Taihoro Nukurangi

Information Stocktakes of Fifty-Five Environmental Attributes across Air, Soil, Terrestrial, Freshwater, Estuaries and Coastal Waters Domains

Prepared for Ministry for the Environment

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Executive summary

Characteristics of our land, water, soil and air—known as attributes—often have clear links to human health or ecological integrity. However, information for deciding which attributes to measure and monitor to understand how our environment is changing, and to maintain human health and ecological integrity at acceptable levels, is lacking.

This project utilised the subject matter expertise of forty-three researchers from NIWA, Manaaki Whenua Landcare Research (MWLR), Cawthron Institute, and Environet Limited to provide a compendium of information on fifty-five environmental attributes in the domains of Air, Terrestrial, Soil, Freshwater, and Estuaries and Coastal Waters.

The fifty-five attribute summaries will provide a robust base for MfE to draw upon when advising on management, policy, and monitoring. Each summary was deliberately organised so that MfE and others can extract information for a range of possible regulatory and non-regulatory approaches.

Collectively, Subject Matter Experts (SMEs) utilised peer-reviewed literature, grey literature, regional and central government reports, and other information sources to generate "information stocktakes" of fifty-five environmental attributes.

Each information stocktake involved the completion of a standard questionnaire/template. The list of attributes and all questions were supplied by MfE. Each attribute information stocktake was iteratively reviewed and refined, with suggestions from a panel of three Māori environmental researchers, Domain Experts at MfE, and Domain Leaders from provider organisations incorporated.

This report consists of fifty-five individual standalone chapters, each containing one finalised "Attribute Information Stocktake". The chapters are grouped according to environmental domain: Air (eleven attributes), Terrestrial (eight attributes), Soil (eleven attributes), Freshwater (seven attributes), and Estuaries and Coastal Waters (eighteen attributes).

At MfE's request, the stocktakes were designed to generate high-level, consensus-type information that will be useful at the national scale. Information from individual chapters can be selected and utilised variously as necessary to facilitate management. The final product was broadly and collaboratively developed.

There was variation in the amount of information available across attributes, and interpretation of questions by individual SMEs differed. The Māori environmental researcher review panel also noted high variation in the familiarity of SMEs with iwi/hapū influence on regional and national policy and planning (e.g., Treaty settlements) and language used by MfE within the context of environmental assessments (e.g., rāhui, tikanga, mātauranga). Māori-driven cultural monitoring and assessment programmes can significantly add value to regional environmental monitoring initiatives by providing a more holistic and integrated assessment of ecosystem and cultural health and wellbeing. How these ways of knowing are effectively empowered in decision making processes and frameworks is critical.

Overall and within each environmental domain, the State of Knowledge of the attributes was determined by SMEs to be "Good", suggesting that there is general agreement on relationships between the selected attributes and ecological integrity / human health. The State of Knowledge was "Poor" for just six of the fifty-five attributes. The relatively high State of Knowledge across attributes is promising, as it is one criterion used to assess their utility in management applications.

1 Introduction

An ability to understand and describe the state of the environment is a critical step in environmental management and achieving desired outcomes for people and nature.

'Attributes' are biophysical characteristics of the environment that can be used to understand and describe environmental state, e.g., in state of the environment (SOE) monitoring programmes. Attributes are even more useful when they clarify desirable environmental states and the possible pathways for achieving them (i.e., determining how and what we need to manage). Robust attributes provide us with cause-and-effect understandings of relationships between resource use stressors and environmental state responses, and they enable evaluations of the effectiveness of management levers. Understanding causal relationships facilitates forecasting, cost-benefit analyses of management interventions, and justifications for resource allocation decisions. This is critical to achieving environmental objectives for the things we value.

In February 2024, Ministry for the Environment (MfE) contracted NIWA to lead a compilation of background information on fifty-five biophysical characteristics of the environment—attributes[1]—within the environmental domains of air, terrestrial indigenous biodiversity, soil, freshwater, estuaries, and coastal waters. For each attribute, the aim was to produce a brief but informative high-level summary of available knowledge for use in various management and national environmental reporting applications. The fifty-five attribute summaries will provide a robust base for MfE to draw upon when advising on management, policy, and monitoring. Each summary was deliberately organised so that MfE and others can extract of information for a range of possible regulatory and non-regulatory approaches. The structure of the summaries will also facilitate the comparison of information across attributes if necessary and where appropriate. For example, 'State of Knowledge' rankings can be assessed within and among environmental domains and for the fifty-five attributes as a whole.

NIWA proposed to jointly supply this work using a consortium of subject matter experts from NIWA, Manaaki Whenua Landcare Research (MWLR), Cawthron Institute, and Environet Limited. Together, these organisations had the personnel and subject matter expertise to cover all five environmental domains and could assign at least one highly qualified subject matter expert to each of the fifty-five attributes.

One "Attribute Information Stocktake" was produced for each attribute (eleven for the Air Domain, eight for the Terrestrial Domain, eleven for the Soil Domain, seven for the Freshwater Domain, and eighteen for the Estuary/Coastal Domain). The names of all attributes grouped by Domain are provided in Table 1, below. Each completed stocktake was written as a standalone chapter, and all stocktakes addressed the same set of questions.

This work has been collaborative, benefiting from diverse skills and experience. MfE staff chose the attributes and questions to be covered in this report based on results of progenitor projects undertaken by MfE and Cawthron Institute (unpublished), which considered whether attributes were important, suitable, and feasible for monitoring and/or management. Te Uru Kahika, the regional and unitary council collective, also contributed to determining both the attributes and the questions addressed in this report. MfE staff directly liaised with stocktake authors, advising and reviewing

[1] The terms 'attribute' and 'indicator' are often used interchangeably. We have tried use 'attribute' consistently in this report.

draft stocktakes as they were completed. NIWA led the project, communicating regularly with MfE, Domain Leaders, Māori environmental researchers, and Subject Matter Experts.

The attributes were selected to complement and expand on MfE's existing internal technical work. Excluded from this report are the attributes developed for the draft National Planning Framework under the now repealed Natural and Built Environment Act. MfE will publish information stocktakes on these attributes later, in a format consistent with the Attribute Information Stocktake chapters provided in this report. There are many additional attributes that were not included here for various reasons (e.g., the emerging cross-cutting issue of microplastics in the environment); MfE will address these in separate horizon scans.

2 General methodology and review procedures

2.1 Attribute information stocktake questionnaire/template

In a Request for Quote (RFQ) document released on 06-Nov-2023 (#1250-01-RFQ), MfE outlined its need for a compilation of focused and relevant information on a set of named attributes. Sixty-four attributes were specified in an Appendix of the RFQ document. Ultimately, a subset of fifty-five attributes selected by MfE were contracted to be completed as part of the project workplan.

MfE also provided a draft set of questions for the attribute information stocktake in the RFQ. The aim was to ensure that the information collated would be focused and relevant, and that information compiled was comparable across attributes. The questions were developed after reviewing MfE's internal templates and processes, including work done for Limits and Targets under the repealed Natural and Built Environment Act, the MfE te ao Māori Analysis tool, and the MBIE Regulatory Impact Statement template. The questions were tailored to elicit responses as to the science and data supporting the attributes.

One of the first methodological steps of the project following contract signing was for MfE and the project's Domain Leaders to discuss, clarify, and finalise the set of questions provided by MfE in the RFQ so that a standardised questionnaire/template could be sent out to all individual Subject Matter Experts (SMEs) for completion.

It was agreed that a few of MfE's draft questions would be removed. Extra directions, clarification, and example answers were also provided on the questionnaire template to ensure, to the greatest degree possible, that similar types of answers and information would be provided by all SMEs. Nevertheless, given the range of attributes covered across multiple environmental domains, and the large number of individual SMEs being relied upon to provide stocktakes, variation in the responses received was expected. Several different phases of review of the stocktakes was specified in the project plan (see Section 2.2, below) to control this variation and provide a more standardised set of answers.

Attribute Information Stocktake questionnaires were sent to SMEs in two tranches. Roughly 25% of the stocktakes were completed before the end of April 2024 (Tranche I). The Tranche I stocktakes were presented to MfE for comment and discussed in a meeting. This allowed MfE and Domain Leaders to ascertain the level of variation in responses received and to make small adjustments to the questionnaire template to clarify or fix issues encountered. It also enabled MfE to highlight examples that fit the purpose well to SMEs who were scheduled to complete stocktakes in the Tranche II phase of the project. Overall, this provided a robust plan for ensuring delivery of useful and informative stocktakes to MfE in a tight delivery timeframe.

The general methodology followed by SMEs to complete a stocktake included:

- Answering all questions.
- Not changing MfE's questions.
- Asking for clarification from respective Domain Leader when required.
- Spending 10–12 hours maximum to complete the questionnaire.
- Providing one paragraph per question (no more than 3 paragraphs per question).
- Providing an overall and national picture rather than specific/local details.
- Supporting answers with references and using a weight-of-evidence approach, based on peer-reviewed literature, grey literature, regional and central government reports, and other information sources.
- Sending completed questions to respective Domain Leaders.

2.2 Review procedures, including review by Māori environmental researchers

2.2.1 Review by MfE Domain Experts and Domain Leaders from NIWA and MWLR

As soon as stocktakes were completed by SMEs, they were shared with MfE. In some cases, the completed stocktakes were deemed by MfE to be suitable "as is" or with very small corrections. In other cases, SMEs worked with MfE staff and Domain Leaders to resolve larger issues and finalise the stocktakes.

Because of tight timeframes and the large number of stocktakes being completed by SMEs simultaneously, it was not always possible to solicit reviews in the same order (e.g., MfE Domain Experts first, then Māori environmental researcher reviews, then Domain Lead reviews). Also, in some cases, preliminary drafts (rather than drafts revised by SMEs in response to MfE comments) were reviewed by the Māori environmental researcher panel or by Domain Leaders. Nevertheless, all sets of review comments were ultimately incorporated into the final stocktake product. The finalised stocktakes were merged into a single document as "chapters" in this report, grouped by environmental domain (Air, Terrestrial, Soil, Freshwater, Estuary/Coastal). A final overall review by the Project Leader and quality assurance checking by NIWA was undertaken.

2.2.2 Review by Māori environmental researchers

Three Māori environmental researchers from Manaaki Whenua and NIWA reviewed all of the information stocktakes, focusing on questions A5, B6, C2 iii and D2 (Table 2-1). One researcher reviewed stocktakes in the Air and Freshwater domains (E. Williams, NIWA, 18 stocktakes), one reviewed those in the Terrestrial and Soil domains (N. Harcourt, MWLR, 19 stocktakes), and the third reviewed stocktakes in the Estuaries and Coastal Watersdomain (A. Kainamu, MWLR, 18 stocktakes). The reviewers collectively self-selected domains that were within their general area of expertise as researchers, though they acknowledge not having familiarity with all areas or attributes within those domains. No three persons can be expected to have the breadth of expertise of a collective of forty-three contributing SMEs.

Table 2-1: Questions A5, B6, C2-iii, and D2. Three Māori environmental researchers reviewed SME answers to these questions in all fifty-five stocktakes.

A5 - Are there examples of this being monitored by Iwi/Māori? If so, by who and how?
B6 - What tikanga Māori and mātauranga Māori could inform bands or allocation options? How? For example, by contributing to defining minimally disturbed conditions, or unacceptable degradation.
C2 – iii. Are there [iwi/hapū driven] interventions/mechanisms being used to affect this attribute? What evidence is there to show that they are/are not being implemented and being effective?
D2 - Where and on who would the economic/environmental/human health impacts likely be felt?

The Māori environmental researcher reviewers noted wide variation in the responses of SME authors to questions A5, B6, C2 iii and D2. Answers depended on their interpretation of the questions, experience working with iwi/hapū, understanding of iwi/hapū influence on regional and national policy and planning (e.g., Treaty settlements), and understanding of language used by MfE within the context of environmental assessments, e.g., rāhui, tikanga, mātauranga.

The panel of Māori environmental researchers conferred with each other and removed text that reflected incorrect understandings of iwi/hapū governance and management influence, te reo, mātauranga, and cultural values/principles. These reviewers also added text where possible to improve te reo usage and increase the standardisation of text across attributes.

2.2.3 Māori review limitations, further context and considerations

Limitations

The co-development and co-implementation of new innovative approaches has enormous potential to acknowledge mātauranga Māori and move beyond approaches that rely heavily on western philosophies to ones recognising cultural expertise and benefit from indigenous knowledge systems. The reviewers did not wish to conflict with, replace, or supersede the distinct perspectives of whānau/iwi/hapū and any respective outputs, actions, or initiatives being lead/influenced by them. Hapū/iwi require access to high-quality information to inform their own debates, decision making, and research, and to assist them to monitor the effects of regional and national government policies and programmes relating to Māori. Many whānau/hapū/iwi wish to contribute to their own environmental management but also to mainstream regional and national policy, monitoring and planning. Further, as identified by the PCE (2019), the development of environmental reporting frameworks and monitoring tools that are underpinned by cultural values and provide a role for hapū/iwi in implementation and interpretation, would be a significant step forward for Aotearoa-New Zealand.

Stocktake authors and reviewers were not resourced to engage with whānau/iwi/hapū to contribute to or endorse this report. There are many examples of hapū/iwi-driven cultural assessment frameworks and monitoring methods/tools (and an increasing number of publications) that could provide guidance around processes for working with hapū/iwi and their mātauranga, capability, and capacity in environmental monitoring and reporting. Rather than relying on a small group of Māori environmental researchers (three people) to cover questions for the entire country, we recommend that MfE fund and facilitate a much larger advisory group of iwi representatives and Māori researchers, similar to the consultation/engagement that occurred during the early development of the National Objectives Framework.

There are examples where tikanga Māori and mātauranga Māori have informed bands and/or allocation options and/or explained "minimally disturbed conditions or unacceptable degradation". However, the knowledge of this body of literature and where it resides, e.g., treaty settlements, cultural impact assessments, environmental court hearings, iwi environmental management plans, journal papers, reports, etc., was very patchy among the SMEs (or they interpreted their attribute very narrowly). Whilst some of these examples may not be directly applicable to particular attributes, there are transferrable learnings and adaptable methods that could be explored in the future.

Within attributes, some questions were difficult for the author/reviewers to assess as more context was needed around the proposed scale (i.e., local, regional and/or national scales) and therefore who, in terms of "Māori", should be involved. This work was scoped by MfE to have a national lens, but MfE acknowledges that this scale may be discordant with the typically place-based mātauranga held by mana whenua.

It must also be acknowledged that the methodologies underpinning the gathering and interpretation of datasets generated by agencies to monitor the state of the environment have generally taken decades of development, with contributions from an international community of scientists and practitioners. The same courtesy needs to be extended to the development of cultural monitoring approaches as it will take time (and continuity of resourcing) to develop all the approaches/methodologies required to generate the holistic datasets (qualitative and quantitative) required by hapū/iwi across all of the attributes and domains included in this review, should they wish to do so.

Further context and consideration

Te Tiriti o Waitangi forms the underlying foundation of the Crown-Māori relationship with regard to Aotearoa-New Zealand's environment. Māori fear that a failure to recognise and provide for their cultural beliefs, values, and their customary practices will ultimately destroy many of the foundations of their culture and identity. This tension has surfaced in many forums, where the challenge Māori confront is not being decision makers in their own right, and/or to convey to decision makers how decisions affect their cultural interests, since many of the existing management methods and models are dominated by western/conventional science techniques which emphasise physical and biological values rather than specifically responding to cultural needs.

Māori-driven cultural monitoring and assessment programmes can significantly add value to regional environmental monitoring initiatives by providing a more holistic and integrated assessment of ecosystem and cultural health and wellbeing. 'Monitoring' is usually one component within a strategic workplan that may be implemented to support iwi and hapū with co-management of environmental resources. Additionally, the methods and processes used by iwi and hapū identify and describe their values, assess the baseline state of these values and the pressures impacting on them, as well as identifying appropriate tools and approaches to monitor changes in state at temporal and spatial scales of relevance to them. This inevitably involves its own iterative processes and methodologies. Ruha et al. (2021) further explains "…cultural health indicators are a geographically specific means of enabling the measurement of a particular attribute of a Hapū or Iwi, then to be appropriately recognised, the method of inclusion in any framework is at least as important. Culture is defined here as a geographically specific expression of identity. Cultural health indicators are the appropriate geographically specific means of representing that identity and its flourishing which enables that culture to be recognised and measured. Therefore, cultural indicators, their definition and their measurement must be the sole prerogative of the relevant Hapū. Moreover, how these ways of knowing are effectively empowered in decision making processes and frameworks is critical as decisions are no longer being made by Iwi and Hapū in isolation".

Settlement legislation is often a motivating factor (e.g., Waikato-Tainui Raupatu Claims (Waikato River) Settlement Act 2010, Ngā Wai o Maniapoto (Waipā River) Act 2012). For example, the aforementioned settlement acts reflect a commitment by the Crown to enter a new era of co-management to restore and protect the health and wellbeing of the Waikato River through relationships with Waikato-Tainui, Raukawa Settlement Trust, Te Arawa River Iwi Trust, Tūwharetoa Māori Trust Board and Maniapoto Māori Trust Board. The co-management instruments (e.g., Waikato River Authority) provide a raft of opportunities for iwi and hapū to engage in good faith and consensus decision-making. Key elements of these settlement include joint management agreements (JMAs) and transfer of powers with local government. That said, all of the five iwi have JMAs with Waikato Regional Council (WRC) and, the WRC reached a landmark decision in 2020, to transfer of certain water quality monitoring functions to Tūwharetoa Māori Trust Board (Tūwharetoa Māori Trust Board 2021).

Iwi and hapū are undertaking a variety of environmental monitoring and assessment initiatives to support their participation in all aspects of decision-making. Various reviews have been undertaken over the years to communicate some of the approaches that have been developed by iwi/hapū/Māori researchers in these spaces (e., Nelson & Tipa 2012, Rainforth & Harmsworth 2019). Often the components (e.g., cultural preference studies, assessment frameworks, monitoring methods/tools/programmes, capability building/training, evaluation approaches) are seen as being one and the same thing by agencies, but this is not the case. Some examples of assessments frameworks and monitoring approaches iwi and hapū have developed and tested across New Zealand include (but are not limited to):

- Cultural Health Index (CHI) for streams (Tipa and Teirney 2003, 2006) and adaptations thereof (including adaptations for kauri/ngahere, lake, estuarine, coastal and urban environs).
- Cultural indicators of wetlands (Harmsworth 1999, 2002) and adaptations thereof.
- State of the Takiwā (Pauling et al. 2007) and adaptations thereof.
- Mauri-ometer (Morgan 2006, 2007; http://mauriometer.org/).
- Mauri Compass (https://www.mauricompass.com/).
- Cultural flow preference and opportunity assessments (https://www.culturalflows.co.nz/).
- Wai Ora Wai Māori assessment tool (Awatere et al. 2017).
- Report cards (Tipa 1999, Williamson et al. 2016).
- Murihiku Cultural Water Classification System (Kitson and Cain 2022; Kitson et al 2018) and Te Mauri o Waiwaia – Maniapoto Cultural Assessment Framework (Kaitiaki contributors et al. 2023).

Some of the above methods may be implemented within a framework that also draws in other data gathering approaches/tools such the Stream Health Monitoring Toolkit and Estuarine Monitoring Toolkit. Iwi and hapū are also exploring the application of new technologies such as in-situ sensors (e.g., for rainfall, water level, water flow and various water quality variables), remote sensing and water/airborne imagery capabilities (e.g., LiDAR, satellite, drones, cameras) that enable the collection of high frequency data in time and space. All of the above methods are dependent on on-going funding.

3 General overview of results

3.1 Variability

Subject Matter Experts attempted to answer all questions in every stocktake. MfE was aware of the potential for high variability in responses due to the:

- large range of material covered by the selected attributes (i.e., multiple environmental domains),
- differing contexts for each attribute (i.e., pertinent to human health, ecological integrity, or both),
- differing amounts of knowledge and published/public information on attributes and their relationships to human health and ecological integrity, and
- differing individual interpretations of questions by the forty-three contributing SMEs.

A key part of Domain Leader reviews was to coordinate with MfE's Domain Experts and SMEs to revise and better standardise the information content and State of Knowledge rankings across stocktakes.

3.2 State of Knowledge

Across attributes, MfE wanted to know when answers were simple or complicated. If the answers were deemed complicated, MfE wanted responses to address whether they were answerable and how. It was expected that answers to some questions would be unknown and would require further research. In short, MfE wanted information on the State of Knowledge of each attribute.

In the questionnaire provided to SMEs, standard definitions of four State of Knowledge classes (Poor, Medium, Good, Excellent; IPBES 2018; Table 3-1) were provided. SMEs were asked to rank the State of Knowledge on the attribute to which they had been assigned. SMEs were reminded that rankings from "Poor" to "Excellent" referred to the amount of information/evidence available and did not necessarily reflect the importance or value of an attribute.

State of Knowledge values from SMEs spanned the entire range from "Poor" to "Excellent" (Table 3-1). The domains of Air, Soil, Freshwater, and Estuaries and Coastal Waters each had at least one attribute for which the State of Knowledge was deemed to be "Poor", whilst all State of Knowledge rankings for the Terrestrial domain were in the top two categories of "Good" or "Excellent" (Table 3-1).

Overall, the average State of Knowledge rankings made by SMEs was deemed to be "Good" (Table 3-2), with 30.5 out of 54 rankings (56%) falling into this category. The domains with the highest and lowest percentages of "Good" rankings were Terrestrial (86%) and Estuaries and Coastal Waters (44%), respectively (Table 3-2). The Air domain had the highest percentage of attributes with a State of Knowledge ranking of "Excellent" (27%).

The overall high percentage of “Good” rankings for State of Knowledge suggests that there is general agreement, although limited data/studies, on relationships between the selected attributes and ecological integrity /human health. The State of Knowledge rankings were promising, in that State of Knowledge is a criterion used in assessing the utility of attributes in management applications.

Table 3-1: State of Knowledge classification by Subject Matter Experts for fifty-five attributes across five environmental domains (Air, Terrestrial, Soil, Freshwater, and Estuaries and Coastal Waters).

			State of Knowledge rank according to SME using IPBES 2018 category definitions				
Domain Name	**Attribute Name**	**Subject Matter Expert (SME) names**	**Poor / inconclusive** based on a suggestion or speculation; no or limited evidence	**Medium / unresolved** – some studies/data but conclusions do not agree	**Good / established but incomplete** – general agreement, but limited data/studies	**Excellent / well established** – comprehensive analysis/syntheses; multiple studies agree	**Foot Note**
Air	Light pollution	Michelle Greenwood [N]			1		
Air	Nitrogen dioxide	Guy Coulson [N] and three others				1	
Air	Sulphur dioxide	Emily Wilton [E]				1	
Air	Carbon monoxide	Emily Wilton [E]				1	
Air	Ozone	Emily Wilton [E]			1		
Air	Black Carbon	Guy Coulson [N] and two others			1		
Air	Benzo(a)pyrene	Jo Cavanagh [M]	0.5		0.5		a

Air	Arsenic	Jo Cavanagh [M]	0.5		0.5		a
Air	Benzene	Jo Cavanagh [M]			1		
Air	Lead	Jo Cavanagh [M]	0.5		0.5		a
Air	Cadmium	Jo Cavanagh [M]	1				
Terrestrial	Wetland extent	Olivia Rata Burge [M]					b
Terrestrial	Dune extent	Al Alder [C] Anna Berthelsen [C]			1		c
Terrestrial	Lowland forest extent	Susan Walker [M]			1		
Terrestrial	Wetland condition index	Olivia Rata Burge [M]			1		
Terrestrial	Dune condition index	Al Alder [C] Anna Berthelsen [C]			1		c
Terrestrial	Indigenous plant dominance	Peter Bellingham [M] Duane Peltzer [M]				1	
Terrestrial	Canopy die back extent	Peter Bellingham [M] Duane Peltzer [M]			1		
Terrestrial	Landscape connectivity	Tom Etherington [M] James McCarthy			1		
Soil	Peatland/peat soils subsidence control	Jack Pronger [M]			1		

Soil	Soil Bacteria composition	Eva Biggs [M]	1				
Soil	Soil Nitrogen and phosphorus	Hadee Thompson-Morrison [M]		0.5	0.5		d
Soil	Soil Carbon	Hadee Thompson-Morrison, Sam McNally [M]		1			
Soil	Landslide susceptibility mitigation	Chris Phillips [M]		1			
Soil	Gully erosion	Chris Phillips [M]		0.5	0.5		d
Soil	Surface erosion/runoff control	Chris Phillips [M] John Drewry [M]			1		
Soil	Riparian protection/streambank erosion control	Chris Phillips [M]			1		
Soil	Soil compaction	John Drewry [M]				1	
Soil	Soil water storage, capacity and fluxes	John Drewry [M]			1		
Soil	Soil contaminants	Jo Cavanagh [M]			1		
Freshwater	Riparian margin establishment protection	Fleur Matheson [N]			1		
Freshwater	Heavy metals in water	Louis Tremblay [M]			1		
Freshwater	Heavy metals in sediment	Louis Tremblay [M]			1		

Freshwater	Groundwater depletion	Channa Rajanayaka [N]				1	
Freshwater	Surface water flow alteration	Doug Booker [N]			1		
Freshwater	Catchment permeability	Christian Zammit [N]	1				
Freshwater	Groundwater nitrates	Chris Tanner [N]			1		
Estuary/Coastal	Seagrass quality and extent	Anna Berthelsen [C]			1		
Estuary/Coastal	Saltmarsh quality	Anna Berthelsen [C] Al Alder [C]			1		
Estuary/Coastal	Mangrove forest extent and quality	Andrew Swales [N] Carolyn Lundquist [N]			1		
Estuary/Coastal	Shellfish bed extent and quality	Drew Lohrer [N]		1			
Estuary/Coastal	Kelp forest extent and quality	Leigh Tait [N] Roberta D'Archino [N]		1			
Estuary/Coastal	Bryozoan thickets extent and quality	Mark Morrison [N]				1	
Estuary/Coastal	Macroinvertebrate community composition	Orlando Lam-Gordillo [N] Drew Lohrer [N]			1		
Estuary/Coastal	Heavy metals in sediment	Louis Tremblay [M]			1		
Estuary/Coastal	Heavy metals in water (or indicator spp.)	Louis Tremblay [M]			1		

Estuary/Coastal	Underwater noise attribute / ocean sound	Deanna Clement [C]			1		
Estuary/Coastal	Extent of mud (broad scale attribute)	Drew Lohrer [N]		1			
Estuary/Coastal	Suspended sediment / water clarity / turbidity	Rob Davies-Colley [N]			1		
Estuary/Coastal	Phytoplankton / Chlorophyll *a* (trophic state)	Mark Gall [N] Matt Pinkerton [N]	0.5	0.5			d
Estuary/Coastal	Dissolved oxygen in water (trophic state)	Bruce Dudley [N] David Plew [N]				1	
Estuary/Coastal	Nutrients in water (trophic state and toxicity)	Bruce Dudley [N] John Zeldis [N]				1	
Estuary/Coastal	Faecal indicator Bacteria in water	Rob Davies-Colley [N] Rebecca Stott [N]		1			
Estuary/Coastal	Faecal indicator Bacteria in Shellfish	Rebecca Stott [N] David Wood [N]		1			
Estuary/Coastal	Cyanobacteria in water	Laura Biessy [C] Susie Wood [C]	1				

a SMEs gave two answers (e.g., good/established for ecological integrity, but poor/inconclusive for human health; 0.5 points allocated to each)
b SME did not answer because answer was deemed highly scale-dependent
c SME provided qualified/equivocal answer
d SME said ranking fell between two adjacent categories (0.5 points allocated to each category in Table)
C Cawthron Institute
E Environet Limited
M Manaaki Whenua Landcare Research
N NIWA

Table 3-2: Average weighted State of Knowledge scores for Domains and overall. Average weighted scores were calculated as Σ(Poor ranks*1, Medium ranks*2, Good ranks*3, Excellent ranks*4) divided by the number of attributes ranked (either by domain or overall). A tabulation of the scores provided in Table 3-1, per domain and overall, are provided in the right-hand columns.

	Average weighted score	Sum (%) of "Poor" rankings	Sum (%) of "Medium" rankings	Sum (%) of "Good" rankings	Sum (%) of "Excellent" rankings
Air (n=11)	**2.82**	2.5 (23%)	0 (0%)	5.5 (50%)	3 (27%)
Terrestrial (n=7*)	**3.67**	0 (0%)	0 (0%)	6 (86%)	1 (14%)
Soil (n=11)	**2.64**	1 (9%)	3 (27%)	6 (55%)	1 (9%)
Freshwater (n=7)	**2.86**	1 (14%)	0 (0%)	5 (71%)	1 (14%)
Estuaries and Coastal Waters (n=18)	**2.69**	1.5 (8%)	5.5 (31%)	8 (44%)	3 (17%)
Overall (n=54*)	**2.84**	**6 (11%)**	**8.5 (16%)**	**30.5 (56%)**	**9 (17%)**

* State of Knowledge was not ranked for one attribute (Wetland extent); see Table 3-1.

3.3 Gaps identified and final thoughts

It is difficult-to-impossible to summarise the volume and diversity of information compiled in this report. We believe that the utility of the compendium lies in the individual stocktake chapters. For people interested in particular research areas, attributes, or domains, the chapters provide access to high-level scans of information and numerous relevant citations. As such, the chapters will serve as an entry point for further investigation if required.

We note that the Estuaries and Coastal Waters domain had the largest number of stocktake chapters (eighteen) and the largest number of low ("Poor" to "Moderate") State of Knowledge rankings. There may be opportunities to prioritise, shortlist, and further develop the attributes in this domain. Poor understanding of relationships between attributes and ecological integrity in the Estuaries and Coastal Waters domain may stem from difficulties in sampling estuarine and coastal areas relative to other domains. For example, remote sensing tools used to study the surface of the Earth are much less effective at capturing images of the seabed and thus estuarine/coastal habitats (Townsend et al. 2018). This may affect quantification of several estuarine/coastal attributes (e.g., Mud extent, Seagrass extent and quality, Bryozoan thicket extent and quality, Shellfish bed extent and quality, Kelp bed extent and quality) to a greater extent than terrestrial attributes such as Lowland Forest extent, Dune Extent, and Canopy dieback extent. Technology, aerial image resolution, and computing power are rapidly improving, so advances in our understanding over time are likely. Therefore, we must be careful not to discard potentially important attributes just because our current understanding is poor.

This report contains information on multiple attributes in each of five environmental domains. A critical research need in New Zealand at present is defining cause-and-effect relationships between stressors and environmental states. This compendium will facilitate the identification of attributes that may, with further research, provide the strongest causal relationships and prove to be the best candidates for evaluating the effectiveness of management interventions. We urge MfE and research providers in New Zealand to use the information provided to make progress in this area.

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5 Air Domain

Eleven attribute information stocktakes for the Air Domain are provided in sections 5.1 to 5.11, below. Peggy Cunningham-Hales and Owen West (MfE Domain Experts), Dr Jo Cavanagh (MWLR, Domain Leader), and the Māori environmental researcher panel reviewed these sections.

5.1 Light pollution

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State of knowledge of "Light Pollution" attribute: Good / established but incomplete

The International Commission on Illumination (CIE) defines light pollution as the "sum total of all adverse effects of artificial light". The impacts of light pollution on the visibility of the night sky are well understood and globally similar[1]. Ecological impacts of light pollution are generally consistent but will vary depending on the light sensitivity, behaviour and lifecycle of the focal organism, with potentially complex impacts across ecosystems due to species interactions[2-4]. While there is good international understanding of the ecological impacts of light pollution on a range of species and ecosystems, more limited New Zealand specific research has been undertaken, with none for marine mammals and herpetofauna[5]. Similarly, the recent and ongoing global conversion of outdoor lights to LEDs and associated shift in lighting spectra is a comparatively new research area[6, 7].

Section A—Attribute and method

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

There is strong evidence globally that light pollution negatively affects terrestrial[2, 8-10], freshwater[11-13], marine[14, 15] and agricultural ecosystems[16] by affecting species behaviour[17, 18], physiology, life-cycle timings, growth rates[13]**,** food availability [19] and interactions with other species[2]. Light pollution can be caused directly by illumination from light sources or indirectly through reflection off surfaces. Sky glow is an increase in apparent brightness of the night sky, exacerbated by reflection off clouds that extends far beyond urban areas[20]. Evidence is strong for multiple mechanisms of ecological impact including reduced visibility of the night sky impacting celestial navigation[21], behaviour and lifecycles regulated by the lunar cycle[14]. Other mechanisms include behavioural changes such as avoidance of lit areas[22], or increased predation opportunities[23, 24], leading to impacts on entire foodwebs [2]. There is more limited, but growing, research for specific impacts on New Zealand ecosystems[5].

The recent global switch to LEDs for outdoor lighting may result in stronger impacts of light pollution, due to common increases in blue light emissions[25]. Many insects are more sensitive to shorter wavelengths[26], and blue light scatters in the atmosphere more[27], penetrates deeper into water waterbodies[2] and may result in stronger impacts on human health [28]**.**

I focus on impacts on ecological integrity, however there is evidence for impacts of light pollution on cultural values (such as Māori astronomical knowledge (Tātai arorangi [29]) and human health[28]).

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

The evidence is strong for impacts of light pollution on ecological integrity globally[10, 30-32]. While there are fewer studies specific to New Zealand[5, 8, 20, 33], the impacts are likely to be similar. Globally the spatial extent of impact is large, particularly for densely populated countries where there may be limited dark refuges[1, 34] and ~47% of the terrestrial land area globally affected by skyglow[35]. In contrast, New Zealand currently has very limited spatial extent of light pollution (~95% of land area with no light emissions)[5], although almost all (97%) of our population lives under skies impacted by light pollution[1]. However, skyglow from urban areas impacts night sky brightness and visibility at distance up to 100km, impacting marine areas[15] and ecological reserves in New Zealand [20, 27] and Key Biodiversity Areas internationally [36]. The spatial extent of light pollution is increasing in New Zealand, particularly around the edges of cities and in rural areas[5].

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

Globally the spatial extent of light pollution is increasing due to population expansion and the development of cheaper lighting sources[1, 37]. The average rate of increase in spatial area of light emissions in NZ was 4.2% between 2012 and 2021 [5], slightly above global average rate[1]. Although areas impacted by light pollution are increasing, the majority of the land area of New Zealand is still unimpacted by light emissions[5]. In addition, increasing numbers of semi-urban areas are being protected through the development of dark sky reserves and parks[38]. Improvements in lighting technology such as use of alternative light spectra, better shielding of lighting units and an ability to dim lights or use sensors or timers to reduce lighting duration, intensity and spatial extent are available. Light pollution is a relatively unique stressor with a straightforward solution to reducing impacts – turn the lights off (or down). We have the technology and enough knowledge of key mechanisms to reduce ecological impacts. It is theoretically relatively straightforward to reduce and reverse light pollution impacts using appropriate technology and policy.

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

Light pollution is measured in a multitude of ways depending on whether the variable of interest is night sky visibility, ecological values, human health or human safety and inconsistencies between disciplines causes confusion [39]. There are recommendations for different methods and units of measurement depending on the goal of the monitoring[39]. Satellite imagery and astronaut photographs from the ISS are freely available to monitor upward light across broad spatial scales (https://eol.jsc.nasa.gov). StatsNZ reports estimates from satellite estimates from 2016 [1] as a national indicator of Artificial Sky Brightness (https://www.stats.govt.nz/indicators/artificial-night-skybrightness). Satellite imagery likely underestimates light pollution changes detected on the ground as not all lights are directed upwards[5, 37]. Citizen science projects exist for ground-based monitoring with free data[37]. In New Zealand there is no regular ground-based monitoring of light pollution, but small datasets exist from several methods (e.g., Sky Quality Meters, lux meters) which are commonly used methods internationally. These data are collected over limited spatial and temporal extents for research projects, community groups, dark sky areas and astronomical societies.

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

Monitoring implementation issues depend on the monitoring method. Satellite imagery is freely available but only monitors upwards light at a broad spatial scale and current satellites are not sensitive to blue light[39], which is problematic given the global conversion to predominantly blue-white LEDs for outdoor lighting. Setting up ground-based networks of continuously monitoring telemetered sensors could be relatively easy, depending on location requirements (e.g., private vs public land) and scale of the monitoring network. Night sky brightness and local light levels are influenced by cloud cover and would benefit from simultaneous monitoring of weather conditions. Remote predictions of cloud cover are available from NASA MERRA-2 climate analysis project but may not correlate well with local ground conditions.

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

Costs for monitoring depend on the data collection methods, which depends on the goal of the monitoring. Satellite imagery of upward light emissions is free with relatively minimal post-processing required. Ground-based Sky Quality Meters are comparatively cheap (~$250 each) although telemetered versions are more expensive. These could be installed on existing infrastructure with the cost dependent on the size of the monitoring network. Digital cameras can also be used or illuminance (lux) meters, which are not prohibitively expensive (approximately $100s to $5000). More expensive tools include more detailed imaging instruments. Most methods require relatively minimal post-processing, but calibration requirements vary. Conversion of collected data to indicator values or interactive live maps would involve a one-off cost to develop the code and set up the system but minimal ongoing costs.

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

Night sky visibility has high cultural relevance given the importance, for example, of tātai arorangi (Māori astronomical knowledge), the maramataka (lunar calendar) in understanding seasons and time, cultural landscapes, celestial navigation and the ability to view the Matariki (Pleaides) constellation [29]. To reduce the impacts of light pollution on their cultural values and knowledge systems, several iwi/hapū/rūnanga are actively working towards dark sky sanctuary status (e.g., Te Rūnanga o Kaikoura [46]. Although we are not aware of any in-depth on-going monitoring of light pollution by iwi/hapū/rūnanga, monitoring of night sky visibility may be undertaken within some of the dark sky reserves (such as Aoraki Mackenzie International Dark Sky Reserve).

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

Light pollution is likely to impact and/or interact with many of the other listed attributes. However, methods to monitor light pollution are different from the other attributes and likely to be stand-alone.

Part B—Current state and allocation options

B1. What is the current state of the attribute?

We have a good understanding of the current state of upward light pollution at a broad national scale from satellite data[5]. StatsNZ reports estimates from satellite measurements from 2016 [1] as a time- static national indicator of Artificial Sky Brightness (https://www.stats.govt.nz/indicators/artificial-night-skybrightness). Currently a limited area of New Zealand is impacted by upward light pollution (<5%), although more will be impacted by sky glow[15, 27] and almost all our population lives under light polluted skies. An indicator could be developed at a national or regional scale using satellite data for upward light pollution. However, linking this to impacts on particular taxa or locations will be more challenging due to the scale mis-match, the fact that satellites don't monitor skyglow, limitations in the spectral sensitivity of the current satellites, and that light from non-upward directions also cause ecological impacts. Although satellite imagery can be a good indicator of local night sky brightness in some locations[15] we have comparatively poor understanding of light pollution at a local scale, at a broad-scale from directions that are not upward and the potential impacts on New Zealand taxa and ecosystems.

Development/choice of an indicator would need to consider what it was designed to monitor (night sky visibility or broad or specific ecological impacts) and the relevant spatial and temporal scale for monitoring would need to be decided before selecting an appropriate indicator. There are a selection of methods and approaches available with good review papers[39].

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

Measurements from the large areas of New Zealand unimpacted by light pollution would provide good reference states for comparison to impacted areas. Light pollution could be monitored in remote areas, using selected/multiple units and measurement methods, to form baseline levels from areas with low/absent light pollution. The impact of cloud cover and lunar phase on lighting in these areas could be quantified and used to help understand these impacts in more lit/impacted areas.

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

There are no national standards in New Zealand for light pollution. Appropriate levels will depend on the purpose light pollution is monitored for. Sky Quality Meters provide a scale for visibility of the night sky that ranges from dark skies with no light pollution to those found in major urban centres, and other scales exist based on the visibility of different constellations. Devising numeric bands for ecological impact would be more challenging as light and spectral sensitivity will vary between taxa.

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

There are thresholds for night sky visibility readings at which different components of the night sky are no longer visible (for example, the milky way). Thresholds for light pollution that have ecological impacts differ depending on the focal taxa or ecosystem and the units used to measure light pollution because species vary in their light and spectral sensitivity. Some general thresholds do co-occur for multiple taxa, however, such as brightness exceeding that present during a full moon, or brightness levels at which the lunar cycle is obscured.

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

Light pollution will generally have limited lag or legacy effects. The immediate impact of light pollution can be removed by turning the lights off. Some lag effects may occur if community composition has shifted due to the impact of the lighting, with rates of community change after the light is removed depending on the activity of the species and proximity of sources of recolonists. The recent conversion of streetlights from older lighting technologies to LEDs may complicate assessment of historical trends of light pollution due to co-occurring shifts in lighting spectra and intensity at the time of the conversion and due to limitations in the spectral sensitivities of some of the monitoring methods. Lunar cycles and cloud cover also need to be accounted for in trend analyses.

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

Star visibility is of cultural importance and use of existing bands that relate night sky brightness to the visibility of certain constellations could be appropriate.

Part C—Management levers and context

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

The relationship between management interventions and light pollution is comparatively easy to understand with use of models and spatial mapping to predict spectral and light intensity outputs occurring in the environment depending on different lighting periods, spectra, light shielding etc.

The relationship between night sky visibility and light pollution is also relatively well understood. The level of lighting reduction required to achieve night sky visibility of a certain level could likely be broadly predicted given local conditions, lunar cycles and atmospheric conditions.

The relationship between light pollution and ecological impacts is more complicated by different light and spectral sensitivities of different species. However, there is a strong consensus that in general, less light is better, and that often a spectral change to reduce the short wavelength light (such as blue light) reduces impacts.

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

Compared to other types of environmental stressors light pollution may be a relatively easy problem to solve[40]. Preventing new areas being lit or using straightforward approaches to reduce light spill, intensity, durations and the use of short-wave length lighting are obvious solutions[2]. A lack of regulations, funding and guidelines are likely limiting implementation of these solutions. A list of management interventions by level of organisation is below:

C2-(i). Local government driven

Almost all district plans contain rules to limit the effects of light spill and glare onto adjacent properties, but few currently include objectives to control the type and quantity of lighting that can be installed to limit ecological impacts. The development of dark sky places encourages such changes. Southland District council has included a plan change for lighting rules on Rakiura-Stewart Island since the development of the Dark Sky sanctuary and the Timaru District Council plan has rules pertaining to the Aoraki Mackenzie International Dark Sky Reserve and recognises the benefits of protecting the night sky.

C2-(ii). Central government driven

There are no central government rules associated with light pollution that I am aware of. New Zealand Transport Agency Waka Kotahi have standards for lighting safety but there are no national policy or standards to reduce the ecological impacts of light pollution.

C2-(iii). Iwi/hapū driven

Iwi/hapū/rūnanga are a driving force behind the development of dark sky areas and have strong public influence and education roles. An example is the Aoraki Mackenzie International Dark Sky Reserve, Dark Sky Project in Tekapo/Takapō, a Ngāi Tahu project, which promotes education around the importance of dark sky preservation.

C2-(iv). NGO, community driven

Community groups and NGO such as dark sky groups and astronomy organisations also contribute strongly to the development of dark sky areas to improve night sky visibility. To be a dark sky area there are lighting requirements that are required to be met before the area can be internationally accredited by the International Dark Sky Association (IDA). The IDA is a non-government not-for-profit organisation established to "preserve and protect the night-time environment and our heritage of dark skies through environmentally responsible outdoor lighting". These include restrictions on the amount of blue light emitted by lighting sources.

Voluntary best-practice guidelines have been developed for several sectors, for example to reduce seabird attraction to commercial fishing vessels[41]. Recognition of the ecological impacts of lighting is supported by the lighting industry in general[42].

C2-(v). Internationally driven

Internationally the importance of darkness within networks of interconnected lit habitats is being recognised but rules to limit lighting are country-specific. Countries with nationwide legislation to reduce the impacts of light pollution include Croatia, France, and Slovenia[43]. Accreditation by the IDA as a dark sky place is strongly attractive and is dependent on meeting lighting criteria. New

Zealand is part of the United Nation's convention on conservation of migratory species, which have endorsed ecological light pollution guidelines.

Part D—Impact analysis

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

The ecological implications of light pollution can be major[2]. More than 50% of New Zealanders cannot see the Milky Way from home [1]. Although the spatial area of New Zealand impacted by direct illumination is very low, these areas can contain important ecological reserves or key populations of taonga species[20]. Larger spatial areas are also impacted by skyglow. Not managing this attribute, particularly as population growth increases the lit area will contribute to species loss and ecological function. There will also be a loss of potential financial gains from astro-tourism, increases in energy costs to light larger areas, and significant cultural impacts. For example, the night sky is integral to tikanga and mātauranga Māori, as evidenced through Māori astronomical knowledge (Tātai arorangi [29]), including Matariki. By not managing this attribute, the importance of the lunar calendar (maramataka) in understanding seasonal processes and the passing of time, and in use of celestial navigation will be impacted.

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

An improvement in light pollution in New Zealand has potential for strong economic benefits as noted in [38]. Using less light will lead to costs savings associated with a reduction in energy use[38]. Astro-tourism can have significant financial benefits and is blooming globally [44]. For example, before the covid pandemic (2020) the Aoraki Mackenzie International Dark Sky Reserve (AMIDSR) had 9 astro-tourism companies operating with visitors to the region spending almost a million dollars a day[38]. New Zealand is world leader in dark sky locations, with the southern-most (Rakiura /Stewart Island) and the largest in the southern hemisphere (AMIDSR). In early 2023 New Zealand had five dark sky places with approximately 15 more hoping to receive such accreditation[38]. An ambitious plan aspires to make New Zealand the second dark sky nation (after Niue): What's a 'dark sky nation' and why does New Zealand want to become one? (nationalgeographic.com)

Costs associated with light pollution causing potential reduction in agricultural productivity[16] or unexpected ecological outcomes in combination with other stressors are likely[45], but difficult to predict or quantify.

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

Environmental changes associated with climate change will not have direct effects on the intensity or spatial distribution of light pollution. Indirect effects may occur through changes in the spatial distribution of urban centres as flood-risk, precipitation or temperature changes alters the habitability of areas. However, the impacts of light pollution will likely have synergistic and potentially unexpected impacts for many taxa and ecosystems in combination with environmental changes caused by climate change[45].

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5.2 Nitrogen dioxide in air

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State of knowledge of '*Nitrogen Dioxide in Air*' attribute: Excellent / well established – comprehensive analysis/syntheses; multiple studies agree.

Part A—Attribute and method

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

NO_2 is a brownish gas mostly emitted by combustion of fossil fuels. Motor vehicles and industrial combustion activities are the main sources of NO_2 in New Zealand (Metcalfe & Sridhar, 2018) although shipping is also a potentially significant source. At an urban scale the spatial distribution in NO_2 concentrations are generally aligned with the road network (Figure 1). NO_2 has also been identified as an indoor air pollutant (Shaw et al 2020, Vardoulakis et al 2020, Gillespie-Bennett et al 2008). Indoor sources of NO_2 include most forms of gas combustion used for cooking or heating.

Health impacts are well known and understood. Short-term exposure to high concentrations of nitrogen dioxide causes a range of respiratory effects include decreased lung function, increased airway hyperresponsiveness, and inflammation of the airways (Health Canada, 2015) and premature mortality (Orellano et al., 2020). It can also cause asthma attacks (US EPA, 2016). Indoor NO_2 has also been found to cause respiratory symptoms in asthmatic children (Health Canada, 2015).

Long-term exposure is associated with all-cause, cardiovascular, and respiratory mortality (Huang et al., 2021). It may cause asthma to develop and lead to decreased lung development in children. It may also increase the risk of certain forms of cancer (US EPA, 2016).

NO_2 is oxidised in the atmosphere to become nitric acid and can cause acid rain if in significant concentrations. This is not generally considered a problem in Aotearoa. Nitric acid can react to become nitrate in aerosol particles, thus contributing to particulate pollution (Xue 2014). NO_2 can cause damage to plants (USEPA, 2024).

Environmental effects also include contribution to the brown haze often seen over Auckland (Dirks et al 2017) through light absorption. Whilst NO_2 is a brown coloured gas, the brown colour of haze typically occurs as a result of scattering of light by particles of varying sizes including by particulate nitrate (Wilton, 2003).

NOx (nitric oxide and nitrogen dioxide) in the atmosphere can also contribute to nutrient pollution in coastal waters. (https://www.epa.gov/nutrientpollution/basic-information-nutrient-pollution)

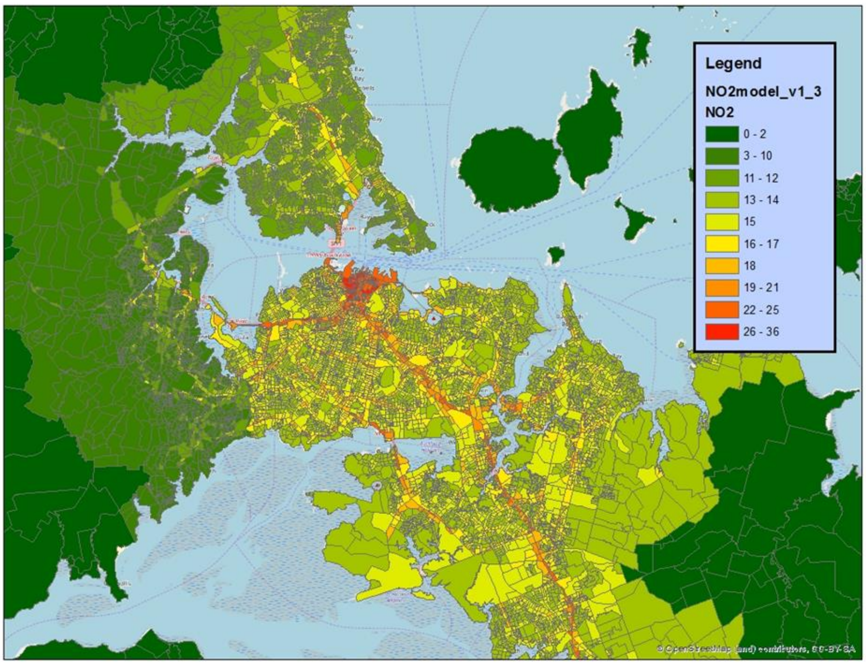

Figure 1. Modelled NO_2 concentrations in Auckland (NIWA, 2019)

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

The weight of evidence on the short term harmful effects of NO_2 is strong (Orellano et al., 2020). The weight of evidence for long term exposures and all cause and respiratory morality is low to moderate and high for chronic obstructive pulmonary disease (Huangfu & Atkinson, 2020).

Health impacts are well understood and characterised, including in Aotearoa where studies have shown NO_2 impacts to include premature mortality, hospital admissions for asthma in children with effect estimates higher than many other reported studies (Hales et al., 2021). Effect estimates have been used to quantify impacts at a population level (Kuschel et al., 2022). Current estimates of the health impacts in Aotearoa are shown in Table 1

Table 1. Health impacts of NO_2 in Aotearoa (from Kuschel et al 2022)

Health impacts in 2016 due to anthropogenic NO_2 air pollution (in case numbers)	
Premature deaths (all adults)	**2,025**
Cardiovascular hospitalisations (all ages)	**1,987**
Respiratory hospitalisations (all ages)	**6,544**
Asthma prevalence (0-18 yrs)	**13,229**

Spatial extent of monitoring is limited to urban areas. Spatial distribution estimates of NO_2 emissions from motor vehicles by NIWA, based on measurements by NIWA and others show that concentrations are highest close to roads and diminish rapidly away from roads (NIWA, 2019). Other peak locations are likely to include Ports and industrial areas.

Cost estimates in Aotearoa are beginning to emerge since the publication of the latest HAPiNZ report (Kuschel et al 2022). The annual health cost of NO_2 is estimated to be NZ$9.4 billion (Table 2)

Table 2. Annual costs of NO_2 in Aotearoa (from Kuschel et al 2022)

Annual costs in 2016 due to anthropogenic NO_2 air pollution (in millions of NZ$)	
Premature deaths (all adults)	$9,168
Cardiovascular hospitalisations (all ages)	$73
Respiratory hospitalisations (all ages)	$208
Asthma prevalence (0-18 yrs)	$2
Total	**$9415**

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

Near roadsides and in residential areas of New Zealand, concentrations of NO_2 have been improving over the past 15 years in a non-linear fashion (Bluett et al., 2016; Ministry for Environment, 2021). Bluett et al., (2016) concludes that emission controls for vehicles have resulted in reduced NO_2 emissions from petrol vehicles but not light duty vehicles (LDV) using diesel. A step change reduction in NO_2 from shipping is likely to have occurred from January 2020 as a result of MARPOL Annex VI.

Existing central government policies are likely to result in significant reductions in NO_2 from motor vehicles over the next 10-30 years. The impact of changes in vehicle fleet composition, including increased electric vehicle use, on NO_2 emissions are integrated into the national vehicle fleet emission model[1] (VEPM 7.0) (New Zealand Transport Agency, 2019). This model indicates an 88% reduction in NO_2 emissions from motor vehicles over the next 26 years (VEPM with default model parameters, https://www.vepm.co.nz/). Waka Kotahi also aim to reduce total kilometres travelled by the light fleet by 20% by 2035 through improved urban form and providing better travel options (Waka Kotahi, 2024) and this will result in further improvements in NO_2. Variations in fleet characteristics that provide an upward pressure on NO_2 emissions include increased proportion of diesel vehicles in the LDV fleet and increased milage of the petrol LDV fleet (more older and gross emitting vehicles) (Bluett et al., 2016).

The impact of National Environmental Standards for Greenhouse Gases from Industrial Process Heat on NO_2 is uncertain as alternative sources adopted and consequent NO_2 emission quantities is not known.

[1] Whilst VEPM includes assumptions around integration of Euro 6/VI vehicles these may not reflect the impact of existing policy and thus improvements may be underestimated.

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

Regulatory monitoring for compliance with the National Environmental Standards for Air Quality (NESAQ) is carried out by regional councils and unitary authorities. The methods used comply with the standard specified in the NESAQ[1].The number of locations varies but it is currently monitored for regulatory purposes in nine sites around the country, in Auckland, Greater Wellington and Christchurch (MfE 2021).

NO_2 is also routinely measured using Palmes type diffusion tubes (DEFRA 2008). The largest of these is run by Waka Kotahi, who maintains a network of 120 NO_2 tubes across the country (Longley and Kachhara 2021). Waka Kotahi has specific objectives in its state highway environmental plan for improving air quality, including understanding the contribution vehicle traffic to air quality, ensuring new state highway projects do not directly cause national environmental standards for ambient air quality to be exceeded and contributing to reducing emissions where the state highway network is a significant source of exceedances of national ambient air quality standards. The NO_2 tubes assist by identifying NO_2 hotspots and providing information on trends in emissions from motor vehicles (Longley & Kachhara, 2021).

NO_2 tubes are less accurate than regulatory methods and have very limited temporal resolution giving at a minimum monthly average concentrations (Longley & Kachhara, 2021). The advantages are that they are much cheaper and easier to use, and this allows much greater spatial coverage, giving a better depiction of population exposure (see e.g. Ma et al 2019, Ma et al 2022).

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

Cost is the main implementation issue although in some areas finding a location that meets siting requirements[2] can be problematic.

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

A new NO_2 instrument that meets the NESAQ monitoring requirements typically costs around $30,000 - $50,000 to purchase and several thousand dollars per year to run. Alternatives such as Palmes tubes are available but have limited adoption owing to the low temporal resolution and non-compliance with the NESAQ. These typically cost less than $2000 per year per site with price varying with bulk runs.

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

We are not aware of any NO_2 monitoring carried out by iwi/hapū/rūnanga.

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[1] Australian Standard AS 3580.5.1:1993, Methods for sampling and analysis of ambient air—Determination of oxides of nitrogen—Chemiluminescence method

[2] Australian/New Zealand Standard 3580.1.1:2016; Methods for sampling and analysis of ambient air - Guide to siting air monitoring equipment

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

Nitrogen oxides (NOx) is a precursor to ozone formation. Whilst there is a relationship between these contaminants it is complex and impacted by meteorology (Nguyen et al., 2022). NO_2 can be correlated with other combustion related air contaminants particularly those predominantly arising from motor vehicles in urban areas (e.g., CO). However, the ratios of contaminants discharged varies with combustion fuel type confounding ambient air correlations when multiple combustion sources contribute (e.g., domestic heating also contributes to CO in urban areas of Aotearoa).

Part B—Current state and allocation options

B1. What is the current state of the attribute?

The current state is reasonably well understood from a combination of regulatory monitoring and passive sampler networks. The Ministry for the Environment issues regular State of the Environment reports with the latest air quality one being in 2021. Monitoring of NO_2 in the larger urban areas of Auckland, Wellington Region and Christchurch show compliance with the NESAQ but elevated concentrations for other exposure durations relative to some other guideline values (Ministry for Environment, 2021).

NIWA has used a national network of passive samplers based on the Waka Kotahi-NZTA network extended by campaign-based sampling in previously unmonitored locations to create a national motor vehicle NO_2 exposure model (Ma et al 2019, Ma et al 2022).

Some air quality data for regulatory monitoring of NO_2 are nationally available on the LAWA website (https://www.lawa.org.nz/).

Regional councils and unitary authorities are responsible for regulatory monitoring. Some publish data on their websites (e.g., https://knowledgeauckland.org.nz/) but for others, although data are usually available, they can be hard to find.

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

Natural concentrations from e.g., lightening are very low, thus natural reference conditions would be expected to be extremely low or zero. Tracking of anthropogenic NO_2 is the only factor important for urban health purposes.

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

The NESAQ contains a one hour average threshold concentration of 200ug/m^3 but no annual standard. The NZ air quality guidelines from 2002 also have a daily guideline of 100 µg/m^3 and an annual guideline value of 40ug/m^3. The WHO (2021) includes a 24-hour average guideline of 25 µg/m^3 and an annual average guideline of 10 µg/m^3. Many urban areas in Aotearoa could exceed this guideline. The ambient air quality guidelines for New Zealand include a critical value for NO_2 for the protection of ecosystems of 30 µg/m^3.

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

Epidemiological studies in New Zealand found no evidence of a threshold concentration value at which there is no effect from NO_2 (Hales et al., 2021). The WHO (2021) have set guidelines based on the lowest concentrations at which effects are robustly demonstrated from studies meeting specified criteria (through meta-analysis). These do not represent no effects thresholds however as the prevalence of lower concentrations without effect is not tested.

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

Some health studies for air contaminants report lag effects between exposures and health endpoints. These do not impact on state or trend assessments. Changes in climate over time may have an impact on concentrations of air contaminants. For example, a predicted decrease in frost days, which are particularly conducive to the build-up of air contaminants in many areas, may result in improvements in concentrations.

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

Good air is considered a taonga and Māori are often disproportionately affected by poor air quality (Telfar Barnard & Zhang, 2018). However, whilst air as a taonga is often mentioned in iwi environmental strategies, we are not aware of any iwi explicitly planning to reduce air pollution exposure of their members.

Part C—Management levers and context

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

MfE regularly publishes a SOE report on air quality, most recently in 2021. The pressures and drivers are reasonably well understood as are (at a general or high level) the response and state. The relationship between emissions and concentrations is spatially and temporally dependent primarily influenced by meteorology and topography. These variables all impact on exposure and subsequent health effects are also influenced by underlying susceptibility (Figure 2).

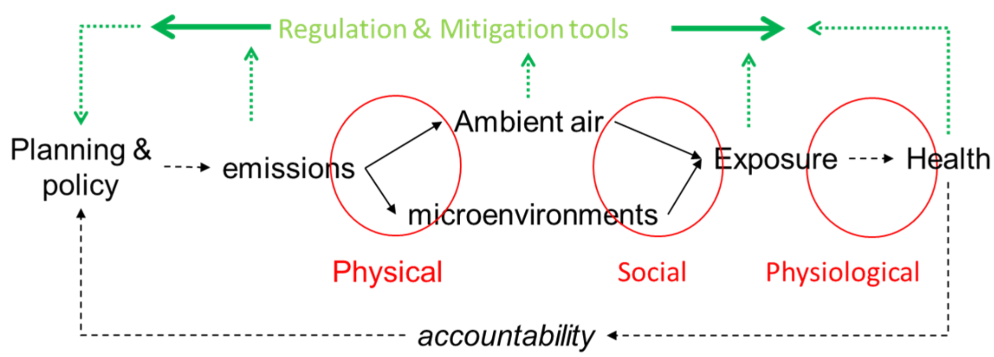

Figure 2. Links in the chain from pollutant emissions to health effects

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

C2-(i). Local government driven

National Environmental Standards for Air Quality include a limit value for NO_2. Whilst these standards assist Regional Councils in managing concentrations of contaminants, they have limited mechanisms available for managing NO_2 emissions from motor vehicles.

C2-(ii). Central government driven

The national introduction of emission standards for motor vehicles has resulted in a reduction in exhaust NO_2 emissions. The effects of this in decreasing ambient NO_2 has been demonstrated over time (Bluett et al., 2016,Ministry for Environment, 2021).

C2-(iii). Iwi/hapū driven

We are not aware of any iwi or hapū activity.

C2-(iv). NGO, community driven

Most community air quality projects we are aware of have concentrated on particle concentrations, usually from wood burning. NIWA has conducted numerous community collaborations on particles, but we are not aware of any involving NO_2. There have been school projects measuring $NO_{2,}$ but it appears not to be a community concern compared to particles.

C2-(v). Internationally driven

The global driver and lead of air quality guidelines is the WHO. Aotearoa's NESAQ were largely based on WHO guidelines published in the 1990s (and reaffirmed in 2005). In 2021 the WHO substantially revised its guidelines, setting much lower values for many pollutants including NO_2 (see above). MfE, along with most environmental agencies around the world is considering its response to the new guidelines.

Part D—Impact analysis

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

Not managing concentrations of NO_2 in the air would result in increased premature mortality rates for all cause, cardiovascular, and respiratory mortality hospitalizations for respiratory conditions as well as an increase in exacerbation of asthma. The health impacts of long term exposures to NO_2 in Aotearoa have been quantified at 9.7% increase in premature all-cause mortality (adults 30+ years) per 10 $\mu g/m^3$ increase in annual average NO_2 concentrations (Hales et al., 2021). Māori will be disproportionately impacted. The contribution of NO_2 to environmental impacts such as Auckland's Brown Haze will also continue.

Climate impacts: Whilst NO_2 does not have direct climate impacts, it is co-emitted with many pollutants that do. Existing policies targeting motor vehicle emissions of CO_2, including the 20% reduction in VKT by 2035 (Waka Kotahi, 2024) will result in improvements in NO_2 as indicated above.

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

The cost of NO_2 air pollution is estimated in HAPINZ at NZ$9.4 billion per year in deaths, illness and lost productivity (Kuschel et al., 2022) (See Tables 1 and 2 above).

The health impacts of NO_2 exposure would affect those in living near to roadways and in urban areas with higher emission densities and where meteorology and topography are more conducive to elevated concentrations.

Māori are disproportionately affected by respiratory and cardiovascular disease (Mason et al., 2019; Telfar Barnard & Zhang, 2018) and will be more susceptible to the health impacts of NO_2 exposure.

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

There is the potential that NO_2 concentrations may decrease slightly in some areas as a result of climate change. This may occur if climate change results in fewer ground frosts as the associated low wind speeds and decreased vertical dispersion increase the potential for degraded air.

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5.3 Sulphur dioxide in air

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State of knowledge of "Sulphur dioxide (SO_2) in air" attribute: Excellent / well established – comprehensive analysis/syntheses; multiple studies agree.

Part A—Attribute and method

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

Sulphur dioxide (SO_2) is a water soluble, colourless, odourous gas that has the potential for adverse health and ecological impacts. It is derived from the combustion of sulphur containing fossil fuels and industry specific processes (e.g., fertiliser manufacturing, oil refinery and aluminium manufacturing). It also occurs naturally from geothermal activity and volcanoes. The main anthropogenic sources of sulphur dioxide in New Zealand are industrial processes including combustion and non-combustion activities and shipping (Metcalfe & Sridhar, 2018). However, in most urban areas where people reside domestic home heating and diesel vehicles are the main source of sulphur dioxide.

Sulphur dioxide oxidises in the air and combines with other chemicals to form particulate sulphate. Like other forms of fine particulate, sulphate can contribute to visibility degradation/ haze. Oxidation of sulphur dioxide in the air also creates sulphuric acid and can cause acid rain and associated ecological impacts if in significant concentrations. This is not generally considered a problem in Aotearoa.

The health impacts of SO_2 exposure are well known and understood. SO_2 is a strong respiratory irritant which acts directly on the upper airways. It causes coughing, mucus secretion and aggravates conditions such as asthma and chronic bronchitis. It can irritate the eyes. Common symptoms include wheezing, shortness of breath and chest tightness, especially during exercise or physical activity. Effects arise from short term exposures and can result in hospital admissions and premature mortality.

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

The evidence on the health impacts of exposure to SO_2 is strong and there are hundreds of studies internationally used to characterise the relationships between premature mortality and respiratory hospital admissions and SO_2 exposure with a high certainty of evidence (Orellano et al., 2020). The certainty of evidence for hourly and 24-hour SO_2 exposures in exacerbation of asthma is moderate (Zheng et al., 2021).

The weight of evidence on spatial variability and exposures is relatively strong owing to solid source characterisation and monitoring networks. Typical population exposures can be characterised by monitoring carried out in a number of residential locations. Concentrations in these areas show consistency and coherence and are below guideline levels. Exposures near to Ports or industrial areas are more variable but have been well characterised through monitoring in high-risk areas. Concentrations in some high-risk areas exceed guideline values on occasion.

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

In residential areas of New Zealand concentrations of SO_2 have been gradually improving over the past ten years (Ministry for Environment, 2021). Areas near to shipping ports will have experienced a significant decrease in SO2 concentrations from January 2020 as a result of the impact of MARPOL Annex VI, which requires the use of low sulphur fuels (or alternative SO2 reducing technology) on ocean going vessels, as demonstrated for the Port of Tauranga by Iremonger, (2023). The National Environmental Standards for Greenhouse Gases includes restrictions for industrial coal combustion which will contribute to further improvements in SO_2 concentrations in areas where industrial coal burning is prevalent.

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

Monitoring of SO_2 has been carried out in residential and industrial locations by Regional Councils. The purpose of the monitoring is typically to assess compliance with national standards and guidelines. Generally, the monitoring method used complies with the standard specified in the NESAQ[1] although in some instances, for example where the objective is to determine spatial variability in concentrations, low-cost passive samplers have been used.

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

Cost is the main implementation issue although in some areas finding a location that meets siting requirements[2] can be problematic.

[1] Australian Standard AS 3580.4.1:2008, Methods for sampling and analysis of ambient air–Determination of sulphur dioxide–Direct-reading instrumental method.

[2] Australian/New Zealand Standard 3580.1.1:2016; Methods for sampling and analysis of ambient air - Guide to siting air monitoring equipment

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

A new SO_2 instrument that meets the NESAQ monitoring requirements can be several tens of thousands of dollars and cost thousands to tens of thousands of dollars per year to run. Low-cost passive samplers have been used for assessing spatial variability in SO_2 concentrations in Auckland (Talbot & Reid, 2017). This method is not suitable for regulatory monitoring owing to the long-term average data collected and the lower level of reliability.

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

We are not aware of any SO_2 monitoring being undertaken by iwi/hapū/rūnanga. The Bay of Plenty Regional Council carry out SO_2 monitoring using a NESAQ compliant method at the Whareroa Marae in Mount Maunganui.

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

SO_2 can be correlated with other air contaminants particularly other by-products of combustion where combustion is a predominant source**.** Airsheds tend to contain a variety of contributing sources each with differing ratios of contaminants which causes variability in the relationships. Additionally, airsheds may contain non combustion sources of SO_2. Thus, there is no reliable proxy for SO_2.

<u>Part B—Current state and allocation options</u>

B1. What is the current state of the attribute?

The current state of SO_2 in New Zealand is reasonably well understood at the national scale being monitored by Regional Councils and reported on by the Ministry for Environment, (e.g., Our Air - 2021)and Regional Councils (e.g., Boamponsem, 2023). In residential areas concentrations are generally well within NES and health-based guidelines including WHO (2021). There is spatial variability in SO_2 concentrations however, and the country has port and industrial hotspots where concentrations have exceeded health-based guidelines including at neighbouring residential locations (e.g., Whareroa Marae in Mount Maunganui).

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

We are not aware of any natural reference states for New Zealand for this attribute.

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

National Environmental Standards for SO_2 include an hourly standard of 570 µg/m^3 that may not be exceeded and an hourly standard of 350 µg/m^3 that may be exceeded nine times in a 12 month period.

The World Health Organization, (2021) include a 24-hour average guideline of 40 µg/m^3 and the weight of evidence for this guideline is high. This value has been used in the Ministry for the Environments 2021 "Our Air 2021" publication for comparing 24-hour average SO_2 concentrations. There is no annual average guideline for SO_2 as impacts are associated with short term exposures. The WHO (2021 and 2005) air quality guidelines includes a 10-minute average SO_2 guideline of 500 µg/m^3.

The ambient air quality guidelines include critical levels of SO_2 for the protection of vegetation including agricultural crops (30 µg/m^3 annual and winter average) forest and natural vegetation (20 µg/m^3 annual and winter average) and lichen (10 µg/m^3 annual average).

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

We are not aware of any thresholds or tipping points with respect to ecological integrity or human health.

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

Some health studies report lag effects between exposures and health endpoints. These do not impact on state or trend assessments. Changes in climate over time may have an impact on concentrations of air contaminants. For example, a predicted decrease in frost days, which are particularly conducive to the build-up of air contaminants in many areas, may result in improvements in concentrations.

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

A high standard of air quality is an outcome sought by iwi/hapū/rūnanga. In addition to discussing this attribute directly with iwi/hapū/rūnanga, in regard to air quality, there is likely to be tikanga and mātauranga Māori relevant to informing bands, allocation options, minimally disturbed conditions and/or unacceptable degradation in treaty settlements, cultural impact assessments, environment court submissions, iwi environmental management and climate change plans etc.

Part C—Management levers and context

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

MfE regularly publishes a SOE report on air quality, most recently in 2021, which includes SO_2. The pressures and drives are reasonably well understood as are (at a general or high level) the response and state. The relationship between emissions and concentrations is spatially and temporally dependent influenced primarily by meteorology and topography. These variables all impact on exposure and subsequent health effects are also influenced by underlying susceptibility (Figure 1).

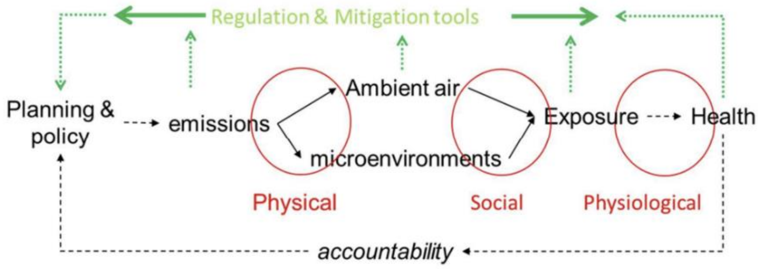

Figure 1. links in the chain from pollutant emissions to health effects

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

C2-(i). Local government driven

C2-(ii). Central government driven

National Engine Fuel Specifications Regulations have progressively reduced the sulphur content of diesel from 3000 parts per million in 2002 to 10 parts per million in 2009 and petrol to 10 parts per million in 2018 (MBIE, 2024). Air quality monitoring of SO_2 in residential areas supports an improvement in concentrations over this period. National Environmental Standards have also been implemented for SO_2 and these assist Regional Councils in managing concentrations of contaminants.

C2-(iii). Iwi/hapū driven

Iwi/hapū planning documents such as Environmental Management Plans and Climate Change Strategies/Plans may contain policies/objectives/methods seeking to influence air quality outcomes for the benefit of current and future generations. We are not aware of other interventions/mechanisms being used by iwi/hapū/rūnanga to directly affect this attribute.

C2-(iv). NGO, community driven

C2-(v). Internationally driven

Shipping SO_2 emissions have decreased significantly as a result of the implementation of Marpol Annex VI, an internationally driven legislation requiring the use of lower sulphur fuels in ocean going vessels. This is evidenced in air quality monitoring data around the Port of Tauranga which shows a significant decrease in SO_2 concentrations coinciding with Marpol Annex VI (Iremonger, 2023)

Part D—Impact analysis

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

Not managing concentrations of SO_2 in the air would result in increased premature mortality rates for all cause and respiratory mortality and increased hospitalizations for respiratory conditions as well as an increase in exacerbation of asthma, chronic bronchitis and respiratory conditions. The health impacts of short term exposures to SO_2 have been quantified at 0.59% increase in premature all-cause mortality and 0.67% increase in respiratory mortality per 10 μg/m^3 increase in 24-hour average SO_2 concentrations (Orellano et al., 2020).

Māori are disproportionately impacted as they share (with Pacific people) the highest respiratory health burden in New Zealand (Telfar Barnard & Zhang, 2018).

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

The cost of air quality related premature mortality in New Zealand has been estimated in the HAPINZ model as around $4,527,300 per life lost ($263,843 per year of life lost) and $31,748 per respiratory hospitalisation (Kuschel et al., 2022). The economic impacts would be spread nationally, e.g., across the health sector and with local hotspots in high-risk areas (e.g., increased lost work days owing to respiratory illness). Māori and Pacific people will be disproportionately impacted.

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

There is the potential that SO_2 concentrations may decrease slightly in some areas as a result of climate change. This may occur if climate change results in fewer ground frosts as the associated low wind speeds and decreased vertical dispersion increase the potential for degraded air.

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5.4 Carbon monoxide in air

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State of knowledge of "Carbon monoxide (CO) in air" attribute: Excellent / well established – comprehensive analysis/syntheses; multiple studies agree

Section A—Attribute and method

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

Carbon monoxide (CO) is a flammable, colourless, odourless and tasteless gas that has the potential for adverse health impacts. It is derived primarily from the combustion of fuels although some non-combustion industrial processes also contribute to carbon monoxide emissions. The main sources of carbon monoxide in New Zealand are domestic heating related combustion activities, motor vehicles and outdoor burning activities (Metcalfe & Sridhar, 2018). Carbon monoxide is also produced by natural processes (e.g., volcanoes, fires and metabolism of organisms).

Health impacts of carbon monoxide exposure occur as inhaled carbon monoxide reaches the blood stream and attaches to haemoglobin that would otherwise carry oxygen around the body. This reduces the amount of oxygen available to the body (Raub & Benignus, 2002). For people with cardiovascular disease, short-term CO exposure can further reduce their body's already compromised ability to respond to the increased oxygen demands of exercise, exertion, or stress. Carbon monoxide can cross the placenta to gain access to the foetal circulation and can impact on the developing brain (Levy, 2015). Exposure to low levels in otherwise healthy people can causes dizziness, weakness, nausea, confusion and disorientation (Graber et al., 2007). As the carboxyhaemoglobin level increases health impacts can include coma, collapse, loss of consciousness and death (Varma et al., 2009).

Carbon monoxide is an indirect greenhouse gas as it reacts with other compounds which can result in an increase in greenhouse gases (Sobieraj et al., 2022).

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

The evidence on the health impacts of exposure to CO for short term exposures (15 minute, hourly and eight-hour averages) is strong and is based on a range of methodologies. The 2021 World health organisation guideline review added a 24-hour average guideline to the existing 2005 WHO guideline values for 15 minute, hourly and eight hour average guidelines (World Health Organization, 2021). The 24-hour guideline is considered of moderate certainty by WHO and is based on an evaluation by Lee et al., (2020) which found relationships between exposure to CO and myocardial infarction.

The weight of evidence on spatial variability and exposures is relatively strong owing to solid source characterisation and monitoring networks. Typical population exposures can be characterised by monitoring carried out in a number of residential locations results for which show consistency and coherence. Exposures near to roadsides can be more variable as concentrations decrease steeply with distance from the road and depend on surrounding topography and meteorology.

Increased indoor exposure to CO also occurs as a result of use of gas heating and cooking, open fires and wood burners, smoking and internal access garages (World Health Organization, 2010). Monitoring of carbon monoxide inside dwellings in Aotearoa found concentrations were low and within guideline values (BRANZ, 2019).

A3. What has been the pace and trajectory of change in this attribute, and what do we expect in the future 10 - 30 years under the status quo? Are impacts reversible or irreversible (within a generation)? [Provide evidence or examples of how fast the attribute has changed, whether the problem historical only or contemporary/ongoing, and what the prospects and pace of recovery might look like (reversable, yes or no)].

Near roadsides and in residential areas of New Zealand concentrations of CO have been gradually improving over the past 20 years (Bluett et al., 2016; Ministry for Environment, 2021). Emission standards for motor vehicles required the installation of catalytic converters from 2000 and these reduce exhaust CO emissions (Bluett et al., 2016). Additionally, the National Environmental Standards for wood burners required that only burners meeting a specified emission and energy efficiency criteria could be installed on properties less than 2 hectares in area from 2005. The latter is likely to have ongoing impact on CO concentrations as older less efficient burners are replaced with more efficient and lower emission alternatives.

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

Monitoring of CO has been carried out in residential and roadside locations by Regional Councils. The purpose of the monitoring is typically to assess compliance with national standards and guidelines. The methods used comply with the standard specified in the NESAQ[1]. Monitoring of CO is not a high priority for most Councils owing to existing low concentrations and good understanding of contributing sources.

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[1] Australian Standard AS 3580.7.1:1992, Methods for sampling and analysis of ambient air–Determination of carbon monoxide-Direct-reading instrumental method

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

Continuous instrumental ambient air quality monitoring has been performed in urban areas for several decades. There do not appear to be major impediments to monitoring (i.e., pole-mounted sensors, or continuous monitoring sensors co-located with meteorological equipment, with analysing equipment in air-conditioned sheds).

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

A new CO instrument that meets the NESAQ monitoring requirements typically costs around $25,000 to purchase and several thousand dollars per year to run.

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

We are not aware of any CO monitoring being undertaken by iwi/hapū/rūnanga.

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

CO can be correlated with other air contaminants particularly other by-products of combustion where combustion is a predominant source**.** Airsheds tend to contain a variety of contributing sources each with differing ratios of contaminants. This introduces variability in the relationships. For example, when traffic is the main contributor to CO concentrations of NOx would be higher than when residential combustion is the predominant source.

Part B—Current state and allocation options

B1. What is the current state of the attribute?

The current state of CO in New Zealand is reasonably well understood with concentrations being monitored by a small number of Regional Councils and reported on by the Ministry for the Environment, Regional Councils and Waka Kotahi. In residential areas and at roadside monitoring sites concentrations are well within health-based guidelines and standards (Ministry for Environment, 2021).

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

We are not aware of any natural reference states for New Zealand.

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

National Environmental Standards for CO include an eight-hour average standard of 10 mg/m^3 that may be exceeded once in a 12-month period and an hourly standard of 30 mg/m^3 that may not be exceeded. The weight of evidence for these standards is strong.

The World Health Organization, (2021) include a 24-hour average guideline of 4 mg/m^3 and the weight of evidence for this guideline is moderate. This value has been used in the Ministry for the Environments 2021 "Our Air 2021" publication for comparing 24-hour average CO concentrations. The WHO (2005 and 2021) guideline values include a 15-minute guideline for CO of 100 µg/m^3, an hourly average guideline of 35 µg/m^3 and an eight hour average guideline of 10 µg/m^3. There is no annual average guideline for CO as impacts are associated with short term exposures.

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

The relationship of carbon monoxide exposure and the COHb concentration in blood can be modelled using the differential Coburn-Forster-Kane equation (Coburn et al., 1965). Epidemiological studies of the impacts on cardiovascular disease mortality are not indicative of thresholds below which effects do not occur (Liu et al., 2018).

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

Some health studies for air contaminants report lag effects between exposures and health endpoints. For example for associations between cardiovascular disease, coronary heart disease, and stroke Liu et al., (2018) report a present day and one day lag with CO concentrations. These do not impact on state or trend assessments. Changes in climate over time may have an impact on concentrations of air contaminants. For example, a predicted decrease in frost days, which are particularly conducive to the build-up of air contaminants in many areas, may result in improvements in concentrations.

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

A high standard of air quality is an outcome sought by iwi/hapū/rūnanga. In addition to discussing this attribute directly with iwi/hapū/rūnanga, regarding air quality, there is likely to be tikanga and mātauranga Māori relevant to informing bands, allocation options, minimally disturbed conditions and/or unacceptable degradation in treaty settlements, cultural impact assessments, environment court submissions, iwi environmental management plans, climate change strategies, etc.

Part C—Management levers and context.

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

MfE regularly publishes a SOE report on air quality, most recently in 2021, which includes CO. The pressures and drives are reasonably well understood as are (at a general or high level) the response and state. There is a non-linear chain from emissions to concentrations to exposure and subsequent health effects that incorporates a number of variables (Figure 1).

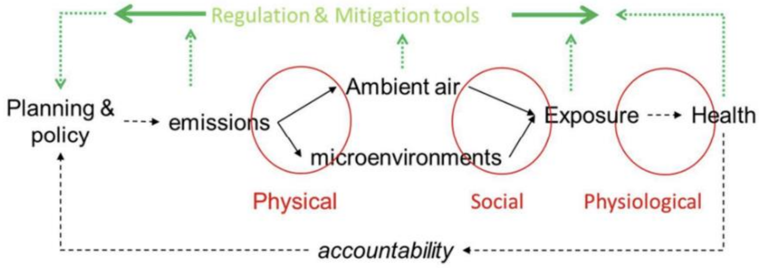

Figure 1. links in the chain from pollutant emissions to health effects

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

C2-(i). Local government driven

C2-(ii). Central government driven

The National Environmental Standards design criteria for wood burners was established for the purpose of reducing concentrations of particulate in urban areas. However, the emission limit and efficiency criteria are likely to also result in improvements in CO concentrations. The impact of this legislation is ongoing in the near future as households replace older more polluting and less efficient burners with compliant models over time.

The national introduction of emission standards for motor vehicles has resulted in a reduction in exhaust CO emissions. The effects of this in decreasing ambient CO has been demonstrated over time (Bluett et al., 2016).

National Environmental Standards for air contaminants include two limit values for CO. These standards assist Regional Councils in managing concentrations of contaminants.

C2-(iii). Iwi/hapū driven

Iwi/hapū planning documents such as Environmental Management Plans and Climate Change Strategies/Plans may contain policies/objectives/methods seeking to influence air quality outcomes for the benefit of current and future generations. We are not aware of other interventions/mechanisms being used by iwi/hapū/rūnanga to directly affect this attribute.

C2-(iv). NGO, community driven

C2-(v). Internationally driven

Part D—Impact analysis

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

If the air were to become degraded and CO concentrations increased to level in excess of standards and guidelines then not managing concentrations of CO in the air could result in increased health impacts including dizziness, weakness, nausea, confusion and disorientation. If extreme short-term concentrations were experienced health impacts could include loss of consciousness and death.

If concentrations became elevated for a period of days, then there would be increased risk of myocardial infarction and associated hospitalisation. Māori are disproportionately affected by cardiovascular disease (Mason et al., 2019) and will be more susceptible to the health impacts of daily CO exposure.

The cost of air quality related premature mortality in New Zealand has been estimated in the HAPINZ model as around $4,527,300 per life lost ($263,843 per year of life lost) and cardiovascular hospitalisations in New Zealand has been estimated in the HAPINZ model as around $36,666 per admission (Kuschel et al., 2022).

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

The health impacts of CO exposure would affect those in living near to roadways and in urban areas with higher emission densities and where meteorology and topography are more conducive to elevated concentrations.

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

There is the potential that CO concentrations may decrease slightly in some areas as a result of climate change. This may occur if climate change results in fewer ground frosts as the associated low wind speeds and decreased vertical dispersion increase the potential for degraded air.

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5.5 Ozone in air

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State of knowledge of "Ozone (O_3) in air" attribute: – Good / established but incomplete – general agreement, but limited data/studies

Section A—Attribute and method

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

Ozone (O_3) is a pale blue, reactive gas with a pungent odour. It is produced naturally in the upper atmospheres where helps protect earth from harmful ultraviolet rays and is referred to as the ozone layer. In the lower atmosphere it occurs as a result of human activities forming through chemical reactions of contaminants such as nitrogen dioxide with volatile organic compounds in the presence of sunlight. Ozone may also be formed by commonly used equipment such as photocopiers, laser printers, and other electrical devices and thus also has the potential to be an indoor air pollutant.

Adverse health impacts of ozone exposure include chest pain, cough, throat irritation and congestion. It exacerbates existing respiratory conditions such as asthma, bronchitis and emphysema and can cause respiratory inflammation and reduce lung function (USEPA, 2024). Short term exposures are also associated with all cause premature mortality (Orellano et al., 2020). Additionally there is potential for long-term exposure impacts including premature all cause and respiratory mortality (Huangfu & Atkinson, 2020). The impacts of ozone on sensitive plants include reduced photosynthesis and increased risk of disease, damage from insects, harmed from severe weather and effects of other pollutants (USEPA, 2023). These impacts can then have ongoing ecological effects such as changes to the specific assortment of plants present, changes to habitat quality and changes to water and nutrient cycles (USEPA, 2023).

A precursor to ozone formation is nitrogen dioxide (NO_2). The main anthropogenic sources of nitrogen dioxide (and thus ozone formation) in New Zealand is motor vehicles, industry (Metcalfe & Sridhar, 2018), and shipping (not fully included in Metcalfe & Sridhar, (2018)). Domestic home heating also contributes to volatile organic compounds (VOC) another a precursor to ozone formation. Farm animals are a source of methane, a VOC that can contribute to ozone formation.

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

The evidence on short term health impacts of exposure to ozone is strong with thousands of studies carried out internationally on the relationship between ozone concentrations and emergency department visits and hospital admissions due to asthma. The certainty of evidence for 8 hourly or 24-hour ozone exposure in exacerbation of asthma is high and for hourly average exposure the certainty of evidence is moderate (Zheng et al., 2021). The certainty of impact for long term exposures including all-cause and respiratory premature mortality endpoint is rated low to moderate (Huangfu & Atkinson, 2020). The evidence of impact on ecosystems appears to be strong (Grulke & Heath, 2020).

The weight of evidence on spatial variability and exposures to ozone is weak owing to limited monitoring networks and complex chemical formation mechanisms. Ozone formation is temperature dependent with highest concentrations occurring during the warmest months of the year. Daily maximum of 8-hour mean concentrations are typically reported rather than 24-hour averages because of the strong diurnal variation in ozone concentration. Thus, concentrations should be highest in temperate areas downwind of high precursor emission areas (particularly nitrogen oxides NOx) such as Auckland.

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

The change in ozone concentrations in New Zealand has been assessed at one site only and was found to be indeterminate (Ministry for Environment, 2021). The formation of ozone is a complicated photochemical reaction that occurs in the lower atmosphere. Changes in ozone concentrations are a combined process of other effects such as anthropogenic emissions, topographic characteristics, and meteorological influences. Nitrogen oxides (NOx) is one of the significant ozone precursors. However, reducing NOx will not guarantee a downward ozone trend. Specific weather phenomena (e.g., anticyclones and sea–land breezes) can enhance specific meteorological parameters that govern the transport and diffusion of ozone and its precursors (Nguyen et al., 2022)

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

Monitoring of ozone is carried out in Auckland and Wellington using a method that complies with the specifications of the NESAQ[1]. Results are compared with hourly, eight hourly and daily guideline values for health purposes.

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

Cost is the main implementation issue although in some areas finding a location that meets siting requirements (AS/NZS 3580.1.1:2016) can be problematic.

1 Australian Standard AS 3580.6.1:1990, Methods for sampling and analysis of ambient air–Determination of ozone-Direct-reading instrumental method

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

A new ozone instrument that meets the NESAQ monitoring requirements can be several tens of thousands of dollars and cost thousands to tens of thousands of dollars per year to run.

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

We are not aware of any ozone monitoring being undertaken by iwi/hapū/rūnanga

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

Nitrogen oxides (NOx) are a significant ozone precursor, and VOCs are also ozone precursors. The relationships are complex and impacted by meteorology. Reducing NOx will not guarantee a downward ozone trend (Nguyen et al., 2022).

Part B—Current state and allocation options

B1. What is the current state of the attribute?

We have a moderate understanding of ozone concentrations in New Zealand. Air quality monitoring for ozone is carried out by two regional councils (Auckland, where precursor emissions are highest, and Wellington). Neither location exceeds the NESAQ for ozone (one hour average of 150 µg/m^3). Ozone concentrations at both sites are within the ambient air quality guideline eight hour value of 100 µg/m^3 and the WHO (2021) value of 60 µg/m^3, although Ministry for Environment, (2021) reports that the maximum eight hour average concentrations at Patumahoe (Auckland) was only just compliant with the latter guideline in 2019 (at 59.3 µg/m^3).

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

Photochemistry involving methane accounts for much of the rise in ozone over the oceans and remote land areas (World Health Organization, 2021). Methane is a VOC that is much less reactive than the other VOCs but is present at much higher concentrations, having risen in concentration over the past 100 years owing to its increasing use as fuel, and is released from rice fields and farm animals (World Health Organization, 2021).

There are no known natural references states suitable for management or allocation options.

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

The NESAQ 1-hour average standard for ozone is 150 µg/m^3 with no allowable exceedances.

The ambient air quality guideline for an eight-hour average exposure is 100 µg/m^3 (Ministry for the Environment, 2002) and the WHO (2021) guideline for an eight hour average exposure is 60 µg/m^3.

The ambient air quality guideline (Ministry for the Environment, 2002) includes critical levels for protection of ecosystems. These are based on cumulative exposure over a concentration threshold of 85.6 µg/m^3 and are calculated for daylight hours only. The critical levels vary from 428 µg/m3 -h, over 5 days, to 21,400 µg/m^3 -h over 6 months.

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

There is a threshold for ecological effects of 85.6 µg/m^3 (40ppb) for an hourly average. A threshold for health impacts was not been able to be determined for the WHO (2021) guideline review (Huangfu & Atkinson, 2020; Orellano et al., 2020).

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

Some health studies for air contaminants report lag effects between exposures and health endpoints. These do not impact on state or trend assessments. Changes in climate over time may have an impact on concentrations of air contaminants. For example, a predicted decrease in frost days, which are particularly conducive to the build-up of air contaminants in many areas, may result in improvements in concentrations.

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

A high standard of air quality is an outcome sought by iwi/hapū/rūnanga (see Section 3.2 for one example). In addition to discussing this attribute directly with iwi/hapū/rūnanga, in regard to air quality, there is likely to be tikanga and mātauranga Māori relevant to informing bands, allocation options, minimally disturbed conditions and/or unacceptable degradation in treaty settlements, cultural impact assessments, environment court submissions, iwi environmental management and climate change plans etc.

Part C—Management levers and context

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

The Ministry for the Environment regularly publishes a SOE report on air quality, most recently in 2021, which includes ozone (Ministry for Environment, 2021). The pressures and drives are generally reasonably well understood for most air contaminants but are more complex for ozone owing to the photochemical formation mechanisms. The state can be monitored. There is a non-linear chain from precursor emissions to concentrations, complicated by variability in temperature and meteorology, to exposure and subsequent health effects that incorporates a number of variables (Figure 1).

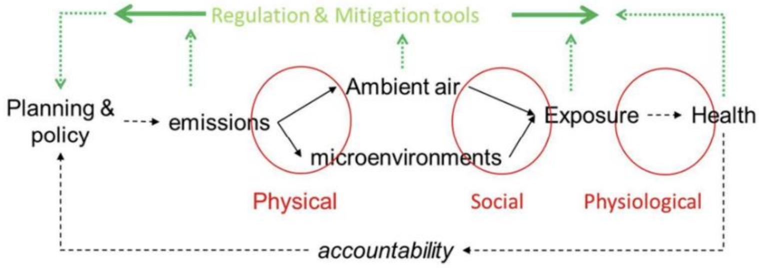

Figure 1. links in the chain from pollutant emissions to health effects

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

C2-(i). Local government driven

C2-(ii). Central government driven

There are no intervention mechanisms directly targeting ozone although emission standards for motor vehicles impact on emissions of NO_2, a precursor to ozone but also a significant contaminant. Monitoring of NO_2 across New Zealand shows a decrease in annual average concentrations from 2011 to 2020 at six out of seven air quality monitoring sites (Ministry for Environment, 2021).

C2-(iii). Iwi/hapū driven

Iwi/hapū planning documents such as Environmental Management Plans and Climate Change Strategies/Plans may contain policies/objectives/methods seeking to influence air quality outcomes for the benefit of current and future generations. We are not aware of other interventions/mechanisms being used by iwi/hapū/rūnanga to directly affect this attribute.

C2-(iv). NGO, community driven

C2-(v). Internationally driven

Part D—Impact analysis

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

Not managing concentrations of precursors to ozone formation may result in increased respiratory health impacts including increased hospitalizations for respiratory conditions and premature mortality. The health impacts of short term exposures to ozone have been quantified at 0.43% increase in premature mortality per 10 µg/m^3 increase in 24-hour average ozone concentrations (Orellano et al., 2020). There is also the potential for reduced yields in crops and damage to forests if ozone concentrations were to increase.

Māori are disproportionately impacted as they share (with Pacific people) the highest respiratory health burden in New Zealand (Telfar Barnard & Zhang, 2018).

The cost of air quality related premature mortality in New Zealand has been estimated in the HAPINZ model as around $4,527,300 per life lost($263,843 per year of life lost) and $31,748 for respiratory hospitalisations (Kuschel et al., 2022).

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

The health impacts, which would lead to economic impacts of ozone exposure would likely be felt in the more temperate areas of New Zealand in locations downwind of precursors sources areas e.g., Auckland.

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

There is the potential for increased ozone formation in New Zealand as the photochemical reaction potential increases with temperature. The relationships are complex, however, and overall impacts are difficult to predict as weather phenomena such as ground frosts, which are predicted to decrease with climate change, impact on the transport and diffusion of O_3 and its precursors.

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5.6 Black Carbon in air

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Preamble: Black Carbon (BC) is a component of particulate matter (PM_{10}) as is defined by WHO 2012 as follows: "*BC is an operationally defined term which describes carbon as measured by light absorption. As such, it is not the same as elemental carbon (EC), which is usually monitored with thermal optical methods*".

Elemental carbon (EC) in atmospheric PM derived from a variety of combustion sources contains the two forms "char-EC" (the original graphite-like structure of natural carbon partly preserved, brownish colour) and "soot-EC" (the original structure of natural carbon not preserved, black colour) with different chemical and physical properties and different optical light-absorbing properties.

A thermal optical reflectance method can be applied to differentiate between char-EC and soot-EC according to a stepwise thermal evolutional oxidation of different proportions of carbon under different temperatures and atmosphere (WHO 2012).

State of knowledge of "Black Carbon in air" attribute: Good / established but incomplete – general agreement, but limited data/studies

Part A—Attribute and method

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

Black carbon is formed through the incomplete combustion of fossil fuels, biofuel and biomass, and is emitted from both anthropogenic and natural sources. It is a component of particulate matter. The key impacts of black carbon are on human health and climate. It also contributes to brown haze through particle light absorption and was found to contribute around 25% of the brown haze in Christchurch in the early 2000s (Wilton, 2003) Health impacts of black carbon include short-term (24-hour) and long-term (annual) exposure impacts on cardiovascular health effects and premature mortality, and are independent of $PM_{2.5}$ impacts (World Health Organization, 2021). Elemental carbon (of which black carbon is a component) is a known carcinogen as a result of carcinogens within combustion processes condensing on the BC particles (World Health Organization, 2021).

Black carbon is a powerful climate-warming agent that acts by absorbing heat in the atmosphere and by reducing the ability to reflect sunlight when deposited on snow and ice (Bond et al., 2013). It is a Short Lived Climate Pollutant (SLCP), also referred to as a Short-lived Climate Forcer by the Intergovernmental Panel on Climate Change (IPCC, 2024). It's lifetime in the atmosphere is days to weeks.

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

Evidence for the health impacts of BC has been available for at least the past decade. For example, WHO 2012, states:

> "The systematic review of the available time-series studies, as well as information from panel studies, provides sufficient evidence of an association of short-term (daily) variations in BC concentrations with short-term changes in health (all-cause and cardiovascular mortality, and cardiopulmonary hospital admissions). Cohort studies provide sufficient evidence of associations of all-cause and cardiopulmonary mortality with long-term average BC exposure."

However, due to a lack of sufficiently detailed spatial measurements, there is still not enough epidemiological data to be able to set exposure guidelines (WHO 2021).

Monitoring data for New Zealand shows that BC concentrations vary both temporally and spatially (Davy & Trompetter, 2017).

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

Receptor modelling results indicate that diesel vehicle emissions and biomass combustion are the primary sources of BC in New Zealand urban areas (Davy & Trompetter, 2017). Concentrations have been gradually improving in the few areas with sufficient records for trend determination as a result of improvements in motor vehicle and solid fuel burning BC concentrations (Davy & Trompetter, 2017).

The National Environmental Standards for Air Quality (NESAQ) require wood burners to meet a specified emission and energy efficiency criteria if they are to be installed on properties less than 2 hectares in area from 2005. The latter is likely to have ongoing impact on BC concentrations as older less efficient burners are replaced with more efficient and lower emission alternatives. Many Regional Councils have adopted additional measures to reduce urban particulate from solid fuel burning which may also accelerate reductions in BC.

Existing central government policies targeting motor vehicles are likely to result in reductions in BC from this source over the next 10-30 years. The impact of changes in vehicle fleet composition, including increased electric vehicle use, on tailpipe $PM_{2.5}$ emissions which include a BC component are integrated into the national vehicle fleet emission model[1] (VEPM 7.0) (New Zealand Transport Agency, 2019). This model indicates a 95% reduction in tailpipe $PM_{2.5}$ emissions from motor vehicles over the next 26 years (VEPM with default model parameters). Waka Kotahi also aim to reduce total

1 Whilst VEPM includes assumptions around integration of Euro 6/VI vehicles these may not reflect the impact of existing policy and thus improvements may be underestimated.

kilometres travelled by the light fleet by 20% by 2035 through improved urban form and providing better travel options (Waka Kotahi, 2024) and this will result in further improvements in BC from motor vehicles.

Impacts of air contaminants such as black carbon are not typically reversable if they contribute to development of chronic impacts through long term exposure.

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

In Aotearoa, there is no requirement to monitor BC, as there are no relevant NESAQ, ambient air quality guidelines nor a WHO guideline and thus there is limited monitoring. Continuous real-time measurements are carried out in Auckland (Customs St) and Wellington (Willis St) and in Christchurch. NIWA and others have carried out campaign based monitoring at various locations, including mobile monitoring (e.g., Olivares et al., 2007).

There is not a New Zealand standard method for black carbon monitoring. Methods used tend to focus on optical approaches including real time aethalometer measurement of absorption by particles (Wilton, 2003) and filter based measurements of light absorption by particles (Davy & Trompetter, 2017). Measurement is complex. Light absorption is only a proxy (indirect technique) for black carbon and direct method measurement systems require complex thermal systems (Wilton, 2003). The mass concentration of BC obtained using different methods varies significantly which confounds impact assessment (Zhang et al., 2023).

Statistics New Zealand report black carbon concentrations in New Zealand for the Auckland monitoring sites on their website under the indicators programme(Statistics New Zealand, 2024). GNS have prepared several reports indicating results of black carbon filter analysis including sources and trends (Davy & Trompetter, 2017, 2018).

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

BC monitoring is usually carried out at existing air quality monitoring sites, so access is not usually problematic.

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

An optical instrument such as an aethalometer will cost several tens of thousands of dollars – typically \$30,000 to \$60,000. Operating costs tend to be minimal at less than a few thousand dollars per year typically.

Indirect measurement methods should be calibrated in-situ to more closely account for source and site related variables impacting on the estimates, which can add to costs. In practice, most users rely on the manufacturer's calibration supplied with the instrument. These are not overly reliable (Zhang et al., 2023). Filter based methods are also impacted on by multiple scattering effects of the filter and particles (Zhang et al., 2023)

Thermal optical methods are the most commonly used approach in the larger European programmes (Zhang et al., 2023). These use temperature to differentiate carbon components and there are still complexities in the assessment. Thermal optical instruments tend be much more expensive (> $100,000).

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

We are not aware of any Black Carbon monitoring being undertaken by iwi/hapū/rūnanga.

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

The correlation between black carbon and $PM_{2.5}$ concentrations is likely to be high as BC is a subset of $PM_{2.5}$ (Boamponsem et al., 2024). Variability in the relationship is likely to be greatest in areas where both motor vehicles and biomass burning are major sources of $PM_{2.5}$ owing to varying contributions of these sources to organic carbon emissions to $PM_{2.5}$.

Source apportionment studies carried out in New Zealand show black carbon is typically within the biomass burning and motor vehicle source profiles as a significant contributor but is also present in the marine aerosol profiles (e.g., (Ancelet et al., 2015). BC can be correlated with other combustion related air contaminants. However, the ratios of contaminants discharged varies with source, confounding ambient air correlations.

Part B—Current state and allocation options

B1. What is the current state of the attribute?

The current state is moderately understood in a small number of locations within Auckland, Wellington and Christchurch. Measurements have also been undertaken on daily filters used in short term (typically around a year) source apportionment studies, in up to ten additional locations.

Whilst some air quality data are nationally available on the LAWA website (https://www.lawa.org.nz/), BC is not included.

Greater Wellington Regional Council and Auckland council publish data on their websites (https://gwrc-open-data-11-1-gwrc.hub.arcgis.com/, https://knowledgeauckland.org.nz/)

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

The largest natural source of BC is wildfires but there are no known background measurements in Aotearoa to act as a baseline.

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

There are currently no guidelines for ambient BC concentrations, but it can be managed indirectly through standards and guidelines for PM_{10} and $PM_{2.5}$. More extensive monitoring internationally is required to better characterise health impacts and evaluate appropriate guideline.

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

BC is a component of $PM_{2.5}$ for which there is no known safe threshold for adverse health impacts. Studies showing independent impacts of BC in conjunction with $PM_{2.5}$ are limited, and no health-based threshold has been identified. BC is a climate pollutant and hence contributes to any climate tipping points or thresholds.

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

Some health studies for air contaminants report lag effects between exposures and health endpoints. These do not impact on state or trend assessments. Because BC is carcinogenic, there may be considerable lag times of decades before effects reveal themselves.

However, because BC is a short-lived pollutant with an atmospheric lifetime of days to weeks, removal of sources will effectively remove any future impacts – this is also true of climate effects. An exception to this would be contributions to disease prevalence and increased frailty which could contribute to health impacts that occur sometime after exposures.

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

Good air is considered a taonga and Māori are often disproportionately affected by poor air quality ((Telfar Barnard & Zhang, 2018)). However, whilst air as a taonga is often mentioned in iwi environmental strategies, we are not aware of any iwi explicitly planning to reduce air pollution exposure of their members.

Part C—Management levers and context

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

MfE regularly publishes a SOE report on air quality, most recently in 2021. Whilst this does not include BC, it does include PM_{10} and $PM_{2.5}$, of which BC is a subset. Analysis undertaken by GNS shows BC to be most strongly associated with the biomass burning and motor vehicle source profiles in New Zealand and notes that the dominant contributor varies with location and proximity of monitoring site to roadside (Davy & Trompetter, 2017). The pressures and drivers of $PM_{2.5}$ are reasonably well understood as are (at a general or high level) the response and state. The relationship between emissions and concentrations is spatially and temporally dependent primarily influenced by meteorology and topography. These variables all impact on exposure and subsequent health effects are also influenced by underlying susceptibility (Figure 1).

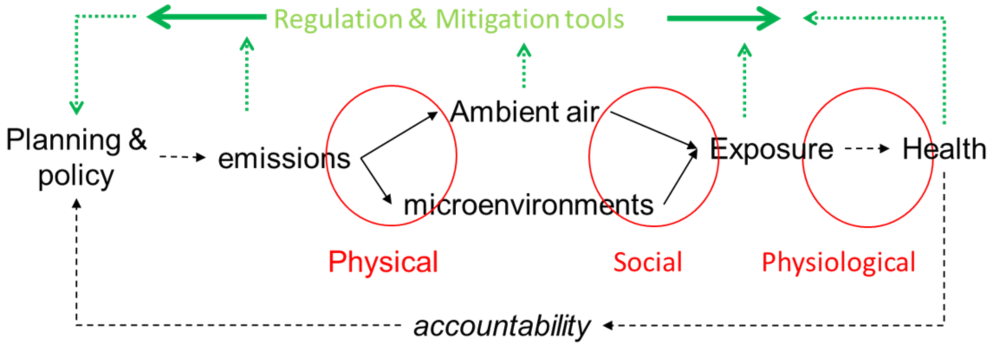

Figure 1. links in the chain from pollutant emissions to health effects

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

C2-(i). Local government driven

There is no NESAQ or other guideline for BC, however any interventions aimed at co-emitted pollutants will have an impact on BC concentrations. These include measures targeting domestic heating and motor vehicles in New Zealand.

C2-(ii). Central government driven

The NESAQ design criteria for wood burners was established for the purpose of reducing concentrations of particulate in urban areas. The impact of this legislation is ongoing in most areas as households replace older more polluting and less efficient burners with compliant models over time. Additionally, many Regional Councils have adopted regulatory measures targeting domestic heating to further improve or accelerate reductions in urban particulate concentrations. BC concentrations are likely to reduce in conjunction with $PM_{2.5}$.

The impact of national measures targeting other contaminants (e.g., fuel and technology specifications) and greenhouse gas emissions from motor vehicles is likely to also impact on black carbon concentrations from this source. Roadside monitoring of black carbon in Wellington CBD shows a reduction in BC concentrations from 2016 to 2023 (https://www.gw.govt.nz/your-council/open-data/).

The Ministry for the Environments 2021 state of the environment report found concentrations of $PM_{2.5}$ to be improving in four out of eight sites from 2011 to 2020 with three sites being indeterminate and one site showing worsening concentrations (Ministry for Environment, 2021).

C2-(iii). Iwi/hapū driven

Iwi/hapū planning documents such as Environmental Management Plans and Climate Change Strategies/Plans may contain policies/objectives/methods seeking to influence air quality outcomes for the benefit of current and future generations. We are not aware of other interventions/mechanisms being used by iwi/hapū/rūnanga to directly affect this attribute.

C2-(iv). NGO, community driven

Most community air quality projects we are aware of have concentrated on particle concentrations, usually from wood burning of which BC is a subset. NIWA has conducted numerous community collaborations on particles, but we are not aware of any directly targeting BC.

C2-(v). Internationally driven

The global driver and lead of air quality guidelines and management strategies is the WHO. Aotearoa's NESAQ were largely based on WHO guidelines published in the 1990s (and reaffirmed in 2005). In 2021 the WHO substantially revised its guidelines, setting much lower values for many pollutants but could not give guidance for BC. It did recommend that regulatory bodies take responsibility for monitoring BC in order to create sufficient evidence to set guidelines in future.

Part D—Impact analysis

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

Existing air quality management including measures targeting domestic heating and motor vehicles are likely to result in ongoing improvements in black carbon. Further management may be required for health protection noting that Māori continue to be disproportionately impacted. Contributions to environmental impacts such as brown haze in Auckland and Christchurch may continue.

Climate impacts: BC has a 1600 times greater global warming potential than CO_2. Therefore, if BC is not managed it will directly contribute to the climate crisis until it is.

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

Health and consequently economic impacts of BC exposure in New Zealand will be greatest in urban areas where concentrations of particulate are high and in areas near to roadways. Māori are disproportionately affected by respiratory and cardiovascular disease (Mason et al., 2019; Telfar Barnard & Zhang, 2018) and will be more susceptible to the health impacts of BC exposure.

Contribution of BC to visibility degradation could have economic impacts by way of reduced tourism for areas such as Christchurch and Auckland where brown haze is noticeable.

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

This attribute contributes to climate change and will be affected by it. A warmer climate increases the risk of wildfire which is a source of BC. There is the potential that BC concentrations in urban areas may decrease slightly as a result of climate change. This may occur if climate change results in fewer ground frosts as the associated low wind speeds and decreased vertical dispersion increase the potential for degraded air.

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5.7 Benzo(a)pyrene in air

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Preamble: BaP is a contaminant that occurs ubiquitously with a range of polycyclic aromatic hydrocarbons (PAHs) primarily as a result of incomplete combustion of fossil fuels and wood. BaP is considered to be a marker for PAHs as it is one of the most strongly carcinogenic of the hundreds of PAHs that exist (of which only 16 are on the US EPA list of priority pollutants and therefore routinely analysed). BaP is primarily associated with particulate matter in air, although it is a semi-volatile compound, and a small fraction may be present in gaseous form. Other PAHs may be more or less volatile than BaP.

State of knowledge of "Benzo(a)pyrene (BaP) in air" attribute: Good / established but incomplete in that studies undertaken agree that elevated BaP is recognised to occur across New Zealand, primarily associated with wood burning for residential heating. Excellent and well-established in relation to the effects on toxicity effects of benzo(a)pyrene on people, although poor / inconclusive regarding the extent of the impact of BaP in ambient air on human health.

Section A—Attribute and method

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

The primary concern associated with BaP in air relates to human health. Several comprehensive reviews of the toxicity of benzo(a)pyrene have been undertaken (1-5). Studies in multiple animal species demonstrate that benzo[a]pyrene is carcinogenic at multiple tumour sites (alimentary tract, liver, kidney, respiratory tract, pharynx, and skin) by all routes of exposure. An increasing number of occupational studies demonstrate a positive exposure-response relationship with cumulative benzo[a]pyrene exposure and lung cancer [5]. The toxicity of PAH mixtures is often determined through the use of potency equivalence factors (PEFs), which express the toxicity of individual PAHs that are carcinogenic relative to that of BaP (BaP-equivalents). Non-carcinogenic PAHs are considered separately.

In terms of non-carcinogenic effects, animal studies show that inhalation exposure to benzo[a]pyrene results in developmental and reproductive toxicity, and available human PAH mixtures studies report developmental and reproductive effects that supports the findings from animal studies. [4]

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

There is no evidence of impact of BaP in ambient air on human-health in NZ. However, there are multiple studies that have assessed BaP concentrations in ambient air in several New Zealand urban areas. The majority of these studies have been undertaken in Christchurch or other South Island towns [6-9], with studies being undertaken in Hamilton [6], Auckland [6] and Masterton on the North Island [10]. These studies show markedly elevated concentrations over winter, which is primarily associated with wood-burning for residential home heating [6-11], with concentrations also influenced by the formation of winter inversion layers [12]. The biological effect associated with particulates from Christchurch and Auckland was assessed by [13] and showed that the organic extracts of Christchurch PM2.5 and PM10 showed higher mutagenicity and CYP1A1 induction compared with PM10 from Auckland. In contrast, water-soluble extracts of Auckland PM were more cytotoxic and resulted in greater TNF-α release than those from Christchurch PM, although they had a lower metal content. The effect associated with the organic fraction was primarily attributed to PAHs.

Observed average concentrations were around 11-15 times higher than the Ambient Air Quality Guidelines (AAQG) 0f 0.3 ng/m3 in Christchurch in 2008/9 and 2012 [6, 7] and 14-16 times higher in Timaru [6, 8].

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

There are limited data available to assess change over time. Studies undertaken in Christchurch show slightly higher annual average BaP concentrations of 4.4. ng/m^3 in 2012 [7], as compared to concentrations of 3.4 ng/m^3 in 2008/9 [6]. This increase follows from an apparent increase in BaP concentrations in Christchurch in 2003/2004 from 3.1 ng/m^3.   Studies in Timaru show a slight decrease in concentrations from 6.4 ng/m^3 in 2007 [6] to ~ 5.5 ng/m3 in 2012 [8].

The comparisons are indicative only, as the studies used different sampling techniques (i.e., sampling of gas-phase and particulates-phase PAHs or particulate-phase only), analytical techniques, frequency and duration of sampling [6, 7].

In the context that BaP in NZ air is primarily associated with the incomplete combustion of wood, and there has been a strong focus on reducing emissions associated with wood burning for home heating (e.g., 14, 15), it might be expected, or hoped, that concentrations would decrease alongside particulate concentrations.

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

No regular monitoring of this attribute is currently undertaken in New Zealand. As noted in A2, there have been some studies that have assessed BaP in air. Particularly in earlier studies, different sampling techniques (i.e., sampling of gas-phase and particulates-phase PAHs or particulate-phase only) were used, with analysis most commonly undertaken using solvent extraction and gas chromatography-mass spectrometry. More recent studies in Christchurch [7] and Timaru [8] used thermal-desorption/gas chromatography-mass spectrometry, partly because these studies were focussed on using a range of organic compounds to assist with source apportionment studies.

Internationally, the European directive 2004/107/EC outlines the requirements for monitoring arsenic, cadmium, mercury, nickel and polycyclic aromatic hydrocarbons in ambient air. Associated with this directive are European Standards that specify the determination of particulate matter concentrations in ambient air (EN1241:2023), and EN 15549:2008, which specifies the standard method for analysis of BaP in PM10 aerosol, through sample extraction and analysis by high performance liquid chromatography (HPLC) with fluorescence detector (FLD) or by gas chromatography with mass spectrometric detection (GC/MS).

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

Monitoring for this attribute would most sensibly be co-located at existing air-quality monitoring sites, thus there are unlikely to be any additional access issues. However, space to fit equipment, if additional is required, may be an issue at some locations.

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

Currently there is no existing ongoing monitoring of this attribute. Where existing air quality sampling includes the use of instruments that collect particulate matter on filters e.g., Partisol samplers, these filters may be able to be used for analysis to determine arsenic concentrations. However, method evaluation is required to determine whether BaP can be detected in the particulate mass typically captured by these instruments or whether a higher volume sampler is required; for example, Partisol samplers can sample at between ~0.6-1.2 m3/hr with the USEPA specifying 1m3/hr (16.7 L/min) for regulatory sampling, however other instruments can sample at different rates, higher or lower.

Currently there is no commercially available method for the determination of benzo-a-pyrene in particulate matter. However, commercial laboratories have previously been used for the analysis of extracts from filters [6], and the general analysis is similar to that required determining PAHs in soils hence it would seem feasible for commercial laboratories to develop the method if there was sufficient demand.

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

We are not aware of any BaP monitoring being undertaken by iwi/hapū/rūnanga.

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

There is likely to reasonable correlation with PM2.5 concentrations, given the association of BaP with incomplete combustion of wood and fossil fuels (i.e. wood-burning for home-heating and vehicle exhaust emissions).

B) Current state and allocation options

B1. What is the current state of the attribute?

As noted in A2, there have been various studies of benzo-a-pyrene in New Zealand, with additional studies providing information on biological effects associated with air particulates [13]. However, the most recent study was undertaken over 10 years ago, hence there is a considerable gap in our current understanding of the state of the attribute.

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

We are not aware of any natural reference states for this attribute.

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

New Zealand ambient air guidelines [4] have an annual average guideline value for BaP in 0.3 ng/m^3. [6] also comments that It is unclear as to how the 0.3 ng m-3 value was actually selected; calculation of the increased risk at 0.3 ng/m3 using the WHO unit risk of 8.7 per ng/m3 yields a risk of 1 in 38 300 or ~3 in 100 000 and that it is unclear why the value of 0.12 ng/m3, which is associated with a risk level of 1 in 100 000, was not adopted as this risk level is consistent with that nominally associated with ambient air quality guidelines for As and other New Zealand documents (e.g., [1]).

Internationally, the EU directive 2004/107/EC provides a target value for BaP of 1 ng/m^3, which is based on the total content in the PM10 fraction averaged over a calendar year. Arsenic is not included in Australian or US air quality standards (but lead is).

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

From toxicological data there are various thresholds that have been identified as leading to different effects (see 1-5). However, there are no known thresholds or tipping points (and no studies undertaken to establish these) associated with benzo(a)pyrene concentrations in ambient air.

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

The existence of lag times and legacy effects for this attribute is unknown/uncertain.

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

A high standard of air quality is an outcome sought by iwi/hapū/rūnanga (see Section 3.2 for one example). Māori are often disproportionately affected by poor air quality (Telfar Barnard et al 20XX). In addition to discussing this attribute directly with iwi/hapū/rūnanga, regarding air quality, there is likely to be tikanga and mātauranga Māori relevant to informing bands, allocation options, minimally disturbed conditions and/or unacceptable degradation in treaty settlements, cultural impact assessments, environment court submissions, iwi environmental management plans, climate change strategies, etc.

Part C—Management levers and context

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

It is recognised that BaP is primarily associated with incomplete combustion fossils fuels and biomass burning e.g., wood, with particulate emissions from wood-burning for residential home-heating considered to be the primary source of particulates in New Zealand air e.g., [6, 11]. Given there has been a strong focus on reducing emissions associated with wood burning for home heating (e.g., 14, 15), it might be expected, or hoped, that concentrations would decrease alongside particulate concentrations.

BaP is also derived from incomplete combustion of fossil fuels e.g., petrol, diesel, coal. With increasingly strict emissions control on vehicle exhaust emissions [21], along with an increasing proportion of electric vehicles, vehicle exhaust emissions could be expected to be reducing in relative significance. Reductions in industrial burning of coal to meet greenhouse gas emission targets will also act to reduce BaP concentrations in NZ air.

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

C2-(i). Local government driven

Councils have campaigns to raise awareness good practices to reduce emissions associated with home-heating.

C2-(ii). Central government driven

No interventions beyond general industrial emissions controls, and requirements for monitoring particulate matter under the National Environmental Standard for Air Quality are being used to affect this attribute.

C2-(iii). Iwi/hapū driven

Iwi/hapū planning documents such as Environmental Management Plans and Climate Change Strategies/Plans may contain policies/objectives/methods seeking to influence air quality outcomes for the benefit of current and future generations. We are not aware of other interventions/mechanisms being used by iwi/hapū/rūnanga to directly affect this attribute.

C2-(iv). NGO, community driven

C2-(v). Internationally driven

Part D—Impact analysis

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

Not managing this attribute could result in increased human health impacts, although it is difficult to gauge the potential magnitude of this increase, or indeed if an increase would occur. As noted in C1, there are a number of current interventions to generally reduce particulate emissions that would also be expected to reduce BaP concentrations in NZ air. Provided those interventions are maintained, it is unlikely that separate interventions to manage BaP are required. However, verification that BaP concentrations are declining alongside any reductions in particulate emissions would be valuable – particularly given the historically comparatively high concentrations of BaP in at least some New Zealand cities.

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

Where and who economic impacts would be felt is largely unknown for this attribute, although the expectation is that managing or not managing this attribute will have minimal economic impacts.

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

Indirectly through primarily through changed winter temperatures, which may result in more or less winter heating, and changes in electricity demand which may influence the extent to which coal-fired power stations are required.

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5.8 Arsenic in air

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Preamble: Arsenic in ambient air is primarily associated with particulate matter in air, hence discussion and measurement of this attribute focusses on analysis of the composition of particulate matter.

State of knowledge of "Arsenic in air" attribute: Good / established but incomplete in that studies undertaken agree that burning of treated timber in residential heating is the primary source of arsenic. Excellent and well-established in relation to the effects on health effects of arsenic on people, although poor / inconclusive regarding the extent of the impact on arsenic in ambient air on human health.

Part A—Attribute and method

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

The primary concern associated with arsenic in air relates to human health. Arsenic is a known human carcinogen, with most studies based on population exposure to drinking water derived from groundwater with naturally high arsenic [1]. These studies show increases in skin cancers and internal cancers of the exposed population. Various studies on occupational exposure also demonstrate causal links between inhaled arsenic and cancer [2,3]. For non-occupational exposed people, dietary sources of arsenic are the primary route of exposure, with food the dominant sources other than in countries where drinking water with naturally elevated arsenic is known to occur [1, 5-7]. The toxicity of arsenic varies depending on its chemical forms with inorganic species of arsenic (most likely to occur in air) considerably more toxic than organic forms (most likely to occur in food) [5].

Overviews of the toxicological effects associated with exposure to arsenic are provided by multiple sources [e.g., 1-7] with cancer, the primary health effect of concern.

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

There is no evidence of impact of arsenic in ambient air on human-health in NZ, although internationally estimates of the impact of atmospheric arsenic on health risks have been made [e.g., 8]. However, there are several studies that show elevated arsenic in air, primarily over winter, in various cities and regions across New Zealand. This includes Auckland [9], Tokoroa (Waikato) [10, 11], Wellington [12, 13, 14], Timaru [15]. Richmond [16, 17], Invercargill [18]. Earlier studies and data are also presented in [19], This elevation is attributed to the burning of copper-chromium-arsenic treated timber for home heating [8, 19, 20], with [20] noting that exceptionally high arsenic concentrations of 1860 mg/kg were observed in particulate matter collected from a pre-1994 wood burner in Tokoroa in which 'old-decking' had been burned.

Additionally, various studies on household dust, which may also be inhaled, have also been undertaken. An international study found that New Zealand had the greatest enrichment of arsenic in household dust compared to soil concentrations, which was attributed to the burning of copper-chrome-arsenic treated timber for home heating [21]. Two further studies provide further research on arsenic in household dust in New Zealand [22, 23].

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

Data from Henderson in Auckland provides the only analysis of data over multiple years, and no trend in arsenic concentrations was observed – rather seasonal variation, with highest concentrations occurring in winter and associated with residential wood-burning was most evident [8, see also B1]. Beyond this there is an unknown historical trajectory of change since arsenic in air has rarely been measured. The primary source of arsenic to NZ air is recognised as residential burning of treated (copper-chrome-arsenic) timber for home heating. As such, the extent of wood-burning in the future – as influenced by winter temperatures - may influence change. There are currently no recognised industrial sources of arsenic emissions, although internationally arsenic from industrial coal-burning activities has been identified as a source of arsenic to air [e.g., 24]. While arsenic from coal burning has been mentioned in a New Zealand context, there is ongoing scrutiny of industrial emissions, and a shift away from coal-fired boilers etc thus it is unlikely any such emissions would increase. Another source of arsenic in ambient air might be expected to be from suspension of soil particles, or localised areas associated elevated arsenic associated with mining waste [25, 26], although these are located away from urbanised areas where people might be exposed. However, the amount of expected change over the next 10-30 years appears minimal.

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

No regular monitoring of this attribute is currently undertaken in New Zealand, although as noted in A2, there have been multiple studies that have assessed arsenic associated with particulate matter in air. The majority of studies have been undertaken for source apportionment purposes, using ion-beam analysis (IBA), a non-destructive multielemental analytical technique. The minimum detection limits for As determination by IBA vary from 3-30 ng m^3 depending on the type and thickness of the filter media used to collect samples This compares to the New Zealand ambient air guideline of 5.5 ng m^{-3} (annual average) [4]. More recently source-apportionment studies have also used measurement by XRF (e.g., [9]), while other studies have used aqua regia extraction of filters combined with inductively coupled-mass spectrometry ICP-MS [12, 13] or water extracts of filters with analysis by ICP-MS [15].

Internationally, the European directive 2004/107/EC outlines the requirements for monitoring arsenic, cadmium, mercury, nickel and polycyclic aromatic hydrocarbons in ambient air. Associated with this directive are European Standards that specify the determination of particulate matter concentrations in ambient air (EN1241:2023), and EN 14902:2005, which specifies the standard method for analysis of Pb, Cd, As and Ni in PM10 aerosol, through microwave digestion of the samples and analysis by graphite furnace atomic absorption spectrometry or by inductively coupled plasma (quadrupole) mass spectrometry.

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

Monitoring for this attribute would most sensibly be co-located at existing air-quality monitoring sites, thus there are unlikely to be any additional access issues. However, space to fit equipment, if additional is required, may be an issue at some locations.

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

Currently there is no existing ongoing monitoring of this attribute. Where existing air quality sampling includes the use of instruments that collect particulate matter on filters e.g., Partisol samplers, these filters may be able to be used for analysis to determine arsenic concentrations. However, method evaluation is required to determine whether arsenic can be detected in the particulate mass typically captured by these instruments or whether a higher volume sampler is required; for example, Partisol samplers can sample at between ~0.6-1.2 m3/hr with the USEPA specifying $1m^3$/hr (16.7 L/min) for regulatory sampling, however other instruments can sample at different rates, higher or lower.

Currently there is no commercially available method for the determination of arsenic in particulate matter. The general method outlined in the European standard EN 14902:2005 is similar to that used for determining arsenic in soils hence it would seem feasible for commercial laboratories to develop the method if there was sufficient demand.

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

We are not aware of any monitoring of this attribute being undertaken by iwi/hapū/rūnanga

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

There may some correlation with PM2.5 concentrations, given the association of arsenic with particulate matter associated with wood-burning – but the association will be dependent on the source of arsenic. Some sources e.g., tyre-wear, brake dust, soil dust will fall into larger particulate size fractions – hence measurement of arsenic will depend on particle size being measured. Regardless of particulate size, any relationship with particulate mass is still likely to be variable depending on the contribution of different sources.

Part B—Current state and allocation options

B1. What is the current state of the attribute?

As noted in A2, various studies have assessed concentrations of arsenic in ambient air, and in household dust in a number of locations across New Zealand. These studies identify the burning of CCA-treated timber in residential wood-burners as the primary source of arsenic in New Zealand air. Outdoor burning of treated timber may also occur e.g., in farm burn piles, although this is unlikely to be captured through existing monitoring. These studies do not provide comprehensive coverage on the state of arsenic in air across all of NZ towns and cities. Given the random and sporadic occurrence of burning of treated timber – in residential wood-burners or outdoor burn piles, the value of undertaking additional monitoring to fill these gaps is perhaps debateable. It does provide some value in ascertaining whether campaigns to not burn treated timber are effective or need to be stepped up, but arguably regular profiling of this issue could be scheduled in the absence of this information to the same effect but without the monitoring cost.

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

To our knowledge, there are no known natural reference states for this attribute.

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

New Zealand ambient air guidelines [4] have an annual average guideline value for inorganic arsenic is 0.0055 $\mu g/m^3$.

Internationally, the EU directive 2004/107/EC provides a target value for arsenic of 6 ng/m^3, which is based on the total content in the PM10 fraction averaged over a calendar year. Arsenic is not included in Australian or US air quality standards.

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

From toxicological data there are various thresholds that have been identified as leading to different effects (see 1-7). However, there are no known thresholds or tipping points (and no studies undertaken to establish these) associated with arsenic concentrations in ambient air. As noted above, general population exposure to arsenic is more likely via dietary sources.

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

The existence of lag times and legacy effects for this attribute is unknown/uncertain.

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

A high standard of air quality is an outcome sought by iwi/hapū/rūnanga. In addition to discussing this attribute directly with iwi/hapū/rūnanga, regarding air quality, there is likely to be tikanga and mātauranga Māori relevant to informing bands, allocation options, minimally disturbed conditions and/or unacceptable degradation in treaty settlements, cultural impact assessments, environment court submissions, iwi environmental management plans, climate change strategies, etc.

Part C—Management levers and context

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

The dominant source of arsenic in New Zealand air is widely recognised to be the burning of copper-chromium-arsenic treated timber in residential wood-burners, with studies in various regions across New Zealand showing marked elevations of arsenic in air over winter periods (see A2). Some elevated arsenic concentrations have also been observed outside of this period, and has been attributed to outdoor burning of treated timber [17]

There are natural sources of arsenic in ambient air, such as soil particles which may contain naturally occurring arsenic, however the contribution of this source has not been quantified.

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

C2-(i). Local government driven

Some councils have campaigns to raise awareness of the risks associated with the burning of CCA treated timber e.g. [26], or regional plans that prohibit the burning of treated timber (e.g., 27]. Beyond this, there are general industrial emissions controls, and requirements for monitoring particulate matter under the National Environmental Standard for Air Quality.

C2-(ii). Central government driven

C2-(iii). Iwi/hapū driven

Iwi/hapū planning documents such as Environmental Management Plans and Climate Change Strategies/Plans may contain policies/objectives/methods seeking to influence air quality outcomes for the benefit of current and future generations. We are not aware of other interventions/mechanisms being used by iwi/hapū/rūnanga to directly affect this attribute.

C2-(iv). NGO, community driven

C2-(v). Internationally driven

Part D—Impact analysis

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

Not managing this attribute could result in increased human health impacts, although it is difficult to gauge the potential magnitude of this increase, or indeed if an increase would occur. As noted above, the primary source of arsenic in air is the burning of treated timber in residential wood-burners. There is an increasing use and availability of low emission wood burners to generally manage emissions from residential wood burners and campaigns to raise awareness of the hazards of residential burning of treated timber. However, the extent to which treated timber may be burned is essentially random. It is most likely to be burned if people have a wood burner, cannot afford to buy firewood and have a source of treated timber that could be burned.

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

Although uncertain, the expectation is that managing or not managing this attribute will have minimal economic impacts.

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

Indirectly through primarily through changed winter temperatures, which may result in more or less winter heating, and also indirectly through dust generated from soil as a result of climate related events. Evaluation of the significance of this hazard is required to ascertain whether, and how management is required to mitigate this.

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5.9 Benzene in air

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Preamble: Benzene is volatile organic compound and is associated with gaseous phase in air. Benzene is commonly found associated with toluene, ethylbenzene and xylenes and collectively are referred to as BTEX. Benzene is typically found in the highest concentrations in ambient air.

State of knowledge of the "Benzene in air" attribute: Good / established but incomplete in that studies undertaken agree that elevated benzene can occur across New Zealand, primarily associated with vehicle emissions although concentrations are generally low and below concentrations that might be of concern for human health.

Part A—Attribute and method

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

The primary concern associated with Benzene in air relates to human health. Several comprehensive reviews of the toxicity of benzene have been undertaken (1-6). Benzene exposure affects the central nervous system (CNS), the haematopoietic (blood cell formation) system and immune system. The bone marrow is the target organ for the expression of benzene haematotoxicity and immunotoxicity, which are consistently reported to be the most sensitive indicators of non-cancer toxicity in both humans and experimental animals.

Benzene is a well-established human carcinogen and is classified as a known human carcinogen (Class 1, Group A) by [5]. Epidemiological studies of benzene-exposed workers have demonstrated a causal relationship between benzene exposure and the production of acute myelogenous leukaemia and also suggest evidence for non-Hodgkin lymphoma, chronic lymphoid leukaemia, multiple myeloma, chronic myeloid leukaemia, acute myeloid leukaemia in children, and cancer of the lung.

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

There is no evidence of impact of benzene in ambient air on human-health in NZ. However, there are several studies undertaken, primarily in the late 1990's and early 2000's. These studies are reported in [7]. Specifically, [7] reported that reported that benzene concentrations in Auckland, Christchurch, Hamilton, Dunedin, Nelson, the Bay of Plenty and on the West Coast mostly fell within the then current guideline value of 10 µg/m^3 annual as well as the guideline value to meet in 2010 of 3.6 µg/m^3 at most 'residential' sites. Exceedances (i.e., >10 µg/m^3) were recorded at peak traffic sites of Khyber Pass Road in Auckland (2002) and Riccarton Road in Christchurch (2001). Subsequent annual monitoring has been undertaken in Auckland to 2013 [8, 9], and Hamilton to 2021 [10]. [8] reports monitoring at 6 sites across Auckland, mostly annual sampling for 5 years. Of these Khyber Pass Road was the only site with markedly elevated concentrations; the remaining sites all showed annual average concentrations <3.6 µg/m^3 over the duration of monitoring. [9] reported monitoring undertaken at Khyber Pass Road over 2001 to 2013, and at another heavily trafficked site over 2009-2012. Benzene was considered to be elevated above guideline values in 2011. Current monitoring in Auckland comprises monthly passive sampling at three sites (two on Khyber Pass Road, one in Penrose) with benzene concentrations below detection limits at all sites in 2022 [18]. Monitoring in 2020 and 2021 found annual average benzene concentrations of 2 and 2.4 µg/m^3 [19].

Eight sites in Hamilton have been monitored over 2003 to 2020 [10]. Monitoring results show that from 2003 to 2005 benzene measured at two high-density traffic sites in Hamilton, would have breached the current National Ambient Air Quality Guideline for benzene, which is 3.6 µg/m^3 (annual average). From 2006 to 2020 annual average concentrations have been <3.6 µg/m^3 at all eight sites, with all sites being <1 µg/m^3 from 2017.

Monitoring was also undertaken at 20 site across the Taranaki in January 2019 [11]. This study determined 1-hr average benzene concentrations at the sites. The highest concentrations were observed at a petrol station, and an unused industrial site.

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

The interest in monitoring benzene was stimulated by the removal of lead from petrol and recognition that lead-free petrol had higher concentrations of aromatic hydrocarbons including benzene [8]. Fuel specification regulations imposed increasingly stringent control on the benzene content of petrol, particularly from 2002, which was attributed to the marked decline in benzene concentration up to 2009 by [8]. After this time, concentrations were considered to have stabilised, with [8] suggesting that the effectiveness of reducing benzene in fuel had reached its limit of effectiveness. Continuing declines in annual average benzene concentrations was observed at 8 sites in Hamilton up to 2020 when monitoring ceased. In 2020, all annual average concentrations were <1 µg/m^3[10].

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

Auckland Council is the only council that currently monitors benzene (along with toluene, xylene and ethylene) using passive samplers [18,19]. Similarly, annual monitoring had been undertaken in Hamilton using passive samplers until 2021. This monitoring, plus most other studies undertaken have tended to use passive samplers with the 3M organic vapour badges most popular. These badges are then sent to commercial laboratories for analysis using by extraction with carbon disulphide and analysed using Gas Chromatography Flame Ionisation Detection. This method was used in original studies by Stevenson and Narsey 1999 (cited in 10] and was further validated by [12]. However, this method is considered to be a screening method only as it does not conform to methods outlined in the Good Practice Guide [13].

A continuous sampling device – a gas chromatograph with photoionisation detector used in Khyber Pass Rd, Auckland and described in [8] is indicated to be the only method used that does conform with methods specified in [13]. However, monitoring via continuous sampling is no longer undertaken in Auckland (L. Boamponsem, Auckland Council, pers. Comm).

Other methods have also been trialled and evaluated [8, 13]

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

Given the size of the passive samplers, and the likely location of deployment i.e. near roadsides, and hence public space, it is unlikely that there would be any implementation issues associated with land access etc. Sampling may also be undertaken at existing air quality sampling sites for which access is already negotiated.

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

Indicative monitoring costs for passive sampling were provided by Waikato Regional Council (J Caldwell, WRC, pers. com). Specifically, 3M organic vapour badges (~$90 each, available from Office Max) were exposed in duplicates at monitoring sites (8) for 3 months and then taken to Hill Labs where the volatile organic compounds (BTEX) were extracted using carbon disulphide and analysed using Gas Chromatography Flame Ionisation Detection. Lab analysis typically worked out to about $145 per badge, resulting in an overall cost of around $7000 in total. Additional costs would be incurred for staff time to deploy and collect the samplers.

The sampling and analysis costs advised by Auckland Council for monthly monitoring at two locations were ~$1,600 (L Boamponsem, AC, pers. Com). These analyses are undertaken at Watercare laboratories.

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

We are not aware of any monitoring of this attribute being undertaken by iwi/hapū/rūnanga

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

Benzene is correlated with other indicators of vehicle emissions including nitrogen dioxide concentrations, black carbon and other volatile organic compounds (e.g., toluene, xylene).

Part B—Current state and allocation options

B1. What is the current state of the attribute?

As noted in A2, there have been various studies of benzene in ambient air in New Zealand. Ongoing monitoring occurs in Auckland, although concentrations were below detection limits in 2022 [18]. Other than this, the most recent monitoring was undertaken in Hamilton in 2020, which showed benzene concentrations at 8 sites were <1 µg/m^3. Monitoring in Auckland includes high-traffic sites, thus, benzene concentrations more generally in New Zealand would be considered very low.

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

To our knowledge, there are no known natural reference states for this attribute.

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

New Zealand ambient air guidelines [6] originally proposed an annual average guideline value for benzene of 10 µg/m^3 with a guideline value of 3.6 µg/m^3 to be achieved by 2010. Internationally, the EU directive 2008/EC/50 provides a limit value for benzene of 5 µg/m^3 to be met from 2010. [14] also provides an overview of additional international standards.

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

From toxicological data there are various thresholds that have been identified as leading to different effects (see 1-6). However, there are no known thresholds or tipping points (and no studies undertaken to establish these) associated with benzene concentrations in ambient air.

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

Reduction of the benzene content in petrol is attributed for the main reason for observed decreases in benzene in New Zealand [8-10], with the most marked reductions occurring in the mid-2000s. Whether further changes in benzene concentrations in fuel would influence ambient benzene concentrations is unclear.

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

A high standard of air quality is an outcome sought by iwi/hapū/rūnanga (see Section 3.2 for one example). In addition to discussing this attribute directly with iwi/hapū/rūnanga, in regard to air quality, there is likely to be tikanga and mātauranga Māori relevant to informing bands, allocation options, minimally disturbed conditions and/or unacceptable degradation in treaty settlements, cultural impact assessments, environment court submissions, iwi environmental management and climate change plans etc.

Part C—Management levers and context

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

Benzene emissions are considered to be primarily associated with petrol vehicle emissions, and with increasingly strict emissions control on vehicle exhaust emissions [15], along with an increasing proportion of electric vehicles, vehicle exhaust emissions could be expected to be continuing to reduce.

While benzene is also recognised to be emitted during wood-burning for residential heating, this does not appear to have a major influence on ambient air concentrations. Nonetheless, given the strong focus on reducing emissions associated with wood burning for home heating (e.g., 16, 17), it might be expected, that concentrations would decrease alongside particulate concentrations.

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

C2-(i). Local government driven

Councils have campaigns to raise awareness good practices to reduce emissions associated with home-heating.

C2-(ii). Central government driven

C2-(v). Internationally driven

Engine Fuel Specifications Regulations are the dominant mechanism used to affect this attribute, by specifying the maximum amount of benzene that can be present in petrol (1%).

C2-(iii). Iwi/hapū driven

Iwi/hapū planning documents such as Environmental Management Plans and Climate Change Strategies/Plans may contain policies/objectives/methods seeking to influence air quality outcomes for the benefit of current and future generations. We are not aware of other interventions/mechanisms being used by iwi/hapū/rūnanga to directly affect this attribute.

C2-(iv). NGO, community driven

Part D—Impact analysis

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

Not managing this attribute, other than through the existing fuel regulations, is unlikely to result in increased human health impacts.

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

Although uncertain, the expectation is that managing or not managing this attribute will have minimal economic impacts.

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

There are no recognized mechanisms by which benzene concentrations in ambient air in NZ would be affected by climate change.

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5.10 Lead in air

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Preamble: Lead in ambient air is primarily associated with particulate matter in air, hence discussion and measurement of this attribute focusses on analysis of the composition of particulate matter.

State of knowledge of the "Lead in air" attribute: Good / established but incomplete in that studies show that lead is present in ambient air, but poor / inconclusive regarding the extent of the impact of human health.

Part A—Attribute and method

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

The primary concern associated with lead in air relates to human health. Lead is a highly toxic element, with the major concern being the cognitive and neurobehavioural deficits that are observed in children exposed to lead [1]. Effects on blood-pressure are the most sensitive effects of lead toxicity on adults with the full range of health effects associated with exposure to inorganic lead and compounds include, but are not limited to neurotoxicity, developmental delays, hypertension, impaired haemoglobin synthesis, and male reproductive impairment. The effects of lead exposure have often been related to the blood lead content, which is generally considered to be the most accurate means of assessing exposure.

Overviews of the toxicological effects associated with exposure to lead are provided by multiple sources [e.g., 1-3].

Since the removal of lead from petrol in New Zealand in 1996, the main source of non-occupational exposure is lead-based paints on and around houses built before 1970 and particularly before 1945 [4]. Beyond this, food intake, and drinking water, in particular where lead has been used in the plumbing system, may also result in exposure [1].

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

There is no evidence of impact of lead in ambient air on human-health in NZ. There are some studies that provide concentrations of lead in air, with some evidence of higher concentrations in winter e.g., Auckland [5], Tokoroa (Waikato) [6], Richmond [7]. These current observations of elevated lead concentrations in winter were attributed to the burning of lead-painted timber [5]. One study that links lead with lower cognitive function in NZ suggests poor air quality as a cause [8], but neither that study nor the cited reference [9] provide any information on ambient lead concentration (but see A3 for more information). Ongoing monitoring in Auckland at an industrial site, Penrose, showed a decrease in annual average concentrations from2.3 ng /m^3 in 2021 to 8 ng /m^3 in 2022 [28].

Despite the banning of lead in petrol, a recent international study suggested that historical gasoline-derived lead remains an important source of lead in the urban environment due to its persistence and effective remobilization [10]. A study in Timaru over from June to August 2010 found water-extractable lead concentrations of 1 ng /m3, and 12.6 ng/mg [11].

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

Data from Henderson in Auckland provides the only analysis of concentrations over time, and this shows a decreasing trend over 2007-2021 [5]. Beyond this there is an unknown historical trajectory of change since lead in air has rarely been measured. The lead concentrations in air in Henderson over the period 2007 to 2021 ranged between 2 to 12 ng /m^3 [5]. This contrast with an earlier study undertaken at two sites in Christchurch over 1987-1989 that found much higher lead concentrations of 70 and 155 ng/m^3, with particulate concentrations of 4820 and 6320 µg lead/g [12]. Vehicle emissions were considered to be the dominant source with a minor contribution from coal and soil. In New Zealand lead in petrol was phased out from 1986 prior to banning in 1996.

Additionally, some studies on household dust, which may also be inhaled, have also been undertaken. An early study of 120 house in Christchurch in 1987 found a geometric mean concentration of 573 µg lead/g with petrol lead and lead-based paints identified as the significant sources of lead in house dust [13]. A more recent international study, which included results from New Zealand houses found that increasing home age was associated with greater lead concentrations, with legacy sources (lead-paint) considered to be the dominant source [14]. [15] also provides further research on lead in household dust in New Zealand.

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

No regular monitoring of this attribute is currently undertaken in New Zealand, although as noted in A2, there have been some studies that have assessed arsenic associated with particulate matter in air. The majority of these studies have been undertaken for source apportionment purposes, using ion-beam analysis (IBA), a non-destructive multielemental analytical technique. Some more recent studies e.g., [5] use XRF, which generally provides better detection limits. Some studies have also used water extracts of filters with analysis by ICP-MS [11].

Internationally, sampling and analysis of lead in ambient air is specified under the Australian National Environmental Protection measure [16] with measurement of lead concentrations in particular matter– Particulate metals high or low volume sampler gravimetric collection – Inductively coupled plasma (ICP) spectrometric method specified in AS/NZS 3580.9.15:2014. The Determination of Suspended Particulate Matter – Total suspended particulate matter (TSP) - High volume sampler gravimetric method is specified under AS/NZS 3580.9.3:2015. In the UK sampling is specified under the UK with sampling of the PM10 fraction of particles is carried out using Digitel DPA-14 ambient air samplers over one-week periods at sites, in accordance with BS EN 12341:2009 and analysis of these samples occurs by Inductively Coupled Plasma - Mass Spectrometry (ICP-MS), in accordance with European Standard EN 14902:2005 at NPL's UKAS-accredited laboratory [17] . In the US, various methods are specified under the list of designated references and equivalent methods [18].

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

Monitoring for this attribute would most sensibly be co-located at existing air-quality monitoring sites, thus there are unlikely to be any additional access issues. However, space to fit equipment, if additional is required, may be an issue at some locations.

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

Currently there is no existing ongoing monitoring of this attribute. Where existing air quality sampling includes the use of instruments that collect particulate matter on filters e.g., Partisol samplers, these filters may be able to be used for analysis to determine arsenic concentrations. However, method evaluation is required to determine whether lead can be detected in the particulate mass typically captured by these instruments or whether a higher volume sampler is required; for example, Partisol samplers can sample at between ~0.6-1.2 m3/hr with the USEPA specifying 1m3/hr (16.7 L/min) for regulatory sampling, however other instruments can sample at different rates, higher or lower.

Currently there is no commercially available method for the determination of lead in particulate matter. Some of the general method outlined in AS/NZS 3580.9.15:2014 is similar to that used for determining lead in soils hence it would seem feasible for commercial laboratories to develop the method if there was sufficient demand.

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

We are not aware of any monitoring of this attribute being undertaken by iwi/hapū/rūnanga

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

There may some correlation with PM2.5 concentrations, given the association of lead with particulate matter – but the association will be dependent on the source of lead. Lead associated with residential wood-burning of lead-painted timber is more likely to be associated with lead derived from e.g., brake dust, soil dust will fall into larger particulate size fractions – hence measurement of lead will depend on particle size being measured. Regardless of particulate size, any relationship with particulate mass is still likely to be variable depending on the contribution of different sources.

Part B—Current state and allocation options

B1. What is the current state of the attribute?

As noted in A2, there are small number studies that have assessed concentrations of lead in ambient air. These studies do not provide comprehensive coverage on the state of lead in air across all of NZ towns and cities. Given the random and sporadic occurrence of burning of treated timber – in residential wood-burners or outdoor burn piles, the value of undertaking additional monitoring to fill these gaps is perhaps debateable.

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

To our knowledge, there are no known natural reference states for this attribute.

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

New Zealand ambient air guidelines [19] have a 3-month moving average guideline value for lead in PM10 of 0.2 µg/m3. This is similar to the US National Ambient Air Quality guideline of 0.15 µg/m3 of Pb in total suspended particles as a 3-month average [20].

Lead is not included in the EU directive 2004/107/EC, although the UK Air quality standards regulation (2010) provide a limit value of lead concentration in PM10 of 0.5 µg/m^3 expressed as an annual mean. This is the same at the Australian air quality standard set under the Australian National Environment Protection (Ambient Air Quality) Measure [16].

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

From toxicological data there are various thresholds that have been identified as leading to different effects (see 1-3). However, there are no known thresholds or tipping points (and no studies undertaken to establish these) associated with lead concentrations in ambient air. As noted above, exposure to leaded paint is likely to result in greater exposure than via ambient air.

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

The existence of lag times and legacy effects for this attribute is unknown/uncertain.

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

A high standard of air quality is an outcome sought by iwi/hapū/rūnanga. In addition to discussing this attribute directly with iwi/hapū/rūnanga, in regard to air quality, there is likely to be tikanga and mātauranga Māori relevant to informing bands, allocation options, minimally disturbed conditions and/or unacceptable degradation in treaty settlements, cultural impact assessments, environment court submissions, iwi environmental management and climate change plans etc.

Part C—Management levers and context

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

The current primary source of lead in New Zealand air is suggested to arise from the burning of lead-painted timber [5, 21], as with arsenic in air associated with the burning of treated timber this is likely to be a random source. Tetraethyl lead is listed as an additive in Avgas, although is restricted to <0.14% weight, equivalent to <0.85 g lead/L fuel, and there is an EPA approval for aviation gasoline and racing gasoline (Avgas 100 and Avgas 100LL) under the Hazardous Substances and New Organisms Act 1996 (HSNO Act) [22]. A recent international study suggested that remobilisation – at least in the highly urbanised city of London [10]. The extent to which this is a source in New Zealand air is unknown.

Lead may also be found in vehicle non-exhaust emissions– including associated with brake-pad wear [23] and lead tyre-weights [24], with some studies indicating that due to increasingly strict emissions control on vehicle exhaust emissions, non-exhaust emissions are increasing as a proportion of total vehicle emissions [25]. Similarly, an increasing proportion of electric vehicles will also increase the relative significance of vehicle non- exhaust emissions.

