BEFORE THE ENVIRONMENT COURT

I MUA I TE KOOTI TAIAO O AOTEAROA **ENV-2018-CHC-000037**

IN THE MATTER	of the Resource Management Act 1991
AND	of appeals under Clause 14 of the First Schedule of the Act
BETWEEN	**SOUTHLAND FISH AND GAME COUNCIL** **Appellant**
AND	**SOUTHLAND REGIONAL COUNCIL** **Respondent**

STATEMENT OF EVIDENCE OF RUSSELL GEORGE DEATH ON BEHALF OF SOUTHLAND FISH AND GAME COUNCIL

Dated: 15 February 2019

Judicial Officers: Judge Borthwick and Judge Hassan

COUNSEL for SOUTHLAND FISH AND GAME COUNCIL:

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1. QUALIFICATIONS AND EXPERIENCE

1.1 My full name is Russell George Death

1.2 I am a Professor in Freshwater Ecology in the School of Agriculture and Environment – Ecology at Massey University where I have been employed since 1993. Prior to that I received a Doctor of Philosophy in Zoology from the University of Canterbury (1991) and was a Foundation for Research, Science and Technology postdoctoral fellow at Massey University (1991-93).

1.3 I have been a Quinney Visiting Fellow at Utah State University, USA and an International Distinguished Visiting Fellow at the Institute of Advanced Studies at the University of Birmingham, UK. I was awarded the 2017 New Zealand Freshwater Sciences Society Medal for an outstanding contribution to our understanding and management of freshwaters.

1.4 I have had twenty five years' experience in professional ecology research, teaching and management. My area of expertise is the ecology of stream invertebrates and fish. I have more than 110 peer-reviewed publications in international scientific journals and books, including 6 invited reviews. I have written more than 45 consultancy reports and given over 80 conference presentations. I have been the principal supervisor for 42 post-graduate research students. I have been researching the invertebrates, periphyton and fish of the lower North Island streams and rivers for the past twenty years.

1.5 I am a member of the Ecological Society of America, the New Zealand Freshwater Sciences Society and the Society for Freshwater Science. I have refereed scientific manuscripts for more than 30 scientific journals and numerous books. I am on the editorial board of the international journal *Freshwater Science*. I am a member of the management committee for One Health Aotearoa, an alliance of New Zealand's leading infectious diseases researchers.

1.6 I have been commissioned by a number of governmental and commercial organisations to provide scientific advice on matters related to the management of freshwater resources. I have provided expert evidence at a variety of resource consent, regional plan, Environment Court and EPA hearings. These include the Canterbury and Greater Wellington Regional

Plan Council-level hearings, Horizons One Plan Environment Court case[1], and the Ruataniwha EPA hearing[2]. I am currently on the Governmental Science and Technical Advisory Group for informing the development of a new National Policy Statement for Freshwater Management.

1.7 As part of my research, in the past I have sampled a large number of streams in the Southland Region for stream invertebrates and associated water quality parameters.

1.8 I have read the Environment Court's Code of Conduct for Expert Witnesses 2014, and I agree to comply with it. I confirm that the issues addressed in this brief of evidence are within my area of expertise. I have not omitted to consider material facts known to me that might alter or detract from the opinions expressed. I have specified where my opinion is based on limited or partial information and identified any assumptions I have made in forming my opinions.

1.9 In preparing my evidence I have reviewed evidence of Roger Hodson, Ewan Rodway and Antonius Snelder (for Environment Southland) and the evidence of Kate MacArthur (for the Royal Forest and Bird Protection Society).

2. SCOPE OF EVIDENCE

2.1 My evidence relates to the following provisions of the Proposed Southland Water and Land Plan (pSWLP):

- Objective 7: *"Any further over-allocation of freshwater (water quality and quantity) is avoided and any existing over-allocation is phased out in accordance with freshwater objectives,*

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[1] *Day v Manawatu Wanganui Regional Council* [2012] NZEnvC 182.

[2] Decision of Board of Inquiry appointed under section 149J of the RMA to consider a plan change request and applications for Notice of Requirement and Resource Consent by Hawkes Bay Regional Council and Hawkes Bay Regional Investment Company in relation to the Tukituki Catchment Proposal, 2014.

freshwater quality limits and timeframes established under the Freshwater Management Unit processes."

- Policy 45: *"Priority of FMU values, objectives, policies and rules"*;[3] and

- Policy 47: *"FMU processes"*.[4]

2.2 I have been asked to provide an opinion on whether Region-wide numerical outcomes can be set, for water quality in the Southland Region, to provide for "ecosystem health", based on current levels of information. This involves:

- an explanation of the meaning of "ecosystem health", referring to the definition in the NPSFM.

- the essential parameters necessary to protect ecosystem health in the streams and rivers of Southland, with a focus on water quality.

- an explanation of why the water quality numerics I propose for water body classes are necessary for maintaining ecosystem health; and

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[3] Fish & Game appeal point to add the words *"the provision in the relevant FMU Section of this plan is not more lenient or less protective of water quality, quantity or aquatic ecology than the Region-wide Objectives and Region-wide Policies"*.

[4] Fish & Game appeal point to amend as follows (insert underlining): *"The FMU sections will <u>support the implementation of region wide objectives by:</u>*
1. identify<u>ing</u> values and establish<u>ing specific</u> freshwater objectives for each Freshwater Management Unit, including where appropriate at a catchment or sub-catchment level, having particular regard to the national significance of Te Mana o te Wai, and any other values developed in accordance with Policies CA1-CA4 and Policy D1 of the National Policy Statement for Freshwater Management 2014 (as amended in 2017); and
2. set water quality and water quantity limits and targets to achieve the <u>region wide and specific</u> freshwater objectives; and
3. set methods to phase out any over-allocation, within a specified timeframe; and
4. assess water quality and quantity taking into account Ngai Tahu indicators of health."

- the proportion of river and stream reaches that currently have water quality characteristics that do not meet the numerical outcomes that I propose (presented in Table 5).

2.3 I am aware there is some dispute over the use of the term “over allocation”, prior to FMU processes occurring. However, from a freshwater science perspective, the term can describe rivers and streams that exceed my recommended numerics.

2.4 My conclusions, showing waterbodies in the Southland Region that exceed thresholds to achieve ecosystem health, differ from Mr Hodson’s conclusions. Mr Hodson has compared current state to water quality attributes in the NPSFM, standards in the Southland Regional Water Plan (“RWP”) and the ANZECC 2000 Guidelines.[5] I explain that the NPSFM currently does not have all the attributes or parameters necessary to protect ecosystem health of rivers and streams. Further, some of the standards in the RWP (also included in Appendix E pSWLP), are inadequate and insufficient to achieve ecosystem health in the various waterbody classes to which they relate.

3 KEY FACTS AND OPINIONS

3.1 “Ecosystem health” is a narrative objective. There are different levels of ecosystem health. However I consider that measurable numeric values, for water quality parameters, can be set in Southland that describe a minimum level of “ecosystem health”. The parameters I recommend in this evidence are for deposited fine sediment, MCI, QMCI, nitrate, and dissolved reactive phosphorous (DRP).

3,2 Water quality and ecological health of streams and rivers draining agricultural land in the Southland is in many cases poor. In contrast, the water quality

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[5] Explained at [44] Hodson.

and ecological health of streams and rivers draining indigenous vegetation is high.

3.3 Where low water quality and ecological health occurs in Southland rivers and streams, there is strong evidence that it is the result of poor agricultural land use practices.

3.4 The predominant detrimental effects of agriculture on water quality are driven through increased nutrients (nitrogen and phosphorus), and deposited fine sediment levels.

3.5 Maintaining ecosystem health requires ensuring *all* the appropriate parameters

3.6 I have provided, in Table 1, scientifically robust thresholds to maintain the appropriate level of water quality and ecological health for the waterbody types identified in the pSWLP.

3.7 Table 2 presents my analysis of what proportion of stream reaches in each waterbody type would not currently meet the Table 1 thresholds for MCI, nitrate and DRP.

3.8 I have utilised the waterbody classifications contained in Appendix E of the pSWLP, which I consider to be an appropriate spatial scale until FMU processes occur. Although FMU processes may seek to refine the parameters to provide for a more improved environmental state, I have set thresholds I consider necessary to achieve the compulsory value of ecosystem health.

3.9 If ecosystem health is to be provided for, as required by the NPSFM, current water quality could not be reduced, and in many instances would need to significantly improve.

4 TERMS AND DEFINITIONS

4.1 Throughout my evidence, I use the term ‘adverse’ and ‘significant adverse’ effect interchangeably. While there may be differences in these terms within the planning and/or legal arena, they are identical in an ecological context.

4.2 I use the term "parameter" to describe different variables within the freshwater environment for which numerical states can be set (including for example deposited fine sediment, MCI, QMCI, nitrate, and dissolved reactive phosphorous).

4.3 A numeric level for each parameter, can be linked to a state. For current purposes, the *desired* state or outcome is the level at which further use of a freshwater body would likely cause it to *fail* to achieve ecosystem health. Some persons describe this state as meeting or exceeding the "assimilative capacity" of a waterbody.

4.4 I explain what I mean by "ecosystem health" in a separate section of my evidence, below.

4.5 I am aware that the terms "*freshwater objective*", "*freshwater management unit*" (FMU), "*attribute*", "*attribute state*", "*limit*" and "*target*" are contained in the NPSFM. I leave the application of those terms to the planning experts.

5 AGRICULTURAL LAND USE ACTIVITIES IN THE SOUTHLAND REGION: ADVERSE EFFECTS

5.1 Land use activities, often associated with agriculture, if not conducted appropriately can lead to a decline in ecological health of waterbodies that occur or flow through that land. As I explain later in my evidence, this includes an excessive increase in periphyton, a change in the chemical and physical characteristics of the habitat (e.g. pH, oxygen levels, substrate composition, deposited fine sediment), and a change in aquatic invertebrate communities.

5.2 These biological changes are a result of a few key driving factors that can occur with agricultural land use practices, including:[6]

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[6] Allan J.D. (2004) Landscapes and riverscapes: The influence of land use on stream ecosystems. *Annual Review of Ecology Evolution and Systematics,* **35,** 257-284, Matthaei C.D., Weller F., Kelly D.W. & Townsend C.R. (2006) Impacts of fine

- increased nutrient levels (nitrogen and phosphorous) from fertiliser use;[7]
- direct and indirect inputs to surface water from livestock;[8]
- increased light and temperature levels from riparian forest removal, changes to hydrology, and instream habitat; and
- increased deposited sediment from land disturbance including cultivation, vegetation removal and livestock access to surface waterbodies and/or riparian margins which destabilise stream banks.

