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'.

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

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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