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

C2-(i). Local government driven

Some councils have campaigns to not burn painted timber e.g. [26], or regional plans that prohibit the burning of painted (e.g., 27]. Beyond this, there are requirements for monitoring particulate matter under the National Environmental Standard for Air Quality.

C2-(ii). Central government driven

C2-(iii). Iwi/hapū driven

Iwi/hapū planning documents such as Environmental Management Plans and Climate Change Strategies/Plans may contain policies/objectives/methods seeking to influence air quality outcomes for the benefit of current and future generations. We are not aware of other interventions/mechanisms being used by iwi/hapū/rūnanga to directly affect this attribute.

C2-(iv). NGO, community driven

C2-(v). Internationally driven

Internationally, the Geneva convention on long-range transboundary air pollution requires parties to reduce emissions of lead (and cadmium and mercury) from industrial sources, combustion processes and waste incineration. The protocol on heavy metals was amended in 2012 to introduce more stringent emission limit values (ELVs) for emissions of particulate matter and of cadmium, lead and mercury applicable for certain combustion and other industrial emission sources that release them into the atmosphere. The emission source categories for the 3 heavy metals were also extended to the production of silico- and ferromanganese alloys. New Zealand does not appear to be a signatory to this convention and hence has no obligations in this regard.

Part D—Impact analysis

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

There is likely negligible impact on human health of not managing lead in ambient air, given the current low concentrations in air, and the absence of significant sources. As noted in A1, exposure to lead-based paints in pre-1960 houses is likely to be more significant source that should be managed [4].

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

Although uncertain, the expectation is that managing or not managing this attribute will have minimal economic impacts.

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

Climate change may indirectly affect this attribute through changes primarily associated with changes in dust generated from soil as a result of climate related events. Evaluation of the significance of this hazard is required to ascertain whether, and how, management is required to mitigate this.

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5.11 Cadmium in air

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Preamble: This attribute stocktake has been prepared assuming the greatest focus is on ambient air, although it should be noted that cadmium is typically associated with particulate matter in air.

State of knowledge of 'Cadmium in air' attribute: Poor / inconclusive

Poor/inconclusive ranking mainly arising from the limited data available to assess state in New Zealand. However, the motivation for collection of that data is minimal as there is low risk associated with this attribute. There may be some value in expanding the national pollution inventory to include cadmium emissions to build a greater understanding of the magnitude of potential emissions and to understand potential points of intervention.

Part A—Attribute and method

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

The primary concern associated with cadmium in air relates to human health. Cadmium is considered as a class 1 carcinogen, with the strongest evidence for carcinogenicity arising from studies indicating occupational cadmium exposure is associated with lung cancer [1]. For non-occupational exposed people, smoking is a significant greater contributor to exposure via the inhalation pathway than ambient air. For non-smokers, dietary sources of cadmium is the primary route of exposure,

Overviews of the toxicological effects associated with exposure to cadmium are provided by multiple sources [e.g., 2-6] with accumulation in kidneys and renal failure the primary health effect of concern [2].

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

There is no evidence of impact of cadmium in ambient air on human-health in NZ. As noted above, the main source of exposure to cadmium is dietary exposure rather than air emissions.

Internationally, one epidemiological study undertaken in Belgium on people in close proximity to a zinc smelter found an increased incidence of lung-cancer associated with exposure to cadmium [7]. The primary source of exposure was suggested to be house-dust arising from the transfer of contaminated soil into houses.

An assessment of metal concentrations in house-dust across NZ, highlighted cadmium as a potential contaminant of concern – based on comparison of house-dust concentrations with NZ soils contaminant standards for rural residential land use of 0.8 mg/kg [8]. Concentrations ranged from 0.08 mg/kg to 15 mg/kg, with a median concentration of 0.6 mg/kg. An earlier study found a geometric mean cadmium concentration of 4.2 mg/kg in house dust from 120 houses in Christchurch in 1993 with no one dominant source of cadmium identified [9].

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

There is an unknown historical trajectory of change since cadmium in air has rarely been measured. Only two studies were found that reported cadmium concentrations in NZ air. The first study undertaken in 1994 reported mean cadmium concentrations of 0.814 ng/m3 in Christchurch with the primary source indicated to be from coal combustion [10], while a study undertaken in Richmond in 2017 found no cadmium concentrations above the limit of detection of 17 ng/m3 [11].

The primary sources of cadmium to the NZ air are unknown. There are no recognised industrial sources of cadmium emissions, although there is ongoing scrutiny of industrial emissions thus it is unlikely any such emissions would increase. The primary source of cadmium in ambient air might be expected to be from suspensions of soil particles. These may contain naturally occurring cadmium from weathered rock, or cadmium derived from anthropogenic sources such as phosphate-fertilisers.

It seems likely there would be minimal change over the next 10-30 years.

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

No monitoring of this attribute is currently undertaken in New Zealand.

A recent study that reported cadmium concentrations in NZ air used X-ray fluorescence spectroscopy (XRF) to determine elements in particulate matter collected on teflon filters using a Partisol sampler [11]. The limit of detection specified in this study was 17 ng/m3, with an uncertainty of 22 ng/m3- this contrasts with the target value of 5 ng/m3 specified for cadmium in the EU directive 2004/107/EC (see also A6). The earlier NZ study [10] used a hi-volume sampler for the collection of particulate matter and acid-extraction and analysis by Graphite Furnace Atomic Absorption Spectroscopy and measured much lower concentrations.

The European directive 2004/107/EC outlines the requirements for monitoring arsenic, cadmium, mercury, nickel and polycyclic aromatic hydrocarbons in ambient air. Associated with this directive are European Standards that specify the determination of particulate matter concentrations in ambient air (EN1241:2023), and EN 14902:2005, which specifies the standard method for analysis of Pb, Cd, As and Ni in PM10 aerosol, through microwave digestion of the samples and analysis by graphite furnace atomic absorption spectrometry or by inductively coupled plasma (quadrupole) mass spectrometry.

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

Monitoring for this attribute would most sensibly be co-located at existing air-quality monitoring sites, thus there are unlikely to be any additional access issues. However, space to fit equipment, if additional is required, may be an issue at some locations.

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

Currently there is no existing monitoring of this attribute. Where existing air quality sampling includes the use of instruments that collect particulate matter on filters e.g., Partisol samplers, these filters may be able to be used for analysis to determine cadmium concentrations. However, method evaluation is required to determine whether cadmium can be detected in the particulate mass typically captured by these instruments or whether a higher volume sampler is required; for example, Partisol samplers can sample at between ~0.6-1.2 m3/hr with the USEPA specifying 1m3/hr (16.7 L/min) for regulatory sampling, however other instruments can sample at different rates, higher or lower.

Currently there is no commercially available method for the determination of cadmium in particulate matter. The general method outlined in the European standard EN 14902:2005 is similar to that used for determining cadmium in soils hence it would seem feasible for commercial laboratories to develop the method if there was sufficient demand.

As noted above, XRF used by [11] in New Zealand appears unlikely to be sufficiently sensitive for the determination of cadmium.

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

We are not aware of any monitoring of this attribute being undertaken by iwi/hapū/rūnanga

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

There may some correlation with PM2.5 concentrations – but the association will be dependent on the source of cadmium. Many sources e.g., tyre-wear, soil dust will fall into larger particulate size fractions – hence measurement of cadmium will depend on particle size being measured. Regardless of particulate size, any relationship with particulate mass is still likely to be variable depending on the contribution of different sources.

Vapour-phase cadmium is only likely to be present in a small number of workplaces and should be managed through workplace safety.

Part B—Current state and allocation options

B1. What is the current state of the attribute?

The state of this attribute in New Zealand is largely unknown. Available studies are reported in A2 (house-dust) and A3 (cadmium in ambient air)

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

To our knowledge, there are no known natural reference states for this attribute.

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

In New Zealand, the only standards related to cadmium in air are provided under Worksafe, which provides a time-weighted average exposure standard for an 8-hr day of 0.004 for respirable dust NZ WES 0.004 mg/m3, which was adopted in 2020 [12].

Internationally, the EU directive 2004/107/EC provides a target value for cadmium of 5 ng/m3, which is based on the total content in the PM10 fraction averaged over a calendar year. Cadmium is not included in Australian or US air quality standards.

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

From toxicological data there are various thresholds that have been identified as leading to different effects (see 2-6). However, there are no known thresholds or tipping points (and no studies undertaken to establish these) associated with cadmium concentrations in ambient air. As noted above, cadmium exposure via dietary sources will be far greater than exposure via ambient air.

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

The only potentially relevant lag time or legacy effects arguably arises from cadmium accumulation in soils as a result of historical application of phosphate fertilisers and suspension of those soil particulates in the ambient air.

From a human health perspective, the effects of cadmium typically arise from chronic exposure over time with accumulation in kidneys and renal effects – although as noted previously inhalation is considered to be a minor route of exposure.

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

A high standard of air quality is an outcome sought by iwi/hapū/rūnanga (see Section 3.2 for one example). In addition to discussing this attribute directly with iwi/hapū/rūnanga, in regard to air quality, there is likely to be tikanga and mātauranga Māori relevant to informing bands, allocation options, minimally disturbed conditions and/or unacceptable degradation in treaty settlements, cultural impact assessments, environment court submissions, iwi environmental management and climate change plans etc.

Part C—Management levers and context

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

There are a wide range of natural and anthropogenic sources of cadmium in ambient air, with soil particles suggested to be the predominant source of natural emissions to the atmosphere, followed by forest and bush fires, sea salt, volcanic emissions and meteoric dust [13]. Other sources of natural Cd emissions are weathering of rocks, airborne soil particles, e.g., from deserts, sea spray, forest fires, biogenic material, volcanoes, and hydrothermal vents [13].

The key anthropogenic emissions of cadmium are fossil fuel and coal combustion, mining and smelting of metals, with additional sources related to the production, use, disposal and recycling of cadmium and cadmium containing products (including phosphate fertilisers with cadmium as impurities) [6, 13, 14]. Cadmium may also be found in vehicle non-exhaust emissions – primarily associated with brake-pad wear [15], with some studies indicating that due to increasingly strict emissions control on vehicle exhaust emissions, non-exhaust emissions are increasing as a proportion of total vehicle emissions [16]. Similarly, an increasing proportion of electric vehicles will also increase the relative significance of vehicle non- exhaust emissions.

The National Pollutant Inventory from Australia [17] provides some perspective on sources of cadmium emission – in this case bushfires as the dominant source, followed by paved and unpaved roads (Table 1).

Source	Air (kg)[1]
Burning(fuel red., regen., agric.)/ Wildfires [*]	20,000
Paved/ Unpaved Roads [*]	13,000
Basic Non-Ferrous Metal Manufacturing [213]	5,100
Motor Vehicles [*]	5,800
Metal Ore Mining [080]	1,800
Windblown Dust [*]	630
Solid fuel burning (domestic) [*]	630
Waste Treatment, Disposal and Remediation Services [292]	540
Electricity Generation [261]	350
Fuel Combustion - sub reporting threshold facilities [*]	270
Coal Mining [060]	220
Aeroplanes [*]	210
Basic Ferrous Metal Manufacturing [211]	180
Glass and Glass Product Manufacturing [201]	140

Table 1. Screenshot of the key sources of cadmium emissions in Australia from the National Pollutant Inventory. (http://www.npi.gov.au/npidata/action/load/emission-by-source-result/criteria/destination/ALL/substance/18/source-type/ALL/subthreshold-data/Yes/substance-name/Cadmium%2B%2526%2Bcompounds/year/2022)

The National Air Emissions Inventory methodology [18] could be extended to include Cd emissions from the various sources, drawing on international data to provide a perspective on key sources of cadmium to ambient air in New Zealand. Many industrial sources do not exist in NZ, and bushfires are much less common than in Australia, so it would be expected that soil -derived sources – including unpaved roads, and motor-vehicles may be primary sources.

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

C2-(i). Local government driven

C2-(ii). Central government driven

To our knowledge, there are no interventions beyond general industrial emissions controls, and requirements for monitoring particulate matter under the National Environmental Standard for Air Quality are being used to affect this attribute. Cadmium is included in Worksafe standards and thus should be being managed in industrial workplaces.

C2-(iii). Iwi/hapū driven

Iwi/hapū planning documents such as Environmental Management Plans and Climate Change Strategies/Plans may contain policies/objectives/methods seeking to influence air quality outcomes for the benefit of current and future generations. We are not aware of other interventions/mechanisms being used by iwi/hapū/rūnanga to directly affect this attribute.

C2-(iv). NGO, community driven

C2-(v). Internationally driven

Internationally, the Geneva convention on long-range transboundary air pollution requires parties to reduce emissions of cadmium (and lead and mercury) from industrial sources, combustion processes and waste incineration. The protocol on heavy metals was amended in 2012 to introduce more stringent emission limit values (ELVs) for emissions of particulate matter and of cadmium, lead and mercury applicable for certain combustion and other industrial emission sources that release them into the atmosphere. The emission source categories for the 3 heavy metals were also extended to the production of silico- and ferromanganese alloys. New Zealand does not appear to be a signatory to this convention and hence has no obligations in this regard.

Part D—Impact analysis

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

There is likely negligible impact on human health of not managing cadmium in ambient air. Dietary intake and smoking will dominate people's exposure to cadmium with inhalation a negligible route of exposure for non-smokers. Cadmium is included in Worksafe standards and thus should be being managed in industrial workplaces.

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

Although uncertain, the expectation is that managing or not managing this attribute will have minimal economic impacts.

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

This attribute may be indirectly affected through changes primarily associated with changes in dust generated from soil as a result of climate related events.

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6 Terrestrial Domain

Eight attribute information stocktakes for the Terrestrial Domain are provided in sections 6.1 to 6.8, below. Dr Patrick Kavanagh and Fiona Hodge (MfE Domain Experts), Dr Duane Peltzer (MWLR, Domain Leader), and the Māori environmental researcher panel reviewed these sections.

6.1 Wetland extent

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State of knowledge of the "Wetland extent" attribute: No short answer given, see longer explanation in the following paragraphs.

Wetland extent is an attribute that can be considered at multiple scales. Wetlands are defined under the Resource Management Act 1991 (section 2): "*includes permanently or intermittently wet areas, shallow water, and land water margins that support a natural ecosystem of plants and animals that are adapted to wet conditions"*.

Extent can be taken to mean the area covered by something. Wetland extent might seem, therefore to be a simple attribute: the area that is covered by permanently or intermittently wet areas within some wider area of interest. However, ecologically, and culturally, how extent is defined (and therefore calculated) will be critical to whether it can provide *relevant* information. For example: for iwi, the historical extent of *an individual wetland*, compared to current extent, may be relevant to its ability to provide cultural provisioning (a form of ecosystem service); this metric may scale up to the historical extent of wetlands within a rohe, compared to current extent. Conversely, at the national scale, current wetland extent will be relevant for consideration of topics like current vegetation carbon stocks in wetlands. Conversely again, at the regional or sub-regional scale, it is likely that the current extent of wetlands as a function of historical wetland extent is likely to be most relevant for questions like flood risk.

Therefore, the answer to 'the State of Knowledge' of this attribute varies across which particular measure or spatial scale is considered. In general, and we note that often a historical attribute of a certain scale is sought to be compared to a current attribute at the same scale, but possibly not the same precision:

1. At the national scale, historical extent of wetlands is **unresolved**, and **inconclusive** at the regional scale and below. The work by Ausseil et al [1] is the most recent work on historical wetland extent, but this work was considered to be a first iteration and is based on superseded data. It has a minimum polygon size of 0.5 ha. It has wetland types attributed to it. The loss of 90% of NZ wetlands in Ausseil et al [1] is consistent with a previous grey literature estimate [2].

2. At the national scale, current wetland extent is mapped to the 1 ha scale (LCDB v5), but lacks a wetland type[1] attribute, making it 'meaningless' per [3] when considering wetland extent ecologically, given the huge variation in ecosystems that occur on wet substrates. As such, we consider this attribute to be **unresolved**. If the wetland type issue were addressed, we would consider this attribute to be **good**. There is a clear issue mapping forested wetlands, which would need to be addressed for this attribute to reach the level of **excellent or well-established**.
3. At the local and regional scale, current wetland extent is **unresolved**, given the difficulties in mapping small wetlands [5], although the current National Policy Statement on Freshwater Management requires that councils map wetlands that are not on public conservation land, down to 0.05 ha, and classify their type, will provide a major step forward in knowledge on extent. However, given that public conservation land is excluded, any catchment scale analyses (or similar) will be incomplete, where a non-trivial amount of public conservation land exists in the area of interest. As such, once regional mapping is complete, we would consider this attribute to be **established but incomplete**, unless and until public conservation land is mapped to the same standard.

Part A—Attribute and method

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

There is strong evidence for a positive correlation between current total wetland extent and ecological integrity, and human health at the catchment or larger scale. However, at the scale of individual wetlands, this is not be confused with biodiversity, as wetland extent may be negatively correlated with biodiversity values (i.e., small wetlands hold high biodiversity [6], [7], but when historically large wetlands become small, this may lead to ecological integrity loss [see below]).

In terms of ecological integrity, loss of historical extent has been linked to decreased ecological condition [8] and is an important predictor of ecological integrity [9]. Loss of wetland extent via drainage has been widespread in New Zealand [10], and drainage near remaining wetlands is surprisingly high [11] and is considered to have a negative impact on ecological integrity [12].

In terms of human health, there is strong evidence of the correlation between current extent of wetlands and ecosystem services that contribute to human health, such as provisioning of critical resources, such as food, fibre, and water, erosion regulating, natural hazard regulation (e.g., floods) [12], [13], [14], [15]; this is also recognised by the Ramsar Convention 1971, to which New Zealand is a signatory. The importance of wetlands to cultural human health has been recognised both internationally [15], and within New Zealand e.g., [16].

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

[1] The authoritative typology in New Zealand, Johnson and Gerbeaux's wetland typology [4], includes nine wetland types: bog, fen, swamp, marsh, seepage, shallow water, ephemeral wetland, pakihi and gumland, and salt marsh.

In terms of ecological integrity, the spatial impact of reductions in wetland loss is well documented in New Zealand, although there is some variation in regional historical losses [1]. Recent wetland loss in New Zealand, in the period 2012-2016, when assessed against multiple human pressure variables [17] was positively correlated with pasture areal extent around wetland edges at the national scale [34]. Evidence for impact on ecological impact has been documented for New Zealand in Ausseil et al. [1], who found that across New Zealand, where ecological integrity could be rated from 1, pristine, to 0, where 0 means complete loss of biodiversity and associated ecological function, that over 60% of wetlands in New Zealand were measured at less than 0.5 on the ecosystem integrity index. This indicated high levels of human-induced disturbance pressure and sustained biodiversity loss.

As wetland extent decreases, linkage to increasing flood impacts once historical loss was over 60% [18]. An international review has found strong evidence that floodplain wetlands reduce flooding [19], but that the evidence for reduction in flooding for other wetland types, evidence was mixed (p 366):

> *Most, but not all, studies (23 of 28) show that floodplain wetlands reduce or delay floods, with examples from all regions of the world. This same influence on floods is also seen, but less conclusively (30 of 66) for wetlands in the headwaters of river systems (e.g., bogs and river margins). A substantial number (27 of 66) of headwater wetlands increases flood peaks.*

Given the documented benefits of wetland extent, it follows that reduced wetland extent will result in reduced benefit; confirmed in a Florida analysis that found after controlling for other relevant variables, the number, type, and location of wetland permits to approve loss of wetlands was a significant predictor of flood damages (a measure of flood impacts) [20]. Additionally, most ecosystem service assessments are quantified on a per-hectare basis [14], [15], and as such, reductions in wetland extent will have a corresponding negative impact on ecosystem services and likely concomitant reductions in the provisioning of resources, but these declines have not been robustly quantified.

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

New Zealand is at the extreme of wetland loss globally; estimates of global loss have been estimated at 50%, with ‘extreme’ losses of >90% in parts of Europe [21]; New Zealand has been estimated to have lost 90% of its wetlands, although this varies across regions and by wetland type [1]. Recent work suggests that wetland loss has not slowed in Southland since the introduction of the RMA in 1991 (i.e., following the introduction of modern policy and law) [22]; and nationally, wetland loss in the period 1996 to 2018 has been estimated at 5,400 ha, most of which is now in high producing grassland indicative of dairying [10]. Losses differed across regions; by 2018, Gisborne had lost 15% of its 1996 extent [10]. As such, while the magnitude of estimates of recent loss differ among regions and time periods, it is clear that losses of wetland extent are continuing. While the NPS Freshwater Management requires mapping of wetlands down to 0.05 ha, which we expect will assist with identifying and reducing loss, recent uncertainty about a workable ‘wetland’ definition in New Zealand (*Page & Crosbie v Greater Wellington Regional Council* [2024] NZCA 51), and variable protection within current regional plans tempers any optimism [10].

Wetland restoration is possible, but varies in achievability by wetland type; humans typically value open water wetlands, which are just one kind of wetland [3]. McGlone [3] points out that talking about wetland loss, or wetland extent, of wetlands overall is meaningless, given the vast range of flora and fauna that inhabit different wetland types. Wetland types are described in Johnson & Gerbeaux [4]. One major issue is that the current land use of many lost areas of wetland is dairy farming, a high value economic activity. As the loss of wetlands is externalised from landowners to the surrounding catchment, but land use change costs with wetland restoration accrue to landowners, this represents a major socio-economic barrier to wetland restoration. That said, if emissions from drained wetlands were included in New Zealand's carbon accounting and accrued to responsible landowners, there are extensive areas particularly in the Waikato region that would be economically unfeasible to continue dryland farming. Technically, wetland restoration is possible in New Zealand, however restoration of certain functions, such as peat-forming and cessation of methane emission in restored peatlands, will take years [23], and therefore there will be a lag between restoration initiation and positive results.

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

Currently wetland extent is monitored at different scales. Nationally, there is the LCDB wetland flag for all LCDB polygons, with a nominal minimum polygon size of 1-hectare. This nominal size means that wetlands smaller than 1-hectare are most likely excluded. There are a vast amount of small wetlands in New Zealand [5]. There is also the LUCAS NZ LUM (Land Use Management layer) which maps two kinds of wetlands: Open Water Wetlands, and Vegetated (non-forest) Wetlands. However, the LUM excludes 90% of the mangroves around New Zealand (that are in fact mapped by the LCDB) and fails to disaggregate forested wetlands from other forests. Forested wetlands are known to be particularly difficult to map, as land cover does not necessarily indicate the hydrological status of the soils below.

At the regional scale, regional and unitary councils are required to map wetlands down to 0.05 ha under the NPS Freshwater Management. Guidance for nationally consistent methods of regional-scale delineation using aerial imagery is now available [35]. As at May 2024, we understand while elements of the NPS Freshwater Management are intended to be repealed, it is not our understanding that the mapping component for wetlands is included in the intended repeal.

Most mapping has been undertaken using desktop imagery, as the cost of field surveying for wetlands across the country is likely unfeasible. The best method to map forested wetlands is unclear given that LCDB relies on vegetation type, and forested wetlands often contain species that may also occur in drylands, such as kahikatea (*Dacrycarpus dacrydioides*) [24]. Forested and woody wetlands have suffered disproportionate loss following human settlement, and even where native dominance has returned to wetlands, the woody components have failed to recover [3], and as such, forested and woody wetlands are considered ecologically significant. Our assessment is that forested wetlands remain a critical knowledge gap in monitoring wetland extent.

As noted, the critical gap to effective monitoring of wetlands is probably forested wetlands at the regional scale, and failure to include mangroves and wetland forests within the LUM at the national scale. While LCDB maps wetlands at the national scale, it fails to assign a wetland type (as does the LUM); as McGlone [3] pointed out: *the term wetland is too broad to be a practical conservation category [4]. Wetlands are only united by their position of a saturated substrate and […] vary enormously on every other physical, biological, and historical dimension. [Statements relating to overall loss] are factually correct but largely meaningless from a conservation viewpoint.*

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

There are likely to be implementation issues. What these are/entail will be better known by regional councils.

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

Unknown: this depends on the minimum size wetland that is sought to be monitored, which will depend on the question that is sought to be answered. On the assumption that most monitoring is sought to be done at the regional and national scale, and therefore will primarily be undertaken with manual mapping using aerial imagery, supplemented by perhaps some modelling to identify candidate sites, we do note that change detection is more advanced of a field than object identification with respect to wetlands, and as such, once wetlands are mapped to 0.05 ha with some degree of manual intervention, a lesser degree of intervention may be required to monitor change. Recent guidance for nationally-consistent methods of regional-scale delineation using aerial imagery has been drafted and provided to MFE (February 2023; MFE report number not yet finalised; provided under contract 2324-23-003 A).

As noted above in section 'State of Knowledge', it is important to also identify wetland ecosystems uniformly to best facilitate conservation prioritisation. However, some wetland ecosystems are disproportionately difficult to measure using current techniques, and therefore, disproportionately under-mapped: this is the case for forested wetlands, where the tree canopy of species that may be present in both wetlands and drylands impedes the use of remote imagery to detect other wetland indictors. The current state of biased mapping is amenable to amelioration. For example, a combination of fieldwork in forested wetlands – and forested drylands, as well as modelling topographic and climatic wetland suitability e.g., [25], [26] could be used to validate that vegetation communities indicative of wetlands, as inferred on the basis of rainfall and topographic metrics, do in fact indicate wetlands.

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

We understand there are health indicators of existing wetlands being monitored. However, this monitoring may not necessarily include assessments of wetland extent. We refer to Te Reo O Te Repo – The Voice of the Wetland [36].

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

There are, as noted above, links between wetland extent and wetland condition, and the wetland condition index for New Zealand takes into account the historical loss of extent of wetlands in the catchment [8]. Additionally, ecosystem attributes that affect the flow through of water in a catchment will affect wetland extent – it is been found that early Māori burning of forests in New Zealand led to an *increase* of fertile, surface water fed wetlands, given the reduction in canopy interception and water uptake that had once occurred within the forests [3].

Part B—Current state and allocation options

B1. What is the current state of the attribute?

As noted earlier, the current attribute state at national scale is unresolved, because although national scale mapping via LCDB is relatively up-to-date (imagery date stated to be summer 2018/2019), it does not include wetland type, and as noted above, wetland type will influence both human benefits such as flood attenuation and extent is meaningless ecologically unless broken down by wetland type. We do not currently have a good understanding of the state at regional scale, because regional-scale mapping is not required to be completed until 2030 and will lack coverage of public conservation land. Our comparison of historical to current extent will be limited as the current historical extent layer is based on legacy (superseded) information. MWLR is currently developing a methodology to revise the historical extent layer to incorporate next generation soils data and LiDAR data, however, there is no funding to scale this to the national-scale.

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

The historical wetland layer exists [1], which estimates pre-human wetland extent and therefore could be considered to a reference state, although this layer suffers from the limitations identified in Question B1.

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

To our knowledge, no numeric or narrative bands have been described for this attribute.

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

Available information suggests that wetlands are vulnerable to tipping points, but it is difficult to generalise to the New Zealand context [27], [28].

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

Yes. Drainage is the major form of reduction in wetland extent in New Zealand [3]. Drainage near wetlands takes multiple years to reach ‘equilibrium’ [29] and even then, peat shrinkage rates continue on drained peatland in the Waikato particularly [30]. Furthermore, carbon sequestration may *apparently* continue in drain-affected wetlands, however this is due to woody plant invasion, which has a time-limited effect on carbon, unlike peat-forming species that sequester carbon into the soil [31]. The effect of drain may also take time to become apparent on the plant community, and as such, there are many wetlands around New Zealand that may be ‘under the influence’ of cryptic drain effects [11].

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

Mātauranga Māori is inherently place-based and so needs to be considered within a local context. Wetlands are valued by Māori as important systems. Discussions with iwi/hapū/rūnanga may reveal tikanga and mātauranga Māori relevant to informing bands, allocation options, minimally disturbed conditions and/or unacceptable degradation in treaty settlements, cultural impact assessments, environment court submissions, iwi environmental management and climate change plans, etc.

Part C—Management levers and context

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

As noted in B5, drainage is the major form of reduction in wetland extent in New Zealand [3]. Drainage has typically been undertaken to render the land suitable for primary production (although drainage and clearance for other land uses has also occurred to a lesser extent). Additionally, vegetation clearance, even where this is *near* but not *in* the wetland, impacts wetland extent while indirect linkages such as failure to enforce relevant council rules relating to drainage and vegetation clearance, insufficiently strong council rules relating to the same also impact wetland extent [10, 32].

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

At the national and regional scale, interventions do not appear to be effective, because wetland loss continues. However, it is not clear the scale of the *averted loss* (loss that has been avoided) as a result of these efforts. Furthermore, it is not clear whether restoration projects restore extent of wetland types that have been lost, rather than just ‘easy to restore’ types. As noted above, wetland type is critical in considering rates and states of loss; different wetland types are not interchangeable.

The following interventions are known and relevant:

C2-(i). Local government driven

Council rules to prevent clearance of wetlands.

C2-(ii). Central government driven

National Policy Statement for Freshwater Management (NPS-FM) and relevant subsidiary policy to prevent drainage of wetlands, and better map what wetlands exist.

C2-(iii). Iwi/hapū driven

Wetland restoration is often a key component of iwi and hapū driven initiatives.

C2-(iv). NGO, community driven

Catchment care groups often play a role in wetland restoration on private land.

C2-(v). Internationally driven

New Zealand is a signatory to the Ramsar Convention, which promotes the wise use of wetlands.

Part D—Impact analysis

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

Changes in the attribute state affect ecological integrity and human health as described in A1 above. Not managing wetland extent will likely lead to continued loss of extent. This will continue to contribute to species loss and displacement and stress ecological function, and impact on flood risk, as well as reducing opportunities for iwi to practice kaitiakitanga of areas that were considered *taonga* food baskets.

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

Economic impacts of wetland loss are likely to affect coastal communities, where they are protected from coastal storm events by wetlands [37], will suffer from increased impacts of coastal storms where wetlands are reduced – such as communities living around coastal mangrove wetlands. Where wetlands are reduced, communities and land owners such as farmers and crop growers in floodplains will be less protected by floodplain wetlands (or no longer protected, where loss is complete) – such as Hawke's Bay. Farmers where expensive flood mitigation schemes are required to replace the natural function of wetlands (such as Hikurangi catchment, Northland). Iwi who are already suffering from the estimated 90% loss of wetland extent in NZ and loss of cultural connection to wetlands.

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

An assessment of the effects of climate change on wetland extent requires a by-wetland-type analysis, as the relevant risks and their magnitude differ. We consider impacts of wetland by type:

- Mangroves may *shift* in distribution, may migrate south in New Zealand with warming water; they may also shift in response to sea level rise (reductions in seaward extent, increases in landward extent).
- Remaining wetlands that receive overland flow (swamps, marshes, fens) may suffer increased sediment and nutrient deposition if the increase in extreme events comes to pass; this may lead to wetland loss, if not reductions in ecological integrity.
- Intermittently closed and open lagoons may be lost under increases in sea levels.
- Coastal wetlands may be lost under increased sea levels, with little room to 'migrate' inland if there are incompatible land uses.
- The bogs in the Waikato are already a climatic oddity [3], [33] and their resilience to a warming climate is unknown. It is possible the peat-forming species will be unable to persist in a warming climate, and therefore the vast bulk of carbon sequestration will cease (depending on the replacement vegetation community).

- Bogs in the Waikato are also under threat of climate change due to salinization, as peat subsidence (due to drainage).
- Across all wetland types, shifts in rainfall patterns may affect wetland extent.
- Across all wetland types, it is unclear to what extent extant wetlands in New Zealand suffer from drainage impacts [11] and the extent to which increased stress via climate change will lead to wetland loss of extent.

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6.2 Dune extent

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Preamble: Aotearoa has multiple types of dune systems that are broadly characterised based on the origin of sand, location (e.g., coastal vs. terrestrial), and the physical activity that led to the structure of these systems (e.g., active[1], stable[2], volcanic[3], inland[4]). While all dune systems in Aotearoa are important and endangered, we interpret this attribute to predominantly refer to coastal active and/or stable sand dunes as these systems have received the greatest share of attention relating to ecological integrity and because our professional expertise is within the estuaries and coastal waters domain. However, the issues and pressures related to the decrease of coastal dune extent can be broadly applied to all dune systems. Also note that dune 'extent' is encompassed under dune 'condition index' given extent is one indicator of dune condition.

State of knowledge of the "Dune Extent" attribute: Overall, we consider the state of knowledge for the dune extent attribute to be 'good / established but incomplete' (though this may need to be changed to poor / inconclusive or medium / unresolved if considering all dune systems). Internationally and nationally, there is excellent evidence relating dune extent to ecological integrity. New Zealand-specific data that quantifies stressor impacts on 'dune extent' and associated ecosystem services are good, and management interventions for coastal dunes are well known (though this may not be the case for volcanic or inland dune systems). Nationally, a standardised protocol for monitoring coastal dune condition exists (which encompasses dune extent), however to our knowledge this has only been adopted for a handful of councils and data on tipping points are lacking. Monitoring of dune extent is not routinely carried out across the country, leading to a lack of national-scale data for comparison of dune extent growth and/or loss.

Part A—Attribute and method

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

[1] See link: Active sand dunes » Manaaki Whenua (landcareresearch.co.nz)
[2] See link: Stable sand dunes » Manaaki Whenua (landcareresearch.co.nz)
[3] See link: Volcanic dunes » Manaaki Whenua (landcareresearch.co.nz)
[4] See link: Inland sand dunes » Manaaki Whenua (landcareresearch.co.nz)

There is excellent evidence globally and in Aotearoa New Zealand (hereafter Aotearoa) to show that dune extent is closely tied with ecological integrity. Dunes are highly energetic habitats that contribute to coastal protection[1-3], support endemic biodiversity[4-6], and biocultural practices[7, 8]. Dunes are a common feature of the landscape throughout Aotearoa[9] but may be most conspicuous along coastlines[1]. Their existence at the land-sea interface makes them important for terrestrial, freshwater, estuarine and nearshore coastal ecosystems.

Nationally, coastal (and terrestrial) dune habitats are endangered and also support various threatened and critically endangered plant and animal species including a number of arachnid[4], lizard[10], and bird species[5, 11]. Dunes serve several physical functions that support the ecological integrity of coastal systems, such as shoreline protection from storm surges, coastal erosion and flooding[1, 3, 12]. Additionally, dunes play a crucial role in nutrient cycling (e.g.,[13]), soil formation (e.g.,[14]), and water regulation (e.g., [15]).

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

Globally and nationally, there is strong evidence of the impact of degraded dune extent on the ecological integrity of coastal systems (e.g., globally[16-22], for Aotearoa[12, 23-27]). Fragmentation and loss of coastal dunes from stressors like coastal development, beach renourishment programs, fire, recreation, invasive species, and climate change has led to a severe reduction in dune extent (i.e., between 60 and 80%, [28, 29]). Notably, this has impacted the national-scale loss or severe reduction of dune habitat for various threatened, endangered and critically-endangered spider and bird species, such as the Katipō spider (*Latrodectus katipo*) and the New Zealand fairy tern (*Sternula nereis davisae*), respectively[4, 5, 30]. In addition, the incursion of invasive plant (e.g., marram grass, *Ammophila arenaria*, [31]) and animal species (e.g., rabbits, [32]) has led to the displacement and loss of native dune flora (e.g., Pīngao, *Ficinia spiralis*; Spinifex, *Spinifex sericeus*), altering dune structure and promoting coastal recession and the loss of native bird nesting habitat [30, 33-38].

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

Both globally and nationally, coastal dune extent has declined significantly over time, particularly due to historic land reclamation (for agriculture, forestry, and/or development, [39, 40]), coastal infrastructure development / hardening (e.g., groins, seawalls, and dykes, [41-43]), livestock grazing [32, 40], recreation [44, 45], and invasive species [31]. Nationally, this is also the case for volcanic and inland dune systems, the latter of which were once regarded as 'useless sand wastes' (see the link at footnotes 3 and 4 for further details)[46-48]. It is likely that over the next 10 – 30 years, the interactions among sustained stressors will continue to reduce dune extent (for all types of dunes, globally and nationally), which also directly threatens the habitats supported by dune systems (e.g., dune slacks, dune deflation hollows, and/or damp sand plains).

While the multitude of stressors are actively interacting to reduce dune extent (to varying magnitudes based on location), most can be considered reversible and many are being managed, to

[1] Other dune types exist throughout Aotearoa, such as volcanic dunes formed from volcanic sediments and inland dunes formed from riverine sediments. Generally, these dunes are uncommon, in part due to decades of land-use change, which have made them endangered and critically endangered nationally.

some degree, by locally-led management and restoration programs (e.g., [49-51]). However, many of these dunes (especially near urban or developed areas) are generally in a degraded state (e.g., moderate to poor, [38]). Furthermore, natural dune recovery is highly variable, depends on sediment supply, the presence of stabilising native plant species, reduced physical disturbance (i.e., from humans, livestock, and/or pest species) and may not fully recover without additional interventions, such as planting of native flora (which can take up to 2 years for rearing plant propagules[52, 53]). This means that retaining or improving dune extent will be heavily dependent on effective legislative action that affords dunes adequate protection, monitoring, risk mitigation, and restoration where needed.

Climate change is also predicted to impact dune extent and stressors associated with this are expected to exacerbate over the next 10-30 years[54]. See Section D3 for climate change impacts and management actions. It should be noted that, internationally, the expansion of dune extent as a result of climate change may also be a wider sign of desertification (e.g.,[55]). However, we are not aware of any evidence of this occurring for Aotearoa.

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

In Aotearoa, active dune extent has been determined (albeit infrequently) from coastal monitoring programmes, published topographic maps, aerial and satellite imagery mapping [29]. We are aware of one standardised monitoring protocol for dune extent throughout Aotearoa produced by the Coastal Restoration Trust of New Zealand[56-58]. This protocol is used by a handful of regional/city councils to determine the state and condition of dunes within their management areas (e.g., Greater Wellington[59], Bay of Plenty[56], Northland[1], Canterbury, Christchurch). Additional councils monitor dune condition as part of their state of environment monitoring, which includes pressures (e.g., livestock, mammalian pest species, human impact) and the ecological state of dune systems (e.g., indigenous animal dominance, indigenous and non-indigenous land cover; e.g., Hawkes Bay Regional Council [60]).

Some work has trialled the use of remote sensing (e.g., aerial and satellite images) to estimate dune extent (with respect to dune condition), however, there often remains a need to ground-truth these data as there is often interest to include the extent and coverage of native plant cover[61-63]. Furthermore, these broad scale mapping methods may be limited for identifying the spatial extent of active and stable sand dunes due to high variation in classification methods for land types, making accurate assessments of extent difficult (e.g., [64, 65]). However, technological advances may help improve the accuracy of this type of data collection with time (e.g., as suggested for UAVs, [66]).

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

In general, mapping dune extent can be achieved through remote sensing platforms (for more see section A4-[i] and [62]). For this, there is a need for technical expertise such as mapping/GIS skills. However, some extent monitoring methods may require on-the-ground fieldwork (i.e., for ground-truthing). For ground-truthing, access is a key consideration given that some dunelands are located on private lands and are therefore subject to the landowner's property and/or customary rights (e.g.,

[1] See link: Coastal State of the Environment monitoring - Northland Regional Council (nrc.govt.nz)

for sites within or near Marae or Urupā). Accessing private property without the owner's consent can be considered trespassing, so clear communication, establishing good relationships, and addressing any concerns or impacts on the landowner's property or operations will be necessary. Formal access agreements or contracts may need to be established. It is possible that some dunes are not permitted to be accessed during certain times of year due to ecological factors such as nesting of rare birds.

Various health and safety factors also need to be considered in relation to fieldwork. These include access to the dunes and whether a 4-wheel drive vehicle is required for transport. Depending on the monitoring method being used, technical expertise such as plant species/taxa identification may also be required.

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

Most costs of monitoring dune extent will relate to personnel time spent collecting data, mapping and reporting results, which can cost more than tens of thousands of dollars of labour time depending on size of the dune (estimated pricing for remote mapping of vegetation structural classes for unmapped dune systems, [61]). Key equipment includes a computer set-up with sufficient computational power to support ARC GIS or equivalent software (estimated as $1000 - $3800 total).

The costs associated with ground-truthing are likely similar to those proposed for surveying estuaries. For example, in 2002 the approximate cost to survey one estuary (for all substrate and vegetation types; this could be analogous to dune extent) following NEMP was estimated to be between $15,000 to $30,000[67]. However, this cost was dependent on the size of estuary (*or dune system*) and whether suitable aerial photographs were available or needed to be obtained for the survey. The approximate cost now (to account for inflation and technological expenses) will likely differ.

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

Māori indicators for dune monitoring mainly relate to the presence of native flora (e.g., such as Pīngao) and fauna [102].

We are currently unaware of any monitoring of dune extent being undertaken by iwi/hapū/rūnanga.

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

Active and stable dunes are part of larger, continuous coastal habitats. This means that unobstructed connections with other habitats leads to higher habitat quality and ecological functions than isolated or fragmented dunes, all of which relates to 'dune condition index' and 'landscape connectivity'. Dunes that offer limited 'access to natural areas' (specifically in relation to human disturbance) may also have less impact to extent and support more diverse ecological communities. However, this does not exclude the potential impacts of stressors on adjacent attributes such as 'wetland extent' or 'surface water flow alteration', which can influence the formation of dunes.

Dunes are often found between multiple ecosystems, meaning there will likely be a crossover in monitoring methods for 'dune condition', 'salt marsh quality', 'seagrass quality', 'lowland forest extent', 'mangrove extent and quality', and, to some extent 'beach litter'. In addition, the 'wetland condition index' and 'wetland extent' are applicable to dune-associated wetlands, such as dune swales [68, 69]. Provided there is sufficient data resolution to assess dune condition (i.e., that can suitably measure indigenous vs non-indigenous species), methods for monitoring 'indigenous plant dominance' in the terrestrial domain may also overlap with those used to measure dune extent.

Part B—Current state and allocation options

B1. What is the current state of the attribute?

There is substantial evidence that dunes have been lost throughout Aotearoa since initial surveys of dune systems in the early 1900s [23, 25-27, 29, 44, 70-72]. The current state of dune extent is well understood at some regional levels (e.g., Hawkes Bay Region [36, 73]) and is reasonably well understood nationally (e.g., [28, 29, 74]). However, there is some indication that, with the spread and intensification of human activity around Aotearoa, the current (as of 2024) extent of dune systems remains unknown (as suggested by [28]).

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

The active nature of sand dunes (i.e., resulting from processes of accretion and erosion) means that assessing natural reference states for dune extent is likely difficult, if not impossible, due to the inherently variable nature of dune systems. However, dune systems that have retained their historical condition and that have limited to no introduced plant species or evidence of human-induced impacts (e.g., vehicular trampling, livestock grazing) could be considered a reference state. Dunes found in remote, protected locations such as those within or in association with national parks may best serve as examples of natural states with limited to no impact from human-induced stressors. However, sites within remote, protected areas may still contain stressors like introduced weeds and mammals and may still be subject to climate change impacts.

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

To our knowledge, there are no known numeric or narrative bands that specifically describe dune extent. The use of a dune condition index (which includes aspects related to dune extent) has recently been included in State of the Environment monitoring for a number of councils throughout Aotearoa[38, 60]. The condition index used by these councils includes indicators for pressures to dune systems (e.g., livestock, mammalian pest species, human impact) and the ecological state of dunes (e.g., indigenous animal dominance, indigenous and non-indigenous land cover). Each indicator is given a score between zero and five with a low score representing negative condition (e.g., low indigenous vegetation cover, high foot traffic) and a high score representing positive condition (e.g., high indigenous vegetation cover, limited physical disturbance). The scores are then added up and compared against a possible maximum score to determine overall condition. Additionally, the narrative bands used to monitor wetland condition index (which includes dune wetlands) could be modified to inform bands for dune extent[69].

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

Tipping points for dune extent have been reported internationally for coastal systems (e.g., 0.25 m of sea level rise, [54, 75, 76]), but data for Aotearoa is lacking. The tipping points reported internationally depend on factors such as dune condition, sediment budget, wind intensity, frequency of storm events, and sea level rise (e.g., [54, 76]). For example, increases in variables such as wind speeds, wave action and sea level rise can increase dune erosion resulting in a deficit of sand trapping, which can signal ongoing loss of dune extent [77].

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

Lag time between stressor and impact on dune condition will be site- and stressor-dependent. For example, there may limited to no lag time in cases of direct impact and severe physical damage, such as coastal development or recreational vehicles[78]. Alternatively, lag times are expected from the impacts of stressors such as sea level rise due to relatively slow encroachment. Additionally, there are lag times expected from the impact of non-indigenous plant species where there will be a time when these exist as seeds/seedlings before becoming established and spreading. There may also be lag times following coastal development and/or alterations to hydrological flow regimes, which can influence sediment budgets for dune systems[79].

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

Mātauranga Māori is inherently place-based and so needs to be considered within a local context. Dunes are valued by Māori as important systems that provide resources for cultural practices (e.g., collecting Pīngao for weaving) and as habitat for taonga species. In addition to discussing this attribute directly with iwi/hapū/rūnanga, there is likely to be tikanga and mātauranga Māori relevant to informing bands, allocation options, minimally disturbed conditions and/or unacceptable degradation in treaty settlements, cultural impact assessments, environment court submissions, iwi environmental management and climate change plans.

Part C—Management levers and context

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

In cases where dunes are lost, for example from agricultural, forestry, or infrastructure development, there can be an obvious and direct detrimental relationship between the stressor and dune extent [39]. Furthermore, there is also some information for Aotearoa documenting the relationship between dune extent and other physical stressors such as vehicle damage, livestock grazing, trampling, and invasive species incursions[25, 31, 44, 72, 78, 80]. However, there are still challenges associated with disentangling interactions among multiple stressors, respective lag times, additional legacy affects, and overall dune extent. In addition, the impact of stressors on ecosystems is usually highly context-specific (i.e., place and history are very important) and so effective management and needs to understand and allow for that context.

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

C2-(i). Local government driven

A number of councils have active dune management / restoration plans, which may be part of larger shoreline management schemes (e.g., Northland, Auckland, Waikato, Hawkes Bay, Tasman, and Nelson[1]). For example, dune habitats are commonly roped / fenced off to exclude public access along many public beaches throughout Aotearoa to allow for dune recovery / persistence[2, 82]. These rope systems and associated signage discourage the public from trampling sensitive dune fauna and increase awareness of the ecological function of dunes (e.g., as nesting habitat for rare and endemic seabirds[5, 30]).

Key management interventions include duneland protection and the elimination or reduction of stressors. From a policy perspective, the RMA (1991) is a key piece of legislation that sets out how we should manage our environment. In addition, the New Zealand Coastal Policy Statement guides councils in their day-to-day management of the coastal environment, which specifically includes dune systems[2]. There are various other relevant government-related directions and management implementations, for example for biosecurity, climate change, wildlife, threatened species and national parks. Some local governments have also instituted specific bylaws for the protection and recovery of dune systems (e.g., vehicle use bylaw, [81]).

Active dune restoration is another management intervention that can be carried out to improve dune extent by community groups, councils, DOC, iwi/hapū or others interested in recovering dune habitat [e.g., [56, 58]. For example, a number of coastal care groups (e.g., Coastal Restoration Trust of New Zealand) are actively involved in re-vegetating dune systems with native flora (e.g., [58]). Dune species are grown in some commercial/specialised nurseries and are widely available for these kinds of restoration projects. There is also potential to consider dune restoration to enhance dune extent for flood and sea level rise mitigation as suggested by [83].

A number of local government-driven initiatives are present throughout Aotearoa aimed at restoring dune habitat. Examples include the restoration of endemic dune plant species such as pīngao and spinifex at sites around Timaru [50]; urban dune habitats along the Coromandel coastline [49]; and Ngarahae Bay, West Coast North Island [51]. Additional projects can be found at the websites for the Coastal Restoration Trust of New Zealand[2], Waikato Regional Council Coastcare groups[3], and in [84].

C2-(ii). Central government driven

Central government can provide key funding for the protection, conservation and restoration of dunelands. For example, the recently completed NZ SeaRise Te Tai Pari O Aotearoa[4] project that projects sea level rise around New Zealand could be used to help prioritise future restoration projects at vulnerable coastal areas. There is also potential to consider dune restoration for shoreline management and/or conservation plans supported by the Department of Conservation (e.g., the

[1] https://www.doc.govt.nz/get-involved/run-a-project/restoration-advice/dune-restoration/
[2] https://www.coastalrestorationtrust.org.nz/coast-care-groups/groups/
[3] https://storymaps.arcgis.com/stories/14b535daa5ae4aae820d1be774f740b7
[4] https://www.searise.nz/

Auckland Regional Council Shoreline Adaptation Programme[1]; Rakiura Conservation Management Strategy [85]).

C2-(iii). Iwi/hapū driven

We understand that Māori could offer protection to dune habitats through rāhui (e.g., Pakiri Beach, Auckland[2]). We are also aware of a small number of Iwi-led restoration projects for dune adjacent habitats (e.g., dune wetlands, [8, 86]). Iwi planning documents such as Environmental Management Plans and Climate Change Strategies/Plans may contain policies/objectives/methods seeking to influence dune quality outcomes for the benefit of current and future generations.

C2-(iv). NGO, community driven

A number of community-driven dune restoration projects exist throughout Aotearoa. A notable example includes the Coastal Restoration Trust of New Zealand, which has developed a comprehensive guide and monitoring scheme for dune restoration projects throughout the country[58]. Additional projects include the Native Forest Restoration Trust[3], DUNE, the Whāngaimoana Dune Restoration Group, and Onetangi Beach Dune Restoration, to name a select few (for a comprehensive list of coastal restoration groups see footnote 2 on previous page). International NGOs, such as The Nature Conservancy, are active in New Zealand[4] and provide support for conservation initiatives and nature-based solutions, which can include increasing dune extent through planting of native vegetation.

C2-(v). Internationally driven

Restoring the vitality of degraded systems (which include dune ecosystems) is crucial for fulfilling the UN Sustainable Development Goals and for meeting the targets of the UN Decade (2021-2030) on Ecosystem Restoration (UN-DER). Under the Convention to Biological Diversity (CBD), Aotearoa is required to have a national biodiversity strategy and action plan through which obligations under the CBD are delivered. Aotearoa has international climate change obligations such as those under the Paris Agreement. We understand that Aotearoa has also signed other free trade agreements (e.g., Free Trade) that require conditions around environmental management to be upheld. Additionally, Aotearoa is a signatory of the Ramsar Convention meaning it plays a part in the international effort to conserve wetlands, which includes dune slacks and lakes[87].

Part D—Impact analysis

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

Failing to manage dune extent poses a significant threat to coastal environments, triggering a cascade of ecological problems. For example, the loss of dunes can lead to a severe reduction or loss of coastal protection and habitat for critically endangered endemic species, which is reflected in the decline of bird populations[11]. Additionally, the loss of dunes can allow for saltwater intrusion into

[1] https://www.aucklandcouncil.govt.nz/plans-projects-policies-reports-bylaws/our-plans-strategies/topic-based-plans-strategies/environmental-plans-strategies/shoreline-adaptation-programme/Pages/shoreline-adaptation-plans.aspx
[2] https://www.localmatters.co.nz/news/tangata-whenua-closes-beach/
[3] https://www.nfrt.org.nz/reserves/oreti-totara-dune-forest/
[4] https://www.nature.org/en-us/about-us/where-we-work/asia-pacific/new-zealand/stories-in-new-zealand/our-work-in-new-zealand/

coastal aquifers and wetlands, which can substantially alter ecosystems, leading to further loss of endemic species[88]. This influx of seawater may disrupt the delicate balance of coastal marine life and can sever vital links in the coastal marine food web, which can have cascading impacts on the overall health and biodiversity of coastal ecosystems[89]. Reduction in dune extent will also likely lead to increased degradation of adjacent shoreline habitats.

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

The economic impacts of dune loss are likely to be felt among coastal infrastructure development and tourism sectors. Reductions in dune extent could lead to a loss of habitat for critically endangered bird species, which limits coastal tourism opportunities for certain groups (e.g., birders) and local businesses (e.g., [90]). Reductions in extent of dunes can also limit their protective capacity as natural buffers that absorb wave energy and lessen the impact of storm surges[3, 12, 23, 91]. The loss of dunes exposes coastlines to increased erosion, leading to a retreat of beaches and a heightened risk of damage to coastal infrastructure and sensitive adjacent habitats such as dune slacks[18, 21, 92].

Many marae are at or near sea level, meaning a reduction in dune extent could severely increase the vulnerability of culturally-important sites, which also include urupā[1],[93]. The loss of dune systems and habitats associated with these systems can directly influence tikanga practices, such as kai gathering, which can diminish mana over associated resources and / or areas [94-96].

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

Sea level rise and increased storm frequency is expected to lead to the erosion and loss of dune habitat extent [88, 89, 97, 98]. Sea level rise may also result in reduction in dune extent due to 'coastal squeeze' if habitat is not available for it to migrate to due to the presence of roads, urban areas, stopbanks, or agricultural land directly inland from current dunes [84, 99]. Increased storm frequency will likely lead to increased flooding, which will likely impact coastal sediment budgets and coastal erosion processes [92, 100]. Changes to vegetative cover, at times due to increasing temperature, can alter dune faunal community structure and lead to further range shifts and incursions of invasive species. In addition, increasing temperatures may also reduce below-ground biomass for certain dune vegetation species, which can reduce dune extent by diminishing accretion and subsequent stabilisation processes (e.g., as seen in China[101]).

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6.3 Lowland forest extent

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Preamble: This discussion is focussed on lowland *indigenous* forest extent and does not focus on exotic forest. It should also be noted that there is no universal definition of lowland: 100m, 200m and 400m are all used in the literature. It is difficult to provide ecological comment on the relationship of this attribute to ecological integrity (in QA1) if the IPBES definition is used. The definition is meaningless in a New Zealand terrestrial ecological context. For example, a wholly exotic plant community (such as ryegrass pasture) could have high ecological integrity if it could maintain its processes and community of organisms. I therefore base my response on the definition arrived at[1] in relevant discussions in the NBEA development process over the last few years. Key points are that ecological integrity cannot be site-based (as it is in the NPS-IB, where it was envisaged to apply only to a particular site) and it must include representation as well as attributes that relate to species occupancy (all species that should be present, are present) and native dominance (indigenous species predominate in composition, structure, and process).

State of knowledge of the "Lowland forest extent" attribute: Good / established but incomplete – general agreement, but limited data/studies

Section A—Attribute and method

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

Survival of native vegetation has been uneven across New Zealand's landscape, with the most extensive tracts of native forest occurring in environments unsuited to intensive human land uses because of their cool temperatures, high rainfall, and/or steep terrain [1,2]. The greatest clearance occurred in warm, dry climates on lowland and mid-elevation landforms prone to fire and/or suited to agriculture such as the Waikato, Manawatū, and in the east from East Cape to Southland. Most of

[1] ecological integrity means the ability of the natural environment to support and maintain the following:
(a) representation: the occurrence and extent of ecosystems and indigenous species and their habitats; and
(b) composition: the natural diversity and abundance of indigenous species, habitats, and communities; and
(c) structure: the biotic and abiotic physical features of ecosystems; and
(d) functions: the ecological and physical functions and processes of ecosystems

NZ's lowland indigenous forests have now been cleared, and few extensive lowland forest tracts remain outside the wettest lowland environments of Westland and Fiordland (see evidence below).

Lowland forests are characterised by relative warmth, moisture and productivity [2,3], and therefore support different suites of species from those in higher, drier and lower-productivity environments. The biological character of lowland forests in the past would also have varied greatly around New Zealand in response to differences in environment. Loss of lowland forests has therefore led to poor representation (a key attribute of ecological integrity) of the indigenous biota and ecological processes that once occupied and occurred in New Zealand's forest ecosystems. In particular, the northern warmer and eastern drier lowland forests which once occurred in New Zealand are now extremely poorly represented.

Indigenous species that could potentially have occurred in the lowlands no longer occur there (loss of species occupancy). For example, loss of forest, and not predation, is the primary limiting factor for indigenous forest birds in most of lowland New Zealand [4], and the same applies for all other forest-specialist biotic groups.

Composition and structure of many remaining indigenous forests has been modified by logging, and/or altered by predators and weeds. In many cases species, processes and ecological functions have been lost, and/or displaced those that are different and non-indigenous. Native dominance of lowland ecosystems has therefore also decreased and in many environments has been lost entirely.

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

There is no universal definition of lowland, and Figure 1 below shows three possible definitions based on elevation (below 100m, below 200m and below 400m).

Spatial databases show that NZ has lost 90% of its pre-human indigenous forests below 100m elevation, 86% below 200m and 80% below 400m (Fig. 1). As the elevation of a chosen contour increases, the amount of 'lowland' forest remaining increases from 10% to 20% (Fig. 1), because the expanded lowland zone includes proportionally more land that has historically been less suitable for human habitation and exploitation.

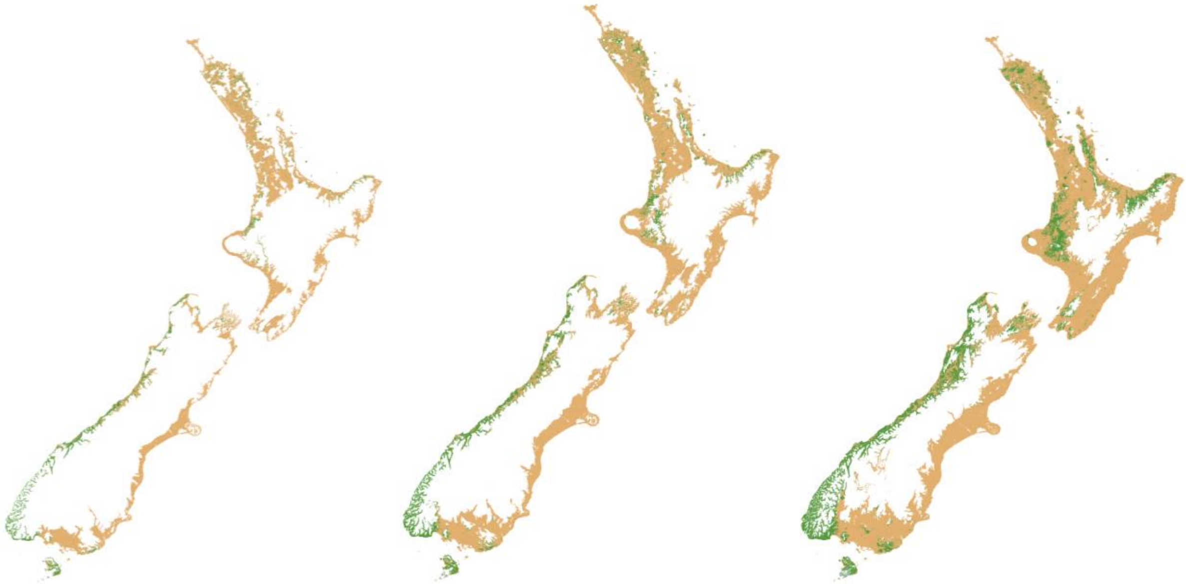

Figure 1. Three different extents of lowland areas defined by elevation (0-100m above sea level left, 0-200m centre, and 0-400m right). The maps show the extent of indigenous forest remaining (green, based on the Ecosat forest layer [5]) and those areas which were forested in prehuman times (assumed to be the extent of the 20 forest classes in the Potential Vegetation of New Zealand dataset [1]) and have now been cleared (brown).

Whichever contour is used to define lowland, most of New Zealand's lowland indigenous forest has been cleared and the majority of lowland forest remaining is in the west and south of the South Island, which are areas that have historically been environmentally unsuitable for habitation or exploitation by humans.

The 'right' contour cannot be chosen using ecological characteristics, because these vary greatly with latitude as well as elevation around New Zealand [3], and it is unlikely that a threshold in any forest ecological characteristic occurs at or near any particular contour.

This is illustrated by Table 1, which shows that the narrowest possible zone (defined by the 100m contour) has the highest proportions of the pure 'Podocarp forest' Ecosat class [5], the 'Unspecified Indigenous forest' class, and the 'Coastal forest' class. The representation of these three classes decrease as elevation increases and the zone increases in size, while the representation of 'Kauri forest', 'Beech forest', and three mixed beech-podocarp-broadleaved classes increase. All classes of Ecosat forest occur in lowland zones irrespective of the contour; it is simply the proportions that change.

Some considerations for choosing a lowland zone are:

- the breadth of ecological characteristics that the attribute is intended to capture, and
- whether the indicator should focus on the narrow elevation zone where the most loss has occurred and national representation of the ecosystems therein is lowest, or whether slightly better represented forests are also relevant.

Table 1. The composition (percentage of Ecosat vegetation classes) of the remaining indigenous forests in lowland zones defined by different lowland contours (100, 200 and 400 m above sea level).

	Lowland contour		
Ecosat vegetation class	**100 m**	**200 m**	**400 m**
Subalpine scrub	0.2	0.3	0.7
Coastal forest	0.6	0.4	0.2
Kauri forest	1.3	2.3	2.9
Podocarp forest	9.4	4.7	2.2
Podocarp-broadleaved forest	29.6	30.3	28.8
Beech forest	5.0	5.8	10.0
Broadleaved forest	3.7	5.8	6.6
Podocarp-broadleaved / Beech forest	13.6	17.3	19.8
Beech / Broadleaved forest	1.2	1.5	1.6
Beech / Podocarp-broadleaved forest	11.9	13.8	15.5
Unspecified Indigenous forest	23.6	17.8	11.6

Lowland indigenous forest extent is strongly linked to all components of ecological integrity. So declines in extent and fragmentation of these ecosystems will reduce ecological integrity. Declines the size and number of lowland forests directly reduces **representativeness** of indigenous-dominated vegetation. The composition of remaining forests is affected by losses of indigenous taxa, but also increased threats from weeds, pests and diseases from fragmentation itself, and proximity to other land uses. Similarly, the structure of lowland forests differs from intact or baseline forests because of edge effects (e.g. more wind, larger temperature fluctuations, reduced moisture) and increased disturbances such as grazing, selective harvesting of tree species, and invasion of weeds in the understory. All of these changes affect multiple ecological and ecosystem functions, which are rarely quantified (e.g., Didham et al. 2015).

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

There was a pulse of deforestation by fire following human arrival, and another when Europeans arrived and settled [6]. The easily exploitable resource ran out more recently.

Compared to historical clearance rates, there has not been a high rate of lowland primary forest clearance since the 1990s [7]. Nevertheless, some areas continue to be cleared in diverse environments, for example on private land on the South Island west coast, and in urban Auckland. Cumulatively, Dymond et al (2017) [8] estimated that about 10,500 ha of indigenous forest was cleared across all of NZ between 1996/97 and 2012/13 (15 years). This suggests that the status quo is for 7,000 to 8,000 ha of indigenous forest to be cleared nationally per decade. Dymond et al (2017) [8] do not provide an estimate of how much of the recent indigenous forest clearance has been of *lowland* indigenous forest, but this could be calculated given a defined 'lowland extent' spatial layer.

If New Zealand policy settings change to remove or weaken current legal protection for remaining lowland indigenous forest, it is reasonable to expect that indigenous lowland forest clearance will accelerate again. However, there likely to be only so much now that it would be economic or socially acceptable to clear, in the author's experience.

Forest stewardship certification (FSC) schemes theoretically help to incentivise protection for indigenous remnants embedded in exotic forest plantations, although the extent to which those schemes have protected indigenous remnants in practice is unclear, because outcomes are rarely measured in the author's experience. FSC protections are market-driven and would not necessarily be weakened if NZ policy or legislation changed.

There is now likely to be an increased rate of loss of extent of indigenous forest through more flooding, slips, and inundation, as climate change sets in [9,10]. For example, swathes of lowland forest are often removed by floods in South Westland – both along the river margins and at the river mouths as the estuaries switch back and forth. Recent changes in the lower Haast and Arawhata rivers in response to storm events have removed sizeable areas of primary forest, for example. Cyclone Ita blew down sizeable areas of west coast forest. Some of this loss would have occurred historically, and then regenerated again, but there may be a step change in scale now, along with a lower likelihood of eventual recovery of forest structure and composition in the changed context of predators, ungulates and weeds.

Mature indigenous lowland forests are centuries old and take at least many human generations to re-establish. Therefore loss of present extent is certainly irreversible within a human generation and given the current changed environmental context, is likely to be permanent.

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

LCDB mapping [11] of indigenous forest and of broadleaved indigenous hardwoods is the only data on extent that is being collected as far as I know. That data collection has been funded as part of NZ's land cover data investment. Those investments have been ad hoc and subject to changes in science and technical funding, and therefore may not continue in future. Larger councils such as Auckland Council and Waikato Regional Council may be collecting their own information, but the author has no knowledge of any such initiatives. There is no baseline monitoring of the background frequency of extreme events and their ecological consequences for lowland forest extent as far as the author knows.

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

Lowland forest can typically be seen in remote-sensing (satellite and aerial) images and its extent is mapped from these. Therefore there is little to prevent or impede implementation. The principal requirement is that New Zealand agencies keep collect or purchase the relevant remote imagery and process it; and the trend is for this to be becoming cheaper and easier.

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

If the resources for LCDB are to continue, most of the up-front costs would be covered and this attribute would not incur much additional cost. Additional costs would be in (ideally) online manual checking of each polygon identified as a loss or a gain, and the costs of extracting and curating the data into the indicator and report. An extent of lowland would need to be decided on, but once that decision is made the time required to create and clip the overlay is negligible.

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

There are few examples of lowland forest extent being monitored by Iwi/Māori, but the forest condition and tree health are being monitored as part of research programmes such as Ngā Rākau Taketake (https://bioheritage.nz/about-us/nga-rakau-taketake/). Both Kauri dieback and myrtle rust diseases have prompted recent work on Mātauranga Māori approaches to tree condition and surveillance largely in lowland forests. More generally, cultural frameworks to monitoring, including biocultural approaches (Lyver et al. 2019), do not focus on a specific attribute or metric like lowland forest extent, but could include this as part of more integrated assessment of forest condition.

There are many examples of hapū/iwi monitoring the health of indigenous forests (ngahere) utilising indicators drawn from mātauranga Māori. See for example Lyver et al. (2017a, b) and McAllister et al. (2019)[23-25]).

A specific example is the development and implementation of the ngahere ora framework to understand the health of the forest by evaluating health state using the perspectives and values of mana whenua. Data are able to be gathered digitally, using Survey123, which is then fed into ArcGIS Pro, a platform that allows creation of maps. This enables hapū/iwi to monitor the extent of forests as well as specific health indicators. (See Reihana et al. 2024 [26]).

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

There are likely to be similarities between mapping lowland forest extent and change and mapping wetland extent and change. Decreases in the extent of lowland forest will be strongly related to decreases in structural and functional connectivity (see landscape connectivity template). Canopy tree dieback may increase and indigenous plant dominance could decrease with lower lowland forest extent because indigenous tree species in these species are likely more exposed to weeds, pathogens or pests and disturbance in forest fragments (e.g., Didham et al. 2015). Lowland forest is usually easier to identify in remote images than wetland, but there may be efficiencies in doing both together (e.g., in the purchase and pre-processing of underlying imagery).

Part B—Current state and allocation options

B1. What is the current state of the attribute?

The current state of knowledge is good because LCDB [11] is available and has been updated quite recently [12]. Currently the total land area under lowland forest is far lower than before human settlement, but the magnitude of decreases varies widely among different forest vegetation classes (Table 1). Less than 2% of coastal and lowland beech/broadleaved forests remain, whereas about 30% of lowland podocarp-broadleaved forests remain (mostly due to relatively large remnant forest areas occurring in Westland). Across nearly all forest types, extent decreases at lower elevations (i.e., extent is smallest closest to the coast). However, as noted above, a 'lowland' definition would be needed before a lowland indigenous forest layer could be created from LCDB.

There are some matters that may need to be explored with land cover mapping experts. For example, there may be a question of whether to include 'broadleaved indigenous hardwoods' within 'indigenous forest'. The distinction between 'broadleaved indigenous hardwoods' with 'indigenous forest' in LCDB is not quantitative[1], and in practice decisions are made subjectively and manually by an operator on the day. And as LCDB is not produced totally *de novo* each time, there is probably a legacy bias (i.e., an operator may stick with a previous assignment, rather than make a change). Those administering the indicator may decide to accept those subjective and manual assignments and include one or both.

A related question is when a recovering (lowland) forest becomes an 'indigenous forest' rather than seral woody vegetation (such as broadleaved indigenous hardwoods), or (say) an induced wetland such as a pakihi. Again, those decisions are usually made subjectively and manually by an operator.

Consequently, quantitative analysis of changes in area between assigned classes may not yield reliable and reproduceable results.

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

Pre-human cover in the Potential Vegetation of New Zealand dataset [1] is the most suitable available dataset to use as a baseline for extent. Unlike some classifications [e.g., 12] it is objective and derived from reliable data [2], and the data and the assumptions used to construct it can be revisited in future if necessary. The 25 classes of the Potential Vegetation of New Zealand can be cut down to the extent of the 20 potential *forest* classes in that layer, omitting duneland, wetland, and other non-forest classes. This was the method followed to construct Fig. 1 (above) and as the basis for determining the present extent of indigenous lowland forests remaining.

Accurate and objective estimates of extent of lowland forest from the times of Māori or European settlement are lacking (to the best of the author's knowledge).

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

There are numerous papers in the international literature which discuss lowland forest extent and loss. With respect to defining 'lowland', a non-exhaustive search for 'which countries report on lowland forest extent?' revealed that 200 and 300m contours have been used [13, 14]. In Ecosat forests Shepherd et al. (2005) [5] distinguished classes of forest based on spectral signature (e.g.,

[1] Indigenous forest is defined as 'vegetation dominated by indigenous tall forest canopy species'.
While Broadleaved indigenous hardwoods are 'Typically found in high rainfall areas associated with Low Producing Exotic Grassland in hill country throughout New Zealand. However. the class also includes low-growing, coastal broadleaved forest. Characteristic is the presence of a mix of broad-leaved, generally seral hardwood species, such as wineberry (*Aristotelia serrata*), mahoe (*Melicytus ramiflorus*), *Pseudopanax* spp., *Pittosporum* spp., *Fuchsia* spp., ngaio (*Myoporum laetum*), and titoki (*Alectryon excelsus*), together with tutu (*Coriaria* spp.) and tree ferns. The presence of this class usually indicates an advanced successional stage back to indigenous forest. Canopy height ranges from 3 - 10m.

beech forests, kauri forests, various types of mixed forest). Those classes could potentially be reported on separately.

For forest extent, there are no quantitatively-defined thresholds based on data for habitat size, biodiversity or structure established within New Zealand. However, semi-quantitative ranks or qualitative goals or aspirations have been developed, for example;

- Setting a long-term restoration target of at least 15% cover over 100 years (Rout et al. 2021);
- The Threatened Ecosystem Classification (Walker et al. 2015) generated six categories based on indigenous vegetation extent (% remaining) and protection from the most highly threatened (Category 1: <10% indigenous cover remaining) to the least threatened (Category 6: >30% remaining and >20% protected);
- The NPS-IB requires that urban areas must have a target of at least 10 per cent indigenous vegetation, and more generally, no further loss of biodiversity or Significant Natural Areas.
- International standards often apply a 30% remaining extent goal for protection (e.g., The Kunming-Montreal Global Framework for Biodiversity Target 3: “Ensure that at least 30 per cent globally of land areas and of sea areas, especially areas of particular importance for biodiversity and its contributions to people, are conserved…”).

There may be other relevant precedents and existing practice internationally.

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

There are no reliable known thresholds or tipping points to the authors’ knowledge. However, it is likely that multiple important thresholds for ecological integrity have already been exceeded in most parts of lowland New Zealand. The exceptions are clearly the largely still forested (but narrow) coast of Fiordland and some lowland parts of the South Island west coast, where forests is still continuous across natural environmental gradients.

There has been no objective test of the applicability to New Zealand of the classic 30% tipping point for forest birds and mammals derived by Andren (1994) [14] in the northern hemisphere. In New Zealand, there are likely to be different tipping points and thresholds for different ecological components (e.g., for different biotic groups, habitat generalists vs specialists) [e.g., 15] and processes and functions (such as regeneration, or provision of breeding or feeding habitat), and those tipping points will also depend on ecological context [16].

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

It is likely that there will be many long lag times and legacy effects. For example, mature indigenous forests are old and therefore take centuries to re-establish, if they ever do given changed ecological contexts [17]. Deforested areas can undergo long lasting or permanent soil changes, and rates and patterns of sedimentation can change for decades or permanently. ‘Priority effects’ of weeds that invade while the sites have little or no vegetation cover [18, 19], and invasive ungulates [21], can have profound effects on long term trajectories. Seed sources are often missing or inadequate in cleared landscapes [22], along with the means to get propagules back into seral vegetation (e.g., key disperser species may be absent, or new ones may introduce unwanted propagules). Extinction debt (i.e. progressive loss of species which initially survived in remaining fragments) is likely to occur in lower forests, but has not been assessed to date. These lags and legacies will usually be so protracted that they are unlikely to affect state and trend assessment over the short-term, but can lead to ongoing declines in biodiversity despite protection or management of current lowland forests.

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

Although we cannot comment directly on māturanga Māori, we do provide suggestions from our experience that there should be condition or states of lowland forest extent or condition described from a te ao Māori perspective. More work is required here to understand place-based goals or aspirations for indigenous forests (e.g., [23]). As noted in A5, there are many examples of mana whenua utilising indicators that are derived from mātauranga-ā-hapū and mātauranga-ā-iwi to measure ngahere health. These indicators are based on knowledge within a local context. See for example [26]. See also the development of a forest health monitoring system with Tūhoe Tuawhenua utilising a Likert scale scoring system where 25 priority indicators were used to form the basis of a field survey approach to monitor forest health [24].

Part C—Management levers and context

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

The main 'stressors' on lowland forest extent through history have been a lack of conservation and environmental legislation and policy to protect lowland forests, or the non-enforcement of that legislation and policy where it did or does now exist. Aside from legal protect, declines in lowland forest extent are closely linked to multiple stresses and other attributes including:

- Historical, and in some regions, ongoing habitat fragmentation increases loss of biodiversity because of habitat size and loss of connectivity (see details in landscape connectivity attribute).
- Close proximity to more intensive land use increases impacts of pests, weeds and diseases, disturbance from wind, fire or grazing which can ultimately drive declines in biodiversity (see indigenous plant dominance and tree canopy dieback attributes).
- In many lowland forests, there are additional historical pressures from selective harvest of podocarps.

Many of these environmental stressors are well understood. Progressive increases in some drivers like extreme weather events under climate change and increasing number and abundance of environmental weeds could lead to further, cumulative, loss of extent [8,9].

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

C2-(i). Local government driven

C2-(ii). Central government driven

C2-(iii). Iwi/hapū driven

C2-(iv). NGO, community driven

Central and local government, iwi/hapū-driven, NGO and private and community driven interventions all apply, in the form of regulations and/or rāhui, and these vary widely across New Zealand depending on district and regional plans, administering authority and land status among other factors.

C2-(v). Internationally driven

It is not clear whether internationally-driven commitments drive local responses: the effects of those commitments and interventions are often indirect and New Zealand does not have experimental control areas to understand the likely trajectories in their absence. For example, it is unclear whether World Heritage status has translated into less active deforestation in South Westland, or whether changes in pace are due to economic or other factors such as social licence [18].

There has been no change in lowland forest extent in response to KunmingMontreal Global Framework for Biodiversity Target 3 as far as the author is aware.

Part D—Impact analysis

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

Assuming that not managing this attribute would result in further lowland forest loss, impacts would include further loss of indigenous plant and animal communities and species and the processes that sustain them, loss of stored carbon in soils and vegetation, changes in hydrology, changes in albedo, loss of flood protection. While beyond this author's expertise, indigenous forests are often of special importance to Māori, and there are spiritual, cultural, environmental and economic aspects to this.

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

Loss of carbon stored in the forests and their soils is likely to add to New Zealand's emissions, and any new land use replacing the forest is likely to be a net emitter of carbon. Those effects could be quantified. There is likely to be locally and/or regionally increased sedimentation and potentially also impacts of slash from deforestation, which may have impacts on agriculture and fisheries. Work could be done to clarify and quantify some impacts; for example, mapping could be used to determine whether areas under indigenous forest eroded and slipped less than exotic forests in cyclones, and research and modelling could quantify relative impacts on industries, businesses, and people.

Allowing indigenous lowland forest to be felled is unlikely to help the brand that export industries and tourism industries rely on. At a smaller scale there may be impacts on community wellbeing, recreation, and economic opportunities (e.g., local tourism) at multiple timescales (short-term to permanent).

This area would benefit from further exploration by economists.

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

As noted above:

"There is now likely to be an increased rate of loss of extent of indigenous forest through more flooding, slips, and inundation as climate change sets in (9,10]. For example, swathes of lowland forest are often removed by floods in South Westland – both along the river margins and at the river mouths as the estuaries switch back and forth. Recent changes in the lower Haast and Arawhata rivers have removed primary forest, for example. Cyclone Ita blew down sizeable areas of west coast forest. Some of this loss would have occurred historically, and then regenerated again, but there may be a step change in scale now, with lower likelihood of eventual recovery."

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6.4 Wetland condition index

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State of knowledge of the "Wetland condition index" attribute: Good / established but incomplete – general agreement, but limited data/studies

There is no 'one' wetland condition index: a condition index could be any combination of biological or biogeophysical chemical indicators that are considered relevant and are practical to measure and monitor.[1] However, there is a wetland condition index (WCI) used by some (not all) councils in New Zealand. The WCI is described primarily in the 'Handbook for monitoring wetland condition' [1] and updated by more recent single council-specific reports [2], [3] and discussed further in a report aimed at Tier 2 (regional) monitoring [4]. There are no known comparisons between the New Zealand WCI and other ground-based indices; however, there will be evidence and studies linking the relevant indicator components (Figure 1) to ecological integrity. The overall structure of the WCI is shown below in Figure 1.

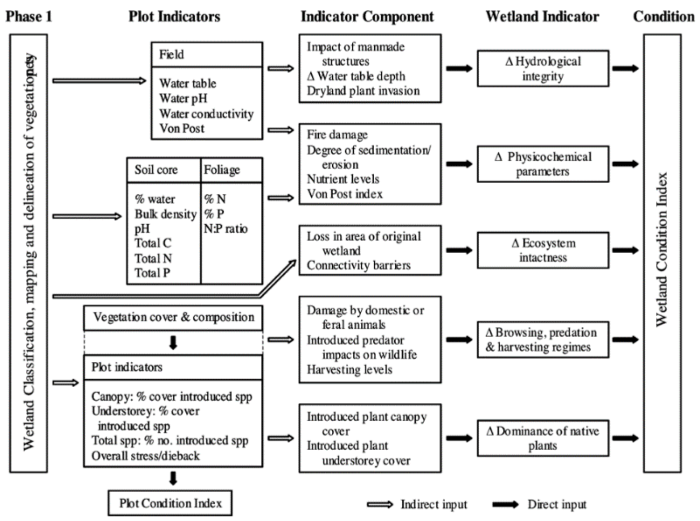

Figure 1: Links between wetland and plot indicators and Phase 1 of the Co-ordinated Monitoring of New Zealand Wetlands project

Figure 1. Structure of the WCI reproduced from the Handbook for Monitoring Wetland Condition [1].

[1] And in fact, assessing questions such as, how well an indicator such as 'ecosystem intactness' is related to 'ecological integrity' may end up being rather circular.

The New Zealand WCI is an example of 'multi-metric indices', where multiple attributes are combined to assess ecological or biological systems [5]. Ruaro et al. [5] note, "*differentiating natural variability from anthropogenic impacts is the major challenge in creating and applying [multi-metric indices]".*

A review (restricted to USA) of a 'rapid' condition assessment (2 people spending half a day in the field, plus a total of half a day office work thereafter) noted critical factors to consider with respect to wetland condition indices were [6]:

- Sampling boundary
- Inclusion of wetland type[1], and reference values for different wetland types due to their different ecological and hydrological settings
- Ideally the index returns one 'integrated' score
- Separate components for ecology and 'value-added components' – e.g., adding 'green space' in a very urban setting should be assessed separate to condition of the wetland itself
- Verification of the index with comprehensive ecological data.

The New Zealand WCI does not delineate separate reference values for different wetland types where there is natural variability, such as the Von Post index [3], which is expected to be scored lower in swamps than bogs, as the peat is naturally more degraded in swamps. This is an opportunity for the future. There was an interim attempt to identify quantitative limits for certain variables for wetlands, by wetland type, however, this was never followed up, and lacked data for certain wetland types [8].

Councils have raised concerns with the 'subjective' nature of the qualitative scoring component of the WCI. There is a lack of knowledge on how much variability in condition scores are due to operator differences, particularly across regional councils or other institutions. Differences in judgement affect even quantitative metrics, such as plant cover, and are ideally (a) minimised and (b) the residual accounted for, in assessing differences in wetland condition [9].

There is also a GIS-based analysis of wetland condition, the EII, described in Ausseil et al [10]. It provides a value where ecological integrity could be rated from 1, pristine, to 0, where 0 means complete loss of biodiversity and associated ecological function, that over 60% of wetlands were measured at less than 0.5 on the ecosystem integrity index. This indicated high levels of human-induced disturbance pressure and sustained biodiversity loss. A recent study [11] has found a reasonable correlation ($r^2 = 0.66$) between the WCI (Clarkson et al 2004) and the EII [10].

Internationally, other countries have also developed wetland condition indices, such as Australia [12], South Africa [13], the USA [14]. Remote-sensed metrics are beginning to be developed [15]. Here we primarily consider the established WCI as a potential attribute of Ecological Integrity.

[1] Wetland type such as bog, fen, swamp. Refer to [7] for wetland types of New Zealand

Part A—Attribute and method

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

The New Zealand WCI is a tool used to assess the ecological health and integrity of wetland ecosystems [1]. Although the WCI is an ecological measure, there are several indirect links between wetland condition and human health:

- Hydrological function of the wetland (a component of the WCI – refer Figure 1) will affect ecosystem services provided to humans (e.g., flood risk mitigation). An assessment of wetland values [16] notes their critical role in the water cycle, thus providing water security for humans.
- Soil nutrients and other variables will indicate the extent to which a wetland is able to provide water filtration ecosystem services – clean water is a key element of human health. Ecosystem services reports set out the value of ecosystem services provided by wetlands in New Zealand [17, 18].
- Climate regulating services are linked to human health via avoidance of heat-related illnesses and spread of infectious diseases. Peatlands (e.g., bogs and peat swamps) are a critical stock of stored carbon, and ecologically intact peatlands continue to sequester carbon [19], [20].
- The WCI is designed to assess ecological integrity, with indicators such as exotic species abundance, soil nutrients, and impacts of introduced herbivores and predators [1]. A recent study [11] has found a reasonable correlation (r2 = 0.66) between the WCI [1] and the wetland ecological integrity index (EII; [10]).

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

There is little field-based assessment of the New Zealand wetland condition index against ecological integrity (which would be circular if we accept that the WCI is meant to represent components of ecological integrity) or human health.

As noted above, the WCI is positively correlated with the EII, and evidence for impact on ecological impact has been documented for New Zealand in Ausseil et al. [10], who found that across New Zealand, where ecological integrity could be rated from 1, pristine, to 0, where 0 means complete loss of biodiversity and associated ecological function, that over 60% of wetlands were measured at less than 0.5 on the ecosystem integrity index. This indicated high levels of human-induced disturbance pressure and sustained biodiversity loss. This survey was GIS-based, no field work involved.

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

There is a limited published literature on the 'attribute' (wetland condition index) and rates of change within it, with 'Grey' reports that set out states and trends in WCI for specific parties/sites [36] and the Clarkson et al report [8] which linked environmental indicators to differences in wetland condition indices across space being the only known available information.

Overall, there has probably been a focus on wetland extent (which is also of critical importance – see wetland extent attribute) and perhaps insufficient attention paid to wetland condition.

Key drivers of degradation in New Zealand inland wetlands are drainage, fire, nutrient addition and invasive species [11], [21], [22], [23], [24], [25], [26], [27]. All these drivers may also affect wetland extent through direct mechanisms (drainage, mainly) and indirect mechanisms (other drivers).

Restoration or rehabilitation of inland wetlands varies widely among different wetland types, but in most cases, requires multiple interventions such as restoring the hydrological regime (i.e., usually reversing the effects of past drainage manipulations), reducing environmental stresses such as nutrients (nitrogen and phosphorus), and management of both biological invaders (especially weeds like willows) and reintroduction or planting of native species [37]. Restoration effort and success vary from relatively 'easy' for open water wetlands by restoring hydrological regimes, to extremely difficult for more complex systems that have undergone a tipping point in condition such as domed peat bogs. In general, most wetland restoration requires major long-term management interventions over the long-term, and the success of these efforts for improving ecological integrity is likely, but not yet known.

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

Our understanding is some councils use the wetland condition index, others, such as Auckland Region, use other methods. Other councils are yet to undertake any comprehensive monitoring. Where MWLR conducts monitoring on behalf of councils, the data may be stored in the NZ Wetland Database (which is in need of modernising). The question of replication is a live one – particularly with respect to the number of plots required for the purpose – regionally representative monitoring will require less than a detailed investigation of one wetland, for example. The Bellingham et al [4] Tier 2 monitoring report recommended the WCI, with some modifications/extensions, as the nationally-consistent reporting method for condition.

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

Access to field sites, some of which may be on private land, is required for assessments. This can create implementation issues (e.g., affect assessment frequency).

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

Labour is the biggest cost: where wetland plots are randomised, the majority of the time may be spent navigating through vegetation to get to a plot. Typically, two people undertake monitoring per plot: one undertakes vegetation measures; the other, soil, foliage, and other environmental variables. The time spent at a plot will depend on factors such as species richness (more species takes longer), plot size (larger plots will take longer), skills of the botanist (less skilled botanists will need to spend more time identifying plants), and ability to dig in the soil (affecting soil sampling and water samples). Some investment in information management, and health and safety gear (normal outdoor gear, plus water gear such as waders), and travel costs.

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

Taura et al [28] set out iwi involvement in wetland monitoring, but not specifically wetland condition index monitoring.

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

There will be clear linkages between other attributes that affect hydrological function and nutrient cycling within wetlands. There will be links between wetland extent and wetland condition, and the wetland condition index for NZ takes into account the historical loss of extent of wetlands in the catchment [1].

Part B—Current state and allocation options

B1. What is the current state of the attribute?

The current state of the attribute is poorly understood due to a lack of widespread, systematic field monitoring. Some councils currently undertake monitoring across a representative set of wetlands in their region, and this number is likely to increase with the uptake of monitoring required under the NPS Freshwater Management. The interim report on potential quantitative limits [8] and the GIS-based EII paper [10] are the only two published syntheses known.

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

No, but this information could be collected. MWLR recently collected information on ~20 'intact' and degraded wetlands included WCI, for a vegetation classification. Areas of lower human impacts [38] could be assessed to derive reference condition scores, and then recent work in quantifying natural variability could be applied to calculate a natural baseline including variability in composition, for a more quantitative baseline [29], in addition to 'natural' variability in WCI scores. Palaeoecology can also provide critical insights into historical wetland and vegetation types, although it cannot infer a historical WCI score [21].

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

Clarkson et al [8], p7 set out the following, but we highlight this was from an interim report that needs further field verification:

The US EPA have developed wetland condition indices by combining biotic metrics into an Index of Biological Integrity (IBI), and biotic metrics and abiotic metrics into an index of ecological integrity (EI) for wetlands (EPA 1998; Faber-Langendoen et al. 2006). According to EPA (1998), although individual metrics may respond differently, the index scores should form a relatively straight line when plotted against a gradient of human disturbance (Fig. 1).

Following this approach, we selected working breakpoints for the states of wetland health around New Zealand. As the Wetland Condition Index (WCI) ranges from 0 to 25, our preliminary working states were evenly distributed scores of:

A: >20–25 (>80%); excellent

B: >15–20 (>60–80%); good

C: >10–15 (>40–60%); moderate

D: <10 (<40%); poor; degraded

The national bottom line is set at the boundary between States C and D (Ministry for the Environment 2014). However, as data from lower condition wetlands were limited (scores mostly above 15), we combined the B and C categories and used three states of condition:

Excellent (A)

Good–Moderate (B–C)

Degraded (or poor) (D)

The ranges may need to be re-assessed following inclusion of data from more degraded wetlands. For example, the national bottom line threshold may be better set at WCI = 12.5 (50% of the WCI maximum) or even at WCI = 15.

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

Tipping points are likely from both hydrological changes and nutrient enrichment/eutrophication, but these have not been quantified or documented to the best of our knowledge.

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

Yes. Drainage is the major form of reduction in wetland extent in New Zealand [21]. Drainage near wetlands takes multiple years to reach 'equilibrium' [30] and even then, peat shrinkage rates continue on drained peatland in the Waikato particularly [31]. Furthermore, carbon sequestration may apparently continue in drain-affected wetlands, however this is due to woody plant invasion, which has a time-limited effect on carbon, unlike peat-forming species that sequester carbon into the soil [32]. The effect of drain may also take time to become apparent on the plant community, and as such, there are many wetlands around New Zealand that may be 'under the influence' of cryptic drain effects [24]. Fertilisation may have cryptic effects (causing lagged invasions) where long-lived plants persist but will not reproduce in eutrophied – or drained, or both – settings [23], [32].

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

There is some work in this area, but it tends to be for complementary streams of monitoring that iwi have autonomy over. A search of New Zealand published materials by authors Garth Harmsworth and Yvonne Taura (MWLR) would be a good entry point for publicly accessible materials.

Part C—Management levers and context

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

Key pressures on wetlands include drainage, eutrophication, invasive species, and fire, which are described sequentially below.

The Ashburton Lakes/O tu Wharekai is an example of where the relationship between a stressor (nutrients) and the receiving wetland has not been quantified well enough to guide management [39]. In other areas, the nutrient limits have been calculated, but are yet to be met.

The relationship between drains and effects on wetlands is a multivariate problem that is affected by soil type, drainage depth, distance from wetland, wetland hydrology, and time since drainage [24]. As such, there is no simple bivariate relationship between drainage and impacts, except perhaps, the more extensive the drainage, the more likely to be negative impacts.

Fire has caused substantial changes, particularly historically, on wetland vegetation and type: see review by McGlone [21].

Invasive species have negative effects on wetlands [22], [33], [34], however, there are few known relationships (for wetlands, and more generally) for impacts as a function of invader biomass.

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

C2-(i). Local government driven

C2-(ii). Central government driven

C2-(iii). Iwi/hapū driven

C2-(iv). NGO, community driven

C2-(v). Internationally driven (e.g., obligations to Convention on Biological Diversity, Kunming-Montreal Global Biodiversity Framework)

There has not been enough monitoring of interventions at the "Tier 2" scale (assessment of management interventions) to have strong evidence on the matter. However, because the WCI assesses clear drivers of decline [1], we expect that actions that are effective in addressing drivers of decline will affect the wetland condition index. Condition index can be broken down into constituent parts to focus on management actions. See wetland restoration handbook [36] and a local council example [40].

Part D—Impact analysis

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

This question is almost too broad to answer, given the number of ecological and hydrological attributes that make up the wetland condition index. Given the number of current pressures on wetlands (refer above – drainage, eutrophication, invasive species, fire – refer answer to QA3), without management, the various ecological and hydrological attributes that make up the wetland condition index will decline. As such, the ecological and human health values that are associated with wetlands (see answer to QA1) would be diminished, however, the specific impacts will depend on the sub-component of the index that is reduced.

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

It is difficult to describe with certainty the economic impacts, because a decline in wetland condition index might be driven by a decline in just one of the indicator components. For example, a decline in a component of ecological integrity might cause a similar drop in WCI score compared to one caused by a decline in hydrological function. Yet the economic implications of this decline are likely to differ. For examples of where decline in extent causes economic issues, see the answer to the extent attribute on this question.

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

We consider impacts of wetland by type. Mangroves may shift in distribution, may migrate south in New Zealand with warming water; they may also shift in response to sea level rise (reductions in seaward extent, increases in landward extent). Remaining wetlands that receive overland flow (swamps, marshes, fens) may suffer increased sediment and nutrient deposition if the increase in extreme events comes to pass; this may lead to wetland loss, if not reductions in ecological integrity. Increased wildfire may allow establishment of weeds. Salinity and inundation due to sea-level rise may cause declines in condition, and ultimately, loss of extent, for coastal, or near-coastal, wetlands.

The bogs in the Waikato are already a climatic oddity [21], [35] and their resilience to a warming climate is unknown. It is possible the peat-forming species will be unable to persist in a warming climate, and therefore the vast bulk of carbon sequestration will cease (depending on the replacement vegetation community). Bogs in the Waikato are also under threat of climate change due to salinization, as peat subsidence (due to drainage).

Across all wetland types, shifts in rainfall patterns may affect wetland condition, particularly in areas that are already suffering from drainage [24].

Management responses will need to be tailored to impact type. For some impacts, such as sea level rise, management may be limited to allowing inward migration of coastal wetlands (if economically and socially feasible). Reducing drainage stressors may increase wetland resilience to extreme events, or reductions in rainfall. Acceptance of plant community change may be required where current species cannot persist in their current geographical niche - management should turn to how to manage successions to the next-most valued plant (and animal) communities. For example: if salinisation affects *Empodisma* in Hauraki Plains wetlands, and the service of carbon sequestration is desired, consider transitioning to *Apodasmia* communities, which can tolerate some degree of salinity, but are also peat formers (to a lesser extent than *Empodisma*).

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6.5 Dune condition index

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Preamble: Aotearoa has multiple types of dune systems that are broadly characterised based on the origin of sand, location (e.g., coastal vs. terrestrial), and the physical activity that led to the structure of these systems (e.g., active[1], stable[2], volcanic[3], inland[4]). While all dune systems in Aotearoa are important and endangered, we interpret this attribute to predominantly refer to coastal active and/or stable sand dunes as these systems have received the greatest share of attention relating to ecological integrity and because our professional expertise is within the estuaries and coastal waters domain. However, the issues and pressures related to the decrease in coastal dune condition can be broadly applied to all dune systems. Also note that dune 'condition index' encompasses dune 'extent' given extent is one indicator of dune condition.

State of knowledge of the "Dune condition index" attribute: Overall, we consider the state of knowledge for the dune condition index attribute to be 'Good / established but incomplete' (though this may need to be changed to Poor / inconclusive or Medium / unresolved if considering all dune systems). Internationally and nationally, there is excellent evidence relating dune condition to ecological integrity. New Zealand-specific data that quantifies stressor impacts on 'dune condition index' and associated ecosystem services are good, and management interventions for coastal dunes are well known (though this may not be the case for volcanic or inland dune systems). Nationally, a standardised protocol exists for monitoring coastal dune condition, however to our knowledge this has only been adopted for a handful of councils and data on tipping points are lacking. Monitoring of dune condition is also carried out haphazardly across the country, leading to a lack of national-scale data for baseline and comparison of changes to dune condition.

Part A—Attribute and method

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

There is excellent evidence globally and in Aotearoa New Zealand (hereafter Aotearoa) to show that dune condition, which relates to dune extent among other measures, is closely tied with ecological

[1] See link: Active sand dunes » Manaaki Whenua (landcareresearch.co.nz)
[2] See link: Stable sand dunes » Manaaki Whenua (landcareresearch.co.nz)
[3] See link: Volcanic dunes » Manaaki Whenua (landcareresearch.co.nz)
[4] See link: Inland sand dunes » Manaaki Whenua (landcareresearch.co.nz)

integrity. Dunes are highly energetic habitats that contribute to coastal protection[1-3], support endemic biodiversity[4-6], and biocultural practices[7, 8]. Dunes are a common feature of the landscape throughout Aotearoa but may be most conspicuous along coastlines[1]. Their existence at the land-sea interface makes them important for terrestrial, freshwater, estuarine and nearshore coastal ecosystems.

Nationally, dune habitats are endangered (for all dune types) and support various threatened and critically endangered plant and animal species including a number of arachnid[4], lizard[9], and bird species[5, 10, 11]. Dunes serve several physical functions that support the ecological integrity of coastal systems, such as shoreline protection from storm surges, coastal erosion and flooding[1, 3, 12]. The presence of intact dunes supports the existence of a number of unique and often sensitive habitats such as dune slacks[2], dune deflation hollows[3], and/or damp sand plains[4]. Additionally, dunes play a crucial role in nutrient cycling (e.g.,[13]), soil formation (e.g.,[14]), and water regulation (e.g., [15]).

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

Globally and nationally, there is substantial evidence of the impact of degraded dune condition on the ecological integrity of coastal systems (e.g., globally[16-22], for Aotearoa[12, 23-27]). The loss of native dune species from stressors like coastal development, beach renourishment programs, fire, recreation, invasive species has led to the fragmentation and loss of coastal dunes, has led to a severe reduction in both dune condition and extent (i.e., between 60 and 80%, [28, 29]). Notably, this has impacted the national-scale loss or severe reduction of dune habitat for various threatened, endangered and critically-endangered spider and bird species, such as the Katipō spider (*Latrodectus katipo*) and the New Zealand Fairy Tern (*Sternula nereis davisae*), respectively[4, 5, 30]. In addition, the incursion of invasive plant (e.g., Marram grass, *Ammophila arenaria*, [31, 32]) and animal species (e.g., rabbits, [33]) has led to the displacement and loss of native dune flora (e.g., Pīngao, *Ficinia spiralis*; Spinifex, *Spinifex sericeus*), altering dune structure and promoting coastal recession and the loss of native bird nesting habitat [30, 34-39].

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

Dune condition has declined significantly over time, particularly due to the loss of native vegetation due to historic land reclamation (for agriculture, forestry, and/or development, [40]), coastal infrastructure development / hardening (e.g., groins, seawalls, and dykes, [41-43]), livestock grazing [33, 44], recreation [45], and invasive species [31, 32]. It is likely that over the next 10 – 30 years, the interactions among sustained stressors will continue to reduce dune condition, which also directly threatens the habitats supported by dune systems (e.g., dune slacks, dune deflation hollows, and/or damp sand plains).

While the multitude of stressors are actively interacting to reduce dune condition (to varying magnitudes based on location), most can be considered reversible and many are being managed, to

[1] Other dune types exist throughout Aotearoa, such as volcanic dunes formed from volcanic sediments and inland dunes formed from riverine sediments. Generally, these dunes are uncommon, in part due to decades of land-use change, which have made them endangered and critically endangered nationally.
[2] See link: Dune slacks » Manaaki Whenua (landcareresearch.co.nz)
[3] See link: Dune deflation hollows » Manaaki Whenua (landcareresearch.co.nz)
[4] See link: Damp sand plains » Manaaki Whenua (landcareresearch.co.nz)

some degree, by locally-led management and restoration programs (e.g., [46-48]). However, many of these dunes (especially near urban or developed areas) are generally in a degraded condition (e.g., moderate to poor, [39]). Furthermore, natural dune recovery is highly variable, depends on sediment supply, the presence of stabilising native plant species, reduced physical disturbance (i.e., from humans, livestock, and/or pest species) and may not fully recover without additional interventions, such as planting of native flora (which can take up to 2 years for rearing plant propagules[49, 50]). This means that retaining or improving dune condition will be heavily dependent on effective legislative action that affords dunes adequate protection, monitoring, risk mitigation, and restoration where needed.

Climate change is also predicted to impact dune condition and stressors associated with this are expected to exacerbate over the next 10-30 years[51]. See Section D3 for climate change impacts and management actions.

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

The use of a dune condition index has recently been included in State of the Environment monitoring for a number of councils throughout Aotearoa[39, 52]. The condition index used by these councils includes indicators for pressures to dune systems (e.g., livestock, mammalian pest species, human impact) and the ecological state of dunes (e.g., indigenous animal dominance, indigenous and non-indigenous land cover). Each indicator is given a score between zero and five with a low score representing negative condition (e.g., low indigenous vegetation cover, high foot traffic) and a high score representing positive condition (e.g., high indigenous vegetation cover, limited physical disturbance). The scores are then added up and compared against a possible maximum score to determine overall condition. Internationally, dune vulnerability indices have been developed, some of which have been used for forecasting tipping points in future climate-change related scenarios[20-22, 51, 53-56].

Some work has trialled the use of remote sensing (e.g., aerial and satellite images) to estimate dune condition, however, there often remains a need to ground-truth these data to accurately characterise native plant cover[57-59]. However, technological advances may help improve the accuracy of this type of data collection with time (e.g., as suggested for UAVs, [60]).

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

Dune condition monitoring methods generally require on-the-ground fieldwork for accurate surveys of indigenous vegetation cover. Access is a key consideration given that some dunelands are located on private lands and are therefore subject to the landowner's property and/or customary rights (e.g., for sites within or near Marae domains or Urupā). Accessing private property without the owner's consent can be considered trespassing (if not tapu), so clear communication, establishing good relationships, and addressing any concerns or impacts on the landowner's property or operations will be necessary. Formal access agreements or contracts may need to be established. It is possible that some dunes are not able to be permitted during certain times of year due to ecological factors such as nesting of rare birds.

Various health and safety factors also need to be considered in relation to fieldwork. These include access to the dunes and whether a 4-wheel drive vehicle is required for transport. Depending on the monitoring method being used, technical expertise such as plant species/taxa identification may also be required.

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

We anticipate that the main cost to undertake monitoring dune condition is paying field staff for their time. The costs associated with ground-truthing are likely similar to those proposed for surveying estuaries. For example, in 2002 the approximate cost to survey one estuary (for all substrate and vegetation types; this could be analogous to dune extent) following NEMP was estimated to be between $15,000 to $30,000[61]. However, this cost was dependent on the size of estuary (*or dune system*) and whether suitable aerial photographs were available or needed to be obtained for the survey. The approximate cost now (to account for inflation and technological expenses) will likely be higher. Additional costs will relate to personnel time spent reporting results, however, key equipment may also include GPS ($300 - $800), camera and / or cell phone ($100 - $600), clip boards, transects, plant ID guides, and if mapping dune extent as part of the condition index as is currently done, ARC GIS or equivalent software ($100 - $3800)[62].

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

To our knowledge, Māori indicators for dune condition monitoring mainly relate to the presence of native flora (e.g., such as Pingao) and fauna (e.g., such as geckos[1]). We are currently unaware of any Iwi/Māori-led initiatives to monitor dune condition specifically. However, there are several Iwi-led projects to monitor and restore dune wetlands (e.g., [8, 63]). Furthermore, a number of Marae are situated within or near dune systems (e.g., [2,3,4,5]), so it is possible that these Iwi are monitoring dune condition locally, especially with respect to anthropogenic disturbances[6].

A6. Are there known correlations or relationships between this attribute and other attribute(s), and what are the nature of these relationships? [Discuss similarities, links, or correlations with other listed attributes. Could the correlated attributes be grouped?]

Active and stable dunes are part of larger, continuous coastal habitats. This means that unobstructed connections with other habitats leads to higher habitat quality and ecological functions than isolated or fragmented dunes, all of which relates to 'dune extent' (encompassed under dune condition) and 'landscape connectivity'. Dunes that offer limited 'access to natural areas' (specifically in relation to human disturbance) may also have less impacted condition and support more diverse, native ecological communities. However, this does not exclude the potential impacts of stressors on adjacent attributes such as 'wetland extent' or 'surface water flow alteration', which can also influence 'indigenous plant dominance'.

[1] See link: Mana whenua guidance key to reptile surveillance - OurAuckland (aucklandcouncil.govt.nz)
[2] See link: Northland marae's concern for wāhi tapu – Te Ao Māori News (teaonews.co.nz)
[3] See link: Te Henga | Maori Maps
[4] See link: Māori cultural sites among most vulnerable to climate change, rising sea levels | Newshub
[5] See link: Bikers tear up eroding Kāwhia Beach dunes, threatening Marae | Stuff
[6] See link: Landowner agrees not to dig Karikari Peninsula's wāhi tapu sand dunes | RNZ News

Dunes are often found between multiple ecosystems, meaning there will likely be a crossover in monitoring methods for ‘salt marsh quality and extent’, ‘seagrass quality and extent’, ‘lowland forest extent’, ‘mangrove extent and quality’, and, to some extent ‘beach litter’. In addition, the ‘wetland condition index’ is applicable to dune-associated wetlands, such as dune swales [64].

Part B—Current state and allocation options

B1. What is the current state of the attribute?

There is substantial evidence that dunes have been lost throughout Aotearoa since initial surveys of dune systems in the early 1900s [23, 25-27, 29, 45, 65-68]. The current state of dune condition is understood at some regional levels (e.g., Hawkes Bay Region [37, 69]) and is reasonably well-understood nationally (e.g., for extent [28, 29, 70]). However, there is some indication that with the spread and intensification of human activity around Aotearoa means that the current (as of 2024) condition of dune systems is likely poor[71].

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

Dune systems that have retained their historical condition and that have limited to no introduced plant species or evidence of human-induced impacts (e.g., vehicular trampling, livestock grazing) could be considered a reference state. Dunes found in remote, protected locations such as those within or in association with national parks national parks may best serve as examples of natural states with limited to no impact from human-induced stressors. However, sites within remote, protected areas may still contain stressors like introduced weeds and mammals and may still be subject to climate change impacts.

Additionally, the United Nations Statistics division has developed a System of Environmental – Economic Accounting, Ecosystem Accounting (SEEA-EA)[1] that assesses a suite of landscape characteristics to quantify ecosystem condition / integrity. The indicators for this are derived from a number of variables in relation to specific reference states for the system in question, e.g., [72]. In the case of dune systems, these indicators include abiotic (e.g., geomorphic alignment of shoreline, number of extreme wave events, and chemical attributes[2]) and biotic (e.g., proportion of native:non-native species, presence of exotic woody species, and the presence of functional, sand-binding species) and were designed to reflect dune condition. Ryan et al. (2023; [72]) implemented this system to provide an updated narrative of dune state throughout Aotearoa and have suggested its use for the development of a globally relevant monitoring framework.

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

[1] System of Environmental Economic Accounting | UN

[2] Though this is data deficient for dune systems throughout Aotearoa.

See section A4-(i) for more information on how current dune condition indices are calculated in New Zealand. Additionally, the narrative bands used to monitor wetland condition index (which includes dune wetlands) could be modified to inform bands for dune extent[73].

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

Tipping points for dune condition (in respect to dune extent) have been reported internationally for coastal systems (e.g., 0.25 m of sea level rise, [51, 54, 56]), but data for Aotearoa is lacking. The tipping points reported internationally depend on factors such as dune condition, sediment budget, wind intensity, frequency of storm events, and sea level rise (e.g., [51, 56]). For example, increases in variables such as wind speeds, wave action and sea level rise can increase dune erosion resulting in a deficit of sand trapping, which can signal ongoing loss of dune extent, and as a result, condition[74]. Recently, the use of spatial modelling has been employed to forecast future 'tipping-point'[1] scenarios for dune condition in response to climate-change, human use, and/or geologic activity[75-77].

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

Lag time between stressor and impact on dune condition will be site- and stressor-dependent. For example, there may limited to no lag time in cases of direct impact and severe physical damage, such as coastal development or recreational vehicles[76]. Alternatively, lag times are expected from the impacts of stressors such as sea level rise due to relatively slow encroachment. Additionally, there are lag times expected from the impact of non-indigenous plant species where there will be a time when these exist as seeds/seedlings before becoming established and spreading. There may also be lag times following coastal development and/or alterations to hydrological flow regimes, which can influence sediment budgets for dune systems[78].

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

Mātauranga Māori is inherently place-based and so needs to be considered within a local context. Dunes are valued by Māori as important systems that provide resources for cultural practices (e.g., collecting Pīngao for weaving) and as habitat for taonga species. Indigenous indicators (i.e., specific tohu and/or taonga species) could be used to inform bands/allocation options.

Part C—Management levers and context

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

In cases where dunes are lost, for example from agricultural, forestry, or infrastructure development, there is an obvious and direct detrimental relationship between the stressor and dune condition [40]. Furthermore, there is also some information for Aotearoa documenting the relationship between

[1] Though not all spatial modelling studies have explicitly referred to these scenarios as 'tipping points'.

dune condition and other physical stressors such as vehicle damage, livestock grazing, trampling, and invasive species incursions[25, 31, 45, 67, 76, 79]. These relationships are currently being quantified as: presence / absence of stressors and certain flora and fauna; % cover / area accessed or disturbed (e.g., as reported by [52, 80]); and in some cases as ratios between native:non-native plant dominance (e.g., for SEE-EA in [72]).

However, there are still challenges associated with disentangling interactions among multiple stressors, respective lag times, additional legacy affects, and overall dune condition. In addition, the impact of stressors on ecosystems is usually highly context-specific (i.e., place and history are very important) and so effective management and needs to understand and allow for that context.

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

Key management interventions include duneland protection and the elimination or reduction of stressors. From a policy perspective, the RMA (1991) is a key piece of legislation that sets out how we should manage our environment. In addition, the New Zealand Coastal Policy Statement guides councils in their day-to-day management of the coastal environment, which specifically includes dune systems[2]. There are various other relevant government-related directions and management implementations, for example for biosecurity, climate change, wildlife, threatened species and national parks. Some local governments have also instituted specific bylaws for the protection and recovery of dune systems (e.g., vehicle use bylaw, [81]).

A number of councils have active dune management / restoration plans, which may be part of larger shoreline management schemes (e.g., Northland, Auckland, Waikato, Hawkes Bay, Tasman, and Nelson[1]). For example, dune habitats are commonly roped / fenced off to exclude public access along many public beaches throughout Aotearoa to allow for dune recovery / persistence[2, 82]. These rope systems and associated signage discourage the public from trampling sensitive dune fauna and increase awareness of the ecological function of dunes (e.g., as nesting habitat for rare and endemic seabirds[5, 30]).

Active dune restoration is another management intervention that can be carried out to improve quality of dune condition by community groups, councils, DOC, iwi/hapū or others interested in recovering dune habitat [e.g., [62, 83]. For example, a number of coastal care groups (e.g., Coastal Restoration Trust of New Zealand) are actively involved in re-vegetating dune systems with native flora (e.g., [83]). Dune species are grown in some commercial/specialised nurseries and are widely available for these kinds of restoration projects. Furthermore, there have been several guides produced to inform the use of various species used in dune restoration to inform restoration efforts [e.g., see[38, 49, 50, 84-87]]. There is also potential to consider dune restoration for flood and sea level rise mitigation as suggested by [88].

C2-(i). Local government driven

A number of local government-driven initiatives are present throughout Aotearoa aimed at restoring dune habitat. Some examples include the restoration of endemic dune species such as Pīngao and spinifex at sites around Timaru [47]; urban dune habitats along the Coromandel coastline [46]; and Ngarahae Bay, West Coast North Island [48], to name a select few. Additional projects can be found at

[1] https://www.doc.govt.nz/get-involved/run-a-project/restoration-advice/dune-restoration/

the websites for the Coastal Restoration Trust of New Zealand[1], Waikato Regional Council Coastcare groups[2], and for international projects, in [89].

C2-(ii). Central government driven

Central government can provide key funding for the protection, conservation and restoration of dunelands. For example, the recently completed NZ SeaRise Te Tai Pari O Aotearoa[3] project that projects sea level rise around New Zealand could be used to help prioritise future restoration projects at vulnerable coastal areas. There is also potential to consider dune restoration for shoreline management and/or conservation plans supported by the Department of Conservation (e.g., the Auckland Regional Council Shoreline Adaptation Programme[4]; Rakiura Conservation Management Strategy[90]).

C2-(iii). Iwi/hapū driven

We understand that Māori could offer protection to dune habitats through rāhui (e.g., Pakiri Beach, Auckland[5]). We are also aware of a small number of Iwi-led restoration projects for dune adjacent habitats (e.g., dune wetlands, [8, 63]). Also refer to footnotes 14-19 for articles related to interventions being used to improve / protect dune condition.

C2-(iv). NGO, community driven

A number of community-driven dune restoration projects exist throughout Aotearoa. A notable NGO is the Coastal Restoration Trust of New Zealand (previously called The Dune Restoration Trust of New Zealand), which has developed a comprehensive guide and monitoring scheme for dune restoration projects throughout the country[83]. This guide has been, or is being, used by several partner councils and government agencies (e.g., Christchurch City Council, Department of Conservation, Environment Canterbury Regional Council, Northland Regional Council, the Greater Wellington Regional Council)[6]. Additional projects include the Native Forest Restoration Trust[7], DUNE, the Whāngaimoana Dune Restoration Group, and Onetangi Beach Dune Restoration, to name a select few (for a comprehensive list of coastal restoration groups see footnote 24). International NGOs, such as The Nature Conservancy, are also active in New Zealand[8] and provide support for conservation initiatives and nature-based solutions, which can include increasing dune condition through planting of native vegetation.

C2-(v). Internationally driven

Restoring the vitality of degraded systems (which include dune ecosystems) is crucial for fulfilling the UN Sustainable Development Goals and for meeting the targets of the UN Decade (2021-2030) on Ecosystem Restoration (UN-DER). Under the Convention to Biological Diversity (CBD), Aotearoa is required to have a national biodiversity strategy and action plan through which obligations under the CBD are delivered. Aotearoa has international climate change obligations such as those under the Paris Agreement. We understand that Aotearoa has also signed other international agreements (e.g.,

[1] https://www.coastalrestorationtrust.org.nz/coast-care-groups/groups/
[2] https://storymaps.arcgis.com/stories/14b535daa5ae4aae820d1be774f740b7
[3] https://www.searise.nz/
[4] https://www.aucklandcouncil.govt.nz/plans-projects-policies-reports-bylaws/our-plans-strategies/topic-based-plans-strategies/environmental-plans-strategies/shoreline-adaptation-programme/Pages/shoreline-adaptation-plans.aspx
[5] https://www.localmatters.co.nz/news/tangata-whenua-closes-beach/
[6] See link: Research Partners • Coastal Restoration Trust of New Zealand
[7] https://www.nfrt.org.nz/reserves/oreti-totara-dune-forest/
[8] https://www.nature.org/en-us/about-us/where-we-work/asia-pacific/new-zealand/stories-in-new-zealand/our-work-in-new-zealand/

Free Trade) that require conditions around environmental management to be upheld. Additionally, the Ramsar Convention of which Aotearoa is a signatory meaning it plays a part in the international effort to conserve wetlands, which includes dune slacks and lakes[91].

Part D—Impact analysis

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

Failing to manage dune condition poses a significant threat to coastal environments, triggering a cascade of ecological problems. For example, increased physical disturbance from vehicle traffic or livestock grazing can lead to the loss of endemic, sand-trapping flora, which can lead to a severe reduction of condition and eventually loss of dunes. With this comes a loss of coastal protection and habitat for critically endangered endemic species, which is partially reflected in the decline of bird populations[10]. The incursion of invasive flora species, like Marram grass, can over-stabilise dunes, upsetting coastal sediment budgets through increased erosion, leading to increased degradation of adjacent shoreline habitats (e.g., [31, 32]).

Additionally, the loss of dunes can allow for saltwater intrusion into coastal aquifers and wetlands, which can substantially alter ecosystems, leading to further loss of endemic species[92]. This influx of sea water may disrupt the delicate balance of coastal marine life and can sever vital links in the coastal marine food web, which can have cascading impacts on the overall health, biodiversity, and thus ecological integrity of coastal ecosystems[72, 93].

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

The economic impacts are likely to be felt among coastal infrastructure development and tourism sectors. Reductions in dune condition could lead to a loss of habitat for critically endangered bird species, which limits coastal tourism opportunities for certain groups (e.g., birders) and local businesses (e.g., [94]). Reductions in dune condition can also limit their protective capacity as natural buffers that absorb wave energy and lessen the impact of tidal flooding and storm surges[3, 12, 23, 77]. The loss of dunes exposes coastlines to increased erosion, leading to a retreat of beaches, coastal wetlands, and lowland forest, which may lead to a heightened risk of damage to coastal infrastructure and sensitive adjacent habitats such as dune slacks[18, 21, 95].

Reductions in dune condition can also impact the loss of culturally important sites, such as Marae and / or Urupā[1,][96]. The loss of dune systems can directly influence tikanga practices, which can diminish mana over associated resources and / or areas [97-99].

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

Sea level rise and increased storm frequency is expected to lead to the erosion and loss of dune habitat [92, 93, 100, 101]. Sea level rise may also result in reduction in dune extent, and thus condition, due to 'coastal squeeze' if habitat is not available for it to migrate to due to the presence of roads, urban

[1] See link: Tairāwhiti marae facing 'devastating' loss of urupā as heavy rain lashes Gisborne region | Stuff

areas, stopbanks, or agricultural land directly inland from current dunes [89, 102]. Increased storm frequency will likely lead to increased flooding, which will likely impact coastal sediment budgets and coastal erosion processes[95, 103]. Changes to vegetative cover, at times due to increasing temperature, can alter dune faunal community structure and lead to further range shifts and incursions of invasive species. Increased intensity of fires as a result of climate related issues, such as drought poses an additional risk to the recovery of dune species, however there is some indication that this does not contribute to dune instability [104, 105]. In addition, increasing temperatures may also reduce below-ground biomass for certain dune vegetation species, which can reduce dune accretion and subsequent stabilisation (e.g., as seen in China[106]).

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6.6 Indigenous plant dominance

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State of Knowledge of the "Indigenous plant dominance" attribute: Excellent / well established – comprehensive analysis/syntheses; multiple studies agree.

The evidence and use of indigenous plant dominance for understanding and reporting on environmental change is overwhelming, and is strongly linked to all components of ecosystem integrity and to many other attributes. More specifically for indigenous dominance, **ecological integrity** (as defined from the NBA Bill) is the ability of the natural environment to support and maintain:

- representation: the appropriate spatial distribution of indigenous species,
- composition: the abundance of indigenous species within different communities,
- structure: the effects of indigenous dominance on physical and abiotic properties of ecosystems, and
- functions: the effects of indigenous dominance on ecological and ecosystem processes.

Indigenous (native) plant dominance is considered an essential biodiversity variable (EBV; see Pereira et al. 2016). Dominance can be measured in multiple ways including cover, biomass, population density, height or size. Dominance measures vary among taxa and systems. In addition, these measures reflect different aspects of abundance or species' effects on representation, composition, structure and function. The spatial scale over which dominance is measured ranges from the biomass of individual trees, through to national-scale reporting of distribution or abundance of specific taxa for biodiversity reporting and carbon accounting.

There are some established targets or limits for indigenous plant species dominance such as maximum height determined for land clearance. Our knowledge and ability to establish targets or limits for indigenous dominance related to ecological functioning or ecosystem processes is rare, because this relies on understanding the per unit impacts of species on ecological process and dominance of ecological process and function (Lee et al. 2005), not biomass or abundance *per se*. Nonetheless, indigenous plant species dominance is one of the most important attributes or indicators of ecological integrity for terrestrial systems.

Part A—Attribute and method

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

In general, increasing indigenous plant species dominance is considered positively related to all aspects of ecological integrity (Lee et al. 2005; McGlone et al. 2020). Indigenous plant dominance directly relates to the ecological integrity of terrestrial ecosystems through composition (i.e., by directly influencing community composition) and structure (i.e., by contributing to vegetation structure). Dominance relates to functions in multiple ways through controlling key ecosystem processes such as primary productivity, nutrient cycling and decomposition, or specific ecological processes like habitat or resource provision for other taxa. Dominance can also be related to representation if changes in abundance or cover of species are evaluated over larger spatial scales (e.g., changes in the cover or biomass of species across a population, sites or across the overall species distribution). An important concept reflecting how dominance varies spatially are abundance-occupancy relationships, which capture the distribution of species throughout their range.

Dominance in some cases is also related to human health through landscape values (e.g., tussock grasslands considered outstanding vegetation in the McKenzie basin), cultural values (e.g., dominance of taonga species), and in a few cases, economic wellbeing (e.g., mānuka honey).

There are some important exceptions to the generalisation that indigenous plant dominance is directly and positively related to ecological integrity. Some examples include:

- The deliberate anthropogenic introduction of, and sometimes invasion by, native plant species beyond their historical distribution (e.g., karaka in coastal habitats).
- Maintenance of indigenous dominance through deliberate management (e.g., maintenance of mānuka monocultures for honey production through repeated disturbance).

In these cases, measures of indigenous dominance alone are insufficient to understand ecological integrity; additional knowledge of ecosystems, management or pressures is required.

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

There is strong evidence that changes in indigenous plant dominance affects ecological integrity and human health, but these effects vary widely among regions and species. Indigenous species dominance is also considered an essential biodiversity variable (EBV), and as a consequence is widely used to report on state and trend of biodiversity and ecosystem condition (Bellingham et al. 2020, McGlone et all 2020; see also A4 below). International and national concerns about long-term declines in biodiversity have driven major data collection and reporting efforts, most of which capture data for indigenous plant species dominance (see A5 and C2 below). Dominant indigenous plant species control many ecosystem properties and processes across different vegetation types such as forests and wetlands (see Allen et al. 2003, 2013, Tierney et al. 2009).

Changes in indigenous plant dominance, both in spatial extent and abundance across scales, are relatively well documented and understood. Shifts in the abundance of many indigenous plant species or vegetation dominated by indigenous species reflect major landscape-scale processes including:

- Altered disturbance regimes, particularly fire: indigenous forest cover and many indigenous plant species declined sharply following anthropogenic fires (Perry et al. 2014, McWethy et al. 2014);
- Land use change and intensification: historical and ongoing land management often replaces indigenous plant species with non-native taxa for a range of purposes or services (Atkinson et al. 1993, Craig et al. 2000, Moller et al. 2008). Land use and management has also fundamentally altered nutrient availability and hydrology or soil properties which often favours dominance of non-native plant species having a strategy of faster growth and resource use compared to most indigenous plant species (Brandt et al. 2021);
- Introduction and impacts of non-native mammalian herbivores: historical and ongoing impacts of different invasive herbivores can drive declines in preferred (palatable, or selected) species but also increases in avoided (unpalatable) species (Coomes et al. 2003, Peltzer et al. 2014).

Additional drivers of change in indigenous species dominance are likely, but poorly understood. These include historical selective harvesting (logging) of a subset of indigenous tree species (McGlone et al. 2022), declines or loss of other indigenous species that affect plant population processes (e.g., loss of mutualistic species such as pollinators), long-term impacts of vegetation fragmentation creating extinction debts (Ewers et al. 2006), and ongoing increases in the number and abundance of non-native environmental weeds that can suppress or displace co-occurring indigenous species (Brandt et al. 2021; PCE2021).

Overall, there is low evidence relating indigenous plant species abundance directly to human health. This is driven by a paucity of information available for most taxa; our understanding and data are restricted to a few species such as:

- Abundance of *Coriaria arborea*/tutu because of potential tutin poisoning of honey.
- Mānuka underpinning honey production and economic wellbeing.
- Declines in major taonga species such as Kauri (*Agathis australis*), linked to multiple cultural values of forest ecosystems (Waipara et al. 2013).
- Declines in broader sets of dominant species such as myrtle rust, potentially having important social, cultural and economic impacts (Lambert et al. 2018).
- Loss of podocarps and related cultural impacts (Lyver et al. 2017a,b).
- Perceptions of loss for wild food and thus human health.

See also details in other templates which consider loss in extent or dominance of indigenous plant species: lowland forest extent, wetland extent, and canopy tree dieback extent.

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

Dominance of many indigenous plant species has changed profoundly historically, and is expected to continue changing over the coming decades, although the pace and trajectory of change will differ widely among taxa and regions.

Historical baselines of plant dominance together with paleo records and biocultural observations provide overwhelming evidence that native plant dominance has declined sharply for many terrestrial ecosystems from pre-human baselines. These declines range from trivial (alpine vegetation) to >95% declines in peat bogs (see Burge's wetland extent attribute) and some lowland forest systems (see Walker's assessment of lowland forest extent attribute). Broad-scale declines in dominance were driven initially by anthropogenic fires, and subsequently by land use changes and introduced pests, weeds and diseases (Perry et al. 2014). In some areas indigenous plant dominance has recovered during the previous decades through changing land management, fire suppression, and discontinuation of harvesting (Allen et al. 2013).

Over the coming decades, dominance of many indigenous plant species is expected to maintain a status quo or increase slightly at the national scale. This is being driven by relatively widespread interest in managing marginal or previously-disturbed vegetation for carbon resulting in increased areas of successional vegetation (Forbes et al. 2023). Moreover, much of this management also improves biodiversity associated with C management, either implicitly (i.e., many woody successions include multiple indigenous plant species) or explicitly (e.g., through market premiums in ETS, or potentially through future biodiversity credits schemes). Both national and regional incentives are driving broad restoration efforts in some wetlands, riparian marginal strips, and enrichment planting; and the effects of these schemes should persist for decades.

Declines in dominance are expected for some indigenous plant species and regions, including taxa susceptible to emerging pathogens (e.g., myrtle rust impacts on myrtaceous plant species that include dominant forest tree species), replacement of some native taxa by non-native species, and by increasing weather extremes and associated disturbances such as cyclones and fire (see also tree canopy dieback extent attribute).

Evaluating current state and change in indigenous plant species dominance requires an explicit consideration of scale. At the national scale, past land use and intensification, species introductions, and landscape-scale changes have reduced the extent and condition of indigenous vegetation in most regions, and altered the distribution and abundance of many common indigenous plant species. These historical legacies have set the trajectory and scale of change possible for the next 30 years and much longer, and are not reversible in the short-term. However, at regional or local scales, changes in dominance can respond far more quickly and are often the focus of data collection efforts and reporting (discussed below). Natural ecosystems and dominance can change quickly, even with natural disturbances (e.g., the 2016 Kaikoura earthquake had immediate impacts on dominance, but recovery will occur over decades; see also Allen et al. 1999).

The pace and trajectory of change in plant species dominance has been evaluated across scales. Nationally, biodiversity monitoring and carbon accounting efforts capture repeated measures of abundance for most indigenous forests and shrublands over the past few decades (e.g., using DOC tier one monitoring). Regional monitoring efforts and plot networks have been established and provide important baselines and often include ancillary information for interpreting change in species dominance (Richardson et al. 2024). Longer-term changes are commonly evaluated at the local or catchment scale, because this is the scale at which management and community interests often emerge.

Overall, long-term (i.e., decadal) change in the dominance of species is the norm in dynamic systems, but there are well established efforts to monitor state and change in many indigenous plant species. Coverage of non-woody vegetation less well monitored (e.g., in alpine systems or marginal/successional systems). The ongoing challenge is assessing these changes against shifting baselines in disturbance regime, climate and other pressures like pests and pathogens (Bellingham et al. 2020, Lyver et al. 2021).

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

Development of major biodiversity and carbon monitoring and reporting efforts over decades have generated **excellent monitoring standards and methods for assessing dominance of indigenous plant species. NHMS developed by DOC (**Lee et al. 2005) built on plot-based assessments of vegetation condition and change (e.g., Bellingham et al. 2000) provide detailed methods, standards and interpretation of biodiversity state and change across spatial scales. In addition, many of these methods are consistent with international efforts (EBVs) and can be used to quantify ecological integrity (McGlone et al. 2020). Ongoing refinement and updating of these methods offers opportunities to leverage data collection, monitoring and use. For example, DOC's tier one monitoring of terrestrial vegetation explicitly measures dominance and representativeness of indigenous plant species (measures for maintaining ecosystem composition, including 'species composition and diversity' and 'Species occupancy of natural range', (Bellingham et al. 2020)).

Excellent compositional data have been collected nationally as a consequence of multiple long-term monitoring efforts for biodiversity, vegetation assessment, soil characterisation, and carbon estimates (Bellingham et al. 2020). Many structural attributes are collected at the same time as compositional data. Structure is driven across scales by abundance and dominance of plant species within sites, reflected by numerous measures of biomass, vertical structure, cover (including in height tiers). Terrestrial vegetation is relatively well-characterised, and quantitatively-derived communities are available (e.g., Wiser et al. 2011). Major national and regional plot networks, monitoring efforts, and land cover spatial data provide fundamental data for quantifying and reporting changes in indigenous plant species dominance (or proxy measures such as vegetation cover class). Cover is the most consistent and commonly-reported measure of dominance (Bellingham et al. 2020). Biomass, density, height are also reported for specific purposes, vegetation-types, or regions. Dominance estimated by proportional contribution to community richness is fraught, because it does not relate strongly to the components of ecological integrity.

Many plot-based monitoring efforts capture information on population abundance and structure and can provide additional insights into population processes at site or larger spatial scales. For example, changes in tree size class distributions both determine forest physical structure, but also reflect habitat quality, effects of disturbance, and numerous ecological functions when considered with functional traits or additional attributes such as disturbance or pest abundance (Peltzer et al. 2014). Similarly, population size or structure is often used as a target or limit for species, both to understand declines or recovery of native species.

Standard methods are widely used but also vary in their application among regions. National standards for biodiversity and carbon are derived from plot-based (point) measures of vegetation composition and structure. This information is a crucial part of Te Mana o te Taiao; regional reporting on biodiversity by RCs and territorial authorities; and LUCAS (Bellingham et al. 2016, 2020). The practical application and ability to apply these standards varies among regions. Tier one measurements used by councils and central government is standardised and widely used but requires point-based measurements. Data are remeasured currently on a 5-yearly return interval across PCL (but not private land), although discussions are in progress for pushing this out to a 10-yr return interval. These methods are applied by Greater Wellington Council across all vegetation-types, by Bay of Plenty/Auckland at the regional scale, and at the catchment or local scales for other regions. Although there is broad consistency in the methods used by different agencies, the spatial coverage and repeat measurement are not consistently applied.

Most measurements link to DOC tier one measurement across all landscapes but are not done in some land cover classes such as pastures or plantations. For example, landscape-scale structure has been relatively well characterised for forests and wetlands, but not for many other systems including naturally rare ecosystems (Williams et al. 2007, Holdaway et al. 2017). Furthermore, plot- or point-based measures are usually amalgamated into land use or cover classes for reporting purposes (Cieraad et al. 2015, Dymond et al 2017). Use of these data for reporting purposes span:

- Convention on Biological Diversity (CBD) reporting.
- MfE State of Environment reporting.
- biocultural monitoring (see B5 below).
- RC regional biodiversity reporting.
- catchment or site-based assessment of changes in indigenous dominance.

Both mapped changes in land use or cover of vegetation classes dominated by indigenous species are commonly reported, but have some inconsistences in underlying data. For example, the resolution of mapping and ground based validation is low (Cieraad et al. 2015) partly because there are underlying errors in threshold and detection from imagery or vegetation classification (e.g., between versions of LCDB). These underlying assumptions or errors require uncertainty estimates for mapping, both false positives and negatives, that are known but rarely implemented in practice. A specific example: ‘depleted grassland’ includes a wide range of indigenous dominance, and this can vary crucially among different height tiers, at the extreme, dominance by tussocks at 30cm height but non-native species like *Hieracium* spp at the ground level (Weeks et al. 2013). Another example is for Auckland, where LCDB spatial resolution is considered too coarse to report changes in peri-urban fringe, or equally, smaller increases through restoration.

An ongoing challenge for reporting changes in indigenous dominance is linking plot-based data and spatial (mapped) information across scales. Emerging technologies like high density LiDAR and high-resolution hyperspectral imagery can be used to quantify structure, but cannot yet distinguish species without ground-based validation (see also Wiser et al. 2021, Ye et al. 2021). Most commonly, the default reporting units are based on LCDB, but reporting could also be done linking point-based measures with other mapped areas or cover classes within LENZ, qualitative ecosystem typologies, or more quantitative typologies (i.e., like Cieraad et al. 2015). These issues could be resolved for some systems, for example, by assessing if declines in wetland extent are mirrored by declines in indigenous plant species dominance. Including both plot-based and spatial data from remote sensing for assessing biodiversity change is the approach championed internationally through EBV development (Bellingham et al. 2020).

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

Implementation of data collection is well established through major monitoring efforts such as LUCAS and regional plot networks (Richardson et al. 2024). Practical or logistical considerations (not barriers *per se*) include:

- Botanical skills (for identifying indigenous taxa) and maintenance of qualified personnel (i.e., for field measurements, data collection, and analyses).
- Provision is in law via the RMA to collect data, but can be logistically difficult for some sites or communities.
- Repeated samples to evaluate changes in dominance require databases, long-term archiving, and access.
- Increasingly, transparent documentation of data processing, analyses and interpretation is required or expected (for both reporting and publication purposes). Similarly, explicit evaluation of assumptions and uncertainty in the data or analyses are required. Remote sensing methods and new technologies (e.g., eDNA) are not immune to these issues; all increasingly require informed consent for data collection, analyses and reporting.
- Intellectual property and data sovereignty issues are a potential barrier to data collection, use and access, and require ongoing consideration as part of monitoring efforts.
- A practical barrier is the lack of sustained/long-term commitment for collecting the primary data by most RCs through lack of funding or prioritisation of efforts elsewhere (see PCE 2019, 2020).
- Current monitoring efforts do not have complete national coverage. There is a bias in data collection against lowlands and rapidly changing (marginal) land use classes. Most reporting has focussed on PCL because one lead agency (DOC) implements monitoring, where potentially changes in indigenous plant dominance have the slowest/modest change. Spatial coverage of data in other land use classes is poor because of multiple agencies involved, has lower priority for many regions compared to other competing issues (e.g., water issues).

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

Costs per plot including data management are well discoverable from DOC and RCs, but do not distinguish costs for measurement of indigenous plant species dominance from other measurements (e.g., plant diversity, pellet counts, deadwood assessment) carried out at the same time. Aside from central and regional government, multiple agencies have the infrastructure, skills and ability to monitor this attribute including Manaaki Whenua and Wildlands. Additional costs for quality assurance of data collection and analyses are also available (e.g., from validation of DOC tier one measurements).

Detailed costs for other monitoring approaches such as remote sensing data are available, but heavily depend on the provider, quality of data collected, and highly variable costs associated with data management and processing or analyses. Costs of data acquisition are available, but depend on provider and data quality captured. Additional costs of specialist skills for data processing, analyses and interpretation are highly variable depending on the scale and reporting needs; normally the purpose is not to monitor indigenous plant dominance but to estimate land (vegetation) cover in which some cover classes are dominated by indigenous species.

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

There are several examples of indigenous plant dominance being monitored. A few of these include:

- Forest diversity and condition monitoring in the Urewera's (e.g., Lyver et al. 2017) as well as shifting baselines in abundance of species and how communities respond or manage.
- Surveillance and impacts of Kauri dieback (see canopy tree dieback attribute).
- Monitoring of pingao abundances as a weaving resources.
- Mānuka as a resource for honey/economic activities (e.g., including Māori-owned businesses such as manawa honey).
- Rongoā plant species abundance (widespread).
- Abundance of dominant plant species such as raupō in wetland restoration (e.g., the Awarua-Waituna Ramsar site, Southland).
- Titi islands. Indigenous plant dominance but by different species because of burning regime associated with muttonbirding on these islands.

Some general considerations for monitoring effects by Iwi/Māori include:

- Standard (i.e., standardised tier one) measures do not apply. Communities do place-based monitoring, decide on what to measure, for what purpose. Fitness of purpose for scaling up monitoring efforts is not a major driver.
- 'Limits and targets' for abundance are set by communities.
- Biocultural monitoring (Lyver et al. 2018, Harcourt et al. 2022, Pou et al 2022) includes more integrated assessment of condition rather than focussing on a single attribute, but these can be used to better interpret change and impact.
- For Māori, everything is connected - biodiversity outcomes are important but cannot be considered separately from socio-cultural outcomes.

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

Indigenous plant species dominance is a unifying attribute across terrestrial ecosystems, and is closely linked to other attributes. Lowland indigenous forest and wetland extent are both directly, positively related to this attribute. The spatial distribution and representativeness of indigenous plant species dominance and other terrestrial attributes are tightly coupled because they respond (negatively) to most of the same drivers of change, i.e., disturbance history, land use change and management, and the impacts of biological invaders.

Part B—Current state and allocation options

B1. What is the current state of the attribute?

The current state and recent change (i.e., over past few decades) in indigenous vegetation cover is well understood at the national scale. Excellent national monitoring data for indigenous plant species is available through tier one DOC monitoring, spatial change of indigenous-dominated vegetation through multiple updates to LCDB, and monitoring effects by many Regional Councils. Since human arrival in New Zealand, indigenous vegetation cover overall is reduced to approximately 44% of New Zealand. However, national and regional monitoring efforts show large variation in the scale of lost indigenous vegetation cover with the greatest declines driven initially by fire, and subsequently by land use changes in low elevation environments (Perry et al. 2014; see also lowland forest extent attribute). For example, the east coast of the South Island and most low-lying areas of the North Island have lost most of their indigenous vegetation cover. Changes in the abundance of individual indigenous plant species can also be quantified, but both spatial resolution of the data, some bias in monitoring efforts toward woody vegetation, and differences among regions in the drivers of species abundance changes (both positive and negative) make detecting changes at the scale of a decade modest for many plant species.

Representation is a major underpinning measure for NHMS and biodiversity monitoring schemes (e.g., Lee et al. 2005, Walker et al. 2016, Bellingham et al. 2020, McGlone et al 2020;). Some of the most widely adopted approaches are large-scale mapping of representative environments derived from multiple environmental, climatic and landscape features, but not diversity (e.g., LENZ, TEC, NZ data stack). Moreover, greater inclusion of biodiversity into ecological representation is likely in the future, given this is an Essential Biodiversity Variable (EBV) (see also Schmeller et al. 2018).

Some indigenous plant species can also be invasive, increasing in dominance through land use change, anthopogenically-altered disturbance regimes, or deliberate movement of species. Some examples include:

- The expansion of native tussock grass species following historical burning and removal of woody vegetation cover (Walker and Lee 2000).
- The deliberate introduction of the native tree species *Corynocarpus laevigatus* (karaka) that is now considered invasive outside of its historically naturally distribution (Costall et al. 2006).

Overall, there are excellent, long-term data available for determining current abundance of many indigenous plant species or vegetation-types. This provides a robust base of evidence for developing indigenous plant species dominance as an attribute for EI.

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

Many areas of indigenous-dominated vegetation have baseline information that can be derived from previous measurements or monitoring, relative intact/undisturbed areas of comparable vegetation, and through historical or paleoecological evidence for abundance and distribution of species. In some areas of highest indigenous vegetation loss, little information is available for natural reference states (e.g., MacKenzie basin pre-fire, many lowland forest areas of the eastern South Island). For long-modified areas having little to no reference state based on monitoring data, paleoecological approaches can provide historical baselines for many species (e.g., Wilmshurst et al. 2007, McWethy et al. 2014, Dietl et al. 2015).

Many drivers of changes for indigenous plant species dominance are chronic, driven by ongoing damage by pests, weeds and diseases. Understanding the speed and magnitude of change in indigenous plant species, and how these respond to different management or allocation options, is a major ongoing effort at national and regional scales.

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

There are several numeric and narrative bands established for the cover of indigenous-dominated vegetation cover, and in a few cases, for dominance of species such as:

- A threshold of LCDB categories classified as indigenous dominance (Cieraad et al. 2015).
- Different bands or levels of indigenous dominance as cover are commonly reported (e.g., by RCs, Bellingham et al. 2016).
- Thresholds of indigenous vegetation have been developed at the property and catchment scale of minimum 15% cover, and ideally 30% cover (e.g., Rout et al. 2021).
- Dominance considered as vegetation height has been applied to restrict land clearance of indigenous woody vegetation to <6m. Maximum extent and magnitude of indigenous vegetation clearance is provided in the NES-PF (e.g., <1.5% of area deemed indigenous vegetation, or <30% damage to indigenous tree crowns);
- Implied thresholds from ‘intact’ to ‘degraded’ status of indigenous vegetation are usually driven by declines in native vegetation, or sometimes increases in weed cover (e.g., due to wilding pine invasion; LCDB classes for grasslands, Weeks et al. 2013);
- There are quantifiable tipping points with fire for successional woody vegetation where risk is a non-linear function of vegetation biomass (Perry et al. 2014, Taylor et al. 2017);
- Internationally, there are many different limits and narrative bands established for indigenous plant species (or biodiversity more generally); there is an entire body of literature on this that cannot be covered here, but provides additional conceptual and practical guidance for establishing limits or thresholds for terrestrial biodiversity (e.g., Nicholson et al. 2021).

Nearly all these approaches require species- and system-specific information to set a limit or threshold. Some additional considerations:

- In some cases the potential future contribution of species is used, for example, presence of a late successional tree species at a site is used to classify the potential for the site to become indigenous forest over time (Mason et al. 2010).
- Successional or other directional changes in indigenous plant dominance are widespread, including the long-term decline of broad-leaved tree species whereas emergent podocarps persistent over the long-term following major disturbance (Richardson et al. 2020).
- Many terrestrial vegetation-types contain a mixture of indigenous and non-native species, and the threshold or limits of these reflect the social or cultural value of the vegetation (e.g., retention or restoration of indigenous woody plant species on farms; retention of some weed species as 'nurse crops' for native plant species).
- There are ongoing consequences of land use change and management effects, but allocation options and priorities are not well co-ordinated, particularly between biodiversity and biosecurity management objectives.

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

There are established or potential thresholds, but no universal thresholds across indigenous vegetation types or species.

Minimum abundance or dominance occurs for species to maintain viable populations at the site to catchment scale. For example, minimum dominance for regeneration success (e.g., dioeceous tree species; pollination efficacy). Similarly, there should be density-dependent effects of plant species that provide habitat or resources to other taxa.

Tipping points in cover have occurred in the past, and these can be used to understand future thresholds. For example:

- Past transitions from indigenous forest to non-woody vegetation have occurred widely, dominance by indigenous plant species was maintained throughout; this suggests that knowledge of representativeness and past disturbance is needed to understand the appropriateness of which indigenous species dominance is suitable for quantifying EI.
- In addition, there is now often a different tipping point toward dominance by non-native woody plant species (pines, legumes, thyme) and their long-term dominance suggesting that increases in non-native plant species dominance are a useful and information indicator of EI (see also Sapsford et al 2020; 2022).
- Novel biogeochemical processes, particularly increasing the availability of macronutrients, have occurred. National-scale terrestrial eutrophication has occurred, linked closely to land use management (Parfitt et al. 2012), and high-nutrient availability feeds-back to maintain dominance of mostly non-native plant species (e.g, Dickie et al. 2022).
- Fire traps/altered disturbance regimes have occurred whereby indigenous dominance has declined because of fire, has been replaced by more fire-adapted non-native

species (or few native species like manuka), creating higher risk of future fires. Climate change is thought to exacerbate such feedbacks.

The links between declines of indigenous plant species and human health are less well known, and more indirect effects appear common. For example, shifts towards dominance of some non-native plant species are a major source of allergens (e.g., from pollen of privet and silver birch, and many introduced grasses). Declines in taonga tree species are thought to pose a major issue for community wellbeing, especially of mana whenua (see tree canopy dieback attribute).

For many naturally uncommon or rare ecosystems dominated by indigenous plant species, some tipping points have been observed that are thought to be driven by climate, weeds, increased nutrients, or altered hydrology; these effects are largely considered tipping points by irreversibly disrupting processes maintaining these highly specialised ecosystems (Holdaway et al. 2017). Some require ongoing management.

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

Much of this question is discussed above in A3. Species' dominance is dynamic, responding to past disturbances, environmental conditions, interactions with other species (i.e., competing plant species, herbivory, pathogens). Lag times of decades are typical, for example, post-fire recovery of vegetation or directional successional changes in vegetation (Wyse et al. 2018). Shorter-term lags of few years are possible for some measures of dominance, for example, cover declines driven by defoliation of canopy trees. Historical fires, ongoing land use, and biological invasions each have major legacies that determine the distribution and abundance of indigenous plant species nationally, although these legacies vary regionally from near complete elimination of dominant species (e.g., lowland forests in Canterbury) to relatively minor legacies (e.g., indigenous forests on some offshore islands).

Both lags and legacies strongly affect current state and trends in indigenous plant dominance. Some of these effects are part of natural disturbance dynamics and successional processes (e.g., Allen et al. 1999, Wyse et al. 2018). However, there are several long-term progressive changes including climate and biological invasions. Increasing fire risk is region-specific, and creates feedbacks to more fire-adapted species; most of which are non-native plant species (Perry et al. 2014). Environmental weeds are increasing in number and distribution over time (Brandt et al. 2021), and these invasions can drive declines in indigenous plant species (Sapsford et al. 2020).

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

Although we cannot comment directly on māturanga Māori, we do provide suggestions from our experience that for indigenous plant dominance, there are condition or states described from a Māori perspective. A few points:

- Place-based goals or aspirations for indigenous plant species exist, but these are specific to that site or hapū. For example, recovery of native tree species canopy in the Raukūmara Range following management of hyperabundant deer and goats.

- Species-specific goals are possible; for example, protecting taonga forest species from tree diseases such as Kauri dieback and more recently, myrtle rust. These examples show that protection of individual species (kauri) is crucial for safeguarding wider values across Iwi for enacting kaitiakitangia, and mauri of the ngahere (Lambert et al. 2018). For myrtle rust, an entire family of plants (the myrtaceae) are at risk and this includes structurally dominant tree species; in this case an observational decline in dominance attributed to the disease is an implied threshold.
- There are a few initiatives where biocultural monitoring of diversity, including dominance of indigenous plant species and shifting baselines of species abundance or condition, has been implemented by mana whenua (e.g., Lyver et al. 2019, 2021).

To summarise, place-based goals or acceptable changes in species dominance require community-specific approaches and cannot be directly applied to other sites. The goals or prioritisation of these activities are not driven by national standards or reporting, and as a consequence, scaling up site-based or biocultural approaches to regional or national scales is difficult.

Part C—Management levers and context

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

Dominance of individual species or indigenous plant species within a community or region is strongly related to the state of environment, stresses or management.

Direct responses are easiest to demonstrate, and this can be through direct harvesting or management of species themselves, or land use change altering local population abundance or the species spatial distribution (occupancy; e.g., Weeks et al. 2013, Cieraad et al. 2015). Anthropogenic disturbances such as fire or grazing also directly affect indigenous plant dominance (negatively for most species; Perry et al. 2014). Restoration or enrichment planting (e.g., of riparian margins) can directly increases the dominance of some indigenous plant species albeit at relatively small spatial scales (Forbes et al. 2023).

Indirect responses are commonplace, but far more difficult to demonstrate. These include altered hydrology, nutrient addition/eutrophication, and the impacts of invasive pests, weeds or diseases. Management interventions are commonly deployed for these stresses or drivers, but usually address one problem in isolation from other drivers. Regardless, a common goal of most management is maintenance or improvement in the abundance of indigenous (plant) species, although the relationship between driver, management and response is usually assumed rather than based on robust evidence (Allen et al. 2023).

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

Multiple interventions or interventions are deployed at all scales from site/property through to national that are too numerous to cover here. Most interventions seek to maintain or increase indigenous plant dominance, either explicitly, or at times, implicitly. Illustrative examples include:

C2-(i). Local government driven

- Regional biodiversity and biosecurity plans prioritise pest animal, weed or disease management for indigenous biodiversity benefit (i.e., to maintain dominance at sites or spatial extent; see also Bellingham et al. 2016, 2021, Hansen et al. 2021).
- Consenting processes (e.g., urban development) include explicit guidance on managing impacts on indigenous (plant) species, or recommendations for their inclusion in revegetation.
- Land use planning: maintenance of land cover classes containing indigenous plant species.
- The maintenance and protection of heritage trees includes indigenous species.