5.3 Poorly managed winter fodder crops, land erosion from landslips, livestock trampling and wallowing, or cultivation on sloping ground or too close to waterways, will deposit sediment into streams to which phosphorous is bound.

5.4 In Southland, winter forage crops grazed by livestock can be significant sources of nitrogen losses to water, particularly on free-draining soil types, and of phosphorus and sediment losses *via* surface runoff from gullies and swales i.e. critical source areas (Monaghan, 2012).

5.5 Subsurface drainage is also an important flow pathway in the transfer of contaminants from agricultural land to water. In Southland artificial agricultural drainage systems are widespread and therefore particularly

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sediment addition to tussock, pasture, dairy and deer farming streams in New Zealand. *Freshwater Biology,* **51,** 2154-2172, Townsend C.R., Uhlmann S.S. & Matthaei C.D. (2008) Individual and combined responses of stream ecosystems to multiple stressors. *Journal of Applied Ecology,* **45,** 1810-1819..

[7] Application of fertiliser can inadvertently end up being applied directly into waterways or be washed into them during rain events.

[8] Livestock, if given access to waterways, have a preference for urinating and defecating directly into the waterway Bagshaw C.S. (2002) Factors influencing direct deposition of cattle faecal material in riparian zones. In: *MAF Technical Paper*, Vol. 2002/19, Wellington, Davies-Colley R.J., Nagels J.W., Smith R.A., Young R.G. & Phillips C.J. (2004) Water quality impact of a dairy cow herd crossing a stream. *New Zealand Journal of Marine and Freshwater Research,* **38,** 569-576..

important contributors of nitrogen, phosphorus, sediment and faecal microorganisms loads to water bodies (Monaghan, 2014).

5.6 The following graphic shows some of these interactions.

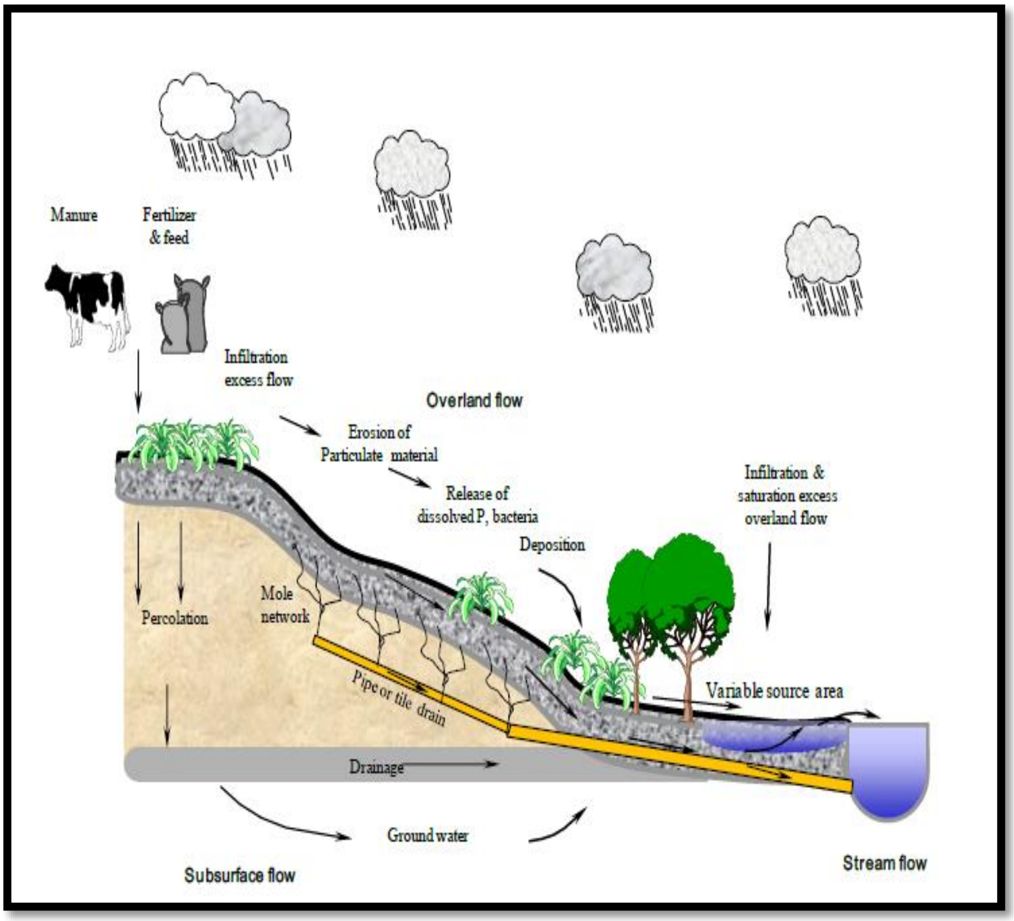

Figure 1 - Conceptual diagram of processes that transport pollutants from the landscape to surface water. From Monaghan, 2014 (as adapted from McDowell et al, 2004).

5.7 These issues are not new ones. There is a comprehensive body of scientific information dating from the 1970’s (Hynes, 1975) that details how land use activities in a catchment’s surrounding waterbodies can have a major detrimental effect on the biological communities living in those waterbodies (Allan, 2004).[9]

5.8 A document prepared for Southland Regional Council, "*Regional Scale Stratification of Southland's Water Quality – Guidance for Water and Land Management*" (March 2014)[10] found that diffuse inputs of nutrients from agricultural land contributed > 90% of total nitrogen, and > 75% of total phosphorus to many Southland rivers at low flows.

5.9 A Southland study also found deposited fine sediment was an even more pervasive stressor, and can interact synergistically with nutrients (Wagenhoff *et al.*, 2011). This means that both nutrients and fine sediment need to be managed.

5.10 Evidence provided by Mr Hodson and Ms MacArthur make it clear that the Southland Region has low water quality and poor ecological health, corresponding with areas with agricultural land use, and that in many cases, water quality is continuing to decline.

5.11 Mr Hodson relies on comparisons against surface water quality standards in the RWP to assess ecological health as well as the bands in the NPSFM and ANZECC guidelines. I suggest this may understate the level of degradation. As set out in my Table 2, the (low) standards of the RWP (and also in Appendix E pSWLP), are not sufficient to ensure ecological health.

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[9] Refer also (Quinn *et al.*, 1997; Townsend *et al.*, 1997; Townsend & Riley, 1999; Quinn, 2000; Clapcott & Goodwin, 2010; Clapcott *et al.*, 2011a; Clapcott *et al.*, 2012; Greenwood *et al.*, 2012; Julian *et al.*, 2017).

[10] Snelder T., Fraser C., Hodson R., Ward N., Rissmann C. & Hicks A. (2014) Regional Scale Stratification of Southland's Water Quality – Guidance for Water and Land Management. . Vol. C13055/02. Prepared for Southland Regional Council Aqualinc Research Limited,, Snelder T. & Ledgard G. (2014) Assessment of Farm Mitigation Options and Land Use Change on Catchment Nutrient Contaminant Loads in the Southland Region. . Vol. C13055/02. Prepared for Southland Regional Council, Aqualinc Research Limited,.

6 ECOSYSTEM HEALTH - NPSFM

6.1 Objective A1 of the NPSFM states:

> *"To safeguard*
>
> *a) the life-supporting capacity, ecosystem processes and indigenous species including their associated ecosystems, of fresh water, and*
>
> *b) the health of people and communities, as affected by contact with fresh water;*
>
> *in sustainably managing the use and development of land, and of discharges of contaminants."*
>
> (My emphasis)

6.2 Appendix 1 to the NPSFM contains "*National values and uses for fresh water*". It contains two "*Compulsory National values*". These are:

- "*Ecosystem health*"; and
- "*Human health for recreation*".

6.3 I use the words "ecosystem health", "ecological health" and "life supporting capacity" interchangeably, because they mean the same in an ecological context.

6.4 Ecosystem health comprises many components. These components are largely encapsulated in the NPSFM definition, as follows:

> "*Ecosystem health – The freshwater management unit supports a healthy ecosystem appropriate to that freshwater body type (river, lake, wetland, or aquifer).*
>
> *In a healthy freshwater ecosystem ecological processes are maintained, there is a range and diversity of indigenous flora and fauna, and there is resilience to change.*
>
> *Matters to take into account for a healthy freshwater ecosystem include the management of adverse effects on flora and fauna of contaminants, changes in freshwater chemistry, excessive nutrients, algal blooms, high sediment levels, high temperatures, low oxygen, invasive species, and changes in flow regime. Other matters to take into account include the essential habitat needs of flora and fauna and the connections between water bodies."* (My emphasis)

6.5 It can be seen that the concept of “maintenance” appears in the NPSFM definition. In a recent review (Canning & Death, in press) completed for an Ecology textbook to be published in 2019, Dr Canning and I adopted a more widely accepted definition of ecosystem health (amongst ecologists), that also contains this concept:[11]

> *“A healthy ecosystem is one that is sustainable — that is, it has the ability to <u>maintain</u> its structure (organization) and function (vigor) over time in the face of external stress (resilience).”*
> (My emphasis)

6.6 Maintenance or stability[12] is multi-faceted, but can be divided broadly into *resistance* (the ability to remain unchanged from stress) and *resilience* (the capacity and timeliness to return to pre-perturbation conditions).

6.7 I note that the concept of “resilience” also appears in the definition of “intrinsic values” in the Resource Management Act.[13]

6.8 There is no definition of “life supporting capacity” in the Resource Management Act, but as stated, from a freshwater science perspective, I consider this to be the same as the concept of “ecosystem health”.

6.9 The NPSFM Appendix 2 contains “Attribute Tables”, including attributes for the compulsory value of ecosystem health.

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[11] Based on Costanza, Norton & Haskell, 1992; Scrimgeour & Wicklum, 1996; Rapport et al., 1998; Boulton, 1999; Norris & Thoms, 1999; Friberg et al., 2011; O’Brien et al., 201.

[12] The maintenance of structure and function over time in the face of external stress, reflects the <u>stability</u> of the following elements of an ecosystem:
- Structure or organization - the assembly of a community - it includes species diversity, community composition and food web topology;
- Function or vigour - an ecosystem’s activity, such as the productivity, throughput, cycling, and metabolism.

