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Setting instream nutrient concentration thresholds for nutrient-affected attributes in rivers

Guidance on implementing Clause 3.13 of the NPS-FM

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Clause 3.13 of the National Policy Statement for Freshwater Management 2020 (NPS-FM) requires regional councils (hereafter councils) to 'at a minimum, set appropriate instream concentrations and exceedance criteria for dissolved inorganic nitrogen (DIN) and dissolved reactive phosphorus (DRP)' to achieve target states (condition) for nutrient-affected attributes. An instream concentration threshold (ICT) delineates concentrations associated with desirable attribute states from concentrations associated with undesirable attribute states. An ICT is, therefore, attribute-specific.

The intent behind the directive to set temporal exceedance criteria was to:

- encourage councils to be aware of the temporal variability present in any instream nutrient time series
- understand how that temporal variability will complicate assessment
- develop methods that specify how you may infer whether nutrient concentrations are above or below an ICT and/or how tolerant councils are of individual observations of nutrient concentrations being above an ICT.

The purpose of this document is to offer guidance on setting ICTs for DIN and DRP to achieve targets for all nutrient-affected attributes in rivers. Although guidance for setting ICTs for the periphyton attribute has already been developed, the periphyton attribute remains within the scope of the present guidance. It is difficult to make effective and efficient decisions about how Clause 3.13 should be implemented when the periphyton attribute is excluded from the decision-making processes. In addition to periphyton, the nutrient-affected attributes covered by this guidance are: macroinvertebrates, fish, dissolved oxygen and ecosystem metabolism.

This guidance covers ICTs for rivers only. The direct effects of nutrients on organisms via ammonia and nitrate toxicity are out of scope. The part of Clause 3.13 that refers to setting ICTs related to downstream-receiving environments is also out of scope.

The authors have developed this guidance from an initial plan, ie, that was presented at an online workshop on 29 September 2021. Workshop attendees included representatives from most councils, and members of the Ministry for the Environment's Science Advisory Group (SAG). Feedback received from councils and the SAG during the workshop was incorporated into a more detailed report outline, which formed the basis of the present report (following further input from the SAG).

The authors of this report have tried to identify effective and efficient strategies for implementing Clause 3.13. We could not offer efficient strategies by limiting our guidance to councils alone. This guidance was written for all stakeholders that could/should play a key role in implementing the NPS-FM, including councils, non-council scientists and the Ministry for the Environment. Accordingly, this guidance uses pronouns such as 'we' and 'us' throughout to acknowledge that implementing the NPS-FM is likely best done through collaboration amongst multiple stakeholders.

This guidance document has two main sections:

- Section 2 — Attributes and their relationship with nutrients — which presents an overview of the relationships that instream nutrient concentrations have with the key nutrient-affected attributes of the NPS-FM
- Section 3 — Setting nutrient exceedance criteria — which presents strategies that may be used to set ICTs for nutrient-affected attributes in rivers.

Attributes and their relationships with nutrients

This section is divided into two major sub-sections:

- Effects of nutrient enrichment on river ecosystems
- Nutrient-affected NPS-FM attributes — properties and sensitivities.

Effects of nutrient enrichment on river ecosystems

This section presents a simple conceptual model and a review of how nitrogen (N) and phosphorus (P) enrichment affects *constituents* of river ecosystems. It differentiates ecosystem constituents from NPS-FM attributes. Any use of the word attribute(s) is reserved for a NPS-FM attribute, which may itself be a statistical summary of an ecosystem constituent (eg, MCI is a summary statistic of the macroinvertebrate constituent). It follows that knowledge of nutrient-attribute relationships is a subset of a broader knowledge of nutrient-constituent relationships. A broader knowledge of nutrient-constituent relationships may improve understanding of the relative sensitivites of NPS-FM attributes to nutrients and the uncertainties around those relationships. This understanding may strengthen the ecological foundations of any attempts to develop ICTs for NPS-FM attributes.

Key observations relevant to all ecosystem constituents are:

- Dissolved nutrients have a direct effect on periphyton (comprising algae and heterotrophic microbes), as well as on macrophytes (aquatic plants), but only indirect effects on other ecosystem constituents (macroinvertebrates, fish, dissolved oxygen and metabolism).
- Experiments have clarified the complex mechanisms by which nutrient enrichment affects river ecosystems, but applying that understanding more broadly to setting the ICTs is difficult due to:
  - the observation that nutrient effects on food webs often depend on the particular location and environmental setting of the experiment
  - the low number of nutrient treatments (usually just ‘control’ and ‘enriched’), make it difficult to identify concentrations at which signficiant deterioraton of an ecosystem constituent occurs.
- Descriptive/correlational studies often report a negative relationship between nutrient enrichment and various indices of ecosystem constituents that represent ecosystem health. However, many of these studies do not adequately separate the effect of nutrients from other correlated variables — including other anthropogenic stressors — and mediating factors. The statistical practices used in these descriptive studies erode confidence in resultant ICTs.

- Numerous environmental and biological factors mediate relationships between nutrients and ecosystem constituents, generally resulting in relationships characterised by a lot of variance and low nutrient-signal:noise ratios. Mediating factors common to all ecosystem constituents include:
  - shade
  - water temperature
  - hydraulic conditions
  - substrate
  - riparian inputs
  - top-down effects of invertebrates that feed on algae and detritus
  - multiple anthropogenic stressors
  - interactions among all of the above.
- Different ecosystem constituents exhibit different sensitivities to nutrient enrichment. As nutrient concentrations in rivers increase it is common to observe significant changes in some variables (eg, benthic algae species composition) before seeing any change in others (eg, benthic algal biomass). This generates the potential for NPS-FM attributes to exhibit different sensitivities to DIN and DRP.
- Among ecosystem constituents, few studies have aimed to compare and contrast the relative sensitivities of constituents to nutrient enrichment. The single study that has was inconclusive.
- Various dissolved nutrient thresholds were gleaned from the literature that councils may use for initial/draft ICTs in their regional plans. These thresholds are presented for chlorophyll *a* (table 2-1) and macroinvertebrates (table 2-2).

Nutrient-affected NPS-FM attributes — properties and sensitivities

This section discusses how the statistical properties of each attribute might affect its spatial and temporal sensitivity to variation in the water column concentrations of dissolved nutrients. A knowledge of the statistical properties and relative sensitivity of attributes may facilitate decisions concerning:

- how to best analyse data to determine ICT
- how to prioritise resources for monitoring attributes (eg, those likely to be the most sensitive indicators of nutrient management actions) and for developing new, attribute-specific models, assuming resources for NPS-FM implementation will be limiting.

No quantitative/objective evaluation of attribute sensitivities to nutrient enrichment is available. Accordingly, the guidance presents a qualitative/subjective evaluation of four factors affecting the sensitivity of individual attributes and/or the relative sensitivity of attributes to nutrient enrichment. These four factors are:

1. Sensitivity of an attribute to changes in nutrient concentrations in time.

- Attributes are likely to differ in the length of their lagged response to nutrient enrichment. For example, the Fish Index of Biotic Integrity (FIBI) is likely to exhibit longer lags in its response to nutrients than the Quantitative Macroinvertebrate Community Index (QMCI).

- By virtue of their statistical properties, certain attributes are likely to be more sensitive to nutrient enrichment than others. For example, community attributes based on presence/absence data (eg, the Macroinvertebrate Community Index (MCI) and the FIBI) are likely less sensitive than community attributes based on relative species abundances (eg, QMCI).

2. Spatial domain of attribute-nutrient relationships.

- Due to, for example, differences in mobility and longevity, attributes are likely to differ in the degree to which attribute state within a river reach is reflective of nutrient enrichment upstream of that reach, versus nutrient enrichment downstream and in adjacent catchments/systems.

3. The extent to which mediating factors will lower the sensitivity of attributes to nutrient enrichment.

- All NPS-FM nutrient-affected attributes are sensitive to multiple mediating factors, which will in turn add variance to nutrient-attribute relationships.

4. Whether non-NPS-FM metrics may respond to nutrient concentrations lower than those inducing a response in the attribute and, if so, consequences for relative sensitivity of attributes.

- For example, it is possible that periphyton species composition (not an attribute) responds to nutrient enrichment before chlorophyll *a* (the NPS-FM attribute). Noting that macroinvertebrate composition can change in response to periphyton composition, it is therefore possible that QMCI (for example) may be more sensitive to nutrient enrichment than the periphyton attribute.

There is much uncertainty about the sensitivity of NPS-FM attributes to spatial and temporal variation in nutrient enrichment.

Setting instream concentration thresholds

The purpose of Section 3 in this report is to:

- offer an interpretation of the directive by Clause 3.13 to set exceedance criteria, and some approaches to (a) quantify key uncertainties around model-based ICTs, and (b) assessment and limit-setting that accommodate those key uncertainties
- describe strategies that could be implemented to set ICTs for DIN and DRP within regions
- outline the consequences and trade-offs associated with choosing one strategy over another
- present some methods that may help implement individual strategies as well as choosing among alternative strategies
- offer some recommendations for how each strategy might be implemented, as well as proposing a pragmatic strategy as a starting point for implementation of Clause 3.13 of the NPS-FM.

We used well-established frameworks in decision science to derive and evaluate strategies. Use of these frameworks required careful explication of the aims/intent of Clause 3.13.

Four strategies for implementing Clause 3.13 were devised in light of two fundamental aims:

1. To establish a set of ICTs that is protective of the target states of all nutrient-affected attributes within regions.
2. To minimise the cost to councils of setting ICTs for nutrient-affected attributes.

The four strategies can be viewed as complementary and linked; they are not mutually exclusive alternatives.

Strategy 1: Use ICTs that have already been developed for a nutrient-affected attribute

Implementing Strategy 1 is straightforward and involves obtaining peer-reviewed, published ICTs from New Zealand technical reports and papers, ideally for all three National Objective Framework (NOF) band thresholds corresponding to attributes.

Strategy 2: Model ICTs for the most sensitive attribute

The objective of Strategy 2 is to generate, for each type of river, a single set of ICTs for an attribute determined to be most sensitive to nutrient enrichment, hence likely to be most protective of target states for all attributes.

Strategy 3: Model ICTs of a subset of attributes for which there is sufficient data

The objective of Strategy 3 is to generate, for each type of river, a set of ICTs for attributes for which there is sufficient data.

The key differences between Strategies 2 and 3 are the determinants of attributes selected for ICTs modelling. In Strategy 2, the aim is to model ICTs for attributes that are likely the most nutrient-sensitive attributes within each type of river *and* for which there is sufficient data. In Strategy 3 the primary determinant is data availability, resulting in a selection of attributes that are not necessarily the most nutrient-sensitive within river types.

Strategy 4: Implement monitoring to obtain data to refine ICTs for a subset of attributes

The objective of Strategy 4 is to evaluate whether collecting further data to refine ICTs of an attribute justifies the cost of that data collection and, if so, design and implement monitoring to obtain that data.

After exploring Strategies 2 and 3, it may be concluded that (a) ICTS are needed for particular attributes and (b) there is not enough data — nationally, regionally or both — to model ICTs for those attributes. In this case, councils, crown research institutes, central government agencies, partners and stakeholders (among others) may opt for designing an adaptive monitoring programme to collect the data required to develop and/or refine ICTs for a specific attribute over time.

Encouraging a phased approach

Councils need ICTs for inclusion in regional plans, so it is practical to implement Strategy 1 in the short term (before plan notification by December 2024).

Strategy 2 arguably provides the most effective and efficient set of ICTs for meeting the two aims listed above, but prioritising subsequent tasks of Strategy 2, let alone tasks of other strategies is challenging without having first completed particular tasks within Strategy 2. A major uncertainty hampering decision-making and prioritisation of activities (including tasks herein) is not knowing the limits of fiscal resources.

MfE and the authors of this report recommend completing particular tasks within Strategy 2 as a next step in implementing Clause 3.13. If resources permit, a national modelling approach to a task within Strategy 3 would complement Strategy 2 well.

We recommend these tasks be completed by a multi-agency research team working collaboratively with councils, towards meeting both of the fundamental aims listed above.

1. Introduction

1.1. Context and purpose

The National Objectives Framework (NOF) of the National Policy Statement for Freshwater Management 2020 (NPS-FM) is provided "to ensure that the health and wellbeing of degraded water bodies and freshwater ecosystems is improved, and the health and wellbeing of all other water bodies and freshwater ecosystems is maintained and (if communities choose) improved" (Policy 5 of the NPS-FM, 2020). The NOF underpins the NPS-FM policies preceding it, in particular Policy 1, which is that "Freshwater is managed in a way that gives effect to Te Mana o te Wai." Te Mana o Te Wai "refers to the fundamental importance of water and recognises that protecting the health of freshwater protects the health and wellbeing of the wider environment" (see Section 1.3, NPS-FM for the full explanation).

A NOF *attribute* is a "measurable characteristic that can be used to assess the extent to which a particular value is provided for" (NPS-FM, 2020). Attributes are presented in appendices 2A and 2B of the NPS-FM. The *values* included in the NPS-FM are either *compulsory values* (eg, ecosystem health) or *other values* (eg, water supply), definitions of which are presented in appendix 1A and 1B of the NPS-FM, respectively. Attributes are not restricted to those included in the NOF and regional councils (hereafter: councils) may identify other attributes for any compulsory value. The attributes discussed in this report are restricted to those currently included in the NOF.

Clause 3.13 of the NPS-FM sets out special provisions for nutrient-affected attributes (eg, periphyton), requiring councils to "at a minimum, set appropriate instream concentrations and exceedance criteria for dissolved inorganic nitrogen (DIN) and dissolved reactive phosphorus (DRP)" to achieve target attribute states "for periphyton, any other nutrient attribute, and any attribute that is affected by nutrients". Clause 3.13 requires councils to set *instream concentrations* and *exceedance criteria*, but these terms are not defined within the NPS-FM. Therefore, some working interpretations[1] of these terms are needed before we can progress with the rest of this guidance.

An *instream concentration threshold* (ICT) delineates concentrations associated with desirable attribute states from concentrations associated with undesirable attribute states. An ICT is, therefore, attribute-specific. The intent behind the directive to set temporal exceedance criteria was to:

- encourage councils to be aware of the temporal variability present in any instream nutrient time series
- understand how that temporal variability will complicate assessment
- develop methods that specify how to infer whether nutrient concentrations are above or below an ICT and/or how *tolerant* councils are of individual observations of nutrient concentrations being above an ICT.

[1] We deliberately use the word 'interpretations' rather than definitions, given the need to present more nuanced terms that more accurately capture the intent of Clause 3.13, before then defining those nuanced terms.

The intent behind the directive to set exceedance criteria in addition to ICTs is, therefore, consistent with Clause 1.6 of the NPS-FM.[2] However, interpreting and implementing the directive to set exceedance criteria is non-trivial. We provide an interpretation of the directive to set exceedance criteria at the beginning of Section 3.

Clause 3.13 requires councils to develop ICTs for "periphyton, any other nutrient attribute, and any attribute that is affected by nutrients" (Clause 3.13 (1)), and "for the upstream contributing water bodies to achieve the environmental outcomes sought for the downstream receiving environments" (Clause 3.13 (2)). Guidance on how to develop ICTs to achieve targets for the periphyton attribute (table 2 in appendix 2A of the NPS-FM) and nutrient-sensitive, downstream receiving environments (eg, lakes and estuaries) has already been developed (Ministry for the Environment, 2021). Clause 3.13 (4) lists examples of non-periphyton NPS-FM attributes affected by nutrients: dissolved oxygen, submerged plants, fish (rivers), macroinvertebrates and ecosystem metabolism.

The purpose of this document is to offer guidance on setting ICTs for DIN and DRP to achieve target states identified for the nutrient-affected attributes in rivers. The guidance is presented within the context of the nutrient-affected attributes identified in Section 3.13 paragraph (4) of the NPS-FM. However, the general principles and strategies presented apply to any nutrient-affected attribute, including those of the NOF and any other nutrient-affected attribute that councils include in their plans. Although guidance for setting ICTs for the periphyton attribute has already been developed, the periphyton attribute remains within the scope of the present guidance. This is because, as explained in Section 3, it is difficult to make effective and efficient decisions about how Clause 3.13 should be implemented when the periphyton attribute is excluded from the decision-making processes. The attributes that this guidance focuses on are listed in table 1.1.[3] For ease of reference, the NOF attribute tables are reproduced in section 7.

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2 Clause 1.6 provides direction around the requirement to use best information available at the time in giving effect to the NPS-FM.

3 We reiterate, however, that this guidance broadly applies to any nutrient-affected attribute; particularly Section 3 of this guidance.

Table 1-1: National Objectives Framework attributes covered in this guidance

No.	Attribute	Attribute unit	NPS-FM reference
1	Periphyton	Chlorophyll *a* (Chl *a*)	Appendix 2A, table 2
2	Fish (rivers)	Fish Index of Biotic Integrity (FIBI)	Appendix 2B, table 13
3	Macroinvertebrates	Macroinvertebrate Community Index (MCI)	Appendix 2B, table 14
4	Macroinvertebrates	Quantitative Macroinvertebrate Community Index (QMCI)	Appendix 2B, table 14
5	Macroinvertebrates	Macroinvertebrate Average Score Per Metric (APSM)	Appendix 2B, table 15
6	Ecosystem metabolism	Gross primary production (GPP) (grams of dissolved oxygen per square metre per day)	Appendix 2B, table 21
7	Ecosystem metabolism	Ecosystem respiration (ER) (grams of dissolved oxygen per square metre per day)	Appendix 2B, table 21
8	Dissolved oxygen	Dissolved oxygen (DO) (milligrams per litre)	Appendix 2A, table 7 Appendix 2B, table 17

1.2. Scope of this guidance

As specified in the project brief, the NOF attributes covered by this guidance are: periphyton, macroinvertebrates, fish (rivers), dissolved oxygen and ecosystem metabolism. Also included in the project brief was a scoping workshop, which involved presenting a draft project plan to, and receiving feedback from, council representatives, the Ministry-appointed Scientific Advisory Group and Ministry for the Environment staff.[4] The scoping workshop was held on 29 September 2021.

Feedback from councils and the SAG obtained through the scoping workshop was used to refine the scope of the guidance and develop a draft table of contents. The draft table of contents was then circulated to the Ministry for the Environment and the SAG as well as Mike Scarsbrook (Waikato Regional Council; attendee of the scoping workshop) and Ned Norton (Land Water People; NIWA sub-contracted advisor to this guidance).

Feedback on the draft table of contents was considered by Ministry and the NIWA project team during November 2021 and then finalised, thus serving as a plan for the guidance presented here. The following are within the scope:

- A section presenting a review of the ecology of the impacts of dissolved nutrients on river ecosystems. This section includes basic conceptual models of nutrient impacts on river ecosystems.
- A section reviewing the statistical properties of NPS-FM nutrient-affected attributes, and how those properties might affect the sensitivity of attributes to variability in dissolved nutrients.

[4] Workshop attendees represented: Southland Regional Council, Greater Wellington Regional Council, Waikato Regional Council, Otago Regional Council, Gisborne District Council, Auckland Council, Bay of Plenty Regional Council, Hawke's Bay Regional Council, Environment Canterbury, Northland Regional Council, Taranaki Regional Council and Tasman District Council; and (for the SAG representatives) James Cook University, Dairy NZ, and Kāhu Environmental, in addition to Ministry for the Environment staff and the NIWA and Cawthron Institute contributors to this report.

- A section aimed at identifying the strategies we may use to develop ICT; strategies that are both effective and efficient. This section was to help councils and other key stakeholders navigate the decisions that need to be made to implement Clause 3.13.

The following are out of scope:

- Co-development of strategies with councils to implement Clause 3.13, due to resource constraints.
- Estimation of ICTs. We offer guidance that *researchers*[5] may themselves use to develop ICTs for non-periphyton attributes.
- Lakes, wetlands and estuaries. This guidance is restricted to ICTs for rivers as specified in the project brief.
- Riverine macrophytes (aquatic plants) as these are not a compulsory NPS-FM attribute. They are considered in this guidance only with respect to how they mediate the effects of nutrients on dissolved oxygen and metabolism. Nevertheless, the strategies offered in Section 3 could be applied to aquatic plants.
- The direct effects of nutrients on organisms *via* ammonia and nitrate toxicity; attribute band thresholds for these attributes are presented in tables 5 and 6 of the NPS-FM.
- Guidance on how to develop and implement nutrient-attribute monitoring networks.
- ICTs related to downstream receiving environments (paragraphs (3)(b) and (3)(c) of Clause 3.13 of the NPS-FM). It is acknowledged that environmental outcomes for downstream receiving environments must be considered when setting exceedance criteria for nitrogen and phosphorus in rivers upstream. Refer to Ministry for the Environment (2021) for guidance on setting nutrient exceedance criteria for downstream receiving environments.

1.3. Who this guidance is for

This guidance has been written for all stakeholders who are likely to, or could, play a role in implementing the NPS-FM and specifically Clause 3.13. These stakeholders include:

- council staff
- Ministry for the Environment staff who — as recommended in Section 3 —could facilitate strategies to implement Clause 3.13
- non-council scientists who — as recommended in Section 3 — could facilitate strategies to implement Clause 3.13.

The authors have tried to identify effective and efficient strategies for implementing Clause 3.13. We could not offer efficient strategies by limiting our guidance to councils alone (Section 3). Accordingly, we deliberately used pronouns such as 'we' and 'us' to acknowledges that implementing the NPS-FM is likely best done through collaborations amongst multiple stakeholders.

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[5] We have deliberately used the word 'researchers' here to include councils as well as other research providers that could complete the task.

1.4. Structure of this guidance and how to use it

In Section 1, we present the background, purpose and scope of this guidance.

Section 2 presents an overview of the relationships that instream concentrations of DIN and DRP (and other forms of nitrogen and phosphorus) have with the key compulsory nutrient-affected attributes of the NPS-FM. Readers who have expert knowledge of both (a) the effects of nutrient enrichment on river ecosystems and (b) the statistical properties and sensitivities of the NPS-FM attributes of Table 1-1 may want to skip Section 2 or just scan the subsection summaries.

Section 3 presents a framework to facilitate the navigation of key decisions that must be made to define ICTs.

1.4.1. Attributes and their relationship with nutrients

Section 2 contains the information needed to understand the non-trivial nature of developing ICTs to satisfy Clause 3.13 of the NPS-FM. Among other outcomes, the content of Section 2 will:

- Facilitate an understanding of which variables to consider when modelling ICTs. For example, Section 2 will highlight the high potential for confounding nutrient gradients with other anthropogenic (eg, fine sediment) and natural (eg, hydraulic) gradients, informing model parameterisation.
- Shed light on epistemic uncertainties about nutrient-attribute relationships in rivers. Epistemic uncertainty refers to our lack of knowledge about the basic causal mechanisms by which — in this case — nutrients affect river ecology. A knowledge of epistemic uncertainties may inform design of monitoring strategies (eg, the knowledge may lead to collection of data to clarify mechanisms, thereby improving confidence in ICTs), and help prioritise strategies to develop ICTs (eg, you may choose to focus limited resources on attributes whose causal links to dissolved nutrients are better understood).
- Set expectations about the levels of ontological uncertainty underpinning nutrient-attribute relationships. Ontological uncertainty is sometimes referred to as statistical uncertainty, and it is affected by such factors as errors in measurement, imperfect parameter estimation (bias and variance) and choice of modelling approach.
- Introduce the relative sensitivities of the compulsory NPS-FM attributes to dissolved inorganic nutrients, thereby facilitating resources to be prioritised when developing ICTs.

Understanding the above four factors is necessary when modelling ICTs and is consistent with best practice in environmental modelling (Steps 3 and 5 of Jakeman et al, 2006).

1.4.2. Setting instream concentraton thresholds

Section 3 presents strategies that may be used to set ICTs for nutrient-affected attributes in rivers. It uses established frameworks of decision analysis,[6] which is a discipline orginally developed by statisticians (Raiffa, 1968) and applied to complex decision problems in environmental policy and management for decades (Hemming et al, 2022).

The emphasis of Section 3 is on broad strategies for implementing Clause 3.13 effectively and efficiently. Choosing this emphasis trades off a detailed consideration of statistical methods that councils may need to be aware of for modelling ICTs. While guidance for setting ICTs overseas focuses exclusively on how to model ICTs for individual attributes, this guidance does not follow that precedent. No assumption has been made that how to model ICTs for every attribute is the key to implementing Clause 3.13. Consideration of statistical details is just one subset of a larger, and likely more important, set of decisions that we need to make to implement Clause 3.13.

Expert, peer-reviewed and highly-cited documents are available that present guidance on best practice in environmental modelling. Repeating that information here was not deemed necessary or of a high priority. Instead, those texts are referenced as required throughout Section 3. Any of the papers cited in Section 3 can be obtained from the NIWA authors.

Section 3 offers processes and methods for answering questions such as:

- How can we reduce the costs of implementing Clause 3.13? That is, how can we maximise the efficiency of NPS-FM implementation?
- Do we need to model ICTs for every nutrient-affected attribute to implement Clause 3.13?
- How can we select a subset of nutrient-affected attributes for ICTs refinement, such that the resultant ICTs are protective of targets for the entire set of attributes?
- For which nutrient-affected attributes should we prioritise ICTs modelling?
- For which nutrient-affected attributes should we collect more data, towards reducing uncertainty about their ICT?

Section 3 does not answer all the questions that councils and other stakeholders may ask of Clause 3.13 but offers a pragmatic starting point for navigating the implementation of Clause 3.13.

[6] Also referred to as structured decision-making (Conroy and Peterson 2013).

2. Attributes and their relationships with nutrients

2.1. Introduction

This section is divided into two main parts:

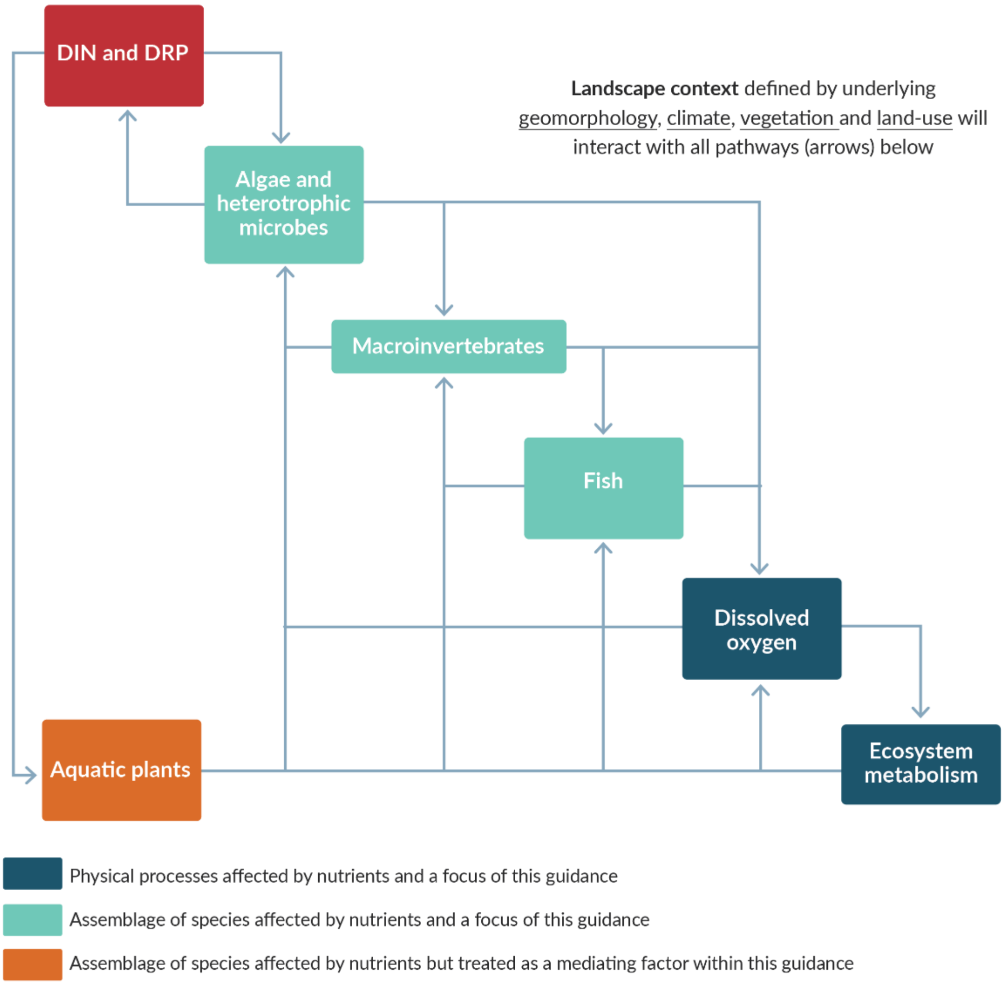
- **Section 2.2** provides a brief review of the effects of nitrogen (N) and phosphorus (P) enrichment in streams on the parts of the ecosystem (hereafter *ecosystem constituents*[7]) represented by the five NOF attributes which are the focus of this guidance. The review is organised according to the conceptual model in Figure 2-1, which presents cause–effect pathways (hereafter, *pathways*) between dissolved nutrients and the ecosystem constituents: three species assemblages and two physical ecosystem processes. Key messages from each review are presented in summary boxes at the start of the subsections covering the five constituents.

- **Section 2.3** addresses relationships between instream nutrient concentrations and the metrics specified for each of the five attributes in the appropriate NOF tables (see table 1-1 and appendix A).

[7] The term *ecosystem constituent* is used instead of *ecosystem component* to distinguish these constituents from the five biophysical components of the value *ecosystem health* listed in appendix 1A of NPS-FM.

2.1.1. A simple conceptual model

Figure 2-1: Simple conceptual model summarising the primary links between nutrients and the constituents of river ecosystems.



Effects of nutrients on many ecosystem constituents are not unidirectional; they involve feedbacks. Not all links are shown. Geomorphology, climate and land- and river-use all interact with each other to mediate the effects of links (arrows) among ecosystem constituents.

This guidance differentiates ecosystem constituents from NPS-FM attributes. Any use of the word attribute(s) is reserved for an NPS-FM attribute, which may be a well-defined statistical summary of an ecosystem constituent (eg, MCI is a summary statistic of the macroinvertebrate constituent). The distinction between ecosystem constituents and attributes is necessary (a) for clarity of guidance and (b) because the review presented in this section covers a broader set of responses within each ecosystem constituent, only a subset of which are NPS-FM attributes.[8]

[8] As an example: our review may cover the effects of nutrients on macroinvertebrate secondary production rate, which has a critical influence on other ecosystem constituents (eg, fish and ecosystem metabolism) but is not an NPS-FM attribute.

Following the definitions above, knowledge of nutrient-attribute relationships is a subset of a broader knowledge of nutrient-constituent relationships. A broader knowledge of nutrient-constituent relationships may improve understanding of the relative sensitivites of NPS-FM attributes to nutrients and the likely epistemic and ontological uncertainties around those relationships, and, consequently, may generally strengthen the ecological foundations of any attempts to develop ICTs for NPS-FM attributes.

Figure 2-1 is a simplification of the complex causal pathways and feedbacks that link dissolved nutrients to river assemblages and physical processes. The purpose of Figure 2-1 is to draw attention to the multiple different pathways among dissolved nutrients and ecosystem constituents. More complex pathways among dissolved nutrients and constituents are described within the subsections of Section 2.1.3, below.