C2-(ii). Central government driven

- Central government: indigenous (plant) dominance is a key indicator for biodiversity and environmental monitoring (e.g., for Te Mana o te Taiao).
- Incentives for afforestation (billion trees initiative) seek to increase the abundance of few indigenous tree species.
- Carbon management. Regulation (ETS) and management of permanent or carbon forests includes retention, or an expectation, that indigenous woody plants will increase in dominance over the long-term (Forbes et al. 2023).
- Management of PCL using NHMS: indigenous plant dominance underpins measures of ecosystem condition and representativeness.
- Species recovery plans have been developed for some vulnerable indigenous plant species (DOC).
- Site-based management (DOC, MPI, LINZ): priority is given for high biodiversity value sites.
- Land cover that is 'predominantly' indigenous vegetation underpins the Threatened Environments Classification (TEC) used by both regional and central government to support identification of high-value habitats or vegetation, and to prioritise protection or management (e.g., see Cieraad 2015).
- Predominantly indigenous vegetation is also considered in the NPS for Plantation Forestry (2017) providing guidance for protection as part of commercial forestry operations.
- National Policy Statement for Indigenous Biodiversity (NPS-IB) sets out 10% indigenous vegetation cover for any urban or non-urban environment (with below 10% cover), providing a guideline for minimum level target of indigenous dominance.

C2-(iii). Iwi/hapū driven

Maintenance or renewal of taonga or key plant species is common and is a crucial component of biocultural restoration (Lyver et al. 2019). A few examples:

- For example, Wakatū has established an extensive regional database of indigenous and endemic vascular plants (Harcourt et al. 2022; Foster 2021 https://issuu.com/wakatu/docs/koekoea_issue3_ngahuru_2021).
- Improving the condition of indigenous forests and species in the Ureweras management included developing an intergenerational vision for restoration activities, and established a global precedent of establishing the forest as a legal personhood requiring protection under law (Lyver et al. 2017a, b; McAllister et al. 2019; Reihana et al. 2024).
- Kauri dieback surveillance, interventions, and rāhui on access to prevent future damage to the species by multiple hapū in Northland and the Coromandel (Waipara et al. 2013); see also Shortland T 2011 (https://www.cbd.int/financial/micro/newzealand-monitoring-kauri.pdf).
- Post-pine restoration using indigenous tree species to speed recovery (Forbes et al. 2021).
- Deliberate management to maintain dominance of mānuka for honey production (widespread management activity).

C2-(iv). NGO, community driven

There are too many community or NGO-driven initiatives that seek to increase indigenous plant dominance usually as part of land management or site restoration to describe here. Maintenance or increases in one or more indigenous plant species are common, and have been increasing over the past decade (Forbes et al. 2023). These activities include:

- Removal of grazing animals (retirement of previously pastoral land) for covenants (e.g., through QEII Trust).
- Hundreds of local/community-driven initiatives for restoring indigenous vegetation (e.g., Peters et al. 2015).
- Some NGO's focussed on increasing the number and availability of indigenous plant species (e.g., Tane's Tree Trust).
- Targeted campaigns by Forest and Bird, including advocating for additional pest animal management for carbon gains and improved ecological condition of indigenous forests (Hackwell and Robinson 2021).

C2-(v). Internationally driven

There are numerous international agreements and obligation for biodiversity, and most include goals for protection of condition and extent of indigenous-dominated vegetation or habitats. We provide only a few examples here:

- Indigenous species dominance is considered an EBV, and thus essential for reporting and meeting international agreements such as the Convention on Biological Diversity and IPBES (see Bellingham et al. 2020).

- Emerging biodiversity credit markets rely heavily on validation of investment using indigenous vegetation cover or management as evidence for biodiversity protection and improvement.
- 'Red-listing' of naturally rare ecosystems have been done qualitatively to meet IUCN criteria of threats to these high biodiversity value ecosystems (Holdaway et al. 2017), and more quantitative approaches across additional indigenous-dominated ecosystems are currently being developed (in collaboration with DOC and MfE).
- The Kunming-Montreal Global Framework for Biodiversity Target 3: "Ensure that at least 30 per cent globally of land areas and of sea areas, especially areas of particular importance for biodiversity and its contributions to people, are conserved…"
- The EU biodiversity strategy has a 2030 goal of legally protecting at least 30% of EU's land, and strictly protecting 10% of land. They have also proposed a new law "Nature restoration law" which calls for binding targets to restore degraded ecosystems. This sets an international standard for protection.

Part D—Impact analysis

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

Not managing indigenous vegetation cover or maintaining dominance of indigenous plant species would lead to continued biodiversity loss and reduction in the extent and representativeness of indigenous species and ecosystems (Ewers et al. 2006, Cieraad et al. 2015, McGlone et al. 2020). This impact would be particularly problematic in already highly reduced environments such as lowland indigenous forests and wetlands.

We would also expect declines in many ecological processes, additional future problems (e.g., accelerated loss of species through extinction debts), and greater risks of crossing thresholds or tipping points. For example, knock-on declines in other species dependent on indigenous plant species (e.g., specialist insects and birds), or greater risks from pests, weeds and diseases for more fragmented or isolated populations or patches of indigenous vegetation (see Landscape Connectivity and Canopy Tree Dieback extent templates).

Human health impacts are largely unknown, but likely to be trivial in most cases (with the exception of declines in foundational or taonga species).

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

Few direct economic impacts are likely. Forestry land uses rely almost exclusively on non-native tree species. However, permanent or carbon forestry owners may enjoy an as yet unknown financial benefit or premium for increasing indigenous woody plant species in marginal sites through carbon or emerging biodiversity credit markets. Pastoral system also rely primarily on non-native herbaceous species, even though some systems also contain indigenous plant species; there are only a few cases where benefits of increasing native plant species could be beneficial (e.g., better animal welfare in pastures containing native shrubs like matagouri, *Discaria tomentosa*).

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

Climate change will affect indigenous plant dominance both directly and indirectly, but most of these effects cannot be managed. Direct effects relate to species' environmental tolerance and distribution, such that dominance could shift throughout the current range, and some species could adjust their range (e.g., upslope adjustment of subalpine species).

Multiple indirect effects of climate are likely, highlighting the problem of the 'twin crises' of climate change and biodiversity loss. Fire regimes are poised to worsen with increases in dry cycles and both natural and anthropogenic ignition sources; most indigenous species are not fire adapted, and thus their dominance is poised to decline with increased fire frequency or intensity. Rapid fire management. Storm damage can exacerbate the impacts of pests, weeds and diseases, including from other native species (e.g., as shown by outbreaks of native beetles causing canopy collapse of tree species following canopy disturbance, Wardle and Allen 1995). The number and distribution of environmental weed species is thought to be exacerbated by warmer conditions and disturbance, creating opportunities for non-native plants to suppress or replace indigenous plant species during successions (e.g., in coastal zones, following glacial retreat, post-fire).

Although the effects of climate change itself cannot be managed, landscape-scale approaches to planning and development can be used. For example, early work shows that land use and species selection can be used to promote more 'climate smart' landscapes in the future that could support dominance of indigenous plant species. Climatic refugia can be considered even for common taxa and included in biodiversity planning. Post-disturbance biosecurity efforts could be timed to reduce the impacts of invasive mammals or weeds. Longer-term, and for some taxa, new technologies could also be deployed to maintain dominance (e.g., RNAi inoculation of rata for myrtle rust). Finally, broader scale interest in afforestation using indigenous tree species could be developed (e.g., by scaling up direct seeding of native species to overcome dispersal or reproductive barriers to range adjustment under climate change; Douglas et al. 2007).

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6.7 Canopy die back extent

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Preamble: Forest canopy dieback extent is a well-established attribute related to the ecological integrity of forest ecosystems. Canopy dieback directly affects the physical structure of forest vegetation, and alters major ecosystem functions like primary productivity and nutrient cycling. Historical high-profile areas of tree dieback in New Zealand have driven efforts to quantify changes in tree canopy condition resulting from both invasive animals (e.g., possum browse and responses to management) and other observed canopy collapse of forest stands (e.g., native beetle outbreaks in *Nothofagus* forests). More recently, both novel pathogens (e.g., myrtle rust) and climate-related increases in disturbance (e.g., cyclone damage to canopies; drought-induced stress or tree mortality) have increased interest in understanding changes in forest canopy condition, the maintenance or recovery of key tree species, and the long-term impacts on forest ecosystems. Quantification of canopy condition have well-established protocols and data collection methods, and emerging technologies in remote sensing (high density LiDAR and hyperspectral imagery) have the potential to better characterise changes through time and at greater spatial scales. Given the multiple potential drivers of forest canopy condition or collapse, this attribute requires additional information to interpret the causes of canopy collapse, and distinguishing between natural canopy declines (i.e., caused by succession) from new and sometimes manageable threats (i.e., emerging diseases, invasive herbivore damage). Most current data collected follow from discrete dieback observations of one or more species at a site; but whether these issues occur at broader spatial scales, or cause longer-term declines in forest condition and ecosystem integrity require more systematic approaches to measurement, and understanding of whether regeneration is offsetting canopy dieback and tree mortality.

State of knowledge of the "Forest Canopy Dieback Extent" attribute: Good / established but incomplete – general agreement, but limited data/studies

Part A—Attribute and method

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

Forest canopy dieback extent is a direct measure of forest structure or state occurring across one or more canopy tree species. As a consequence, this attribute should relate to all components of ecological integrity (EI) including representation and composition (where dieback is related to increased species-specific mortality; discussed below), indigenous dominance (structure, by altering canopy structure) and functions (through alteration of ecological processes such as primary productivity or reproduction, often resulting from altered community composition). Moreover, given the importance of canopy tree species for driving forest ecosystem processes, this attribute will also indirectly be related to human health through alteration of some ecosystem services, and potentially well-being through values of forest health or condition, or direct risk from canopy collapse (although these possibilities have rarely been investigated).

In terms of human health, a reason this attribute commands attention is because people dislike seeing large areas of dead trees. Most people with only a passing knowledge of natural forests often don't appreciate that large-scale dieback of trees often reflects their large-scale recruitment at some stage in the past and them all reaching, simultaneously, a physiological stage where their canopies cannot be maintained. Moreover, most people do not grasp that decay rates differ among dead tree species such that some remain visibly dead in landscapes for at least 50 years (Peltzer et al. 2003; Mason et al. 2013; Fig. 1). Therefore, a key need is to draw distinctions (where it is possible) between natural processes of canopy dieback and those that are induced by either modern pressures (e.g., climate change) or novel pests and pathogens, some of which can be mitigated (see also Wyse et al. 2021).

The spatial scale and magnitude of canopy dieback extent, and over what time period this occurs, matters. For this exercise, canopy dieback is described as population-level (i.e., canopy dieback of individual trees is out of scope; Mueller-Dombois 1985). We are uncertain when a treatment of species-specific population-level dieback in a mixed species forest where most species remain alive constitutes canopy dieback (i.e., declines of canopy cover in one species are often compensated for by increases in canopies of other species). For example, does death of most kohekohe attributable to browsing possums, or death of taraire or mamaku attributable to drought but where most other trees remain alive constitute canopy dieback? We think not, but this raises the issue that most canopy dieback measurements consider individual species, whereas EI should be more closely linked to overall forest canopy condition (see Tierney et al. 2009; see more detail comments below on methods). We suggest that canopy dieback is best measured at the forest stand-level or catchment scale, and represents death of structurally dominant tree species, and could be reported at the regional scale for widespread forest types.

Figure 1. Long-persistent, slow-decaying southern rātā c. 40 years after its canopy dieback in the Kokatahi River valley, central Westland (Photo, Peter Bellingham, 1997; Mason et al. 2013).

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

The spatial extent and magnitude of canopy dieback determines impact on EI (e.g., Tierney et al. 2009).

Maintenance of dominance or shifts in dominance. We know of no examples where indigenous tree species dominance has been lost as a result of canopy dieback. Compositional shifts can occur whereby declines in the canopy cover or extent of one species are replaced by increases in one or more other tree species (Fig. 2), but not always (i.e., canopy dieback in beech canopies usually results in replacement by the same species; Wardle and Allen 1985, Ogden 1985, 1988). We are not aware of any stand-level or catchment-scale tree canopy dieback that results in non-native plant invasions, but this requires further investigation. Furthermore, major canopy dieback or disturbance events such as cyclones coupled with increased numbers of environmental weed species could increase invasions into indigenous forests in the future. In terms of bird communities, canopy dieback can provide new habitat for birds that rely on wood decay (i.e., insect prey) and can provide nesting sites for hole nesters.

Representativeness. There is no evidence of loss of representativeness of forest composition caused by stand or catchment-scale canopy dieback, but this has not been well investigated. Novel pathogens could change this through selective damage to few tree species (e.g., as has occurred for Dutch elm disease or chestnut blight elsewhere).

Function. Changes in function are likely, but have not been investigated. Because canopy condition regulates major energy fluxes and nutrient cycling in forest ecosystems, major canopy dieback or population declines should affect multiple ecosystem functions and ecological processes. Declines or loss of 'foundation species' is of particular concern (Ellison et al. 2005, Genung et al. 2020).

Evidence of impact on human health. We are not aware of direct linkages between tree canopy dieback extent and human health. Visual assessment of forest or bush 'health' is predicated on a healthy forest being one without dieback, and this represents visual landscape value to people (e.g., Handford et al. 2021). Perceptions of the public to environmental condition have also been evaluated used repeated national surveys, but canopy dieback is not considered separately as a part of forest health or condition (see https://www.landcareresearch.co.nz/discover-our-research/environment/sustainable-society-and-policy/environmental-perceptions-survey/).

Figure 2. Reduced forest structure after canopy dieback, Pohangina Valley, Ruahine Range. Podocarps (mostly miro, *Pectinopitys ferruginea*, and rimu, *Dacrydium cupressinum*) that were formerly in or emergent above a canopy dominated by kāmahi (*Pterophylla racemosa*) show the former height of the canopy that showed widespread canopy dieback in the 1950s (Rogers and Leathwick, 1997), while the wholly native community present now (mostly māhoe, *Melicytus ramiflorus*, and kātote, *Cyathea smithii*) is much shorter (Photo, Peter Bellingham, March 1996; Bellingham et al. 1999). Rogers and Leathwick (1997) attributed this canopy dieback to browsing by possums and their non-replacement in the new canopy to browsing by goats and red deer.

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

There is little evidence of change over the past few decades generally. Some exceptions include:

- Kauri (*Agathis australis)*, where increased canopy death is frequently attributed to a phytophthora pathogen (Kauri Dieback - *Phytophthora agathidicida*). However, the trajectory of change is complicated by previous concerns that canopy death was often attributed to illegal bleeding of gum in the 1920s or earlier (Orwin 2019), and this may interact with the current pathogen. Regardless of the original cause, all size and age classes of kauri are susceptible to infection and death (Bellgard et al. 2016, Bradshaw et al. 2020).

Figure 3. Kauri dieback is a high profile disease causing tree canopy tree dieback (https://www.kauriprotection.co.nz/).

- A plant bacterium, *Candidatus Phytoplasma australiense*, spread by insects has selectively damaged and caused dieback ('sudden decline') of mature cabbage trees, but most young trees are left untouched (Beever et al. 2006). The magnitude of dieback has declined over the past 25 years, and varies regionally (Brockie 2020).

- The recent incursion by Myrtle rust on an entire family of plants (Myrtaceae) over several regions has the potential to create major dieback of indigenous tree species including structural dominant species (Teulon et al. 2015, McCarthy et al. 2021).

- Climate change is likely to interact with other drivers such as herbivores or resident pathogens to increase the frequency, extent or magnitude of canopy dieback extent (e.g., McCarthy et al. 2021). Novel pathogens (e.g., rapid ʻōhiʻa dieback) are highly likely to further compound causes of canopy dieback at the decadal timescale.

These effects are reversible only if canopy dieback does not result in population-level declines beyond background variation. If canopy dieback increases tree mortality but also increases growth or reproduction, the net demographic effects can be neutral. This highlights that canopy dieback is one (visible) aspect of more complex forest dynamics, and additional information on tree regeneration and threats or management are needed to understand if the impacts of canopy dieback are reversible. As an example, seasonal or temporary loss of tree foliage and canopy condition is common, but this is reversible over one or few years if individual trees survive. In contrast, canopy dieback resulting in increased tree mortality, coupled with regeneration failure, would be the worst-case scenario and unlikely to be reversed over several decades given the longevity of most indigenous tree species (see also McGlone et al. 2016, 2017).

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

Nearly all canopy dieback assessment, whether stand-based or catchment-based, focusses only on one demographic component of populations (i.e., tree mortality). Without commensurate assessment of population recruitment, the effects on population maintenance, forest ecosystem condition, and thus EI are unknown. Most scales of measurement or monitoring are relevant: site, catchment, regional, and national. Stand-level monitoring is done using different approaches including Foliar Browse Index (FBI) (canopy cover), standard plot-based vegetation measurements, and remote sensing (including LiDAR). Systematic assessment is rarely done among regions, but has been established with sites or catchments for some long-term plot-based studies (e.g., tier two networks, DOC) that occur in areas of canopy dieback and provide population-level assessments of consequences (e.g., Bellingham et al. 1999a; Richardson et al. 2024). Some of these assessments have shown that the mortality rates of some canopy trees are exceeded by recruitment rates of new stems, e.g., of kāmahi in four Westland valleys in which it has shown widespread canopy dieback (Bellingham and Lee 2006; see also Allen and Rose 1985, Rose et al. 1992). Nearly all monitoring is done at the site or catchment-scale but could be scaled up to report at regional or national scales. The main barrier to scaling up monitoring is variation in monitoring efforts among regions, rather than available methods or standards. Canopy dieback is often not the focus of monitoring aside from site- and species-specific areas of concern but can use information collected for biodiversity assessment or C accounting purposes.

Given the complementary ground- and remote-sensing based approaches to monitoring, there are opportunities for monitoring and reporting that link these information sources (e.g., Meiforth et al. 2020). Major efforts internationally to better link remote sensing or earth observation data with plot or ground-based monitoring have generated several emerging indicators of EI related to canopy condition or dieback such as the Forest Structure Condition Index and Lost Forest Configuration (see summary of Hansen et al. 2021). In many cases the underpinning data needed to develop these attributes is available in NZ, but integrating these data require additional research effort and implementation. Similarly, linking plot-based and remote sensing methods is an ideal for EBV's, suggesting there are efficiencies to be made with biodiversity monitoring and reporting (Bellingham et al. 2020).

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

Implementation of data collection is well established through major monitoring efforts such as LUCAS and regional plot networks (Richardson et al. 2024). Practical or logistical considerations (not barriers *per se*) include:

- botanical skills (for identifying indigenous taxa) and maintenance of qualified personnel (i.e., for field measurements, data collection, and analyses).
- provision is in law via the RMA to collect data, but can be logistically difficult for some sites or communities.
- repeated samples to evaluate changes in dominance require databases, long-term archiving, and access. Increasingly, transparent documentation of data processing, analyses and interpretation is required or expected (for both reporting and publication purposes). Similarly, explicit evaluation of assumptions and uncertainty in the data or analyses are required.
- Remote sensing methods and new technologies (e.g., eDNA) are not immune to these issues; all increasingly require informed consent for data collection, analyses and reporting.
- Intellectual property and data sovereignty issues are a potential barrier to data collection, use and access, and require ongoing consideration as part of monitoring efforts.
- A practical barrier is the lack of sustained/long-term commitment for collecting the primary data by most RCs through lack of funding or prioritisation of efforts elsewhere (i.e., as highlighted in multiple investigations by the Parliamentary Commissioner for the Environment).
- Current monitoring efforts do not have complete national coverage. There is a bias in data collection against lowlands and rapidly changing (marginal) land use classes. Most reporting has focussed on PCL because one lead agency (DOC) implements monitoring, where potentially changes in indigenous plant dominance have the slowest/modest change. Spatial coverage of data in other land use classes is poor because of multiple agencies involved, has lower priority for many regions compared to other competing issues (e.g., water issues).

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

Costs per plot including data management are well discoverable from DOC and RCs, but do not distinguish costs for measurement of indigenous plant species dominance from other measurements (e.g., diversity, pellet counts, deadwood assessment) carried out at the same time. Individual plots range in cost from $2000-5000 depending on access and complexity. Aside from central and regional government, multiple agencies have the infrastructure, skills and ability to monitor this attribute including Manaaki Whenua and Wildlands. Additional costs for quality assurance of data collection and analyses are also available.

Efficiencies can be made with data collection and analyses for other needs. For example, remeasurement of several networks of vegetation plots on Rakiura have been prioritised by DOC (maximizing forest C programme) because interest in potential C loss from forests dominated by palatable tree species such as kāmahi, and for potential responses to planned predator control operations and eradication across the island. Additional networks are now being considered for remeasurement, and much of the information gathered for this activity (Richardson et al. 2024). There are also opportunities for efficient re-use of data collected for other monitoring purposes to quantify changes in canopy condition; here, the cost of data collection and hosting is already covered.

Detailed costs for other monitoring approaches such as remote sensing data are available. Costs of data acquisition are available, but depend on provider and data quality captured. Additional costs of specialist skills for data processing, analyses and interpretation are highly variable depending on the scale and reporting needs; normally the purpose is not to monitor indigenous plant dominance but to estimate land (vegetation) cover in which some cover classes are dominated by indigenous species (Cieraad et al. 2015).

Direct costs of monitoring vegetation plots or canopy condition are well documented, however, additional interpretive data to understand attribution or manage the causes of canopy dieback is not included in these estimates. Single drivers of dieback are rare, requiring this additional effort.

Human perception of dieback is a key driver of monitoring or interventions (e.g., cabbage tree dieback), so there is a largely unknown opportunity and cost of including communities or citizen science to monitor canopy dieback (see also https://www.landcareresearch.co.nz/discover-our-research/environment/sustainable-society-and-policy/environmental-perceptions-survey/).

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

There are few examples of canopy dieback being monitored by Iwi/Māori. See Raukūmara Pae Maunga, one stated objective is to monitor canopy recovery across habitats (see https://www.raukumara.org.nz/objectives).

Relatively high-profile research programmes such as Ngā Rākau Taketake (https://bioheritage.nz/about-us/nga-rakau-taketake/) explicitly supported community-based responses to these pathogens (Lambert et al. 2018). Kauri dieback has been monitoring by several communities in Northland and the Coromandel, driven in part by Kauri being considered a cultural keystone/taonga species. The recent invasion by myrtle rust (*Austropuccinia psidii*) threatens a range of taonga tree species (Teulon et al. 2015). These diseases have prompted recent work on Mātauranga Māori approaches to surveillance including researchers directly working with kaitiaki and rangatira from ten tribal regions, whose taonga are affected by myrtle rust and kauri dieback to monitor the presence and impacts of these diseases (see https://bioheritage.nz/research/integrated-surveillance/). See also https://bioheritage.nz/new-forest-health-tool-helps-mana-whenua-capture-culturally-important-data/.

More generally, cultural frameworks to monitoring, including biocultural approaches (Lyver et al. 2019), do not focus on a specific attribute or metric, but could include specific indicators as part of more integrated assessment of forest condition, or ngahere ora/mauri state (Waipara et al. 2013; Reihana et al. 2024).

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

Tree canopy dieback extent is directly linked to indigenous plant dominance for indigenous forests. Canopies are formed by dominant tree species, so declines in canopy extent directly reflect change in biomass or dominance of canopy tree species. This relationship works at a stand or catchment scale, but at regional or national scales, it is the spatial extent of canopy dieback affecting species distribution and abundance that is related to dominance (i.e., species occupancy and range). Lowland forest extent should related to canopy dieback through several mechanisms but this requires investigation. These include forest fragmentation effects, increased edge effects, and closer proximity to novel weeds, pests and diseases, all of which can increase canopy disturbance (and suppress tree species recruitment in many cases). As a specific example, lowland forest fragments are often disturbed, increasing the invasion of some understorey weeds (e.g., *Tradescantia fluminensis*) that, in turn, suppress tree recruitment, leading to canopy and population declines over the long-term (Standish et al. 2001).

Contemporary drivers of declines in tree canopy condition include invasion herbivores (primarily possums) and increased disturbance (including fire, pathogens, and storms). Disturbance itself is not an attribute, but better information on disturbance is needed to interpret changes in canopy condition or dieback. In contrast, attributes that consider the distribution, abundance and impacts of pests (foliar herbivores) and diseases should be directly, positively related to canopy dieback. These relationships will be species-specific because both herbivores and diseases damage species or functional groups of species differently (Nugent et al. 2000).

Part B—Current state and allocation options

B1. What is the current state of the attribute?

There is no consistent regional or national-scale reporting of tree canopy dieback extent. Most reporting is for a focal species of concern at one or few sites. No long-term commitment to remeasurement is currently in place. However, other measurement or monitoring schemes can be used to derive canopy condition or potential dieback from remeasurements, including the national vegetation plot network (DOC tier one; LUCAS) and several regional networks (Richardson et al. 2024).

Current measurement activities together with management could be used to develop this attribute as an indicator of ecological integrity. For example, comparing plot-based or remote-sensed measures of canopy condition between management units for herbivore control, predator elimination, or presence of pathogens could be used to establish baselines of dieback, and responsiveness of this to management interventions (including failure to manage; see also Peltzer et al. 2024).

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

Baseline or reference states for canopy condition, dieback and long-term changes in tree populations are available from several sources. These include:

- **Paleo-ecological information.** Long-term pollen records of stand to catchment-scale dieback. Multiple species and dieback events have been observed. This demonstrates that some tree species have gone through canopy dieback in the past (McGlone et al. 2016, 2017).
- **Plot-based measurements.** National to regional-scale networks of temporary and permanent vegetation plots focus largely on indigenous forests, and usually provide cover estimates of species. These provide excellent quantitative estimates of variation in tree canopy cover along major environmental gradients. This information can be used in combination with other information for management or disturbance history effects on different tree species. For example, evaluating the spatial impacts of an earthquake on canopy condition and mortality of mountain beech forest (e.g., Allen et al. 1999).
- **Observations.** Most canopy dieback is a visible feature of forest condition. Semi-quantitative or qualitative measures can be used to compare changes in 'healthy' vs. 'dieback' forest stands using repeated images, perceptions of forest health, or observations (e.g., Jamieson et al. 2014).
- **Remote sensing** methods including aerial photography, high-density LiDAR and hyperspectral imagery provide data for evaluating canopy condition and changes through time at larger scales. These approaches can be used to detect overall changes in canopy cover, leaf area or 'greenness' (NDVI), but usually do not distinguish responses of different tree species. However, these approaches have been used to monitor changes in some emergent canopy species such as kauri, to quantify extent of canopy dieback.

These approaches can be used to understand canopy condition or dieback across different scales, time periods and resolutions. Some specific measurements include quantification of structure (vegetation cover by species, live cover fraction, foliar browse index (FBI), Foliar cover index (FCI), leaf area index (LAI)) and function (NDVI, Aboveground Net Primary Production (ANPP)). Only plot-based measurements can be used to understand whether tree canopy dieback is a symptom of population declines and longer-term changes in EI.

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

There are no current bands or limits for canopy dieback extent. However, thresholds or limits could be developed from existing data, and should be guided by population processes (i.e., to related canopy condition to increased tree mortality rate for different species, and to evaluate whether mortality greatly exceeds recruitment rates; see also Table 1). Given the high variability among key tree species in life history strategies such as response to disturbance, longevity, and susceptibility to herbivores, pathogens or other drivers of canopy condition (Allen et al. 2003, 2013, Richardson et al 2009), bands or limits will vary among species and potentially regions. The LUCAS plot network data could be used to understand baselines and changes at national to regional scales.

Table 1. Summary of canopy dieback. Although no quantitative bands are currently available, current data could be used to establish an ordinal scale of dieback extent for different species, forest classes, or regions. Both the canopy lost and duration of declines should be used in combination to establish bands or thresholds for describing canopy dieback extent. Spatial scale is not explicitly included, but these criteria could be applied at any spatial scale of interest.

Canopy dieback extent	Proportion canopy lost	Duration of canopy loss	Impacts on forest ecosystem integrity	Notes on potential causes of dieback
Low	<10%	<1yr	Negligible	Seasonal variability or minor disturbances. Could be an early warning of pathogens requiring surveillance.
Moderate	10-30%	1-2 yrs	Variable	Requires ancillary information to understand loss. Can result from weather events, environmental stresses. Warrants evaluation.
High	>30%	3+ yrs	Variable-high	Increased tree mortality and population declines likely. Prioritise for monitoring or intervention.

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

No. However, data are available to assess the links between canopy dieback extent and tree population declines, and this would reveal thresholds and tipping points for forest structure, function and EI. Similarly, the links to human health have, to our knowledge, not yet been investigated explicitly, but could be. For example, the relationship between visible tree death and peoples' wellbeing could be evaluated for canopy dieback for different species or locations. The narratives for kauri dieback recently imply community concerns about forest health and the wellbeing of communities (Lambert et al. 2018).

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

There are multiple causes of tree canopy dieback extent, and these include several lags and legacy effects that cannot be covered in detail here (but see Mueller-Dombois 1985). The major considerations include:

- Recognising and distinguishing seasonal or short-term tree canopy declines that do not cause increase tree mortality or overall population declines. This can include seasonal changes in canopy condition, short-term damage from wind, snow or browsing, and resident pathogens.
- The longevity of visible death varies widely among species, from a few years to many decades (Peltzer et al. 2003, Mason et al. 2010).
- Past, infrequent disturbance is an important legacy; many of our tree species recruit as a cohort following disturbance, but then also die as a cohort. For example, podocarps following volcanic eruptions (Richardson et al. 2009; Smale et al. 2015). Thus, infrequent historical and contemporary disturbances are a major driver of tree population structure and canopy condition.
- Tree population processes generate multiple lags including in recruitment, growth and mortality, often spanning several decades.

Despite these and other lags and legacies involved in forest dynamics (and thus canopy condition), relatively rapid changes in canopy condition or dieback occur that can be attributable to specific causes (like novel pathogens, increased or chronic herbivory by invasive animals). As a consequence, state and trend analyses will be relatively easy to apply at stand or catchment scales, but more difficult to disentangle from background processes at regional or national scales (with some exceptions, like the broad impacts of new diseases, Jo et al. 2023).

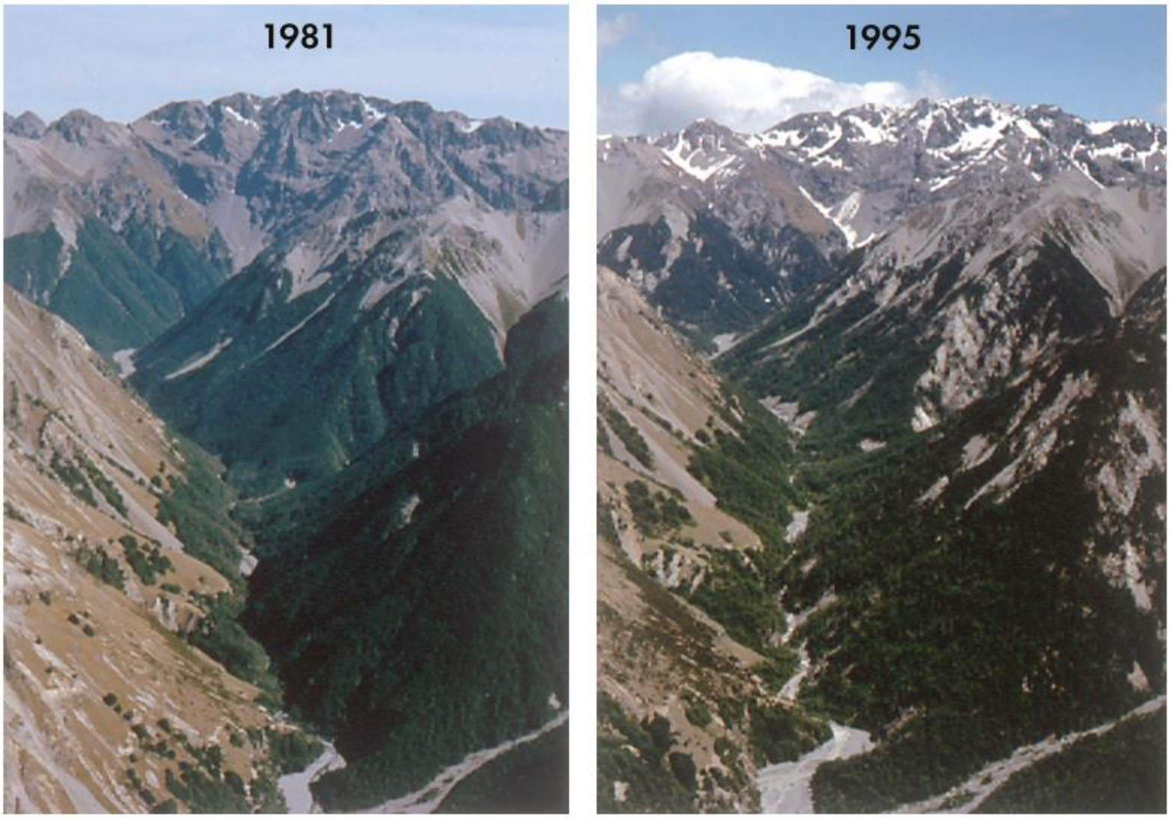

Figure 4. Major lags and legacies drive changes in canopy dieback extent including major natural disturbances like earthquakes. Ongoing mortality of trees and canopies can occur for several years following disturbance (from Allen *et al*. 1999 *Ecology 80*, 708–714).

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

Both tikanga Māori (e.g., for biocultural monitoring) and mātauranga Māori (e.g., for understanding changes in forest condition) are emerging as crucial approaches for evaluating the condition of the environment and people, interdependencies, and acceptable conditions or goals of management (Lyver et al. 2019). Forest canopy dieback extent is a specific concern, and has driven several activities in recent years including:

- Mana whenua led or inclusion in surveillance and management of kauri dieback and myrtle rust.
- Co-ordination or inclusion of Iwi in monitoring or strategic long-term management of long-term forest declines (e.g., in the Raukūmara and Kaweka Ranges).
- Greater inclusion of Māori in major conservation efforts such as predator free (collaborative groups) and Te Mana o te Taiao can be linked, at least in part, to changes in forest health and tree canopy condition. Although specific bands or allocation options have not yet been developed, these efforts provide the system-level approach needed for mana whenua to identify targets, limits and thresholds.
- More generally, the scale of canopy dieback processes is usually at the stand- or catchment-scale, and this matches the scale of concern by hapū and Iwi for aspirations (e.g., Tūhoe goals, management and monitoring of Te Urewera; Lyver et al. 2017, 2018).

Part C—Management levers and context

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

Distinguishing among the multiple, and sometimes interacting drivers of tree canopy declines is complex (see B5). Tree canopy condition or dieback can be driven by cultural disturbance (harvesting), biological invaders (possum browsing, pathogens, disease), climate change (increased wind or storm damage, increased fire risk), and land use change (including forest fragmentation). Although there is a wealth of information and knowledge for forest dynamics and responses to many of these pressures, this information has not been used to relate state of the environment to canopy dieback, but rather consider individual stresses individually, and in some cases, the effectiveness of management interventions. For example, canopy defoliation by invasive possums has been widely quantified, and this information used to relate defoliation intensity or duration to tree mortality for some tree species (Urlich and Brady 2005, Gormley et al. 2012, Sweetapple et al. 2016); but whether stresses interact or compound changes in canopy dieback (e.g., defoliated trees are more suspectable to pathogens) is largely unknown.

All of the stresses mentioned could increase canopy dieback extent, and some are manageable. Relating invasive mammal impacts to forest condition, or to forest recovery following their management is a long-term issue (Husheer and Tanentzap 2023). Recent interest in linking pest animal management to C sequestration provides ample information that could link management to tree canopy dieback because the data collected to assess ecosystem C usually includes measures of canopy cover for different species at regional to national scales (see Holdaway et al. 2012, Peltzer et al. 2024). Cultural disturbance (harvesting) is trivial because forestry involving indigenous tree species was largely stopped in the mid 1980's (McGlone et al. 2022). Land use and management could contribute both negatively (e.g., through fragmentation of vegetation) or positively (e.g., restoration, enrichment planting), but the effects of this on tree canopy condition or dieback have not been assessed.

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

Visible death has driven management/interventions, but often to no quantifiable effect. This is, in part, due to a lack of well-designed monitoring of canopy condition, and additional information needed to evaluate the efficacy of management interventions.

The most common intervention to date has been possum control following observation of dieback for few canopy tree species. Often herbivore control is applied at local (stand or site) scales, and can be effective for increasing canopy health and reducing tree mortality (e.g., Nugent et al. 2000 documented recovery of kohekohe). Often both herbivory and control are repeated, but few studies have considered how long herbivory or management is needed to improve canopy condition (but see Payton et al. 1997, Holland et al. 2013, Sweetapple et al. 2016).

Possum management can improve canopy condition and foliage cover, ultimately improving the survival of affected tree species. The best data available were part of a 'how long' study at Waihaha forest (Sweetapple et al. 2016), in which the canopy condition of 4 tree species were assessed following possum management (or unmanaged experimental controls) over 20 years. that showed:

- Three possum-palatable tree species had increased foliage cover and reduced crown dieback over the 20 years of monitoring.
- Increases in foliage cover were modest (8–19%), but consistent with other studies (Nugent et al. 2010; Duncan et al. 2011; Gormley et al. 2012).
- Canopy recovery of heavily browsed tree species took about a decade.
- These results reflect that less than half (20–49%) of trees were possum browsed at the start of the study.
- Trees that were initially heavily browsed by possums had large increases in foliage cover (36–89%) during the first 6 years of the study.

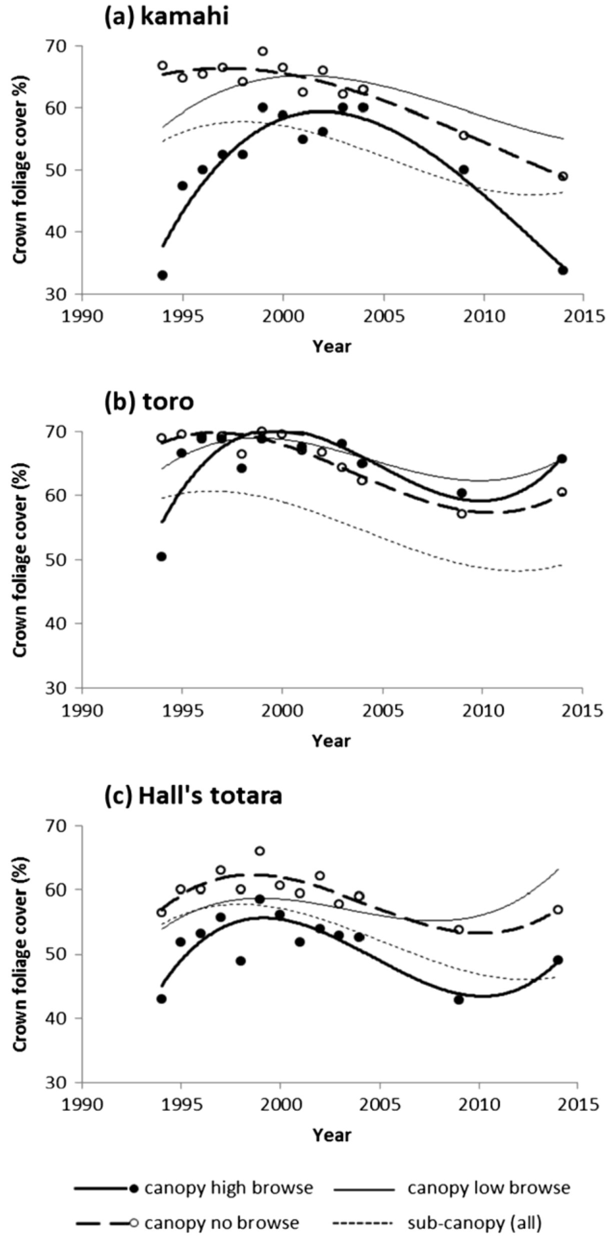

Figure 5. Interannual trends in mean foliage cover for three tree species at Waihaha. Data for each species are divided in 4 groups based on 1994 possum browse class (none, low, high). Data points are shown for canopy trees in the high browse (solid circles) and no browse (open circles) classes. From Sweetapple (et al 2016).

A few recent management options have been developed to manage kauri dieback, ranging from rāhui on access to prevent movement of the disease, phosphite injection to increase the health of infected trees (Horner et al. 2016), and Mātauranga Māori approaches using rongoā derived from indigenous plant species. All of these approaches could maintain or reduce canopy dieback of kauri individuals or sites, and appear effective over the short-term (<5 years).

How such management effects scale up to the catchment- or regional-levels has not been evaluated, but could be done by integrating management information (i.e., mapped areas of aerial possum control alongside measures of possum abundance) with repeated measures of canopy condition (see also Peltzer et al. 2024).

Early or major impacts of invasive herbivores have occurred, but this is largely restricted to observations rather than based on monitoring efforts. For example, some successional species are thought to have been locally extirpated by herbivores (i.e., fuschia, tutu).

C2-(i). Local government driven

- Auckland Council are actively monitoring integrity, and have been actively managing several current or potential tree diebacks in the region; these are included in state of the environment reporting (Griffiths et al. 2021).
- Kauri dieback. Auckland and Northland councils have active programmes for awareness, policies to limit movement of pathogens including rāhui, and have carried out direct operations (e.g., phosphite injection of trees) to protect this species. All of these interventions are relatively short-term solutions for managing the disease, and more strategic/long-term solutions are sought.
- Cyclone damage (e.g., Ita in Kahurangi, Cyclone Gabrielle in Hawke's Bay) has required several councils to prioritise protection of indigenous forests, improve monitoring, or consider the impacts of these storms on land use.
- Cabbage tree decline (Beever et al. 1986, Brockie 2020) was a major national (primarily North Island) and regional concern that prompted many councils to carry out additional monitoring, awareness campaigns, and some management to contain spread of the disease.
- Often local dieback events or perceptions have been raised by communities or councils, and used to prioritise management of presumed drivers of decline. This has most often been possum control, and the effectiveness of such management for improvements in canopy condition or tree population improvement are rarely considered (e.g., kaikawaka/mountain cedar declines on Mt. Taranaki; beech dieback at Moa Stream canterbury, rātā-kamahi dieback in parts of Westland).

C2-(ii). Central government driven

Although most management is targeted at sites/stand-level dieback, at times concern for broad-scale dieback requires more co-ordinated approaches nationally such as:

- myrtle rust response, mostly aimed at understanding the vulnerability of key indigenous tree species to this pathogen.
- kauri dieback has required both DOC and Biosecurity NZ to manage this taonga species through awareness campaigns, restricting access to vulnerable sites, and monitoring tree canopy condition (see also https://www.mpi.govt.nz/biosecurity/exotic-pests-and-diseases-in-new-zealand/long-term-biosecurity-management-programmes/protecting-kauri-from-disease/).
- concern over potential broad-scale declines in key tree species such as kāmāhi using the national network of vegetation plots. Current plans and co-ordinated budgets to address this decline are in progress (e.g., via the 'Maximising Forest C' programme involving DOC, MPI and MfE).

C2-(iii). Iwi/hapū driven

Several examples of Iwi or hapū management include:

- Kauri dieback and access to sites or areas using rāhui.
- Aspirations and long-term plans for improving forest health, and linked cultural values (Tūhoe Tuawhenua Trust management of Te Urewera).
- Raukūmara Pae Maunga Trust seek to monitor indigenous forest canopy recovery across habitats within the Raukūmara Range, largely resulting from goat and deer control efforts (https://www.raukumara.org.nz/).
- Māori community responses to myrtle rust (Black et al. 2019).

This is not an exhaustive list, but indicates that community/hapū approaches to management of forests are widespread. In many cases, Iwi/hapū driven approaches utilise mātauranga Māori and conventional scientific understandings for planning and monitoring efforts.

C2-(iv). NGO, community driven

Several community-driven efforts to raise awareness and intervene or manage canopy dieback have occurred such as:

- Campaigns by Forest and Bird raising the issue of national-scale forest condition, and potential declines or dieback of tree species such as kāmāhi, invoking pest animals as the primary driver of these declines that require management.
- Public concern over rātā-kāmahi dieback: several campaigns or efforts over perceived dieback of these species several regions including Westland, Rakiura, Bay of Plenty (Pūtauaki).
- Several community-based activities around kauri dieback and myrtle rust ranging from awareness of these tree diseases, citizen science initiatives, and mana whenua-led management activities (e.g., Black et al 2019, Sutherland et al. 2020, Hill et al. 2021).
- Landowners and community trusts also commonly use photo points for documenting changes in forest canopy condition and change following changes in management like pest animal control, restoration or retirement from grazing (e.g., in some QEII Trust covenants).
- Many forest health monitoring schemes are in place, primarily as part of industry biosecurity activities for plantation forests (e.g., FOAs, NZFFA).
- See also Peters et al. (2016) discussion of community-based monitoring, which includes aspects of forest health.

Figure 6. View from the Kauri Museum at Matakohe, Northland, in 1995. By 2000, the single remaining tree at the site stood dead. From Brockie (2020).

C2-(v). Internationally driven

Tree canopy dieback itself is not required for international agreements or obligations. However, national reporting requirements for Convention on Biological Diversity, led by DOC, include tier one monitoring nationally that can be used to report on changes in indigenous plant species dominance and extent (see template for that attribute), and which are considered internationally to be an Essential Biodiversity Variable (EBV; Pereira et al. 2013; see also Bellingham et al. 2020).

Similarly, forthcoming efforts to apply the IUCN of 'red listing' of ecosystems is in progress (mid-2024) with DOC and MfE, which will include indicators of ecosystem state and change for international reporting obligations to the IUCN.

Part D—Impact analysis

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

Few to no negative changes are likely given the lack of current management, or it's effective for few species at local scales only. However, high-profile tree species dieback of foundation species (structural and functional dominant species) can have major impacts on both environmental condition and human well-being. This is exemplified by concerns about the progression of tree diseases such as kauri dieback, whose declines could cause major negative impacts on forest ecosystem integrity as well as multiple communities. For Māori, declines of taonga species can have multiple impacts including undermining the mauri of the ngahere and multiple connections to mana whenua (Waipara et al. 2013, Black et al. 2019).

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

Few direct economic impacts are likely from tree canopy declines, and have not generally been considered for indigenous tree species. There is potential that landscape values would decline in some areas for vulnerable species, potentially having impacts on tourism or property values. Growing interest in claiming carbon credits from management of indigenous forests could also create a direct economic impact (through inability to claim C credits because of dieback sufficient to); this could be national in scale for vulnerable forest types (i.e., as suggested for forests containing kāmahi in Hackwell and Robinson 2021). Consequences of canopy dieback on ranges that are critical for water supplies to urban areas or to horticulture (e.g., drought-induced dieback of high-elevation forests in the Kaimai Range; Jane and Green 1983a,b), such as water interception, stem flow, and hydrology in soil are unknown but potentially important.

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

Several climate-change-driven effects are likely:

- Sudden decline in cabbage tree is suggested to move south as climate changes (Brockie 2020), although the underpinning evidence or forecasts for this are lacking.
- Multiple new or emerging pathogens are likely with climate change and increased transportation/trade frequency (Sturrock et al. 2011).
- Major likely or known drivers of canopy declines are likely to increase in the coming decades, including environmental weeds, climate-induced drought (Jane and Green 1983a,b; Grant 1984), increased frequency and intensity of fires, and extreme weather events (Wyse et al. 2018).

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6.8 Landscape connectivity

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Preamble: Much of the basis of our understanding of landscape connectivity comes from theoretical modelling work. The very nature of landscape connectivity precludes the traditional experimentational manipulation needed to produce robust evidence, as it is not feasible to manipulate the connectivity of whole real landscapes many times. Therefore, while there is general agreement on how landscape connectivity could affect ecological processes, there is little data and few studies that demonstrate obvious real-world effects in New Zealand. Overall, landscape structural connectivity is widely considered nationally and internationally in policy, but currently lacks agreed measurement or a standardised approach for evaluating current state or change in response to management or land use decisions.

State of knowledge of the "Landscape connectivity" attribute: Good / established but incomplete – general agreement, but limited data/studies

Part A—Attribute and method

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

Landscape connectivity is the degree to which landscape features facilitate or impede movement (Taylor et al. 1993) and can be further divided into structural connectivity that ignores species characteristics and simply measures habitat contiguity, and functional connectivity that considers species-specific responses to landscape features (Tischendorf & Fahrig 2000).

In terms of ecological integrity, habitat fragmentation, which includes a reduction in landscape connectivity, is considered one of the major drivers of biodiversity loss (Fahrig 2003). Conversely, habitat fragmentation often results in increases in ease of movement and dispersal for invasive plant and animal species and diseases (Meentemeyer et al. 2012; Brearley et al. 2013; Rodewald & Arcese 2016).

Overall, structural connectivity is best evaluated at regional to national scales because this attribute captures landscape-scale processes related to land cover and management, and is related to the distribution, abundance and function of both indigenous and non-native species.

In terms of human health, landscape epidemiology (Lambin et al. 2010) stresses an important link between landscape connectivity and human disease prevalence. For example, as landscapes become more fragmented via human modification, human and natural ecological systems have more interactions, and this leads to emerging infectious diseases (Despommier et al. 2006).

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

Within New Zealand there is very little evidence of landscape connectivity having direct impacts on ecological integrity or human health. However, as changes in landscape connectivity are largely driven by anthropogenic habitat fragmentation we can expect that the spatial extent of degradation of structural connectivity will be associated with more human-dominated landscapes. However, the exact magnitude of any degradation will vary depending on the functional connectivity of individual species, and in some instances, landscape connectivity may be improved rather than degraded.

Despite a current lack of evidence linking landscape structural connectivity to ecological integrity, there are examples or likely declines in integrity due to increased habitat fragmentation and loss of connectivity such as:

- Declines in absolute habitat availability are commonly thought to reduce population viability and cause long-term declines in indigenous biodiversity through extinction debt (e.g., Velland et al. 2006, Kuussaari et al. 2009), i.e., that increased habitat fragmentation and isolation lead to declines in multiple functions like dispersal among habitats, reproductive failure, or declines in other processes like pollination services.
- Declines in the extent and connectivity of wetlands could limit the spread and population genetic diversity for keystone species (Rayne et al. 2022).
- Understorey wood plant invasions into indigenous forests are increased largely by disturbance and close proximity to forest edges, implying decreased connectivity will be positively related to plant invasions (Jo et al. 2024).
- Increased landscape connectivity is thought to increase population viability of mobile taxa, and is suggested as crucial for scaling up restoration efforts (Norton et al. 2018).
- Reduced functional connectivity for birds between fenced ecosanctuaries and surrounding habitat (Burge et al. 2021).

A well-connected conservation network is one where ecological processes and functions connect between different sites. This includes sustaining the ability of individuals or populations of species to move between sites, providing resilience against climate change, and is considered an essential component of healthy ecosystem functioning (e.g., Tucker *et al.* 2018). Most efforts to increase connectivity have focussed on species-level conservation activities, but whether these activities are sufficient to sustain biodiversity is uncertain (Watson et all 2020).

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

Although landscape connectivity itself has not been quantified over time, several lines of evidence suggest that structural connectivity has declined. Declines of indigenous vegetation cover due to initially to fire, and subsequently from ongoing land use changes, has led to declines in forest cover and increased fragmentation of >90% for some vegetation types (e.g., see lowland forest extent attribute; Ewers et al. 2006, Dymond et al. 2017). Similarly land management for drainage has reduced both the number, extent and condition of wetlands (see wetland extent and condition). In other regions or communities (e.g., Westland forests, most alpine and subalpine vegetation), there have been far smaller changes in land cover and thus connectivity.

The largest declines in connectivity were caused over a century ago from fire, and then by major land use changes nationally with the expansion of farming and grazing operations (Greasley and Oxley 2009; Perry et al. 2014). Over the past 30 years, the rate and pace of change has slowed but is variable among regions, with some regions and vegetation-types expanding or increasing in area (and thus presumably having greater landscape connectivity) due to natural regeneration of woody species, and in some cases, active restoration of marginal vegetation (e.g., riparian plantings, wetland restoration; MacCleod and Moller 2006, but see Lee et al. 2010). On the other hand, in some areas, agricultural intensification and ongoing urbanisation have had the largest effects on fragmentation over the past decade, and this is likely to continue over the short-term (10 years; Curran-Cournane, et al. 2021).

Overall, landscape connectivity is driven largely by changes in landscape structure. Except for large-scale natural disturbance events such as major floods or earthquakes, naturalistic landscapes change relatively slowly via processes such as progressive erosion and vegetation succession. In contrast, anthropogenic disturbance has the potential to change landscape structure and connectivity very rapidly as highlighted above. Therefore, the pace of change in landscape connectivity largely depends on changes in land use, and can be both positive or negative. For example, large-scale changes in afforestation have been driven over the past decade by carbon farming and permanent forests through national-scale incentives (i.e., the NZ emissions trading scheme, the billion trees initiative). As a consequence, at least for woody vegetation, there’s likely to be increased cover and connectivity if these large-scale land use changes persist over the next 30 years.

The trajectory of change is complicated if the functional or ecological effects of connectivity are considered. Different species or functional groups of taxa respond differently to landscape structure and fragmentation, making universal generalisations about baselines and change in connectivity difficult. Rather, connectivity should be considered in terms of the structure of certain landscape features or the function for certain species, but these effects have rarely been quantified (e.g., minimum habitat requirements for population connectivity have been considered for birds, but few other taxa; MacCleod and Moller 2006). For example, both fragmentation and spillover of nutrient effects from pasture to forests have contrasting effects on different species (Didham et al. 2015).

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

No monitoring or reporting of landscape connectivity is currently done in New Zealand. This is somewhat surprising given the long-term interest in management and reporting of land cover change and its consequences for biodiversity (e.g., Lee et al. 2005; Bellingham et al. 2020). In general, landscape connectivity metrics are most commonly used for measuring the potential movement of organisms, but can also include the movement of abiotic factors such as nutrients and water.

New Zealand currently has no standard longitudinal monitoring or reporting of landscape connectivity. More generally, there is no consensus or agreed standards for quantifying and monitoring structural connectivity, but there is a body of knowledge that could be used to develop monitoring and reporting. Structural landscape connectivity is conceptually simple and there is a wide array of metrics, mostly derived from graph theory based network measures, with which to measure it (Keeley et al. 2021).

Unfortunately, as with landscape metrics more generally, these landscape connectivity metrics are imperfect, numerous, and often correlated, so choosing an appropriate measure, or measures requires further evaluation and work. Choice of what are the most appropriate measures of landscape connectivity depends upon the specific objectives, landscape, and species involved, and there is guidance available to support such decision-making (Keeley et al. 2021).

Functional landscape connectivity is most commonly approached via a cost or resistance geographic information system map that quantifies landscape features that facilitate or impede connectivity (Zeller et al. 2012). There is a wide array of computational tools available that can support functional landscape connectivity analyses (Dutta et al. 2022), many of which are well developed and could allow landscape connectivity to be easily measured at national extents and sub-hectare resolution. The most popular methods are least-cost modelling (Etherington 2016) and circuit theory (McRae et al. 2008). Functional landscape connectivity is largely unstudied, with only one example for a native bird (Richard & Armstrong 2010) and an invasive species (Etherington et al. 2014). This is perhaps not surprising given we know very little even about the distances native forest birds will disperse (Innes et al. 2022) to measure structural connectivity, let alone the landscape features that may variably affect each individual species to measure functional connectivity.

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

There are relatively few implementation issues for measuring and reporting state and change for structural landscape connectivity. Structural landscape connectivity could be measured from land cover or land use datasets, and as such would not require any land access or sampling. Functional connectivity would be harder to achieve, as this would require some form of land access to directly measure a population in some manner. These population connectivity measurements could be from one-off sampling using a landscape genetic approach, or via repeated sampling using a mark-recapture approach (Zeller et al. 2012).

State and change through time could also be assessed using past datasets such as changes in land cover (i.e., versions of LCDB) or remote sensed imagery (e.g., using repeated measured of overlaps of imagery collected over time; e.g., Parracciani et al. 2024). If habitat quality is to be taken into account, possibly as a metric for the quality of interconnected habitat as a whole, this may necessitate on-site visits to private (and public) land.

Functional connectivity requires improvements to knowledge and underpinning data, so implementation requires additional research and a stronger evidence base prior to using it for monitoring or reporting. Overall, most structural connectivity metrics use data including the size, shape, and distances between habitat patches that are readily available. In contrast, measures of functional connectivity require information about the species behaviour, population responses or ecological processes, and this information is available only for few, well-studied species.

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

For structural landscape connectivity of vegetation-types or discrete habitats, data or information are already collected for other purposes (e.g., LCDB updates, satellite imagery, LiDAR in some areas). Structural landscape connectivity can be quantified through analysis of land cover or land use datasets, and this would be a relatively low-cost monitoring approach requiring resource for collation, analyses and interpretation of information. No additional up-front costs for equipment or operations are required assuming the thematic and spatial resolution of existing data products provide the required levels of geographic information. In some cases, imagery of higher resolution may be required, incurring additional costs for purchasing from commercial sources (e.g., for some higher resolution satellite imagery) or data collection (e.g., expanded LiDAR collection). Additional, ground-truthing or ground-based data collection may be required to validate remotely sensed information that underpins connectivity metrics, or to assess habitat quality.

Functional connectivity is much harder and expensive to measure. One of the more promising methods for rapid large-scale studies would be landscape genetics/genomics approaches to measure landscape connectivity via gene flow within a population – though the suitability of this approach will vary by species and landscapes (Keeley et al. 2021; Rayne et al. 2022).

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

We are not aware of any landscape connectivity monitoring being undertaken by iwi/hapū/rūnanga.

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

Landscape connectivity influences the distribution of all organisms, and in its broadest sense also influences the flow of physical material and energy through ecosystems. In this regard landscape connectivity is connected to and will influence all other attributes as it underpins where everything is or could be. Attributes like lowland forest extent, and wetland extent will be positively associated with landscape structural connectivity because declines in connectivity are often driven by loss of total area and fragmentation of vegetation. Connectivity could also be positively related to canopy tree dieback extent but for different reasons; increased connectivity could exacerbate some of the drivers of tree canopy declines such as providing contiguous habitat or facilitating movement of pest species or some pathogens.

Part B—Current state and allocation options

B1. What is the current state of the attribute?

The state of landscape connectivity has not been monitored or reported at the national scale (see A4i). However, most metrics of connectivity use spatial change in habitat or vegetation to estimate state and change in connectivity. As a consequence, given ongoing data collection to understand changes in land cover including reductions in extent and size of land cover classes or vegetation-types, and the national declines in many vegetation-types like forests and wetlands (see A3), the current state of this attribute is undoubtably lower than historical baselines. Landscape connectivity could be evaluated using current national spatial databases such as LCDB to determine its suitability as an indicator (see A4ii).

Some effort to understand fragmentation and potential risks of declining habitat availability or connectivity has been done at regional scales. For indigenous forests, fragmentation effects vary widely among regions, with smallest average habitat fragment area in Northland and Auckland (of ca. 20ha), and greatest edge density effects in Northland, Auckland, Taranaki, Tasman, Nelson and West Coast (Ewers et al. 2006). Loss of connectivity is also assumed to be an issue for relatively small remnant wetlands, lowland indigenous forests, shrublands, some grasslands and riparian ecosystems that occur in anthropogenically-modified landscapes; these are areas that have sustained greater habitat loss already (Walker et al. 2006; Weeks et al. 2013; DOC 2015).

Combinations of data including abiotic land environments (LENZ), land cover (LCDB) and protected status of land cover have been used to determine rates of change in areal extent of environments as a proxy of biodiversity and the role of protection (e.g., Brokerhoff et al. 2008, Walker et al. 2008, Cieraad et al. 2015). These analyses have not considered state or change of spatial connectivity of protected areas. Global assessments of protected area connectivity (e.g., ProcConn metric) have been developed and show, at a national scale, that connectivity is >17% (e.g., Saura et al. 2018; WWF2020); this metric was developed based on the dispersal of mobile/migratory taxa over >10km, but could be downscaled to understand connectivity at finer spatial scales.

Overall, there are both data and potential metrics that could be applied to understand the current, and recent historical, change in connectivity, but this has not been done to date.

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

There are no fine-scale maps representing natural reference states from the distant past that we are aware of. Broad-scale pre-human land cover maps do exist (e.g., Weeks et al. 2013, Fig. 2a), to which current levels of connectivity could be compared, however any such comparison would be limited by the accuracy of these maps. Historic information that could be evaluated for spatial connectivity such as aerial imagery have been captured decades after major land use changes and declines or fragmentation of many habitats or vegetation-types from fire and land clearance, and many areas are actually reverting to woody cover as marginal farming lands are retired. As a result, natural reference states or baselines are unavailable although changes at the decadal scale, and comparisons between protected or managed areas and other management regimes could be assessed to inform the effects of management interventions on connectivity.

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

None that we are aware of. International efforts to quantify the connectivity of only protected areas have been developed (e.g., Saura et al. 2018, WWF 2020), and considered for reporting to meet international obligations and goals such as the Kunming Montreal Target 3 in the Global Biodiversity Framework). Most goals or levels establish a total area of protected land rather than setting bands or targets for connectivity itself; rather, increased connectivity is usually assumed to occur from increasing total area under protection.

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

In theory yes, but these have not been described or reported for New Zealand. Early modelling work demonstrated that as habitat fragmentation increases, we can expect there to be a tipping point in structural connectivity where a landscape can suddenly shift from being fully connected to fully disconnected (Gardner et al. 1987). While these theories are generally well accepted, we are unaware of any empirical studies that have demonstrated such tipping points in landscape connectivity in real landscapes.

Given the international focus on protected areas for highly mobile or migratory species such as birds, there is an opportunity to quantify thresholds or tipping points for our indigenous bird species that are highly variable in both habitat requirements and dispersal ability by considering how reintroductions of species succeed or fail in different landscapes (e.g., Miskelly et al. 2013; Innes et al. 2022).

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

There is certainly the potential for ecosystem effects to lag behind changes in landscape connectivity. For example, a reduction in landscape connectivity may result in a species population becoming fragmented into isolated non-viable sub-populations, but the actual extinction of the sub-populations may take some time to become evident, especially for long-lived species. These processes are part of 'extinction debt', where past habitat loss and fragmentation cause longer-term declines in biodiversity over decades (e.g., Velland et al. 2006).

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

Both tikanga Māori (e.g., for biocultural monitoring) and mātauranga Māori (e.g., for understanding changes in ecosystem condition) might be used for evaluating the condition of the environment and people and interdependencies (Lyver et al. 2019), but we are not aware of any bands or allocation options with respect to landscape connectivity.

Although we cannot comment directly on mātauranga Māori, we suggest that for structural or functional connectivity, there could be condition or states described from a te ao Māori perspective, but we are unaware of any examples. Given the crucial importance of interconnectedness of people and environment, connectivity is a likely (but undeveloped) indicator (e.g., Lyver et al. 2021). Place-based goals or acceptable changes in structural or functional connectivity will require community-specific approaches that cannot be directly applied to other sites.

Part C—Management levers and context

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

The current state of landscape connectivity in New Zealand is unknown, which precludes relationships with stresses from being quantified. However, the relationship between landscape connectivity and environmental stresses could be evaluated from currently available data sources (see B1).

The well documented declines in habitat or indigenous vegetation from past disturbance and land use change and intensification (e.g., Ewers et al. 2006; Walker et al. 2008; Weeks et al. 2013) has likely driven losses of connectivity through reduction in the extent, number and size of habitats. For example, we have a good general understanding of how vegetation clearance leads to loss of structural connectivity. However, there is a great deal of complexity in how structural connectivity (or the physical layout of habitat) leads to functional connectivity or fragmentation. Different species and processes operate at different scales, and this depends on their dispersal ability, range size requirements, and tolerance to disturbance. Additionally the permeability of the matrix affects functional connectivity (e.g., what the habitat patches are surrounded by).

Structural connectivity is relatively simple but functional connectivity is more complex and highly dependent on local focus (e.g., which particular species or processes exist in a locality and which local people are concerned about protecting). Overall, landscape connectivity strongly reflects land use and management over decades, and these relationships could be quantified with currently available information. Functional connectivity is far more complex, but likely responds to a greater number of environmental stresses associated with habitat fragmentation, but will require additional research to understand how population or ecological functions are altered by connectivity. Bird movement including from restoration or translocation is relatively well studied, and knowledge of bird population movement and connectivity could be used to better understand the effects of fragmentation and other stresses at the landscape scale (Innes et al. 2021, 2022; Allen et al. 2023).

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

C2-(i). Local government driven

Several local government policies refer to the importance of habitat fragmentation or connectivity of habitats, but connectivity is usually poorly defined and not quantitatively assessed. For example, connectivity is considered an important characteristic of environmental condition by local government NZ (e.g., the Willis 2017 report ‘‘Biodiversity and the role of Regional Councils’). However, landscape structural connectivity is not currently included in National Environmental Monitoring Standards (NEMS) or as part of legal biodiversity protection (e.g., in the EMaR biodiversity M18; see Bellingham et al. 2016). Despite this uncertainty of how to measure and monitor connectivity, at least two councils are monitoring or measuring connectivity: Wellington City Council (2015), and Auckland Council (2021).

C2-(ii). Central government driven

Landscape structural or functional connectivity *per se* is not reported in national state of environmental reporting by MfE and Stats NZ. Land fragmentation is reported but focusses on the effects on urbanisation on high quality soils or productive lands rather than biodiversity or indigenous ecosystems.

Connectivity of natural areas is considered in the National Policy Statement – Indigenous Biodiversity, which recognises connectivity as important for maintaining and indigenous biodiversity and promoting resilience to climate change and requiring prioritisation (amongst other considerations) in restoration. Connectivity is also referenced in the NZ Coastal Policy Statement (2010, Policy 11) to avoid significant adverse effects on ecological corridors. Similarly, recent strategies by DOC for public conservation land (and public-private partnerships) in Te Mana o te Taiao for freshwater management (see also Lee et al. 2005 and Bellingham et al. 2020) highlight a goal of restoring river corridors at landscape scales; here the goal is to increase connectivity of riparian and floodplain systems usually administered by LINZ.

Overall, despite fragmentation or connectivity being included in multiple national strategies across central government departments, current state and change of connectivity is not yet part of biodiversity or environmental monitoring and reporting.

C2-(iii). Iwi/hapū driven

We are not aware of any interventions to improve connectivity specifically led by Iwi/hapū; but this should be investigated further given there is general awareness of the importance of connectivity in many restoration initiatives and in Iwi Environmental Management Plans (e.g., Te Kotahitanga o Te Atiawa) and part of collective restoration efforts (e.g., the Kotahitanga mō te Taiao Alliance).

C2-(iv). NGO, community driven

Although spatial connectivity is usually not explicitly stated as a management goal for community-based restoration or management, improving the number and proximity of habitats containing indigenous biodiversity is. For example, sheep and beef farms as an industry have considered the role and benefits of indigenous vegetation on farms at the national scale, and restoration or retiring of grazing could contribute to increased connectivity of habitats (see Pannell et al. 2021).

In general, there are only a few examples of both structural and functional landscape connectivity interventions (Etherington 2015). Within New Zealand there are efforts such as the Te Ara Kākāriki Greenway Canterbury Trust whose aim is to promote structural connectivity via native plantings primarily for forest birds (https://kakariki.org.nz/) but the landscape-scale effectiveness of habitat restorations remains unclear. Another New Zealand example would be the use of predator-proof fences to reduce functional connectivity for invasive predators, and the positive effect of such interventions on native birds is beyond doubt (Innes et al. 2012).

Overall, there are practical efforts to restore habitat for species taking the landscape and spatial availability of habitat into consideration, but these efforts are largely driven by landscape-scale conservation management efforts rather than explicit regional or national strategies or international obligations.

C2-(v). Internationally driven

There are numerous international efforts, agreements and strategies that consider habitat fragmentation and spatial connectivity of habitats. We mention only a few of these here. Overall there is general agreement and commitment to improving the connectivity of habitats internationally, as demonstrated in:

- The “2030 Nature Compact” agreed by the 2021 G7 Leaders Summit advocates for “improved quality, effectiveness and connectivity of protected areas”.
- The United Nations General Assembly in 2021 adopted Resolution 75/271, which encouraged member States to “maintain and enhance the connectivity of habitats…”
- The International Union for the Conservation of Nature (IUCN) Policy Resolution 073 on “Ecological connectivity conservation in the post-2020 global biodiversity framework: from local to international levels” emphasizes the importance of ecological networks and corridors to sustaining biodiversity and nature’s contributions to people, and recommends that all IUCN Members work to conserve connectivity by documenting it across ecosystems, informing policies, laws, and plans, identifying key drivers and building synergies across institutions and borders to implement solutions.
- The UK is developing an indicator (D1: Quantity, quality and connectivity of habitats) as part of their 25 Year Environment Plan, but there are not yet any proposed targets associated with habitat connectivity (DEFRA 2023).
- The EU’s recently released Biodiversity 2030 Strategy aims to protect 30% of EU land and sea area and includes references to ecological corridors and a ‘coherent network’. Although connectivity is not explicitly monitored in this strategy, effective mesh density for landscape fragmentation is (i.e., the number of landscape elements/km^2); see EU biodiversity strategy for 2030).
- International efforts to understand the extent and connectivity of protected areas has developed methods and a global assessment (e.g., the Protected Planet Reports), for example, using Protected Connected (ProtConn) and PARC-Connectedness metrics for evaluating progress against Aichi Target 11. Based on the ProtConn method, 7.84% of global terrestrial ecosystems are considered both protected and connected, but this is far below the 17% required by Aichi Target 11 (Saura *et al.* 2019).
- The IUCN WCPA Connectivity Conservation Specialist Group (CCSG) published the IUCN Guidelines for Conserving Connectivity through Ecological Networks and Corridors (Hilty *et al.*, 2020). These Guidelines are an important step towards a coherent global approach for connectivity conservation, providing clarity on the role of ecological corridors.

What these examples demonstrate is a growing international interest in landscape-scale increases in protected or managed areas that are more structurally and functionally linked. This goal is strongly reflected in major biodiversity strategies and obligations.

Part D—Impact analysis

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

Many ecological processes required to maintain ecosystem health, such as seasonal food migrations or seed dispersal, are reliant on sufficient landscape connectivity. Therefore, a loss of sufficient levels of landscape connectivity could result in a collapse of ecosystem health. But landscape connectivity can have both positive and negative impacts. For example, a heavily forested landscape would be highly connected for native forest birds, and if combined with human access could provide recreational opportunities to support physical and mental health. However, this highly forested and connected landscape could exacerbate the potential for massive wildfires, or could provide vectors for wildlife diseases that could affect livestock and humans, or some plant pathogens (e.g., kauri dieback).

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

There are no obvious or documented economic impacts of changes in landscape connectivity.

We would envisage that economic impacts would be most likely be felt by people living in rural areas and working in production industries as these are more exposed to more natural areas that will be affected by changes in landscape connectivity. Remnant habitat patches often occur on privately held land and the subset of owners with habitat patches on their land (potentially as a result of good stewardship) will be impacted by policy changes. Additionally, Māori land contains proportionately more indigenous vegetation cover, and may also have more remnant habitat patches.

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

Climate change will be most likely to influence landscape connectivity via gradual changes in habitat distributions in natural areas, changes to the ability for species to passively improve connectivity through natural regeneration, or by potentially sudden changes in human land use and hence land cover in human dominated landscapes. Identification and management of areas critical to landscape connectivity would help to mitigate any effects of climate change on landscape connectivity. Planning for 'climate smart landscapes' (Lavorel et al. 2022) can include consideration of connectivity and its potential role in maintaining indigenous diversity and ecosystem processes in the face of multiple climate change impacts like increasing disturbance, drought and climatic variability.

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7 Soil Domain

Eleven attribute information stocktakes for the Soil Domain are provided in sections 7.1 to 7.11, below. Dr Nina Koele, Dr Fiona Curran-Cournane, and Bruce Croucher (MfE Domain Experts), Dr Jo Cavanagh (MWLR, Domain Leader), and the Māori environmental researcher panel reviewed these sections.

7.1 Peatland/peat soils subsidence control

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Preamble: This attribute is in the soil domain and focuses on peat subsidence control in drained peatlands. These peats, which are mapped as Organic Soils in the New Zealand Soils Classification, were formed in wetland ecosystems where waterlogged conditions prevented the complete decomposition of organic plant material. Consequently, the deposition of organic matter exceeded decomposition and over time the partially decayed organic matter accumulated to form peat that can be many metres deep, making intact peatlands significant carbon stores[1]. The drainage of peatlands for agricultural use lowers the water table and results in land subsidence and decomposition of previously protected organic material. The decomposing organic matter produces CO_2 emissions and, over time, a more consolidated and mineralised peat soil forms.

In Aotearoa-NZ, drained peatlands account for up to 8% of net GHG emissions[2]. This section focuses on peat soil subsidence control for drained peat soils and does not focus on GHG emissions from drained peat soils. The reader should also see wetland condition and extent components in the terrestrial and indigenous biodiversity domain for information that encompasses natural intact peat wetlands.

State of knowledge of the "Peatland/peat soils subsidence control" attribute: Good / established but incomplete – general agreement, but limited data/studies

Part A—Attribute and method

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

Intact peat wetlands represent unique hydrological and ecological environments that support threatened endemic flora and fauna, are a carbon sink, and can represent a taonga for Māori[3]. Drainage of peatlands for agriculture diminishes these ecological values and results in ongoing land subsidence through shrinkage and consolidation (and CO_2 emissions through biochemical oxidation). Long term ongoing consolidation and oxidation ultimately leads to the complete loss of the peat soil, representing a loss of soil natural capital and its associated specific ecosystem services (e.g., denitrification potential can be high in peat soils). Globally, intact and undrained peat wetlands have been identified as having an important role to play in maintaining biodiversity and climate change mitigation[4].

Subsidence of drained peatlands (and CO_2 emissions[5]) is strongly correlated to water table depth, shallower water tables and wetter conditions reduce subsidence[6, 7]. The consequences of ongoing subsidence and eventual loss of peatland soils include increasingly severe impacts on adjoining/adjacent wetland ecosystems as the surrounding land subsides, and increased risk and frequency of flooding and inundation of drained land that reduces productivity (an example of this is the lower portion of the Muggeridge's catchment in the Hauraki[8], Waikato region). This can result in a requirement to upgrade, repair or install drainage and other infrastructure (e.g., flood protection, pumping, roads and utilities) or ultimately the need to abandon current land-use. The impact of subsidence is likely to be exacerbated by sea level rise where drained peatlands are close to the Coast (e.g., low-lying areas in the Hauraki Plains).

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

Drainage of peatlands for agriculture and the resulting subsidence is well documented globally[6, 9] and locally[10-12]. Subsidence impacts adjacent intact wetlands and peat lakes, for example the small 114 ha Moanatuatua Scientific Reserve, in the Waikato, which is an intact peat wetland remnant is severely impacted by surrounding drainage[13], resulting in drier conditions on the edges, which encourages colonisation by weed species. Moanatuatua Scientific Reserve is one of the few remaining sites where a natural population of the relict endemic plant species *Sporodanthus ferrugineus* exists. If the long-term viability of such ecosystems is under threat from the surrounding land-use, then this conflicts with the National Policy Statement for Freshwater Management[14], which requires the significant values of wetlands be protected. In the Waikato Region, there is strong evidence that drained peatlands are subsiding at a mean rate of 20 mm/yr but with high spatial variability[10]. In other regions, local and regional authorities have some information on subsidence rates (e.g., Whangarei District Council at Hikurangi).

In Manaaki Whenua's most recent analysis of peat soil area provided to MPI and MfE[15], peatlands were calculated to cover about 220,500 ha of NZ, and only about 73,200 ha was in non-drained land use (for example vegetated wetland or forest), indicating that about 147,300 ha (67% of NZs Organic Soils) are drained and therefore subsiding. These total areas are lower than those used in NZs current GHG inventory which uses area calculated from the now outdated Fundamental Soils Layer (FSL). S-map was used where available for our most recent estimates[15] and area estimates from FSL (where S-map was not available) were improved by extracting information on proportional contributions for mixed mineral/peat soil areas that was not used in previous estimates. The difference in the area estimates between FSL and S-map occur because of uncertainty in both historic and contemporary soil maps and loss of peat soils from ongoing decomposition[2].

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

Based on Waikato knowledge, peatland drainage for agriculture (and therefore subsidence) began in the early 1900s and accelerated in the mid-1900s as mechanized approaches to drainage were more available[3], and this is likely similar across NZ. In the future 10-30 years we do not expect further expansion of the drained area because of policy designed to prevent this (National Policy Statement for Freshwater Management). However, on already drained peatlands, subsidence will continue until the peat no longer remains. The rate of subsidence and time to loss of peat will be dictated by land and drainage management, and peat depth. Potentially, there could be some rewetting of drained peatlands to mitigate subsidence and GHG emissions, but the likely trajectory of potential rewetting activity is unclear.

There is strong evidence that rewetting will slow or stop subsidence (based on international literature) within a generation[3]. Subsidence rates are relatively high (~20 mm/yr[10]) but peat growth and accumulation is much slower (~1 mm/yr[16]) so recovery of already lost peat will take 100s to 1000s of years. In contrast to the shorter time required to slow or stop subsidence through rewetting, timeframes to slow or stop GHG emissions from rewetting of peatlands is highly uncertain. This is largely because reductions in CO_2 emissions following rewetting may be offset by increases in CH_4 and N_2O emissions from nutrient rich rewetted peat soils, especially if surface flooding occurs [17].

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

There is currently no standard or monitoring program related to subsidence of drained peat soils at the national scale. However, the Waikato Regional Council (WRC) manage a continuous monitoring network that measures subsidence and water table levels at 11 point locations across their region. This approach could be expanded nationally but requires ongoing site maintenance so is potentially best led through regional authorities. WRC also fly LiDAR transects with a helicopter every 5 years to gain greater spatial understanding of subsidence (the first baseline flight was done in April 2021). A design summary of both these monitoring networks can be found in [18], and [19] reviewed a range of potential monitoring techniques for WRC.

Historically, WRC has measured subsidence using depth probing, and as such, current regional scale estimates of peat subsidence are based on that monitoring. Outputs from this monitoring have previously been reported as technical reports (e.g., [20]), a paper[10], and as a state of the environment indicator on the WRC website (Peat subsidence | Waikato Regional Council). Depth probing for subsidence monitoring in the Waikato region has now been replaced with the continuous monitoring network and LiDAR transects referred to in the previous paragraph.

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

Yes, access to private land is important for setting up continuous monitoring sites and ground truthing airborne LiDAR based approaches.

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

Continuous peat surface level/subsidence and water level logging at single point location:

Based on WRCs experience, materials to set up each single continuous monitoring site cost about $5000 in June 2022. This included two levelloggers, a barologger, dipwells, mesh, rods, and associated mounting hardware and site fencing. Labour, including site selection and set up, was about $6500 per new site and required access to high precision GPS based survey gear. WRC estimate about another $5500 per year to maintain the sites including downloading, checking/plotting and storing the data. However, there will be some synergistic gains when this is done over multiple sites meaning, ongoing per site cost could be lower.

Helicopter based LiDAR:

In 2021, [18] calculated that for a regionally representative peat subsidence monitoring network, WRC should budget $100,000 for 5-yearly helicopter based LiDAR measurements with a coverage of about 10,000 ha of drained peatland. About 20% of the budget was for project management (job safety assessment, flight planning, liaising with landowners), 25% for helicopter time, 10% for ground truthing, 30% for data processing and DEM differencing, and 20% for reporting and communication of results.

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

I am not aware of specific peatland subsidence monitoring being carried out by representatives of iwi/hapū/rūnanga. However, there is iwi-led monitoring being done of quality/condition for intact peat wetlands (see wetland condition and extent components in the terrestrial and indigenous biodiversity domain for information that encompasses natural intact peat wetlands).

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

There are relationships between peat subsidence and wetland extent and wetland condition in the terrestrial indigenous biodiversity domain. Drained peatlands were previously vegetated peat wetlands, so their drainage is linked to the large loss of wetlands in NZ. Continued subsidence of drained peat soils will negatively impact the sustainability of adjoining wetland ecosystems. For example, drainage of peatland adjacent to Kopuatai Peat Dome (a globally unique Ramsar site and the largest unaltered restiad peat bog in New Zealand – a bog type most extensively found in NZ, formed from jointed rush-like herbs from the Restionaceae family[21]) and Moanatuatua Scientific Reserve results in drier conditions on the edges, which encourages colonisation by weed species[13].

There are also potential relationships to surface and shallow groundwater quality. Highly organic (usually more recently drained and developed) peat soils can have low anion storage capacity (ASC<60%), so added P is highly mobile relative to most mineral soils[22]. In the absence of careful nutrient budgeting and application, P added to low ASC peat soils can infiltrate to shallow groundwater or surface water. Typically, as the peat soil becomes more mineralised and consolidated, anion storage capacity increases (ASC>60%) and risk of P loss decreases[23]. Rewetting of drained peatlands that have a legacy of heavy fertilisation application and animal excreta returns could also mobilise these nutrients and result in eutrophication of receiving waterways[17, 24].

Part B—Current state and allocation options

B1. What is the current state of the attribute?

The extent of peatland drainage is well understood at national scale and can be quantified by intersecting Organic Soils area with either the LUCAS LUM (land-use map) or land cover database (LCDB). Although, some regions have more up-to-date and accurate area mapping (e.g., S-map) than others (e.g., FSL). As outlined in section A2, about 147,300 ha of NZs total Organic Soils area (220,500 ha) are under a LUCAS LUM land-use that indicates drainage, representing about 67% of the area[15]. A more complete national drains spatial layer would help to better quantify drainage intensity and options for rewetting.

In contrast, the subsidence rates of drained peat soils are not well quantified at a national scale, but we do know that with drainage these soils will be subsiding, and this will continue until the peat no longer remains if drainage is maintained. The only region with information that could feed into a national monitoring programme is likely the Waikato Region (monitoring summarised in A4). The Waikato region has about 40% of NZs drained peatland area, and historic and contemporary subsidence information is available (e.g.[10], WRC monitoring (Peat subsidence | Waikato Regional Council)). This monitoring shows that contemporary subsidence rates average about 20 mm/yr with high spatial variation in subsidence rates. Other regions have some ad hoc data on subsidence rates from intermittent consultancy work (e.g., Whangarei District Council for Hikurangi swamp).

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

Yes there are remaining intact peatland areas, and work done to characterise these includes detailed research[13, 25-27] and condition monitoring (e.g., NZ National Wetland Database[28]). Manaaki Whenua manages the NZ National Wetland Database (about 249 wetlands around NZ), which holds data on soil bulk density, carbon and nutrient content in the top 7.5 cm for remaining intact peatlands together with wetland condition scores. This data could be used to inform the relative condition of drained peatlands at selected monitoring sites compared to the natural state and potentially inform progress toward recovery where peatlands were rewet/restored.

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

We are not aware of numeric bands set for monitoring peatland subsidence in NZ or elsewhere in the world. Waikato based research[10] and regional council monitoring shows drained Waikato peatland are subsiding at an average rate of 20 mm/yr with high spatial variation. While any subsidence is an indicator of a poor state, potentially a band could be calculated based on average rates and values above the determined average range could be indicative of a poorer state. Any potential band developed would need to account for the relationship between subsidence rates and time since drainage from international literature[10]. In general, subsidence rates are expected to reduce over time in the absence of drain deepening or infrastructure upgrades (e.g., drainage pump installation/upgrade). General guidance for minimising peat soil subsidence under agricultural management has been provided by WRC[29], but this guidance needs updating.

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

Drainage of peat soils results in ongoing consolidation and decomposition of the peat soil and ultimately the complete loss of peat and reclassification of the soil to a mineral soil (e.g., Gley Soil) when the surface organic layer is reduced to less than 30 cm cumulative depth. This is a loss of soil natural capital and will result in a change in soil flora/fauna and associated specific ecosystem services (e.g., denitrification potential which is high in peat soils). The time taken for this to occur will depend on depth of drainage, management practices, and peat depth. In some circumstances there maybe a risk that the underlying sediments contain iron sulphides (Acid Sulphate Soils) that when exposed to oxygen can form sulphuric acid resulting in very low pH (<4) that can have deleterious effects on flora and fauna[30].

There is also likely a bulk density threshold or tipping point where long-term consolidation and decomposition would make it very difficult to reestablish the dominant peat forming vegetation (*Empodisma robustum*) required for successful restoration. Further research is required to better understand how much degradation/consolidation will make restoration particularly challenging (further detail in B5 below).

In relation to nutrient loss from peat soils there is some evidence that an anion storage capacity of about 60% represents a tipping point where P loss reduces[22]. Highly organic peat soils have low anion storage capacity (ASC<60%) so P loss from peat soils to ground water (by infiltration) can be higher than from mineral soils. As the peat soil becomes more mineralised and consolidated anion storage capacity increases (ASC>60%) and risk of P loss decreases.

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

Subsidence rates of drained peat soils is typically high following drainage and slows over time (see Figure 2 in [10]). Long-term subsidence is likely to continue at rates of about 20 mm/yr [10]). In contrast, peat growth and accumulation is much slower (~1 mm/yr [16]). Therefore, recovery of already lost peat will take 100s to 1000s of years.

For a drained peatland, the long-term trend is surface subsidence but superimposed on this long-term subsidence is a seasonal cycle of surface swelling and shrinking, known as peat surface oscillation (PSO) [31]. PSO occurs in both drained [32] and natural peatlands [31] and is driven largely by soil moisture and water table dynamics. During extended dry periods, the surface shrinks and during extended wet periods the surface swells resulting in a seasonal cycle of increase and decrease in surface height. Understanding this cycle is important for any peat subsidence monitoring programme.

During the preparation of drained peatlands for agriculture they were typically heavily cultivated and large quantities of soil nutrients were added to ensure productivity[11]. Ongoing cultivation is often required to work lime into the soil profile, reducing acidity and allowing adequate plant rooting depth. Through time, the soil bulk density increases along with nutrient content. Bulk density is an indicator of peat development. In their natural state, bulk density is often <0.05 t/m^3, while in drained and consolidated peat soils, it can range considerably but often sits in the range of 0.2-0.5 t/m^3[11, 29]. Highly decomposed consolidated peat soils typically have higher nutrient content as a result of long-term agricultural nutrient input. This higher nutrient content increases the risk for nitrous oxide and methane emissions, and also mobilisation and loss of these nutrients to receiving water bodies, if rewetting occurs [17].

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

I am not aware of examples of tikanga Māori and mātauranga Māori informed bands or allocation options, this is an area that requires further exploration which must be done in consultation with Māori. However, there is iwi led monitoring being done of quality/condition for intact wetland/peatland (also see wetland condition and extent components in the terrestrial and indigenous biodiversity domain for information that encompasses natural intact peat wetlands).

Part C—Management levers and context

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

Historically the ecosystem service value of intact peat wetlands was not widely appreciated or understood and pressure for increased agricultural land resulted in the drainage of these areas. Much of this drained peatland area is now high value agricultural land, largely used for dairy production, and raising of water tables would threaten some areas of this high-return land-use.

Subsidence of drained peatlands (and CO_2 emissions) is strongly correlated to water table depth, based on strong consensus in international data[5-7]. However, there is little data on the relationship between water table depths and subsidence rates for NZs drained peatlands, although, in the Waikato region a dataset of subsidence rates relative to water table depth is growing from monitoring on drained peatlands by WRC.

Water table depth is controlled through drainage infrastructure (drains/pumps/weirs) when water is in excess (i.e., winter months), which is managed by private landowners and by regional and district councils. However, many smaller farm surface drains often run dry during summer or autumn periods, indicating drains are no longer controlling the water table depth. During these dryer periods, available energy and surface evaporation rates have increasing control over water table depths in drained agricultural peatlands. This is in contrast to intact natural restiad peatlands where the vegetation strongly limits evaporative water loss during dry periods[33].

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

C2-(i). Local government driven

Waikato Regional Council are likely the most proactive council with respect to monitoring of peat soil subsidence and exploring options to reduce subsidence (the Waikato region has about 40% of NZs peat soils). The Waikato Regional Policy Statement (Policy LF-P10) requires WRC to manage adverse effects of activities from use of peat soils. Under this, Method LF-M38 directs WRC regional plans to slow the rate of subsidence and carbon loss, mitigate adverse effects from the use and development of peat soils, and ensure drainage infrastructure minimises any adverse effects on peat soils and peat lakes. Methods LF-M39 and LF-M40 promotes research and advocacy into better peatland management.

However, there is currently limited robust local information to implement Policy LF-P10. Recognising the need for robust information, the soil and land science team at WRC have applied for and received funding through sequential Long Term Plan processes. This work began in 2018 and resulted in development of a regional subsidence monitoring programme with 5-yearly monitoring[18]. They have also funded reviews to examine management practices to slow subsidence[3] and reviews are in progress to identify risks and alternative land-use opportunities. Going forward, they plan to design field-based mitigation testing, develop decision support tools for peatland managers, and extend and update good practice guidelines[29].

In 1999 and again in 2006, WRC published good practice guidance for peatland management[29]. However, uptake of this information by land and drainage managers on peat is unknown, and the published guidance is overdue for an update.

Note that WRC has responsibility to meet particular levels of service to maintain and manage drainage and this is, to some extent, conflicting with policy to mitigate adverse effects of activities from use of peat soils.

C2-(ii). Central government driven

I am not aware of central government driven work directly focused on interventions to better understanding and mitigate subsidence of drained peat soils. However, central government has funded work to better understand and quantify GHG emissions from drained peat soils through the GHG Inventory Fund managed by MPI and approaches to reduce emissions through a joint collaboration fund set up between MPI and their equivalent in Ireland (DFAM). These projects will indirectly help understand and mitigate peat soil subsidence by exploring options and effectiveness of raising water tables and alternative potential wet land-use options.

Historically there was an attempt to develop strategic direction and policy for the management of peatlands in NZ in 1982[34]. North and South Island working groups were established to identify conflicts and requirements for different land-uses on peat soils. Water table control and drainage aspects were deemed to be priorities. The report included recommendations for management, conservation, and further research. Some of the recommendations have been implemented over time but many are still relevant and unresolved, for example, 'We recommend the true cost of agricultural development be determined. We further recommend that peat shrinkage rates be investigated in some detail, by comparing the effects of various agricultural, horticultural and pastoral regimes'[34].

C2-(iii). Iwi/hapū driven

Iwi are involved in projects to restore peatland environments. For example Iwi driven mitigation work is occurring in the lower Waikato and this group in collaboration with others have reviewed options for alternative wet land uses [35]. In the Hauraki, iwi is involved in restoration projects adjacent to Kopuatai and may also be initiating work around Torehape. We are not able to provide detail on these projects but representatives of Ngāti Hako may be able to provide more information.

C2-(iv). NGO, community driven

I am aware of a large farming operation in the Waikato region that manages land adjacent to Moanatuatua peat reserve who is exploring opportunities to retire and restore a buffer strip beside the reserve to reduce the impact of farming activities and drainage on the reserve. There are also community driven projects involving local farmers focused on protecting Waikato peat lakes and wetlands that have involved planting buffer zones around the lakes. I am not aware of any monitoring activity to assess the effectiveness of these activities.

C2-(v). Internationally driven

I am not aware of any international obligations directly related to peat soil subsidence mitigation. However, commitments under the Paris Agreement (2016) require NZ to meet 2030 and 2050 GHG targets. Drained peat soils likely contribute about 8% on NZs net GHG emissions from less than 1% of its land area[2], so rewetting to help meet these targets could occur and would also mitigate peat soil subsidence.

Part D—Impact analysis

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

Continued subsidence of drained agricultural peatland threatens adjacent remaining peat wetlands through lowering of the regional water table during dry periods and increased risk of flooding and inundation during wet periods due to a lowering of the land surface, especially toward the edge of intact wetlands. Both increased drying and inundation threaten unique and endemic flora and fauna found in peat wetlands. Additionally, remaining peat in both drained and intact peatlands represents a large irrecoverable[36] (in our lifetimes) carbon store, which drainage destabilises, contributing to NZs total GHG emissions and climate change. Intact peat wetlands also represent a taonga for Māori, however, Māori researchers are better placed to elaborate on this aspect. Ongoing subsidence of drained peatlands also increases risk and frequency of flooding and inundation of managed agricultural land, reducing productivity. Ultimately the peat soil can be completely lost and reclassified as mineral soil (e.g., Gley Soil). The time taken for this to occur will depend on depth of drainage, management practices, and peat depth. The impact of this loss will vary spatially and in some circumstances there may be a risk that the underlying sediments contain iron sulphides (Acid Sulphate Soils) that, when exposed to oxygen, can form sulphuric acid, resulting in very low pH (<4) that can have deleterious effects on flora and fauna[30].

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

Continued subsidence will result in increased risk and frequency of flooding and inundation leading prolonged high soil moisture and ponding. Excessively wet drained peatlands result in loss of agricultural production for farmers and lead to an ongoing cyclic requirement to upgrade, repair or install drainage and other infrastructure (e.g., flood protection, pumping, roads and utilities), which is costly for both responsible councils and their rate payers. Urban development has also occurred on peatlands in the Waikato and the building and engineering challenges associated with building on drained peat soils cannot be understated [37]. There are many personal accounts of rural buildings in the Waikato region where continued subsidence resulted in the need to build up the land surface around houses or jack up buildings and cut piles down to lower buildings including the need to modify attached infrastructure. In some cases, farm milking sheds have been abandoned because the concrete pads they were built on were poured over piles and the shed floor is now so far above the land surface they are no longer useable (examples can be viewed off Valentine Road, Gordonton).

We are aware of areas (e.g., parts of Hikurangi and parts of Hauraki) where prolonged high soil moisture is making pastoral farming uneconomic, and farmers need an equitable exit strategy. Therefore, ongoing subsidence poses environmental, economic, and social challenges for future land management. However, there is currently no national level strategy for the management of peatlands along with little robust New Zealand-based information or examples of how to slow or stop subsidence. Under contract to Waikato Regional Council, [3] reviewed international literature to identify potential approaches to slow or stop subsidence. Lifting water tables and reducing cultivation (associated with cropping and pasture renewal) were likely the most certain ways to reduce subsidence, but this work also identified the need for better decision support tools for land managers and policy incentives to drive change [3].

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

Increased frequency of extreme weather events (both floods and drought) will exacerbate existing challenges for the management of drained peatlands. For example, subsidence rates are higher during prolonged dry periods (drought years) and more frequent and extreme rainfall events will exacerbate the frequency of inundation and flooding risk. Increased subsidence and flooding risk will require increased frequency of infrastructure upgrades (e.g., flood protection, pumping, roads and utilities), and greater levels of intervention, which is costly for responsible councils and rate payers. The impact of subsidence is likely to be exacerbated by sea level rise in low coastal catchments, for example the lower Waihou Piako catchment on the Hauraki Plains in the Waikato Region.

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7.2 Soil Bacteria composition

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Preamble: The most numerous organisms in soil are bacteria, which play a crucial role in nutrient cycling and other soil biogeochemical processes. Bacteria respond to major soil disturbance, such as droughts, floods, and land management activities e.g., tillage, lime addition. changes and thus there can be a change in composition of bacteria. Bacterial composition can be measured in different ways which provides different levels of detail about composition. For example, composition described by phospholipid fatty-acid analyses can be expressed as amount of gram positive and gram-negative bacteria with only selected bacterial taxa identified, while more recent analyses have focussed on using molecular approaches, in particular eDNA metabarcoding approaches, which provide a finer level of detail about the composition of bacteria in a given soil sample.

State of knowledge of the '*Bacteria composition in soil*' attribute: Poor / inconclusive – based on a suggestion or speculation; no or limited evidence

There is some knowledge on bacterial compositional change in relation to land use types or activities. However, while we can answer the question of 'Who is there?', we are still struggling to understand the 'What are they doing?', and therefore the significance of compositional changes. 'What are they doing?' can be done on a smaller scale, but our current limitation is cost effective analysis of soil samples at a national scale.

Part A—Attribute and method

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

Soil bacteria are key players in soil nutrient cycling and their composition is known to influence the ecological integrity by either enabling or stalling some of those nutrient cycling functions. When examining bacterial composition in soil, information on which bacterial taxa are present can be provided and some information on abundance and function they perform in soil can be provided although these relationships are variably known. In relation to human health, soil bacterial composition in itself is not a risk, however if there are pathogenic bacteria present and growing in soils, that can become an issue from a human health perspective if these bacteria enter waterways or food sources. However, bacteria are not the only soil organisms that contribute to biogeochemical processes. Fungi, protists, and invertebrates are just as important, and they all contribute to soil nutrient cycling. They are also just as sensitive to external changes in soil as bacteria and it is important to encompass total soil microfauna when talking about soil biodiversity and soil health [1].

Bacterial composition in soils can be measured in different ways which provides different levels of detail about composition. For example, composition described by phospholipid fatty-acid analyses can be expressed as amount of gram positive and gram-negative bacteria with only selected bacterial taxa identified – alongside information on fungal community. More recent analyses have focussed on molecular analysis using eDNA metabarcoding approaches, which provides a finer level of detail about the described taxa of bacterial population present in a given soil sample. When talking about bacterial composition, we often compare alpha and beta-diversity measures to look for patterns occurring between different soil samples. Bacteria are often very sensitive to soil disturbances and with their (generally) fast growth rate, their composition gives us a good insight into which bacterial communities respond to an external factor of interest (e.g., fertilisation, tillage, drought, heavy metals). Since bacteria are major drives of soil biogeochemical processes, we can extrapolate what soil functioning changes in soil based on the taxa present.

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

Several studies looked at the relationship between soil bacterial composition to soil health, but they are mainly limited in the scope or defined by the specific case study. When studies look into bacterial composition as an indicator of soil health, they often correlate microbial alpha and beta diversity measures to soil disturbance. Internationally, [2] found that bacterial and fungal diversity is lower in undisturbed systems (woodlands) and high in grasslands and disturbed environments (croplands). [3] searched for bacterial biomarkers that would be directly correlated to soil health after major disturbance and found that genome size, interpolated from bacterial community data is the best predictor of soil disturbance (tillage) that they measured. Nationally, [4] also looked at correlations between bacterial composition in soil and soil chemical properties. However, the evidence of direct impact of bacterial composition on ecological integrity is very weak, with studies only looking at correlation of bacterial beta diversity with soil type. While there are very clear trends emerging from individual studies when examining correlations between bacterial diversity and soil type, these analyses are very site specific. When looking at different soil types and land uses in different environments, the correlation will change and that makes current microbial measurements not suitable for predicting soil health metrics to be used across the globe [5, 6]. In the recent years, there has been increasing international evidence that microbial functioning (spanning wider than just bacterial) is a better soil health indicator than diversity measures [7].

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

This is unknown as there have been no studies that provide consistent information over time in New Zealand. How soil disturbance affects bacterial composition depends on the readiness of the soil bacterial community to change. Some soils can be considered more resilient and more ready for change when multiple members of soil bacterial community can support the same soil function (e.g., nitrification). These soils will perform better over time and will adapt to new conditions faster [8].

Currently, studies are only studying one or two disturbances in the study system. This gives us important oversight into potential resilience of soil community, but we are still unable to predict the response of microbial community when they are exposed to multiple disturbances, such as successive weather events (e.g., prolonged drought, followed by prolonged flooding). These are already happening as our climate keeps changing and it will be an even more common occurrence in the future [9].

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

The attribute is not currently monitored or reported as such either nationally or internationally and there is no clarity of use [6].

Improvements in technology accelerated the scale of which the soil bacterial composition can be measured. Initially, phospholipid-fatty acid profiles were the main way to determine bacterial composition by determining Gram-positive:negative ratios which is correlated to C, N availability and drought resistance, however this method is limited by the number of samples that can be processed at the same time. With development of next generation sequencing, metabarcoding became a primary way to sequence and determine bacterial composition in soils, with the ability to sample a wide range of samples in one sequencing run. However, this still only gives us information on microbial taxonomy which can only be linked to soil health and functioning via correlation. While there has been an evolution in methodology, it is still unclear how to best measure microbial response to soil disturbance and how to interpret/standardise [9].

There has been an EU funded study (Benchmark programme - https://soilhealthbenchmarks.eu/scientific-articles/) that looked at soil indicators in Europe and while they have acknowledged soil bacteria composition as an option, it is not consistent enough to include it on the list on indicators.

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

Private land may be required to be accessed for soil sampling for regulatory informing purposes, and thus there can be issues with access.

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

To monitor bacterial composition via eDNA, soil must go through DNA extraction and then barcoding and library preparation before being sequenced. While the cost has reduced significantly in the past decade, the attribute needs to be analysed in a molecular laboratory. Sequencing is the main cost of the analysis and it is costed per run, which can include ~200 samples in one MiSeq (Illumina) run.

One MiSeq run with 200 samples cost can be estimated between $6000 and $8000 (from DNA extraction to sequence output), with additional time needed for a professional bioinformatics analysis. The analysis time can be estimated to 20h for a standard community analysis, with more time required if there is any more detailed analysis needed. This costing is under the assumption that soil samples do not need any further optimisation and DNA can be extracted using standard methodology. Some soils (such as allophanic soils) bind DNA to soil particles and can be particularly difficult in DNA extraction [10]. Those soil samples would increase the cost of any molecular analysis by at least 3-fold or more.

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

We are not aware of any soil bacterial profile monitoring being undertaken by iwi/hapū/rūnanga. However, we are aware of many soil health monitoring programmes being led by Māori (see, e.g., [18].

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

Bacterial composition is directly linked to Soil nitrogen and phosphorus and Soil carbon, especially since bacteria are major drivers of cycling associated with those attributes. Soil compaction and Soil water storage, capacity and fluxes will have an impact on bacterial composition in relation to water and oxygen availability that would drive the bacterial composition. Soil bacterial composition is also heavily dependent on aboveground plant diversity, so Indigenous plant dominance attribute would be related to the Bacterial composition attribute.

Part B—Current state and allocation options

B1. What is the current state of the attribute?

Currently there is no routine monitoring of the soil bacterial composition and there is also no consensus on how to interpret the diversity data to link it to soil quality indicators, since these relationships are primarily based on correlation, rather than establishing causality. There is a call internationally for validation and standardisation of microbial bioindicators (i.e., specific bacterial taxa or genome size) as soil health attributes [11].

Overall, current state of bacterial diversity attribute is only known for local samples (i.e., individual studies), but it remains unknown on a national or even regional scale.

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

The study by Hermans *et al.* [4] is the most extensive study to date on the use of bacterial composition as a soil health metric in New Zealand. They were able to demonstrate that soil bacterial community composition corelates to land use and that when modelling the site land use, the best way forward was when combining chemical/physical data with microbial diversity data. However, that still only gives us a high-level granularity – land use and is not easily converted into the quantitative measure for soil health metrics. Generally, native forest is considered a reference state however as a study by Hermans *et al.* [4] demonstrate, native forest has a lower bacterial diversity than managed land (often considered "unhealthy"), which makes it difficult to set a reference point for this attribute.

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

To our knowledge, there are no existing numeric or narrative bands.

Nationally and globally, there is a recognition that soil microbiota (combined bacteria, archaea, fungi, viruses, nematodes) is a major factor in soil health and it can be influenced by land use and management practices, but no numeric or narrative bands have been defined for bacterial composition. Even if the bands would be well defined, this would be very difficult to do on a national scale, but it would have to be on a more local scale and land-use dependant.

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

To our knowledge, there are no known thresholds or tipping points for bacterial composition.

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

On a broader scale, it is well understood that land use change leads to a lasting effect on soil microbial community composition. That has been demonstrated both in New Zealand and abroad [12, 13]. However, these broad community patterns do not take into consideration differences between active part of the bacterial community and inactive, dead, or even just free-soil DNA. Bacterial communities in soil react extremely quickly to any changes in their environment, often within minutes, however the growth of the microorganism or persistence of DNA in soil, lags behind those changes and is not responsive enough to make conclusion with high confidence. There are different techniques available, that can overcome these issues (such as stable isotope probing), but they are not cost-effective for routine monitoring. At the moment we are unable to predict how fast bacterial composition adapts to changes in ecological integrity, since this is directly connected to bacterial resilience at individual soils and is site specific.

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

Māori have high interest in soil health, and this encompasses a holistic approach to assessing the condition of all parts of the soil ecosystem (and beyond). In addition to discussing this attribute directly with iwi/hapū/rūnanga, there is likely to be tikanga and mātauranga Māori relevant to informing bands, allocation options, minimally disturbed conditions and/or unacceptable degradation in treaty settlements, cultural impact assessments, environment court submissions, iwi environmental management and climate change plans, etc.

Part C—Management levers and context

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

No, since the use of microbial composition measurement itself is not very clear. There are patterns in soil bacterial composition that directly correlate to various land management practices, but the measurements of beta diversity are too variable to base management interventions on [4, 14].

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

C2-(i). Local government driven

Local government, e.g. the regional councils via the land monitoring forum have an interest in using biological indicators to assess soil health in SOE soil health monitoring and have supported research into the use of bacterial composition using eDNA metabarcoding approaches, although use as a soil health indicator is still in a scoping phase.

C2-(ii). Central government driven

None existing, other than the current attribute scoping.

C2-(iii). Iwi/hapū driven

Iwi planning documents such as Environmental Management Plans and Climate Change Strategies/Plans may contain policies/objectives/methods seeking to influence soil bacterial quality outcomes for the benefit of current and future generations. We are not aware of any other interventions/mechanisms being used by iwi/hapū/rūnanga to directly affect this attribute.

C2-(iv). NGO, community driven

C2-(v). Internationally driven

Internationally, soil microorganisms are well recognised as key drivers of soil functioning, but there is no consensus on how to measure that in a validated methodology. Bacterial composition measurements (alpha and beta-diversity) were the primary way to assess the state of soil bacteria, but research focus has now shifted into the area of measuring functions using metagenomics and looking at genes involved in soil nutrient cycling [11] as better ways to provide meaning in the context of environmental state and ecological integrity.

Part D—Impact analysis

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

This is unknown since connections between bacterial composition and environmental or human health effect are largely inconclusive outside of specific cases where there was an issue with release of contaminated water onto land or pathogenic bacteria entering drinking water reservoirs from soils due to agricultural or industrial practices.

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

Soil bacteria are often a source of new antibiotic resistance genes [15], which would impact public health sector. Soil can also harbour pathogenic bacteria if there is a spill of wastewater or effluent on crop, which would have an economic impact both on agriculture (via loss of crop) and public health.

Biosecurity is another economic threat, with pathogens in soil can not only target humans, but also agricultural products (plants and animals).

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

Bacterial populations in soil will and already are reacting to climate change via changes in the community compositions and functioning they perform. There is a call from national and international scientific community to improve the way we measure bacterial response to environmental stressors using new methodologies and more scales that would represent future climate change conditions [6, 9, 16, 17]. There is also a greater need to properly standardise the measures of soil bacterial response and resilience if we want to use this information to form management strategies.

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7.3 Soil nitrogen and phosphorus

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Preamble: There are multiple ways to measure soil nitrogen (N) and phosphorus (P) individually, including:

- Total P concentrations in soils.
- Olsen P – a measure of plant available P in soil [1], commonly used as an indicator of soil fertility and of the risk of excess phosphorus loss to the wider environment [2]. Olsen P can be measured gravimetrically or volumetrically, with these methods producing different values, although it is possible to convert between them if the volume weight or bulk density of the soil is known [3]. The gravimetric method is specified for State of the Environment (SOE) reporting [40] while the volumetric method is predominantly used for fertiliser recommendations.
- Anion sorption capacity/P retention – an inherent soil property that measures the ability of a soil to bind P in the soil matrix. This measure can inform soil characterisation and inherent risk of P loss from soils. Irrigation with high P-loading has been shown to decrease the anion sorption capacity of soils over time, leading to increased risk of P loss [4].
- Alternative estimates of plant available P, based on different extraction methods are also available e.g., Melich P, Bray P [5].
- Several less common measures of soil P include using fractionation methods to identify organic and inorganic forms of soil P to inform the mechanisms controlling the plant availability of soil P over time [5], spectroscopy methods (e.g., NMR, XAS, NanoSIM) that identify P concentrations and bonding forms, and novel methods to trace soil P reactions (e.g., 33P isotope dilution, zymography, DGT) [6].
- Total N concentrations in soils.
- C:N ratio – this measure can indicate whether there are potential N limitations to plant growth and has therefore been used to indicate mineralisation rates of organic matter in soils [7]. This measure provides information about the nature of the biological communities in soils – soils with higher C:N ratios have more fungal-dominated communities [8]. Other nutrients are also linked to SOC cycling, including P and S,

therefore nutrient stoichiometry of soil organic matter may be appropriate to use if considering organic matter cycling.

- Anaerobically mineralisable N (AMN) has been used as a measure of the nitrogen available to plants over the course of a growing season and correlates well with microbial biomass [7,8].
- Hot water extractable N (HWEN) is suggested as a more robust, replacement measure for AMN, measuring biological activity in soils [9,10,11].
- Hot water extractable carbon (HWEC), like HWEN has been proposed as a replacement for AMN in regional soil quality monitoring, also used as a measure of biological activity in soils [12,13].
- Mineral N, comprised of nitrate and ammonium forms of nitrogen in soils, measures the nitrogen available to plants and can be used as a measure of soil fertility.
- Quick N – a measure developed to help with fertiliser decisions, that estimates the nitrate available in soils for plant uptake.
- Nitrous oxide emission losses measure the loss of excess N as a greenhouse gas from soils to the atmosphere.

State of knowledge of 'Soil nitrogen (N) and phosphorus (P)' attribute: Good / established but incomplete (movement to and impact on waterways), and Medium / unresolved (impact on ecological integrity on soil)

There are a plethora of studies that provide information on the concentrations of P (typically as Olsen P) and N (various measures), primarily in agricultural soils (specifically, see sections A3 and B1). These studies most often assess the state, or relationship to agricultural production of pasture or plant crops, or to inform fertiliser requirements. There are few studies that assess the response of soil biota to P or N additions – with the response of the microbial community most frequently assessed. Studies that provide soil N and P under indigenous vegetation have also been undertaken and most often provide an assessment of state (noting that work on chronosequences has assessed nutrient cycling, ecosystem development and retrogression), rather than assessing impact of changes or additions over time.

However, the primary concern about soil N and P is in agricultural areas and relates to impacts on aquatic systems, in particular freshwater systems. There is considerable knowledge about the factors influencing loss of P and N from soil to water, which is strongly influenced by land management practices. Movement of N and P to waterways requires that there is a transportation pathway of N and P from soil to water, and that there is a lack of attenuation processes along this pathway and is exacerbated by the presence of N and P that is surplus to plant requirements. There is also considerable knowledge of the effects of excess soil P and N on aquatic systems.

Part A—Attribute and method

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

Ecological integrity. N and P are essential elements for the growth and functioning of all soil organisms, including plants. Beyond this critical role, P additions to soils affect soil microbes' (bacteria and fungi) population composition and function. Strong correlations were identified between relative abundances of individual soil taxa and concentrations of Olsen P [14]. Prolonged soil P additions can lead to microbial growth becoming C and N limited, in the absence of additions of these elements [15], and decoupling of the interactions between plants and soil organisms, resulting in plant reliance of fertiliser P rather than on biological mineralisation of P [16]. Arbuscular mycorrhizal fungi functioning is suppressed by excess P can negatively affect the productivity and temporal stability of plant communities [17].

The primary concern associated with elevated P in soils is surface run-off and movement into surface waters, negatively affecting freshwater quality and aquatic ecosystems [18]. This loss to surface water occurs when nutrients surplus to plant requirements are present in the soil, there is a transport pathway from soil to surface water, and there is a lack of sufficient attenuation processes along this transport pathway to decrease the P (and N) lost [19]. These factors can be influenced by land use and management. Excess P and N can also leach into deeper soil horizons, and may enter groundwater, negatively affecting groundwater quality [20,21]. Leaching of P to groundwater has been observed in soils under intensive land use receiving P additions, and is more likely in soils that support rapid transport of P, i.e., soils that are sandy, stony, shallow, or recent with low anion sorption capacity [20].

Addition of N to soils through fertiliser, cow urine, and effluent applications also affects microbial community function. Fertiliser N additions can lead to lower levels of biological N fixation [22]. Excess soil N further negatively affects ecological integrity by resulting in leaching of nitrate to groundwater which negatively affects groundwater quality [23,24], runoff of N to surface water [25], and emissions of nitrous oxide, a greenhouse gas, to the atmosphere [26].

Human health. Effects on human health arise indirectly through contamination of surface water with excess soil nutrients resulting in the growth of harmful algae e.g., cyanobacteria. If groundwater sources used for drinking water become sufficiently polluted to exceed drinking water standards for nitrate and nitrite, there is a risk to human health through consumption of this water [27,28,29]. There are no human health effects directly associated with elevated soil P and N.

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

The impact of P and N additions to soil on the ecological integrity of soil is largely unknown. A handful of studies have addressed the relationship between soil P and N and soil biology, primarily bacteria and fungi composition and function (Section A1). One study assessed the impacts of long-term P and N additions on soil biology, namely bacteria, fungi and earthworms [30], finding prolonged additions of both P and N decreased fungal biomass, N additions decreased microbial biomass, and higher earthworm abundance was associated with increasing P.

The evidence of impact of P and N on the ecological integrity of freshwater is strong: Studies have established the management factors and transport pathways involved in the loss of soil P and N to surface and groundwater [2,18,20,31], and have demonstrated evidence of this occurring in New Zealand [21,23]. Specifically, where nutrient additions (including fertiliser, effluent and wate-water applications) exceed plant requirements, and there is a transport pathway with limited attenuation of nutrients, the risk of negative impact on surface and groundwater quality increases due to the potential for transfer of soil P and N to drainage and to direct discharge to surface waters [19,21,32].

The extent and magnitude of degradation of waterways due to excess soil P and N varies by land use, soil type, catchment characteristics, topography, and farm management practices. Losses are not uniform or nationally consistent, with some intensive land uses are more prone to excess nutrients in soils due to larger quantities of fertiliser used or effluent and/or waste-water applied to soils, e.g., dairy farming [23]. Losses from critical source areas on farms and from soils with low anion sorption capacity and/or macropore flow contribute disproportionally to total nutrient losses, and can be influenced by farm management practices [33,34].

N-additions to soil also impact on greenhouse-gas emissions as 94% of New Zealand's N_2O emissions are from agriculture with N_2O emissions from N fertiliser use making up approximately 3.9% of agricultural emissions [35].

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

There are limited studies that assess the change in soil P concentrations over time, and even fewer studies assessing the change in soil N over time. The most extensive study assessing P concentration over time analysed ~450,000 samples collected over 2001-2015 and processed by commercial laboratories [36]. This study found a national mean rate of increase in the median concentration of Olsen P in soils in different regions by land use and soil order group combinations of 1.2% [36]. Median concentrations varied from 9 to 52 mg/L in the different groups. Across the 124 combinations, there were 32 significant trends – both increases and decreases – over this time period. In terms of specific land uses, Olsen P in soils under drystock and cropping sites increased between 1996-2018 while total N decreased in cropping and indigenous vegetation soils over this time period [37,38]. Analysis of data from the Greater Wellington region over 2000-2018 showed statistically significant increases in Olsen P concentrations in cropping systems and total N concentrations in drystock systems [39].

The pace or trajectory of change in the future 10-30 years is unknown, and depends on the extent to which known management practices are adopted e.g., ensuring application of fertilisers is sufficient and not excessive for agricultural crop-growth. Elevated soil P and N is reversible, where ongoing inputs are reduced or stopped and plant growth uses the available P and N. Predicted median timeframes to decrease Olsen P in soils with elevated concentrations across New Zealand to a water-extractable P concentration proposed as an environmental target of 0.02 mg/L has been assessed as within a year for most soils, while some land uses are predicted to take up to 11.8 years [36]. However, as some environmental targets are more conservative than the one used in the above prediction, timeframes for reductions to actual targets may be longer.

The pace and trajectory of change of N and P in aquatic systems is beyond the scope of consideration in this attribute.

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

Most regional councils in New Zealand monitor some measure of soil nutrients under their soil quality monitoring programmes for SOE reporting. Olsen P, total N and AMN have been three of the seven key soil quality indicators since the implementation of SOE monitoring in the early 2000s. More recently, HWEC has been more widely monitored since it has been proposed as a replacement for AMN [9,10,13]. The NEMS for soil quality and trace elements [40] specifies a standard for sampling analysis of soils for Olsen P, total N and AMN. Data are usually compared to provisional target values for these soil quality indicators [41]. Long-term field trials at Winchmore and Ballantrae have also assessed and reported soil P (as Olsen P) and N (Ballantrae only, as mineralisable N and C:N ratio) under grazed grassland in multiple reports and journal papers [30,42,43].

Soil P and N, generally as Olsen P and mineral N, are also measured in relation to crop requirements to inform fertiliser requirements by farmers. The Fertiliser Association of New Zealand has produced multiple guides for this with agronomic targets for different crops and production systems [44].

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

For all direct soil measures, there is a need to access privately owned land to collect repeat samples for monitoring of this attribute. Landowners may be more, or less, willing to provide access to land for sampling and to have data from their land used for regulatory informing purposes.

Indirect measures of this attribute, i.e., measures that indicate cumulative losses from soil such as total P and DRP in surface water, nitrate in groundwater and nitrous oxide emissions to the atmosphere are less straightforward to attribute to a specific source or location. However, the general health over time of receiving waters can be measured and this may require access to privately owned land to access rivers and lakes to collect repeat samples.

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

Variables estimates provided by Regional Council scientists to MfE:

- $10,000 per year estimated by Marlborough Regional Council, broken down as: Chemical laboratory analyses of which Olsen P, total N and AMN are included, for ~20 sites/ soil samples. Two people sampling eight full time days per year.
- $85,000 total cost per year (pers comm Waikato Regional Council), broken down as follows: ~$1000 per sample/site for all seven basic soil quality indicators (including Olsen P, total N and AMN). For approximately 30 sites, one scientist spends approximately one third of their time on soil quality monitoring.
- $80-100,000 per year (pers comm Horizons Regional Council), for monitoring of the seven soil quality indicators, not including staff training and farmer outreach.
- $250,000 per year monitoring costs plus Regional Council soil scientists' time (unspecified, 5 staff in team) (pers comm Environment Canterbury).

Various measures of soil P and N are available from commercial laboratories, often as part of a suite of different tests e.g., Basic soil profile analysis (includes Olsen P), organic matter suite (includes Total N), with costs ranging from $27 to $140.

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

Although N and P monitoring by representatives of iwi/hapū/rūnanga may be uncommon, measurement of P and N via standard techniques is common in Māori agribusiness. Perhaps the question here is whether iwi/hapū/rūnanga are using techniques other than standard Olsen P measures to measure the nutrient profiles of soils? We are not aware of any mātauranga Māori-led measures of N and P specifically, but emphasise that Te Ao Māori measures of soil health take a holistic approach (a measure of the overall mauri of the soil ecosystem).

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

Soil attributes –

- Bacteria composition: There are known nutrient changes with composition, however the specific and defined relationships are not yet fully described.
- Soil C: Through the C:N ratio of soils. Lower C:N ratios with high total N are associated with increased losses of N [50].
- Soil contaminants: Trace element contaminants including Cd, F and U can originate in phosphate fertilisers. As such, applications of phosphate fertilisers can lead to accumulation of these trace elements in soils [51,52]
- Surface erosion/runoff (and other erosion related attributes): Soil P is often lost through erosion and runoff, as it adheres to soil particles. Therefore, erosion and runoff of soil to surface water can also result in loss of P to surface water.
- Soil compaction: Runoff of P and N are more likely from compacted soils (indicated by low macroporosity) [53]. Compaction reduces soil porosity, which can result in runoff that can contain unattenuated P and N due to the soil's reduced ability to infiltrate and store water [54,55].
- Soil water storage, capacity and fluxes: As noted in the point above, when the water storage capacity of soil is reduced due to human activities, runoff and leaching of soil P and N are increased.

Freshwater attributes –

- Multiple National Policy Statement for Freshwater Management (NPS-FM) attributes can be influenced by the movement of soil P and N into freshwater including N, P, plants and algae (specifically phytoplankton and periphyton, as trophic state attributes), and cyanobacteria [56].
- Groundwater nitrates: Groundwater nitrates are a direct result of N losses from soils, where there are excessive N inputs, e.g., fertiliser application, livestock (urine), effluent or wastewater application

- Riparian margin establishment/protection: This can reduce the loss of P and N to surface water from surrounding agricultural land [57,58].

Estuaries and coastal waters –

- Nutrients in water (trophic state and toxicity): Nutrients P and N in water, causing changes to trophic state and toxicity risks, result from P and N losses from surrounding soils.
- Cyanobacteria in water: Cyanobacteria blooms are caused by excess nutrients in water, which can result from P and N losses from surrounding agricultural soils.

Part B—Current state and allocation options

B1. What is the current state of the attribute?

Olsen P and total N are measured for SOE and were last reported at a national scale in *Our Land 2021* [38]. Recently, *Our Land 2024* was released however soil N and P were not updated in this report. Data in *Our Land 2021* includes data up to 2018. Therefore at a national scale, we have some understanding of the state of this attribute, however, the representativeness of the current SOE monitoring framework for providing a national assessment has not been determined. Olsen P status of soils varies depending largely on land use. *Our Land 2021* reports that 61% of sites under both cropping and dairy, 46% of sites under orchard/vineyards, and 30% of monitored dry stock sites exceeded targets. As detailed in A3 above, Olsen P has increased in soils over the past ~30 years. Total N concentrations in soils monitored between 2014-2018 were within the target range for more than 72% of monitored dairy, drystock and forestry sites [38]. Cropping and orchard/vineyards sites do not currently have target ranges for total N to compare data to due to the variations in N requirements of different crops. Despite the majority of dairy, drystock and forestry sites being within total N targets, between 2016-2020, 69% of New Zealand's river length had modelled N concentrations indicating risk of environmental impairment compared to reference conditions [59].

Extensive additional information on the state of this attribute (both N and P) will be held by fertiliser companies or farm consultants as a result of soil-testing to inform fertiliser requirements. The most likely measures here are Olsen P and some measure of plant available N – e.g., mineral N or quick N.

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

SOE soil quality monitoring of Olsen P, total N and AMN in soils under indigenous vegetation could potentially represent reference states for these measures in New Zealand soils, however there are few indigenous sites included in SOE monitoring, and where included, they may not necessarily represent undisturbed indigenous vegetation. Additional studies of soils under indigenous vegetation could add to this knowledge base [60, 62]. However, New Zealand soils are known to be naturally low in plant available P due to New Zealand's geology and climate influencing soil development processes, and indigenous ecosystems are typically adapted to these soil conditions [61,62]. Thus, the relevance of this reference state to inform N and P concentrations in soils under primary production use is debateable since N and P requirements will be driven by requirements of specific crop or pasture, which can vary considerably. Similarly, this reference state is unlikely to be suitable for soils where exotic plants are desirable e.g., grass on sports fields.

SOE targets for this attribute are generally used in productive systems where optimum levels of soil fertility are targeted for crop and pasture growth. Suggested target values for these indicators have historically been grouped by soil type and/or land use [41], and as these measures can inform soil fertility, productive systems strive for higher levels (and specific levels vary by crop) [44,63]. Therefore concentrations under indigenous vegetation are unlikely to relevant for production land use.

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

'Target values' have been developed for use in regional council state of the environment reporting, with a review of the derivation of these values recently undertaken by Manaaki Whenua – Landcare Research [41]. These values were based on combining production optima and environmental considerations, although limited environmental data was available at the time of development [41]. These values vary with land use and soil order, with the current recommendations for Olsen P concentrations in soils shown in Table 1. The values in Table 2 are similar but not identical to those used by MfE and StatsNZ in *Our Land 2021* [38]. Revision of these target values currently being contracted by MfE (Revision of Soil Quality Indicator Target Ranges).

Table 1. Suggested Olsen P target ranges. Units not specified but assumed to be mg/kg. Original source from Mackay et al. [64], taken directly from Cavanagh et al. [41].

Land use	**Soil type**	**Minimum**	**Maximum**
Pasture; horticulture and cropping	Volcanic	20	50
Pasture; horticulture and cropping	Sedimentary and Organic soils	20	40
Pasture; horticulture and cropping	Raw sands and Podzols with low P retention	5	5
Pasture; horticulture and cropping	Raw sands and Podzols with medium and above P retention	15	25
Pasture; horticulture and cropping	Other soils	20	45
Pasture; horticulture and cropping	Hill country	15	20
Forestry	All soils	5	30

Table 2. Olsen P target ranges used by MfE and StatsNZ [38].

Soil order	Land use	Min	Max	Unit
Raw	All	5	25	µg/g
Podzol	All excl. cropping and orchard/vineyard	5	25	µg/g
Podzol	Cropping, orchard/vineyard	20	50	µg/g
Allophanic, Pumice	Exotic forestry	5	30	µg/g
Allophanic, Pumice	All excl. exotic forestry	20	50	µg/g
Brown, Gley, Granular, Melanic, Oxidic, Pallic, Recent, Semi-arid, Ultic	Dairy, Drystock, Lifestyle, Scrub, Tussock, Urban Park/Reserve	15	45	µg/g
Brown, Gley, Granular, Melanic, Oxidic, Pallic, Recent, Semi-arid, Ultic	Cropping, orchard/vineyard	20	45	µg/g
Organic	All excl. exotic forestry	20	40	µg/g
Organic	Exotic forestry	5	30	µg/g

NB: Anthropic soils not mentioned

Total N targets used by MfE and StatsNZ [38] are those specified in guidance developed by the LMF for use in SOE monitoring [65](Table 3), summarised as 0.25-0.7% for pasture, and 0.1-0.7% for exotic forestry.

Table 3. Total nitrogen target ranges (% w/w). Bold values indicate target values.

Land use	Very depleted	Depleted	Normal	Ample	High	
Pasture	0	**0.25**	0.35	0.65	**0.70**	1.0
Forestry	0	**0.10**	0.20	0.60	**0.70**	
Cropping and horticulture			excluded			

Source: LMF [65].

AMN targets used by StatsNZ [38] are those proposed by the LMF [65] (Table 4), summarised as a minimum of 20 mg/kg for cropping, horticulture and exotic forestry, and a minimum of 50 mg/kg for pasture. No land uses have maximum targets for AMN.

Table 4. AMN target ranges (mg/kg). Bold values indicate target values.

Land use	Very low	Low	Adequate	Ample	High	Excessive	
Pasture	25	**50**	100	200	200	250	300
Forestry	5	**20**	40	120	150	175	200
Cropping and horticulture	5	**20**	100	150	150	200	225

Source: adapted from LMF [65] using information from Mackay et al. [64].

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

Water extractable P is an additional measure that has been used to assess potential P loss via surface runoff [36]. This is a modelled attribute, derived from Olsen P and anion storage capacity. An environmental target of 0.02 mg/l was used in McDowell et al. [36]. This value was selected to limit eutrophication (and is based on all default guideline values for DRP concentrations across lowland rivers or lakes or cool climates set by the Australian and New Zealand Governments, ranging from 2-20 µg P/L [66]). However, pursuing low water extractable P concentrations may result in Olsen P concentrations lower than the agronomic target in many soils, impairing production [36].

There are various recognised 'changes' on ecological integrity arising from different nutrient conditions e.g., high nutrient conditions will favour pasture grass growth over native plant species, there are no specific thresholds or tipping points.

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

As stated in Section A3, the trajectory of change of this attribute, and therefore lag times and legacy effects on the ecological integrity of soil depends on the extent to which known management practices are adopted. Elevated soil P and N is reversible, where ongoing inputs are reduced or stopped and plant growth uses the available P and N.

There is suggested to be a lag time between interventions to manage elevated soil P and N and the impacts of these excess nutrients on freshwater quality [36, 67]. An assessment of lag times for nitrates arising from livestock farms entering surface water in 34 New Zealand catchments found that the median lag time was 4.5 years, with a total range of 1-12 years [67]. Lag time was influenced by factors including catchment size and slope, illustrating legacy effects.

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

As noted previously, soil health is an area of high interest to Māori and there are many tohu/indicators that are utilised according to mātauranga-ā-hapū and mātauranga-ā-iwi [78,79]. In addition to discussing this attribute directly with iwi/hapū/rūnanga, there is likely to be tikanga and mātauranga Māori relevant to informing bands, allocation options, minimally disturbed conditions and/or unacceptable degradation in treaty settlements, cultural impact assessments, environment court submissions, iwi environmental management and climate change plans, etc.

Part C—Management levers and context

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

Fertiliser application to soils in agricultural systems is a key factor influencing the state of this attribute [23,36]. Intensified land use in turn can drive fertiliser applications to soils. A fertiliser code of practice [68] and guides for fertiliser management of productive systems [44] have been developed by the Fertiliser Association of New Zealand to inform management of this issue through fertiliser application. Adoption of the code of practice is voluntary and adoption rates are not assessed.

Similarly, wastewater or effluent application – including municipal wastewater, dairy shed effluent, or wastewater from food manufacturing plants, e.g., dairy factories, to land can also negatively influence the state of this attribute.

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

C2-(i). Local government driven

SOE monitoring by is undertaken by Regional Councils, however the extent of interventions to effect change in soil N or P status, is largely limited to reporting soil quality monitoring results, which may include reporting back to the individual landowners on whose properties the sampling has been undertaken. Elevated Olsen P concentrations have been reported in agricultural soils since the commencement of the 500 Soils programme in 2000 [69], and concentrations have continued to increase since this time [36].

Regional and District Plans and Policies may specify requirements and require resource consents for farming activities that are likely to result in P and N loss from soil. They may require the use of Overseer or other tools or plans to calculate and manage nutrient budgets.

C2-(ii). Central government driven

National Environmental Standard for Sources of Human Drinking Water [70] specifies that drinking water sources comply with the New Zealand Drinking-water Standard [29], which specifies maximum accepted values for nitrate and nitrite of 50 and 3 mg/L, respectively, also providing that the sum of the ratio of the concentrations of nitrate and nitrite to each of their respective maximum accepted values must not exceed 1.

The National Environment Standard for Freshwater [71] refers to the National Policy Statement for Freshwater Management [72]. This National Policy Statement regulates some of the same drivers (fertiliser inputs, animal stocking density) that are important for soil ecological integrity as affected by excess soil P and N, however there is no specific legislation or policy for soil P or N.

Freshwater Farm Plans are required in some regions for certain farming activities. These are legislated by the Resource Management (Freshwater Farm Plans) Regulations 2023 [73] and compliance is monitored by Regional Councils. They have no specific provisions for soil P and N however are intended to protect freshwater quality from farming activities.

New Zealand's national emissions reduction plan [35] includes N_2O emissions from agriculture in the net-zero emissions target for 2050.

C2-(iii). Iwi/hapū driven

As noted above, we note that hapū/iwi take a holistic approach to environmental monitoring. Iwi planning documents such as Environmental Management Plans and Climate Change Strategies/Plans may contain policies/objectives/methods seeking to influence soil quality outcomes for the benefit of current and future generations. We are not aware of any other interventions/mechanisms being used by iwi/hapū/rūnanga to directly affect this attribute.

C2-(iv). NGO, community driven

Catchment management groups to improve water quality outcomes, generally relating to farming activities, exist in many catchments throughout New Zealand.

C2-(v). Internationally driven

The Paris Agreement [74] on climate change includes mitigating N_2O emissions from agriculture in both the near-term target: close 10% of emissions gap by 2020 to achieve 2°C warming target; and the long term target: cumulative emission reduction of up to 60 Gt CO2e and 3500 ozone depletion potential kt by 2050.

Part D—Impact analysis

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

Changes in the attribute state affect ecological integrity and potentially human health as described in A1 above. Continuing to have excess P and N in agricultural soils leads to a greater risk of contamination of surface water and groundwater, negatively affecting freshwater quality and aquatic ecosystems. This can affect human health if surface water becomes eutrophic, causing the growth of harmful algal blooms, or if drinking water sources become polluted.

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

The issue of excess P and N in soils primarily applies to agricultural land – in particular more intensive land uses including dairy and vegetable cropping that apply large quantities of fertilisers. The impacts of excess N and P in soils are likely mostly externalities, affecting other environmental domains. However, the application of excess P and N fertilisers is an inefficient use (waste) of money. This is often not realised, as application of fertiliser can be perceived as 'more is better', and growers can be risk averse to potential decreases in production.

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

Increases in extreme weather events may result in more leaching and runoff of excess P and N from soils to ground- and surface water, as the processes of runoff and leaching are stimulated by rainfall (and irrigation).

Warmer temperatures and wetter conditions are also predicted to increase nitrous oxide emissions from soils [75] resulting in a positive climate feedback, whereby emissions continue to increase [76].

Microbial cycling of soil nutrients will be affected by temperature variations. Specifically, warmer temperatures are likely to enhance N-cycle processes and P utilization [77] which may result in reduced losses from soils.

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7.4 Soil Carbon

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Preamble: There are multiple measures for soil carbon (C) discussed in this document. We have listed some of the commonly used measures for soil carbon below. The measurements most used for soil carbon are total carbon, soil organic carbon, water extractable carbon, and the carbon to nitrogen (C:N) ratio. More detail on these measurements are outlined as follows:

- Total C concentration (%) in soils, commonly used in soil quality monitoring of mineral soils and expressed as a percentage of soil by weight (i.e., g C/100 g soil). Total carbon (and nitrogen) in soils is usually quantified using combustion. In New Zealand, total carbon is generally considered to be analogous to organic carbon due to negligible amounts of inorganic carbon. However, total carbon would include organic and inorganic carbon.

- Soil organic carbon (SOC). A measure of the total organic carbon pool in soils. This is determined following removal of any inorganic carbon after acid digestion. In New Zealand, this acid digestion step is usually not carried out. SOC can be further differentiated into different C fractions that are considered to differ in function and turnover. For example, particulate organic C (POC) is considered more labile and vulnerable to loss than mineral associated organic C (MAOC) which is more stable and persistent due to the interaction of C with minerals. Resistant organic C (ROC) is a fraction that is considered resistant to turnover and is typically associated with charcoal or pyrogenic carbon. In New Zealand soils, SOC generally consists of POC and MOAC as ROC is usually very negligible although there is limited data on this.

- Soil inorganic carbon (SIC) relates to carbon associated with inorganic constituents such as carbonates. Generally, New Zealand soils contain negligible amounts of SIC. However, in some soils (e.g., pH>7) SIC will contribute to the Total C measured in soils. The inorganic component would be the difference between total carbon and soil organic carbon.

- Soil organic C stock is the quantity of SOC in a soil for a given layer, usually defined by depth. The stock is calculated using the bulk density and total C concentration for a given layer of soil over a given area. The unit for stocks of SOC is typically given in tC/ha. The stock of SOC is considered a better measure than total carbon as it accounts for changes in bulk density that are often associated with changes in management. The stock of SOC should be assessed on an equivalent soil mass basis rather than a fixed depth basis particularly if comparisons between treatments, through time, are being considered.

- Water extractable carbon is the soluble fraction of carbon in soils that is typically associated with carbon cycling and microbial activity. Hot water extractable carbon (HWEC) has been proposed as a replacement for anaerobically mineralisable N in regional soil quality monitoring, also used as a measure of biological activity in soils [1, 2].

- Soil organic matter (SOM) comprises the organic material in soils, including organic carbon, nitrogen and other nutrients. This is typically determined using loss on ignition and measuring the weight lost. This method is cheaper than other methods but generally not preferred as there is no universal standard protocol and results can vary due to furnace settings.

- C:N ratio – this measure can indicate whether there are potential nitrogen (N) limitations to plant growth and has therefore been used to indicate mineralisation rates of organic matter in soils [3]. This measure provides information about the nature of the biological communities in soils – soils with higher ratios have more fungal-dominated communities [4]. It is important to note that other nutrients are also linked to SOC cycling, including P and S, therefore nutrient stoichiometry of soil organic matter may be more appropriate [5].

- CO_2 respiration: this represents the carbon that has been mineralised by microbial activity. The amount of carbon respired over a three-month period has been demonstrated to correlate well to the quantity of water extractable carbon [6].

- The CO_2 burst method is another method for measuring CO_2 respiration from soils. This measures microbial respiration under disturbed conditions and is a quicker method than described above. This method is used in soil quality monitoring in the US [7].

- Less common methods of measuring organic carbon in soils include spectroscopy methods, namely mid-infrared spectroscopy (MIR), nuclear magnetic resonance (NMR) spectroscopy [8]. It should be noted that some commercial laboratories in New Zealand routinely use spectroscopy, as opposed to combustion, as the main method to determine total carbon (and nitrogen) in soils due to the rapid assessment.

Remote/Hyperspectral sensing of SOC has received increasing interest recently due to the low cost associated with acquiring data. However, many of the studies that have assessed this approach rely on the use of existing large datasets where spectral data is already collected (e.g., LUCAS dataset in Europe) for ground truthing [9 - 11]. Another caveat is that generally the hyperspectral imagery is collected from bare soil with no plant cover. To our knowledge, this technique has not been applied for New Zealand soils.

State of knowledge of "Soil C" attribute: Medium / unresolved – some studies/data but conclusions do not agree. While there are some studies in New Zealand on soil carbon these have largely focussed on changes related to land use or specific management practices and related to greenhouse gas emissions. These studies have been summarised in review papers but knowledge gaps remain particularly related to impacts on ecological integrity.

Part A—Attribute and method

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

Ecological integrity: Soil C affects many soil properties and functions influencing soil resilience [12-15]. Soil C is linearly correlated to aggregate stability, an indicator of soil structural quality [16], therefore effects on soil C will impact the resilience of soil to water stress and drought - a feature dependent on good soil structure [17]. Soil C also provides nutrients through cycling of organic matter and climate regulation through carbon sequestration in soil [12] therefore declines in soil C will negatively impact these soil functions.

Greenhouse gas emissions: Soil contains the largest pool of the Earth’s terrestrial C. Loss of this stored C will cause feedback to and contribute to increasing atmospheric CO_2 concentration [18]. There has been a large amount of research in New Zealand, and globally, on management practices that could increase this pool of soil carbon or reduce losses. Internationally, the 4per1000 initiative was created recognising the importance that increases to the global stock of soil carbon could have for mitigating greenhouse gas emissions. However, carbon is important for soil functions and health beyond just the GHG mitigation potential.

There are also links between soil and other attributes. For example, soil C relates to the ecological integrity of water, as organic matter cycling in soils affects nutrient loss (namely N and P) to waterways which can result in negative impacts on aquatic ecosystems [5].

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

Weight of evidence for C depletion under certain management (e.g., irrigation [19]) and land use (e.g., cropping [6, 20-22]) is high.

Land use change is generally considered the largest driver of change in soil carbon in New Zealand and the impacts on land use change on soil carbon between 1990 and 2016 was estimated by Whitehead et al. [23]. Management effects on soil carbon are less certain and inconclusive for New Zealand although this was limited to grassland soils only [24].

On-going monitoring is carried out on soil carbon (e.g., shallow sampling in SOE monitoring) and a recent large scale monitoring programme (National Soil Carbon Monitoring, NSCM [25]) has been established to sample approximately 500 sites across New Zealand in the dominant land use classes (e.g., Dairy, Sheep and Beef, Horticulture and Cropping). These sites will be visited multiple times over an extended period and will give information on any changes to carbon stocks to 60 cm depth. This programme will contribute to improving understanding on impacts of soil carbon change across New Zealand. However, it does not specifically address management effects on soil carbon.

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

At a national level the New Zealand Greenhouse Gas inventory [26] uses IPCC methodology and the CMS model to quantify change in SOC stock with land use change (0-30 cm depth). Within this CMS model, grassland soils are assumed to contain higher mean stocks of SOC than other land uses (e.g., annual cropping, perennial cropping, natural and planted forest). This change in SOC, through land use change, is assumed to occur over a 20-year period before a new steady state is reached. Furthermore, if any land use was to return to its original state the assumption is that the SOC would return to what it started as (e.g., reversible). However, there is limited evidence to support this assumption. There is also very little evidence to support the 20-year timeframe to reach a new steady state with some studies internationally demonstrating change can carry on for much longer.

The original 500 Soils project, from 1995-2001 constituted the inception of soil quality monitoring in New Zealand. Results from this initial project showed total C was depleted in soils under cropping land use [20-21]. Recent monitoring data showed that 26% of cropping sites monitored between 2014-2018 had total C lower than the target range [27-28]. This consistency between older and more recent data indicates that this loss of C under cropping has remained somewhat steady over the past ~30 years. There has been no overall improving trend or increases in soil carbon [28].

The pace or trajectory of change in the future 10-30 years is unknown and depends on the extent to which current management practices known to affect soil C change or remain the same [29]. Research indicates that some soils, particularly those under cropping, have a larger C deficit (due to their higher levels of C depletion) and therefore have the capacity to sequester more C [30]. Whether or not this eventuates will be dependent on land management practices.

The national soil carbon monitoring (NSCM) programme is currently underway with baseline sampling completed. This programme samples multiple productive land uses across New Zealand and assesses C stocks from 0-60 cm [25]. In time this programme will provide better information on trajectory and size of change for SOC. Currently, the baseline sampling has been completed but the data is not yet available.

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

Most regional councils in New Zealand monitor total C (%) under their soil quality monitoring programmes for SOE reporting. Environment Canterbury also have a wealth of data within their arable and pastoral programme. Total C has been one of the seven key soil quality indicators since the implementation of SOE monitoring in the early 2000s. More recently, HWEC has been more widely monitored since it has been proposed as a replacement for anaerobically mineralisable nitrogen, used as an indicator of microbial activity on soils [1; 31-32]. The NEMS [33] for soil quality and trace elements specifies a standard for sampling and analysis of total C. This specifies a shallow sampling depth of either 0-10 cm (primary method) or 0-15 cm (alternative method). Data are usually compared to provisional target values for this soil quality indicator, with the development of these target values described in a recent report [34].

Long-term field trials at Winchmore, Ballantrae and Tara Hills have also assessed and reported soil C and under grazed grassland in multiple reports and journal papers [35-38]. A review on changes in soil C under grassland soils summarising many of these trials and additional studies has been published [24].

More recently, New Zealand's National Soil Carbon Monitoring (NSCM) programme has been established with the aims of providing a benchmark for soil organic carbon stocks across agricultural land use classes in New Zealand and monitoring changes over time [25]. The initial benchmarking has been completed however the results are not yet available.

Several measures of soil C including total C, total organic C and C:N ratio are also measured in relation to productive requirements to inform farmers of their soil characteristics. The Fertiliser Association of New Zealand (FANZ) has produced a guide for cropping farms that mentions laboratory testing for soil carbon and C:N ratio [39].

As stated in A3, at a national level, the GHG inventory applies IPCC methodology and uses the CMS model to quantify change in SOC stock with land use change (0-30 cm depth). At a simple level, reference soil carbon stocks (0-30 cm depth) are quantified for each land use and these are applied to change in land use area quantified using spatial layers. This methodology assumes a linear change to the new reference stock over a 20-year period. Any change is considered reversible if land use is reverted.

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

For all direct soil measures, there is a need to access privately owned land to collect repeat samples for monitoring of this attribute. Landowners may be more, or less, willing to provide access to land for sampling and to have data from their land used to inform SOE or for other purposes.

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

Variables estimates provided by Regional Council scientists to MfE:

- $10,000 per year estimated by Marlborough Regional Council, broken down as: Chemical laboratory analyses of which total C is included, for ~20 sites/ soil samples. Two people sampling eight full time days per year.
- $85,000 total cost per year (pers comm Waikato Regional Council), broken down as follows: ~$1000 per sample/site for all seven basic soil quality indicators (including total C). For approximately 30 sites, one scientist spends approximately one third of their time on soil quality monitoring.
- $80-100,000 per year (pers comm Horizons Regional Council), for monitoring of the seven soil quality indicators, not including staff training and farmer outreach.
- $250,000 per year monitoring costs plus Regional Council soil scientists' time (unspecified, 5 staff in team) (pers comm Environment Canterbury).

Various measures of soil carbon are available from commercial laboratories, typically in combination with N or as part of a wider suite of different tests e.g., organic matter suite (includes Total carbon C:N ratio), with costs typically ranging between $25 to $75.

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

We are not aware of any monitoring being carried out by representatives of iwi/hapū/rūnanga. However, we know that soil carbon is of high interest to Māori through partnership with hapū/iwi entities. We know that Māori monitor soil health holistically, and that mātauranga Māori indicators are applied to the soil ecosystem rather than single attributes like soil carbon. See for example https://www.landcareresearch.co.nz/assets/Discover-Our-Research/Land/Soil-resilence/Maori_soil_health_research-v2.pdf.

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

Soil attributes are correlated in various ways with:

- Peatland/peat soil subsidence control: Drainage of Organic, or peat soils, results in large amounts of soil C loss [45-46].
- Bacteria composition: Cycling or organic matter in soils is dependent on soil microbiology, and therefore bacterial composition [47].
- Soil N and P: Through the C:N ratio of soils. Lower C:N ratios with high total N are associated with increased losses of N [48]. Loss of soil C results in reduced nutrient supply from organic matter [17], meaning agricultural land use may become reliant on inorganic fertilisers.
- As above other nutrients are also linked to SOC cycling, including P and S, therefore soil C is linked to other nutrients through stoichiometry of soil organic matter [5].
- Soil contaminants: The mobility and bioavailability of trace element contaminants in soils including Cd, Zn, As and Pb can be affected by soil C. Generally, increases in soil C result in decreased bioavailability of these trace elements [49].
- Surface erosion/runoff (and other erosion related attributes): Soil C contained in topsoils (and subsoils, if affected) is lost through erosion and runoff.
- Soil water storage, capacity and fluxes: Soil C is linearly correlated to aggregate stability, a measure of soil structural quality [16]. Reductions in soil C reduces soils' resilience to water stress/drought [17].

Part B—Current state and allocation options

B1. What is the current state of the attribute?

New Zealand soils contain moderate to high stocks of SOC (approximately 100 tC ha^{-1} in top 30 cm) compared to other countries partly due to soils being geologically young in development [50].

Current SOE monitoring only reports on SOC concentration of topsoil (mostly 0-10 cm) and knowledge now considers this depth to be inadequate for a true indicator of SOC change to depth. For example, shallow sampling as in SOE monitoring, will not capture any redistribution of SOC due to some tillage practices. Therefore, new programmes (e.g., NSCM) have been established to measure changes to soil carbon to depth (e.g., 0-60cm). However, the topsoil (0-10 cm) is generally where most of the root activity in agricultural crops will be contained, so this data is still useful as an indicator of soil health.

While we know that some land use and land management result in large losses of SOC, many of these studies have been focussed on specific regions or managements. The NSCM programme will give a better coverage of the state of soil C stocks across multiple land use and soil types throughout New Zealand. However, forest soils are not represented in this programme.

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

Within the GHG inventory methodology, reference states are assumed for each land use class. These states represent a mean stock of soil carbon within each land use across New Zealand. While there are some slight differences in this reference state driven by climate and soil differences, they are generally small. The benchmarking sampling of the NSCM could potentially be used to improve the reference states within these land uses and will provide better coverage of soils across New Zealand.