[13] “*Intrinsic values*”, in relation to ecosystems, means those aspects of ecosystems and their constituent parts which have value in their own right, including—
(a) Their biological and genetic diversity; and
(b) The essential characteristics that determine an ecosystem's integrity, form, functioning, and <u>resilience</u>.

6.10 However, Appendix 2 currently does not have all the attributes necessary to protect ecosystem health of rivers and streams. It does outline A, B, C, and D (D = environmental bottom lines) bands and criteria for some attributes that can be used to ensure freshwater ecosystem health, within those bands.

6.11 The NPSFM lacks many of the key parameters that most freshwater ecologists would contend are important for protecting freshwater ecosystems. For example, although nitrate is included for river waterbodies, it is only included at concentrations where nitrates are toxic (Hickey and Martin 2009). Nitrates, nitrite and ammonia (NH3) can be directly toxic to many aquatic animals (Hickey & Martin, 2009), however declines in ecological health occur long before toxic levels are achieved.[14]

6.12 In the next section of my evidence, I discuss additional critical parameters to manage for ecosystem health.

6.13 I note that Appendix 2 to the NPSFM is currently under review and I am a member of the Governmental Science and Technical Advisory Group that is advising on that review.

7 CRITICAL PARAMATERS TO MANAGE FOR ECOSYSTEM HEALTH

7.1 The following diagram shows influences on the ecological community composition of a waterbody:

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[14] The NPSFM does have nitrogen and phosphorus attributes for lakes.

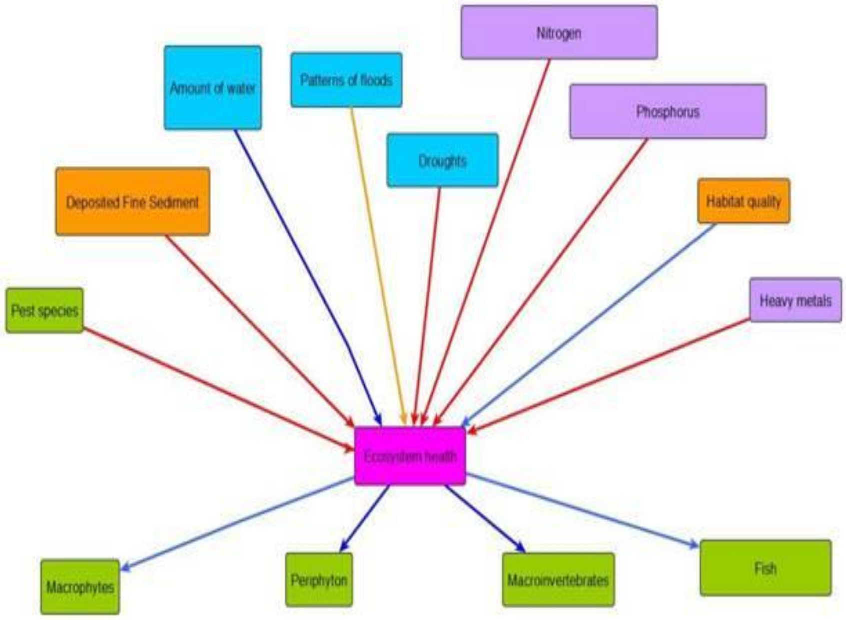

Figure 2. Conceptual diagram of the main parameters that determine ecosystem health and some of the components of ecosystem health.

7.2 In my opinion the critical parameters to manage for ecological or ecosystem health are instream habitat quality (i.e. natural character), water quantity, flow pattern (i.e. floods and droughts), nutrients (nitrogen and phosphorus), deposited fine sediment and riparian margins. If these are managed appropriately then this should lead to healthy periphyton and invertebrate communities that in turn will maintain sustainable and healthy fish populations and a healthy resilient ecosystem.

7.3 It is important to manage for all these parameters. Ecosystems are complex interconnected entities. The parameter that is most stressed or at its minimum will be the factor limiting the health of an ecosystem. For example, in a river, nutrient levels may be low, water quantity high, deposited sediment absent and habitat appropriate, but yet if arsenic levels are too high few animals will be able to survive and the ecosystem will thus be unhealthy.[15]

7.5 As aquatic ecological communities are complex ecosystems that are affected by multiple interacting stressors, the effects for ecological communities of specific management practices that focus on controlling only one of these stressors (e.g. reductions in nitrogen loadings) is difficult to predict. For example if deposited sediment levels are very high, reducing nutrient levels is unlikely to improve ecological health. Improvement in the ecological health of these waterbodies will require the management of all the interacting stressors. However, we can expect that any reductions in nutrients, deposited sediment, faecal contamination, and restriction on stock access to waterbodies will result in an improvement from the current degraded state.

7.6 There has been a considerable body of research in New Zealand over the last 25 years to investigate the numerical relationships between many of these physiochemical stressors, and ecological health.[16] I draw upon these

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[15] This is a fundamental principle in ecology - Liebig's law of the minimum Begon M., Townsend C., R. & Harper J., L. (2005) *Ecology : from individuals to ecosystems,* Blackwell Pub., Malden, MA, Dodds W.K. (2009) *Laws, Theories, and Patterns in Ecology,* University of California Press, Berkley. It states that growth is dictated **not** by the total resources available, but by the scarcest resource (limiting factor).

[16] Quinn J.M. (2000) Effects of pastoral development. In: *New Zealand Stream Invertebrates: Ecology and Implications for Management.* (Eds K.J. Collier & M.J. Winterbourn), pp. 208-229. New Zealand Limnological Society, Hamilton, Parkyn S. & Wilcock B. (2004) Impacts of agricultural land use. In: *Freshwaters of New zealand.* (Eds J. Harding & P. Mosely & C. Pearson & B. Sorrell). New Zealand Hydrological Society and New Zealand Limnological Society, Clapcott J., Young R., Goodwin E., Leathwick J. & Kelly D. (2011a) Relationships between multiple land-use pressures and individual and combined indicators of stream ecological integrity. In: *DOC Research and Development Series*, p. 57. Department of Conservation, Wellington, Clapcott J., Young R., Harding J., Matthaei C., Quinn J. & Death R. (2011b) Sediment Assessment Methods: Protocols and guidelines for assessing the effects of deposited fine sediment on in-stream values. Cawthron Institute, Nelson, Clapcott J.E., Collier K.J., Death R.G., Goodwin E.O., Harding J.S., Kelly D., Leathwick J.R. & Young R.G. (2012) Quantifying relationships between land-use gradients and structural and functional indicators of stream ecological integrity. *Freshwater Biology,* **57,** 74-90, Clapcott J., Wagenhoff A., Neale M., Death R., Storey R., Smith B., Harding J., Matthaei C., Collier K., Quinn J. & Young R. (2017) Macroinvertebrate metrics for the National Policy Statement for Freshwater Management project: Report 1. In: *Prepared*

in the next sections of my evidence, to recommend numerical measures to achieve a minimum level of ecosystem health, for different waterbody types.

8 NUMERIC ECOSYSTEM HEALTH OUTCOMES FOR THE SOUTHLAND REGION

8.1 Table 1 includes the key water quality parameters and the associated numeric that I consider are appropriate to ensure ecosystem health (as per NPSFM Objective A1), for each of the waterbody types used in the pSWLP.

8.2 Table 1 includes numerics for fine deposited sediment, nitrate, DRP, MCI, or QMCI. As the NPSFM already has Chlorophyll *a* as a compulsory attribute, I do not include periphyton.

8.3 As Mr Hodson explains, nitrogen can be in an aquatic environment in a number of forms and nitrate, nitrite, ammonium and particulate organic nitrogen are directly available for plant uptake. Although Mr Hodson[17] uses Nitrate-Nitrite-Nitrogen (NNN) as the measure that Environment Southland has historically tested, below I derive Nitrate-Nitrogen (N-N) values for use in Southland. I consider these comparable, as nitrite is almost always very low.

8.4 The evidence of Ms MacArthur discusses indigenous fish communities, and their conservation threat status. While Fish & Game's interests relate to trout and salmonids, I have not singled out the requirements for trout and salmonids, but recommend critical parameters for ecosystem health. There

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for Ministry for the Environment., p. 99 p. plus appendices., Vol. No. 3012. Cawthron Report, Neslon, Gluckman P. (2017) *New Zealand's fresh waters: Values, state, trends and human impacts,* Office of the Prime Minister's Chief Science Advisor, Auckland, Julian J.P., De Beurs K.M., Owsley B., Davies-Colley R.J. & Ausseil A.G.E. (2017) River water quality changes in New Zealand over 26 years: response to land use intensity. *Hydrology and Earth System Sciences,* **21,** 1149-1171, Ministry for the Environment & Stats NZ. (2017) *Our fresh water 2017: Data to 2016,* Ministry for the Environment & Stats NZ, Wellington, Oecd. (2017) *OECD Environmental Performance Reviews: New Zealand 2017,* OECD Publishing, Paris..

[17] Hodson at [30].

is generally good correlation between the habitat requirements of indigenous fish and salmonids. Trout and salmonids are the most studied fish species in the world, and the provision of salmonid habitat requirements provides protection for the health of most other species in aquatic ecosystems.

Table 1. Water quality numerics to protect ecosystem health in each waterbody type.*

Waterbody classification	***Deposited fine sediment <2 mm diameter (maximum bed cover %)***	***MCI (median over 3 years)***	***QMCI (median over 3 years)***	***Nitrate (mg/l) (median over 3 years)***	***Dissolved reactive phosphorous (mg/l) (median over 3 years)***
	Less than or equal to	Greater than or equal to	Greater than or equal to	Less than or equal to	Less than or equal to
“Natural State Waters”					
“Mountain”	20	120	6	0.10	0.006
“Lake Fed”	20	100	5.5	0.46	0.019
“Spring Fed”	20	120	6	0.10	0.006
“Hill”	20	100	5.5	0.46	0.019
“Lowland hard bed”	20	100	5.5	0.46	0.019
“Lowland soft bed”	30	90	4.5	0.89	0.038
“Mataura 1, 2 & 3”	20 20 20	120 100 100	6 5.5 5.5	0.10 0.46 0.46	0.006 0.019 0.019

* Some Natural State Waters may on rare occasion breach these numerics because of the unusual biophysical conditions e.g. tannin rich peat streams.