Dissolved nutrients (ie, DIN and DRP) have a more direct effect on periphyton (comprising algae and heterotrophic microbes) than on other ecosystem constituents. DIN and DRP are essential raw materials for the growth of algae. Thus fewer mediating variables separate water column nutrients from periphyton growth than from (for example) macroinvertebrate growth (see figure 2-1). This conceptualisation is appropriate within the context of this guidance because:

- the direct effects of dissolved nutrients on macroinvertebrates and fish are manifest as toxicity effects, which are out of scope (see Section 1.2)
- excluding toxicity effects, dissolved nutrients will affect macroinvertebrates and fish through their effects on food web processes (ie, food quality and quantity)
- dissolved nutrients affect dissolved oxygen *via* the influence of nutrients on periphyton growth rates and biomass, and on subsequent secondary production (eg, in macroinvertebrates), and thence respiration and photosynthesis of these species assemblages
- ecosystem metabolism is estimated directly from balances of dissolved oxygen production (photosynthesis) and consumption (respiration), and is linked to nutrient concentrations only *via* these biological processes

Aquatic plants are not a focus of this guidance (see Section 1.2, Scope of this Guidance), but they may mediate the influence of dissolved nutrients on assemblages and physical processes, and so are included in Figure 2-1.

The response of all the ecosystem constituents to any specified level of nutrient enrichment will depend on the landscape context within a river reach, including:

- geomorphology (eg, riverbed substrates influence periphyton and microbial assemblages, hence rates of nutrient uptake into the food web)
- climate (eg, the influence of rainfall on river flow regimes, or temperature on algal growth rates, which, in turn, will influence nutrient retention within a reach [in, for example, algal mats], versus export)
- land- and river-use context (eg, riparian coverage) will influence light and water temperature, hence rates of nutrient uptake. They will also influence inputs of terrestrial carbon into streams, which provides a food source for instream organisms; and factors

like the magnitude of water abstraction may also influence instream nutrient concentrations.

2.1.2. About DIN and DRP and how they relate to other measures of instream nutrients

Dissolved inorganic nitrogen (DIN) comprises nitrite-nitrogen (NO_2-N), nitrate-nitrogen (NO_3-N) and ammoniacal nitrogen (NH_4-N). DIN is the most bioavailable form of nitrogen for plants and algae, including periphyton and aquatic plants in rivers. Many studies report on the relationships between river ecology and total nitrogen (TN), which is the sum of all types of nitrogen found in a water sample, including DIN and the nitrogen in organic substances like amino acids and plant tissues.[9]

Dissolved reactive phosphorus (DRP) comprises mainly the P in dissolved phosphate ions (PO_4^{3-}). Most P initially enters rivers attached (adsorbed) to sediment within runoff. While DRP in the water column is the most immediately bioavailable form of P, the P attached to sediment can be released within periphyton mats under certain conditions (eg, high pH and low oxygen concentrations), forming a source of DRP that is independent of water column concentrations. Total dissolved phosphorus (TDP) includes DRP and the P attached to small particles (that pass though a 0.45 µm filter) in the water column. Total phosphorus (TP) is a measure of all forms of P present within a sample and includes the phosphate bound to sediment and the P incorporated into organic molecules (like fats and nuleic acids), as well as DRP.[10]

2.1.3. Uncertainties about how DIN and DRP affect river ecosystems and use of best information under Clause 1.6 of the NPS-FM

Sections 2.2 and 2.3 of this guidance highlight major uncertainties about how dissolved nutrients affect ecosystem constitutents and nutrient-affected NPS-FM attributes. Consistent with Clause 1.6 of the NPS-FM (Best Information), uncertainties in this guidance are presented to inform (a) prioritisation of which information may be best to use for estimating ICT; and (b) collection of data to refine ICTs in the long term.

9 More information: Land, Air, Water Aotearoa (LAWA) — Nitrogen.

10 More information: Land, Air, Water Aotearoa (LAWA) — Phosphorus.

2.2. Effects of nutrient enrichment on river ecosystem constituents

2.2.1. Periphyton

Key messages

In cobble/gravel-bedded rivers – the primary river type in New Zealand – dissolved nutrients affect river ecosystems via the effects they have on the periphyton (comprising mainly algae and heterotrophic microbes) at the base of river food webs.

Algae

When conditions are suitable for biomass accrual (ie, in the absence of high flows that remove biomass), the growth of algae in freshwater is commonly limited by the availability of DIN or DRP, or both. Enrichment of rivers with these nutrients generally increases periphyton biomass. Theoretically, algal biomass should increase with dissolved nutrients up to a certain level of enrichment, beyond which non-nutrient resources (like space) become limiting and biomass should approach an asymptote. Some correlational/descriptive studies present empirical evidence for this asymptotic relationship aross sites/rivers.

Changes in the concentrations of N and P in rivers and alteration of their relative abundances (N:P ratio) can, respectively, alter the nutrient content and N:P ratio of algae, which affects food quantity and quality for invertebrate grazers.

Nutrient enrichment can change the species composition of algal assemblages, which, in turn, alters the nutritional value of algae to invertebrate grazers. Some evidence shows that alteration of algal species composition caused by nutrient enrichment is also associated with declining food quality for invertebrate grazers.

Algal species compositon and nutritional value may be more sensitive to nutrient enrichment (that is, respond at lower DIN and DRP concentrations) than algal biomass (as Chl *a*). It is possible that ICTs based on Chl *a* may not be protective of nutrient-affected attributes higher up the food chain, and therefore not consistent with Te Mana o te Wai.

Enrichment of rivers with nutrients can increase the abundance of toxic algal species within the algal assemblage.

Heterotrophic microbes

In many or perhaps all streams, microbial processes are just as critical to ecosystem functioning as algal processes and can respond to dissolved nutrients just as strongly as benthic algae.

Changes in DIN and DRP concentrations within rivers can:

- alter the abundance of microbes in rivers
- change the taxonomic composition of microbial communities
- increase (or decrease) the nutrient content of the microbial community.

All of these changes can, in turn, affect the quality of periphyton as food for invertebrates and fish.

Mediating factors

The functional relationship between dissolved nutrients and periphyton (algae/microbes) is mediated by:

- shade
- water temperature
- hydraulics
- substrate
- riparian inputs
- effects of invertebrates including top-down effects of consuming periphyton/microbes
- multiple anthropgenic stressors (eg, turbidity and deposited fine sediment)
- the complex interactions among all of the above.

Consequently, the observed nutrient-algae and nutrient-microbe relationships observed at landscape scales, among rivers, are typically characterised by a lot of variance. A proportion of the variance is due to spatial variability of biomass at small scales (ie, within sites) and it is important to minimise this variability by following standardised sampling protocols.

Periphyton — the organic material associated with rocks and other stable substrates in streams — is one of the main pathways by which dissolved nutrients enter river food webs, in particular in the hard-bottom rivers (ie, riverbeds dominated by gravel-sized or larger particles) that dominate New Zealand's river network (Ministry for the Environment, 2021).

Periphyton usually comprises a complex community of autotrophs (ie, primary producers – algae, including cyanobacteria) and heterotrophs such as fungi and bacteria.[11] The autotrophic part of periphyton (hereafter *algae* or *benthic algae*) often dominates biomass, so that periphyton can be represented by measurements of the areal concentration chlorophyll *a*, the photosynthetic pigment found in all algae. In some circumstances (such as shaded streams and streams with a lot of dead plant material), periphyton can be dominated by heterotrophic organsims (hereafter *heterotrophic microbes*), including those that break down dead/dying plant material within the river. This part of periphyton is not represented in chlorophyll *a* measurements.

Regardless of the composition of periphyton, and excluding direct toxicity effects, dissolved nutrients affect river food webs through their impacts on periphyton at the base of the food web.

[11] Note that this definition of periphyton is inconsistent with the meaning of the word "periphyton" (ie, peri = around, phyton = plants) but nevertheless is in common usage, especially in the USA and New Zealand. Therefore, in this section, we use separate terms for the autotrophic and heterotrophic parts of periphyton, viz. "algae" (represented by chlorophyll *a*) and "heterotrophic microbes" (not included in chlorophyll *a*). Note that the autotrophic component of periphyton includes algae attached to many surfaces as well as stones (eg, plants and wood). In New Zealand rivers, streambed stones are the predominant substrate for benthic algae.

Algae

Three effects of N and P enrichment on algae are relevant to this guidance (Ardón et al, 2021):

- changes in the nutrient content of algae
- increases in the rate of growth, often translating to increased biomass, of algal cells
- changes in the species composition of the algal assemblages in periphyton.

Algae can sequester and store N and P in their cells[12] (Sterner & Elser, 2003). Consequently, altering the relative concentrations of N and P in rivers can, in turn, affect the N:P ratios in algae (Iannino et al, 2020; Liess & Hillebrand, 2006; Stelzer & Lamberti, 2001). Shifts in the N/P *stoichiometry*[13] of periphyton may affect the efficiency with which periphyton is transferred into macroinvertebrate biomass (see Section 2.2).

The accrual of benthic algae within New Zealand's rivers can be limited by availability of N and P (Biggs, 2000) as is common with algal growth in many rivers of the world, under natural conditions, and when flow conditions are suitable (ie, in the absence of scouring high flows). Evidence of nutrient limitation of periphyton accrual comes primarily from three sources: nutrient diffusing substrates (NDS), experimental streams and mesocosms, and observational (correlative) studies. Numerous small-scale studies (eg, NDS and mesocosm experiments) have shown that, on average, algal biomass[14] is higher under enrichment with both N and P or N alone than under enrichment with P alone, indicating that algal biomass can be *co-limited* by N and P (Elser et al, 2007). In individual rivers, the limiting nutrient determined from *in situ* NDS experiments can vary over both space and time (Reisinger et al, 2016). Long-term ecosystem experiments have also shown that when rivers are enriched with N and/or P, algal biomass generally increases (Ardón et al, 2021). These controlled experiments allow the effects of nutrients to be isolated against a background of other mediating factors (Cross et al, 2006; Rosemond et al, 2015). A limitation of some of these experiments is they test for responses to very few nutrient concentrations, such as just a control/background concentration and an enriched concentration. A low number of nutrient treatments makes it difficult to (a) ascertain how a continuous gradient of nutrient concentrations affects algae and/or heterotrophic microbes; and (b) derive ICT.

Descriptive, landscape-scale studies have developed statistical models of average[15] benthic algal biomass as a function of average DIN and DRP concentrations across sites/rivers. These studies have shown there is much uncertainty concerning the shape and magnitude of periphyton biomass–nutrient relationships (Dodds et al, 2002). Some of the uncertainty arises from the variability inherent in sampling material that is distributed unevenly on streambeds and whose abundance is determined by multiple factors over a range of spatial scales from millimeters to kilometers. Following tested sampling protocols (eg, NEMS periphyton, Periphyton » National Environmental Monitoring Standards (NEMS)) can help minimise this source of variation but cannot eliminate it.

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[12] This trait of algae is sometimes referred to as *luxury uptake.*

[13] The study of how elemental compositions and/or ratios of organisms affect ecological processes is called *ecological stoichiometry.*

[14] Methods for measuring periphyton biomass vary across studies, and frequently involve use of chlorophyll *a* concentrations as a surrogate measure of algal biomass. For simplicity we use the term biomass in Section 2.1.2.

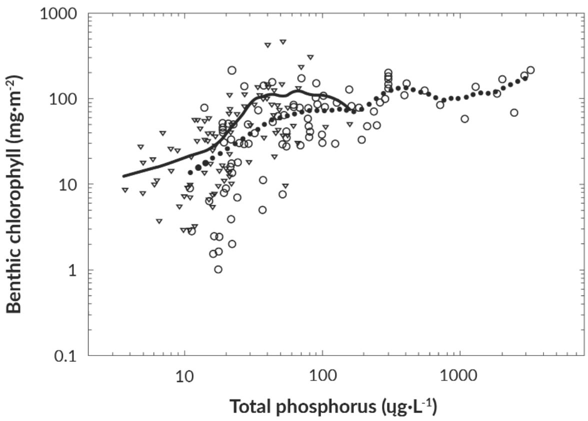
[15] We use the word *average* here for convenience, but the method of summarising periphyton biomasses and/or nutrients within sites, over time, varies across studies (eg, some may calculate medians, not means).

Nevertheless, even including multiple sources of variability, average periphyton biomass within sites/rivers generally increases as a function of average DIN and/or DRP, with some evidence for an asymptote at high nutrient concentrations (Dodds et al, 2006; Dodds et al, 2002; Snelder et al, 2022).

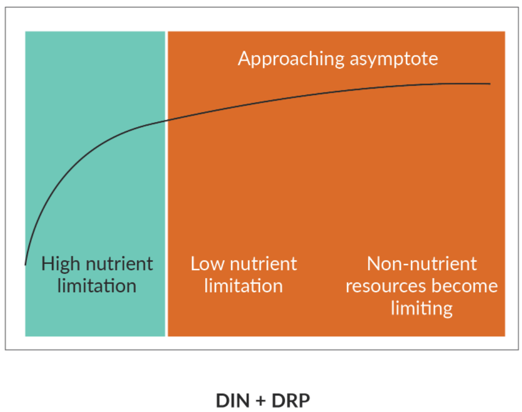
Assuming nutrient–benthic algae relationships are consistent within sites/rivers (see Mediating factors, below), you could reasonably expect an asymptotic relationship between average dissolved nutrients and average periphyton biomass across rivers. During accrual periods, algal biomass at individual sites will increase, with growth rate dependent on dissolved nutrient concentrations, until other resources — like available space on hard substrates — become limiting. When non-nutrient resources become limiting, or loss processes — such as sloughing (Bouletreau et al, 2006) or macroinvertebrate grazing are dominant — increasing dissolved nutrient concentrations will not necessarily increase algal biomass, leading to relationships across sites like those shown in Figure 2-2. Once nutrient concentrations have become saturating — that is, concentrations exceed the capacity of instream processes to take up the nutrient — then surplus nutrients can only be exported downstream (Mulholland et al, 2008).

Figure 2-2: Relationship between average nutrient concentrations and average benthic algal biomass (Chl *a*) across sites/rivers.

(a) Data from Dodds et al. (2006)



(b) Heuristic – expected form of functional relationship between dissolved nutrients and Chl *a*, in the absence of mediating factors



(a) Data and fitted splines from Dodds et al (2006).

(b) Conceptual model of the expected asymptotic relationship between dissolved nutrients and periphyton biomass. Two phases of the relationship are delineated: a phase of nutrient-limited periphyton growth, defined by high increase in periphyton as a function of nutrients (green); a phase where periphyton growth becomes limited by non-nutrient resources (orange).

Experiments have shown that increasing nutrient concentrations in rivers can change the species compositon of benthic algae (Iannino et al, 2020). For example, Rosemond et al (1993) demonstrated that diatom taxa responded positively to nutrient enrichment in streams, while abundance of other algal taxa was reduced. In a whole-of-ecosystem experiment, Slavik et al (2004) showed that long-term enrichment of streams with DRP and DIN resulted in strong shifts in communities of benthic primary producers, from algae to mosses. Stelzer and Lamberti (2001) showed the taxonomic compositon of periphyton in their experiment was related to concentrations of DIN and DRP, as well as to DIN:DRP ratios. Their experiment also showed periphyton taxonomic composition exhibited a stronger response to nutrient enrichment than that of chlorophyll *a* (hereafter Chl *a*), periphyton biovolume and total periphyton carbon. This was particularly so after four weeks of enrichment by which time biomass sloughing had begun (Bouletreau et al, 2006). Experimental work by Cashman et al (2013) showed the stoichiometry (carbon and nitrogen content) and fatty acid composition[16] of periphyton may significantly change following nutrient enrichment, without a significant increase in periphyton biomass.[17] It is possible, therefore, that ICTs based on algal biomass (ie, chlorophyll *a*) may not be protective of nutrient-affected attributes higher up the food chain.

Enrichment of rivers with nutrients may increase the abundance of toxic algal species within the periphyton assemblage. In New Zealand, for example, abundance of the toxic cyanobacterium *Microcoleus autumnalis*[18] has been shown to increase as DIN increases, at least up to moderate concentrations (~0.6 mg L^{-1}, Wood et al (2020)), although the field observations were not supported by experiments (McAllister et al, 2018).

We are aware of few studies of the effects of nutrient mitigation on periphyton. One example (Suplee et al, 2012) reported declines in periphyton chlorophyll *a* in a river over a period of ~12 years in response to improvements in waste-water treatment that reduced TP concentrations. However, the decline was not observed at all sites. The authors concluded that successful mitigation efforts “require achievement of concentrations below saturation and likely close to natural background.”

Heterotrophic microbes

Effects of nutrient enrichment on heterotrophic microbes are less well known than effects on algae. We do know, however, that in many streams and, in particular, shaded forested streams, microbial processes can be as important to ecosystem functioning as algal processes are in streams that are well illuminated (Ardón et al, 2021). Microbial processes can also respond strongly to DIN and DRP enrichment (Cross et al, 2006). Effects of inorganic nutrients on riverine heterotrophic microbes are mostly studied within the context of microbe-dependent processes, like leaf litter breakdown and subsequent effects on macroinvertebrate production. Accordingly, nutrient–heterotrophic microbe relationships is covered further in Section 2.2.2. For now, three important effects of increasing DIN and DRP on heterotrophic microbe communities within rivers (Ferreira et al, 2015) are as follows:

- an increase in microbe abundance (Rubin & Leff, 2007; Woodward et al, 2012)
- changes in their taxonomic composition (Zeglin, 2015)
- an increase in their nutrient content (Gulis & Suberkropp, 2003).

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[16] Fatty acids are lipids (fats), some of which are essential requirements for macroinvertebrates and must be obtained through their diets (eg, from benthic algae or terrestrial detritus).

[17] The Cashman et al (2013) experiment was in streams with background concentrations of 0.49 to 1.15 mg DIN/L, and 24–30 mg TDP/L and concentrations were increased by ~2-3 percent (N) and ~12 percent (P).

[18] Previously known as *Phormidium autumnale.*

Mediating factors

With respect to periphyton (ie, the combination of benthic algae and heterotrophic microbes), several mediating factors relevant to this guidance have already been discussed in other New Zealand guides (Ministry for the Environment, 2021; Snelder et al, 2022). Accordingly, only a brief summary is provided below of the dominant factors that mediate the responses of benthic algae and heterotrophic microbes to dissolved nutrients. Freshwater scientists know less about the mediation of responses by heterotrophic microbes to dissolved nutrients than they do about that of algae.

1. *Shade*. Shade limits algal growth rates (Halliday et al, 2016; Hill et al, 1995) and so reduces the maximum benthic algal biomass observed at sites at a given concentration of dissolved nutrients (Figure 2-3). Heterotrophic microbes do not require light for energy production, so the impact of shade on microbial processes is likely minor.
2. *Temperature*. Water temperature is one of the strongest regulators of biological processes (Brown et al, 2004), and exerts a strong influence on biological processes in rivers (Caissie, 2006). Temperature increases the rate of both algal and heterotrophic microbial growth in rivers (Ardón et al, 2021), such that for any specified level of dissolved nutrients, microbial and/or algal (Figure 2-3) accrual will be faster, and therefore potential maximum biomass higher when water is warmer.
3. *Hydraulics*. Hydraulic mediating factors include the effect of floods and sustained velocities within a river reach.[19] Floods increase water velocities and mobilise substrates, both of which can reduce periphyton biomass. Most New Zealand rivers experience frequent flood events, meaning that the algal and heterotrophic microbial assemblages of our rivers are likely most often in a non-equilibrium state — a state of recolonising substrates and community succession (Biggs & Stockseth, 1996). Consequently, rivers experiencing more frequent flooding will support lower time-averaged biomass of both algae and heterotrophic microbes (Hoyle et al, 2017) (Figure 2-3). Higher sustained water velocities within a river reach (eg, > 1.5 m s^{-1}) generally suppress periphyton biomass (Flinders & Hart, 2009) (Figure 2-3). On the other hand, increased velocities up to a certain point (eg, as in riffle habitats) can facilitate delivery of dissolved nutrients to algae adapted to high velocity conditions, which explains observations of high biomass in some low-nutrient streams (Larned et al, 2004).
4. *Substrate*. Substrate composition of the streambed will mediate the nutrient-periphyton relationship. For example, for any specified nutrient concentration, in hard-bottom streams, large, stable substrate will support higher periphyton biomass than smaller, more mobile substrates (Biggs et al, 1999; Hoyle et al, 2017). Within soft-bottom rivers, the primary substrate for algal colonisation may be aquatic plants,[20] which are likely to support different algal assemblages than inorganic substrates. Rates of nutrient uptake are likely to differ between rivers dominated by plants and those dominated by inorganic substrates.
5. *Riparian inputs*. The quantity of riparian inputs, hence the quantity of dead plant material in a river, will affect microbial biomass (Ferreira et al, 2015; Woodward et al, 2012). Both total microbial and algal biomass will affect the rate of nutrient uptake into the food web (Cross et al, 2006; Gulis & Suberkropp, 2003). It follows that riparian inputs will mediate nutrient-algae and nutrient-microbe relationships.

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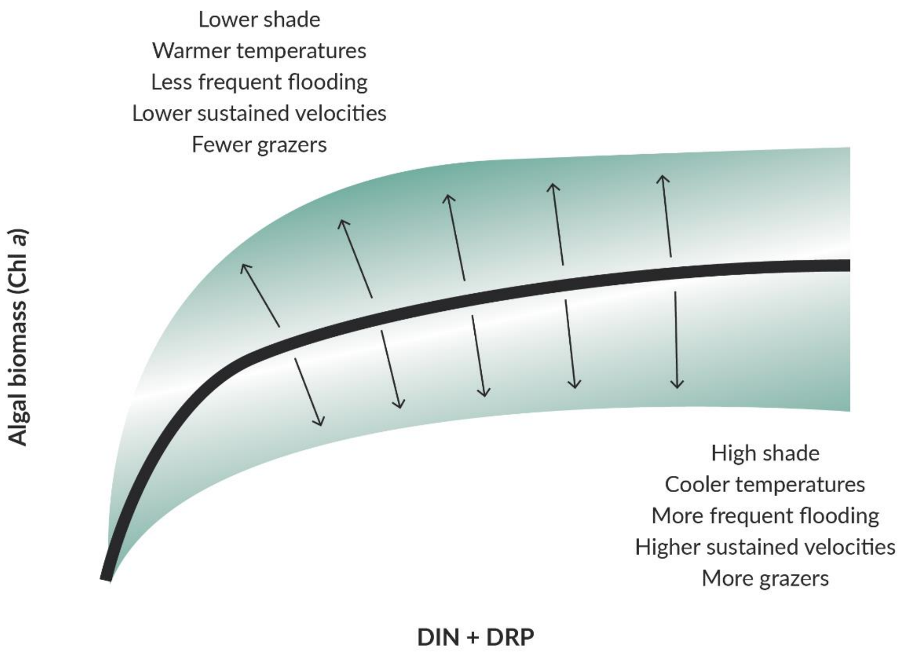
[19] River geomorphology (eg, segment slope) will affect the velocities experienced within a river reach.

[20] Note that algal mats can develop on soft substrata under very slow-flowing stable conditions.

6. *Consumers*. Grazers[21] (eg, snails, many mayfly and caddisfly larvae) and shredders (eg, some beetle and caddisfly larvae) are primary consumers that feed on periphyton, channelling nutrients up the food chain (Elwood et al, 1981). Grazers typically harvest algae and shredders harvest coarse organic matter including microbial heterotrophs. Thus, at any specified level of dissolved nutrients, high densities of these primary consumers may confound nutrient-algae and nutrient-microbe relationships (Figure 2-3) (Rosemond et al, 1993).
7. *Multiple stressors*. Effects of anthropogenic nutrient enrichment on both algal and microbial communities will be mediated by other anthropogenic stressors. For example, suspended and deposited, fine, inorganic sediment can hamper algal production, weakening the nutrient-algae relationship through restricting light. However, P-rich sediment deposits may favour growth of some algae (Wood et al, 2015b). Other stressors include increased temperature (from removal and shade) and increased inputs of contaminants such as herbicides.
8. *Interactions*. While all of the above mediating factors can complicate nutrient-algae and nutrient-microbe relationships on their own, they are likely to also interact with each other to further increase uncertainty in nutrient-periphyton and nutrient-microbe relationships (Munn et al, 2010; Sturt et al, 2011).

Figure 2-3: Many factors mediate the relationship between dissolved nutrients and algal biomass



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[21] Classifications of macroinvertebrate functional feeding groups vary across the literature, but generally include: grazers (also called scrapers or browsers that consume mainly algae); shredders (that consume mainly coarse organic matter, leaves, etc); collectors (that consume mainly fine organic deposits or suspended particles from the water column); and predators (that consume other animals).

2.2.2. Macroinvertebrates

Key messages

Descriptive studies and correlational studies

Descriptive, correlational studies report threshold nutrient concentrations, above which significant declines in various macroinvertebrate community indices may be observed. These threshold concentrations have been recommended as ICTs by some authors.

Factors lowering confidence in the thresholds reported by correlational studies include: (a) landscape-scale gradients in dissolved nutrients are often confounded by several other anthropogenic stressors — the forms of land use that elevate nutrients in streams also degrade invertebrate habitat in other ways — and (b) the fact that these confounding effects cannot be completely removed using statistical models.

Experimental/mechanistic studies

It is possible that a positive relationship between nutrients and macroinvertebrates (the *subsidy* effect, where a positive effect is taken as increasing taxonomic diversity or other metrics indicating a 'healthy' community) occurs until a threshold level of enrichment, beyond which nutrients may have a negative effect on macroinvertebrates (the *stress* effect). The prevalence of a nutrient *subsidy–stress relationship* in macroinvertebrate communities is unclear. The mechanisms underlying the subsidy (positive) side of the relationship are well represented in the literature. The mechanisms underpinning the stress (negative) side of the relationship may include altered habitat and food quality caused by nutrient-driven algal proliferations.

Sustained enrichment of rivers with nutrients can change the species composition and N:P ratio of periphyton, which can in turn:

- alter the species composition of macroinvertebrate communities
- change the nutritional composition of periphyton, with some studies reporting a decline in nutritional value to macroinvertebrates, and other studies reporting an increase in nutritional value.

Changes to periphyton species composition and nutritional status can occur at nutrient concentrations lower than those causing significant changes in periphyton biomass.

Increased primary and secondary production associated with nutrient enrichment can increase river respiration, creating hypoxic conditions lethal to macroinvertebrate fauna in some contexts.

There is some evidence that ICTs based on periphyton Chl *a* may not be protective of macroinvertebrate communities and so may not satisfy Te Mana o te Wai. However, there is much uncertainty concerning the relative sensitivity of ecosystem constituents to dissolved nutrients.

Mediating factors

Dissolved nutrients affect macroinvertebrates through their effects on algae and heterotrophic microbes. It follows that the factors mediating nutrient-periphyton and nutrient-microbe relationships will also mediate nutrient-macroinvertebrate relationships. In addition, reduced oxygen concentrations associated with nutrient stimulation of both algae and heterotrophic microbes can negatively affect sensitive macroinvertebrate taxa.

Numerous studies have reported correlations between nutrient enrichment and macroinvertebrate community indices and species richness (King & Richardson, 2003; Niyogi et al, 2007; Wang et al, 2007; Weigel & Robertson, 2007). These studies often report a threshold nutrient concentration above which a signficiant decline in the health of the macroinvertebrate community is observed (Evans-White, Dodds, Huggins et al, 2009; Wang et al, 2007; Weigel & Robertson, 2007). These threshold concentrations may be deemed ICTs (Wang et al, 2007). However, to identify such thresholds, these studies sampled macroinvertebrate community structure along land-use gradients where variation in nutrient concentration is often confounded by other anthropogenic stressors (Weigel & Robertson, 2007). Some of these correlational studies document the extent to which anthropogenic stressors are confounded, and use analyses that facilitate the partitioning of variance contributed by multiple interacting stressors (Wagenhoff et al, 2017). Nevertheless, it is impossible to completely remove confounding effects in correlational studies, lowering confidence in the reported effects that dissolved nutrients have on macroinvertebrates (Suter, 2001). Some studies have greatly improved confidence in ICTs derived in correlational studies by corroborating correlational results with controlled experiments (King & Richardson, 2003), but such comprehensive studies are rare.

Many studies have aimed to improve understanding of, and confidence in, nutrient-macroinvertebrate relationships by accounting for processes at multiple levels of the food web. A brief review of these studies is presented below. Assuming dissolved nutrient concentrations are below those known to induce toxic effects on macroinvertebrates, dissolved nutrients can only affect macroinvertebrates indirectly, through the direct effects that dissolved nurients have on periphyton and microbes (Figure 2-1).

Effects *via* benthic algae

Section 2.2 discussed the major effects of dissolved nutrients on the algae in periphyton, which can, in turn, have the following consequences for macroinvertebrates that consume algae:

- An increase in the rate of benthic algal production can be beneficial to many grazer species, increasing grazer biomass. In the context of the algae-macroinvertebrate pathway, it is possible the positive relationship between nutrients and macroinvertebrates (subsidy effect) occurs until a threshold level of enrichment, beyond which nutrients may have a negative effect on macroinvertebrates (stress effect).

- Changes in the nutrient content and species compositon of algae may alter the nutritional value and toxicity of foods available to grazers, and also the physical habitat available to macroinvertebrates, which may, in turn, affect grazer biomass and grazer species composition.

- Very high levels of plant biomass (including algae) may raise ecosystem respiration rates to levels where hypoxia may occur at night, potenially impacting on sensitive macroinvertebrate taxa (this pathway is covered in Section 2.2.4).

Nutrient enrichment increases algal growth rates (Section 2.2), which, in turn, increases production of grazing invertebrates, as well as the invertebrate predators that feed on grazers (Ardón et al, 2021). It is possible the response of macroinvertebrate variables is unimodal or 'hump-shaped', indicating a subsidy-stress response of macroinvertebrate communities to dissolved nutrients (Quinn, 2000). Subsidy-stress relationships have been recorded in a streamside mesocosm experiment for the density of certain macroinvertebrate taxa as well as for the density and richness of all taxa combined or just those belonging to the sensitive insect orders, Ephemeroptera, Plecoptera and Trichoptera (EPT) (Wagenhoff et al, 2012). However, field surveys in New Zealand have not shown strong evidence for such unimodal patterns, possibly due to the influence of other stressors (Wagenhoff et al, 2017). In cases where such patterns occurred, the inflection point where subsidy turns into stress was at relatively low levels of nutrient concentrations (Wagenhoff et al, 2011). The mechanisms underpinning the subsidy (positive) side of the nutrient-macroinvertebrate relationship are relatively well understood (Elwood et al, 1981; Rosemond et al, 1993; Slavik et al, 2004). In contrast, there is much uncertainty about the prevalence of the stress (negative) side of the nutrient-macroinvertebrate relationship, as well as the mechanisms that might drive any such stress effect (but see below, the discussion of Iannino et al, 2020).

Nutrient enrichment alters the species composition of the algae in periphyton (Section 2.2), which in turn alters the availability of food to grazers (Dudley et al, 1986). Algal species vary in their morphology, and some morphologies may be more difficult to graze than others. For example, algal-piercing caddis flies of the family *Hydroptilidae* generally become more dominant when streambeds are covered by thick mats or long filamentous algae, and grazers of thin biofilms such as the common mayfly *Deleatidium* become less common or disappear (Quinn, 2000).

Changes in the nutrient and species composition of benthic algae can affect its nutritional value to grazers. Enrichment of rivers with N and/or P will affect the N:P stoichiometry of the periphyton assemblage (Section 2.2). Controlled experiments have shown these changes in the stoichiometry of periphyton can then go on to increase production of grazing invertebrates (Liess & Hillebrand, 2006; Stelzer & Lamberti, 2002; Tibbets et al, 2010). Based on experimental results, you cannot necessarily infer a negative effect of nutrient-enriched stoichimetry on macroinvertebrate grazers; production of these macroinvertebrates often increases, depending on which nutrients are limiting.

Descriptive studies along landscape-scale gradients in nutrient enrichment have correlated periphyton stoichiometry with macroinvertebrate community structure. Liess et al (2012) studied periphyton stoichiometry and macroinvertebate communities along a New Zealand riverine nutrient gradient. The studies showed that periphyton N:P stoichiometry was correlated with and reflected instream nutrient concentrations. As the concentration of N in periphyton relative to P increased (higher N:P), macroinvertebrate taxon richness declined. For example, a four-fold increase in periphyton N:P was related to the loss of 50 percent of invertebrate species, but with very high uncertainty (less than 15 percent of the variance in taxon richness was explained by this negative relationship) (also see Evans-White, Dodds, Huggins, et al, 2009).