SOE soil quality monitoring of total C in soils under indigenous vegetation could potentially represent reference states for these measures in New Zealand soils, however there are few indigenous sites included in SOE monitoring, and where included, they may not necessarily represent undisturbed indigenous vegetation. Additional studies of soils under indigenous vegetation, including those on C stocks (rather than the total C measure), could add to this knowledge base.

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

'Target values' have been developed for use in regional council state of the environment reporting, with a review of the derivation of these values recently undertaken by Manaaki Whenua – Landcare Research [34]. These values were based on combining production optima and environmental considerations, although limited environmental data was available at the time of development [34]. These values vary with land use and soil order, with the current recommendations for total C concentrations in soils detailed in Table 1. The values in Table 1 are similar but not identical to those used in *Our Land 2021 [27]* (Table 2).

Table 1. Provisional soil quality target ranges for total C (%). The numbers in bold are generally used as the minimum target value. Table from Cavanagh et al. [34].

Soil type	Very depleted		Depleted	Normal	Ample
Allophanic	0.5	**3**	4	9	12
Semi-arid, Pallic, & Recent	0	**2**	3	5	12
Organic			excluded		
All other soil orders	0.5	**2.5**	3.5	7	12

Table 2. Minimum target values for total C used by MfE and StatsNZ in Our Land 2021 [27]

Soil order	Minimum target value (% gravimetric)
Pallic, Pumice, Raw, Recent, Semi-arid	2
Brown, Gley, Granular, Melanic, Oxidic, Podzol, Ultic	2.5
Allophanic	3
Organic	NA

NB: Anthropic soils not mentioned

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

There is very limited data on effects associated with soil carbon loss particularly in the context of tipping points or thresholds. There is limited evidence that production of certain crops decreases once SOC concentration gets below certain thresholds [51] However, these thresholds don't exist for New Zealand. Furthermore, New Zealand soils (e.g., Allophanic) which have high stocks of SOC may experience negative impacts on production and soil quality well above these global thresholds.

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

While soil carbon losses can be rapid (e.g., in cropping/fallow periods, [22; 52-53]) gains in soil carbon are very slow and difficult to quantify without an intensive sampling approach. Legacy effects influencing changes in SOC are likely to occur although evidence is limited, particularly within New Zealand.

Within the CMS model (used in the LUCAS framework) any change is assumed to occur, linearly, over a 20 year period before a new steady state is reached. There is evidence that changes in SOC are not always linear, and that changes can occur for much longer [24].

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

Māori utilise a holistic approach to assessing the condition of all parts of the soil ecosystem (and beyond). In addition to discussing this attribute directly with iwi/hapū/rūnanga, there is likely to be tikanga and mātauranga Māori relevant to informing bands, allocation options, minimally disturbed conditions and/or unacceptable degradation in treaty settlements, cultural impact assessments, environment court submissions, iwi environmental management and climate change plans, etc. Observations of ill-health of plants/animals/soil through a Māori lens may provide some insight for bands for this attribute.

Part C—Management levers and context

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

Drainage of Organic, or peat soils, results in large amounts peat subsidence and soil C loss [45-46].

Decreases in soil C can occur following change from pasture to cropping (e.g., as rotational cropping in pastoral systems) or through changes in management though the severity of this can vary with soil type [16, 54, 60]. Allophanic and Organic soils naturally contain more C than other soil orders and Allophanic soils appear to retain more soil C in the medium-long term following conversion to cropping [16]. However, losses of soil carbon were observed under Allophanic soils under more intensive management [54]. This loss of soil C is attributed to tillage practices on cropping farms, resulting in oxidation of C exposed to air, the period of time within cropping cycles that have no plant inputs (e.g., fallow period [22]), and or biomass removal under cropping (P. Mudge, pers. comm.). There have been two reviews summarising the current knowledge on the drivers of soil carbon change within New Zealand with very little evidence to demonstrate management practices that increase soil C [24, 29]. However, some management practices that have been observed to decrease soil C are irrigation and frequent cultivation for cropping [6, 19].

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

C2-(i). Local government driven

SOE monitoring is undertaken by Regional Councils, however the extent of interventions to effect change in soil C status, is largely limited to reporting soil quality monitoring results, which may include reporting back to the individual landowners on whose properties the sampling has been undertaken. Lower soil C concentrations have been reported in agricultural – namely cropping – soils since the commencement of the 500 Soils programme in 2000 [20-21].

C2-(ii). Central government driven

New Zealand has obligations to create an inventory and report on the greenhouse gas emissions as part of the UNFCCC. Within this inventory, SOC change due to land use change is quantified and reported.

New Zealand's national emissions reduction plan [55] includes very limited reference to soil carbon but acknowledges increasing knowledge on practices that may increase soil carbon (e.g., regenerative agriculture) and offset emissions will contribute to the net-zero emissions target for 2050.

C2-(iii). Iwi/hapū driven

As above, we note that hapū/iwi take a holistic approach to soil health monitoring. Iwi planning documents such as Environmental Management Plans and Climate Change Strategies/Plans may contain policies/objectives/methods seeking to influence soil health outcomes for the benefit of current and future generations.

C2-(iv). NGO, community driven

Voluntary carbon markets where SOC stocks are measured using measurement, reporting and verification (MRV) protocols to credit SOC sequestration. Multiple MRV protocols exist as summarised by Oldfield et al. [56]. To our knowledge, these are not being extensively used in New Zealand at present.

C2-(v). Internationally driven

GHG inventory reporting (mineral soil carbon change through land use change) as required as part of New Zealand's obligation to UNFCCC. Reducing losses of SOC are of benefit to New Zealand's reporting obligations. International markets may also play an increasingly important role in these indicators.

Part D—Impact analysis

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

Soil carbon is considered natural capital and is important for many soil functions and ecosystem services including food and fibre production, climate regulation through carbon sequestration in soil and nutrient cycling [12]. Substantial losses of SOC through lack of monitoring and management will likely contribute to the loss of soil structure and function impacting many ecosystem services.

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

Loss of SOC could cause economic impacts to agricultural/horticultural production. Loss of SOC would also be associated with increased loss of other nutrients within SOM (e.g., N, P, S) which could compromise food production and increase costs associated with providing these nutrients through fertiliser.

Losses of SOC also contribute to increasing atmospheric carbon dioxide concentrations which could be valued as the C credit cost required to offset this loss (e.g., using the NZ ETS).

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

Changes to soil temperature and moisture because of increasing air temperature and changes in rainfall distribution will impact the cycling and change in SOC. Respiration of SOC generally increases as soil temperature and moisture increase [57]. Recent studies have explored the response of microbial respiration of soil organic matter to better understand and predict soil carbon losses due to climate change [58-59]. This increase in SOC respiration may be partly offset by increases in C inputs to soil through greater plant production though evidence of this is unclear.

Loss of stored soil C will result in increased atmospheric CO_2, contributing to climate change, which is predicted to drive a positive feedback loop that intensifies further C losses from soils [18].

Microbial cycling of nutrients has been demonstrated to be affected by temperature variations [47].

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7.5 Landslide susceptibility mitigation

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Preamble: Landslide susceptibility describes the inherent properties of terrain which make it more or less susceptible to failure e.g., geology, slope angle, elevation, hydrological conditions, etc. It does not include the frequency of landslides, or the consequences (e.g., areas that may be impacted by landslides are not identified).

In practical terms the 'landslide susceptibility mitigation' attribute aims to reduce erosion from rainfall-triggered shallow landslides on land at high risk from this process by requiring councils to proactively define areas where susceptibility is high and then plan and monitor relevant mitigation such as soil conservation planting, afforestation and reversion.

It is not currently monitored nationally or regionally.

It would be difficult to focus specifically on this as an attribute as it risks being conflated with more general 'measures' of 'erosion susceptibility' and current progress towards reducing erosion in general. It (landslide susceptibility) varies widely across NZ making it less suitable as a national attribute, particularly for much of the South Island.

Landslide susceptibility is a specific subset of 'erosion susceptibility' which is a more encompassing term describing the susceptibility of land to some/all erosion processes, e.g., Highly Erodible Land (HEL – [1]; NZeem® [2]). Highly erodible land has been defined as land at risk of severe mass-movement erosion (landslide, earthflow, and gully) if it does not have protective woody vegetation". 'Erosion susceptibility' has been, and continues to be incorrectly used, to cover landslide susceptibility, mass wasting (as in HEL), and any or all erosion processes.

Because landslide susceptibility is lumped with other erosion processes, it is difficult to assess the state of knowledge of this specific attribute, particularly its control, i.e., separate it from more general erosion control. However, our knowledge of rainfall-triggered landslides being the dominant erosion process in many parts of Aotearoa is well established [3] as are the significant impacts that arise from events that trigger landslides [4, 5].

A well-established woody vegetation cover can reduce the number and density of landslides triggered by storms [6, 7, 8]. Thus, vegetation is the main control measure for most erosion processes, including rainfall-triggered shallow landslides, and this is well understood [7, 8]. Our knowledge of land specifically susceptible to rainfall-triggered landslides across Aotearoa is not well-advanced, i.e., there are no national maps nor agreed approaches for producing such layers. Statistical landslide susceptibility models have been developed for several regions [9], and when incorporating LiDAR-derived digital elevation models, promise to provide high spatial resolution of shallow landslide susceptibility. These data-driven models are likely to be the most suitable approach for developing national layers.

If such an attribute were to be considered, the current state would need to be defined at a suitable scale. A 'response' indicator such as 'area of works/area treated' (planting, afforestation, reversion, soil conservation planting) would then need to be defined to provide evidence that land that was susceptible was being treated and progress was being made against a baseline state.

State of knowledge of 'Landslide susceptibility mitigation' attribute: Medium / unresolved

Part A—Attribute and method

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

Soil erosion is a wide-spread and long-standing issue in New Zealand [3, and many others]. Rainfall-initiated shallow landsliding is the key/dominant erosion process in many regions of New Zealand [3]. However, its occurrence and effects are episodic in both magnitude and spatial extent.

It affects ecological health through reducing soil depth and integrity at source locations and through the deposition of sediment in receiving environments. It can affect human health indirectly via mental well-being, being a recurring natural hazard that impacts livelihoods, infrastructure and communities. Knowledge and understanding of these impacts are not well correlated other than in a general sense.

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

Landslide susceptibility is an inherent property of the landscape. Geology, steepness, vegetation cover, historical and current land use, and the exposure to hydrological factors such as intense rainfalls determine a location's susceptibility to rainfall-triggered shallow landslides. Landslide susceptibility is often conflated with erosion susceptibility, i.e., all erosion processes combined.

Statistical models are used to determine the probability of one location's susceptibility over another e.g., [10]. Such models are data driven and require an inventory of where landslides occur and where they don't and how each point relates to rock type, vegetation cover, slope steepness, etc.

Impacts can be local or regional and depend on the characteristics of the triggering rainfall event (and/or antecedent conditions). The evidence is strong that rainfall-initiated shallow landslides are a key environmental and societal issue e.g., [3, 10, 11] as is the evidence that vegetation is the primary method to reduce most, but not all impacts [6].

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

In some places, susceptibility has temporarily reduced in the last 60 years as steep pasture covered slopes have been treated with soil conservation plantings, allowed to revert, or afforested [6, 12]. In other areas the opposite has occurred as susceptible land remains without woody vegetation.

Vegetation removal including forest harvesting, can increase landslide susceptibility [13].

Climate change projections suggest increasing storminess in many regions indicating more occurrences of rainfall-triggered shallow landsliding will occur [14, 15].

Impacts are only partially or temporarily reversible. In short timescales (years to decades), closed canopy woody vegetation will reduce local incidence of landslides for small to moderate rain events. At decadal to century timescales and in the most severe rain events (e.g., cyclones like Gabrielle, Bola, etc) geomorphic thresholds are crossed and many landslides will be initiated, including those under a woody vegetation cover. Further, soil will take centuries to rebuild and soil loss is not easily reversible. Vegetation, weeds, grass, colonisers will establish quickly, weeks to months across shallow landslide scars and on deposits (woody vegetation longer). Impacts of pasture production and forestry production of landslides are well known [25, 26].

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

This attribute (*sensu stricto*) is not routinely monitored.

In regions where it might be monitored, it is usually combined with other erosion processes such as 'highly erodible land', 'erosion susceptibility' or other proxies such as 'bare land'.

Where it might be monitored, the metric is usually the spatial extent of 'highly susceptible' land and how much of the 'worst' classes have been, or are being treated, by either soil conservation planting, afforestation or reversion.

There is no consistent methodology in use and no nationally agreed monitoring methodology.

Some Councils assess 'bare ground', or the proportion of LUC classes that have changing vegetation coverage, or similar approaches as part of SOE monitoring. Other councils may report how much is invested annually in soil conservation programmes, or poles planted etc. None of these approaches are focused on land that is <u>specifically</u> susceptible to rainfall-initiated shallow landslides.

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

There are no major implementation issues for this attribute. While there is no consistent approach to assessment of land susceptible to rainfall-initiated shallow landslides and its treatment, remote sensing is clearly the most suitable approach for determining treated areas relative to a baseline state. As the baseline state does not exist, it is somewhat moot.

There is unlikely to be issues accessing private land unless there was a requirement for validation of remote-sensed information. Many remote sensing tools can provide resolution down to individual trees. The main barrier will be the cost of acquiring imagery and setting up automated processes to enable change comparisons. Another barrier will be the development of consistent and standardized data reporting, including systems and infrastructure to manage, store and report on monitored data.

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

It is not being done routinely therefore hard to assess.

At least one regional council undertakes a 5 yearly survey of changes in “erosion status” assessed through remote-sensed imagery coupled with local records, while others assess bare ground on an annual basis as a proxy for ‘erosion’. Some councils report on how many farms are treated with soil conservation works, others on the number of poles planted, etc.

If high resolution, cloud-free satellite imagery was regularly and cheaply available, algorithms could be developed to routinely assess different treatments, such as areas planted, counts of individual trees, etc. between one year and the next.

To date, most assessment of rainfall-triggered shallow landslides is manually done following a storm event and is therefore costly. Semi-and fully-automated approaches utilising satellite imagery for mapping landslides (or eroded area) are still in their infancy in NZ [16]. Separating shallow landslide erosion from other erosion types and the proportion of landslide-susceptible land treated or not, will be problematic.

The attribute would need to be monitored by consistent, spatial and standardised reporting of soil conservation work overseen by regional councils and/or central government to progress the evidence base that work is being done towards environmental improvement.

Some councils already report on assessments of proxies of erosion and the kinds and area of land treated with soil conservation plantings and are well positioned with data systems in place. Data standardisation across regional councils is likely to be achievable with direction, coordination, and funding.

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

Land slide susceptibility risk is of high interest to Māori (see, e.g., [38], Hōretireti Whenua Sliding Lands programme, etc.), although I am not aware of any monitoring of this attribute being carried out by representatives of iwi/hapū/rūnanga. GNS are working with Ngāti Porou representatives to understand detailing learnings from recent landslide events, co-design a landslide response framework for hapū, iwi and the community, and enhance natural hazard preparedness and response in Kura Kaupapa Māori/schools. See https://www.gns.cri.nz/research-projects/sliding-lands/

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

Shallow landslide erosion is usually grouped with other erosion processes in a “general erosion assessment”. It may or may not be correlated with the other listed land attributes.

During large rain events that initiate shallow landslides, gully processes are likely to be exacerbated particularly where shallow landslides transform into debris flows and are channelised within a gully or stream. In this situation, toe slope removal may initiate mass failure. Alternatively, in the absence of landslides being connected to gullies, concentrated runoff in the gully may scour the toe slopes resulting in 'streambank' failure (mass failure), i.e., in certain conditions there is a feedback loop between channel and slope.

Shallow landslide erosion is usually lumped with other erosion processes, e.g., gully erosion, surface erosion, and even bank erosion in assessments of erosion. In the LUC/Land Resource Inventory the key/dominant erosion process for that polygon is described along with its severity, e.g., in LUC as Ss soil slip, Da debris avalanche, Df debris flow [17], along with secondary erosion processes. In the NES-CF ESC, erosion processes are combined within one of 4 erosion susceptibility classes (the ESC is based on the national LUC).

Part B—Current state and allocation options

B1. What is the current state of the attribute?

Current state of landslide susceptibility is not well understood at the national scale. Understanding is not advanced enough for this to be used as an indicator.

Some North Island Councils (Gisborne, Hawkes Bay) have recently acquired high-resolution shallow landslide susceptibility layers derived from statistical landslide susceptibility models [10, 18, 19].

To be used as an indicator would require further development of statistical landslide susceptibility models that incorporate different geological rock types and empirical data from South Island regions before a national layer could be derived. Once a national layer was available, the areas deemed to be most susceptible could be monitored (annually or 5-yearly) to determine how much land had been 'treated' or was under a permanent tree cover relative to a starting baseline.

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

I am unaware of the existence of natural reference states for this attribute.

Removal of the original indigenous forest increased the susceptibility of the landscape to erosion (of many processes) across the motu. We no longer have a natural system - it is highly modified - vegetation was removed, cities were built, hydrological regimes were changed etc. Consequently, susceptibility to erosion from all processes has likely increased - by how much by when it is difficult to know as we start getting into broader landscape evolution concepts which operate over much longer time frames 10^2 to 10^4 years (e.g., [26]). Assessments in lakes and estuaries and flood plains of sediment build up as being a proxy for erosion that span 10s to 100s to 1000s of years provides a glimpse of this but it relates to all erosion and is therefore an estimate of what natural erosion might have been [27-29].

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

I am unaware of any existing numeric or narrative bands for this attribute.

Globally, landslide susceptibility models (and their outputs) are routinely developed to inform landslide hazard and risk assessments [26]. The aim of landslide susceptibility analyses is to assign different likelihoods for landslide occurrence and classify different spatial locations in different susceptibility levels. However, practical uses of susceptibility analyses are often limited by large uncertainties and inconsistencies of various input data, and difficulties to understand the different susceptibility maps based on numerous methods [26-27]. Thus, it is important to have sufficient empirical data from landslide inventories to underpin these approaches. These data are more commonly available in the North Island than South Island.

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

I am unaware of thresholds or tipping points that specifically relate to effects on ecological integrity or human health.

'Tipping points' exist but in a general sense. These are more correctly described as geomorphic thresholds and embody concepts of magnitude and frequency, and they are difficult to quantify.

Thresholds will vary from region to region, i.e., there is no 'one-size-fits-all' which is why we see different responses in different places to the same 'size' rain event.

The annual recurrence interval of a storm that will initiate shallow landslides is likely to vary by an order of magnitude across New Zealand.

Large storms/events such as Cyclone Bola and Gabrielle do have significant effects on ecological integrity and human health. We do not know enough to suggest what the minimum recurrence interval of an event will be that will result in 'significant' effects on ecological integrity, either locally, regionally, or nationally.

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

Although uncertain, lags and legacy effects are likely to be present for this attribute.

Vegetation removal creates a period in which landslide susceptibility increases. Typically, this is from a few years up to several decades.

The original clearance of indigenous vegetation off steep hill country 150 years ago clearly demonstrates this. Conversely, planting/reversion will with time reduce susceptibility on land prone to shallow landsliding for the smaller to moderate-sized events. Tree planting and reversion are the key treatments for such land.

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

As noted above, there are current studies underway within GNS that are exploring how mātauranga Māori informs landslide risk. In addition to discussing this attribute directly with iwi/hapū/rūnanga, there is likely to be tikanga and mātauranga Māori relevant to informing bands, allocation options, minimally disturbed conditions and/or unacceptable degradation in treaty settlements, cultural impact assessments, environment court submissions, iwi environmental management and climate change plans, etc.

Anecdotally, many iwi do not like pine forests or exotic trees as mitigation options for reducing shallow landslide susceptibility, preferring native forest [39].

Part C—Management levers and context

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

The relationship between erosion-prone land (including land that is susceptible to rainfall-triggered shallow landslides) and vegetation cover is well understood and has been for decades [11, 22, 23].

Past approaches have often been to afforest significant areas of marginal pastoral farmland with exotic pines to treat multiple erosion processes including land susceptible to rainfall-initiated shallow landslides. Alternatively, wide-spaced soil conservation plantings of poplars and willows is used where pastoral farming continues. Increasingly, planting of natives such as manuka, managed reversion, and permanent forestry are also seen as additional options [28, 29].

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

C2-(i). Local government driven

All Regional and Unitary Councils have or have had soil conservation or land management programmes that aim to reduce the amount of erosion (and sediment) in their regions. However, the size, scale and cost of these programmes varies widely. In some regions there is not enough resource available to treat the current state of erosion.

Afforestation, soil conservation planting, and reversion are the primary treatment methods.

C2-(ii). Central government driven

Current and past initiatives have been directed towards erosion control, e.g., East Coast Forestry Project, One Billion Trees, Hill Country Erosion Fund.

Central government funding is the primary source of funds for regional erosion control. It may be delivered through local government and/or iwi/NGO/catchment/industry initiatives.

C2-(iii). Iwi/hapū driven

Landslide risk and mitigations to prevent these occurring is of high interest to hapū/iwi, especially in the areas severely impacted by Cyclone Gabrielle (e.g., the GNS - Ngāti Porou partnership study outlined above. Iwi planning documents such as Environmental Management Plans and Climate Change Strategies/Plans may contain policies/objectives/methods seeking to influence erosion outcomes for the benefit of current and future generations.

C2-(iv). NGO, community driven

Unknown.

C2-(v). Internationally driven

Unknown.

Part D—Impact analysis

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

Not managing this attribute and erosion in general, would result in further losses to ecological integrity (sedimentation in rivers, wetlands, hydro dams, estuaries and oceans), reduced clarity in freshwaters, woody debris impacts of bridges, etc. It would also result in the degradation of hill country soils leading to reduced productivity [24, 25].

Increased erosion may also lead to a decline in farmer well-being and could lead to the further demise of rural communities and those who live in highly susceptible/erosion-prone areas, e.g., East Coast Māori communities.

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

North Island hill country farming communities are the most likely to be affected as this is the area where rainfall-induced shallow landsliding is more common [3]. Farming is marginal on many hill country properties [30]. If there were further requirements to plant more trees on farms it could make some properties uneconomic and lead to further rural decline. The Beef & Lamb and forestry sectors are linked, in a general sense, with soil loss and erosion, including rainfall-induced shallow landsliding, and their management, including social license to operate [31].

Horticulture on floodplains in Gisborne, Hawkes Bay and other regions are also impacted by erosion in catchments expressed as increased flood risk and sedimentation on flood plains (e.g., [32, 33].

Manawatu, Whanganui, Wairarapa, Hawkes Bay, East Coast, Northland, Te Tau Ihu, etc., are regions prone to rainfall-induced shallow landslides on steeplands (e.g., [1], [18]).

When a storm occurs cannot be predicted, nor can its spatial extent (other than in a near-time weather forecast). Thus, how often and where a locality is impacted becomes a matter for natural hazard and risk assessment. Current research by GNS, NIWA, and Manaaki Whenua Landcare Research is aimed at improving this, though there are no near-term national products available.

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

Climate change is projected to result in more storminess and thus more rainfall-triggered landslides [15].

Many hill country areas currently under pasture will require more trees to ameliorate landslide risk. Better classes of land could consider a change to silvopastoral systems (e.g., [34, 35] while the most susceptible parts of the landscape will require a permanent tree cover.

The challenge will be to embrace the diversity in NZ's landscapes and match land-use to both land capability and susceptibility to provide a mosaic of use and cover at a finer spatial scale than is seen today [36].

Large-scale afforestation with exotic species on the most landslide-susceptible terrain will merely repeat the failures of the past, e.g., slash issue (https://www.landcareresearch.co.nz/news/living-with-nature-for-a-sustainable-future/).

The pace of transition needs to increase faster than the perceived, modelled, or real changes arising from climate change [15].

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7.6 Gully erosion

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State of knowledge of the "Gully erosion" attribute: between Medium / unresolved and Good / established but incomplete

Medium-good state of knowledge because we understand the process, there are some studies, and there is agreement on what gullies are, how they are treated, and the effectiveness of treatment. But there is limited to no spatial coverage of either gully location or 'areas' of gully erosion (other than general noted presence within a polygon in the NZLRI) which limits this as a national attribute.

In practical terms the 'gully erosion protection' attribute would aim to reduce gully erosion from the processes that contribute to its development or enlargement. This would require councils to define areas where gullies and/or gully erosion exists or has a high likelihood of developing. Councils would then plan and monitor relevant mitigation towards reducing the activity or severity of erosion within a gully, or the prevention of gullies developing or enlarging.

Part A—Attribute and method

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

Soil erosion is a wide-spread and long-standing issue in New Zealand (Basher 2013; and many others). Gully erosion (including tunnel gully erosion) is not as ubiquitous as some other erosion processes in many regions of New Zealand, though it is significant in some, e.g., East Coast, Manawatu-Wanganui, Taupoūhō, Marlborough, Banks Peninsula (Basher 2013) (See Figure 1). Gully erosion is caused by runoff and seepage from rainfall and storms and is thus 'episodic'. The activity and severity of erosion within gullies is generally related to the intensity and duration of rainfall as is their initial development, though land disturbance is also important. Gullies are usually local features in the landscape and may be restricted in terms of size, severity and spatial extent (Basher 2013; Marden et al., 2017).

The protection of land with existing gully erosion and/or land in which gullies are likely to develop is largely achieved with vegetation and runoff controls. It, like other erosion process control, benefits the ecological integrity of waterbodies by reducing sediment generation and transport (Frankl et al., 2021).

Gully erosion, like other erosion processes, also affects ecological integrity through reducing soil depth and integrity at source locations (reduction of soils fertility, decreases in water-holding capacity, and decline in ecosystem function) resulting in making land more challenging to manage (Rosser & Ross 2011; Walsh et al., 2021). Gully erosion, like other erosion, can have adverse effects on the environment, including habitat destruction, loss of biodiversity, and degradation of water quality (Larned et al., 2019). Sediment runoff from erosion, including gully erosion, can smother aquatic habitats, disrupt stream ecosystems, and contribute to algal blooms in water bodies (Ryan 1991). It can affect human health indirectly via mental well-being, being a recurring and permanent feature in the landscape that can impact farming and some infrastructure. Knowledge and understanding of these impacts are not well correlated other than in a general sense with erosion and sedimentation and their impacts on freshwater and marine ecosystems (e.g., Cavanagh et al., 2014).

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

A gully is a landform created by running water, mass movement, or commonly a combination of both eroding into soil or other relatively erodible material, typically on a hillside or in river floodplains or terraces (e.g., Poesen et al., 1998; Day & Shepherd 2019).

Gully erosion is predominantly driven by fluvial processes, i.e., removal of material by channelised running water. Sheet or surface erosion may lead to rills which in turn may develop into gullies. Tunnel gullies are a special form in which subsurface flow or piping leads to collapse at the surface. Gullies may be small linear features of a few 10s of square metres up to large 'amphitheatre-like' 'badass' gullies of 10s of hectares (Marden et al., 2017).

Gullies are characterised by a distinct headscarp or headwall, generally have steep sides, and tend to enlarge headward or upstream rather than laterally.

Gully erosion is a significant issue in New Zealand affecting both rural and urban areas and is but one of a suite of erosion processes (Figure 1). It is more common in some regions than others, e.g., in the soft rock hill country of the East Coast North Island, on crushed argillite and mudstone, and in the North and South Island mountainlands (Basher 2013). Human activities such as deforestation, agriculture, urbanization, and improper land management practices can exacerbate erosion, including gully erosion, by increasing surface runoff and soil disturbance (Dotterweich 2013; Stats NZ 2018)

Gullies are landscape features resulting from erosion. Geology (including rock type and faulting and fracturing), steepness, vegetation cover, historical and current land use, and the exposure to rainfall may determine where a gully is formed and how big it becomes (Basher 2013). Gully erosion is often combined with other erosion processes in reference to erosion, i.e., the term 'erosion' is generally used as an umbrella term to describe multiple erosion processes. The evidence is strong that gullies, as but one of several erosion processes, through the production of sediment impacts ecological integrity (Larned et al. 2019). The evidence is also strong regarding how to treat them, though effectiveness is largely qualitative and anecdotal (Phillips et al., 2020).

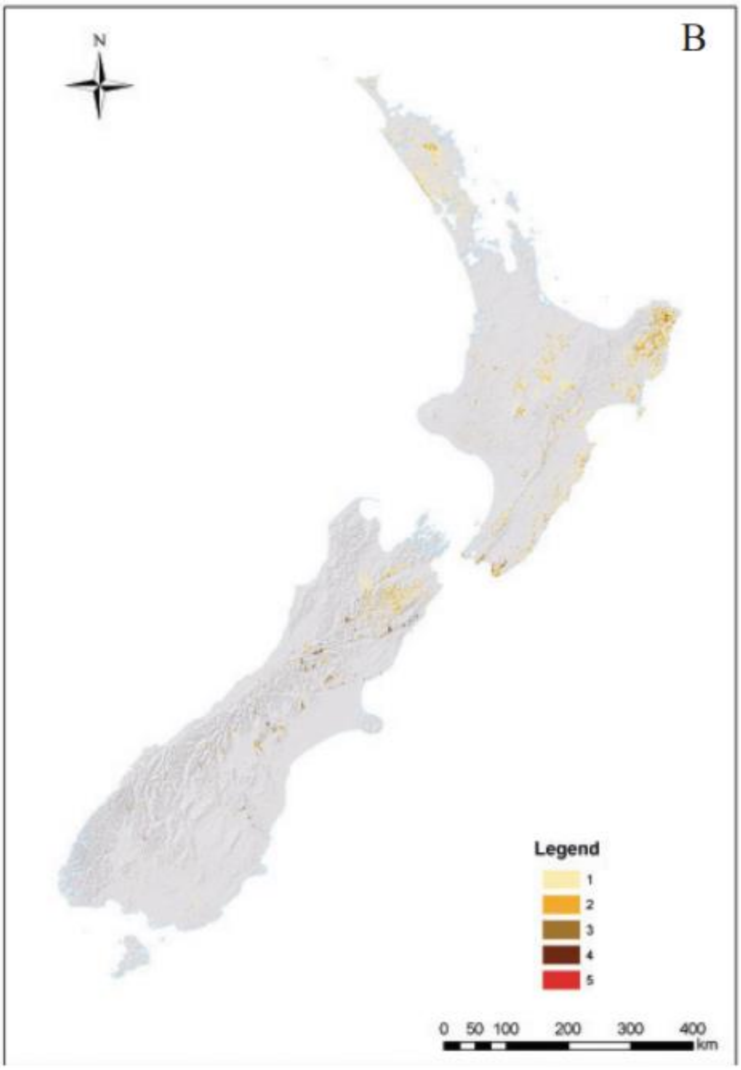

Figure 1. Distribution and severity of gully erosion in New Zealand derived from the New Zealand Land Resource Inventory (from Basher 2013). The lighter colours represent lower severity.

Mapping (on the ground, aerial photography or remote sensing) is used to identify gullies in the landscape. These may be mapped in terms of spatial extent, counted and their activity/severity assessed. There is no national nor regional inventories of gullies. Land in which gullying is present and/or has a high potential to develop will be recorded in the Land Use Inventory underpinning the national Land Use Capability mapping (Lynn et al., 2021) (see Figure 1). Gully inventories developed as part of science projects exist for some places like the Gisborne-East Coast region (Michael Marden pers. comm), though these are not publicly accessible.

Techniques such as soil conservation planting, afforestation, and retirement/reversion or other effective land management techniques are used to reduce or limit gully erosion (Basher 2013; Thompson & Luckman 1993; Phillips et al., 2008). Structural measures to lessen scouring such as drop structures, debris dams, rock chutes and some vegetative techniques can also be used.

Gullies can be assessed by number, size, severity, likelihood of stabilisation, and types of land prone to gullying.

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

In some localities and regions, gully erosion has reduced in the last 50 years as steep pasture covered slopes have been treated with soil conservation plantings, allowed to revert, or afforested (Marden et al., 2011; 2014). In other areas the opposite has occurred as new gullies have developed, and others have enlarged (Marden et al., 2017). Apart from local knowledge, there is unlikely to be any quantitative data on the pace and trajectory of change in this attribute.

Climate change projections suggest increasing storminess in many regions indicating a concomitant increase in erosion is likely. How this will preference gully erosion over other erosion processes is unknown. As rainfall is a key driver of gullying, gully erosion, where it exists, is likely to increase with increases in rainfall (Basher et al., 2020; Neverman et al., 2023).

Impacts are only partially or temporarily reversible. In short timescales (years to decades), closed canopy woody vegetation will reduce and "shut down" some gullies as will treatment with soil conservation (Marden et al., 2014; 2017). At decadal to century timescales and in the most severe rain events (e.g., cyclones like Gabrielle, Bola, etc) geomorphic thresholds are crossed and new gullies will be initiated, existing gullies may be enlarged, and old 'stable' gullies may be re-activated including those under a woody vegetation cover (Marden et al., 2017).

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

This attribute (*sensu stricto*) is not routinely monitored either nationally or regionally. In regions where it might be monitored, it is usually combined with other erosion processes such as 'highly erodible land', 'erosion susceptibility' or other proxies such as 'bare land'.

Gullies have been mapped in the Gisborne-East Coast region (Marden pers. comm) and recently re-mapped post Cyclone Gabrielle (for MPI). However, this is not monitoring *per se* but two snapshots in time of gully distribution in this region.

Where it could be monitored, the metric would be the number of gullies or the spatial extent of gullies (area). In terms of treatment monitoring, it would be the number of gullies whose activity status changed from active to inactive, or a spatial measure of land with or prone to gullying being treated. Frequency of monitoring less than 5 yearly would not be warranted as the changes are time dependent.

There is no consistent methodology in use and no nationally agreed monitoring methodology. Some Councils assess 'bare ground', or the proportion of LUC classes that have changing vegetation coverage, or similar approaches as part of SOE monitoring. Others may report how much is invested annually in soil conservation programmes, or poles planted, or the number of farm plans +/- works, etc. None of these approaches are focused <u>specifically</u> on gullies or land susceptible to gully erosion.

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

There do not appear to be any major impediments or implementation issues for this attribute. While there is no consistent approach to assessment of gullies, gully erosion, and gully erosion mitigation, remote sensing is likely to be the most suitable and cost-effective method for determining gully numbers, land susceptible to gully processes, or treated areas relative to a baseline state. As the baseline state does not exist, it is somewhat moot.

There is unlikely to be issues accessing private land unless there was a requirement for validation of remote-sensed information. Many remote sensing tools can provide resolution down to individual trees or sub-metre. The main barrier will be the cost of acquiring imagery and setting up automated processes to enable change comparisons. As with other 'erosion monitoring', another barrier will be the development of consistent and standardized data reporting, including systems and infrastructure to manage, store and report on monitored data.

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

To our knowledge, monitoring of this attribute is not being done routinely anywhere, therefore it is hard to assess costs.

At least one region (East Coast) has whole or partial spatial coverage of gully distribution (snapshot in time of gully locations), though this information is not easily accessible (M. Marden pers. comm).

Some councils report on how many farms are treated with soil conservation works (which would include treatment of active gullies if present), others on the number of poles planted, etc.

If high resolution, cloud-free satellite imagery was regularly and cheaply available, algorithms could be developed to routinely assess gully distribution and their treatment such as areas planted, counts of individual trees, 'activity' of gullies, etc., between one time period and another. Given issues around change detection, assessment of change in less than 5 years would not be warranted or practical.

To date, most assessment of gullies (similar to other erosion features) is manually done using aerial photography/satellite imagery and field mapping and is therefore costly. Semi-and fully-automated approaches such as OBIA (Object based image analysis) utilising satellite imagery for mapping gullies are not yet commonly used in NZ, though have been tried (Hölbling et al., 2016; Hölbling et al., 2022; Abad et al 2023). Separating gully erosion from other erosion types and the proportion of land treated or not, will be problematic. Bare ground has been used as a proxy for erosion and/or erosion risk (North et al., 2022; Norris & Wyatt 2023) the corollary of which would be a reduction in bare ground.

The attribute would need to be monitored by consistent, spatial and standardised reporting of numbers of gullies, areal extent of those gullies, activity status, and the amount of soil conservation work and treatment by regional councils and/or central government to progress the evidence base that work is being done towards environmental improvement and gully erosion was reducing.

Some councils already report on assessments of erosion 'proxies' and the kinds and area of land treated with soil conservation plantings and are well positioned with data systems in place. Data standardisation across regional councils is likely to be achievable with direction, coordination, and funding and/or using a platform such as LAWA (https://www.lawa.org.nz/).

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

I am not aware of any monitoring being carried out by representatives of iwi/hapū/rūnanga.

However, erosion is of very high interest to Māori and various hapū/iwi are focused on monitoring erosion and mitigating risk. See for example, the Waiapu River Restoration project (Led by Te Rūnanganui o Ngāti Porou) that focuses on erosion in the Waiapu catchment https://www.ngatiporou.com/nati-news/the-waiapu-river-restoration. See also the Waiapu Koka Huhua initiative https://ourlandandwater.nz/wp-content/uploads/2022/02/TMOTW-Case-Study-Waiapu-Kokahuhua.pdf

Successful erosion control within the catchment is required to achieve the cultural aspirations of Ngāti Porou (Scion, 2012). Measures of erosion (including visual observations and drone aerial measurements) are just parts of a holistic approach to assessing the state of the catchment.

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

Gully erosion is usually grouped with other erosion processes in a "general erosion assessment".

During rain events that initiate shallow landslides, gully processes are likely to be exacerbated particularly where shallow landslides transform into debris flows and are channelised within a gully or stream. In this situation, toe slope removal may initiate mass failure. Alternatively, in the absence of landslides being connected to gullies, concentrated runoff in the gully may scour the toe slopes resulting in 'streambank' failure (mass failure), i.e., in certain conditions there is a feedback loop between channel and slope (Ionita et al., 2015). However, gully processes operate whenever it rains rather than only in extreme events.

Gully erosion is usually lumped with other erosion processes, e.g., shallow landslide erosion, surface erosion, and even bank erosion in assessments of erosion (e.g., New Zealand Empirical Erosion Model (NZEEM), Highly Erodible Land (HEL)). In the Land Use Capability (LUC)/Land Resource Inventory the key/dominant erosion process for that polygon is described along with its severity, e.g., in LUC as G gully erosion (Lynn et al., 2021), along with secondary erosion processes. Severity is rated in 6 classes based on area, i.e., size of gully with 'slight' being < 0.05 ha, and extreme being > 5 ha (Lynn et al., 2021). A special form of gully erosion – tunnel gully is referred to as T and its severity is ranked based on the percentage of the mapping polygon that has this erosion process.

In the National Environmental Standards for Commercial Forestry (NES-CF) Erosion Susceptibility Classification (ESC), all erosion processes are combined within one of 4 erosion susceptibility classes (the ESC is based on the national LUC).

Part B—Current state and allocation options

B1. What is the current state of the attribute?

Current state of gully erosion is not well understood at the national scale (the process is well understood and locally/regionally its distribution might be). Understanding is not advanced enough for this to be used as a national indicator, though it has potential in some regions where it is more prevalent, e.g., Gisborne–East Coast. At the regional scale, some regional councils will have a good idea where the large gullies are, but they may not monitor the development of new gullies or the activity of existing gullies.

To be used as an indicator would require establishment of a baseline state for each region of gully number, their area (individually and/or collectively), their activity status and their treatment. Once a national layer was available, the numbers/areas could be monitored (5-yearly) to reduction in gully area (treated) relative to the starting baseline.

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

I am unsure of the existence of natural reference states for this attribute. A pre-European reference state, while potentially attractive as a natural reference state, would be difficult to quantify and would not be attainable in contemporary New Zealand. Gully erosion would have existed in New Zealand pre-Europeans, but it was the clearance of indigenous forest from 'hill country' in both islands for farming that exacerbated all erosion types, including gully erosion (Basher 2013).

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

I do not know of any existing numeric or narrative bands for this attribute.

Globally, gully erosion is a problem in some countries more than others. For example, it is a key erosion process in many parts of Australia, China, Spain etc., (Boardman et al., 2022). However, our understanding on spatial patterns and rates of gully erosion remain little understood (Poesen et al., 2003).

Gully erosion, if it is monitored, is usually monitored using change quantification techniques (Aber et al., 2010). In Europe, attempts have been made to integrate a soil erosion module into the Land Use/Cover Area frame statistical Survey (LUCAS) Topsoil Survey (Borrelli et al., 2022).

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

I do not know of any thresholds or tipping points, but there are some "rules of thumb" which relate to the size of gullies in some regions and whether they can be treated or not (Marden et al., 2011; 2017; Marden & Seymour 2022).

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

Yes, lag times and legacy effects are likely. Vegetation establishment in the margins or within a gully and/or its contributing catchment may take some years to become effective and shut down the gully, similar to other vegetative approaches. Once sufficient density of vegetation is established or canopy closure occurs, the effectiveness of rainfall will be reduced. Some gullies may only be partially shut down and continue to contribute small amounts of sediment even though they appear to be successfully treated. Non-vegetated measures may be immediately effective or not, i.e., there is likely to be variability in effectiveness of treatment depending on what is implemented and the activity of the gully itself. Hard engineered mitigations are not common in New Zealand.

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

In addition to discussing this attribute directly with iwi/hapū/rūnanga, there is likely to be tikanga and mātauranga Māori relevant to informing bands, allocation options, minimally disturbed conditions and/or unacceptable degradation in treaty settlements, cultural impact assessments, environment court submissions, iwi environmental management and climate change plans etc.

I suspect that many iwi will see eroded gullies as being a scar on Papatūānuku. Continual on-going bleeding of sediment from the large gullies will not be acceptable. In terms of gully treatment, not all vegetation solutions, i.e., species, may be agreeable to iwi.

Part C—Management levers and context

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

The relationship between erosion-prone land (including land that is susceptible to gully processes or has gullies on it) and vegetation cover is moderately well understood (Basher 2013, etc). In the Gisborne - East Coast region gullies are characteristic of both forested and grassed landscapes (Parkner et al., 2006; 2007).

Various strategies and techniques are employed to avoid, manage and mitigate gully erosion in New Zealand (Basher 2013). These may include revegetation of gully margins (and/or contributing catchment area) with exotic or native plants to stabilize soil (Marden et al., 2011), construction of targeted erosion control structures such as check dams (Howie 1968), gabion baskets, rock riprap and rock bunds in concentrated flow zones to slow water flow, and trap sediment and implementation of sustainable land management practices to reduce surface runoff and soil disturbance (Phillips et al.,, 2020). The latter include measures that increase topsoil resistance and the use of vegetation barriers (Frankl et al., 2021). Once deeply incised, the development of gullies may be partially controlled by diverting runoff away from the channel, but this comes at the risk of relocating the problem (McIvor et al., 2017).

Terracing and contouring are also used to help reduce the speed and volume of water flow, allowing it to infiltrate the soil rather than causing erosion. This is not commonly done in NZ, though has been trialled in the past. It is more commonly related to significant earthworks associated with urban developments or roading (Leersnyder et al., 2018).

Prevention is much more effective than repair. New Zealand experience suggests that once erosion is into the bedrock it is very difficult to get trees (or any other plant material) established. If gullies are treated when they are small (< 1ha) the likelihood of treatment leading to reduced erosion and soil loss is high (see Figures2, 3, 4 – Marden et al., 2011; 2017; Marden & Seymour 2022). When they get larger (threshold > 10 ha in area, commonly called 'badass' gullies) they are much more difficult if not impossible to treat. Large gullies cannot be repaired, e.g., Tarndale Slip, Barton's Gully on the East Coast (Figure 4) (Marden et al., 2017).

The contributing catchment surrounding the gully needs to be retired from grazing and planted in closed canopy trees to reduce the effective runoff, or if the gully is small, soil conservation planting along the edge and within the gully may stabilise it. Eroding surfaces within the gully likely need to be repeatedly planted with willow wands to create a vegetated surface (McIvor et al., 2017). Structures such as debris and check dams in association with planting is another 'bioengineering' treatment commonly used in the past and but less so now (Phillips et al., 2020).

Figure 2. Typical East Coast linear gully showing untreated state (left) and treated state (right) with paired poplars and/or willows.

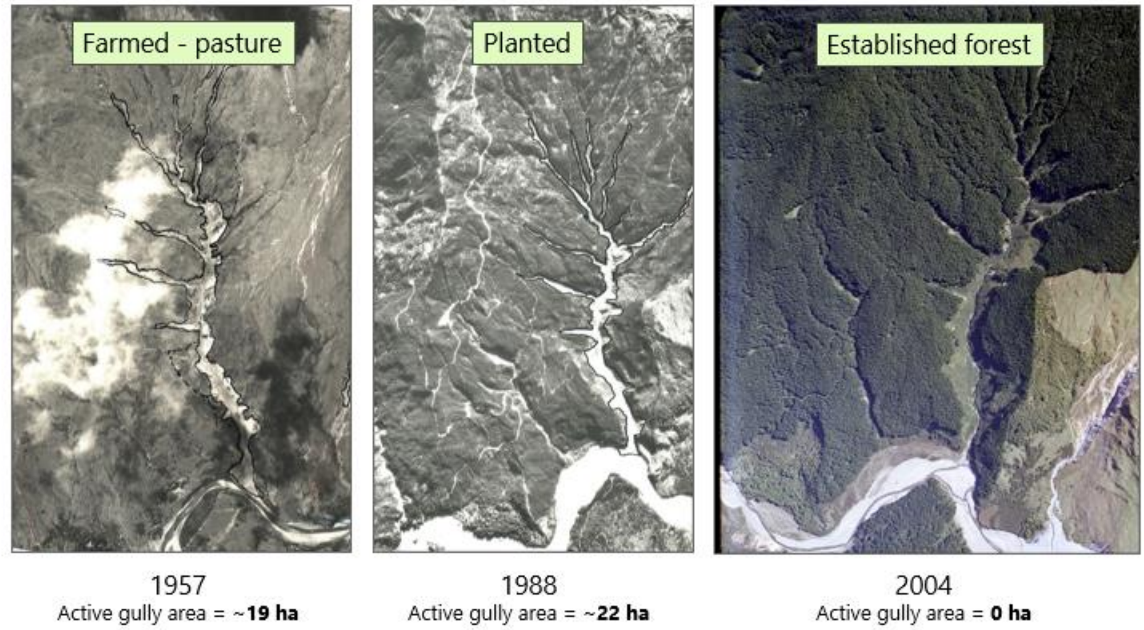

Figure 3. East Coast (Owhena, North Island, New Zealand) gully showing effective treatment with exotic forest (Marden pers. comm.).

Figure 4. Large amphitheatre or badass gullies from the East Coast region. Similar photos can be found in Marden et al. (2017).

Like other aspects of erosion control, gully treatment or protection is expected to improve the ecological integrity of land and water bodies as described in A2.

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

C2-(i). Local government driven

All Regional Councils have soil conservation or land management programmes that aim to reduce the amount of erosion (and sediment) in their regions from different erosion processes including gullies. However, the size, scale and cost of these programmes varies widely. In some regions there is not enough resource available to treat the current state of erosion and therefore some gullies will go untreated.

Afforestation, soil conservation planting, and reversion are the primary treatment methods though gully treatment may involve a range of other methods (see C1 above).

C2-(ii). Central government driven

Current and past initiatives have been directed towards more general erosion control which includes gully erosion control, e.g., East Coast Forestry Project, One Billion Trees, Hill Country Erosion Fund. I am unaware of any funding specifically targeting gully erosion treatment.

C2-(iii). Iwi/hapū driven

Erosion risk and mitigations to prevent erosion are of high interest to iwi/hapū/rūnanga, especially in the areas severely impacted by Cyclone Gabrielle (for example). Iwi planning documents such as Environmental Management Plans and Climate Change Strategies/Plans may contain policies/objectives/methods seeking to influence erosion outcomes for the benefit of current and future generations.

C2-(iv). NGO, community driven

Unknown.

C2-(v). Internationally driven

Unknown.

Part D—Impact analysis

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

Not managing this attribute (or erosion in general) would result in further losses to ecological integrity (sedimentation in rivers, wetlands, hydro dams, estuaries and oceans), reduced clarity in freshwaters, etc., particularly in the areas where gully erosion is common. It would also result in the continuing degradation of hill country soils leading to reduced productivity, though unlike pasture production recovery on shallow landslides (Rosser & Ross 2011) no such information exists for gullies.

Not managing this attribute (and erosion in general) may lead to the further demise of rural communities and those who live in highly susceptible/erosion-prone areas, particularly East Coast Māori communities. It might be seen as an affront not to treat this type of erosion, as unlike other erosion forms, it can be identified, is specific to a location, and its severity can be assessed.

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

North Island hill country farming communities are the most likely to be affected. Farming is marginal on many hill country properties, including those where gullies exist. If there were further requirements to plant more trees on farms it could make some pastoral properties uneconomic and lead to further rural decline. The Beef & Lamb and forestry sectors are linked with soil loss and erosion and their management, particularly on steeplands and hill country in North Island.

Horticulture on floodplains in Gisborne, Hawkes Bay and other regions are also impacted by erosion in catchments expressed as increased flood risk and sedimentation on flood plains, of which gully erosion is a part, and in some localities can be significant.

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

Many hill country areas currently under pasture with or without active gullies will require more trees. Better classes of land under agriculture could consider a change to silvopastoral systems (Mackay-Smith et al. 2024), while the most susceptible parts of the landscape will require a permanent tree cover. The challenge will be to embrace the diversity in NZ's landscapes and match land-use to both land capability and susceptibility to provide a mosaic of use and cover at a finer spatial scale than is seen today (PCE 2024).

Large-scale afforestation with exotic species on the most susceptible terrain will merely repeat the failures of the past, and could give rise to future unintended consequences, e.g., slash issue on steep Tertiary hill country on the East Coast.

The pace of transition needs to increase faster than the perceived, modelled, or real changes arising from climate change (Neverman et al., 2023).

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7.7 Surface erosion/runoff control

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State of knowledge of the "Surface erosion / runoff control" attribute: Good / established but incomplete

Good state of knowledge because we understand the surface erosion process, there are quite a few studies that have been carried out across the country and there is agreement on what surface erosion is, and what the benefits are of reducing it. Similarly, runoff control. There is, however, little to no spatial coverage of either current active surface erosion or of land where past areas of surface erosion have been treated. Surface erosion may be persistent over time, or it may change rapidly. Bare ground has been used as a proxy for surface erosion but not all bare ground is eroding. This limits this attribute as a potential national attribute.

Part A—Attribute and method

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

The control of runoff and surface erosion can benefit the ecological integrity of adjacent waterbodies in several ways. Principally this is aimed at reducing soil loss and sediment deposition in receiving environments and avoiding sediment affecting drinking water and water quality, in general, which may be an issue if contaminants such as E. coli 'attach' to sediment particles (Davies-Colley et al., 2018). Reductions in water clarity also affect swimming quality and may mask submerged hazards posing risks to swimmers (West et al. 2015)

Surface erosion is a natural geomorphic process (along with other erosion processes) which occurs on all land surfaces to varying degrees (Basher 2013). Land disturbance (earthworks) and land use practices (ploughing, tillage, grazing, etc) can increase surface erosion (Basher 2013).

Unlike mass movement processes (landslide/soil slip, slump, earthflow), surface erosion involves the movement of a thin layer of particles across the ground by water, wind or gravity. There are several sub-process types – splash, sheet, and rill erosion (Morgan 1986).

The impacts of surface erosion (and all erosion) are loss of soil and productive capacity on land (Rosser & Ross 2011) and impacts on waterbodies via changes in water clarity and deposition of sediment in the beds of rivers, lakes and estuaries (Ryan 1991; Gluckman 2017).

The extent of benefit afforded by runoff control and reduction in surface erosion on ecological integrity and human health is spatially and temporarily variable and depends on many factors (landscape, climatic, biological, and land use). Factors that affect control performance include land use and practice(s), degree of physical disturbance, vegetative cover, topography, rainfall intensity and duration, land use legacy, exposure time, etc., (Basher 2013).

Where erosion and sediment control (ESC) practices have been implemented, the ecological integrity and health of adjacent waterbodies is expected to improve, especially locally. However, complete control is rarely achieved as many ESC practices are not completely effective, i.e., their treatment performance is always less than 100% (Basher & Moores 2016; Phillips et al. 2020).

Various studies have measured surface runoff (and its contaminants e.g., nutrients, sediment) to help quantify the effects on water quality, from plot-scale field experiments (e.g., Smith & Monaghan 2003; McDowell et al. 2005), performance of straw mulch at earthworks sites (e.g., ARC 2000), in-situ rainfall simulation from within paddocks (Russell et al. 2001), and paddock-scale small catchments (e.g., Monaghan et al. 2017). Surface runoff measurements from plot-scale field experiments can be highly variable and dependent on many factors. These types of measurements in plot studies usually require material (e.g., wood, metal, plastic) to be inserted in the soil, to divert flows so they can be measured and collected at the downhill point and contain a known area of ground. Other studies have measured surface runoff (and its contaminants) from small catchments to evaluate control measures, such as sediment traps in agriculture, detainment bunds, etc (Levine et al. 2021; Smith & Muirhead 2023) and sediment retention via silt fences, vegetated buffers, decanting earth bunds and sediment ponds (e.g., Winter 1998; Babington and Associates 2004) in urban earthworks.

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

It is well known that land clearance and land use practices can disturb soils and thus expose the soil surface to wind, rain and gravity. Loss of soil by erosion (including surface erosion) will have contributed to the degradation of freshwater and downstream receiving environments by enabling sediment to enter water bodies where it affects water clarity and is deposited on the beds of streams, potentially smothering habitat. It will continue to do this wherever bare soil or disturbance coincides with rainfall, i.e., it is almost a ubiquitous process, though the amount of soil eroded and transported will vary widely.

Awareness of soil erosion and the need for soil conservation became a matter of national concern by the 1940s resulting in the passing of the 1941 Soil Conservation and Rivers Control Act and the establishment of catchment boards to manage erosion and sediment problems. It was also the time when a wide variety of techniques were developed for controlling erosion.

In the urban environment Auckland Regional Council published a set of ESC guidelines for earthworks in 1995 that was significantly revised and published as TP90 in 1999 (Auckland Regional Council 1999). TP90 has formed the basis of subsequent ESC guidelines across New Zealand. Specific ESC guidance has also been produced for the horticulture industry (Franklin Sustainability Project 2000; Barber & Wharfe 2010; Barber 2014) and the forestry sector (e.g., Bryant 2007; New Zealand Forest Owners Association 2007; Gilmore et al. 2011).

In the rural environment, emphasis has been on biological erosion control using trees and plants because of its relatively low cost and its effectiveness, particularly in reducing the incidence of rainfall-triggered shallow landslides (Phillips et al., 2020). There is generally less focus in New Zealand's rural environment on surface erosion than other erosion processes, in part because the volume of soil lost by other processes is orders of magnitude greater. However, focus has intensified in recent years on surface erosion and runoff control as intensive grazing and winter forage cropping on both hilly and flat land have bought these issues to the public's attention (Donovan & Monaghan 2021; Monaghan et al. 2021). With such practices the ecological impacts tend to be more 'continuous' or chronic rather than episodic or acute as the rainfall required to result in 'muddy' water from surface erosion will be significantly less than that to trigger a mass movement.

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

Land use and land practices, together with rainfall, are the primary drivers of surface erosion (Morgan 1986). Land slope is also a significant driver. Surface soil erosion will also vary with soil type (Basher 2013; Lynn et al., 2021).

Climate change projections suggest increasing storminess in many regions indicating a concomitant increase in erosion is likely (Neverman et al., 2023). How this will preference surface erosion over other erosion processes is unknown, though if rainfall becomes more intense, it is likely that surface erosion will increase.

Surface erosion has been included in several erosion models and in models aimed at understanding climate change impacts, e.g., Basher et al. 2020; Neverman et al. 2023; Donovan 2021.

Historically, erosion modelling focused on surface erosion processes and many models had/have at their heart the Universal Soil Loss Equation (USLE – Wischmeier & Smith 1978) or variants thereof. The USLE consists of an empirical equation which calculates the mean annual soil-erosion rates on agricultural slopes as a function of six variables (R, precipitation factor; L, slope length factor; S, slope factor; C, vegetation cover and tillage factor; K, soil-erodibility factor; P, erosion-protection factor).

The SedNetNZ sediment budget model (Dymond et al. 2016) was developed to represent the range of erosion processes that occur in New Zealand (e.g., shallow landslides, earth flows, gully erosion, surficial erosion, and stream bank erosion). This makes the model more suited to New Zealand's diverse landscape and environmental management and planning needs compared to models like RUSLE (the Revised Universal Soil Loss Equation) or SWAT (Soil and Water Assessment Tool) which largely focus on surface erosion processes. For example, RUSLE represents a limited range of erosion processes (sheet and rill erosion), and SWAT (Soil and Water Assessment Tool) uses MUSLE (the Modified Universal Soil Loss Equation) to model hillslope erosion but does not include mass movement processes such as landslides, which is a significant erosion process in New Zealand.

SedNetNZ is not available nationally. National erosion models include the Highly Erodible Land model (HEL – Dymond et al. 2005) and New Zealand Empirical Erosion Model (NZEEM, Dymond et al. 2010) but these include all erosion processes. Donovan (2021) provided the first national-scale surface soil erosion model based on RUSLE and found surface erosion rates for winter-forage paddocks (11 t ha^{-1} y^{-1}) were substantially higher than pastoral grasslands (0.83 t ha^{-1} y^{-1}), woody grasslands (0.098 t ha^{-1} y^{-1}), forests (0.103 t ha^{-1} y^{-1}) and natural soil production rates (≤1–2 t ha^{-1} y^{-1}). Model results were validated with empirical measurements from sediment traps, sediment cores, and chemical fingerprinting. Further, the study suggested that surface erosion could account for up to 24–32% of sediment yield over timescales sufficiently long to allow 100% sediment delivery.

As with other erosion forms, impacts are partially reversible and rely on ESC and soil conservation measures to reduce erosion in the first place.

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

This attribute is not routinely monitored either nationally or regionally.

Some councils may carry out periodic surveys and bare ground assessments (as a proxy for erosion). At finer and local scales, councils and catchment groups may undertake stream monitoring (water clarity, stream bed cover assessment, etc) or catchment surveys to determine areas where erosion (including surface erosion is likely to occur).

There is no consistent methodology in use and no nationally agreed monitoring methodology for assessing surface erosion in New Zealand. Internationally, plot studies have been used to determine the influence of factors on surface erosion (e.g., Anache et al., 2017; Carollo et al., 2024). Similarly, there is unlikely to be any standard methodology to determine treatment monitoring, other than perhaps by modelling (Morgan et al., 1998) or by remote sensing (e.g., Sepuru & Dube 2018; North et al., 2022).

There is also no consistent methodology to measure surface runoff from small catchments to evaluate control measures from its retention in sediment traps and detainment bunds etc (Levine et al. 2021; Smith & Muirhead 2023).

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

If this attribute were to be monitored, it would require access to private land to firstly undertake an erosion assessment and secondly, determine what treatment has been carried out and if it was successful.

Remote sensing may be effective in monitoring bare ground as a proxy for surface erosion (and vice versa previous bare ground now vegetated as evidence of control) (North et al., 2022). Repeat aerial photographs or other remote sensing methods including LiDAR may also be suitable.

Differentiating sediment sources by erosion process at the catchment scale is currently only possible using sediment fingerprinting techniques (Vale et al, 2022). While these have been used in New Zealand, there is some disagreement on the results each method provides (Vale et al., 2022) and in relation to contributions of sediment from different land uses.

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

Monitoring of this attribute is not being done routinely and, therefore, costs are hard to assess.

Repeat remote sensing and/or repeat on-ground bare ground assessment might be a useful methodology but currently this is likely to be expensive. It would also require validation because all bare ground may not be losing soil from surface erosion, i.e., it just looks bare.

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

We are not aware of any monitoring being carried out by representatives of iwi/hapū/rūnanga. However, erosion is of high interest to Māori and some hapū/iwi are focused on monitoring erosion and mitigating risk. Successful erosion control within the Waiapu River catchment, for example, is required to achieve the cultural aspirations of Ngāti Porou. Measures of erosion (including visual observations and other remotely-sensed measurements) are parts of an holistic approach to assessing a catchment's state.

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

Surface erosion is correlated and could be grouped with other erosion processes, e.g., shallow landslide erosion, surface erosion, and gully erosion in assessments of erosion (e.g., NZEEM, HEL, SedNetNZ).

In the LUC/Land Resource Inventory the key/dominant erosion process for that polygon is described along with its severity. Surface erosion includes sheet (Sh), wind(W), and scree (Sc) (Lynn et al. 2021), along with secondary erosion processes. Severity of erosion is rated in 6 classes based on % of the area affected by that process (Lynn et al. 2021).

Part B—Current state and allocation options

B1. What is the current state of the attribute?

Surface erosion is well understood in a broad qualitative sense but not quantitatively, spatially or temporally, at least at the regional and national levels. The process is very well understood as are the main ways to reduce or control it (Basher 2013; Phillips et al., 2020). 'Understanding' in the context of quantitatively monitoring surface erosion spatially and temporally at regional and national levels is not advanced enough for this to be used as a national indicator, though locally it may be possible to monitor it once a baseline state is determined.

Like other erosion processes, to be used as an indicator would require significant investment and assessment to establish a baseline state for each region of active surface erosion, and then its treatment. Once a national layer was available, it could be monitored (5-yearly) relative to the starting baseline.

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

We are unaware of any known natural reference states for this attribute.

As surface erosion is a natural geomorphic process understanding what the situation was like pre-European is impossible, i.e., we know that erosion was less under a natural forest cover and prior to human settlement, but we do not know what erosion process was dominant or the relative balance between different processes.

A pre-European reference state, while potentially attractive, would be difficult to quantify and would not be attainable in contemporary NZ. Surface erosion would have existed in NZ pre-Europeans, but it was the clearance of indigenous forest in both islands for farming that exacerbated erosion of all types, including surface erosion. Land use and land use practices on erosion-prone land are still contributing to further erosion, e.g., Donovan (2021).

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

We are unaware of any existing numeric or narrative bands for this attribute.

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

We are unaware of specific thresholds or tipping points for this attribute.

Severe surface erosion (and erosion in general) is likely to have significant local effects on ecological integrity, i.e., within metres of source areas but effects would dissipate rapidly from the active source.

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

Many ESC approaches such as mulching are effective immediately they are implemented on bare ground. Others, such as hydroseeding and planting may take longer to become effective.

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

In addition to discussing this attribute directly with iwi/hapū/rūnanga, there is likely to be tikanga and mātauranga Māori relevant to informing bands, allocation options, minimally disturbed conditions and/or unacceptable degradation in treaty settlements, cultural impact assessments, environment court submissions, iwi environmental management and climate change plans etc.

Erosion or disturbed land may be one metric in cultural health assessments (Tipa & Tierney 2003).

Part C—Management levers and context

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

The state of surface erosion protection is driven by the need for land to continue to be used for agriculture and urban purposes. Where surface erosion is severe, ‘productive’ land may be impacted by rain events, requiring landowners to employ soil conservation or ESC methods to reduce soil loss. Most councils require earthworks to be controlled and, in some regions, regional rules may require that farmers follow prescribed practices either as part of consent conditions or central Government policies. Where land has protective measures in place and active areas of surface erosion have been treated and stabilised, such measures are expected to benefit the ecological health of adjacent waterbodies in ways described in Section A.

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

C2-(i). Local government driven

There are several mechanisms employed to affect control of surface erosion and runoff in New Zealand. Principally these are delivered to landowners and developers through regional and district councils via regulatory processes (consents), land development levies, voluntary actions or in some cases via incentives. Council staff usually provide advice to landowners/developers via urban planning, farm planning or catchment planning processes. Most councils have clear ESC guidelines that land developers are required to adhere to (MfE 2023). This may or may not apply to other land users such as farmers and foresters. The NES-CF has an erosion susceptibility map, the classes of which trigger a range of requirements in terms of consenting regime. Measures used to control surface erosion and reduce runoff are many and most can be found related to earthworks and urban and transport development (e.g., Auckland Regional Council 1999).

C2-(ii). Central government driven

Examples of current and past funds that have supported erosion control (in general) using a range of methods include One Billion Trees Fund, the Provincial Growth Fund, Hill Country Erosion Fund, Jobs for Nature, etc. Such funds are usually implemented via MPI or MfE. Most of these programmes report metrics on funding inputs rather than on what has been achieved. MfE (2023), as part of the NPS for Freshwater provide information relating to the management and monitoring of surface erosion and sediment control.

C2-(iii). Iwi/hapū driven

Iwi planning documents such as Environmental Management Plans and Climate Change Strategies/Plans may contain policies/objectives/methods seeking to influence landscape outcomes for the benefit of current and future generations.

C2-(iv). NGO, community driven

Increasingly, community catchment groups undertake aspects of erosion control by planting and action on the ground, either in partnership with councils and landowners or increasingly directly with collectives of landowners within catchments. Such actions would include treatment of areas of surface erosion, but they are generally targeted at a broader range of processes and outcomes such as biodiversity improvement. Some of these initiatives fall under the umbrella of organisation such as the NZ Landcare Trust, Tane’s Tree Trust, NZ Farm Forestry Association, Forest & Bird, The Nature Conservancy of Aotearoa New Zealand, Pure Advantage (O Tatou Ngahere) while others are developed independently.

Some primary sector bodies via industry levies (e.g., Dairy NZ, Beef&Lamb NZ, Fonterra, etc) also provide advice to landowners on what and how to implement erosion control and may require landowners to keep records as part of farm plan implementation.

There are many pathways for landowners to get advice and to implement action on the ground to protect land, water, habitats, and biodiversity which erosion of all forms can impact. While there are many initiatives, coordination is often lacking and recording of what is being done where is less than optimal.

C2-(v). Internationally driven

There may be some international agencies, including WWF, The Nature Conservancy, IUCN, etc., that also contribute advice and/or funding to projects in which surface erosion is controlled either directly or as a co-benefit of other actions.

Part D—Impact analysis

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

If this attribute (or erosion in general) was not managed, further losses to ecological integrity (sedimentation in rivers, wetlands, hydro dams, estuaries and oceans), reduced clarity in freshwaters, etc., particularly in the areas where surface erosion is active would likely result. Continuing degradation of both hill country and lowland soils would lead to reduced productivity (Rosser & Ross 2011), and in some cases loss of high value land on flood plains.

Not managing this attribute may lead to further cultural impacts for Māori particularly in some sensitive locations. It may also impact the mental wellbeing of some landowners who try unsuccessfully to manage erosion if there is no support from councils to assist in mitigation, especially after large events, i.e., they see their soil and land wash away.

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

Economic impacts from not managing this attribute (and erosion in general) will affect both urban and rural sectors in New Zealand. Ultimately it affects all New Zealanders as taxes and rates are the major sources of funding for managing this.

Farmers and landowners, iwi and urban dwellers are all affected. Impacts are likely to include further decline in freshwater health, increased costs of managing sediment in water bodies, drinking water etc, and increased costs associated with repairing flood damage resulting from large storm events (MfE 2020).

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

Climate change is projected to result in more storminess, but some regions may experience either a decline in total rainfall or an increase. As rainfall is a key driver of most erosion, climate change impacts are likely to be variable, though erosion overall is expected to be greater (Neverman et al., 2023).

Managing this attribute will improve overall resilience to future climate changes. It may not be as significant as retiring land prone to mass movement erosion and planting or reverting to more intact forest cover. However, the pace of implementation of measures to manage erosion (in general) is unlikely to keep up with the perceived, modelled, or real changes arising from climate change (e.g., Vale et al. 2022; Vale & Smith 2023).

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7.8 Riparian protection/streambank erosion control

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Preamble: Riparian streambank protection or 'treatment' may occur through a range of techniques including hard engineering approaches such as rip rap, groynes, dykes, gabion baskets, with soil bioengineering such as brush layering, fascines, and plantings.

State of knowledge of "Riparian protection / streambank erosion control" attribute: Good / established but incomplete

Good state of knowledge because we understand the process, there are quite a few studies and many observations of bank erosion in the field (though unlikely to be comprehensively recorded), and there is agreement on what stream bank erosion is, what riparian streambank protection is, and what the benefits are. However, there is little to no spatial coverage of either current active bank erosion or of riparian areas that have been 'protected' which currently limits this as a national attribute.

<u>Part A—Attribute and method</u>

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

Established/vegetated riparian margins and stable streambanks can benefit the ecological integrity and human health of adjacent waterbodies in several ways. This attribute should be read in conjunction with Riparian margin establishment/protection attribute (Matheson 2024). The stream bank protection attribute focuses on ensuring stream bank soil is stabilised (either via hard engineered or nature-based (plant) solutions). Stable streambanks help reduce erosion and thus reduce sediment impacts in water bodies (Marden et al., 2005; Hughes 2016, 2021).

Streambank erosion is a natural geomorphic process (along with other erosion processes) which occurs in all channels as adjustments of channel size and shape are made to convey the discharge and sediment supplied from the stream or river catchment.

As a natural process, bank erosion is generally beneficial to the ecology of waterways, since erosion and deposition create a variety of habitats for flora and fauna which contributes to ecological diversity (Environment Agency 1999). Conversely, an increase in sediment supply due to accelerated stream bank erosion, which can often be linked to land-use change, is a major cause of non-point source pollution within river systems. In such cases mitigations (hard and soft engineering solutions) are required to reduce those impacts.

Stream bank erosion can be classified into two basic groups. Those dominated by gravitational or mechanical failures (mass movement) and those where hydraulic-induced failure mechanisms (fluvial erosion) dominate. Gravitational failures include both mass movement failures and individual grain failures. The two process groups are often linked (e.g., a hydraulic-induced mechanism, such as bank undercutting, can cause a gravitationally-induced collapse such as a cantilever failure). Identification of bank erosion processes, via a bank assessment survey, is important for determining suitable measurement techniques and for choosing appropriate remedial options. The conditions under which different processes occur are determined by bank material characteristics and local soil moisture conditions (O'Neill and Kuhns 1994).

Active stream bank erosion is often treated as part of river engineering and/or catchment management. Hard engineering approaches such as rip rap, groynes, dykes, gabion baskets, bank grading, crib walls, geotextiles, etc may be used singly or in combination with each other and in combination with soil bioengineering including brush layering, fascines, wattle fences, hydroseeding, planting, etc (e.g., Schiechtl & Stern 1994; Rey et al., 2018).

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

How well stable streambanks and riparian margins protect ecological integrity varies through space and time. Factors influencing this protection include upstream and upslope land use, rainfall, streamflow, geology, past land use and river history (legacy effects), storm frequency, protection/interventions, etc.

Often effects from unstable and eroding banks are felt locally though if bank erosion is extensive, it is a process that may contribute significantly (up to 100%) to the stream's sediment load (De Rose 1999; Hughes 2021). However, in New Zealand there is little data on the contribution of bank erosion to measured river sediment yields and it has been a relatively poorly studied process (Watson & Basher 2006; Hughes et al., 2021; De Rose & Basher 2011; Smith et al., 2019).

Since the establishment of Catchment Boards and later Regional Councils under the Soil Conservation and Rivers Control Act 1941, many rivers have been managed for flood protection. Much of this work relates to addressing issues of bank erosion. Thus, at least for the main stem of rivers on flood plains, councils will have a reasonable idea of where areas of bank erosion are, their severity, and the degree of controls in place. However, for smaller rivers and streams this information is unlikely to be known. There are no national inventories or surveys of bank erosion as catchment sources of sediment. This limits the effective implementation of nation-wide catchment prioritisation and rehabilitation measures.

Clearance of indigenous vegetation in the 19th and 20th centuries has accelerated erosion (including bank erosion) in many places in New Zealand. It is likely that many New Zealand catchments are still responding to these disturbances (Hughes 2021). Catchment-wide channel erosion in response to land disturbance is a widely reported occurrence (e.g., Wasson et al., 1998; Kondolf et al., 2002; Downs & Gregory 2004). Such disturbance-induced channel adjustment will continue until a new steady state is reached under the modified discharge and sediment supply regime (Harvey and Watson 1986).

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

As land use has changed, the associated hydrological regime in rivers changes. Hence regions that have experienced significant changes in land use or land use practices (vegetation removal, increased soil compaction, land drainage, etc) bank erosion is likely to have increased, e.g., Southland (Ellis et al., 2018; Poole 1990). In other areas where bank erosion has been identified and mitigation measures implemented, e.g., in many main stem rivers, bank erosion is likely to have declined. Many lowland rivers now have fully armoured channels and associated stop banks, mostly to reduce flooding risk, but also to reduce and loss of land by bank erosion (Brierley et al., 2023).

Climate change projections suggest increasing storminess in many regions indicating a concomitant increase in erosion is likely. How this will preference bank erosion over other erosion processes is unknown. Bank erosion has not generally been included in modelling exercises aimed at understanding climate change impacts, (e.g., Basher et al., 2020; Neverman et al., 2023) though it is represented in some sediment yield models such as SedNetNZ (Dymond et al., 2016).

Impacts are only partially or temporarily reversible. In short timescales (years to decades), hard engineering and soil bio-engineering (vegetation +/- hard engineering) will reduce and “treat” localities where bank erosion is prevalent, i.e., river engineering. At decadal to century timescales and in the most severe rain events, geomorphic thresholds are crossed and bank erosion may appear in places where it previously didn’t exist and/or treated areas may be re-activated.

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

This attribute is not routinely monitored either nationally or regionally.

Some councils may carry out periodic surveys and bank erosion assessments (e.g., as part of monitoring stream bank vegetation), e.g., Norris et al., (2020). Similarly, at much finer scales, many communities, catchment groups, and iwi undertake periodic stream and bank assessment surveys as part of catchment ‘health’ and cultural assessments.

There is no consistent methodology in use nor nationally agreed monitoring methodology for assessing bank erosion. Many approaches are qualitative or at best semi-quantitative. Similarly, there is unlikely to be any standard methodology to determine treatment monitoring.

A possible metric might be the length (or %) of a stream or river that has active bank erosion (often assessed as bank retreat) and the proportion of that active bank has been treated. Measures of bank erosion may also be incorporated into more general erosion and river bank vegetation surveys.

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

If this attribute were to be monitored it would require access to private land to firstly undertake a stream/bank assessment and secondly, determine what treatment has been carried out and if it was successful.

Unlike other erosion processes, remote sensing may not be as effective in monitoring stream bank erosion and its treatment largely because of spatial resolution, i.e., rivers and streams are linear features and eroding stream banks (and treated stream banks) may not be visible on typical aerial photographs or via other remote sensing methods, possibly with the exception of repeat LiDAR. Private land would also have to be accessed in order to validate any remote sensed mapping.

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

This attribute is not being routinely monitored and, therefore, costs are hard to assess.

Repeat or differential LiDAR might be a useful future methodology but currently this is very expensive relative to potential benefits.

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

I am not aware of any monitoring being carried out by representatives of iwi/hapū/rūnanga. However, erosion is of very high interest to Māori and various hapū/iwi are focused on monitoring erosion and mitigating risk. Successful erosion control within the catchment is required to achieve the cultural aspirations of Ngāti Porou. See, for example, the Waiapu River Restoration project (Led by Te Rūnanganui o Ngāti Porou) that focuses on erosion in the Waiapu catchment https://www.ngatiporou.com/nati-news/the-waiapu-river-restoration. See also the Waiapu Koka Huhua initiative https://ourlandandwater.nz/wp-content/uploads/2022/02/TMOTW-Case-Study-Waiapu-Kokahuhua.pdf, where restoration ofriparian plantings is a key mitigation.

Bank erosion assessment may be included in iwi ‘cultural health’ monitoring of local streams.

Our Land & Water National Science Challenge has developed a “Register of Land Management Actions” (https://ourlandandwater.nz/project/register-of-land-management-actions/) which may include aspects of bank erosion treatment.

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

Stream bank erosion is correlated and can likely be grouped with other erosion processes, e.g., shallow landslide erosion, surface erosion, and gully erosion in assessments or models of erosion (e.g., NZEEM, HEL). It may or may not be directly correlated with the other listed land/soil attributes.

In the LUC/Land Resource Inventory the key/dominant erosion process for that polygon is described along with its severity, e.g., in LUC as Sb stream bank erosion (Lynn et al., 2021), along with secondary erosion processes. Severity is rated in 6 classes based on % of a reach in which stream bank erosion occurs along both banks and the lateral erosion or stream bank retreat (Lynn et al., 2021).

Part B—Current state and allocation options

B1. What is the current state of the attribute?

Current state of stream bank erosion is not well understood at the national scale (the process is well understood and locally/regionally its general distribution (or prominence as an erosion process) might be, at least for the main stem lowland rivers). Understanding in my opinion, is not advanced enough for this to be used as a national indicator across all streams and rivers.

To be used as an indicator would require significant investment and assessment to establish a baseline state for each region of active stream bank erosion, and its treatment. Once a national layer was available, it could then be monitored (5-yearly) relative to the starting baseline.

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

I do not know of any natural reference states for this attribute. As bank erosion is a natural geomorphic process understanding what the situation was like pre-European is impossible.

A pre-European reference state, while potentially attractive, would be impossible to quantify and would not be attainable in contemporary New Zealand. Stream bank erosion would have existed in New Zealand pre-Europeans (and pre-humans), but it was the clearance of indigenous forest in both islands for farming in the 19th and 20th centuries that exacerbated erosion of all types, including stream bank erosion. Land use and land use practices on erosion-prone land are still contributing to further erosion of all types. Vegetation clearance altered the hydrologic response of catchments and would have contributed to an increase in bank erosion.

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

I am unaware of any existing numeric or narrative bands for this attribute.

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

Thresholds or tipping points for this attribute have not yet been defined.

Severe bank erosion is likely to have significant local effects on ecological integrity, i.e., within metres but effects would dissipate rapidly from the active source. Cumulatively, if a large proportion of both stream banks were active, and not treated, then sediment loads would increase and impacts to downstream receiving bodies such as lakes and estuaries would be elevated, and locally stream beds would be inundated with sediment. Differentiating sediment sources by erosion process at the catchment scale is only possible using sediment fingerprinting techniques (Vale et al 2022).

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

Lags and legacy effects for this attribute are likely. Depending on the treatment method, it may take some years for bioengineering methods to become effective. Hard engineering such as rip rap are effective immediately they are implemented.

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

In addition to discussing this attribute directly with iwi/hapū/rūnanga, there is likely to be tikanga and mātauranga Māori relevant to informing bands, allocation options, minimally disturbed conditions and/or unacceptable degradation in treaty settlements, cultural impact assessments, environment court submissions, iwi environmental management and climate change plans, etc.

River/streambank condition is one of several metrics in cultural health assessments (Tipa & Tierney 2003).

Part C—Management levers and context

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

The state of streambank and riparian protection is driven by the need for land to continue to be used for agriculture and urban purposes. This includes vegetation removal from banks which has a direct effect and vegetation removal/change within the contributing catchment which changes the hydrological regime (indirect effect). Where bank erosion is severe, productive land is lost during high river flows, requiring landowners to move fences and often requiring regional councils to invest in river engineering approaches such as rip rap, bank grading, planting etc. Where land has protective measures in place and eroding stream banks have been stabilised, such measures are expected to benefit the ecological health of adjacent waterbodies in ways described in A1. At a general level the relationships between the activities that result in bank erosion and how it is treated are understood, but at the local level they are often not, i.e., we are often treating the symptom rather than the cause (Ellis et al., 2018). For example, bank erosion is recognised locally, and interventions are made to treat it locally, but it may be occurring in response to changes in the catchment that alter the hydrological regime far removed from where the bank erosion is manifested (Hughes et al., 2016).

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

C2-(i). Local government driven

There are several mechanisms employed to affect riparian protection-streambank control in New Zealand. Principally these are delivered to landowners through regional and district councils via incentives, voluntary actions or in some cases via regulatory processes. Council staff usually provide advice to landowners via farm planning or catchment planning processes. Financial support for fencing and planting on a cost-share basis is the primary mechanism for action on the ground. Often, central government through large funding programmes (see C2) complement local rates investment. Where bank erosion is severe and critical infrastructure is threatened, councils usually fund hard engineering directly with or without landowner contributions.

C2-(ii). Central government driven

Examples of current and past funds that have supported riparian and streambank protection include One Billion Trees Fund, the Provincial Growth Fund, Hill Country Erosion Fund, Jobs for Nature, etc. Such funds are usually implemented via MPI or MfE. Most of these programmes report metrics on funding inputs rather than on what has been achieved. Some information may be available through https://ourlandandwater.nz/project/register-of-land-management-actions/. Further, few if ever, evaluate if the desired outcome is delivered, in part because there is a delay or lag of years to decades between when the intervention is implemented and when a potential outcome might be reached. Where data are recorded, they tend to be simple metrics such as length of stream retired from grazing, number of plants planted, etc.

C2-(iii). Iwi/hapū driven

Erosion risk and mitigations to prevent these occurring is of high interest to hapū/iwi, especially in the areas severely impacted by Cyclone Gabrielle (for example). Iwi planning documents such as Environmental Management Plans and Climate Change Strategies/Plans may contain policies/objectives/methods seeking to influence erosion outcomes for the benefit of current and future generations.

C2-(iv). NGO, community driven

Increasingly, community catchment groups undertake planting and action on the ground, either in partnership with councils and landowners or increasingly directly with collectives of landowners within catchments. Some of these initiatives fall under the umbrella of organisation such as the NZ Landcare Trust, Tane's Tree Trust, NZ Farm Forestry Association, Forest & Bird, The Nature Conservancy of Aotearoa New Zealand, Pure Advantage (O Tatou Ngahere), etc., while others are developed independently.

Some primary sector bodies via industry levies (e.g., Dairy NZ, Beef & Lamb NZ, Fonterra, etc.) provide advice to landowners on what and how to do riparian protection and may also require landowners to keep records of metrics such as length of streams fenced, numbers of plants planted in riparian margins etc.

C2-(v). Internationally driven

As discussed above, there are many pathways for landowners to get advice and to implement action on the ground to protect land, water, habitats, and biodiversity. While there are many initiatives occurring across the country, coordination is often lacking and recording of what is being done where is less than optimal.

Several international agencies also contribute advice and/or funding to projects in this space including WWF, The Nature Conservancy, IUCN, etc.

Part D—Impact analysis

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

If this attribute (or erosion in general) was not managed, further losses to ecological integrity (sedimentation in rivers, wetlands, hydro dams, estuaries and oceans), reduced clarity in freshwaters, etc., particularly in the areas where streambank erosion is active would likely result. Continuing degradation of both hill country and lowland soils would lead to reduced productivity, and in some cases loss of high value land on flood plains (Heaphy et al., 2014; Soliman & Walsh 2020; Walsh et al., 2021).

Not managing this attribute and erosion in general may lead to further cultural impacts for Māori particularly in some sensitive locations. It may also impact the mental wellbeing of some landowners who successively lose their land due to stream and river bank erosion if there is no support from councils to mitigate such occurrences, especially after small to medium floods.

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

Economic impacts from not managing this attribute (and erosion in general) will affect both urban and rural sectors in New Zealand. Ultimately it affects all New Zealanders as taxes and rates are the major sources of funding for managing erosion and its impacts.

Farmers and landowners, iwi and urban dwellers are all affected. Impacts are likely to include further decline in freshwater health, increased costs of managing sediment in water bodies, drinking water etc, and increased costs associated with repairing flood damage resulting from large storm events.

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

Climate change is projected to result in more storminess, but some regions may experience either a decline in total rainfall or an increase. As rainfall is a key driver of most erosion, climate change impacts are likely to be variable, though erosion overall is expected to be greater.

Managing this attribute will increase overall resilience to future climate changes. However, the pace of implementation of measures to manage erosion (in general) is unlikely to keep up with the perceived, modelled, or real changes arising from climate change (e.g., Vale et al., 2022; Vale & Smith 2023).

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7.9 Soil compaction

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Preamble: There are multiple measures for soil compaction. These indicators for soil compaction can be measured at a single points in time or multiple points in time to determine the degree to which soil is compact when comparing against established target values. Soil compaction is responsive to physical actions, including biological activity – this means multiple points in time are more robust; near-surface compaction can change seasonally. Indicators of compaction include:

- Macroporosity (typically volumetric percentage of soil pores greater than 30 or 60 microns, or other as specified in the literature) [1, 2]. The definition using pores greater than 30 microns tends to be used by AgResearch and the Land Monitoring Forum[1], with pores greater than 60 microns used in studies by Manaaki Whenua – Landcare Research (MWLR) [2], and the MWLR laboratory [3]. Compaction is indicated by low macroporosity.
- Bulk density – well used in the literature and by the Land Monitoring Forum but is useful within similar soils, and a poor measure across different soils (e.g., Organic and Pumice Soils intrinsically much lower than Pallic and Ultic Soils)
- Aggregate stability – well used as an indicator under cropping, in the literature for cropping land use and by the Land Monitoring Forum particularly for cropping land use, and in some reviews, studies [4, 5]. It could be used as an indicator of compaction (compact soil has more 'massive/blocky' aggregates), but more commonly used as indicator of structural degradation.
- Penetration resistance – well-used in the literature particularly to indicate restrictions to root growth but is responsive to moisture content, can be subject to operator influence (so not a good indicator in my opinion)
- Infiltration and other measures of water flow through pores e.g., saturated or unsaturated hydraulic conductivity.

State of knowledge of the "Soil Compaction" attribute: Excellent / well established – comprehensive analysis/syntheses; multiple studies agree. There is a comprehensive body of knowledge on how soil compaction through animal treading and machinery for arable cropping affects soil physical properties, with good knowledge on crop yields.

Part A—Attribute and method

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

The attribute relates to ecological integrity (not human health). Soil compaction usually occurs in unsaturated soil (moderately wet or below field capacity) [1]. Soils can only be compacted when air is present (otherwise they are deformed); the response of a soil to compaction is related to soil moisture (soils being more resistant when dry). Typically, some large air-filled pores in soil get compressed when soil is compacted by various means e.g., stock-treading, machinery trafficking. The resulting soil structure can restrict air, gas and water movement in soil, thus affecting plant root and shoot growth, arrangement of pores, and therefore a range of ecosystem services such as storage and filtering of water, biomass production, nutrient cycling, carbon storage, soil biodiversity, and physical stability [1, 5]. Soil compaction can also affect gas exchanges, and biological processes such as C and N mineralisation, nitrification and denitrification [6]. Soil function underpins key ecosystem services such as pasture and crop production, nutrient cycling and contaminant losses [5, 6], so soil compaction affects soil function. Compact soils limit root growth through loss of aeration and/or resistance to root penetration – this can limit plant establishment e.g., pine trees [7], crops, and plant succession (favouring species tolerant of compaction), drought resistance and vulnerability to pathogens [8]. For urban areas and perennial species impacts of compaction are greater in non-irrigated areas and undrained areas. Over-compaction of the soil when heavy machinery is used in urban areas impacts on drainage capacity and the ability of roots to penetrate the subsoil [9].

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

There is a good body of evidence on how compacted soil affects pasture and crop yields, plant root growth; as soil becomes more compact, plant yield and root growth generally decline [1, 10, 11]. There is some evidence on the impact of soil compaction on macrofauna, such as worms where compaction limits earthworm abundance and activity [5, 10]. There are only a few recent studies on the impact of soil compaction on soil bacterial communities and their diversity [13, 14] and the mycorrhizal community.

The weight of evidence and state of knowledge is very strong on how soil compaction affects soil properties from as shown by multiple literature reviews [1, 5, 6], and from New Zealand regional and national reporting [19-25]. Macroporosity, bulk density and aggregate stability are commonly used indicators [1, 5, 6], [19-25]. Spatial extent is clearly linked to soil management practices and moisture variation at the time soils are vulnerable.

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

Considerable research on soil compaction has been conducted in New Zealand, primarily in the context of agricultural, and plantation forestry [7, 28], and to a lesser extent in horticultural production [27]. Such studies evaluate how soil compaction is affected by different grazing practices, soils or mitigations [6, 30-34], crop management and practices [5, 35], and farm system modelling of the impact of pugging on pasture yield across a farm system [36, 37], and plantation harvesting and site preparation [7, 28]. Pugging primarily occurs when the soil is very wet, causing visual soil deformation, whereas compaction is more 'hidden' i.e., below the surface. A very few studies have studied soil compaction associated with large-scale infrastructure (dams, mines, roads and urbanisation) [38, 39].

Changes in soil compaction over long periods of time have primarily been assessed in regional council state of the environment soil quality monitoring but few trends have been observed over the long-term [19, 22, 40]. No trend in macroporosity was found in drystock farming, dairy farming, orchard/vineyard land use, or cropping from 1995 to 2018 [20, 40]. Similarly, no trend in bulk density was found for drystock farming land use from 1995 to 2018 [40]. An exception is changes in compaction following remediation actions to prepare sites for establishment of plantation forests – these studies show responses last >20 years, being 'locked in' as root systems proliferate in loosened zones.

Most experimental research studies that have examined changes in soil compaction over time, have typically been up to about three years duration, or up to five years duration [41,42]. Most studies have tended to be short term changes over time (< several years) following treatments to make the soil compact. However studies examining recovery/rejuvenation of compaction are much fewer, e.g., in cropping [43] or pasture [41].

Compaction is reversible (through natural process and active management for remediation) to some extent especially at shallow soil depths, e.g., 0–10 cm, but is much more difficult to reverse at deeper depths to 20 or 30 cm under dairy cattle grazing especially with irrigation [46,47]. Recent evidence indicates from several indicators e.g., macroporosity, bulk density, available water capacity etc, soil compaction under dairy farming is occurring to depths of about 30 cm, i.e., typical depth of topsoil [46, 47]. Some reversibility can be from natural processes (e.g., cracking, shrink/swell, frosting, worm activity etc) given sufficient time [41] in shallow depths, or from management practices including mechanical subsoiling/aeration equipment using a tractor for deeper soil e.g., subsoil depths [48-50]. Cultivation is not generally practical for pasture except at pasture renewal.

On cropping farms, continued vehicle movement or ploughing on wet soil, can lead to soil compaction sometimes called a 'plough pan' or a 'pan' i.e., not a natural feature. But a pan can occur at deeper depths caused by ploughing, and so is more difficult to reverse, unless something like subsoiling or soil aeration is used. Soil compaction on cropping farms in also reviewed by [5], and there are numerous New Zealand studies [46], and international studies [47] (if more detail is needed). Vineyard compaction mitigation is through compost application, managing vegetation cover, changing machinery ground-pressure, etc [96].

Compaction caused by ground-based harvest of plantation forests is ameliorated to at least 50 cm depth using one-off mechanical loosening between harvests that also raise the soil surface and can be designed to also enhance drainage. For example, the effects of deep ripping in forest, on a Pumice soil to 80 cm depth lasted 25 years [53]. These techniques are also used to ameliorate deep compaction associated with infrastructure developments (R Simcock pers comm). Deep ripping in pine forest soils reduced penetration resistance and increased the stem volume of *Pinus radiata* [54], and increased seedling height and survival [55].

New Zealand has experienced significant land use intensification such as expansion of the dairy industry and irrigated land, and both in combination, over the last 20–30 years [2, 5, 47]. The change of land use to irrigated dairying or irrigated beef cattle grazing has resulted in more soil compaction than under non-irrigated land [15, 47].

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

Regional council SoE soil quality monitoring includes macroporosity and bulk density at 0–10 cm depth across New Zealand [24, 56, 57], with methods specified in the NEMS for soil quality and trace elements. Data are usually compared to provisional target values for these soil quality indicators [41]. Many councils undertake monitoring yearly and report per year, e.g., Wellington region, whereas others undertake monitoring and reporting 5-yearly e.g., Taranaki region. Several studies provide an assessment of regional soil quality monitoring over time [19, 21, 22, 40], with some additional targeted studies on aspects of compaction also available [25, 58, 59]. At the time of writing, most councils have undertaken monitoring and have published results through reports. There appears to be few details or reports published from some councils (e.g., Horizons, Gisborne), while other councils have recently commenced monitoring (Otago) or have undertaken monitoring intermittently (Southland). No soil quality monitoring has been undertaken in the West Coast region.

In Canterbury, under long-term arable production, aggregate stability (used a lot in cropping due to aggregate break-down), macroporosity, bulk density, penetration resistance etc are used to monitor soil quality [58].

There are variable terms used, and variable ways in which macroporosity can be measured, which has led to confusion and inconsistency in the reporting of macroporosity in previous studies [60]. The NEMS specifies uses the term air-filled porosity, for macro-porosity measured at -10 kPa.

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

For direct soil measurements, there is a need to access privately owned land to collect repeat samples for monitoring of this attribute. Landowners may be more, or less, willing to provide access to land for sampling and to have data from their land used for regulatory informing purposes.

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

A key cost is staff time to undertake sampling of about an hour plus travel, with additional staff required for interpretation and reporting etc.

Bulk density and macroporosity are sampled using undisturbed cores (not sieved) using stainless steel rings. The Land Monitoring Forum/NEMS recommends 3 samples per site (0-10cm depth only is monitored, but this is a limitation for perennial deeper-rooted plants), each of which get analysed in the laboratory. Topsoil and upper subsoils should be measured in non-pastoral sites with deeper rooting species to better inform implications for resilience and production. The Manaaki Whenua – Landcare Research physics laboratory routinely does bulk density and macroporosity testing, along with aggregate stability, and a range of other tests. Further information is available from:

https://www.landcareresearch.co.nz/partner-with-us/laboratories-and-diagnostics/soil-physics-laboratory/

Current costs for bulk density and macroporosity measurements combined are approximately $60+GST/sample, but noting that 3 samples per site are needed (total ~$180).

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

Yes. Ngāi Tahu Farming is involved in a soil health project with AgResearch, where soil macroporosity (air-filled porosity at −10 kPa) and bulk density were measured [60]. More broadly, soil health is of high interest to Māori, and soil macroporosity is just one attribute within a suite of holistic measures that can be used to assess health status.

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

Surface runoff of nutrients and other contaminants (P, sediment, *E. coli*) (and hence impacts on freshwater quality) are more likely from compacted soils [5, 61]. Some correlations of compaction measures with changes in the composition of soil bacteria have been observed based on NZ soil quality monitoring data [13,14].

Part B—Current state and allocation options

B1. What is the current state of the attribute?

Macroporosity and bulk density, have been reported in several regional and national studies, and numerous other studies have assessed soil compaction using a wider range of soil physical properties e.g., [59]. In national reporting of state of the environment soil quality monitoring [60] and other review [5, 6] Soil compaction (as measured by macroporosity) has been reported as a key issue in the Waikato, Auckland, Marlborough and Wellington regions [5] – primarily associated with land used for dairy [6, 21].

However, no trend in macroporosity over time were observed in soils used for drystock, dairy, orchard/vineyard, cropping land use, or other land uses, 1995 to 2018 [40, 63]. Similarly, no decreasing trend or improvement in bulk density was found for drystock farming land use from 1995 to 2018 [40].

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

No. Compaction primarily occurs on land which is disturbed though human activity, i.e., primary production land, urban land development [39], but is also recorded in native forests from wild ungulates (deer) [61]. Comparison of measures used to assess compaction (macroporosity, bulk density) in undisturbed fence line areas may help to identify typical ranges associated with undisturbed agricultural land. However, indigenous forest soil sampling to compare agricultural land may not be useful for monitoring. In just a few studies, soil compaction has been measured under farm paddock fence lines [65] to compare with paddock areas, as those areas are typically not trampled nor affected by vehicles. My opinion is that indigenous vegetation reference states would not be useful to inform specific agricultural management for this indicator (compaction). Use of undisturbed fence line areas may be useful as a reference state for a predominantly undisturbed paddock area, but its use would be depended on the objectives of the study.

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

There are existing 'target values' for macroporosity, bulk density, aggregate stability that are used in council SOE monitoring [57] although revision of these target values is being undertaken through a contract with MfE (Revision of Soil Quality Indicator Target Ranges).

For macroporosity in New Zealand, there is some literature about minimum macroporosity to support crop and pasture yield, much of which is reviewed in [1, 66]. The reference [1] reviews some of the science behind the typically used value of <10% macroporosity. Similarly, bulk density and cone penetration resistance affects crop and pasture yield, typically via impedance to roots) and this is more commonly described in the literature, and is included in [1, 66] and other studies.

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

There is some information about minimum values of macroporosity that can impact on crop and pasture productivity [62] [1, 66] [67]. The reference [1] reviews some of the science behind the typically used value of <10% macroporosity. Bulk density is more commonly used as a measure of compaction (as it is easier to measure and useful) and effects on crop and pasture yield and this is more commonly described in the literature, and is included in [1, 66] and other studies.

Specific impacts on ecological integrity occur but thresholds are generally ecosystem, plant species and plant-growth stage specific and mediated by climate – such impacts include vulnerability to disease, (especially root rots), anoxic conditions, drought and wind-throw.

Tipping points result in changes in relative competitiveness of different species, e.g., from kiwifruit or avocado (intolerant of poor drainage or waterlogging, i.e., wet soil;[97]) to pipfruit (more tolerant); titoki and kanuka (less tolerant) to kahikatea and manuka (more tolerant) (R Simcock pers comm). There are also differences in tolerance to drought and drainage/waterlogging between cultivars in grapes and apples, as different cultivars have different drought and drainage tolerances. Soil drainage, aeration and water storage properties should be used to guide plant selection [66]. Different species have different tipping points as they have different vulnerability to water stress (exacerbated in droughts and where there is no supplemental irrigation) and to anaerobic soil conditions (linked to higher water tables or higher rainfall, thus decreasing the air-filled volume in soils). Tipping point is also influenced by the stresses plants experience as they grow, especially long-lived species such as trees – plants with larger and deeper root systems are more resilient to drought (R Simcock pers comm). However, tipping points and changes in species occur in crops where compacted soil restricts root systems, so smaller root systems have access to a smaller soil water volume, and reflected in plant indicators [98]. Areas that are compacted or degraded, especially in wetter hollows, have lower aeration and/or slower drainage reflected as poorer crop emergence, stunted plants, more root diseases, increases in species (often weeds) that are more competitive [67, 99] (R Simcock pers comm).

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

There are 'lag times' for the recovery of compacted soil, which also depends on the extent of deliberate intervention to remediate compactions, but more importantly the natural processes that help rejuvenate shallow soil (cracking, roots, worms). Without intervention, there can be natural recovery – or reversibility - in shallow soil depths through soil cracking, drying, root and macrofauna activity, especially in surface soils where root and biological activity is abundant [41, 68]. A study in Southland showed that compaction at various soils depths typically occurred in Spring, recovery occurred in Summer due to drying and cracking of the soil, and winter (when stock are off-farm). However, compaction is much more difficult to remediate at greater soil depths (>15 cm depth) [41, 44, 45]. Deliberate interventions to remediate compaction in shallow soils include tillage while mechanical aeration (subsoiling, deep-ripping), and spot mounding, remediate compaction in deeper soils [48-50].

Impacts of compaction may only be seen under specific climate conditions because of its impact on water and air-filled pore volume; typically near-surface compaction effects are highlighted either when plants are actively growing under wet conditions, reflected as nitrogen stress or spread of root diseases – with young drops most vulnerable. In contrast, deep compaction that limits root volume is revealed during climate cycles that induce drought stress or have unusually strong storms, and/or when trees reach specific heights; in this case long-lived crops and ecosystems can be highly vulnerable to toppling or failure [69, 70].

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

As noted previously, soil health is an area of high interest to Māori and there are many tohu/indicators that are utilised according to mātauranga-ā-hapū and mātauranga-ā-iwi [94, 95]. In addition to discussing this attribute directly with iwi/hapū/rūnanga, there is likely to be tikanga and mātauranga Māori relevant to informing bands, allocation options, minimally disturbed conditions and/or unacceptable degradation in treaty settlements, cultural impact assessments, environment court submissions, iwi environmental management and climate change plans, etc.

Part C—Management levers and context

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

The relationship between attribute state (as measured by macroporosity and bulk density), land management pressures and potential management interventions is generally well understood, particularly for soils where soil compaction is a problem, hence the studies on such soil types. Some studies on, for example, pasture or crop yield relationships (response curves) to macroporosity etc are presented in [1, 11, 71]. For soil compaction and/or compaction with pugging, pasture damage occurs from treading in wet conditions [68, 72, 73]. Note that pugging tends to be more visibly damaging and is a separate process. There are five critical factors that influence the degree of soil compaction damage under livestock pastoral grazing [6]: inherent soil susceptibility to damage (strength); soil wetness (rainfall or irrigation); livestock loading (weight/hoof contact area); grazing management: intensity (animals/ha), duration (time on soil), and livestock movement (stationary or walking); and type/extent of vegetative cover.

On cropping farms, continued vehicle movement or ploughing on wet soil, can lead to soil compaction or a 'plough pan'. Soil compaction on cropping farms is also reviewed by [5], and there are numerous New Zealand studies on soil compaction [76, 77], and international studies [47]. Soil compaction occurs in plantation forestry especially associated with ground-based harvesting [7, 100]. Soil compaction can occur in urban areas and land development – primarily through use of heavy machinery [9, 34]. Compaction in vineyards is likely under the wheel track areas, but not under vines. Soil quality samples taken from Marlborough vineyards, showed compaction (low macroporosity) but the soil was much less compact under the vines [48, 78], but this has not been related to production.

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

C2-(i). Local government driven

Some regional councils produce brochures or information on soil management including compaction (e.g. GWRC, Waikato). Information on soil compaction may be less common. GWRC produced a brochure on soil compaction by the science and land management teams, 'Soil compaction and pugging on dairy farms'. I'm not aware of specific regulations etc for this attribute.

C2-(ii). Central government driven

I'm not aware of specific regulations etc for soil compaction. However, regulations were introduced for soil pugging (which can compact soil too) to protect water quality, and the related MfE-produced guidance material for farmers and consultants that is promoting the avoidance of soil pugging under forage crops [78-81]. There are recommendations to create soil profiles that have subsoil conditions (primarily relief of compaction) that allow trees to grow in urban environments [9].

C2-(iii). Iwi/hapū driven

We are not aware of other interventions/mechanisms being used by iwi/hapū/rūnanga to directly affect this attribute.

C2-(iv). NGO, community driven

The dairy industry and cropping industry have produced information on soil compaction (and pugging), and off-paddock grazing such as stand-off pads etc. Examples include [83, 84]. Since the introduction of intensive winter grazing regulations, pastoral industries have soil compaction /mitigation information on their websites.

A factsheet by MWLR is available: https://www.landcareresearch.co.nz/assets/Discover-Our-Research/Land/Soil-health-resilence/factsheet-compaction-pugging.pdf

AgResearch produced a booklet called 'Managing treading damage on dairy and beef farms in New Zealand: booklet for farmers and industry', (5000 copies printed) which was distributed to farmers to help avoid and mitigate soil compaction and pugging [84]. Various guides for farmers and material have been produced by Plant and Food Research to manage soil compaction and soil quality [85,86].

C2-(v). Internationally driven

Recent legislation on soil health has been proposed in the EU. The new law aims to address key soil threats in the EU, including compaction.

Part D—Impact analysis

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

Refer to A1 for context on ecological and environment affects.

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

Economic impacts are likely be felt by farmers directly, and industry where crop or pasture production is affected, but these farmers are also the people who have the direct ability to prevent or minimise compaction occurring, if it is practical to do so. There have been few specific economic studies, other than farm economic modelling that have been undertaken [37, 49, 87]. Studies that have assessed the agronomic benefit (in terms of crops and pasture yields) from remediation of compacted soil through subsoiling/aeration or restricting grazing duration [48, 88, 89], can of course be translated into economic benefits.

Economic impacts via environmental effects are difficult to determine on a larger scale, and few studies have attempted this, but some are available [5, 90]. Others have evaluated ecosystem services affected by soil compaction [91].

There is also impact of compaction in urban areas on flooding and flood risk (notably in urban catchments) where soils lose their sponginess, and health of perennial species and trees can be affected [9, 92].

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

Soil compaction is affected by climate through cracking in drying conditions, cracking in frosty conditions, but compaction is likely to get worse (due to susceptibility of soils to become compact under wet conditions), under increasing frequency of high soil moisture and rainfall events, and crop and pasture growth can therefore be affected [1, 5]. It is likely under wetter soil conditions, with treading and machinery present in wet conditions, there may be more soil compaction, including if there is greater extent of irrigation use [93]. In drier conditions, there may be some benefit from limited compaction in 'dry years' (vs 'wet years') as soil water can be more tightly held in compacted soils and therefore less likely to evaporate and therefore benefit crops [1]. However, where soil compaction limits root volume and rooting length, drought effects on plants are exacerbated. Here, deep compaction is a more severe issue than shallow compaction. Effects of soil compaction are likely greater in cities due to heat island effect increasing air temperatures, therefore increasing the moisture demand of vegetation (for cooling). Soil compaction that results in lower infiltration and/or soil water storage will result in more flooding under most climate change scenarios of higher-intensity rainfalls.

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7.10 Soil water storage, capacity, and fluxes

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Preamble: There are multiple measures for soil water storage, and capacity listed below, which are used for estimating water availability and storage for pastures and crops, irrigation scheduling, soil water management, and modelling in hydrological and plant production studies etc.

Much of the information below is written for a predominantly agricultural context because many of these indicators have been developed for and are used in agricultural contexts.

Field capacity (FC) is generally considered to be the water content after rapid gravity-driven drainage has effectively ceased, normally after 1–2 days, but this can vary between soils particularly in imperfectly and poorly-drained soils, and therefore, Available water capacity (AWC) can also vary [1, 2]. Field capacity is commonly measured at –10 kPa in the laboratory, but in some studies it can vary from –5 to –20 kPa, and in USA typically is at –33 kPa.

Permanent wilting point (PWP) generally considered to be the water content at which water is tightly held and unavailable to be extracted by plants, causing them to become permanently wilted [1]. This is nominally –1500 kPa although some plants (and with assistance of mycorrhizae) can access water >1500 kPa.

Available water capacity (AWC) and Profile available water (PAW). Describes the amount of water a soil can store. The profile available water is defined by New Zealand's digital soil mapping system, S-map, as the amount of water that can be stored in the soil profile and is potentially available to the plant for growth [1, 3]. It is generally considered to be between field capacity and permanent wilting point. According to [1], "PAW is an alternative to the more common term AWC. AWC is expressed as an equivalent depth (mm) or a percentage of soil volume or weight. Management such as cultivation and compaction can affect AWC. In New Zealand, PAW is commonly expressed as mm of water storage per 100 cm soil depth, or to 60 cm depth in irrigation scheduling or nutrient budget models, such as Overseer.". In the S-map factsheets, for example, PAW is expressed as 'mm' and is listed for 3 profile depth classes of 0–30, 0–60 and 0–100 cm.

Readily available water capacity (RAWC) and Profile readily available water (PRAW) generally considered to be the difference in water content between field capacity and the 'stress point', where a plant becomes stressed with a risk of growth limitation from drought stress [1]. RAWC is typically defined as the proportion of the soil water storage drained between the pressure levels of –10 and –100 kPa, but the stress point may vary depending on the crop [1, 2].

Gravimetric water content is the mass of water per mass of dry soil.

Volumetric water content is the ratio of the volume of water to the unit volume of soil reported on a volume basis.

Soil moisture deficit (SMD). NIWA provides modelled soil moisture deficit data: Soil Moisture Deficit (SMD) | NIWA. SMD is calculated based on rainfall, potential evapotranspiration, and a fixed available water capacity (this is assumed for all soil orders). SMD is useful for evaluating how dry the soil is, drought severity and for modelling.

Matric potential. This is equal to the potential energy due to the attraction of the solid matrix, and is equated to the absorption and capillary forces which hold water to soil particles [4]. It's a measure of pressure/suction (kPa) and gives an indication of how tightly soil holds onto water.

There are multiple measures for 'fluxes' including measures of:

Saturated hydraulic conductivity, Ksat. Typically measured using approx. 10 cm diameter undisturbed soil cores, and adequate replication is needed as Ksat can be quite variable. Small cores can have artefacts so modelling approaches can also be used [5]. Further details for the various flux measurements are available in [6]. Field measures are also used with results influenced by size of ring, whether single or double rings are used, and the ponding depth of water. For measures for stormwater, engineers use different methods which are generally falling head using single rings (R Simcock pers comm).

Unsaturated hydraulic conductivity, Kunsat. Typically measured using approx. 10 cm diameter undisturbed soil cores, at a range of matric potentials. For example, Manaaki Whenua – Landcare Research has cores equilibrated to −0.1, −0.4, and −1 kPa tensions [7] to measure unsaturated flow in several pore sizes. Typically, unsaturated hydraulic conductivity is much less than saturated hydraulic conductivity, and variation is also less as the influence of macropores is reduced.

Infiltration. Manaaki Whenua – Landcare Research has recently developed a field infiltrometer, but some aspects are still being developed and tested. More info and photos are available here: https://www.landcareresearch.co.nz/publications/soil-horizons/soil-horizons-articles/a-farmer-friendly-infiltrometer/

The relationships describing hydraulic conductivity and water retention (soil water storage), as functions of matric potential are two key attributes to describe a soil's hydraulic character [8]. Internationally, other terms can be used such as in Australia for drained upper limit (DUL) in place of field capacity, and crop lower limit (CLL) in place of PWP [9]. Similarly plant available soil water capacity (PAWC) can be used in place of RAWC [10].

State of knowledge of the "Soil water storage, capacity and fluxes" attribute: Good / established but incomplete – general agreement, but limited data/studies.

The multiple measures for soil water storage, capacity are generally well studied, but usually for research purposes rather than for monitoring per se. Soil water storage and capacity are generally used for irrigation scheduling, soil water management, and modelling soil water balances such as in modelling of plant production or farm management effects. However, fluxes such as Ksat, Kunsat, and infiltration are used less often in NZ.

Part A—Attribute and method

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

This attribute relates to ecological integrity but not to human health. Soil water is critical for ecosystem functioning as it provides water for plants, transports nutrients, enables nutrient cycling processes and (microbial) life cycles based in soils. Soil water is important as it influences soil properties including aeration (air movement), temperature and consistence [2]. How much water a soil can store (how big is the bucket) within soil pores depends on several soil properties particularly texture (hence the arrangement and size of soil pores) and organic matter, but also soil management [1, 2]. For a soil profile, taking into account the depth of the soil is also important. How much water a soil actually holds at any one time (how much is in the bucket) depends on these factors but also plants, their transpiration, climate (drought vs rain, evaporation), whether irrigated or not etc [2]. Ecosystems develop depending on the ecoclimatic zone, which is in part due to soil water availability. The status of soil moisture affects ecohydrological processes such as runoff, infiltration and evaporation and plant function such as through transpiration and photosynthetic rate [11]. Plants also affect soil moisture and dynamics through the plant's involvement in the water cycle such as root uptake, transpiration etc [11].

This group of attributes has a wide range of indicators, so some caution is required when considering them. It is important to realise that some, such as AWC and RAWC, etc can be slow to change, and are typically considered to be quite stable because they are related to physical properties and arrangement of the soil (but they can change with soil management). PWP is unlikely to change much at all as it is very depending on soil texture. In contrast, soil water content is a measure of water in the soil at any one time, and it can rapidly change with evaporation, irrigation, rainfall, soil management etc.

How water flows (water fluxes) depends on soil properties particularly pore arrangement and connectivity, and where sufficient water is available, water flow may influence movement of nutrients and contaminants through soil and into groundwater. For agriculture, soil water content can be increased by irrigation where needed.

Measurements of soil water storage (e.g., AWC, RAWC) have been used for understanding soil water balances for agricultural and agronomic purposes and pasture/crop/nutrient uses and dynamics (including modelling) [11-13], runoff or leaching losses [14, 15], and for farm irrigation scheduling. Measurements of soil water storage allow an informed approach to farm irrigation water use and the optimisation of plant yield [11, 16, 17]. Direct use of RAWC and AWC can be made for evaluating water storage for irrigation scheduling – but usually in conjunction with software or decision support tools e.g., [19]. Available water capacity (AWC) is important for accurately simulating (modelling) crop yield in dry conditions and under irrigation [16, 19]. Knowledge or representation of these measures of soil water storage is also important for modelling water and diffuse nutrient losses from agriculture.

Unsaturated hydraulic conductivity measurements data are scarce [21]. Hydraulic conductivity measurements of soils have been used to improve modelling of soil drainage and pasture yield [21].

Many of these water storage and flux attributes are also of critical importance for developing modelling, pedo-transfer functions (to predict soil water storage e.g., RAWC, AWC, PAW and transport e.g., Ksat and estimates and the uncertainty) associated with S-map [22-24] (NZ's digital soil map). Significant efforts have gone into improving modelling with these attributes within S-map in the last 10 years with significant sampling campaigns to improve data from a range of soil orders. For example, in recent years S-map has relied on a pedotransfer function developed in 2014. A major effort had been made to substantially increase the amount of measured data from which the model is derived; further information is available from:
https://www.landcareresearch.co.nz/publications/soil-horizons/soil-horizons-articles/new-water-retention-model-in-s-map/

Similarly, a significant effort has been made in recent years to significantly improve the hydraulic conductivity model computed from water retention curves for a range of New Zealand soils [5, 25, 26], thereby improving estimates of AWC, PAW, Ksat etc for a wide range of New Zealand soils – and therefore also improving environmental and farm systems modelling, e.g., Overseer nutrient budgeting, and in the Agricultural Production Systems Simulator (APSIM).

More recently, a new pedotransfer function for S-map was developed in 2024 [27], and a webinar gives latest information for NZ soils as at April 2024 on "Improving information in S-map - Important updates on soil water storage characteristics"
https://www.landcareresearch.co.nz/events/linkonline/

Water storage indicators (RAWC, AWC) and saturated hydraulic conductivity are also used in stormwater / flood modelling to model performance of catchments and different water retention and detention devices [28] and living roof substrates [29]. Soil water storage in specific urban stormwater treatment devices is fundamental to their performance – most critically in greenroofs/living roofs. Auckland Council developed a greenroof media that has an available water storage to meet stormwater quality targets when placed at a depth of 100 mm [30].

Finally, regarding human health, soil stores water, so there may be an indirect pathway for nutrients and contaminants that affect human health to travel when water moves through soil.

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

The ability of soils to store water for ecological functioning differs with their properties, different soil types and spatially in the landscape. Dominant factors influencing the various indicators of soil water storage capacity for ecological functioning are soil texture, presence of stones, pore sizes and pore arrangement. Soil type influences texture and stones, but both soil type and management of soil influence pore sizes and their arrangement. Managing soil water availability in soils with low AWC is challenging (e.g., in shallow, stony soils).

In agriculture, more specifically under irrigation, the spatial extent and magnitude of degradation of water storage properties is not well known for modern sprinkler irrigation and farm grazing systems, with some studies conducted only recently [1, 17]. However, there have also been a number of studies under older border-dyke irrigation systems, reviewed in [1]. An example is a study across Canterbury, where the effects of irrigation on soil physical properties under pastoral grazing were evaluated. Under irrigation there was a shift towards a greater abundance of smaller soil pores, thus reducing readily available water capacity to pastures under irrigation than under no irrigation [17].

When there is limited water storage it is important that as much capacity as possible is preserved through good soil management on farms [31]. Knowing more about the attribute would help to better manage irrigation and water use, as would knowledge of their spatial variability across the landscape. Improved knowledge on a paddock scale could be combined with soil mapping and use of variable rate irrigation, measurements of soil physical properties in the lab, sensor-based technology for soil or plants, and soil-water balance irrigation scheduling etc, which can all improve crop yields and water use efficiency [12, 20, 31, 32].

Flood risk is increased where soil water storage is reduced. In urban and rural context, reduced storage is associated with loss of organic matter, compaction and/or reduction of soil profile depths through earthworks [33].

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

Significant changes in New Zealand in the last two decades include a 10% expansion in urban areas between 1996 and 2012 with a loss of high-quality soils for food production [34]. Areas of irrigation have expanded, often resulting in dairying and arable cropping expansion [1, 35]. The area of irrigated agricultural land increased by 91% between 2002 and 2019 [35].

The pace and trajectory of change and impacts associated with these land use changes over time for this soil attribute however is largely unknown. In agriculture, research of soil water storage in soil types in border-dyke irrigation systems was undertaken in the 1960s and 1970s where agricultural development, climate and irrigation on soils were evaluated [1], or more recently under modern irrigation systems but typically this has been only a single point in time or for short time periods [16, 31].

Rapid changes have occurred in urban areas – most soils having reduced AWC through compaction and profile truncation. However, specific anthropic soils are specified for stormwater devices and tree pits that have increased AWC through provision of deeper profiles, organic amendments to topsoils, and use of high AWC components (e.g., pumice) [33].

Some impacts are reversible. Common techniques increase water held by adding amendments (organic materials, biochar, pumice, clays), increasing topsoil depths, or, where rooting depths are limited, using imported soils or physical amelioration such as ripping or subsoiling. This is often done in combination with increasing infiltration rates and slowing water runoff to maximise the rate at which water enters the soil profile (i.e., reducing runoff). Such techniques are core to regenerative agriculture and urban stormwater management. Adding municipal compost has been shown to increase water storage pores [37].

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

Several councils (Auckland, Waikato) have invested in soil moisture content monitoring [38, 39] for the benefit of farmers and council. Auckland Council indicate for their monitoring, "The purpose of this network was to assist with agricultural farm management and for drought information purposes" [39]. A National Environmental Monitoring Standard (Soil quality and trace elements. Sampling, measuring, and managing soil quality and trace element data. Version 1.0.0) [40] has been developed. However, regional council methods for routine monitoring of soil water content vary considerably [38-39].

For example, Auckland Council established their soil moisture monitoring network in 2014 with ten soil moisture sensors (aquaflex) installed across the Auckland region between 2014 and 2016 and in accordance with the National Environmental Monitoring Standard [39]. Several other councils (e.g., Hawke's Bay, Otago) have information available on their websites. Southland Regional Council also monitors soil water. These councils monitor soil water content on a routine basis. My understanding is that GWRC has monitored soil moisture using small in-situ probes, but this information may be out of date.

There is no routine monitoring of soil water capacity or fluxes e.g., Ksat, that I'm aware of.

NIWA provides modelled soil moisture deficit data: Soil Moisture Deficit (SMD) | NIWA. Soil moisture deficit is reported on their web page with historical soil moisture deficit, the soil moisture deficit at the same time last year, and the current soil moisture deficit.

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

For direct soil measurements for many of these attributes, or for ongoing soil water content monitoring, there would be a need to access privately owned land. Landowners may be more, or less, willing to provide access to land for sampling and to have data from their land used to inform monitoring.

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

A key cost will be staff time to undertake sampling, with additional staff required for interpretation and reporting etc.

AWC, RWAC, FC, PWP etc are sampled using undisturbed cores using lab-supplied stainless-steel rings, i.e., not 'bulked' soil samples. The Manaaki Whenua – Landcare Research physics laboratory routinely does these as part of a range of measurements, generally for research purposes. Further information is available from: https://www.landcareresearch.co.nz/partner-with-us/laboratories-and-diagnostics/soil-physics-laboratory/

Current costs for RAWC and AWC from the soil physics lab at Manaaki Whenua – Landcare Research are approx. $110 per sample.

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

Yes. Ngāi Tahu Farming is involved in a soil health project with AgResearch, where field capacity (−10 kPa), available water capacity and wilting point (−1500 kPa) were measured [41]. There may be other monitoring being carried out by representatives of iwi/hapū/rūnanga that we are unaware of. As land owners and farmers, aspects of this attribute are likely to be of high interest to Māori.

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

Soil properties that influence soil water storage, capacity and flux were described in sections A1 and A2. Attributes that relate to some of those properties are: soil carbon, soil compaction, erosion attributes, peat soil subsidence, wetland extent, and soil nutrients. Other factors that may influence soil water content are climate and weather events and catchment properties, attributes relating to those factors are catchment permeability, surface water flow alteration, groundwater depletion.

There is an established relationship between size ranges of soil water storage pores, matric potential, and therefore how tightly soil water is held [1, 8]. There is a relationship with the attribute macroporosity, in terms of the soil matric potential (tension) commonly used to define the pore size boundary (–10 kPa) that is also typically the upper boundary for AWC and RAWC. The relationships describing hydraulic conductivity and water retention (soil water storage), as functions of matric potential are two key attributes to describe a soil's hydraulic character [8].

Part B—Current state and allocation options

B1. What is the current state of the attribute?

We don't know the current state of many indicators of this attribute, namely AWC, RAWC, FC, PWP, and fluxes (Ksat, Kunsat), as they are not routinely monitored. Information and data on these attributes has typically been gained through research studies include characterising soil water storage attributes under land uses, land management practices, characterising soils etc [5, 16, 19, 22-25, 37-39], and farm or crop system studies [11, 19, 29, 40]. Other studies use these attributes or estimate them for a range of uses [41, 42]. Soil water storage attributes like AWC are linked from S-map to models such as Overseer and APSIM. Research has been conducted in NZ to evaluate more cost effective methods [48].

In contrast, soil water content (which can change rapidly with weather events) is monitored by several regional councils as described in section A4-(i). However, this monitoring of soil water content has limited coverage within regions. Monitoring soil water content is useful for specific situations, e.g., irrigation and effluent application for agriculture, growing pasture and crops etc. However, it can change rapidly so may not be relevant/suitable as national scale indicators or attributes, but that is dependent on the purpose and scale of the intended monitoring.

NIWA provides modelled soil moisture deficit (SMD) data on their web page: https://niwa.co.nz/nz-drought-indicator-products-and-information/drought-indicator-maps/soil-moisture-deficit-smd. SMD is calculated based on rainfall, potential evapotranspiration, and a fixed available water capacity (this is assumed and is not monitored). SMD is useful for evaluating how dry the soil is, drought severity and for modelling. This modelled attribute would be useful as a national attribute, but currently it is too coarse, so finer resolution and using soil type (soil sibling) data from S-map would be required. Other refinements may also be required.

Measuring AWC, RAWC, FC, and PWP are useful for specific situations, e.g., irrigation and effluent application for agriculture, growing pasture and crops etc, but are unlikely to change rapidly (depending on circumstances described in A1), so are unlikely to be relevant/suitable as national scale indicators or attributes, but that is again dependent on the purpose and scale of the intended monitoring.

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

The concept of reference states does not seem particularly relevant for the practical use of this attribute in agriculture, but has been applied in the urban environment. For example, the concept of reference states is relevant for the practical use of this attribute in a) the context of urban soils, where stormwater quality and flood volumes in areas that will be urbanised are modelled using 'pre' and 'post' development that uses soil water storage (with 'before' being the reference state) (R Simcock pers com) and b) in the context of soil rehabilitation (with the non-depleted, high-organic matter soil being the reference state) [49-51].

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

The following presents existing numeric or narrative bands described for this attribute 'state'. These are not related to regulation or any specific management, but in general are related to plant growth and the availability of water that can be used by plants during dry periods (more water storage means greater resilience to drought). In terms of regulation and guidance, good practice for dairy effluent application is typically using application depths of 10 mm or less [44], but the current attribute is not typically specified in council regulations.

The MWLR S-map factsheets present modelled information for the public for example 'profile available water' as bands: "PAW of 100 mm implies that 10% of the soil volume is water available to plants. Low PAW is <60 mm, moderate is between 60 and150 mm, and high is ≥150 mm." Source: https://smap.landcareresearch.co.nz/support/glossary#profile_available_water_paw.

Similarly, MWLR S-map factsheets present modelled information for the public for example 'permeability of slowest horizon' as bands, "The permeability of the slowest permeable layer of the soil. As well as being expressed as 'slow', 'moderate' or 'rapid', this is also expressed as the movement of water in millimetres per hour.". Source: https://smap.landcareresearch.co.nz/support/glossary#permeability_of_slowest_horizon.

Some base information/rating systems for permeability classes including those used in S-map are found in [52]. This is expressed at the soil family level, and there are more permeability classes at the Functional Horizon level – these are documented in the S-Map manual (L Lilburne pers comm). The bands for permeability classes used at the soil family level in S-map are 'slow' <4mm/h, 'moderate' ≥4 to<72 mm/h, and 'rapid' ≥72 mm/h (L Lilburne pers comm).

A number of classes, bands or ratings are available for available water capacity, readily available water capacity, profile available water and soil permeability for NZ soils, and these are explained with further details in [4], with some of these band values copied in the images below.

Available-water capacity

These value are suitable for subsoils or profile average values, but are rather low for topsoils.

	%
Very high	>20
High	15–20
Moderate	10–15
Low	5–10
Very low	<5

Readily available water capacity

	%
High	>10
Moderate	5–10
Low	2–5
Very low	<2

Profile available water

Based on a potential rooting depth of 100 cm or actual rooting depth, whichever is the lesser.

Storage class	*mm of water*
Very high	>300
High	250–300
Moderate–high	200–250
Moderate	100–200
Moderate–low	50–100
Low	25–50
Very low	<25

Internationally, other terms can be used such as in Australia for drained upper limit (DUL) in place of field capacity, and crop lower limit (CLL) in place of PWP [9]. Similarly plant available soil water capacity (PAWC) can be used in place of RAWC [10].

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

Yes. Specific effects on ecological integrity occur when soils are changed to the extent that AWC is reduced and cause a) increased flooding or b) reduced resilience to drought. This can occur with changes to rooting depth or to changes in volume of water held per unit of soil (most commonly reduced organic matter) (R Simcock pers comm).

Tipping points result in changes in relative competitiveness of different species, e.g., from kiwifruit or avocado (intolerant of waterlogging/wet soil) [53] to pipfruit (more tolerant) (R Simcock pers comm). There are also differences in tolerance to drought and waterlogging between cultivars in grapes and apples, with drought stress reducing photosynthesis, and transpiration rate [54]. Soil drainage, aeration and water storage properties should be used to guide plant selection [55]. Different species have different tipping points as they have different vulnerability to water stress (exacerbated in droughts and where there is no irrigation) and to anaerobic soil conditions (linked to wet soils). Tipping point is also influenced by the stresses plants experience as they grow, especially long-lived species such as trees – plants with larger and deeper root systems are more resilient to drought (R Simcock pers comm). Areas that have wetter hollows, have lower aeration reflected as poorer crop emergence, stunted plants, more root diseases, increases in species (often weeds) that are more competitive [56, 57] (R Simcock pers comm).

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

Legacy effects include peat drained and subsiding soils [58,59], so potentially they could be losing water storage capacity. This will have links to the peat subsidence attribute.

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

As noted previously, soil health is an area of high interest to Māori and there are many tohu/indicators that are utilised according to mātauranga-ā-hapū and mātauranga-ā-iwi [67, 68]. In addition to discussing this attribute directly with iwi/hapū/rūnanga, there is likely to be tikanga and mātauranga Māori relevant to informing bands, allocation options, minimally disturbed conditions and/or unacceptable degradation in treaty settlements, cultural impact assessments, environment court submissions, iwi environmental management and climate change plans, etc.

Part C—Management levers and context

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

The effects of management were briefly described earlier (sections A2, A3). For example, the effects of irrigation on soil physical properties under pastoral grazing suggests a reduction in readily available water capacity in soils under irrigation than under no irrigation [17]. AWC and the other attributes are affected by soil texture, structure, soil depth and layering, organic matter, and stone content, and also management such as cultivation and compaction.

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

C2-(i). Local government driven

No. As indicated previously, some councils monitor soil water content, but it is unknown if the information is used for regulatory purposes. Auckland Council indicate for their monitoring, "The purpose of this network was to assist with agricultural farm management and for drought information purposes" [39]. Auckland Council has developed a guidance for flood modelling and stormwater quality device design [60].

C2-(ii). Central government driven

We are not aware of interventions/mechanisms being used by central government to directly affect this attribute.

C2-(iii). Iwi/hapū driven

We are not aware of interventions/mechanisms being used by iwi/hapū/rūnanga to directly affect this attribute.

C2-(iv). NGO, community driven

We are not aware of NGO or community driven interventions/mechanisms being used to affect this attribute.

C2-(v). Internationally driven

We are not aware of any internationally driven interventions/mechanisms being used to affect this attribute.

Part D—Impact analysis

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

Not managing soil water content, water use through irrigation etc is likely to lead to water over use, wastage of water, greater nutrient losses, and less than optimum management of pastures and crops for production [61-64].

As discussed earlier, adequate AWC, RWAC, etc, along with good management of these, are valuable for managing crops, pasture etc, and associated environmental and production modelling is important [12]. These will become increasingly important for climate change resilience.

In cities, increased flooding and increased water use for garden irrigation occurs where soil water storage is reduced [65]; increased urban vegetation drought stress leads to slower growth and a narrower range of species, with less temperature moderation (links to climate change) [33].

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

Economic impacts are likely be felt by farmers directly, and industry where crop or pasture production is affected, or where irrigation water is rationed. I'm not aware of economic studies.

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

Arrangement of soil pore space, and thus ability to store water can be affected by climate (and therefore climate change) via drying and soil wetness, and so crop and pasture growth can therefore be affected by climate change. It is likely under wetter conditions, from increased storm intensity or frequency under climate change, there may be more soil compaction, thus affecting RAWC and the other water storage attributes indirectly. Climate change is likely to directly affect soil carbon and soil compaction [66] (and therefore RAWC, AWC etc. indirectly).

In cities, climate change is likely to increase flooding in more intense, frequent high rainfall, and in drier conditions increase temperatures resulting in vegetation drought stress, such as for canopy trees [33].

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7.11 Soil contaminants

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Alternative attribute name: Trace element contaminants in (residential) soils

Preamble: Trace elements occur naturally in the environment with concentrations in soils dependent on geological and pedological processes. There are many anthropogenic activities that resulting in the release of trace elements to soil including various industrial and agricultural processes, fossil fuel combustion, waste disposal process and residential use.

The term 'heavy metals' is most often intended to be used to describe metal *and* metalloid elements that are considered to cause deleterious effects. However, this is a technically inappropriate term as not all metals and metalloids are heavy nor do they all cause deleterious effects – the latter is more dependent on the concentration at which the metal and metalloid elements occur. For example, zinc and copper can be considered heavy metals but they are also essential micronutrients required for plant and animal growth. At low soil concentrations they may be deficient for some plant and animal species, while at higher concentrations can become toxic.

A more appropriate term might be trace element contaminants, to encompass both metal and metalloid elements and providing a focus on when the trace elements may become contaminants/present as elevated concentration that results in deleterious effects.

The common suite of contaminant elements includes arsenic, cadmium, chromium, copper, lead, nickel, mercury (variably), zinc.

This information stocktake was constrained to a focus on metal and metalloid (trace element) contaminants in residential soils.

State of knowledge of the "Soil Contaminants" attribute: Good / established but incomplete – general agreement, but limited data/studies. Reasoning is that it is established that elevated trace element concentrations can occur on residential soils, but there isn't comprehensive coverage of residential soils across. There is excellent/well established information (globally) that elevated concentrations can cause human health or environmental effects but there poor / inconclusive information that trace elements in residential soils have negatively impacted on human health or ecological receptors.

Part A—Attribute and method

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

Heavy metals in residential soils relate to both ecological integrity and human health. There is comprehensive discussion and assessment of the effects of a range of 'priority contaminants' for human health in [1, 2], and ecological receptors in [3-5] and references therein. These documents provide an overview of the effects of different trace element contaminants on people [1] and soil biota (plants, microbes, invertebrates) [3] and establish guideline values to indicate concentrations to indicate concentrations at which exposure might need to be managed to avoid negative effects [2, 4]. Further considerations in relation to both human-health and ecological integrity for soil-derived contaminants is the potential for leaching into groundwater and movement into surface water systems.

There are many factors that influence the actual effect of the different trace elements– this includes the way in which people or ecological receptors are exposed to the trace elements, and the bioavailability of the soil-associated contaminant. Bioavailability in turn is influenced by soil properties such as pH, soil carbon content, cation-exchange capacity [e.g., 6] and can also be directly assessed through different extraction processes designed to simulate release of trace elements [e.g., 7] from ingested soil particles, or weak-acid extraction to provide an indication of the availability of trace elements to plants [e.g., 8]. However, the specific bioavailability to plants and other soil organisms will also differ between species, and different species have different sensitivities to different trace element contaminants [e.g., 3].

Arsenic and lead are typically of greater human health concern i.e., soil contaminant standards for protection of human health are lower than ecological soil guideline values, while copper and zinc are typically of most concern for ecological receptors (in soils and water)[1,3].

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

There is evidence of soil contamination from a plethora of site investigations that have been undertaken for the purposes of managing contaminated land under the National Environmental Standard for Assessing and Managing Contaminants in Soil to Protect Human Health (NES-CS) [9]. These investigations occur on sites where an identified Hazardous Activity or Industry [10, 11] is likely to have occurred and typically where subdivision or a change in land use is proposed. These investigations determine the extent to which remedial action is required to inform its future use. Where remediation has been undertaken, site validation reports are to ensure remedial action has been effective to make the site 'safe' for use – in terms of protection of human health. While there is a regulatory requirement under RMA section 31 for territorial authorities to prevent or mitigate any adverse effects of the development, subdivision, or use of contaminated land', the existence of the NES-CS, which is for the protection of human health, and the general expectation that environmental issues fall into the remit of regional councils there is effectively no regulatory requirement to consider protection of ecological receptors [12] although this may vary between different councils, and some consultants will also consider this. The individual reports are held by different councils with no ready ability to collate the data.

As noted, the NES-CS only applies at the point of subdivision or land use change – with land likely to become residential property relevant for this attribute. Specifically, the NES-CS doesn't allow for the assessment of land not undergoing change, but which may be contaminated i.e., has been identified as a property on which a Hazardous Industries and Activities List (HAIL) activity has, or may have, occurred [12]. Regional Council have the responsibility to undertake the investigation of land for the purposes of identifying and monitoring contaminated land (RMA 1991 s.30(1)(ca)). However, further site investigation is required to determine if the site actually has concentrations present at concentrations of concern.

Beyond this, regional council state of the environment monitoring, which includes assessment for trace element contaminants, provides information on concentrations in agricultural land, which may then be sub-divided for residential use. A particular issue is associated with the sub-division of agricultural land, which may have cadmium concentrations above the rural residential soil contaminants standard [2, 13].

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

This is unknown as there has been no regular monitoring over time. In the context that there is greater awareness of the potential for contamination associated with various anthropogenic activities, and that resource consenting processes are intended to prevent soil from being contaminated, then future contamination of residential soils should be minimal and it is more a case of managing legacy contamination.

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

No regular monitoring of this attribute is currently undertaken in New Zealand, although as noted in A2 there are numerous site investigations that have been undertaken for the purpose of managing soil contamination for the protection of human health, but there is a gap for residential properties that don't trigger the NES-CS.

There is guidance in contaminated land management guideline #5 [14] for undertaking site investigations, although this doesn't specify methods used for analysis of soil. However, the National Environmental Monitoring Standard for soil quality and trace elements [15] does specify the standard method for determining total 'recoverable' concentrations. Some information on bioavailability testing relevant for human health is provided in [16].

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

There would undoubtedly be issues as access to private properties would be required to fully assess the state of this attribute. However, under the current regulatory regime there is a gap in the requirement to assess residential properties that are not undergoing sub-division or land use change [12].

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

The cost of analysis of a soil sample for a trace element 'contaminant' suite, is generally around $80-$90 per sample based on total recoverable concentrations.

A simulated gastric extraction of arsenic and lead is also commercially available at Hill Laboratories for around $160 / sample with an additional set-up fee. This test gives some indication of the bioavailability of these trace elements from ingested soil.

Some bioavailability testing relevant for plants is available commercially, although these tests (e.g., melich-3, edta extractions) are typically undertaken to inform trace element fertiliser rather than assessing potential toxicity.

Further costs are incurred through the collection of samples – which will be determined by the number of sites, how sampling is undertaken at each site, including the number of samples and the depth of samples.

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

Māori have high interest in rehabilitating or remediating areas and sites considered contaminated or degraded [3]. However, we are not aware of specific monitoring being carried out by representatives of iwi/hapū/rūnanga.

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

There is some relationship of trace element concentrations in residential soils to other soil domain attributes, specifically:

- Soil carbon,
- Contaminants in surface water, groundwater, sediment and air, and
- Erosion.

However, the nature of these relationships is highly variable depending on the individual trace elements of concern, the specific location and soil characteristics. The bioavailability of contaminants in soils depends on various soil properties including soil carbon content. The movement of soils with elevated concentrations into aquatic systems e.g., through erosion, or the leaching of contaminants into groundwater, can contribute to elevated concentrations in water and sediments. Resuspension of the contaminated soils (e.g., those containing lead, arsenic or cadmium) could contribute to their presence in ambient air.

Part B—Current state and allocation options

B1. What is the current state of the attribute?

The current state is partially known. As noted in A2, there are a multitude of site investigations and site validation reports for land that is subdivided and/or will become residential properties. These sites will be 'safe' for people as required by the NES-CS, although risks to ecological receptors may still arise. As also noted in A2, there is limited knowledge of the state of residential soils that haven't undergone land use change, but where a Hazardous Activity or Industry has, or may have occurred.

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

Yes and no. Naturally occurring soil concentrations provide one form of a natural reference state and have been determined at a national level [17]. However, a focus on background soil concentrations in the context of managing soil contamination is a significant driver for the excess disposal of soil to landfill [12, 17-18]. Rather than focussing on background concentrations, more attention should be directed to managing the risk associated with elevated soil contamination, which includes using additional information such as the bioavailability of contaminants, to refine risk assessments [16].

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

Yes. Soil contaminant standards have been developed for the protection of human health in the New Zealand regulatory regime [2]. Similarly, ecological soil guideline values have also been developed for use in a New Zealand context [4]. These soil guideline values provide useful information to help screen the potential risk associated with contaminant concentrations but further information, including the use of bioavailability assessment should be used to inform appropriate management approaches.

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

From toxicological data there are various thresholds that have been identified as leading to different effects on people as well as ecological receptors [1, 3], and leading to the development of soil guideline values that can be useful to screen for potential risk to human health or environment [2, 4].

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

Yes. Some of the observed contamination is a result of historical activity, however, new contamination may arise from improper disposal of waste products e.g., legacy wastes, ongoing application of fertilisers (e.g., cadmium), fungicides (e.g., copper), and animal remedies (e.g., zinc for facial eczema treatment), disposal of wastes to land (e.g., wastewater application to land) e.g., [11]. A key point to note is that ongoing application of trace elements, which do not degrade, will lead to accumulation in soils [e.g. 13, 19], potentially to concentrations where negative effects may be observed.

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

Māori have a keen interest in being involved in the management and rehabilitation of contaminated or degraded land [4, 16]. In addition to discussing this attribute directly with iwi/hapū/rūnanga, there is likely to be tikanga and mātauranga Māori relevant to informing bands, allocation options, minimally disturbed conditions and/or unacceptable degradation in treaty settlements, cultural impact assessments, environment court submissions, iwi environmental management and climate change plans, etc.

Part C—Management levers and context

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

Beyond naturally occurring concentrations of trace element contaminants, the primary factor influencing their concentrations in residential soils depends on how the land was used BEFORE it became a residential property e.g., [9,11]. This includes agricultural activities (e.g., cadmium, arsenic from sheep dips), industrial activities, and diffuse pollution such as lead from leaded petrol. See also section C2.

The significance of the concentrations present depends also on how the land is being used as residential property, and therefore the exposure pathways for residents and ecological receptors, e.g., whether vegetables and fruit-trees are being grown, whether chickens for egg production are present, whether it is high-density residential property, and therefore exposure to soil in minimised.

Some of these factors are considered in the current management of contaminated land, briefly outlined in C2.

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

C2-(i). Local government driven

C2-(ii). Central government driven

The management of contaminated land is required under the Resource Management Act (1991), and the National Environmental Standard for Assessing and Managing Contaminants in Soil to Protect Human Health (NES-CS) is the primary tool for managing trace element contaminants in residential soil, and more generally soil contamination. The NES-CS is supported by a User Guide [9], as well as a series of Contaminated Land Management Guidelines and other documents [14, 21-27] to support the assessment of, and management of information related to potential contamination. The Hazardous Industries and Activities List (HAIL) [10] provides a list of activities and industries that may lead to soil contamination, with additional guidance recently released to aid with the identification of HAIL sites [11].

However, the NES-CS only applies at the point of land use change, and thus doesn't allow for the assessment of land not undergoing change, and which may be contaminated [12]. Potentially contaminated sites may be identified where regional councils have undertaken city-wide assessments to identify HAIL sites (given the Regional Council responsibility to undertake the investigation of land for the purposes of identifying and monitoring contaminated land (RMA 1991 s.30(1) (ca)). However, further site investigation is required to determine if the site actually has concentrations present at concentrations of concern.

Another key gap of the NES-CS is that it only applies to the management of soils for the protection of human health rather than also considering ecological receptors or aquatic systems (groundwater, surface water). However, many consultants and councils are starting to incorporate ecological soil guideline values in their detailed site investigations.

C2-(iii). Iwi/hapū driven

Iwi planning documents such as Environmental Management Plans and Climate Change Strategies/Plans may contain policies/objectives/methods seeking to influence managing contaminant outcomes for the benefit of current and future generations.

C2-(iv). NGO, community driven

C2-(v). Internationally driven

Part D—Impact analysis

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

Not managing this attribute would likely lead to environmental and human health impacts. However, focussing only on residential soils is likely to not be an effective means of managing soil contamination or the associated environmental and human health risks, as assessing residential land is effectively managing the symptoms, not the cause i.e.. contamination on residential properties is more often a result of legacy land use for previous non-residential purpose.

Perhaps a notable exception is lead. The most common source of lead contamination on residential properties is lead-based paint [27]. The management of the risk associated with lead also requires an assessment of the internal sources of lead e.g., internal lead-based paints, and the residential lead working group has highlighted the need for a cross-agency approach to the management of this issue [28]. Some residential activities such as the disposal of wood-ash, including wood-ash from the burning of treated timber, can give rise to hot-spots of contamination on a residential property.

For soils that are currently not contaminated, the most effective way of managing soil contamination is to minimise the accumulation of trace elements to concentrations that can cause negative effects, though managing any ongoing inputs to soil e.g., use of copper fungicides, zinc for facial eczema treatment [4].

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

The greatest economic impact associated with this attribute is arguably on residential property prices. The perception of contamination – arising from the identification of residential that has previously been used for hazardous activities and industries–gives rise to concerns around the potential impact on property prices although there is no evidence of this risk being realised. Property price impacts are the concern most commonly raised in public meeting when regional councils have undertaken city-wide HAIL assessment; concern around the potential human health effects arising from potential soil contamination are less discussed.

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

The effects of climate change on this attribute are unclear. Increasing storm and flood events could result in the wider movement of soil that contains elevated concentrations of trace elements. At a more subtle level, changes in temperature and moisture regimes could influence changes in the bioavailability of the trace elements.

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8 Freshwater Domain

Seven attribute information stocktakes for the Freshwater Domain are provided in sections 8.1 to 8.7, below. Dr Graeme Clark (MfE Domain Expert), Dr Fleur Matheson (NIWA, Domain Leader), and the Māori environmental researcher panel reviewed these sections.

8.1 Riparian margin establishment/protection

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State of knowledge of the "Riparian margin establishment/protection" attribute: Good / established but incomplete – general agreement, but limited data/studies

Part A—Attribute and method

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

The establishment and protection of riparian margins by fencing and planting with a combination of trees, shrubs, sedges, and grasses can benefit the ecological integrity and human health of adjacent waterbodies in several ways:

- by reducing contaminant losses in surface runoff and shallow subsurface flows from land, including sediment, nitrogen, phosphorus (McKergow et al. 2022) and *E. coli* (Collins et al. 2004).
- by stabilising waterbody banks (Marden et al. 2005; Simon et al. 2023; Stantec, Auckland Council 2023) and improving bank condition (Hughes 2016);
- by shading waterbodies (Rutherford et al. 2021) to reduce water temperatures (Rutherford et al. 1997) and nuisance growths of aquatic plants and algae (Matheson et al. 2017);
- by providing physical habitat for aquatic (Parkyn et al. 2003, Jowett et al. 2009, Hickford and Schiel 2011), semi-aquatic and terrestrial biota.

The extent of benefit afforded by riparian margin establishment/protection on ecological integrity or human health is spatially and temporally variable and depends on landscape, climatic and biological factors that affect the riparian margin, the land surrounding it and adjacent waterbody. Factors that affect performance include upslope land use and associated contaminant loading rates, rainfall, temperature, geology, land slope, soil type, hydrological flow pathways, the extent, soil and vegetation characteristics of the riparian margin, and the physical, chemical, and biological characteristics of the adjacent waterbody.

In catchments where fencing and planting of riparian margins have been carried out, the ecological integrity and human health of adjacent and downstream waterbodies is expected to improve, especially where the establishment and protection activities have been extensive. However, catchment or reach-scale studies that have monitored indicators of ecological integrity and human health in adjacent waterbodies before and after, or upstream and downstream of, fencing and planting of riparian margins have often reported inconclusive results, especially for water quality variables (Parkyn et al. 2003, Davies-Colley and Hughes 2020) or findings of improvements are correlative only (Wrightstow and Wilcock 2017, Graham et al. 2018, Graham and Quinn 2020). Interpreting the results of catchment and reach scale investigations of riparian fencing and planting is difficult given the influence of the many factors listed above which can vary spatially and temporally over the course of a study. Therefore, the strongest evidence of improvements afforded by riparian margin establishment and protection is likely to come from:

- investigations that limit the influence of confounding factors and/or quantify them to allow their influence to be accounted for in the development of relationships (e.g., inflow-outflow water quality studies to determine contaminant attenuation by riparian buffers; McKergow et al. 2020a);
- investigations that use models with a sound physical basis to determine relationships (models of radiation and sunpath that can incorporate riparian and stream characteristics to determine stream shade; Rutherford et al. 2021).

The resultant relationships can be combined in data-driven catchment modelling approaches to generate catchment scale predictions of impact (e.g., Matheson et al. 2021).

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

Indigenous vegetation in New Zealand was removed as land was cleared for settlement and farming (Ewers et al. 2006). This included removal of indigenous vegetation from the riparian margins of waterbodies particularly in lowland areas (Miller 2002). Loss of vegetated riparian margins will have contributed to the degradation of freshwater and downstream receiving environments by:

- enabling greater quantities of contaminants to enter waterbodies;
- lessening stream bank stability;
- reducing shading of waterbodies; and
- removing habitat for aquatic, semi-aquatic and terrestrial biota.

Beginning in the 1970s efforts to set-aside, fence, and plant riparian margins in New Zealand (and elsewhere; Mohan et al. 2022) began in sensitive lake and river catchments (i.e., Taupō and Upper Kaituna; McKergow et al. 2016). Early this century, a partnership between organisations in the dairy sector introduced national requirements for riparian margins to be fenced to exclude dairy stock from entering streams wider than a stride (>1m) and deeper than a gumboot (>30cm) (Fonterra Co-operative Group, Regional Councils, Ministry for the Environment, and Ministry of Agriculture and Forestry 2003). Recently enacted national stock exclusion guidelines (New Zealand Government 2020) require stock (beef cattle, dairy cattle, dairy support cattle, deer, or pigs) to be kept at least 3m away from waterbodies (wetlands, lakes, and rivers) >1m wide.