Notes:
The parameters in Fish & Game’s appeal on Appendix E are consistent with the above, subject to the following:

- MCI and QMCI numerics adjusted downward from Fish & Game’s appeal for Hill and Lowland Hard bed waterbodies.
- Deposited fine sediment (maximum bed cover) adjusted from Fish & Game’s appeal because, for the purpose of this evidence, these parameter numerics are focused on ecosystem health, rather than the specific parameters for Salmonids.
- Nitrate and Dissolved Reactive Phosphorus columns have been added to achieve the relevant MCI/QMCI states.

9. EXPLAINING CRITICAL PARAMETERS: STREAM BIOLOGICAL COMMUNITIES

9.1 Mr Hodson's evidence discusses the effects of elevated nutrient concentrations. I agree with his description of the effects of elevated nutrient concentrations and nuisance algal growth[18] and expand on it briefly.

9.2 The two main nutrients that can result in excessive periphyton growth, are nitrogen (N) and phosphorous (P).[19]

9.3 Excessive periphyton growths are not only aesthetically unappealing, they can also result in dramatic changes to the biological communities in rivers and streams.

9.4 Periphyton is the algae (often only visible microscopically or as a coating of slime) that forms the basis of most stream and river food webs. Aquatic invertebrates consume this periphyton either directly (along with other organic sources) or by predating the smaller grazing invertebrates.[20] Some periphyton is required as food for many aquatic invertebrates;

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[18] Hodson at [25] – [35].

[19] (Biggs, 1996; Dodds, Jones & Welch, 1998; Biggs, 2000; Death, Death & Ausseil, 2007). As also stated by Mr Hodson at [43]: *"The Council's SOE reporting for rivers includes a number of indicators of ecosystem health: macroinvertebrates, benthic periphyton, NNN, amoniacal nitrogen, and DRP.* ***These nutrients are the most readily available to drive instream algal and macrophyte growth****, which can adversely affect ecosystem health".* (my emphasis).

[20] Both as larvae within the river, and as flying adults, invertebrates form an important dietary component for both aquatic (e.g., fish Mcdowall R.M. (1990) *New Zealand Freshwater Fishes: A Natural History and Guide,* Heinemann Reed, Auckland. and terrestrial (e.g., birds, spiders, bats Polis G.A., Power M.E. & Huxel G.R. (2004) Food webs at the landscape level. pp. xviii, 548. University of Chicago Press, Chicago, Winterbourn M. (2004) Stream Invertebrates. In: *Freshwaters of New Zealand.* (Eds J.S. Harding & M.P. Mosley & C.P. Pearson & B.K. Sorrell), pp. 16.11-16.14. New Zealand Hydrological Society Inc. and New Zealand Limnological Society Inc., Christchurch, Burdon F.J. & Harding J.S. (2008) The linkage between riparian predators and aquatic insects across a stream-resource spectrum. *Freshwater Biology,* **53,** 330-346.) food webs.

however, too much algal growth can dramatically change the ecology and habitat conditions of a river.

9.5 As the abundance of periphyton increases and becomes excessive, the nature of the invertebrate community changes, from mayfly, stonefly and caddisfly dominated communities to ones with worms, snails and midges that do not support the same abundance, biomass or diversity of fish that the former communities do. The types of invertebrate present in a river will indicate the nature of the river habitat and to what extent it is affected by human activities. This is utilised by scientists to create indices (e.g., Macroinvertebrate Community Index, MCI) that measure the ecological health and/or water quality of a stream or river.

9.6 Changes to the invertebrate communities can have significant impacts on the health of aquatic and terrestrial ecosystems and widespread effects on ecosystem functioning both in the waterbody and within the wider catchment (Fig. 3).

Figure 3. Excessive periphyton growth and smothered substrate in the Otapiri Stream, Southland.

9.7 The periphyton can also build up to such a biomass that the lower layers start to rot. This can dramatically reduce the oxygen levels and change the pH of the water, leading to significant adverse effects on many invertebrates and fish.

9.8 The change to habitat structure and quality (in particular pH and oxygen levels) as a result of excessive algal growth will result in fish emigrating, growing more slowly, being more susceptible to disease, or in the worst case dying. Large fish kills can be a result of reduced oxygen levels from excessive periphyton growth particularly on warm summer days.

9.9 Dr Mike Joy has also shown that juvenile native fish (Galaxias and Gobiomorphus) can detect the difference between water coming from high and low level nutrient waterbodies as they migrate upstream and actively avoid the high nutrient rivers altogether. Therefore, elevated nutrient levels could act as a barrier to fish migration.

9.10 The nutrient (N or P) that is limiting periphyton growth is the one that when added to a waterbody will result in an increase in periphyton biomass. To illustrate this, you could consider a pot plant that needs light and water to grow; you can grow it in the best light possible, but if you do not water it then the plant will die. Water becomes the limiting resource because it is the scarcest resource; addition of any water (as long as the plant has not died) will result in the plant growing. Thus, the resource (nutrient) that is at the lowest level in the waterbody is the one that can have the biggest impact on controlling periphyton growth in a waterbody.

9.11 However waterways need to be managed for both N and P to prevent any significant adverse effects on the ecosystem health of those waterways. Management of both these nutrients will be required if MCI states are to be met.

9.12 There is an extensive body of research, including several New Zealand studies, that illustrate that although one particular nutrient may be limiting at one point in space or time, sustainable management must involve managing both nutrients simultaneously.[21]

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[21] Francoeur S.N., Biggs B.J.F., Smith R.A. & Lowe R.L. (1999) Nutrient limitation of algal biomass accrual in streams: seasonal patterns and a comparison of methods. *Journal of the North American Benthological Society,* **18,** 242-260, Francoeur S.N. (2001) Meta-analysis of lotic nutrient amendment experiments: detecting and quantifying subtle responses. Ibid.**20,** 358-368, Wilcock B., Biggs B., Death R., Hickey C., Larned S. & Quinn J. (2007) Limiting nutrients for controlling undesirable

10. RECOMMENDED MCI, QMCI, N AND P STATES

10.1 In this section of my evidence I recommend annual median N-N and DRP levels to assist in meeting the MCI and QMCI desired states set out in Table 1.

10.2 It is possible to use modelled data to set desired nutrient concentrations for each waterway classification in Southland.

10.3 In New Zealand the Macroinvertebrate Community Index (MCI) and its quantitative variant (Quantitative Macroinvertebrate Community Index or QMCI) are the most widely used indices to assess and manage ecosystem health.[22] MCI is essentially a model of how impacted a stream is, based on the invertebrates in a stream. It assesses the effects of some common anthropogenic stressors such as nutrients, and is a good integrator of temporal and spatial changes in many chemical measures.

10.4 MCI is highly correlated with its QMCI.[23] I believe the QMCI is more appropriate for assessing ecological health as it accounts for quantitative changes (such as an increase in Chironomidae from 1 individual to 10,000 individuals, while the MCI treats this as no change).

10.5 The pSWLP Appendix E also refers to the Semi Quantitative Macroinvertebrate Index (SQMCI). This is an intermediate between the MCI and QMCI. Individual taxa are assessed as rare, common and abundant but not counted. In my view the SQMI neither quantifies the effect of numerical changes in animal abundance or ignores it. I am not aware of many regional councils that use the SQMCI. Appendix E, pSWLP MCI

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periphyton growth. p. 38. National Institute of Water & Atmospheric Research, Hamilton, Keck F. & Lepori F. (2012) Can we predict nutrient limitation in streams and rivers? *Freshwater Biology,* **57,** 1410-1421.

[22] (Stark, 1985; Boothroyd & Stark, 2000; Stark et al., 2001).

[23] (Stark 1985; Boothroyd and Stark 2000; Stark and Maxted 2007).

states are compared to my recommended values as follows (SQMCI is included for completeness)[24].

Waterbody classification	**MCI (minimum) RDeath values in brackets**	**SQMCI (minimum) RDeath QMCI values in brackets.**
Natural State	Natural quality of the water shall not be altered	
Lowland Soft Bed	80 (90)	3.5 (4.5)
Lowland Hard Bed	90 (100)	4.5 (5.5)
Hill	100 (100)	5.5 (5.5)
Mountain	120 (120)	7 (6)
Lake Fed	90 (100)	4.5 (5.5)
Spring Fed	90 (120)	4.5 (6)
Mataura 1, 2 and 3	120 (120), 100 (100), 90 (100) as river progresses from mountain, hill to lowland hard bed	

Table 2: Recommended MCI and QMCI states compared to RWP/pSWLP Appendix E.

10.6 The waterbody classification contained in the pSWLP is based on the National River Environment Classification (REC) framework and is an appropriate spatial scale to assign desired states, pending FMU processes. Under the FMU processes, the community may adopt more stringent parameters for MCI, QMCI, N and/or P to deal with site-specific factors.

10.7 I agree with the general approach adopted for the MCI standards in the pSWLP, but I believe an MCI of 80 is too low for a bottom line. As reflected in Table 2, in my opinion an MCI of 90 is an appropriate bottom line for ecological health in New Zealand streams. Having sampled streams throughout New Zealand I have rarely found MCI values below 80. Once a

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[24] A footnote to Appendix E states these indices are to be determined using Environment Southland’s SOE sampling protocol and MfE’s Protocol P2 for sample processing (Stark et al. 2001).

stream gets an MCI of 80 there are so few, very hardy, taxa left that further MCI degradation does not appear to occur no matter how 'worse' the water quality gets. This is referred to in the Paper at Appendix 1 of my evidence (Death *et al.*, 2018).[25]

10.8 In my experience an MCI of 100 is an appropriate level for a stream in a managed landscape. This represents a reasonable diversity of taxa, including some sensitive species such as mayflies, stoneflies and caddisflies and few midges, snails and worms (Table 3).

10. 9 Table 3 illustrates the change in community composition from an MCI of 100 where the EPT (Ephemeroptera, Plecoptera and Trichoptera) insects dominate to an MCI of 80 where midges, snails and worms dominate.

Table 3 Invertebrate communities that could yield MCIs of 100, 90 and 80.