Nutrient enrichment may affect the micronutrient (eg, fatty acid) composition of benthic algae (see Section 2.2), which may, in turn, affect macroinvertebrate production. Guo et al (2016) showed experimentally that nutrient enrichment may increase highly unsaturated fatty acids, which, in turn, increased growth rates of grazers. In a study of periphyton fatty acid composition along a P-enrichment gradient, Iannino et al (2020) offered a possible causal explanation for the stress effect that is sometimes observed in nutrient-response relationships. Along a DRP gradient from 5 to 100 µg L^{-1} essential polyunsaturated fatty acids peaked at intermediate DRP concentrations (ca 50 µg DRP L^{-1}). Consumer performance increases with the polyunsaturated fatty acid content of food.

Descriptive and experimental studies have documented changes in macroinvertebrate densities and/or community composition along gradients in nutrient enrichment, but without an observable change in algal biomass (Elwood et al, 1981; Liess et al, 2012; Rosemond et al, 1993). These observations, coupled with studies showing that periphyton compositon and nutritional values may be more sensitive to nutrient enrichment than periphyton biomass, provide some evidence that ICTs based on periphyton Chl *a* may not be protective of macroinvertebrate communities. There is, however, much uncertainty concerning the relative sensitivity of ecosystem constituents to dissolved nutrients (Ardón et al, 2021).

Nutrient enrichment may increase the abundance of toxic algae within the periphyton assemblage (Section 2.2), which may, in turn, be toxic to some macroinvertebrates (Anderson et al, 2018; Camargo & Alonso, 2006). The extent to which this pathway is a problem in New Zealand is under investigation (Wood et al, 2014).

Effects *via* heterotrophic microbes

Section 2.2 discussed the key effects of dissolved nutrients on heterotrophic microbes, which can then have the following consequences for macroinvertebrates that feed on detritus (grazers/shredders):

- an increase in the rate of microbial production can be beneficial to many shredder/grazer species, increasing macroinvertebrate biomass
- long-term enrichment of microbial-based, river food webs can result in changes in the macroinvertebrate species composition
- very high levels of microbial production may increase ecosystem respiration during day and night, resulting in the depletion of dissolved oxygen and possibly hypoxia for some macroinvertebrate species. (This pathway is covered in Section 2.2.4.)

Experimental enrichment of rivers containing high quantities of plant material (ie, substrate for microbial assemblages) has shown that increased rates of microbial production (Section 2.2) can lead to strong increases in macroinvertebrate biomass (Ardón et al, 2021; Cross et al, 2006; Davis et al, 2010; Rosemond et al, 2015). In a long-term, whole-river experiment, Davis et al (2010) documented a shift in the species composition of a detritivorous, macroinvertebrate community in response to nutrient enrichment. Long-term enrichment resulted in an increase in the abundance of large-bodied species within the macroinvertebrate community. A novel outcome was that this change in the size structure of the invertebrate community impaired production of a salamander — the top predator in the experimental system — whose gape was too small to consume the large macroinvertebrate species. Although the authors of this study interpreted this effect as a reduction in ‘food web efficiency’, the experimental system was quite unique, and further review is needed to establish the generality of the result.

Mediating factors

Dissolved nutrients affect macroinvertebrates through their effects on periphyton (Figure 2-1). It follows that the factors mediating nutrient-algae and nutrient-heterotrophic microbe relationships (Section 2.2) will also mediate nutrient-macroinvertebrate relationships. Additional factors that can influence macroinvertebrate assemblages include riparian condition (eg, habitat for adult stages of aquatic insects), fine and coarse organic matter inputs, sources of drifting colonists, river-flow regime (including water velocity effects) and substrate size and stability.

2.2.3. Fishes

Key messages

Freshwater scientists know less about nutrient-fish relationships than they do about relationships among nutrients, periphyton (comprising algae and heterotrophic microbes) and macroinvertebrates.

Descriptive and correlational studies

Correlative studies that have compared the relative sensitivity of periphyton, macroinvertebrates and fish to nutrient gradients in rivers have generally concluded that fish communities are the least sensitive to nutrient enrichment. These results contrast with a global meta-analysis of experimental results, which concluded that riverine fish are not necessarily less sensitive to nutrient enrichment than periphyton and macroinvertebrates.

Nutrient enrichment has been correlated with changes in fish community composition, with some fish species increasing with nutrient enrichment and other species decreasing in abundance.

Certain indices of fish community health have been negatively correlated with dissolved nutrients in rivers and these negative relationships have been used to identify potential ICTs in North America.

Experimental/mechanistic studies

Ascertaining nutrient-fish relationships that can be generalised from controlled experiments is difficult due to the highly context-dependent nature of results.

Mediating factors

Dissolved nutrients affect fishes due to how they affect algae and heterotrophic microbes, then macroinvertebrates. It follows that the factors mediating nutrient-algae, nutrient-microbe and nutrient-macroinvertebrate relationships will also mediate nutrient-fish relationships. In addition, *mass effects* will mediate fish-nutrient relationships. Mass effects refer to the effects of fish movement from neighbouring populations on local populations. Mass effects mediate local environment-assemblage relationships, by weakening the measured effects of local conditions.

Effects of nutrients

In a recent global meta-analysis of how dissolved N and P affect river ecosystems, Ardón et al (2021) concluded that uncertainty about nutrient-fish relationships is greater than uncertainty about how nutrients affect periphyton and macroinvertebrates. Relationships between dissolved nutrients and riverine fish populations and/or communities are much less studied than those of periphyton and macroinvertebrates, likely because of the more indirect causal pathways between nutrient enrichment and fish (Figure 2-1) (Ardón et al, 2021).

As is the case for periphyton and macroinvertebrates, some descriptive studies have found that various fish indices and fish species composition metrics correlate with dissolved nutrients among rivers (Justus et al, 2010; Qu et al, 2019; Taylor et al, 2014; Wang et al, 2007; Weigel & Robertson, 2007). Similar to the periphyton- and macroinvertebrate-nutrient correlations documented across rivers, correlational studies of fish-nutrient relationships are often complicated by colinearity among numerous environmental stressors, hence confounding any nutrient gradients. Some studies do a better job in accounting for this colinearity (eg, Taylor et al, 2014) than others (eg, Wang et al, 2007). Correlative studies that have compared the relative sensitivity of periphyton, macroinvertebrates and fish to nutrient gradients among rivers have generally concluded that fish communities are the least sensitive to nutrient enrichment (Justus et al, 2010; Qu et al, 2019). These results contrast with the global meta-analysis of experimental results mentioned above, which concluded that, although uncertainty might be higher, riverine fish are not necessarily less sensitive to nutrient enrichment than periphyton and macroinvertebrates (Ardón et al, 2021).

Nutrient enrichment has been correlated with changes in fish community composition, with some fish species increasing with nutrient enrichment, while other species may decrease in abundance (Qu et al, 2019; Taylor et al, 2014). In some cases, changes in fish community composition as a function of nutrients is nonlinear, with abrupt changes occuring across narrow ranges of nutrient concentrations (Taylor et al, 2014). Certain indices of fish community health have been negatively correlated with dissolved nutrients in rivers and these negative relationships have been used to identify potential ICTs in North America (Wang et al, 2007; Weigel & Robertson, 2007).

It is difficult to ascertain generalisable nutrient-fish relationships from controlled experiments due to the highly context-dependent nature of results. For example, in a long-term nutrient enrichment experiment in an estuarine system, fish growth and biomass increased in the enriched areas for the first ca. six years. Thereafter, nutrient enrichment changed the structure of edge vegetation, destabilising banks, which, in turn, altered fish habitat. This caused fish growth and biomass to decline sharply to below reference (no enrichment) levels after eight years of enrichment (Nelson et al, 2019). This experiment illustrates the complex mechanisms linking nutrient enrichment to fish populations. Similarly complex and context-dependent results have emerged from experiments into whole-river nutrient enrichment. In some cases, nutrient enrichment was detrimental to higher consumers (Davis et al, 2010) while, in others, it increased production of higher consumers (Deegan et al, 1999; Slavik et al, 2004).

Although effects of nutrient enrichment on riverine fishes are not well researched and generally poorly understood, the high dependence of fish populations in New Zealand on macroinvertebrate communities[22] implies the general direction — ie, deterioration or improvement — and magnitude of fish response to enrichment is likely to follow that of macroinvertebrates.

Mediating factors

Dissolved nutrients affect fishes through their effects on periphyton and microbes, then macroinvertebrates (Figure 2-1). It follows that the factors mediating nutrient-periphyton, nutrient-microbe and nutrient-macroinvertebrate relationships (Section 2.1.3) will also mediate nutrient-fish relationships. A further factor that will also mediate nutrient-fish relationships is called *Mass effects*.

Fish are the most mobile aquatic organisms within riverine food webs. They move among rivers and different ecosystems as they mature, and their reproductive and feeding behaviour can result in them moving among rivers and habitats on scales of days to months. These movements mean the

[22] All of Aotearoa’s freshwater fishes are invertivores, either wholly or in part.

fish sampled at a location have not necessarily been at that location throughout their development. It follows that local fish assemblage structure is the result of both local conditions and conditions elsewhere, from which fish have moved. Such effects are referred to as *mass effects* by population biologists. Mass effects mediate local environment-assemblage relationships, by weakening the measured effects of local conditions. Mass effects will be magnified when sampled rivers are adjacent
to source populations (populations from which fish have dispersed) from rivers with different environmental conditions.

2.2.4. Dissolved oxygen

Key messages

Minimum levels of dissolved oxygen (DO) are needed to sustain the life-supporting capacity of rivers. DO concentrations are controlled by the balance between DO exchange between the air and the water (re-aeration), DO released during photosynthesis by algae and macrophytes, DO consumed by heterotrophic microbes during the breakdown of organic matter, and DO consumed by primary and secondary producers during respiration.

The different sources and sinks of DO may lead to diurnal fluctuations in rivers, with maxima in the late afternoon and minima just before dawn. Continuously logged DO (eg, 15-minute intervals) can therefore be represented by various metrics, including daily range (termed diel DO flux), mean or minimum.

Descriptive/correlational studies

Few studies have focused on identifying correlations between DO and nutrient concentrations, and mostly these report relationships with diel DO concentration fluctuations. While correlations between diel DO flux and benthic chlorophyll *a* were identified in some cases, the relationships contained high levels of variance and were not quantified in a way that would allow derivation of ICTs.

DO concentration metrics can be more closely related to indices of land-use impairment (eg, percentage of catchment under intensive agriculture) than to nutrient concentrations, likely reflecting the influence of multiple consequences of catchment activities on DO dynamics.

Mediating factors

Mediating factors in the relationship between DO and dissolved nutrient concentrations can be thought of as a hierarchy in which the overarching effects of factors at the top override the effects of those further down. Temperature (including season) is the most important mediating factor because oxygen is less soluble in warm water than in cooler water and respiration rate (consuming oxygen) increases with temperature. Therefore, warmer waters are always more susceptible to low DO than cooler waters. Next in the hierarchy come stream hydraulics (ie, water velocity and turbulence); algal and macrophyte growth characteristics (and the factors that in turn control them); and organic matter inputs — although the order of importance may vary across stream types.

This hierarchy may be a useful framework for assessing the potential risk of low DO.

Dissolved oxygen (DO) is a fundamental part of the life-supporting capacity of rivers and requires minimum levels to sustain the river assemblages depicted in Figure 2-1. Therefore, minimum levels of DO are the most critical. The main sources and sinks of DO in rivers (Davies-Colley et al, 2013) are:

- re-aeration (source and sink): atmospheric oxygen is transferred to water and *vice versa* under supersaturation
- photosynthesis (source): plants and algae release oxygen into the water
- respiration (sink): plants, algae and secondary producers consume oxygen from the water
- biochemical oxygen demand (BOD)[23] (sink): oxygen is required by microorganisms as they break down organic matter in the water
- sediment oxygen demand (SOD) (sink): oxygen is required by microorganisms as they break down organic matter in the sediments.

Effects of nutrients

Nutrient enrichment may lead to lower than desirable DO minima through:

1. Accumulation of algae in periphyton (in hard-bottomed streams) or of aquatic plants (in soft-bottomed streams), to biomass levels at which oxygen is consumed by aerobic respiration at a faster rate than it can be replenished. Typically, the most severe effects are at night in the absence of oxygen produced by photosynthesis, leading to maximum DO in the late afternoon and night-time minima just before dawn (eg, Wilcock & Nagels, 2001).
2. The effect of nutrients on heterotrophic microbes through oxygen consumption during breakdown of organic litter and detritus, particularly downstream of point-source inputs to rivers such as from waste-water treatment plants (Davies-Colley et al, 2013).

Because DO concentrations in rivers are often characterised by diurnal fluctuations that are linked to changes in photosynthesis and respiration/oxygen demand over the course of a day, continuously logged DO concentrations provide choices of metrics for comparison with nutrient concentrations. These include mean DO over a defined period and, more commonly, mean diel DO flux calculated as the average of daily maximum minus daily minimum DO concentrations over the period of interest from continuously logged data (typically 15-minute intervals).

Studies that include continuous logging of DO in streams generally focus on calculating measures of ecosystem metabolism (see Section 2.2.5) rather than simpler metrics of DO concentration. When simple measures of DO are included in published studies, relationships with nutrient concentrations are rarely the focus (eg, Waite et al, 2019, treated DO as a stressor rather than a response). These publication patterns translated into few reports of relationships between DO concentration metrics and nutrient concentrations.

Relationships between DO concentration metrics and nutrient concentrations that have been identified may also indicate the importance of mediating factors. For example, Stevenson et al (2012) identified positive correlations between mean DO (and DO variability), and TP concentration (and also periphyton biomass and percentage cover by filamentous algae) in spring, but the correlations were negative in summer. The authors of this study attributed the difference between seasons to the effects of higher discharge in spring with higher DO re-aeration. The DO data were weekly spot measurements randomised over time of day.

[23] Biochemical oxygen demand (BOD) is a laboratory assay that measures the oxygen used by microorganisms (eg, for organic matter breakdown) and inorganic matter oxidation in a water sample over a specified time.

Most predictive relationships between nutrient concentrations (or related variables) and DO have been with diel DO flux. The outcome of a study aiming to identify ICTs using relationships between nutrient concentrations (DIN, TN and TP), benthic chlorophyll *a* and DO flux (Miltner, 2010) was that while benthic chlorophyll *a* explained only 7 percent of the variance in DO flux, there was a “change-point” between diel DO flux and benthic chlorophyll *a* at 182 mg m^{-2}. The study suggested that diel DO flux > 6 mg/L carried a significant risk of stream health “impairment”. The direct relationship between nutrient concentrations and DO was not investigated. Over shorter timescales, the diurnal cycling of DO in streams can lead to diurnal fluctuations in other elements. In particular, NO_3-N concentrations can be related to DO through daily cycles of nitrification and de-nitrification and other processes (Harrison et al, 2005). Note these associations are unlikely to be relevant to ICTs based on relationships with nutrient concentrations over much longer time scales.

Other studies have reported positive correlations between diel DO flux and various measure of algal abundance and nutrient concentrations but have not quantified those relationships (Heiskary & Bouchard Jr, 2015; Morgan et al, 2006). Graham and Franklin (2017) detected an association (using a multivariate analysis) between daily minimum DO and chlorophyll *a* in summer/autumn, using individual chlorophyll *a* observations amalgamated across six sites in the Manawatū-Whanganui region, although maximum daily temperature accounted for most of the variance in DO.

Experimental studies have been more successful than field studies in demonstrating links between DO metrics and nutrient enrichment (eg, Suplee et al, 2019). However, the ability of small-scale experimental enrichment to reflect ecosystem-scale responses is limited (Ardón et al, 2021).

DO concentration metrics can be more closely related to indices of land-use intensification (such as the percentage of catchment under agriculture) than to nutrient concentrations (Feio et al, 2010; Justus et al, 2019), which may reflect the influence of multiple consequences of catchment activities on DO dynamics (see Section 2.2.5 for more details).

While the development of algal proliferations in response to nutrient enrichment can lead to low night-time DO (Suplee et al, 2009), nutrient enrichment also increases the DO-depleting effect of heterotrophic microbes, which can be just as significant (Munn et al, 2020).

Focusing on the autotrophic production versus enrichment response as a driver of stream degradation and low DO may lead to oversimplification of the issues, as illustrated by the case of increased cover by algal mats in spring-fed, macrophyte-dominated systems in Florida over several decades. The algal mats were attributed to increasing DIN concentrations over the same period. However, a more plausible explanation for algal proliferations was shown to be declining DO over a 30-year period (likely reflecting natural trends in groundwater age), which released the algal mats from macroinvertebrate grazing pressure (Heffernan et al, 2010). This case illustrates the importance of keeping an open mind to alternative cause and effect pathways in relation to DO and nutrient concentrations.

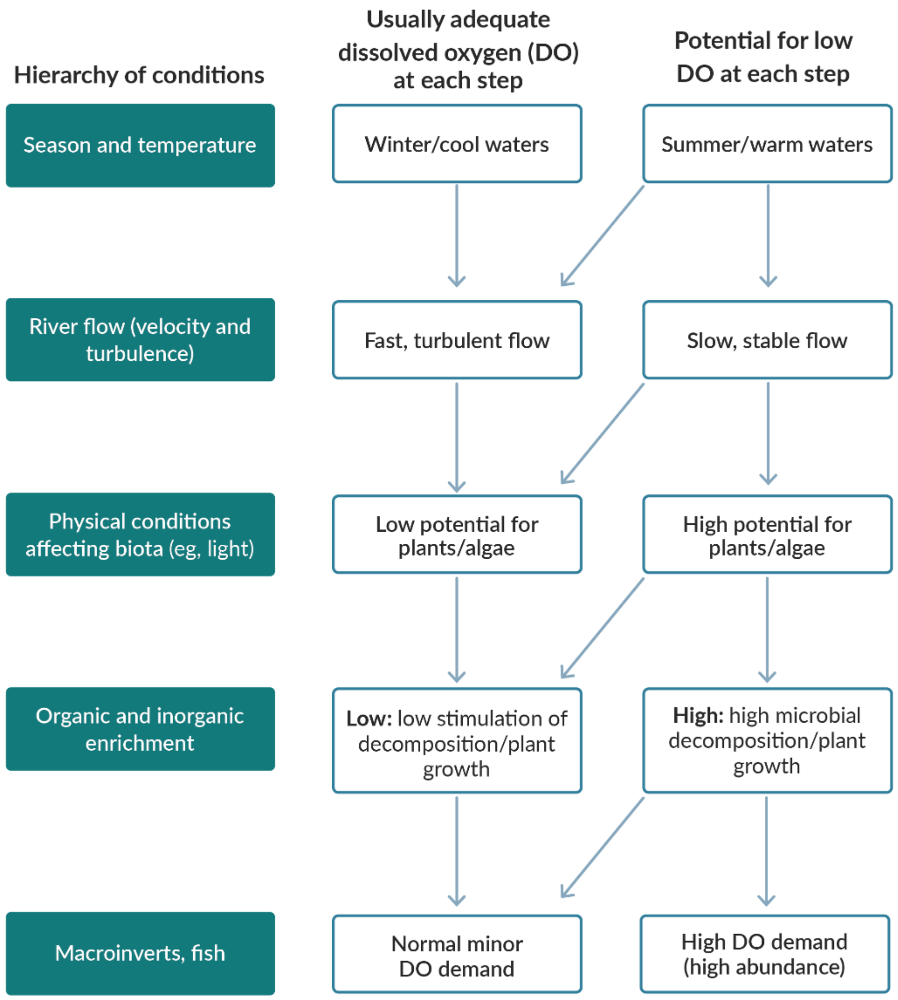
Mediating factors

The general lack of straightforward relationships (such as simple correlations) between nutrient concentrations and DO concentrations (summarised in various metrics) identified from the literature can be explained by the high variability in sources and sinks for DO in fresh waters. The rates of DO added or removed by sources and sinks are themselves influenced by external factors that mediate the indirect response of DO to dissolved nutrient concentrations.

1. *Temperature (including season).* Water temperature is a critical mediating factor in nutrient-DO relationships because oxygen is less soluble in warm water than in cooler water. All other factors being equal, the effect of temperature on DO solubility means that warmer waters (eg, shallower waters at lower altitudes in summer) are more susceptible to harmful DO minima than cooler waters. In addition, instream respiration rates increase as water temperature rises, consuming oxygen and further reducing DO.
2. *Stream hydraulics*. The rate of transfer of oxygen to the water column from the air (ie, re-aeration) is strongly controlled by stream flow, and is a function of water depth and velocity, stream gradient and bed roughness (Young & Huryn, 1999). In slow-flowing, low-gradient streams, low re-aeration may set up the potential for low night-time DO; conversely, high re-aeration means that steeply sloping streams would rarely experience DO low enough to be problematic (Garvey et al, 2007).
3. *Algal growth and biomass.* All of the factors that moderate relationships between nutrients and benthic algal biomass in hard-bottomed streams also affect stream DO through their effect on the amount of algal material available for photosynthesis (ie, light, temperature, nutrient supply and disturbances, see Section 2.2).
4. *Macrophyte growth and biomass.* In soft-bottomed waterways, abundant macrophytes can drive large swings in diel DO concentrations. The effects on DO may depend on growth form (eg, floating versus rooted macrophytes (Wilcock & Nagels, 2001)). Thus, the suite of conditions that drive macrophyte community composition (Riis & Biggs, 2001) also indirectly influence DO response.
5. *Organic inputs to streams.* Both biochemical and sediment oxygen demand are driven by the breakdown of organic matter by heterotrophic microorganisms, which is enhanced by nutrient enrichment (Tank et al, 2010)). BOD may be high downstream of wastewater treatment plant discharges, but such oxygen demand in most streams is likely to be predominantly from SOD, from the processing of fine sediments (eg, from agricultural runoff (Feijo-Lima et al, 2018)), and coarse organic matter such as leaves, wood and plant detritus (eg, from floating macrophytes (Wilcock & Nagels, 2001)).
6. *Groundwater effects.* Groundwater inputs to lowland streams can affect DO measurements because DO in groundwater can be substantially lower than that in the surface water it discharges to (McCutchan et al, 2002).
7. *Interactions.* A nice example of how many of the factors above interact to determine a seasonal pattern of DO fluctuations was presented by Halliday et al (2016). In that case, the primary driver was algal growth driven by nutrient enrichment as water temperatures increased during spring, but suppressed by riparian shading during summer, and interrupted by occasional floods.

The factors that control DO concentrations in streams, and therefore moderate relationships with nutrient concentrations, can be thought of as a hierarchy of effects in which the overarching effects of factors at the top override the effects of those farther down in the hierarchy (Garvey et al, 2007). This hierarchy may be a useful framework for assessing the potential risk of low DO, although it may need re-interpretation for different types of streams (ie, the order of importance of factors may vary). The thresholds separating low and high risk at each level will need to be determined regionally (Figure 2-4).

Figure 2-4: Suggested hierarchy of environmental drivers of low DO concentrations in rivers



In this system, the hierarchy places temperature as the top controller of DO. Low temperature rivers will rarely experience low enough DO to be a problem. The risk of low DO declines at each successive step in the 'Usually adequate' column but increases at each step in the 'Potential for low' column. Thresholds separating the two columns will need to be defined at a regional or river scale. Adapted from figure 9 in Garvey et al (2009).

2.2.5. Ecosystem metabolism

Key messages

Ecosystem metabolism is the combination of gross primary productivity (GPP, reflecting release of oxygen during photosynthesis in algae and plants) and ecosystem respiration (ER, reflecting uptake of oxygen by all aerobic stream biota during respiration and decomposition).

Metabolism can be estimated from continuously logged, dissolved oxygen concentrations (along with temperature, stream width and light data) in a stream reach. Nutrient enrichment affects the rate of both GPP (through stimulation of algal growth) and ER (via nutrient effects on all food web constituents).

Descriptive/correlational studies

While the overall response of metabolism to nutrient enrichment (both N and P) is positive, responses reported from individual studies vary widely and depend on context. Most studies reviewed reported only weak correlations between either GPP or ER and nutrient concentrations across sites.

Mediating factors

The factors that mediate relationships between nutrient concentrations and GPP or ER reflect those that mediate nutrient – DO relationships, but differ between the two elements of metabolism. Light and flow regime strongly control GPP and moderate the relationship with nutrients. ER is most strongly influenced by organic matter inputs to a stream reach. Water temperature affects both processes, and, in combination with other factors including light and flow, drives seasonal patterns in metabolism.

Ecosystem metabolism in streams can be estimated from continuous (eg, 15-minute intervals) data on DO[24] over at least a diel cycle in a stream reach (ideally multiple cycles), and refers to the combination of:

- gross primary production (GPP) — the rate at which inorganic carbon is converted to organic material (making it available to the food web) through photosynthesis in algae and plants
- ecosystem respiration (ER) — the rate at which all stream biota – including heterotrophic bacteria, algae, aquatic plants, micro invertebrates, macroinvertebrates and fish – metabolise carbon (ie, deplete oxygen and release carbon dioxide).

GPP and ER together give an estimate of net community primary productivity, which indicates whether a system is accumulating carbon (ie, positive, net *autotrophic* — higher GPP than ER) or depleting carbon (ie, negative, net *heterotrophic* — lower GPP than ER) at the time of the measurement. Both absolute measures of GPP and ER and the ratio between the two can help to inform the state of ecosystem health. Collecting the data required to estimate both ER and GPP in streams has become considerably more straightforward over recent decades, as automatic *in situ* logging devices have become more available, cost efficient and reliable (Rode et al, 2016).

Effects of nutrients

Nutrient enrichment is linked to metabolism via the same pathways that influence DO concentrations, since metabolism is determined from DO measurement. Nutrients influence GPP through stimulating primary production (eg, algal growth) when a nutrient was previously limiting growth, and ER through indirect or direct effects of nutrient concentrations on all constituents of the

[24] Other data required to complete the calculations include records of water temperature, water depth and light.

food web (ie, primary producers, heterotrophic microbes, macroinvertebrates and fish) (Figure 2-1). Ardón et al (2021) reported consistent positive responses to nutrient enrichment by both primary production (as GPP) and "integrated ecosystem responses" (eg, whole-stream respiration), although mostly based on the results of small-scale experiments or larger experiments testing a small number of levels of nutrient enrichment. Across multiple large-scale experimental studies, metabolism metrics responded more strongly to N and P enrichment together than to either N or P alone (Ardón et al, 2021).

The outcomes from individual survey-based and experimental studies that have explored relationships between metabolism (GPP and ER) and nutrient concentrations across multiple sites have been varied. Mulholland et al (2001) identified a strong positive correlation ($R^2 = 0.56$) between DRP concentration and ER across eight streams in the USA. TN concentration explained 59 percent of the variance in GPP (positive correlation) across 25 cobble-bedded streams in Queensland (Fellows et al, 2006). In New Zealand, modelled N concentration was positively correlated with both GPP ($R^2 = 0.26$) and ER ($R^2 = 0.30$) across 84 stream sites (Clapcott et al, 2010). Data from 15 river sites in the Tasman region indicated a positive association between DIN and DRP and ER across sites (although not quantified) (Young & Collier, 2009).

On the other hand, a common finding has been that the percentage of GPP or ER explained by nutrient concentrations across sites is low (eg, Alberts et al, 2017; Burrell et al, 2014; Clapcott & Doehring, 2014; Hall et al, 2016; Izagirre et al, 2008; Mejia et al, 2019; Munn et al, 2020; Yates et al, 2013). In these cases, other variables such as turbidity, flow, shade, catchment land-cover metrics, or chlorophyll *a*, were better predictors of metabolism than nutrient concentrations, although such relationships in isolation tell us little about the underlying drivers of metabolism. Responses are not necessarily linear because ER can be low in both cool, low-nutrient streams and in streams in highly modified landscapes (Young & Collier, 2009).

Negative relationships between metabolism and nutrient concentrations are occasionally reported. For example, significant relationships between GPP and TP concentrations in two of four small regional datasets in the USA ($n < 9$) were negative (Frankforter et al, 2010). The explanation was that suspended sediment (which was correlated with TP) reduced light at the streambed, which limited GPP. This study also hinted at regional differences because no significant correlations were detected between any of the explanatory variables and GPP or ER when data from the four regions were combined. In a study of 21 streams in Canterbury, New Zealand, the relationships between DRP and both GPP and ER were negative (although explaining < 10 percent of the variance), but corresponding relationships with nitrate-N were positive (Burrell et al, 2014).

In all the above examples, GPP and ER were measured using the whole-stream metabolism method (Young et al, 2016) (see Section 2.3.5 below). Metabolism can also be determined on a smaller scale based on dissolved oxygen measurements in a sealed chamber in both light and darkness (Acuna et al, 2008; Tank et al, 2010). Relationships between nutrient concentrations and metabolism using the latter technique might be expected to be stronger (because the total contribution to ER from sediments may not be included). However, reported correlations with nutrient concentrations are still typically weak or absent. For example, across five streams and four seasons, background nitrate-N (0.170 to 4.3 mg/L) explained only 13 percent of the variance in GPP measured using the chamber method (Reisinger et al, 2016). ER measured in small chambers was positively correlated across 37 streams in Portugal with ammonium concentrations, and a range of indices of land-use intensity, but no correlations with DIN or DRP were reported (Feio et al, 2010).

Links between nutrient concentrations and ecosystem metabolism also depend on the range of nutrient concentrations being considered, partly reflecting that primary production (represented by benthic chlorophyll *a*) reaches an asymptote at high concentrations (see Section 2.2). As an example, Gucker et al (2006) found only minor and seasonal differences in ER upstream and downstream of two wastewater treatment plant (WWTP) discharges despite increases in TN and TP downstream. The small responses were attributed to existing enrichment upstream. In another case, declining ER, but not GPP, over time downstream of a WWTP discharge was explained by improvements in treatment by removal of organic material from the discharge. Concentrations of nitrate-N and DRP also declined but the final concentrations were still above those observed to saturate periphyton biomass so that there was no effect on primary production (Uehlinger & Brock, 2005). In contrast, nutrient concentrations in the low (ie, growth limiting) range can be strong predictors of GPP across streams (Myrstener et al, 2021).

The high variability in nutrient concentration-metabolism relationships discussed above partly reflects that nutrient *concentration* does not necessarily indicate biological demand. Consistent with this, several studies have shown correlations between metabolism metrics and nutrient *uptake* metrics (eg, (Hoellein et al, 2009; Reisinger et al, 2015),[25] and examples in Tank et al (2010), rather than concentrations. The variability in correlations may also reflect differences in the study design (eg, duration of measurement of metabolism and number of sites samples). The optimal duration for metabolism measurements was explored by Munn et al (2020) who recommended measurements of metabolism over at least 14 days and sample sizes (number of sites) > 30 for establishing relationships between metabolism and biological responses.

There may also be a disconnect between water-column, nutrient concentrations and ER in particular, because metabolism in streams occurs mainly at the bed and in the hyporheic zone. Recycling of nutrients between sediment and biota can occur, driving biological activity independently of the nutrient supply in the overlying water (Hoellein et al, 2013).

Many of the published studies on metabolism across multiple sites highlight the typical state of many streams is heterotrophic, ie, net ecosystem metabolism is negative, which indicates that stream metabolism is commonly dominated (to varying degrees) by respiration in the heterotrophic compartment rather than by primary productivity (eg, Frankforter et al, 2010; Hoellein et al, 2013; Young & Collier, 2009; Izagirre et al, 2008; Dodds, 2006). Net autotrophy is generally associated with moderately impacted streams in which enrichment enhanced GPP (ie, algal growth; Alberts et al, 2017; Young & Collier, 2009). While the overall magnitude of responses to nutrient enrichment by heterotrophic and autotrophic ecosystem constituents may be generally similar (Ardón et al, 2021), mediating factors (see below) affect the relationships differently, so that GPP and ER are rarely correlated across sites (but see Mejia et al, 2019).