Regional councils have also provided advice and funding to assist landowners to set-aside, fence, and replant riparian margins and in some regions these activities have been extensive (e.g., Taranaki, Graham et al. 2018). Nevertheless, on a national scale, the available evidence suggests that the riparian margins of most waterbodies still lack protection, especially those of smaller streams (<1m wetted width). Where riparian margins have been fenced and planted, they typically have non-woody vegetation and narrow widths (<5m; Greenwood et al. 2012; Renouf and Harding 2015; Norris et al. 2020).

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

Removal of indigenous vegetation from riparian margins would have been a rapid process in those areas of New Zealand that were first and most extensively used for settlement and farming. As more of the country was settled, additional areas of vegetated riparian margin would have been removed, leaving only remnants, especially in lowland areas. Reports documenting extents of indigenous riparian forest loss and remnant extents remaining are limited (but see Miller 2002). Nevertheless, it seems reasonable to assume that the pace and trajectory of degradation for this attribute, would track similarly to degradation in extent of indigenous vegetation cover, generally.

Anecdotal evidence suggests that set-aside, fencing and planting riparian margins has taken place at many locations around New Zealand, but, in general, the pace and scale of change to reverse the degradation in this attribute has been slow and limited. This is expected to continue in the next 10-30 years under the status quo.

The degradation in national extent of riparian margin establishment and protection is mostly reversible, the main impediment is cost. It will be more costly (or even cost-prohibitive) to set-aside, fence, and plant riparian margins in some locations than in others due to variation in:

- lost opportunity costs associated with current use of the land;
- costs to remove existing built infrastructure (if applicable, i.e., in urban areas); and
- costs to fence, plant and maintain the margin areas.

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

There is no national monitoring or reporting of riparian margin establishment/protection or consensus on the most appropriate measurement method. Some local authorities assess the condition of stream riparian margins as a component of stream habitat assessments for State of Environment monitoring (e.g., Collier and Kelly 2005) and a national habitat assessment protocol has been developed (Clapcott 2015). Other monitoring protocols with riparian components include the stream ecological valuation method (SEV, Rowe et al. 2006), the stream habitat assessment protocol (SHAP, Harding et al. 2009), the restoration indicators toolkit (Parkyn et al. 2010), and the stream health monitoring and assessment kit (SHMAK, NIWA 2019). Habitat assessments typically evaluate several aspects of riparian condition including canopy cover (over stream), the width of the fenced margin, and vegetation composition, bank protection and stability.

The riparian management classification (RMC; Quinn et al. 2001) was developed to assess stream riparian state and functions and it has been applied to streams in Canterbury, Waikato, and Nelson (Phillips and Marden 2004, Quinn 2009). The Jobs for Nature biodiversity monitoring protocols (Clapcott et al. 2021) which form the basis of the Department of Conservations' online Freshwater Biodiversity Guide (FBG) tool, has been designed for monitoring of stream riparian restoration projects. The FBG protocols cover a much broader set of riparian parameters than prior tools, including aspects of plant survival, canopy height, plant and animal pests, terrestrial invertebrates, herpetofauna, bats and birds. Both the RMC and FBG also include protocols for monitoring the state of many instream parameters.

To the authors knowledge, local authority State of Environment riparian margin monitoring data have not been used to report on national or regional scale state and trend in this attribute, except for the Waikato Region by Pingram et al. (2023). Graham et al. (2020) also analysed trends in several riparian indicators from the Waikato State of Environment ecological data specifically for the Waikato River catchment. The Waikato Region conducts a 5-yearly survey specifically focused on the riparian characteristics of pastoral waterways (Norris et al. 2020). The survey includes random stratified sampling site measurements of the proportion of bank length effectively fenced, bank length with woody vegetation, and bank length with erosion and has been used to track their state over time since 2002. Lake riparian margins have been less commonly assessed by local authorities (but see Wildland Consultants 2011).

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

Current riparian margin monitoring approaches use survey sites that are often on privately owned land requiring staff conducting the survey to contact landowners to gain access permissions (Collier and Kelly 2005, Norris et al. 2020). Iwi/hapū/rūnanga should also be kept informed of any survey work being conducted in their rohe. In future, increased availability of remotely-sensed products (e.g., aerial and satellite imagery, LiDAR) may allow survey data to be collected without the need for physical site visits (for example see Pattle Delamore Partners (PDP) 2023), provided there are no legal barriers to obtaining data on privately owned land in this way.

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

Current riparian margin monitoring approaches rely primarily on visual assessments. They incur the following costs:

- Initial set up costs including staff time to design the monitoring programme, purchase of equipment, training staff to undertake the survey;
- Staff time and travel expenses to carry out the field surveys;
- Staff time to enter and analyse the data gathered;
- Staff time to report the results;
- Staff time and expenses involved in maintaining survey equipment.

These costs will vary by region depending on the number of sites to be monitored and the set of riparian margin indicators to be measured.

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

There are several examples of riparian margin establishment and protection being monitored by iwi/hapū/rūnanga. For example, in the Waitao catchment, Bay of Plenty, riparian vegetation, and its functions, were assessed in 2010, seven years after a programme of stream fencing and riparian planting was initiated by Ngā Pāpaka O Rangataua and other partners in the Te Awa O Waitao Restoration Project (Cooper et al. 2007, Blackett et al. 2011). In another example, riparian margin use, vegetation extent and composition, and riverbank condition were assessed as part of the development of a Cultural Heath Index (CHI) for streams and waterways, which gathered monitoring data on the Kakaunui, Taieri, Hakatere, and Tukituki rivers (Tipa & Tierney 2003). The CHI has since been adapted by many other iwi/hapū/rūnanga around New Zealand across a range of habitats (streams, rivers, lakes, wetlands, marine environs)[1]. More recently, bank vegetation and protection have been assessed as part of the development of Ngāti Maniapoto indicators and methodologies to support watercress assessment in streams and rivers and tested in the Turitea stream (Herangi and Ratana 2021). Further, in a summary of Kaupapa Māori Freshwater Assessments (Rainforth and Harmsworth 2019) most of the 13 tools identified appear to have a riparian habitat component. This suggests that one or more riparian parameters are being monitored by iwi/hapū/rūnanga using a variety of tools and approaches. Waterway access to stock (proxy for riparian protection) is also a component of many assessment methods being used by iwi/hapū/rūnanga.

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

Riparian margin establishment/protection conceivably influences all other attributes listed under the Freshwater Domain in this project, i.e., trace metals in water and sediment, groundwater nitrate, surface water flow alteration, groundwater depletion, and catchment permeability.

Riparian margin establishment/protection is likely to result in the interception of trace metals transported to waterbodies in surface runoff and shallow subsurface flows. Trace metals adsorbed to sediment can be deposited with sediment in vegetated riparian margins (Tang et al. 2014), especially those with plantings that encourage sediment deposition (e.g., with a dense, uniform grass filter, McKergow et al. 2022). Trace metals dissolved in water are likely to be assimilated into riparian vegetation biomass via root uptake as surface runoff water infiltrates into riparian soil, or as shallow subsurface flows pass through the vegetation root zone. Groundwater nitrate, transported in shallow subsurface flows through the vegetation root zone is also likely to be assimilated by vegetation. Groundwater nitrate may also be denitrified to nitrogen gases (i.e., N_2 or N_2O) in the vegetation root zone which typically has organic-carbon-rich soils that support this process. However, groundwater nitrate and trace metals in deeper aquifers, below the plant root zone will not be subject to plant uptake (and, in the case of nitrate, less likely to be denitrified).

The vegetation planted in riparian margins may affect surface water and groundwater availability, by altering water losses due to evaporation and plant evapo-transpiration processes. The extent of alteration to rates of plant evapo-transpiration rates will depend on the species planted, and what existing species they replace (e.g., pasture grasses). It will also depend on how much the new vegetation alters evaporative losses. Vegetation is generally expected to decrease evaporative losses

[1] For some examples, see https://www.stats.govt.nz/indicators/cultural-health-index-for-freshwater-bodies

via enhanced rainfall interception, build-up of organic soils which retain water, root biomass increasing soil permeability and shading of land and water surfaces.

The set-aside, fencing and planting of riparian margins is also likely to improve the permeability of soils in these areas, especially in grazed areas, as soil compaction from livestock treading will be eliminated (Cooper et al. 1995). Increased root biomass and organic matter is also likely to improve soil permeability.

Part B—Current state and allocation options

B1. What is the current state of the attribute?

At national scale, anecdotal evidence suggests that the riparian margins of most waterbodies still lack protection, especially those of smaller streams (i.e., streams <1m wetted width). Where riparian plantings have been established it seems that the width of plantings is typically narrow. It is expected that most regional councils and unitary authorities include some form of riparian assessment in their State of Environment monitoring programmes. Therefore, it may be possible to bring these data together in a national state assessment. It might also be possible for riparian datasets from other sources, such as iwi/hapū/rūnanga cultural assessments and community-based projects (e.g., funded by Jobs for Nature) to be included in a national scale assessment.

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

Areas of riparian margin located within remnant tracts of indigenous vegetation could possibly serve as natural reference states. However, even these remnant margin areas are likely to have suffered some degree of modification, especially where surrounding land use is highly modified. It may be possible to predict natural reference state using multiple lines of evidence and modelling approaches like those applied for groundwater nitrate (Daughney et al. 2023) and catchment nitrogen and phosphorus loads (Snelder et al. 2018).

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

Many of the visual assessment protocols used to assess riparian margin condition use narrative descriptions to assign a score for specific parameters. These narratives and scores are amenable to conversion to narrative bands (i.e., A-D bands as used in the National Policy Statement for Freshwater Management (NPS-FM) attributes). Protocols used to assess riparian margin condition that generate quantitative data for specific parameters could be converted to numeric bands but the maximum and minimum values possible for these parameters need to be determined as well as a justifiable process applied to identify the points of transition between A-B, B-C and C-D bands. Examples of how narrative and numeric bands can be developed for stream and lake riparian condition parameters are illustrated in the pilot Waikato River Report Card (Williamson et al. 2016), funded by the Waikato River Authority (WRA)[1] and guided by a Waikato River Iwi Advisory Group, where the following numeric and narrative parameters were scored A-D:

- Stream fencing state
- Stream shade state
- Lake riparian condition

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

There are known thresholds for riparian canopy cover (over stream) which creates sufficient shading to regulate growths of nuisance periphyton and macrophytes to levels recommended to protect stream ecological values. An analysis of data from Waikato Regional Councils' Regional Ecological Monitoring of Streams Programme showed that with ≥65-70% canopy cover of riparian vegetation the weighted composite cover (WCC) of periphyton was always <30% and channel clogginess by macrophytes (MCC) was always <50% (Matheson et al. 2017). These findings have also been supported by research in Canterbury streams (Collins et al. 2018). The <30% periphyton WCC and <50% MCC thresholds are provisional stream ecological integrity guidelines identified by Matheson et al. (2012).

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

It will take time for tree, shrub and sedge vegetation planted on riparian margins to mature and reach their full potential for reducing contaminant losses from land, stabilising banks, improving bank condition, and providing shade and physical habitat. It is likely to be necessary, therefore, for any state and trend assessment of riparian margin establishment/protection to consider the development stage of the vegetation and how close it is to maturity.

Some vegetation types will mature and reach their maximum height and density more quickly (e.g., grasses). In contrast, it will take many years for larger tree species to mature, and for their root structures to reach their full below-ground biomass and extent. Enrichment of riparian soils with leaf litter and plant organic exudates is also likely to be a longer-term process, even in areas planted with fast-maturing and/or annual species.

The state of the riparian margins prior to fencing and planting is also likely to influence the pace and trajectory of recovery. As noted under A1, the extent of benefit afforded by riparian margin establishment/protection on ecological integrity or human health is spatially and temporally variable and depends on various landscape, climatic and biological factors which includes the state of the riparian margins at the time that recovery was initiated.

[1] The Waikato River Authority was set up on 25 November 2010 under section 22(1) of the Waikato-Tainui Raupatu Claims (Waikato River) Settlement Act 2010 (the Act) and section 23(1) of the Ngāti Tuwharetoa, Raukawa, and Te Arawa River Iwi Waikato River Act 2010.

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

In addition to the WRA report card example explained in B3, cultural health assessment protocols (e.g.,Tipa and Tierney 2003, Herangi and Ratana 2021) could help to inform narrative bands. For example, the CHI suggests the following metrics which are scored between 1 (unhealthy) and 5 (healthy):

- river bank condition,
- vegetation – banks & margins,
- indigenous species – margins & adjacent land,
- use of the river banks + margins.

In the watercress assessment (Herangi and Ratana 2021) users of the methodology are asked to give the following metrics scores between 1 (unsatisfied) to 5 (satisfied):

- if they are satisfied that bank vegetation is healthy and that it is the right vegetation to support tuna/swimming/drinking water,
- if they are satisfied that the banks are protected to support tuna/swimming/drinking water,
- if they are satisfied that the effects of pest plants and invasive species are minimised at this site.

Part C—Management levers and context

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

The state of riparian margin establishment/protection is driven by the need for land to be used for rural and urban purposes. Evidence from regional surveys suggests that in rural locations, riparian margins are often in pasture and used for livestock grazing (Norris et al. 2020). They may also be planted in crops. In urban locations, riparian margins are likely to be mown or rank grass areas, have footpaths or even built structures. Where land is set aside from these uses, riparian margin establishment and protection is expected to benefit the ecological integrity and human health of adjacent waterbodies in the ways described in A1.

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

There have been several interventions/mechanisms employed to affect riparian margin establishment and protection in New Zealand as follows:

C2-(i). Local government driven

These include council staff providing advice to landowners and financial subsidies to support riparian fencing and planting activities. Analysis of council data in the Taranaki region indicates that where significant tracts of fenced and/or planted margins have been re-established there is a correlation with improving trends for *E. coli* and macroinvertebrates (Graham et al. 2018).

C2-(ii). Central government driven

These include investments to fund riparian fencing and planting through the One Billion Trees Fund and the Provincial Growth Fund, and for Ministry for Primary Industries to provide extension services to support these initiatives through the Productive and Sustainable Land use package (Wetlands and waterways gain from 1BT funding | Beehive.govt.nz). These are recent initiatives that are in the process of being evaluated. Projects are often only required to report on simple indicators of effort such as kilometres of stream bank fenced and/or hectares of riparian margin planted but see Clapcott et al. (2021) for a broader range of recommended riparian protocols.

C2-(iii). Iwi/hapū driven

There are many examples of iwi/hapū/rūnanga-driven interventions/mechanisms affecting this attribute around New Zealand. Examples include investments in riparian fencing and planting projects through the Waikato River Authority to deliver Te Ture Whaimana – the Vision & Strategy for the Waikato River (Waikato River Authority Projects (morphumdata.com)) and projects funded or partially funded by other central and local government initiatives. As for (ii) funded projects are often only required to report on simple indicators of effort such as kilometres of stream bank fenced and/or hectares of riparian margin planted. The CHI allows iwi/hapū/rūnanga users to assess whether they are satisfied that the riparian vegetation is healthy/protected and that it is the right vegetation to support their cultural values (e.g., mahinga kai).

C2-(iv). NGO, community driven

These include direction and/or guidance for riparian fencing and planting projects through NZ's primary sector organisations including DairyNZ, Beef&LambNZ and Fonterra. As noted under A2, industry guidelines have required riparian margins to be fenced to exclude dairy stock from entering streams wider than a stride (>1m) and deeper than a gumboot (>30cm) (Fonterra, Ministries for Agriculture and Environment, Local Government NZ 2003). The Dairy Best Practice Catchment study (Wrightstow and Wilcock 2017) found that improved stream fencing and effluent disposal was associated with improving trends in suspended sediment in all catchments and improving trends in other water quality parameters and certain macroinvertebrate metrics in some catchments. Other organisations that provide guidance and support for riparian planting include the New Zealand Landcare Trust, New Zealand Farm Forestry Association, Tane's Tree Trust and Tirohanga Ngahere Canopy and Rural Design.

C2-(v). Internationally driven

Restoring the vitality of degraded systems including riparian ecosystems is crucial for fulfilling the UN-Sustainable Development Goals in a timely manner and essential for attaining the targets of the UN-Decade (2021-2030) on Ecosystem Restoration (UN-DER) (Mohan et al. 2022). The Nature Conservancy is an international agency that is active in New Zealand (see Our Work in New Zealand | The Nature Conservancy) and provides support for conservation initiatives and nature-based solutions which can include riparian fencing and planting.

Part D—Impact analysis

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

Changes in the state of the riparian margin establishment and protection attribute affects ecological integrity and human health. If this attribute were not managed there are likely to be the following impacts:

- increased contamination of water that will detrimentally affect human water uses including drinking water, crop and stock water supplies and use of waterbodies for recreational activities;
- increased contamination of water that will detrimentally affect survival of aquatic life including taonga species important to iwi/hapū/rūnanga;
- continued loss of native aquatic, semi-aquatic and terrestrial biodiversity as physical habitat availability declines, and they are exposed to higher air and water temperatures;
- decreased water availability as less water is retained in the landscape through recharge of aquifers, retention in organic soils, and reduced evaporative losses.
- loss of cultural values, practices and mātauranga associated with (the appropriate) riparian vegetation (e.g., rongoā, raranga).

Not managing this attribute represents a missed opportunity to improve many aspects of ecological integrity and human health through riparian margin establishment and protection.

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

Economic impacts from not managing this attribute will affect both urban and rural sectors in New Zealand. Impacts are likely to include:

- increased costs of removing contaminants from water extracted from degraded waterbodies for various uses, especially for human drinking water and crop and livestock water supplies;
- increased costs associated with development of water retention infrastructure to ensure sufficient water supplies for urban and rural uses during times of drought;
- increased costs associated with repairing flood damage resulting from higher rates of flow and contaminant mobilisation during storm events.

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

Managing this attribute will increase landscape resilience to the anticipated effects of climate change. Riparian margin establishment and protection should lead to the following:

- Reducing flood and drought risk and associated damage as these events become more frequent and extreme by retaining water on the land, through recharge of soil and groundwater aquifers;
- Reducing the mobilisation of contaminants from land as storm events become more frequent and intense;
- Protecting native terrestrial, semi-aquatic and aquatic life that inhabit riparian margins and adjacent waterbodies including taonga species from increased air and water temperatures.

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8.2 Heavy metals in freshwater

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Alternative attribute name: Trace metals in freshwater

State of Knowledge for the "Heavy Metals in freshwater" attribute: Good / established but incomplete – general agreement, but limited data/studies

Part A—Attribute and method

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

Trace metals are naturally present in the environment. Their distribution depends on the presence of natural sources (e.g., volcanoes or erosion) and human activities through extraction from ores [1]. The main anthropogenic activities resulting in the discharge of metals include fossil fuel combustion, industrial and agricultural processes and many metals are used in daily home activities [2].

The term heavy metals is often used to describe metals in general. However, it is not appropriate as not all metals are heavy or non-essential. For instance, cadmium and mercury are heavy metals but other metals of environmental concern including zinc and copper are essential metals. It is estimated that one-third of all proteins requires a metal cofactor for normal functions [1]. However, even essential metals can be toxic and that depends on the concentration. This relates to the concept of essentiality as illustrated in Figure 1. For essential metals like copper, zinc and selenium, there is a "window of essentiality" which represents a range of concentrations that will maintain a level of health in an organism- as illustrated in Figure 1A. For non-essential metals like cadmium, when concentrations reach levels that overcome the defence capacity of an organism, then it becomes toxic (Figure 1, panel B). This is why using trace metals is the appropriate term to use as it covers all metals. The most appropriate term would be trace elements as arsenic is defined as an element or metalloid.

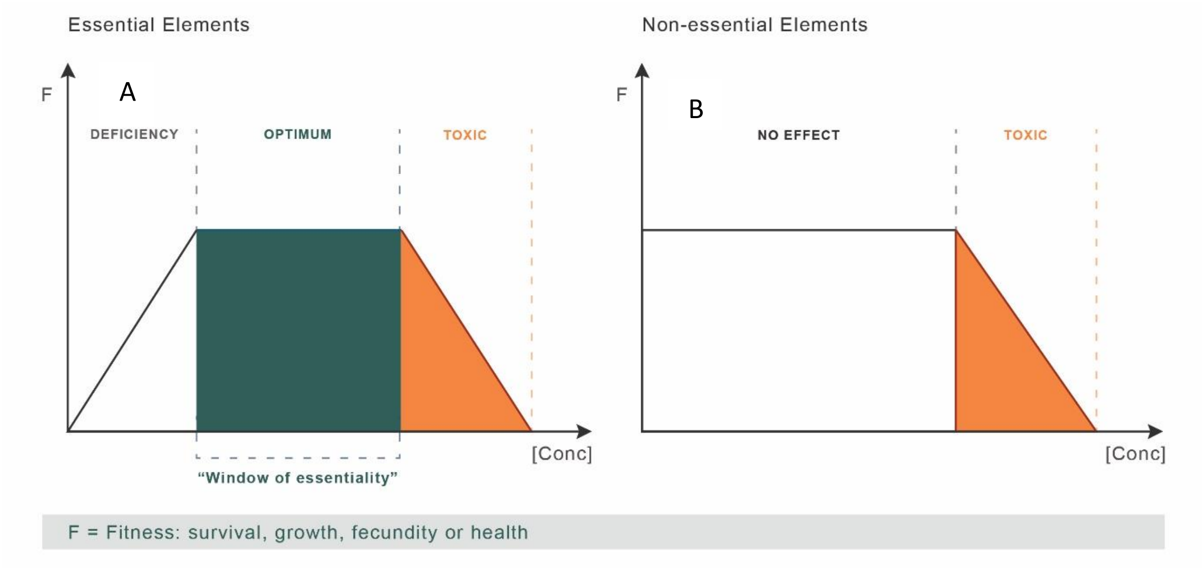

Figure 1. Conceptual diagrams illustrating the differences in concentration–response relationships with respect to organism health between A) essential metals and B) non-essential metals.

The toxicity of trace metals is well established and can impact both ecosystem and human health. Metals and metal compounds can interfere with functions of the central nervous system (CNS), the haematopoietic system, liver and kidneys [2].

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

There is strong evidence globally of the adverse effects on human metabolism resulting from exposure to metal-contaminated drinking water [3]. Exposure to non-essential metals is potentially harmful as they do not have physiological roles in the metabolism of cells. In addition, the ingestion of metals via food or water can modify the metabolism of other essential elements including zinc, copper, iron and selenium [2]. The general mechanism of heavy metal toxicity is through the production of reactive oxygen species (ROS) leading to oxidative damage and subsequently, adverse effects on health [3]. The disruption of metal ion homeostasis leads to oxidative stress through the formation of ROS which overwhelm body antioxidant protection and subsequently induces DNA damage, lipid peroxidation, protein modification and other effects, all symptomatic of numerous diseases, including cancer, cardiovascular disease, diabetes, atherosclerosis, neurological disorders (Alzheimer's disease, Parkinson's disease), chronic inflammation and others [4]. Another important mechanism of toxicity is the bonding of redox inactive metals like cadmium, arsenic and lead to sulphydryl groups of proteins and depletion of glutathione [4]. The mechanisms of toxicity are conserved, and metals affect ecosystem health in a similar way.

Waterbodies in areas of high anthropogenic activity like urban centres and areas of intensive agriculture are more likely to be contaminated with metals. Urban areas have larger areas of impervious surfaces such as roofs, roads and paved areas that are sources of metals [5]. Stream water quality changes in urban and rural areas as development both increases the generation of contaminants and changes the transport and processing of contaminants. Many urban and rural streams are also the receiving environment for untreated sewage, via leakage or overflows from wastewater networks and treatment plants.

Increasing population pressure and urbanization of the coastal zones have resulted in a variety of chronic impacts operating on coastal and estuarine ecosystems. Land-based activities affect the runoff of pollutants and nutrients into fresh and coastal waters affecting biodiversity and ultimately the provision of ecosystem services . Local studies in the Auckland coastal zone and the Tauranga Harbour showed ecological health decline, based on community structure composition changes along a pollution gradient, occurring at metal levels below guideline threshold values . These are good examples that coastal ecosystems are often exposed to multiple stressors and robust management frameworks are required to consider the presence of multiple physical and chemical stressors. Similar considerations apply in the freshwater domain.

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

There are multiple sources of metals. Natural sources of metals include rocks, ore minerals, volcanoes, and weathering releases of metals during soil formation transported to the surface and/or aquifer waters. The primary anthropogenic sources of metals are related mostly to the mining, extraction, and refining stages of ore that can lead to air, water, and soil pollution [11]. Metals are elements that can neither be created nor destroyed so once they are extracted from ore, they may be dispersed into the environment where they can accumulate [1]. Metals are transported from secondary sources into waterways via stormwater from roads (zinc from tyre wear, copper from brake pad wear); roofs (zinc from galvanised roofing); and other impervious surfaces (including paved areas around industrial sites) [12].

Copper and zinc are major contaminants in urban streams and frequently used as indicators of stormwater inputs [5]. For state of the environment reporting, the accumulation of metals has been monitored over the years, primarily in the main centres of Auckland, Wellington and Christchurch. The data is publicly available. For instance, between January 2015 and December 2017 median concentrations of dissolved zinc exceeded the Australian and New Zealand Guidelines for Freshwater and Marine Water Quality default guideline values (DGVs) at 8 of 11 Auckland sites, 3 of 5 Wellington sites, and 13 of 39 Christchurch sites[1]. The trends for dissolved zinc and copper at sites monitored between 2011 and 2017 varies from improving to worsening. The analysis of a range of parameters including dissolved zinc indicates that if urban development continues its' current trend, increases in urban land cover around New Zealand can be expected to result in further declines in water quality and a reduced likelihood that water quality objectives will be achieved at impacted locations [5].

For metal physiology and toxicology, the importance of chemical speciation cannot be overstated. Perhaps the simplest feature of speciation is whether the metal is in the dissolved or particulate form. This reflects the general recognition that particulate metals exhibit negligible toxicity and bioavailability to aquatic organisms relative to dissolved metals. The speciation chemistry of different metals varies greatly, but in general lower pH increases the free ion concentration, thereby increasing toxicity, whereas alkalinity (i.e., bicarbonate - HCO_3) and inorganic anions tend to complex metal ions, thereby decreasing toxicity. The hardness cations (Ca^{2+} and Mg^{2+}) as well as Na^+ and K^+ (and sometimes H^+) may also decrease toxicity by competing for metal binding sites on the gills of fish. The presence of dissolved organic matter (DOM) in most water bodies is another effective agent of protection against most metals [1]. Therefore, as metals continue to accumulate and partition into

[1] https://www.stats.govt.nz/indicators/river-water-quality-heavy-metals

the receiving environments, the bioavailability and ultimately the risk of these metals are highly dependent on the speciation conditions specific to that environment.

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

Some metals are monitored as part of the State of Environment (SoE) reporting. Councils are conducting routine analyses for the occurrence and trends of metals for coasts, rivers, lakes and groundwaters as part of SoE monitoring and to meet consent condition requirements [13]. The SoE monitoring by regional councils focuses on a set of metals as reported in the recent Parliamentary Commissioner for the Environment (PCE) report on regulating the environmental fate of chemicals [1]. Monitoring of metal residues in relation to determining compliance with consent conditions is also often conducted for landfill leachate, wastewater and stormwater discharges [13].

The Australian and New Zealand Guidelines for Fresh and Marine Water Quality (https://www.waterquality.gov.au/anz-guidelines) are key tools to help planners, regulators and researchers to manage the quality of our water in New Zealand, especially for metals which are not currently covered by the NPS-FM. They provide default guideline values (DGVs) for all metals. These DGVs have been jointly developed by the Australian and New Zealand governments.

There have been notable advances in the development of bioavailability models for assessing toxicity as a function of water chemistry in freshwater ecosystems. For instance, the biotic ligand model (BLM), the multiple linear regression model, and multimetal BLM have been developed for most of the common mono- and divalent metals. Species sensitivity distributions for many metals are available, making it possible for many jurisdictions to develop or update water quality criteria or guidelines [15].

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

The author is not in a position to comment, but Regional Councils have selected sites where they monitor trends for SoE reporting. Consent holders would also have access to sites for monitoring as part of their consent conditions.

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

The analytical methods using inductively coupled plasma mass spectrometry (ICP-MS) instruments can measure elements and metals and are well established and validated. Several commercial laboratories including Hill Labs and AsureQuality can measure metals at competitive prices.

A Jacobs investigation reported limitations that councils have identified that prevent the expansion of current monitoring programmes including the high costs for both laboratory analysis and council staff time spent doing monitoring and reporting [13].

[1] https://pce.parliament.nz/publications/regulating-the-environmental-fate-of-chemicals/

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

We are not aware of any monitoring of this attribute being regularly undertaken by iwi/hapū/rūnanga. Resourcing is difficult for iwi/hapū/rūnanga to obtain, and any monitoring by agencies is generally infrequent, inconsistent, and ad hoc, and most programmes fail to provide information on whether chemical contaminants will have impacts of concern to Māori [28]. The Waikato River Report Card and other environmental assessment frameworks being developed by/with iwi/hapū/rūnanga include "safe to eat" or "safe to swim" outcomes [29-31]. Data/indicators required to fully realise these holistic cultural assessment frameworks will require information about heavy metals in water, sediment, and/or mahinga kai species.

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

Metals are often not the only type of contaminants at impacted sites. The issue of multiple stressors – e.g., stormwater and wastewater contain a range of other contaminants. Therefore, management of fresh and coastal waters must contend with multiple drivers in concert as the coordination of regulating agencies for urban and agricultural runoff is warranted [8]. As such, metals are only one component within a range of other contaminants that can accumulate in the environment.

Metals can be assimilated by, and bioaccumulate within organisms. Riparian vegetation would be expected to assimilate metals from surface and subsurface water passing through their root zones and from contaminated sediments deposited in riparian areas (see Riparian margin establishment and protection attribute).

Part B—Current state and allocation options

B1. What is the current state of the attribute?

The ecotoxicological effects of metals and their speciation under a range of environmental conditions are well understood and documented (as per references cited above). The key anthropogenic sources are well characterised to assist the management of these contaminants. The main challenge is that the management of metals requires a holistic/system approach as there are multiple factors to consider. For instance, roof material often contains zinc that can leach overtime. Some effort is required to find alternative types of material with less impacts. This needs to be underpinned by appropriate policy and evidence shows that this can be effective. For instance, the global phase-out of leaded petrol use has contributed to the decline of concentrations in the ocean [18].

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

Finding reference sites with low levels of anthropogenic pressure could provide a baseline to confirm adverse impacts of metals and other stressors on receiving ecosystems. However, it is very difficult to find reference sites that experience no anthropogenic pressure.

The hazards of metals and their mechanisms of toxicity have been extensively characterised using model test species under controlled laboratory conditions. The data generated are used to derive the default guideline values (DGVs) which provide threshold values over which adverse impacts are expected. A metal concentration above a DGV should trigger further investigations to fully assess the impacts of the metal on the receiving ecosystem. Good baseline values can complement this approach by providing numeric evidence of concentrations expected in a healthy, unmodified ecosystem looks like is important. There are options to compensate for the lack of proper reference sites by monitoring across a gradient of stressors.

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

There are well established default guideline values (DGVs) for a number of metals that have recently been reviewed by the Australian and New Zealand Guidelines for Fresh and Marine Water Quality [1]. These threshold values cover a range of protection levels of 80, 90, 95 and 99 % relevant to the particular ecosystem of interest, e.g., from industrial areas to national park and reserve areas.

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

There are threshold value guidelines available. The ANZG DGVs have been developed to provide threshold values for metals and other contaminants. They are set to provide a range of protection as per point B3.

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

As discussed in the above sections, metals have multiple anthropogenic sources and they can continue to accumulate in various environmental compartments including surface water, groundwater, and coastal waters due to the non-degradability of metals.

Natural background levels of metals in lakes and rivers may vary widely because of differences in local geology, and the aquatic organisms that live there tend to be genetically adapted to the local levels of metals. This adaptation is described as the “metalloregion concept” [8]. This is particularly relevant to New Zealand where levels of some metals in the environment are associated with our unique soil and volcanic activity. For instance, in the central North Island, arsenic is released from geothermal systems into the Waikato River [19]. The receiving ecosystems may have adapted to higher background levels, although in the case of the Waikato River system this has been extensively modified via the creation of hydrolakes for electricity generation which is likely to have altered the biota assemblages that now reside in those waterbodies.

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

A high standard of water quality is an outcome sought by iwi/hapū/rūnanga. In addition to discussing this attribute directly with iwi/hapū/rūnanga, in regard to heavy metals in water, there is tikanga and mātauranga Māori relevant to informing bands, allocation options, minimally disturbed conditions

[1] www.waterquality.gov.au/anz-guidelines

and/or unacceptable degradation residing in treaty settlements, catchment/species restoration strategies, cultural impact assessments, environment court submissions, iwi environmental management plans, reports, etc.

For example, as result of the Waikato-Tainui Deed of Settlement, Te Ture Whaimana o te Awa o Waikato (the Vision & Strategy) is the primary direction setting document for the Waikato River and activities within its catchment affecting the river. In order to realise Te Ture Whaimana, 13 objectives and 12 strategies guide the restoration of the health and wellbeing of the Waikato River, including: "The restoration of water quality within the Waikato River so that it is safe for people to swim in and take food from over its entire length". The pilot Waikato River Report Card (Williamson et al. 2016), funded by the Waikato River Authority (WRA) and guided by a Waikato River Iwi Advisory Group, scored 'arsenic in water' between A-D using the ANZECC guidelines.

There are one-off-studies where iwi/hapū/rūnanga are influencing research initiatives exploring the state and impacts of environmental contaminants (including heavy metals) on the outcomes they are seeking (e.g., mauri is protected, kai is safe to eat, water is safe to swim) (e.g., [32-34]).

<u>Part C—Management levers and context</u>

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

The SoE reporting for MfE highlights the level of environmental degradation in both freshwater and marine domains [20, 21]. Metals are one of the multiple stressors that have been identified with sources including stormwater, municipal treated wastewater and agricultural discharges. There is evidence of interactive effects between copper and temperature on freshwater mussel species, suggesting the increased stress of elevated temperatures and copper exposure occurring together [22].

The toxicity and ecotoxicity of individual metals are well characterised and understood. However, predicting or assessing the environmental impacts of an individual chemical is a challenge in a field situation as contaminants are often found in complex mixtures. For instance, exposure to low levels of multiple chemicals in mixtures can cause toxicity at concentrations where exposure to an individual chemical might cause no effect based on their DGVs. This is because multiple physiological processes may be affected by chemicals having different mechanisms of toxicity. This is a strong argument for in favour of a systems approach to the management of aquatic systems.

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

C2-(i). Local government driven

C2-(ii). Central government driven and C2-(v). Internationally driven

he Australian and New Zealand guidelines for Fresh & Marine Water Quality trigger values are designed to lead to further site-specific risk assessment. In a recent survey on the type and range of chemical contaminants that councils monitor, the emphasis was on the type of chemicals, but the implications of exceedance of DGVs was not assessed [13]. The author is not aware of any follow up studies in New Zealand in response to a DGV exceedance.

C2-(iii). Iwi/hapū driven

(iii) Treaty of Waitangi settlements have resulted in waterways/lakes/wetlands being returned to Māori ownership and/or management, many of which are in a highly degraded state. Many settlements include cultural redress packages to address the protection, restoration or rehabilitation of values, uses and services (and at scales) that have not previously been a strategic priority for research, restoration and monitoring by agencies. Treaty Settlements have also been the key drivers in the provision of new innovative approaches that bring together multiple knowledge systems together to inform co-management and restoration regarding values, species, catchments, and/or at scales that have not previously been prioritised by agencies.

Iwi/hapū/rūnanga are also influencing resource consent conditions which may include contaminant/bioaccumulation assessments that include heavy metals in water, sediments and/or mahinga kai species; however, generally these reports are not accessible in the public domain.

C2-(iv). NGO, community driven

Part D—Impact analysis

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

Metals are not degradable so any continuous discharges will accumulate metals into the various environmental compartments and biota. The impacts of human activities have pushed estuarine and coastal ecosystems far from their historical baseline of rich, diverse, and productive ecosystems [23]. The impacts on freshwater ecosystems from human activities have also been significant. Managing the sources of harmful contaminants is a priority to ensure the protection of these valuable ecosystems and to protect water supplies required for private and commercial uses from contamination. Encouragingly, there are examples of declining metal concentrations from improved environmental controls on emissions and discharges of metals and other contaminants, e.g., [24].

There are multiple challenges to reduce the discharge of metals in urban and rural environments, particularly non-point sources like stormwater. There are examples of options to reduce metals at-source summarised in the PCE report but these may be challenging to implement. For example, an initiative to impose restrictions on the maximum amount of zinc in galvanised or zinc coated roofing materials may be opposed by manufacturers of those materials [25].

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

Fishery and aquaculture industries are most likely to be impacted by pressure from metal contamination. Healthy and functional ecosystems and healthy fish stocks are important for the fisheries industry [20]. There are other aspects to consider including natural beauty and recreational use of our freshwaters, estuaries, coastal and open ocean areas that are central to our culture and national identity. Furthermore, there are likely to be increased costs and economic impacts associated with the need for removal of trace metals from water supplies to ensure its safe use for drinking water, stock water, irrigation and other industrial and agricultural uses if this attribute were not managed.

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

Climate change may alter physical, chemical and biological properties of ecosystems, affecting organisms but also the transport and fate of chemical pollutants. For example, an increase is storms is likely to mobilise contaminants in runoff in dissolved or particulate form. However, key concern with the effects of climate change on the risks associated with metal contamination is that changes to temperature and pH can modulate the speciation of metals or basically, their bioavailability. The importance of metal speciation cannot be overstated as it affects the bioavailability and toxicology of these materials. The simplest feature of speciation is whether the metal is in the dissolved or particulate form. Originally, environmental regulations were based on total metals present in the water as assayed by hot acid digestion of the samples. However, there has been a gradual change in many jurisdictions to regulations based on the dissolved component only. This reflects the general recognition that particulate metals exhibit negligible toxicity and bioavailability to aquatic organisms relative to dissolved metals [1]. Increases in temperature have been correlated with increasing toxicity of metals to aquatic organisms [26]. As such, temperature should be accounted for in risk assessment, because it may modify the effects of chemicals on the structure and functioning of aquatic communities, especially at higher levels of biological organization [27].

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8.3 Heavy metals in freshwater sediment

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Alternative attribute name: Trace metals in freshwater sediment

Preamble: Pressures from human activities, such as agriculture, effluent discharges from landfill and wastewater treatment plants (WWTPs), urbanisation, and industrial wastes increase sediment metal concentrations [1]. Metals are of growing concern in terms of water quality management, as they cannot be degraded in the environment although some metal species can be transformed into other species which may be more or less toxic [1].

State of knowledge of "Trace metals in freshwater sediment" attribute: Good / established but incomplete – general agreement, but limited data/studies

Part A—Attribute and method

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

Trace metals are naturally present in the environment. Their distribution depends on the presence of natural sources (e.g., volcanoes or erosion) and human activities through extraction from ores [2]. The main anthropogenic activities resulting in the discharge of metals include fossil fuel combustion, industrial and agricultural processes and many metals are used in daily household activities [3]. It is important to recognise the types of metals. For instance, cadmium and mercury are heavy metals but other metals of environmental concern including zinc and copper are essential metals. It is estimated that one-third of all proteins requires a metal cofactor for normal functions [2]. However, even essential metals can be toxic and that depends on the concentration. This relates to the concept of essentiality as illustrated in Figure 1. For essential metals like copper, zinc and selenium, there is a "window of essentiality" which represents a range of concentrations that will maintain a level of health in an organism- as illustrated in Figure 1A. For non-essential metals like cadmium, when concentrations reach levels that overcome the defence capacity of an organism, then it becomes toxic (Figure 1, panel B). This is why using trace metals is the appropriate term to use as it covers all metals. The most appropriate term would be trace elements as arsenic is defined as an element or metalloid.

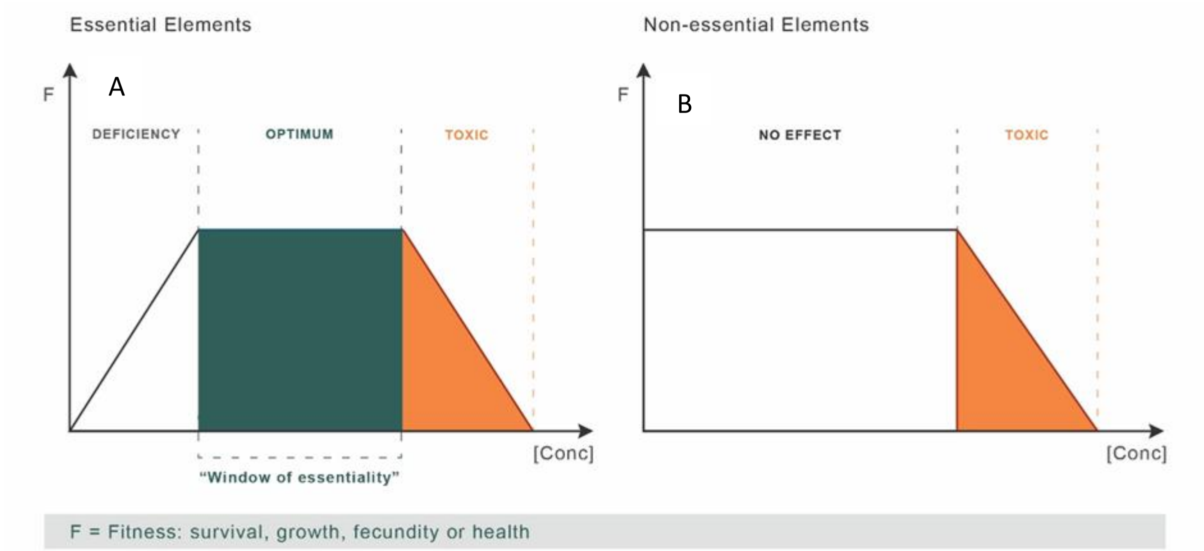

Figure 1. Conceptual diagrams illustrating the differences in concentration–response relationships with respect to organism health between A) essential metals and B) non-essential metals.

The toxicity of trace metals is well established and can impact both ecosystem and human health [4]. The relationship of metals to human and ecological health has been covered in the Attribute of trace metals in water. The hazards remain similar with sediment as another source of metal exposure with receptor species most at-risk being sediment dwelling organisms. Metals in sediment can enter the food chain through bioaccumulation posing a risk to exposed biota higher in the food chain and humans [5]. As sediment is the major compartment where metals accumulate, it is also the major source of exposure posing the highest risk [6], although metals in dissolved form are considered more toxic (see Trace metals in water A3 and D3) .

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

There is strong evidence globally of the adverse effects on human metabolism resulting from exposure to metal-contaminated drinking water [3]. Exposure to non-essential metals is potentially harmful as they do not have physiological roles in the metabolism of cells. In addition, the ingestion of metals via food or water can modify the metabolism of other essential elements including zinc, copper, iron and selenium [4]. Metals and metal compounds can interfere with functions of the central nervous system (CNS), the haematopoietic system, liver and kidneys [2].

Waterbodies in areas of high anthropogenic activity like urban centres or rural areas with intensive agriculture are more likely to be impacted bymetal contaminations. Urban areas have larger areas of impervious surfaces such as roofs, roads and paved areas that are sources of metals [7]. Many urban and rural streams are also the receiving environment for untreated sewage, via leakage or overflows from wastewater networks and treatment plants.

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

The status quo would result in the continuous accumulation of metals in the environment as they are not biodegradable. Worldwide, in addition to the issue of anthropogenic zinc contamination in urban areas, contamination of soils with zinc has increased in some agricultural sectors, such as dairy farming and horticulture. The most significant concern for freshwater lakes relates to the partitioning of zinc to bed sediments, where over time it may gradually build up beyond ecotoxic thresholds for macroinvertebrates and other bed-dwelling organisms, which are integral components of aquatic ecosystems [8]. Accumulation of zinc in sediments from rural lakes is now evident in the Waikato region. While 86 per cent of lakes assessed have at least twice background concentrations, three lakes presented values above the interim sediment quality guideline low value of 200 mg/kg (meaning that further investigation is required to assess the extent of risk posed by the chemical) [8]. A recent study of water quality in urban streams indicated that if urban development continues in its current form, increases in urban land cover around New Zealand can be expected to result in further declines in water quality at impacted locations [7].

There is evidence that better management of trace metal sources can reverse the trends. For instance, the global phase-out of leaded petrol use has contributed to the decline of concentrations in the ocean [9]. Also, a UK study showed that reductions in industrial activity and improved environmental controls on emissions resulted in a decline in trace metal concentrations in sediments [10].

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

A report commissioned by the PCE provided a national-level summary of the chemical contaminants including metals that Regional Councils/Unitary Authorities include in consent-based monitoring requirements and routine State of the Environment (SoE) monitoring programmes [11]. It stated that copper, zinc and lead were the most frequently listed trace metals monitored as part of consent conditions [11]. It is interesting that to date, there are no published studies have quantitatively assessed relationships of copper and zinc with intensity of urban land use, despite these metals being key contaminants in urban streams and frequently used as indicators of stormwater inputs [7].

One important aspect that is not commonly included in current monitoring frameworks is the use of biological indicators, or bioindicators. Bioindication is the use of an organism, a part of an organism, or a community of organisms, to assess the quality of its/their environment [5]. A definition of bioindicator was suggested to be an anthropogenically induced variation in biochemical, physiological, or ecological components or processes, structures, or functions (i.e., a biomarker) that can be causally-linked to biological effects [12].

Macroinvertebrate abundance can be influenced by the level of stressors as taxon richness declines across pollution gradients. Pollution sensitive taxa respond to levels of contaminants leading to alterations to benthic macroinvertebrate assemblages (e.g., [13]). Effects of trace metals on benthic communities in New Zealand streams were similar to those reported for metal-polluted streams in North America and Europe, suggesting that responses to metal contamination are predictable [14].

There have been notable advances in the development of bioavailability models for assessing toxicity as a function of water chemistry in freshwater ecosystems. For instance, the biotic ligand model (BLM), the multiple linear regression model, and multimetal BLM have been developed for most of the common mono- and divalent metals. Species sensitivity distributions for many metals are available, making it possible for many jurisdictions to develop or update water quality criteria or guidelines [15]. Sediment bioavailability models are also emerging including models that allow for prediction of toxicity in sediments for copper and nickel [15].

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

The author is not in a position to comment but Regional Councils have selected sites where they monitor trends for the SoE. It is possible that consent holders would also have access to sites for monitoring as part of their consent conditions. Accessing sites for monitoring should use appropriate engagement practices with all stakeholders and partners.

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

The analytical methods using inductively coupled plasma mass spectrometry (ICP-MS) instruments can measure elements and metals and are well established and validated. Several commercial laboratories including Hill Labs and AsureQuality can measure metals at competitive prices.

A recent investigation reported limitations that councils have identified that prevent the expansion of current monitoring programmes including the high costs for both laboratory analysis and council staff time spent doing monitoring and reporting [11]. However, it should be noted that consent holders cover agreed conditions monitoring costs.

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

We are not aware of any heavy metals monitoring being regularly undertaken by iwi/hapū/rūnanga. Resourcing is difficult for iwi/hapū/rūnanga to obtain, and any monitoring by agencies is generally infrequent, inconsistent, and ad hoc, and most programmes fail to provide information on whether chemical contaminants will have impacts of concern to Māori [32]. The Waikato River Report Card and other environmental assessment frameworks being developed by/with iwi/hapū/rūnanga include “safe to eat” or “safe to swim” outcomes [33-35]. Data/indicators required to fully realise these holistic cultural assessment frameworks will require information about heavy metals in water, sediment, and/or mahinga kai species.

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

Contaminants are mostly found as complex mixtures of which metals are one family of pollutants at impacted sites. The issue of multiple stressors relates to the range of sources that put pressure on the receiving environment – e.g., stormwater and wastewater contain a range of other types of contaminants. Cumulative effects, through additional new industries, climate change and other stressors, can reduce environmental resilience and increase the risk of environmental degradation or economic collapse of enterprises relying on the environment [20]. The importance of sediments as stressors will depend on site ecosystem attributes and the magnitude and preponderance of co-occurring stressors [21]. Management approaches must contend with multiple drivers in concert. The coordination of regulating agencies for urban and agricultural runoff is warranted for as metals are only one component within a range of other contaminants that can accumulate in sediment [22]. The sources of metals in urban areas of New Zealand have been well-characterised providing direction for reducing metal concentrations in stormwater through source control (e.g., reducing metal leaching from roofing materials) and at-source treatment in key locations [7].

Metals can be assimilated by, and bioaccumulate within, organisms. Riparian vegetation would be expected to assimilate metals from surface and subsurface water passing through their roots zones and from contaminated sediments deposited in riparian areas (see Riparian margin establishments and protection attribute). Furthermore, metals have been positively related to the proportion of imperviousness in upstream catchments (see B1 and Catchment permeability attribute).

Part B—Current state and allocation options

B1. What is the current state of the attribute?

The information to date indicates that trace metals are accumulating in our environment. For example zinc has been positively related to the proportion of urban land cover and imperviousness in upstream catchments [7]. The ecotoxicological effects of trace metals and their speciation under a range of environmental conditions are well understood and documented (as per references cited above). The key anthropogenic sources are well characterised to assist the management of these contaminants. The main challenge is that the management of metals requires a holistic/system approach as there are multiple factors to consider. For instance, roof material often contains zinc that can leach overtime. Some effort is required to find alternative types of material with less impacts which needs to be underpinned by appropriate policy. There are examples of recovery following policy changes, e.g., the global phase-out of leaded petrol use has contributed to the decline of concentrations in the ocean [9].

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

Finding reference sites with low levels of anthropogenic pressure is important to provide a baseline to confirm adverse impacts of metals and other stressors on receiving ecosystems. However, it is very difficult to find reference sites that experience no anthropogenic pressure.

One option to consider is to use a ranking of environmental targets in line with the ecosystems to protect. The widespread and serious degradation of urban streams has been documented and their improvement should be ranked high to achieve agreed level levels of protection but it will be challenging [7].

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

Sediment quality guideline values (SQGVs) for trace metals have been derived and updated [23]. These values are now used as default guideline values (DGVs) n the Australian and New Zealand

Guidelines for Freshwater and Marine Water Quality as Toxicant Default Guideline Values for Sediment Quality[1].

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

There are threshold value guidelines available. The Australian and New Zealand Guidelines for Freshwater and Marine Water Quality Toxicant Default Guideline Values for Sediment Quality have been developed to provide threshold values for metals and other contaminants. They are set to provide a range of protection of 80, 90, 95 and 99 % relevant to the particular ecosystem of interest, e.g., from industrial areas to national park and reserve areas.

The sediment DGVs indicate the concentrations below which there is a low risk of unacceptable effects occurring, and should be used, with other lines of evidence, to protect aquatic ecosystems. In contrast, the 'upper' guideline values (GV-high), provide an indication of concentrations at which there might already have toxicity-related adverse effects. As such, the GV-high value should only be used as an indicator of potential high-level toxicity problems, not as a guideline value to ensure protection of ecosystems.

If a DGV is exceeded or even where toxicant concentrations in the sediment are trending towards the DGV, it is recommended to use a multiple lines of evidence approach as part of the weight-of-evidence process to better assess the risk to the sediment ecosystem.

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

As discussed in the above sections, metals have multiple anthropogenic sources and they can continue to accumulate in various environmental compartments including sediment due to the non-degradability of metals.

Natural background levels of metals in lakes and rivers may vary widely because of differences in local geology, and the aquatic organisms that live there tend to be genetically adapted to the local levels of metals. This adaptation is described as the "metalloregion concept" [22]. This is particularly relevant to New Zealand where levels of some metals in the environment is associated with our unique soil and volcanic activity. For instance, in the central North Island, arsenic is released from geothermal systems into the Waikato River [24]. The receiving ecosystems will have adapted to higher background levels, although in the case of the Waikato River system this has been extensively modified via the creation of hydrolakes for electricity generation which is likely to have altered the biotic assemblages that now reside in those waterbodies.

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

A high standard of water quality is an outcome sought by iwi/hapū/rūnanga. There is tikanga and mātauranga Māori relevant to informing bands, allocation options, minimally disturbed conditions

[1] https://www.waterquality.gov.au/anz-guidelines/guideline-values/default/sediment-quality-toxicants

and/or unacceptable degradation residing in treaty settlements, catchment/species restoration strategies, cultural impact assessments, environment court submissions, iwi environmental management plans, reports, etc.

For example, as result of the Waikato-Tainui Deed of Settlement, Te Ture Whaimana o te Awa o Waikato (the Vision & Strategy) is the primary direction setting document for the Waikato River and activities within its catchment affecting the river. In order to realise Te Ture Whaimana, 13 objectives and 12 strategies guide the restoration of the health and wellbeing of the Waikato River, including: "The restoration of water quality within the Waikato River so that it is safe for people to swim in and take food from over its entire length". The pilot Waikato River Report Card [33], funded by the Waikato River Authority (WRA) and guided by a Waikato River Iwi Advisory Group, scored 'arsenic in water' between A-D using the ANZECC guidelines.

There are one-off-studies where iwi/hapū/rūnanga are influencing research initiatives exploring the state and impacts of environmental contaminants (including heavy metals) on the outcomes they are seeking (e.g., mauri is protected, kai is safe to eat, water is safe to swim) (e.g., [36-38]).

Part C—Management levers and context

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

It is important to address the increasing trends of metal accumulation and develop solutions to revert the increasing trends using better management frameworks for the sources. The National State of Environment reporting for MfE highlights the level of environmental degradation in both freshwater and marine domains [26,27]. Metals are one of the multiple stressors that have been identified with sources including stormwater, municipal treated wastewater and agricultural discharges.

The toxicity and ecotoxicity of individual metals are well characterised and understood. Predicting or assessing the environmental impacts of an individual chemical is a challenge in a field situation as contaminants are often found in complex mixtures. For instance, exposure to low levels of multiple chemicals in mixtures can cause toxicity at concentrations where exposure to an individual chemical might cause no effect based on their DGVs. This is because multiple physiological processes may be affected by chemicals having different mechanisms of toxicity. This is a strong argument for the need for a systems approach to the management of aquatic systems.

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

C2-(i). Local government driven

C2-(ii). Central government driven and C2-(v). Internationally driven

The Australian and New Zealand Guidelines for Freshwater and Marine Water Quality Toxicant Default Guidelines for Sediment Quality are designed to trigger further site-specific risk assessment based on a weight of evidence approach. In a recent survey on the type and range of chemical contaminants that councils monitor, the emphasis was on the type of chemicals, but the implications of exceedance of DGVs was not assessed [11]. The author is not aware of any follow up studies in New Zealand responding to a DGV exceedance.

C2-(iii). Iwi/hapū driven

Treaty of Waitangi settlements have resulted in waterways/lakes/wetlands being returned to Māori ownership and/or management, many of which are in a highly degraded state. Many settlements include cultural redress packages to address the protection, restoration or rehabilitation of values, uses and services (and at scales) that have not previously been a strategic priority for research, restoration and monitoring by agencies. Treaty Settlements have also been the key drivers in the provision of new innovative approaches that bring together multiple knowledge systems together to inform co-management and restoration regarding values, species, catchments, and/or at scales that have not previously been prioritised by agencies.

Iwi/hapū/rūnanga are also influencing resource consent conditions which may include contaminant/bioaccumulation assessments that include heavy metals in water, sediments and/or mahinga kai species; however, generally these reports are not accessible in the public domain.

C2-(iv). NGO, community driven

Part D—Impact analysis.

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

A business-as-usual scenario would lead to on-going increase of metals in sediment and have detrimental impacts on exposed ecosystems. There is no doubt that the accumulation of anthropogenic pollutants in the environment is causing harm and scientists need to work with other stakeholders to reduce pollution [28]. Metals are not degradable so any continuous discharges will accumulate in the various environmental compartments including biota. The impacts of human activities have pushed estuarine and coastal ecosystems far from their historical baseline of rich, diverse, and productive ecosystems [26]. The impacts on freshwater ecosystems are also considered to be significant. Managing the sources is a priority to ensure the protection of these valuable ecosystems and to protect water supplies from contamination. Encouragingly, there are examples of declining metal concentrations from improved environmental controls on emissions and discharges of metals and other contaminants, e.g., [10].

There are multiple challenges to reduce the discharge of metals in urban and rural environments, particularly non-point sources like stormwater. There are examples of options to reduce metals at the sources summarised in the PCE report, but they may be challenging to implement [8]. For example, an initiative to impose restrictions on the maximum amount of zinc in galvanised or zinc coated roofing materials may be opposed by those who manufacture these materials [8].

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

Freshwater, coastal and ocean ecosystems provide commercial, cultural, recreational and economic benefits as well and they support diverse habitats and species of local and global significance [20]. It is well-recognized that healthy and thriving coastal and freshwater ecosystems are essential for economic growth and food production [20]. The key impacts from the pressure that metals place on receiving environments is the potential loss in biodiversity and disruption of ecosystem functions and services through shifts in distributions of key species. The economic implications resulting from the impacts of metals would be loss of revenue for fishery and aquaculture industries in both freshwater and marine environments that are most likely to be impacted by pressure from metal contamination. Healthy and functional ecosystems and healthy fish stocks are important for the freshwater and marine fishery industries [29]. There are also other aspects to consider including natural beauty and recreational use of our freshwaters, estuaries, coastal and open ocean areas that are central to our culture and national identity and support our tourism industry. Furthermore, there are likely to be increased costs and economic impacts associated with the need for removal of trace metals from water supplies to ensure its safe use for drinking water, stock water, irrigation and other industrial and agricultural uses if this attribute were not managed.

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

Climate change will have multiple effects in modulating the accumulation and bioavailability of metals. Climate change increasingly affects the variation in volume and frequency of stormwater events and runoff which can increase transport of trace metals in dissolved and particulate form and resuspension and direct exposure of sediments in water bodies [1]. A key concern with the effects of climate change on the risks associated with metal contamination is that changes to temperature and pH can modulate the speciation of metals and consequently their bioavailability. The importance of metal speciation cannot be overstated as it modulates the bioavailability and toxicology of trace metals. The simplest feature of speciation is whether the metal is in the dissolved or particulate form. Originally, environmental regulations were based on total metals present in the water as assayed by hot acid digestion of the samples. However, there has been a gradual change in many jurisdictions to regulations based on the dissolved component only. This reflects the general recognition that particulate metals exhibit negligible toxicity and bioavailability to aquatic organisms relative to dissolved metals [2]. Increases in temperature have been correlated with increasing toxicity of metals to aquatic organisms [30]. As such, temperature should be accounted in risk assessment, because it may modify the effects of chemicals on the structure and functioning of aquatic communities, especially at higher levels of biological organization [31].

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8.4 Groundwater depletion

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State of knowledge of the "Groundwater depletion" attribute: Excellent / well established – comprehensive analysis/syntheses; multiple studies agree.

Part A—Attribute and method

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

Groundwater depletion, primarily caused by excessive extraction for agricultural, industrial, and domestic purposes [1-4], presents significant challenges to both ecological integrity [5-7] and human well-being [8,9], despite its susceptibility to natural phenomena such as prolonged low precipitation.

Ecologically, depleting groundwater affects interconnected surface water systems because they form a single, hydraulically connected resource [10,11]. Consequently, groundwater depletion can lead to the drying up of wetlands, streams, and rivers, disrupting habitats and endangering species reliant on these vital water sources. Furthermore, it adversely affects vegetation health, resulting in diminished biodiversity and altered ecosystems within groundwater aquifers [12] and above ground environments alike. Additionally, depleting fresh groundwater resources increases the potential for seawater intrusion in coastal areas, and the consequences of contamination can be detrimental, requiring lengthy remediation processes through careful management.

Groundwater depletion is the primary cause of land subsidence, which poses a potential threat to the ecological integrity of the area due to loss of support or underground caving in certain geological settings [13-15]. The repercussions of groundwater depletion extend to human health, as lowered groundwater levels often result in the cessation or reduction of access to groundwater extraction from shallow drinking water wells [16], leading to physical health issues (e.g., kidney stones, urinary tract cancers and some colon cancers) due to inadequate water consumption [17,18] and mental health concerns [19,20].

Groundwater depletion also frequently results in diminished water quality within aquifers and surface water bodies, particularly pronounced during periods of low rainfall when these resources heavily rely on groundwater recharge [1,21]. This heightened vulnerability to pollutants in the remaining groundwater exacerbates health risks.

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

There is a substantial body of international and national literature that demonstrates the impact of groundwater depletion on ecological integrity and human health. Studies have shown that reduced water levels in some streams, both in Aotearoa-NZ [22,24] and overseas [25-28], have compromised ecological integrity due to depleted groundwater levels. Recent evidence also highlights the effects on groundwater-dependent ecology caused by decreasing groundwater levels [27,29,30]. Ecological impacts due to land subsidence, resulting from excessive groundwater abstraction and consequently depleted groundwater levels, have been reported in Auckland [31], Wairakei [32,33] and potentially in Canterbury [15], and is a significant problem in many countries, including populated areas such as California, Florida, Arizona, Helsinki, Bangkok, Mexico City, and Jakarta [13,14,34-37].

Nationally, the depletion of groundwater wells has impacted human physical and mental health. Shallow wells in the Canterbury region began to run dry in the 1970s [16], and Environment Canterbury identified that the situation worsened over the last two decades, particularly around the Ashburton and West-Melton area [38]. There is also substantial international evidence showing health issues related to groundwater depletion [39-44]. Groundwater depletion is linked to increased concentrations of pollutants, due to lower water volumes available for dilution, which heightens health risks. These risks include gastrointestinal illnesses [45-46], reproductive disorders [47], and neurological ailments [46], especially in communities reliant on groundwater for drinking and irrigation purposes.

The spatial extent and magnitude of degradation caused by depleting groundwater vary considerably due to factors such as geology, aquifer characteristics, heterogeneity of the aquifer, recharge rates (due to both natural [precipitation] and anthropogenic influence [irrigation]), groundwater abstraction volumes and their patterns, as well as surface-groundwater interactions. Therefore, it is crucial to consider aquifer or site-specific properties to minimise degradation resulting from groundwater depletion [48].

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

The pace and trajectory of groundwater depletion varies significantly in Aotearoa-NZ and overseas. Typically, humans tend to access surface water, if available, before developing groundwater resources, as it is generally cheaper to abstract surface water than to pump from deep aquifers. Globally, the pace of groundwater depletion is most rapid in arid and semi-arid regions, as well as in highly populated areas [49-52].

Similar trends can be seen in regions such as Canterbury in Aotearoa-NZ due to the high demand for groundwater resources. In Canterbury, groundwater abstraction for domestic and stock water purposes started in the early 1900s. Environment Canterbury data show a significant increase in consents for groundwater use for domestic supply and industrial purposes around 1970.

Since the early 1990s, consents for groundwater use have increased considerably, mainly for irrigation [53]. With the current trend of high demand for groundwater, it is predicted that groundwater depletion will continue over the next 10 to 30 years. Increased temperatures and more drought-prone conditions are projected under climate change scenarios for many parts of the world.

Along with associated longer growing seasons, it is likely that groundwater demand for irrigation will increase, resulting in further groundwater depletion [54-56], unless interventions are taken to remedy the adverse trend. The reversal of impacts can be achieved through appropriate management of groundwater abstractions and approaches such as managed aquifer recharge (MAR) [1]. However, the rate of reversal is dependent on many factors, including aquifer properties.

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

There are several methods for monitoring groundwater depletion. The most widely used methods in Aotearoa-NZ are groundwater level measurements using observation wells or piezometers. Piezometers are specialised wells used to measure the pressure head at specific depths, providing detailed information about groundwater levels in different aquifer layers. Regional authorities maintain strategically positioned groundwater level networks to monitor groundwater levels [57]. These networks are typically installed in areas of high groundwater demand and different hydrogeological units to support predicting trends, identifying potential risks linked to excessive extraction, and supporting sustainable water use. The monitoring is usually conducted according to the National Environmental Monitoring Standards (NEMS).

Historically, groundwater levels were measured manually using a dipping device (inserting a metal tape into the well until it contacts water). However, the majority of councils now utilise automated methods for continuous measurements (telemetered sites). The telemetered data should be cross-checked periodically against manual measurements to ensure accuracy. The groundwater level measurements are typically reported relative to the New Zealand Vertical Datum 2016 for meaningful comparison between sites and other water bodies (e.g., rivers and lakes). Groundwater levels are generally reported as time series plots for each well or a combination of wells within a cluster such as a hydrogeological unit.

Given the high cost associated with installation, most monitoring is targeted in high-demand areas, and the monitoring network can potentially be insufficient to provide adequate information about the complete groundwater level dynamics within a large area, such as an entire catchment or region. This limitation arises because groundwater flows in the "downhill" direction, similar to surface water, and due to the potential energy associated with pressure [11] (hydraulic head). Excessive abstractions in one area can potentially impact other areas of the aquifer to which they are hydraulically connected due to the pressure differential.

Other approaches used to support understanding groundwater depletion include Water Budget Analysis [58] (calculating the balance between inputs [recharge, inflow] and outputs [extraction, outflow]), Geochemical Methods such as isotope analysis [59] and water quality monitoring [60]. Additionally, remote sensing techniques are increasingly used to understand groundwater depletion, especially over large geographical areas. The most common remote sensing approaches are the NASA’s Gravity Recovery and Climate Experiment (GRACE) and GRACE Follow-On (GRACE-FO) [61,62], and InSAR (Interferometric Synthetic Aperture Radar) [63,64]. The GRACE satellites measure changes in Earth's gravity field, which can be used to infer changes in groundwater storage over large regions (approximately 400 km by 400 km). InSAR uses satellite radar data to detect ground surface deformation, which can be related to groundwater extraction.

There is no consensus on the most appropriate measurement method, as the spatiotemporal scale of accuracy and cost associated with each method vary significantly. For example, a monitoring well, which involves considerable investment, is more appropriate for assessing groundwater depletion at a single location, while remote sensing provides information for a larger area at relatively low cost, but lower accuracy for any single location. Therefore, the monitoring method needs to align with the decision-making needs. However, by combining the above methods, along with scientific approaches such as trend analysis of groundwater levels [65] and considering the dynamics of interconnected systems (e.g., climate, river flow, water abstractions, land use changes, tidal effects), a detailed and accurate picture of groundwater depletion can be obtained.

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

The implementation issues of monitoring and reporting are primarily dependent on the approach used. For example, using 'invasive' methods such as groundwater wells or piezometers requires obtaining permission from landowners to install the well or piezometer and ensuring ongoing access for monitoring and maintenance. It may also require compliance with privacy laws to use or publicise information, if relevant. The cost of telemetry approaches for monitoring depends on the availability of telecommunication options for the site. Using satellite technology for data transfer in areas without mobile coverage can be expensive.

In addition, the performance of wells can be compromised due to sedimentation and damage from natural events (e.g., earthquakes) or vandalism. Therefore, it is important to take necessary precautions and implement QA-QC measures to ensure the data is reliable. Failing to use consistent measurement standards and a common datum can impact the ability to accurately utilise monitored data for regulatory and management purposes.

While this is not a significant issue for assessing groundwater level changes, which are typically slower than surface water level changes, temporal resolution of data availability and cloud cover and weather conditions may hinder the use of remote sensing data for high temporal resolution analysis. Similarly, remote sensing methods like GRACE only offer coarse spatial resolution (e.g., 400 km by 400 km), which may not be suitable for detailed local or regional analysis [66,67].

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

The costs associated with monitoring groundwater depletion vary significantly depending on the approach used and site-specific conditions. The up-front cost of monitoring wells, commonly used by regional councils to measure location-specific groundwater levels, depends on factors such as the well's depth and diameter, screen length, site accessibility for drilling, and the time required to develop the well (to remove sediment and clean the well). The cost can range from less than $10,000 to more than $100,000 per well.

The capital cost for data transmission also depends on the medium used, such as cellular or satellite, and ranges from less than $10,000 to more than $20,000. The ongoing costs can be considerable for manual sites due to staff time requirements. For automated sites, the ongoing cost may range from less than $100 to more than $250 per year, depending on whether it uses cellular networks or satellite. However, the cost can increase to more than $5,000 if the temporal resolution of data transfer is high, such as 15-minute intervals. Additionally, ongoing costs include data storage, QA-QC of data, and data analysis.

The costs associated with the use of remote sensing data also vary with many factors. For example, while the raw data from GRACE is freely accessible, the total cost of using GRACE data can vary significantly depending on the user's specific needs and resources, including software, computational resources, expertise, and potential consulting services.

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

The author is not aware of any groundwater depletion monitoring being specifically undertaken by iwi/hapū/rūnanga. However, several organisations use groundwater level data for their research (e.g., Te Rūnanga o Ngāi Tahu, Te Kura Taka Pini research unit) and business operations (e.g., Ngāi Tahu Farming).

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

Groundwater depletion has known correlations or relationships with attributes including riparian margin establishment/protection, surface water flow alteration, catchment permeability and groundwater nitrate [3,27,68-71].

Part B—Current state and allocation options

B1. What is the current state of the attribute?

Understanding the current state of groundwater depletion varies significantly between and within regions. For example, monitoring by Environment Canterbury indicates that groundwater levels have been affected by abstraction [72]. In the Kaituna catchment in the Bay of Plenty, where the demand for groundwater is the highest in the region, trend analysis of groundwater levels reveals mixed results. Among the 22 groundwater level time series analysed, only two showed statistically significant long-term decreasing trends. Additionally, the study found that climate (i.e., natural factors), rather than groundwater use, is the primary factor controlling groundwater levels [65].

Based on these examples, it is reasonable to conclude that while understanding the current state and trends in groundwater depletion is crucial for freshwater management, this attribute should be assessed in conjunction with interconnected systems such as climate, land surface recharge, stream flows, and water abstractions to accurately ascertain changes in state.

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

Typically, currently available monitoring does not inform the natural reference states in Aotearoa-NZ for two main reasons: (1) most groundwater level monitoring sites are located in high-demand areas where groundwater levels are affected by abstractions, and (2) current land cover and use do not represent natural conditions. Measurements would need to be taken in wilderness areas such as Fiordland which would be difficult and expensive. Due to these challenges, it is generally not feasible to directly develop natural reference states of groundwater levels through monitoring in most areas.

An alternative approach to develop natural reference states for groundwater levels is to utilise hydrological modelling approaches [3,73] to support management and allocation options, including the limit-setting process under the NPS-FM. However, these modelling approaches can only represent a “pseudo-natural state”, such as no water abstractions and no irrigation, and typically simulate the current land cover rather than a "pure" natural state due to a lack of detailed information about the former natural state. The accuracy of these models may also be affected by other anthropogenic influences such as dams, diversions, and discharges, which impact groundwater level dynamics.

Nevertheless, these models can provide a useful tool for informing groundwater management by simulating conditions that approximate natural states as closely as possible given existing constraints.

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

There are no globally accepted default numeric or narrative bands for groundwater depletion levels to ensure sustainable water resource management, ecological integrity, and human health. This is primarily due to the spatial heterogeneity in aquifer dynamics, where the same absolute or proportional change in groundwater levels can produce different outcomes in different locations. For example, surface-groundwater interactions vary spatially, and the rate of outflow from an aquifer through groundwater discharge to surface water features will spatially differ if the same level of change is applied. Sustainable groundwater management ideally requires a location-specific balance of recharge, discharge, and extraction.

Regional councils typically develop location-specific numeric or narrative bands based on water balance assessments or modelling approaches. In the absence of a detailed assessment, adaptive management thresholds can be used in groundwater management. This approach allows for responding to changing conditions and new information through appropriate monitoring and a review process.

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

As described above in B3, there are no universal numeric or narrative bands to determine thresholds or tipping points that specifically affect ecological integrity or human health. The most effective approach is to develop location-specific abstraction limits that maintain a sustainable level of groundwater depletion. This ensures an appropriate level of outflow to surface water bodies (baseflow) and oceans, which is crucial for reducing the potential risk of seawater intrusion. These measures support long-term ecological integrity and human health by maintaining the natural hydrological balance and preventing adverse environmental impacts.

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

In general, groundwater dynamics are associated with lag times. However, lags are widely variable due to many factors including geology, surface-water interaction, climate and water abstraction patterns. For example, a recent study of radiocarbon dating of groundwater samples estimated that the groundwater in some coastal South Canterbury areas is up to 11,000-21,000 years old [74], indicating a very slow flow through the aquifer. In contrast, other sites contain younger groundwater [75,76], implying more rapid flow rates through the aquifer. Consequently, lag time of impact of groundwater abstraction from an aquifer and climate cycles, and subsequent effect of groundwater depletion, varies between and within aquifers [77,79].

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

Māori have a range of values, beliefs and practices associated with groundwater ecosystems that are underpinned by a holistic and integrated understanding of the water cycle and the environment as a whole [90]. Groundwater-dependent cultural values, beliefs and practices encompass cultural landscapes and settlements, wāhi ingoa (place names), wāhi tapu (sacred places) and wāhi taonga (treasured places), rongoā (healing) and ceremonies (e.g., burials), mahinga kai (e.g., spring-fed streams), tuhitera neherā (rock art), marae water supplies and indigenous biodiversity (e.g., [91-93]).

There are examples of where groundwater features have been afforded the status of wāhi tapu and wāhi taonga. For example, Ōmaru puna wai in the Canterbury region was registered with the New Zealand Historic Places Trust as a wāhi tapu in 2005 [94]. Ratana et al. (2017)[95] developed an approach for Ngā Tai o Kāwhia whānau to express their aspirations for wetlands and puna, and the enhancement of important taonga species that utilise them. A framework was developed to support whānau prioritisation of sites for restoration based on their uses and associations with repo (swamps) and puna (springs) in the Kāwhia rohe.

As explained in B3 and B4, it is not feasible to develop universal spatial thresholds to limit groundwater depletion and achieve uniform outcomes. However, determining location-specific levels of allowable groundwater depletion due to human activities should consider the impact on surface water bodies and, consequently, on tikanga Māori and/or mātauranga. This approach supports cultural flow preference studies (or adaptation of this published method), such as those developed for harvesting mahinga kai from various freshwater habitats [80,81].

Part C—Management levers and context

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

There is a well-understood direct relationship between stressors and groundwater depletion. These stressors can be divided into two main groups: natural and anthropogenic. Natural stressors primarily include low precipitation, increased evaporation, and low river flows in areas that recharge groundwater. Anthropogenic stressors include groundwater abstractions, river flow alterations (e.g., surface water abstraction, diversions, impoundments), and changes in land use and management.

Although there is a relationship between stressors and groundwater depletion, it is generally challenging to quantify and attribute the influence of each stressor due to the complex interconnections within the system, as well as lags and legacy effects. To better understand these relationships, process-based distributed hydrological models can be used. However, developing such models is resource-intensive and often prohibitive for many areas. Additionally, these models are often developed to be specific to particular sites and cannot easily be applied to different spatial locations [82]. To overcome these limitations, several statistical modelling approaches have been developed to estimate the relative contributions of changes in natural systems and human activity to groundwater depletion [38,83,84].

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

Key management interventions used to address groundwater depletion include developing sustainable water allocation limits for aquifers using cumulative impact assessments and restricting water abstractions when groundwater levels fall below certain thresholds, based on predefined rules and groundwater level monitoring of a reference well. The former can be specified for each aquifer in the regional or freshwater management unit plan. The latter can be implemented by including conditions in each water take resource consent to restrict water abstractions.

However, there are many barriers to setting sustainable limits and implementing water abstraction restrictions to achieve the desired outcomes. For example, determining sustainable water allocation limits may be challenging due to limitations in available data needed for a robust assessment. Similarly, the impact of restrictions on water abstraction from different spatial locations, even within the same aquifer, can vary due to the heterogeneity of the aquifer and its hydraulic connection to surface water [48]. Therefore, designing an accurate restriction regime with variable spatial restriction limits with the same aquifer is challenging and can be perceived as an unfair management strategy by some water users.

Given the complexity and resource-intensive nature of developing sustainable water allocation limits and water restriction management plans, the uncertainty associated with the developed numerical values (for both allocation limits and restrictions) can be significant. Therefore, it is advisable to utilise an adaptive management plan with appropriate monitoring and a review process. Such adaptive management should also include the ability to alter the developed numeric values for limits and water restrictions as soon as new knowledge is gained to avoid environmental degradation, as some aquifer systems may take years to recover from prolonged adverse activities.

Both the regional plan and consent conditions should refer to separate schedules that can be altered through adaptive management for allocation limits and restriction triggers, rather than specifying them directly in the regional plan and consent, if legally allowed. For example, specifying water take restriction conditions as numeric values in the resource consent ("hard coded") may prevent changes until the consent is due for renewal, even if adaptive management findings illustrate that the consented water take regime is unsustainable.

C2-(i). Local government driven

Regional councils and unitary authorities set allocation limits for each groundwater aquifer or Groundwater Allocation Zone (GAZ) using a variety of approaches. These approaches include percentages of average annual rainfall, soil-crop-water balance modelling, water balance assessments, field investigations (e.g., use of lysimeters), mathematical or statistical modelling, and physically based modelling.

Full or partial groundwater abstraction restrictions are implemented by regional authorities through consent conditions to achieve various outcomes, such as preventing groundwater level depletion, supporting sufficient groundwater discharge to surface water bodies to prevent ecological degradation, and preventing saltwater intrusion. In addition to using consent conditions for water take restrictions, regional authorities can also use Section 329[1] of the RMA to issue water shortage directions.

When implemented appropriately, full or partial groundwater abstraction restrictions can be an effective intervention. For example, Tasman District Council (TDC) identified seawater intrusion into the coastal areas of the Hau Plains during low rainfall periods (droughts) through their monitoring network. TDC implemented water restrictions in the affected area for droughts exceeding a 1-in-10 year event, as stipulated in their Resource Management Plan, which has proven to be an effective means of reducing the risk of seawater intrusion.

C2-(ii). Central government driven

The central government policies related to this attribute are outlined in the NPS-FM 2020, which applies to all freshwater (including groundwater) and, to the extent they are affected by freshwater, to receiving environments such as estuaries and the coastal marine area.

NPSFM clause 3.16 (3) states that “Environmental flows and levels must be expressed in terms of the water level and flow rate, and may include variability of flow (as appropriate to the water body) at which: (c) for levels of groundwater: any taking, damming, or diversion of water meets the environmental outcomes for the groundwater, any connected water body, and receiving environments.”

Under Te Tiriti o Waitangi, it is the duty of the partnership to manage freshwater [85], including groundwater, in a manner that meets the cultural values of mana whenua.

C2-(iii). Iwi/hapū driven

Iwi/hapū/rūnanga provide input into the development of regional plans and have developed their own iwi environmental management plans that include objectives, policies and/or methods to support the groundwater-related outcomes they are seeking. For example, the Mahaanui Iwi Management Plan [Canterbury region] policy WM8.6 requires that: “aquifers are recognised and protected as wāhi taonga. This means: (a) The protection of groundwater quality and quantity, including shallow aquifers; (b) The protection of aquifer recharge; (c) Ensuring a higher rate of recharge then abstraction, over the long term; (d) Continuing to improve our understandings of the groundwater resource, and the relationship between groundwater and surface water” [94].

There are examples of where groundwater features have been afforded the status of wāhi tapu or wāhi taonga. For example, Ōmaru puna wai in the Canterbury region was registered with the New Zealand Historic Places Trust as a wāhi tapu in 2005 [94].

C2-(iv). NGO, community driven

It is difficult for NGOs or community-driven organizations to implement large-scale interventions to address groundwater depletion beyond contributing to resource management planning. However,

[1] Under s329 of the RMA, regional councils and unitary authorities can issue water shortage directions at any time there is a serious temporary shortage of water in its region or any part of its region. The direction may apportion, restrict, or suspend the taking, use, damming, or diversion of water.

many NGOs, community groups, and conservation organisations have voiced concerns about environmental degradation due to depleting groundwater levels. For example, the Environment and Conservation Organisations of New Zealand, a non-profit network of over 45 organisations concerned with conservation and the environment, works with international networks and engages with central and regional governments to advocate for environmental issues. Another example is Fish & Game, which has reported several instances of water shortages in some Canterbury streams and considers there to be a link between low stream flows and groundwater abstractions in inland Canterbury [22].

Interventions currently being employed by NGOs, community groups (including iwi/hapū/rūnanga) and conservation organisations in New Zealand to improve freshwater quality can also conceivably help to increase water availability in catchments including groundwater. These interventions include creation of new hard or soft infrastructure to increase water storage and surface water/aquifer recharge in catchments (e.g., reservoirs, ponds, wetlands, plantings and organic soils.

C2-(v). Internationally driven

From an international perspective, the United Nations General Assembly has recognised the human right to safe and clean drinking water and sanitation. Groundwater depletion undermines this human right. The UN-Water Summit on Groundwater 2022 called on governments for accelerated actions towards sustainable groundwater development.

In 2005, the UN's Educational, Scientific and Cultural Organisation, along with the Food and Agriculture Organisation, developed a landmark document titled “Groundwater in International Law - Compilation of Treaties and Other Legal Instruments” to highlight the importance of groundwater resources in international law [86].

Another international initiative is the Groundwater Project, a non-profit organisation committed to advancing education by creating and providing free high-quality groundwater educational materials online for everyone (https://gw-project.org/).

Other international obligations to prevent adverse groundwater depletion include climate change commitments under the Paris Agreement and conditions under Free Trade Agreements.

Part D—Impact analysis.

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

Failing to manage long-term groundwater depletion can have significant environmental and human health impacts. Environmentally, many ecosystems depend on groundwater. Wetlands, rivers, and lakes that rely on groundwater inputs can dry up, leading to the loss of biodiversity and the degradation of aquatic habitats. Over-extraction of groundwater can cause land subsidence, resulting in permanent loss of aquifer storage capacity and damage to infrastructure such as buildings, roads, and pipelines. In coastal areas, reduced groundwater levels can lead to saltwater intrusion, contaminating freshwater aquifers with seawater, making the water unusable for drinking and irrigation.

Human health is also significantly impacted by groundwater depletion. Depleted groundwater resources mean less availability of water for drinking, sanitation, agriculture, and industrial use, leading to water scarcity and associated health issues such as dehydration, poor sanitation, and food shortages, and mental health issues. Lower groundwater levels can lead to the concentration of contaminants like arsenic, fluoride, and nitrate in the remaining water, posing serious health risks such as poisoning, dental and skeletal fluorosis, and other chronic illnesses. Reduced agricultural productivity due to water scarcity can lead to food insecurity, malnutrition, and increased poverty, further exacerbating health problems. Effective management of groundwater is crucial to prevent these adverse environmental and human health impacts. Sustainable practices and policies should be implemented to ensure the long-term availability and quality of this resource.

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

The economic impacts groundwater depletion would likely be felt across various sectors and communities, with significant repercussions for both rural and urban areas due to adverse levels of groundwater depletion.

- Agricultural sector: The agricultural sector, particularly viticulture, horticulture, and dairy farming, relies heavily on groundwater for irrigation across the country. Depletion would lead to reduced water availability for irrigation, resulting in lower crop yields, reduced crop quality, and decreased productivity. Insufficient water for other needs, such as produce processing, stock water, milk cooling, and dairy shed washing, would directly affect farmers' incomes. While pastoral farmers (e.g., dairy) may be able to purchase supplementary feeds (e.g., seeds) for animals, potentially at a higher cost, to mitigate the effects of a drought, this option is not feasible for horticulture or vegetable production [87]. Such impacts on horticulture are likely to be severe in areas like Pukekohe, Heretaunga Plains in Hawke's Bay, and Poverty Bay Flats in Gisborne. Businesses that supply agricultural inputs, such as seeds, fertilizers, and equipment, would also suffer as farmers cut back on spending due to reduced revenues. Lower agricultural output could increase the cost of raw materials, impacting food processors and manufacturers.
- Communities: Communities in rural areas often depend on groundwater for drinking water and domestic use. Depletion could lead to water shortages, necessitating costly alternatives like water trucking or infrastructure investments to access alternative sources. Reduced agricultural productivity can have a ripple effect on local economies, leading to decreased spending in rural towns and potentially causing job losses in agricultural and supporting sectors.
- Hydropower generation: Groundwater depletion reduces river baseflows, particularly during summer, leading to lower water levels in reservoirs. This decreases the availability of water for hydropower generation, reducing electricity output and potentially causing energy shortages. Consequently, reliance on alternative, often less sustainable energy sources may increase, impacting both the economy and environment.
- Tourism sector: Impacts can threaten the tourism sector by diminishing water resources essential for hospitality services and outdoor activities. It undermines the

country's "clean and green" image, deterring visitors attracted to pristine natural environments and potentially leading to economic losses in tourism-dependent communities.

- Coastal areas: Groundwater depletion can lead to saltwater intrusion, affecting freshwater supplies and ecosystems that support fisheries and agriculture sector. This would impact businesses and communities reliant on these industries.
- Public health and environment: Degraded water quality due to groundwater depletion can increase the incidence of waterborne diseases and chronic illnesses, leading to higher healthcare costs for communities and the government. Costs associated with environmental degradation, such as loss of wetlands and decreased river flows, may require expensive restoration projects funded by taxpayers.
- Infrastructure: Groundwater depletion can cause land subsidence, damaging infrastructure such as roads, bridges, and pipelines, leading to costly repairs and maintenance.
- Government and policy: The government may face higher costs related to increased regulation, monitoring, and enforcement efforts to manage the remaining groundwater resources effectively. There may also be a need for economic support programs to assist affected farmers and communities, placing additional strain on public finances.

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

Groundwater depletion will be significantly affected by climate change, as altered precipitation patterns, increased evaporation rates, and prolonged droughts are likely to reduce natural groundwater recharge, particularly in eastern parts in Aotearoa-NZ [88], and consequently reduce discharge to rivers [89]. Rising temperatures can exacerbate water scarcity by increasing demand for irrigation in agriculture [54], leading to more intensive groundwater extraction. In coastal areas, sea-level rise and reduced freshwater recharge can increase the risk of saltwater intrusion into freshwater aquifers, further degrading groundwater quality and availability. These changes necessitate a proactive management response to mitigate the impacts of climate change on groundwater resources.

Effective management strategies will need to include enhanced monitoring of groundwater levels and quality, coupled with adaptive management practices that can respond to changing conditions. This will involve the implementation of more robust water allocation frameworks that prioritise sustainable use and protect critical recharge zones. Additionally, the development and promotion of water-efficient technologies and practices in agriculture, industry, and domestic use will be crucial. Managed aquifer recharge (MAR) and water storage infrastructure projects (see Section C2 iv) can help augment natural recharge, and regulatory measures must be put in place to limit over-extraction and prevent contamination.

If no action is taken to address groundwater depletion, the consequences will be severe. Groundwater levels will continue to decline, leading to the deterioration of ecosystems, reduced agricultural productivity, and compromised water supply for communities. This will exacerbate water scarcity, potentially leading to conflicts over water resources. The increased concentration of contaminants in dwindling groundwater supplies will pose serious health risks, making the need for protection and sustainable management even more critical. Protecting groundwater resources is not just important but essential to ensure water security and environmental health in the face of climate change.

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8.5 Surface water flow alteration

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Preamble: Surface water flow regimes are a fundamental driver of river ecosystem integrity and support human health. Alteration of surface water flow regimes must be managed to ensure ecosystem integrity and human health is not compromised. However, surface water flow alteration (SWFA) cannot be directly related to ecological integrity or human health.

There are significant technical barriers to each step needed to translate the concept of SWFA into a National Objectives Framework (NOF)-type SWFA attribute (i.e., defining SWFA, representing SWFA as a metric, relating the SWFA metric to ecological integrity and/or human health, setting bands or limits for the SWFA metric, measuring/estimating SWFA, linking management actions to outcomes in the metric of SWFA).

Quantified relationships between different aspects of surface water flow regimes (magnitude, duration, frequency, timing, rate of change) and ecological integrity and/or human health are essential for environmental management and reporting because many values are influenced by various aspects of surface water flow regimes. Predictive relationships would allow decision makers and communities to assess what combination of management practices might be required to meet freshwater objectives.

See the following reference for full discussion: Booker, D.J., Franklin, P.A., Stoffels, R (2022) A proposed framework for managing river flows to support implementation of the NPS-FM. NIWA client report prepared for MfE.

State of knowledge of "Surface water flow alteration" attribute: Good / established but incomplete – general agreement, but limited data/studies

Part A—Attribute and method

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

It is widely accepted that river flows are a master variable that is linked to various physical, chemical, and ecological states that are in-turn linked to ecosystem health, human health, cultural wellbeing, landscape character, recreation, and water supply for out-of-stream use (Poff et al., 1997; Sofi et al. 2020). Surface water flows in rivers, lakes, wetlands, and estuaries are a strong driver of ecological integrity and human health because surface water flow is the fundamental process determining the size, shape, structure, and dynamics of riverine ecosystems (Zeiringer et al., 2018). There is a strong case for surface water flows being a main driver of ecological integrity in freshwater systems. Adequate flow of water in rivers, wetlands, lakes, and estuaries is vital for maintaining ecological integrity associated with in-stream values such as healthy ecosystems, basic human health (Gleick, 1998), and local customary practices (Stewart-Harawira, 2020). However, the effects of surface water flows on ecological integrity and/or human health are complicated for a variety of reasons.

- The effects of surface water flows on ecological integrity and/or human health are mediated by landscape settings relating to climate, vegetation, sediment supply, geology, topography setting (altitude, slope, etc.), and local ecological processes. The same flow will produce different flow-driven effects in different locations.
- Different aspects of ecological integrity and/or human health will have different relationships with surface water flows. The strength, magnitude, and direction of flow-ecology relationships will vary depending on the in-stream value. For example, some species would benefit from more disturbed flow regimes whereas other species may prefer more benign flow regimes. Flow-ecology relationships will also vary depending on landscape setting (Snelder et al., 2011). For example, in highly disturbed aquatic habitats the frequency of flows above a threshold for sediment transport could be seen as important for ecosystem functioning (Jellyman et al., 2013), whereas in spring locations the threshold for sediment transport may be reached very rarely.
- Surface water flows are constantly changing due to weather patterns. “River flow regime” is a phrase often used to describe the collective long-term properties of a river flow time-series, including the duration, magnitude, and variability of low flows, high flows, and flow seasonality.
- Aside from cessation of flow, it is very difficult to define and isolate the influence a single component of the flow regime as being particularly important for overall ecological integrity and/or human health. For example, higher flows relate to flushing of sediment, mid-range flows relate to cuing of fish movements, whereas low flows relate to provision of space, maintenance of temperature or dissolved oxygen.
- Many components of natural flow regimes are highly correlated, but these correlations can be broken under altered flow regimes. Thus, unintended consequences can arise from selecting a sub-set of statistically independent hydrological variables as attributes to represent all aspects of the flow regime for environmental management or environmental reporting purposes. For example, summer minimum flow may be statistically representative of summer flow conditions, but sole application of this variable to set limits to hydrological alteration will allow changes in summer mean flow or variability, and sole application of this variable for environmental reporting will obscure changes in summer mean flow or variability. Conversely, application of many hydrological variables to set limits to hydrological alteration is not feasible because it is impossible to link management actions (e.g., via consent conditions) with multifaceted hydrological characteristics.

The following concepts are important when considering the appropriateness of SWFA as an attribute for environmental management or national environmental reporting.

- Surface water flows should be expected to be constantly varying due to changes in weather regardless of local human activities or management interventions. For example, even in natural catchments, different weather patterns in each year will produce different surface water flows. Thus, the influence of management actions on SWFA could be masked by weather/climate patterns, and numeric or narrative bands for SWFA could be exceeded naturally.
- Surface water flow alteration cannot be measured directly. River flows can be measured. Continuous flow time-series are usually measured at gauging station sites. Instantaneous flow can be measured at discrete times usually using spot gauging techniques and cannot capture flow variability. In theory, the sum of measured consumptive water use/manipulation can be compared to measured river flows to estimate surface water flow alteration. However, such a comparison can only be applied at gauging stations, and the hierarchical nature of river flows means that gauging stations are not necessarily representative of their upstream catchments or nearby catchments. More importantly, estimates of SWFA will be highly uncertain because many abstractions are not measured, some abstracted water may return to supplement surface water flows, and the timing and extent of groundwater abstractions on surface water flows is highly uncertain, especially if aquifer properties are unknown (Zipper et al., 2021). In some cases, uncertainty in quantifying SWFA may be greater than the expected effect of alternative management actions.
- SWFA occurs over multiple temporal and spatial scales. Measurement location is particularly important when considering either the cumulative effects of multiple small abstractions distributed across the landscape (SWFA may be higher in tributaries and lower in main stems) or the large effects of a single large abstraction (SWFA may be very large just downstream of the abstraction but reduce with distance downstream as flow accumulates). See Booker et al. (2014) for demonstration of why the same rules for water take limits can lead to different outcomes across a catchment.
- Surface water flow alteration can be expressed as a relative alteration (i.e., a proportional reduction or a percentage reduction) or as an absolute alteration (i.e., a reduction in litres per second). Absolute alteration is meaningful within a site, but not meaningful when comparing between sites. Furthermore, the meaning of relative alteration for ecological integrity or human health may not be constant between sites or between flows within a site.
- SWFA requires a baseline state to be defined. This is a technically challenging task because it is not clear whether a baseline state should just account for water abstractions, or also account for landcover changes, and climate changes even though all three factors combine to alter flows (see Lapides et al., 2022).

If “attribute” is used in the same way as in the National Policy Statement for Freshwater Management-National Objectives Framework (NPSFM-NOF), then the concept of SWFA is difficult to reconcile as a NOF attribute that can be directly related to ecological integrity and human health. A NOF attribute would typically meet the following criteria:

- There are proven or theoretical links between the attribute and in-stream values, including ecological integrity and/or human health.
- The attribute is manageable because there are proven or theoretical links between the attribute and a management intervention.
- The attribute is measurable because there is a clearly described measurement and analysis procedure for quantifying the attribute as a single numeric value (e.g., the median of discrete samples taken at monthly intervals collected over a 5-year period).
- The attribute has bands that represent target states to be compared with observed data.
- Temporal patterns in the attribute can be tested for trend to determine whether its state is steady, deteriorating, or recovering.

The first and second criteria are more compatible with the concept of SWFA, but the remaining criteria are less compatible with the concept of SWFA. The concept of SWFA would have to be disaggregated into a set of definitive variables to meet the latter three technical criteria. However, many hydrological variables would likely be needed to represent SWFA or river flow regimes, and the meaning of alteration in hydrological variables would be different between sites and between variables within sites.

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

There is a large body of international and local literature that relates surface water flow regimes to various aspects of ecological integrity and/or human health conceptually. However, there are few studies quantifying impact of particular components of SWFA on particular components of ecological integrity and/or human health due to difficulties in defining and measuring SWFA combined with a lack of monitoring data suitable for detecting flow-driven impacts. See the following references for more details:

- Stoffels, R.J., Booker, D.J., Franklin, P.A., Holmes, R. (2024) Monitoring for the adaptive management of rivers. Journal of Environmental Management, 351, p.119787.
- Stoffels R.J., Booker, D.J., Franklin P.A., Holmes, R. (2022) Monitoring and Evaluation to Support Adaptive Management of River Flows, Part 1: rationale for design. NIWA client report prepared for MBIE-Envirolink.

<u>Extent and magnitude of degradation</u>. Before considering impacts of SWFA on ecological integrity and/or human health, we must first consider the extent and magnitude of SWFA itself. Water accounting methods are available to estimate the extent and magnitude of SWFA. Although uncertainties could be reduced by refining water accounting methods, they are not the limiting factor in calculating extent and magnitude of SWFA. Incompleteness of water abstraction data and sparsity of river flow data mean that reliable estimates of the extent and magnitude of SWFA are lacking. The influence of inter-annual climate variability (shorter-term cycles, or longer-term trends) is a further challenge to desegregating SWFA from background variability in surface water flows.

<u>Evidence of impact</u>. There is a wide body of international literature describing theoretical links between river flow regimes and ecological integrity and/or human health (e.g., Poff and Zimmerman,

2010; Poff et al., 2012). There is also a body of literature that reinforces the same general types of theoretical links in the NZ setting. Many NZ studies have investigated empirical, but not necessarily causal, links between river flow regimes and ecological integrity and/or human health. Examples include the following.

- Links between surface water flows and Māori cultural values (e.g., Crow et al., 2018; Tipa and Nelson, 2012). In relation to cultural values, readers are referred to the statement of Crow et al., (2018) that: “From the perspective of Māori cultural values, beliefs and practices supported by fresh water can be seen to compete with economic uses [irrigation and hydroelectric generation in the Waitaki catchment]. If the interests of Māori are to be weighed alongside the many other uses, and if environmental streamflow assessments and allocative decision-making are to benefit from the knowledge of whanau, hapū and iwi, new techniques are needed to assess the appropriateness of streamflows in culturally sensitive ways”. Further information on the role of surface water flows from a mātauranga Māori perspective can be found in Harmsworth et al. (2011), Harmsworth et al. (2016), Tipa et al. (2016), Te Aho (2019), Taylor et al. (2021), Taylor (2022), Tadaki et al. (2022) and references therein.
- Links between surface water flows and the distribution and abundance of fish because fish are influenced by several factors, particularly migration and habitat suitability, which are in turn strongly influenced by flow regimes (Crow et al., 2013, Booker et al., 2016). The specific effects of flow regimes on fish migration and habitat suitability are discussed in detail by Closs et al. (2016).
- Links between surface water flows and periphyton biomass. For example, Biggs (2000) linked periphyton biomass with nutrient concentrations and a hydrological variable defined as the frequency of high flow events exceeding three times the median flow (FRE3). Further work linking periphyton to nutrients, substrate and flows includes (Snelder et al., 2014, Snelder et al., 2019; Neverman et al., 2018)
- Links between surface water flows and stream macrophyte abundance, especially the frequency of high flow events exceeding seven times the median flow (FRE7) (Riis et al. 2003) and relationships to other variables including light, nutrients and temperature (Matheson et al. 2012).
- Links between surface water flows and invertebrate communities. For example, Greenwood et al. (2016) described a hydrologically sensitive invertebrate community index for New Zealand rivers, and Townsend et al. (1997) calculated the frequency of events exceeding different ecologically relevant thresholds when comparing various surface flow metrics of disturbance to macroinvertebrate species traits and species richness.
- Links between surface water flows and riverbed substrate composition (e.g., Jellyman et al., 2013; Haddadchi et al, 2018).
- Links between surface water flows and river water temperature (Booker and Whitehead, 2022).
- Links between surface water flows and dissolved oxygen (Franklin, 2014).

Despite the aforementioned studies, there are relatively few easily applied methods or tools that provide predictive relationships between SWFA (or just surface water flow magnitude and variability) on ecological integrity and/or human health. The main challenge for developing predictive models is disentangling the influence of SWFA from climate, other habitat, or biological (e.g., invasive species) influences.

Historically, physical-habitat modelling has often been used to make empirical links between magnitude of surface water flow and hydraulic conditions (water depth, water velocity, wetted width), and therefore availability of physical habitat deemed suitable for a target species (e.g., a life-stage of a fish species) (e.g., Jowett, 1990). However, suitability of available physical habitat has been criticized for being a metric that is not rationally linked to organism states, processes, or behaviours, and therefore not an appropriate representation of ecological integrity (e.g., Lancaster and Downes, 2010). Physical habitat has also been criticized for ignoring the need for flow regime variability, as well as links between flow regimes and both water quality and geomorphology.

<u>Extent and magnitude of degradation</u>. Water accounting methods are available to calculate the extent and magnitude of SWFA (e.g., Rouse et al., 2014; Booker et al., 2015). Water accounting has been applied to estimate streamflow depletion from measured abstractions (e.g., Booker, 2020). However, several factors have combined to limit knowledge about the extent and magnitude of SWFA.

- Lack of complete, high-quality, well-collated water abstraction data for all water abstractions across catchments, regions, or nationally, including data associated with abstractions that are:
  - allowable as permitted activities (e.g., stock drinking);
  - not obliged to provide records under current legislation (e.g., consented abstractions with low rates of maximum allowable abstraction); and
  - obliged to provide records under current legislation but not providing data.
- Lack of a systematic estimation of naturalised river flows to inform on SWFA, especially in relation to magnitude and temporal lags in streamflow depletion associated with groundwater abstractions.

Nationwide estimates of potential SWFA associated with consents for consumptive water abstraction have been used for national environmental reporting in lieu of information on actual SWFA (Booker et al., 2018). Consents indicate that water is abstracted from surface water and groundwater for various purposes, but the dominant purpose is irrigation (Figure 1). Maps of estimated potential flow alteration (Figure 2) show how cumulated consented water abstraction (the sum of all upstream rates of consented abstraction) compares to estimated median flow across the national river network (Figure 3; Booker, 2018). Results indicated the extent and magnitude of pressure for surface flow alteration under the worst-case-scenario of groundwater flow being assumed to be fully depleting nearby river flow and all consents being simultaneously exercised. The extent of potential flow alteration was widely spread across several regions and of high magnitude (e.g., summed rates of maximum allowable abstraction exceeded the estimated median flow) in many locations due to the position and magnitude of consented abstraction upstream of some river reaches with relative low flows.

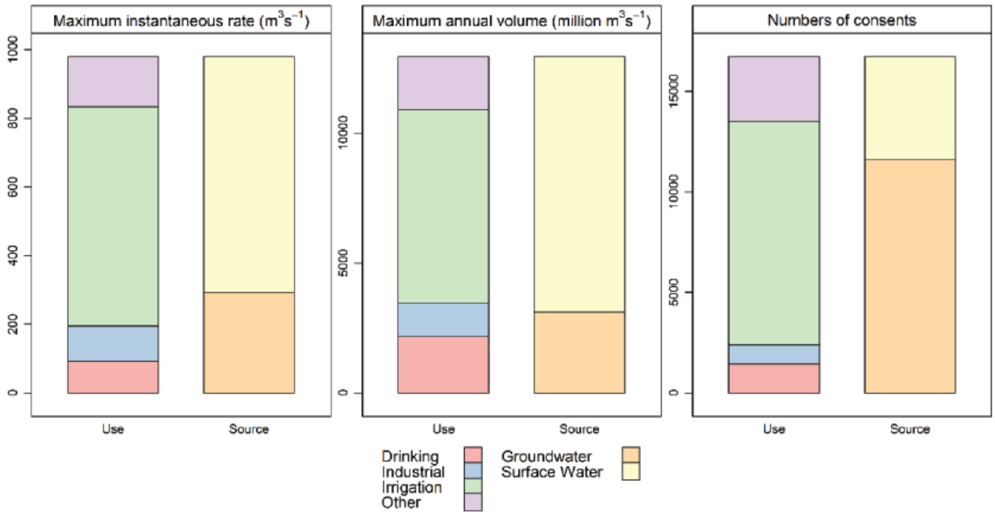

Figure 1. Water take consents (non-hydropower) active in February 2018 across New Zealand. Derived using the definitions and data of Booker and Henderson (2018).

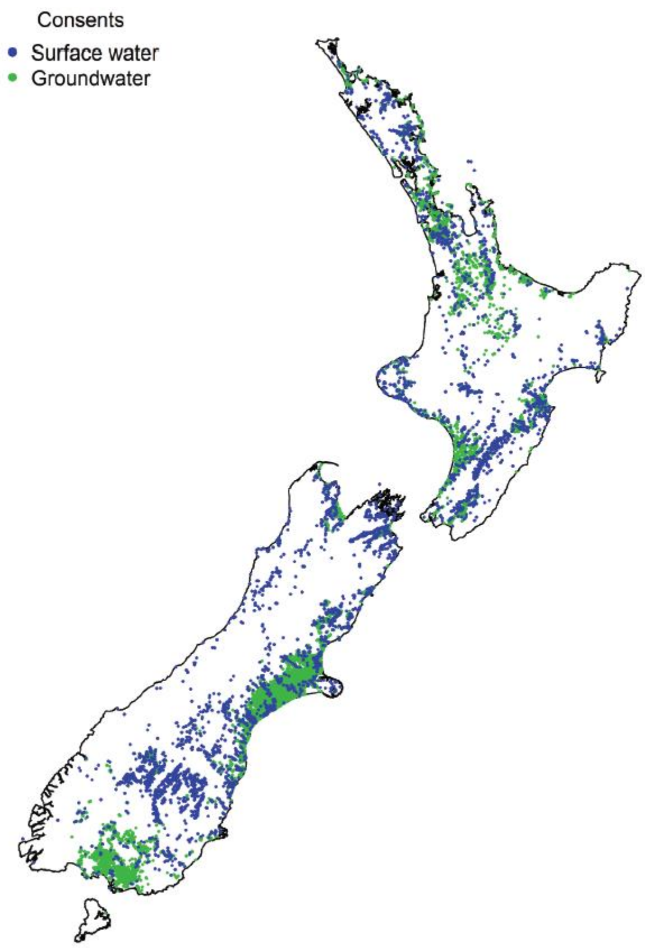

Figure 2. Locations of water take consents (non-hydropower) active in February 2018 across New Zealand. Derived using the data of Booker and Henderson (2018).

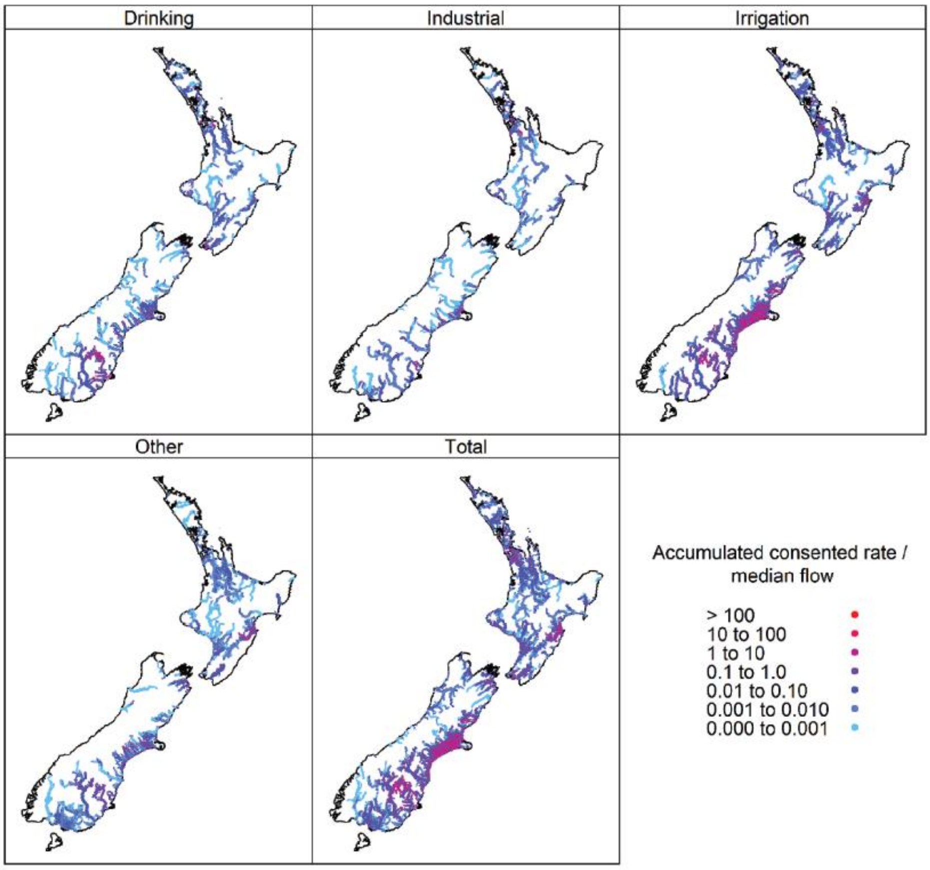

Figure 3. Map of accumulated upstream consented rate of take (non-hydropower) relative to median flow for all rivers across New Zealand. Derived using the data, definitions and methods of Booker and Henderson (2018).

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

Technical difficulties defining and measuring SWFA have limited ability to precisely quantify the pace of change in SWFA over the recent past. Modelling methods could be applied to inform change in SWFA if suitable data were available. For example, a method was recently devised and applied for detecting changes in summer river flow conditions and attributing their causes to changes in climate versus local anthropogenic activities (Booker and Snelder, 2023). The findings from Booker and Snelder (2023) indicate changes to both climate and local activities have combined to alter flow regimes, suggesting that SWFA resulting from local activities should be considered alongside climate change for environmental reporting or when making river flow management decisions.

Estimates of irrigated area (e.g., Dark, 2020) and cumulated consented rates of maximum allowable abstraction (e.g., Figure 4) have been used as proxies to SWFA. Those proxies indicate that SWFA does have large potential to alter surface water flows and therefore influence ecological integrity and/or human health. However, inconsistencies in data provision meant that consent data used for environmental reporting in 2018 could not be compared with consent data used in the previous round of environmental reporting (Booker and Henderson, 2018). Similarly, comparison of three representations of irrigated area at different times may indicate that irrigated area has increased, but these data were not obtained using consistent methods (e.g., Dark, 2020).

Expansion of irrigation and increased water demand may have been driven by a combination of regulatory drivers or economic incentives associated with increased production (Ma et al. 2020). Analysis of irrigated area is complicated by technological advancements such as centre pivot irrigation systems (Duncan et al. 2016) that may have altered water utilisation, legislative changes that have encouraged efficient irrigation practices in the region (Dench and Morgan 2021), and the possibility that local activities may be adapting to changes in climate (Cradock-Henry et al. 2020).

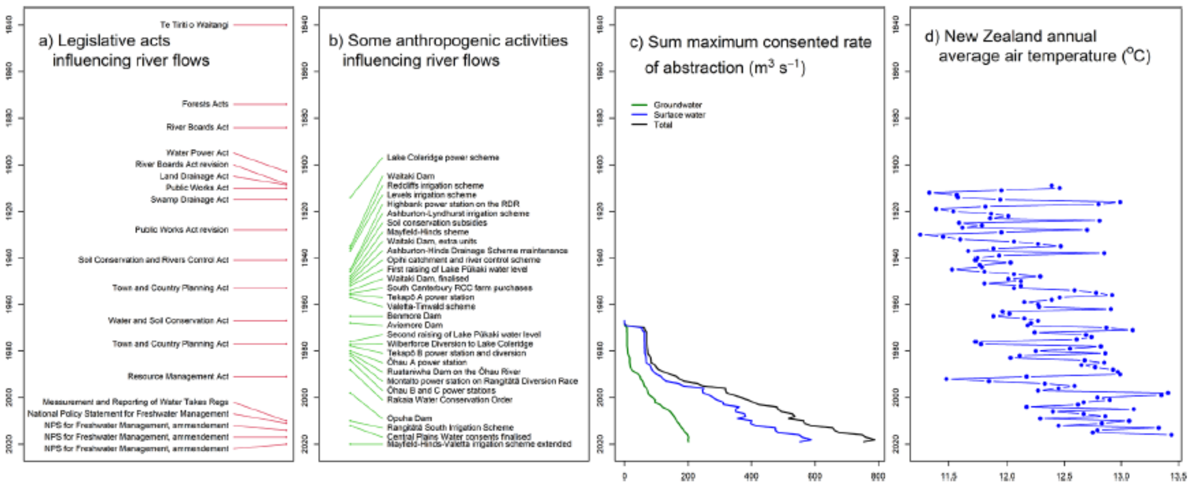

Figure 4. Timelines showing: (a) some regulations relating to water-land management; (b) examples of flow-influencing activities in the Canterbury region; (c) maximum allowable rate of abstraction summed across all consents to take water in the Canterbury region (data source: Environment Canterbury); (d) annual air temperatures.

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

SWFA cannot be measured directly because it must be estimated from monitored abstraction data and river flow data. Under current regulations, water abstractions with larger consented allowable rates must provide recorded abstraction data using verified methods. Regional councils and other organisations are monitoring river flows and applying agreed monitoring standards. However, there is no operational system for national collation or systematic analysis of either water abstraction or river flow data. Furthermore, river flow gauging stations are not strategically positioned to represent SWFA and an operational method for calculating SWFA has not been devised, accepted, or tested. Filling missing data and accounting for climate variability are particular challenges when estimating SWFA.

Monitoring water abstraction. The Resource Management (Measurement and Reporting of Water Takes) Regulations (2010, amended 2016 and 2020) require holders of consents to use verified measurement devices and then report actual water take to their local authority. These regulations were designed to ensure consistent measuring and reporting of water take at national, regional and catchment levels. However, unless required in the relevant regional plan, the regulations do not apply to holders of consents for: abstractions at a rate of less than 5 litres per second; non-consumptive uses (irrespective of the rate of that use); or abstractions of coastal or geothermal water. The regulations also do not apply to abstractions that do not require a resource consent such as: individual households or businesses that abstract water from a reticulated supply; abstractions that are specifically permitted in section 14 of the RMA, including those for an individual's domestic purposes, for animals' drinking water or for firefighting; or abstractions that are permitted by a general rule in a regional plan.

The regulations should ensure that data on consumptive water abstraction are available for estimating SWFA. However, data requests to regional councils on behalf of MfE have not yielded complete, high-quality, well-collated water abstraction databases due to a mixture of physical difficulties in measuring abstraction, difficulties in applying QA-QC procedures for abstraction data, funding considerations, and lack of technical expertise and data infrastructure within some regional councils. Ideally, water abstraction data should be linked to consent information to facilitate data checking and allow calculation of consent utilisation.

Monitoring of water abstraction is a technically challenging task, with potentially high uncertainties when viewed at high frequencies (e.g., 15-minutely, or even daily). Collation of accurate data quantifying the actual amount of water taken for many water consents has been lacking. Data supplied to councils has been irregular and of poor quality in some cases despite the regulations (OAG, 2018; MfE & Stats NZ, 2020). Uncertainties in abstraction data are very difficult to check because they can contain either legitimate or erroneous zeros, one-off high values, or long periods of the same recorded value (Booker et al., 2022).

<u>Monitoring river flows</u>. Many regional councils monitor river flows at gauging stations. NIWA and private companies (e.g., hydroelectric power companies) also monitor river flows at particular gauging stations. Stage-discharge relationships are often applied in the form of a rating curve to convert water level to river flow. However, rating curves contain uncertainties and may become obsolete due to changes in riverbed height, which often occur in gravel-bed rivers during high flow events, or channel roughness, which often occurs in low slope channels with in-stream vegetation growth (McMillan et al., 2012). All flow measurements should conform to national environmental monitoring standards for open channel flow measurement (National Environmental Monitoring Standards 2013) and should be subject to quality assurance-quality control procedures equivalent to National Environmental Monitoring Standards (2019).

<u>Consensus on the most appropriate measurement method</u>. Removing human influences from streamflow time-series is a process often referred to as river flow naturalisation. Estimates of naturalised river flows are essential to express SWFA for various aspects of river flow regimes. However, clear definition of natural river flow or natural flow regimes is needed because methodological definitions can vary. Several methods for flow naturalisation have been developed, including adding observed water abstractions to observed river flows, simulation by physically-based models, and substituting re-scaled observed time-series from a "natural" reference site to a site of interest. Terrier et al. (2021) provides detailed discussion of naturalisation methods and why naturalised flows should not necessarily be considered true natural flows. Methods have been applied for estimating the degree of hydrological alteration at ungauged sites across regions overseas (Sengupta et al., 2018), but not systematically across New Zealand.

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

Water users are required to submit records of water abstraction to local authorities. These data are typically used for consent compliance purposes but systematic QA-QC and collation for estimating SWFA are rare. River flow measurement is a technically challenging task that is more easily applied in some locations (e.g., gorges or larger rivers) than others (e.g., small alluvial channels), leading to a bias in the locations of gauging stations. Longer-term complete river flow records are more valuable than shorter-term records containing missing data. Gauging stations are not deployed using a strategic design for national benefit in relation to monitoring the effects of river flows.

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

There would be a significant cost associated with adequate monitoring of abstraction and flow followed by data post-processing, including QA-QC procedures, and calculation of SWFA.

For monitoring water abstraction, up-front costs depend on whether abstracting from groundwater or surface water, equipment associated with measurement methods, and location. Regulations require abstraction data to be telemetered at 15-minute intervals, which adds to installation and maintenance costs compared to non-telemetered data. The large number of takes distributed across the country (>16,000 consents) implies significant up-front and on-going costs are associated with monitoring water abstraction. The concept of "representative sites" cannot be applied to monitoring water abstraction because the cumulative effects of multiple abstractions is of most interest.

For monitoring river flows, up-front costs depend on equipment associated with measurement methods (e.g., a weir versus open channel measurement), and location (e.g., the need to install a cableway for measurement of high flows). On-going costs are likely to be higher in locations with shifting riverbeds due to the need to intermittently check and update rating curves (see McMillan et al., 2012 for further details). The concept of "representative sites" can be applied to monitoring natural river flow regimes but may not translate to monitoring SWFA due to between-catchment patterns in water abstraction and between-regional patterns in floods and droughts (Caruso et al., 2016). Monitoring flows in wetlands and lakes is even more technically challenging and costly than for rivers.

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

This attribute is of high interest to Māori. There are examples of work/monitoring in this area by iwi/hapū/rūnanga. For example, Cultural Flow Preference Studies (CFPS) support whānau, hapū and iwi to identify their preferences with respect to the flow regimes that they want to see in streams and rivers (Tipa & Nelson, 2012; Tipa & Associates 2018). This method has been applied by iwi/hapū/rūnanga in several catchments to inform their SWFA decision-making (e.g., Tipa & Associates 2012; 2013).

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

Flow has a conceptual and empirically demonstrated link with many other potential attributes including NOF-type attributes representing periphyton (cover or biomass; Snelder et al., 2019), stream macrophytes (volume/clogginess; Riis et al. 2003, Matheson et al. 2012), nutrients (nitrogen concentration or loads; Snelder et al., 2020), invertebrates (Macroinvertebrate Community Index; Greenwood et al., 2016), and sediment (percent cover of fine sediment; Stoffels et al., 2021).

Part B—Current state and allocation options

B1. What is the current state of the attribute?

It is reasonable to state that SWFA is a major environmental management issue for many regions across NZ, given large irrigated areas and high consented allowable rates of abstraction, as well as the prominence of SWFA in regional planning, the NPSFM, and Environment Court proceedings. However, please see comments above regarding challenges of defining and measuring SWFA which mean that it is not possible to precisely quantify SWFA at the broad scale.

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

Monitored reference states of river flows (i.e., no SWFA) exist but are rare and not representative of locations impacted by abstraction. The most useful information about SWFA is likely to come from modelled estimates, but methods would be contestable, and results may be associated with considerable uncertainties.

Given that both abstraction and landcover alteration contribute to SWFA, natural reference states would have to be made in catchments with very limited abstractions or alterations to landcover. Such catchments are present in some NZ catchments (e.g., in Fiordland) but river flows in these locations are not well monitored, and they are not representative of locations where SWFA would be expected to be high (e.g., agricultural catchments with high irrigation demand, hydroelectric dams, or large urban populations).

Spatial patterns in specific hydrological indices have been estimated at ungauged sites across NZ. For example, Booker and Woods (2014) compared and tested various methods for predicting MALF, mean flow, and flow duration curves for ungauged sites. Similar studies were conducted by Booker (2013) for FRE3 (a variable indicating mid-range flow variability, flashiness, and frequency) and by Singh et al. (2019) for baseflow index (a variable indicating low flow relative magnitude), including seasonal components. Predictions from these studies were described as representing “reasonably natural conditions” because the studied databases contained data from sites that were not affected by large engineering projects such as dams, diversions, or substantial abstractions, according to information given by data providers. These predictions were generally shown to be unbiased but uncertain, with the level of accuracy varying with the hydrological variable, and location (Booker and Whitehead, 2018). The predictions can be used for informing river flow management in locations where little observed hydrological information exists, and where relatively low pressure on water resources occurs. At the time of writing, estimates of various hydrological indices across the national river network are available from MfE’s data portal and NIWA’s nzrivermaps tool (Whitehead and Booker, 2019).

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

Take limits have been set in regional plans to constrain SWFA. Take limits often comprise cease-to-take trigger flows and a rate of maximum allowable abstraction. Take limits do not have global bands because they are set in relation to local in-stream values proposed by communities and tangata whenua.

The equivalent of NOF-type bands have been proposed in the international literature using a mixture of narrative and numeric concepts that seek to apply a uniform target for relative SWFA to all flows (e.g., less than 10% alteration of naturalised daily flows equates to "a high level of protection"). This type of band is difficult to implement as a take limit to control multiple abstractions distributed across the landscape because it cannot easily be applied to control management levers. The maximum rate of abstraction for each water user needed for collective abstraction to continuously keep within the band would: a) be continuously changing; b) be influenced by other users; c) be highly uncertain to estimate for the administering authority; and d) not provide certainty about reliability of supply for water users.

For the purposes of this question, let us equate bands for SWFA to environmental flow regimes which have been defined in the NZ context as "a description of the quantity and timing of river flows required to maintain or improve the structure, functions, processes and resilience of aquatic ecosystems and the human values supported by those ecosystems, including values associated with conservation, culture, recreation, and landscape character" (Booker et al., 2022).

Take limits (aka "water resource use limits") in the NPSFM 2020 have been interpreted as sets of rules in regional plans that constrain water use to restrict the degree of SWFA arising from collective operation of flow-altering activities. Take limits guide control of human activities in order to provide environmental flow regimes. Take limits can be thought of as predefined rules that guide authorities to deliver environmental flow regimes by controlling flow-altering activities, and also clarify water availability for out-of-stream use. Take limits typically apply a cease-to-take trigger flow to protect conditions at low flows and a rate of maximum allowable abstraction to limit alteration to other parts of the flow regime. It should be noted that multiple cease-to-take trigger flows and allowable rates could be applied to allow fine control of hydrological alteration to deliver environmental flow regimes and predictable levels of water supply (Booker et al., 2023; Booker and Rajanayaka, in review). The NPSFM 2020 did not apply the concept of SWFA or environmental flow regimes and did not contain bands for take limits. Please see Section 4 of Booker et al. (2022) for detailed discussion of the relationship between take limits and environmental flow regimes.

Globally, environmental flow regimes have been proposed and adapted using different approaches, many of which are based on some combination of: 1) limiting alterations from the natural flow baseline to maintain biodiversity and ecological integrity; and/or 2) designing and purposefully manipulating flow regimes to achieve specific ecological and ecosystem service outcomes. Acreman et al. (2014) argued that the former "limit to hydrological alteration" approach is more applicable to natural and semi-natural rivers where the primary objective and opportunity is ecological conservation. The latter "designer" approach is better suited to modified and managed rivers where returning to natural conditions is no longer feasible and the objective is to maximize natural capital, as well as support economic growth, recreation, or cultural history. In both approaches, environmental flow regimes often intend to mimic naturalised flow patterns and ecological outcomes of the natural flow regime.

The limit to hydrological alteration approach seeks to equate changes to all components of the flow regime from their natural state to desired risk of altering aquatic ecosystems, whereas the designer approach seeks to identify and deliver the parts of the flow regime necessary to uphold desired ecosystem states. Acreman et al. (2014) argued that in a future characterized by altered climates and intensive regulation, where hybrid and novel aquatic ecosystems predominate, the designer approach may be the only feasible option. This conclusion stems from a lack of natural ecosystems from which analogue conditions may be drawn, and the need to support broader socioeconomic benefits and valuable configurations of natural and social capital. However, application of the designer approach requires well known flow-ecology relationships, which is challenging given the complex and dynamic (i.e., unpredictable) nature of flow effects on river ecosystems, or leeway to apply adaptive management principles to alter flow regimes (see Stoffels et al., 2018; Stoffels et al., 2024).

The “presumptive standard” method of Richter et al. (2012) for river flows or Gleeson and Richter (2018) for groundwater abstraction are examples of a limit to hydrological alteration approach. The “building block” method of De Villiers et al. (2008) is an example of a designer approach. A mixture of these two approaches can be applied because flow-ecology relationships used in designer approaches are often derived from hypotheses about, or observations from, near-natural situations. For example, the ecological limits of hydrologic alteration (ELOHA) framework of Poff et al. (2010) mixes the two approaches.

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

Cessation of flow is a known tipping point because aquatic ecosystems cannot function without flow. However, it should be noted that some rivers systems can experience flow intermittency under natural conditions (Datry et al., 2014; Larned et al., 2010). Other thresholds have been investigated but are very difficult to prove due to reasons mentioned throughout Section A.

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

There are known, but variable, lags between abstraction of water from groundwater wells and streamflow depletion in rivers. Inter-annual climate variability, climate cycles, and climate trends are known to be drivers of river flows, but climate patterns also interact with water demand and biological processes. See Booker and Snelder (2023) for further discussion.

Lags between SWFA and ecological integrity and/or human health are a theoretical certainty, but are difficult to quantify, and are likely to be space and value dependent.

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

Cultural Flow Preference Studies (CFPS) involve a series of steps (including interviews, site selection, identification of cultural uses and attributes that describe healthy vibrant flows to support those cultural uses, rating scales/descriptions/themes, on site assessments, identification of flow related issues, comparison of average assessor ratings with average daily flows recorded for the assessment date etc) to produce data across themes (e.g., mahinga kai, health & wellbeing, cultural landscapes, Wai Māori) and inform 'critical flow' thresholds (in cumecs) that describe what flows whānau will/will not be satisfied that their cultural rights and interests are being recognised and provided for (e.g., Tipa & Associates 2012; 2013). The outcomes of these studies have been used by hapū/rūnanga to inform SWFA decision making.

Part C—Management levers and context

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

The relationship between the state of the environment and stresses on that state cannot be easily described with respect to SWFA. Please see river flow management guidelines within Booker et al. (2022).

Booker et al. (2022) proposed a framework that can be used to facilitate an approach to managing river flows to achieve environmental outcomes defined under the NPSFM. The framework, which is currently being applied by regional councils (e.g., Horizons, NRC, GWRC), sets out a transparent approach for linking environmental flow regimes to ecosystem states that represent ecosystem health through controls on flow-altering activities. The framework comprises a cascade of steps which: a) starts with a broad definition of environmental flow regimes; b) includes consideration of climate change; c) incorporates a loop for monitoring and adaptive management; and d) ends with controls on flow-altering activities. The framework (illustration below) can be operated in a manner consistent with the NPSFM by mapping each flow-related NPSFM clause to a step in the framework.

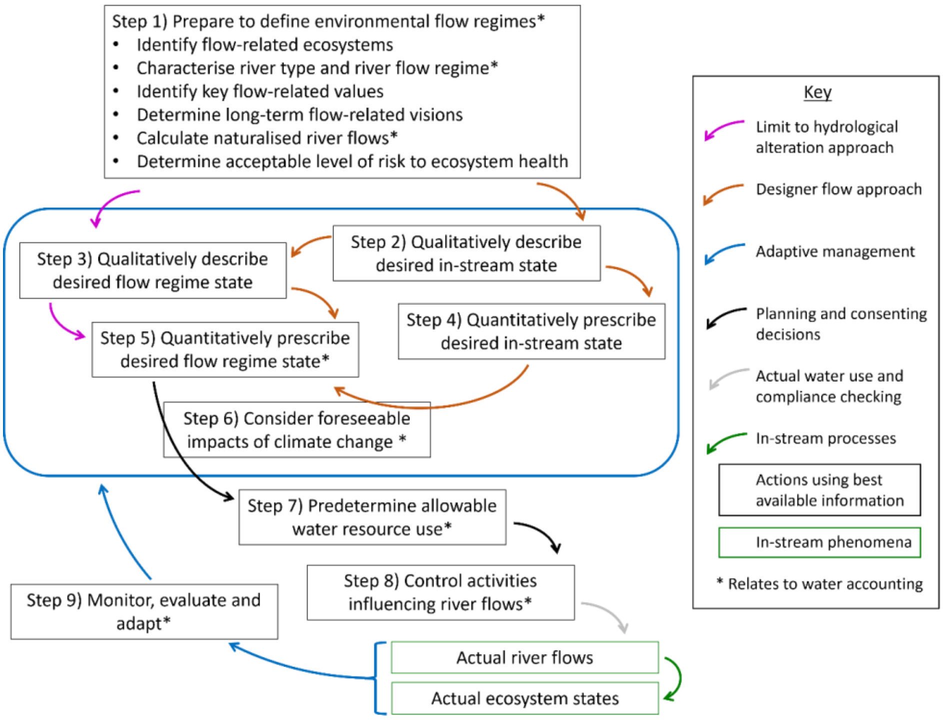

Figure 5. Simplified depiction of a proposed framework for river flow management (after Booker et al., 2022).

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

Take limits in regional plans are the main mechanism used to affect SWFA. Take limits are sets of rules in regional plans that constrain water use to restrict the degree of SWFA arising from collective operation of flow-altering activities. Take limits can be thought of as predefined rules that guide authorities to deliver environmental flow regimes by controlling flow-altering activities. Take limits also clarify water availability for out-of-stream use. See further comments in Section B3 on this topic.

Conditions are written into water resource use consents in order to restrict activities that influence surface water flows. Consent conditions should control collective abstraction in order to meet take limits.

See comments in Section A2 describing barriers to evidence about the impact of particular components of SWFA on particular components of ecological integrity and/or human health due to difficulties in defining and measuring SWFA combined with a lack of monitoring data suitable for detecting flow-driven impacts.

C2-(i). Local government driven

Management of water quantity, including river flows and groundwater levels, is a necessary consideration for local government for all 15 NPSFM policies and all of the values outlined in Appendix 1 of the NPSFM.

C2-(ii). Central government driven

As above, in the context of the NPSFM developed by central government, the process of water allocation can be broadly described as limiting locally-induced hydrological alteration to sustain healthy natural environments by controlling taking of water to meet various economic and human health requirements whilst recognising the foreseeable effects of climate change.

C2-(iii). Iwi/hapū driven

All iwi/hapū planning documents such as Environmental Management Plans and Climate Change Strategies/Plans will contain policies/objectives/methods seeking to influence SWFA outcomes for the benefit of current and future generations. There are many examples of interventions/mechanisms being used by iwi/hapū/rūnanga to directly affect this attribute.

C2-(iv). NGO, community driven

Regional councils are obliged to engage with communities and tangata whenua to identify scope of physical environments for which flows are to be managed. However, the manner, completeness, and transparency of engagement processes have been criticised (Stewart-Harawira, 2020; Taylor, 2022).

C2-(v). Internationally driven

International organisations such as the UN, IUCN, and RAMSAR provide broad level guidance about environmental management in relation to SWFA. See the following links as examples:

- IUCN report on the essentials of environmental flows
- IUCN poster about flow allocation in natural and managed river systems
- RAMSAR report about the importance of flows for wetlands
- UN International Recommendations for water statistics

Part D—Impact analysis

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

Not managing flow regimes in rivers, lakes, and wetlands would present significant risk to environmental values such as healthy ecosystems, basic human health, and all aspects of iwi/hapū/rūnanga health and wellbeing.

SWFA needs to be managed in a way that acknowledges the connectedness between people and water for interdependent social, economic, cultural, and environmental reasons.

Wai (water) is integral to the intricate relationships iwi/hapū/rūnanga have with te taiao, and the spiritual and cultural significance of fresh water can only be determined by mana whenua.

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

Economic impacts of changes in SWFA would be felt across several sectors and regions due to changes in water availability:

- Agriculture and horticulture and where water is used for irrigation, frost protection, stock water drinking, washdown.

- Hydroelectric power generation where water is dammed or diverted.
- Tourism as river flows are essential for natural landscape character and various recreational activities (e.g., kayaking, jet boating).
- Various ancillary industrial purposes (e.g., where water is used for cooling purposes).
- Authorities responsible for municipal water supplies.

See Map in Figures 2 and 3 for broad locations where pressure for water abstraction is greatest. Essentially, nearly all parts of NZ use water for various purposes, but pressure is particularly high for irrigation purposes (see Figure 1) in parts of Canterbury, Otago, Marlborough, Greater Wellington, Hawke's Bay. Economic impact assessments should note that water quantity is intrinsically linked with water quality.

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

Climate change is expected to exacerbate the need for river flow management due to the dual effects of lowering water availability (e.g., river flows) and increasing water demand (e.g., for irrigation).

The direction and magnitude of effect of climate change on river flows in rivers, lakes, and wetlands is likely to be spatially variable due to interactions between topography and predominant weather patterns (Collins 2021). In general, climate change on its own would be expected to reduce water availability due to increases in evaporation associated with increases in temperature. A reduction in summer flows associated with climate changes during the past 30 years has already been shown for some rivers across Canterbury (Booker and Snelder, 2023). However, there are complex and interrelated dynamics within human-induced hydrological systems that are influenced by climate and local activities. The dual influences of climate and local activities on river flow regimes are important because these factors combine to confound analysis of observed patterns and introduce uncertainty when predicting future hydrological conditions (AghaKouchak et al. 2021).

Management responses to mitigate the effects of climate change on SWFA could depend on interactions between several factors:

- Climate change driven reductions on water availability.
- Climate change (and population increase) driven increases in water demand.
- Climate change driven flow changes presenting increased risk to environmental values.
- Climate change driven temperature or water quality changes presenting increased risk to environmental values (e.g., water temperature effects regardless of flow changes).
- The degree to which environmental managers wish to penalise endogenous-local water users to compensate for exogenous-globally driven factors that have increased risk to flow-driven environmental values for the same level of water use due to decreases in water availability.

Management response could include the following possibilities:

- More stringent water take limits are needed to reduce local water use in order to maintain the current level of risk to flow-driven environmental values in the face of climate driven reductions in water availability (i.e., globally derived SWFA is mitigated by local actions).
- Water take limits and water demand are maintained at their current levels in the face climate driven SWFA, resulting in increased risk to flow-driven environmental values (i.e., globally derived SWFA is not mitigated by local actions).
- Water take limits are maintained at their current levels, there are climate-driven increases in water demand, and climate driven decreases in water availability, which combine to increase risk to flow-driven environmental values (i.e., globally derived SWFA is exacerbated by local actions).

For the latter two scenarios, other interventions to increase water availability would then be required to avoid increasing risk to flow-driven environmental values. These could include creation of new hard or soft infrastructure to increase water storage and surface water/aquifer recharge in catchments (e.g., reservoirs, ponds, wetlands, plantings and organic soils).

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8.6 Catchment permeability

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Preamble: To be useful, the "catchment permeability" attribute needs to be defined in relation to what stress is being managed and what outcome or objective will be impacted. The following examples illustrate the current issue for three different contaminant stressors:

- Heavy metals concentration is linked to (among other things) the level of imperviousness in urban catchments, but less so in rural catchments [1-4].
- Diffuse nitrogen pollution is linked to (among other things) the interaction of land use with surface water/groundwater interaction and pathways, the presence/absence of an aquifer, aquifer movement and characterisation and/or presence of a denitrification zone in the aquifer [5-9].
- Sediment source and transport is linked to (among other things) the soil-type, land cover characteristic, position in the catchment and rainfall intensity [10-12].

The lack of a more precise definition translates to different measurements or estimations of catchment permeability ranging from GIS analysis (e.g., level of imperviousness in relation of discharge of metal contaminants to waterways [1]) to inverse hydrological and transport modelling (e.g., estimation of contaminant residence time within groundwater system or watershed soil hydraulic conductivity [7]).

In the absence of a formal definition the author considers catchment permeability to be "how quickly or readily water percolates through the soil profile/subsurface environment".

State of knowledge of the "Catchment permeability" attribute: Poor / inconclusive – based on a suggestion or speculation; no or limited evidence

The status of knowledge answer is associated with the multiple definitions attached to the attributes in regards of the multiple stressors that we aim to manage for.

<u>Part A—Attribute and method</u>

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

Depending on the contaminant or hydrological stressor considered, there is a body of evidence that catchment permeability is associated with ecological integrity and human health. For example, the level of imperviousness in urban catchments is often linked to trace metal concentrations in urban streams and their impacts on ecological integrity [3].

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

Previous studies have identified “catchment permeability” as one of the delivering pathways of impact of land management activities on receiving water bodies [2,5,7]. As such, catchment permeability is an indicator of various aspects of water delivery to receiving environments. High permeability is the inverse of high impervious (paved) surface, the latter of which can increase the “flashiness” of water delivery to receiving environments and transport urban contaminants (metals, hydrocarbons, etc) that are mobilised in runoff from paved surfaces into waterways. Increased permeability is anticipated to reduce surface runoff and allow water to percolate below-ground and to be subsequently transported by subsurface flows to receiving waters.

A clear and concise interpretation of this attribute is likely to be important if it is to be used as a metric of stressors and ecology integrity. Moreover, the coarser the spatial resolution under consideration, the more complex the conceptualisation of the catchment permeability attribute is likely to become.

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

Catchment permeability is likely to have decreased over time with increased urbanisation and associated creation of impervious surfaces (e.g., roads, paving, buildings). This trend is expected to continue under the status quo. Use of mitigation systems can help to slow or reverse that trajectory. For example, mitigation systems can be put in place to collect impervious area runoff and to control metal concentrations in stormwater (e.g., https://niwa.co.nz/freshwater/stormwater-management/characterising-stormwater-quality). However, the development of mitigation systems such as porous pavement and porous road asphalt to reduce flood risk are also likely to impact pathways of delivery of contaminants to waterways [13]. For example, water and contaminants will percolate into soil and groundwater and be transported to receiving waters via subsurface flow pathways (i.e., interflow, groundwater) instead of via surface runoff. Some contaminants might be better removed from water if they are transported by these flow pathways (e.g., nitrate removed by denitrification or vegetation uptake).

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

A catchment permeability attribute is not currently monitored or reported nationally or internationally. However, councils or other organisations may hold information on impervious surface layers in GIS databases. There has been research attempting to link aspects of catchment permeability to receiving water contamination by specific stressors (e.g., impervious surface area to trace metals [2-4]). Attempts have been made through numerical modelling to estimate catchment permeability [7, 9], but results generally depend on the numerical model used. Most of the models are also small scale. It is likely that a national model would need to be developed to understand changes in catchment permeability, with significant initial work required to understand how different land uses and land management impact upon catchment permeability.

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

Impervious surface area as an aspect of catchment permeability is likely to be best monitored by identifying paved areas using aerial imagery (at the scales of cities, catchments, or regions). However, catchment permeability (the named attribute, which is more than impervious surface area) is likely to be more difficult and nuanced to measure. Access to private land may be required to measure catchment permeability.

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

The answer to this question depends on the definition of "catchment permeability". As an aspect of catchment permeability, some impervious surface area data may already be held in GIS databases by councils and other organisations.

Measuring or monitoring a more nuanced attribute of catchment permeability could be significantly more difficult, especially if an assemble of stressors associated with this attribute need to be characterised.

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

The author is not aware of any catchment permeability monitoring being undertaken by iwi/hapū/rūnanga. However, we are aware of the Urban CHI which could potentially be adapted to include imperviousness as an attribute (Gail Tipa, pers. comm, 20 June 2024).

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

A correlation or relationship is likely to exist between the catchment permeability attribute and other freshwater attributes when looking at specific stressors, as catchment permeability (which includes impervious surface area) is expected to affect the delivery pathway of other attributes to receiving water bodies (e.g., trace metals, nitrate) and the quantity of water in surface waterbodies (surface water flow alteration) and subsurface aquifers (groundwater depletion). However, few studies have characterised integral/holistic relationships between catchment permeability and other attributes.

Part B—Current state and allocation options

B1. What is the current state of the attribute?

Compared to many other countries worldwide, New Zealand, overall, has a low percentage of paved/impervious land area. Although land cover has been substantially modified since pre-human times, the human population density and amount of paved area in New Zealand is very low relative to other countries. High catchment permeability (which includes low paved/impervious surface) is generally considered to be associated with a healthy environmental state.

Our current knowledge of the attribute state in New Zealand is generally poor at regional and catchment scales. It is possible to quantify the attribute accurately at point scale, but upscaling that degree of accuracy from point scale to catchment scale is difficult. It is possible to quantify and monitor the level of imperviousness in a watershed-based on the presence of road network but the level of watershed surface imperviousness will also depend on factors including the type of road and buildings, the state of the vegetation (e.g., flushed, dry, or burned) especially in rural areas, the time of the year (summer/winter), soil compaction, and local scale topography.

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

The natural reference state of catchment permeability is expected to be higher than present given the existence of our current road networks and urban areas which affect imperviousness. However, the natural states of other factors that affect catchment permeability (as listed in B1) are not well known.

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

The author is not aware of any proposed bands or guidelines for a catchment permeability or impervious surface area attribute in use in New Zealand. However, there may be some guidelines for impervious surface area in relation to stormwater contaminants in the overseas literature.

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

The author is not aware of any specific thresholds relating catchment permeability to ecological integrity. The author considers that specific thresholds may depend on the individual contaminant or hydrological stressors that are affected by catchment permeability (e.g., nitrate, trace metals, surface water flow alteration, groundwater depletion) being considered. These stressors, in turn, affect ecological integrity in ground and surface waters.

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

Depending on the stressor considered, lag times and legacy effects may be important. Subsurface and groundwater pathways associated with nutrient management will be affected by catchment permeability. Currently hydrological/water quality modelling is used in association with isotope hydrology to assess lag times in the delivery of nitrogen (usually as nitrate) to rivers. However, this may be a stressor specific response, and a holistic approach considering multiple stressors and using consistent approaches to determine lag time for multiple stressors (e.g., nitrate, trace metals, surface water flow alteration, groundwater depletion) is likely to be required.

In general, the amount of impervious surface area in New Zealand is expected to continue to rise as it is highly unlikely for road networks or urban areas to ever be retired once they are constructed. However, use of water sensitive design elements in urban areas (e.g., porous pavements, raingardens, stockholm tree pits, wetlands) and in rural areas (e.g., wetlands, riparian buffers) may help to improve the permeability of modified land surfaces.

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

In addition to discussing this attribute directly with iwi/hapū/rūnanga, in regards to catchment permeability, there may be tikanga and mātauranga Māori relevant to informing bands, allocation options, minimally disturbed conditions and/or unacceptable degradation in documents like iwi environmental management and/or climate change plans etc. For example, the Iwi Management Plan for Ngāti Whātua Ōrākei outlines tikanga (engagement protocols) and their priorities on any matters which effect the lands, air and water within the rohe which includes “any proposal which creates an impervious area greater than 5000 m^2” [14]. The Ngāti Whātua Ōrākei Kaitiakitanga Framework also specifically references objectives for water sensitive urban design [15].

Many iwi economic entities themselves have substantial urban property portfolios that include commercial and residential investments [e.g. 16], where three iwi, Ngāi Tahu, Ngāti Whātua o Ōrākei, and Waikato-Tainui, currently own the greatest share [17]. Iwi urban property investment is predicted to increase as more groups settle their claims [18, 19].

Part C—Management levers and context

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

Catchment permeability, or the ease at which water can percolate through the soil profile or subsurface environment, is expected to decrease in response to an increase in urban and rural development activities including construction of roading, paved surfaces, buildings, soil surface sealing and compaction.

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

C2-(i). Local government driven

In urban areas, porous pavements, stockholm tree pits, raingardens and wetlands are being used by some councils to retain water and lessen the downstream impact of stormwater flows and to reduce contaminant loads to receiving waters. In rural areas, many councils support the establishment of riparian buffers and the protection and/or creation of wetlands. Establishment of these urban and rural interventions are expected to increase catchment permeability.

C2-(ii). Central government driven

Central government requires the protection of natural wetlands >500 m^2 in size which is expected to enhance catchment permeability.

C2-(iii). Iwi/hapū driven

As mentioned in B6, iwi/hapū planning documents such as Environmental Management Plans and Climate Change Strategies/Plans may contain policies/objectives/methods seeking to influence catchment permeability outcomes for the benefit of current and future generations. The author is unaware of any other iwi/hapū driven interventions/mechanisms being used to affect this attribute except for projects that create riparian buffers, protect and create wetlands (e.g., funded by the Waikato River Authority) and encourage water sensitive urban design (including by iwi entities themselves)

C2-(iv). NGO, community driven

The establishment of riparian buffers and protection and creation of wetlands by NGO's (e.g., New Zealand Landcare Trust) and catchment groups is expected to increase catchment permeability.

C2-(v). Internationally driven

We are unaware of internationally driven interventions/mechanisms being used to affect this attribute. However, there is the IPBES and the Nature Futures Framework which may have parts that are relevant.

Part D—Impact analysis

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

Without any checks or controls, urban expansion and uncontrolled growth will result in the widespread paving of our natural areas, which would have negative impacts on numerous aspects of terrestrial, freshwater, and coastal marine ecosystems. In the rural sector, high intensity farming with livestock can lead to soil compaction decreasing catchment permeability. The negative consequences of decreased catchment permeability include: increase in surface runoff leading to increased erosion and risk of downstream flooding, decrease in recharge of groundwater and river baseflows and decrease in removal of contaminants from water associated with biological assimilation and transformation processes.

In turn, the consequences of decreased catchment permeability can have detrimental effects on ecology integrity and human health including i) insufficient river flows to support aquatic life, ii) contaminant levels in groundwater and surface water that are harmful to aquatic life and iii) depletion of groundwater and surface that can concentrate contaminants that are pose human health risks (e.g., nitrate, trace metals).

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

The economic impacts of decreased catchment permeability are likely to be felt by those communities that are affected by increased erosion and heightened risk of downstream flooding that threatens homes, livelihoods (e.g., agricultural enterprises) and infrastructure, by decreased groundwater and surface water supplies for drinking water and irrigation, and costs associated with the treatment of contaminated ground and surface waters for drinking water, irrigation and other purposes. Both urban and rural communities throughout New Zealand are expected to be economically impacted by the effects of reduced catchment permeability, but the impacts are likely to be most strongly felt in lowland areas of large catchments, with a high degree of urbanisation and/or intensive agriculture.

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

Rainfall and storm frequency/intensity is expected to increase with climate change. This could interact with catchment permeability to affect runoff volumes and water quality. Fire and drought frequency could also increase which could lead to an increase in the sealing of soil surfaces. Climate change could affect catchment permeability in a number of ways, but we do not currently have the tools to make confident predictions about the significance or direction of the changes.

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8.7 Groundwater nitrates

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State of knowledge of the "Groundwater Nitrates" attribute: Good / established but incomplete – general agreement, but limited data/studies

Part A—Attribute and method

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

Elevated concentrations of nitrate are known to impact subterranean aquatic organisms (composed mainly of small crustaceans, oligochaetes, nematodes, acari, and molluscs, less than 1 mm to several centimetres in body size) and microbial communities that inhabit underground aquifers and springs. Collectively known as stygofauna, these organisms are adapted to complete darkness and restricted space and exhibit an unexpectedly high biodiversity, which is recognised but poorly documented [1, 2], The invertebrates and fish that inhabit spring-fed streams (including their hyporheic zones), lakes, wetlands and caves (karst ecosystems) are also known to be sensitive to elevated nitrate [3-7]. Effects include chronic and acute toxicity and impacts on microbial biogeochemistry and overall ecological functioning. Additionally, elevated nitrate in groundwater discharges to surface water impact on aquatic organisms and their interrelationships in receiving waters [8], and contribute to eutrophication and associated problems such as proliferation of algae and toxic cyanobacteria.

Elevated concentrations of nitrate in drinking water sources from groundwaters via wells, bores, springs and spring-fed streams are known to affect human health. Impacts range from methemoglobinemia or blue baby syndrome in infants to adverse reproductive and birth outcomes, thyroid disease and a range of cancers, particularly colorectal [9-14].