Taxa	Taxa MCI score	Stream 1	Stream 2	Stream 3
Ephemeroptera (mayflies)				
Coloburiscus humeralis	9	120		
Deleatidium sp	8	450	60	2
Nesameletus ornatus	9	10		
Trichoptera (caddisflies)				
Aoteapsyche	4	130	23	45
Beraeoptera	8	195		
Hydrobiosis	5		12	
Costachorema sp	7	1	2	5
Helicopsyche sp	10	5		
Pycnocentrodes	5	120	23	120
Diplectrona	9	2		
Plecoptera (stoneflies)				

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[25] Where the authors state: *"Perhaps the only concern we have in using this approach is that the established bottom line for MCI/QMCI of 80/4 appears to be very low. Once ecological health reached that point the long flat tail of the relationship (e.g. Fig. 2) along the right of the nutrient axis meant there could be large increases in nutrient levels with only a very small decline in health. In other words, once the ecological health is at the bottom line, condition is relatively unaffected no matter how many more nutrients are added. This suggests the bottom line for the MCI/QMCI may be better at a slightly higher level (e.g., 90 or 4.5 for the MCI and QMCI, respectively)."*

Megaleptoperla grandis	9	1		
Spaniocercoides	8		12	
Stenoperla	10	12		
Zelandobius sp	5		23	
Zelandoperla sp	10	25		
Diptera (two-winged flies)				
Austrosimulium sp	3		1	50
Chironomus	1		12	1020
Orthocladiinae	2		10	250
Crustacea				
Amphipod	5	2	56	120
Oligochaeta (worms)				
Oligochaete sp	1		12	12
Mollusca (snails)				
Potamopyrgus	4			56
Number of taxa		13	12	10
Number of animals		1073	246	1680
MCI		103	90	80
QMCI		7.38	5.28	1.99

10.10 Mountain streams and Spring-fed streams should have the highest ecological health as light, sediment, nutrients and water chemistry should all be of good quality yielding MCIs of 120 or more.

10.11 Hill streams, when managed appropriately can have characteristics that favour some of these healthier invertebrate communities: however these streams are still far from pristine with increased light, sediment and nutrients.

10.12 Lake fed (lake outlets) streams often have high levels of organic material and can have a high abundance of species such as net building caddis. So again, ecological health can be reduced from an MCI of 120.

Nitrogen and Phosphorus

10.13 N and P are essential parameters for management because managing activities to a measure such as periphyton biomass is extremely difficult - periphyton is highly variable in space and time (you can’t ask a farmer to limit the excess periphyton in his or her stream). Because of ease of collection, measurement and the strong correlation with periphyton, dissolved nutrients are what is managed and reported by almost all regional councils.

10.14 The EPA Board of Inquiry hearing for the Ruataniwha scheme in Hawkes Bay opted for managing MCI using my derived DIN levels (Death, 2013). This was preferred over management using periphyton biomass from the TRIM model proposed by HBRIC (Chisholm *et al.*, 2014; Death, 2015), in effect because periphyton biomass can only be managed by a control on nitrogen and phosphorus.[26] Although parameters such as MCI and periphyton are good measures of an environmental outcome, numerical parameters for N and P are more suited to making comparisons with resource use.

10.15 To determine the nutrient concentrations to achieve the a healthy ecosystem appropriate for each waterway type, I used the methodology set out in Appendix 1. I consider this approach improves on existing nutrient guidelines for New Zealand's rivers, because it is based on emprical and/or modelled data.

10.16 In Appendix 1, a statistical approach is used to derive nutrient concentrations to achieve ecosystem health QMCI or MCI levels, based on the weight of evidence analysis. This research established that the critical nutrient concentrations differentiating rivers in each of the A, B, C and D states from the NPSFM are 0.10, 0.46 and 1.32 mg/l for nitrate-nitrogen and 0.006, 0.019 and 0.057 mg/l for DRP respectively:

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[26] Above footnote 2: Report and decision of Board of Inquiry Volume 1 page 148 and at [357] *"The Board finds Dr Death's correlation of the desired MCI levels with an empirical DIN level useful. An indicator of ecological health such as MCI which is not related to a measurable water quality nutrient concentration would be problematic. … as water quality science advances a different DIN limit may emerge as a more appropriate level. In the meantime the Board sees the DIN limit of 0.8 mg/l as a pragmatic level that appropriately protects ecological health while enabling more intensive land use."*

MCI	QMCI	Nitrate-Nitrogen (mg/l)	DRP (mg/l)
120	6.0	0.10	0.006
100	5.0	0.46	0.019
80	4.0	1.32	0.057

Table 4: Nitrate-Nitrogen and DRP concentrations to achieve MCI and QMCI states

10.17 The weight of evidence approach involves transparent application of individual weights to individual results/lines of evidence. Ten pieces of evidence were compiled from around New Zealand to assess relationships between nutrients and MCI. This included New Zealand National Network Monitoring data (Unwin & Larned, 2013), published reports and papers (e.g. (Biggs, 2000; Joy, 2009; Matheson, Quinn & Unwin, 2016), my own data from 964 streams (Death *et al.*, 2015), and the ANZECC guidelines (Davies-Colley, 2000). This analysis included pieces of evidence on links between nutrients and invertebrates, links between fish and nutrients and links between periphyton and nutrients as well as the statistical distribution of nutrient levels in New Zealand waterways. These yielded remarkably consistent nutrient limits across the lines of evidence.

10.18 Weighted averaging was applied based on whether linkages were direct or indirect. Direct linkages were allocated twice the weight of purely statistical or less direct linkages. Only numbers from significant relationships were included in the final assessment.

10.19 Reference condition DRP levels in North island streams are generally higher than those in the South Island because of volanic activity in the former. Thus a case could be made for less stringent limits for North Island streams. However derived DRP levels for the South Island are clearly appropiate.

10.20 Wagenhoff et al., 2017 have also found a threshold for impact on macroinvertebrate communities at total N of ~ 0.5 mg/l, slightly above my comparable level of 0.46 mg/l. Wagenhoff et al., 2017 looked at regional field survey data from 58 stream sites in the Manawatu Whanganui Region, in order to derive stressor-response shapes for environmental indicators, to consider whether clear management (freshwater) objectives could be derived. Although the authors found limited information for setting P objectives, common macroinvertebrate metrics, including MCI, responded to TN.

11 SEDIMENT

11.1 Reducing sedimentation is critically important for maintaining ecosystem health in Southland rivers (Wagenhoff *et al.*, 2011).

11.2 Fine sediment is defined as substrates with a diameter smaller than 2 mm (Clapcott *et al.*, 2011b).

11.3 As I explain in this section of my evidence, increased levels of fine suspended and deposited sediment can have dramatic effects on stream ecosystems. Increased sediment loads can:[27]

- smother natural benthos;
- reduce water clarity and increase turbidity;
- decrease primary production because of reduced light levels;
- decrease dissolved oxygen;
- cause changes to benthic fauna;

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[27] Ryan P.A. (1991) Environmental effects of sediment on New Zealand streams: a review. *New Zealand Journal of Marine and Freshwater Research,* **25,** 207-221, Waters T.F. (1995) Sediment in streams: sources, biological effects, and control. *American Fisheries Society Monograph,* **7,** 251, Matthaei C.D., Weller F., Kelly D.W. & Townsend C.R. (2006) Impacts of fine sediment addition to tussock, pasture, dairy and deer farming streams in New Zealand. *Freshwater Biology,* **51,** 2154-2172, Townsend C.R., Uhlmann S.S. & Matthaei C.D. (2008) Individual and combined responses of stream ecosystems to multiple stressors. *Journal of Applied Ecology,* **45,** 1810-1819, Clapcott J., Young R., Harding J., Matthaei C., Quinn J. & Death R. (2011c) Sediment Assessment Methods: Protocols and guidelines for assessing the effects of deposited fine sediment on in-stream values. Cawthron Institute, Nelson, Collins A.L., Naden P.S., Sear D.A., Jones J.I., Foster I.D.L. & Morrow K. (2011) Sediment targets for informing river catchment management: international experience and prospects. *Hydrological Processes,* **25,** 2112-2129..

- kill fish;
- reduce resistance to disease;
- reduce growth rates; and
- impair spawning, and successful egg and alvein development.

11.4 Sediment occurs as a natural component of many natural aquatic systems, which is transported as suspended sediment and bedload, mostly at times of high river flows and floods. Small particles, such as clay and silt, are generally transported in suspension, whereas larger particles, such as sand and gravel, usually roll or slide along the riverbed.

11.5 Deposited sediment can smother animals directly (Fig. 4A and 4B) and/or motivate them to leave. It can also smother and bind with the periphyton on rock surfaces that is the food for many aquatic invertebrates and lower the nutritional quality of this food. It fills in the interstitial spaces between rocks (Fig. 4C) where many of the fish and invertebrates live during the day (most are nocturnal) or during flood events. Stream invertebrates and many fish (e.g. eels) can live at least up to a metre under the stream bed if there are suitable interstitial spaces (Williams & Hynes, 1974; Stanford & Ward, 1988; Boulton *et al.*, 1997; McEwan, 2009).

Figure 4A. Koura struggling in deposited sediment.

Figure 4B. Banded kokopu struggling in deposited sediment.

Figure 4C. Stream substrate with interstitial spaces partly clogged with deposited sediment.

11.6 A number of fish species, particularly trout, are visual feeders, thus any increase in suspended sediment or corresponding reduction in water clarity reduces their ability to feed efficiently. The reduced water clarity results in visual feeding fish spending more time and energy foraging which in turn reduces growth rates, general heath, and causes potential reductions in reproductive fitness (Kragt, 2009).

11.7 Increases in suspended sediment have the potential to adversely affect macroinvertebrate communities. Reductions in water clarity can cause reductions in primary production, periphyton biomass and food quality. Invertebrate community composition may be altered as a result of sedimentation generally with a loss of stonefly and mayfly species, and an increase in chironomids and oligochaetes that can burry into silt. Sediment may also cause a reduction in dissolved oxygen by clogging substrate interstices leading to a reduction in gas exchange with more oxygenated surface water.

11.8 Fish, such as salmonids, that lay their eggs in the substrate of the stream are also particularly sensitive to deposited sediment. The sediment can smother eggs directly or reduce oxygen levels in the area directly below the stream bed dramatically (Olsson & Persson, 1988; Crisp & Carling, 1989; Weaver & Fraley, 1993; Waters, 1995).

11.9 Along with specific regulatory and non-regulatory mechanisms to reduce sediment inputs from land use activities into waterways current research suggests these effects would be best dealt with by an upper limit of 20% or 30% cover for deposited sediment, depending on the waterbody type (Clapcott *et al.*, 2011b; Burdon, McIntosh & Harding, 2013).

11.10 Changes in macorinvertebrate communties start to occur when the deposited fine sediment levels starts to exceed 20% cover (Fig. 5) (Clapcott *et al.*, 2011b; Burdon, McIntosh & Harding, 2013). At 30%, invertebrate communities are declining in health but still maintain a reasonable level of ecosystem health. At 40% cover ecological health is below an acceptable bottom line.