Mediating factors

A recent review (Bernhardt et al, 2018) set out current understanding of the drivers of, and constraints on, river metabolism, focusing on the idea that every river has a metabolic regime that varies over time. The review emphasised the roles of light regime, temperature regime and flow regime as drivers of ecosystem metabolism rates, as did previous reviews (eg, Young et al, 2009).

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[25] Water column metabolism estimated in closed bottles.

Ultimately, GPP is related to nutrients via benthic algal biomass (and also macrophyte biomass); and ER is related to nutrients via the varied effects of nutrients on all river biota. Thus, as was clear from the literature review above, the strongest models for predicting either GPP or ER include multiple predictors that may or may not include nutrient concentrations (Bernot et al, 2010). Furthermore, studies of metabolism within single sites over time have highlighted strong connections between both ER and GPP and non-nutrient factors (eg, light, temperature and river flow), and weak or no relationships with N or P (Beaulieu et al, 2013; Houser et al, 2015).

1. *Light.* Light is the primary driver of the photosynthesis that determines GPP in both algae and macrophytes. Shaded stream reaches typically support lower algal biomass and lower GPP than unshaded reaches, and responses to nutrient enrichment are muted in the former (Greenwood & Rosemond, 2005). Annual variation in instream GPP partly depends on whether riparian shading changes over the seasons (deciduous vs evergreen). Thus, longer-term changes in GPP may be driven by changes in shading by riparian vegetation as a part of land-use change.

2. *Temperature (including season).* Water temperature controls the rate of all biological processes and therefore the thermal regime of a river determines the timing of minimum and maximum GPP and ER (in the context of other constraints on, and drivers of, the two processes including nutrient enrichment). The thermal regime of a river has an overriding seasonal pattern reflecting climate, but is modified by factors such as shade, groundwater inputs (Burkholder et al, 2008) and the presence of lakes or dams (Young et al, 2004), each of which also affects rates of metabolism.

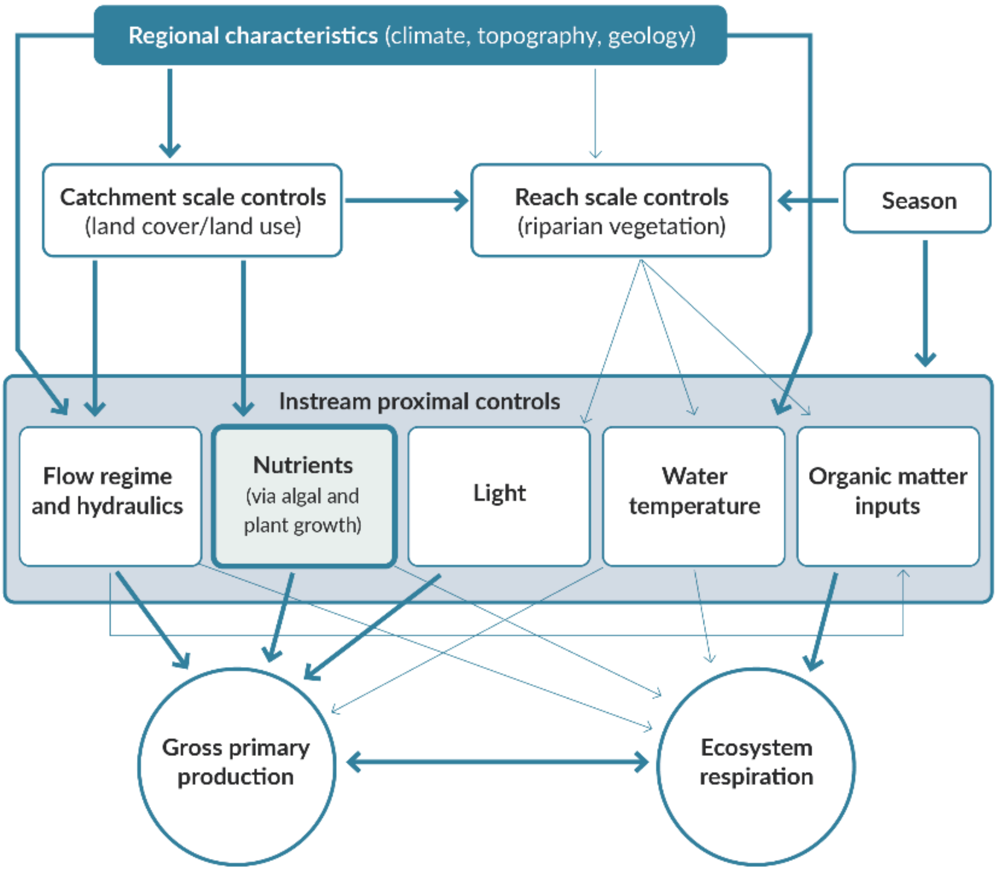
3. *Stream flow and flood frequency (disturbances)*. The removal of periphyton (ie, algae and heterotrophic microbes) biomass during flow disturbances suppresses both GPP and ER. Recovery from floods is a function of the combined drivers of metabolism, with faster recovery in nutrient-enriched conditions (Lohman et al, 1992). Flow regime changes that reduce floods can affect metabolism, both through providing longer accrual periods for algal accumulation (ie, GPP) and for detritus accumulation, which increases ER.

4. *Organic matter supply*. Organic matter inputs to streams drive the ER side of the metabolism budget (Tank et al, 2010). While such allochthonous inputs (from leaves to dissolved organic compounds leached from soils) are largely independent of nutrient concentrations, the rate at which they are broken down (contributing to ER) is usually increased by nutrient enrichment (Tank et al, 2010).

5. *Land use and land cover*. It was noted above that ecosystem metabolism across sites is sometimes more strongly correlated with land-use indices than with nutrient concentrations. The links between metabolism and land cover are indirect but somewhat consistent: stream reaches in undeveloped catchments tend to be net heterotrophic with high ER driven by microbial breakdown of allochthonous (ie, imported material such as leaves) inputs, and low GPP (suppressed by shade and low nutrient concentrations). In contrast, runoff from developed catchments may transport more fine sediment to streams than from undeveloped catchments (Davis et al). Retained sediments can suppress ER in unshaded streams (Houser et al, 2005), even under nutrient enrichment (McTammany et al, 2007). However, moderate inputs of sediment can be associated with higher GPP (eg, through facilitating nutrient recycling that stimulates algal growth (Wood et al, 2015a).

6. *Interactions.* All of the above factors interact to moderate the effects of nutrient concentrations on ecosystem metabolism in different ways, as illustrated in figure 2-5.

Figure 2-5: Interactions amongst factors understood to influence stream ecosystem metabolism separated into gross primary production (GPP) and ecosystem respiration (ER)



Controls operate on decreasing spatial scales going from the top to the bottom of the figure. Thick arrows indicate the main controlling factors and thin arrows show minor factors. Nutrients can directly influence both GPP and ER and both relationships are modified by other factors. Adapted from figure 1 in Alberts et al (2017).

2.2.6. Tables of dissolved nutrient thresholds from literature

In this section, examples are presented of nutrient thresholds required to achieve various periphyton (algae/Chl *a* only) and macroinvertebrate target attribute states. The thresholds are from New Zealand studies. Many more nutrient thresholds for various periphyton and macroinvertebrate targets from around the world are presented in appendix B.

There are no existing ICTs for the fish, DO or ecosystem metabolism attributes.

Table 2-1: Dissolved nutrient thresholds to achieve various periphyton (as chlorophyll *a*) target attribute states in New Zealand.

Concentrations are medians calculated from monthly time series. Blank cells mean that there are no criteria for that nutrient.

Region	TN or DIN* criteria (mg/m^3)	TP or DRP* criteria (mg/m^3)	Target or state	Reference and notes
New Zealand	<20*	<1*	50 mg/m^2 Chl *a*, max, 20 days of accrual	(Biggs, 2000) Refer to New Zealand periphyton guideline for complete table with range of days of accrual. The ICTs apply to unshaded hill- or mountain-fed rivers with no lakes upstream.
	<295*	<26*	120–200 mg/m^2 Chl *a*, max, 20 days of accrual	
	<10*	<1*	50 mg/m^2 Chl *a*, max, 50 days of accrual	
	<19*	<1.7*	120–200 mg/m^2 Chl *a*, max, 50 days of accrual	
New Zealand	9–120	0.1–8.2*	50 mg/m^2 Chl *a*, 92nd percentile	(Snelder et al, 2019) ICTs allowing exceedance of NPS-FM 2020 periphyton Chl *a* targets at no more than 20% of sites (20% spatial exceedance). Range across 21 source-of-flow REC classes. Based on cover data rivers in the National River Water Quality Monitoring Network, with cover converted to chlorophyll *a* using an independent relationship.
	47–607	0.2–114*	120 mg/m^2 Chl *a*, 92nd percentile	
	113–1440	1.5–289*	200 mg/m^2 Chl *a*, 92nd percentile	
New Zealand	2–146 1–83*	0–9 0–3*	50 mg/m^2 Chl *a*, 92nd percentile	(Snelder et al, 2022) ICTs allowing exceedance of NPS-FM 2020 periphyton Chl *a* targets at no more than 20% of sites (20% under protection risk); range across 21 source of flow REC classes (unshaded sites only). Concentrations at the top of the range were uncertain and are shown as approximate concentrations beyond which chlorophyll a did not respond to nutrient concentration increases.
	164–~1000 76–>1000*	6–~50 1–~25*	120 mg/m^2 Chl *a*, 92nd percentile	
	2154– ~1000 1678– ~1000*	120– ~50 18– ~25*	200 mg/m^2 Chl *a*, 92nd percentile	
Canterbury	20–500*		200 mg/m^2 Chl *a*, 92nd percentile	(Kilroy et al, 2017) Hill-fed rivers only, ICTs depending on conductivity and accrual period (positive effect on chlorophyll a) and % fine substrate (negative effect). GLM prediction so 50% under-protection risk. (See Snelder et al, 2022)

Region	TN or DIN* criteria (mg/m³)	TP or DRP* criteria (mg/m³)	Target or state	Reference and notes
Manawatū – Whanganui	120–240*		50 mg/m² Chl *a*, 92nd percentile	(Kilroy, 2019) Across selected sites in MW region where chlorophyll *a* met or exceeded One Plan targets, but DIN did not happen to be consistent with existing One Plan chlorophyll *a* targets.
	680–1080*		120 mg/m² Chl *a*, 92nd percentile	
Northland	<10–580	<4–21*	50 mg/m² Chl *a*, 92nd percentile	(Kilroy & Stoffels, 2019) Criteria derived for sites with particular characteristics (eg, REC classes), using multiple regression models. Criteria for 50% under-protection risk (2.5% risk also calculated). Values with > mean that the chlorophyll *a* threshold could not be achieved by increasing DIN or DRP.
	405–>1000	7–>50*	120 mg/m² Chl *a*, 92nd percentile	
	805– >1000	43–>50*	200 mg/m² Chl *a*, 92nd percentile	
Bay of Plenty	3–85		50 mg/m² Chl *a*, 92nd percentile	(Kilroy et al, 2020) TN criteria determined using a single model for 7 REC classes at Geology level, 20% under-protection risk (10%, 30% and 50% also provided). > means that the chlorophyll *a* threshold could only be achieved by increasing TN to above a "saturating" concentration, reflecting high uncertainty in the model.
	33–>1500		120 mg/m² Chl *a*, 92nd percentile	
	1371–>1500		200 mg/m² Chl *a*, 92nd percentile	

Table 2-2: Dissolved nutrient thresholds to achieve various macroinvertebrate targets in New Zealand

Concentrations are medians calculated from monthly time series.
Blank cells mean that there are no criteria for that nutrient.

Region	TN or DIN* criteria (mg/m³)	TP or DRP* criteria (mg/m³)	Target or state	Reference and notes
Manawatū-Whanganui	<500		Point at which MCI, %EPT taxa and EPT richness ceased to respond	(Wagenhoff & Liess et al, 2017) Impact cessation threshold for community metrics after negative response.
	<500 (150)		Point at which significant macroinvertebrate assemblage turnover	(Wagenhoff et al, 2017) Assemblage threshold range within which significant turnover of multiple taxa occur.

Region	TN or DIN* criteria (mg/m³)	TP or DRP* criteria (mg/m³)	Target or state	Reference and notes
			subsided (and peak of turnover)	
	<500		Point at which multiple metrics of food web function had ceased to respond or shown inflection	(Canning & Death, 2021) Study of food web stability and function, including community respiration via energy flows in riverine networks.
Southland	144*		Inflection point of subsidy-stress relationship for %EPT abundance	(Wagenhoff et al, 2011) Subsidy-stress relationship only at lower levels of deposited fine sediment.
New Zealand	600* (QMCI) 1100* (MCI) 1100* (ASPM)	20* (QMCI) 28* (MCI and ASPM)	Nutrient criteria to support NPS-FM national bottom-line targets for MCI, QMCI and ASPM	(Canning et al, 2021) Minimisation-of-mismatch analysis of national dataset. Criteria shown are medians (approximate values) based on measured nutrient data. Data accompanying the paper provides criteria for other target attribute states.
New Zealand	10*, 330*, 1470*	1*, 9*, 28*	Nutrient criteria to support NPS-FM A/B, B/C and C/D (bottom-line) macroinvertebrate attribute thresholds, respectively	(Canning, 2020) Based on linear regression relationships for MCI, QMCI and ASPM in national dataset.
New Zealand, mesocosm experiment	728*	70*	Inflection point of subsidy-stress relationship for EPT metrics and for total taxon richness	(Wagenhoff et al, 2012) Threshold values treated with caution due to temporal and spatial limitations of such experiments (experimental values expected to be higher than in real streams).

2.3. Nutrient-affected NPS-FM attributes

This section presents a brief discussion of how the statistical properties of each nutrient-affected attribute (Table 1-1) might affect its spatial and temporal sensitivity to variation in dissolved nutrients. A knowledge of the statistical properties and relative sensitivity of attributes will facilitate decisions concerning how to:

- best analyse data to determine ICT
- prioritise resources for (a) monitoring attributes (eg, which attributes are likely to be the most sensitive indicators of nutrient management actions) and (b) developing new, attribute-specific models, assuming resources for NPS-FM implementation will be limited.

The basic statistical properties of each nutrient-affected attribute are described. A qualitative evaluation of four factors affecting the sensitivity of individual attributes and/or the relative sensitivity of attributes to nutrient enrichment are:

1. **Sensitivity of attribute to changes in nutrient concentrations through time**. This topic covers possible time lags in the response of attributes to nutrient enrichment and to nutrient reductions. It also discusses how *inert* attribute scores may be to nutrient enrichment through time. Attribute inertia is important since, for any fixed level of nutrient enrichment, the statistical properties of some attributes may increase their sensitivity, while the properties of others may result in them not responding to nutrient enrichment.
2. **Spatial domain of attribute-nutrient relationships**. The topic covers the degree to which attribute state reflects local versus regional environmental conditions, that is, how reach- or catchment-specific, nutrient-attribute relationships may be.
3. **The extent to which mediating factors will lower the sensitivity of attributes to nutrient enrichment**. This looks at how sensitive is the attribute to nutrient enrichment relative to that attribute's sensitivity to other natural and anthropogenic factors.
4. **Whether non-NPS-FM metrics may respond to nutrient concentrations lower than those inducing a response in the attribute and, if so, consequences for relative sensitivity of attributes.** For example, is it possible that periphyton species composition (not an attribute) responds to nutrient enrichment before Chl *a* (the NPS-FM attribute)? If so, what does that mean for the sensitivity of other attributes (eg, QMCI) relative to the periphyton attribute?

2.3.1. Periphyton

The NPS-FM periphyton attribute is measured as concentration of chlorophyll *a* (mg/m^2) (hereafter Chl *a*), which correlates tightly and positively with algal standing crop biomass. Following the National Environmental Monitoring Standards (NEMS), Chl *a* is monitored to yield a monthly time series at each site. To evaluate the state of periphyton at a site against the NOF, the 92nd or 83rd percentile of Chl *a* is estimated using at least three years of a site's monthly time series.[26] The percentiles are equivalent to an average of, respectively, one or two exceedances of a NOF *band threshold* per year at, respectively, *default* and *productive* sites. Thresholds separating NPS-FM bands are:

[26] It is assumed that periphyton surveys that provide the data for grading river sites against the NOF are carried out using standard methodology as described in the current National Environmental Standard (NEMS) for periphyton (Periphyton » National Environmental Monitoring Standards (NEMS)), or the preceding publications on which the NEMS was based (eg, Biggs and Kilroy 2000).

- A-B, 50 mg Chl *a* /m^2
- B-C, 120 mg Chl *a* /m^2
- C-D (*National Bottomline*), 200 mg Chl *a* /m^2.

The thresholds were set based on published work on perceptions of what constitutes "nuisance" levels of algae in rivers, and on reported negative relationships between Chl *a* and macroinvertebrate indices, as set out by Snelder et al (2013). The relationships were derived from those described in previous guidelines (Biggs, 2000) and in more recent work (Matheson et al, 2012).

The periphyton attribute was developed recognising that:

- short periods of high periphyton biomass may have minimal ecosystem effects (hence the exceedance allowance)
- periphyton can vary a lot from year to year (hence the minimum of three years of data)
- some rivers are naturally productive (hence the allowance of two exceedances per year on average at these productive sites compared to one exceedance at default sites)
- periphyton Chl *a* in rivers usually conforms to an exponential distribution over time (ie, most observations have relatively low values and just a few have high values; Snelder et al, 2014). Therefore, the 92nd (or 83rd) percentile(s) are a good representation of *peak biomass*, or the site's carrying capacity for periphyton biomass (see above) at most sites. (There are exceptions, for example, some lake-fed rivers have consistently high biomass over time, and the exponential distribution of biomass does not apply.)

Sensitivity of the attribute to changes in nutrients through time

During the nutrient-limited growth phase (Figure 2-2) and assuming unrestricted accrual (ie, no scouring floods) Chl *a* is expected to respond to changes in DIN and DRP within days/weeks (Francoeur et al, 1999). If periphyton growth has reached the point where it is limited by non-nutrient resources (asymptotic phase in Figure 2-2), then Chl *a* may be relatively insensitive to changes in DIN and DRP.

In contrast to Chl *a per se*, the periphyton attribute (ie, the 92nd or 83rd percentile of the three-year Chl *a* series) is relatively insensitive to changes in nutrient concentrations over short to medium time periods (months to a few years). Using state of the environment (SoE) data you can calculate periphyton attribute values (92nd and 83rd percentiles) over numerous three-year periods within sites, then compare variation in those attribute values to variation in mean Chl *a* values calculated over the same periods. Such calculations show that a 92nd percentile of 220 mg m^{-2} (for example) corresponds to mean Chl *a* concentrations anywhere between ca. 40 and 200 mg m^{-2}. Given the exponential nature of the frequency distribution of Chl *a* values through time (Snelder et al, 2014) (ie, most observations have relatively low values and just a few have high values) within a site, much more temporal variability will be observed in the responses of mean Chl *a* values to DIN and DRP than those of NPS-FM periphyton attribute value.

Given the periphyton attribute at time *t* is estimated using a three-year time series preceding time *t*, you can expect a lag in the response of the NPS-FM periphyton attribute to increases and decreases in DIN/DRP.

Spatial domain of attribute-nutrient relationship

Periphyton assemblages are attached to benthic substrates and are largely immobile. Consequently, the periphyton attribute is a good indicator of DIN/DRP enrichment upstream of a site and is relatively uninfluenced by nutrient concentrations at sites downstream or in adjacent aquatic habitats.

Will mediating factors lower sensitivity of the attribute to nutrients?

The mediating factors of lowered light, increased grazer biomass and lower temperatures (among other mediating factors; see Section 2.2) will reduce the sensitivity of the periphyton attribute to changes in DIN and DRP concentrations.

Non-nutrient anthropogenic stressors will likely lower the sensitivity of the periphyton attribute to DIN or DRP. In particular, increases in suspended and deposited fine sediment can lead to lower periphyton biomass through reducing light availability and increased sloughing.

Are metrics based on periphyton taxonomic composition more sensitive to DIN/DRP than the NPS-FM attribute, and, if so, how would that affect the relative sensitivity of other attributes?

There is some evidence indicating that metrics based on periphyton assemblage composition and nutritional status may significantly respond to an increase in DIN and DRP when Chl *a* and, in particular, the NPS-FM periphyton attribute, does not (see Section 2.2). Changes in periphyton assemblage and nutritional composition may affect macroinvertebrate communities (see Section 2.2), creating the potential for macroinvertebrate NPS-FM attributes to be more sensitive to variation in DIN and DRP than the periphyton attribute.

2.3.2. Macroinvertebrates: MCI, QMCI and ASPM

Based on the NOF, the Macroinvertebrate Community Index (MCI), Quantitative MCI (QMCI) and the Average Score Per Metric (ASPM) are calculated from annual samples taken between December and March (inclusive), with current states calculated as medians over the preceding five years.

New Zealand's MCI and QMCI are calculated from taxon-specific values of tolerance to organic pollution and nutrient enrichment for all taxa in the community (Stark, 1985; Stark & Maxted, 2007) following the method in Stark and Maxted (2007). The MCI score is calculated from presence-absence data which is determined from a composite sample collected with a hand- or kick-net (semi-quantitative collection method). The QMCI is calculated from count data, with either fixed counts of at least 200 individuals or full counts, determined from a composite sample collected with a Surber sampler (fully quantitative collection method, yielding densities per unit area) or, often in state of the environment monitoring, a hand-net (ie, semi-quantitative, yielding percentage composition).[27]

[27] Samples collected for calculation of QMCI can always be used to calculate MCI, but the reverse is not always true.

Formulae for the two indices are:

$$MCI = 20\frac{\sum_{i=1}^{S} a_i}{S} \qquad \text{(Eqn 1)}$$

$$QMCI = \sum_{i=1}^{S} \frac{n_i a_i}{N} \qquad \text{(Eqn 2)}$$

where S = the total number of scoring taxa in the sample, a_i = the tolerance value for the ith taxon, n_i = the abundance for the ith taxon and N = the total abundance of the scoring taxa for the entire sample. The scalar of 20 (Eqn 1) for the MCI has been added to distinguish between MCI site scores and site scores of the QMCI.

The NPS-FM specifies when taxa tolerance values for soft-bottomed streams rather than the standard taxa tolerance values for hard-bottomed streams should be used, with all tolerance values being used as defined in table A1.1 in Clapcott et al (2017).

The ASPM is calculated from three metrics:

1. the MCI
2. the number of taxa belonging to the insect orders *Ephemeroptera* (mayfly), *Plecoptera* (stonefly) and *Trichoptera* (caddisfly) excluding *Hydroptilidae* (EPT richness)
3. the percentage of EPT individuals within the entire community (%EPT abundance) according to the method of Collier (2008).

The metrics are aggregated by firstly scaling (normalising) the observed site scores by the observed maximum value across a set of sample sites. This results in values between 0 and 1. The mean of these scaled metrics is then used to calculate the ASPM (Collier, 2008). The NPS-FM specifies that the following minima and maxima be used when normalising scores: MCI (0-200), EPT richness (0-29), %EPT abundance (0-100).

Thresholds separating NPS-FM bands are:

- A-B, MCI = 130, QMCI = 6.5, ASPM = 0.6
- B-C, MCI = 110, QMCI = 5.5, ASPM = 0.4
- C-D (*national bottom line*), MCI = 90, QMCI = 4.5, ASPM = 0.3.

The notes in the macroinvertebrate attribute tables related to protocols for collecting macroinvertebrates acknowledge the following (noting that some of these assumptions have been updated during development of the more recent National Environmental Monitoring Standards (NEMS) for Macroinvertebrates[28]):

[28] In addition to the details given in the NPS-FM, refer to standard protocols published in the National Environmental Monitoring Standards (NEMS) for Macroinvertebrates published in draft format in November 2020 (http://www.nems.org.nz), which was based on Stark J, Boothroyd I, Harding J, Maxted J, & Scarsbrook M. (2001). *Protocols for sampling macroinvertebrates in wadeable streams.* New Zealand Macroinvertebrate Working Group Report No 1. Prepared for the Ministry for the Environment, Sustainable Management Fund Project No. 5103.

- The summer period is likely the time of year when DIN/DRP will exert its strongest effects, and therefore samples taken around this time are most likely to detect enrichment effects.
- Restricting sampling to riffles/runs reduces the possibility that inter-river and temporal patterns in attributes are confounded by sampling location within sites. Riffles/runs also support the most diverse macroinvertebrate assemblages, possibly increasing the sensitivity of attribute scores to nutrient enrichment.
- Effects of flow variability on macroinvertebrate attributes are reduced by not sampling the community within a specified period following bed-moving floods.
- Interannual variability in macroinvertebrate attribute values is reduced by calculating current state as a five-year median.
- The macroinvertebrate attributes are broadly applicable across New Zealand as indicators of the effects of nutrient and organic enrichment within each site given that the responses of individual taxa to enrichment are unlikely to vary across the country. Exceptions may be sites where there are naturally few macroinvertebrates present due the hostility of natural environmental factors or they are naturally unproductive (see bullet point below).
- Natural environmental factors also shape macroinvertebrate communities, and that different stream types in their reference condition may therefore have different metric scores which must be taken into account when setting targets for these metrics. The recognition that naturally soft-bottomed streams have generally lower MCI values in their reference condition prompted development of a second set of tolerance values for such streams so as to use the same band thresholds that are defined for hard-bottomed streams (Stark & Maxted, 2007).

Sensitivity of the attribute to changes in nutrients through time

The tolerance values for the MCI and its variants have specifically been developed to respond to increasing nutrient and organic enrichment and most EPT taxa (excluding *Hydroptilidae*) are sensitive to enrichment. The relative sensitivity of MCI, QMCI and ASPM to DIN/DRP is partly determined by how taxa contribute to attribute calculation. MCI and EPT richness — the latter being one of three metrics contributing to the ASPM — require localised extirpation of a taxon before the attribute value changes. By contrast, %EPT abundance or the QMCI only require a change in the relative abundance of taxa. Therefore, QMCI should be the most sensitive to effects of DIN/DRP enrichment, followed by the ASPM and then the MCI. However, these hypotheses concerning the relative sensitivity of the three macroinvertebrate attributes to DIN/DRP have not been scientifically tested.

When compared with fish, macroinvertebrates have relatively short life cycles. Most New Zealand macroinvertebrates are univoltine or plurivoltine (Phillips & Smith, 2018), meaning they complete at least one reproductive cycle per year. Short life-cycle durations of taxa should result in the community responding to nutrients on a scale of one to five years.

Given the macroinvertebrate attributes at time *t* are estimated using five-year time series preceding time *t*, you can expect a lag in the response of the NPS-FM macroinvertebrate attributes to increases and decreases in DIN/DRP.

Spatial domain of attribute-nutrient relationship

The larval stages of aquatic macroinvertebrates can disperse into and out of a river reach by drifting. Thus, the macroinvertebrate community observed within a reach is the product of environmental conditions within that reach, but also environmental conditions upstream and possibly within upstream tributaries as well. Nutrient concentrations at monitoring sites also integrate nutrient concentrations within upstream catchments. Thus, if both DIN/DRP and invertebrate populations are integrating the same part of the FMU, it follows that the influence of larval dispersal on sensitivity of macroinvertebrate attributes is unlikely to be significant.

Will mediating factors lower sensitivity of the attribute to nutrients?

Factors mediating the response of the periphyton attribute to DIN/DRP will also mediate the responses of macroinvertebrate attributes to DIN/DRP.

Will other metrics based on macroinvertebrate assemblages respond to DIN/DRP enrichment before the NPS-FM attributes do, and, if so, how would that affect the relative sensitivity of other attributes?

Given that QMCI and metrics contributing to ASPM integrate changes in taxon-specific relative abundance, these attributes should be sensitive to change in macroinvertebrate assemblage composition. This contrasts with the periphyton attribute, where we are likely to see changes in the periphyton assemblage in response to DIN/DRP before the periphyton attribute itself changes (see Section 1).

2.3.3. Fish: The Index of Biotic Integrity

The Fish Index of Biotic Integrity (FIBI) was developed for application to New Zealand by Joy and Death (2004). The FIBI is an adaptation of similar indices of biotic integrity developed for North America (Angermeier & Karr, 1986; Joy & Death, 2004; Karr, 1981). Unlike the macroinvertebrate indices, the FIBI was not developed as an indicator of organic pollution or dissolved nutrient enrichment, nor does it have a long history of application to freshwater management in New Zealand. Multi-decadal deterioration in national FIBI scores has been related to very coarse land-use classes (Joy et al, 2019), but its use as an indicator of DIN/DRP effects is unclear. Similar FIBIs in North America were originally developed as a coarse index of the effect of “man’s activities” on the state of river ecosystems (Karr, 1981).

The FIBI integrates six metrics calculated from a sample of fish species’ presences/absences:

1. the number of native species
2. the number of native riffle-dwelling species
3. the number of native, benthic, pool-dwelling species
4. the number of native pelagic species
5. the number of native intolerant or sensitive species
6. the proportion of species that were non-native.

For each metric, the observed value is divided by the theoretical maximum value for that site, yielding a proportion. These proportionate values are then assigned a score of 5, 3 or 1 if they lie within the intervals > 0.67, ≤ 0.67 and ≥ 0.33, or < 0.33, respectively.

For metric 6 — the proportion of the sample comprising non-native species — the maximum value is, of course, one. For metrics 1 to 5, maximum values depend on a site's distance from the coast and its elevation. This site-dependence of maximum species richness acknowledges that fish species richness in New Zealand naturally declines with both distance from the coast and elevation. Maximum species richness lines (MSRLs) are regression lines defining the relationships between maximum species richness and either distance from the coast or elevation. The NPS-FM recommends the method of Joy and Death (2004) for application of the FIBI, in which case MSRLs are subjectively estimated by eye. Objective methods of estimating MSRLs (eg, various forms of quantile regression) could be employed to ensure methods and results are transparent and repeatable.

Once the scores have been obtained for each of the six metrics, they are summed to give the FIBI value for a site. If no native fish were found at a site, the FIBI value is zero.

Sensitivity of the attribute to changes in nutrients through time

The sensitivity of the New Zealand FIBI to spatial/temporal gradients in nutrient enrichment has not been investigated. The FIBI will likely be less sensitive to DIN/DRP than the periphyton and macroinvertebrate attributes for the following reasons:

- The FIBI is based on presence-absence data and so only changes value when a fish species is not detected at a site. Assuming the sampling design is sufficient to yield a high probability of detection, a species must be locally extirpated for the FIBI to change. It follows the FIBI will not detect any nutrient-induced reductions in local fish abundance, which may continue for several years before local extirpation. By contrast, the macroinvertebrate attributes and the periphyton attribute detect changes in abundance.
- The macroinvertebrate attributes were explicitly developed to be indicators of organic pollution, part of which is nutrient enrichment. By contrast, the FIBI is a more generic, non-specific indicator of stress on a local fish community (Karr, 1981).

Spatial domain of attribute-nutrient relationship

Fishes are more mobile than periphyton and macroinvertebrates. They also have longer lifespans and, for many of New Zealand's fishes, more complex life histories (eg, diadromy) than periphyton and macroinvertebrates, which generates greater potential to move between rivers/streams/lakes/estuaries throughout their life. Relative to periphyton and macroinvertebrates, increased mobility may result in a higher degree of spatial decoupling of the relationship between DIN or DRP and the FIBI within rivers.

Will mediating factors lower sensitivity of the attribute to nutrients?

Factors mediating the response of the periphyton and macroinvertebrate attributes to DIN/DRP will also mediate the responses of the FIBI to DIN or DRP.

Will fish assemblages respond to DIN/DRP enrichment before the FIBI does, and, if so, how would that affect the relative sensitivity of other attributes?