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

(a) Evidence is strong for effects on aquatic organisms, based on New Zealand and international ecotoxicity studies [2-7]. Evidence for ecosystem-level effects on spring-water fed freshwaters is moderate, while effects on subterranean groundwater communities is poor, primarily due to a lack of (and difficulties accessing) ecosystem-level monitoring and assessment below-ground, and lack of baseline ecological records. It has often proved difficult to detect ecosystem level responses to elevated nitrate in spring fed streams where multiple compounding factors interact [15, 16].

(b) Evidence is moderate to strong for impacts of groundwater nitrate on human health, with assessments carried out in multiple countries including New Zealand, and systematic reviews and metanalyses undertaken [9-14, 17]. Effects on new-born babies fed milk formula prepared from nitrate-rich water are widely accepted, but there is still some contention regarding the wider risks from nitrate in drinking water [17-19], given significant additional exposure from food. Neither the WHO nor USEPA (or the NZ government) has yet revised drinking water guidelines to account for such risks, but evidence is mounting.

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

In 2020 StatsNZ [20] reported nitrate levels in 49% of monitored sites (128 of 262) sites were likely or very likely improving, and 35% (92 out of 262) were likely or very likely worsening. Continued upward trends in nitrate concentration have continued or accelerated in many aquifers across the country, particularly those associated with intensifying agriculture including vegetable growing and horticulture. Natural baseline for groundwater nitrate concentrations and longer-term trends have been assessed in the National Groundwater Monitoring Programme using isotopic tracing techniques [21, 22]. More recently assessments have been made using both the National Groundwater Monitoring Programme and council monitoring results compiled by LAWA using hierarchical cluster analysis, relationships to groundwater age, and regression against a measure of land-use impact [23]. There can be long lags in response of groundwaters to contamination due to the size and complexity of underground aquifers and associated travel times between source and monitoring site. These issues can be better understood and accounted for by aging groundwaters and modelling aquifer recharge and flow rates.

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

Groundwater quality (including nitrate) is monitored at the regional level as part of council State of the Environment (SOE) monitoring programmes[1]. At a national level, regional authorities collaborate with GNS Science as part of the National Groundwater Monitoring Programme[2]. This monitoring data is made publicly available on the LAWA[3], MfE[4] and GNS[5] websites and summary data presented by StatsNZ [20]. This long-term research and monitoring programme involves sample collections across New Zealand by council staff following standard protocols, and analysis by the GNS Science laboratory. LAWA presents data from the NGMP as well as from wider council monitoring programmes. Overall groundwater sampling and analytical methods are generally comparable between the NGMP dataset and the LAWA dataset. Regional council staff collect samples following

[1] https://www.lawa.org.nz/learn/factsheets/groundwater/monitoring-groundwater-in-new-zealand#:~:text=Groundwater%20quality%20is%20monitored%20by%20regularly%20collecting%20groundwater,samples%20are%20sent%20to%20accredited%20laboratories%20for%20analysis.
[2] https://www.gns.cri.nz/data-and-resources/national-groundwater-monitoring-programme/
[3] https://www.lawa.org.nz/explore-data/groundwater-quality
[4] https://data.mfe.govt.nz/table/104571-groundwater-quality-trend-1999-2018/
[5] https://www.gns.cri.nz/data-and-resources/gns-geothermal-and-groundwater-ggw-database/

the National Environmental Monitoring Standards (NEMS) for groundwater quality data[1], and samples are sent to accredited laboratories for analysis. Each well may be sampled annually, quarterly or monthly depending on the purpose of the monitoring. Monitoring tends to be focussed on areas where groundwater is used as a source of drinking water and there are concerns regarding its safety, primarily for human consumption.

A recent assessment of the NZ groundwater monitoring network [24] concluded that it was unfit for the purpose of detecting nitrate reductions within practical timeframes (5-10 years) or budgets. The need for information on mean residence time data was emphasized to provide additional information on lag and temporal dispersion effects.

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

Overall, there are minimal practical limitations to monitoring, except significant current reliance on existing wells and bores and the set depths that they source water from (i.e., not ideal in terms of monitoring for change). Modifying the monitoring network, as recommended to increase its ability to detect changes resulting from management actions [24], would mean starting again at a significant number of new sites where there will not be a baseline of historical information. Therefore, I would conclude that measuring this attribute is practical and feasible, but will require significant investment to redevelop a fit-for-purpose monitoring programme.

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

A monitoring programme following standard procedures is already in place collecting samples from selected wells and bores. The spatial extent and frequency of measurements could be usefully increased. There is also potential for high frequency monitoring using sondes (Electrical conductivity, dissolved oxygen, redox, pH, temperature) and optical nitrate probes. These provide near continuous data enabling high temporal resolution to discern short-term, seasonal and long-term variability and trends to be discerned, However, they are relatively expensive to purchase and run and still require calibration, maintenance and careful quality assurance and control to achieve quality results.

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

We understand that Lincoln Agritech has worked with iwi/hapū/land trusts (e.g., Ngāti Tahu-Ngāti Whaoa in the Waiotapu Stream catchment) on groundwater nitrates as part of the Critical Pathways research programme funded by MBIE. It appears that the Section 33 transfer of certain water quality monitoring functions from WRC to Tūwharetoa Māori Trust Board in 2020 included: "Groundwater: Biannual groundwater quality monitoring at two schools (Kuratau and Waitahanui); and Groundwater: Six weekly groundwater level measurements at 62 sites in the Taupō catchment" [45].

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

[1] https://bucketeer-54c224c2-e505-4a32-a387-75720cbeb257.s3.amazonaws.com/public/Documents/NEMS-Water-Quality-Part-1-Sampling-Measuring-Processing-and-Archiving-of-Discrete-Groundwater-Quality-Data-v1.0.0.pdf

The other key contaminants commonly impacting groundwater (in addition to nitrate) are faecal microbes (as commonly measured using the indicator bacteria *E. coli*,), ammonium-N, and phosphorus. Other persistent chemical pollutants such as pesticides, PFAS, PAHs, and estrogens may cause localised contamination of groundwaters. Pesticides are subject to targeted lower-level monitoring as part of the four-yearly national survey of pesticides in groundwater (since 1990), which in 2018 included analysis for emerging organic contaminants [26]. All are associated with agricultural or urban/industrial sources, but the relationships to nitrate are generally variable and indirect.

Recent studies in New Zealand [23] have found that land-use intensity provides a good predictor of nitrate concentrations in oxic (but not anoxic) groundwaters. Nitrate concentrations are reduced where groundwaters are anoxic [27], which promotes denitrification (conversion to gaseous forms of N emitted to the atmosphere). This would suggest that geochemical indicators of groundwater anoxia (e.g., redox state) would also be needed if using land-use as a predictor [28].

Part B—Current state and allocation options

B1. What is the current state of the attribute?

We have a reasonably good understanding of the current state of groundwater nitrate across the motu from current monitoring networks, and this is reported regularly on LAWA. However, there is a bias towards collecting data in areas where groundwater is used for drinking water and there are concerns about safety (see A4i). Also, lag times for nitrate to be detected in wells and receiving waters i can be considerable so the ability of the current network to detect changes is relatively poor [24]. Additionally monitoring coverage tends to be clustered in alluvial aquifer areas (especially Canterbury) where nitrate contamination of ground water is known to be significant, and less common elsewhere [23]. Groundwater nitrate concentrations are trending steadily upwards in many New Zealand agricultural regions where monitoring is in place. Understanding is sufficient to use nitrate concentration as an indicator of human health and ecosystem integrity risk.

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

Natural reference states and rates of change have recently been defined across NZ [23, 29], The three different approaches compared all provided similar estimates [23].

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

No specific numeric bands have been identified specifically for groundwater nitrate. Ecosystem toxicity for groundwaters have generally been assessed using standards proposed for surface waters. For example, Moreau and Daughney[29] made comparison using trigger values from the ANZECC 2000 guidelines[1] (since superseded by the Australian and New Zealand Guidelines for Fresh and Marine Water Quality, 2018)[2]. Alternatively, the nitrate toxicity values in the National Objectives Framework (NOF) could also be applied, providing three attribute states (A-C) and a national bottom

[1] https://www.waterquality.gov.au/anz-guidelines/guideline-values/default
[2] https://www.waterquality.gov.au/anz-guidelines

line (D) based on annual medians and 95th percentile values. The numeric nitrate guideline values for the NOF framework are based on the statistically-derived 'no observed effect concentration' (NOEC) and 'threshold effect concentration' (TEC) values for 22 surface-water species [7]. The applicability of these thresholds for groundwater fauna would need to be evaluated further, as it is unclear whether groundwater species are more [30] or less [31, 32] or similarly sensitive to nitrate-N. The relative stability of natural groundwater environments is characterised by low ecological trait variability, and low biomass and reproductive rate in stygofauna, This suggests that recovery potential following disturbance is likely to be poor [33] making groundwater fauna particularly vulnerable to flushes of high nitrate and other toxicants.

For drinking water the maximum allowable value (MAV) has been set at the WHO and NZ drinking water guideline standard of 11.3 mg Nitrate-N/L. Groundwater nitrate-N levels at or above this are generally noted as high risk, and levels below this medium or low risk (e.g., ECAN 2022 [34]), but no specific numerical bounds for the medium and low risk categories have been identified.

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

Thresholds for toxicity of nitrate to different aquatic organisms have been identified for New Zealand [7], but are generally gradual rather than sharp tipping points. In New Zealand, the 20th and 80th percentiles are often used as thresholds to define default guideline values for surface water quality, and similar thresholds have also been proposed as an appropriate national-scale default threshold for groundwater quality.

Thresholds for nitrate in drinking water that pose unacceptable risk from methemoglobinemia in infants are reflected in the drinking water guidelines. Thresholds for other potential health risks are currently uncertain, although some meta-analysis studies have derived statistical thresholds.

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

There are variable lag times and legacy effects for groundwater nitrate that may affect aquatic ecosystems and human health. These relate primarily to the volume of aquifers relative to groundwater recharge rates and the travel times for groundwater flows to reach downslope sites where they are sourced (e.g., via groundwater bores) or mix with surface waters [23, 24]. Long-term climate cycles and trends may also affect the state and trends in groundwater aquifers with relatively short residence times.

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

A high standard of groundwater quality is an outcome sought by all iwi/hapū/rūnanga. In addition to discussing this attribute directly with iwi/hapū/rūnanga, in regards to groundwater nitrates, there is highly likely to be tikanga and mātauranga Māori relevant to informing bands, allocation options, minimally disturbed conditions and/or unacceptable degradation in treaty settlements, cultural impact assessments, environment court submissions, iwi environmental management and climate change plans, etc.

Part C—Management levers and context

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

Relationships of groundwater nitrate concentrations and their temporal trends to agricultural and urban land use pressures are well established for oxic (but not anoxic) groundwaters [20, 23, 35]. Nitrate concentrations tend to be reduced where groundwaters are anoxic [27], promoting denitrification (conversion to gaseous forms of N emitted to the atmosphere). This would suggest that geochemical indicators of groundwater anoxia (e.g., redox state) are also needed if using land-use as a predictor [28].

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

It has been estimated that implementation of farm mitigation actions between 1995 and 2015 driven by central and local government, and industry policies reduced the nitrogen losses (predominantly nitrate) per hectare from dairy land use that would have occurred in the absence of mitigation actions by 25-30% [36]. However, these reductions were only able to marginally off-set overall pastoral farming (Dairy and drystock) increases of N loss by ~25%, which mainly resulting from intensification and expansion of dairy land use over the same period [36]. This increased N loss occurred largely in irrigated, free-draining areas of the country which contribute disproportionally more to elevated groundwater nitrate loads. This suggests that farm mitigation actions driven by policy and industry actions are only partly ameliorating the problem.

C2-(i). Local government driven

Resource consenting processes, policies and rules, often selectively applied in recognised problem areas (e.g., sensitive lake and coastal catchments) or areas of high or increasing population density, are being used to promote greater treatment of domestic wastewaters (e.g., advanced on-site treatment systems) before land application and reticulation to communal treatment systems. Similarly, improved construction of farm effluent storage and treatment ponds (e.g., properly lined to reduce seepage to groundwater), and effluent land application practices (e.g., deferred and low-rate irrigation, reduced N application rates) are gradually being implemented that reduce the risk of nitrate losses to groundwater. Managed aquifer recharge is also being trialled to dilute groundwater contaminants and maintain stream flows. Overall, there is little specific evidence that these actions specifically are having significant effects on groundwater nitrate, although improving trends are common in some areas [20] as indicated by 49% of monitored sites with improving trends (see A3).

C2-(ii). Central government driven

Improvements to wastewater treatment infrastructure have been a concern of central government for a number of years. Labour's "Three waters"/Affordable water reforms, have been replaced with National's "Local Water done well" to address long-term under-investment and inaction on improved water and wastewater infrastructure. Fixing leaking sewerage networks, reducing wet-weather overflows, upgrading onsite and small community and town wastewater treatment systems, and improved design and management of land treatment systems will likely reduce localised hot-spots of elevated groundwater (and surface water) nitrate.

As part of the NPS-FM 2020, NZ $140 million per year was provided by government to aid implementation, 60% of which was focused on temporary Jobs for Nature projects under the COVID-19 Relief and Recovery Programme [37]. In addition, the One Billion Trees programme (NZ$24 million per year) supports native and exotic plantings that take social, environmental, cultural and economic priorities into account to help meet climate change objectives (Te Uru Rākau 2018; Climate Change Response (Zero Carbon) Amendment Act 2019, Public Act 2019 number 61). These government schemes provided support to catchment groups and supported a wide range of environmental projects with potential water quality co-benefits that could include mitigation of groundwater nitrate.

The NPS-FM 2020 instigated by the previous government also attempted to put controls on winter grazing, stock exclusion from watercourses, maximum N application rates, and requirements for farm management plans to address excess nutrient loads to ground and surface waters. This strengthening of freshwater management policy also influenced resource consenting decisions and associated requirements. With repeal of the Natural and Built Environments Bill, and the Spatial Planning Bill and introduction of the Fast-track Approvals Bill, there is uncertainty about central government future intentions.

C2-(iii). Iwi/hapū driven

Iwi/hapū trust farms are grappling with the challenge of protecting te taiao (the natural world) and safeguarding mahinga kai (cultural food resources and practices) as part of their kaitiaki (environmental guardianship) responsibilities, whilst generating income to support and advance the wellbeing of their iwi/hapū/whanau. Different iwi/hapū are at different stages along the path of reconciling and addressing these challenges.

Iwi/hapū consultation requirements under the RMA, and their involvement in co-management and co-governance including recognition of ownership of various river and lake beds, and incorporation of treaty obligations, guiding documents such as the Vision and Strategy for the Waikato River, establishment of legal personhood (e.g., Whanganui River), and concepts such as Te Mana o te Wai, have undoubtedly shifted the bar in terms of freshwater management expectations.

Overall, this has increased the emphasis on environmental protection above economic exploitation, and provided a more holistic approach to freshwater management than was possible under the largely site-by-site consenting approach taken under the RMA. Due to the normal long-term chronic nature of groundwater nitrate pollution, methods such as rāhui are unlikely to be effective. Overall, there is little tangible evidence that these actions specifically are having significant effects on groundwater nitrate, although improving trends are common in some areas [20] as indicated by improving trends at 49% of monitoring sites (see A3).

C2-(iv). NGO, community driven

Farmer and/or community catchment and river care groups undertake local actions to address environmental issues, including those affecting groundwater quality. These include riparian planting which can remove nitrate from shallow groundwater flows that pass through the vegetation root zone by plant uptake and by provision of organic soils which promote nitrate removal by microbial denitrification. It is also possible for protected and restored natural wetlands and constructed wetlands to remove nitrate from shallow groundwater that seeps into them.

C2-(v). Internationally driven

Many biodiversity protection and enhancement actions (as might be implemented as part of international conventions) also have potential co-benefits for surface and groundwater nitrate concentrations. The impacts are likely to be variable depending on hydrogeological characteristics, but overall considered to be relatively minor unless it involves large-scale retirement of agricultural land-use and conversion to native bush.

Part D—Impact analysis

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

Changes in the attribute state affect ecological integrity and human health as described in A1 above.

Not managing groundwater nitrate is likely to result in:

- Reduced ecological integrity of groundwater and spring-fed ecosystems, with potential repercussions for downstream freshwater, estuarine and coastal ecosystems.
- Reduced quality of groundwater-sourced water supplies with increased health risks and associated costs to ameliorate them; e.g., increased treatment requirements or provision of alternative water supplies/bottled water. There is potential for increased infant deaths and wide variety of human health effects and associated costs if not managed.

Once aquifers become contaminated, remediation becomes logistically challenging using current practical options, and will often require long timeframes for sustainable improvement due to enduring legacies of N loading within catchments [38].

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

Impacts would primarily be felt in vegetable growing areas (Pukekohe, Gisborne, Hawkes Bay, Horowhenua) with high fertiliser use, and in areas with intensive irrigated dairying (e.g., Canterbury). There is also evidence of nitrate-contaminated groundwater aquifers in the Waikato River catchment [46]. Groundwater nitrate contamination could also affect markets for products and prices obtained, where negative environmental and human health effects associated with production are incompatible with consumer and market preferences.

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

Climate change effects on groundwater hydrology are complex [39, 40], and corresponding effects on contaminant levels in groundwaters still uncertain [41, 42]. Climate change will likely also lead to land use and management changes that will affect groundwater utilization and recharge and associated hydrological water balances and contaminant flows. Areas that become drier or experience more frequent droughts are likely to increasingly rely on irrigation to safeguard productivity and reduce uncertainty. Depending on the source of irrigation water (groundwater or river flows) and efficiency of its use, this is likely to affect the reliability of irrigation supply [43], the recharge of groundwater, rate of groundwater turnover and the accumulation of nitrate in the aquifer [44]. In areas where rainfall and/or its variability is likely to increase there is potential for greater leaching of nitrate to groundwater, but also greater denitrification losses back to the atmosphere (due to more saturated soil conditions), and greater dilution and flushing with low nitrate rainfall.

Generally, there is low confidence regarding the outcomes of climate change on groundwater nitrate. Further modelling studies are required for different NZ regions to better understand the likely effect of climate change on groundwater availability, transit times and nitrate concentrations and loads [40, 41].

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9 Estuaries and Coastal Waters Domain

Eighteen attribute information stocktakes for the Estuarine and Coastal Domain are provided in sections 9.1 to 9.18, below. Dr Hannah Jones (MfE, Domain Expert), Dr Drew Lohrer (NIWA, Domain Leader), and the Māori environmental researcher panel reviewed these sections.

9.1 Seagrass quality and extent

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Preamble: Aotearoa has only one species of seagrass, *Zostera muelleri*, which is commonly also referred to as karepō, nana, rehia, rimurehia and eelgrass. I have interpreted the seagrass quality attribute to exclude other plant species sometimes referred to as 'seagrass', including *Ruppia* spp.

Note that I have used the terms 'habitat' and 'meadows' interchangeably to reflect areas of seagrass regardless of size. Also note that seagrass 'extent' has been encompassed under the seagrass 'quality' attribute given extent is one indicator of wetland condition as per (41).

State of Knowledge of the "Seagrass quality and extent" attribute: Good / established but incomplete – general agreement, but limited data/studies

Overall, I consider the state of knowledge for the seagrass quality and extent attribute to be 'good / established but incomplete'. Internationally and nationally, there is excellent evidence relating seagrass quality (including extent) to ecological integrity. NZ-specific data that quantify stressor impacts on seagrass quality and tipping points are good. Seagrass monitoring guidance for local councils has recently been outlined. However, monitoring of seagrass quality (beyond extent and percent cover) is only routinely carried out in a limited number of areas around the country, leading to a lack of national-scale data and baselines for comparison. Management interventions to protect and restore seagrasses are well known although emerging restoration techniques (seed-based), to facilitate large-scale restoration, are in development.

Part A—Attribute and method

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

There is excellent evidence globally and in Aotearoa New Zealand (hereafter Aotearoa) to show that seagrass quality is closely tied to ecological integrity. Seagrass meadows are one of the world's most valuable coastal ecosystems (1), offering an array of ecosystem services that benefit society and the environment (2, 3). These services include (but are not limited to) supporting biodiversity and food security, regulating water quality and mitigating climate change (4). Seagrasses are naturally found along coastlines throughout Aotearoa, usually on soft sediments and in low-energy environments such as estuaries and harbours. Their existence within the land-sea interface makes them important for terrestrial, freshwater, estuarine and nearshore coastal ecosystems.

In Aotearoa, seagrass habitats are important for supporting various fish species, including Australasian snapper (tāmure, *Chrysophrys auratus*), trevally (araara, *Pseudocaranx georgianus*) and mullet (e.g., yellow-eyed mullet, aua, *Aldrichetta forsteri*)(5, 6). Seagrass meadows also harbour diverse benthic macrofauna and small mobile invertebrate communities, and are often used by foraging birds – e.g., see reviews by (6-9), and see also (10) and (11). Furthermore, a study in Aotearoa demonstrated that seagrass (artificial mimics) was important for juvenile fish settlement with increased fish numbers associated with higher seagrass blade density (12, 13). Some animal species are closely associated with seagrass meadows. For example, the black (or wide-bodied) pipefish (*Stigmatopora nigra*) is most often observed in subtidal seagrass meadows compared to other habitats (14).

In Aotearoa and overseas, it has been demonstrated that seagrass habitats help to regulate the climate by sequestering carbon dioxide through photosynthesis and storing organic carbon in the sediments beneath them (15-17). Studies have also demonstrated the role played by seagrass meadows in regulating coastal water quality by acting as a natural filter, trapping fine sediments and taking up nutrients and contaminants (e.g., metals from the water) as they grow, thereby improving or maintaining coastal water quality (18-21). Seagrasses also have wave attenuation properties that help to protect coastal shorelines from erosion (22-25).

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

Globally and nationally, there is excellent and good evidence respectively of the impact of degraded seagrass quality on the ecological integrity of coastal systems. For example, overseas, loss of seagrass has led to release of carbon (26, 27) and disrupted marine ecosystems including loss of habitat and food (28, 29). Historical records from Aotearoa indicate the detrimental impacts of seagrass loss on biodiversity such as fish, invertebrates, and birds (see review by [30]). The results of the many recent studies on benefits of seagrass in Aotearoa (see Section A1 above) also support the concept that numerous animal species will be impacted detrimentally, as well as various other ecosystem services, if seagrass is degraded / lost.

Significant historic reductions in seagrass extent have been documented in various estuaries and harbours in Aotearoa (7). Seagrass loss continues today in many places e.g., (31-34).

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

Seagrass quality in respect to extent in Aotearoa has declined over time (see review by (7). As such, seagrass is listed as 'at risk – declining' (35). It appears that subtidal seagrass is more vulnerable than intertidal seagrass, likely due to impacts of poor water quality. At least 39 threats were identified overall for seagrass habitat in Aotearoa by (36). Threats deemed to have major impact on seagrass meadows were sedimentation, reclamation, benthic accumulation of debris from marine farms, causeway construction, and nitrogen and phosphorus loading. The interactions among sustained stressors continue to reduce habitat suitability, and thus seagrass quality. Furthermore, lag times between management actions and stressor reduction remain for some cases (see Section B5). While these multitude of stressors are actively interacting to reduce seagrass quality (to varying magnitudes based on location), most can be considered reversible. Natural recovery of seagrass over large scales (i.e., multiple hectares or more) can take a relatively long time (i.e., between five to fifty years - see review by (37)). Furthermore, it is possible that there may not be full recovery without additional interventions. This means that retaining or improving seagrass quality, will be heavily dependent on effective legislative action that affords seagrasses adequate protection, monitoring, risk mitigation, and restoration where needed.

Climate change also impacts seagrass quality (36) and stressors associated with this are predicted to exacerbate over the next 10-30 years (38). See Section D3 for climate change impacts and management actions.

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

Beyond extent and percent cover, there is no routine monitoring of seagrass quality collected on a nation-wide scale across the country. Seagrass extent is monitored by local government for state of the environment purposes within many estuaries nationally[1]. For intertidal meadows, extent mapping is usually conducted following the standardised broadscale method under the national estuary monitoring protocol (NEMP, (39)). Mapping methods for subtidal meadows differ to those in the intertidal, especially for ground truthing, due to the underwater environment e.g., as per (40, 41). Seagrass loss compared to historical extent is sometimes reported in association with mapping where baseline information is available e.g., indicator called '% decrease from baseline' (42). Additionally, information on seagrass percent cover (which relates to quality) is being routinely collected in many estuaries to complement broadscale mapping e.g., (42). There are also some cases where fine-scale monitoring following the NEMP is carried out within a seagrass habitat and data such as percent cover and environmental parameters are collected e.g., (43, 44). Seagrass health (i.e., quality) beyond extent and percent cover is monitored more specifically by councils in some areas e.g., (41, 45). This monitoring can encompass a range of seagrass health indicators such as epiphyte and macroalgal cover, prevalence of fungal wasting disease and biomass and chemical and environmental parameters.

Besides mapping under the NEMP, various monitoring protocols relevant to seagrass quality exist for Aotearoa. Recent guidance for councils outlines seagrass monitoring approaches and methods for seagrass extent/percent cover and seagrass and environmental health indicators (46). The Estuarine Trophic Index also mentions various seagrass/related supporting indicators for assessing estuary trophic state (47). A recent scoping review of the NEMP includes seagrass condition surveys as 'targeted investigations' (48), so perhaps these surveys will be embedded in the NEMP in future. Work that will propose indicator metrics for seagrass quality (to MfE) is underway (Stevens et al. in prep. 2024). The 'wetland condition index' by (49) is also technically applicable to seagrass, although it doesn't appear to be specifically targeted at this habitat type and some of the indicators at least are not relevant e.g., 'dryland plant invasion' and 'fire damage'.

[1] Some seagrass extent data from across Aotearoa are reported on the Department of Conservation 'Our Estuaries Hub'.

Some seagrass quality data are present for Aotearoa across various scattered sources (besides local council SOE monitoring). These data are gathered for individual studies / research or for resource management act (RMA) related processes e.g., (40, 50-54).

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

Most seagrass quality monitoring methods require on-the-ground fieldwork and therefore come with various health and safety considerations for field personnel. Weather and sea state including ocean currents are particularly important safety considerations for monitoring subtidal meadows in relation to boating and / or SCUBA diving activities. Safety considerations for monitoring also relate to how the meadow will be accessed in respect to presence of any water channels and areas of deep substrate and whether a 4-wheel drive vehicle or a boat is required. The safest and / or most cost-effective site access option may also require travel over private (including Māori-owned) land in some cases. Accessing private property without the owner's consent can be considered trespassing, so clear communication, establishing good relationships, and addressing any concerns or impacts on the landowner's property or operations would be necessary. Some seagrasses may be in or nearby culturally significant areas.

Depending on the monitoring method used, technical expertise such as mapping / GIS skills and laboratory testing may be required. Monitoring may also require specialised equipment such as light and temperature loggers. Aerial imagery of the seagrass meadow, of suitable quality, taken at low tide and with no cloud cover and likely within a certain season or month, is also required for mapping extent of intertidal meadows. This imagery can be collected by aeroplane, drone or satellite (46). Furthermore, SCUBA diving and boating expertise/equipment and underwater cameras may also be required for monitoring subtidal seagrass.

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

I anticipate that the main cost to undertake seagrass quality monitoring (as per 46) is paying field staff for their time. However, depending on the monitoring approach / methods followed, other costs include laboratory analysis of various parameters associated with the seagrass (e.g., leaf TN and TC – approximately $60 each) and sediment (e.g., grain size, organic matter, nutrients – approximate cost range between ten/s to > one hundred dollars each depending on the analysis[1]). There are also various equipment items although most of these are relatively inexpensive except for field loggers to measure temperature and light (cost is at least a couple of hundred dollars each) and GPS (see following section). In general, costs for monitoring subtidal seagrass meadows will probably be higher than for those in the intertidal given they usually require a boat including skipper and SCUBA divers (may also cost multiple thousands of dollars per day) and/or underwater drop- or towed video- cameras (purchase cost is hundreds to thousands of dollars, depending on type/model).

As an example, further details on costs to monitor intertidal seagrass extent following NEMP, one indicator of quality, are as follows. In 2002, the approximate cost to survey one estuary (for all substrate and vegetation types) following the NEMP was estimated to be between $15,000 to $30,000 (39). However, this cost was dependent on the size of estuary and whether suitable aerial

[1] Based on current prices from Hill Laboratory

photographs were available or needed to be obtained for the survey. The approximate cost now will likely differ e.g., to account for inflation and technological advances. Most costs will relate to personnel time spent collecting and analysing data and reporting results, however, key equipment includes GPS ($300 - $800) and ARC GIS or equivalent software ($100 - $3800). New more cost-effective techniques such remote mapping and machine learning, may be used in the future (46) e.g., as per (55, 56).

Monitoring frequency will also dictate costs over time. For mapping extent, for example, the NEMP recommends broadscale monitoring every 5 years while (46) outlines this to be carried out every one to three years depending on whether a gold or silver standard respectively is being followed.

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

In Aotearoa, tohu (i.e, ecological and cultural health indicators) have been developed / used for monitoring and management of local estuaries. One example of iwi-led estuarine monitoring methods are those by (57) for Whakatū / Nelson, which include seagrass under the category 'estuarine vegetation (wet part)'. This is specific to Whakatū and the whānau who co-developed the monitoring assessment, and further review would be required to understand the many estuarine and coastal monitoring assessment throughout Aotearoa and Te Waipounamu. However, it is unknown by the author at what scale this monitoring has been undertaken.

Māori indicators for wetland monitoring were outlined by (49). To my knowledge monitoring of these indicators has not been carried out for seagrasses in Aotearoa. The "Ngā Waihotanga Iho – The Estuary Monitoring Toolkit"[1] provides tools to measure environmental changes that occur in estuaries over time. Among other things, this involves flora identification (broadly for estuaries), but again I am unsure whether it has been implemented for seagrass.

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

Seagrasses that are part of larger, continuous coastal habitats tend to have higher habitat quality and ecological functions than isolated or fragmented meadows, all of which relates to 'landscape connectivity'. Seagrass quality links to sediment- and nutrient-related attributes as both will likely respond to catchment sediment loads. Other characteristics of mud or tidal flats (e.g., 'sediment carbon', 'sediment microbial processes', 'sediment bacteria') can also be influenced (and vice versa) by the establishment, distribution, and quality of seagrass and associated habitats, e.g., mangroves and saltmarsh (15, 58, 59). Seagrass also relates to other stressor-related attributes such as those associated with heavy metals and temperature changes brought about by climate change. Seagrasses that offer limited 'access to natural areas' (specifically in relation to human disturbance) also often have higher habitat quality and support more diverse ecological communities.

Seagrasses are transitional habitat found between multiple ecosystems, meaning there will likely be a crossover in monitoring methods with the quality and extent of habitats such as dunes, saltmarshes and mangroves.

[1] https://niwa.co.nz/te-kuwaha/tools-and-resources/ng%C4%81-waihotanga-iho-the-estuary-monitoring-toolkit

Part B—Current state and allocation options

B1. What is the current state of the attribute?

There is good evidence that seagrass habitats have been lost nationally over time (7), although historical baselines aren't necessarily known. The current state of remaining seagrass quality (beyond extent) is not well understood at the national and local level. There is information available from estuaries around the country on seagrass extent and percent cover based on habitat mapping. However, besides for a small number of locations, current knowledge and reporting of the quality of existing seagrass habitats, beyond extent / cover, in Aotearoa are poor.

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

Seagrass systems that have retained their historical extent and that have limited to no evidence of human-induced impacts (e.g., pollution, vehicle or anchor / mooring damage) could be considered a reference state with respect to quality. Seagrasses associated with remote, protected locations such as national parks may best serve as examples of natural states with limited impact from human-induced stressors. However, sites within remote, protected areas may still contain stressors such as from boat anchoring or mooring activities and be subject to climate change impacts. For subtidal seagrasses, offshore islands, where water clarity is good, may provide the best reference areas containing meadows even if the nearby land is not necessarily protected.

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

Advice on ecological quality status thresholds for seagrass (for extent and percent cover/density at least) in Aotearoa are currently in draft (60). Examples of thresholds previously applied or proposed for seagrass in Aotearoa include those by (47) and (61) based on current extent vs baseline. I know of no existing numeric or narrative bands specifically for fine-scale seagrass health monitoring in Aotearoa. (46) states that this type of monitoring can help inform future development of early warning indicators of decline.

Guidelines for allocating scores, on a scale of 0–5, to the various indicator components of the 'wetland condition index' are outlined in (49). Quantitative limits to maintain the ecological integrity of freshwater wetlands are detailed in (62). These are based on attribute states ranging from A (excellent condition) to D (poor condition). However, I do not know of any examples where the 'wetland condition index' or its indicator scores or states have been applied to seagrass (see Section A4-(i) for comment on relevance of this index to seagrass).

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

There is good evidence for seagrass quality tipping points or thresholds reported internationally (e.g., (29, 63-65) and nationally. In Aotearoa, for example, macrofaunal species diversity can increase with seagrass colonisation in relation to plant % cover (11) and fish abundance can increase when seagrass is present and has a higher plant shoot density - based on artificial seagrass (12, 13). Due to the importance of seagrass for ecological integrity, loss or degradation of this habitat type can directly impact ecological integrity (see Sections A1 and A2). Tipping points or thresholds for seagrass loss / degradation reported for Aotearoa depend on factors including seagrass coverage, biomass, water clarity, nitrogen loads/levels, storm events and sea level rise (e.g., (51, 52, 54, 66-69). Multiple stressors will likely need to be considered given that individual stressors in estuarine systems can be conditional on the state of other stressors (70).

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

Lag time between stressor and impact on seagrass quality will be site- and stressor-dependent. For example, there may be no lag time in cases of direct and severe physical damage, such as shoreline infilling / land reclamation for coastal development. Alternatively, lag times are expected from the impacts of stressors / factors such as land-based nutrient and sediment runoff (71, 72). In terms of the impact of non-indigenous species (such as exotic *Caulerpa*), there will be a timeframe when these are first present before becoming established and spreading. In respect to naturally occurring processes, seagrass extent / biomass / shoot density is known to vary across seasons and can also be influenced by long-term climatic patterns such as El Niño (8). Plant recovery may take time and be slowed (or impossible) when stressors such as poor light penetration to the seabed due to sediment loading are present (see summary of unsuitable parameter conditions in [37]).

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

Mātauranga Māori is place-based and so the local context and ecology is important. Coastal wetlands are highly valued by Māori as important systems that provide habitat for taonga species and as sources of mahinga kai (73). Seagrass (in some areas at least) is viewed by hapū and iwi as crucial to the mauri (life force) of the taiao (environment) (e.g., (40)). Seagrass rhizomes may have been used by historically for food and the leaves for adorning clothing (30). Indigenous-based tohu/indicators (i.e., specific tohu and/or taonga species, see Section A5) could be used to inform bands / allocation options. However, given it is context based, the bands should be informed by the approach exemplified by various iwi and hapū specific environmental assessments. For instance, the estuarine indicators for Whakatū, Nelson (57).

Part C—Management levers and context

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

In cases where seagrass is destroyed, such as shoreline infilling / land reclamation, there is obviously a direct detrimental relationship between the stressor and seagrass quality. Furthermore, there is also some evidence for Aotearoa on the impacts of other physical stressors such as vehicle and mooring damage, bird herbivory and human perturbation on seagrass quality (e.g., (41, 50, 74, 75). Various studies quantify relationships between seagrass quality and stressors in the water column and sediments caused by issues such as sedimentation and excessive nutrients in Aotearoa (see review by (76). However, there are still challenges associated with disentangling interactions among multiple stressors, respective lag times, additional legacy affects, and overall seagrass quality. In addition, the impact of stressors on ecosystems is usually highly context-specific (i.e., place and history are very important) and so effective management needs to understand and allow for that context.

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

Key management interventions include seagrass protection and the elimination or reduction of stressors. From a policy perspective, the RMA (1991) is a key piece of legislation that sets out how we should manage our environment. In addition, the New Zealand Coastal Policy Statement (77) guides councils in their day-to-day management of the coastal environment. There are various other relevant government-related directions and management implementations, for example for freshwater, climate change mitigation and adaptation, biosecurity, marine protected areas and fisheries. Despite this, government management is failing to fully protect seagrasses in Aotearoa as evidenced by their continued decline in many places.

Seagrass restoration is another management intervention that can be carried out to improve quality of degraded or lost seagrass by government, iwi / hapū, community groups or others interested in recovering estuarine habitat[1]. Removal / management of existing stressors and catchment-based restoration, such as riparian planting and fencing, can indirectly facilitate natural seagrass recovery (e.g., as potentially indicated by (78). The traditional method for actively carrying out direct seagrass restoration involves transplantation from wild meadows e.g., (79). Seed-based restoration also holds promise over larger scales but has not yet been carried out in the field in Aotearoa. A practical guide for carrying out seed-based seagrass restoration in Aotearoa has recently been created and has remaining knowledge gaps outlined (80).

C2-(i). Local government driven

Local governments can take action to protect (e.g., through policy / plans) and restore seagrass meadows at the regional level. A number of local government-driven initiatives have occurred at certain sites within Aotearoa aimed at restoring seagrass habitat directly using the technique of transplantation e.g., (79). There are also many examples of council-supported / led catchment-based restoration, which has potential to facilitate seagrass protection and recovery.

C2-(ii). Central government driven

Central government can provide key funding for the protection, conservation and restoration of seagrass across the country and for improvement of catchment health in general. For example, for relevant DOC activities, projects under the Freshwater Improvement Fund and MfE's 'At risk' catchments programme such as the Te Hoiere/ Pelorus Catchment Restoration Project.

C2-(iii). Iwi/hapū driven

Hapū and iwi are driving the Hinemoana Halo project alongside Conservation International (88), which includes supporting nature-based solutions including restoration of coastal wetlands and seagrass. There are many examples of the suite of tools that hapū and iwi have towards supporting

[1] Department of Conservation. Restoring estuaries. Restoring Estuaries Map. Retrieved March 25, 2024, from https://www.doc.govt.nz/nature/habitats/estuaries/restoring-estuaries-map/

ecological health (89), although are usually restricted to fisheries decisions rather than habitat improvement in estuaries and coastal environments (90).

In addition, in terms of research to inform management interventions, a project on potential estuarine thresholds and interventions, from land to sea, in collaboration with iwi, has aimed to achieve impact by combining knowledge streams to inform catchment-based solutions that lead to improved mauri of catchments and estuaries (82).

C2-(iv). NGO, community driven

A number of community-driven land-based restoration projects (that help to improve catchment health) exist throughout Aotearoa. Some examples include the Manawatū Estuary Trust, the Cobden Aromahana Ecological Restoration Group, and the Waimea inlet restoration project, to name a select few. Additional projects can be found at the DOC Restoring Estuaries Map[1]. NGO's and community groups may wish to support / carry out seagrass restoration in the future.

C2-(v). Internationally driven

Internationally-driven obligations relevant to the protection of seagrass include the Ramsar Convention of which Aotearoa is a signatory meaning it plays a part in the international effort to conserve wetlands[2]. There are multiple Ramsar sites around the country that contain seagrass habitats. Under the Convention to Biological Diversity (CBD), Aotearoa is required to have a national biodiversity strategy and action plan through which obligations under the CBD are delivered. Aotearoa has international climate change obligations such as those under the Paris Agreement. I understand that Aotearoa has also signed other international agreements (e.g., Free Trade) that require conditions around environmental management and climate change to be upheld. Restoring the vitality of degraded systems (which include seagrass ecosystems) is crucial for fulfilling the UN Sustainable Development Goals and for meeting the targets of the UN Decade (2021-2030) on Ecosystem Restoration (UN-DER).

Part D—Impact analysis

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

Failing to manage seagrass habitats poses a significant threat to coastal environments, triggering a cascade of ecological problems. For example, the degradation of seagrass quality can lead to a reduction or loss of habitat for biodiversity such as fish, invertebrates and birds (See Section A1 for importance of seagrass for biodiversity). Reductions in seagrass quality can sever vital links in the marine food web, which can have cascading impacts on the overall health and biodiversity of coastal ecosystems. Lost / degraded seagrasses may also lose their ability to filter pollutants and excess nutrients from the water, potentially leading to increasingly polluted coastal areas. Increased pollutants may disrupt the delicate balance of marine life and could trigger harmful algal blooms and

[1] Department of Conservation. Restoring estuaries. Restoring Estuaries Map. Retrieved March 25, 2024, from https://www.doc.govt.nz/nature/habitats/estuaries/restoring-estuaries-map/

[2] Ramsar Wetlands - National Wetland Trust of New Zealand | Learn more. (2021, September 7). National Wetland Trust of New Zealand. https://www.wetlandtrust.org.nz/get-involved/ramsar-wetlands/

oxygen depletion zones. Reduction in seagrass quality may also lead to increased erosion of shoreline habitats (e.g., as shown overseas (24)). Overall, impacts on marine ecological health and biodiversity have a detrimental flow-on effect for mauri and cultural practices such as mahinga kai (83, 84).

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

The economic impacts of seagrass degradation / loss are likely to be felt among fisheries, as fish-habitat associations have been quantified between subtidal seagrass and commercially important species such as snapper and trevally (see review by (5)). Furthermore, historical seagrass loss in certain areas within Aotearoa is indicated to have led to a decline of multiple marine-related fisheries (30) and references within. Reduced seagrass quality causing poorer water quality can lead to various detrimental implications for commercial activities such as aquaculture, fishing, tourism, and recreational industries such as water-sports events. Reductions in seagrass quality could also potentially limit their protective capacity as natural buffers that absorb wave energy and lessen storm surge impacts and erosion (e.g., (24, 25)), which is relevant for maintaining coastal infrastructure and tourism.

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

Climate change places additional pressure on seagrass meadows, for example, through exposure to more frequent and intense storm events, poorer water quality, increasingly frequent extreme temperature events (air and water) and sea-level rise (4, 85, 86).

Reducing / stopping anthropogenic greenhouse gas emissions is crucial for mitigating climate change impacts. Resilience of seagrass habitats to climate change may be improved by limiting impacts from other stressors, such as catchment sediment and nutrient runoff and protection from physical impacts such as boat anchoring / mooring and vehicle driving. Another management action to reduce climate change impacts on seagrass includes providing sufficient space for this habitat to migrate inwards, which can be accomplished through the removal of hard structures on the estuary seafloor such as seawalls and roads to reduce impacts of 'coastal squeeze' e.g., (87). The quality of seagrass habitats that are currently degraded can also be improved through actions such as increased protection and restoration.

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9.2 Saltmarsh quality and extent

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Preamble: 'Salt marsh' is a collective term that refers to the many species of salt tolerant plants that are found in bays, lagoons, estuaries, river mouths and sheltered coastal areas, usually at or above the high tide mark [2]. They are also characterised by the absence of species that are not salt tolerant [31].

Also note that we have encompassed salt marsh 'extent' under salt marsh 'quality' given extent is one indicator of wetland condition as per [31].

State of Knowledge of "Saltmarsh quality and extent" attribute: Good / established but incomplete – general agreement, but limited data/studies

Overall, we consider the state of knowledge for the salt marsh quality and extent attribute to be 'good / established but incomplete'. Internationally and nationally, there is excellent and good evidence, respectively, relating salt marsh quality (including extent) to ecological integrity. However, NZ-specific data that quantifies stressor impacts on 'quality' and associated ecosystem services are limited, and data on tipping points are lacking. Nationally, there is a standardised protocol for monitoring, and management interventions are well known. However, monitoring of salt marsh quality is not routinely carried out across the country, leading to a lack of national-scale data and baselines for comparison.

Part A—Attribute and method

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

There is excellent evidence globally and good evidence in Aotearoa New Zealand (hereafter Aotearoa) to show that salt marsh quality is closely tied to ecological integrity. Salt marshes are highly productive biogenic habitats that support biodiverse communities; these habitats are often comprised of various specialised plant and animal species that contribute to complex food webs, nutrient cycling processes and carbon storage [see review by [1,73,74]]. Salt marshes are naturally found along coastlines throughout Aotearoa. Their existence at the land-sea interface makes them important for terrestrial, freshwater, estuarine and nearshore coastal ecosystems.

Nationally, salt marshes support various threatened or at-risk plant and animal species including birds[2; 3, 4, 5, 6, 7, 8]. They can also provide habitat and foraging opportunities for native fish species[1, 9]. Salt marshes serve several physical functions, such as shoreline protection to control terrestrial erosion and natural wastewater treatment, improving water quality by trapping sediments and pollutants (e.g., excessive nutrients) from runoff[10; 11]; the latter of which helps prevent the contamination of coastal waters and may reduce the risk of harmful algal blooms. Furthermore, salt marshes sequester carbon dioxide, mitigating climate change by storing carbon in plant biomass and sediments [e.g., 12, 13].

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

Globally and nationally, there is good evidence of the impact of degraded salt marsh quality on the ecological integrity of coastal systems [e.g., for Aotearoa, 1; globally, 14]. Fragmentation and loss of salt marsh from stressors like coastal development, recreation, and livestock grazing / trampling has led to reduced filtering capacity, release of carbon, and decreased shoreline protection[11,12,15,16,17]. Notably, this has impacted the national-scale loss or severe reduction of salt marsh habitat for various threatened and at-risk bird species, the impacts of which can be seen in reduced population trends[18]. In addition, the incursion of invasive, weedy species (e.g., *Spartina* spp.) have displaced and/or outcompeted indigenous salt marsh flora[19]. However, it is worth noting that introduced plants can still carry out key ecosystems services such as wave attenuation[16] and / or carbon storage and sequestration[20].

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

Salt marsh quality in Aotearoa has declined over time, particularly due to habitat loss as a result of historic land reclamation [e.g., for agriculture and/or development, 1, 2, 21, 22, 23]. At least 31 threats have been identified for salt marsh habitat in Aotearoa. Threats deemed to have extreme effects on salt marsh are further land reclamation and sea level rise, both of which have the capacity to eliminate existing salt marsh habitat[24]. Threats that can have a major impact on salt marsh are causeway construction, increased sediment loading on rivers, and oil pollution[24]. The interactions among sustained stressors continue to reduce habitat suitability, and thus salt marsh quality, for key indigenous saltmarsh-forming species [e.g., 1, 25]. Furthermore, lag times between management actions and stressor reduction remain for some cases (see Section B5). While these multitude of stressors are actively interacting to reduce salt marsh quality (to varying magnitudes based on location), most can be considered reversible. However, natural salt marsh recovery is generally slow (up to 40 years) and may not fully recover without additional interventions[26, 27]. This means that retaining or improving salt marsh quality will be heavily dependent on effective legislative action that affords salt marshes adequate protection, monitoring, risk mitigation, and restoration where needed [22, 28, 29].

Climate change is also predicted to impact salt marsh quality and stressors associated with this are expected to exacerbate over the next 10-30 years[24, 30]. See Section D3 for climate change impacts and management actions.

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

To our knowledge, there is no routine monitoring or reporting of salt marsh quality (beyond extent), *per se*, on a national scale for Aotearoa. As a result, there is consensus lacking on the most appropriate measurement method for assessing this attribute in Aotearoa. A standardised monitoring protocol for measuring wetland condition, based on both broad- and plot-scale data collection and applicable to salt marsh, was developed by [31]. This 'wetland condition index'[1] does not appear to be routinely used in Aotearoa for state of the environment (SOE) monitoring of salt marshes, although some councils have applied it in individual cases [see [32,33]].

Data on salt marsh extent (including dominant vegetation types), which is relevant to salt marsh quality, is routinely collected through SOE estuary habitat mapping by councils e.g., following the National Estuary Monitoring Protocol [NEMP] broadscale methods[34]. Historical loss in salt marsh area is also often reported during broadscale estuary SOE monitoring where this information is available (e.g., indicator called '% of historical remaining')[35]. A recent scoping review of the current NEMP outlines a conceptual approach for future salt marsh monitoring that includes 'condition' in respect to factors such as erosion, vehicle damage and weed incursion[36]. National-level monitoring of saline wetland extent (which includes salt marsh and mangroves) is reported in a Stats NZ environmental indicator called "Wetland Area"[2]. However, this assessment does not capture small wetlands less than 0.5 ha in area that are known to have important ecological value, e.g., [21].

Some salt marsh quality data are present for Aotearoa across a number of scattered sources. Some of these data are gathered for individual studies or research, e.g., impacts of livestock grazing, [37]; vehicle use [29]. Some of it will also likely result from resource management act (RMA) related processes. Additional information on the presence of non-indigenous plant species (e.g., the invasive weed *Spartina* spp.) is being collected by councils and / or other government organisations such as Department of Conservation (DOC) to inform eradication activities, for e.g., as part of regional council pest management plans [38]. Data on the number / type of non-indigenous species trapped (e.g., for pest mammals) or eradicated (e.g., for *Spartina*) along with biodiversity indicators (e.g., indigenous bird abundance), is likely to be collected and stored haphazardly by various community groups[3] and governmental organisations like DOC.

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

Current salt marsh quality monitoring methods require on-the-ground fieldwork. Access is a key consideration given that some salt marshes are located on private land and are therefore subject to

[1] The 'wetland condition index' encompasses five wetland condition indicators: change in hydrological integrity, change in physicochemical parameters, change in ecosystem intactness, change in browsing, predation and harvesting regimes, and change in dominance of indigenous plants.
[2] https://www.stats.govt.nz/indicators/wetland-area/
[3] For example, Tasman Environmental Trust's 'Battle for the banded rail' project. https://www.tet.org.nz/projects/battle-for-the-banded-rail/

the landowner's property rights. The safest and / or most cost-effective site access option may require travel over private (including Māori-owned) land in some cases. Accessing private property without the owner's consent can be considered trespassing, so clear communication, establishing good relationships, and addressing any concerns or impacts on the landowner's property or operations will be necessary. Formal access agreements or contracts may need to be established. It is possible that some salt marshes are not able to be accessed during certain times of year due to ecological factors such as nesting of rare birds. Some salt marshes may also be in or nearby culturally significant areas.

Various health and safety factors also need to be considered in relation to fieldwork. These include access to the marsh in respect to presence of any channels of water and whether a 4-wheel drive vehicle or a boat is required for transport. Depending on the monitoring method being used, technical expertise such as plant species / taxa identification and mapping / GIS skills may also be required. Aerial imagery of the salt marsh, of suitable quality, taken with no cloud cover and likely within a certain season or month, is also required for mapping habitat extent. This imagery can be collected by aeroplane, drone or satellite. Additionally, monitoring may need specialised scientific equipment such as field pH and/or conductivity meters depending on the method being used.

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

We anticipate that the main cost to undertake monitoring of the five indicators encompassed under the wetland condition index by [31] is paying field staff for their time. However, other costs include laboratory analysis of various parameters within soil samples. There are also a number of required equipment items although most of these are relatively inexpensive except for field meters to measure pH and conductivity (which could cost approximately $6000 combined) and GPS (see following section). Refer to the attribute template for the 'Wetland condition index' for further detail.

As an example, in 2002, the approximate cost to survey one estuary (for all substrate and vegetation types) for salt marsh extent (i.e., one indicator of quality) following NEMP was estimated to be between $15,000 to $30,000 [34]. However, this cost was dependent on the size of estuary and whether suitable aerial photographs were available or needed to be obtained for the survey. The approximate cost now will likely differ e.g., to account for inflation and technological advances. Most costs related to personnel time spent collecting data and reporting results, however, key equipment includes GPS ($300 - $800) and ARC GIS or equivalent software ($100 - $3800). New, more cost-effective techniques, such remote mapping and machine learning e.g., [39], may be used in the future but this will need to gather the relevant information required to inform salt marsh quality [e.g., as per [31]].

Monitoring frequency will also dictate costs over time. For mapping extent, for example, the NEMP recommends broadscale monitoring every 5 years.

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

There are te ao Māori based indicators for wetland monitoring, as outlined by [31]. To our knowledge monitoring of these indicators has not been carried out on a regular (or even infrequent) basis for salt marshes in Aotearoa. The "Ngā Waihotanga Iho – The Estuary Monitoring Toolkit"[1] provides tools to measure environmental changes that occur in estuaries over time. Among other things, this involves flora identification (broadly for estuaries), but again we are unsure whether it has been implemented.

Environmental-based indicators including cultural health indicators, have been co-developed for monitoring and management of local wetlands and some estuaries. An example is development of cultural health assessments by Te Rūnanga o Ngāi Tahu for Te Ihutai/Avon-Heathcote Estuary to ensure their values are included [40]. It is noted that within the assessment were observations related to modifications of vegetation communities regarding the incursion of invasive species [40]. Another example of iwi-led estuarine monitoring methods that includes salt marsh is outlined by [41] for Whakatū/Nelson, however, we are unsure whether this monitoring has been implemented.

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

Salt marshes that are part of larger, continuous coastal habitats and have unobstructed hydrological connections tend to have higher habitat quality and ecological functions than isolated or fragmented marshes, all of which relates to 'landscape connectivity'. Salt marshes that offer limited 'access to natural areas' (specifically in relation to human disturbance) also often have higher habitat quality and support more diverse ecological communities. However, this does not exclude the potential impacts of pollution on salt marsh quality from 'microplastics' and 'heavy metals', which can come from distant sources. Additionally, salt marshes often occur in areas with significant mud deposition from tidal action, meaning 'mud extent' and characteristics of mud or tidal flats (e.g., 'soil bacteria composition', 'soil carbon', 'surface erosion/runoff control') can influence the establishment, distribution, and quality of salt marsh and associated habitats, e.g., mangroves and seagrass[1].

Salt marshes are transitional habitat found between multiple ecosystems, meaning there will likely be a crossover in monitoring methods with the quality and extent of habitats such as dunes, seagrass, and mangroves. In addition, the 'wetland condition index' is applicable to other wetlands besides salt marshes. Provided there is sufficient data resolution to assess salt marsh quality (i.e., that can suitably measure indigenous vs non-indigenous species), methods for monitoring 'indigenous plant dominance' in the terrestrial domain may also overlap with those used to measure salt marsh quality.

Part B—Current state and allocation options

B1. What is the current state of the attribute?

There is good evidence that salt marshes have been lost nationally over time[22, 21], although historical baselines aren't necessarily known. The current state of remaining salt marsh quality is not well understood at the national and local level. There is information available from estuaries around the country on salt marsh extent based on habitat mapping. However, in general, current knowledge and reporting of the overall quality of existing salt marsh habitats in Aotearoa are poor.

[1] https://niwa.co.nz/te-kuwaha/tools-and-resources/ng%C4%81-waihotanga-iho-the-estuary-monitoring-toolkit

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

Salt marsh systems that have retained their historical extent and have limited to no introduced plant species or evidence of human-induced impacts (e.g., vehicular trampling, pollution, livestock grazing) could be considered a reference state with respect to quality. Salt marshes found in remote, protected locations such as national parks may best serve as examples of natural states with limited impact from human-induced stressors. However, sites within remote, protected areas may still contain stressors like introduced weeds and mammals and be subject to climate change impacts.

Information regarding historical reference state could also be gathered through methods such as sediment coring under existing and former salt marsh habitats, which can avail reference states via analysis of pollen, plant remains, and geochemical indicators [e.g., [42, 43]].

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

Guidelines for allocating scores, on a scale of 0–5, to the various indicator components of the 'wetland condition index' are outlined in [31]. Additional, quantitative limits to maintain the ecological integrity of *freshwater* wetlands are detailed in [44]. These are based on attribute states ranging from A (excellent condition) to D (poor condition). There is at least one example where these freshwater wetland attribute states have been applied to salt marshes [e.g., [32]].

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

International thresholds / tipping points for salt marsh quality have been reported, but data for Aotearoa is lacking. The thresholds / tipping points depend on salt marsh type and local structure of associated salt marsh floral and faunal communities. For example, shifts in characteristics of nekton and vegetation communities from a natural state (e.g., indigenous plant and fish dominated communities) to impacted states (e.g., invasive plant and invertebrate dominated communities) can signal ongoing degradation[45]. Slowed salt marsh recovery rates, increased similarity in saltmarsh quality (i.e., low quality) across time and sites, increasing variability in seasonal salt marsh state, and increased variability in data structure (i.e., increased data skewness) have also been related to tipping points in salt marsh states[46].

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

Lag time between stressor and impact on salt marsh quality will be site- and stressor-dependent. For example, there may be no lag time in cases of direct and severe physical damage, such as shoreline hardening for coastal development. Alternatively, lag times are expected from the impacts of stressors / factors such as contaminant incursion, droughts and land-based nutrient and sediment runoff [e.g., [47, 48, 49]]. In terms of the impact of non-indigenous plant species, there will be a time when these exist as seeds / seedlings before becoming established and spreading.

Plant growth and recovery of species interactions can be highly influenced by legacy effects following historic land reclamation, coastal development[1], drainage [50], nutrient enrichment[51], and fires[28].

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

Mātauranga Māori is place-based and so the local context is important. Salt marsh and coastal wetlands are highly valued by Māori as important systems that provide resources for cultural practices [e.g., weaving, construction, 51], as habitat for taonga species and mahinga kai species [e.g.,53]. Understanding Indigenous-based indicators (i.e., specific tohu and/or taonga species, see Section A5) with whānau, hapū and iwi could be used to inform bands/allocation options.

Part C—Management levers and context

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

In cases where salt marsh is destroyed, such as shoreline hardening, there is obviously a direct detrimental relationship between the stressor and salt marsh quality. Furthermore, there is also some information for Aotearoa documenting the relationship between salt marsh quality and other physical stressors such as livestock grazing and trampling and vehicle damage [e.g., 29, 37]. However, there are still challenges associated with disentangling interactions among multiple stressors, respective lag times, additional legacy affects, and overall salt marsh quality. In addition, the impact of stressors on ecosystems is usually highly context-specific (i.e., place and history are very important) and so effective management and needs to understand and allow for that context.

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

Key management interventions include salt marsh protection and the elimination or reduction of stressors. From a policy perspective, the RMA (1991) is a key piece of legislation that sets out how we should manage our environment. In addition, the New Zealand Coastal Policy Statement[54] guides councils in their day-to-day management of the coastal environment. There are various other relevant government-related directions and management implementations, for example for freshwater, biosecurity, fisheries, wildlife, climate change, threatened species and national parks. Despite this, [22] and [55] outline how government management is still failing to fully protect wetland / salt marsh habitats in Aotearoa.

Salt marsh restoration is another management intervention that can be carried out to improve quality of degraded salt marsh by government, iwi/hapū, community groups or others interested in recovering estuarine habitat [e.g., 56]. Salt marsh species are grown in some commercial nurseries and so are widely available for restoration projects, and there have been several studies on survival and growth of various species used in salt marsh restoration that can inform restoration efforts [e.g., see review by 1].

C2-(i). Local government driven

Local governments can take action to protect (e.g., through policy / plans) and restore salt marsh at the regional level. A number of local government-driven initiatives are present throughout Aotearoa aimed at restoring salt marsh habitat. Some examples include the restoration of endemic salt marsh species such as oioi (Salt marsh Ribbonwood, *Apodasmia similis*) at sites along the Taranaki coastline; salt marsh and adjacent habitats at Charlesworth Reserve, Christchurch; and the restoration of Wainono Lagoon, Canterbury. Additional projects can be found at the DOC Restoring Estuaries Map[56].

C2-(ii). Central government driven

Central government can provide key funding for the protection, conservation and restoration of salt marshes. For example, for relevant DOC activities, projects under the Freshwater Improvement Fund and MfE's 'At risk' catchments programme such as the Te Hoiere / Pelorus Catchment Restoration Project. There is also potential to consider salt marsh restoration for carbon credits, for example within the Government's Emissions Trading Scheme [12, 57].

C2-(iii). Iwi/hapū driven

Indigenous taonga and their ecosystems are protected under Te Tiriti o Waitangi. Hapū and iwi are driving the Hinemoana Halo project alongside Conservation International [75], which includes supporting nature-based solutions including restoration of coastal wetlands, which holistically can support supporting saltmash habitats. There are many examples of the suite of tools that hapū and iwi have towards supporting ecological health [76]. In addition, in terms of research to inform management interventions, a project on potential estuarine thresholds and indicators, included collaboration with iwi, has aimed to achieve impact by combining knowledge streams to inform catchment-based solutions that lead to improved mauri of catchments and estuaries[58]

C2-(iv). NGO, community driven

A number of community-driven wetland restoration projects exist throughout Aotearoa. Some examples include the Manawatū Estuary Trust, the Cobden Aromahana Ecological Restoration Group, and the Waimea inlet restoration project, to name a select few. Additional projects can be found at the DOC Restoring Estuaries Map[56]. There is also potential to consider salt marsh restoration for carbon credits including through voluntary offset schemes e.g., as is being explored by The Nature Conservancy (an NGO) for Aotearoa[59].

C2-(v). Internationally driven

Internationally-driven obligations relevant to the protection of salt marsh quality include the Ramsar Convention of which Aotearoa is a signatory meaning it plays a part in the international effort to conserve wetlands[60]. There are multiple Ramsar sites around the country that contain salt marsh habitats. Under the Convention to Biological Diversity (CBD), Aotearoa is required to have a national biodiversity strategy and action plan through which obligations under the CBD are delivered. Aotearoa also has international climate change obligations such as those under the Paris Agreement. We understand that Aotearoa has also signed other international agreements (e.g., Free Trade) that require conditions around environmental management and climate change to be upheld. Restoring the vitality of degraded systems (which include salt marsh ecosystems) is crucial for fulfilling the UN Sustainable Development Goals and for meeting the targets of the UN Decade (2021-2030) on Ecosystem Restoration (UN-DER).

Part D—Impact analysis

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

Failing to manage salt marshes poses a significant threat to coastal environments, triggering a cascade of ecological problems. For example, the degradation of salt marsh quality can lead to a reduction or loss of salt marsh nursery functions, which is reflected in the decline of bird populations [1]. Additionally, impacted salt marshes may lose their ability to filter pollutants and excess nutrients from runoff water before it reaches the ocean leading to increasingly polluted coastal areas. This influx of pollutants may disrupt the delicate balance of marine life and can trigger harmful algal blooms and oxygen depletion zones. Reductions in salt marsh quality can sever vital links in the marine food web, which can have cascading impacts on the overall health and biodiversity of coastal ecosystems. Reduction in salt marsh quality may also lead to increased erosion of shoreline habitats. Overall, impacts on marine ecological health and biodiversity have a detrimental flow-on effect for mauri, cultural practices and mātauranga such as those interconnected with mahinga kai (site, species and habitats).

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

The economic impacts of salt marsh degradation / loss are likely to be felt among fisheries, coastal infrastructure development, and tourism sectors. Reductions in salt marsh quality could lead to a loss of habitat for commercially-important species[1, 9, 61]. Similarly, reductions in salt marsh quality can limit the suitability for threatened and endangered bird species, which limits coastal tourism opportunities for certain groups (e.g., birders) and local businesses[62]. Reduced salt marsh quality causing poorer water quality, can lead to various detrimental implications for commercial activities such as aquaculture, fishing, tourism, and recreational industries such as water-sports events. Reductions in salt marsh quality can also limit their protective capacity as natural buffers that absorb wave energy and lessen the impact of storm surges[63]. The loss of salt marsh exposes coastlines to increased erosion, leading to a retreat of beaches and a heightened risk of damage to coastal infrastructure[64].

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

Sea level rise and increased storm frequency is expected to lead to the erosion and loss of existing salt marsh [e.g., from 'drowning', 30, 65, 66]. Sea level rise may also result in reduction in salt marsh extent (and therefore quality) due to 'coastal squeeze' if suitable space is not available for it to migrate to due to the presence of roads, urban areas, stopbanks, or agricultural land directly inland from current salt marsh[67, 68]. Increased storm frequency will likely lead to increased run-off of sediment, nutrients and contaminants. Although note that sediment supply is key for salt marsh vertical accretion, which can improve marsh resilience to sea level rise[69]. Alterations to water quality, which includes changes in salinity, can alter salt marsh community structure and lead to further incursion of invasive species. Increasing temperatures due to climate change is already leading to range shifts of indigenous and invasive species. Increased frequency of fires as a result of climate related issues, such as drought poses an additional risk as many salt marsh species, overseas at least, are shown to be not well adapted to wildfire regimes and demonstrate even slower recovery after burns compared to non-burned salt marshes [e.g., [70]]. However, increasing temperatures may yield some-short term benefits for certain salt marsh vegetation species, such as increased biomass[71].

Reducing / stopping anthropogenic greenhouse gas emissions is crucial for mitigating climate change impacts. Besides this, management actions to reduce climate change impacts on salt marsh include providing a habitat buffer along current salt marsh margins to provide sufficient space for this habitat to migrate inwards. This can be accomplished through the removal of any hard structures such as seawalls and roads, to reduce impacts of ‘coastal squeeze’[68, 72]. Resilience of salt marsh habitats to climate change can be further improved by reducing impacts from other stressors, such as catchment sediment and nutrient runoff and protection from livestock grazing and development. The quality of salt marsh habitats that are currently degraded can also be improved through actions such as increased protection and restoration.

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9.3 Mangrove forest extent and quality

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State of knowledge of the "Mangrove Habitat Extent and Quality" attribute: Good / established but incomplete – general agreement, but limited data/studies

Our overall assessment of the state of knowledge for the attribute Mangrove Forest Extent and Quality is good/established but incomplete. There is generally good information on the broadscale spatial extent and distribution of mangrove forest in New Zealand's northern estuaries. Information sources include aerial photographic surveys extending back to the 1930s, with progressive improvements over time (i.e., black & white colour, scale, accuracy). Present-day satellite coverage (e.g., ESA Sentinel) provides high-resolution multi-spectral products for mapping mangrove forest extent and some aspects of attribute quality. LiDAR coverage of intertidal habitats has also steadily improved in frequency, resolution, and accuracy. 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. The quantity and frequency of on-the-ground monitoring of mangrove forest characteristics and the quality thereof varies between regions, as do the variables measured. Research interest in NZ mangrove forest systems has grown substantially since the 1970s and particularly over the last 20 years. The knowledge generated by this research encompasses the biophysical and social sciences and research and applied studies on coastal wetland blue carbon and biodiversity, which has gathered pace within the last decade. Understanding of the drivers of NZ mangrove forest development and ecosystem function are good, although much of this work has been conducted at a handful of locations. Knowledge gaps remain in aspects of mangrove forest characteristics/quality. Understanding of the future resilience of NZ mangrove forests to climate warming and relative sea level rise across the range of environmental setting where they occur is at an early stage.

Part A—Attribute and method

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

Mangrove habitat expansion in New Zealand's northern estuaries from the mid-1800s onwards has largely occurred due to the expansion and vertical accretion of (unvegetated) intertidal flat habitat, suitable for mangrove colonisation. 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). In this sense, the general pattern of mangrove habitat expansion in many estuaries is a symptom of excessive soil erosion during the historical era [e.g., reviews (Horstman et al. 2018; Morrisey et al. 2010; Swales et al. 2020b)]. Although monitoring changes in the extent of mangrove habitat is readily amenable to remote sensing, the range of mangrove in New Zealand, limited to the upper North Island, means that mangrove habitat extent is not a suitable indicator of ecological quality at a national scale. Below we describe the key processes driving mangrove habitat expansion in New Zealand's northern estuaries and their ecological values.

Due to their physiology, mangroves require regular tidal exposure to the air with a maximum hydroperiod (i.e., duration and frequency of inundation) that coincides with mean sea level. NZ mangrove forests are composed of monospecific stands of the grey mangrove (Mānawa, *Avicennia marina* var. *australasica*) and occur south to Kawhia Harbour on the west coast and to Ōhiwa Harbour on the east coast (i.e., 38.1°S). Their national distribution has been controlled by low winter air temperatures, frost frequency, biogeography and oceanography that limit mangrove propagule dispersal [e.g., de Lange & de Lange (1994)] and the limited availability and relative remoteness of suitable estuarine environments south of their present range. Although climate warming may change their potential range (e.g., landward migration facilitated by sea level rise), biogeographic limitations to their southern expansion remain.

In general, the expansion of mangrove habitat in New Zealand's northern estuaries has coincided with adverse effects on the ecological integrity of estuaries. This has largely been driven by the delivery of mud (i.e., particle size < 62.5 micron) associated with increased catchment soil erosion in the historical era. In many estuaries, increased fine sediment loads from catchments as well as ongoing impacts today of legacy fine sediment have resulted in order of magnitude increases in sedimentation rates, shift from sand- to mud-dominated systems and reduction in the areal extent of subtidal habitats. Ecosystem degradation has been characterised by loss of plants and animals sensitive to increased water turbidity, reduced light levels, and smothering by fine sediment deposition (e.g., seagrass meadows, filter-feeding bivalves [Bainbridge et al. 2018; Booth 2019; Inglis 2003; Thrush et al. 2004; Zabarte-Maeztu et al. 2021]). The substrate elevation in many mangrove forests is also close to the upper limit of the present tidal frame (i.e., Mean High Water Spring) so that hydroperiods are short and the opportunity for estuarine fish to utilise mangrove forest is limited. Although many bird species make extensive use of mangrove habitat for roosting, feeding or breeding, there are no New Zealand birds that are exclusively found in mangroves (reviewed in Morrisey et al. 2010). Even in fine-sediment degraded estuaries, mangroves do provide important ecosystem services which include sequestration of fine sediment and associated stormwater contaminants (e.g., heavy metals and carbon [Bulmer et al. 2020; Swales et al. 2002]). Mangrove forests may also reduce the risk of coastal erosion and inundation associated with storm tides (Gijsman et al. 2021; Swales et al. 2015). Mangrove forests provide long-term carbon sequestration, serving as important carbon sinks, and mitigating climate change by reducing overall greenhouse gas

(GHG) emissions (Bulmer et al. 2020; Ross et al. 2024; Suyadi et al. 2020) and there is increasing interest in restoration of mangrove forests as a climate mitigation strategy (Stewart-Sinclair et al. 2024).

In a small number of estuaries that have not experienced large fine-sediment loadings, the ecological integrity of these systems may be enhanced by mangrove forests. Estuaries with relatively small catchments and/or retain areas of indigenous forest landcover fall into this category. These systems may include subtidal and intertidal seagrass meadows and mangrove habitat on sandy substrates (e.g., Rangaunu, Whangateau).

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

There is substantial weight of evidence that mangrove habitat expansion during the historical era is associated with adverse impacts on the ecological integrity of New Zealand's northern estuaries. This association is coincidental rather than causal, as the evidence indicates that mangrove habitat expansion is symptomatic of excessive fine sediment loading and sedimentation in NZ estuaries (Morrisey et al. 2010; Suyadi et al. 2019; Swales et al. 2020b) in comparison to pre-deforestation/baseline conditions (Suyadi et al. 2019).

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

Aerial photographic surveys since the late 1930s indicate the rate of mangrove habitat in New Zealand's northern estuaries expansion has averaged 4.1% yr^{-1} (range 0.2–20.2% yr^{-1}) (Morrisey et al. 2010; Suyadi et al. 2019). This expansion of NZ mangrove habitat has occurred at twice the rate as observed in the temperate *Avicennia* forests of southeastern Australia (average 2.1% yr^{-1}, range 0.7%–9.1% yr^{-1}, see review (Morrisey et al. 2010)). There is also evidence that the main phase of mangrove habitat expansion in some estuaries occurred at an early stage, during the period of catchment deforestation in the mid–late-1800s, and forest development was largely complete by the early-1900s [e.g., Kaipara Harbour, see review (Swales et al. 2020b)]. Mangrove habitat loss has also occurred over the last century due to land reclamation for ports and urban development, agriculture, road and causeway construction and landfills. Historical changes prior to the 1930s (both losses and gains) in the extent of New Zealand's mangrove forests have not been accurately quantified because substantial habitat change occurred prior to systematic aerial photographic surveys.

Comparison of the Landcover Database (Ver. 5, 2021) mangrove layers for 1996 and 2018 suggests that there has been virtually no measurable change in mangrove habitat in since the mid-1990s (i.e., 28,204 ha (1996), 28,172 ha (2018) (Hicks et al. 2019). However, there are several potential factors that suggest the LCDB mapping may not provide a reliable evaluation of recent changes in mangrove habitat extent. The LCDB Ver. 5 does not include more recent coastal vegetation mapping as part of updates to regional coastal plans or spatial planning processes (e.g., Northland Regional Coastal Plan; Tai Timu Tai Pari Hauraki Gulf Marine Spatial Plan [Waikato Regional Council 2017], or from mangrove removal activities which have been substantial in some harbours [Bulmer et al. 2017; Lundquist et al. 2012; Lundquist et al. 2014]). Further, broadscale metrics (i.e., hectare as per LCDB) are unlikely to adequately detect small scale changes in mangrove habitat boundaries, including increasing density of canopy cover on expanding forest fringes that cumulatively account for substantial expansion (see, e.g., Suyadi et al. 2018a, 2019). Mangrove seedling recruitment events resulting in large increases in habitat extent (i.e., tens of metre) are also infrequent, largely due to physical controls on the success of seedling establishment. This in turn is determined by the local wind-wave climate in estuaries and timing of propagule fall during the spring–neap tidal cycle. Large recruitment events depend on propagule fall coinciding with an extended period of calm weather (i.e., 1–2 weeks) and a declining spring to neap high tide level, when bed disturbance by wave action is minimal (Balke et al. 2015; Gijsman et al. 2024). Such is the case in the southern Firth of Thames, where large-scale seedling recruitment has not occurred since the mid-1990s (Swales et al. 2015). The detection of changes will depend on the spatial resolution of the base data for mapping (i.e., aerial photography, LiDAR, satellite). The LCDB mapping is also unlikely to have captured the gradual colonisation of saltmarsh habitat by mangroves that has been occurring, as described below.

Observations and anecdotal evidence also suggest, however, that mangroves have been colonising estuarine saltmarsh habitats over the last decade or so, which may not have been adequately captured by aerial surveys/remote sensing. Mangrove colonisation of saltmarsh has been observed by the authors in many estuaries, for example Whangateau Harbour, Tauranga Harbour and the southern Firth of Thames, and is no doubt occurring in other locations. In southeastern Australia, mangrove habitat expansion (*Avicennia* spp.) into saltmarshes is well documented. This habitat expansion has been attributed to climate warming and resulting southern extension of temperature thresholds coincident with SLR, although complicated by limitations on mangrove propagule dispersal, among other factors (Saintilan et al. 2014). This process also appears to be enhanced by meteorological events that provide windows of opportunity for mangrove propagule transport and establishment into saltmarsh habitat, including annual to decadal scale ENSO-cycles that influence sea level on the coasts during periods of onshore wind and/or lower than average atmospheric pressure (Swales et al. 2015; Swales et al. 2020b). Extreme storm tides and elevating sea level also provide a key mechanism for mangrove colonisation of saltmarsh habitat. For example, a storm tide in the southern Firth of Thames that occurred January 2018 (largest event since 1938) introduced vast numbers of propagules into a glasswort marsh (*Salicornia quinqueflora*) and resulted in the rapid loss of supratidal saltmarsh ribbonwood, most likely due to storm-induced salinisation (Swales et al. 2023). Both saltmarsh habitats pre-dated the development of the mangrove forest from the early 1960s. These climate-warming related processes will ultimately result the displacement of saltmarsh and mangrove dominance on temperate shorelines, including in the northern estuaries of New Zealand.

Future changes in mangrove habitat extent over the next 20–30 years will be influenced by sea level rise (SLR) associated with climate warming and physical barriers (i.e., natural or anthropogenic) to landward migration. To persist into the future, mangroves must maintain their elevation in the upper intertidal zone, above MSL. This can be achieved by one of two main mechanisms: (1) landward migration “upslope” and/or (2) vertical accretion of mineral sediment (from rivers) or organic sediment produced by the mangrove forests themselves (McBride et al. 2016; Swales et al., 2020a). In NZ mangrove systems, sedimentation is dominated by river-borne mineral sediment, with much smaller contributions from organic sediment several produced by the trees themselves. Over the next 20–30 years mangrove habitat migration into saltmarsh is likely to accelerate, with episodic storm tides perturbing the system and resulting in step changes in coastal wetland habitats. Section D3 describes the potential longer-term impacts of climate change.

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

Monitoring can occur at a range of scales, from monitoring of boundaries and expansion/contraction of individual mangrove forests using hand-held GPS (see techniques described in Swales et al. 2011) to broad scale habitat mapping using satellite remote sensing. Ground-truthed surveys by regional councils vary in frequency, from opportunistic responses to required updates for coastal policy statements (e.g., Northland Regional Plan, 2020) to scheduled updates.

Waikato Regional Council routinely measures the extent (but not quality) of mangrove habitats through two different projects. These projects use different methods and are undertaken for different purposes, which are mapping the extent of mangrove habitat for (1) a subset of Waikato estuaries as component of an environmental indicators [Extent of coastal habitats | Waikato Regional Council]; and (2) more detailed habitat surveys conducted every 10 years (plan to increase frequency to ~ 4 years subject to funding) [Intertidal habitat mapping for ecosystem goods and services: Waikato estuaries | Waikato Regional Council]. The mapping includes the mangrove/saltmarsh ecotone where these habitats are difficult to separate (e.g., Whangapoua Harbour) (Source: Dr Steve Hunt, WRC).

Bay of Plenty Regional Council map estuarine wetlands (including mangrove) along with freshwater wetlands using aerial photography, LiDAR/DEM and obliques. Wetland classes mapped follow Johnson and Gerbeaux (2004). Landcare Research is currently developing standards/guidelines for mapping extent of wetlands for MfE. Methods for freshwater and estuarine wetlands should align if possible. Mangrove and saltmarsh condition is also monitored (SOE) at ~150 vegetation plots established in five estuaries. Methods are adapted from those developed for freshwater wetlands (Clarkson et al. 2004). There is no standard for this type of monitoring that BoPRC are aware of (S. Dean, BOPRC, pers. comm.). Baseline data has been collected and the first survey will likely occur in 2024/2025. No data analysis has been undertaken to date. BoPRC are also undertaking other projects relevant to the mangrove and saltmarsh attributes:

- Long running study comparing mangrove sites to sites where mangroves have been removed – data on macrofauna, sediment quality and epifauna – data not yet analysed.
- Mangrove encroachment into saltmarsh: mapping was done in around 2011 but has not been repeated.
- Blue carbon coring and emissions work in saltmarsh and areas being restored into saltmarsh.
- Rod Surface Elevation Tables installed in saltmarsh and mangrove habitats in Athenree Estuary and Nukuhou Inlet, Ohiwa Harbour.
- The Coastal SIG also have an Envirolink-funded project to develop a tool for using satellite imagery to map coastal wetland habitats. (Source: Ms Shay Dean, BoPRC).

In the Auckland Region (Source: Grant Lawrence, AC), fine scale wetland-change mapping was conducted for the 2010-11 and 2017 period. Between these surveys, mangrove habitat expansion typically occurred seaward or sideways of existing mangrove habitat onto unvegetated intertidal flat (forming sparse or dense monocultures depending on the location). There are some examples of landward expansion, although these are typically onto unvegetated intertidal flat. At Puhinui and Pollen Island, where substantial areas of saltmarsh occur, mapping based on 2010-11, 2017 and 2023 imagery, shows a complex mosaic where Saltmarsh and Mangrove intergrade. This ecotone presently appears to be stable with no obvious signs of mangrove encroaching on/or displacing saltmarsh over the last decade.

Northland Regional Council mapped saltmarsh and mangrove habitat in 2020. This mapping is summarised in 19 worksheets covering the various harbours and estuaries (https://www.nrc.govt.nz/resource-library-summary/research-and-reports/saltmarsh-and-mangroves/), with mapping methods reported at https://www.nrc.govt.nz/media/5ynp3hea/northland-intertidal-vegetation-mapping-methodology-2020-2.pdf, and the GIS layers available at: https://localmaps.nrc.govt.nz/localmapsviewer/?map=55bdd943767a493587323fc025b1335c

More recently, NRC have mapped all of Northland's wetlands, including saltmarsh and mangrove, using a slightly different mapping method. The 2024 mangrove and saltmarsh layer was generated by combining 2014-2016 imagery and 2019 LiDAR data. These new map products will be released in 2024. (Source: Mr Richard Griffiths, Resource Scientist -Coastal).

Analysis of satellite remote sensing has also been opportunistic, including council-funded analyses, academic theses, and a recent MBIE-funded project on blue carbon that describes a national mapping exercise of mangroves, saltmarsh, and seagrass; Bulmer et al. 2024). There is no consistency between councils on coastal wetland typologies, with diversities of mangrove categories from differences based on morphology (scrub v. fringe forest) to mixed communities (i.e., ecotones) such as saltmarsh or seagrass within mangroves, or for differences in mangrove density and patchiness (see Suyadi et al. (2018a, 2019) for mangrove forest landscape characteristics). Mangrove density, height, and characteristics of patches (e.g., width) have strong influence on ecosystem services provided (Horstman et al. 2014; Suyadi et al. 2018a).

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

Primary monitoring of mangrove forest extent can occur via satellite remote sensing, thus there are no access issues, unless ground-truthing is required. Much of the information on mangrove habitat quality, associated with ecological and environmental characteristics (e.g., macrofaunal communities, sedimentation rates, sediment properties etc.) does require field surveys. In the Bay of Plenty, some sites have been excluded from the BoPRC mangrove and saltmarsh SOE monitoring programme due to a lack of landowner permissions.

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

Quantitative techniques for extracting hyperspectral signatures indicating presence of mangroves from satellite remote sensing have been developed based on current satellite technology (Sentinel-2 images). Analytical techniques were available for aerial photographs from prior analyses (Suyadi et al. 2018b; Swales et al. 2009). Sentinel-2 satellite images are open source, but operational costs (technical analyses) would be required to update mangrove maps on a regular basis. Timeframes of

5–10-year intervals are likely suitable to quantify changes in extent. During these longer timeframes, satellite technology may change, requiring development of new techniques, though global communities regularly provide open-source code for mangrove image analysis.

The costs associated with monitoring the mangrove-forest quality attribute in any given estuary will depend on a range of factors - the number and type of parameters measured, spatial density of measurements, temporal frequency, logistics, etc. Monitoring costs, consequently, could range from 10s to 100s thousands of dollars per year. BoPRC's condition/SOE monitoring for saltmarsh and mangrove habitat has an estimated cost of $50,000/year over three years. No data analysis has been undertaken to date. NRC (Richard Griffiths) comments, "*Depends on what you are going to monitor. Mapping their extent is relatively straightforward but time consuming (even using remote sensing methods still requires a lot of manual checking and QA*. *Northland has a big coastline and a lot of mangroves, so the cost* (*time*) *of mapping the extent depends on these factors* (*length of coastline + amount of mangroves*). *Ecological monitoring of mangrove habitat as a whole could be expensive (macroinvertebrates) or cheap* (*sediment quality, eDNA, birds*) *depending on what you did"*.

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

There are no formal examples acknowledged by the authors of iwi monitoring mangrove habitats as far as we are aware. Mangrove extent, and more so expansion, has long been part of formal and informal wānanga and discussions in hapū, iwi, public and academic forums (Maxwell 2018; LeHeron et al 2022), thus leading to specific modules being highlighted in estuary monitoring toolkits as co-developed by NIWA (see below).

Mangrove habitats can be challenging environments to work in, particularly in muddy substrates. In some situations, there are also health and safety considerations when working in soft mud. An estuarine monitoring toolkit, Ngā Waihotanga Iho, was developed by NIWA (Swales et al. 2011) to provide potential guidelines of how whānau, hapū and iwi members may want to utilise tools to measure environmental changes in their estuaries. One of the modules, includes estuarine plant and habitat modules that provide methods to monitor a range of parameters including change in habitat extent.

The rapidly growing interest in blue carbon and coastal wetland restoration and enhancement of biodiversity are being driven by hapū and iwi, and or driven by other organisations who acknowledge the role of Indigenous Peoples efforts in protecting whenua and moana. For instance, NGOs, including The Nature Conservancy and Conservation International are partnering with hapū and iwi Māori, including the internationally recognised Hinemoana Halo, which includes supporting coastal wetland and seagrass restoration (Conservation International, 2024). Within Aotearoa, DOC has also recommended best practices to align with hapū and iwi Māori towards Blue Carbon initiatives that include wetland/estuary environments in Aotearoa (Kettles et al 2024). This includes a recent study that maps the current extent of blue carbon habitats alongside the following iwi Ngāti Porou, Te Whānau a Apanui, Ngāti Wai, Te Rarawa, Ngāti Ruanui, Ngā Rauru, Ngāi Tahu and Ngāti Kuri (Bulmer et al. 2023).

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

As noted in Sections A1 and A2, there are associations between mangrove habitat extent and substrate muddiness and saltmarsh quality attributes. Mangrove habitat extent in many estuaries coincides with intertidal flat areas above MSL elevation. These upper intertidal, typically muddy, environments have been substantially increased in extent in many estuaries due to soil erosion and elevated sediment loads from catchments during the historical era. Mangrove habitat expansion is symptomatic of excessive fine sediment loading and sedimentation in NZ estuaries [see reviews (Horstman et al. 2018; Morrisey et al. 2010; Swales et al. 2020b)] in comparison to pre-deforestation/baseline conditions (Hicks et al. 2019). Saltmarsh extent and quality is likely to progressively decline due to mangrove habitat colonisation of saltmarsh habitat, facilitated by climate warming and SLR.

Part B—Current state and allocation options

B1. What is the current state of the attribute?

The current state of mangrove habitat extent is well understood in northern New Zealand's estuaries where mangroves occur, and is included in the national land cover map LCDB (Manaaki Whenua Landcare Research 2019); monitoring occurs somewhat regularly by regional councils. Although the LCDB suggests that there has been virtually no measurable change in mangrove habitat extent since the mid-1990s [latest estimate (2018) 28,172 ha (Manaaki Whenua Landcare Research 2019)], this is unlikely to be accurate for the reasons described in Section A3.

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

Natural reference states have not been described for New Zealand mangrove habitat extent and quality, as far as we are aware. Historical records and sediment core studies indicate that mangroves substantially increased their distribution in northern estuaries as a direct result of increased fine sediment loads and rapid accretion of intertidal flat habitat suitable for colonisation. This process was triggered by large-scale catchment deforestation and increased soil erosion (Hicks et al. 2019; Morrisey et al. 2010). In many estuaries, this disturbance was manifest by order of magnitude increases in sedimentation rates, shift from sand- to mud-dominated systems and reduction in the areal extent of subtidal habitats (Morrisey et al. 2010; Swales et al. 2015; Swales et al. 2002; Thrush et al. 2004).

A natural reference state for NZ mangrove habitat would therefore include some or all of the following characteristics:

- Sand or muddy-sand substrate with a generally low terrigenous mud input during the historical era.
- Low sediment accumulation rates (SAR) over decadal time scales. SAR approach pre-deforestation values (e.g., < 1.2 mm yr^{-1}; Swales et al. 2020a).
- Low concentrations of major stormwater contaminants (e.g., heavy metals; Swales et al. 2002), organic compounds associated with fossil fuel), and nutrients (Schwarz 2002; Thomson et al. 2024).

- Low concentrations of suspended fine sediment in the water column to enable penetration of photosynthetically available radiation (PAR), sufficient to support subtidal seagrass habitat.
- Coastal wetland sequence, with increasing elevation, consisting of (all/some of): subtidal and intertidal seagrass habitat, mangrove habitat, saltmarsh habitat and supratidal habitat (e.g., saltmarsh ribbonwood, manuka, kahikatea).
- Presence of diverse benthic fauna in mangrove dominated by mud burrowers e.g., species of crabs and shrimp that have some sensitivity to mud deposition (Ellis et al. 2004; Ellis et al. 2015).
- Mangrove habitat and creeks utilised by fish species that are sensitive to fine sediment. For example, the foraging success and health of juvenile snapper are adversely affected by elevated levels of suspended fine sediment (Lowe et al. 2015; Swales et al. 2016).

Estuaries, or parts thereof, that may approach the reference state include the Rangaunu Harbour and Wairakau Creek, Whangaroa Harbour (Northland), southern Whangateau Harbour (Auckland), and the Purangi River (Coromandel). These systems share some/all biophysical characteristics – relatively small catchment sediment loading during the historical era that reflective catchment attributes (size, steepness, erodibility, climate, indigenous forest landcover etc) and/or estuary biophysical characteristics that favour fine-sediment export to the sea. The Rangaunu Harbour, for example, has a relatively small catchment, some 80% of which is composed of lowland sand dune country. The saltmarsh, mangrove and subtidal seagrass habitats fringing the south-west shoreline of the harbour are also largely isolated from the Awanui River's fine-sediment load, with river plumes transported along the harbour's northern shore to the sea.

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

There are no known numeric bands for New Zealand mangroves. Spatial metrics to assess condition via satellite remote sensing have been developed globally to inform mangrove forest patch characteristics (Hai et al. 2022) with respect to patchiness and fragmentation, and could be applied in New Zealand. Other more comprehensive indices of mangrove forest quality have also been 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.

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

There are no known thresholds or tipping points for mangrove forests; however, key environmental characteristics such as hydroperiod (inundation time), the potential effects of sea level rise, and barriers to shoreward expansion are key drivers of future distributions of mangrove forests.

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

There are several important temporal lags/legacy effects that have influenced mangrove habitat extent in estuaries:

- Sediment delivery from catchments.
- Vertical accretion of intertidal flats to elevation (i.e., mean sea level) within the hydroperiod tolerance for *Avicenna*.
- Mangrove seedling recruitment - windows of opportunity.

The historical pattern and pace of estuary infilling and creation and expansion of intertidal habitat potentially suitable for mangrove colonisation and forest development has varied depending on catchment and estuary characteristics. A non-linear relationship between catchment sediment loads, tidal prism volume and the area of intertidal habitat above MSL suitable for mangrove colonisation has been determined for a range of estuary types in the Auckland Region (Section B5). The model shows that drowned river valley estuaries typically have relatively larger areas on intertidal flats suitable for mangrove than do estuarine embayments (Swales et al. 2020b).

<u>Sediment delivery from catchments:</u> New Zealand studies show that annual sediment loads are dominated by fine sediment (mud [≤ 62.5 microns] and sand [62.5–2000 microns]) transported during storm events (e.g., Basher & Dymond 2013; Hicks et al. 2000; Hughes et al. 2012) that dominate sedimentation in marine receiving environments. Most catchments in northern New Zealand, where mangroves occur, are relatively small (i.e., < 1000 km^2), so time lags in sediment delivery to estuaries and coastal marine systems will mainly depend on sediment transport characteristics. Time lags for sediment delivery from the catchments to estuaries varies markedly between sand and mud. Mud is readily maintained in suspension in rivers due to relatively low settling velocities (i.e., 0.1–2 mm s^{-1}; Lamb et al. 2020) so that a large fraction of the mud will be delivered to estuaries during floods (i.e., hours to days), unless retained in a catchment sediment sink during over-bank flow conditions (e.g., flood plain, vegetated areas). Fine sand has much higher settling velocities (up to ~30 cm s^{-1}) so that transport rates are typically much lower than for mud. Thus, the bulk of the mud load is likely to be transported along the entire length of a river channel network to estuaries during the course of a flood event.

<u>Vertical accretion of intertidal flats:</u> Mangroves may colonise intertidal flats once they become ecologically suitable, by vertically accreting above MSL elevation, so that hydroperiod is within the physiological tolerance for *Avicenna* spp. The rate at which this vertical accretion and formation of intertidal flats occurs is dictated by the catchment sediment load, spatial dimensions of the receiving estuary (i.e., fetch and depth), sub-environment and hydrodynamic properties that dictate sediment transport. Historical sediment accumulation rates (SAR) measured on tidal flats in ~30 estuaries (mainly upper North Island) have averaged 3.2 mm yr^{-1} (range: 2–5.2 mm yr^{-1}; summarised in Huirama et al. 2021), equivalent to ~ 30 cm of accretion over 100 years. SAR are typically much higher near catchment outlets, tidal creeks and where sediment loading has been excessive. In tidal creeks, SAR of 10 mm yr^{-1} are not uncommon, and these environments hold some of the oldest mangrove forests. In the Firth of Thames, the intertidal flats accreted rapidly (SAR ~ 20 mm yr^{-1}) so that large areas became suitable for colonisation by the early 1960s. This process has been sustained by legacy sediment that has accumulated in the southern Firth that has accumulated in the southern Firth over the last ~170 years or so. In summary, the time lag for intertidal flat formation to the MSL threshold will vary widely, but typically over decades to centuries.

Mangrove seedling recruitment: events resulting in large increases in habitat extent (i.e., tens of metre) are infrequent, largely due to physical controls on the success of propagule anchoring and seedling establishment. This recruitment process is determined by the local wind-wave climate in estuaries and timing of propagule fall during the spring–neap tidal cycle. Large recruitment events depend on propagule fall coinciding with an extended period of calm weather (i.e., 1–2 weeks) and a declining spring to neap high tide level, when bed disturbance by wave action is minimal (Balke et al. 2015; Gijsman et al. 2024). Such is the case in the southern Firth of Thames, where large-scale seedling recruitment has not occurred since the mid-1990s (Swales et al. 2015). In tidal creeks and small estuaries with limited wave fetch (e.g., 100s m – km), colonisation will largely depend on tidal flat accretion and propagule production and dispersal.

Strong physical controls on seedling recruitment mean that mangrove habitat extent typically does not match the area of intertidal flat that is potentially suitable for colonisation. This is demonstrated in the east-coast estuaries of the Auckland Region, where mangroves occupied only 58% (i.e., 27 km^2) of the total intertidal flat area above MSL elevation (Swales et al. 2009). The proportion of suitable intertidal flat occupied varied from 22 to 75% and the lower elevation of mangrove trees and seedlings averaged 0.35 m above MSL. The Waitemata Harbour accounted for 32% of the 2009 mangrove habitat, and in the central harbour area had not substantially increased since the 1950s. The largest percentage increases in mangrove habitat extent occurred in the smallest estuaries with high-tide areas less than 1.5 km^2 (i.e., Okura, Orewa, Waiwera).

These considerations suggest that mangrove habitat extent in most estuaries is unlikely to ever attain its potential extent (i.e., current intertidal flat > MSL) due to physical constraints on seedling recruitment. This effect will be exacerbated in the future by sea level rise, as mangrove (and saltmarsh) habitat will progressively migrate landward where submerging low-lying land is available or will be reduced in extent by coastal narrowing and coastal squeeze in the extreme upper intertidal zone. In this later scenario, mangrove will replace saltmarsh, and mangrove habitat would only persist where sediment supplied by rivers was sufficient for these forest remnants to keep pace with SLR.

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

Mātauranga Māori with respect to mangrove forests is context-dependent, with ecosystem-dependent species associated with mangrove forests varying between locales. Examples include use of mangroves as wood, as dye materials, as habitat for kaimoana including oysters (including non-native oyster species), and for mangrove tidal creeks as fishing locales for tuna. Therefore, understanding bands would be best done alongside whānau, hapū and iwi.

Part C—Management levers and context

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

Mangrove habitat expansion in New Zealand's northern estuaries from the mid-1800s onwards has largely occurred due to accelerated estuary infilling. This has occurred due to elevated sediment loading associated with large-scale catchment deforestation and conversion to pastoral agriculture and more recent land-use intensification. In this sense, the general pattern of mangrove habitat expansion in many estuaries is a symptom of excessive soil erosion and environmental degradation of New Zealand's northern estuaries during the historical era [e.g., reviews (Morrisey et al. 2010; Swales et al. 2020b)]. The proportion of the suitable intertidal habitat is substantially larger than present mangrove habitat extent (McBride et al. 2016). Models have been developed to understand the interplay of factors controlling seedling recruitment on wave-exposed intertidal flats (Balke et al. 2015; Gijsman et al. 2024), although this has not been mapped at estuary scale.

Looking forward, sea level rise will likely drive changes in mangrove habitat extent, with landward/upslope migration to maintain their position above MSL. The relatively wide range of elevation (i.e., MSL to Highest Astronomical Tide [HAT]) that mangrove occupies, relatively large tidal ranges, as well as generally sediment-rich estuarine systems means that New Zealand mangrove forests are unlikely to be lost to inundation during this century (Lovelock et al. 2015a). It should be noted that mangrove habitat is already expanding into saltmarsh habitat and will highly likely displace saltmarsh without management interventions. For both mangrove and saltmarsh, interventions that remove built physical barriers to migration, as lowland areas are inundated, will mitigate the likelihood of habitat loss and generate opportunities for restoration of freshwater–estuarine wetland ecosystems. The pattern and pace of potential changes in coastal wetland habitats under SLR and exploring management interventions are major research objectives of the Future Coasts Aotearoa programme.

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

C2-(i). Local government driven

C2-(ii). Central government driven

C2-(iii). Iwi/hapū driven

C2-(iv). NGO, community driven

C2-(v). Internationally driven

Mangrove forests continue to decline across most of the world, particularly in tropical habitats. Aotearoa is an exception, where mangrove trends are of expanding temperate mangroves (Morrisey et al. 2010).

National and international examples of interventions are provided by hapū and iwi. Recent interest in mangrove forest restoration as a climate adaptation/blue carbon mitigation strategy, in partnership with hapū and iwi, includes more recent mapping of existing and potential mangrove habitats (Bulmer et al. 2023; Bulmer et al. 2024). International support of Indigenous partnership by NGOs in blue carbon climate mitigation potential (Conservation International 2024), and funding of mapping/monitoring exercises, with focus on carbon sequestration values of mangrove forests (see Stewart-Sinclair et al. 2024).

The rapidly growing interest in blue carbon and coastal wetland restoration and enhancement of biodiversity are being driven by hapū and iwi, and or driven by other organisations who acknowledge the role of Indigenous Peoples efforts in protecting whenua and moana. For instance, NGOs, including The Nature Conservancy and Conservation International are partnering with hapū and iwi Māori, including the internationally recognised Hinemoana Halo, which includes supporting coastal wetland and seagrass restoration (Conservation International, 2024). Within Aotearoa, DOC has also recommended best practices to align with hapū and iwi Māori towards Blue Carbon initiatives that include wetland/estuary environments in Aotearoa (Kettles et al 2024). This includes a recent study that maps the current extent of blue carbon habitats alongside the following iwi Ngāti Porou, Te Whānau a Apanui, Ngāti Wai, Te Rarawa, Ngāti Ruanui, Ngā Rauru, Ngāi Tahu and Ngāti Kuri (Bulmer et al. 2023).

Within the four northern regional authorities where mangrove occur (i.e., Northland, Auckland, Waikato and Bay of Plenty), specific sections of regional coastal plans are dedicated to mangrove management. These sections include rules and guidelines to inform consent activities with respect to mangrove removals. Mangrove removals occur regularly to support infrastructure (i.e., access, transit, power lines etc.), whereas more restrictive mangrove removal guidance has been implemented for removals related to other values such as recreation, accessibility and viewscape. Mangrove removal applications have declined significantly following large consent processes (e.g., Tauranga, Whangamata, Tairua) in the 2010s, and review of limited recovery of most mangrove removal locations (though see Whangamata for exceptions within an adaptive management framework) (Bulmer et al. 2017; Lundquist et al. 2012; Lundquist et al. 2014). Central government interventions with respect to mangroves have been primarily in assessing expansion in Ramsar sites (Miranda, Firth of Thames) where expansion was perceived as a threat to shorebird habitat.

Part D—Impact analysis

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

Left unmanaged, it is anticipated that mangrove expansion rates within northern New Zealand may continue to increase in areas with ongoing sediment supply and estuarine infilling. However, for many other areas, particularly in narrow tidal creeks, the intertidal area that is suitable for mangrove colonisation based on inundation frequency is already colonised, and little additional expansion is likely to occur. As noted earlier, mangrove expansion is correlated with higher levels of land-based sediment supply, which are typically correlated with degraded estuaries. Other impacts are more likely to be related to mental health and human wellbeing, as mangrove expansion is associated with reduced coastal access and recreational access due to muddier sediments and vegetation blocking access to sandflats and to boat launch sites.

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

Economic impacts of mangroves are anticipated to occur with the potential for voluntary (or government supported) credit schemes for mangrove restoration to support climate mitigation. This opportunity is currently being explored, with key barriers of land tenure and regulatory context requiring further consideration (Stewart-Sinclair et al. 2024).

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

Sea level rise (SLR) due to climate warming will have a major impact on mangrove habitat extent. Around the fringes of some estuaries, natural topographic features will prevent appreciable landward migration of mangrove (and saltmarsh) habitat. This scenario is referred to as coastal narrowing (Swales et al. 2020a). The shorelines of many NZ estuaries in lowland catchments dominated by agriculture and urban centres and their hinterland have commonly been modified by built infrastructure, including stop banks, roads, railway lines, reclamations for port, urban and industrial development. These built barriers, unless breached or removed will prevent natural adaption to SLR through habitat migration, which is referred to as coastal squeeze (Swales et al. 2020a). Under this condition, mangrove forests will require a sufficient supply of sediment to vertically accrete to keep pace with SLR. The southern Firth of Thames is a notable example of this process, with the mangrove forest tidal platform presently being some two metres higher than farmland inside the stopbank (Swales et al. 2015) due to the high availability of fine sediment supplied by rivers during the historical period (i.e., legacy sediment) and today.

The extent to which SLR will result in changes in mangrove habitat extent will also depend on the local geological and geomorphic setting. Key processes include the rate of vertical land motion (VLM, subsidence or uplift) and sediment supply rate. Relative, or local, SLR (i.e., RSLR) incorporates VLM (i.e., with the regional SLR trend). Again, the southern Firth of Thames provides pertinent example, where subsidence of a deep sedimentary basin is occurring at ~ 8 mm yr^{-1}, resulting in RSLR of ~10 mm yr^{-1} in the mangrove forest (Swales et al. 2016), some five-fold higher than the regional rate of SLR. Despite this extreme RSLR, the mangrove forest today occupies ~11 km^2 of upper intertidal flat and accumulated a ~ 2 m thick deposit of fine sediment. The forest continues to expand seaward today. This rapid habitat expansion and persistence of the mangrove forest has been possible due to the supply of contemporary and legacy sediment delivered to the Firth by rivers that maintain the intertidal flat elevation above MSL (Hicks et al. 2019; Swales et al. 2015). This relationship between catchment sediment loads, tidal prism volume and the area of intertidal habitat above MSL suitable for mangrove colonisation has been determined for a range of estuary types in the Auckland Region. A simple exponential model ($r^2 = 0.69$, $P < 0.001$) also shows that drowned river valley estuaries typical have relatively larger areas suitable for mangrove habitat than so estuarine embayments (Swales et al. 2020a).

As observed elsewhere in the Indo-Pacific (Lovelock et al. 2015b), mangrove habitat will likely persist in many of our northern estuaries beyond 2100, where landward migration is possible and/or sediment supply from catchments is sufficient to keep pace with RSLR. Rod Surface Elevation Tables (RSET) (Cahoon et al. 2002)) are used globally as a primary tool to monitor the elevation trajectories of mangrove forests and saltmarshes relative to rising sea levels, e.g., (Lovelock et al. 2015b; Webb et al. 2013). In New Zealand, RSET have been used by NIWA and Waikato Regional Council for research and monitoring mangrove forest elevation trajectories in the southern Firth of Thames since 2007 (Swales et al. 2016; Swales et al. 2019). The network is now being expanded in the MBIE-funded *Future Coasts Aotearoa* Endeavor Programme, in partnership with Regional Councils and DOC, so far including sites in the Bay of Plenty, Auckland and Canterbury Regions.

The natural range of NZ mangrove habitat has historically been controlled by number of physiological and biogeographic factors. These include low winter air temperatures, frost frequency, biogeography and oceanography that limit mangrove propagule dispersal [e.g., (de Lange & de Lange 1994)] and the limited availability and relative remoteness of suitable estuarine environments south of their present range. Although climate warming may change their potential local extent (e.g., landward migration facilitated by sea level rise), biogeographic limitations to their southern expansion remain.

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9.4 Shellfish bed extent and quality

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Preamble: When referring to "shellfish" in this information stocktake, we are referring to bivalve mollusc shellfish only. Although there are hundreds of species of bivalve molluscs in New Zealand, the term "shellfish beds" is generally applied to 8-10 species only. Bed-forming shellfish are generally large, common, and well-known species including: **cockles, dog cockles, pipi, wedge shells, oysters, green-lipped mussels, horse mussels, and scallops**. Most of the bed-formers listed above are recognised as kaimoana or as ecologically important 'key' species. Although some bivalves such as mussels and rock oysters occur on hard substrates, the term "shellfish bed" usually refers to bivalve-dominated soft-sediment habitats. 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). Green-lipped mussels and oysters are farmed in many estuarine and coastal areas throughout New Zealand, but we are not including cultured bivalves in our information stocktake of "Shellfish bed extent and quality".

State of knowledge of the "Shellfish bed extent and quality" attribute: Medium / unresolved – some studies/data but conclusions do not agree

Shellfish biology is generally well understood. Shellfish populations are monitored in many parts of the country to keep track of stocks. However, shellfish bed extent is not often monitored, especially for infaunal and subtidal species that are difficult to directly observe. "Quality" is also not usually assessed, though it may be possible to assess quality using a combination of abundance, size structure, and other metrics (e.g., body tissue contaminant concentrations). Therefore, we generally have a medium / unresolved state of knowledge of this attribute.

Part A—Attribute and method

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

Shellfish are a key indicator of ecological integrity in intertidal and shallow subtidal coastal and estuarine systems [1-5]. 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.

Almost all the shellfish species mentioned in the pre-amble are sensitive to sediment eroded from land (suspended and deposited sediments) and shellfish population collapses throughout New Zealand have been attributed to multiple stressors such as direct overharvesting, indirect damage/disturbance from trawling and dredging, mass mortalities in heat waves, smothering under eutrophication-related nuisance macroalgae outbreaks, ocean acidification effects on larval/juvenile life-stages, and more. Different shellfish perform different ecological roles, therefore, having a diversity of shellfish bed 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.

Shellfish are critical to ecological integrity because of the key ecological roles they perform and the ecosystem functions/services they deliver [1,2,6-8]. Bed forming shellfish stabilise and armour seafloor sediments. Bivalve shell hash (dead/broken shell material) creates habitat heterogeneity in soft-sediment habitats and has a positive influence on soft-sediment macroinvertebrate community diversity [9]. Banks comprised of dead bivalve shells are utilised by rare and threatened shorebirds (e.g., as high tide roosts), and living shellfish are eaten by birds and fish (e.g., oystercatchers, eagle rays). Some of the bed forming shellfish are relatively mobile (e.g., cockles) and bioturbate surface sediments, influencing primary production and nutrient release rates [10,11]. Others (e.g., wedge shells) create porewater pressure gradients that influence fluxes of solutes across the sediment water interface. Green-lipped mussels, horse mussels, and oysters create hard structure and vertical relief above the sediment-water interface in soft-sediment seafloor habitats, creating biodiversity hotspots. Organisms settle on their shells (sessile invertebrates) or take refuge in the shell clusters (mobile invertebrates and fish). Most bed forming shellfish are filter-feeders and have the potential to cleanse/clarify turbid water [12,13]. Some shellfish are used as time-integrative biomonitors (e.g., to track environmental contaminants such as metals) because of the large volumes of water they filter over periods of weeks to months. Biodeposition of faeces and pseudo-faeces by horse mussels organically enriches surrounding sediments, affecting macrofaunal communities and microbial remineralisation rates [2,14]. All bivalve shellfish have calcium carbonate shells, though their potential role in blue carbon sequestration is generally thought to be related to the trapping and burial of organic carbon rich particles within beds. Some studies suggest that shellfish beds positively influence denitrification, a microbially mediated process that converts nitrate to di-nitrogen gas in a series of dissimilatory steps [15]. Inorganic N removal is an important ecosystem service in N-enriched (eutrophic) systems [16,17]

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

One of the most recognisable indicators of estuarine degradation has been the collapse of natural shellfish populations. The scale of shellfish bed declines is massive and nationwide. We have lost an estimated 500 km^2 of green-lipped mussel beds (*Perna canaliculus*) in the Hauraki Gulf and 100 km^2 from the Marlborough Sounds [18-22]. High density horse mussel beds (*Atrina zelandica*) have almost completely disappeared, with just relict beds remaining [1-4]. Lucrative scallop fisheries have crashed nationwide, and populations have not rebounded despite harvesting bans including both rāhui and national-scale MPI fisheries closures. Pipi beds (*Paphies australis*) at the mouth of Whangārei Harbour covered 0.5% of the area in 2017 that they covered in 2005 [23,24], 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 [25,26]. Shellfish 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.

The inability to find, collect, and safely consume shellfish is devastating to mana whenua, whose identity and wellbeing has relied upon connections to shellfish and their wider ecosystems 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 (e.g., mussel and oyster aquaculture; scallop fisheries).

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

Shellfish populations have declined rapidly in some areas, with “collapse” (rather than “steady decline”) often used to describe the pace of change. The extremely large green-lipped mussel populations that covered 500 km^2 of seafloor habitat in the Hauraki Gulf were decimated in just 50 years coincident with a bottom-contact dredge fishery (1910-1960). Ten tonnes of pipi disappeared from Mair/Marsden bank in Whangarei Harbour in ~10 years (2005-2017). Horse mussels in Mahurangi Harbour declined from densities of 10-20 m^{-2} to <0.5 m^{-2} in ~ 10 years (1998-2009) and continue to be scarce where they once occurred in dense beds (e.g., Pakiri, eastern Coromandel, Tauranga Harbour, Marlborough Sounds).

Survey data indicate serial depletions of scallop populations in the Marlborough Sounds as fishers move from overharvested beds to new beds. There is some indication that seafloor habitat quality, rather than the supply of larvae to those habitats, is the factor most responsible for the lack of scallop recovery [27]. Habitat quality has been impacted by bottom-contact fishing and terrigenous sediment inputs, which have resulted in muddy seafloor sediments with insufficient biogenic structure [27].

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 [28], potentially limiting recovery prospects further. However, it is hypothesised that in-estuary interventions, combined with catchment management, can create places and times in which stressors are sufficiently reduced and aligned with biological requirements to enable shellfish recovery. Restoration that involves moving adult shellfish from one area to another is a zero-sum gain and poses biosecurity risks. Therefore, advances in our ability to consistently/successfully produce new spat for restoration is critical, though significantly technically challenging [29].

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

Some shellfish populations are monitored (using various standard survey techniques suited to the species of interest). However, the attribute “Bed extent and quality” is rarely quantified.

Infaunal bivalves including cockles, wedge shells, and pipi are monitored at sentinel monitoring sites by many councils using standard sized cores [30-32]. 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, >40 mm). An Estuarine Toolkit published by NIWA (in English and te reo Maori) provides guidance on standard shellfish monitoring methods for intertidal shellfish (cockle, wedge shells, juvenile pipi; [33]. Most councils have started reporting estuarine monitoring data on the Land, Air, Water Aotearoa (LAWA) website [34]. 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 [35-37]. Semi-quantitative 'rapid habitat assessment' techniques developed by NIWA have been used by some councils to define the spatial extent of 'high density cockle' and 'high density pipi' beds [38,39]. The 'rapid habitat assessment' method is semi-quantitative because broad areas are walked with regular spot checking to assign habitats to pre-defined categories (i.e., "High Density Pipi habitat" = areas with >10 pipi sized >40 ml shell length in a 15 x 15 cm square quadrat). Some iwi groups have mapped cockle, pipi and green-lipped mussel beds using quantitative (usually quadrat-based) techniques [40].

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-funded 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 [41-43]. Because of the destructiveness of the technique, methods are being developed to transition towards underwater towed camera surveys [44]. Transitioning to camera-based surveys may also increase the availability of useful ancillary information on the appearance/condition of the habitat.

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

Practical/logistical barriers to conducting surveys of shellfish bed extent and quality are species-specific. Intertidal sandflats where cockles, wedge shells and pipi are common are generally highly accessible and present very few practical/logistical barriers. Measuring shellfish bed extent and quality in subtidal areas (e.g., for green-lipped mussels, horse mussels, dog cockles, scallops) is much more difficult; such surveys may require access to Worksafe accreditations for boating and diving, access to expensive dive and camera gear, and the securing of permits to sample. Suspension-feeding shellfish often occur in areas of high current flow, which can pose risks to divers and affect the positioning/speed of towed cameras (affecting the quality of the footage). Diving and camera work in areas with low water clarity reduces the scales of observation, and observing large areas of seabed with either type of technique is generally difficult in coastal/estuarine areas.

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

Costs are extremely variable and depend on type of shellfish bed being assessed. For example, almost anyone can sample intertidal cockle and pipi populations using a garden sieve, a quadrat/core made from PVC plastic, a ruler, and a cell phone. Surveys of scallops funded by MPI (involving divers, cameras, and large vessels), in contrast, can cost hundreds of thousands of dollars.

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

There are likely many examples of iwi and hapū representatives monitoring (formally or informally) shellfish bed extent and quality using a range of mātauranga Māori based and western science based methods. Cultural practices surrounding the collection of shellfish have been handed down through generations and declines in shellfish bed extent and quality are well known and deeply impact hapū and iwi throughout Aotearoa (e.g., [67]).

Cockles, pipi, and mussels are monitored by local kaitiaki throughout Aotearoa. 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), the monitoring of cockles by Ngāti Whakehemo in intertidal soft-sediment habitats of Waihī Estuary, and the monitoring of subtidal mussel populations and beds, as well as other species of shellfish by Ngāti Awa and the Te Ūpokorehe Resource Management Team in Ōhiwa harbour. Similarly, many hapū and iwi led have co-led and/or driven the assessment of shellfish and associated ecosystems, including Te Papatipu Rūnanga o Koukourarata, Te Papatipu Rūnaka o Ōraka-Aparima, Te Papatipu Rūnaka o Kāti Huirapa ki Puketeraki, and many more. In addition to this, there are many hapū and iwi led shellfish species and ecosystem assessments, e.g., including co-development of appropriate indicators of estuarine mahinga kai [68-70].

Standard methods that local kaitiaki and mana whenua can use to monitor shellfish are described in Ngā Waihotanga Iho (Estuary Monitoring ToolKit; [33]). The degree of use and uptake of Ngā Waihotanga Iho by iwi and hapū, and the degree of method standardisation across New Zealand, is unclear.

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

Shellfish bed extent and quality is likely to be inversely correlated with “Mud Extent” and “Suspended sediment/water clarity/turbidity” (as most bed-forming shellfish are intolerant of high suspended sediment concentrations, high rates of sediment deposition, and high bed sediment mud content). Bed-forming shellfish are likely to be positively correlated with “Phytoplankton/Chlorophyll *a*” as this is a food source for sessile benthic bivalves. For both Suspended Sediment and Phytoplankton, intermediate concentrations are likely most favourable for shellfish (water that is too clear does not have enough food, but water that is too turbid or eutrophic is harmful). Shellfish Bed Extent and Quality could potentially be measured in intertidal estuarine habitats at the same time as “Mud Extent”, “Seagrass Extent” and “Seagrass Quality” (e.g., using the Rapid Habitat Assessment techniques developed by NIWA), though it is not advisable to group these attributes given that they indicate different elements of ecological integrity. The “Macroinvertebrate Community Composition” attribute may provide information on the densities and sizes of some bed-forming shellfish species (e.g., cockles, wedge shells, pipi) but it will not inform or necessarily correlate with the Shellfish Bed Extent and Quality attribute.

Part B—Current state and allocation options

B1. What is the current state of the attribute?

The current state of Shellfish bed extent and quality—poor due to impacts from multiple stressors—is relatively well understood at the National scale. However, most of the monitoring of shellfish is for population density, rather than bed extent or quality, and our understanding is considerably better for some species (e.g., cockles) than it is for others (horse mussels, dog cockles). Much of the change in shellfish bed extent and quality may have occurred before coastal and estuarine benthic habitats were sampled effectively and broadly using modern survey techniques, so it is difficult to know exactly what has been lost and when it happened (although mana whenua recollections may fill gaps). Rates of terrigenous sediment input to estuarine and coastal areas increased 10-100 fold following the arrival of Europeans in New Zealand [45], and widespread trawl fisheries were established as early as 1910. Despite all this, increases in the extent and quality of bed-forming shellfish from today's generally poor state could be used as a indicator of improving ecological integrity. Targets could be set for shellfish bed extent and quality, and restoration efforts could be aimed at improving shellfish recovery prospects.

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

Natural reference states for Shellfish bed extent and quality are not known. Moreover, reference states would be species specific. 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.

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

To my knowledge, numeric or narrative bands do not exist for the attribute Shellfish bed extent and quality.

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

Relatively sudden collapses of several bed-forming shellfish species in New Zealand suggests the existence of tipping points and thresholds. However, disentangling the underlying causes of bed-forming shellfish collapse is difficult. A recent review of cumulative effects of stressors on scallops and scallop habitats in the Marlborough Sounds suggested a clear negative impact of specific human activities both on land (land clearance and forestry) and in adjacent coastal zones (bottom contact fishing). However, the spatial and temporal resolution of the available data on stressors, and on the specific times and places of scallop population declines, precluded tight linkage of cause and effect [27].

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

Shellfish bed extent and quality within estuaries has likely contracted as terrigenous sediment loading and Mud Extent in estuaries have expanded. It has likely taken many decades for Mud Extent to build to its current levels, and reductions will also likely take many decades. Lags in recovery of bed-forming shellfish are likely to be longer than for Mud Extent due to Allee effects; most of the bed-forming shellfish are broadcast spawners whose reproductive success (fertilisation probability) depends on them being present in high density beds. Moreover, some bed-forming shellfish species are long-lived and may take years to reach full size and maximum fecundity. Therefore, legacies of past degradation and lags in recovery need to be considered by mana whenua kaitiaki and other resource managers.

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

To our knowledge, there are no bands for shellfish bed extent and quality in existence in New Zealand. Thus tikanga Māori and mātauranga Māori have not been utilised to develop bands, targets, or allocation options. Tikanga Māori and mātauranga Māori should be included, in collaboration with whānau, hapū and iwi, in decision making and band/threshold definition.

Part C—Management levers and context

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

There is evidence that the natural extent of shellfish beds in New Zealand, and the size of shellfish within those beds, was much greater prior to the arrival of Europeans than it is today [19,20]. Shellfish beds once covered hundreds of square kilometres of seafloor but have dramatically declined or disappeared due to overharvesting, habitat destruction, terrigenous sediment loading, and other stressors. Declines in Shellfish Bed Extent and Quality are not unique to New Zealand. However, declines in this attribute have likely occurred more recently in New Zealand relative to elsewhere. Although the drivers of declines in Shellfish Bed Extent and Quality are generally understood, specific causal relationships with stressors are not well quantified. The options available to managers to reverse shellfish declines are unclear because of multiple stressor interactions and biological and physical factors that promote hysteresis (e.g., density-dependent spawning and Allee effects; legacies of past sediment loading and biogenic habitat removal that may take years to improve). Nevertheless, there are large-scale oyster restoration projects being undertaken overseas (e.g., Chesapeake Bay) and there are promising new examples of green-lipped mussel restoration success from New Zealand, suggesting that shellfish bed recovery may be possible for some species with a combination of stressor reduction and active mitigation.

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

C2-(i). Local government driven

Many councils and local/regional authorities have taken steps to control sediment and nutrient inputs, which should result in the eventual improvement of shellfish bed extent and quality. However, current sediment and nutrients inputs to coastal ecosystems are likely still high and driven by external events (such as Cyclone Gabrielle). Some of the Jobs for Nature initiatives (while Central government driven) are being implemented locally, but outcomes for shellfish bed extent and quality are not yet known. Some of the 'local' initiatives are being undertaken on relatively large scales (e.g., the $100m Kaipara Moana Remediation project; [46]).

C2-(ii). Central government driven

Fisheries NZ (MPI) is tasked with managing Quota Management Species, which includes some of the bed-forming shellfish (e.g., scallops, green-lipped mussels, cockles, pipi, horse mussels). Due to recent collapses of scallops, FNZ has instituted (almost) National scale scallop fisheries closures.

The Jobs for Nature programme (administered by five central government agencies) has directed hundreds of millions of dollars towards riparian planting in catchments to prevent sediments and nutrients from entering freshwater and coastal receiving environments downstream. Obviously, the Jobs for Nature funding was not targeted at improving the Shellfish bed extent and quality attribute, though the attribute may be useful at tracking the successes of individual catchment interventions (with the caveat that there will be temporal lags and legacy effects). Several central government agencies are commissioning work on the effects of catchment contaminants in estuarine/coastal ecosystems and/or have strategies for catchment contaminant load reductions, but specific actions are not likely being widely implemented yet.

C2-(iii). Iwi/hapū driven.

Rāhui on collections of bed-forming shellfish have been implemented in many areas of New Zealand. Rāhui is often misappropriated and confused with 'temporary closures' and their protection status contrast to each other; for instance, rāhui has no legal teeth, and are therefore followed voluntarily, while temporary closures are a 'two-year' fishery ban that can be applied once a Customary Management Area is established [71]. There are numerous examples of how CMA tools including temporary closures are not providing for the needs of whānau, hapū, iwi and their taonga/marine ecosystems. For example, CMAs have failed to deliver meaningful governance, timely responses, or localised and ecologically relevant solutions [72-75].

For scallops, at least three temporary closures have been set for the Hauraki Gulf. Ngāti Manuhiri set a two-year scallop closure. Coromandel residents declared a voluntary rāhui on scallop collecting on the eastern side of the peninsula. Patuharakeke have supported a series of two-year temporary closures for the collection of pipi from Mair/Marsden bank. Ngāti Awa have set a temporary closure for the collection of recently settled (restored) seabed mussels in Ōhiwa Harbour.

Iwi and hapū have been heavily involved in Jobs for Nature projects across New Zealand to address land and water quality, including downstream estuary health (though not necessarily shellfish bed extent and quality specifically). Iwi and hapū are also leading and contributing to shellfish restoration initiatives in partnership with Councils and National Science Challenge researchers from various universities, CRIs, and other research institutes/providers [47,48].

C2-(iv). NGO, community driven

Revive Our Gulf is a broad partnership designed to restore mussel reefs in the Hauraki Gulf [49] and there are similar shellfish restoration initiatives at the Top of the South Island.

C2-(v). Internationally driven

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

Part D—Impact analysis

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

Shellfish population collapses have impacted Māori identity and wellbeing (e.g., whakapapa, mātauranga, taonga) and cultural values and practices (e.g., kaitiakitanga, kaimoana harvest) [50,51]. Shellfish restoration is high priority for many hapū and iwi nationwide [51,52]. Māori also have substantial economic interests (e.g., fisheries, aquaculture, tourism) that will benefit from restored estuaries and shellfish populations. Shellfish restoration can clean estuarine waters of nitrogen pollution [53-55] and increase fish diversity and abundance through habitat provision [56,57]. Shellfish also provide ecosystem services such as food web support and carbon sequestration [58].

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

Food budgets, regional economies, and national exports have been impacted by the decline in shellfish populations resulting from estuarine degradation. The scallop fishery of Te Tauihu, worth over $70m [59] in the mid-1970s (approx. $700m today) is closed. Habitat restoration would make sustainable harvests of 10-20% of this level achievable, with a value in the order of $100m/year. Restoration of other degraded shellfish habitats and populations (e.g., the inner Hauraki Gulf/Firth of Thames, Bay of Islands) could support sustainable fisheries with a combined value of a further $100m/year. The mussel industry anticipates that restoring the supply of seed mussels in Kenepuru Sound alone is worth >$15m/year. It is estimated that a $350m investment in estuary repair could be returned in <5 years from improved commercial fisheries of fish, shellfish and crustaceans [109].

While the value of job creation is not presently quantifiable for Aotearoa-NZ, we note that shellfish restoration in Chesapeake Bay, Maryland, created 313 jobs while increasing fisheries output and nitrogen removal value by US$22.3m and US$3-18m, respectively [60]. Restoration would create a range of job opportunities in aquaculture, tourism and restoration-focused businesses.

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

There is evidence that heat waves (specifically, extreme air temperatures coincident with mid-day low tides) contributed to mass mortality in an intertidal cockle bed in Whangateau Estuary, Auckland [61,62]. Heat waves and thermal stress are likely to become more and more problematic in a warming world. Although there is little to no evidence that marine heatwaves and heat stress contributed to bed-forming shellfish population collapses, shellfish recovery may be affected by climate change if heat stress continues to increase. Coastal sea surface temperatures (SST) near the mouths of several North Island estuaries have increased over the last 20-30 years and SST was a significant driver of estuarine macroinvertebrate variables in some long-term time-series datasets [63]. SST around New Zealand in 2024 was the warmest on record, and the long-term trend of increasing SST is predicted to continue.

Some bed-forming shellfish inhabit intertidal flats and banks. Climate-related increases in sea level will eventually permanently inundate these areas, thereby reducing the suitability of the habitat for intertidal species [64]. Sea level rise also has the potential to alter current flow regimes and the positions of tidal channels, which could affect various bed-forming shellfish species (as most of suspension feeders that rely on high-current flows to bring them suspended particulate food material).

Finally, the shells of bivalve shellfish are made of calcium carbonate. Ocean acidification (due to increasing atmospheric CO_2 concentrations) negatively affects the calcification and growth of shellfish [65,66]. Acidification can also result from excess nutrient/organic matter loading, which depletes oxygen and elevates CO_2 production. Eutrophication-related acidification may be the greater risk to coastal shellfish in New Zealand, relative to climate-related acidification, though a slowly shifting baseline towards lower pH waters associated will not help.

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9.5 Kelp forest extent and quality

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State of Knowledge of the "Kelp forest extent and quality" attribute: Medium / unresolved – some studies/data but conclusions do not agree

Kelp forests are well studied in some regions with some consistent monitoring occurring, however, in other regions there are few observations or studies, and most regions lack consistent monitoring. For example, some marine reserves (e.g., Goat Island, Poor Knights, Fiordland) are regularly monitored, but few consistent monitoring programmes exist nationally. Furthermore, the focus of many monitoring studies has been the relative occurrence of *Ecklonia radiata* and urchin barrens (Wing et al 2022). However, satellite remote sensing has been successfully used to monitor giant kelp (*Macrocystis pyrifera*) across its geographic range in Aotearoa, New Zealand (Tait et al. 2021). These products, while covering large spatio-temporal scales, have limitations and caveats. In particular, they do not resolve populations unless they reach the sea-surface, and only provide broad metrics of total coverage, and do not provide estimates of density or biomass.

Kelp forests in Wellington region have been mapped testing whether machine learning and computer vision techniques approaches could be used to identify dominant macroalgal species. The trial of machine learning approaches produced very compelling results and indicated great future potential for monitoring subtidal kelp forests and ground truth remote sensing data (D'Archino et al 2021). *Macrocystis pyrifera* in Wellington harbour was mapped in 2017 and its distribution compared with historical data (Hay, 1990). Several areas of decline, absence or persistence were identified (D'Archino et al. 2019). The citizen science Project baseline Wellington (https://projectbaseline.org/wellington/) has been providing data (underwater videos, photos and mapping) of kelp forests at Kau Point (Wellington harbour) since 2016. These data provide a snapshot of *Macrocystis pyrifera* and *Carpophyllum* spp. bed fluctuations over the years and the large invertebrates associated with, including number of sea urchins.

<u>Part A—Attribute and method</u>

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

Kelp forests are a vital indicator of marine ecosystem health in temperate ecosystems (D'Archino & Piazzi 2021). Loss of kelp forests leads to a cascading shift in ecological assemblages which are often associated with a loss of key ecosystem functions and biodiversity (Filbee-Dexter & Scheibling 2014). Large macroalgae (e.g., kelp) are capable of altering the biophysical environment of rocky reef ecosystems through physical (e.g., wave dampening; Dubi & Tørum 1995) and physiological processes (e.g., oxygen production, carbon fixation; Pfister et al 2019). These processes support a rich variety of species, including many commercially important species such as paua, kina, koura and finfish. The health and integrity of temperate rocky reef ecosystems is directly linked to the presence and abundance of kelp forests.

Healthy functioning kelp forests provide numerous ecological services that can benefit human health, directly or indirectly. For example, kelp and seaweeds are photosynthetic, they produce oxygen and absorb CO_2. Furthermore, because they are at the base of the food web, they provide energy for other organisms and are themselves a notable source of food. Combined the provision of sustenance from kelp forests is potentially high, and in some cases important pharmaceutical products are extracted from seaweeds (Lomartire & Gonçalves 2022).

Additionally, the ability to absorb and integrate nutrients (particularly nitrogen) is another service that can improve water quality, thereby reducing risks to human health. Kelp and fucoid algae provide coastal protection from erosion, dampening the wave's energy (Morris et al 2020) and they mitigate ocean acidification by up taking CO2 (Hepburn et al. 2011).

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

There is a large body of literature that details the decline of kelp forests under a range of scenarios and environments. Loss of kelp forests is frequently associated with dramatic shifts in ecological states and a loss of functioning and integrity (Filbee-Dexter & Scheibling 2014).

Evidence suggests that up to 30% of fished reef habitats in northeastern New Zealand are likely degraded and dominated by Urchin barrens (Kerr et al. 2024). Although Tait et al. (2021a) identified declines in giant kelp (*Macrocystis pyrifera*) in southern New Zealand following marine heat-waves, many beds recovered 6-12 months after the event. Localised loss of other kelp forests (e.g., Durvillaea spp) following marine heat waves appears to be more catastrophic (Thomsen et al. 2019; 2021).

Population of giant kelp (*Macrocystis pyrifera*) at their northern limit (Wellington region) are particularly at risk for increased temperature.

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

At the scale of individual reef systems, the pace of change has been rapid. For example, urchin barrens can form quickly (i.e., months), and the impacts of marine heat-wave events can be almost immediate (Thomsen et al. 2019; Tait et al. 2021a). These events have decreased dramatically in the past 10 years (Thoral et al 2022). Urchin barrens can be reversed through active harvesting or culling of urchins (Miller et al. 2024), however, the impacts of marine heat-waves may be harder to reverse, and active restoration of kelp can be challenging (Thomsen et al. 2019).

Ocean warming and increase in frequency and duration of marine heat waves will negatively affect cold water species e.g., *Durvillaea poha*, *D. willana*, *Macrocystis pyrifera*, particularly populations at their northern limit, with consequent loss of associated species. Increasing temperature might not affect directly population of *Ecklonia radiata* in the Northern New Zealand but could promote an expansion in the distribution of the subtropical sea urchin *Centrostephanus rodgersii* leading to urchin barrens (Cornwall et al.2023, Shear et al. 2024).

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

There is no standardisation of monitoring methodology or responsible agencies. In most cases monitoring is completed for different reasons and with different tools or methods. However, ultimately each monitoring method can be converted to one of two metrics that can be compared and standardised at a national level. These metrics are density (e.g., number of plants per m^2) or cover (percentage cover per m^2). Ensuring consistent collection of metadata (e.g., depth, location, fetch, exposure) will be vital to nation-wide comparisons and standardisation for temporal comparisons.

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

While there are typically few access barriers to implementing monitoring, there are a range of practical limitations that have historically affected the application of monitoring. For example, marine habitats with high levels of wave and wind exposure are difficult to safely monitor and access. Furthermore, some regions of Aotearoa are remote and difficult to access from the sea. Such areas have typically been accessed by land or not at all. Increasingly, aerial imagery from satellites (Tait et al. 2021a) and drones (Tait et al. 2019; Tait et al. 2021b) can be used to monitor kelp and seaweed populations that occur in exposed and hard to reach areas.

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

Costs for monitoring kelp ecosystems are highly context specific, as kelp forests come in a variety of types in a range of settings. For example, giant kelp (*Macrocystis pyrifera*) can be observed from freely available satellites, while subtidal *Ecklonia radiata* forests require in water monitoring by divers, or possibly remote operated or towed cameras. Intertidal kelp forests can be monitored in situ using cheaper equipment. However, subtidal kelp monitoring requires a reasonable investment in diving equipment or camera systems. The use of vessels for any method greatly adds to the overall costs of monitoring.

Georeferenced underwater videos can be acquired with small boats, able to access shallow water (2-20 m) and automated video analysis can significantly reduce the cost (D'Archino et al 2021). Algorithms could be developed to estimate percentage coverage and biomass of subtidal species.

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

Mana whenua are holders of a wealth of information on the status of kelp beds over time due to historical and present day uses of bull kelp and other kelp and seaweeds for various purposes. There are likely many examples of iwi and hapū monitoring kelp forest attributes, particularly in intertidal and shallow subtidal habitats. One example known to us is in the Wellington region, where Taranaki Whānui is actively involved in seaweed monitoring (intertidal surveys) and restoration. To complement this, they are testing the use of harakeke and pingao for growing seaweed as an alternative biodegradable substrate in laboratory settings at NIWA.

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

Kelp forests are indicative of overall ecosystem health and are a key ecological indicator in temperate reef ecosystems (D'Archino & Piazzi 2021). Poor kelp forest health is often associated with degradation of a range of fisheries species, particularly paua, kina and koura. Abundance of kelp can be considered an excellent indicator of whole ecosystem health.

Part B—Current state and allocation options

B1. What is the current state of the attribute?

While there are in some cases clear signs of decline of kelp beds (Thomsen et al. 2019; Thomsen et al 2021), not all kelp forests from all regions are well represented at a national level. Other kelp forests (e.g., *Macrocystis pyrifera*) show concerning trends under elevated sea surface temperatures (Tait et al. 2021a) but also bounce back quickly during cooler years. In some regions, particularly Northland and Bay of Plenty, *Ecklonia radiata* beds are grazed by kina (*Evechinus chloroticus*), but increasingly the black-spinned urchin (*Centrostephanus rodgersii*) (Balemi & Shears 2023).

Baseline data are needed to benchmark the state and condition of the system being managed, for trends to be identified. This lack of sound baseline information constrains the ability to evaluate the effectiveness of management methods. This lack of critical information impedes the detection of range shifts in response to local or global stressors, as well as the occurrence of local extinctions and the introduction of non-native species.

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

There are reports and resources that have attempted to describe natural reference states for kelp in New Zealand. In 2015 MPI funded a project to understand the role of macroalgae as ecological indicators and their distribution at national scale (D'Archino et al. 2019). This study summarised international and national literature about decline of kelp and monitoring techniques, tested different approaches for mapping, including machine learning and tested stressors (e.g., temperature, light, sediment) on different species. In addition, distribution data gathered from scientists and members of the Local Government Coastal Special Interest Group (C-SIG) involved in ecological research and environmental monitoring, were used to identifying gaps or areas where monitoring work could be repeated.

Kelp forest in Cook Strait and the Marlborough Sounds have been documented by tow-video surveys (Anderson et al 2019) and their role, as biogenic habitat assessed (Anderson et al. 2018).

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

Giant kelp (*Macrocystis pyrifera*) and bull kelp (*Durvillaea spp.*) are native New Zealand seaweed and classified as 'Threatened - At Risk – Declining' (Nelson et al 2019). Due to its decline in Victoria, South Australia and Tasmania the giant kelp is protected under Australia's national environmental law, the Environment Protection and Biodiversity Conservation Act 1999 (EPBC Act) as a threatened ecological community. However, to our knowledge, there are no known numeric or narrative bands describing the state of this attribute (apart from "barrens" vs not).

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

Known tipping points exist for some key parameters. For example, urchin densities above 2 urchins/m2 are often sufficient to turn kelp forests into urchin barrens, yet fewer urchins are required to maintain barrens (Ling et al 2015). Light intensities less than 1 mol/m^2/d is also a key limit for healthy and productive kelp forest ecosystems (Tait 2019).

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

There may be time-lags related to reversing poor water clarity (especially if it is the result of resuspension of legacy sediment), and it may take time following marine reserve establishment for the typical kelp-urchin-predator cascades to re-establish.

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

Māori are intimately familiar with their moana, especially with kina being a kaimoana resource. They hold significant knowledge on what constitutes the "natural state" or minimally disturbed conditions.

Part C—Management levers and context

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

The general relationship is that high kelp cover on rocky reefs is associated with high ecosystem health (high diversity and trophic transfer), whereas key stressors such as overfishing (reducing keystone predators) and sediment loading (reducing light and increasing deposition) can reduce lush canopies of macroalgae to practically unvegetated barrens. There is evidence for key thresholds of light and temperature, e.g., giant kelp (*Macrocystis pyrifera*) appears to be somewhat limited to sea water temperatures less than 19°C (Hayes 1990). Furthermore, many kelp species require light intensities greater than 1 mol/m^2/d (Tait 2019). Sites with light intensities below this threshold show compromised health attributes in kelp forests (Tait et al. 2019). Likewise, more than two exposed urchins per m^2 is associated with excess grazing and loss of kelp (Kerr et al. 2024). All of these relationships are quantified, however, not every species that form kelp forests are equally studied.

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

Interventions around the culling and harvesting of urchins have been shown to be highly effective for restoring kelp forests (Miller et al. 2024). While it is expected that improving water quality attributes (e.g., sedimentation) would lead to improved kelp forest health, there are no examples where land-based restoration has been of sufficient scale to examine potential recovery.

C2-(i). Local government driven

Some regional councils have rocky reef monitoring and water quality monitoring programmes, but there are few-to-no examples of management interventions designed to maintain or restore coastal rocky reef kelp forests.

C2-(ii). Central government driven

The main intervention/mechanism that has helped reverse urchin barrens in some locations is the establishment of Marine Protected Areas.

C2-(iii). Iwi/hapū driven

There have been partnerships with Māori designed to eliminate urchin barrens in places (e.g., Little Barrier Island), and rāhui and temporary closures are used to stop overfishing (which could indirectly help improve kelp forest cover and health).

C2-(iv). NGO, community driven

The community project Love Rimurimu with the assistance of NIWA and Victoria University of Wellington is aiming to restore kelp bed in Wellington harbour and identify the drivers for giant kelp (*Macrocystis pyrifera*) decline.

C2-(v). Internationally driven

Kina barrens and kelp loss are global problems and there are examples of kelp restoration projects involving the provisioning of artificial hard substrate in Southern California that have shown signs of success.

Part D—Impact analysis

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

Degradation of kelp forests will impact key environmental health metrics and compromise critical ecosystem functions. This could influence human health through a lack of suitable food resources (e.g., related fisheries). Allowing kelp forests to continue to degrade will impact the environment and to a lesser degree human health.

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

Fisheries in particular would feel the economic impacts of degradation of kelp forests. Many key fisheries (e.g., paua, kina, koura and many finfish) rely on healthy and functioning kelp forests. Loss of kelp forests would quickly lead to declining fish stocks. These impacts may be greatest felt in southern New Zealand, however, areas of northern New Zealand where barrens occur may also experience loss of economic value in associated fisheries.

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

Evidence already shows that climate change, particularly extreme events like marine heat-waves, greatly impact kelp forests (Thomsen et al, 2019; Tait et al. 2021a; Thomsen et al. 2021). There is also evidence that poor water quality (particularly turbidity), compromises ecological resilience of giant kelp forests (Tait et al. 2021a). Management interventions that focus on improving water clarity will likely improve the resilience of kelp forests to climate change stressors.

Ocean warming and increase in frequency and duration of marine heat waves will negatively affect cold water species e.g., *Durvillaea poha, D. willana, Macrocystis pyrifera*, particularly populations at their northern limit, with consequent loss of associated species. Increasing temperature might not affect directly population of *Ecklonia radiata* in the Northern New Zealand but could promote an expansion in the distribution of the subtropical sea urchin *Centrostephanus rodgersii* leading to urchin barrens (Cornwall et al.2023, Shear et al. 2024).

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9.6 Bryozoan thickets extent and quality

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State of knowledge of the "Bryozoan thicket extent and quality" attribute: Excellent / well established – comprehensive analysis/syntheses; multiple studies agree.

Part A—Attribute and method

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

There is a strong record of evidence in New Zealand of bryozoan thickets providing ecological integrity to significant local areas of the coastal zone and continental shelf, including: a) increasing biodiversity (invertebrates, fish)[1]; b) providing important nursery habitats for the juveniles of valuable fisheries species (e.g., blue cod, tarakihi, leatherjackets, snapper and dredge oysters)[2-5]; c) increasing foraging (food) resources for adults of these and other fisheries species[4]; d) increasing bentho-pelagic coupling through the consumption of phytoplankton and the subsequent expelling of waste products[6]; and e)providing stability to coarser bottom sediments as a biogenic cover, increasing the resilience of such areas to physical forces (currents, waves, storms)[7].

Human health is supported through the production of fisheries catch that supports healthier diets and lifestyles, by providing economic activities for local communities (including in more remote areas) and supporting recreational fishing activities for mental wellness. The ecological integrity provided by bryozoan thickets is directly proportional to the human health benefits.

Here we are referring to frame-building bryozoans, defined as "*species that regularly grow to ≥ 50 mm in three dimensions*" [8], and of these, those that are 'habitat-formers'. The most relevant habitat-former scale are defined as being "*those cases where frame-building bryozoans dominate (at least) square metres of seafloor and thereby contribute significantly to the habitat complexity of the locality*" (at least 27 New Zealand species) [8]. Singularly and collectively, these form 'bryozoan thickets [11-12].

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

There is very strong evidence of negative impacts on ecological integrity, across a range of New Zealand locations. Spatial losses and/or habitat quality declines include the impacts of historical and ongoing commercial fishing activities (Foveaux [7-9], South Taranaki [13], Chetwode Banks Marlborough Coast, Tasman Bay [8,10,16], and from human-driven land-derived sedimentation (Separation Point) [14-16]. Losses are probably under-represented as some bryozoan thicket areas were almost certainly lost early on in human occupation of New Zealand (e.g., areas of the Hauraki Gulf, as suggested from 'death assemblages') [17]. Cascades of effect will have flowed out into far-field non-bryozoan thicket areas, such as a reduction in the production of juvenile fish and by association, the abundance of adults.

Specific examples include a) oyster dredging in western Foveaux Strait from 1977 to 1998, with the loss of extensive long linear *Cinctipora elegans* bryozoan reefs (up to 1 metre high, 4 to 40 metres long, 3 to 6 metres wide) that formed current aligned reef clusters, 300 to several kilometres wide [7]; b) historical trawling in Tasman and Golden bays, which eliminated circa 300 km^2 of *Hippomenella vellicata* ('paper coral') at Torrent Bay [8], and a similarly large area west of D'Urville Island (with the addition of scallop dredging). The Separation Point bryozoan field (*Celleporaria agglutinans*) was estimated to cover circa 200 km^2 in its original state; following fisher concerns on spatial losses of this juvenile fish habitat from trawling impacts, a central 156 km^2 was closed to bulk fishing methods in 1981 [18]. In 2002 about 55 km^2 remained [14], but by 2021 all of the bryozoan habitat was lost, attributed to sedimentation from Cyclone Gita in 2018 [19]. On the South Taranaki Bight, some 2000 km^2 of bryozoan habitat is thought to be reduced in quality and height from the ongoing effects of trawling [6].

Human health has been affected by likely reduced fisheries catches and economic activity, through direct estimates of this have not been quantified [5-6].

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

The pace of change has varied between the regions, but the trajectory of change has been overwhelming negative. The examples given in A2 all involve loss of habitat. For Foveaux Strait bryozoans, significant loss was quantified from 1977 to 1998 (21 years) [7], with earlier large historical losses likely to have occurred since the oyster dredge fishery started operating in the 1880s. For Tasman/Golden Bays, complete losses of two bryozoan areas from trawling impacts had occurred by the late 1960s (over 500 km^2) [8,15-16], while the Separation Point field continued to decline until a complete loss some time before 2021 [19] (likely in 2018 in association with Cyclone Gita), despite protection from bulk fishing. Similarly, Chetwode Bank (circa 100 km^2, coastal Marlborough Sounds) was reported to have been historically covered in bryozoan reefs along with other biogenic habitats, with trawling eliminating this cover somewhere around the 1960s or earlier. Conversely, some bryozoan thicket areas appear to have remained largely unchanged (e.g., 100 km^2 of high quality habitat on the Otago shelf) [8-9,20], through some historical loss from fishing is likely.

In the next 10 to 30 years, recovery is likely for those areas where fishing has been the main driver of loss, if fishing pressure is removed, and the environment remains suitable for bryozoan thickets re-establishing. For Foveaux Strait, some areas of high-quality bryozoan thickets remain [21], possibly from those areas never having been heavily fished. Recovery would be expected across wider Foveaux Strait if oyster dredging was more spatially restricted, but the rate of recovery would be likely to vary widely [22]. Conversely, the extensive bryozoan area losses from Tasman/Golden Bay are unlikely to recover, as increased sedimentation has made the seafloor unsuitable for bryozoans, and significant source populations for larvae may no longer exist within the region. Emerging threats including ocean warming and acidification may increase in their negative impacts over coming decades [23].

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

Little ongoing temporal monitoring is currently done for any bryozoan area, with the exception of a) a high biodiversity area (sponges, bryozoans) off Spirits Bay, upper North Island, closed to scallop dredging following its discovery, assessed for change at decadal scales [24,25]; and b) broad-scale ongoing monitoring of significant biogenic habitat areas across the Marlborough Sound by the Marlborough District Council (MDC) [26,27]. One-off surveys have been carried out to quantify change over time for some bryozoan areas that have older historical data with which to compare, including camera drops at the Ulva Marine Reserve, Stewart Island, and on the Otago shelf [19,20]; as well as camera drops, towed video, and multibeam sonar mapping of the Separation Point field [19, 28].

Measurement methodologies are not standardised, through there has been a call to do so [19], with methods including visual assessment from diving, dropped and towed cameras, from dredging, and the use of sidescan and multibeam sonar mapping. An ideal standardised methodology would include sonar mapping the extent of bryozoan fields, stratification of that imagery/data into different putative bottom types and stratified random ground-truthing using drop or towed cameras, including quantifying metrics such as colony counts (density) and % cover. Such an approach has now been used for several potential marine farm applications (south Foveaux Strait [21], offshore Marlborough Sounds), as well as for Separation Point, Chetwode Bank, and outer Queen Charlotte Sound bryozoan fields [28,29].

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

A suitable vessel is needed for bryozoan thicket surveys, ranging from small run-abouts for more coastal locations, to large seagoing vessels for areas on the continental shelf. Boat size is also driven by the type of sampling equipment being deployed.

Logistical barriers are largely the need to deploy expensive survey vessels and equipment, along with staff, to remote areas.

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

Depending on the location and depth of the bryozoan thickets to be monitored, significant up-front costs may be incurred through the purchase of expensive equipment such as cameras, and the building of bespoke towed camera arrays. Costs vary from low-cost simple systems such as a Go-Pro camera set within a drop frame/on a stand (<$1000), through moderate cost towed cameras (circa $16,000) to expensive high-quality camera arrays ($150,000–$250,000). Other costs include the availability of suitable hardware and software for post-processing and video analysis, and the high cost of human labour needed to process video (through A.I. may help offset this in the future). The use of sidescan and/or multibeam sonar systems is also a significant cost, including the skilled operators needed. Full mapping/monitoring of larger areas (e.g., Separation Point) using sonar approaches is likely to cost several hundred thousand dollars for the mapping alone, although sub-sampling transects could be deployed as a cheaper intermediate approach.

The need for a survey vessel is a significant cost, depending on the size of vessel needed. Small runabouts are relatively cheap to operate, but larger vessels may cost several thousand dollars a day to charter and consume relatively large fuel volumes. The use of day vessels invokes further costs of providing onshore accommodation and meals for the survey team (up to $1000 day for a team of four), while live-aboard vessels require the purchase of stores, and in some cases the services of a cook.

Significant monitoring work requires multiple days of survey, which may include higher labour costs through the need to work through weekends. Direct labour costs vary widely, depending on whether the work is being done in-house by an agency, or contracted out to a research provider.