11.11 For completeness I note that trout require a more stringent requirement at spawning sites. Generally less than 10% sediment cover is considered good for trout spawning and none is optimal (Clapcott et al., 2011c), however the recommendations do not single out requirements for trout.

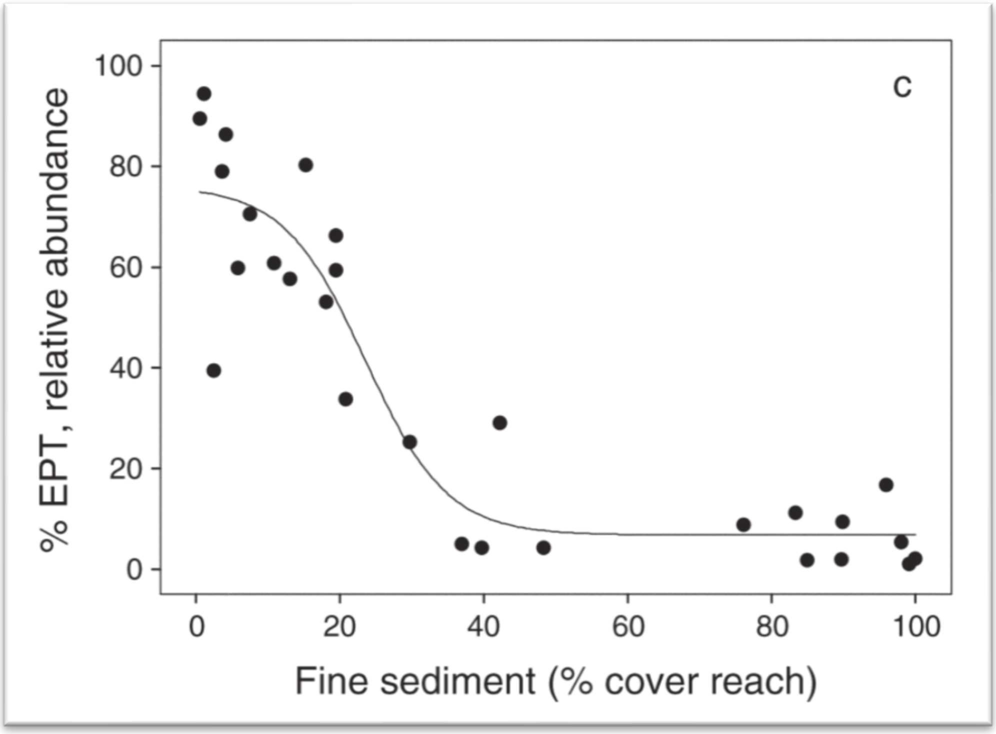

Figure 5 Plot of EPT (sensitive taxa) against the percent coverage of fine sediment (from Burdon et al. 2013).

12 THE CURRENT STATE COMPARED TO THE RECOMMENDED NUMERIC OUTCOMES

12.1 Using modelled data of nitrate, DRP and MCI from (Unwin & Larned, 2013),[28] I was able to calculate the percentage of stream reaches in each of the waterbody types that had values that exceeded (nitrate and DRP) or were

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[28] Unwin and Larned (2013) compiled data, from 786 water quality sites, collected from 2006 to 2011 around New Zealand. They modelled nitrate-nitrogen and dissolved reactive phosphorus (DRP) using random forests (a modelling technique that uses decision trees) and 28 site-specific catchment descriptors as predictors: Reach elevation, Catchment elevation, Mean slope, Catchment area, Lake index, Mean flow, Rain variability, Min temperature, Max temperature, Rain days > 10, Rain days > 50, Rain days > 200, Evapotranspiration, %alluvium, %glacial, %peat, Calcium, Hardness, Particle size, Phosphorous, %bare, %exotic forest, %indigenous forest, %pastoral heavy, %pastoral light, %scrub, %urban, %wetland. The models explained 66% and 57% of the variation in the data, for nitrate and DRP, respectively.

below (MCI) thresholds identified in Table 1. For the purpose of this analysis a stream reach was defined as the NZ reach; a section of stream between tributaries.

Table 5. Stream reaches in Southland with water quality that breaches water quality numerics in Table 1. That is MCI below the numeric in Table 1 or Nitrate-Nitrogen, or DRP above the numeric in Table 1.

Waterbody classification	**Total number of reaches**	**MCI (median over 3 years) Percent in brackets**	**Nitrate-Nitrogen (mg/l) (median over 3 years) Percent in brackets**	**Dissolved Reactive Phosphorous (mg/l) (median over 3 years) Percent in brackets**
		Parameter less than or equal to numeric	Parameter greater than or equal to numeric	Parameter greater than or equal to numeric
“Natural State Waters”	33258			
“Mountain”	649	577 (89%)	13 (2%)	9 (1%)
“Lake Fed”	317	57 (18%)	30 (9%)	21 (7%)
“Spring Fed”	413	413 (100%)	401 (97%)	390 (94%)
“Hill”	4347	814 (19%)	521 (12%)	124 (3%)
“Lowland hard bed”	7150	5330 (75%)	4544 (64%)	1786 (25%)
“Lowland soft bed”	7560	3494 (46%)	2829 (37%)	3 (0.04%)
“Mataura 1”	7	7 (100%)	7 (100%)	7 (100%)
“Mataura 2 ”	3	2 (67%)	3 (100%)	0
“Mataura 3”	9451	2979 (32%)	2549 (27%)	352 (4%)
Total as percentage		33	17	8

12.2 The relevant water body classifcations are mapped in pSWLP Map Series 1. Table 5 illustrates that:

a. The majority of “Mountain” reaches are below proposed Nitrate-Nitrogen and DRP concentrations, however, a high proportion still do not meet an acceptable bottom line for ecological health. This indicates that variables, other than Nitrate-Nitrogen and DRP concentrations, are the cause of low ecological health.

b. Some “Lake Fed” reaches are below acceptable bottom lines for ecological health.

c. Most, if not all, “Spring Fed” reaches are below acceptable bottom lines for ecological health.

d. Some “Hill” reaches are below acceptable bottom lines for ecological health.

e. “Lowland hard bed”, “Lowland soft bed” and “Mataura 3” reaches have the greatest number of reaches that are below acceptable bottom lines for ecological health.

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APPENDIX 1

“Clean but not green: a weight-of-evidence approach for setting nutrient criteria in New Zealand rivers” (Death *et al.*, 2018)

Clean But Not Green: A Weight-of-Evidence Approach for Setting Nutrient Criteria in New Zealand Rivers

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Right running head: Nutrient criteria for New Zealand rivers

Abstract

Eutrophication of waterbodies is a major stress on freshwater ecosystems globally, and New Zealand is no exception. Expanding agricultural intensification is increasing nutrient levels in rivers throughout the country and, as a response, the New Zealand Government has established a policy of freshwater management where waterbodies are managed within four states ranging from high to low ecosystem health. We compiled a large range of data sources, used a weight-of-evidence approach to determine nitrate, and dissolved reactive phosphorus (DRP) limits objectively to categorise rivers and streams into these four states. The compiled evidence establishes nutrient concentrations differentiating rivers into each of these states at 0.10, 0.46 and 1.32 mg/l for nitrate and 0.006, 0.019 and 0.057 mg/l for DRP. While a wide range of interacting stressors affect the ecological health of rivers, nutrients are among the most important stressors and we believe the evidence supports managing to these nutrient criteria will provide for better ecological condition in New Zealand’s rivers and streams.

Keywords Ecological health · Eutrophication · New Zealand · Multiple lines of evidence · Nutrient criteria · Nutrients · River management

Introduction

Globally freshwater biodiversity is under considerable threat from a wide variety of anthropogenic stressors (Dudgeon 2010; Dudgeon et al. 2006; Vorosmarty et al. 2010). This decline in biodiversity has resulted from multiple interacting stressors (Leps et al. 2015; Matthaei et al. 2010; Piggott et al. 2012; Wagenhoff et al. 2011) including water abstraction for consumptive and agricultural needs (Dewson et al. 2007; McDowell et al. 2011; Poff and Zimmerman 2010), invasive species (Collier and Grainger 2015; Olden et al. 2010), channelization, sedimentation, eutrophication (Allan 2004; Carpenter et al. 1998b) and changing climate regimes (Death et al. 2015b; Heino et al. 2009; Palmer et al. 2008). Eutrophication is among the most widespread and problematic stressors: high nutrient levels are associated with the loss of biodiversity, reduced recreational and property values, and increased costs for drinking water treatment (Foote et al. 2015). Eutrophication of freshwaters, therefore, not only comes with a cost to the organisms that inhabit these systems but also financially to the agencies managing them (Dodds et al. 2009; Jarvie et al. 2013; Pretty et al. 2003). The main culprits of eutrophication requiring the greatest attention for management and policy development are nitrogen and phosphorus (Carpenter et al. 1998a; Elser et al. 2007).

As in most developed countries, there has been considerable concern over the declining water quality, ecological health and biodiversity of many of New Zealand’s freshwater bodies (Ballantine and Davies-Colley 2010; Foote et al. 2015; Joy 2015; Joy and Death 2014; Larned et al. 2016; Ministry for the Environment & Stats NZ 2017; Parlimentary Commisioner for the Environment 2013; Verburg et al. 2010). Over the last 25 years many measures of water quality have declined at monitored sites throughout the country, particularly in lowland rivers with catchments dominated by agriculture (Ballantine and Davies-Colley 2010; Davies-Colley and Nagels 2002; Foote et al. 2015; Ministry for the

Environment & Stats NZ 2017; Unwin and Larned 2013). Most sites in lowland pastoral catchments and all sites in urban catchments exceed safe swimming standards for pathogens, and 55% and 25% of monitored sites have increasing nitrate-nitrogen and dissolved reactive phosphorus (DRP) levels, respectively (Larned et al. 2004; Ministry for the Environment and Statistics New Zealand 2015). Thirty-two percent of monitored lakes are now classed as polluted with nutrients as are 84% of lakes in pastoral catchments (Verburg et al. 2010). Groundwater ecosystems are less well monitored, but at 39% of monitored sites nitrate levels are rising and at 21% pathogen levels exceed human drinking standards (Daughney and Wall 2007).