Fish assemblage composition must change considerably before the FIBI changes value. The FIBI only responds to species extirpation. Although the relative sensitivity of NPS-FM attributes to nutrient enrichment has not been formally analysed, it is likely the periphyton and macroinvertebrate attributes will be more sensitive to changes in DIN or DRP.

2.3.4. Dissolved oxygen

The attribute for dissolved oxygen (DO) included in the first version of the NOF (NPS-FM 2014), and in the subsequent 2017 amendment, was applicable only to rivers below point-source discharges because low DO is commonly associated with wastewater discharges (Davies Colley, 2013; Hamdhani et al, 2020). The original DO attribute is retained in the NPS-FM 2020 (as table 7 in appendix A), along with a second DO attribute requiring an action plan (where objectives are not met), and it applies to all rivers (table 17 in appendix B of the NPS-FM 2020). Bands A to D of both DO attributes were defined by the same metrics and thresholds and have the same narratives.

The attribute was developed by Davies-Colley (2013), who provided background to, and justification for, the attribute metrics and thresholds separating bands A, B, C and D. Attribute states are defined using two metrics (as recommended by Davies-Colley et al, 2013), both measured as the concentration of DO in water (mg/L):

- the seven-day mean minimum of DO (defined as the mean value of seven consecutive daily minimum values)[29]
- the one-day minimum (defined as the lowest daily minimum across the whole summer period).

The metrics are the same as those adopted by the US EPA (1986), recognising that:

- minimum DO occurs at night (reflected by the one-day minimum)
- the minimum DO concentration is important in determining the effect on biota (reflected by the seven-day mean minimum).

The NPS-FM unit is concentration rather than percentage saturation, recognising the effect of water temperature on the solubility of oxygen.

Thresholds separating NPS-FM bands are:

- A-B, seven-day mean minimum, 8 mg/L; one-day minimum, 7.5 mg/L
- B-C, seven-day mean minimum, 7 mg/L; one-day minimum, 5 mg/L
- C-D (*national bottom line*), seven-day mean minimum, 5 mg/L; one-day minimum, 4 mg/L.

[29] Davies-Colley et al (2013) stated: "Seven-day duration alone is insufficient to avoid chronic impacts. It is intended that in any continuous seven-day period throughout the year, this threshold will be met, ie, this is the annual minimum seven-day mean."

The only difference between table 7 and table 17 in appendix A of the NPS-FM 2020 is that table 7 specifies that metrics are calculated using data from the "summer period: 1 November to 30th April", while table 17 does not have this qualifier. In both tables, the second footnote reads: "The one-day minimum is the lowest daily minimum across the whole summer period." Therefore, it is unclear whether the metrics for table 17 are to be calculated using summer data only or not. We assume the latter (ie, data collection all year) but expect that the overall minimum would occur in summer in any case, as would the lowest seven-day mean minima.[30]

Sensitivity of the metrics to changes in nutrient concentrations over time

The metrics report mean minimum DO over a short period (7 days) and absolute minimum value over the summer period (a single measurement with duration equal to the logging interval). Sensitivity of these two metrics to day-to-day variability in measurements is, respectively, moderate and high. This is appropriate for DO because low DO, even for a short time, can have serious detrimental effects on stream ecosystems, with longer-term (chronic) or instantaneous (acute) modes of action. The footnote (in NPS-FM appendix A tables 7 and 17) for the one-day minimum specifies the minimum value across the whole summer period, which implies there must be continuous monitoring between 1 November and 30 April. Thus the one-day metric is a single DO measurement which is sensitive to the combination of conditions in each summer period.

While the DO attributes have high temporal resolution, determining the sensitivity of the DO attributes to DIN and DRP variation requires further investigation and may not be possible to assess.

Spatial domain of attribute-nutrient relationship

Generally, minimum DO reflects ecosystem metabolism unfolding within and upstream of river reaches. It is unlikely to be influenced by processes downstream, nor in adjacent catchments/systems. The DO attributes are, therefore, comparable with periphyton and macroinvertebrate attributes with respect to their spatial domains of influence.

Will mediating factors lower sensitivity of the attribute to nutrients?

Numerous factors will mediate the response of DO to nutrients. Minima of DO are expected to vary from day to day according to temperature, cloud cover, shade, river flow (ie, water velocity and turbulence), organic enrichment of the waterway and abundance of plant growth (including periphyton), as well as nutrient concentrations. The requirement to calculate the metrics over the summer months is appropriate for potentially detecting the worst-case scenario each year. However, observations over many summers may be required to determine nutrient-DO relationships.[31]

It is appropriate to compare summer DO minima and seven-day means with median nutrient concentrations measured across the whole year rather than just over the period of DO measurements, because:

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[30] Note that the Dissolved Oxygen (Rivers) table in STAG (2019) includes a footnote that states: "Objectives apply year-round." This footnote was not transferred to table 17 in the NPS-FM (2020).

[31] Alternatively, trends over time in the DO metrics and nutrient concentrations could be compared using more fine-scaled time-series data. (This was the strategy used by Suplee et al (2019) to detect parallel changes in periphyton biomass and nutrient concentrations in a river.)

- the usual nutrient metric for defining nutrient criteria is an annual or longer-term median value[32]
- summer nutrient concentrations (ie, taken over the period of the DO measurements) may not accurately represent the overall, average nutrient supply to primary production at that site (because of uptake of DIN and DRP by algae in periphyton or phytoplankton in the water column in summer). While DO production will reflect the standing crop of algae (measured as Chl *a*) at a site, Chl *a* is typically more closely related to preceding higher nutrient concentrations (eg, over several months) than to ambient concentrations.[33]

Will any other metrics of DO respond to nutrient concentrations before the NPS-FM attribute metric does?

The literature review (Section 2.2.4) indicated that diel DO flux (maxima minus minima; a range) may be correlated with nutrient and/or chlorophyll *a* concentrations but there was sparse information on relationships between nutrient concentrations and DO minima, or between DO flux and DO minima. Therefore, the relative sensitivity of DO metrics (eg, flux or daily median *versus* NPS-FM DO minima) is unknown.

2.3.5. Ecosystem metabolism

The Science Technical Advisory Group (STAG) (2019) recommended inclusion of ecosystem metabolism in the NPS-FM 2020 because:

- "There are currently no measures of ecosystem production in the NPS-FM, despite it being one of the five key components of the ecosystem health framework."
- "Ecosystem respiration provides a holistic ecosystem indicator, that includes the large, yet often ignored, microbial component. Ecosystem respiration not only signals changes in microbial processing but can also indicate changes to invertebrates and fish communities/population demographics as body size, temperature, nutrients and food supplies can all impact their respiration." (STAG, 2019)

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32 Other metrics such as shorter-term averages, or continuous values, could be used.

33 For this reason, monthly nutrient concentrations are rarely positively correlated with corresponding monthly chlorophyll *a* measurements at the same site. The relationships are more likely to be negative (eg, Kilroy C, Greenwood M, Wech J, Stephens T, Brown L, Mathews M & Patterson M. (2018). *Periphyton — environment relationships in the Horizons region: Analysis of a seven-year dataset.* NIWA Client Report No: 2018123CH. For: Dairy NZ/Horizons Regional Council. 188 p.).

The two elements of ecosystem metabolism, GPP and ER (the attribute variables), are specified for grading rivers against the NOF. This attribute differs from all others in that no numeric bands are specified because of a lack of data to support band definitions (STAG, 2019).[34] Grading is against three levels of stream health, 'healthy', 'satisfactory' and 'poor', which are defined in a separate publication (Young et al, 2016).

The attribute unit is the mean daily rate of production of DO per square metre of riverbed (in g O_2 m^{-2} d^{-1}), which is to be determined using the 'single-station open channel approach', as described in Young and Collier (2009).

The attribute description implies that a stream may be graded based on one-off measurements made over any seven-day period during summer (December to March). Data requirements are:

- a seven-day continuous (10-minute interval) time series of DO (in mg/L) and corresponding measurements of water temperature (°C)
- light intensity over the same period (to determine times of onset of darkness and daylight)
- average water depth determined from measurements across five cross sections upstream of the site.

DO should be logged as close as possible to the main flowing channel (thalweg) at the site. GPP and ER are calculated from the time series using a spreadsheet model (RiverMetabolismEstimator, Young and Knight 2005). Clapcott and Doehring (2017) present an example application of the Young and Knight (2005) method. ER is estimated using data collected in darkness (< 2 µmol m^{-2} s^{-1}) and requires an estimate of the DO deficit (difference between DO concentration at 100 percent saturation and actual concentration) and a re-aeration coefficient (K, which quantifies gas exchange between the air and water), with a correction for the effects of temperature.

Note that calculations of ER and GPP using this method can 'fail' (ie, return unrealistic values) partly because the assumptions that K and ER are constant (at given temperature) may not hold for all rivers (Parker, 2020). In addition, the assumption that the mass balance of O_2 is homogeneous for the whole study reach has generated some controversy (Demars et al, 2011; Demars et al, 2015; Hall & Tank, 2005; McCutchan & Lewis, 2006; McCutchan et al, 2002). It is beyond the scope of this guidance to discuss measurement in further detail, but we note that (STAG, 2019) included the following comments on the inclusion of metabolism as an attribute (additional qualifiers in italics):

- These processes *[GPP and ER]* have been measured in only a small number of streams *[in New Zealand]*. It is unclear how representative those streams and rivers are and how transferrable any conclusions from these measurements.

[34] From the STAG report (2019): "The bands we have proposed are based on international literature, as well as research from New Zealand. Whilst some members are confident that ecosystem metabolism metrics are robust indicators, the database of information for this metric in New Zealand is currently small and relationships between driving variables, such as land use, organic load and periphyton biomass, are not well understood. The group is less certain regarding where the bottom line should lie; therefore, we do not recommend setting a bottom line at this stage. This recommendation will need to be revisited as more data become available." The metrics "should be calculable whenever there are continuous dissolved oxygen measurements available and more data will allow these relationships to be refined".

- Further work is required to develop a national bottom line for ecosystem metabolism, and these bottom lines may need to vary with river type.

Regardless of the calculation methods, collecting data to estimate ER and GPP in streams has become more straightforward recently, as automatic *in situ* logging devices have become more available, cost efficient and reliable (Rode et al, 2016).

Continuous logging of DO (eg, at 10-minute intervals), with the addition of light and temperature loggers (both widely measured at sites across New Zealand) and depth measurements, would cover both the DO and ecosystem metabolism attributes: both require a minimum of seven days of continuous monitoring.

Attribute numeric values

The numeric values referred to in the NOF (table 8.1, Young et al, 2016) are based on those derived by (Young et al, 2008) using an analysis of data from 16 international and New Zealand studies in which sites were categorised as reference (draining unmodified catchments) and impacted (draining intensively modified land).[35] A percentile system was used to define ranges of GPP and ER in three categories (healthy, satisfactory and poor). Numeric values were also provided for the ratio of GPP to ER (P/R in table 8.1 of Young et al, 2016), which indicate whether a system is net heterotrophic (P/R < 1) or autotrophic (P/R > 1) and the strength of the autotrophy (eg, strongly autotrophic when P/R > 2.5).

Table 8.1 in Young et al (2016) also includes numeric values for the ratios of GPP and ER at sites of interest (test sites) with, respectively, GPP and ER at relevant reference sites. These ratios were based on the *a priori* categorisation of sites in the dataset as reference or impacted sites (Young et al, 2008). It is assumed the primary metrics are the raw measures of GPP and ER and the P/R ratio.

NOF thresholds separating the three stream health categories are:

GPP

- Healthy/Satisfactory – 4 g O_2 m^{-2} day^{-1}
- Satisfactory/Poor – 8 g O_2 m^{-2} day^{-1}

ER

- Healthy/Satisfactory – either 1.5 g O_2 m^{-2} day^{-1} (higher values in healthy range)
or 5.5 g O_2 m^{-2} day^{-1} (lower values in healthy range)
- Satisfactory/Poor – either 0.7 g O_2 m^{-2} day^{-1} (higher values in satisfactory range)
or 10 g O_2 m^{-2} day^{-1} (lower values in satisfactory range)

P/R

- Healthy/Satisfactory – 1.3 (ratio)
- Satisfactory/Poor – 2.5 (ratio)

[35] Note the numeric values in table 8.1 of Young et al (2016) differ slightly from those in Young et al (2008).

The alternate thresholds for ER reflect the unimodal relationships between land-use intensity metrics and ER, so that sites in impacted landscapes can be characterised by both very low and very high ER (see Section 2.2.5).

Sensitivity of the metrics to changes in nutrient concentrations over time

The literature review (see Section 2.2.5) indicated that nutrient concentrations only occasionally explain significant proportions of GPP and ER variability across multiple sites.

Substituting time for space, it is expected that relationships between measurements of GPP and ER (seven-day mean values in summer) and median annual nutrient concentrations over multiple years at the same site may be difficult to detect.[36]

Both GPP and ER measured over a given period are strongly dependent on non-nutrient factors at the time of the measurement, as shown by studies on temporal patterns in metabolism in single rivers (see Section 2.2.5); thus the annual metabolism measurement will reflect water temperature, light, flow and organic matter present (Section 2.2.5) at the time of the measurement, as well as the algal biomass (Chl *a*) that is the primary link to nutrient concentrations (Figure 2-5).

Spatial domain of attribute-nutrient relationship

Gross primary production and ER generally reflect balances in DO unfolding within and upstream of river reaches. Both GPP and ER are unlikely to be influenced by processes downstream, nor are they likely to be affected by adjacent catchments/systems. GPP and ER are, therefore, comparable with DO, periphyton and macroinvertebrate attributes with respect to their domains of spatial influence.

Will mediating factors lower sensitivity of the attribute to nutrients?

Factors mediating the response of the periphyton (Chl *a*, representing primary production) attribute and the DO attribute to DIN or DRP will also mediate the responses of the metabolism attribute to DIN or DRP. In addition, factors that contribute to ER (in particular the activities of heterotrophic microbes, driven mainly by the quantity of organic inputs to the stream) will influence the nutrient-metabolism relationship. Single measurements of either attribute over a random seven-day period in summer are therefore subject to high variability.

Will any other metrics of metabolism respond to nutrient concentrations before the NPS-FM attribute metric does?

As highlighted by STAG (2019), information on metabolism across New Zealand streams is sparse. The additional metrics suggested in table 8.1 of Young et al (2016) (ie, comparisons with data from reference sites), and their relationships with DIN or DRP concentrations have yet to be explored.

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36 Note the use of other nutrient metrics such as shorter-term averages, or continuous values, is not precluded.

3. Setting instream concentration thresholds

Section 2 summarised:

- our understanding of how nutrient enrichment affects (a) river ecosystems in general; and
  (b) NPS-FM attributes
- some key challenges and essential things to consider when developing ICTs for nutrient-affected river attributes in New Zealand.

In light of the context presented by Sections 1 and 2, the purpose of this section is to:

- offer an interpretation of the directive to set exceedance criteria
- describe strategies that could be implemented to set ICTs for DIN and DRP within regions
- outline the consequences and trade-offs associated with choosing one strategy over another
- present some methods that may help implement individual strategies as well as choosing from alternative strategies.

3.1. Interpreting Clause 3.13's directive to set exceedance criteria

Clause 3.13 directs councils to set temporal exceedance criteria as well as ICTs. However, the term *exceedance criterion* is open to interpretation and before progressing with the rest of Section 3 an explanation of this directive is required.

The intent of the directive to set temporal exceedance criteria was to:

- encourage councils to be aware of the temporal variability present in any instream nutrient time series
- understand how that temporal variability will complicate assessment
- develop methods that specify how you may infer whether nutrient concentrations are above or below an ICT and/or how *tolerant* councils are of individual observations of nutrient concentrations being above an ICT.

In this subsection:

- A brief hypothetical case study illustrates how spatial variation in nutrient-attribute relationships complicates setting ICTs and subsequent assessment of nutrient concentrations against ICT. It also generates risks to nutrient management. Some approaches to managing those risks are presented. The case study is provided to show that setting tolerances to *temporal variation* in nutrient concentration is one half of the tolerance problem associated with Clause 3.13; tolerances of *spatial variation* are at least as important to those of temporal variation.

- A brief discussion follows about how temporal variation in nutrient concentration generates uncertainty about the observed state of nutrient concentration in a river and, in turn, whether the observed state is above or below the ICT.

The case studies presented in this section are illustrative only to highlight some important issues to be aware of. Approaches that differ strongly from these could be used for estimating ICTs and for assessing instream concentrations against ICTs.

3.1.1. Tolerance of ICTs to under- and over-protect rivers

Environmental scientists and managers often wish to predict how ecosystems will change through time, as a result of management interventions (eg, reducing nutrient inputs to a river). Rather than explicitly modelling temporal change in an attribute as a function of temporal change in the stressor, a common approach is to assume *space-for-time-substitution* (SFTS). This involves developing a model of attribute state as a function of a spatial gradient in the stressor. This is assuming that within ecosystems the attribute's response to temporal changes in the stressor is reasonably approximated by its response to a spatial gradient in the stressor across ecosystems.[37]

The SFTS approach is commonly used for estimating stressor thresholds within the context of the NPS-FM (Franklin et al, 2019; Snelder et al, 2022). However, alternatives to SFTS could be used for predicting the response of attributes to nutrients. For example, process-based models may be better at capturing some of the complex responses of rivers to changes in fertiliser use within catchments (eg, time-lags/hysteresis; Geary et al, 2020; Robson et al, 2008).

Figure3-1a (overleaf) presents a hypothetical relationship between DIN concentration and MCI score among rivers within a region. If you suppose each point in the figure is a pair of long-term medians estimated at an individual site within a region,[38] the fitted regression line can be viewed as the (negative) relationship between DIN concentration and MCI 'on the average' among rivers within the region. The expression 'on the average' is important as not all points fall on the regression line and

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[37] A discussion of the advantages and disadvantages of SFTS is beyond the scope of this guidance, but we provide some references as a starting point:
Banet AI & Trexler JC. (2013). Space-for-time substitution works in everglades ecological forecasting models. *Plos One*, *8*(11), e81025. https://doi.org/10.1371/journal.pone.0081025;
Cowan JJH, Grimes CB & Shaw RF. (2008). Life history, history, hysteresis and habitat changes in Louisiana's coastal ecosystem. *Bulletin of Marine Science*, *83*(1), 197–215. https://www.ingentaconnect.com/content/umrsmas/bullmar/2008/00000083/00000001/art00011;
Lester RE, Close PG, Barton JL, Pope AJ & Brown SC. (2014). Predicting the likely response of data-poor ecosystems to climate change using space-for-time substitution across domains. *Global Change Biology*, *20*(11), 3471–3481. https://doi.org/https://doi.org/10.1111/gcb.12634;
Mac Nally R. (2008). The lag dæmon: Hysteresis in rebuilding landscapes and implications for biodiversity futures. *Journal of Environmental Management*, *88*(4), 1202–1211. https://doi.org/https://doi.org/10.1016/j.jenvman.2007.06.004

[38] When it comes to modelling ICT, you should carefully consider the choice of DIN/DRP variable. Although this case study uses medians, we have not justified this choice and others are available, eg, the periphyton attribute uses the 92nd percentile of monthly Chl *a* observations under the explicit assumption it is the most extreme high values of Chl *a* that reduce ecosystems' health, and acknowledging that Chl *a* concentrations are highly variable through time. Use of the 92nd percentile increases the tolerance of our attribute to that temporal variability.

the scatter of points about the regression line may represent river-specific DIN–MCI relationships and/or measurement errors in either variable. As such, the scatter of points about the regression line is a form of uncertainty about the region-wide parameters that define the slope and vertical position (intercept) of the regression.

The SFTS regression in Figure 3-1a may be used to estimate the DIN ICTs that — on the average for the region under consideration[39] — corresponds with a specific MCI target (Figure 3-1a). Extending the horizontal and vertical lines of Figure 3-1a that correspond with, respectively, the MCI target and the ICT, generates four quadrants in Figure 3-1b:

1. Sites in the upper-left quadrant are healthy with respect to MCI scores and DIN concentrations. Their MCI scores are 'better' than the target,[40] and their median DIN concentrations are below the ICT. Let *H* (healthy) denote the set of sites in the upper-left quadrant. There are N(*H*) = 12 healthy sites.

2. Sites in the lower-right quadrant have DIN concentrations above the ICTs and MCI scores worse than the target. These sites are obvious candidates for setting limits on nutrient use within their catchments. Let *L* (limits/action required) denote the set of sites in the lower-right quadrant. There are N(*L*) = 30 sites clearly requiring nutrient limits and/or an action plan to reduce nutrient inflows.

3. Sites in the upper-right quadrant have MCI scores that are better than the target despite having DIN concentrations above the ICT. These sites do not require reductions in DIN to ensure their MCI scores are at or above the target.[41] As such, these sites are referred to as *over-protected* relative to the regional ICT. There are N(*O*) = 8 over-protected sites.

4. Sites in the lower-left quadrant have MCI scores that are 'worse' than the target even though their DIN concentrations are below the regional ICT. These sites may require further reductions in DIN below the ICT[42] to elevate their MCI scores towards the regional target. As such, these sites are referred to as *under-protected* relative to the regional ICT. There are N(*U*) = 10 under-protected sites.

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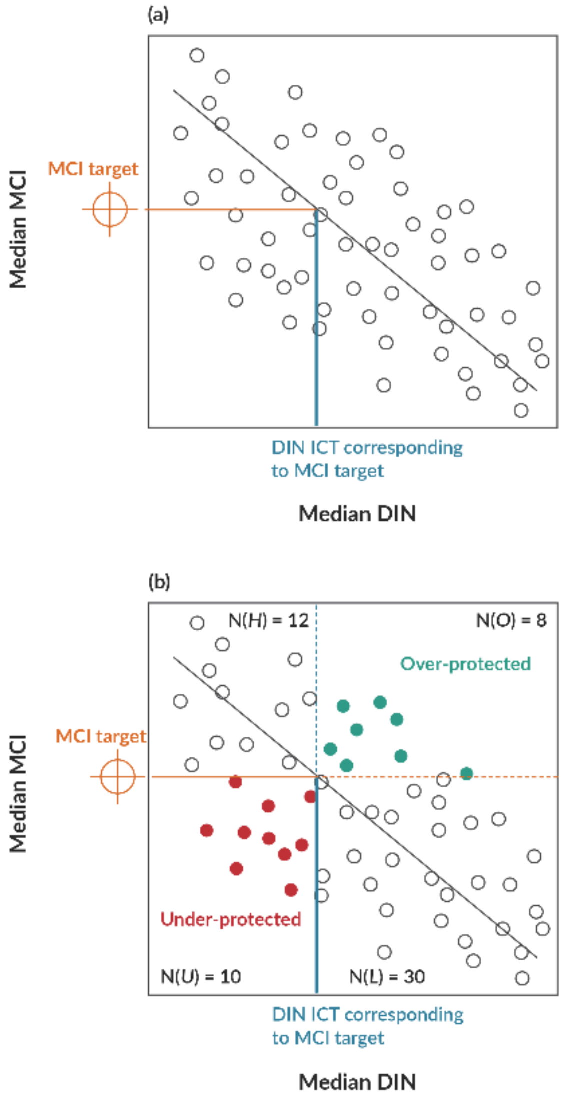
[39] Given the data and the model used, this is important context because a different data set (eg, one from sites randomly distributed throughout a region rather than located in a non-random arrangement) and/or a different model (eg, a process-based model or a correlational/statistical model containing multiple stressors) may yield different attribute-ICTs relations.

[40] We use the expressions *better* or *worse than* the target because they are more explicit relative to objectives than *greater* or *less than*. For some attributes, high scores are desirable (eg, MCI) whereas for other attributes high scores are undesirable (eg, Chl *a*, *E. coli*).

[41] But may still be adversely affected by observed nutrient concentrations, hence may have attribute states that could improve following reductions in nutrient concentrations.

[42] Or may not, if the low MCI scores are due to the influence of other, non-nutrient stressors.

Figure 3-1: Diagrams explaining how ICTs may be derived using a very basic statistical model



(a) Demonstration of an SFTS regression can be used to determine the ICTs corresponding to an attribute target.

(b) Explanation of this approach to deriving an ICTs generates four sets (quadrants of the plot space) of importance regarding assessment and subsequent nutrient management.

Under-protected sites highlight the potential that our ICTs fail to meet ecological targets in a subset of rivers. In Figure 3-1b, there are N(*H*) + N(*U*) = 22 sites with DIN concentrations below the ICTs and N(*U*) = 10 of those sites are under-protected. Hence, ([10/22]*100) 45 percent of sites that satisfy the regional ICTs may be under-protected with respect to macroinvertebrate targets.[43] If we assume the sites in Figure 3-1a and Figure 3-1b are representative of the region as a whole, then we may say there is a probability of 0.45 of under-protection (P(*U*) = 0.45) associated with the ICT, given the attribute target, data and model. A useful question is: what value of P(*U*) are we willing to *tolerate*?

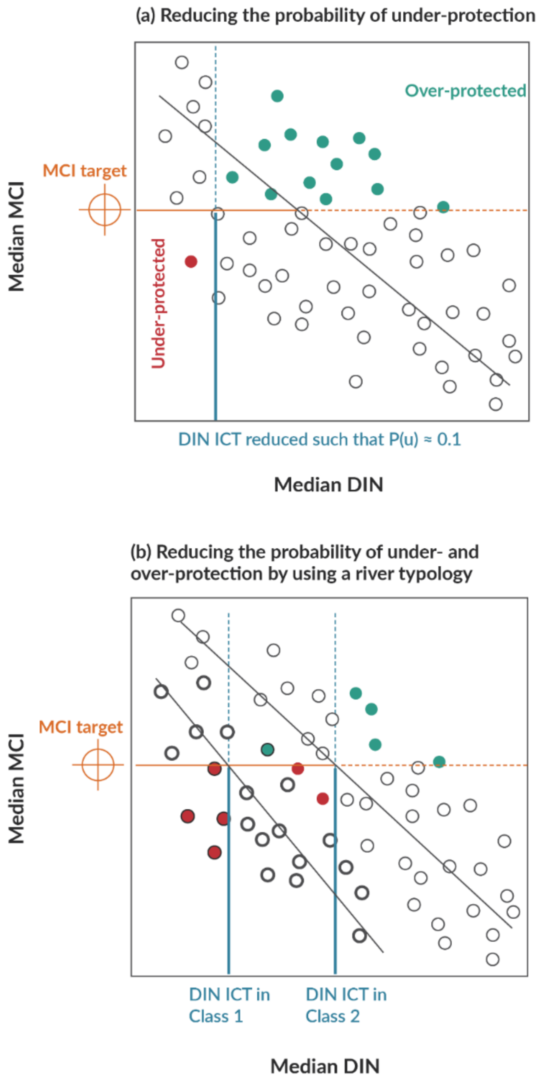
43 We say *may* be under-protected because we do not know that reductions of DIN will improve MCI in those streams. It is possible, for example, that MCI scores are low in the under-protected sites because of other stressors (eg, fine sediment).

Thus, P(*U*) can be viewed as one metric of *tolerance* associated with our ICT.[44]

Suppose P(*U*) = 0.45 represents too much tolerance and we want to reduce P(*U*) to 0.1. Figure 3-2a shows the result of reducing our tolerance of under-protection. Reducing P(*U*) to 0.1, while being environmentally conservative — hence consistent with Te Mana o te Wai — comes with trade-offs and assumptions that should be made explicit.

Figure 3-2: Diagrams showing how the probabilities of under- and over-protection can be reduced.

Noting the trade-offs and assumptions that come with use of the approaches explained in this figure (see text).



One trade-off associated with reducing P(*U*) is an increase in the proportion of sites that are over-protected (P(*O*)). In Figure 3-1b and Figure 3-2a there are N(*H*) + N(*O*) = 20 sites with MCI scores better than the target. When P(*U*) = 0.45 there were 8 over-protected sites, such that P(*O*) = 0.4 (Figure 3-1b). Reducing our tolerance of under-protection to 0.1 increases P(*O*) to 13/20 = 0.65. Thus, decreasing tolerance of under-protection may increase tolerance of over-protection and vice versa.

44 Note the term *exceedance criterion* is inappropriate and/or misleading in this context: under-protected rivers are not *exceeding* some threshold in space or time; they are in fact below a target threshold and the ICT.

The overarching principle of Te Mana o te Wai within the NPS-FM implies that under-protection should be of greater concern than over-protection. It is nevertheless worthwhile to consider the possible consequences of over-protection. For example, over-protection can be a problem if it:

- leads to unnecessary allocation of limited resources (eg, setting limits and implementing nutrient action plans in catchments with naturally higher DIN levels and whose ecological targets are already satisfied) or
- unnecessarily imposes heavy restrictions on agriculture (eg, reducing stock density in catchments whose rivers have high ecosystem health despite also having DIN concentrations greater than the ICT).

A second trade-off that comes with reducing P(*U*) (and increasing P(*O*)) is that it increases the number of sites clearly requiring limit-setting and/or action, N(*L*). When P(*U*) = 0.45 (figure3-1b), N(*L*) = 30, but when P(*U*) = 0.1, N(*L*) = 39. Increasing tolerance of over-protection increases the need to allocate council resources to nutrient management.

If we decide to minimise P(*U*) and reduce the DIN ICT, then we are assuming DIN is having a significant detrimental effect on the MCI scores at sites classified as under-protected. Before imposing the trade-offs that come with reducing ICTs to reduce P(*U*), you should decipher the influence of other, non-nutrient stressors on the sites in Figure 3-1b that are classified as under-protected.

Possible solutions and the need for pragmatism

Approaches such as the one used by Canning et al (Canning et al, 2021) aim to minimise the difference between P(*U*) and P(*O*), thus balancing the risks of over- and under-protection. Whether or not balancing P(*U*) and P(*O*) is consistent with Te Mana o te Wai requires more discussion and is beyond the scope of this guidance.

Risks of under- and over-protection are exacerbated by managing rivers (ie, limit-setting and designing action plans) under the assumption that stressor-response relationships are spatially homogeneous (Stoffels et al, 2021). One solution is to classify a region's rivers into a *typology* that groups them based on their ecological responses to instream nutrients, then develop ICTs for each *class* of river within the typology (Franklin et al, 2019; Snelder et al, 2022).

Figure 3-2b presents a hypothetical example. Let subscripts denote the class (either Class 1 or Class 2; Figure 3-2b), such that the:

- number of healthy rivers in Class 1, $N(H)_1 = 5$
- number of rivers requiring limit-setting in Class 1, $N(L)_1 = 12$
- number of over-protected rivers in Class 1, $N(O)_1 = 1$
- number of under-protected rivers in Class 1, $N(U)_1 = 4$
- number of healthy rivers in Class 2, $N(H)_2 = 10$
- number of rivers requiring limit-setting in Class 2, $N(L)_2 = 22$
- number of over-protected rivers in Class 2, $N(O)_2 = 4$
- number of under-protected rivers in Class 2, $N(U)_2 = 2$.

Thus, by dividing our rivers into two classes, the risk of under-protection is now $N(U)_1 + N(U)_2$ / $[N(U)_1 + N(U)_2 + N(H)_1 + N(H)_2]$ = 6/21 = 0.29 (reduced from 0.45 without the typology in Figure 3-1b). Similar calculations can be made to show that by using a typology the risk of over-protection is reduced from 0.4 (Figure 3-1b) to 5/20 = 0.25. Developing class-specific ICTs simultaneously reduces the probabilities of under- and over-protection.

Challenges such as those presented above are an unavoidable consequence of policy statements that direct management towards quantitative targets of both stressors and responses. Note the above complications concern only one nutrient-affected attribute and one nutrient. Several other solutions to developing ICTs that facilitate simultaneous reductions in the risk of under- and over-protection are possible. They are not presented in this guidance as they are more complicated and would require more detailed worked examples; such approaches should be a focus of future research.