The simplest lowest cost approach for monitoring of individual sites is the use of a Gro-Pro on a frame/stand, deployed from a suitable sized vessel for the area being surveyed.

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

Although of interest to Māori, I am not aware of any bryozoan thicket monitoring being carried out by representatives of iwi/hapū/rūnanga.

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

Bryozoan thickets and the individual bryozoan species that contribute to them have not been studied in detail in terms of the environmental bounds within which they can exist, aside from the observation that they are generally associated with coarser clean bottom sediments (notably pebbles) and higher current flows [8-10]. Suspended sediment is a known stressor of these and other filter feeders and is thought to be responsible for the loss of the Separation Point bryozoan field. Phytoplankton / chlorophyll *a* in water (trophic state) and dissolved oxygen are almost certainly also important associated attributes for bryozoan thicket extent and quality, but no quantitative work or qualitative observations are available. Horse mussels are found in low densities in association with some bryozoan thickets, but whether there are any direct inter-relationships is unknown. Dredge oysters are considered to have been historically strongly associated with bryozoan reefs [7], but there are strong contrasting views also.

Part B—Current state and allocation options

B1. What is the current state of the attribute?

The current state of bryozoan thickets is generally well known at the broad scale of regional areas, where such thickets have or once covered a significant spatial extent. That state varies by area, e.g., Tasman/Golden Bay thickets – lost; South Taranaki Bight field – reduced quality; Otago Shelf –

healthy and probably static; Foveaux Strait – extensive biogenic reefs lost but recovering in some areas.

The central issue for monitoring and the use of bryozoan thickets is spatial scale, which would need to be addressed by good spatial stratification and associated stratified random sampling. Remote sensing methods such as side-scan and/or multibeam sonar are ideal for large scale detection and mapping but become cost-prohibitive at increasing big scales. They may also not be able to provide bryozoan discrimination for some bryozoan/seafloor combinations, such as the pebbles, sand and low height patchy bryozoans of the Otago Shelf [30,31].

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

No, but such natural reference states could be developed for some areas using past or present data sets. Currently there are no formal definitions of natural reference states for any bryozoan area; these would need to be species mix specific as bryozoan thickets vary in their species composition around New Zealand.

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

No. Numeric or narrative bands could be developed through the use of habitat/environment classification schemes such as the Coastal and Marine Ecological Classification Standard (CMECS). Work by NIWA is looking at this, although it would need to be accepted and adopted by regulatory agencies.

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

No there are not. Current work on Habitats of Particular Significance to Fisheries Management is looking at the relationships between bryozoan thickets/reefs (and other biogenic habitats) and the juveniles of commercially important finfish species (notably blue cod, and to a lesser extent tarakihi), for the Marlborough Sounds region [28]. Indications are that fish density increases with increasing habitat cover and complexity.

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

Yes. Areas where the general environment has changed to the extent it will no longer support the growth and health of historically present bryozoan thickets, are very unlikely to ever recover. A model of recovery by succession has also been proposed for Foveaux Strait bryozoan reefs, where recovery require a succession of intermediate biogenic habitats [22]. Naturally occurring processes such as El Nina and La Nino are also likely to affect recovery through wind and current changes, that in turn may influence larval retention and supply.

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

I know of no relevant tikanga Māori and mātauranga Māori approaches directly at bryozoan thickets, although these may exist outside of the public domain.

Part C—Management levers and context

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

Bulk mechanical fishing methods (e.g., trawling, dredging) are well established as being very detrimental to bryozoan thickets. Extensive bryozoan reef areas have been lost from Foveaux Strait though oyster dredging [7] and are likely to be being prevented from recovery in some areas by ongoing fishing (although other factors may be involved). Bryozoan patches do currently exist within the general area available for oyster dredging [21], including areas that are seldom if ever targeted for oyster harvesting.

High sedimentation and associated suspended sediment, and seafloor deposition, is a known negative stressor of bryozoan thickets in Tasman/Golden Bay [14,32], and almost certainly drove the complete loss of the Separation Point bryozoan field following its protection from bulk fishing in 1981. It is also likely to be one of/the key stressor preventing the recovery of bryozoan fields in Torrent Bay, with its legacy and ongoing effects probably having shifted the seafloor state to one where bryozoans cannot re-establish (muddy, with little harder substrate). The lack of recovery of the west D'Urville Island bryozoan area is less clear in terms of stressors; fishing removed the bryozoans but in the present day the seafloor still holds lots of clean shell cover, and suspended sediment levels are low [31].

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

C2-(i). Local government driven

The Marlborough District Council (MDC) has an ongoing programme of identifying and listing significant marine ecological areas within its region [26,27]. This includes both bryozoan thickets, and mixed biogenic habitats that include bryozoan patches. This listing enables these areas to be excluded from potentially harmful activities such as the establishment of marine farms but does not prevent damage from activities such as anchoring. Land-based sedimentation issues are not addressed by this designation.

C2-(ii). Central government driven

Interventions/mechanisms that are being used to protect the extent and quality of bryozoan thickets are limited to the voluntary (Otago Shelf) [20] or regulatory (Separation Point) [18] closure of important bryozoan thicket areas. The Otago Shelf bryozoans have shown no change over decades, whereas the Separation Point bryozoans have been completely lost [19,28].

C2-(iii). Iwi/hapū driven

I am not aware of interventions/mechanisms being used by iwi/hapū/rūnanga to directly affect this attribute.

C2-(iv). NGO, community driven

I have no knowledge of initiatives to improve bryozoan thickets spatial extent and quality being carried out by representatives of NGOs.

C2-(v). Internationally driven

I have no knowledge of obligations to internationally initiatives that would require improvement of improve bryozoan thickets spatial extent and quality.

Part D—Impact analysis

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

The environmental cost of not managing bryozoan spatial extent and quality is the ongoing loss of coastal biodiversity [1,33], juvenile fish production, and general carrying capacity of the coastal region [5,6].

Human health impacts will include a reduction in the production of fisheries species that support economic activity [5,6] and the associated benefits of consuming fish, as well as a reduction in recreational fishing benefits, for both food gathering and mental wellbeing.

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

The impacts are likely to be felt in inshore fisheries for species whose juveniles directly use bryozoan thickets as juvenile habitat, as well as for larger fish that forage within bryozoan thickets. The spatial scale of these impacts will vary by species and region, depending on the movement/migration ranges of the associated fish species [5]. For example, blue cod impacts are likely to be at the scale of hundreds of metres to a kilometre, snapper impacts at the scale of regional stocks, and tarakihi potentially at the national scale, due to a large scale ontogenetic migration of juveniles from the lower South Island up the entire east coast of New Zealand, and of juveniles from the Tasman/Golden Bay region to the entire west coast of New Zealand.

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

The impact of climate change on bryozoan thickets spatial extent and quality is unknown, but probably likely to be negative as many key species are more dominant and widespread in colder southern waters [10,34]. Ocean acidification has been identified as another major likely threat [23]. There is little opportunity for bryozoan thickets to extent their range further south, as they already occupy suitable depths and areas there. Range and spatial extent retraction seems likely. Protecting and restoring bryozoan thicket areas in southern regions where climate change impacts may be less severe is an obvious key response.

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9.7 Macroinvertebrate community composition

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

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

Part A—Attribute and method

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

Part B—Current state and allocation options

B1. What is the current state of the attribute?

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

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

B2. Are there known natural reference states described for New Zealand that could inform management or allocation options?

One of the ways to understand natural or reference conditions is to seek and sample "pristine" locations. Given the widespread influence on humans on our land, oceans, and climate, finding "pristine" locations can be difficult. Secondly, there are many permutations and variations of "healthy"—which impedes the identification of a single clear reference state. Nevertheless, it is relatively easy to identify degraded states in this attribute, which can be compared to healthier locations nearby.

Several indices used internationally and in New Zealand utilise references states in some way. These include the Traits Based Index (TBI, [53]), the Benthic Health Models (BHM, [8]), the Estuary Trophic Index (ETI, [52]), and the Benthic Index of Biotic Integrity (B-IBI, [54]). At least one overseas index has been successfully adapted for use with New Zealand macroinvertebrate community data (e.g., AMBI [13, 55]). However, the ability of the various indices to track stressors and indicate health varies widely [56]. Some authors have tried combining indices to take advantage of each one's individual strengths [57]. However, testing and validation of indices outside of the regions where they were originally developed remains an issue.

B3. Are there any existing numeric or narrative bands described for this attribute? Are there any levels used in other jurisdictions that could inform bands? (e.g., US EPA, Biodiversity Convention, ANZECC, Regional Council set limit)

There are several metrics and numeric bands being used in New Zealand to describe the status of macroinvertebrates communities. The TBI provides an indicator of coastal ecological integrity and resiliency based on macroinvertebrate traits and abundance [53]. Scores > 0.4 are considered 'good', 0.3 – 0.4 are considered 'moderate', and <0.3 indicate 'poor' health and low functional redundancy. However, the use of this scoring system outside of Auckland and Waikato is not advised at this time, nor is the comparison of TBI scores across intertidal and subtidal habitats [58].

Bands for the AMBI have been used in New Zealand to categorise site health [e.g., [59]. However, the appropriateness of the banding system that was developed overseas [healthy to unhealthy reported as 'Very low' 0.0-1.2; 'Low' 1.2-3.3; 'Fair' 3.3-5.0; 'High' 5.0-6.0; and 'Very High' >6.0 to azoic] has not to our knowledge been checked or validated. The original Borja et al. publication [13] had eight benthic community health categories.

Numeric bands for BHMs are also available. Local BHMs developed for Auckland and Bay of Plenty estuaries were expanded to the national level in a 2020 [8]. Although the methods underpinning the BHMs are complex (based on multivariate canonical analysis of principle coordinates), results can be summarised relatively simply in five equally sized categories [Level of impact from lowest to highest: 'Very Low' 1.0 to <2.0, 'Low' 2.0 to >3.0, 'Moderate' 3.0 to <4.0, 'High' 4.0 to <5.0, and 'Very high' ≥5.0]. The National BHMs have been shown to perform well in two estuary types (e.g., tidal lagoons and shallow river valleys) and across five to six regions of New Zealand (Mud BHM: Abel, Banks, Chalmers, Portland, Raglan and Northeastern; Metals BHM: Abel, Southeastern, Portland, Raglan and Northeastern)[8]. Councils appear supportive of the use of the National BHM models, with further testing and refinement urged as more data become available.

B4. Are there any known thresholds or tipping points that relate to specific effects on ecological integrity or human health?

There is general consensus that >25% mud content in intertidal sediments will lead to decreases of macroinvertebrate community integrity [2, 11, 23, 25, 26, 29, 60]. Recent research suggests that this threshold may be slightly higher in the subtidal zone [61]. It has been suggested that >4-5% of organic content in sediment is associated with degraded macroinvertebrate communities [2, 24, 25]. Thresholds for over environmental variables, such as Chlorophyll *a* and coastal sea surface temperatures are not well delimited at this time.

B5. Are there lag times and legacy effects? What are the nature of these and how do they impact state and trend assessment? Furthermore, are there any naturally occurring processes, including long-term cycles, that may influence the state and trend assessments?

Macroinvertebrate communities respond to changes in environmental conditions (individuals immigrate or emigrate, alter their reproduction, or die). These responses are unlikely to be immediately detectable (i.e., they are temporally lagged). Temporally lagged responses of macroinvertebrate communities may occur depending on life history traits (i.e., timing of reproduction, hatch, or settlement) and ecological thresholds (i.e., how tolerant individual species and communities are to particular levels of stress). A recent study suggested that responses of macroinvertebrate communities were site-dependent and lagged in relation to oceanic, climatic, freshwater, and local environmental conditions [32].

At broader scales, legacies of past loadings—particularly the infilling and substantial expansion of mangroves and muddy habitats in our estuaries—may be masking or interfering with the detection of responses to newly loaded contaminants. There is substantial ecological theory on the topic of alternative stable states and hysteresis [62-65]. This work suggests that although elevated loading of catchment contaminants has led to estuarine degradation, we cannot expect estuaries to immediately respond to catchment contaminant reductions. This type of lag is much longer than the ecological lags described above. Although the existence of lagged responses is unequivocal, exploring the significance of lags is difficult.

B6. What tikanga Māori and mātauranga Māori could inform bands or allocation options? How? For example, by contributing to defining minimally disturbed conditions, or unacceptable degradation.

To our knowledge, tikanga Māori and mātauranga Māori has not been considered during the development of macroinvertebrate health/integrity bands. Macroinvertebrate communities, specifically kaimoana (e.g., seafood), are highly valuable for Māori as crucial economic and cultural resources. Tikanga Māori and mātauranga Māori should be included in decision making and band/threshold definition where possible.

Part C—Management levers and context

C1. What is the relationship between the state of the environment and stresses on that state? Can this relationship be quantified?

The relationship between macroinvertebrate communities and environmental stressors is reasonably well understood. There are several studies showing the effects of single and multiple stressors on the macroinvertebrate communities [1, 2, 9, 24, 25, 27, 30-33]. In most of the cases the relationships are quantified and showed deterioration of macroinvertebrate communities with increased environmental stressors. The BHM also quantified the relationship between macroinvertebrate communities and specific stressors such as mud content and metals concentration in sediment [8].

C2. Are there interventions/mechanisms being used to affect this attribute? What evidence is there to show that they are/are not being implemented and being effective?

Macroinvertebrate community composition has utility as an attribute because it is an integrative and responsive indicator of trends in estuarine/coastal ecological integrity. However, most catchment and estuarine interventions are not targeted at improving “macroinvertebrate communities” *per se*. Macroinvertebrates are generally small and cryptic (and tend to be overlooked by non-specialists), but their critical roles in food webs and maintaining a range of life-supporting ecosystem functions demonstrate their importance. Interventions that could affect macroinvertebrate communities include land/freshwater management practices aimed at reducing sediment and nutrient loads into coastal receiving environments. Recent updates to the National Policy Statement for Freshwater Management (NPSFM) are directed as such. Evidence that land/freshwater interventions are leading to improvements in macroinvertebrate community health states has not emerged yet (probably due to the presence of lags and legacy effects mentioned above).

C2-(i). Local government driven

Many councils and local/regional authorities have taken steps to control sediment and nutrient inputs, which should result in the eventual improvement of macroinvertebrate communities. However, current sediment and nutrients inputs to coastal ecosystems are likely still high and driven by external events (such as recent Cyclone Gabrielle). Some of the Jobs for Nature initiatives (while Central government driven) are being implemented locally, but outcomes for macroinvertebrates and estuarine health are not yet known. Some of the ‘local’ initiatives are being undertaken on relatively large scales (e.g., the $100m Kaipara Moana Remediation project; [66]).

C2-(ii). Central government driven

Central government developed the Essential Freshwater Package to improve and maintain sustainable outcomes from freshwater management and updates the NPSFM approximately every three years. The Jobs for Nature programme (administered by five central government agencies) has directed hundreds of millions of dollars towards riparian planting in catchments to prevent sediments and nutrients from entering freshwater and coastal receiving environments downstream. Obviously, the Jobs for Nature funding was not targeted at improving the macroinvertebrate community composition attribute, though the attribute may be useful at tracking the successes of individual catchment interventions (with the caveat that there will be temporal lags and legacy effects). Several central government agencies are commissioning work on the effects of catchment contaminants in estuarine/coastal ecosystems and/or have strategies for catchment contaminant load reductions, but specific actions may not be widely implemented yet.

C2-(iii). Iwi/hapū driven

Iwi and hapū have been heavily involved in Jobs for Nature projects across New Zealand. Iwi and hapū are also leading estuarine restoration initiatives in partnership with Councils, CRIs, as part of the National Science Challenge programmes from various universities, and with other research institutes/providers [67, 68]. Iwi and hapū have implemented Customary Management Areas tools, including temporary closures in coastal areas to protect shellfish resources (e.g., scallops, pipi). Again, however, we do not know of any iwi/hapū driven initiatives that were specifically designed to improve the condition of macroinvertebrate communities in coastal ecosystems of New Zealand.

C2-(iv). NGO, community driven

As above, although there are many estuary/coast orientated community groups, we know of no initiatives that are being specifically designed to improve the condition of macroinvertebrate communities in coastal ecosystems of New Zealand. Revive Our Gulf is a broad partnership designed to restore mussel reefs in the Hauraki Gulf [69], and there are similar shellfish restoration initiatives at the Top of the South Island.

C2-(v). Internationally driven

To the best of our knowledge there are no initiatives to improve the condition of estuarine/coastal ecosystems or macroinvertebrate communities in coastal ecosystems of New Zealand driven by international entities. However, the Department of Conservation and other agencies set many of their policy goals to align with Convention of Biological Diversity targets, e.g., CBD Aichi Target 11 (biodiversity and ecosystem services are conserved using effective area-based conservation measures integrated into wider landscapes and seascapes).

Part D—Impact analysis

D1. What would be the environmental/human health impacts of not managing this attribute?

Degradation of macroinvertebrate communities would affect the ecological integrity of coastal ecosystems, as explained in previous sections. Ignoring the management of macroinvertebrate communities could lead to reductions in the delivery of key ecosystem functions and services. For example, less diverse and abundant macroinvertebrate communities may result in reduced organic matter cycling and nutrient removal, essential processes for healthy estuarine/coastal ecosystems. It will also result in less food available for higher trophic levels, and reduced pollutant detoxification capacity. Macroinvertebrates are often habitat-defining species (e.g., “cockle bed”, “tube-worm mat”, “crab burrow habitat”) that modify the environment and facilitate/inhibit other organisms—

signifying their fundamental roles in estuarine/coastal ecosystems. Not managing macroinvertebrate communities could impact Māori, particularly with estuarine/coastal kaimoana supporting whānau nutritionally, economically, and culturally.

D2. Where and on who would the economic impacts likely be felt? (e.g., Horticulture in Hawke's Bay, Electricity generation, Housing availability and supply in Auckland)

Impacts of macroinvertebrate community degradation may manifest as increased eutrophication, reduced food available for fish and birds, increased deposition of sediment, and altered sediment geochemistry/oxygenation [2, 3, 30, 31, 33, 35, 48]. Furthermore, coastal ecosystems will be impacted economically by reduced shellfish fisheries and less cultural activities (e.g., tourism).

D3. How will this attribute be affected by climate change? What will that require in terms of management response to mitigate this?

Increasing temperatures, frequency of storms, and heatwaves (i.e., climate change), are expected to exacerbate the pressures/stressors on macroinvertebrate communities [25, 27, 32]. For example, increasing temperatures and storms will result in increased nutrient and sediment inputs, anoxia, and eutrophication events. Sea level rise is also expected to alter the proportions of intertidal and subtidal habitats, and there is a great likelihood that estuarine morphology (the positions of tidal creeks and sediment accretion/deposition zones) may change. Management actions should focus on limiting sediment and nutrient inputs from terrestrial sources entering to coastal ecosystems, to reduce the risk of worsening the condition of the macroinvertebrate communities. Although carbon reduction strategies and a reduced reliance on fossil fuels is essential, the ability of New Zealand to influence overall climate change trajectories may be small.

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9.8 Heavy metals in sediment

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Alternate attribute name: Trace metals in coastal sediment

Preamble: Pressures from human activities, such as agriculture, effluent discharges from landfill and wastewater treatment plants (WWTPs), urbanisation, and industrial wastes increase sediment metal concentrations [1]. Metals are of growing concern in terms of water quality management, as they cannot be degraded in the environment although some metal species can be transformed into other species which may be more or less toxic [1].

State of knowledge of "Trace metals in coastal sediment" attribute: Good / established but incomplete – general agreement, but limited data/studies

Part A—Attribute and method

A1. How does the attribute relate to ecological integrity or human health?

Trace metals are naturally present in the environment. Their distribution depends on the presence of natural sources (e.g., volcanoes or erosion) and human activities through extraction from ores [2]. The main anthropogenic activities resulting in the discharge of metals include fossil fuel combustion, industrial and agricultural processes and many metals are used in daily household activities [3]. It is important to recognise the types of metals. For instance, cadmium and mercury are heavy metals but other metals of environmental concern including zinc and copper are essential metals. It is estimated that one-third of all proteins requires a metal cofactor for normal functions [2]. However, even essential metals can be toxic and that depends on the concentration. This relates to the concept of essentiality as illustrated in Figure 1. For essential metals like copper, zinc and selenium, there is a "window of essentiality" which represents a range of concentrations that will maintain a level of health in an organism- as illustrated in Figure 1A. For non-essential metals like cadmium, when concentrations reach levels that overcome the defence capacity of an organism, then it becomes toxic (Figure 1, panel B). This is why using trace metals is the appropriate term to use as it covers all metals. The most appropriate term would be trace elements as arsenic is defined as an element or metalloid.

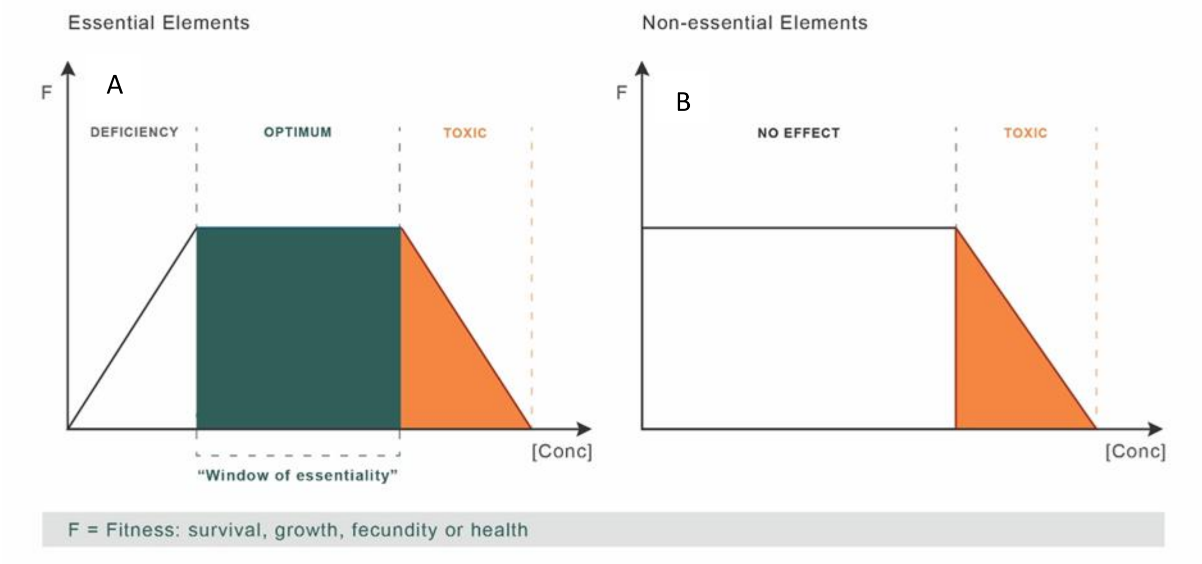

Figure 1. Conceptual diagrams illustrating the differences in concentration–response relationships with respect to organism health between A) essential metals and B) non-essential metals.

The toxicity of trace metals is well established and can impact both ecosystem and human health [4]. The relationship of metals to human and ecological health has been covered in the Attribute of trace metals in water. The hazards remain similar with sediment as another source of metal exposure with receptor species most at-risk being sediment dwelling organisms. Metals in sediment can enter the food chain through bioaccumulation posing a risk to exposed biota higher in the food chain and humans [5]. As sediment is the major compartment where metals accumulate, it is also the major source of exposure posing the highest risk [6].

A2. What is the evidence of impact on (a) ecological integrity or (b) human health? What is the spatial extent and magnitude of degradation?

There is strong evidence globally of the adverse effects on human metabolism resulting from exposure to metal-contaminated drinking water [3]. Exposure to non-essential metals is potentially harmful as they do not have physiological roles in the metabolism of cells. In addition, the ingestion of metals via food or water can modify the metabolism of other essential elements including zinc, copper, iron and selenium [4]. Metals and metal compounds can interfere with functions of the central nervous system (CNS), the haematopoietic system, liver and kidneys [2].

Areas of high anthropogenic activity like urban centres are more susceptible to the impacts of trace metals. Urban areas have larger areas of impervious surfaces such as roofs, roads and paved areas that are sources of metals [7]. Many urban streams and coastal zones are also the receiving environment for untreated sewage, via leakage or overflows from wastewater networks and treatment plants. The concentration and bioavailability of metals bound to estuarine sediments depends on many factors including, redox potential, pH, salinity, dissolved metal species and sediment composition [8].

It is well documented that estuaries and coastal zones have been key locations for human settlement and marine resource use. Centuries of overexploitation, habitat transformation, and pollution have led to estuarine degradation and biodiversity loss undermining their ecological resilience [9]. Local examples of estuarine and coastal ecosystems degradation have been reported. Ecological health declines in benthic community structures have been against gradients of metal contamination at concentrations below guideline thresholds (e.g., [10,11]. Ecotoxicity studies of sediment samples from the Ahuriri (Hawke's Bay) and New River (Southland) Estuaries revealed the presence of metals, including lead and zinc that were at concentrations exceeding sediment quality guideline values [12]. The situation is the same nationwide as dissolved zinc was found to be positively related to the proportion of urban land cover and imperviousness in upstream catchments [7].

A3. What has been the pace and trajectory of change in this attribute, and what do we expect in the future 10 - 30 years under the status quo? Are impacts reversible or irreversible (within a generation)?

The status quo would result in the continuous accumulation of metals in the environment as they are not biodegradable. Worldwide, in addition to the issue of anthropogenic zinc contamination in urban areas, contamination of soils with zinc has increased in some agricultural sectors, such as dairy farming and horticulture. The most significant concern for freshwater lakes relates to the partitioning of zinc to bed sediments, where over time it may gradually build up beyond ecotoxic thresholds for macroinvertebrates and other bed-dwelling organisms, which are integral components of aquatic ecosystems [13].

Yes, there is evidence that better management of trace metal sources can reverse the trends. For instance, the global phase-out of leaded petrol use has contributed to the decline of concentrations in the ocean [14]. A UK study showed that reductions in industrial activity and improved environmental controls on emissions resulted in a decline in trace metal concentrations in sediments [8].

A4-(i) What monitoring is currently done and how is it reported? (e.g., is there a standard, and how consistently is it used, who is monitoring for what purpose)? Is there a consensus on the most appropriate measurement method?

A report commissioned by the PCE provided a national-level summary of the chemical contaminants including metals that Regional Councils/Unitary Authorities include in consent-based monitoring requirements and routine State of the Environment (SoE) monitoring programmes [15]. It stated that copper, zinc and lead were the most frequently listed trace metals monitored as part of consent conditions [15]. It is interesting that to date, there are no published studies that have quantitatively assessed relationships of copper and zinc with intensity of urban land use, despite these metals being key contaminants in urban streams and frequently used as indicators of stormwater inputs [7].

One important aspect that is not commonly included in current monitoring frameworks is the use of biological indicators, or bioindicators. Bioindication is the use of an organism, a part of an organism, or a community of organisms, to assess the impacts the quality of its/their environment [5]. A definition of bioindicator was suggested to be an anthropogenically induced variation in biochemical, physiological, or ecological components or processes, structures, or functions (i.e., a biomarker) that can be causally-linked to biological effects [16].

Macroinvertebrate abundance can be influenced by the level of stressors as taxon richness declines across pollution gradients. Pollution sensitive taxa respond to levels of contaminants leading to alterations to benthic macroinvertebrate assemblages (e.g., [17]).

There have been notable advances in the development of bioavailability models for assessing toxicity as a function of water chemistry in freshwater ecosystems. For instance, the biotic ligand model (BLM), the multiple linear regression model, and multimetal BLM have been developed for most of the common mono- and divalent metals. Species sensitivity distributions for many metals are available, making it possible for many jurisdictions to develop or update water quality criteria or guidelines [18]. Sediment bioavailability models are emerging including models that allow for prediction of toxicity in sediments for copper and nickel [18].

A4-(ii) Are there any implementation issues such as accessing privately owned land to collect repeat samples for regulatory informing purposes?

I am not in a position to comment but Regional Councils have selected sites where they monitor trends for the SoE. It is certainly possible as consent holders would also have access to sites for monitoring as part of their consent conditions. This should be part of using proper engagement practices with all stakeholders and partners.

A4-(iii) What are the costs associated with monitoring the attribute? This includes up-front costs to set up for monitoring (e.g., purchase of equipment) and on-going operational costs (e.g., analysis of samples).

The analytical methods using inductively coupled plasma mass spectrometry (ICP-MS) instruments can measure elements and metals and are well established and validated. Several commercial laboratories including Hill Labs and AsureQuality can measure metals at competitive prices.

A recent investigation reported limitations that councils have identified that preventing the expansion of current monitoring programmes including the high costs for both laboratory analysis and council staff time spent doing monitoring and reporting [15]. However, it should be noted that consent holders cover agreed conditions monitoring costs.

A5. Are there examples of this being monitored by Iwi/Māori? If so, by who and how?

We are not aware of any sediment heavy metals monitoring being regularly undertaken by iwi/hapū/rūnanga. Resourcing is difficult for iwi/hapū/rūnanga to obtain, and any monitoring by agencies is generally infrequent, inconsistent, and ad hoc, and most programmes fail to provide information on whether chemical contaminants will have impacts of concern to Māori [34]. Some of the environmental assessment frameworks being developed by/with iwi/hapū/rūnanga include “safe to eat” or “safe to swim” outcomes [35-37]. Data/indicators required to fully realise these holistic cultural assessment frameworks will require information about heavy metals in water, sediment, and/or mahinga kai species. See also [13], [19], [20].

A6. Are there known correlations or relationships between this attribute and other attribute(s), and what are the nature of these relationships?

Contaminants are mostly found as complex mixtures of which metals are one family of pollutants at impacted sites. The issue of multiple stressors relates to the range of sources that put pressure on the receiving environment – e.g., stormwater and wastewater contain a range of other types of contaminants. Cumulative effects, through additional new marine industries, climate change and other stressors, can reduce environmental resilience and increase the risk of environmental or economic collapse [24]. The importance of sediments as stressors will depend on site ecosystem attributes and the magnitude and preponderance of co-occurring stressors [25]. Therefore, management of coastal waters must contend with multiple drivers in concert as the coordination of regulating agencies for urban and agricultural runoff is warranted as metals are only one component within a range of other contaminants that can accumulate in sediment [26].

Part B—Current state and allocation options

B1. What is the current state of the attribute?

The information to date indicates that trace metals are accumulating in our environment, e.g., zinc was positively related to the proportion of urban land cover and imperviousness in a upstream catchments [7]. The ecotoxicological effects of trace metals and their speciation under a range of environmental conditions are well understood and documented (as per references cited above). The key anthropogenic sources are well characterised to assist the management of these contaminants. The main challenge is that the management of metals requires a holistic/system approach as there are multiple factors to consider. For instance, roof material often contains zinc that can leach overtime. Some effort is required to find alternative types of material with less impacts which needs to be underpinned by appropriate policy. There are examples of recovery following policy changes, e.g., the global phase-out of leaded petrol use has contributed to the decline of concentrations in the ocean [14].

B2. Are there known natural reference states described for New Zealand that could inform management or allocation options?

Finding reference sites with low levels of anthropogenic pressure is important to provide a baseline to confirm adverse impacts of metals and other stressors on receiving ecosystems. However, it is very difficult and nearly impossible to find reference sites that experience no anthropogenic pressure.

One option to consider is to use a ranking of environmental targets in line with the ecosystems to protect. For instance, New Zealand is recognized internationally for its environmental management and innovative regulatory frameworks, as demonstrated, for example, by the implementation of the first no-take marine reserve [24].

B3. Are there any existing numeric or narrative bands described for this attribute? Are there any levels used in other jurisdictions that could inform bands? (e.g., US EPA, Biodiversity Convention, ANZECC, Regional Council set limit)

Sediment quality guideline values (SQGVs) were derived and updated [27]. These values are now used as default guideline values (DGVs) for toxicants in sediment in the Australian and New Zealand Guidelines for sediment quality [1].

B4. Are there any known thresholds or tipping points that relate to specific effects on ecological integrity or human health?

[1] https://www.waterquality.gov.au/anz-guidelines/guideline-values/default/sediment-quality-toxicants

Yes, there are threshold value guidelines available. The ANZG DGVs have been developed to provide threshold values for metals and other contaminants. They are set to provide a range of protection as per point B3. The sediment DGVs indicate the concentrations below which there is a low risk of unacceptable effects occurring, and should be used, with other lines of evidence, to protect aquatic ecosystems. In contrast, the 'upper' guideline values (GV-high), provide an indication of concentrations at which there might already have toxicity-related adverse effects. As such, the GV-high value should only be used as an indicator of potential high-level toxicity problems, not as a guideline value to ensure protection of ecosystems.

If a DGV is exceeded or even where toxicant concentrations in the sediment are trending towards the DGV, it is recommended using multiple lines of evidence approach as part of the weight-of-evidence process to better assess the risk to the sediment ecosystem.

B5. Are there lag times and legacy effects? What are the nature of these and how do they impact state and trend assessment? Furthermore, are there any naturally occurring processes, including long-term cycles, that may influence the state and trend assessments?

As discussed in the above sections, metals have multiple anthropogenic sources and they can continue to accumulate in various environmental compartments including sediment due to the non-degradability of metals. There are indications of degradation of coastal ecosystems globally and in New Zealand (e.g., [9-11,26]. Despite the occurrence of some extinctions, most species and functional groups persist in coastal areas but in greatly reduced numbers. The potential for recovery remains, and where remediation efforts have focused on protection and restoration, recovery has been observed but over long time periods [9].

B6. What tikanga Māori and mātauranga Māori could inform bands or allocation options? How? For example, by contributing to defining minimally disturbed conditions, or unacceptable degradation.

It has been recognised that indigenous peoples, knowledge frameworks, and values are critical in orienting international efforts for the management of chemicals and waste that are more sustainable and equitable for all [28]. A high standard of sediment quality is an outcome sought by iwi/hapū/rūnanga. There is tikanga and mātauranga Māori relevant to informing bands, allocation options, minimally disturbed conditions and/or unacceptable degradation residing in treaty settlements, catchment/species restoration strategies, cultural impact assessments, environment court submissions, iwi environmental management plans, reports, etc.

There are one-off-studies where iwi/hapū/rūnanga are influencing research initiatives exploring the state and impacts of environmental contaminants (including heavy metals) on the outcomes they are seeking (e.g., mauri is protected, kai is safe to eat, water is safe to swim) (e.g., [38-40]).

See also [19], [20], [22], and [23].

Part C—Management levers and context

C1. What is the relationship between the state of the environment and stresses on that state? Can this relationship be quantified?

The key here is to address the increasing trends of metal accumulation and come up with solutions to revert the increasing trends by developing better management frameworks for the sources. The SoE reporting for MfE highlights the level of environmental degradation in both freshwater and marine domains [9,29]. Metals are one of the multiple stressors that have been identified with sources including stormwater, municipal treated wastewater and agricultural discharges.

The toxicity and ecotoxicity of individual metals are well characterised and understood. Predicting or assessing the environmental impacts of an individual chemical is a challenge in a field situation as contaminants are often found in complex mixtures. For instance, exposure to low levels of multiple chemicals in mixtures can cause toxicity at concentrations where exposure to an individual chemical might cause no effect based on their DGVs. This is because multiple physiological processes may be affected by chemicals having different mechanisms of toxicity. This is a strong argument for the need of a system approach to the management of aquatic systems.

C2. Are there interventions/mechanisms being used to affect this attribute? What evidence is there to show that they are/are not being implemented and being effective?

C2-(i). Local government driven and C2-(ii). Central government driven and C2-(v). Internationally driven

The Australia New Zealand guidelines for sediment quality DGVs are designed to trigger further specific site risk assessment based on a weight of evidence. In a recent survey on the type and range of chemical contaminants councils do, the emphasis was on the type of chemicals, but the implications of exceedance of DGVs was not assessed [15]. I am not aware of any follow up studies in New Zealand responding to a DGV exceedance.

C2-(iii). Iwi/hapū driven and C2-(iv). NGO, community driven

We are not aware of interventions/mechanisms being used by NGOs or iwi/hapū/rūnanga to directly affect this attribute.

Part D—Impact analysis

D1. What would be the environmental/human health impacts of not managing this attribute?

A business-as-usual scenario would lead to on-going increase of metals in sediment and have devastating impacts on exposed ecosystems. There is no doubt that the accumulation of anthropogenic pollutants in the environment is causing harm and scientists need to work with other stakeholders to reduce pollution [30]. Metals are not degradable so any continuous discharges will accumulate into the various environmental compartments and biota. The impacts of human activities have pushed estuarine and coastal ecosystems far from their historical baseline of rich, diverse, and productive ecosystems [9]. Managing the sources is a priority to ensure the protection of these valuable ecosystems. However, there are examples of declining metal concentrations from improved environmental controls on emissions and discharges of metals and other contaminants, e.g., [8].

There are multiple challenges to reduce the discharge of metals in urban environments, particularly non-point sources like stormwater. There are examples of options to reduce metals at the sources [13]. There are solutions to consider but their implementation would be challenging, e.g., an initiative to impose restrictions on the maximum amount of zinc in galvanised or zinc coated roofing materials [13].

D2. Where and on who would the economic impacts likely be felt? (e.g., Horticulture in Hawke's Bay, Electricity generation, Housing availability and supply in Auckland)

Coastal and ocean ecosystems provide commercial, cultural, recreational and economic benefits as well as support diverse habitats and species of local and global significance [24]. It is well-recognized that healthy and thriving coastal and freshwater ecosystems are essential for economic growth and food production [24]. The key impacts from the pressure that metals place on receiving environments is the potential loss in biodiversity and disruption of ecosystem functions and services through shifts in distributions of key species. Fishery and aquaculture industries are most likely to be impacted by pressure from metal contamination. Healthy and functional ecosystems and healthy fish stocks are important for the fisheries industry [31]. There are other aspects to consider including natural beauty of our estuaries, coastal and open ocean areas that are central to our culture and national identity.

D3. How will this attribute be affected by climate change? What will that require in terms of management response to mitigate this?

Climate change will have multiple effects in modulating the accumulation and bioavailability of metals. Climate change increasingly affects the variation in volume and frequency of stormwater events and runoff which can increase resuspension and direct exposure of sediments in water bodies [1]. The key concern with the effects of climate change on the risks associated with metal contamination is that changes to temperature and pH can modulate the speciation of metals or basically, their bioavailability. The importance of metal speciation cannot be overstated as it modulates the bioavailability and toxicology of trace metals. The simplest feature of speciation is whether the metal is in the dissolved or particulate form. Originally, environmental regulations were based on total metals present in the water as assayed by hot acid digestion of the samples. However, there has been a gradual change in many jurisdictions to regulations based on the dissolved component only. This reflects the general recognition that particulate metals exhibit negligible toxicity and bioavailability to aquatic organisms relative to dissolved metals [2]. Increases in temperature was correlated with increasing toxicity of metals to aquatic organisms [32]. As such, temperature should be accounted in risk assessment, because it may modify the effects of chemicals on the structure and functioning of aquatic communities, especially at higher levels of biological organization [33].

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9.9 Heavy metals in water (or indicator spp.)

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Alternate attribute name: Trace elements in water (or indicator spp.)

Preamble: Heavy metals is the commonly but wrongly used term to describe all metals. It would be more appropriate to use trace metals or even better, trace elements including arsenic, a metalloid. The term 'trace metals' will be used in this document to provide a more accurate description as it still contains the term 'metals'. Ideally 'trace elements' should be used but it can cause some level of confusion.

State of knowledge of "Trace metals in estuary/coastal water (or indicator species)" attribute:
Good / established but incomplete – general agreement, but limited data/studies

Part A—Attribute and method

A1. How does the attribute relate to ecological integrity or human health?

Trace metals are naturally present in the environment. Their distribution depends on the presence of natural sources (e.g., volcanoes or erosion) and human activities through extraction from ores [1]. The main anthropogenic activities resulting in the discharge of metals include fossil fuel combustion, industrial and agricultural processes and many metals are used in daily home activities [2].

The term heavy metals is often used to describe metals in general. However, it is not appropriate as not all metals are heavy or non-essential. For instance, cadmium and mercury are heavy metals but other metals of environmental concern including zinc and copper are essential metals. It is estimated that one-third of all proteins requires a metal cofactor for normal functions [1]. However, even essential metals can be toxic and that depends on the concentration. This relates to the concept of essentiality as illustrated in Figure 1. For essential metals like copper, zinc and selenium, there is a "window of essentiality" which represents a range of concentrations that will maintain a level of health in an organism - as illustrated in Figure 1A. For non-essential metals like cadmium, when concentrations reach levels that overcome the defence capacity of an organism, then it becomes toxic (Figure 1, panel B). This is why using trace metals is the appropriate term to use as it covers all metals. The most appropriate term would be trace elements as arsenic is defined as an element or metalloid.

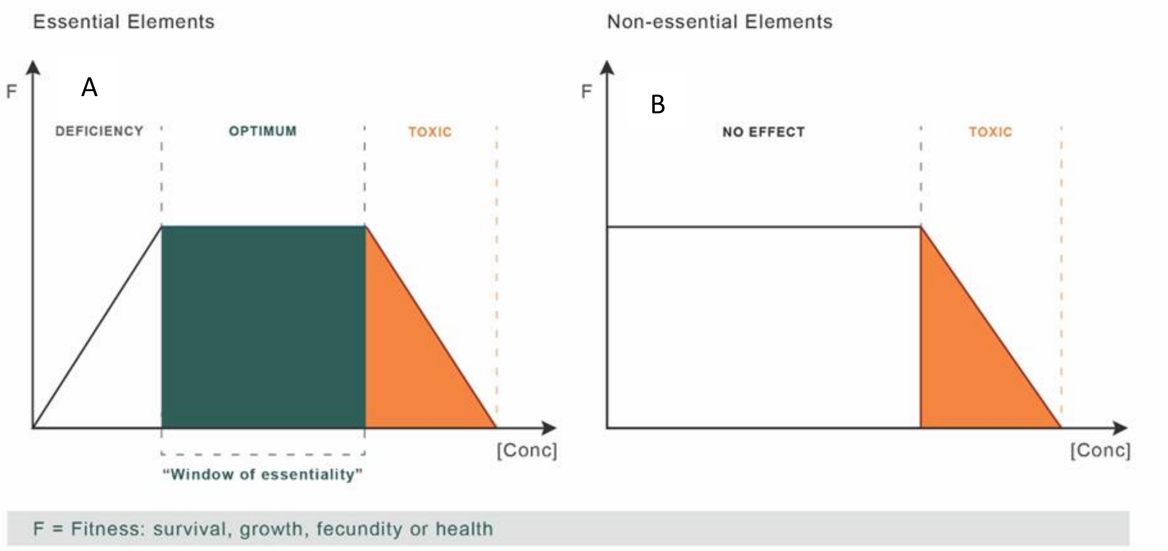

Figure 1. Conceptual diagrams illustrating the differences in concentration–response relationships with respect to organism health between A) essential metals and B) non-essential metals.

The toxicity of trace metals is well established and can impact both ecosystem and human health. Metals and metal compounds can interfere with functions of the central nervous system (CNS), the haematopoietic system, liver and kidneys [2].

A2. What is the evidence of impact on (a) ecological integrity or (b) human health? What is the spatial extent and magnitude of degradation?

There is strong evidence globally of the adverse effects on human metabolism resulting from exposure to metal-contaminated drinking water [3]. Exposure to non-essential metals is potentially harmful as they do not have physiological roles in the metabolism of cells. In addition, the ingestion of metals via food or water can modify the metabolism of other essential elements including zinc, copper, iron and selenium [2]. The general mechanism of heavy metal toxicity is through the production of reactive oxygen species (ROS) leading to oxidative damage and subsequently, adverse effects on health [3]. The disruption of metal ion homeostasis leads to oxidative stress through the formation of ROS which overwhelm body antioxidant protection and subsequently induces DNA damage, lipid peroxidation, protein modification and other effects, all symptomatic of numerous diseases, including cancer, cardiovascular disease, diabetes, atherosclerosis, neurological disorders (Alzheimer's disease, Parkinson's disease), chronic inflammation and others [4]. Another important mechanism of toxicity is the bonding of redox inactive metals like cadmium, arsenic and lead to sulphydryl groups of proteins and depletion of glutathione [4]. The mechanisms of toxicity are conserved, and metals affect ecosystem health in a similar way.

Areas of high anthropogenic activity like urban centres are more susceptible to the impacts of metals. Urban areas have larger areas of impervious surfaces such as roofs, roads and paved areas that are sources of metals [5]. Stream water quality changes in urban areas as development both increases the generation of contaminants and changes the transport and processing of contaminants. Many urban streams and coastal zones are also the receiving environment for untreated sewage, via leakage or overflows from wastewater networks and treatment plants.

Increasing population pressure and urbanization of the coastal zones have resulted in a variety of chronic impacts operating on coastal and estuarine ecosystems. Land-based activities affect the runoff of pollutants and nutrients into coastal waters affecting global biodiversity and ultimately the provision of ecosystem services [6-8]. Local studies in the Auckland coastal zone and the Tauranga Harbour showed ecological health decline, based on community structure composition changes along a pollution gradient, occurring at metal levels below guideline threshold values [9,10]. These are good examples that coastal ecosystems are often exposed to multiple stressors and robust management frameworks are required to consider the presence of multiple physical and chemical stressors.

A3. What has been the pace and trajectory of change in this attribute, and what do we expect in the future 10 - 30 years under the status quo? Are impacts reversible or irreversible (within a generation)?

The status quo would result in the continuous accumulation of metals in the environment as they are not biodegradable. Worldwide, in addition to the issue of anthropogenic zinc contamination in urban areas, contamination of soils with zinc has increased in some agricultural sectors, such as dairy farming and horticulture. Another poorly managed key source is the of run-off from stormwater, which can contain complex mixtures of industrial chemicals, pharmaceuticals, metals and nutrients. Metals are transported into waterways via stormwater from roads (zinc from tyre wear, copper from brake pad wear); roofs (zinc from galvanised roofing); and other impervious surfaces (including paved areas around industrial sites) [11]. A recent study of water quality in urban streams indicated that if urban development continues in its current form, increases in urban land cover around New Zealand can be expected to result in further declines in water quality at impacted locations [5]. Current chemical stressors combined with the significant impacts of legacy metals remain a concern for water quality, e.g., like in the Sydney Harbour [12]. The analysis of a range of parameters including dissolved zinc indicates that if urban development continues in its current trend, increases in urban land cover around New Zealand can be expected to result in further declines in water quality and a reduced likelihood that water quality objectives will be achieved at impacted locations [5].

There is evidence that better management of trace metal sources can reverse the trends. Better waste management like depositing treated wastewaters offshore and the introduction of more articulate environmental protection laws, regulations and enforcement can lead to improved water quality [12]. For instance, the global phase-out of leaded petrol use has contributed to the decline of concentrations in the ocean [13].

A4-(i) What monitoring is currently done and how is it reported? (e.g., is there a standard, and how consistently is it used, who is monitoring for what purpose)? Is there a consensus on the most appropriate measurement method?

Some metals are monitored as part of the State of Environment (SoE) reporting. Councils are conducting routine analyses for the occurrence and trends of metals for coasts, rivers and groundwaters as part of SoE monitoring and to meet consent condition requirements [14]. The SoE monitoring by regional councils focuses on a set of metals as reported in the recent Parliamentary Commissioner for the Environment (PCE) report on regulating the environmental fate of chemicals [15]. Monitoring of metal residues in relation to determining compliance with consent conditions is also often conducted for landfill leachate, wastewater and stormwater discharges [14].

The Australia and New Zealand Guidelines for fresh and marine water quality are the key tools to help planners, regulators and researchers to manage the quality of our water in New Zealand[1]. They provide default guideline values (DGVs) for all metals. These DGVs are jointly developed by the Australian and New Zealand governments.

A4-(ii) Are there any implementation issues such as accessing privately owned land to collect repeat samples for regulatory informing purposes?

Regional Councils have selected sites where they monitor trends in trace metals for State of Environment reporting. Consent holders would also have access to sites for monitoring as part of their consent conditions.

A4-(iii) What are the costs associated with monitoring the attribute? This includes up-front costs to set up for monitoring (e.g., purchase of equipment) and on-going operational costs (e.g., analysis of samples).

The analytical methods using inductively coupled plasma mass spectrometry (ICP-MS) instruments can measure elements and metals and are well established and validated. Several commercial laboratories including Hill Laboratories and AsureQuality can measure metals at competitive prices.

A Jacobs investigation reported limitations that councils have identified that preventing the expansion of current monitoring programmes including the high costs for both laboratory analysis and council staff time spent doing monitoring and reporting [14]. However, it should be noted that consent holders cover agreed conditions monitoring costs.

A5. Are there examples of this being monitored by Iwi/Māori? If so, by who and how?

We are not aware of any heavy metals in water monitoring being regularly undertaken by iwi/hapū/rūnanga. Resourcing is difficult for iwi/hapū/rūnanga to obtain, and any monitoring by agencies is generally infrequent, inconsistent, and ad hoc, and most programmes fail to provide information on whether chemical contaminants will have impacts of concern to Māori [32]. Some of the environmental assessment frameworks being developed by/with iwi/hapū/rūnanga include “safe to eat” or “safe to swim” outcomes [33-35]. Data/indicators required to fully realise these holistic cultural assessment frameworks will require information about heavy metals in water, sediment, and/or mahinga kai species. See also [16-19].

A6. Are there known correlations or relationships between this attribute and other attribute(s), and what are the nature of these relationships?

Contaminants are mostly found as complex mixtures of which metals are one family of pollutants at impacted sites. The issue of multiple stressors relates to the range of sources that put pressure on the receiving environment – e.g., stormwater and wastewater contain a range of other types of

[1] https://www.waterquality.gov.au/anz-guidelines

contaminants. Cumulative effects, through additional new marine industries, climate change and other stressors, can reduce environmental resilience and increase the risk of environmental or economic collapse [12]. The importance of sediments as stressors will depend on site ecosystem attributes and the magnitude and preponderance of co-occurring stressors [20]. Therefore, management of coastal waters must contend with multiple drivers in concert as the coordination of regulating agencies for urban and agricultural runoff is warranted as metals are only one component within a range of other contaminants that can accumulate in sediment [8].

Part B—Current state and allocation options

B1. What is the current state of the attribute?

The ecotoxicological effects of metals and their speciation under a range of environmental conditions are well understood and documented (as per references cited above). The key anthropogenic sources are well characterised to assist the management of these contaminants. The main challenge is that the management of metals requires a holistic/system approach as there are multiple factors to consider. For instance, roof material often contains zinc that can leach overtime. Some effort is required to find alternative types of material with less impacts. This needs to be underpinned by appropriate policy. For instance, the global phase-out of leaded petrol use has contributed to the decline of concentrations in the ocean [13].

B2. Are there known natural reference states described for New Zealand that could inform management or allocation options?

Finding reference sites with low levels of anthropogenic pressure is important to provide a baseline to confirm adverse impacts of metals and other stressors on receiving ecosystems. However, it is very difficult and nearly impossible to find reference sites that experience no anthropogenic pressure.

The hazards of metals and their mechanisms of toxicity have been extensively characterised using model test species under controlled laboratory conditions. The data generated is used to derive the default guideline values (DGVs) which provide threshold values over which adverse impacts are expected. A metal concentration above a DGV should trigger further investigations to fully assess the impacts of the metal on the receiving ecosystem. This is where having good baseline values of what a healthy ecosystem looks like is important. There are options to compensate for the lack of proper reference sites by monitoring across a gradient of stressors.

B3. Are there any existing numeric or narrative bands described for this attribute? Are there any levels used in other jurisdictions that could inform bands? (e.g., US EPA, Biodiversity Convention, ANZECC, Regional Council set limit)

There are well established default guideline values (DGVs) for several metals that have recently been reviewed by the Australian and New Zealand Guidelines for Fresh and Marine Water Quality [1]. These threshold values cover a range of protection levels of 80, 90, 95 and 99 % relevant to the particular ecosystem of interest, e.g., from industrial areas to national park and reserve areas.

[1] www.waterquality.gov.au/anz-guidelines

B4. Are there any known thresholds or tipping points that relate to specific effects on ecological integrity or human health?

Yes, there are threshold value guidelines available. The ANZG DGVs have been developed to provide threshold values for metals and other contaminants. They are set to provide a range of protection as per point B3.

B5. Are there lag times and legacy effects? What are the nature of these and how do they impact state and trend assessment? Furthermore, are there any naturally occurring processes, including long-term cycles, that may influence the state and trend assessments?

As discussed in the above sections, metals have multiple anthropogenic sources and they can continue to accumulate in various environmental compartments including surface water, groundwater, and coastal waters due to the non-degradability of metals.

Natural background levels of metals in lakes and rivers may vary widely because of differences in local geology, and the aquatic organisms that live there tend to be genetically adapted to the local levels of metals. This adaptation is described as the "metalloregion concept" [8]. This is particularly relevant to New Zealand where levels of some metals in the environment is associated with our unique soil and volcanic activity. For instance, in the central North Island, arsenic is released from geothermal systems into the Waikato River [21]. The receiving ecosystems will have adapted to higher background levels.

B6. What tikanga Māori and mātauranga Māori could inform bands or allocation options? How? For example, by contributing to defining minimally disturbed conditions, or unacceptable degradation.

It has been recognised that indigenous peoples, knowledge frameworks, and values are critical in orienting international efforts for the management of chemicals and waste that are more sustainable and equitable for all [22]. A high standard of environmental quality is an outcome sought by iwi/hapū/rūnanga. There is tikanga and mātauranga Māori relevant to informing bands, allocation options, minimally disturbed conditions and/or unacceptable degradation residing in treaty settlements, catchment/species restoration strategies, cultural impact assessments, environment court submissions, iwi environmental management plans, reports, etc.

There are one-off-studies where iwi/hapū/rūnanga are influencing research initiatives exploring the state and impacts of environmental contaminants (including heavy metals) on the outcomes they are seeking (e.g., mauri is protected, kai is safe to eat, water is safe to swim) (e.g., [36-38]).

See also [16-19].

Part C—Management levers and context

C1. What is the relationship between the state of the environment and stresses on that state? Can this relationship be quantified?

The SoE reporting for MfE highlights the level of environmental degradation in both freshwater and marine domains [23,24]. Metals are one of the multiple stressors that have been identified with sources including stormwater, municipal treated wastewater and agricultural discharges. For instance, cadmium, chromium, copper, nickel, lead, platinum and zinc are on the list of selected stormwater priority pollutants [25].

The toxicity and ecotoxicity of individual metals are well characterised and understood. Predicting or assessing the environmental impacts of an individual chemical is a challenge in a field situation as contaminants are often found in complex mixtures. For instance, exposure to low levels of multiple chemicals in mixtures can cause toxicity at concentrations where exposure to an individual chemical might cause no effect based on their DGVs. This is because multiple physiological processes may be affected by chemicals having different mechanisms of toxicity. This is a strong argument for the need of a system approach to the management of aquatic systems.

C2. Are there interventions/mechanisms being used to affect this attribute? What evidence is there to show that they are/are not being implemented and being effective?

C2-(i). Local government driven and C2-(ii). Central government driven and C2-(v). Internationally driven

The Australia New Zealand guidelines for fresh & marine water quality trigger values are designed to lead to further specific site risk assessment. In a recent survey on the type and range of chemical contaminants councils do, the emphasis was on the type of chemicals, but the implications of exceedance of DGVs was not assessed [14]. I am not aware of any follow up studies in New Zealand in response to a DGV exceedance.

C2-(iii). Iwi/hapū driven and C2-(iv). NGO, community driven

We are not aware of interventions/mechanisms being used by NGOs or iwi/hapū/rūnanga to directly affect this attribute.

Part D—Impact analysis

D1. What would be the environmental/human health impacts of not managing this attribute?

A business-as-usual scenario would lead to on-going increase of metals in sediment and have devastating impacts on exposed ecosystems. There is no doubt that the accumulation of anthropogenic pollutants in the environment is causing harm and scientists need to work with other stakeholders to reduce pollution [26]. Metals are not degradable so any continuous discharges will accumulate into the various environmental compartments and biota. The impacts of human activities have pushed estuarine and coastal ecosystems far from their historical baseline of rich, diverse, and productive ecosystems [27]. Managing the sources is a priority to ensure the protection of these valuable ecosystems. However, there are examples of declining metal concentrations from improved environmental controls on emissions and discharges of metals and other contaminants, e.g., [28].

There are multiple challenges to reduce the discharge of metals in urban environments, particularly non-point sources like stormwater. There are examples of options to reduce metals at the sources summarised in the PCE report. For instance, the challenge of an initiative to impose restrictions on the maximum amount of zinc in galvanised or zinc coated roofing materials [15].

D2. Where and on who would the economic impacts likely be felt? (e.g., Horticulture in Hawke's Bay, Electricity generation, Housing availability and supply in Auckland)

Coastal and ocean ecosystems provide commercial, cultural, recreational and economic benefits as well as support diverse habitats and species of local and global significance [12]. It is well-recognized that healthy and thriving coastal and freshwater ecosystems are essential for economic growth and food production [12]. The key impacts from the pressure that metals place on receiving environments is the potential loss in biodiversity and disruption of ecosystem functions and services through shifts in distributions of key species. Fishery and aquaculture industries are most likely to be impacted by pressure from metal contamination. Healthy and functional ecosystems and healthy fish stocks are important for the fisheries industry [23]. There are other aspects to consider including natural beauty of our estuaries, coastal and open ocean areas that are central to our culture and national identity.

D3. How will this attribute be affected by climate change? What will that require in terms of management response to mitigate this?

Climate change will have multiple effects in modulating the accumulation and bioavailability of metals. Climate change increasingly affects the variation in volume and frequency of stormwater events and runoff which can increase resuspension and direct exposure of sediments in water bodies [29]. The key concern with the effects of climate change on the risks associated with metal contamination is that changes to temperature and pH can modulate the speciation of metals or basically, their bioavailability. The importance of metal speciation cannot be overstated as it modulates the bioavailability and toxicology of trace metals. The simplest feature of speciation is whether the metal is in the dissolved or particulate form. Originally, environmental regulations were based on total metals present in the water as assayed by hot acid digestion of the samples. However, there has been a gradual change in many jurisdictions to regulations based on the dissolved component only. This reflects the general recognition that particulate metals exhibit negligible toxicity and bioavailability to aquatic organisms relative to dissolved metals [1]. Increases in temperature were correlated with increasing toxicity of metals to aquatic organisms [30]. As such, temperature should be accounted in risk assessment, because it may modify the effects of chemicals on the structure and functioning of aquatic communities, especially at higher levels of biological organization [31].

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9.10 Underwater noise / ocean sound

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State of Knowledge of the "Underwater noise / ocean sound" attribute: Good / established but incomplete – general agreement, but limited data/studies

Part A—Attribute and method

A1. How does the attribute relate to ecological integrity or human health?

Underneath the ocean's surface is an environment filled with noise generated from natural sources (i.e., plate tectonics, under sea volcanoes, hydrothermal vents), climatic events (i.e., ice, waves, storms, rain, wind) and marine fauna undertaking every day biological activities (i.e., communication, orientation, foraging, predator interactions, reproduction) [1-4]. As sound propagates differently in water compared to air, underwater noise can be detected several kilometres from the source, much further and faster than would be possible with vision or other senses [1]. However, underwater noise propagation is also complex in that transmission is affected differently in deep water versus shallow water, in cold water versus more temperate waters, in channels or canyons versus flat, homogenous seafloors, and in silty versus clear water [1, 3].

The efficiency of sound transmission through water means most marine fauna use underwater noise as their primary sense for most aspects of their lives [1- 4]. For example, several invertebrate and fish species rely on natural underwater sound cues from reefs or rocky shores to guide larval stages to suitable habitats for settlement. Other species like blue whales communicate with conspecifics across whole ocean basins using low frequency underwater vocalisations. The range of hearing capabilities in marine animals dictates their potential responses to different underwater sounds while making them vulnerability to impacts from different sources of underwater noise [2]. For these reasons, changes in ambient (or background) noise or particular sound frequencies can be a hinderance for marine fauna that are reliant on sound for survival. A sound can only be detected if the received level of the sound is equal or exceeds a detection threshold, usually the ambient noise level [1-4]. Hence, a health ecosystem is dependent on its organisms being able to detect and react to important underwater sounds [4].

Marine fauna likely cope with naturally occurring large , but short duration, variations in ambient noise levels and the distances over which sound is effective [3]. However, elevated ambient noise levels caused by an increase in anthropogenically generated underwater noise can prevent or interfere with the detection of sounds important to marine fauna. Termed underwater noise pollution, the detrimental effects (acute) of increased human-generated underwater noise on the marine environment is a well-studied but more data is needed on longer-term chronic effects [1-5].

A2. What is the evidence of impact on (a) ecological integrity ~~or (b) human health~~? What is the spatial extent and magnitude of degradation?

Strong international evidence recognises the increasing adverse effects that underwater noise levels are having on marine resources and ecosystems (ecological integrity) [2-4]. Based on this growing evidence, anthropogenic underwater noise is now recognised as a concern by international organisations, industries and regulatory agencies around the world [6-11].

Adverse effects to marine fauna associated with increases in underwater noise include reduced detection, behavioural responses (e.g., changes in surfacing or diving patterns), auditory masking (e.g., interruptions in type or timing of vocalisations) and possible auditory injury (e.g., auditory threshold shifts and stress) [2-5, 12]. Acute effects that are associated with high impact sounds (i.e., seismic surveys, pile driving, underwater explosions) can have immediate effects on nearby individual animals over limited distances (frequency dependent). More chronic effects from less intense, wide-spread sounds of longer duration (primarily shipping traffic) can affect individuals as well as populations. Known as noise-dependent or physiological stress, research suggest this latter effect is the greater impact of underwater noise pollution as it can lead to negative consequences for whole ecosystems [2, 4, 13].

A3. What has been the pace and trajectory of change in this attribute, and what do we expect in the future 10 - 30 years under the status quo? Are impacts reversible or irreversible (within a generation)?

Most international literature reports that the ocean has become louder by an average of 3-4 dB* per decade since the 1960 [2-5), although recent research suggests the current trend may be slowing to 1-2 dB per decade. This increase is attributed mainly to an exponential increase in maritime traffic since the end of World War II [14-16], specifically commercial shipping, which accounts for up to 90% of current internationally traded goods [17], but also includes fishing, military fleets, tourism and transport fleets.

Future estimates suggest that with the current rate of growth in ship traffic and economic trading, ambient noise is projected to continue to rise globally [16]. However, several international initiatives are forcing rapid developments towards quieter, more efficient propulsion technology such as electric or hybrid systems for new ships (see question C2(v)) [18]. At the same time, noise abatement technology (i.e., bubble curtains) and noise threshold limits for construction are continuing to be refined and implemented in the United States and several European countries to mitigate and / or manage other anthropogenic noise pollution [13, 19].

The Covid-19 pandemic gave us a rare opportunity to study the impact of reducing shipping and recreational traffic on the global soundscape [20-21]. While the results of this relatively short-term 'experiment' are not consistent [21], the data demonstrate that when underwater noise pollution is decreased, listening and communication distances are immediately improved [20]. For example, in some parts of the ocean, a vocalising whale would have been audible twice as far away during the pandemic relative to 2019. Whether the wider biological and ecological consequences of long-term, elevated underwater soundscapes are as reversible at the individual or population level has yet to be fully tested.

* It is important to note that the decibel scale is logarithmic, which means an increase of 3 dB represents a doubling of intensity.

A4-(i) What monitoring is currently done and how is it reported? (e.g., is there a standard, and how consistently is it used, who is monitoring for what purpose)? Is there a consensus on the most appropriate measurement method?

As far as we are aware, there is no routine or long-term monitoring of underwater noise levels within Aotearoa New Zealand waters. The majority of monitoring done with Aotearoa New Zealand waters is highly localised (i.e., specific port or bay) and short-term (i.e., several months, very few more than one year) with most underwater noise monitoring undertaken by industry for resource consent or RMA applications and / or university student projects.

Several international governments and regulatory agencies are continuing to research, review and revise appropriate standards and methods (including units) for measuring a variety of different underwater noise components; general soundscape levels, ship noise limits, and adverse hearing and behavioural threshold limits for marine fauna [13, 19, 22-24]. There are currently no agreed upon national guidelines or standards used for underwater noise in Aotearoa New Zealand. The exception is a section in the Auckland Unitary Plan that has policies relating to the management of underwater noise from high-impact construction activities (i.e., pile-driving or blasting) and its effects on marine mammals [25]. Most Aotearoa New Zealand ports undertaking infrastructure upgrades are currently voluntarily adhering to the United States' NOAA standards for pile-driving and construction activities as part of their resource consent condition requirements [19, 22-23].

A4-(ii) Are there any implementation issues such as accessing privately owned land to collect repeat samples for regulatory informing purposes?

Aotearoa New Zealand presently has several real-time monitoring buoys operating throughout coastal waters [26-28], however, none of these current systems monitor underwater noise levels. While several real-time, telemetered systems have been designed and successfully trialled internationally and nationally, the cost and complexity of sending large amounts of continuous digital underwater noise data (live or stored) over months or years make this method presently unaffordable. Hence, the greatest issue restricting wide-spread monitoring is the cost of sending and storing large data files.

Another consideration when placing and leaving scientific recording gear in the marine environment (for short or long periods of time) is that it always involves a moderate level of risk, potential for loss of gear (and the data) either due to natural causes (i.e., water leak, mooring shifted or lost in storm) or human-related ones (i.e., trawled by commercial or moved, stolen by recreational fisheries).

Finally and perhaps most importantly, Aotearoa New Zealand currently lacks trained and experienced acousticians to process and analyse underwater noise data to the expected international standards.

A4-(iii) What are the costs associated with monitoring the attribute? This includes up-front costs to set up for monitoring (e.g., purchase of equipment) and on-going operational costs (e.g., analysis of samples).

Standard underwater noise monitoring in Aotearoa New Zealand is currently undertaken using a stand-alone, purpose-built mooring with 1 or 2 passive acoustic recording devices (i.e., noise levels are recorded and stored) attached close to the seafloor. Designed and built in Aotearoa, the *SoundTrap* recorder costs between $4,500 and $5,500 USD. With the cost of a suitably insulated frame to hold the recorder and mooring floats, ropes and anchor gear, an initial deployment will cost a total of approximately $10,000 to $15,000 NZD (depending on location and recovery needs). Autonomous recorders, like *SoundTraps*, can record and archive approximately 3-6 months of underwater noise data using SD cards, depending on the location and duty cycle.

As a single recorder can store up to 5TB of audio data over one deployment, AI-aided software is a necessity in auditing the data for the various sounds of interest. Even with purpose-built algorithms for automated auditing and processing, each day of data collected can take between three and four days to process and analyses on a consumer-grade computer. Parallelisation and GPU arrays can substantially improve this processing time. Data processing costs are in addition to the hardware.

A real-time option for monitoring underwater noise includes placing an integrated hydrophone onto an existing coastal monitoring buoy. A new purpose-built monitoring buoy costs approximately $30,000 to $200,000 (depending on size, processing, storage and sending capabilities). However, integrating and sending noise data can be costly depending on the type and intervals of data collected and the level of on-board processing needed. Often edge processing and AI is used, whereby the acoustic data are processed inside the buoy itself, transmitting only small data payloads containing detection data as they occur, or sound pressure level statistics over predefined time periods. These payloads are transmitted to the cloud, where further processing can occur as required.

An alternative to moored devices would be a cabled hydrophone (or hydrophone array) from a land-based station. This setup would only be applicable at locations in which the hydrophone could be regularly supervised and maintained, and in which the hydrophone was stationary and well-protected but deep enough to avoid noise contamination (i.e., rocky shore nearby). Cable laying can also be expensive and require relays when the cable exceeds 700-750m to maintain the integrity of the hydrophone's data cable. With the on-board processing occurring on land, cabled systems can have the advantage of being powered via mains supply or ethernet in some circumstances, which permits long-term, even permanent, placements.

A5. Are there examples of this being monitored by Iwi/Māori? If so, by who and how?

We are not aware of any specific underwater sound related monitoring being carried out by representatives of iwi/hapū/rūnanga.

There are mātanga moana (Māori expertise in marine knowledge) who have long monitored tohorā (whales) and their wider ecosystems and are just beginning to engage with science researchers on local projects. Collaboration and partnership in this space would lend itself to understanding mātauranga and the range of variables that mātanga incorporate in their assessment of estuarine and coastal health.

A6. Are there known correlations or relationships between this attribute and other attribute(s), and what are the nature of these relationships?

The state of water quality (i.e., turbidity, type of sediments) can have minimal or indirect effects on underwater sound. However, such factors mainly affect the speed of sound rather than pressure levels.

Part B—Current state and allocation options

B1. What is the current state of the attribute?

The current state of underwater noise levels in Aotearoa New Zealand is only partially understood at a national scale. Sporadic sampling has been undertaken within most of our ports and industrial coastal areas, which allows for relative comparisons of current underwater noise levels [20, 29]. Additional underwater noise data from protected bays to open coastal zones also exist through one-off studies by different universities or institutes [30]. However, as there are no national or regulatory requirements for the collection of underwater noise data, there is no coordinated, standardised data collections or network of existing monitoring efforts.

We are aware that the Royal New Zealand navy has an array of hydrophones off Great Barrier Island, starting in 1961 [31-32], to monitor sounds and movements in eastern, North Island waters. To the best of our knowledge, these data are not available for public use, but could represent an important long-term monitoring database that would help quantify the current state of underwater noise levels in our northern coastal waters.

B2. Are there known natural reference states described for New Zealand that could inform management or allocation options?

There are no currently described reference state(s) for underwater noise levels in Aotearoa New Zealand coastal waters, with the possible exception of the inaccessible, long-term naval monitoring database. However, it is possible that underwater noise data exists on representative habitats within our coastal waters (i.e., isolated fiords, rocky shore reefs, surf breaks) that represent a natural state [30]. The Covid lockdown gave us the opportunity to glimpse what the 'natural soundscape' of some of our ports (Auckland, Lyttelton, and Picton) might have been previously relative to recorded noise levels both before and post-lockdown [20, 33].

B3. Are there any existing numeric or narrative bands described for this attribute? Are there any levels used in other jurisdictions that could inform bands? (e.g., US EPA, Biodiversity Convention, ANZECC, Regional Council set limit)

There are currently no attribute bands, standards, or thresholds for underwear noise levels in Aotearoa New Zealand. The two exceptions are narrative management objectives and policies in the Auckland Unitary Plan for high-impact construction activities (i.e., pile-driving or blasting) and its effects on marine fauna [25]. The Department of Conservation has an industry wide voluntarily code of conduct to minimise any effects of seismic operation noise on marine mammals in Aotearoa water [34].

There are several overseas narrative standards and numeric thresholds for assessing potential impacts of different underwater noise components that have been developed by international organisations and regulatory agencies [13, 19, 22-24]. High-impact or impulsive noise (i.e., pile driving, seismic surveys) tend to be discrete and are generally controlled through a consent or regulatory process. For instance, to determine at what distance predicted noise levels could cause any physical impairment or injury (i.e., permanent or temporary hearing threshold shifts) to marine mammal species, the United States' National Oceanic and Atmospheric Administration (NOAA) developed relevant underwater acoustic thresholds based on established functional hearing groups to distinguish between different marine mammal species [19]. Appropriate sound level thresholds for behavioural disturbance of marine mammals from high-impact noise sources are currently being assessed and revised overseas [22-23]. In the interim, a two-tiered approach in which a lower behavioural responses threshold is used for impulse noise levels with more moderate responses at higher sound levels of all species are being used overseas studies [24, 35].

Numeric hearing thresholds are also available for non-impulsive sounds, but usually applied to similar construction activities (i.e., dredging, increase in construction traffic) rather than regular shipping traffic. Behavioural response and auditory masking ranges are based on a continuous noise approach known as dose-response curves [22-24, 35]. This approach estimates the probability of a response occurring at different noise levels (i.e., distances from the source) and can be species-specific where data are available. The only legislation globally that directly addresses chronic underwater noise pollution (primarily shipping) and requires that noises levels do not adversely affect marine ecosystems is the European Union's Descriptor 11 of the Marine Strategy Framework Directive (MSFD) developed in 2008 [13]. In 2021, the EU added a narrative action to reduce underwater noise pollution in its waters. A preliminary indicator for this initiative aims at tracking low frequency ambient noise level using annual average sound levels across three different frequency bands (63Hz, 125Hz and 2000Hz bands) within a specified affected area [13, 36].

B4. Are there any known thresholds or tipping points that relate to specific effects on ecological integrity ~~or human health~~?

As noted in answer to B3, underwater noise thresholds have been developed overseas for several marine fauna (particularly marine mammals and some fish) based on their hearing and vocalisation capabilities. Thresholds have been developed mainly for two types of noise - continuous and impulse. Thresholds are provided to protect marine fauna against permanent (PTS; i.e., injury, mortality) and temporary (TTS: i.e., injury, discomfort) hearing shifts, which are considered more acute and relate to discrete, high-impact noises (i.e., pile driving, seismic surveys). But both PTS and TTS thresholds have been developed for continuous noises (i.e., dredging) as well. Several preliminary behavioural responses and masking thresholds have been proposed, but they are not yet species-specific, instead focusing on dose-exposure risk. Internationally, there are currently no agreed upon tipping points or thresholds for underwater noise pollution levels that distinguish a health ecosystem from an impacted one. However, international and national research is underway into the use of acoustic indices as proxies for monitoring marine biodiversity with habitats [37-38] as well as sound impact mapping [39].

B5. Are there lag times and legacy effects? What are the nature of these and how do they impact state and trend assessment? Furthermore, are there any naturally occurring processes, including long-term cycles, that may influence the state and trend assessments?

In terms of high-impact underwater noise, hearing and physiological effects are generally immediate and site-dependent, similar to a point source effect. However, the chronic effects of longer-term construction projects and /or busy shipping areas on individuals and their populations are more difficult to assess due to potential generational lags (i.e., reproductive impacts) and natural variations in density / abundance of some marine fauna, masking survivorship or emigration impacts. In addition, marine fauna's lagged responses to the more pronounced effects of large-scale climate drivers (i.e., marine heatwaves) potentially conceal chronic noise pollution effects [29].

Potential legacy effects relate mainly to the pace at which underwater noise monitoring technology has evolved over the past two decades. Prior to the 2000s, sound files were still recorded from reels on to tapes and cassette, potentially affecting the sound quality. In addition, historical noise data was limited to low and medium frequencies, again, due to technological limits. Hence, the use of historical datasets can involve lots of complex adjustments and calibrations.

B6. What tikanga Māori and mātauranga Māori could inform bands or allocation options? How? For example, by contributing to defining minimally disturbed conditions, or unacceptable degradation.

Based on our limited knowledge, we understand mātauranga Māori considers sound to be an important connector between the land and the water. For example, species on land (e.g., kauri trees) share a connection to a species in the water (e.g., parāoa - sperm whales) through sounds /songs.

Part C—Management levers and context

C1. What is the relationship between the state of the environment and stresses on that state? Can this relationship be quantified?

The relationships between underwater noise levels and natural sources as well as climatic and daily events are well studied with generalised spectra curves available [1]. However these sources, while very noisy at times, are transient, lasting minutes to days. The propagation of underwater noise generated from discrete anthropogenic activities (such as pile-driving and seismic surveys) have also been well documented, and with location-specific data (i.e., seafloor sediment composition, depth, stratification and temperature), can be modelled and predicted [19, 22-23]. Hence, why there are several applicable guidelines and thresholds for these types of activities (see B3).

In relation to marine shipping traffic, underwater noise levels increase linearly with an increase in the number of ship present, noting however that decibels are logarithmic. However, this relationship is not always clear as not all ships are the same, the increase in general noise pollution along a busy port or channel will vary across different frequencies depending on the size, weight and type of population of the ships present. The impact of shipping noise is continuous and travels over long distances, at the same time noise levels also change as individual marine ships come and go. As a result, annual average sound pressure levels are proposed to be considered against a representative condition (i.e., 'good noise' year based on long-term data) with the aim of an overall spatial reduction percentage for a particular area i.e., Hauraki Gulf) [13,39].

C2. Are there interventions/mechanisms being used to affect this attribute? What evidence is there to show that they are/are not being implemented and being effective?

C2-(i). Local government driven

The Auckland Unitary Plan is the only local government example of policies aimed at limit or reducing underwater noise levels in relation to high-impact activities on marine fauna. However, no specific interventions are recommended or required [25].

Most underwater noise management or mitigation measures come through resource consent conditions, either offered by the client or enforced through the local regulator. As a result, this process provides considerable inconsistencies and uncertainty for both operators and regulators. To our knowledge, only one port infrastructure project has monitored, analysed and reviewed the efficacy of their underwater noise intervention and mitigation measures in regard to marine mammals [29]. This work found that enforced shutdowns based on TTS, requirements for qualified observers, and daylight limits for pile driving helped reduce shorter-term impacts. The review also recommended additional measures were warranted to reduce noise production at the source (e.g., bubble curtains). The project was not able to conclusively determine if longer-term declines in dolphin detections were due solely to the construction projects or in conjunction with other climate factors (e.g., simultaneous marine heatwave).

C2-(ii). Central government driven

While section 16 of the RMA mentions noise, it is not specific to underwater noise and is rarely mentioned in resource consent cases.

The Department of Conservation’s national code of conduct to minimise effects of seismic operation noise on marine mammals has been reviewed and revised internationally to ensure efficacy [34].

C2-(iii). Iwi/hapū driven

We are not aware of interventions/mechanisms being used by iwi/hapū/rūnanga to directly affect this attribute.

C2-(iv). NGO, community driven

With advice from researchers, the shipping industry and Ports of Auckland developed a voluntary transit protocol to minimise Bryde’s whale collisions in the Hauraki Gulf region [40-41]. By limiting speed for all commercial ships travelling within the Gulf to 10 knots (and reducing noise generation at the same time), the estimated probability of a lethal ship strike with Bryde’s whales has reduced from 51% to 16% [42]. Unfortunately, the reduction in underwater noise levels was not quantified.

Bubble curtain technology to reduce noise levels generated from pile-driving was also trialled by KiwiRail for up-coming infrastructure upgrades at ferry terminals in both Wellington and Picton. Similar to overseas studies, preliminary results demonstrated large reduction in middle and higher frequency ranges, which overlap with several marine mammal species [43]. However, more work in needed to reduce lower frequency noise levels, ranges that affect whale and some fish species [4].

C2-(v). Internationally driven

In addition to the large amount of work being undertaken by international organisations to monitor, reduce and mitigate underwater noise as described in B3, other overseas ports have also implemented voluntary slow down protocols for shipping traffic in their areas [44-45] while passenger ferries servicing large cities are now actively monitoring for marine fauna [45].

Maersk Shipping and universities are researching how ship design changes affect underwater noise production while the company retrofitted one of their ship classes to carry more containers [46]. They found a reduction up to 5dB, likely due to changes in the propeller and bow design. The International Maritime Organization (IMO) recently revised the guidelines for the reduction of noise emissions from shipping, while also taking into account technical innovations and adaptations in shipbuilding [18].

Part D—Impact analysis

D1. What would be the environmental/human health impacts of not managing this attribute?

Changes in the attribute state affect ecological integrity as described in A1. Not managing underwater noise levels will lead to continued displacement of soniferous fish and mammal species from preferred coastal / inshore habitats and eventual avoidance by more migratory species, particular noise sensitive life stages of vertebrates and invertebrates [1-3]. For those species with restricted home-ranges and unable to move away from areas with increasing noise levels (i.e., ports, marina, shipping channels, oil / gas fields), increased chronic ecological stress at a regional and population level may eventually affect reproductive and survival capabilities [2-4]. Such impacts are greatest for non-migrating, taonga and indigenous species, such as Hector’s and Maui dolphins and southern right whales, but also important iconic Aotearoa New Zealand ecosystems / habitats.

D2. Where and on who would the economic impacts likely be felt? (e.g., Horticulture in Hawke’s Bay, Electricity generation, Housing availability and supply in Auckland)

Shipping is the largest contributor to underwater noise levels [1-5, 14-16] yet restricting commercial shipping speeds within Aotearoa New Zealand waters equates to longer shipping times and costs that are passed on to the industry, ports and eventually Aotearoa New Zealand taxpayers. However, as noted from Covid pandemic research, slowing shipping speeds and limiting recreational boats can have a significant reduction in the amount of underwater noise produced in our shipping channels [20]

Slowing shipping speeds and lowering noise levels also has an important secondary advantage of mitigating the risk of ship collision with whales and other large marine fauna [40]. The voluntary transit protocol to minimise Bryde’s whale collisions initiated in 2013 between the shipping industry and the Ports of Auckland for the Hauraki Gulf region is a noteworthy Aotearoa New Zealand example [41]. As discussed in answer to C2(v), other overseas ports have successfully undertaken similar initiatives to reduce shipping noise effects even in light of potential economic impacts [44-45].

D3. How will this attribute be affected by climate change? What will that require in terms of management response to mitigate this?

Overseas researchers [16, 49] trying to understand the potential effects that climate change might have on ocean noise levels note that the 2022 IPCC assessment of climate change impacts [48] does not acknowledge any impacts of climate change on the ocean soundscape. In general, climate driven increases in the frequency and intensity of storm events, windier and / or wetter seasons may generally increase ambient noise levels within most habitats, but these effects might be moderated by the countering effects of raising ocean temperatures and decreasing ocean pH, that tend to increase the speed at which underwater noise travels [2, 49-52]. Simulation modelling predicts that some northern oceans will be up to 7dB noisier while locations in the Pacific and Southern Oceans are expected to be quieter by the next century (based on the sound field of a single ship) [16]. Drivers of these effects are mainly stratification, and to a lesser extent absorption, due to the creation or disappearance of sound ducts that will affect sound speeds and propagation distances at various depths differently [16, 53].

Future changes in sound propagation have the potential to significantly affect those marine fauna that rely on specialised auditory systems, such as marine mammals, however, such implications have not been investigated. In addition, the performance of anthropogenic acoustic sensor systems, on which maritime organisations such as naval military depend, will also likely be substantially affected [54]. The only current mitigation or management actions to reduce such effects are tied to those associated with reducing climate change. However, an overall reduction in shipping traffic and any increased efficiencies in furthering noise reduction technologies will help slow or reduce these impacts.

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9.11 Extent of mud (broad scale attribute)

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Preamble: "Mud" refers to a cohesive mixture of deposited (benthic) material consisting of water, clay minerals, silt particles, and organic matter (biopolymers). "Sediment mud content" refers to the proportion of sediment particles (i.e., percent of a sediment sample's dry weight) ≤ 63 µm. In intertidal areas, muddy sediment can be identified by its lack of firmness underfoot (e.g., how far a person sinks into the sediment when walking or standing on it). Some researchers can identify sediments with high mud content by other means, for example, by the presence of indicator species (e.g., crabs) or their traces (e.g., burrows). Note, however, that sediment muddiness is difficult to assess accurately at broad scales, especially in submerged (subtidal) habitats and using drone or satellite imagery. The "Extent of mud" in an estuarine area may be defined as the proportional area where sediment mud content is greater than a particular threshold, e.g., 25% mud content. However, Mud Extent in an estuarine area will likely be determined through interpolation of point samples of mud content (as percent mud content is not easy to determine from drone/remotely-sensed imagery or by walking/observing an estuary). Interpolation accuracy depends on the number of sampling points and their spatial distribution across estuaries. This will need to be considered when comparing Mud Extent in different estuaries.

State of knowledge of "Extent of Mud (broad scale, estuarine)" attribute: Medium / unresolved – some studies/data but conclusions do not agree

In soft-sediment habitats, sediment mud content is widely and well understood to affect many sediment parameters, including macroinvertebrate-based estuarine health metrics and human use/amenity values (such as walkability, firmness, and odour). Thus the attribute is likely important. However, knowledge of how differences in "extent of mud" relate to estuarine ecological integrity is medium / unresolved. Obtaining accurate measurements of the extent of muddy habitat at estuarine scales is challenging. Therefore, relationships between this attribute and ecological integrity remain unclear.

Part A—Attribute and method

A1. How does the attribute relate to ecological integrity or human health?

Human Heath: The "Extent of mud (broad scale, estuarine)" attribute—hereafter Mud Extent—does not relate to Human Health. A possible exception is that food (e.g., shellfish) gathered from muddy sites may have higher concentrations of contaminants than food collected from sandier sites, as trace metals and other anthropogenic contaminants often bind to fine sediments. Therefore, food safety may be worse in estuaries with high Mud Extent, although this would be better assessed directly using other attributes, e.g., 'Trace metals in sediment' or 'Trace metals in water/indicator species' (report sections 9.8 and 9.9).

Ecological Integrity: Mud content is inversely related to various measures of ecological integrity. Therefore, estuaries and coastal areas with greater Mud Extent are likely to be in a poorer state of health overall. Muddy sediments, particularly when mud content exceeds 10 to 25%, are associated with significantly reduced abundance, richness, and diversity of benthic macroinvertebrates [1-4]. Muddy sediments also have fewer large-sized suspension-feeding bivalves (pipi, cockles, scallops, horse mussels, green lipped mussels) that are key contributors to ecological integrity. Ecosystem functions (benthic primary production rates, nutrient regeneration) have been shown to be affected by experimental additions of mud [5,6] and to change across natural gradients of sediment muddiness [7-9]. The resuspension of muddy sediments by waves can affect water clarity, which can in turn affect feeding by visual predators such as fish and wading birds. In conclusion—based on numerous individual experiments and observations across mud content gradients [10-13]—estuaries with high Mud Extent are likely to have lower ecological integrity than estuaries with low Mud Extent. However, Mud Extent is difficult to measure accurately and the shape/slope of the overall negative relationship between Mud Extent and ecological integrity remains unclear.

A2. What is the evidence of impact on (a) ecological integrity or (b) human health? What is the spatial extent and magnitude of degradation?

Although the 'natural baseline' (pre-human) state of Mud Extent is unknown, it is widely accepted that Mud Extent has increased dramatically in estuaries throughout New Zealand in modern times. Humans have greatly elevated rates of fine-sediment input into estuaries through land-use practices (e.g., widespread deforestation). The most pronounced changes in fine sediment accumulation rates (and thus muddiness and Mud Extent) have occurred during the last 100 years. Sediment accumulation rates are presently estimated to be 10 to 100 times greater than natural [14]. Some harbours that were once navigable to ships are now highly infilled. The spatial extent of coastal mangroves in North Island estuaries has expanded markedly over the last 100 years (~4% per year since the 1940s), coincident with and possibly driven by increases in Mud Extent [15]. The expansion of Mud Extent in estuaries tends to proceed from head to mouth, i.e., muddiness increases first in upper estuarine areas where rivers are introducing sediments that have eroded from land. Later, as the estuary infills, Mud Extent may expand outward from the upper estuary tidal creeks into the main body of the estuary and towards the mouth.

A3. What has been the pace and trajectory of change in this attribute, and what do we expect in the future 10 - 30 years under the status quo? Are impacts reversible or irreversible (within a generation)?

Mud Extent has not been monitored or estimated in estuarine/coastal areas until relatively recently, however, it is safe to assume that Mud Extent has increased markedly over the last 150 years since the arrival of Europeans in Aotearoa New Zealand. Road building, urban expansion, and agricultural intensification during the last 30 years has likely contributed to the highest rates of increase in Mud Extent (though this is speculative because Mud Extent is not widely measured). Under the status quo, Mud Extent will continue to increase, especially with climate change expected to increase the loadings of terrigenous sediments to coastal receiving environments, driven by more frequent and higher intensity storms. However, with large-scale afforestation, riparian planting, and exclusion of livestock from river margins, the loading of new terrigenous sediment to estuarine/coastal areas is predicted to decrease, which would in turn allow Mud Extent to slowly decrease. Mud Extent may decrease fastest at sites and in estuaries with positive net sea level rise, as increased inundation of muddy fringing habitats may help to flush muddy sediments out of estuaries, though this is speculative and requires testing/verification.

One important point is that Mud Extent can increase rapidly following major storms. For example, there was evidence of muddy deposits covering wide extents in Auckland, Waikato (Coromandel), and Hawke's Bay estuaries following the passage of Cyclone Gabrielle (NIWA Hamilton Marine Ecology team pers. obs., 2023). These increases in Mud Extent were detected in the immediate post-event period, but the persistence of the muddy areas was not well studied.

A4-(i) What monitoring is currently done and how is it reported? (e.g., is there a standard, and how consistently is it used, who is monitoring for what purpose)? Is there a consensus on the most appropriate measurement method?

Mud Extent has been assessed in many South Island estuaries, and in a few North Island estuaries. Researchers at Wriggle and Salt Ecology have used a method called "Broad scale mapping", which is intended to provide a rapid overview of estuary condition based on visible features (e.g., aerial photographs), supported by ground-truthing to validate the visible features. The usual reporting metric for Mud Extent is "Soft Mud Percent Cover", defined as "percent of available intertidal habitat with >25% mud"[16-18].

It must be noted that mud content is not generally detectable from aerial photographs, therefore, on-the-ground observations of underfoot substrate firmness by field staff and grainsize samples are used for validation. For this method, "soft mud" is identified when an adult sinks 2-5 cm, and "very soft mud" is defined when an adult sinks >5 cm. Together they are called "total soft mud" [18], or possibly "mud-elevated substrate (>25% mud content)" [Leigh Stevens, Salt Ecology Ltd., pers. comm.]. However, correlations between "firmness" and grainsize are not always clear [19]. Furthermore, from a distance, proponents acknowledge that soft mud looks visually similar to firm muddy sand, firm sandy mud, firm mud, and very soft mud. This raises questions about standardisation and the ability to measure this metric accurately.

Other types of "Rapid Habitat Assessment" techniques have been undertaken in a number of Auckland and Waikato estuaries, although these assessments are focused on biotic habitats (defined by dominant species or biological features), with Mud Extent not typically assessed directly [20-23]. The forthcoming update of the National Estuarine Monitoring Protocol may provide guidance on a standard method or consistent technique for quantifying Mud Extent. However, Mud Extent is likely a difficult variable to measure accurately at the scale of whole estuaries—particularly when the estuaries are too large for researchers to cover on foot and when the estuaries have large proportions of subtidal habitat.

A4-(ii) Are there any implementation issues such as accessing privately owned land to collect repeat samples for regulatory informing purposes?

Assessing Mud Extent requires field work to access estuarine areas that are exposed at low tide. All parts of the estuary need to be assessed (not just specific sentinel sites), meaning that boats and multiple points of access from shore will likely be required to characterise Mud Extent. Permissions may be needed to gain access from private lands. Ancestral and sacred areas, such as areas near burial grounds, are likely to be off limits for assessments (possibly even by unmanned aerial drone). It is always advisable to communicate with mana whenua to understand access issues. In general, clear communication, good relationships, and addressing concerns or impacts to landowners' property or operations is necessary. Formal access agreements may need to be established in some cases.

Several health and safety indications need to be considered for fieldwork. Use of boats and kayaks generally requires health & safety training and Worksafe qualifications. Sinking into deep mud or traversing channels on incoming tides can be fatally hazardous if this risk is not managed.

A4-(iii) What are the costs associated with monitoring the attribute? This includes up-front costs to set up for monitoring (e.g., purchase of equipment) and on-going operational costs (e.g., analysis of samples).

Costs of assessing the Mud Extent attribute will vary widely depending on estuary size and morphology. Costs will be determined by the staff time required to safely and thoroughly assess all parts of an estuary. It may be possible to gather information from local people (and possibly enlist their help in surveys) to determine Mud Extent while limiting cost. In most cases, Mud Extent would be just one of many attributes measured (therefore, costs could be shared). It is possible that use of UAVs (aerial drones) will increase estuarine coverage and measurement accuracy, provided that standard protocols for operation and processing are developed. Use of drones will require upfront costs (drone purchase, training).

A5. Are there examples of this being monitored by Iwi/Māori? If so, by who and how?

We are unaware of any iwi or hapū that have assessed mud extent in estuaries in their rohe moana. It is highly likely, however, that knowledge about estuarine muddiness and extent is held by mana whenua and could facilitate quantification of this attribute. An assessment of co-developing appropriate indicators for estuarine mahinga kai was co-led with a rūnaka within the takiwā of Ngāi Tahu ki Murihiku, which highlighted the importance of sediment characteristics as key to mahinga kai ecosystems [56]. As such, an additional scope to the project had led to working with Environment Southland to incorporate a significant site for sediment monitoring as part of the annual assessment, and which whānau will additionally monitor at key periods to align with seasonal relevance [56].

A6. Are there known correlations or relationships between this attribute and other attribute(s), and what are the nature of these relationships?

Bed sediment muddiness is correlated with a range of other attributes, e.g., heavy metals in sediment (positively), water clarity/turbidity (positively), and macroinvertebrate community composition (negatively). However, Mud Extent—percent of available intertidal habitat with >25% soft mud—is probably most closely correlated with other estuary scale attributes such as mangrove extent and quality (positively), seagrass extent and quality (negatively), and shellfish bed extent and quality (negatively). While the correlated attributes should not necessarily be grouped, it may be possible for them to be assessed together (to save costs and provide a greater range of information).

Part B—Current state and allocation options

B1. What is the current state of the attribute?

Our understanding of Mud Extent attribute (broad scale in estuaries) is moderate at the National Scale. Mud Extent has been quantified in many South Island estuaries and a few lower North Island estuaries. Mud Extent has not, to our knowledge, been estimated in any Auckland, Waikato, or Northland estuaries. Where it has been assessed, questions remain about the accuracy/repeatability of the estimates. Mud extends into subtidal areas but the attribute is only assessed in intertidal habitats. Mud Extent is calculated as percent of available intertidal area *excluding saltmarsh*, but no guidance has been presented to date on whether to exclude mangroves (which are only distributed in central and upper North Island estuaries). Thus, in summary, there are gaps and potential issues in using this attribute as an estuarine health indicator. However, with guidance on how to standardise the assessment technique, and with the incorporation of new technologies such as aerial drones, the Mud Extent attribute has promise for use as a national indicator, particularly if it can be shown to drive or correlate with changes in other spatial attributes such as Shellfish extent/quality and Seagrass extent/quality.

B2. Are there known natural reference states described for New Zealand that could inform management or allocation options?

There are no known reference states for this attribute. Oral histories and sediment cores 20-50 cm deep (with deeper layers of sediment being progressively older) may be able to fill this information gap. It should be noted that some estuaries and parts of estuaries are likely to have been dominated by fine sediments / soft mud even in their natural state (e.g., for example in protected estuary arms and downstream of catchments with silt-clay dominated soils).

B3. Are there any existing numeric or narrative bands described for this attribute? Are there any levels used in other jurisdictions that could inform bands? (e.g., US EPA, Biodiversity Convention, ANZECC, Regional Council set limit)

The following bands have been circulated by researchers at Wriggle and Salt Ecology:

Very Good – Band A	Soft mud **<1%** of intertidal substrate outside of saltmarsh
Good – Band B	Soft mud **1-5%** of intertidal substrate outside of saltmarsh
Moderate – Band C	Soft mud **5-15%** of intertidal substrate outside of saltmarsh
Poor – Band D	Soft mud **>15%** of intertidal substrate outside of saltmarsh

The evidence/justification provided for the setting of these bands was:

> *"Soft Mud Percent Cover. Soft mud (>25% mud content) has been shown to result in a degraded macroinvertebrate community (Robertson et al. 2015, 2016), and excessive mud decreases*

water clarity, lowers biodiversity and affects aesthetics and access. Because estuaries are a sink for sediments, the presence of large areas of soft mud is likely to lead to major and detrimental ecological changes that could be very difficult to reverse. In particular, its presence indicates where changes in land management may be needed". [from 16]

Percent vs hectares. It should also be noted that, in large estuaries with hundreds of hectares of intertidal habitat, low "Soft Mud Percent Cover" values may equate to a relatively large areal extent (many hectares) of soft mud—a factor that may need to be considered when evaluating health (as it is for some other estuarine attributes).

B4. Are there any known thresholds or tipping points that relate to specific effects on ecological integrity or human health?

To our knowledge, there are no known examples of thresholds or tipping points for the Mud Extent attribute. The bands have been determined based on observations of experts who have worked in many estuaries (albeit mostly southern estuaries without mangroves). There do not appear to be any step changes or tipping points across the defined band boundaries.

B5. Are there lag times and legacy effects? What are the nature of these and how do they impact state and trend assessment? Furthermore, are there any naturally occurring processes, including long-term cycles, that may influence the state and trend assessments?

It has likely taken several decades for Mud Extent to build to its current levels. Contraction of Mud Extent will also likely take many decades. In other words, yes, historical legacies must be considered.

Reductions of catchment sediment loading rates are likely to first reduce sediment deposition/accretion rates, with bed sediment muddiness much slower to change. The speed and patterns of Mud Extent contraction will likely depend on local estuarine morphology (degree of exposure of estuarine areas to wind-waves and ocean flushing). The effects of mangroves or other biogenic structures that reduce wind-wave energy may need to be considered.

Mud Extent may decrease fastest at sites and in estuaries with positive net sea level rise, as increased inundation of muddy habitats may help to flush muddy sediments out of estuaries. However, sediment loading to estuaries is predicted to increase with climate change associated increases in rainfall intensity/frequency, thus mitigations on land are likely going to be required just to stop Mud Extent from expanding.

B6. What tikanga Māori and mātauranga Māori could inform bands or allocation options? How? For example, by contributing to defining minimally disturbed conditions, or unacceptable degradation.

Mātauranga Māori and extensive knowledge of estuarine spaces that have been handed down over generations is likely to be useful in defining the natural state of mud extent in estuaries. Iwi and hapū observations and knowledge of how muddiness (and other sediment characterisation) has changed over time along with other practices (e.g., shellfish collection) could help with the setting of bands. It is also highly likely that knowledge of the current state of estuarine muddiness and extent is held by mana whenua and could facilitate quantification of this attribute. Mana whenua should be asked to participate in estuarine surveys and information gathering as part of Mud Extent attribute evaluation and band refinement.

Part C—Management levers and context

C1. What is the relationship between the state of the environment and stresses on that state? Can this relationship be quantified?

With Mud Extent, relationships between environmental state and stressors are generally understood. Land-cover change (conversion of native forested hillslopes into pastureland or rotationally harvested pine) has greatly accelerated hillslope and streambank erosion, in turn leading to 10-100 fold increases in sediment accumulation rates estuaries [14]. Sediment-source tracing using compound specific stable isotope (CSSI) techniques has demonstrated the presence of terrigenous sediments with pine and streambank signatures in estuarine receiving environments [24]. Relationships between rainfall levels (river discharge volumes) and sediment loading rates are also relatively well understood, as are predictions of future loading under various climate change scenarios [25,26].

There are dozens of New Zealand studies demonstrating the deleterious effects of terrigenous fine sediments (mud) on biodiversity and ecosystem functioning in estuaries—so the consequences of the expansion of muddy habitats in our estuarine spaces is well understood [1-13,27-44].

The options for mitigating sediment loading to estuaries are known [25,26], and many of these options are already being exercised (e.g., exclusion of stock from stream and riverbanks; riparian planting; afforestation; sediment retention ponds/sumps at construction sites; controls on logging and road building).

Key knowledge gaps are knowing (1) where the greatest gains in sediment retention can be made with catchment mitigations and (2) how to evaluate the effectiveness of catchment mitigations given that positive responses in estuaries may take many years and be far from where mitigations were implemented. With the Mud Extent attribute, where quantification (identifying "soft mud" across whole estuaries) is problematic and where responses to mitigations are likely to be extremely slow/lagged, evaluating mitigation effectiveness will be extremely difficult.

C2. Are there interventions/mechanisms being used to affect this attribute? What evidence is there to show that they are/are not being implemented and being effective?

Many of the interventions/mechanisms for controlling the erosion and discharge of sediments from land to sea are collaborative and cross-cutting, i.e., involving iwi/hapū, local government, NGOs, and/or central government.

C2-(i). Local government driven

Most resource consents for land development (housing, construction, road building) that are considered by Councils require sediment controls to prevent entry of (especially) fine sediments into waterways. Councils can oppose plans that they think do not sufficiently protect against adverse effects. Lengthy consent timeframes (35 years) sometimes prevent local governments from mitigating potentially adverse effects (e.g., cannot prevent logging of steep coastal hillslopes if the activity was consented many years prior). Some of the Jobs for Nature initiatives (while Central government driven) are being implemented locally, but efficacy in terms of reducing Mud Extent is not yet known. Some of the 'local' initiatives are being undertaken on relatively large scales (e.g., the

$100m Kaipara Moana Remediation project; [45]). The Waikato River Authority funded riparian planting, and other councils have also likely undertaken similar initiatives.

C2-(ii). Central government driven

Central government agencies such as MfE and DOC have sediment-related strategies and plans [46,47]. Updates to MfE's National Policy Statement for Freshwater Management [48] and urging from PCE [49] have emphasised a more holistic and integrated land-to-sea approach to estuarine management. The Jobs for Nature programme (administered by five central government agencies) has directed hundreds of millions of dollars towards riparian planting in catchments to prevent sediments from entering freshwater and coastal receiving environments downstream. Fisheries New Zealand has funded studies on the effects of land-based stressors (including sediments) on coastal fisheries [40]. MPI funded a response to the major sediment loading events to the coastal zone off Hawke's Bay and Gisborne following Cyclone Gabrielle.

C2-(iii). Iwi/hapū driven

Iwi and hapū are aware of the deleterious effects of mud in estuaries and have reported changes in muddiness over time. Iwi and hapū have been heavily involved in Jobs for Nature projects across New Zealand. Iwi and hapū are also leading estuarine restoration initiatives in partnership with Councils and National Science Challenge researchers from various universities, CRIs, and other research institutes/providers [50,51]. Mana whenua have also co-led tangata whenua approaches to improve estuarine mahinga kai management, that includes the importance of sediment characteristics, and are important to developments of attribute decisions [56].

C2-(iv). NGO, community driven

There are catchment care groups and other local groups that are working on riparian planting and other sediment control measures. The Nature Conservancy is involved in catchment planting and restoration.

C2-(v). Internationally driven

We are unaware of any internationally driven initiatives that are specifically designed to address Mud Extent in estuarine/coastal ecosystems of New Zealand.

Part D—Impact analysis

D1. What would be the environmental/human health impacts of not managing this attribute?

Estuaries are vital to Aotearoa-NZ's socio-cultural identity and economy [49,52]. Many of Aotearoa-NZ's estuaries and coastal environments have been devastatingly degraded, in part due to the loading of terrigenous sediments entering via freshwater, which has greatly increased Mud Extent—impacting amenity values, decreasing native biodiversity, increasing invasions by non-indigenous species, affecting key ecosystem functions, and decreasing food for fish and bird species. If we do not manage this attribute, our estuaries are likely to continue to degrade (i.e., healthy estuarine area will shrink).

A range of culturally and commercially important shellfish species are found in estuaries, including mussels, scallops, cockles, and pipi. Increased Mud Extent and degraded estuarine health are barriers to iwi/hapū food security and wellbeing aspirations, and are likely to limit job and business opportunities for Māori and other New Zealanders (e.g., mussel and oyster aquaculture; scallop fisheries).

D2. Where and on who would the economic impacts likely be felt? (e.g., Horticulture in Hawke's Bay, Electricity generation, Housing availability and supply in Auckland)

The main areas of economic impact related to Mud Extent would be on customary fisheries, mussel and oyster aquaculture businesses*, on- and in-water tourism enterprises (such as kayak rentals, scuba charters, glass bottom boating), and potential new Blue Carbon initiatives. There is some suggestion that severe infilling and expansion of Mud Extent can affect navigation by sea into city centres (case in point Invercargill City). Investment in activities that reverse estuarine degradation is a potential economic growth area (i.e., development of Restorative Marine Economies).

*Note that oysters are relatively mud tolerant and note that commercial oysters and mussels are both cultured above the bed, not on it, so impacts on these activities from increased mud extent in the upper arms of estuaries may be small.

D3. How will this attribute be affected by climate change? What will that require in terms of management response to mitigate this?

Climate change is predicted to result in more frequent and higher intensity storms, which will likely elevate rates of terrigenous sediment input to estuaries [26], thus exacerbating issues of high Mud Extent. In contrast, sea level rise [53] may increasingly inundate currently muddy intertidal flats and potentially ameliorate high Mud Extent by resuspending mud from the edges of estuaries [54,55].

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9.12 Suspended sediment / water clarity / turbidity

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State of knowledge of the "Suspended sediment / water clarity / turbidity" attribute: Good / established but incomplete – general agreement, but limited data/studies

Section A—Attribute and method

A1. How does the attribute relate to ecological integrity or human health?

Suspended sediment, clarity and turbidity are strongly related entities/concepts [1]. Suspended particulate matter (SPM) in waters, usually measured as total suspended solids (TSS; g/m^3), is typically the dominant light attenuating constituent in coastal waters [2] and therefore controls water clarity (transmission of light through water) and turbidity (an index of light scattering [3]). Other major light-attenuating constituents in coastal waters, in common with natural waters more generally, include coloured dissolved organic matter (CDOM), phytoplanktonic algae, and (pure) water itself [2] [4].

There are two main aspects to water clarity, and both strongly affect ecological integrity of coastal and other waters [1]:

- Penetration of (diffuse) sunlight with depth in water, and
- Visual range in water (visual clarity; visibility).

Although broadly correlated (e.g., [2]) these two aspects relate to *different* 'types' of light attenuation such that light penetration cannot be accurately predicted from visual clarity or vice versa. Penetration of sunlight into waters is quantified by the down-welling irradiance (diffuse light, usually photosynthetically available radiation, PAR) attenuation coefficient [5] [6]

$$K_d(\mathrm{PAR}) = \ln(1/T_{\mathrm{PAR}})/\Delta z \qquad (1)$$

Where z is depth below the water surface and T_{PAR} is transmission of down-welling PAR, over the depth interval Δz. A useful index of light penetration is the euphotic depth, defined as the depth of the 1% PAR level, which is given approximately by $z_{eu} = 4.6/K_d(\mathrm{PAR})$.

Visual range in waters, for humans as well as fish, marine mammals and aquatic birds, is mainly controlled by the light beam (radiance) attenuation coefficient [7] [8]

$$c = \ln(1/T)/r \qquad (2)$$

in which light beam transmission is: $T = L/L_o$ where the incident light beam (radiance) is L_o and the beam is attenuated to L over path length r. Visual range is a complicated function of reflectivity and angular size of the visual target, underwater lighting and direction of viewing [8], but a very valuable (simple) index of underwater visibility is the sighting range of an (optically large) *black body* (with zero reflectance) in the *horizontal direction* (eliminating any dependence on change of illumination with depth). A practical approach is to observe a matte black disc, for which horizontal visual range underwater is: $y_{BD} = 4.8/c(550)$, where $c(550)$ is the light beam attenuation coefficient at 550 nm (near peak sensitivity of the human eye) [9]. The coefficient of 4.8, accounts for the contrast sensitivity of the human eye, and was confirmed to high accuracy by Zanevald & Pegau [8]. Black disc visibility has been measured in New Zealand for many years, for example since 1989 in rivers [10].

A traditional index of visual clarity is the Secchi depth (sighting range of a white or black-and-white) disc viewed vertically), which is less useful as a visibility index because it depends on underwater lighting as well as light beam attenuation [11]. Furthermore, Secchi observations can be very misleading in often depth-stratified coastal waters.

Sunlight penetration largely controls depth limits of aquatic plants and benthic algae in waters [5]. For example, keystone seagrasses, including the one New Zealand species, *Zostera muelleri*, are ‘light-demanding’ plants that can only grow with sufficient sunlight penetration to the seabed [e.g., 12, 13]. The controlling factor is then, *not* K_d(PAR)) as such but *optical depth*: $K_d(\text{PAR})*z$ (dim.) – which, from the definition in Eqn 1 above, corresponds to PAR at the bed being reduced to a particular fraction of that incident on the water surface (in unstratified water). Interestingly, fine sediment affects seagrasses *both* when suspended (by attenuating sunlight) and when deposited (by exerting an oxygen demand while also reducing oxygen transfer to the substrate from overlying water). These two stressor modes of fine sediment *interact* because seagrasses need sufficient light to generate oxygen fast enough to counter deoxygenation of muddy substrate near their roots [14].

Visual clarity strongly affects behaviour of sighted animals including fish and aquatic birds [7] Reactive distance of sighted predators is necessarily less than visual range. Some New Zealand coastal fishes, including commercially valuable snapper, are very sensitive to visual clarity being preferentially visual predators. They have to switch over to less favoured modes of prey detection when their visual range is constrained [15]. Certain aquatic birds, notably diving predators like shearwaters and gannets [16], are obliged to detect prey visually, and may be expected to be very sensitive to visual clarity – perhaps more so than fish which can augment vision with lateral line or olfactory capabilities. Marine mammals may also be very sensitive to visual habitat, although toothed whales can substitute their highly developed sonar for vision in low clarity waters, so may be *less* sensitive to visual clarity than preferential visual feeding animals.

Visual clarity also strongly affects human uses of waters, including coastal waters, as regards amenity and for recreation [17]. Although visual water clarity is often considered primarily an aesthetic concern, visual range in waters also affects contact recreation safety as regards avoidance of submerged hazards [18]. Measurement of visual clarity in tandem with bacterial indicators in beach surveillance is strongly encouraged – both because of safety concerns with low clarity water [19] and in view of the broad inverse correlation of faecal contamination and visual clarity [18]. (Refer the FIB stocktake by Davies-Colley & Stott.)

Turbidity, usually measured by nephelometry (light side scattering), has long been used as a convenient index of SPM concentration and clarity of waters. Unfortunately, nephelometric turbidity is not a 'proper' scientific quantity – being measured in 'informal' (non-SI) units *relative to formazine*. [20]. Because natural SPM scatters light very differently from formazine, measurements relative to formazine are of limited usefulness[1]. Only passing reference to 'turbidity' will be made herein – confined mainly to recognising the (continuing) utility of high-frequency nephelometers to provide a proxy for SPM concentration and water clarity in coastal waters, necessarily with local calibration.

A2. What is the evidence of impact on (a) ecological integrity or (b) human health? What is the spatial extent and magnitude of degradation?

There is evidence from laboratory experiments and field observations that the light climate affects growth (and depth limits) of coastal benthic plants such as keystone seagrasses [12]. Indirect evidence for degradation of the light climate of New Zealand's coastal waters comes particularly from the historical declines of seagrasses in some of our estuaries [13]. A range of stressors may be responsible including herbicides, unusually large storms, and substrate disturbance by boats, but an association with river-borne mud seems clear in many systems in New Zealand. For example, Pauatahanui Inlet lost most of its seagrass meadows in recent decades plausibly due to mud loading from urbanisation and pastoral farmland [13].

Evidence is less strong on ecological effects of visual range on reactive distance of fish and birds – mainly because most studies have used formazine-based turbidity as the metric rather than visual clarity (or TSS) [e.g., 15]. However, there are very good reasons for expecting strong effects of 'visual habitat' on aquatic animals [7]. Research on visual clarity tolerance of New Zealand coastal species seems highly desirable. Visual clarity is also known to strongly affect human aesthetic preference and safety for contact recreation in waters [17].

It is hard to assess the optical impact of fine sediment on coastal waters in the absence of a baseline. However, a reasonable assumption is that NZ's coastal waters are less clear than in pre-human and, particularly, pre-European, times owing to increased fine sediment concentrations reflecting forest clearance causing accelerated erosion of the NZ land mass [[21]. Several lines of evidence suggest that clearance of indigenous forest on hill land for pastoral agriculture amplifies sediment loads indefinitely by about an order of magnitude.

It is easier to assess optical impacts in rivers than in coastal waters because spatial changes down-river can be related, for example, to the confluence of turbid tributaries draining erosion prone terrain [22]. Because rivers are the main influence on coastal water quality [23], degraded river visual clarity may be expected to translate into our coastal waters. However, the 'translation' is not linear with salinity because river SPM tends to be flocculated on contact with salt ions promoting

[1] Turbidity has long been known to be instrument-specific, such that different turbidity sensors give different responses on the *same* waters. In one laboratory study (Rymszewicz et al. 2017) a five-fold range of output was measured across 12 different turbidity sensors on the *same* river. Even turbidity sensors of the same principle differ in response due to subtle design differences combined with differing natural SPM optics. Davies-Colley et al. (2021) reported a two-fold range for six nephelometric turbidity sensors compliant with the international standard (ISO7027; specifying 90° scattering of near-infrared radiation) that are commonly used in NZ. These authors asserted that (1) turbidity is not suitable for environmental standards, (2) turbidity should not be used as a measure of SPM effects on aquatic life, and (3) formazine-based turbidity should be abandoned and replaced by light beam attenuation.

settling [2] [24]. That is, estuarine mixing of river and seawaters is typically strongly *non-conservative* (non-linearly related to salinity).

A3. What has been the pace and trajectory of change in this attribute, and what do we expect in the future 10 - 30 years under the status quo? Are impacts reversible or irreversible (within a generation)?

The pace and direction of change in the optical water quality of NZ's coastal waters is largely unknown because only recently has monitoring of appropriate metrics been standardised [25].

Fortunately, NZ has been 'evolving' towards a better understanding of SPM and its relationship to light attenuation in waters over recent years, culminating in standards for protecting freshwaters from fine sediment damages in terms of visual clarity [26] [27]. Advances have also been made in optics of NZ coastal waters [2]. In future we may expect increased direct monitoring of visual clarity in coastal waters backed up with beam transmissometry [25] Likewise, increased monitoring of (sun)light penetration of waters is expected, particularly with PAR sensors mounted on CTD/sondes for convenient depth profiling [25].

Extreme weather events produce severe erosion and sediment delivery and cause temporarily increased 'muddiness' of rivers and coastal receiving waters. For example, the Hudson River Estuary exhibited hysteresis of suspended sediment and resulting turbidity for at least two years following tropical cyclones in 2011 [28] Similarly Cyclone *Gabrielle* (14 February 2023) may be expected to increase muddiness of rivers and coastal receiving waters for several years, particularly in the strongly impacted regions of Gisborne and Hawkes Bay.

A4-(i) What monitoring is currently done and how is it reported? (e.g., is there a standard, and how consistently is it used, who is monitoring for what purpose)? Is there a consensus on the most appropriate measurement method?

There are no NZ *standards* for SPM or related optical metrics applicable to coastal waters. However, the Ministry for the Environment [29] gives *guidelines* as follows: a black disc visibility of 1.6 m is recommended for swimming in coastal waters as well as freshwaters (for safety as well as amenity protection). Visual clarity should not be changed by more than 20%, and euphotic depth (index of light penetration) should not be changed by more than 10%, to avoid changes, respectively, to 'visual habitat' of aquatic animals (and people) and light climate for benthic plants. Clearly an appropriate baseline is required from which to gauge *change* in these optical metrics, but such baselines are not generally available for NZ coastal waters.

Standards have recently been promulgated for visual clarity to protect river ecosystems from sediment damage (NPS-FM2020: Table 8) [26]. Implementation of these standards in freshwaters has implications for downstream receiving coastal waters and the river standards themselves might be broadly applicable in coastal waters at least until refined by research.

Most monitoring of SPM and clarity (both aspects) of NZ estuaries and coastal waters is carried out by regional councils as part of SoE reporting [23]. The NEMS for discrete coastal water quality monitoring [25] is a very valuable guide that is improving cross-regional comparability and should eventually improve national reporting on NZ's coastal water quality. Depth profiling with sondes is increasingly conducted in coastal monitoring, so we can expect increasing deployment of PAR sensors on these multi-parameter devices to quantify PAR penetration [25] [2]. Visual clarity can be indexed by Secchi or black disc observations, backed up where necessary, if not actually substituted, by field deployment of beam transmissometers [25] or laboratory measurement of light beam attenuation [20]

High-frequency optical monitoring of coastal waters on moorings or structures is potentially very informative, notably. for 'sea-truthing' of remote sensing, and for capturing 'events' like wind-wave disturbance, and plumes from river stormflows or dredging. However, high-frequency optical monitoring in coastal waters is quite challenging. Nephelometric turbidity sensors are probably best for this purpose due to relatively low cost, wide dynamic range (up to 1000-fold) and comparative resistance to biofouling. But, for nephelometric turbidity data to be useful, *local calibration* to better metrics (TSS, visual clarity, light penetration) is needed [20]. Fairly frequent (e.g., monthly) visits are required, for sensor cleaning as well as on-site calibration and sampling.

Beach surveillance typically emphasises faecal indicator bacterial (FIB) measurement (usually weekly through the bathing season). Visual clarity measurement at times of beach surveillance is strongly recommended [19] – because visual clarity also affects swimmability (safety as well as aesthetics) and because a rough, but useful, inverse correlation typically exists for these variables [18] such that clear water reliably indicates uncontaminated water that is safe for recreation.

The guideline for visual clarity to protect contact recreation of 1.6 m [29] (still) applies to coastal waters – just as it does in rivers and lakes. The 1994 guidelines also recommend no more than 20% reduction in visual clarity and no more than 10% reduction in euphotic depth – while acknowledging the difficulty of establishing the relevant baseline from which to measure change.

A4-(ii) Are there any implementation issues such as accessing privately owned land to collect repeat samples for regulatory informing purposes?

Boat access is usually most appropriate for monitoring coastal water quality at central (relatively deep) sites [25], including for TSS, visual clarity and light penetration. Measurement of surface visual clarity can be conducted from a boat in tandem with sampling of SPM. Depth profiling, including for PAR penetration, requires access to deep waters, and typically can't be done from the shore unless a suitable structure such as a bridge or jetty is available. Shore (wading) access is, however, usual and appropriate for beach sampling for swimming water quality,

Several regional councils conduct coastal water quality sampling by helicopter [25] which is unsuitable for depth profiling (including of light penetration) and direct visual clarity observations. If the water is sufficiently turbid (visibility < 0.6 m) a SHMAK tube could be used for direct visibility observations on helicopter samples [25], else visual clarity, can be estimated from light beam attenuation measurements on the water samples in the laboratory.

A4-(iii) What are the costs associated with monitoring the attribute? This includes up-front costs to set up for monitoring (e.g., purchase of equipment) and on-going operational costs (e.g., analysis of samples).

Boat access to deep central sites will normally be required for measurement of a wide range of water quality variables in coastal waters [25], and boat operation is recognised as a major cost item. However, once boat sampling is underway there is little *marginal* extra cost to conducting on-site measurements of visual clarity and PAR penetration and obtaining (relatively large volume) samples for laboratory processing for TSS (costing $40/sample). If a CTD/sonde (costing perhaps $30,000 per unit, depending on optional sensors (such as turbidity, Chla-fluorescence, DO, PAR) is used for depth profiling [25], K_d(PAR) can be estimated from the simultaneous PAR profile for no extra on-site time. Alternatively, paired PAR sensors displaced by a known (adjustable) depth interval, with two-channel logger readout, of the type used by Gall et al. [2], currently costs about $5-7000, but requires extra on-site time (perhaps 5-10 minutes) for dedicated PAR profiling.

Visual clarity is best measured on-site by the black disc method [25] A float attached to the black disc target at the end of a telescoping pole makes for easier observations (but still challenging in choppy water) when the sighting range is longer than the boat length. A closely equivalent alternative to direct observation [8] is to deploy a beam transmissometer in the field (sensor plus logger costing about $20,000) or commission measurements of beam-c on water samples in the laboratory ($30/sample). A SHMAK tube is convenient to use on-site or back at the laboratory to give black disc visibility when this is low (< 600 mm) [25].

The traditional (vertical) Secchi depth is a useful index of visual clarity that is somewhat easier to measure from a boat than the black disc (horizontal) visibility, but Secchi observations can be misleading in the optically stratified water commonly encountered in estuaries. Secchi depth (in vertically mixed, i.e., unstratified water) is about 25-30% higher than horizontal black disc visibility [30]. Observer time (perhaps 5 minutes per observation) is the main cost component for visibility observations since equipment costs are low.

Increasing deployment of CTD/sondes on fixed structures or moorings may be expected in future in NZ coastal waters so as to provide high-frequency records that ‘capture’ events. However, there are significant costs associated with maintaining optical sensors on such platforms including turbidity sensors as convenient proxies for TSS, visual clarity and/or light penetration. These monitoring platforms do not obviate the need for fairly frequent (e.g., monthly) and ongoing (so costly) site visits for removal of biofouling, plus sampling and on-site measurements for calibration and validation of high frequency sensors.

A5. Are there examples of this being monitored by Iwi/Māori? If so, by who and how?

The SHMAK equipment set, including equipment and methods for black disc visual clarity observations, can easily be used in coastal waters as well as fresh waters by community monitors, and practical guidance is given in the estuary monitoring toolkit [31].

Water clarity is one of forty important environmental indicators provided by hapū and iwi in association with estuarine ecosystems throughout Waitaha (a.k.a. Canterbury) [49]. Visual clarity in freshwater is also one of the eight key indicators of the cultural health indicators, which are part of the wider state of the takiwā monitoring programme [50,51]. Visual clarity is measured within the freshwater area that feed into to significant estuaries, and contributes to the overall monitoring of estuary and its catchments [50,51].

A6. Are there known correlations or relationships between this attribute and other attribute(s), and what are the nature of these relationships?

Visual clarity correlates strongly, inversely, with measures of SPM, such as TSS [2]. Vant [32] showed that the ratio of light beam attenuation to TSS (known as ‘optical cross-section’ in view of the units), is appreciably lower for estuarine waters (about 0.4 m^2/g) than freshwaters (averaging about 1 m^2/g) – plausibly due to the flocculation of river fines on contact with salt ions in estuaries.

Sunlight penetration into coastal water also correlates with SPM, usually somewhat more weakly [2]. Kirk [33] showed why PAR attenuation with depth depends *non-linearly* on TSS – typically following a power law with exponent in the range 0.5-1.0 depending on the ratio of light absorption to light scattering by the SPM [2]. Light penetration and visual clarity are broadly correlated, but the contrasting definitions of these quantities (Eqn 1 versus Eqn 2) shows why estimating one from the other should not be attempted without local optical data. Nephelometric turbidity is not very useful in itself [20], but can provide a valuable (local, high-frequency) proxy for both TSS and optical parameters [2].

Visual clarity is often very usefully, inversely, correlated with faecal contamination of river waters [18]. This has important practical ramifications because bathers can protect themselves from exposure to faecal pathogens by avoiding low clarity water. The correlation of these variables in rivers is expected to also apply to coastal receiving waters, given that rivers are the main influence on water quality in estuarine and coastal waters [23]. For example, visual clarity, TSS, salinity and *E. coli* were all strongly inter-correlated in a stormflow plume from the Hutt River within Wellington Harbour, and nephelometric turbidity provided a valuable local proxy for these variables across the plume [24].

Part B—Current state and allocation options

B1. What is the current state of the attribute?

Currently we have only a patchy national picture of optical water quality of the New Zealand coastal zone because of the lack of in-water monitoring in several regions [23]. However, more effort on coastal monitoring is being encouraged by the NEMS for discrete coastal water quality monitoring [25], so the quality and quantity of the national dataset should improve over time into the future.

Seas, Coasts and Estuaries NZ (SCENZ), based on satellite imagery, probably gives a reasonable overall picture of certain remote sensing 'products' (including TSS, K_d(PAR), black disc visibility and Secchi depth) [34], despite very limited 'sea-truthing'. Visual clarity of coastal waters is extremely variable with time in response to meteorological forcing. River stormflows 'inject' large quantities of muddy water as plumes [e.g., 24], and wind-driven wave action over shallow waters resuspends flocculated fine sediment from the seabed [e.g., 35].

Visual clarity data is available for coastal water in some NZ regions, but not others and Dudley et al. [23] were forced to use (less satisfactory) nephelometric turbidity to attempt a national picture. These authors showed that water quality generally, including turbidity, of our estuaries and coastal waters is strongly affected by river inflows such that water quality increases with increasing salinity. Visual clarity is often highest (sometimes exceeding 40 m) in coastal waters remote from major river inflows and after prolonged periods of dry calm weather. Visual clarity is episodically degraded by phytoplankton in eutrophied estuaries and coastal waters, particularly in spring when growth is accelerated by nutrient availability combined with increasing day length and warming waters.

Visual clarity is certainly capable of being adopted as a national indicator – and arguably should be so adopted given its widespread use as a freshwater indicator in NZ and multiple useful correlations (e.g., with TSS and light penetration as well as salinity). Black disc (horizontal) visibility is more useful than Secchi depth as an index of visibility, not least because the latter can be confounded by optical stratification. Black disc visibility also has the virtue of being precisely related to light beam attenuation [8] and independent of sunlight conditions so is more useful for optical modelling.

B2. Are there known natural reference states described for New Zealand that could inform management or allocation options?

The clearest seawaters on earth are those in oceanic 'desert' waters such as the SE Pacific gyre between Tahiti and Rapanui [36]. The euphotic depth (depth at which PAR irradiance is reduced to 1% of that incident on the water surface) is about 100 m within the clearest waters. Visual clarity in such clear ocean waters has not been measured *directly* so far as we are aware [37], but black disc range would probably approach the estimated 80 m for distilled water. Such extremely high visibility has actually been measured in some remarkably clear freshwater, such as Blue Lake/*Rotomairewhenua* [37] and inferred in others from instrumental measurements (e.g., Waikoropupu Springs).

In extremely clear waters, almost devoid of suspended particles [37], a slight increase in SPM concentration greatly reduces visual range and light penetration. For example, marine waters around NZ are probably more typically about 40 m visibility (regarded as "very clear' by divers), and coastal waters even this clear are uncommon owing to river suspended sediment inputs and wave disturbance of the bed, plus seasonal phytoplankton blooms.

Because inflowing rivers strongly affect water quality of receiving waters, NZ's coastal waters remote from river mouths and with adjacent land in conservation estate may be among the clearest. Indeed, Dudley et al.[23] found strong evidence for the influence of livestock agriculture and urban land on NZ estuarine and coastal water quality.

B3. Are there any existing numeric or narrative bands described for this attribute? Are there any levels used in other jurisdictions that could inform bands? (e.g., US EPA, Biodiversity Convention, ANZECC, Regional Council set limit)

There are no numeric or even narrative target attributes states in NZ for SPM metrics such as TSS. Collins et al. [38] reviewed standards for fine suspended sediments and related metrics in other countries. Turbidity is the metric most commonly used, although this is arguably very inappropriate given the 'informal (non-SI) units and weak numerical comparability of different sensors [20].

Numeric attribute states have been defined for visual clarity in NZ freshwaters (NPS-FM2020 Table 8; [26]) – with 'bottom lines' ranging from 0.61-2.22 m, depending on classification of the sediment regime. Turbidity was originally proposed for the NPS, but dropped after our laboratory tank experiments ([20] showed 'uncomfortably' weak numeric agreement among different nephelometers. These target attribute states for visual clarity to protect freshwater ecology might be a starting point for coastal receiving waters – at least until appropriate research is conducted.

As mentioned above, *guidelines* (rather than *standards*) for visual clarity and euphotic depth (light penetration) have been enumerated by MfE [29]. No more than 10% change in euphotic depth is recommended to protect the light climate of benthic plants, and no more than 20% change in visual clarity to protect the visual field of aquatic animals (and humans). A visual clarity of 1.6 m is recommended for swimming – on the basis that swimmers need to be able to avoid submerged hazards when wading in chest-deep water [29].

B4. Are there any known thresholds or tipping points that relate to specific effects on ecological integrity or human health?

Thresholds for light 'starvation' resulting in extinction of keystone benthic plants like seagrasses are fairly well defined in theory and practice [12]. Laboratory experiments or field measurements at depth limits can establish numeric light thresholds (PAR in mol/m^2/day or % of surface average PAR) [e.g., 39]. For seagrasses, thresholds are known to *increase* in muddy systems owing to the *interaction* of light extinction by fine suspended sediment and deoxygenation under deposited fines [14].

In some coastal ecosystems, light starvation may result in 'tipping' in the classic 'catastrophe theory' sense of change to a new metastable state. Seagrass meadows are known to be vulnerable to 'tipping' to a devegetated seabed where plant recovery may be precluded by substrate muddiness [13].

Thresholds, but not tipping points, are expected for visual clarity as it affects behaviour of aquatic animals such as fish, as shown for snapper in NZ by Lowe et al. [15]. However, visibility thresholds are poorly known for NZ coastal species suggesting the need for indigenous research.

B5. Are there lag times and legacy effects? What are the nature of these and how do they impact state and trend assessment? Furthermore, are there any naturally occurring processes, including long-term cycles, that may influence the state and trend assessments?

Lag effects occur in coastal waters due to flocculation and settling of inflowing river sediment on contact with salt ions. This settled, flocculated sediment, sometimes referred to as a 'nepheloid layer, is easily resuspended by wind waves or other hydraulic disturbance of the bed. So, terrigenous fine sediment, delivered mainly by river stormflows, may go through multiple cycles of settling and resuspension before being "winnowed'' from coastal waters.

Petersen et al. [40] showed that seabed lighting could be much more strongly shaded by nepheloid layers than calculated from extrapolating the water column light gradient – which may explain, in part, the legacy effect of seagrasses often failing to recolonise after extinction by mud loading. Transplantation experiments in NZ were successful in former seagrass meadow sites in Whangarei Harbour [41] but not in Pautahanui Inlet (despite apparently sufficient bed PAR) – apparently because of mud deposition and intrusion into the substrate in the latter estuary [13].

B6. What tikanga Māori and mātauranga Māori could inform bands or allocation options? How? For example, by contributing to defining minimally disturbed conditions, or unacceptable degradation.

Visual clarity of waters is of high concern to Māori – as demonstrated, for example, by the close inverse correlation of visual clarity and cultural health reported by Harmsworth et al. [42]. Furthermore, the broad inverse correlation of FIB and visual clarity [18] provides a means to avoid contact with faecal contamination, simply by avoiding low clarity waters. In practice, if visual clarity is close to or better than the MfE [29] guideline of 1.6 m for swimming, the risk of infection by faecal pathogens is very low [18]. Low clarity coastal water is best avoided in view of the difficulty of detecting submerged hazards [29] even if demonstrably free of faecal contamination.

Part C—Management levers and context

C1. What is the relationship between the state of the environment and stresses on that state? Can this relationship be quantified?

The main pressure on optical water quality of coastal waters is mobilisation of fine sediment from adjacent land with conveyance to the coast via rivers, notably during storm flow events [43]. However, the relationship between the *pressure* (land erosion) and coastal optical water quality is highly complex because of:

- Displacement in space of land sources from coastal receiving waters
- Variation in time, particularly with river flow conditions and coastal plume hydrodynamics,
- Uptake and storage of fines (and faecal microbes) in the hyporheic zone of rivers [44].
- Flocculation of fine sediment on contact with salt ions from seawater, with subsequent settling to the seabed[1].

C2. Are there interventions/mechanisms being used to affect this attribute? What evidence is there to show that they are/are not being implemented and being effective?

Because most fine sediment loading on NZ's coastal waters comes from erosion of adjacent land, the interventions must focus on land. Such interventions are underway in NZ, focussed mainly on general water quality (including nutrients and faecal contamination as well as fine sediment) of *rivers* rather than coastal waters.

C2-(i). Local government driven

Regional councils are the agencies most actively intervening to improve water quality in NZ, including optical water quality – by promoting soil conservation and stream fencing (to reduce cattle ingress) and riparian setbacks (to trap sediment in overland flow). Such land management has been shown to improve stream water quality. For example, integrated catchment management at Whatawhata generally improved sediment loads and stream water quality [45-47], and improved visual clarity and reductions in *E. coli* have been reported for catchments with land management action in Horizons Region [48]. These water quality improvements can be expected to 'translate' downstream to coastal receiving waters.

Regional councils are keen to inform the recreating public of swimming suitability of waters. In future, modelling of visual clarity informed by high-frequency turbidity monitoring could, in principle, be used to 'now-cast' swimming suitability. NIWA currently has 'Smart Idea' funding of a project (WaiSpy MBIE contract: C01X2204) that is attempting to develop a system for nowcasting 'swimmability' of rivers – most strongly affected by faecal contamination and visual clarity. The approach can potentially be translated to downstream coastal receiving waters.

C2-(ii). Central government driven

[1] Resulting nepheloid layers are easily resuspended by hydraulic disturbance, notably wind wave exposure over tidal flats. Nepheloid layers consolidate over time, but leave the substrate 'muddier' than previously, more subject to deoxygenation, degraded as habitat, and less suitable for re-establishment of keystone seagrasses.

C2-(iii). Iwi/hapū driven

Iwi and hapū are aware of the deleterious effects of suspended sediment in estuarine and coastal waters and have reported changes in this attribute over time. Iwi and hapū have been heavily involved in Jobs for Nature projects across New Zealand (many of which address sediment and nutrient entry into waterways, which are related to this attribute) and are leading estuarine restoration initiatives in partnership with Councils and National Science Challenge researchers from various universities, CRIs, and other research institutes/providers.

C2-(iv). NGO, community driven

Community-driven initiatives such as catchment care groups and 'Mountains to Sea' mobilise community interests in stream fencing, restoration planting and water monitoring. These efforts should reduce the burden of fine sediment loading of rivers and downstream coastal waters. We are not aware of improved coastal water quality in NZ being explicitly linked to land management, however such connections have been established overseas. Improved coastal water quality is difficult to attribute to land management because of the complexity of land-coastal connections (Refer C1).

C2-(v). Internationally driven

Part D—Impact analysis

D1. What would be the environmental/human health impacts of not managing this attribute?

Not managing sediment load to coastal waters and resulting degradation of optical water quality is likely to lead to decreased recreational amenity and swimming suitability, and downgraded perception of NZ as 'clean and green' among tourist visitors. Not managing fine sediment and water clarity (both aspects) is also likely to severely impact coastal ecosystems – reducing visual range of fish and birds and degrading the light climate of benthic plants. Seagrass meadow ecosystems, in particular, seem particularly vulnerable to 'tipping' by muddy stormflow plumes into a new de-vegetated state with no guarantee of eventual seagrass recovery [12].

Managing optical water quality of coastal waters requires, mainly, management of sediment loads of inflowing rivers – in turn by reducing land erosion. So, land activities that mobilise sediment, primarily livestock agriculture, plantation forestry and urban land use, need to be isolated so far as possible from waters. Important controls on sediment mobilisation include:

- *In livestock pasture:* soil conservation activities (e.g., poplar space-planting) and fencing to exclude livestock from waters, plus riparian fencing and planting to entrap sediment in overland flow,
- *In plantation forestry:* care with roading and culverting to prevent erosion, slash management and riparian set-backs to avoid slash entry and minimise sediment entry to streams.
- *In urban areas:* street-sweeping to reduce mobilisation of fines and associated contaminants in stormwaters, and detention of roof and paved area runoff to prevent bank erosion of urban streams by high peak flows.

To *manage* optical water quality of coastal waters requires its *measurement* – which, currently, is not adequate in NZ because sampling is mainly discrete at a sparse distribution of sites. What is needed for improved management is *modelling* to fill in the measurement gaps in time and space – ideally informed by high-frequency instrumental monitoring (in contributing rivers as well as coastal receiving waters) or new modelling approaches using satellite remote sensing such as SCENZ [34]. The 'Coastwatch' research programme, proposed by NIWA to the MBIE Endeavour fund, would integrate relevant sources of monitoring data (notably coastal satellite imagery and high-frequency river monitoring) within an artificial intelligence (AI) framework for a 'quantum leap' in NZ's coastal management capability.

D2. Where and on who would the economic impacts likely be felt? (e.g., Horticulture in Hawke's Bay, Electricity generation, Housing availability and supply in Auckland)

The main economic impact of *not* managing coastal water optical quality would be on NZ's tourist industry – which trades strongly on NZ's image as a 'clean green', environmentally responsible country. Loss of aquatic species of conservation concern is also possible, including marine mammals and birds. Commercially valuable fish species, such as snapper, could be reduced in abundance by poor visual habitat and loss of 'nursery' seagrass meadow habitat.

The general public of NZ would be impacted in a difficult-to-quantify way if our coastal waters were increasingly perceived by NZ citizens as of degraded optical water quality, resulting in reduced recreational opportunity for fear of underwater hazards (and, likely, faecal contamination) and reduced sport fishing opportunity and biodiversity values.

D3. How will this attribute be affected by climate change? What will that require in terms of management response to mitigate this?

Increased storminess due to global warming may be expected to decrease optical water quality of coastal water, by increasing SPM concentrations and light attenuation. More frequent large storms are forecast to cause more land erosion [21], resulting in increased fine sediment loads conveyed by rivers with associated degradation of habitat and recreational amenity in coastal receiving waters. Neverman et al. [21] modelled erosion response to global warming from the NZ land mass under defined IPCC scenarios, and predict that sediment yield from soft-rock areas (such as Gisborne, and Hawkes Bay) could double by 2090.

Higher summer temperatures with global warming may be expected to drive people to swim and recreate more often in NZ's coastal waters despite poor visual clarity (and more frequent faecal contamination). Higher summer temperatures will also exacerbate deoxygenation in coastal substrates under deposited (flocculated) river fines – which exert an oxygen demand as well as reducing oxygen transfer from overlying water.

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9.13 Phytoplankton/chlorophyll *a* in estuarine/coastal water (trophic state)

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Alternate attribute name: Trophic state using chlorophyll *a* concentration as a proxy for phytoplankton biomass in estuarine and coastal ocean waters.

State of knowledge of the "*Chlorophyll a (Chl.a)*" attribute

- Coastal-ocean (open coast): Good / established but incomplete.
- Estuaries: Medium / unresolved to Poor / inconclusive

National-scale satellite remote sensing of "surface" ocean colour and derived water quality attribute maps, of which chlorophyll *a* (Chl.*a*) is one, provides observations relevant to most coastal (< 12 nm limit) and some estuarine conditions, on moderate space (500 m) and time (days to decades) scales [32, 34]. A "coastal-ocean" ranking - good / established but incomplete - highlights on-going research effort and challenges in improving and confirming accuracy near coastlines in turbid waters, need for increased spatial resolution in many areas (10-100 m), and gap-filling and extrapolation techniques due to pixel failure (cloud cover, atmospheric correction, land adjacency and bottom reflection) [50]. Despite these challenges, observations on space and time scales aligned with natural dynamics in coastal and oceanic physical processes[16], provides long-term observation from mid-2002 on natural variability, suitable for establishing patterns and trends[37]

For the purposes of this document, "estuarine" waters require a distinction from – but are connected to – "coastal-ocean" waters, to reflect their more enclosed nature, connection to freshwater river inputs and their tidal/subtidal influence. For many estuaries, a medium / unresolved to poor / inconclusive ranking, highlights the porosity of location and time appropriate observations in consideration to their dynamic nature, particularly in connection with episodic and transient storm flow plumes [96, 24]. As outlined above, at times, in some areas, valid satellite observations in estuaries are useful and should improve with advancements in multiple satellite sensor resolutions and processing, with more in-water collections and sensor monitoring needed for calibration/validation in these complex and tidally dynamic water bodies, globally and nationally [3, 50, 49, 34].

Part A—Attribute and method

A1. How does the attribute relate to ecological integrity or human health?

Phytoplankton are microscopic algae (single-celled 'plants') living in the upper parts of the water which grow by photosynthesis depending on nutrient and light availability[52]. As all phytoplankton contain a green pigment Chl.*a* (mg m^{-3}), its concentration is commonly used as a proxy for algal carbon biomass (C – mg m^{-3}) which is often related to primary productivity rate (mg C m^{-2} d^{-1})[26].

Chl.*a* concentration is a widely accepted (globally and nationally) indicator of 'overall' biological productivity and "trophic state" (which was originally proposed for lakes[9]). Trophic state based on Chl.*a* has advanced into generally accepted classes across lake, estuarine and coastal waters (e.g., Table 1). Except where sub-surface Chl.*a* maxima occur, a near surface measurement of Chl.*a* is indicative of water-column integrated phytoplankton abundance and therefore its primary production particularly in ocean studies[4].

Some species of algae are toxic and are called Harmful Algal Blooms (HABs) and these can cause severe environmental and human health issues. A significant increase in the frequency of occurrence of HABs around the world has occurred, likely due to climate change and land-use changes[40, 113]. Adverse effects may include mass fish and wildlife death, human toxic reactions in consumed seafoods and in extreme cases, death. HABs are a subset of naturally-occurring blooms, and algal blooms in general can often be detected and tracked by satellite remote sensing of Chl.*a* (and other colour methods)[6]. With ground truth historical data (species ID), novel machine learning techniques have been used to identify specific events[45]. For local perspective, HAB occurrences and societal impacts for Australia and New Zealand have recently been reviewed [43].

Table 1: Trophic classes. Chlorophyll *a* concentration (Chl.*a*, mg m^{-3}) and water clarity (measured vertically as Secchi Disk depth - z_{SD}, m) ranges for descriptive trophic classes used in New Zealand lakes and estuaries; extended to include coastal-ocean conditions. Note: Microtrophic and Ultra-Microtrophic are not commonly present around New Zealand. Visibility can also be measured horizontally using a black disk (y_{BD}, m) which is broadly comparable to z_{SD}. Freshwater and Estuarine NPS states: A (Minimal); B (Moderate); C (High); D (Very High) – see footnotes.

Trophic class	Level	Lakes				Estuaries and coastal-ocean			
		Chl.a_{min}	Chl.a_{max}	z_{SDmax}	z_{SDmin}	Chl.a_{min}	Chl.a_{max}	z_{SDmax}	z_{SDmin}
Ultra-Micro	1	0.13	0.33	31	24	*0.01*	*0.05*	*80*	*40*
Microtrophic	2	0.33	0.82	24	15	*0.05*	*0.10*	*40*	*30*
Oligotrophic	3	0.82	2	15	7.8	0.1	1	30	8.5
Mesotrophic	4	2	5	7.8	3.6	1	3	8.5	4.0
Eutrophic	5	5	12	3.6	1.6	3	8	4.0	2.1
Supertrophic	6	12	31	1.6	0.7	8	12	2.1	1.6
Hypertrophic	7	31	~100	0.7	<0.1	12	~50	1.6	<0.1

Table 1 Footnotes: For New Zealand lakes, regional councils historically have used a seven-class trophic level index (TLI)[8], which has evolved into the trophic state (ecosystem health) system in the National Policy Statement for freshwater management (Freshwater NPS)[70]. The 4-attribute state (A-D) eutrophication system (minimal – very high) is also used in the New Zealand Estuary Tropic Index (ETI)[92] adapted[88] for four salinity estuary types. We extend the euhaline type estuaries (>30 ppt salinity) Chl.*a* ranges to include oligotrophic,

microtrophic and ultra-microtrophic classes guided by ocean studies [73] (Chl.*a* < 1 mg m^{-3}), as these can be associated with oceanic waters surrounding New Zealand. Secchi disk depths are included as a water clarity (visibility) guide only, being not as reliable as Chl.*a* for eutrophication as measures are impacted (reduced) by TSS and CDOM if present. Water clarity however is an important metric to consider as levels of irradiance drive the photosynthesis of primary producers. Water clarity is also important for sighted aquatic predators. Coastal marine eutrophication primary productivity (gC m-2 day-1) ranges for oligotrophic (< 100), mesotrophic (100-300), eutrophic (301-500) and hypertrophic (>500) systems that can be broadly associated with the trophic classes[76]. A worldwide compilation of 131 estuarine-coastal ecosystems also provides useful Chl.*a* context[11].

A2. What is the evidence of impact on (a) ecological integrity or (b) human health? What is the spatial extent and magnitude of degradation?

Estuaries. Globally, strong evidence of change and/or degradation in estuarine trophic states through eutrophication was established over 20 years ago[97] and more recently there is growing evidence and concern for HABs impacts[40, 113]. Nationally in New Zealand, we have limited direct evidence of the spatial extent or magnitude of degradation or adverse impacts resulting from changes in Chl.*a* in estuarine environments. Traditionally, monitoring Chl.*a* based on in water collections has been much more limited than long-term New Zealand river monitoring practices[96] because of difficulties with sampling estuarine/coastal systems at an adequate scale. Many in situ coastal monitoring sites should be considered as 'case study' indicators of local rather than national relevance [23, 31].

Degrading water quality at estuarine sites have been linked to changes in river flow and land-use (related to agriculture and urban land cover)[24]. In the absence of adequate long-term consistent and widespread monitoring of estuarine Chl.*a*, there has been increasing use of New Zealand estuarine trophic index (ETI) studies and tools for assessing eutrophic susceptibility and trophic state from alternative measured indicators [88] and use of predictions from Bayesian Belief Networks (BBN) when limited or no data are available [118]. Both ETI-based and BBN-based studies highlight many degraded and suspectable estuaries.

Coastal systems. Further offshore, a recent 2023 update on monitoring ocean health around New Zealand utilizing satellite Chl.*a* (with sea surface temperature and total suspended solids) documents a range of natural patterns (seasonal and interannual variability), states and range of trend directions (from mid-2002) in different areas around NZ[84]. This national scale assessment highlights the complexities of the choice of scale in spatial (site, area, region, water mass) and temporal (month, season, year, overall) interpretation of states, natural patterns, and change. A more comprehensive analysis of coastal hydrosystems[1], regions and water masses are needed to address spatial extent and magnitude of degradation, or change, to usefully assess potential impacts.

Coastal HABs are less well researched on a national scale, particularly in their relationship to Chl.*a*. However, species have been monitored since 1993 as part of the human health Marine Biotoxin Monitoring Programmes encompassing both commercial and non-commercial shellfish harvesting[93]. Their timely New Zealand review emphasizes the wide-ranging effects on commercial productivity and future directions and improvements needed in predicting movement and size of HABs,

[1] We use the term "coastal hydrosystem" following [47] to describe coastal features that span a gradient from near coast freshwater lakes/wetlands to marine.

something that satellite Chl.*a* monitoring can augment. This is particularly relevant in consideration to the advancement of machine learning approaches which use these type of data[13].

Summary. The spatial extent and magnitude of estuarine and coastal degradation varies with factors like nutrient input, water circulation patterns, and local environmental conditions. In highly impacted areas, such as those with intensive agricultural or urban runoff, the degradation can be substantial, leading to widespread ecological disruption and public health concerns. These anthropogenic effects are also embedded in larger scale climate change influences on oceanic water masses. There is a critical role of Chl.*a* monitoring in assessing estuarine and coastal degradation across various space (from local to regional) and time (days to decades) scales and informs targeted mitigation efforts to address the root causes of ecosystem degradation and protect human health.

A3. What has been the pace and trajectory of change in this attribute, and what do we expect in the future 10 - 30 years under the status quo? Are impacts reversible or irreversible (within a generation)?

Estuaries. Change of Chl.*a* in estuaries is principally affected by land-use and riverine input, as well as modifications/activities in the estuaries themselves. The trajectory of change is directly influenced by river inputs and storm flow events carrying contamination loads and could be reasonably inferred or modelled using available tools[116]. Recovery responses will vary depending on catchment characteristics and the specific coastal hydrosystem. Managing and mitigating contaminant inputs, particularly those influencing algal growth such as macro-nutrients and light attenuation effects of suspended sediment, can modify eutrophication levels, reduce the potential for HABs and improve trophic status. It's crucial to recognize that the effectiveness of these measures is contingent upon the removal rates and buffering capacity of the system. Downstream effects in receiving environments may be reversible over time, but the rate of recovery is dependent on various factors including the extent of land-use practices and the resilience of the ecosystem. Additionally, the impacts of climate change on river hydrology are likely to have important long-term implications [12], with concurrent impacts on receiving environments such as estuaries.