The condition of New Zealand's freshwater has become such an issue that both national and regional government have responded with a large variety of regulatory, non-regulatory or funding initiatives in an attempt to improve water quality (Cullen et al. 2006; Hughey et al. 2010; Joy 2015; Ministry for the Environment 2004; Ministry for the Environment 2014). However, regulation and/or limit setting with respect to waterbody nutrient levels has become one of the most contentious issues in improving New Zealand's water quality (Chisholm et al. 2014; Rutherford 2013; Wilcock et al. 2007). This is undoubtedly because of the perceived negative economic consequences associated with constrained nutrient discharge to waterbodies, particularly by the dairy farming industry, although the cost of preventing nutrients reaching waterways is considerably less than trying to remove them once they are there (Foote et al. 2015; Joy 2015; USEPA 2015). The government has established total nitrogen and total phosphorus criteria for lakes, but in the case of bioavailable nutrient forms in rivers, the government has only established criteria associated with toxic endpoints (i.e., nitrate and ammonia) not to manage ecological health (Ministry for the Environment 2010; Ministry for the Environment 2014). There are guidelines for nutrient management for particular river types (e.g., ANZECC 2000) and/or taxa (e.g., Biggs 2000b). However, despite the obvious and extensively documented links between high nutrient levels in rivers and declines in ecological health (Biggs 1996; Biggs 2000a; Clapcott et al. 2012; Collier et al. 2013; Death et al. 2007; Death et al. 2015a), current government policy does not provide mechanisms to manage nutrients to safeguard overall ecological health.

Many countries have established nutrient criteria or thresholds to protect aquatic life in their waterways (Camargo and Alonso 2006; Dodds and Welch 2000; Heiskary and Bouchard 2015; Jarvie et al. 2013; Smith and Tran 2010). There are four broad approaches, ecological, statistical and expert-opinion, that can be used alone or in combination (Birk et al. 2012). The ecological approach establishes critical levels of a potential stressor at which ecological condition shifts markedly. Statistical approaches partition all available records of a stressor into *a priori* determined numerical groups (e.g., 25th, 50th 75th percentile). Expert-opinion uses the knowledge of a range of experts to determine the critical levels of a stressor where ecological change occurs. In setting the current numerical thresholds for toxicity, the New Zealand Ministry for the Environment appears to have relied predominantly on expert opinion (e.g., Snelder et al. 2013b). While this approach can be useful when there is insufficient data to make more objective decisions, this is not the case in New Zealand where multiple parameters of river water quality and ecological health have been monitored for nearly three decades (e.g., Clapcott et al. 2012; Larned et al. 2004; Scarsbrook et al. 2000; Smith and Maasdam 1994; Smith and McBride 1990; Unwin and Larned 2013).

In this study we adopt the weight-of-evidence approach of Smith and Tran (2010) to develop nitrogen and phosphorus nutrient limits for New Zealand rivers and streams to protect ecosystem health. We adopt the New Zealand Ministry for the Environment approach detailed in the ‘National Policy Statement for Freshwater Management (NPS-FM)’ whereby a number of measures (termed attributes: nitrogen and phosphorus in this case) are identified by numerical thresholds into one of four states (from A to D). State D is termed the ‘National

Bottom Line' or 'minimum acceptable state' (actually an unacceptable condition of impairment), with the intention that waterbodies will need to be improved to at least above the national bottom lines over time (Ministry for the Environment 2014). This approach differs from that in the USA where nutrient limits are derived for impaired / not-impaired waterways (Dodds and Welch 2000; USEPA 2000), but is similar to that of the European Union Water Framework Directive, which also characterises water bodies as belonging to one of five states of ecological status from bad to high (Birk et al. 2012; European Commission 2000; Poikane et al. 2014). Our work improves on the existing nutrient guidelines for New Zealand's rivers with multiple lines of evidence from empirical and/or modelled data rather than expert opinion, and, by defining states to safeguard ecological health for periphyton, macroinvertebrates and fish rather than a few key taxa. Our approach is the first we know of where the ecological requirements of all riverine food web components are considered concurrently in developing in-stream nutrient concentrations.

Materials and Methods

Methods for Nutrient Identifying Criteria

There are four established methods for identifying nutrient limits (Smith and Tran 2010; USEPA 2000). These are 1) division of known nutrient measures into equal classes (percentile analysis); 2) identification of significant thresholds in the relationship between nutrient values and ecosystem health metrics (Baker and King 2010; King and Richardson 2003; Nelson and Shober 2012; Smith and Tran 2010); 3) identification of signification relationships between nutrient values and ecosystem health metrics at predetermined points; and 4) experimental manipulation of the effect of nutrient values on ecosystem health metrics. Classification and Regression Tree analysis of the data did identify some thresholds of change (option 2 above), but these thresholds had low accuracy (only 30% of cases were correctly classified). Furthermore, they were often binary splits more in line with impaired/non-impaired waterway classification, than degrees of impairment implicit in the New Zealand policy framework. Therefore, to be consistent with the derivation of existing NPS-FM attribute criteria, we used approaches one and three to define potential criteria for both nitrate-nitrogen and dissolved reactive phosphorus (DRP) (Davies-Colley et al. 2013; Hickey 2014; National Objectives Framework Reference Group 2012; Snelder et al. 2013a). A combination of empirical and modelled data sourced from a variety of publications and agencies (Table 1) were used to determine biological or percentile variables. Some data sets allowed the derivation of multiple metrics for determining criteria; so to avoid potential non-independence of these metrics we averaged the nutrient criteria derived from metrics from a single data source and used them as a single piece of evidence. The contribution of each piece of evidence to an overall threshold was determined by weighted averaging of the 10 numerics based on whether linkages were direct or indirect.

Data Sets and Preparatory Analyses

Percentile Analysis of Modelled Nutrient Data for National Environmental Monitoring and Reporting

Collection of data on water chemistry in New Zealand rivers is relatively extensive, but highly variable in space and time, with proportionally more sites in lowland areas than higher altitude conservation land (Ballantine and Davies-Colley 2010; Larned and Unwin 2012; Larned et al. 2004; McDowell et al. 2009; Unwin and Larned 2013). Unwin and Larned (2013) have compiled data, from 786 water quality sites, monitored from 2006 to 2011, around New Zealand (Table 1: dataset 1). They modelled nitrate-nitrogen and dissolved reactive phosphorus (DRP) using random forests and 28 site-specific catchment descriptors as predictors. The models explained 66% and 57% of the variation in the data; for nitrate and

DRP, respectively, and provided predicted median nitrate and DRP values for every river reach in New Zealand (n = 566,563). The predicted medians were strongly correlated with independent measures (r=0.64 and r=0.83, for nitrate and DRP, respectively) made at 22 Manawatu rivers and streams (R Death, unpublished) and at 77 National River Water Quality Network (NRWQN) sites (r=0.86 and r=0.73, for nitrate and DRP, respectively). Although it might have been better to have actual nutrient data for all sites, it requires several years of monthly collection to estimate accurate medians for nutrients. Furthermore, sites where such records are available are highly skewed to large lowland sites of particular interest to environment agencies, not the smaller streams that collectively represent a longer length of stream. Modelled data also has the advantage of removing the considerable ‘white noise’ that occurs with actual nutrient measures (Özkundakci et al. 2018). As the modelled data gave a more extensive, consistent and spatially unbiased measure of nitrate and DRP, the use of this data is appropriate and the modelled medians from Unwin and Larned (2013) were used for the percentile analysis. Modelled data are increasingly being used for practical, planning and legal resource management decisions because of their many advantages (Özkundakci et al. 2018; Schmolke et al. 2010)

To assign sites into percentile groups, based on their nitrate and DRP values, we used the percentile analysis approach of Smith and Tran (2010) and the USEPA (2000). The USEPA recommend the 25th percentile when all sites (pristine and impaired) are combined; and the 75th percentile for pristine sites only. We used the 25th, 50th and 75th percentiles for the modelled medians to yield A, B, C and D thresholds for nitrate and DRP. For reaches in Conservation land like National Parks (n = 242,521) that are relatively pristine we used the 95th, 99th and 99.9th percentiles. These sites will reflect natural geographic and geological variation in nitrate and DRP levels but have little or no anthropogenic nutrient influences (Fig. 1); thus, our pre-defined values were at the high extremes of what can occur. Even the A state allows for minimal degradation, while B, C and D allow for increasing degradation levels.

Nutrient - Ecosystem Health Metric Relationships

Several data sources were used to examine the relationship between nutrient concentrations and metrics of ecosystem health (Table 1: datasets 2-8). New Zealand has well established biological indicator criteria for benthic invertebrates: the Macroinvertebrate Community Index (MCI) and its quantitative variant (QMCI) (Quantitative Macroinvertebrate Community Index) (Boothroyd and Stark 2000; Stark 1985; Stark and Maxted 2007). These have been in place since 1985 and are now widely used in all environmental assessment in New Zealand (Boothroyd and Stark 2000; Ministry for the Environment and Statistics New Zealand 2015). Although there is some suggestion they may respond to a variety of stressors in New Zealand waterways, they were specifically developed to assess organic enrichment and eutrophication (Stark 1985) and have been shown to be insensitive to heavy metals, acid mine drainage and deposited sediment (Death and Death 2014; Gray and Harding 2012; Hickey and Clements 1998). The standard MCI and QMCI states (120, 100 and 80, and 6, 5 and 4, for MCI and QMCI, respectively) provide ideal criteria against which to assess A, B, C and D criteria for nutrients.

There are no similar criteria for other potential invertebrate metrics like the proportion of Ephemeroptera, Plecoptera and Trichoptera (EPT). We, therefore, derived criteria for determining nutrient thresholds for these metrics by examining the distribution of EPT(taxa) (percent of taxa that are EPT at a site) and EPT(animals) (percent of animals that are EPT individuals at a site) in 513 streams sampled in conservation land. The A, B and C/D attribute classes for percent EPT(taxa) and EPT(animals) were set at values for the 10th, 1 and 0.1% of these sites. For the metrics EPT(taxa) these were 46, 37 and 22%, respectively and for EPT(animals) 26, 11 and 1% for A, B and C/D, respectively.

The datasets used to explore the relationship between nutrient concentration and ecosystem health metrics (Table 1) are independent and were derived using different methodologies including modelled metric and nutrient values, measured metric and modelled nutrient values at a reach-scale, and measured metric and nutrient values. Each dataset and the approach used to describe the relationship with nutrients is outlined below and in Table 1. Where multiple metrics were averaged for a single dataset, the individual regressions are presented in the Supplementary Material and the averages in Table 1.