A key message from the above is that ontological uncertainties will mean that, irrespective of the modelling approach taken, some degree of under- and over-protection of rivers must be tolerated. If possible, we recommend documenting the tolerances of under- and over-protection that come with any ICTs implemented. The amount of tolerance councils are willing to accept may depend on resource constraints and other practical considerations (Stoffels et al, 2021). For example, a simple approach may identify rivers that clearly are well above the ICT and also have low attribute scores — ie, rivers in the lower-right, *L* set (Figure 3-1b). If councils have limited resources, rivers in the *L* set may be prioritised for more stringent levels of on-ground nutrient management, while rivers in set *U* (under-protected), may — at least initially — be targeted for further investigation towards an improved understanding of the influence of other stressors.

Furthermore, when deciding on how much to invest in refining ICT models, keep in mind the environmental outcomes specified in plans, that is, what we are trying to achieve 'on the ground'. Ontological uncertainties are merely one source of uncertainty affecting environmental outcomes (Van Der Sluijs et al, 2005). Arguably, greater sources of uncertainty arise from, for example, the behavioural response of farmers to stricter limits on fertiliser use, and the response of catchment-scale nutrient cycles to on-ground nutrient management plans.

3.1.2. Analysis and assessment of a nutrient time series against an ICT

Exceedance is typically defined as the frequency with which observations exceed some threshold. This definition is intuitive and frequently encountered in disciplines such as risk analysis. However, our research[45] has indicated that multiple interpretations of exceedance criteria are circulating amongst stakeholders responsible for, and affected by, Clause 3.13.

45 Phone and email correspondence.

One way to interpret exceedance is the frequency with which observations in a sample exceed a metric of location[46] (MOL) of that sample.[47] We emphasise 'of that sample' to reiterate this particular interpretation of exceedance refers to the frequency of observations above a MOL of that sample, not some threshold defined by other data not included in that sample or, for example, a model-based threshold. For clarity, we refer to this type of exceedance as exceedance above a sample MOL (abbreviated to exceedance of a sample's location; ESL). Exceedance above a sample MOL is different to exceedance above a threshold (ET), which we explain later.

To better understand ESL, consider the time series of monthly DIN observations from a site in Southland (Figure 3-3). The black, orange and red lines in Figure 3-3 are the long-term median, mean and 75th percentile, respectively. The ESL (as a proportion) of the median, mean and 75th percentile (Q3) by all monthly observations in Figure 3-3 is, respectively, 0.5, 0.45 and 0.25. These values of ESL are consistent with what you would expect based on the metrics of location used (median, mean and Q3) and the observed distribution of DIN values in Figure 3-3:

- The median divides the ranked set of observations equally into two subsets of equal size: one subset contains observations greater than the median; the other subset contains observations less than the median. With respect to the median, ESL must, by definition, be 0.5.
- Similarly, Q3 is the individual observation that divides the ranked set of observations into the top 25 percent of observations, and the bottom 75 percent of observations – its ESL must, by definition, be 0.25.
- ESL of the mean will equal that of the median when the distribution of individual observations about the mean is symmetrical (eg, a normal or a uniform distribution). If the distribution of observations is right-skewed[48] – as is the case in Figure 3-3 – then the mean will be greater than the median, resulting in the ESL of the mean being lower than that of the median.

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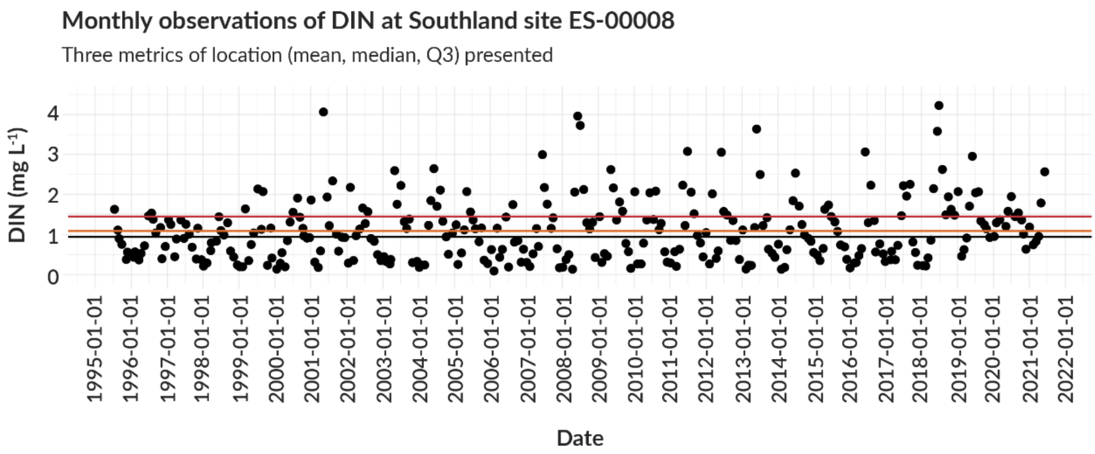
46 Two types of metrics are often used to describe a sample/distribution of observations. Metrics of location define what value the sample of observations is centered on; the mean and median are examples. Metrics of dispersion (eg, variance) define how variable observations are around metrics of location.

47 A sample is defined as a set of individual observations. For example, a sample may consist of several individual, point observations of DIN concentration. This definition of *sample* and *individual observation* follows that of Sokal and Rohlf (1994) Biometry: *The Principles and Practice of Statistics in Biological Research*. 3rd Edn. Freeman.

48 A distribution of observations is *right-skewed* when the distribution has a longer tail associated with larger values and/or outliers associated with larger values. The higher frequency of larger values relative to that of smaller values results in the mean of a right-skewed distribution being higher than the median.

Figure 3-3: Time series of individual DIN observations from a site in Southland



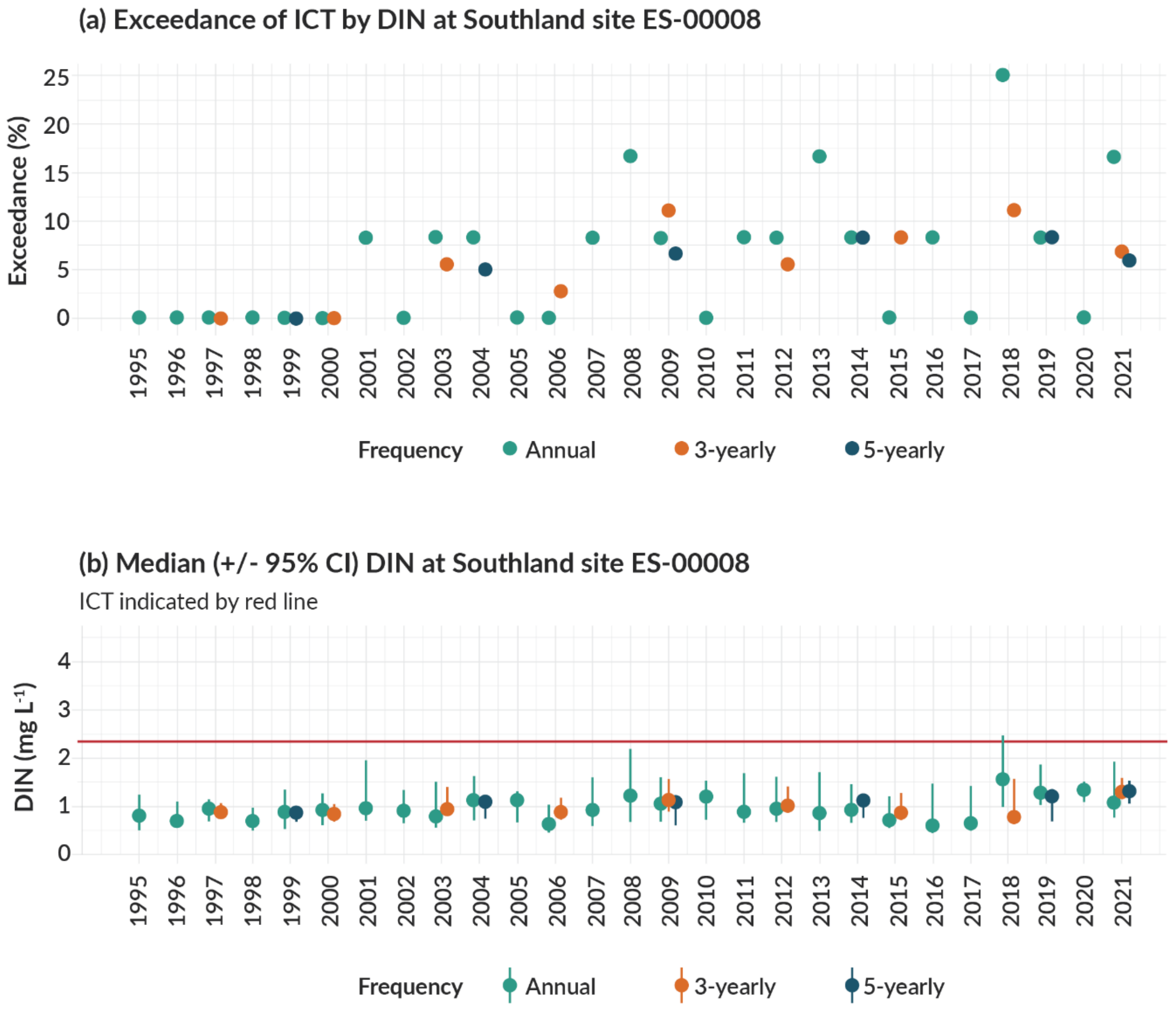
Black, orange and red lines denote, respectively: the long-term median DIN concentration; the long-term mean; and the long-term 75th percentile (Q3; third quartile).

In the context of the NPS-FM, values of ESL familiar to readers will be, for example, 8 percent for the periphyton metric. That is, the 92nd percentile[49] of (at least three years of) monthly Chl *a* observations is how the periphyton attribute is calculated. The 92nd percentile of observations was selected as the attribute to acknowledge that high periphyton biomass was the primary threat to ecosystem health. Note that, in the context of the periphyton attribute, the ESL of 8 percent says nothing about the location of a Chl *a* time series relative to any of the periphyton attribute band thresholds. Because the ESL of the periphyton attribute will always be 8 percent, irrespective of the 92nd percentile of Chl *a* sample itself, it provides no information about the frequency with which Chl *a* at a site has exceeded any threshold, except in one special case: If the 92nd percentile of a sample of Chl *a* observations happens to equal a periphyton band threshold, then we know the exceedance of that band threshold by that particular sample is 8 percent. Only in this very special case does ESL provide information about exceedance of NOF band thresholds. In reality, the probability of Chl *a* samples from rivers yielding 92nd percentiles that equal band thresholds will almost certainly be very small.

More generally, knowing the ESL of an attribute is of marginal value when assessing whether a time series of attribute measurements is above or below attribute targets. For successful implementation of Clause 3.13, ESL of attributes is of minor importance relative to ET, as explained below.

[49] For default sites.

Figure 3-4: Summaries of the time series of monthly DIN samples at Southland site ES-0000



In (a) exceedance rate of a point defined as the percentage of monthly observations leading up to that date, over the period specified (either 1, 3 or 5 years). (b) Red line denotes ICT.

Suppose the ICT of site ES-00008 is 2.35.[50] Figure 3-4a shows the exceedance (as a percentage of individual observations) of ICT = 2.35 by monthly DIN observations taken at ES-00008. Exceedance rates at three different time intervals are presented: over 1-year, 3-year and 5-year intervals. Thus, ET is one way of summarising a time series of nutrient observations *relative to the ICT*. It is easy to see how, in contrast with ESL, ET is a useful concept for assessment of instream nutrient concentrations against ICTs under the NPS-FM.

Clause 3.13 does not mandate assessment of nutrient time series using exceedance rates as shown in Figure 3-4a. An alternative approach is to summarise a time series of DIN concentrations using MOLs and some measure of confidence about those MOLs. Figure 3-4b presents 1-year, 3-year and 5-year median DIN concentrations for ES-00008. Confidence intervals (CIs; 95 percent) about those

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50 2.35 mg L^{-1} is actually the 90th percentile of all DIN time series collected in Southland up to and including June 2021.

medians are also presented. As you could expect, medians calculated over longer periods are less variable in time than those calculated over shorter periods (Figure 3-4b). None of the 95 percent confidence intervals in Figure 3-4b are above the (hypothetical) ICT. If the lower limit of a 95 percent confidence interval was above the ICT, then we may reasonably infer the DIN concentrations at the site during that period were significantly above the ICT.

When comparing assessment of DIN series using ET rates (Figure 3-4a) and medians (Figure 3-4b) the following observations are noted:

- Rates of ET fluctuate through time much more than medians. Like medians, however, temporal variability in exceedance rates can be reduced by increasing the period over which they are estimated.
- Rates of ET are based on counts of observations above the ICT, so assessments based on rates of ET alone do not provide an indicator of the magnitude of DIN enrichment. By contrast, medians present a more transparent assessment of the magnitude of DIN fluctuations through time.
- The two different approaches to assessment leave contrasting impressions. Rates of ET create the impression of excessive DIN, while medians +/- confidence interval create the impression of there not being a problem with excessive DIN at ES-00008.

What are the consequences to planned environmental outcomes of an exceedance rate of 10 percent? Alternatively, what are the consequences to planned environmental outcomes of a median DIN concentration of 1.5 mg L^{-1}? These questions highlight that, when choosing a method to assess nutrient concentration series, it may be wise to use statistics that summarise DIN or DRP series in a way that best represents the ecological relationships between nutrients and nutrient-affected attributes. For example, if episodic excursions of DIN above the ICT are of little consequence to attributes, then rates of ET may be a poor choice of assessment method. Choice of assessment method must be guided by a knowledge of ecological relationships (Section 2 of this guidance) (Jakeman et al, 2006).

Two additional approaches to assessment of DIN time series against an ICT are presented in Figure 3-5. Both of these approaches involve fitting a simple model — Generalised Additive Model (GAM)[51] — to running statistics of DIN concentration over time. In Figure 3-5a, a GAM has been fitted to running ET rate. To estimate running ET for a specific month, we estimated the rate of ET over a specified time window (1-year, 3-years and 5-years) before that month. In Figure 3-5 b, a GAM has been fitted to running medians. The points of difference noted for Figure 3-4 above also apply when comparing approaches presented in Figure 3-5. In addition, we note the following:

- trends in rates of ET and median DIN concentration are better deciphered when modelled as in Figure 3-5, than when presented as discrete observations as in Figure 3-4
- modelled running statistics (refer Figure 3-5) smooth out the fluctuations observed for discrete, periodic estimates (refer Figure 3-4). Thus, using models like that used for Figure 3-5 render assessments that are less sensitive to short-term fluctuations in nutrient concentrations. This may or may not be desirable, depending on objectives in regional plans.

We reiterate that the significance of rates of ET to attribute state may be unknown for specific attributes.

[51] For illustrative purposes only. A more defensible modelling exercise would require a more elaborate approach.

Figure 3-5: Additional ways of analysing a time series against an ICT

Generalised Additive Models (GAMs) fitted to running statistics. Presented are a (a) running exceedance model (+/- 95 percent CI) and a (b) running median model (+/- 95 percent CI) over 1-year, 3-year and 5-year windows.

Suggestions

The directive of Clause 3.13 to set temporal exceedance criteria could arguably be more usefully framed as the need to clearly define how time series of individual nutrient samples will be analysed and assessed against the corresponding ICT. We must define:[52]

1. **Which metrics will be used to summarise a series of individual observations.** For example, we must decide between rates of ET and medians, at a minimum.[53] Choice of metric should be guided by an understanding of the ecological relationships between the metric used to summarise a nutrient time series and the attribute under consideration. For example, if episodic excursions of nutrient concentration above an ICT (ET rates) are likely to have less ecological impact than a sustained increase in median concentrations, then not choosing ET would be logical and defensible.
2. **The time period over which the metric of location will be applied.** The longer the time period over which the metric of location is applied, the more temporal variation in nutrient concentrations is smoothed out, and the less sensitive the metric is to temporal variation in nutrient concentration.
   It may also be appropriate to specify how the period is defined. For example, should the metric be calculated over a series of discrete years such as January to December, or can the calculation start in any month, producing a monthly time series of the metric? Time series modelling allows the user to 'borrow strength' for inference from the entire series, but it does lower sensitivity to short-term fluctuations in nutrient concentrations.
3. **Confidence and/or levels of tolerance about the metric.** Confidence intervals about the metric and/or fitted model will be required to infer statistical significance of differences between ICTs and observed DIN series. Metrics such as standard deviation, standard error and confidence intervals indicate the dispersion of observations about an MOL. Confidence intervals of 95 percent (95 percent CI) are commonly used to assess the statistical significance of the difference between two measures (Anderson et al, 2000). It is less easy to estimate confidence intervals about quantiles and rates of ET and this may require a numerical (eg, bootstrap) approach (Manly & Navarro Alberto, 2021). If rates of ET are preferred, you need to define what rate of ET can be tolerated, before inferring that observed nutrient concentrations are significantly elevated above an ICT.
4. **The rule that is applied to determine whether nutrient concentrations over the specified period are significantly higher than the ICT.** For example, you may conclude nutrient concentrations within a river are significantly higher than the ICT[54] if the lower limit of the confidence interval is above the ICT. In Figure 3-4b, for example, there is no significant exceedance of the ICT given the MOL (median) and confidence (95 percent CI) used for that analysis.

We do not recommend particular metrics and rules to solve the four decision-problems above. Such recommendations are beyond the scope of this guidance and will require research so that the costs and benefits associated with the choices available are better understood.

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[52] Noting that much more sophisticated approaches to time series analysis could be taken. We only present these four steps to show some key things to consider when developing approaches to assessing instream nutrient concentrations against ICTs.

[53] Other metrics are possible. Rates of ET and medians are just two possibilities.

[54] Given the metric of location used, and the time period over which that metric is applied.

3.2. Best practice when navigating difficult decisions – the PrOACT framework

Implementing Clause 3.13 — within the context of implementing the NPS-FM as a whole — involves councils navigating a complex set of decisions (refer to Section 1.4.2 for more context). Similar decision problems have been tackled for decades by various management agencies globally, resulting in the development of frameworks for structured decision-making. These frameworks have been so extensively applied and refined they can be considered best practice in structured decision-making (Conroy & Peterson, 2013; Gregory et al, 2012).

This guidance follows the PrOACT framework for smart decision-making (Gregory & Keeney, 2002; Keeney, 2004), which expert environmental scientists have widely adopted (Hemming et al, 2022). There are five key elements of the PrOACT framework (Keeney, 2004):

1. ***P**roblem*: Define the decision problem carefully so the right problem will be solved. Section 1 of this guidance presents the problem to be solved.
2. ***O**bjectives*: Clearly define and differentiate fundamental and means objectives (aims in this case) that must be met to solve the problem. This is done in Section 3.2.
3. ***A**lternatives*: As far as practicable, present the full range of alternative strategies for meeting the fundamental aims. This is critical as it frames the entire approach to solving the problem, ensuring the choices available to decision-makers are preserved.
4. ***C**onsequences*: Describe how well the alternative strategies enable the fundamental aims to be met.
5. ***T**rade-offs*: Balance the pros and cons of the alternative strategies that can be chosen to meet the fundamental aims.

Other elements of effective decision-making are often added to extend the PrOACT framework (Keeney, 2004):

6. Identify major *uncertainties* associated with alternative strategies.
7. Consider the relevant *risk tolerances* associated with each alternative strategy — each alternative comes with some risk of not meeting the fundamental aims, so how much risk are we willing to tolerate?
8. When evaluating alternative strategies against the fundamental aims, keep in mind *linked decisions.* These are decisions associated with solving a latter problem, but likely to influence how we evaluate strategies for solving the current problem.

The rest of this section is concerned with elements 2 to 8 of the PrOACT framework. In Section 3.2, we define fundamental and means aims of possible strategies to implement Clause 3.13. In Section 3.4, we present four strategies we may follow to meet fundamental aims. Each strategy considers elements
2 to 8, while also describing some methods available to give effect to each strategy.

The material within each strategy is presented in four subsections:

1. a brief overview of the strategy
2. *Advantages:* a summary of each strategy's advantages, particularly the consequences for meeting means and fundamental aims

3. *Trade-offs and uncertainties*: a summary of the trade-offs and uncertainties of implementing the strategy, relative to other strategies
4. *Application and recommendations*: descriptions of how the strategy may be applied — including some available methods/tools that may help — as well as our recommendations.

Under the *Application and recommendations* subsections we were mindful of linked decisions that come after developing ICTs to meet fundamental aims. These were those decisions associated with setting limits and implementing action plans to reduce nutrient inputs to rivers.

Our treatment of risk tolerances under *Application and recommendations* is cursory for two reasons:

- First, it is not yet clear how risk could/should be estimated for particular tasks of strategies outlined below. Approaches to risk calculation will likely emerge as the strategies are implemented.
- Second, risk tolerances are often subjective judgements made by decision-makers (Conroy and Peterson 2013), and so need to be set by councils themselves or, if facilitated by non-council personnel, be co-developed with councils.

Various tools are available to implement each strategy (eg, regression models for estimating ICTs) and these are referred to in the *Application and recommendations* subsection of each strategy. However, tools are not covered in any detail, nor is one tool advocated over another. Instead, some general guidelines are provided to help you select the most appropriate tools in light of constraints and intended use.

Before presenting strategies, however, it is necessary to clarify the aims we hope to meet when implementing Clause 3.13 to guide strategy development. We differentiate between two types of aims; fundamental and means.

3.3. Fundamental aims *versus* means aims of strategies to implement Clause 3.13

A large body of literature has shown that effective and efficient policy implementation is compromised by failure to (a) identify all fundamental aims[55] of policy clauses; (b) differentiate fundamental aims from means aims; and (c) differentiate the aims of clauses from the strategies that may be implemented to meet those aims (Conroy & Peterson, 2013; Gregory et al, 2012; Hemming et al, 2022). In the context of this guidance:

- *Fundamental aims* represent relevant nutrient-affected values (compulsory and other; appendix 1 of the NPS-FM) and are the environmental outcomes (Clause 3.9 of the NPS-FM) we wish to achieve through implementing Clause 3.13. Fundamental aims may go beyond desired environmental outcomes and include the minimisation of constraints that hamper our ability to meet environmental outcomes relevant to Clause 3.13, as well as outcomes relevant to other clauses of the NPS-FM.

[55] The discipline of decision science now consistently uses the expressions fundamental and means *objectives*, not fundamental and means *aims*. However, the word 'objective' is used throughout the NPS-FM, and so is likely to be interpreted in particular ways by the reader. To avoid confusion, we use the word 'aim' to refer to aims we may wish to achieve by employing strategies to implement Cause 3.13.

- *Means aims* are the aims we may need to meet to achieve our fundamental aims; means aims are a 'means to an end'. Failure to meet a means aim may be of little concern if our fundamental aims are met, yet the converse is not true.

Clause 3.13 of the NPS-FM states: "To achieve a target attribute state for periphyton, any other nutrient attribute, and any attribute that is affected by nutrients, every regional council must, at a minimum, set appropriate instream concentrations and exceedance criteria for DIN and DRP" (Clause 3.13(1) of the NPS-FM).

Clause 3.13(1) requires some interpretation to ensure we do not conflate fundamental and means aims and, consequently, embark on an inefficient implementation pathway. For example, you may interpret Clause 3.13(1) as a directive to quantitatively develop separate ICTs for all nutrient-affected attributes within freshwater management units (FMUs). Choosing to interpret Clause 3.13(1) in this way would result in work to develop at least:

> 3 (thresholds between bands) x 8 (minimum number of nutrient-affected attributes; see below) x 2 (nutrient species; DIN and DRP) = 48 ICTs.

If ICTs become region- and/or river-type-specific, then this number of 48 could easily double or triple, for example.

It has become apparent in preparing this guidance[56] that stakeholders have interpreted the aims of Clause 3.13 in different ways. If we are to identify effective and efficient strategies for implementing Clause 3.13 then we must establish some fundamental and means aims to be used for strategy development. To be clear, the fundamental and means aims presented below have been defined by the authors of this guidance based on (a) our interpretation of Clause 3.13 and (b) our understanding of the constraints councils are experiencing, as highlighted during the scoping workshop[57] for this project.

- **Fundamental Aim (FA) 1** is to establish a set of ICTs that protects the target states of all nutrient-affected attributes within regions.

Fundamental Aim 1 does not necessarily imply we must develop statistical relationships between DIN, DRP and every nutrient-affected attribute of the NPS-FM in every region.

We acknowledge that:

- all NPS-FM requirements relating to nutrient management sit alongside those relating to other stressors (eg, those pertaining to other anthropogenic stressors like environmental flows, fine sediment)
- meeting FA1 is the first step of many associated with nutrient management within FMUs, including setting limits on resource use (Clause 3.14 of the NOF) and developing action plans (Clause 3.15 of the NOF), resulting in changes to catchment management practices.

Acknowledging the above two points, implementing the NPS-FM could be resource intensive.

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[56] Through personal communications. For example, during the scoping workshop (see next footnote) communications that informed the writing of this guidance, as well as through conversations with stakeholders in other meetings.

[57] A scoping workshop was held with councils and the Ministry for the Environment to scope this guidance.

Recognising this leads to a second fundamental aim to guide strategy development:

- **Fundamental Aim (FA) 2** is to minimise the cost to councils of setting ICTs for nutrient-affected attributes.

Fundamental Aim 2 is not an objective of the NPS-FM, nor is it implied by the NPS-FM, but we include it in this guidance to highlight that — given limited resources — resource allocation to one particular NPS-FM implementation activity (eg, intensive monitoring programs to refine ICTs) will likely come at the expense of resources allocated to another activity (eg, working with landholders to change land-use practices to reduce nutrient inputs).

Taken together, FA1 and FA2, respectively, concern maximising efficacy and efficiency of nutrient management. The means aims that we have to meet in order to achieve FA 1 are:

- **Means Aims (MA) 1–8** are to define DIN and DRP ICTs that allow councils to meet the target states for each of the following attributes:
  - Chl a (MA1)
  - MCI (MA2)
  - QMCI (MA3)
  - ASPM (MA4)
  - FIBI (MA5)
  - DO (MA6)
  - GPP (MA7)
  - ER (MA8).

Means aims to meet FA2 include:

- **Means Aim 9** is to minimise the number of attribute-specific ICTs required by councils.

Completing certain strategies in Section 3.4 may lead to us concluding that, for example, setting ICTs to meet MA3 satisfies FA1. If so, then it is not strictly necessary to have ICTs for the eight different attributes corresponding to MA1 to MA8.

- **Means Aim 10** is to minimise unnecessary data analyses employed to derive ICTs.

If, for example, analyses have already been conducted to yield ICTs for an attribute, then is it necessary to re-analyse data to further refine those ICT? Or could the resources that would have been directed towards re-analysis of data be better used on other NPS-FM implementation tasks? At what stage should ICTs be updated with new knowledge/data?

- **Means Aim 11** is to minimise the duplication of effort.

Certain actions taken towards setting ICTs (refer Section 3.4) may not need a council-specific approach. Instead, the required output/information could be provided by a single research team, eliminating duplication of effort and increasing efficiency of NPS-FM implementation.

- **Means Aim 12** is to minimise unnecessary collection of data.

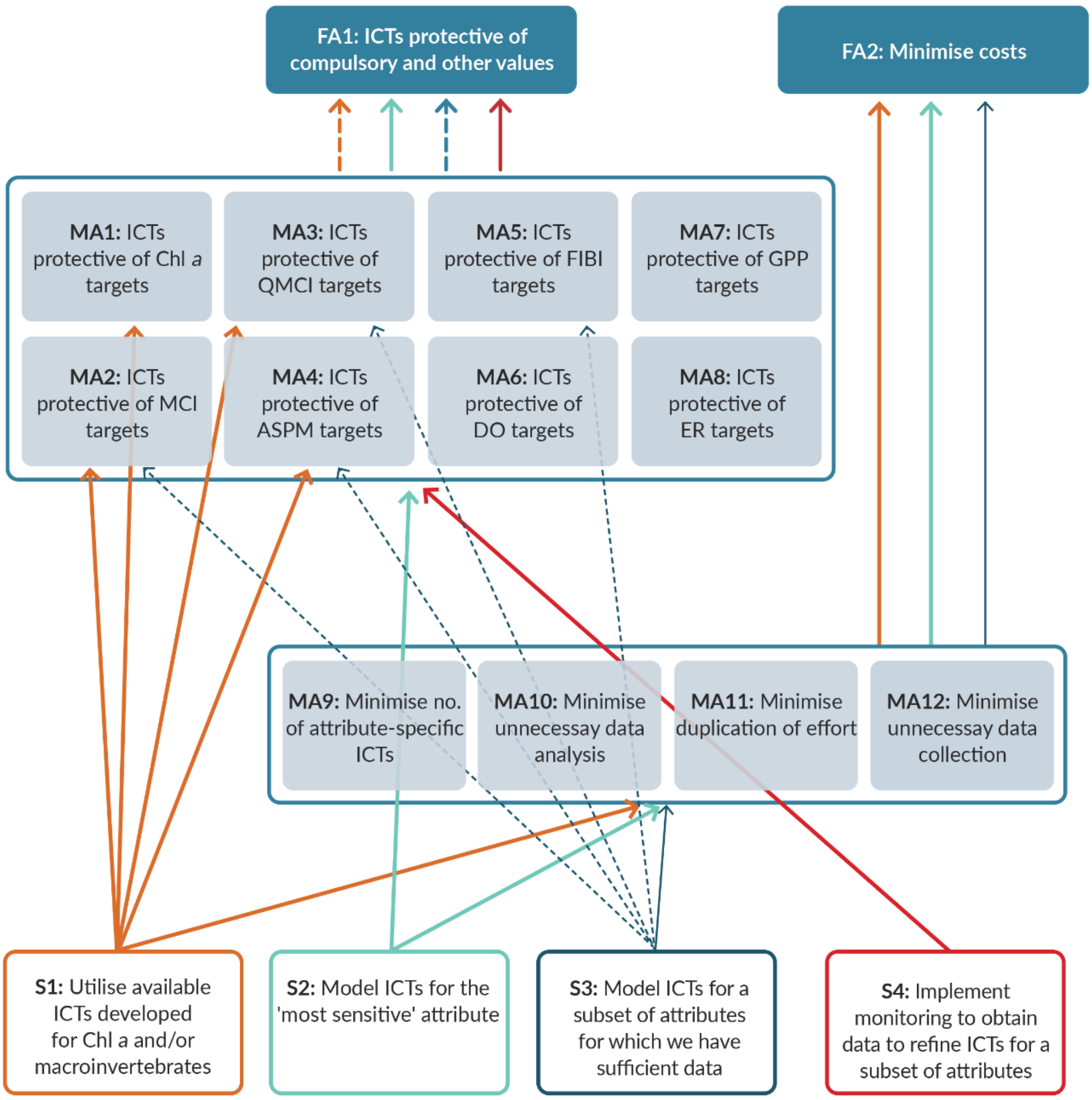
If ICTs have been set such that we are confident that FA1 can be met, then further monitoring for estimation of ICTs *per se* may not be necessary. In this case, then the purposes and priorities of future data collection must be carefully considered.

3.4. Strategies to set nutrient exceedance criteria

An aims network (*sensu* Conroy and Peterson 2013) is presented in Figure 3-6. In defining the aims network, we have avoided over-simplifying the problem (following the recommendation of Keeney, 2004). We have not ignored the complexity of implementing Clause 3.13 of the NPS-FM, but have nevertheless aimed to distil the decision-making process into as few strategies as possible (four).

The four strategies are complementary and linked; they are not mutually exclusive alternatives. For example, Strategy 4 cannot sensibly be seen as a standalone alternative to other strategies and is only recommended after having tried to implement other strategies (see Section 3.4.4).

Figure 3-6: Aims network linking four strategies for obtaining ICTs (S1-S4) to means aims (MA) and fundamental aims (FA).