Coastal systems. For the bulk of the coastal-ocean, where catchment influences are more “dilute” and open-ocean processes dominate, changes are principally due to the effects of climate change, with local modification due to land-use/riverine systems/coastal development. Changes to Chl.*a* in coastal systems due to climate variability and change are observable, ongoing, inevitable and unlikely to be reversible in the short term (decades). Earth-system models project changes to 2100 in coastal water temperature, mixed layer depth (light), macronutrients and primary productivity to varying degrees [57]. There remains uncertainty over how these large-scale climate changes will affect continental shelf (< 200 m) waters which are in the transition zone between land and oceanic influences. Climate-driven changes to coastal Chl.*a* will specifically depend as to how nutrient supply from river inputs and offshore upwelling/downwelling events, wave resuspension changes on light climate, mixed layer depths and water current changes will be modified under future conditions. Robust and accurate satellite observations of Chl.*a* across these physical process scales can be used for state of the environment monitoring for these areas.

Summary. Without significant intervention, trends in estuarine and coastal Chl.*a* are expected to persist and potentially intensify over the next 10 to 30 years. While targeted management efforts, such as reducing nutrient and suspended sediment inputs, offer some hope for reversal of changes in

Chl.*a* in some near shore coastal hydrosytems, the effectiveness and pace of recovery depend on various factors, including the severity of degradation and the scale of interventions. While some impacts may be reversible with concerted action, prolonged or severe degradation could result in partially irreversible effects within a single generation, emphasizing the need for proactive measures to safeguard coastal ecosystems and human well-being.

A4-(i) What monitoring is currently done and how is it reported? (e.g., is there a standard, and how consistently is it used, who is monitoring for what purpose)? Is there a consensus on the most appropriate measurement method?

<u>In situ (discrete) sampling</u>. In-water measurement methodologies for Chl.*a* have been rationalised by the National Environment Monitoring Standards (NEMS) for discrete (in situ) sampling and analysis of coastal water quality, including Chl.*a*[72]. Standardization of measurement methods ensures consistency and comparability of data across different regions, but historical data prior to this relatively recent (2018) implementation differs making assessment of change problematic. In these in situ/laboratory-based analyses of water quality, typically, particles from a known volume of collected water are concentrated onto filter paper, extracted with acetone and the concentration determined against Chl *a* standards using either fluorescence or spectrophotometric methods (mg m^{-3}).

For New Zealand, regular monitoring of Chl.*a* is primarily conducted by Regional Councils, responsible for monitoring specific coastal areas within their jurisdictions and to their most relevant need, mainly for estuarine water quality and summer period recreational swimability guidance. They provide annual[48, 79] and multi-year SOE reports[28, 80, 27], with their own regional flavours and no national consistency in reporting. Regional sampling effort is guided by the Department of Conservation (DOC) Policy 21: “Enhancement of Water Quality” as part of the 2010 New Zealand Coastal Policy Statement (NZCPS) where it has deteriorated[77]. The Ministry for the Environment (MfE) oversees national environmental monitoring programs which use Regional Council data to fund broader insight[23, 31], jointly with StatsNZ in “Our marine environment 2022” reporting series[71].

The Department of Conservation (DOC) is responsible for the implementation, management, and monitoring of 44 marine reserves, and provides guidance in the Marine Monitoring and Reporting Framework (MMRF) Theme 6: Water quality [18]. MMFR is designed for both local and national stakeholders to guide long-term marine monitoring taking a whole system approach.

Coastal and estuarine water quality data are available through some Regional Councils website, with curated data nationally reported as summaries or aggregates at: LAWA (Land Air Water Aotearoa): Estuary Health [58], but this is yet to include water quality attributes, instead linking to StatsNZ as states and trend indicators[101]; and DOC in the “Estuaries spatial Database”[19]. These are compiled from the most recent MfE funded analysis[31].

<u>Autonomous field instrumentation</u>. NEMS protocols for monitoring Chl.*a* by other methods do not currently exist for in situ deployable sensors such as Chl.*a* fluorometers[119]). The accuracy of observations of estuarine/coastal Chl.*a* by autonomous instrumentation relies on robust relationships between in-water Chl.*a* and fluorescence [83] which is a field of active research overseas but less so in New Zealand.

<u>Remote sensing</u>. Satellite (remote-sensing) technology for observing water colour in specific wavebands (e.g., remote sensing reflectance Rrs[115]) in coastal systems has existed since 1997 (4 km)

with reasonable spatial resolution (0.5-1 km) since 2002. Satellite methods of observing estuarine/coastal Chl.*a* rely on relationships between in-water Chl.*a* concentrations and water colour[41]. For satellite Chl.*a* maps, empirical "band-ratio algorithms" (e.g., blue and green Rrs) has proven effective for open-ocean and coastal waters where phytoplankton and its by-products primarily contribute to colour, so called Case 1 conditions[41] and [78]. However, in turbid coastal waters (so called Case 2) Case 1 algorithms are inaccurate due to the presence of other light attenuating components, such as suspended sediments and coloured dissolved organic matter from rivers, shore erosion and wave resuspension events. Case 2 waters require more complex "semi (or quasi) analytical algorithms (QAA)" to tease apart the contributions of absorption and scattering characteristics of each component, prior to determining their levels for improved accuracy[60, 61, 37].

The Ministry for the Environment (MfE) jointly with StatsNZ has recently started using satellite observations for longer temporal and broader spatial Chl.*a* assessments in coastal systems[[86, 84]). NIWA-SCENZ (Seas, Coasts and Estuaries New Zealand) provides web-access to all satellite water quality attributes for the NZ coastal-ocean from mid-2002 in weekly, monthly, seasonal, and yearly aggregate states, climatologies and anomalies [85, 37]. Satellite monitoring is expected to advance state of the environment assessment and reporting with future research effort and national support, to provide improved national maps for standard, consistent monitoring of Chl.*a*, with a range of other water quality attributes and analytics (e.g., climatologies, anomalies, trends)[32]. MPI have also funded recommendations in ocean indicators (including Chl.*a* and primary producers) as part of the Atmospheric and Ocean Climate Change Tier 1 Statistic[87].

<u>Other monitoring</u>. Aquaculture New Zealand relies on clean coastal environment and sustainable farming practices, with expected future growth in offshore (open ocean) farming, particularly with Integrated Multi-Tropic Aquaculture (ITMA)[103]. In some cases aquaculture monitoring datasets are not be freely available, but they form part of long-term monitoring requirements by the Ministry of Primary Industries (MPI), which have funded a range of water quality monitoring guidelines (including Chl.*a*) and standards for the aquaculture industry, including bivalve shellfish (mussels and oysters)[51], salmon in Marlborough Sounds[25] and open-ocean finfish culture[38]. Additionally, Fisheries NZ (MfE) have recently completed (Aug-23) year three review of National Environmental Standards for Marine Aquaculture (NES-MA) recommending inclusion in regional coastal plans and developing best practice guidelines on the scope and scale of monitoring.

Appropriate observation of Chl.*a*, primary producers and HABs are needed for monitoring the "Carrying capacity" of a system, particularly in consideration to bivalve (mussel and oyster) filter feeders on trophic food sources[68], finfish farm eutrophication and potential HAB effects[30], and aquaculture impacts in general which are of environmental, social, economic, and cultural concern globally[7]. Multi-scaled sampling using in situ monitoring (water collections and moored instrumentation) with satellite observations are all required to capture fit-for-purpose Chl.*a* dynamics across different scales.

A4-(ii) Are there any implementation issues such as accessing privately owned land to collect repeat samples for regulatory informing purposes?

<u>In situ (discrete) sampling</u>. The practicality, feasibility and cost effectiveness of in situ methods of monitoring Chl.*a* is dependent on the characteristics of the coastal system of interest. The activity of in-water sampling in estuaries and the coastal-ocean is challenging, that usually requires trailered

boat surveys (< 6m), a qualified skipper with local knowledge, trained staff and all the appropriate health and safety requirements for the activity. Weather provides challenges of its own for regular, large-area, coastal sampling. Consequently, sampling is often biased to calmer periods unless suitable research vessels are available (typically in offshore coastal waters), examples of which are mentioned in a 15-yr Firth of Thames seasonal samplings[36]. These medium sized vessels (15-25m) with several crew and on-board, food, and accommodation, ensure sampling is achieved. Due to high costs, often these are part of larger area surveys that might be combined with moored instrument deployments, recovery, cleaning, servicing, and redeployment. Larger oceanographic vessels (~75-100m) such as the NIWA Tangaroa may be required for coastal-ocean surveys on the continental shelf or deeper waters with voyage plans supporting multiple projects and objectives for multi-week samplings.

In New Zealand, where a considerable portion of the coastline may be privately owned, gaining access permissions for sampling from the shore may be challenging, especially if landowners are unwilling to grant access or if there are legal complexities regarding property rights. Additionally, logistical challenges such as navigating rugged coastal terrain, coordinating sampling schedules with landowners, and ensuring the safety of monitoring personnel in remote or inaccessible areas further complicate the monitoring process. These implementation issues can hinder the collection of consistent and representative data, potentially impacting the accuracy and reliability of regulatory assessments and decision-making processes.

Remote sensing. Satellite remote sensing data relies on availability through the earth-observation community worldwide, as well as specifically the contributions of the NASA Goddard Space Flight Center and the MODIS project for access to remote sensing globally. Without their support, access to satellite data would not be possible. Much of the imagery is provided by the Land Atmosphere Near-real-time capability for EOS (LANCE) system and the services offered by the Global Imagery Browse Services (GIBS), both operated by the NASA/GSFC/Earth Science Data and Information System (ESDIS), funded by NASA/HQ. The NOAA Center for Satellite Applications and Research (STAR) the Ocean Biology (OB) group, the European Union Copernicus Marine Environment Monitoring Service (CMEMS), and the European Space Agency (ESA) Ocean Colour (OC) Climate Change Initiative (CCI) all provide satellite data for free. At present for NIWA-SCENZ, we are using MODIS-Aqua data from NASA but will need to utilise and incorporate other ocean colour sensors such as VIIRS (NASA) and Sentinel-3 (ESA) in future.

A4-(iii) What are the costs associated with monitoring the attribute? This includes up-front costs to set up for monitoring (e.g., purchase of equipment) and on-going operational costs (e.g., analysis of samples).

In situ (discrete) sampling. The laboratory analytical cost for a Chl.*a* analysis are small (about $15 per filter, $20 if filtration needed) compared to the overall cost of sample collection. The overall cost of monitoring Chl.*a* in New Zealand estuarine and coastal systems at the national scale using in situ sampling is obviously dependant on how many samples are needed. Specifically, the costs depend on how many sites and how often sampling is needed. This has not yet been properly assessed, but it is clear that current levels of in situ sampling are inadequate in both regards: too few sites are monitored, and the sampling is too infrequent to account for the variability. Sampling from the shore, jetties, or wharfs at high tides, in many systems, does not adequately capture its variability.

See the relevant section in the discrete water quality sampling and testing Vol. 4 (Coastal) NEMS[72] for further advice and details on this type of sampling.

Autonomous field instrumentation. Autonomous field instrumentation may provide a more cost-effective monitoring system for Chl.a than in situ (discrete) sampling but this is not proven. A comparison of costs will need to include upfront costs to purchase of monitoring equipment such as fluorometers, spectrophotometers, water quality probes, and boats or vessels for sample collection and staff time. Additionally, investments may be required for establishing monitoring stations, deploying sensors on buoys or moorings, and acquiring laboratory facilities for sample processing and analysis (for calibration/validating in situ instrumentation). Other recurring expenses may include maintenance and calibration of monitoring equipment, quality control measures, data management software, and communication infrastructure for transmitting real-time data. Ideally, autonomous field instrumentation would be combined with (seasonal) surveys to combine time-series measurements with a suite of collections of other environmentally relevant parameters and scientific objectives (e.g., Hauraki Gulf time-series).

Remote sensing. Suitable satellite data for monitoring estuaries and coastal Chl.a at the New Zealand national scale is available for free from NASA, NOAA and EU (Copernicus). There is a relatively-small ongoing cost within New Zealand to access, quality control/validate/calibrate (using local knowledge/measurements), and apply the data for tracking change in Chl.*a*. Maintaining New Zealand expertise in coastal remote sensing to get the most out of the free-to-air satellite data requires ongoing funding which has no long-term stable funding.

A5. Are there examples of this being monitored by Iwi/Māori? If so, by who and how?

The Ngā Waihotanga Iho (Estuary Monitoring Toolkit) is a useful example of the developments relevant the collection of estuary attributes[91, 105, 75]. While collections and observations of Chl.*a* are not directly included at present, they could be considered in the future to provide information on aspects of "decline in water quality, for example reduced water clarity, contamination of water and kaimoana due to runoff of human and animal waste". This has particularly relevance to potential changes in trophic states, HABs, and human health.

The observation of phytoplankton (as indicated by *Chl. A*) would also be important in their relationship with kaimoana, in particular mātaitai (shellfish) are part of the holistic approach to understanding ecosystem health in estuaries. That is beyond, an indicator for decline in water quality (mentioned above), they are an indicator of the benthic-pelagic coupling, and thus advocate for supporting a healthy population size and structure of taonga species, such as mātaitai (shellfish), which has been assessed in engagement with mana whenua in Otago [[120]].

There are other "in-direct" observational methods that could be more practical for Iwi/Māori and citizen science in general. For example, Munsell water colour standards [15] and observations can be used as a proxy for Chl.*a* and trophic state that also provide a more direct aesthetic connection to human perception[114]. Supporting this approach are efforts to use water colour aspects from satellite remote sensing[109, 111, 62, 110] which can provide a connection between the different types of monitoring effort across different scales.

A6. Are there known correlations or relationships between this attribute and other attribute(s), and what are the nature of these relationships?

There are many correlates of Chl.a in coastal and estuarine waters of relevance to understanding change in water quality for the purposes of improving environmental management, stewardship and wealth creation. Chl.a in estuarine and coastal systems is intricately linked to attributes including: (1) turbidity/scattering; (2) total suspended solids; (3) water colour (Munsell water colour standards, Apparent Visible Wavelength); (4) water clarity (visibility, light attenuation, Sechhi depth); (5) macro-nutrients); (6) Dissolved Oxygen Concentration; (7) Faecal contamination; (8) primary production by phytoplankton growth; (9) light reaching the seabed (related to benthic autotroph primary production). The exact relationships between Chl.a and these other factors depend on complex ecological interactions and environmental processes[53]. High Chl.*a* can reduce water clarity[35], influence dissolved oxygen dynamics[117], and serve as a proxy for phytoplankton biomass and diversity[94]. These interrelationships highlight the interconnected nature of coastal ecosystems and the importance of considering multiple attributes in monitoring and managing coastal water quality and ecosystem health[53].

Part B—Current state and allocation options

B1. What is the current state of the attribute?

In situ (discrete) sampling. The current state of change in estuarine Chl.*a* at the national scale is poorly understood based on direct observational measurements (in situ sampling and/or autonomous field instrumentation). Regional Council monitoring sites are clustered unrepresentatively around urban centres (about 350 sites), and many types of estuaries (and likely types of change) are under-represented spatially across the system[23, 31]. Sampling frequencies at these sites (typically monthly) are also inadequate to have the statistical power to capture whare are usually small long-term changes and trends against a backdrop of substantial variability on time-scales from minutes to years. Despite these challenges, the latest MfE-funded national analysis in 2021, compiled from a curated subset, documents a mixture of current states and direction of any trends in Chl.*a* with high variability in ETI classes [31]. A wide range of changes based on this analysis is not unexpected because of regional differences and natural spatial and temporal variability within each system. StatsNZ updates and publishes a higher-level assessment on their website [102] (e.g., “For nine coastal and estuarine water quality measures, more sites had improving trends that worsening trends (2006-2020)”. While this provides a degree of national rationalisation, it highlights inadequacies and challenges of sampling and interpretation across different coastal hydrosystems.

Remote sensing. For the coastal-ocean (excluding may small estuaries), satellite Chl.*a* provides a more complete assessment, but not one that is without it’s own limitations in terms of detecting and understanding the drivers of change in Chl.*a* [84]. From mid-2002 to early-2023, satellite studies show that Chl.*a* has increased along the mid-lower North Island and all of South Island, especially during winter, suggesting higher productivity likely due to nutrient inputs or upwelling. Conversely, Chl.*a* has decreased along the west coast of Northland and northeast New Zealand shelf, indicating potential issues with productivity possibly linked to changes in nutrient availability or environmental factors. Lower-than-normal Chl.*a* during Marine Heatwave (MHW) events along the west coast of

South Island in December 2017 to March 2018 and December 2022 to January 2023 may signify reduced productivity, necessitating further investigation into its causes and potential impacts on marine ecosystems and fisheries. The satellite-observed trends in Chl.a may also help to prioritise in situ sampling and design a more representative system of sentinel coastal sites.

A recent update in satellite Chl.*a* trends (up to Aug-2023) was provided as part of a Cyclone Gabrielle (Feb-2023) impacts on fisheries, for a long-term context[59]. Web access to data is provided through the NIWA-SCENZ[74] project page application "Trendy-SCENZ"[74], which illustrates national scale maps of Chl.*a* trend directions (absolute and percentage of climatology median) per decade, and statistical likelihoods [67]. Web access of national scale maps allows users to interactively explore the complex spatial patterns ranging from broad coastal-ocean (1000 km) to estuary (0.5 km) scales, highlighting the significance of delineating aggregation boundaries for precise interpretation.

B2. Are there known natural reference states described for New Zealand that could inform management or allocation options?

Identifying natural reference states for Chl.*a* in New Zealand's estuaries is challenging and would need to be inferred from pre-European settlement conditions, remote and pristine comparable estuaries, or indigenous ecological knowledge. Regional Councils are facing challenges in determining appropriate reference states for State of the Environment (SOE) reporting, given that many coastal hydrosystems have undergone significant modifications due to landscape changes over time. These changes would align to pre-human reference states (before 1200), modifications during the pre-European period post-Māori burning (1840), and further modifications up to the present post-European settlement era[112]. Hindcasting estuary ecological states (ETI scores) using sediment cores is a promising method for gaining insight into identifying step changes and reference states of attributes of interest[42]. Satellite Chl.*a* in coastal-ocean waters, particularly in regions adjacent to remote and pristine coastlines (if any can be identified at present), may also reflect reference states, however, a thorough analysis is yet to be made. Coastal regions would need to consider adjacent coastal catchments, water mass types, physical oceanic processes, and seasonality to determine reference states and natural variability.

B3. Are there any existing numeric or narrative bands described for this attribute? Are there any levels used in other jurisdictions that could inform bands? (e.g., US EPA, Biodiversity Convention, ANZECC, Regional Council set limit)

The Australian and New Zealand Environment and Conservation Council (ANZECC) and the Australian Water Quality Guidelines for Fresh and Marine Waters provide guidance on Chl.*a* for maintaining ecosystem health, and "precautionary" default trigger values of 4 mg m^{-3} for estuaries and 1 mg m^{-3} for marine due to lack of data[1]. This was advanced in 2016 to a New Zealand relevant Estuary Trophic Index in 2016[92], adapted in 2020[88] and an extension proposed in 2022 to include coastal-ocean waters[33](Table 1). While there may not be universally (globally) established bands for Chl.*a*, some regional councils may set different limits or guidelines based on water quality standards or ecosystem health targets in their region, particularly in consideration to returning to reference state conditions.

B4. Are there any known thresholds or tipping points that relate to specific effects on ecological integrity or human health?

There are not any known "general" (i.e., national scale) thresholds or tipping points in Chl.*a* that relate to specific effects on ecological integrity or human health. It is likely that the significance of a change or particular level of Chl.*a* depends on the particular characteristics of the estuary and coastal hydrosystem, including the ecosystem products and services that humans value in that system. New Zealand lacks an understanding of appropriate "general level" national-scale Chl.*a* tipping points in estuaries and the coastal-ocean, except for ongoing efforts to streamline eutrophication thresholds and define the four attribute states outlined in Table 1, which aid in identifying states of concern. This underscores the importance of fundamental research in understanding processes as well as ecosystem goods and services in natural estuarine/coastal systems, as predicting tipping points is impossible without the underlying understanding[5].

B5. Are there lag times and legacy effects? What are the nature of these and how do they impact state and trend assessment? Furthermore, are there any naturally occurring processes, including long-term cycles, that may influence the state and trend assessments?

Yes, lag times and legacy effects can significantly impact state and trend assessments of Chl.*a* in estuarine and coastal waters. Legacy effects occur when past disturbances or inputs continue to influence Chl.*a* after the cessation of the initial driver. For example, changes from historical land use practices may continue to affect nutrient and sediment inputs in estuarine/coastal systems for years or decades after the change in land use occurred, contributing to change in Chl.*a*, potential eutrophication and algal blooms. These lag times and legacy effects can complicate state and trend assessments by masking short-term changes or obscuring the true extent of ecosystem degradation. Furthermore, climate variability and change, such as the El Niño-Southern Oscillation (ENSO) or the Pacific Decadal Oscillation (PDO), co-occurring with changes to land use or coastal development can influence Chl.*a* through changes in sea surface temperature, precipitation patterns, and ocean circulation, leading to variability in state and trend assessments over long-term cycles. Accounting for these complicating factors (multiple stressors) is needed to accurately interpret Chl.*a* data and inform effective management and policy decisions in coastal environments. It has been estimated that unambiguously separating the signal of anthropogenic climate change from decadal variability on ocean Chl.*a* and productivity may take 20-50 years of satellite observations so we are only just beginning to have time-series long enough to do this around New Zealand[44].

B6. What tikanga Māori and mātauranga Māori could inform bands or allocation options? How? For example, by contributing to defining minimally disturbed conditions, or unacceptable degradation.

Tikanga Māori and mātauranga Māori offer valuable insights for defining coastal ecosystems states, examples being the connection of location knowledge and palaeoecology[63], and supporting the sustainability of aquatic biological heritage[2]. These can relate to the productivity (Chl.*a*) of local coastal waters, for cultural, social, economic, ecological and ecosystem services, including customary harvest and health of keystone or kaitiaki species and taonga associated with marine resources.

Part C—Management levers and context

C1. What is the relationship between the state of the environment and stresses on that state? Can this relationship be quantified?

Macro-nutrients, light availability (which depends in part on suspended sediment), water-column structure and temperature are the main drivers of Chl.*a*, trophic state and HABs. Nutrient and sediment are affected through various pathways of contaminant transport (dissolved macro-nutrients and suspended sediment) either from adjacent land runoff and river discharge (particularly during stormflow events) or from coastal physical process such as mixing and upwelling transport of oceanic nutrients[10, 11]. Many of the drivers of Chl.*a* can be quantified at a local scale with adequate observation, knowledge and understanding on various time-scales (tidal, episodic, seasonal, interannual, decadal). Location-specific relationships and dynamics are linked to coastal land-catchment properties and inputs of river water quality[98, 54, Larned, 2020 #4202], the type of hydrosystem[47], and their physical oceanographic process connections[104]. Spatial maps using ecological Marine Environment Classification (MEC)[99] and marine Ecosystem Services (ES)[107] can provide useful context information at the national scale.

C2. Are there interventions/mechanisms being used to affect this attribute? What evidence is there to show that they are/are not being implemented and being effective?

C2-(i). Local government driven

A significant portion of nutrient loading in coastal waters originates from urbanization and pastoralization of adjacent land based on robust evidence of their effects on freshwater ecosystems[56] and recent analysis on nitrogen loads to aquatic receiving environments in comparison to regulatory criteria[100]. Regional Council interventions have hence tended to prioritize land-based actions to first enhance freshwater quality. Regional Councils face resource constraints while balancing obligations to monitor the effectiveness of regional policies and plans under the Resource Management Act (RMA), the MfE National Policy Statement for Freshwater Management (NPS-FM), the New Zealand Coastal Policy Statement (NCPS) and to meet community and tangata whenua priorities. These often conflict with the approaches necessary for a national overview of the state of the environment[39, 106]. The recent Envirolink report on challenges and needs of Regional Councils in freshwater monitoring[106], underscores the pressing need for increased scientific investment in environmental monitoring at the national scale. Key priorities included better quantification of land-use impacts, establishment of infrastructure to support community monitoring efforts, advancement in modelling techniques, and the development and utilization of remote sensing and environmental DNA (eDNA) technologies.

Regional Councils support and are often involved in partnerships with other agencies and groups proactively engaging mitigation efforts and research into remediating environmental damage and expected responses. One example is the Whatawhata Integrated Catchment Management (ICM) project in the Horizons region, which is the longest continuously monitored before and after control impact catchment scale study (since 1995) in New Zealand, tracking responses in stream ecosystem health and water quality[46]. A compilation of work to improve the health of estuaries (mountains to the sea) is provided by DOC as an interactive map with links to agency and community initiatives, strategic plans, plant lists and restoration guides[22]. Most Regional Councils have riparian management programmes and plans in place for landowners to achieve water quality and biodiversity outcomes by excluding livestock and planting natural vegetation.

C2-(ii). Central government driven

Central government actions have primarily arisen from Resource Management Act (RMA) environmental regulations and national policy statements produced by MfE for freshwater management (NPS-FM2020[70]) and DOC for coastal (NCPS2010[17]). Regional Councils, with communities and tangata whenua develop their plans and manage their regions from these policy statements. The National Objectives Framework (NOF) in the NPS-FM2020 sets out the process which requires identifying freshwater values and attributes, target states and monitoring and reporting requirements. The NPS-FM applies to receiving environments, such as estuaries and coastal waters where they are connected and affected by freshwater inputs. The stipulated response to degradation in an attribute (e.g., lake phytoplankton trophic state/ Chl.*a*) is to take action to halt or reverse the change, by making or changing a regional plan or preparing an action plan which must identify the deterioration cause, methods to address and evaluation of effectiveness. However, since the 1940's the science-policy interface has often functioned poorly in the face of continued environmental degradation, and this led to five imperatives for implementing the relatively new NPS-FM 2020: (1) inclusiveness, (2) partnership with Māori, (3) strategic planning, (4) funding mandate, and (5) authorising agency[55]. There are real challenges and opportunities ahead in the values, state, trends, and human impacts on New Zealand's freshwaters[39]. However, the demonstration of the cost effectiveness and benefits of a national riparian restoration programme in New Zealand[14], suggests that a pathway could be found toward effective improvements in trophic states (Chl.*a*) in both freshwater and marine receiving environments.

There is no direct funding from central government for environmental monitoring. MBIE has a range of investment funds that support scientific research effort and infrastructure directly impacting our state of the environment, knowledge, impacts of environmental stressors, mitigation opportunities and management levers. Non-contestable funding (Strategic Science Investment Fund, SSIF) provides long-term funding to research providers and some of this SSIF funding has supported long-term environmental monitoring and research. With the decline of SSIF funding in real terms, much of the long-term environmental time-series is under threat.

The National Science Challenges (2016-2026) advanced some specific objectives, projects, and tools applicable to improving water quality and providing management insight. Our Land and Water[65] has the Healthy Estuaries | Ki Uta Ki Tai programme to assess the interaction between loadings of different contaminant from freshwaters on the health and functioning of estuaries. Mapping of Freshwater Contaminants programme has a series of projects to classify landscapes and management practices according to their contaminant delivery from source to sink. The Monitoring Freshwater Improvement Actions programme is monitoring and effectiveness of interventions and mitigation actions and the development of a WebApp. Sustainable Seas[66] also has several relevant programmes such as Degradation and Recovery (assessing cumulative effects caused by human activities and potential for recovery), and a range of ecosystem-based management (EBM) approaches: improving decision making (embedded in marine management and governance), enhance practices (tailoring practice, policy, regulation, and legislation), and in action (real world trials with stakeholders and Māori). With the NSC coming to an end, and no replacement, advances in understanding and time-series observation of estuarine/coastal Chl.*a* is likely to slow.

C2-(iii). Iwi/hapū driven

Many of the programmes mentioned above in National Science Challenges have strong collaboration with iwi and hapū Māori. Specifically, in Sustainable Seas the "Empowering Mana Moana" programme focussed on marine management and governance; in Our Land and Water included a component "Matarau: Empowering Māori Landowners in Land Use Decisions". The DOC "Monitoring estuaries map"[20] and "Restoring estuaries map"[20], also illustrate historical and active hapū and iwi efforts with relevance to improving water quality and estuarine health. One example is the Omarumutu Marae Papakainga and Waiaua Estuarine Restoration Project[81].

C2-(iv). NGO, community driven

The DOC "Monitoring estuaries map"[20] and "Restoring estuaries map"[20], documents a range of historical and active community driven effort with relevance to improving water quality and estuarine health. One example is the Friends of Mangemangeroa, a community group restoring native vegetation in the valley in river catchment to improve water quality entering the estuary [64]. Fonterra also have a range of sustainable catchment projects around the country with various central and region government agencies, iwi, scientists, and other primary sector organisations with the national movement on catchment restoration[29].

C2-(v). Internationally driven

There are a range of indirect interventions and international mechanisms relevant to monitoring, understanding and managing factors affecting Chl.*a* in estuaries and coastal water surrounding New Zealand. The Ministry of Foreign Affairs and Trade (MFAT) serves as New Zealand's representative in global discussions and the implementation of international agreements concerning marine governance and fisheries management[69] (which are affected by changes to Chl.*a*). Notably, MFAT plays a key role in upholding agreements such as: The UN Convention on the Law of the Sea (UNCLOS), which governs activities within New Zealand's Exclusive Economic Zone (EEZ) and continental shelf; Marine biological diversity beyond national jurisdiction (high seas); Ocean Acidification and its impacts.

New Zealand is a signatory to the United Nations 2030 Agenda for Sustainable Development Goal 6, which specifically addresses clean water and sanitation[108]. While it does not provide a direct legal obligation to New Zealand, it reflects the countries' commitment to ensuring access to clean water and improving water quality.

New Zealand is also a party to the Ramsar Convention on Wetlands[90]. The Convention aims to conserve and sustainably use wetlands, including freshwater and estuarine ecosystems. There are several designated several Ramsar sites, which are internationally recognized wetlands of importance (Farewell Spit, Firth of Thames, Kopuatai Peat Dome, Manawatu Estuary, Awarua Waituna Lagoon, Wairarapa-Moana, and Whagamarino wetland) [21].

Part D—Impact analysis

D1. What would be the environmental/human health impacts of not managing this attribute?

Unmanaged eutrophic Chl.*a* in estuarine and coastal waters can lead to harmful algal blooms, oxygen depletion, and degraded water quality, posing risks to aquatic ecosystems, human health, and Māori cultural practices.

D2. Where and on who would the economic impacts likely be felt? (e.g., Horticulture in Hawke's Bay, Electricity generation, Housing availability and supply in Auckland)

Negative economic impacts of eutrophic states of Chl.*a* in estuarine and coastal waters can arise due to adverse effects on: (1) tourism; (2) coastal/inshore wild-caught fisheries (finfish and shellfish; commercial, recreational and cultural fisheries); (3) aquaculture; and (4) water-based recreational industries in coastal regions. Immediate effects include tourism declines and aquaculture and fisheries closures, particularly due to algal blooms. Longer-term implications may involve shifts in higher trophic organisms (e.g., foraging birds, fish, and migrating species such as sharks, dolphins, and whales), tourism patterns and changes in aquaculture practices, requiring ongoing monitoring and research for mitigation.

D3. How will this attribute be affected by climate change? What will that require in terms of management response to mitigate this?

Climate change is likely to significantly impact trophic state and Chl.*a* in estuarine and coastal systems waters [89]. Warmer temperatures will likely stimulate algal growth by increasing the metabolic rates of phytoplankton, but effects of climate change will also be felt through a range of other processes. Alterations in nutrient cycling patterns will occur from increased rainfall and runoff, leading to the transportation of higher amounts of nutrients like nitrogen and phosphorus into water bodies. Additionally, changes in water temperature, wind, waves and precipitation will alter vertical stratification (water-column structure) and changes to mixed layer depths which affect nutrient availability and light availability, with effects on primary production and Chl.*a*. In extreme cases these will lead to reduce water quality, clarity, HABs, modification of habitat and loss of natural resources, and periods of oxygen depletion (hypoxia). Climate change will also affect zooplankton grazers of phytoplankton and microbial processes, thereby affecting loss rates and export of algal carbon to depth.

In summary, Chl.a in New Zealand estuaries and coastal waters are likely to be sensitive to the effects of climate change, and yet the processes involved are complex, overlapping, poorly-observed or understood, and variable in time and space[12,57]. Climate change drivers of changes in Chl.*a* will act in addition to changes arising from land-use, hydrological and coastal development.

Global moves towards zero carbon dioxide emission and potentially carbon dioxide removal (CDR) strategies would help slow (and potentially reverse) global climate change. In the meantime, management options for Chl.a in estuarine and coastal systems are related to measures aimed at enhancing catchment resilience by addressing land-use practices, landscape management, and hydrological processes[82]. Reforestation, particularly in low-land areas, implementation of riparian buffers, and enhancement and protection of wetlands will help to reduce soil erosion, nutrient runoff, and sedimentation into coastal water bodies. Additionally, enhancing soil health through practices like conservation agriculture and cover cropping can improve water retention and reduce the leaching of nutrients and contaminants into streams and estuaries[95].

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9.14 Dissolved oxygen in water (trophic state)

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State of Knowledge of the "Dissolved oxygen conc. in estuary/coastal water" attribute: Excellent / well established – comprehensive analysis/syntheses; multiple studies agree.

Part A—Attribute and method

A1. How does the attribute relate to ecological integrity or human health?

There is a strong record of evidence in New Zealand and globally to show that dissolved oxygen content in seawater relates to ecological integrity of coastal waters [1-3]. 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 [4]. Tolerance of marine biota to oxygen depletion is well understood; some marine and estuarine species are more tolerant to low oxygen levels than others [5], so changes in oxygen availability change which species are present in coastal waters [1, 6]. Increased respiration during decomposition of organic material tends to drive down oxygen levels. This process typically shows daily and seasonally cycles - oxygen minima occur at times when respiration exceeds primary production (photosynthesis) [4].

A2. What is the evidence of impact on (a) ecological integrity or (b) human health? What is the spatial extent and magnitude of degradation?

There is strong evidence of impact of seawater hypoxia on biological integrity of coastal waters globally [2, 7, 8], and in New Zealand [3, 9, 10]. Recorded impacts include mortality events such as fish kills but can include changes to growth, movement, and behaviour of marine organisms. It is well documented that the magnitude of these impacts are not consistent spatially. Waters with the greatest tendency to become hypoxic are those that receive high nutrient loads [11], and those that stratify [12]. Relatively deep and poorly flushed hydrosystems have greater tendency to stratify [13] and thus deep waters within these systems have a greater tendency to become hypoxic. However, reductions in oxygen content of waters have also been observed in well-mixed estuaries receiving high nutrient loads from land [14]. Reduced seawater oxygen concentrations are also common in waters around intensive marine aquaculture, especially those that add large amounts of nutrients to confined areas such as fish farms [15, 16]. Reduced seawater oxygen concentrations are known to impact aquaculture production in some species [17].

A3. What has been the pace and trajectory of change in this attribute, and what do we expect in the future 10 - 30 years under the status quo? Are impacts reversible or irreversible (within a generation)?

Reduction in dissolved oxygen in coastal waters is largely a result of increased flows of nutrients (which increases growth of plants and algae) [18] and direct addition of labile organic matter from land. These additions have increased due to land use change globally, including in New Zealand, with worsening impacts to dissolved oxygen levels [3, 10, 11]. The number of hypoxic zones globally in the coastal margin is approximately doubling every decade [19]. We would expect the trajectory of this attribute to track the future pace and trajectory of loading of nutrients and organic material from land to the ocean. However, several factors may affect that relationship. The susceptibility of ecosystems depends on factors including warming of coastal waters, the depth of the water body, and the tendency of waters within the water body to mix [20]. Water bodies that stratify will tend to develop zones of low oxygen availability near the seabed, where declines in seawater oxygen levels caused by respiration are not balanced by oxygen produced by photosynthesis (which occurs mostly in surface waters). Increasing seawater temperatures are expected to exacerbate coastal de-oxygenation by reducing the solubility of oxygen in seawater, increasing ecosystem metabolism rates, and increasing the tendency of the ocean to stratify [20, 21]. The degree of reversibility of hypoxia in coastal waters depends on flushing, as well as organic content of bottom sediments.

A4-(i) What monitoring is currently done and how is it reported? (e.g., is there a standard, and how consistently is it used, who is monitoring for what purpose)? Is there a consensus on the most appropriate measurement method?

Most monitoring of coastal waters carried out by regional council scientists uses in situ probe measurement of dissolved oxygen concentration and saturation [22-26]. These samples are almost always carried out during the day, in the top 30 cm of the water column, and are most commonly at monthly frequency. This sampling is likely to miss most of the 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. There is a standard for measurement of dissolved oxygen in coastal waters [27]. Measurement of dissolved oxygen across depth profiles (e.g., using boat-deployed instruments) is carried out more rarely [22]. Continuous monitoring, e.g., using sensors deployed on moorings, or attached to submerged structures, is carried out by some councils and the feasibility of this approach is being investigated by others [28].

A4-(ii) Are there any implementation issues such as accessing privately owned land to collect repeat samples for regulatory informing purposes?

While depth profile sampling is the most appropriate method of monitoring coastal waters for oxygen depletion, expense hinders its use. Depth profile sampling typically can't be done from the shore unless a suitable structure such as a bridge is in a useful position across an estuary. Because most (but not all) regional council state of the environment (SoE) sampling for coastal water quality is conducted from shore or helicopter [22], depth profile sampling of dissolved oxygen levels would require considerable extra expense.

A4-(iii) What are the costs associated with monitoring the attribute? This includes up-front costs to set up for monitoring (e.g., purchase of equipment) and on-going operational costs (e.g., analysis of samples).

Up-front costs differ depending on measurement method. For discrete sampling from land, a hand-held dissolved oxygen sensor is required [27]. Deployable dissolved oxygen loggers are also available and some offer both relative reliability and low cost (e.g., PME Minidot loggers cost ca. $5,000 per unit). Upfront costs are markedly higher for depth-profile sampling performed from boats/ships. In addition to purchase or hire of a boat and qualified crew, the instruments used (e.g., YSI EXO Sondes with dissolved oxygen measurement capability) typically cost in the order of $NZ40,000. If sensors for continuous long-term measurement are attached to moorings, the mooring itself requires substantial additional up-front costs. All dissolved oxygen sensor options require periodic calibration, as well as staff expertise for measurement and interpretation of data, databasing and reporting. Deployed sensors for continuous long-term measurement in coastal environments require periodic maintenance in the order of 1-2 months, including cleaning, batteries, and recalibration.

A5. Are there examples of this being monitored by Iwi/Māori? If so, by who and how?

We (the authors and John Zeldis, also of NIWA) are not aware of any monitoring being carried out by representatives of iwi/hapū/rūnanga. The exception may be Māori-owned marine businesses (e.g., green-lipped mussel farmers) who may be required to monitor water quality including DO as part of consent conditions.

A6. Are there known correlations or relationships between this attribute and other attribute(s), and what are the nature of these relationships?

Eutrophication is a main cause of fluctuations in dissolved oxygen. Because of this, seawater dissolved oxygen tends to co-vary with other seawater indicators of eutrophication. These include chlorophyll *a* concentrations, and total nutrient concentrations (e.g., total nitrogen (TN), and total phosphorus (TP))[29]. Seawater hypoxia and acidification also co-occur in coastal waters globally because respiration of organic matter increases dissolved inorganic carbon (DIC) and drives down pH and O_2 [3, 30]. These respiration-driven reductions in pH can significantly outpace other drivers of ocean acidification [31], and reach levels that cause substantial harm to marine life [32].

Part B—Current state and allocation options

B1. What is the current state of the attribute?

The current state of seawater dissolved oxygen in New Zealand’s deep (stratifying) estuaries is not well understood because of practical issues associated with sampling. As described above, depth profiles of seawater dissolved oxygen are expensive to measure and high frequency or continuous monitoring is required to measure diel fluctuations in dissolved oxygen concentration. For these reasons there is a lack of national-scale monitoring at the sites and times where problems are likely to occur. However, limits/thresholds for life are well understood [5]. Measurement methods are well established, and we have good understanding of state in a few areas of New Zealand [3]. Dissolved oxygen could be used as a national indicator.

B2. Are there known natural reference states described for New Zealand that could inform management or allocation options?

Yes. Those systems with consistently high but not supersaturated oxygen saturation values provide reference conditions. Reference conditions sit consistently near 100% saturation. Undersaturation associated with other indicators of eutrophication (e.g., high TN, TP, chlorophyll-*a*, DIC) indicates a problem.

B3. Are there any existing numeric or narrative bands described for this attribute? Are there any levels used in other jurisdictions that could inform bands? (e.g., US EPA, Biodiversity Convention, ANZECC, Regional Council set limit)

Yes, this is well understood. Metadata studies provide thresholds of dissolved oxygen concentrations required for survival of various taxa across hundreds of estuaries [5]. The 2000 ANZECC guidelines give maximum and minimum values for dissolved oxygen in estuaries at 110% and 80% saturation, respectively [33]. Regional councils also provide site-specific standards for estuaries (e.g., [34]).

B4. Are there any known thresholds or tipping points that relate to specific effects on ecological integrity or human health?

Yes, for ecological integrity there is consensus on the requirements for life for varying taxa [5, 21, 35]. This attribute relates to those species that take their oxygen from seawater so does not relate directly to human health.

B5. Are there lag times and legacy effects? What are the nature of these and how do they impact state and trend assessment? Furthermore, are there any naturally occurring processes, including long-term cycles, that may influence the state and trend assessments?

It is likely that lag-times to both hypoxia onset and recovery are site-dependent, and recovery of ecosystems tends to be more linear in response to reductions in nutrients and labile organic matter when the reductions are from point sources (such as sewage outfall diversion), rather than attempts to reduce diffuse sources [36]. System flushing and muddiness are also likely controlling factors of recovery lag times [37]. As described above, cyclical or long-term changes in climate that affect water temperatures (and thus water column stratification and ecosystem carbon balance) are also likely to affect seawater dissolved oxygen levels.

B6. What tikanga Māori and mātauranga Māori could inform bands or allocation options? How? For example, by contributing to defining minimally disturbed conditions, or unacceptable degradation.

Mana whenua have long advocated for more holistic approaches to inform estuarine and coastal health (e.g., ki uta ki tai) and this drive has seen for instance efforts towards understanding ecological condition and the need for better protection of significant areas such as Ōreti (New River Estuary; e.g., [50]. A key example of how mana whenua have shaped the approaches to improving management for land, freshwater and estuaries are evident within Murihiku (aka Southland). For instance, having estuaries included within Freshwater Management Units (FMUs) have been strongly advocated for by iwi, including Ngāi Tahu ki Murihiku [51, 52]. The involvement of mana whenua within decision-making, including the provisions of management policy statements (e.g., NPSFM), and the values set out within Iwi Environmental Management Plans is essential. It is therefore advised that an approach towards developing bands and allocation is done more appropriately. The requirement for engagement and collaboration with mana whenua is shared in the following example, where a multi-disciplinary study (mātauranga and Western science), co-lead with kairangahau Māori (who have expertise within the economic, freshwater ecology, marine ecology and mātauranga Māori) within the National Science Challenge, have suggested expanding beyond upstream, leading with three steps: (1) understanding iwi aspirations for place, (2) estuarine ecologists being able to identify freshwater contaminant thresholds or load limits for achieving or moving towards those aspirations, and (3) catchment modellers determining the necessary mitigations or changes in land use to achieve the necessary loads [53].

Part C—Management levers and context

C1. What is the relationship between the state of the environment and stresses on that state? Can this relationship be quantified?

Process relationships between loading of nutrients and organic matter to estuaries, eutrophication, and impacts to seawater dissolved oxygen content are well understood [3, 39, 40]. The New Zealand Estuarine Trophic Index gives an example of tools to quantify these relationships in a management context [41, 42], however links within these tools between nutrient loading and seawater oxygen levels are indirect (based on a Baysian belief network [43]) and tenuous due to lack of available data. See https://shiny.niwa.co.nz/Estuaries-Screening-Tool-1/

C2. Are there interventions/mechanisms being used to affect this attribute? What evidence is there to show that they are/are not being implemented and being effective?

Key mechanisms to affect this attribute are controls on nutrient loading initiated by councils to give effect to the (central government initiated) National Policy Statement for Freshwater Management (NPSFM), and resource consents on point sources of nutrients (e.g., wastewater).

C2-(i). Local government driven

Diversion of the Christchurch wastewater treatment plant outflow to the Ihutai (Avon-Heathcote) estuary provides perhaps the best example nationally of the potential to improve seawater oxygen content by removing point source discharges. This diversion represented a reduction of around 90% of the total nitrogen load to this estuary. Seawater and sediment chemistry, and indices of primary production in the estuary were measured before and after the diversion [37, 44, 45]. The diversion resulted in improvement in oxygen content of waters at the sediment/seawater interface at sites near the outfall.

C2-(ii). Central government driven

There is currently little evidence to suggest that controls on nutrient loading initiated by councils to give effect to the NPSFM are causing improvement in dissolved oxygen levels in estuaries; despite long-term trends indicating strongly that dissolved oxygen levels in estuaries are increasing nationally [22, 23]. This is because council sampling is carried out in surface waters during daylight hours, where we would expect photosynthesis to drive oxygen saturation. As such, these increasing trends in seawater dissolved oxygen are not necessarily reflective of improving ecological integrity of estuaries, in fact, the opposite may be true if increasing daytime productivity is balanced by higher respiration/decomposition rates at night and in deeper waters [3]. Long-term, continuous sampling of seawater oxygen content across depth gradients is required to infer changes in this attribute. Conducting these measurements over timescales when changes in nutrient loads take place would provide evidence to show whether load changes are effective in improving seawater oxygen conditions.

C2-(iii). Iwi/hapū driven

We are not aware of interventions/mechanisms being used by iwi/hapū/rūnanga to directly affect this attribute. However, there are many examples of where variables, such as oxygen/dissolved oxygen (and indicated by for instance changes in colour and smell), are considered within a holistic approach to supporting estuarine and coastal health (e.g., [54, 55]). For instance, within land to sea management plans led and co-developed with hapū and iwi towards informing approaches towards key indicators for supporting environmental health and associated culturally important ecosystems (e.g., [54]). It is difficult to measure the improvements given the legacy issues in estuaries, and the more recent collaborations that acknowledge a systems approach is required [53].

C2-(iv). NGO, community driven

We are unaware of initiatives to improve dissolved oxygen in coastal waters being carried out by representatives of NGOs.

C2-(v). Internationally driven

We are unaware of international initiatives that are driving improvement of dissolved oxygen conditions in coastal waters.

<u>Part D—Impact analysis</u>

D1. What would be the environmental/human health impacts of not managing this attribute?

Changes in the attribute state affect ecological integrity as described in A1 above. Not managing eutrophication processes in coastal waters to provide sufficient oxygen for biota will likely lead to degradation of inshore fisheries, including mahinga kai species, especially in already degraded environments [46, 47]. Reduced oxygen will continue to contribute to species loss and displacement, and stress ecological function of eutrophic waters [48]. Frequency of hypoxia-driven fish kills is likely to increase [49].

D2. Where and on who would the economic impacts likely be felt? (e.g., Horticulture in Hawke's Bay, Electricity generation, Housing availability and supply in Auckland)

Impacts have long been acknowledged and addressed by hapū and whānau, for instance Ngāi Tahu whānau and hapū in relation to estuarine and coastal ecosystems, including inshore fisheries [55-58].

These impacts may be direct (e.g., via fatally hypoxic waters [5]), or indirect (e.g., via decreased food availability for fisheries species [48]). Demersal and benthic species (those that live and feed on or near the bottom of seas) are likely to be disproportionally affected [2]. Reduced seawater oxygen concentrations are known to impact aquaculture production in some species [17].

D3. How will this attribute be affected by climate change? What will that require in terms of management response to mitigate this?

Even under current loading levels of nitrogen to coastal waters from land, increasing seawater temperatures are expected to exacerbate coastal de-oxygenation both by reducing the solubility of oxygen in seawater, increasing ecosystem metabolism rates, and increasing the tendency of the ocean to stratify [20, 21]. An appropriate management response would be to manage loads of organic material and nitrogen from land to levels that are unlikely to cause damaging hypoxia in coastal waters, with sufficient tolerance to negate climate-change-driven effects.

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9.15 Nutrients in water (trophic state and toxicity)

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State of knowledge of the "Nutrients in water (trophic state and toxicity)" attribute: Excellent / well established – comprehensive analysis/syntheses; multiple studies agree.

Part A—Attribute and method

A1. How does the attribute relate to ecological integrity or human health?

A key impact of increased nutrient loads from land to estuaries and coastal waters is increases in coastal eutrophication, the process whereby the extra nutrients stimulate excessive primary production. 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 [1-3]. Models suggest that phosphorus (P) may limit phytoplankton blooms in a smaller fraction of New Zealand estuaries [4, 5].

As eutrophication progresses, the excessive production of aquatic plants and algal biomass result in an over-accumulation and respiration 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 [6, 7]. In coastal waters, these effects include increased water column and sediment hypoxia and anoxia and acidification[8], degradation of benthic habitat quality, and reductions in biodiversity [2, 7, 9]. A further indirect impact of increased N and P (hereafter 'nutrient') loads and nutrient ratios can be increases in the abundance of toxic algal species in coastal waters, including those of New Zealand [10].

Nutrients are present in the waters of estuaries and other coastal waters in a variety of chemical forms; these are summarised in Table 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 [7]. 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 trophic state changes [11]. For these reasons, measured concentrations of the water column nutrient forms listed in Table 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 [4, 12, 13], or biological indices of nutrient availability [7, 14-16] are commonly preferred metrics for quantifying relationships between nutrient pressure and trophic response in coastal waters.

Table 1. Nutrient species components of nutrient loads to coastal waters. Adapted from Sutula, Fong [7]

Form	Components of Total Nitrogen	Components of Total Phosphorus
Dissolved Inorganic	Nitrate (NO_3^-) + nitrite (NO_2^{2-})	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.
	Ammonium (NH_4^+; in dynamic equilibrium in natural waters with unionized or free ammonia)	
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 organic phosphorus (detritus left from undecayed or partially decayed organic matter)
	Particulate inorganic nitrogen (insignificant in natural waters and usually not considered)	Particulate inorganic phosphorus (typically associated with minerals)

With the exception of ammonia, and to less extent nitrate and urea, nutrients are generally not considered to directly impair beneficial uses of estuaries [7]. Ammonia and nitrate can be toxic to estuarine flora such as seagrasses at high concentrations [17-19] and high concentrations of ammonia are toxic to fauna such as fishes [20]. For direct toxic effects, concentrations of nutrients can be related to ecological impacts through laboratory testing. For example, the ANZECC [21] guidelines provide default values for ammonia toxicity.

A2. What is the evidence of impact on (a) ecological integrity or (b) human health? What is the spatial extent and magnitude of degradation?

There is a very strong record of evidence in New Zealand and globally to show that increased nutrient inputs to land and their subsequent passage via freshwater flows (from both diffuse and point sources) to estuaries relate to ecological integrity of coastal waters [4, 6, 22, 23]. Studies provide clear and consistent evidence that nutrient loads to land in New Zealand have changed greatly in the past two centuries, and that much of this additional load passes via freshwater flows to increase nutrient concentrations in coastal waters [5, 24-26]. At regional scale, spatial extent and magnitude of coastal degradation follows patterns of increased nutrient availability in New Zealand estuaries [4, 5, 11, 23, 26] and estuaries globally [27-30]. In part because coastal eutrophication is so widespread, there is also a good understanding of the physical attributes of individual estuaries that increase or reduce the magnitude of degradation in ecological integrity resulting from increased nutrient load pressure from land. Important physical attributes include dilution within and flushing of water from estuaries. Higher residence times of water within estuaries allow nutrients to be taken up by algae, and for phytoplankton to multiply before being flushed to the ocean [4, 7, 31]. Geomorphological attributes are also important – for example, sandy estuaries have been shown to be more resilient to eutrophication than muddy ones[32], and lagoonal estuaries are more sensitive to macroalgal blooms than riverine estuaries[4].

New Zealand's most nutrient-enriched estuaries provide the strongest evidence of trajectories of trophic change resulting from increases in nutrient availability. For example, the Firth of Thames, which receives nutrient loads ca. 82% higher than it did in its pre-human state [11, 24], has in recent decades shown ecological and biogeochemical changes due to eutrophication. These changes include acidification and hypoxia [11], increased proliferation of toxic algal species [10], and reduced denitrification that reduces the capacity of this system to assimilate N loading [33]. The New River Estuary in Southland provides an example of a shallow, intertidal dominated (lagoonal) estuary that now receives nutrient loads many times higher than it did in its pre-human state [23]. That estuary has shown eutrophication impacts over the last 100 years that have closely tracked the rate of nutrient additions from land [15, 23, 34].

Nutrient concentrations in coastal waters do not relate directly to human health.

A3. What has been the pace and trajectory of change in this attribute, and what do we expect in the future 10 - 30 years under the status quo? Are impacts reversible or irreversible (within a generation)?

As described in the section above, New Zealand has examples of highly eutrophic estuaries that provide trajectories of eutrophication under time series of increasing nutrient loads [8, 10, 23]. The number of hypoxic zones globally in the coastal margin resulting from increases in coastal nutrient availability is approximately doubling every decade [35]. We would expect the trajectory of eutrophication impacts to track the future pace and trajectory of loading of nutrients and organic material from land to the ocean. Factors that may affect this relationship include interactions between nutrient availability and climate warming; for example Tait, Zeldis [36] documented increased severity of macroalgal blooms in Avon Heathcote Estuary (Canterbury) in years of sea surface temperature anomalies. Notably, within this same estuary the anomalous periods of high water temperatures also corresponded to periods of lower nutrient concentrations in surface waters [37] (despite little change in nutrient <u>loads</u> to estuaries) as nutrients were rapidly taken up by primary producers. This provides evidence that eutrophication impacts were not well-described by measured seawater nutrient concentrations.

The time necessary for remediation and/or recovery of estuaries is likely to increase if nutrient loads to New Zealand's estuaries are kept at current levels. The first reason for this is that physical and chemical conditions in many estuaries are degrading under current nutrient loads. This degradation causes feedbacks that hinder subsequent recovery, as described below in section B5.

Pace and trajectories of change of measured nutrient concentrations in New Zealand estuaries are regularly analysed at a site level using trend analyses in reports for MfE [38-40]. Trend analyses have all indicated that phosphorus concentrations (as soluble reactive phosphorus (SRP) and total phosphorus (TP)) have decreased at most sites across New Zealand estuaries in the last 10-15 years. Nitrogen trends are less uniform in direction, with nitrate concentrations decreasing at most sites, while ammonium has shown increases at many sites.

A4-(i) What monitoring is currently done and how is it reported? (e.g., is there a standard, and how consistently is it used, who is monitoring for what purpose)? Is there a consensus on the most appropriate measurement method?

Most monitoring of nutrient content of coastal waters in New Zealand is carried out by regional council scientists via collection of discrete samples of water [38, 39, 41-43]. These samples are almost always carried out during the day, in the top 30 cm of the water column, and are most commonly at monthly frequency. There is a standard for measurement of nutrients in coastal waters [44].

As described above, nutrients entering estuaries from land can be rapidly taken up by primary producers (particularly during summer months), so temporal patterns (trends) of water column nutrient concentrations monitored in this way may carry considerable noise [7, 11]. Measurement of nutrient loads at terminal reaches entering estuaries is carried out more rarely, but would be useful for managing catchment nutrient loads to control eutrophication in estuaries [41].

A4-(ii) Are there any implementation issues such as accessing privately owned land to collect repeat samples for regulatory informing purposes?

Methods to model the impact of changes to nutrient loads on ecological integrity of estuaries such at the Estuary Trophic Index (ETI) use mixing models, which require determinations of concentrations of nutrients in open ocean coastal water and terminal river reaches, as well as river flow. Some regional council state of the environment (SoE) sampling for coastal water quality is conducted by helicopter [39], which facilitates measurement of concentrations of nutrients in open ocean waters. However, offshore ocean sampling is still not common across regional councils due to its expense. Terminal river reach sampling is not common amongst regional council monitoring programmes and so would require extra expense.

A4-(iii) What are the costs associated with monitoring the attribute? This includes up-front costs to set up for monitoring (e.g., purchase of equipment) and on-going operational costs (e.g., analysis of samples).

Up-front costs differ depending on measurement method. For discrete sampling from land the major costs are labour, transport, shipping, and laboratory analyses. Laboratory analysis costs are currently roughly NZ$100 in total per sample for the dissolved <u>inorganic</u> nutrients listed in Table 1, and total nitrogen (TN) and total phosphorus (TP) content. Labour, transport, and shipping will be site dependent. Costs are markedly higher for sampling performed from boats/ships which would require purchase or hire of a boat and labour costs of qualified crew.

All nutrient analysis requires some staff expertise for sample collection and interpretation of data, databasing and reporting. If nutrient loads from terminal river reaches are required, flow measurements are required in addition to nutrient concentrations. Setup and maintenance of a flow station would cost roughly NZ$20,000 for its first year of operation.

A5. Are there examples of this being monitored by Iwi/Māori? If so, by who and how?

We are not aware of any nutrient concentration measurements in coastal waters being carried out by representatives of iwi/hapū/rūnanga Māori groups. The exception may be Māori-owned marine businesses (e.g., green-lipped mussel farmers) who may be required to monitor water quality including nutrients as part of consent conditions.

A6. Are there known correlations or relationships between this attribute and other attribute(s), and what are the nature of these relationships?

As described above, nutrient availability in seawater exerts control over accumulation of labile organic matter in surface waters and sediments, the balance of basic biogeochemical cycles in sediments and surface waters, and a cascade of ecological processes [6, 7]. Some of the other indicators covered in this report are known to relate to biogeochemical and ecological processes affected by eutrophication. These include seagrass health and extent [45], macroinvertebrate community composition [46, 47], water clarity [48, 49], phytoplankton / chlorophyll *a* in water [4, 50], and dissolved oxygen content of water [8]. Other indicators used in tools to manage eutrophication in estuaries include sediment organic matter and redox potential depth [12, 51].

Because nutrient availability can limit growth of marine algae, some algal species have evolved to rapidly assimilate nutrients into their tissues [52, 53]. As a result, measured concentrations of nutrients in seawater can relate poorly to the impact of nutrients on ecosystems [7]. Across tools to assess eutrophication impacts on estuaries in New Zealand and overseas, seawater nutrient concentrations are often not included as an indicator or are grouped alongside a suite of other indicators. Those that group nutrients alongside other indicators of eutrophication include the Assessment of Estuarine Eutrophic Status (ASSETS) approach for US estuaries [54, 55], and its updates [56]. Measured water column nutrients concentrations are not included as an indicator of eutrophication in the ETI Tools, which instead relate nutrient loads (adjusted for dilution within and flushing from estuaries) to other indicators of eutrophication including those listed above [12]. Similarly, the modification of ASSETS developed for Spanish Basque Country Water Framework Directive (WFD-BC) estuary evaluations [57] also adjusts nutrient load based on dilution and flushing as an index of nutrient 'pressure'.

Part B—Current state and allocation options

B1. What is the current state of the attribute?

Current state of nutrient concentrations in coastal seawater is quite well understood at the national scale. Nutrient concentrations in coastal waters are regularly monitored by most regional councils, and summary reports of regional council data (including state and trend analyses) have been prepared several times in the last 10 years [38-40]. High concentrations of nutrients identified in national reporting correspond to known areas of coastal eutrophication, notwithstanding the noise often found in nutrient trends noted above. In particular, New Zealand's urban estuaries feature as exhibiting both nutrient pressures [26] and eutrophication symptoms [4]. Collation and analysis of regional council data from across New Zealand has shown that land-sourced nutrient pollution, conveyed by rivers, is the main cause of degraded water quality in New Zealand estuaries [26].

B2. Are there known natural reference states described for New Zealand that could inform management or allocation options?

There are few rivers that drain to the sea in New Zealand unaffected by land development for agriculture or urbanisation within their catchments. As a result, there are few estuaries known to exhibit reference state trophic conditions in New Zealand. However, the study of Plew, Dudley [5] mapped both historical and current trophic state of New Zealand estuaries, while the study of Snelder, Larned [24] mapped anthropogenic increases in nutrient concentrations to New Zealand rivers. These studies could be used to attempt to find estuaries that exhibit reference trophic state conditions. Such estuaries are likely to be backed by catchments that retain reference state (pre-human) landcover, so could be indicated by landcover analyses.

B3. Are there any existing numeric or narrative bands described for this attribute? Are there any levels used in other jurisdictions that could inform bands? (e.g., US EPA, Biodiversity Convention, ANZECC, Regional Council set limit)

As described above, various approaches for managing eutrophication impacts include bands of nutrient concentrations or loads to estuaries as a quantification of nutrient 'pressure' [54-57]. The New Zealand National Policy Statement for Freshwater Management 2020 (NPSFM) requires local authorities to "manage freshwater, and land use development, in catchments in an integrated and sustainable way to avoid, remedy or mitigate adverse effects on the health and well-being of water bodies", including estuaries [58]. The NPSFM provides target levels for various water quality parameters in freshwater bodies but does not do so for estuaries. Instead, it requires local authorities to determine the nutrient limits needed to achieve desired environmental outcomes for estuaries. Some regional council plans include target concentrations of nutrients in coastal waters to achieve these outcomes. These councils include Horizons Regional Council [59, 60], Northland Regional Council [41, 61] and Waikato Regional Council (see https://www.waikatoregion.govt.nz/assets/WRC/ProposedRegionalCoastalPlan.pdf).

ANZECC [21] provides default guideline values for nutrient concentrations, however these have been found to be inappropriate for some regions (for example where naturally occurring nutrient concentrations from oceanic seawater exceed the guideline values). The updated guidelines (see https://www.waterquality.gov.au/anz-guidelines) recommend developing statistically based bandings based on local measurements. This has been carried out to develop guideline nutrient concentrations for some areas of New Zealand [42, 62].

The ETI and similar approaches (e.g., Garmendia, Bricker [57] and the Dissolved Concentration Potential (DCP) approach [63]) include numeric bands of calculated 'potential' nutrient concentrations corresponding to bands of trophic state and other indicators of eutrophication [4, 12, 64]. While estimating 'potential' concentrations avoid issues relating to uptake or chemical transformation of water column nutrients, they are subject to error associated with calculation of nutrient loads to estuaries, measurement of oceanic nutrient concentrations, and estuarine mixing. They also do not measure the same parameter as nutrient concentrations obtained by within-estuary grab sampling, as they measure nutrients potentially available to primary producers [64] with bandings set using cases of measured, co-occurring trophic response [4, 12].

B4. Are there any known thresholds or tipping points that relate to specific effects on ecological integrity or human health?

There are known thresholds for direct toxic effects, but not for eutrophication effects.

For direct toxic effects, concentrations of nutrients can be related to impacts through laboratory or field testing. For example, the ANZECC [21] guidelines provide default values for ammonia toxicity. Numerous studies have examined direct toxic effects of other nutrients on estuarine biota (See section A1) from which guidelines can be established.

At a whole-of-estuary scale, impacts to trophic state tend to worsen progressively with increasing nutrient loads, without clear tipping points [4, 31, 57]. An example is the linear relationship between potential nutrient concentration and macroalgal ecological quality rating (EQR; [4]).

That said, the underlying dose-responses of biota such as macroalgae are non-linear, for example, asymptotic growth responses of macroalgae to nutrient dose. This means that ecological damage associated with eutrophication is likely to increase rapidly at low inorganic nitrogen concentrations [34, 65]. Furthermore, tipping points have been identified in the relationship between macroalgal biomass and sediment accretion as described in the next section. This represents a tipping point in that the recovery from the impacted state will not follow the same trajectory as when the problems developed [23, 66].

B5. Are there lag times and legacy effects? What are the nature of these and how do they impact state and trend assessment? Furthermore, are there any naturally occurring processes, including long-term cycles, that may influence the state and trend assessments?

The combination of high nutrient and high sediment loads from rivers can cause the accumulation of nutrient-rich, oxygen-poor, fine sediments in New Zealand estuaries [15, 67, 68]. Dense beds of opportunistic seaweed (macroalgae) can flourish under high nutrient input conditions, and increasing density of macroalgae enhances fine-sediment trapping [67]. These sediments provide an additional source of the nutrients that stimulate algal growth. In turn, high algal biomass displaces and hinders the recovery of other biological communities after nutrient loads from rivers are reduced [69]. The New River Estuary in Southland provides an example of an estuary where recovery would likely be slow due to the buildup of nutrient-enriched sediments over recent decades [15, 23]. Figure 2 shows an example of sediment and macroalgal accumulation within this estuary. Other estuaries in New Zealand that are subject to elevated sediment and nutrient inputs are on similar trajectories of degradation [64, 70]. In estuaries where fine sediment deposition rate is slow and the sediments are coarse, sediments do not hold large amounts of nutrients that can be remineralised, and recovery from excessive nutrient availability can be rapid. For example, the Avon-Heathcote estuary in Christchurch showed a relatively rapid recovery following diversion of Christchurch's major wastewater outfall (which discharged to the estuary) to an offshore site (Barr, Zeldis [65] and Zeldis, Depree [32]. In cases where the denitrifying environment in the sediments becomes overwhelmed by organic matter deposition and anoxia, negative feedback arises, furthering eutrophication [7, 32, 33] and reducing the capacity of these systems to assimilate further nitrogen loading without increasing degradation.

Figure 2: Photographs illustrating the change in sediment trapping and retention following the establishment of persistent beds of macroalgae (*Gracilaria chilensis*). These photographs were taken at Bushy Point, New River Estuary (Southland) 2007, 2012 and 2016 [67].

The interactions described above between nutrient availability, climate and trophic state may affect state and trend analyses of nutrients in coastal waters [36, 37]. For example, we would expect higher algal growth and lower nutrient concentrations in coastal waters when other conditions required for algal growth (such as light and temperature) are met. Therefore, sea-surface temperature anomalies (such as marine heat waves), or cycles (such as El Niño-Southern Oscillation (ENSO)) may affect state and trend analyses of nutrient concentrations in coastal waters [39].

Because nutrient loads from land are a key factor in determining nutrient concentrations in estuaries [26], changes in estuary morphology or river flow rates that alter freshwater content at a given measurement site are likely to alter measured nutrient concentrations. This could be addressed by accounting for salinity changes in trend analysis of estuarine nutrient concentrations.

B6. What tikanga Māori and mātauranga Māori could inform bands or allocation options? How? For example, by contributing to defining minimally disturbed conditions, or unacceptable degradation.

Mana whenua have long advocated for more holistic approaches to inform estuarine and coastal health (e.g., ki uta ki tai) and this drive has seen for instance efforts towards understanding ecological condition and the need for better protection of significant areas such as Ōreti (New River Estuary; e.g., [86]. A key example of how mana whenua have shaped the approaches to improving management for land, freshwater and estuaries are evident within Murihiku (aka Southland). For instance, having estuaries included within Freshwater Management Units (FMUs) have been strongly advocated for by iwi, including Ngāi Tahu ki Murihiku [87, 88]. The involvement of mana whenua within decision-making, including the provisions of management policy statements (e.g., NPSFM), and the values set out within Iwi Environmental Management Plans is essential. It is therefore advised that an approach towards developing bands and allocation is done more appropriately. The requirement for engagement and collaboration with mana whenua is shared in the following example, where a multi-disciplinary study (mātauranga and Western science), co-lead with kairangahau Māori (who have expertise within the economic, freshwater ecology, marine ecology and mātauranga Māori) within the National Science Challenge, have suggested expanding beyond upstream, leading with three steps: (1) understanding iwi aspirations for place, (2) estuarine ecologists being able to identify freshwater contaminant thresholds or load limits for achieving or moving towards those aspirations, and (3) catchment modellers determining the necessary mitigations or changes in land use to achieve the necessary loads [89].

Part C—Management levers and context

C1. What is the relationship between the state of the environment and stresses on that state? Can this relationship be quantified?

Nutrients, conveyed by rivers, are the main cause of increased nutrient availability and eutrophication in New Zealand estuaries [4, 26] and estuaries globally [27] . Methods to quantify relationships between management interventions to control nutrient loads to freshwaters upstream, nutrient loads to estuaries, and trophic state in estuaries are still developing.

The causal relationships between nutrient availability in estuaries and impacts to trophic state are well understood [8, 33, 71]. The New Zealand Estuarine Trophic Index gives an example of tools to quantify these relationships in a management context [4, 13], however quantitative links within the ETI tools between nutrient availability and some eutrophication indicators are unevenly distributed regionally across New Zealand (more information for southern than northern New Zealand) and also with respect to estuary type (e.g., lagoons are better understood than riverine estuaries). See https://shiny.niwa.co.nz/Estuaries-Screening-Tool-1/ and Zeldis and Plew [12]. Standardisation of coastal eutrophication indicators measured more evenly across our estuaries would provide data to improve quantification of these relationships.

As described above, biogeochemical processes occurring within estuaries add noise to relationships between nutrient concentrations measured in estuaries (stressors) and eutrophication indicators. To address this issue, tools (such as the ETI Tools, or ASSETS [57]) quantify relationships between stressor (nutrient) <u>loads</u> adjusted for dilution and flushing, and eutrophication indicators. However, a limitation of our load models (such as CLUES [72]) is that they provide 'steady-state' nutrient load estimates. This may be important because:

1. Loads to estuaries differ naturally between years, e.g., between years with different river flow rates [37].
2. Timing of contaminant (e.g., sediment and nutrient) loading to estuaries may alter effects on estuary attributes.

Development of time-varying models of nutrient loads to estuaries, and regular monitoring of nutrient concentrations and flow at terminal river reaches (ideally at sites representative of freshwater management units [41]) could improve quantification of relationships between management interventions upstream, and trophic state in estuaries.

C2. Are there interventions/mechanisms being used to affect this attribute? What evidence is there to show that they are/are not being implemented and being effective?

C2-(i). Local government driven

Key mechanisms that affect this attribute are controls on nutrient loading initiated by local government to give effect to the (central government initiated) National Policy Statement for Freshwater Management (NPSFM), and resource consents on point sources of nutrients (e.g., wastewater). Diversion of the Christchurch wastewater treatment plant outflow to Ihutai (Avon-Heathcote) estuary provides perhaps the best example nationally of the potential to improve seawater nutrient concentrations (and associated eutrophication) by removing point source discharges. This diversion represented a reduction of around 90% of the total nitrogen load to this estuary. Seawater and sediment chemistry, and indices of primary production in the estuary were measured before and after the diversion. The diversion resulted in improvement in indicators of trophic state across the estuary [32, 65]. To date, however, effective diffuse source nutrient management has been difficult to achieve in New Zealand, although it is generally recognised as the major source of estuarine degradation attributable to nutrients. Phosphorus levels have declined in several time series, attributable to point source improvements (e.g., Waikato region rivers [73], and Canterbury region rivers [39]), although nitrogen levels often continue to increase. However, because estuaries are generally nitrogen limited, the benefit of these improvements has been muted.

C2-(ii). Central government driven

As noted above, there is some evidence to suggest that controls on nutrient loading initiated by councils to give effect to the NPSFM may be causing improvement in nutrient availability in estuaries; long-term trends indicate strongly that phosphorus and dissolved inorganic nitrogen (especially nitrate) concentrations in estuaries are decreasing in some regions nationally [38-40], but not in others [8]. However, caution is needed in making this interpretation, as New Zealand's coastal climate is changing [74], and may be affecting processes of nutrient uptake in estuaries within the timescales examined in trend analyses [11, 33, 36, 37]. For example, in Avon Heathcote estuary there is evidence that marine heatwave-driven increased winter temperatures have led to rapid algal growth, accompanied by decreasing nutrient concentrations. In such conditions, trophic state may degrade, even under unchanging nutrient loads [37]. Council sampling is carried out in surface waters during daylight hours, where we would expect nutrient uptake to be high. We would suggest that improved understand of nutrient <u>loads</u>, as well as monitoring of other indicators of estuary trophic state (e.g., bioindicators of nutrient availably [12, 14, 15, 65]) are the best option to infer changes in nutrient availability in estuaries.

C2-(iii). Iwi/hapū driven

There are many examples of iwi and hapū driven initiatives to improve estuarine health overall. The specifics to nutrient status does not necessarily align with holistic approaches. It is difficult to measure the improvements given the legacy issues in estuaries, and the more recent collaborations that acknowledge a systems approach are required [89].

C2-(iv). NGO, community driven

I have no knowledge of initiatives to improve nutrient availability or estuary trophic state in coastal waters being carried out by representatives of NGOs.

C2-(v). Internationally driven

I have no knowledge of obligations to internationally initiatives that would require improvement of nutrient availability or estuary trophic state in coastal waters.

<u>Part D—Impact analysis</u>

D1. What would be the environmental/human health impacts of not managing this attribute?

Changes in the attribute state affect ecological integrity as described in A1 above. Not managing eutrophication processes in coastal waters will likely lead to degradation of inshore fisheries, including shellfish and other mahinga kai species, especially in already degraded environments [75, 76]. Excess nutrient availability and primary production can lead to degradation of seagrass beds, severely impacting habitat quality for the juveniles of commercial fish species [76]. Reduced oxygen (a result of advanced eutrophication) will continue to contribute to species loss and displacement, and stress ecological function of eutrophic waters [77]. Frequency of hypoxia-driven fish kills and toxic algal blooms is likely to increase [78]. Excess macroalgal growth will form a public nuisance when biomass displaced from beds washes ashore and decomposes, causing unpleasant odours.

Not managing this attribute in the near term would also have implications for any future remediation efforts, including increased time for recovery, increased difficulty of remediation and requirements for additional types of remediation [79].

D2. Where and on who would the economic impacts likely be felt? (e.g., Horticulture in Hawke's Bay, Electricity generation, Housing availability and supply in Auckland)

The impacts are likely to be felt in inshore fisheries (including mahinga kai, site, species and habitat health) and aquaculture operations in areas where nutrient loads from land are high. These impacts may be caused by damaging or fatal hypoxia in waters and sediments [80], with demersal and benthic species (those that live and feed on or near the bottom of seas) likely to be disproportionally affected [81]. Impacts to wild fisheries can be driven by decreased habitat for juveniles e.g., seagrass [76, 82] or via decreased food availability [77]. Impacts to fisheries may extend well beyond the range of impacted juvenile habitat, if these habitats supply juveniles to adult populations covering a greater spatial range [82].

Impacts to amenity of coastal waters (such as caused by rotting algal biomass) are likely in estuaries and coasts near where nutrient loads from land are high.

D3. How will this attribute be affected by climate change? What will that require in terms of management response to mitigate this?

Severity of harmful algal blooms may increase even under current nutrient loading rates if other conditions for algal growth improve [36, 83]. Additionally, even under current loading levels of nitrogen to coastal waters from land, increasing seawater temperatures are also expected to exacerbate coastal de-oxygenation both by reducing the solubility of oxygen in seawater, increasing ecosystem metabolism rates, and increasing the tendency of the ocean to stratify [84, 85].

An appropriate management response would be to manage loads of nitrogen from land to levels that are unlikely to worsen primary drivers of eutrophication (algal growth) and secondary impacts of eutrophication (including hypoxia in coastal waters), with sufficient tolerance to negate climate-change-driven effects.

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9.16 Faecal indicator bacteria in water

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State of knowledge of the "Faecal Indicator Bacteria in estuary/coastal water" attribute: Medium / unresolved – some studies/data but conclusions do not agree.

Part A—Attribute and method

A1. How does the attribute relate to ecological integrity or human health?

Faecal Indicator Bacteria (FIB) indicate (fairly recent) faecal contamination of water by faeces of warm-blooded animals. The preferred FIB for freshwaters in NZ is *Escherichia coli* (*E. coli*) and for marine waters is enterococci. This is primarily because *E. coli* are more persistent in sunlit freshwaters, with enterococci being more persistent in saline waters – because of salt toxicity to sun-damaged *E.* coli [1]. The 'best' FIB in intermediate salinity ('brackish') estuarine waters is either (or both) depending, not so much on salinity per se, as on flushing time [2].

The main influence on NZ's coastal microbial water quality is river inputs from adjacent land [3]. McBride et al. [2] showed that *E. coli* is the more appropriate FIB In rapidly flushed estuaries, particularly near inflowing rivers. Recognising the complexity of their advice, these authors recommend *both* indicators be monitored in estuaries.

NZ health-based guidelines for enterococci in marine waters are established from epidemiological studies encompassing the overall gastrointestinal and respiratory risk from a variety of pathogens. In contrast, the guideline for *E. coli* in freshwater relies on numerical modelling to estimate the risk of infection from just one pathogen – *Campylobacter.* Consequently, the enterococci guideline actually has a sounder basis for protection of human health. NZ does not have threshold levels of *E. coli* in coastal waters for risk protection, so we are stuck, for now, with enterococci for routine coastal surveillance monitoring – despite that the change in indicator from fresh to saline water complicates potential modelling of coastal faecal pollution from land sources.

FIB per se are not relevant to ecological integrity. However, faecal pollution as indicated by FIB may well have ecological impacts due to organics, oxygen demand and other contaminants accompanying FIB in faecal matter. For example faecal pollution has a detrimental effect on aquatic microbial community structure and correlates with reduced microbial diversity [4]. This would affect microbial processing within ecosystems and hence ecosystem functioning.

Conversely ecological integrity can compromise FIB as reliable indicators for health-based water quality monitoring. FIB can persist or even grow within estuarine plankton and on seaweeds [5] [6] whilst blooms of cyanobacteria may inactivate FIBs [7].

FIB are rather tenuously related to human health – because the actual hazard to human health is (infection by) a number of enteric pathogens that *may* be present (episodically) in faecally-contaminated water. FIB themselves do not normally cause disease [2] although there are some types such as *E. coli* 0157 that are pathogenic.

Correlation of FIB with risk to human health is at best only moderate for several reasons including that different animal and bird sources have very different risk 'profiles', ranging from low-risk for bird contamination to high risk (similar to that for sewage containing human wastes) for cattle sources [8].

Significant correlations between FIB and health risk are often detected following wet weather events and at locations impacted by recent faecal contamination [9]. Poorer relationships exist with multiple sources and some pathogens (e.g., viruses due to differential fate and behaviour) because health risks from mixed sources are not necessarily driven by the source(s) with the greatest load of FIB [10]. Certain strains of FIB can persist or even grow in the environment, further complicating risk relationships [11].

A2. What is the evidence of impact on (a) ecological integrity or (b) human health? What is the spatial extent and magnitude of degradation?

Several overseas studies of bather health after swimming at coastal beaches have shown a weak to moderate correlation of human health effects (gastroenteritis, respiratory problems...) with faecal contamination of the bathing water – as reviewed by McBride et al. [2].

The correlations in such studies are at best moderate because of the variety of pathogens that may be (episodically) present in faecally-contaminated water and the different type and severity of health effects. Additionally, FIB concentrations in marine waters can vary widely over time due to various inactivation or removal processes and coastal hydrodynamics. Resuspension of beach sands by wave action can remobilise stores of FIB [12]. Nevertheless, while FIB may not reliably predict the presence of specific faecal pathogens, they indicate an increased potential for pathogens to be present.

The extent to which FIB indicate the presence of waterborne pathogens and associated potential health risks in New Zealand is currently being assessed with a revision of the MfE/MoH [13] freshwater recreational guidelines [14]. The outcome of this assessment will have implications for the suitability of FIBs for assessing public health risk and for comprehending contamination in freshwaters and implications for downstream estuarine waters.

A3. What has been the pace and trajectory of change in this attribute, and what do we expect in the future 10 - 30 years under the status quo? Are impacts reversible or irreversible (within a generation)?

The FIB, *E. coli* and enterococci, and similar or related FIB (e.g., faecal coliforms) have been used for many decades (nearly a century for faecal coliforms).

Despite that these attributes (FIB) do not *directly* measure health-impacting pathogens, they are hard to improve upon short of measuring actual pathogens. That is very challenging and usually too expensive for surveillance monitoring because particular pathogens (which must be individually tested for) are usually absent (only episodically present) and typically require technically sophisticated methodology for detection.

The problem of faecal contamination of coastal waters remains a fairly major concern in NZ. Fortunately, faecal contamination of coastal waters is self-correcting once sources are cut off (e.g., inflowing contaminated river floodwaters abate) because of fairly rapid natural disinfection (primarily by sunlight) [1] combined with sorption and sedimentation and hydrodynamic dispersion. These natural processes of attenuation of faecal contamination should be recognised as a major ecosystem service.

In the future we may expect to see increasing measurement of

- Certain actual pathogens (e.g., *Campylobacter* in NZ where campylobacteriosis is a major reportable disease and is endemic in our dairy herds)
- Microbial source tracking (MST) by genetic markers to identify animal sources with different risk factors (e.g., low risk from avian sources versus high risk from bovine sources)
- Phenotypic differentiation between enteric (fresh and aged faecal sources) and non-enteric sources of FIB
- Proxy instrumental monitoring (e.g., turbidity and visual clarity often correlate roughly-but-usefully with – co-mobilised – FIB),
- On-site automatic portable laboratory monitors (e.g., Coliminder) for high-frequency analysis of biochemical proxies of FIB in waters, and
- Modelling of FIB concentrations based on high-frequency monitoring, particularly in inflowing rivers combined with hydrodynamic modelling and satellite (optical) remote-sensing of covarying water tracers.

A4-(i) What monitoring is currently done and how is it reported? (e.g., is there a standard, and how consistently is it used, who is monitoring for what purpose)? Is there a consensus on the most appropriate measurement method?

FIB are routinely measured in NZ as part of two main categories of monitoring:

- SoE water quality monitoring (usually monthly) as one of a broad suite of variables, and
- Bathing beach surveillance (usually weekly) over the summer bathing season. (This monitoring also contributes to beach grading.)

Sampling and laboratory methods for FIB are (have to be) very well standardized in order to achieve reliable (comparable) results. This does have the advantage, however, in permitting data aggregation across waters and monitoring agencies in NZ. For example, the LAWA website 'hosts' FIB data on NZ waters obtained mainly by regional councils in both types of FIB monitoring.

Current monitoring and reporting fail to fully meet public health objectives for several reasons including retrospective microbial risk information (laboratory tests for FIB typically take at least 24 hours to culture the organism), information on risk is spatially and temporally limited, and reporting of human health risk is limited in scope – primarily focusing on FIB while risks presented by cyanobacteria and hazards posed by poor water clarity are overlooked [15]. There are also no guidelines or standards specifically for the microbial quality of estuarine waters.

A4-(ii) Are there any implementation issues such as accessing privately owned land to collect repeat samples for regulatory informing purposes?

There are no substantial practical or logistical barriers to monitoring FIB in coastal waters – *except* that a boat is usually needed for access for SoE monitoring (as is common to coastal water quality more generally). Once on-site, sampling is quick and easy (although care is needed to prevent contamination) and can be cost-effectively combined with sampling for a variety of other variables and attributes (as is routinely done in SoE monitoring in NZ (NEMS2019 – Part 4 Coastal waters) [16]. Monitoring of bathing beach water quality is usually by wading from the shore, so a boat is not normally needed.

An insulated bin ("chilly bin") is mandatory with FIB sampling to prevent inactivation by sunlight during (prompt – typically within 24 hrs is often specified) transfer to the laboratory. Chilling water samples to slow biochemical reactions is also advised [16].

A4-(iii) What are the costs associated with monitoring the attribute? This includes up-front costs to set up for monitoring (e.g., purchase of equipment) and on-going operational costs (e.g., analysis of samples).

Since FIB sampling of coastal waters is typically combined with other attributes in routine SoE monitoring, the costs of access (notably travel time and boat deployment) are distributed. The only costs of sampling specific to FIB is for sterile sample containers (e.g., 100 mL vials).

Bathing beach sampling is routinely done by wading from the shore, ideally using a pole sampler.

Laboratory charges are currently about NZ$40 per sample for both membrane filtration and multiple-well methods (Colilert, Enterolert) [17].

A5. Are there examples of this being monitored by Iwi/Māori? If so, by who and how?

Faecal contamination of waters is a particular concern for iwi as regards swimming exposure and contamination of moana kai. Iwi groups are currently using the Petrifilm® method in the NIWA-designed SHMAK kit to measure *E. coli* in NZ river waters. Methods to enable community measurement of FIB in coastal waters have been developed for Estuary-SHMAK by Rebecca Stott (NIWA-Hamilton) [19].

Examples of hapū and iwi monitoring include use of the SHMAK faecal indicator tools, the Murihiku Cultural Water Classification System by Ngāi Tahu ki Murihiku [27,28], and the assessment of river health input into estuaries for the State of the Takiwā [29].

A6. Are there known correlations or relationships between this attribute and other attribute(s), and what are the nature of these relationships?

Yes – very often useful (local) correlations of FIB with other attributes in coastal water can be established, notably with visual clarity and salinity [2].

The dominant source of faecal contamination of most coastal waters in NZ is river inflow. For example, Dudley et al. [3] showed that coastal water quality varied strongly inversely with salinity which is reduced by river inflow. FIB in rivers sometimes exhibit rough but useful correlations with flow and visual clarity, and these correlations are expected to translate into coastal receiving waters. In pastoral catchments, the correlation of *E. coli* and visual clarity can be relatively strong due to co-mobilisation of FIB and fine sediment by livestock activities. Correlations of FIB with visual clarity are usually stronger than flow correlations in such catchments [18].

Fairly strong correlations between salinity, visual clarity and FIB occur within faecally-contaminated coastal plumes produced by river floods [20].

Part B—Current state and allocation options

B1. What is the current state of the attribute?

We have a broad understanding of faecal contamination of NZ coastal waters at the national scale from monitoring by regional councils for SoE and marine recreational bathing sites.

Faecal contamination of NZ coastal waters is extremely variable over time – mainly with varying river inputs. This reflects rivers being the main source of the FIB in coastal waters and that rivers have extremely variable fluxes (cfu/s) to the coast. As a consequence, the state of faecal contamination in coastal waters varies very widely over time. Most estuaries and almost all embayments are typically clean and clear of faecal contamination (except when subject to wind-wave disturbance) but may become heavily contaminated for a few hours or days by flood plumes from rivers that are contaminated by livestock pasture or urban drainage [20]. Additionally, resuspension of FIB populations in beach sands and decaying vegetation may contribute to inputs into coastal waters [21] [22].

Although hydrodynamics plays a major role in determining coastal FIB levels, the faecal inputs of rivers appears to be the single largest influence. NZ rivers vary widely in characteristic faecal contamination. For example, Davies-Colley et al. [18] reported median *E. coli* concentrations ranging from (about 1 cfu/100 mL in the near-pristine upper Motueka River at the Gorge, to 310 cfu/100 mL in the predominantly pastoral Mataura River at Seaward Downs. Rivers, and therefore downstream coastal recieving waters, also vary greatly in FIB concentration with state-of-flow.

As a consequence, only those estuaries and embayments with adjacent land catchments in near-pristine condition (lacking pastoral agriculture or urban development) can be expected to have swimmable FIB levels after heavy rain.

B2. Are there known natural reference states described for New Zealand that could inform management or allocation options?

A relatively few NZ estuaries with near-pristine catchments might be useful as references as regards faecal contamination status, such as Whanganui Inlet and Okarito Lagoon (Westland).

Given that coastal water faecal contamination is most strongly affected by river inputs, we can infer general coastal contamination levels based on the condition of adjacent land. Faecal contamination levels are expected to be relatively high (particularly after rain events) where FIB are mobilised from catchments in adjacent land by certain activities, particularly livestock agriculture and urban runoff.

B3. Are there any existing numeric or narrative bands described for this attribute? Are there any levels used in other jurisdictions that could inform bands? (e.g., US EPA, Biodiversity Convention, ANZECC, Regional Council set limit)

New Zealand has existing numeric guidelines for coastal water contact recreation (also for recreational shellfish harvesting) [13]. These guidelines have been more recently reviewed and endorsed by McBride et al. [2]. Note, however, that these guidelines should not be used where wastewater discharges dominate, because the relationship between indicator and pathogens may be substantially changed during wastewater treatment particularly with technical disinfection.

NZ now has new standards (“target attribute states”) for *freshwaters* in the National Policy Statement for Freshwater Management- Tables 9 and 22 [23]. These standards may be expected to contribute strongly to achieving swimmable conditions in downstream *coastal* receiving waters that are strongly degraded (albeit episodically) by contaminated river flood plumes.

B4. Are there any known thresholds or tipping points that relate to specific effects on ecological integrity or human health?

No, the concept of tipping points does not really apply to human health effects of exposure to faecally-contaminated water. So far as we know, human health risk increases monotonically, although not necessarily *linearly*, with FIB concentration – without inflexion points, let alone singularities associated with change to a new stable state. Target attribute states and guidelines (“thresholds”) are therefore based on somewhat arbitrary levels of estimated health risk.

B5. Are there lag times and legacy effects? What are the nature of these and how do they impact state and trend assessment? Furthermore, are there any naturally occurring processes, including long-term cycles, that may influence the state and trend assessments?

The main legacy effect of faecal contamination of both rivers and coastal receiving waters is uptake by the bed sediments in which natural dieoff of FIB (and pathogens) is greatly slowed compared to overlying water due to screening from sunlight. For example, Drummond et al. [24] modelled uptake of *E. coli* by river beds (the hyporheic zone) during declining flows and subsequent mobilisation of these faecal stores due to accelerating water currents on flood fronts. Similarly, faecal microbes stored in coastal bed sediments or beach wrack during quiescent conditions may be mobilised by currents or waves [12].

B6. What tikanga Māori and mātauranga Māori could inform bands or allocation options? How? For example, by contributing to defining minimally disturbed conditions, or unacceptable degradation.

Contamination of ecosystems is, understandably, of great concern to tangata whenua.

Practices such as rāhui on environments, including the activity of shellfish harvesting, demonstrate the importance for preventing risks, however it is just one of a suite of tools.

There are examples of mātauranga a-iwi, and multidiscplinary approaches given above (e.g., [27]) that provide insight into more culturally appropriate approaches towards answering this query. Engagement with the mana whenua, with Māori researchers, and those who engage with appropriate methodology is fundamental.

<u>Part C—Management levers and context</u>

C1. What is the relationship between the state of the environment and stresses on that state? Can this relationship be quantified?

The main pressure on faecal contamination status of coastal waters is mobilisation of FIB from adjacent land with conveyance to the coast via rivers, notably during high flow events. However, the relationship between the *pressure* (FIB mobilisation on land) and coastal faecal contamination status is highly complex because of:

- Displacement in space of land sources from coastal receiving waters
- Variation in time, particularly with river flow conditions and coastal plume hydrodynamics
- Dieoff of FIB (and pathogens) in waters, often referred to as 'natural disinfection', depending most strongly on sunlight exposure
- Uptake and storage of FIB (and pathogens) in the hyporheic zone of rivers and coastal bed sediments (that are subject to hydraulic disturbance), and
- Poor wastewater infrastructure

C2. Are there interventions/mechanisms being used to affect this attribute? What evidence is there to show that they are/are not being implemented and being effective?

Because most faecal contamination of coastal waters comes from adjacent land the interventions must focus on land. Interventions are underway in NZ, focussed mainly on general water quality (including fine sediment and nutrients as well as FIB) of *rivers* rather than coastal waters.

C2-(i). Local government driven

Regional councils are the agencies most actively intervening to improve water quality in NZ, including faecal contamination status – by promoting stream fencing (to reduce direct livestock pollution – cattle have a known attraction to waters) and riparian setbacks (to trap FIB in land runoff – reducing indirect livestock pollution). Such riparian management has been shown to improve stream water quality with reductions in *E. coli* observed in relatively flat pastoral land [26]. However, effectively managing FIB losses in hill country sheep and beef farming poses challenges especially on steep slopes. Improving sewage infrastructure should also reduce faecal contamination.

Regional councils are also keen to inform the recreating public of faecal contamination status – based currently, mainly on so-called beach-grading. In future, modelling of FIB status informed by high-frequency monitoring of flow and FIB proxies such as salinity and turbidity [2] could, in principle, be used to warn swimmers in near real-time of the likelihood of contamination. NIWA currently has 'Smart Idea' funding of a project (WaiSpy MBIE contract: C01X2204) that is attempting to develop a system for informing swimmers of 'swimmability' of rivers, and potentially, also downstream coastal receiving waters, based on monitoring of contributing rivers.

C2-(ii). Central government driven

C2-(iii). Iwi/hapū driven

There are examples of mātauranga a-iwi, and multidiscplinary approaches given above (e.g,. [27]) that demonstrate culturally appropriate approaches. Tikanga Māori are well known to prevent risks to contamination, however whānau and hapū have long advocated for more holistic approaches that prevent contamination, and improved the health of catchments, ki uta ki tai [28]. Implementation of tikanga Māori is difficult given the legislative barriers to mātauranga preventing the implementaiton of hapū and iwi decision-making within waterway management [30,31].

C2-(iv). NGO, community driven

Community-driven initiatives such as 'Mountains to Sea' mobilize community interests in stream fencing, restoration planting and water monitoring. These efforts should reduce the burden of faecal pollution of rivers and downstream coastal waters. We are not aware of improved coastal water quality in NZ being explicitly linked to land management, however such connections have been made overseas. Improved faecal contamination status of coastal waters is difficult to attribute to land management because of the complexity of land-coastal connections (Refer C1).

C2-(v). Internationally driven

Part D—Impact analysis

D1. What would be the environmental/human health impacts of not managing this attribute?

Not managing faecal contamination of coastal waters is likely to lead to increased disease burden on recreational swimmers [2] and downgraded perception of NZ as 'clean and green' among tourist visitors. It would also have severe implications for shell fisheries and exports as well as cultural impacts for kai moana (e.g., rāhui on shellfish harvesting, and the healthy reciprocal relationship between Tangata and their Whenua/Moana).

Managing faecal contamination of coastal waters requires, mainly, management of faecal contamination of inflowing rivers – in turn by reducing faecal mobilisation from land. So, land activities that mobilise FIB (and potentially pathogens, episodically), primarily livestock agriculture and urban land use, need to be isolated so far as possible from waters. Important controls on FIB mobilisation in waters are:

- In livestock pasture: fencing to exclude livestock and riparian set-backs to entrap FIB in runoff water,
- In semi-rural areas: improved operation and maintenance of on-site wastewater systems, and

- In urban areas: maintenance of foul sewers (reducing wet weather surcharging and overflows) plus street-sweeping to reduce stormwater contamination by domestic and feral animals.

To manage faecal contamination status of coastal waters requires its measurement – which, currently, is deficient in NZ because of necessarily discrete sampling for FIB and sparse distribution of sites. What is needed for improved management is *modelling* to fill in the measurement gaps in time and space – ideally informed by high-frequency instrumental monitoring of proxy variables like turbidity (in contributing rivers as well as coastal receiving waters) or new modelling approaches using satellite remote sensing – integrated within an artificial intelligence framework (refer 'Coastwatch' currently proposed by NIWA to the MBIE Endeavour fund.)

D2. Where and on who would the economic impacts likely be felt? (e.g., Horticulture in Hawke's Bay, Electricity generation, Housing availability and supply in Auckland)

The main economic impact would be on NZ's tourist industry – which trades strongly on NZ's image as a 'clean green', environmentally responsible country.

The general public of NZ would be impacted in a difficult-to-quantify way if our coastal waters were increasingly perceived by NZ citizens as contaminated, resulting in reduced recreational opportunity and sporting activities for fear of illness.

A decline in the microbial quality of coastal waters would also be expected to have economic impacts on bivalve shell fisheries, especially oyster farming.

D3. How will this attribute be affected by climate change? What will that require in terms of management response to mitigate this?

Increased variability of river flows due to global warming may be expected to increase variability of water quality of coastal water, including faecal contamination status. More frequent large floods can be expected to cause more over-land runoff, resulting in more faecal contamination being conveyed episodically to coastal waters via rivers with associated increased risk of waterborne faecal-related diseases [2].

Higher summer temperatures may be expected to drive people to swim and recreate more often in coastal waters despite frequent contamination, further increasing the disease burden from swimming exposure with children being typically identified as being at higher risk than adults.

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9.17 Faecal indicator bacteria in shellfish

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Preamble: Bivalve molluscan shellfish (BMS) including oysters and mussels are filter feeders and are known to concentrate pathogenic microorganisms from the surrounding water. Grazing shellfish like paua, kina and pupu (catseyes) generally pose a lower human health risk compared to filter feeding BMS. To minimise the risk of human health disease from consumption of commercially grown or recreationally harvested shellfish, shellfish safety continues to revolve around two categories a) the quality of waters in which shellfish grow and, b) the flesh conditions of harvested and processed shellfish. Both these categories use levels of faecal indicator bacteria to minimise the risk of human health disease from consumption of shellfish. Discarding the gut (hua) from shellfish before cooking and eating them further reduces the risk[1]. Criteria for commercial shellfish are driven by multiple market standards with exported shellfish needing to comply with a standard based on *E. coli* in shellfish flesh (e.g., in the EU), and a standard based on faecal coliforms used to classify growing areas (e.g in the USA) [1]. For recreational harvesting of shellfish, guidelines refer only to the use of faecal coliforms to determine the quality of waters and assess the risk of faecal pollution of shellfish harvesting areas [2].

Information regarding the attribute of FIB in shellfish is considered in a wider sense of FIBs in the environment because the presence of faecal microbial contaminants in shellfish reflects the microbial quality of shellfish growing waters.

State of knowledge of attribute: Faecal indicator bacteria in shellfish (estuary/coastal): Medium / unresolved – some studies/data but conclusions do not agree.

While FIB provide valuable information about the faecal contamination status of shellfish harvesting waters and flesh, evidence relating FIB in shellfish to human health is moderate at best as their presence does not always reliably predict the presence of pathogens, nor do they relate to non-faecal derived pathogens or marine biotoxins which can present a significant risk to shellfish consumers. In addition, the relationship between water and shellfish flesh contamination is often poor, especially when the samples are taken contemporaneously, so a guideline based on microbes in water does not always provide assurance of shellfish safety in regards to flesh. Monitoring of

[1] https://www.mpi.govt.nz/dmsdocument/1058-Food-safety-for-seafood-gatherers

shellfish safety for recreational or cultural consumption is not routinely carried out and monitoring of growing water quality is spatially and temporally limited leading to a lack of national-scale data and reference sites for comparison.

Part A—Attribute and method

A1. How does the attribute relate to ecological integrity or human health?

There is medium to good evidence from New Zealand and international studies that Faecal Indicator Bacteria (FIB) in shellfish are a weak indicator of the risk to human health from consuming raw or lightly cooked shellfish that have been exposed to faecal contaminated water. FIBs are generally not used to assess ecological integrity, but they can be used in combination with other indicators to assess overall water quality [3]. Faecal matter, particularly from humans, but also from warm-blooded animals such as birds and domestic animals (cows and sheep), may contain pathogens which are harmful to human health. These faecal microbial contaminants can enter estuarine and coastal environments through agricultural runoff, discharge of treated effluent from wastewater treatment plants into freshwater or marine environments, direct deposition into water (by birds), accidental sewerage overflows, and/or discharge from boats [4, 5], [6-8]. Shellfish, such as Bivalve Molluscan Shellfish (with two shells), can accumulate indicator bacteria and pathogens through their filter feeding activities. When consumed raw or lightly cooked, the contaminated BMS can make people ill.

Given the impracticality of routinely monitoring pathogens in shellfish, due to technical difficulties and costs, the use of faecal indicator bacteria FIB as a proxy for risk is a traditional approach. But it has limitations. There is only a moderate but positive correlation between norovirus, a common shellfish-associated pathogen, and indicator organisms [9]. Individual observations with low levels of FIB in shellfish do not imply low risk, and sanitary surveys, water quality monitoring, and the analysis of historical data should be used to assess risk. Not all health risks from the consumption of shellfish are associated with pathogens; biotoxins are a significant hazard, and FIBs do not provide any insight into these risks.

A2. What is the evidence of impact on (a) ecological integrity or (b) human health? What is the spatial extent and magnitude of degradation?

There is good evidence of widespread but intermittent faecal contamination of New Zealand shellfish growing waters and flesh particularly for norovirus [10, 11] [12] . Despite regulations to mitigate microbial contamination, outbreaks of disease linked to shellfish consumption continue in New Zealand [13] and elsewhere [14].

Areas used for commercial shellfish production and recreational harvesting in New Zealand are often located in shallow estuarine and coastal systems and would be vulnerable to extreme spatial and temporal variability in faecal microbial contaminant concentrations [15] [16]. However, not many estuarine or coastal areas near large river outflow are monitored for microbial contaminants[1] so understanding of the spatial extent of faecal contamination in water and shellfish is limited. The use of FIB as surrogates for human health assumes that FIB consistently correlates with pathogen

[1] https://www.stats.govt.nz/indicators/coastal-and-estuarine-water-quality/

presence. There is strong evidence that many potential pathogens co-occur with high densities of faecal coliform bacteria in shellfish harvesting waters following rainfall events [17]. However, FIB often show poor correlation with viral pathogens such as norovirus during other times [12] or with autochthonous pathogens like *Vibrio* sp. [18].

Shellfish habitats are highly susceptible to runoff or discharge from adjacent catchments and river inputs which transport and disperse faecal microbial contaminants downstream into shellfish growing waters [4]. The nature and extent of this contamination are significantly influenced by land use practices such as urban development and agriculture, which determine the types and quantities of microbial contaminants entering these waters. Additional contaminant sources include direct defecation into the water by birds and the discharge of ballast or sewage from ships [8].

Enteric viruses occur frequently in non-commercial shellfish, especially near sewage outfalls following accidental sewage discharge events [12]. In contrast, sites impacted by diffuse sources such as agricultural runoff are more likely to be contaminated with bacterial pathogens. Consequently, multiple sources of faecal contamination can be present in shellfish areas and the health risks from shellfish consumption vary depending on these sources [19].

The extent to which FIB indicate the presence of waterborne pathogens and associated health risks in New Zealand is currently under review with the revision of MfE/MoH freshwater recreational guidelines [20]. This will have implications for the suitability of using FIBs to evaluate public health risks in downstream estuarine waters.

A3. What has been the pace and trajectory of change in this attribute, and what do we expect in the future 10 - 30 years under the status quo? Are impacts reversible or irreversible (within a generation)?

Faecal bacteria such as faecal coliforms and *E. coli* have been used as surrogates for excreted microbial pathogens for many years to assess the faecal contamination and microbiological quality of BMS and their growing waters [14].

At estuarine and coastal sites in NZ with substantial freshwater inputs, a large proportion of FIB are land-derived consistent with international understanding of susceptibility of coastal zones to land-based activities [16]. But there is progress in improving estuarine and coastal water quality. Between 2006 to 2020, 50% of NZ estuarine and coastal sites showed improving trends in FIB water quality[5] inferring improved conditions for shellfish growing waters and reduced potential contamination of shellfish.

However, challenges remain. While efforts to improve land management practices and reduce microbial losses from agricultural sources are underway (refer Section C2), aging urban infrastructure poses a threat of point source pollution from human sources with a high risk profile [21].

Faecal contamination of estuarine and coastal waters remains a major concern in New Zealand. Urban pollution, coastal development, land use intensification and climatic events considerably influence the faecal microbial quality of shellfish growing waters presenting ongoing challenges to managing microbial contamination and ensuring shellfish safety in New Zealand. Reducing the influx of faecal microbial contaminants into shellfish areas will assist in their recovery. Water quality can also rapidly improve after contamination events aided by factors like sunlight inactivation, sorption, sedimentation and hydrodynamic dispersion [22]. However, tidal currents and wave exposure can resuspend contaminants back into the water [23] prolonging the persistence of microbial contamination in shellfish waters and flesh. Variability in environmental conditions, sources of pollution, and hydrometeorological conditions further complicate efforts to maintain shellfish safety [8].

There is a growing range of methods that can be used to assess human health risks including phenotypic differentiation between enteric (fresh and aged faecal sources) and non-enteric sources of FIB [24], alternative viral indicators and faecal source tracking to identify the likely source and risks from faecal contamination [25] as well as improved pathogen monitoring techniques [26].

A4-(i) What monitoring is currently done and how is it reported? (e.g., is there a standard, and how consistently is it used, who is monitoring for what purpose)? Is there a consensus on the most appropriate measurement method?

FIB are monitored to assess faecal contamination and potential health risks associated with shellfish consumption. However, monitoring differs for commercial and non-commercial shellfish harvesting purposes and among different regulatory agencies monitoring for recreational harvesting. These differences include the species of FIB monitored, whether water or flesh is tested, the sampling and testing method used, the locations and frequency of testing, data analysis and information reporting. Inconsistences in data collection and limited spatial and temporal coverage presents challenges for aggregating and comparing data at a national level. This complicates efforts to obtain a comprehensive understanding of faecal contamination and the safety of shellfish for consumption.

Monitoring requirements for the commercial harvest of BMS, are set out in the Animal Products Regulations and Notice for Bivalve Molluscan Shellfish administered by MPI [27] [1]. BMS harvested from areas classified for human consumption are monitored for faecal coliforms in water (MPN/100mL) and *E. coli* in shellfish flesh (MPN/100g) [28] to ensure shellfish safety for consumption. Monitoring frequency is prescribed for each classification area, but most areas are only sampled 5 times per year. Results are reported to MPI annually to demonstrate compliance with regulations against bacteriological standards for that classified area. Measurement methods are specified by MPI but there is provision for seeking approval to use equivalent methods.

Councils may monitor across several different programmes to provide information on the safety of shellfish for consumption e.g., as part of weekly surveillance monitoring of recreational waters, monthly SoE water quality monitoring or for compliance monitoring for resource consents [29] [30]. Water samples are typically collected but sampling methods are not standardised e.g depth of sampling, tidal state (and bias towards high or low salinity conditions). The microbial quality of shellfish gathering waters is compared to the guidelines for recreational harvesting included in the MfE and MoH Microbiological Water quality Guidelines for Marine and Freshwater Recreational Areas [2]. These guidelines specify thresholds for faecal coliform levels in water over the shellfish gathering season. These criteria align with those for approved growing waters for commercial shellfisheries for which shellfish are expected to have suitable microbiological quality for safe public consumption [1]. Guidelines should be applied alongside a sanitary survey to confirm the absence of point sources of contamination. This precaution is necessary because water meeting faecal coliform criteria may still pose a risk if a contamination source is identified.

There are no specific microbiological guideline criteria for routine flesh testing of recreationally or customarily harvested shellfish which would provide greater confidence in shellfish safety for consumption. Few councils appear to consistently monitor shellfish flesh. Council monitoring for consent compliance may include pathogen testing and FIB especially for wastewater discharges, to better understand the relationship between microbial concentrations in water and flesh and associated health risks.

IANZ-accredited multi-tube MPN methods are recommended for determining faecal coliforms in shellfish gathering water and *E. coli* in flesh [2] [28]. However, other methods, such as those reporting results as CFU, are used by councils. This discrepancy complicates data aggregation across monitoring agencies particularly where recreational and commercial shellfish monitoring overlap.

There is no national monitoring for faecal microbial contaminants in recreationally harvested shellfish unlike for biotoxins[1]. Available FIB information collected as part of SoE monitoring of estuarine waters or other focused monitoring programmes (e.g., see [30], and which may overlap with recreational shellfish locations, is not included on LAWA for estuary health[2] but could be to provide a broader understanding of estuary conditions for shellfish harvesting.

A4-(ii) Are there any implementation issues such as accessing privately owned land to collect repeat samples for regulatory informing purposes?

Monitoring for FIB such as faecal coliforms and *E. coli* is practical and feasible; however, FIB do not differentiate the human health risks associated with various faecal sources. This limitation means that while FIB provide an indication of faecal contamination, they do not specify the origin or potential pathogenicity of the contamination.

Monitoring deeper coastal waters poses logistical challenges, often requiring the use of boats, unlike the more accessible shallower estuarine waters. This can impact the frequency and coverage of monitoring efforts. Monitoring sites are not necessarily representative at a national level due to the omission of suitable monitoring sites that are inaccessible or where access is prohibited, and differences in resourcing and capability across councils. This restricts the understanding of the extent and magnitude of FIB in shellfish and how this attribute relates to human health risks.

Monitoring frequency for shellfish gathering waters may be insufficient, as monthly monitoring for SoE purposes often fails to capture temporal variations in water quality due to hydrometeorological effects like rainfall and tides. In addition, regulatory monitoring might only monitor shellfish gathering waters during the summer bathing period to align with marine water surveillance or during a “shellfish-gathering season”. However, this approach should recognise local practice and a season defined according to local usage and in consultation with the community or even year-round.

Current monitoring and reporting practices fall short of fully meeting public health objectives. Microbial risk is retrospective, spatially and temporally limited, and human health risk is constrained by the limitations of using FIB to detect faecal pathogens in shellfish growing waters or in flesh.

Detection and quantification methods for pathogens like norovirus in shellfish exist [13], but there are not established microbiological standards for norovirus or other pathogens in BMS. Proposed enteric virus concentrations in commercial shellfish growing waters [31] are yet to be included in NZ legislation. There is a reluctance in implementing viral testing due to uncertainty about regulatory response if positive results are found. The absence of clear guidelines for viral pathogens complicates microbial contamination management and assurance of shellfish safety in New Zealand.

[1] https://www.mpi.govt.nz/fishing-aquaculture/recreational-fishing/where-unsafe-to-collect-shellfish/shellfish-biotoxin-alerts/
[2] https://www.lawa.org.nz/explore-data/estuaries#/tb-national.

A4-(iii) What are the costs associated with monitoring the attribute? This includes up-front costs to set up for monitoring (e.g., purchase of equipment) and on-going operational costs (e.g., analysis of samples).

Shallow intertidal waters can be effectively sampled using a pole sampler whilst deeper waters require a boat. Sampling for FIB in water can be integrated with other attributes for SoE assessments or routine monitoring during the summer bathing season and thus included within survey costs. This approach optimises resources and streamlines sampling efforts. Shellfish sampling requires collection by hand from inter-tidal or sub-tidal areas and is more labour intensive. Thus, main survey costs are related to field staff labour costs.

In contrast to pathogen testing which can be more complex and time-consuming, the enumeration of FIB using culture-based assays offers a relatively quick and cost-effective method for assessing water quality [32] and shellfish. Laboratory charges are approximately NZ$40 per sample for both membrane filtration (CFU) and multiple-well (MPN) methods [33]. However, most costs are associated with personnel time spent on data collection, analysis and reporting results.

A5. Are there examples of this being monitored by Iwi/Māori? If so, by who and how?

Shellfish species have significant customary value as taonga for Māori and faecal contamination of water is a particular concern for iwi regarding the potential contamination of kai moana. Methods to enable tangata whenua to measure FIB in estuarine waters have been developed for the "Ngā Waihotanga Iho – The Estuary Monitoring Toolkit [34] but we are not aware of whether it has been implemented by iwi. Similar low-cost methods have been evaluated for detecting FIB in shellfish and could be similarly used by iwi and communities [35].

There are examples of the development of cultural health indicators and indices that have been used at local scale and/or to provide baseline measures in estuarine management [36]. Cultural health assessments have been used to assess the cultural health of the Te Ihutai (Avon-Healthcote Estuary) using the State of the Takiwa system developed from Ngāi Tahu values [37, 38] which included laboratory analysis for *E. coli* in water. Iwi-led observations included catchment land use, visual clarity and silt deposits that all influence suitability for safe shellfish harvesting.

It is important to highlight that, the level of bacteria in shellfish for consumption, is only part of the baseline of health that is considered by Tangata Whenua when considering harvesting kaimoana for consumption. For instance if there site and wider catchment health has a history of contamination and there is degradation within a site, kaimoana won't be collected from that area. The standard of health considers a whole suite of indicators that are already assessed by whānau, hapū and iwi that suggest an area is environmentally unsafe for interacting with [74]. Second, the local government includes bacteria in their suite of monitoring protocols. Today there are numerous areas of significance to tangata whenua that have been restricted from harvesting due to the impacts of contaminants on mahinga kai and overall cultural environmental health.

A6. Are there known correlations or relationships between this attribute and other attribute(s), and what are the nature of these relationships?

Establishing useful local correlations between FIB with other attributes in coastal water can assist in determining periods when harvesting of shellfish is most likely unsafe. The dominant source of faecal contamination in most estuarine and coastal waters in NZ is river flow. Strong correlations between salinity and FIB occur within faecally contaminated coastal plumes produced by river floods [39]. These plumes result in an influx of microbial contaminants due to runoff from adjacent catchment areas resulting in contamination of shellfish with FIB after rainfall events often from ruminant sources [40].

Commercial shellfisheries have established specific criteria for each growing area that trigger closures for harvesting. These criteria rely on threshold values for environmental indicators of contamination such as rainfall in growing area catchments and salinity levels at the shellfish farm. By implementing harvest closures e.g during high rainfall events, the risk of exposure to potentially contaminated shellfish is mitigated. Harvest closures are also instigated in response to notification of wastewater discharge events such as overflows.

Part B—Current state and allocation options

B1. What is the current state of the attribute?

In New Zealand, there is a broad understanding of faecal microbial contamination of waters as these are routinely monitored for FIB either weekly during summer season surveillance of recreational bathing sites, or monthly for SoE monitoring. The preferred FIB for freshwaters in NZ is *Escherichia coli* (*E. coli*), while enterococci are used for marine waters. In estuarine waters, with intermediate salinity ('brackish'), either or both indicators may be used depending more on flushing time than salinity itself [41]. However, guidelines for shellfish harvesting use faecal coliforms.

While some councils assess water quality at coastal and estuarine locations to determine suitability for shellfish gathering, not all councils monitor waters specifically for this purpose. If faecal coliforms are included in their SoE monitoring programmes, the data might only be used to determine the overall health status of the estuary rather than explicitly addressing shellfish contamination. Periodic monitoring of microbial contaminants in recreationally harvested shellfish has been done by councils [42, 43]. As a result, the understanding of shellfish FIB contamination is spatially and temporally limited at the national scale.

Estuarine and coastal waters receive multiple inputs from point and diffuse sources of contamination with the potential to impact on water quality. River flows are the primary influence on downstream water quality with extremely variable fluxes (#/s) of FIB [44]. The extent and nature of this influence varies between seasons and years. Most estuaries and nearby coastal areas are heavily contaminated by flood plumes from rivers polluted by livestock runoff or urban drainage – the effect of which may last hours or days [39]. Through monitoring, some councils identify unacceptable health risks to recreational shellfish gatherers and erect warning signage while investigating problematic sites. However, only shellfish harvesting areas in remote areas with adjacent land catchments free from pastoral agriculture or urban development are likely to have safe-to-eat shellfish after heavy rain.

B2. Are there known natural reference states described for New Zealand that could inform management or allocation options?

Given the strong influence of river inputs on the faecal contamination status of downstream estuarine and coastal water quality, the general contamination levels in these areas can be inferred from land use practices in adjacent catchments. A few NZ estuaries with near-pristine catchments would serve as useful reference sites regarding faecal contamination status. Some councils have identified high-quality and typically “unimpacted” areas in their coastal environmental plans e.g., [45].

Establishing background contamination levels at these pristine sites would be particularly valuable for benchmarking viral markers such as F-specific RNA bacteriophage , crAssphage and pepper mild mottle virus (PMMoV) and their use in indicating the potential risk of viral contamination of human origin in shellfish [25] [46] [47] . Most commercial shellfish growing areas are located in remote regions away from large population centres, and consequently, the risk of contamination from domestic wastewater is perceived to be low [48]. However, testing from non-commercial sites near polluted urban areas has shown a high prevalence of norovirus highlighting the potential risks associated with proximity to human settlements [12].

Shellfish can serve as valuable bioindicators, bioaccumulating contaminants and providing a means to evaluate the effectiveness of management strategies in maintaining or improving water quality suitable for various purposes including shellfish harvesting and recreational activities. The use of shellfish as sentinels also allows for the detection of changes in environmental conditions over time and provide insights into how these changes affect water quality.

B3. Are there any existing numeric or narrative bands described for this attribute? Are there any levels used in other jurisdictions that could inform bands? (e.g., US EPA, Biodiversity Convention, ANZECC, Regional Council set limit)

New Zealand has established numeric guidelines for recreational shellfish harvesting that use faecal coliforms as an indicator of water quality and the suitability for gathering shellfish [2]. These guidelines specify that the median faecal coliform level over the shellfish gathering season should not exceed 14 MPN/100mL, and no more than 10% of samples should exceed 43 MPN/100ml. These numeric thresholds have also been incorporated into regional council coastal environment plans where they manage for shellfish gathering waters [45]. Current recreational harvesting guidelines assess compliance at the end of the season potentially posing health risks during the season. To enable a short-term health risk assessment, McBride et al (2019) recommended using a single sample maxima for surveillance criteria, changing the 90th percentile to a maximum where no sample should exceed 43MPN/100mL[41].

Councils are concerned about overly conservative numeric indicator bacteria values without a technical explanation correlating them with actual human health risks [49]. In response, reformulating the use of faecal coliforms as the indicator for recreational shellfish gathering waters has been suggested based on a risk assessment approach, potentially replacing faecal coliforms with enterococci with a requirement that the median is less than 7 enterococci/100 ml and the maximum does not exceed 22 enterococci per 100 mL [41].

Levels of faecal coliforms in water and *E. coli* in flesh used to classify commercial shellfish growing waters into 6 categories, could be used to determine bands for recreational shellfish harvesting waters.

B4. Are there any known thresholds or tipping points that relate to specific effects on ecological integrity or human health?

It is generally regarded that as the concentration of FIB in water or shellfish increases, the presence of pathogens becomes more likely, thereby increasing the probability of experiencing adverse health effects from the consumption of shellfish. Although the relationship between FIB concentration and health risk is unidirectional, it is not necessarily linear nor indicative of a change to a new state. Hence tipping points do not generally apply for human health exposure.

To manage risk, regulatory agencies monitor recreational shellfish areas against threshold levels for FC in waters. These thresholds are based on median concentrations and a proportion of samples not exceeding a specific criterion. If these criteria are exceeded, there is no formal requirement to conduct a further specific risk assessment in the MfE/MoH (2003) guidelines.

B5. Are there lag times and legacy effects? What are the nature of these and how do they impact state and trend assessment? Furthermore, are there any naturally occurring processes, including long-term cycles, that may influence the state and trend assessments?

A legacy of faecal contamination in estuarine/coastal receiving waters is the uptake of faecal microbial contaminants by sediments and beach sands. These environments provide protection for faecal microbes from various biotic and abiotic stressors allowing them to persist [50]. These sediments act as reservoirs for microbes such as FIB which can be resuspended back into the water column by wave action and tides, significantly impacting microbial water quality and intertidal exposure of shellfish to microbial contaminants in the absence of fresh inputs [51] [52]. Additionally, the resuspension of FIB populations from decaying vegetation can contribute to the contamination of coastal waters and uptake by shellfish [53].

Shellfish bioaccumulate microbial contaminants from the surrounding water, creating a lag between the contamination of the water and the contamination of shellfish flesh. This has severe implications for pathogens such as norovirus which can persist and remain in shellfish for extended periods (weeks to months) and is not effectively removed by depuration practices [13].

B6. What tikanga Māori and mātauranga Māori could inform bands or allocation options? How? For example, by contributing to defining minimally disturbed conditions, or unacceptable degradation.

Estuarine and coastal areas are highly valued by tangata whenua, as their papa kāinga, and including mahinga kai (sites, species, and practices) [54] [55]. Faecal contamination of these areas impacts environmental safety, including the safety to gather kai, thus impacting long term relationships with place, and practices [56]. Several kaupapa Māori frameworks have been developed for estuarine/coastal assessments in NZ [57, 58] and may provide indicators to benchmark perspectives of shellfish safety for harvesting and consumption. Integrating tikanga and mātauranga Māori regarding cultural practices around collecting and processing of shellfish with scientific approaches (e.g attribute states) may provide a more holistic assessment of safety of shellfish for consumption.

The State of the Takiwā is a complementary monitoring framework that integrates mātauranga Māori and science to gather environmental data and takes into account Māori cultural values. The approach has been used to establish baseline conditions for assessing the health of estuaries [38]. The Marine Cultural Health Index[1] has been added to provide a protocol where kaitiaki can assess the health of their mātaitai, taiāpure or area where a temporary closure has been imposed. Recently, a

[1] https://www.mahingakai.org.nz/community-tools/marine-cultural-health-index/

marine cultural health index (MCHI) has been developed as part of the Marine Cultural Health Programme, to monitor the health of the marine environment e.g., for mahinga kai in the Ahuriri/Napier area [59,]

Part C—Management levers and context

C1. What is the relationship between the state of the environment and stresses on that state? Can this relationship be quantified?

Contamination of shellfish with faecal microbes is influenced by the quality of overlying waters in growing areas. The state of faecal contamination of estuarine and coastal waters with FIB is strongly affected by land-derived contaminants delivered by contaminating river flows to downstream locations notably during high flow events. Potential sources of contamination include runoff from agriculture, stormwater, treated wastewater discharges, leaking septic tanks and /or liveaboard vessels and other marine craft and wildfowl. There is clear evidence that agricultural and urban land use pressures negatively impact the microbiological quality of freshwater systems [60] [61] and that estuarine water is generally of poorer quality than at coastal sites [30]. However, the relationship between pressure variables and faecal contamination status of estuarine/coastal water and shellfish is complex and a challenge to understand due to;

- the simultaneous presence of multiple sources of contamination – FIB do not provide information on the source of faecal contamination so distinguishing between multiple sources is difficult,
- spatial displacement of land use sources from estuarine/coastal receiving waters,
- temporal variability as a result of river flow conditions and contaminating plume hydrodynamics in estuarine and coastal waters,
- tidal movements affecting the distribution and resuspension of faecal microbial contaminants in water,
- uptake and storage of FIB (and pathogens) in estuarine/coastal sediments or other habitats (e.g wrack) that may be released back into the water column by hydraulic disturbance,
- inactivation and die-off of FIB (and pathogens) in water – depending strongly on sunlight exposure,
- bioaccumulation within shellfish and differential persistence of pathogens – rate and extent of bioaccumulation can depend on the type of shellfish, filter feeding rates, and local environmental conditions. This makes it difficult to correlate levels of FIB in shellfish directly with those in the surrounding water,
- poorly maintained wastewater infrastructure.

Uncertainty about the source and causes of degradation presents a risk that management interventions will target pressures that have little effect on the impact of safe shellfish for consumption. Interventions should be tailored to the specific sources of contamination identified in shellfish (guided by sanitary surveys) with regular testing of waters and implementing temporary harvesting closures when bacterial (or proxies) levels exceed safety thresholds.

C2. Are there interventions/mechanisms being used to affect this attribute? What evidence is there to show that they are/are not being implemented and being effective?

C2-(i). Local government driven

Since most faecal contamination of coastal waters originates from land-based sources in New Zealand, efforts to improve water quality are primarily focussed on the water quality of rivers rather than coastal water quality. Intensive pastoral agriculture is a significant contributor to faecal microbial degradation of waterways; reducing FIB loss to surface waters also reduces the downstream delivery of zoonotic pathogens (disease causing organisms transmitted between humans and animals). Council promotion of measures such as stream fencing to exclude livestock from water bodies, and establishing riparian buffers to attenuate FIB in land runoff, mitigate contaminants entering river networks leading to evidence of water quality improvements under base and elevated flow conditions [62] [63] [64]. Various interceptive mitigations are available across a range of farmed landscapes to reduce contaminant losses to surface waters although managing FIB losses on steep slopes presents challenges [65]. Implementing a package of mitigation measures for erosion-sediment control at multiple locations within a region can produce valuable co-benefits including improved microbial water quality on a regional scale though upgrading point source discharges provides stronger evidence of regional water quality improvements [66].

C2-(ii). Central government driven

Contamination of surface waters with pathogens is a national water quality issue. Policies and targets for human health are part of the NPS-FM 2020 [67] which provides attribute states (standards) for *E. coli* as the preferred faecal indicator bacteria for freshwaters, to manage the average level of health risk for contact recreation. Management to ensure risks to human health are within an acceptable limit is a statuary requirement of the NPS-FM and requires regional councils to set limits on resource use to achieve this outcome. A recent study estimated that a mean reduction of 73% of current load of *E. coli* was required across NZ to meet minimum bottom lines for contact recreation [68]. The NPS-FM does not have attributes for estuaries, but measures taken to reduce loss of *E. coli* into rivers are expected to contribute strongly to reduction of other FIB and faecal pathogens into downstream shellfish areas.

C2-(iii). Iwi/hapū driven

To safeguard the sustainability of shellfish populations, rāhui and/or temporary closures are employed by iwi, effectively prohibiting the harvesting of shellfish during designated periods [69]. These customary practices not only protect shellfish populations but can also prevent and protect communities from consuming contaminated shellfish. By restricting access to shellfish beds known to be affected by faecal contamination, rāhui and temporary closures help mitigate health risks. Given that Māori may be more likely to consume shellfish more frequently than the general population and thus have a higher risk of exposure [13], these protective measures are important for minimising exposure to potential faecal microbial contaminants from cultural practices.

C2-(iv). NGO, community driven

Ki uta ki tai – from mountains to sea – is a philosophy that acknowledges the connectivity between the land-to sea and people with the environment. Given the land use pressures and mobilisation of

FIB from upstream to downstream waters, this approach would provide an improved management approach to shellfish safety for consumption. Participatory approaches are necessary for implementation and the efforts from iwi and community driven interests and initiatives (e.g restoration planting and water monitoring using community-based water monitoring methods) should reduce the burden of faecal contamination of rivers and downstream coastal waters. Attributing improved faecal contamination status of coastal waters is challenging due to the complex connections between land and coastal environments (see Section C1).

C2-(v). Internationally driven

For commercial enterprises, current criteria for determining when BMS are safe to harvest are driven by multiple international market standards which either require compliance with a standard based on *E. coli* in shellfish flesh (EU) or compliance with a standard based on faecal coliforms in growing waters (US).

Part D—Impact analysis

D1. What would be the environmental/human health impacts of not managing this attribute?

Changes in the attribute state will affect human health. Failing to manage faecal contamination of estuarine and coastal waters would likely lead to increased disease burden among shellfish consumers and harm shellfish exports. Managing the sources of contamination (e.g. human and animal faecal sources) is a priority to ensure the protection of public health. Recent estimates indicate that around 8% of all norovirus infections in New Zealand are due to shellfish consumption, with commercially harvested oysters implicated in 85% of these outbreaks [13]. Since shellfish, through their filter-feeding activities can bioaccumulate faecal microbes particularly viruses from water and the median infectious dose for norovirus (i.e. the infectious dose at which there is a 50% chance of becoming ill) is less than 30 genome copies, even minimal faecal contamination of overlying waters poses a significant public health risk.

D2. Where and on who would the economic impacts likely be felt? (e.g., Horticulture in Hawke's Bay, Electricity generation, Housing availability and supply in Auckland)

The aquaculture industry particularly oyster farming, is highly susceptible to impacts from faecal microbial contamination. Oyster farms are typically located in shallow intertidal areas making them more vulnerable to episodic contamination from river plumes. In the event that a shellfish growing area is impacted by a wastewater pollution event, the area is generally closed for 28 days following the end of the event. The area can only reopen once evidence shows the contaminating event has ceased and microbial contaminant levels in water and flesh have returned to background levels [1]. Closure of shellfish farms has significant economic implications including revenue loss, increased operational costs, potential loss of market share, disruption of supply chains, reputational damage and additional regulatory compliance costs.

Recreational and cultural harvesting of shellfish would also be affected with associated socio-cultural impacts and impacts on cultural values.

D3. How will this attribute be affected by climate change? What will that require in terms of management response to mitigate this?

Climate change is likely to affect disease burden from exposure to pathogens though the direction of change may vary as environmental and climatic drivers of transmission differ among pathogens [70].

Key features of climate change expected are increased temperatures and variability in extreme rainfall events [71]. Runoff effects following short intense rainfall will increase the variability of river flows and expected variability in the faecal contamination status of estuarine and coastal waters. Bacteria indicator and pathogen concentrations can increase by 1 – 3 orders of magnitude during high flow conditions in pastoral streams [72] and can remain elevated. An increase in rainfall will also increase the chance of sewage overflows, and infrastructure damage [73]. Extreme weather events including more frequent flooding events, are expected to cause more land runoff resulting in episodic delivery of increased faecal contamination to coastal waters via rivers with an associated increased risk of waterborne faecal related diseases. Studies also suggest a decrease in FIB during summer months due to reduced runoff and increased temperatures that enhance bacterial die-off processes [8]. However, warming waters may also lead to increased persistence and survival of pathogenic bacteria including naturally occurring bacterial pathogens (eg *Vibrio* sp), posing additional risks.

The uptake and accumulation of FIB and pathogens by shellfish typically reflect the concentrations present in the overlying water. Temperature and salinity are key factors influencing shellfish filter-feeding activities and the rates of faecal microbe bioaccumulation. Warmer water temperatures increase shellfish clearance rates, while reduced salinity decreases feeding rates. Studies on FIB accumulation kinetics in shellfish show that they can respond quickly to environmental contamination with maximum concentrations accumulating within 30 minutes of exposure and persisting in flesh for at least a week after rainfall events.

Management responses to mitigate these impacts include developing climate-adaptive management strategies, such as adjusting harvesting times and locations, improving water quality monitoring, and enhancing shellfish treatment processes, e.g., depuration strategies to maintain shellfish safety.

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9.18 Cyanobacteria in water

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Preamble: There are three different concerns related to cyanobacteria in coastal waters.

- Planktonic cyanobacteria (PC) that live in the water column of estuaries/coastal waters.
- Benthic cyanobacterial mats (BCM) that live on substrates (i.e., mud, other organisms) in estuaries/coastal ecosystems.
- Freshwater cyanobacteria (FW) that live in upstream rivers or lakes, which flow into estuaries/coastal ecosystems. These freshwater species can survive some exposure to saline conditions. Additionally, the toxins produced by freshwater species, also flow into coastal waters and can accumulate in marine species.

There are different degrees of knowledge for each of the above and different response and mitigation actions are required. In each question below we will note how the answer/s related to each "type" of cyanobacteria.

State of knowledge of the "Cyanobacteria in coastal waters" attribute: Poor / inconclusive – based on a suggestion or speculation; no or limited evidence

Part A—Attribute and method

A1. How does the attribute relate to ecological integrity or human health?

Ecological integrity (PC, BCM)**.** Marine cyanobacteria can be beneficial with some species forming symbiotic relationships with other micro- and macro-eukaryotes. They can also play vital roles in the biogeochemical cycle of carbon and nitrogen and represent the primary nitrogen-fixing microorganisms in marine environments [1]. However, cyanobacterial blooms and proliferations of benthic mats can also alter trophic structure and functionality, and in the case of planktonic species, can also causes water column deoxygenation, leading to fish mortalities and decreasing water quality [2].

PC blooms can cause light limitation and the death of competing phytoplankton species and rooted aquatic vegetation. Cyanobacteria are often of low food preference for herbivores when compared to other phytoplankton [3, 4]. Planktonic cyanobacterial blooms are relatively common in estuaries and brackish waters in Aotearoa New Zealand. For example, there is a long history of planktonic cyanobacterial blooms in Te Roto o Wairewa / Lake Forsyth and Te Waihora /Lake Ellesmere [5].

BMC blooms can smother the underlying aquatic system, resulting in food web changes and in some cases killing the underlying substrate (i.e., sponges and corals; [6]). Once formed, the mats can impact the recruitment of corals and other benthic taxa, impacting food webs. For example, data from overseas indicates there can be shifts in reef fish communities [7].

Ecological integrity (FW)**.** FW species and their toxins can have profound impacts on coastal ecosystems. Internationally freshwater cyanotoxins (see below) have been responsible for the death of marine mammals and have been detected in a range of other marine organisms [8].

Human health (PC, BCM, FW)**.** Cyanobacteria can be hazardous to aquatic and terrestrial organisms, as some species produce highly toxic secondary metabolites, known as cyanotoxins that have killed numerous animals worldwide and endanger human health [9]. Exposure to cyanotoxins can occur in various ways, however, the oral route is the most important. In the marine environment, the most likely exposure route is though the consumption of contaminated seafood (e.g., [10]). Dermic exposure and inhalation are also possible especially during proliferations of BCM. Some species of BCMs, including those in Aotearoa New Zealand, produce lyngbyatoxin-a, a cyanotoxin that can cause acute dermatitis as well as eyes and throat irritation and respiratory issues [11].

A2. What is the evidence of impact on (a) ecological integrity or (b) human health? What is the spatial extent and magnitude of degradation?

Internationally, data and publications on cyanobacteria in coastal waters is relative limited, especially compared to their freshwater counterparts.

PC – Internationally, there is considerable evidence to show that PC blooms and the toxins they produce can have negative ecosystem effects. For example, the toxins commonly produced by the cyanobacteria in estuaries in Aotearoa New Zealand have been shown to affect the reproduction and survival or aquatic organisms (e.g., [12]) and when blooms are severe, they can create water quality issues such as low dissolved oxygen. These cyanobacteria also pose a health risk to human through exposure via by ingestion of containment seafood or water, inhalation of aerosolised cells or toxins, or dermal contact with cells / toxins. To date these blooms are constrained to a small number of brackish water lakes / estuaries in Aotearoa New Zealand

BMC – There is limited data from Aotearoa New Zealand, but there is an increasing body of international literature that has highlighted a range of negative ecological impacts from smothering other organisms to changing food webs [6]. Studies on the ecological impact of BCM have not been undertaken in Aotearoa New Zealand. There is strong evidence from international studies that shows a detrimental impact of aerosols from BCM [13], and some evidence from anecdotal reports in Auckland that this may also be an issue in Aotearoa New Zealand. Recently the toxins produced by BCM mats have been detected in shellfish but the toxicity to humans via shellfish consumption is unknown (Biessy L, in prep).

FW - There is evidence that some freshwater cyanobacteria can survive at different salinity levels in estuaries [14, 15]. As noted above, the main impact of FW species on coastal waters is likely through the contamination of organisms that ingest the toxins through contaminated food or water. The extent of the problem is unknown in Aotearoa New Zealand.

A3. What has been the pace and trajectory of change in this attribute, and what do we expect in the future 10 - 30 years under the status quo? Are impacts reversible or irreversible (within a generation)?

We anticipate that the frequency and severity of all three cyanobacterial types will increase over the next 10-30 years with rising eutrophication and sea temperatures [2]. However, assessing these changes will be challenging as currently very little data is collected on cyanobacteria in coastal waters. BCM are known to bloom seasonally, but the prevalence and duration of BCM blooms are increasing at an accelerating rate worldwide [6]. In Aotearoa New Zealand, there are records of blooms of the filamentous marine cyanobacterium, identified at the time as *Lyngbya majuscule* (now reclassified as species of *Lyngbya*, *Moorea*, *Okeania*, *Dapis* and others) in the Hauraki Gulf [16]. Blooms of benthic marine cyanobacteria at sites in eastern Auckland in the Hauraki Gulf as well as sites in the Manukau Harbour and Waitematā Harbour were reported with large rafts of this species washed ashore during warm summers in the years 1999-2001, 2003 and 2005 [17], with over 100 tonnes washing up on the beach for the 2005 event. For the last two summers (2022 and 2023), over 400 tonnes of BCM, identified as *Okeania* sp. (previously classified as *L. majuscula*), have washed up on Waiheke Island, affecting local communities and the environment.

The impacts of cyanobacteria in coastal waters should be reversible but these will take considerable catchment wide restoration. If the drivers / stressors were removed or reduced over time, we anticipate that cyanobacterial blooms / proliferations will decrease in severity. However, effects from cyanobacteria might not be reduced if these blooms were caused by larger scale climatic drivers, but due to the lack of research, this is unknown.

A4-(i) What monitoring is currently done and how is it reported? (e.g., is there a standard, and how consistently is it used, who is monitoring for what purpose)? Is there a consensus on the most appropriate measurement method?

To our knowledge, there is no current monitoring for any of the cyanobacterial types in coastal waters.

Auckland Council monitored BCM between 1999 and 2007 at selected sites. They used a graduated approach that included; (1) a regular assessments of algal mat build-up through the summer months, (2) working closely with the public health services regarding public health risks posed and the level of public nuisance, and (3) monthly sampling of the location of interest using a systematic grid and quadrats [18].

Australia, in particular the Queensland Region, has experienced frequent BCM proliferations over the last decade. The Moreton Bay Regional Council published a Harmful Algal Bloom Response Plan in 2018 including monthly monitoring (i.e., visual inspections from boats, combined with shore-based inspections) and a three-level response plan shown in Table 1 [19].

Table 1. Three-level response plan for marine benthic cyanobacterial blooms in Moreton Bay, Australia [19].

Alert level	Detection	Response
1	Small to moderate bloom material at locations away from developed areas	No action required to remove material, but signs to inform the public of the presence of a potentially

		harmful algal bloom may be appropriate. Activate stakeholder communications.
2	Large quantities of bloom material washing ashore or forming rafts adjacent to developed areas or areas of high public use	Activate or install signs immediately. Issue media release. Physically remove material from foreshores.
3	Very large quantities of material washed ashore or beginning to form large rafts adjacent to developed areas or areas of high public use	Same response as for Level 2, but closure of beaches may also be required, particularly where large amounts of blooms are growing close to the water's edge.

The Cawthron Institute, alongside Health New Zealand | Te Whatu Ora and the New Zealand Ministry of Health | Manatū Hauora, is currently working on a project titled "Managing marine harmful algal blooms (HABs) in recreational settings" [20]. This is a review of international material on recreational management of marine HABs (bathing, water sports and aerosols) to determine the feasibility of developing guidelines for Aotearoa New Zealand. The project is also identifying knowledge gaps required to form robust risk management approaches and including for BCM.

A4-(ii) Are there any implementation issues such as accessing privately owned land to collect repeat samples for regulatory informing purposes?

We do not envision significant barriers or issues related to implement monitoring for this attribute, although as noted above, there is a need to develop appropriate monitoring protocols. Most estuaries/ coastal waters are not on privately owned land, but access across private land might be required to reach suitable boat launching or monitoring sites. Iwi and hapū should be kept informed of any work and sample collection being undertaken in their rohe.

A4-(iii) What are the costs associated with monitoring the attribute? This includes up-front costs to set up for monitoring (e.g., purchase of equipment) and on-going operational costs (e.g., analysis of samples).

Field work would be required to monitor all types of cyanobacteria in coastal waters (PC, BCM, FW). As noted above, there are currently no national (or regional) monitoring protocols for any of the types of cyanobacteria in coastal waters and these need to be developed.

<u>PC and FW.</u> Sampling could be undertaken using similar approaches to those used to monitor lakes. This would likely involve travelling to the site(s), taking a boat to the sampling site(s), and taking grab or depth integrated water sample. The samples then need to be sent to a laboratory with algal identification expertise. The species and concentrations of cyanobacteria need to be determined using microscopy and this usually costs about $150 per sample. In addition to the above, there would be initial set up costs such as staff time to design monitoring programme and select high-risk sites.

<u>BCM</u>. Monitoring protocols still need to be developed. It is likely that they will involve diver surveys as well as assessments of beaches. Initially, the causative/dominant species needs to be identified using microscopy or genetic testing. Thereafter, it may be possible to use macroscopic surveys to assess the abundance of BCM.

A5. Are there examples of this being monitored by Iwi/Māori? If so, by who and how?

We are not aware of any monitoring of PC, BCM, or FW being undertaken by representatives of iwi/hapū/rūnanga. However, it is worth noting that Ngāti Paoa iwi and whānau have been severely impacted by recent BCM events on Waiheke Island over the last two summers. Restrictions on place (i.e., rāhui or temporary closures) are generally applied by iwi and hapū to ensure the safety of whānau and the wider community.

A6. Are there known correlations or relationships between this attribute and other attribute(s), and what are the nature of these relationships?

There are possible links to coastal water nutrient and sediment attributes (PC/ BCM). However, because there is insufficient data on the causes of increases in cyanobacteria in coastal waters in Aotearoa New Zealand, we are currently unable to establish this. If cyanobacteria in freshwater systems increase, this will impact on the occurrence of FW species and their toxins in coastal waters. There are well established links between cyanobacterial blooms in lakes and total nitrogen and total phosphorus levels. It would be reasonable to assume a relationship between FW cyanobacteria in coastal waters and the lake ecosystem attributes of total nitrogen and total phosphorus [21].

Part B—Current state and allocation options

B1. What is the current state of the attribute?

There is almost no information on the current state and distribution of marine cyanobacteria in Aotearoa New Zealand. A few small studies have characterised cyanobacteria communities in coastal waters (e.g., [22]) and work is currently underway on BCM in the Auckland / Waiheke Island region. These studies have primarily focused on biodiversity or identification of causative toxin-producing/bloom forming species and did not focus on studying spatial or temporal patterns.

As noted above, there is a long history of cyanobacterial blooms in Te Roto o Wairewa / Lake Forsyth and Te Waihora / Lake Ellesmere – which are often classified as estuaries or as Intermittently Closing and Opening Lakes and Lagoons [5]. More recent studies have explored bloom dynamics and the accumulation of cyanotoxins in tuna (eels; [23]) and the causes of these blooms [24].

We assume that all cyanobacterial types described here will become more problematic with increasing eutrophication and climate change. There is some evidence to suggest this in Aotearoa New Zealand with an increase over the last two summers in BCM the Auckland region / Waiheke Island and more frequent and intensive cyanobacterial blooms in freshwater bodies that flow into coastal systems.

We believe that there is **currently insufficient information** for marine cyanobacteria in coastal waters to be used as an indicator.

B2. Are there known natural reference states described for New Zealand that could inform management or allocation options?

There is no information on natural reference states for cyanobacteria in coastal waters. Paleolimnological studies have been undertaken in lakes [25, 26] in Aotearoa New Zealand, and this approach might be useful to estimate the natural reference state of FW cyanobacteria in marine systems. To date, the areas where paleolimnology studies of cyanobacterial communities have been undertaken are not lakes which flow directly into coastal systems. Paleolimnological studies could also be undertaken in estuaries, and this might help establish natural reference states for BMC and PC.

B3. Are there any existing numeric or narrative bands described for this attribute? Are there any levels used in other jurisdictions that could inform bands? (e.g., US EPA, Biodiversity Convention, ANZECC, Regional Council set limit)

We are unaware of any existing numeric or narrative bands for marine cyanobacteria. There are National Objective Frameworks for freshwater cyanobacteria, but these are not relevant for cyanobacteria in coastal waters.

B4. Are there any known thresholds or tipping points that relate to specific effects on ecological integrity or human health?

We are unaware of any specific thresholds or tipping points that relate to specific effects on ecological integrity or human health for cyanobacteria in coastal waters. Internationally, there has been some work undertaken to explore safe levels of cyanotoxin consumption in seafood (e.g.,[27]).

B5. Are there lag times and legacy effects? What are the nature of these and how do they impact state and trend assessment? Furthermore, are there any naturally occurring processes, including long-term cycles, that may influence the state and trend assessments?

There is insufficient data on cyanobacteria in coastal waters in Aotearoa New Zealand to assess any potential lag times, legacy effects, or natural oscillations.

B6. What tikanga Māori and mātauranga Māori could inform bands or allocation options? How? For example, by contributing to defining minimally disturbed conditions, or unacceptable degradation.

A high standard of water and sediment quality is an outcome sought by iwi/hapū/rūnanga. There is tikanga and mātauranga Māori relevant to informing bands, allocation options, minimally disturbed conditions and/or unacceptable degradation residing in treaty settlements, catchment/species restoration strategies, cultural impact assessments, environment court submissions, iwi environmental management plans, reports, etc.

<u>Part C—Management levers and context</u>

C1. What is the relationship between the state of the environment and stresses on that state? Can this relationship be quantified?

There are insufficient data on all cyanobacterial types (PC, BCM, FW) in coastal waters in Aotearoa New Zealand to determine this. However, based on our expert opinions and work on cyanobacteria in freshwater systems, we suggest that it is highly likely that increases in all types of cyanobacteria in coastal waters will occur with increasing nutrients, sediment and other contaminates, as well as warming sea temperatures.

C2. Are there interventions/mechanisms being used to affect this attribute? What evidence is there to show that they are/are not being implemented and being effective?

- **C2-(i). Local government driven**
- **C2-(ii). Central government driven**
- **C2-(iii). Iwi/hapū driven**
- **C2-(iv). NGO, community driven**
- **C2-(v). Internationally driven**

At present, we do not know of any interventions/mechanisms being used to affect this attribute, other than general freshwater and marine water and sediment quality management activities (which are covered by discussions of other attributes). Although cyanobacteria in coastal waters are becoming an increasing problem, data is so deficient that we are currently a long way from been able to implement interventions. However, as noted above, because of the likely close relationship with blooms / mat proliferation and eutrophication, any measures taken to reduce the input of nutrients and contaminants into coastal waters will likely also reduce cyanobacteria in coastal waters.

Part D—Impact analysis

D1. What would be the environmental/human health impacts of not managing this attribute?

This is unknown because sufficient data and information on cyanobacteria in coastal water in Aotearoa New Zealand is lacking. Based on the limited international research, we assume that if this attribute is not managed in the future, all three types (PC, BCM, FW) of cyanobacterial in coastal waters will increase in severity and frequency. This could have wide-reaching environmental, economic, and social impacts. For example, it could result in harvesting of seafood being restricted over certain periods and closure of beaches for recreational activities, and BCM mats could smother or impact important marine ecosystems (e.g., when tons of BCM mats were removed from Waiheke Island beaches, most of the shellfish beds that are necessary to feed colonies of native birds, were removed at the same time, impacting the food web).

D2. Where and on who would the economic impacts likely be felt? (e.g., Horticulture in Hawke's Bay, Electricity generation, Housing availability and supply in Auckland)

There is insufficient data on all cyanobacterial types (PC, BCM, FW) in coastal waters in Aotearoa New Zealand to determine this. However, given the recent issues in Auckland, we highlight this region where blooms of BCM have had a significant impact. BCM can wash up on the beach which results in access being restricted, impacting locals and tourism. The toxin detected in shellfish may impact the aquaculture industry. For example, the harvest and consumption of shellfish was prohibited by the Ministry for Primary Industries during the 2023 bloom (https://www.mpi.govt.nz/news/media- releases/public-health-warning-shellfish-biotoxin-alert-for-waiheke-island/). Removal of the BCM bloom from beaches on Waiheke Island cost approximately $250k in 2022 and $450k in 2023. An example of where FW bloom have had an impact on coastal waters occurred in 2004 in the Hokianga Harbour. Lake Omapere experienced severe cyanobacterial blooms, and these reached the Hokianga Harbour via the Utakura River. Cyanotoxins were then detected in oysters in the harbour prohibiting harvesting for a significant period [22, 28].

D3. How will this attribute be affected by climate change? What will that require in terms of management response to mitigate this?

Given the limited knowledge on cyanobacteria in coastal waters in Aotearoa New Zealand, we are unable to comment on this with any certainty. There is strong evidence that climate change will impact cyanobacteria in freshwater in Aotearoa New Zealand [29]. If freshwater blooms increase, then it is likely that the effect of FW cyanobacteria on coastal waters will also intensify. International studies suggest that climate change will increase the occurrence of BMC [6].

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