Heavy solid lines = strongly facilitates aim; light solid line = weakly facilitates aim; dashed line = extent to which aim has been met unknown.

3.4.1. Strategy 1: Use ICTs that have already been developed for a nutrient-affected attribute

Implementing Strategy 1 is straightforward and involves obtaining peer-reviewed, published ICTs from New Zealand technical reports and papers, ideally for all nutrient-affected attributes.

Advantages

- A subset of MA1–8 will be met yielding initial/draft ICTs for including in regional plans, due to be notified by late 2024.
- Strategy 1 is implemented at minimal cost, hence meets MAs 9–12 and FA2.
- This guidance presents ICTs that councils could use now in regional plans, so there is also minimal cost in terms of time. This strategy will not delay development of regional plans.

Trade-offs and uncertainties

- Published New Zealand ICTs are only available for periphyton and macroinvertebrates, so the extent to which these ICTs enable us to meet FA1 is unknown (refer Figure 3-6).
- Periphyton and macroinvertebrate attributes may be ineffective indicators of nutrient impacts in soft-bottom and/or plant-dominated streams.
- For all macroinvertebrate attributes, a single set of ICTs was developed for both DIN and DRP for all of New Zealand. They are not stratified by possible mediating factors. A single set of ICTs for DIN and DRP for all of New Zealand may be seen as advantageous and/or practical by some, in that it is easy to implement. Others may view this as being unrealistic and a biased threshold of nutrient impacts in many catchments of New Zealand.

Application and recommendations

Nutrient exceedance criteria have been developed for the NPS-FM periphyton attribute (Snelder et al, 2022). Instream concentrations for both DIN and DRP corresponding to the periphyton A-B, B-C and C-D band thresholds have been provided in Table 2-1. Canning et al (2021) estimated DIN and DRP ICTs for the C-D threshold of all three macroinvertebrate attributes. The corresponding instream concentration thresholds have been provided in Table 2-2.

The approach of Snelder et al (2022) to developing ICTs differs very strongly to that of Canning et al (2021). The modelling approach used by Canning et al (2021) does not account for the mediating effects of landscape context or other anthropogenic stressors on nutrient-macroinvertebrate relationships. Accordingly, their approach did not yield numerous ICTs to be applied to different landscape contexts throughout New Zealand. The analysis of Canning et al (2021) resulted in a single C-D (bottom line) ICTs for both DIN and DRP, nationwide.

By contrast, the analysis of Snelder et al (2022) accounted for the mediating effects of several factors including:

- climatological and topographical variables as defined in the River Environment Classification (Snelder & Biggs, 2002)
- hydrological variables
- shaded versus unshaded streams
- deposited fine sediment.

Their analysis also yielded context-specific ICTs at three different levels of risk of 'under-protection' at a site (see Snelder et al, 2022, for details). Consequently, their analysis yielded 126 ICTs for each of DIN and DRP, for each periphyton band threshold (126 x 2 x 3 = 756 ICTs in total).

We do not recommend one set of published ICTs over another. There is currently no consensus on which of these two different approaches and their resultant sets of ICTs will be most effective and efficient for managing nutrient inputs to New Zealand's rivers. Councils will need to decide which published ICTs to use, based on their local knowledge, practical requirements and constraints. For example, councils may ask themselves: where are monitoring sites to be located within a region? The answer will shed light on how many different landscape contexts DIN and DRP will be monitored in, hence how different ICTs might need to be throughout a region.

If councils use the ICTs based on periphyton developed by Snelder et al (2022) then they should do so according to the associated guidance (Snelder et al, 2022). No such guidance is available for implementation of the ICTs based on macroinvertebrate response developed by Canning (2021).

If councils have not yet developed their own ICTs using sound approaches (see Strategy 2), then we recommend implementing Strategy 1 in the short term, for inclusion in regional plans. Strategy 1 is not, however, a long-term solution, given the uncertainties about how its implementation will meet FA1.

3.4.2. Strategy 2: Model ICTs for the most sensitive attribute

The objective of Strategy 2 is to generate, for each type of river (see Task 2.1), a single set of six[58] ICTs for an attribute determined to be most sensitive to nutrient enrichment. Implementing Strategy 2 involves four tasks (circles), four outputs (diamonds) and one decision/question (rectangle; Figure 3-7). The process is as follows:

1. **Task 2.1**: The aim of this task is for councils to inform Task 2.2 by suggesting a coarse typology of streams in their region.[59] It is unlikely a single attribute will be the most sensitive indicator of nutrient impacts across all stream types. For example, consider soft-bottomed, macrophyte-dominated streams and hard-bottomed streams with few macrophytes. It is possible QMCI is the most sensitive indicator in hard-bottomed streams, but DO is the most sensitive in soft-bottomed streams. The coarse typology should be based on (a) which types of rivers are of concern with respect to nutrient enrichment, hence which types of rivers need to be a focus of monitoring and adaptive nutrient management; and (b) the types of rivers that are likely to

[58] Three band thresholds for each of DIN and DRP = 6 ICT.

[59] Councils may already have a working typology; in which case this task is already complete.

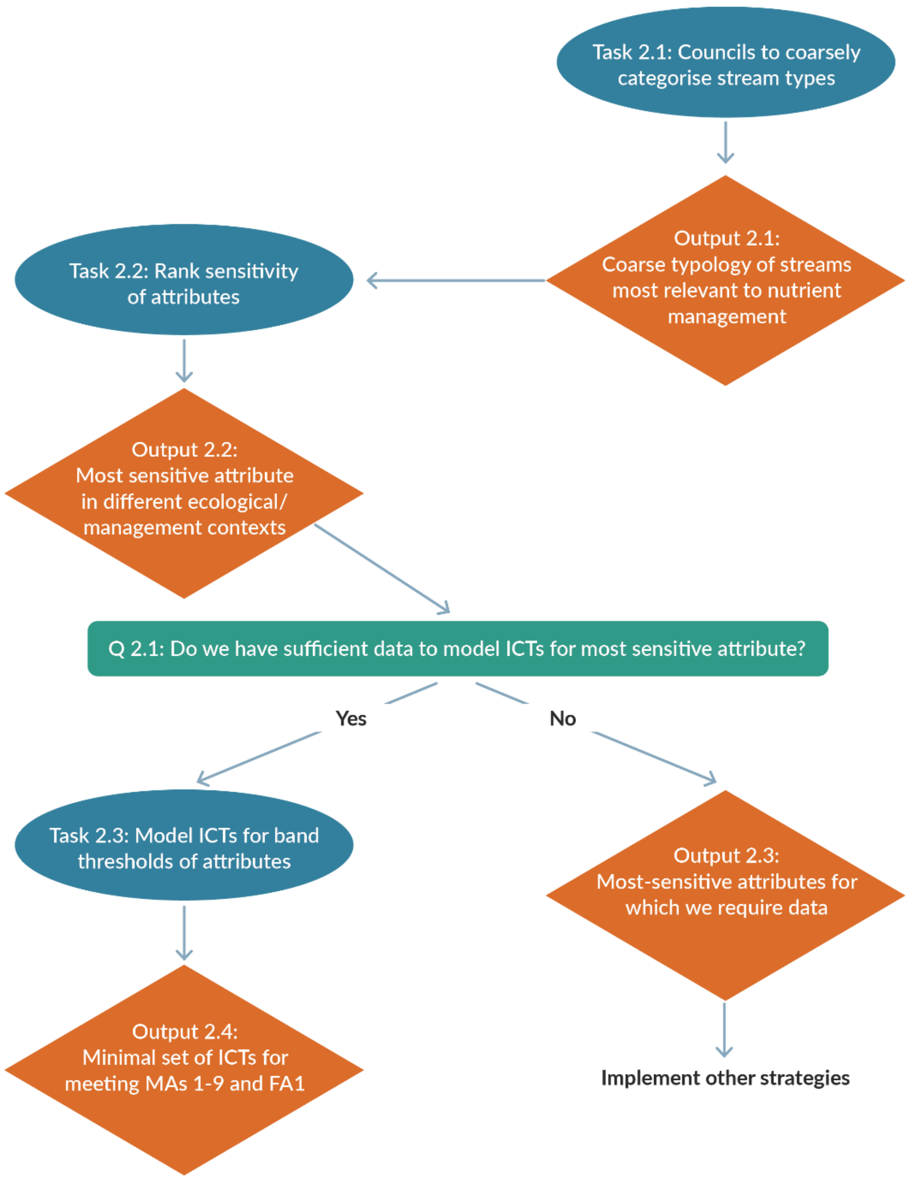
support different nutrient-sensitive attributes (eg, rivers where the primary producer biomass is macrophyte-dominated, *versus* those that are periphyton-dominated).

2. **Output 2.1**: A coarse typology of rivers developed by councils will determine how attributes will be grouped for subsequent meta-analyses that is aimed at ranking the sensitivity of attributes to nutrient enrichment.[60] Grouping of attributes before meta-analysis ensure that the most nutrient-sensitive attribute per river type is obtained, hence to ensure ICTs are fit for purpose.
3. **Task 2.2**: Structured meta-analyses (see Application and recommendations, below) could be conducted[61] to rank the sensitivity of attributes to nutrient enrichment, within each stream type. The meta-analysis involves synthesising quantitative and qualitative information across published papers and reports which have examined nutrient effects on ecosystems. Ideally, this task will also give you some estimates of confidence in the rankings.
4. **Output 2.2**: Task 2.2 will yield the attribute most sensitive to nutrient enrichment within each coarse stream type.
5. **Question 2.1**: Once you have obtained the set of most sensitive attributes (one per type of river), you then need to determine data availability for modelling. Question 2.1 must be answered after gaining some appreciation of data requirements of different approaches, hence after some exploration of approaches to Task 2.3. If there are insufficient data to model the ICTs of a sensitive attribute, then you may need to consider Strategies 3 and 4.
6. **Output 2.3**: The set of attributes (one per type of river) that are likely the most sensitive attributes within a river type, yet for which you require more data.
7. **Task 2.3**: Statistical models could be developed to determine ICTs for the most sensitive attributes (one per type of river) within a region.
8. **Output 2.4**: The final output of Strategy 2 is a minimal set of ICTs for meeting FA1.

[60] Councils may have already developed a typology, fit for purpose.

[61] Not necessarily by councils; see *Application and recommendations.*

Figure 3-7: Process for implementing Strategy 2 (See text for explanation.)



Advantages

- Meta-analysis capitalises on several decades of work around the world that has aimed to determine the effects of nutrient enrichment on rivers. Drawing upon this large bank of knowledge/data may reduce uncertainty around the relative sensitivity of attributes at little cost in terms of both time and money. Use of existing knowledge/data facilitates meeting MA10 and MA12, hence FA2.

- By minimising the number of attributes for which we model ICT, we facilitate the meeting of MA9 and MA10, hence FA2.

- Having councils drive development of a stream typology for nutrient monitoring and management ensures the typology is fit for their purposes (relevant to their needs) and legitimate (Cash et al, 2003).

- There is no need for numerous researchers (including councils) to complete Task 2.2 or 2.3 — these tasks could be completed by a single research team (*Application and recommendations*), thus facilitating MA11, hence FA2 (but see *Application and recommendations*).
- Strategy 2 yields a minimal set of ICTs for implementation within regions, so should be relatively simple to implement in the context of regional monitoring and management plans.
- The ranking of attributes also enables us to prioritise reduction of uncertainties in ICT. That is, we may better target data collection for a smaller subset of attributes, such that we can refine ICTs in the long term.
- Task 2.2 will strengthen our conceptual understanding of nutrient impacts, which, in turn, is a critical factor affecting confidence in model outputs (Allen-Ankins & Stoffels, 2021; Dormann et al, 2012; Kelly et al, 2022).
- FA1 is met at minimal cost.

Trade-offs and uncertainties

The extent to which the meta-analysis of Task 2.2 can confidently rank attribute sensitivity is unknown. Ardon et al (2021) recently completed a meta-analysis of experimental studies of nutrient effect on river ecosystems. One of their objectives was to determine the relative sensitivity of different trophic levels (ie, primary producers, primary consumers and so on) to nutrient enrichment. They were unable to conclude that trophic levels exhibit different effect sizes as a result of nutrient enrichment. However, their meta-analysis was not aimed at ranking NPS-FM attributes (eg, they included experiments examining effects of nutrient enrichment on periphyton composition and primary production rate, which would have increased the sensitivity of primary producers to enrichment (cf. periphyton biomass/Chl *a*)). Their meta-analysis also did not include descriptive studies. A more targeted and comprehensive meta-analysis aimed at ranking NPS-FM attributes may be more insightful than the analysis of Ardon et al (2021). However, there is a risk we cannot confidently rank attributes by their nutrient sensitivity, which is an uncertainty associated with Strategy 2.

Strategy 2 ranks attributes by their nutrient sensitivity based on data gleaned from rivers around the world. You could argue this introduces uncertainty about the extent to which rankings based on ecosystems in other countries are applicable to New Zealand. This argument/source of uncertainty would have to be addressed as part of Task 2.2.

By focusing on the most sensitive attributes per river type, we may trade off the development of ICTs for less nutrient-sensitive attributes. In turn, you could argue this results in a less complete suite of ICTs to implement within regions.

Application and recommendations

Councils need to drive Task 2.1 to ensure the river typology is fit for the purpose of Task 2.2 as well as for the purposes of monitoring and managing nutrients ‘on the ground’, within regions. Council staff spanning science, consents policy and catchment management should all contribute to the typology. This will help balance the influence of ecological details (science) with the practical constraints of consenting, planning and catchment management. If councils already have a typology that is fit for purpose, Task 2.1 can be skipped.

We recommend Task 2.2 is completed by a single research team with a broad set of subscriptions to scientific journals and experience in data synthesis and meta-analysis. This team may or may not include council staff. Completing Task 2.2 in a council-specific fashion is unnecessary and would not satisfy MA11 nor MA10. Numerous robust approaches to meta-analysis are available (Chen & Pollino, 2012; Pappalardo et al, 2020; Pollino et al, 2007; Stewart, 2010; Webb et al, 2013; Webb et al, 2012).

If Task 2.2 yields a set of nutrient-sensitive attributes, then Question 2.1 needs to be answered after considering the modelling approaches to be used for estimating ICT. Some modelling approaches need more data (eg, Snelder et al, 2022) than others (eg, Canning et al, 2021). It follows there is some flexibility in the data requirements of Task 2.3.

If there are insufficient data for the attribute most sensitive to nutrient enrichment within river types, within regions, then Strategies 3 and 4 will need to be considered.

Completing Task 2.3 will involve fitting some form of regression model in which the response, *Y*, is some function of a set of covariates, X, including DIN and/or DRP. Many and varied model types can be used for regression, including:

- generalised linear models (GLMs) (McCullagh & Nelder, 1989)
- generalised additive models (GAMs) (Wood, 2017)
- the mixed-effects versions of these two families (GLMMs and GAMMs) (Gelman & Hill, 2007; Zuur et al, 2009)
- various machine learning approaches like boosted regression trees (BRTs) and random forests (RFs) (Hastie et al, 2009)
- other computer-intensive approaches including 'minimisation of mismatch' (Phillips et al, 2019).

Once a good regression model *Y* ~ X has been obtained, ICTs are derived by determining the values of DRP/DIN that result in the band thresholds of *Y*.

We do not recommend specific regression models for Task 2.3 for three reasons:

1. A lot of guidance on how to select certain regression models to estimate ICTs in light of data availability/constraints has already been developed and published (Kelly et al, 2022; Phillips et al, 2018; Poikane et al, 2021). Phillips et al (2018) have already developed Excel-based and R-based tools for applying their recommended approaches. We do not repeat that information here.
2. It is extremely difficult to recommend specific models or modelling frameworks for estimating ICTs due to the plethora of different modelling approaches available, including the type of regression models covered by Phillips et al (2018) through to process-based models, Bayesian belief networks and rule-based models (eg, decision trees). Each approach has strengths and weaknesses and, as suggested by Jakeman et al (2006), a it may be better to highlight best-practice procedures that are more generally applicable to all families of models (see Section 3 of Jakeman et al, 2006).
3. Selection of specific modelling approaches is only one step among many in best-practice modelling (Jakeman et al, 2006).

Instead of recommending specific statistical models, here are some general rules for good practice in regression modelling you can apply to any specific regression approach selected. Most of these are presented in advanced regression modelling texts (Gelman & Hill, 2007; Hastie et al, 2009). The rules presented below are special cases — adapted to the specific problems this guidance addresses — of the more general cases presented by Jakeman et al (2006), Bennet et al (2013) and Rose et al (2015).

Before applying any of the rules below to Task 2.3, you should refer to Jakeman et al (2006) and Bennet et al (2013). Any of the papers cited below can be obtained from the NIWA authors of this guidance.

Some basic rules for regression modelling to obtain ICTs include:

A. **When parameterising[62] the model, keep in mind the intended use of the model so that it is fit for purpose.** Quality, defensible regression modelling is not easy (see the rules below), but avoid the temptation to get lost in the scientific and statistical details. Let the model parameterisation be guided by its intended use. For example, if councils feel they have no use for hundreds of ICTs for multiple contexts, then why parameterise the model that way?

B. **Carefully document the intended use of the model and, in light of intended applications, the assumptions and parameterisation decisions made.** No approach to regression modelling is perfect as all approaches come with trade-offs. Expert modellers know this, but too often the assumptions and modelling decisions are poorly documented and explained (Schmolke et al, 2010; Zurell et al, 2020).

C. **In addition to DIN and/or DRP, include covariates that are likely to interact with DIN/DRP and change the magnitude and/or direction of the attribute's response to DIN/DRP.** As shown in Section 2, effects of nutrients on attributes are strongly dependent on spatial context. Context-dependencies can be accounted for to some degree by including contextual covariates in the model, thereby partitioning their influence on attribute response (Zuur et al, 2009). See Snelder et al (2022) for a worked ICTs example and their influence on attribute response (Zuur et al, 2009).

D. **Assess how well the data represent the environments for which the ICTs will be applied.** Were any of the data used to develop ICTs collected within your region? Were data used to develop ICTs representative of the broad landscape types (eg, climate, geology, topography) relevant for nutrient management within your region? Questions such as these must be asked to answer Question 2.1.

E. **Check for collinearity amongst covariates and be aware of potential confounding of nutrient gradients**. In Section 2 we highlighted the potential for nutrient gradients to be confounded by other covariates, particularly other anthropogenic stressors (eg, rivers subject to high nutrient inputs are often subject to other stressors like fine sediments). Confounding of DIN/DRP gradients occurs when those nutrient variables are collinear with other variables. Collinearity amongst covariates has the potential to significantly distort ICTs estimation. The problem cannot be completely removed (Dormann et al, 2013), but Dormann et al (2013) and Mac Nally (2000) offer some practical advice for handling it professionally.

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[62] Parameterisation refers to how the regression model is designed, which covariates are included and how those covariates interact with each other.

F. **Check how well the model fits the data, throughout the covariate domain[63] of interest**. As recently highlighted by expert statisticians, environmental scientists often place too much faith in model-fit statistics output by software (Bennett et al, 2013; Mac Nally et al, 2018). The residuals of the fitted model need to be plotted against all covariates of interest to check for biased fits, hence potentially biased ICT. Zuur et al (2009) explains how to check model fit in detail and in simple, accessible language. External validation of predictive performance is also recommended (Mac Nally et al, 2018). Simulation can be used to examine the sensitivity of model predictions to spatial and temporal variability in covariates throughout domains of interest (Gelman and Hill 2007; see Stoffels et al (2020) for a worked example). Bennet et al (2013) presents thorough guidance on how to assess the performance of a model.

G. **Do not extrapolate model predictions beyond the domain of training data**. The *training data* is the subset of all available data used to estimate model parameters — the data used to 'train' the model. Once a final model has been obtained, it is not uncommon to apply the fitted model beyond the domain of the training data. Doing so usually leads to spurious inferences/predictions and it is not recommended (Hastie et al, 2009; Wood, 2017; Zuur et al, 2009). Modellers need to ensure when estimating ICTs, they are doing so well within the domains of *all covariates* of the fitted model; not just within the domains of DIN and/or DRP. Modellers also need to be cautious with predictions within the margins of all covariate domains. Training data is often sparse within the margins of domains, which again can lead to high uncertainty in predictions.

H. **Analyse uncertainties in ICT.** Be transparent about the ontological/statistical uncertainties about estimated ICTs (Jakeman et al, 2006). Present confidence intervals about ICT. For any given set of ICTs, irreducible spatial variation in nutrient-attribute relationships will result in unavoidable risks of observing sites with (a) good attribute scores even though they have failed the ICT; and (b) bad attribute scores even though they have passed the ICT. Awareness of these risks can then shape how ICTs are estimated to minimise one form of risk over another (eg, Snelder et al, 2022) or balance the probability of both risks equally (eg, Canning et al, 2021).

I. **If resources permit, employ ensemble modelling for improved understanding and presentation of uncertainties about ICT**. All models have trade-offs and no single approach is perfect. Ensemble modelling is the application of numerous different types of models to the same problem. By using several models to draw inferences and make predictions we (a) improve our understanding of both epistemic and ontological uncertainties; and (b) better present those uncertainties in a transparent manner (Jakeman et al, 2006). Ensemble modelling is now standard practice in, for example, climate science (Stainforth et al, 2007) and is becoming commonplace in modelling to support conservation decisions (Buisson et al, 2010; Diniz-Filho et al, 2009; Thuiller et al, 2019).

J. **If expert modelling capability is unavailable within councils, employ help from outside councils**. Environmental modelling standards are improving rapidly (Robson, 2014). With that improvement comes increased sophistication of practice and higher demands on technical expertise. It is easy to make mistakes (Jakeman et al, 2006), so consult expert analysts where possible.

[63] In the context of this guidance, the *domain* of a covariate, **x**, is defined by the set of values $\{x_1, x_2, \ldots, x_i, \ldots, x_n\}$ used in the training data.

K. **Document the modelling approach and outputs and subject that documentation to peer review**. In addition to the documentation covered in recommendation B, above, document model performance (Bennet et al, 2013) and outputs (eg, the model objects/software) and then subject that document to expert, independent peer-review, to improve quality assurance (Rose et al, 2015).

To what extent should Task 2.3 be completed by individual councils for their own application? There are good reasons to have a single research team complete Task 2.3 using data from across all regions to develop a model of national extent (henceforth, a *national model*).

First, use of a national data set increases precision of parameter estimates and may, in some cases, reduce bias by incorporating environmental contexts for which some regions do not have data. Second, if developed by expert analysts, a national model of ICTs for the most sensitive attributes can serve as a useful benchmark and/or point of ICTs reference for regional models (Kelly et al, 2022). Third, a national model may facilitate the provision of ICTs to councils with insufficient data. Individual councils may not have sufficient data to complete Task 2.3 for the most sensitive attribute within a river type, but such data may be available from other nearby regions.

Developing a national model for Task 2.3 does not preclude councils from completing Task 2.3 on their own if sufficient data are available. Indeed, there is value in cross-validating ICTs arising from both national and regional models (Kelly et al, 2022).

3.4.3. Strategy 3: Model ICTs of a subset of attributes for which we have sufficient data

The objective of Strategy 3 is to generate, for each type of river (see Task 3.1), a set of ICTs for attributes for which we have sufficient data.

The key differences between Strategies 2 and 3 are the determinants of attributes selected for ICTs modelling. In Strategy 2, we aim to model ICTs for attributes that are likely the most nutrient-sensitive attributes within each type of river *and* for which we have sufficient data. In Strategy 3, the main determinant is data availability, resulting in a selection of attributes that are not necessarily the most nutrient sensitive within river types.

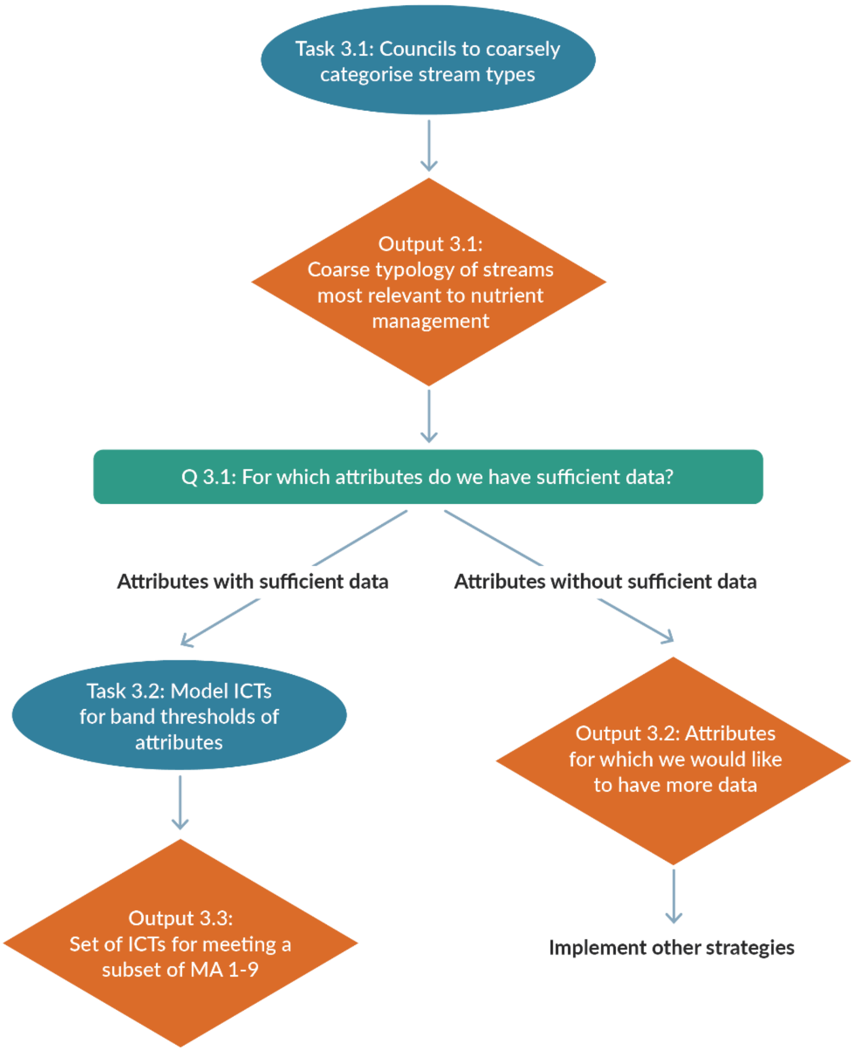
We may opt for Strategy 3 over Strategy 2 for several reasons, including:

- Councils may wish to make as much use of available data as possible.
- Councils may require coupling of Strategies 3 and 2 if the answer to Question 2.1 (Figure 3-7) is 'No' for certain attributes/river types within their region. That is, councils may opt for Strategy 2 but then choose to model ICTs for what is likely, for example, the second or third most nutrient-sensitive attribute within a river type, if that is the attribute for which they have sufficient data. It is important to distinguish this coupling of Strategies 2 and 3 from the use of Strategy 2 alone, to ensure the uncertainties and trade-offs associated with using Strategy 3 are not discounted from the coupled strategy.
- Certain stakeholder groups may ask councils to develop ICTs for attributes that are not necessarily the most nutrient sensitive, but of particularly high social, economic or cultural value.

Implementing Strategy 3 involves two tasks (circles), two outputs (diamonds) and one decision/question (rectangle; Figure 3-8).

1. Task 3.1 and Output 3.1 are equivalent, respectively, to Task 2.1 and Output 2.1.
2. Question 3.1 differs from Question 2.1 in that we determine data availability for a subset of nutrient-sensitive attributes, where that subset is not necessarily the set of most sensitive attributes per river type.
3. Output 3.2 is similar to Output 2.3 in that it is a set of attributes for which we would ideally have more data. However, because the attributes in Output 2.3 are the most sensitive attributes, we require more data for them to meet FA1. By contrast, the attributes in Output 3.2 are not necessarily the most sensitive, and so getting more data for them may not be needed to meet FA1.
4. Task 3.2 is equivalent to Task 2.3.
5. Output 3.3 differs from Output 2.4 in that the set of ICTs produced is not necessarily minimal (councils may have estimated ICTs for multiple attributes within river types). Unlike Output 2.4, Output 3.3 does not necessarily meet MA 1-8, and so does not necessarily meet FA1.

Figure 3-8: Process for implementing Strategy 3 (See text for explanation.)



Advantages

- A subset of Means Aims 1-8 will be met.
- Councils maximise use of available data.
- Having councils drive development of a stream typology for nutrient monitoring and management ensures the typology is fit for their purposes (relevant to their needs).
- There is no need for numerous researchers/councils to complete Task 3.2 — this task could be completed by a single research team (see *Application and recommendations*), thus facilitating MA11, hence FA2 (but see *Application and recommendations*).
- Strategy 3 can be coupled with Strategy 2 to yield a defensible set of ICTS, given data limitations for certain nutrient-sensitive attributes.
- If deemed efficient to do so, national models of ICTs for several nutrient-affected attributes could be developed in a standardised way, then compared to shed further light on which NPS-FM attributes yield the most protective ICT.

Uncertainties and trade-offs

The extent to which Strategy 3 meets FA1 is uncertain, due to uncertainty about the relative nutrient sensitivity of attributes and the degree to which ICTs developed for a subset of attributes are protective of other attributes.

If councils choose to let data availability be the main criterion by which ICTs are developed, MA9-12 will not be met, resulting in a failure to meet FA2.

If Strategy 2 is skipped altogether, we may remain uncertain as to which attributes are most impacted by nutrient enrichment. Failing to reduce such uncertainties may, for example, impair our ability to communicate the anticipated benefits of nutrient management plans to stakeholders.

If Strategies 2 and 3 are coupled, that coupled strategy does not default to acquiring the advantages of Strategy 2 — potential uncertainties and trade-offs of Strategy 3 will apply to a subset of the ICTs developed.

Application and recommendations

Given the methodological equivalence of tasks across Strategies 2 and 3, details concerning application have been covered under Strategy 2.

If councils choose to implement Strategy 3, they need to be aware of the advantages of Strategy 2 that they are trading off, and the resultant uncertainties that come with implementing Strategy 3.

Our answer to the question 'To what extent should Task 3.2 be completed by individual councils for their own application?' is the same as the answer we provided to the analogous question (for Task 2.3) in Strategy 2.

We do not recommend implementing Strategy 3 on its own. We nevertheless present Strategy 3 as an option for councils to ensure that no valid strategies have been missed.

3.4.4. Strategy 4: Implement monitoring to obtain data to refine ICTs for a subset of attributes

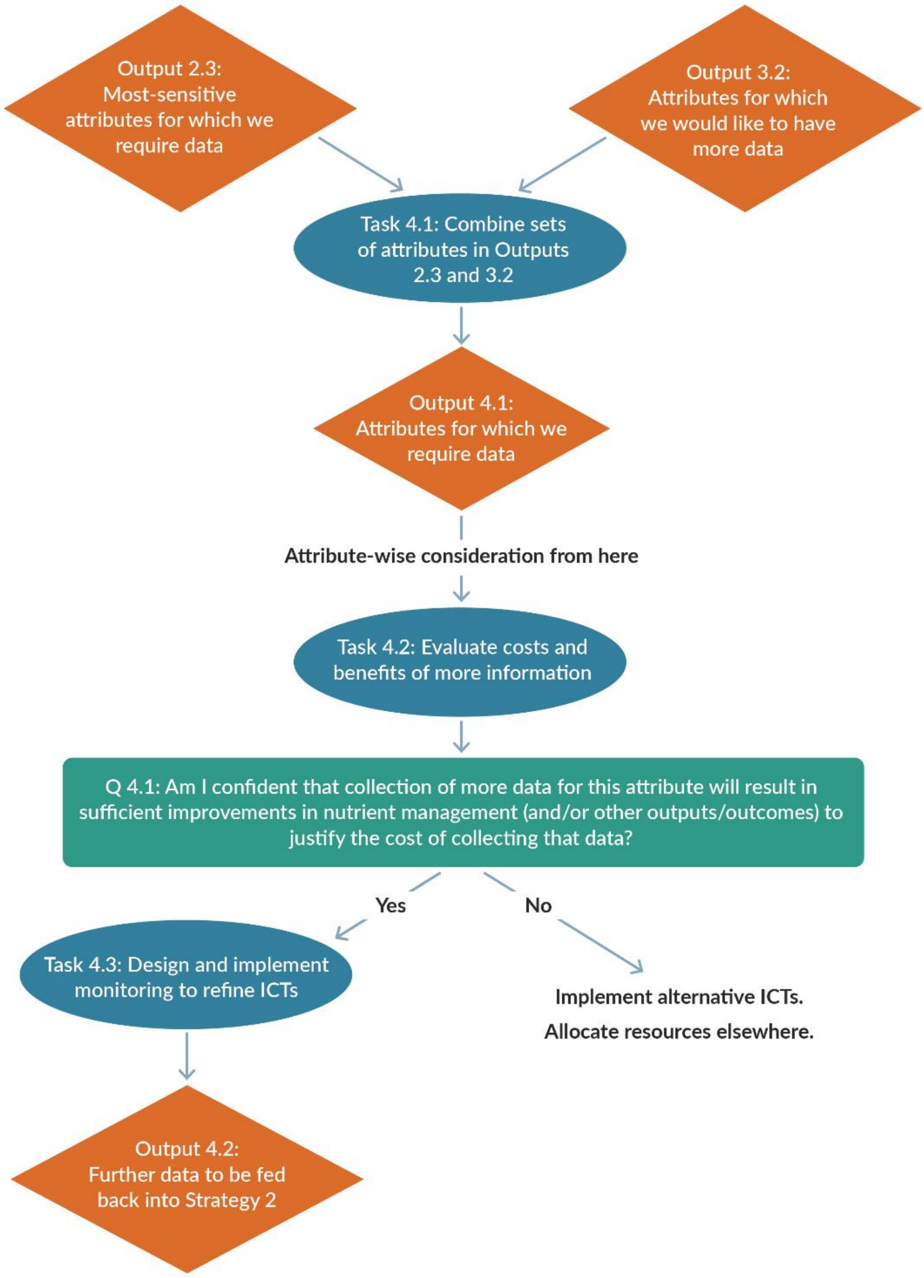
The objective of Strategy 4 is to evaluate whether collecting more data to refine ICTs of an attribute justifies the data collection cost and, if it does, then design and implement monitoring to obtain that data.

After exploring Strategies 2 and 3, we may conclude that (a) we require ICTs for particular attributes; and (b) we have insufficient data — nationally, regionally or both — to model ICTs for those attributes. In this case, we may opt for designing an adaptive monitoring programme to collect the data we require to develop and/or refine ICTs for a specific attribute over time.

Implementing Strategy 4 involves up to three tasks (circles), two outputs (diamonds; not including the two outputs from other strategies that are inputs to Strategy 4) and one decision/question (rectangle; Figure 3-9):

1. **Task 4.1** is simple and involves combining the sets of attributes within Outputs 2.3 and 3.2 (see Strategies 2 and 3, respectively).
2. **Output 4.1** is a single set of attributes for which we require further data.
3. Following collation of **Output 4.1**, we then proceed to implement Strategy 4 in an attribute-wise fashion (one attribute at a time).
4. **Task 4.2** is aimed at generating the information we need to answer Question 4.1. Task 4.2 involves careful analysis of the costs and benefits associated with allocating resources to collecting more data to develop/refine ICTs *versus* allocating those resources to other NPS-FM implementation activities.
5. **Question 4.1** is difficult to answer but it must be included as a check on any decision that involves further monitoring, given the cost of monitoring and the implications for FA2, as well as the inevitable trade-offs we would be making concerning investment in other regional priorities.
6. If the answer to Question 4.1 is 'yes' then **Task 4.3** involves designing and implementing monitoring to refine nutrient-attribute relationships and ICT. We do not expand on Task 4.3 in this guidance, as design of nutrient-attribute monitoring programmes is outside the scope (see Section 1.2).
7. Additional monitoring of nutrient-attribute relationships would yield data (**Output 4.2**) to feed back into Strategy 2 and/or 3.
8. If the answer to Question 4.1 is 'no' then councils have — after completing **Task 4.2** — decided to allocate resources to other aspects of NPS-FM implementation or their broader regional objectives.

Figure 3-9: Process for implementing Strategy 4 (See text for explanation.)



Advantages

- Task 4.2 encourages careful consideration of the costs and benefits of further data collection and so may facilitate more efficient use of resources available for NPS-FM implementation.
- Further nutrient-attribute data to facilitate meeting certain means aims (Figure 3-6), hence FA1.
- Reduction of epistemic and ontological uncertainties about how instream nutrients affect freshwater ecosystems.

Uncertainties and trade-offs

Unless we decide that further data collection is strictly necessary, implementing Strategy 4 may erode our ability to meet FA2. However, Task 4.2 should minimise unnecessary expenditure.

It is often difficult to accurately estimate the expected gain in management efficacy arising from further data collection. That is, there will likely be great uncertainties about how (in our case) a refinement of ICTs will improve on-ground nutrient management, hence nutrient concentrations in rivers and, in turn, freshwater values. These uncertainties make it difficult to estimate the expected benefits of collecting more information.

Allocating resources to monitoring often comes at substantial cost to other policy implementation activities.

Allocating time and personnel to reducing uncertainties about ICTs may delay taking on-ground management action to reduce nutrient input into rivers (ie, inconsistent with Clause 1.6 of the NPS-FM).

Application and recommendations

Task 4.1 is self-explanatory.

Task 4.2 potentially requires its own guidance document. We recommend application of the PrOACT framework to Task 4.2 and Question 4.1 (Gregory & Keeney, 2002; Keeney, 2004). However, application of the PrOACT framework — and decision analysis more broadly — to Task 4.2 is beyond the scope of the present guidance. The Problems and Objectives (Aims) elements of the PrOACT framework applied to Task 4.2 extend well beyond those presented in Section 3.1 of this guidance. Consequently, the Alternatives, Consequences and Trade-off elements of PrOACT applied to Task 4.2 also extend beyond the scope of Clause 3.13 and this guidance. Application of the PrOACT framework to Task 4.2 would involve councils considering (a) their annual budget for NPS-FM implementation; (b) their NPS-FM investment priorities as a whole (beyond Clause 3.13 and nutrient management); and (c) the consequences and trade-offs associated with alternative regional investment strategies.

To complete Task 4.2 and answer Question 4.1 it will be critical to frame the problem correctly. After employing Strategies 1, 2 and/or 3, you may arrive at Strategy 4 with a very limited amount of information, namely: *I have identified attributes for which I do not have ICT, and for which there are insufficient data to estimate ICT*. That is, the problem is currently framed as:

- There are attributes for which we do not have ICTs nor sufficient data to estimate ICT.

You cannot complete Task 4.2 and answer Question 4.1 when the problem is framed this way. Reframe as:

- *Question 4.2*: Given the ICTs we already have (see Strategies 1–3), to what extent will we observe improved ecosystem health by extending our monitoring of nutrient-attribute relationships to refine ICT?

Question 4.2 is part of Task 4.2. To answer Question 4.2 you would need to consider further questions, such as (Li et al, 2017; Runge et al, 2011; Wintle et al, 2010):

- **To what extent will new ICTs change nutrient use limits and action plans?** For example, suppose that nutrient concentrations in all nutrient-affected rivers are higher than those in the ICTs we already have. In this case, councils already know they need to implement limits on resource use to reduce riverine nutrient inputs. Councils may also need to facilitate on-ground remediation activities with farmers. You could argue that, in this case, (a) having new ICTs is unlikely to change resource use limits and action plans; and (b) allocating resources to refining existing ICT or developing new ICT, is unjustified at this stage of policy implementation (under the assumption that resources are limited).

- **What are the major uncertainties about how we can achieve our broadest fundamental aims?** When it comes to achieving Te Mana o te Wai and improved ecosystem health through nutrient management where do the key uncertainties lie? Are we most uncertain about the precise values of ICT, or more uncertain about how any given resource use limit and on-ground management action affects nutrient inputs to rivers? If the latter, then perhaps we should be allocating monitoring to reduce uncertainty around land-water nutrient interactions, rather than refinement of ICTs *per se*.

- **What is the approximate cost of monitoring to reduce uncertainty about attribute-specific ICT?**
  It is possible that, in certain circumstances, implementing monitoring to reduce uncertainty about ICTs of an attribute is cost effective. For example, consider the case where (a) we have no ICTs for an attribute deemed to be most sensitive to nutrients within an important river type; and (b) most of the data required for refining that attribute's ICTs is already required for some other purpose (eg, reporting requirements of regional plans). In such cases, the decision to implement the monitoring may be a 'no-brainer' (sensu Hemming et al, 2022)

If the scope of completing Task 4.2 extends beyond the scope of the current guidance how then should councils implement Strategy 4? At a minimum, we recommend councils implement the PrOACT framework in light of clear (highest-level) objectives of NPS-FM policy implementation. Even a qualitative implementation of the PrOACT framework — like we have outlined in this guidance — can guard against some common decision traps (Gregory et al, 2012; Hammond et al, 1998).

Quantitative, decision-analysis frameworks are available to complete Task 4.2: *Value of Information* (VoI) analysis allows us to make defensible and smarter decisions about when to invest in further data collection (Canessa et al, 2015; Nicol et al, 2018; Runge et al, 2011). Value of Information analysis comprises more advanced statistical methods and would require high levels of capability in councils or partnering with specialist science providers.

Towards improving efficiency and meeting FA2 of this guidance, we recommend a coordinated approach to Task 4.2 and the broader problem of deciding when and how to allocate resources to further monitoring under the NPS-FM. For example, using Strategy 4 as a case study, it may be worthwhile having a single research team implement PrOACT (including VoI) to Task 4.2, such that councils can examine that case study and decide whether they wish to apply similar processes and tools to the broader set of decision problems centred on when to collect further information.

3.4.5. Encouraging a phased approach

Implementing environmental policy is extremely challenging. We recommend not getting too far ahead of where you need to be when it comes to implementing Clause 3.13 and instead taking a phased approach, that is, one step at a time.

Councils need to include ICTs in regional plans due for notification by December 2024 so it is practical to implement Strategy 1 in the short term. Strategy 2 arguably provides the most effective and efficient set of ICTs for meeting the fundamental aims, but how to prioritise its subsequent tasks, let alone tasks of other strategies, is challenging without first completing Task 2.2.

A major uncertainty hampering decision-making and prioritisation of activities (including tasks herein) is not knowing the extent to which fiscal resources will be limiting.

The authors of this report and MfE recommend completing Tasks 2.1 and 2.2 as a next step in implementing Clause 3.13. If resources permit, a national modelling approach to Task 3.2 will complement Tasks 2.1 and 2.2 well. We have recommended (refer to Section 3.4.2 and 3.4.3) these tasks be completed by a single research team working collaboratively with councils, towards meeting both FA1 and FA2. Work on these tasks could begin during 2022.

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5. Glossary of abbreviations and terms

attribute	In this document, the term attribute is reserved for NPS-FM attributes only.
benthic algae	The living algae attached to, or associated with, inorganic substrata on streambeds. It is also the primary constituent of periphyton and quantified by measurements of chlorophyll *a* in samples collected from known areas.
biomass	Refers to levels of periphyton abundance. In the NPS-FM periphyton attribute, biomass is quantified as milligrams of chlorophyll *a* per square metre of riverbed.
ecosystem constituent	The biological assemblages and processes in rivers that represent attributes in the NPS-FM that are affected by nutrients.
epistemic uncertainty	Uncertainty arising from lack of knowledge about the basic causal mechanisms by which — in this case — nutrients affect river ecology.
fatty acids	Fatty acids are lipids (fats), some of which are essential requirements for macroinvertebrates and must be obtained through their diets (eg, from benthic algae or terrestrial detritus).
fundamental aim	In the context of the NPS-FM and river ecosystems, fundamental aims represent relevant, nutrient-affected values (compulsory and other; refer appendix 1 of the NPS-FM) and are the environmental outcomes (refer clause 3.9 of the NPS-FM) we wish to achieve through implementing clause 3.13. They may go beyond desired environmental outcomes and include minimising constraints that hamper our ability to meet environmental outcomes relevant to clause 3.13, as well as outcomes relevant to other clauses of the NPS-FM.
heterotrophic microbes	Bacteria and fungi that break down dead and/or dying plant material within the river. These organisms are included in the definition of periphyton.
macrophyte	Aquatic plants that grow in or on water. Macrophyte types include emergent (ie, rooted in the sediment with part of the plant above the water surface), submerged (ie, growing entirely below the water) and floating (ie, the entire plant lives at the water surface).
means aim	In the context of the NPS-FM and river ecosystems, means aims are those we may need to meet to achieve our fundamental aims. They are a ‘means to an end’. Failure to meet a means aim may be of little concern if our fundamental aims are met, yet the converse is not true.
model parameterisation	The process of choosing variables to include in a model. Changing the values of those variables changes the value of the thing being modelled.

ICTs (nutrient exceedance criterion)	A statistical rule describing how time series on the measurements of nutrient concentrations are summarised over time. This is combined with a concentration (numeric value) that delineates concentrations that are likely to cause a specified ecological condition to deteriorate from concentrations likely to prevent deterioration of that value.
ontological uncertainty	Uncertainty arising from methodological or statistical factors such as errors in measurement, imperfect parameter estimation (bias and variance) or the choice of modelling approach.
periphyton	The organic material associated with rocks and other stable substrates in streams, comprising a complex community of autotrophs (primary producers – algae, including cyanobacteria) and heterotrophic microbes such as fungi and bacteria.
PrOACT framework	PrOACT (Problem, Objectives, Alternatives, Consequences and Trade-offs) is a framework for decision-making used by environmental scientists.
stoichiometry	The elemental composition of organisms. The study of how the elemental composition of organisms affects ecological processes (particularly consumer-resource interactions) is called ecological stoichiometry.

6. References

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7. Appendix A: Attribute tables

Tables reproduced from NPS-FM appendix 2A (Attributes requiring limits on resource use) and appendix 2B (Attributes requiring action plans). Tables are presented in the order in which they are discussed in Section 2.3. The table numbers refer to those in the NPS-FM.

Appendix 2A

Table 2 — Periphyton (trophic state)

Value (and component)	**Ecosystem health (Aquatic life)**	
Freshwater body type	Rivers	
Attribute unit	mg chl-*a*/m^2 (milligrams chlorophyll-*a* per square metre)	
Attribute band and description	**Numeric attribute state (default class)**	**Numeric attribute state (productive class)**
	Exceeded no more than 8% of samples	Exceeded no more than 17% of samples
A Rare blooms reflecting negligible nutrient enrichment and/or alteration of the natural flow regime or habitat.	≤50	≤50
B Occasional blooms reflecting low nutrient enrichment and/or alteration of the natural flow regime or habitat.	>50 and ≤120	>50 and ≤120
C Periodic short-duration nuisance blooms reflecting moderate nutrient enrichment and/or moderate alteration of the natural flow regime or habitat.	>120 and ≤200	>120 and ≤200
National bottom line	**200**	**200**
D Regular and/or extended-duration nuisance blooms reflecting high nutrient enrichment and/or significant alteration of the natural flow regime or habitat.	>200	>200

At low-risk sites, monitoring may be conducted using visual estimates of periphyton cover. Should monitoring based on visual cover estimates indicate that a site is approaching the relevant periphyton abundance threshold, monitoring should then be upgraded to include measurement of chlorophyll-*a*.

Classes are streams and rivers defined according to types in the River Environment Classification (REC). The productive periphyton class is defined by the combination of REC "Dry" climate categories (that is, Warm-Dry (WD) and Cool-Dry (CD)) and REC Geology categories that have naturally high levels of nutrient enrichment due to their catchment geology (that is, Soft-Sedimentary (SS), Volcanic Acidic (VA) and Volcanic Basic (VB)). Therefore, the productive category is defined by the following REC defined types: WD/SS, WD/VB, WD/VA, CD/SS, CD/VB, CD/VA. The default class includes all REC types not in the productive class.

Based on a monthly monitoring regime. The minimum record length for grading a site based on periphyton (chlorophyll-*a*) is three years.

Appendix 2B

Table 14 – Macroinvertebrates (1 of 2)

Value (and component)	**Ecosystem health (Aquatic life)**	
Freshwater body type	Wadeable rivers	
Attribute unit	Macroinvertebrate Community Index (MCI) score; Quantitative Macroinvertebrate Community Index (QMCI) score	
Attribute band and description	**Numeric attribute states**	
	QMCI	MCI
A Macroinvertebrate community, indicative of pristine conditions with almost no organic pollution or nutrient enrichment.	≥6.5	≥130
B Macroinvertebrate community indicative of mild organic pollution or nutrient enrichment. Largely composed of taxa sensitive to organic pollution/nutrient enrichment.	≥5.5 and <6.5	≥110 and <130
C Macroinvertebrate community indicative of moderate organic pollution or nutrient enrichment. There is a mix of taxa sensitive and insensitive to organic pollution/nutrient enrichment.	≥4.5 and <5.5	≥90 and <110
National bottom line	**4.5**	**90**
D Macroinvertebrate community indicative of severe organic pollution or nutrient enrichment. Communities are largely composed of taxa insensitive to inorganic pollution/nutrient enrichment.	<4.5	<90

MCI and QMCI scores to be determined using annual samples taken between December and March (inclusive) with either fixed counts with at least 200 individuals or full counts, and with current state calculated as the five-year median score. All sites for which the deposited sediment attribute does not apply, whether because they are in river environment classes shown in table 25 in appendix 2C or because they require alternate habitat monitoring under clause 3.25, are to use soft-sediment sensitivity scores and taxonomic resolution as defined in table A1.1 in Clapcott et al (2017) *Macroinvertebrate metrics for the National Policy Statement for Freshwater Management*, Cawthron Institute: Nelson, New Zealand (see clause 1.8).

MCI and QMCI to be assessed using the method defined in Stark JD and Maxted JR (2007) *A user guide for the Macroinvertebrate Community Index*. Cawthron Institute: Nelson, New Zealand (See Clause 1.8), except for sites for which the deposited sediment attribute does not apply, which require use of the soft-sediment sensitivity scores and taxonomic resolution defined in table A1.1 in Clapcott et al (2017) *Macroinvertebrate metrics for the National Policy Statement for Freshwater Management*, Cawthron Institute: Nelson, New Zealand (see clause 1.8).

Appendix 2B

Table 15 – Macroinvertebrates (2 of 2)

Value (and component)	**Ecosystem health (Aquatic life)**
Freshwater body type	Wadeable rivers
Attribute unit	Macroinvertebrate Average Score Per Metric (ASPM)
Attribute band and description	**Numeric attribute states ASPM score**
A Macroinvertebrate communities have high ecological integrity, similar to that expected in reference conditions.	≥0.6
B Macroinvertebrate communities have mild-to-moderate loss of ecological integrity.	<0.6 and ≥0.4
C Macroinvertebrate communities have moderate-to-severe loss of ecological integrity.	<0.4 and ≥0.3
National bottom line	**0.3**
D Macroinvertebrate communities have severe loss of ecological integrity.	<0.3

ASPM scores to be determined using annual samples taken between December and March (inclusive) with either fixed counts, with at least 200 individuals, or full counts and with current state calculated as the five-year median score. All sites for which the deposited sediment attribute does not apply, whether because they are in river environment classes shown in table 25 in appendix 2C or because they require alternate habitat monitoring under clause 3.25, are to use soft-sediment sensitivity scores and taxonomic resolution as defined in table A1.1 in Clapcott et al (2017). *Macroinvertebrate metrics for the National Policy Statement for Freshwater Management*, Cawthron Institute: Nelson, New Zealand (see clause 1.8).

When normalising scores for the ASPM, use the following minimums and maximums: %EPT-abundance (0-100), EPT-richness (0–29), MCI (0–200) using the method of Collier, K (2008). *Average score per metric: An alternative metric aggregation method for assessing wadeable stream health*. New Zealand Journal of Marine and Freshwater Research, 42:4, 367–378, DOI: 10.1080/00288330809509965 (see clause 1.8).

Appendix 2B

Table 13 – Fish (rivers)

Value (and component)	**Ecosystem health (Aquatic life)**
Freshwater body type	Wadeable rivers
Attribute unit	Fish Index of Biotic Integrity (FIBI)
Attribute band and description	**Numeric attribute state (average)**
A High integrity of fish community. Habitat and migratory access have minimal degradation.	≥34
B Moderate integrity of fish community. Habitat and/or migratory access are reduced and show some signs of stress.	<34 and ≥28
C Low integrity of fish community. Habitat and/or migratory access is considerably impairing and stressing the community.	<28 and ≥18
D Severe loss of fish community integrity. There is substantial loss of habitat and/or migratory access, causing a high level of stress on the community.	<18

Sampling is to occur at least annually between December and March (inclusive) following the protocols for at least one of the following methods: backpack electrofishing, spotlighting, or trapping as set out in Joy M, David B and Lake M (2013), *New Zealand Freshwater Fish Sampling Protocols (Part 1): Wadeable rivers and streams,* Massey University: Palmerston North, New Zealand (see clause 1.8).

The F-IBI score is to be calculated using the general method defined by Joy, MK and Death RG (2004), Application of the Index of Biotic Integrity Methodology to New Zealand Freshwater Fish Communities. *Environmental Management*, 34(3), 415–428 (see clause 1.8).

Appendix 2A

Table 7 – Dissolved oxygen

Value (and component)	**Ecosystem health (Water quality)**	
Freshwater body type	Rivers (below point sources only)	
Attribute unit	mg/L (milligrams per litre)	
Attribute band and description	**Numeric attribute state**	
	7-days* mean minimum (summer period: 1 November to 30 April)	1-day* minimum (summer period: 1 November to 30 April)
A No stress caused by low dissolved oxygen on any aquatic organisms that are present at matched reference (near-pristine) sites.	≥8.0	≥7.5
B Occasional minor stress on sensitive organisms caused by short periods (a few hours each day) of lower dissolved oxygen. Risk of reduced abundance of sensitive fish and macroinvertebrate species.	≥7.0 and <8.0	≥5.0 and <7.5
C Moderate stress on a number of aquatic organisms caused by dissolved oxygen levels exceeding preference levels for periods of several hours each day. Risk of sensitive fish and macroinvertebrate species being lost.	≥5.0 and <7.0	≥4.0 and <5.0
National bottom line	**5.0**	**4.0**
D Significant, persistent stress on a range of aquatic organisms caused by dissolved oxygen exceeding tolerance levels. Likelihood of local extinctions of keystone species and loss of ecological integrity.	<5.0	<4.0

*The 7-day mean minimum is the mean value of 7 consecutive daily minimum values.

*The one-day minimum is the lowest daily minimum across the whole summer period.

Appendix 2B

Table 17 – Dissolved oxygen

Value (and component)	**Ecosystem health (Water quality)**	
Freshwater body type	Rivers	
Attribute unit	mg/L (milligrams per litre)	
Attribute band and description	**Numeric attribute state**	
	7-days* mean minimum	1-day* minimum
A No stress caused by low dissolved oxygen on any aquatic organisms that are present at matched reference (near-pristine) sites.	≥8.0	≥7.5
B Occasional minor stress on sensitive organisms caused by short periods (a few hours each day) of lower dissolved oxygen. Risk of reduced abundance of sensitive fish and macroinvertebrate species.	≥7.0 and <8.0	≥5.0 and <7.5
C Moderate stress on a number of aquatic organisms caused by dissolved oxygen levels exceeding preference levels for periods of several hours each day. Risk of sensitive fish and macroinvertebrate species being lost.	≥5.0 and <7.0	≥4.0 and <5.0
National bottom line	**5.0**	**4.0**
D Significant, persistent stress on a range of aquatic organisms caused by dissolved oxygen exceeding tolerance levels. Likelihood of local extinctions of keystone species and loss of ecological integrity.	<5.0	<4.0

*The seven-day mean minimum is the mean value of 7 consecutive daily minimum values.

*The one-day minimum is the lowest daily minimum across the whole summer period.

Appendix 2B

Table 21 — Ecosystem metabolism (both gross primary production and ecosystem respiration)

Value (and component)	**Ecosystem health (Ecosystem processes)**
Freshwater body type	Rivers
Attribute unit	g O_2 m^{-2} d^{-1} (grams of dissolved oxygen per square metre per day)

Derived from at least seven days of continuous dissolved oxygen monitoring to be collected at least once during summer (December to March inclusive), using the method of Young RG, Clapcott JE, Simon K (2016), Ecosystem functions and stream health, *Advances in New Zealand Freshwater Science*, NZ Freshwater Sciences Society, NZ Hydrological Society (see clause 1.8).

Method	Assessment Parameter	Criterion	Assessment
Absolute value	GPP at test site (g O_2/m^2/day)	GPP < 4.0	Healthy
		GPP = 4.0–8.0	Satisfactory
		GPP > 8.0	Poor
	ER at test site (g O_2/m^2/day)	ER = 1.5–5.5	Healthy
		ER = 0.7–1.5 or 5.5–10.0	Satisfactory
		ER < 0.7 or >10.0	Poor
	P/R at test site	P/R < 1.3	Healthy
		P/R = 1.3–2.5	Satisfactory
		P/R > 2.5	Poor
Comparison with reference	Ratio of GPP at test (GPP_t) and reference (GPP_r) sites	GPP_t:GPP_r = 0.4–1.5	Healthy
		GPP_t:GPP_r = 0.1 0.4 or 1.5–3.0	Satisfactory
		GPP_t:GPP_r = <0.1 or >3.0	Poor
	Ratio of ER at test (ERt) and reference (ER_r) sites	ER_t: Er_r = 0.4–1.4	Healthy
		ER_t: Er_r = 0.2–0.4 or 1.4–2.5	Satisfactory
		ER_t: Er_r = < 0.2 or > 2.5	Poor

8. Appendix B: Tables of nutrient thresholds from the literature

Table B-1: Periphyton nutrient thresholds from the literature

Note that all the criteria shown are for TN and TP.

Region	TN criteria (mg/m^3)	TP criteria (mg/m^3)	Target or state	Reference and notes
Canada, Atlantic maritime	870–1200	10–30	80% prediction interval, 100 mg/m^2 chl *a* (mean summer values)	(Chambers et al, 2012) Five approaches (percentile or relationships) used to identify “impaired” sites. Criteria developed for different regions
Canada, Montane cordillera	210	20		
Canada, Mixedwood plains	~1100	~30		
Canada, interior prairies	390–980	~10		
Global (North America, NZ data)	700	25	60 (max), 20 (mean) mg/m^2 chl *a*	(Dodds et al, 1998) Derived thresholds separating oligo-meso- and eutrophic streams
	1500	75	200 (max), 70 (mean) mg/m^2 chl *a*	
Global and USA dataset	537, 515	43, 27	mean chl *a*, wide range, not specified	(Dodds et al, 2002; 2006) Breakpoints assumed to indicate saturation effect
	602, 367	62, 27	maximum chl *a*, wide range not specified	
Eastern Canada	780, 1590, 1850	66, 114, 73	Eutrophic	(Lavoie et al, 2014) Derived criteria for increasing pH, to define four trophic states, based on diatom community composition
	700, 890, 1360	61, 52, 58	Meso-eutrophic	
	300, 530, 1330	24, 26, 28	Mesotrophic	
	290, 360, 790	22, 16, 24	Oligotrophic	
Ohio, USA	435*	38	107–182 mg/m^2, seasonal average	(Miltner, 2010) Based on change points in relationship

Table B-2: Macroinvertebrate nutrient thresholds from the literature

Region	TN or DIN* criteria (mg/m³)	TP or DRP* criteria (mg/m³)	Target or state	Reference and notes
Manawatu-Whanganui, New Zealand	<500		Point at which MCI, %EPT taxa and EPT richness ceased to respond	(Wagenhoff, Liess et al, 2017) Impact cessation threshold for community metrics after negative response
	<500 (150)		Point at which significant macroinvertebrate assemblage turnover subsided (and peak of turnover)	(Wagenhoff et al, 2017) Assemblage threshold range within which significant turnover of multiple taxa occur
	<500		Point at which multiple metrics of food web function had ceased to respond or show inflection	(Canning & Death, 2021) Study of food web stability and function including community respiration via energy flows in riverine networks
Southland, New Zealand	144*		Inflection point of subsidy-stress relationship for %EPT abundance	(Wagenhoff et al, 2011) Subsidy-stress relationship only at lower levels of deposited fine sediment
New Zealand	600*	20*	Nutrient criteria to support NPS-FM national bottom-line targets for MCI, QMCI and ASPM	(Canning et al, 2021) Minimisation-of-mismatch analysis of national dataset
New Zealand	10*, 330*, 1470*	1*, 9*, 28*	Nutrient criteria to support NPS-FM A/B, B/C, and C/D (bottom line) macroinvertebrate attribute thresholds, respectively	(Canning, 2020) Based on linear regression relationships for MCI, QMCI and ASPM in national dataset
European Union		16–105*, 40–170*	Possible range of nutrient boundary values for very large rivers to support the High/Good and Good/Moderate ecological status boundaries (invertebrates), respectively	(Phillips et al, 2018) Geographical intercalibration dataset covering data from multiple EU member states
Central Europe	30–120 (NH_4N)	70–110	Thresholds relating to ecological status as determined by macroinvertebrate communities	(Kail et al, 2012) Use of various statistical methods including those that investigate community metrics and multiple responses of individual taxa
Minnesota, USA		42–233 (135)	Range (mean) of state-wide threshold concentrations for relationships between TP and macroinvertebrate and fish metrics	(Heiskary & Bouchard Jr, 2015) Based on breakpoints and changepoints in relationships; final TP criteria given based on weight-of-evidence of a variety of approaches (50, 100 and 150 for three regions, respectively)
New York State, USA	300*, 700	30	Nutrient thresholds based on shifts in macroinvertebrate	(Smith & Tran, 2010)

Region	TN or DIN* criteria (mg/m³)	TP or DRP* criteria (mg/m³)	Target or state	Reference and notes
			and diatom community structure	Weighted average of results from different statistical approaches
USA, several states	610–1920	40–150	Range of estimated threshold values for various macroinvertebrate metrics	(Evans-White et al, 2013) Reviews criteria determined from three studies using different statistical methods (including from Evans-White, Dodds, Huggins et al, 2009)
Central Plains, USA		41–150	Breakpoints or thresholds in relationship between macroinvertebrate richness and TP	(Dodds et al, 2010) Estimate varied three-fold depending on statistical method used; data from Evans-White et al 2009 (see above)
Everglades, USA		14.5	Community changepoint of negatively responding macroinvertebrate indicator taxa	(Baker and King, 2010) Threshold indicator taxa analysis (TITAN)
Germany	~900 (371–4468) (NO_3N)	~50* (4–252)	Median (range) change point of macroinvertebrate taxa showing negative responses	(Sundermann et al, 2015) TITAN analysis to report change points of various macroinvertebrate taxa to stressor gradients
Canada	590–2830	21–63	Range of change points derived from relationships between nutrients and invertebrate metrics including EPT metrics	(Chambers et al, 2012) Final nutrient criteria proposed were based on nutrient-invertebrate and nutrient-algal relationships as well as on chemically based methods. These were 870–1200 for TN and 10–30 for TP
Ohio, USA	440*	40	Protection of macroinvertebrate communities	(Miltner, 2010) Based on change points in relationships between nutrients and secondary response indicators dissolved oxygen and benthic chlorophyll and macroinvertebrate indicators
New Zealand, mesocosm experiment	728*	70*	Inflection point of subsidy-stress relationship for EPT metrics and for total taxon richness	(Wagenhoff et al, 2012) Threshold values treated with caution due to temporal and spatial limitations of such experiments (experimental values expected to be higher than in real streams)