Modelled Nutrient - Modelled Ecosystem Health Metric Relationships (Table 1: datasets 1 vs. 2)

Clapcott et al. (2013) modelled MCI values calculated from invertebrate collections in 1033 unique stream segments between 2007 and 2011 using Random Forests to yield predictions of MCI scores for all river reaches in New Zealand (r=0.83 between observed and predicted MCI). We regressed these modelled MCI values against the modelled nutrient values from Unwin and Larned (2013) for each reach in New Zealand. QMCI was calculated for the Clapcott et al (2013) MCI predictions by deriving a regression equation between measured MCI and QMCI from 963 North Island sites (Death et al. 2015a) ($F_{1,961}$ =1761 $p<0.001$; r^2=0.65). These QMCI values for each river reach in New Zealand were also regressed against the modelled nutrient values from Unwin and Larned (2013). Finally, predicted MCI (in the absence of landuse change) and observed MCI expressed as a ratio of Observed/Expected (O/E) (Clapcott et al. 2017) were regressed against the same modelled nutrient values from Unwin and Larned (2013). Thresholds for A, B and C/D for the O/E were determined at 0.9, 0.85 and 0.8 following Clarke and Murphy (2006).

Modelled Nutrient – Measured Metric Relationships (Table 1: datasets 1 vs. 3 and 6)

Biological indices calculated for invertebrate data were collected at 962 sites sampled in the lower North Island between 1994 and 2007 were used for the regression (Death et al. 2015a). Most of these sampling occasions involved 5 replicate 0.1 m^2 Surber samples from riffles, although some collections comprised a single 1-minute kick-net sample (see Death et al. (2015a) for more details). Samples were filtered through a 500 μm mesh sieve and identified to the lowest possible taxonomic level (usually genera) using Winterbourn et al. (2006). Where repeat samples were collected from a site in multiple years, only the most recent sample was used in the analysis. The MCI and QMCI is relatively independent of sampling effort and season (Duggan et al. 2002), thus we are confident that the measures of biological water quality used are an accurate representation of ecological condition, even though data were collected for a variety of reasons. MCI, QMCI, EPT(taxa) and EPT(animals) were regressed against the modelled nutrient values from Unwin and Larned (2013).

The Index of Biotic Integrity (IBI) (Joy and Death 2004), a bioassessment metric used for fish assemblages in New Zealand, was calculated for data collected nationally but irregularly (New Zealand Freshwater Fish Database https://www.niwa.co.nz/our-services/online-services/freshwater-fish-database (Jowett 1996)) between 1970 and 2007 (Joy 2009). These measures were regressed against the modelled nutrient values from Unwin and Larned (2013) for the corresponding reach. IBI thresholds for A, B, C and D were set at 42, 32 and 24 following Joy (2009).

Measured Nutrient – Measured Metric Relationships (Table 1: datasets 4 and 5)

Median metrics calculated from collected invertebrates and nutrients were regressed against each other for two datasets. One collected at 24 Manawatu streams and rivers (Death 2013) and the other at 64 nationwide NIWA monitoring rivers (Larned and Unwin 2012; Unwin and

Larned 2013). Samples were collected on multiple occasions (monthly for nutrients, yearly for invertebrates) between 1999 to 2011 and 1989 to 2014, for Death (2013) and NIWA, respectively.

Relationships between biological metrics and nutrient measures were assessed with linear regression using the lm function in R (R Development Core Team 2015). Regressions of $y=x$, $y=\ln(x)$, $\ln(y) = x$ and $\ln(y) = \ln(x)$ were analysed for the best fit. Nutrient thresholds were determined by back calculating from the regression equation at y= 120, 100 and 80 for MCI, y= 6, 5 and 4 for the QMCI, y= 46%, 37% and 22% for EPT(taxa), and y = 26%, 11% and 1% for EPT(animals).

Previously Published Numerics and Ecosystem Health Metric Relationships

Several previous publications have investigated nitrate-nitrogen and DRP thresholds for water management in New Zealand. The ANZECC (ANZECC 2000) guidelines derived nitrate and DRP thresholds for upland and lowland rivers in New Zealand (Table 3.3.10 (ANZECC 2000)) based on monitoring data collected by Davies-Colley (2000) (Table 1: datasets 9 and 10). These have been used widely in New Zealand over the last two decades for management decisions around water quality (e.g., Manawatu Wanganui Regional Plan). Biggs (2000a) collected a variety of periphyton and nutrient measures from 30 rivers throughout New Zealand and derived regression equations for maximum chlorophyll *a* and nitrate / DRP (Table 1: dataset 7). This information has also been used in management recommendations on water quality in New Zealand (Biggs 2000b; Biggs and Kilroy 2000). The current National Policy Statement for Freshwater Management 2014 lists A, B, C and D thresholds for periphyton of 50, 120 and 200 mg chlorophyll *a* m^2, so these were used with the Biggs' equations to derive nitrate and DRP numerics (Ministry for the Environment 2014). Matheson et al. (2016) have also used quantile regression on data from several regions

(Wellington, Manawatu Wanganui, Canterbury and Hawkes Bay) to derive nutrient guidelines to achieve the NPS periphyton attribute states above (Table 1-3 in report). These derived numerics were also included as lines of evidence (Table 1: dataset 8).

Weighting Lines of Evidence

Thresholds of change between the above ecological classes were derived from each nutrient ecosystem health metric relationship regression and combined with percentiles or previously published limits in Table 2 (see also Supplementary Material). The final nutrient limits were determined by calculating a weighted average of those 10 nutrient limits for each dataset / line of evidence multiplied by their allocated weighting. Following Smith and Tran (2010), direct linkage relationships between ecosystem health measures and nutrients were allocated twice the weight in the analysis of purely statistical or less direct linkages (e.g. percentile analysis and Fish IBI). Where relationships were not significant they were not included as a line of evidence i.e. they were allocated a weighted value of 0. To evaluate the influence of a single piece of evidence (i.e. sensitivity) the weighted criteria were recalculated by removing one line of evidence in turn, for all lines of evidence.

Results

Are National or Regional Criteria More Appropriate?

New Zealand is geologically active with high mountains, frequent earthquakes, geothermally active areas and volcanoes. This geological activity in turn results in a spatially variable geology that might suggest regional nutrient criteria will be necessary to account for the natural differences in ‘pristine’ environmental conditions. However, a plot of the median and range of nutrient values from Unwin and Larned (2013) in catchments with predominantly (>80%) native vegetation (Fig. 1) indicates that although the median is lower and range greater as one moves south, there are no dramatic regional differences. For nitrate, all regions have 75% of ‘pristine’ reaches well below the A band upper nutrient threshold (see below for derivation), and all reaches are well below the B band upper threshold, except for a few outlying points in the South Island (Fig. 1). There are more distinct differences between the North and South Islands in DRP because of the preponderance of volcanic activity in the former. A different threshold for category A in the North and South Islands may be warranted, but given the greater simplicity and understanding associated with one set of national criteria, rather than multiple regional criteria, we have opted for the former.

Ecosystem Health Metric Relationships

The relationships between the health metrics and nutrient concentrations were predominantly exponential (Supplementary Material) with health declining more rapidly for increasing nutrient concentrations at low levels and plateauing as ecological health approached poor condition (e.g., Fig. 2). That is, once low health was achieved, further increasing nutrient levels had little additional detrimental effect. As variables other than nutrients will also potentially be affecting ecosystem health it is not surprising that there is a large spread in the data. Only numbers from significant relationships were included in the final assessment.

Numerical Nutrient Thresholds

Table 2 presents the numerical nutrient thresholds for the A, B, C and D states derived from each line of evidence. The weighted evidence yielded nitrate concentrations of 0.10, 0.46 and 1.32 mg/l, and DRP concentrations of 0.006, 0.019 and 0.057 mg/l for the A, B, C and D states (Table 2). Criteria from each individual line of evidence (where these were significant) were remarkably consistent across all the lines of evidence (Table 2, Supplementary

Material). The only real exception was that criteria derived from the percentile analysis were generally lower than those from the regression analysis.

Sensitivity analysis (i.e., removing one line of evidence in turn and recalculating weighted criteria) had very minor effects on the final weighted criteria. For example, in this sensitivity analysis the nitrate criteria ranged from 0.10-0.15, 0.43-0.81 and 1.35-1.93 for the A/B, B/C and C/D criteria, respectively. The DRP criteria ranged from 0.005-0.006, 0.017-0.022 and 0.039-0.064 for the A/B, B/C and C/D criteria, respectively. There was also no indication of differences in criteria derived from regionally focused data (e.g. Manawatu (FAT) data) or those from more geographically spread data.

A small percentage of New Zealand river reaches, based on modelled median nitrate or DRP levels from Unwin and Larned (2013), would be classified as below the bottom line for ecosystem health (Table 3). The majority of river reaches would be classed as A for nitrate (58.2%) and B for DRP (52%).

Discussion

Although the ecological health of rivers and streams is determined by a wide range of potentially interacting stressors, nutrients are one of the most pervasive and detrimental stressors for the fauna and flora of rivers globally (Allan 2004; Carpenter et al. 1998a; Stevenson and Sabater 2010). Environmental stress from excess nutrients is particularly detrimental to river health in New Zealand where the dominant business and land use is agriculture (Foote et al. 2015; Joy 2015; Weeks et al. 2016). Our weight-of-evidence assessment produced the following nutrient criteria: 0.10, 0.46 and 1.32 mg/l for nitrate, and 0.006, 0.019 and 0.057 mg/l for DRP. These criteria represent objective, data-driven numbers for use in policy tools to maintain or improve the ecological health of rivers in good, moderate or poor condition. Additionally, Wagenhoff et al. (2017) in a study of 58

Manawatu, New Zealand rivers, published subsequent to our data compilation, have also found a threshold for impact on macroinvertebrate metrics at total N = ~ 0.5 mg/l.

Although there can be many situations where expert opinion, rather than data, are necessary to establish management objectives, this is not the case in the nutrient management of rivers and streams for ecological health in New Zealand. There is a large amount of data available to draw on to make decisions; the only issue can be how to draw all that information together into some firm conclusions. The weight-of-evidence approach offers an objective, scientifically rigorous, multiple lines of evidence method to compile a variety of data sources to set nutrient thresholds to meet the four attribute states of ecological health adopted by current New Zealand Government policy. Given the large environmental, economic and social costs these limits may create (Foote et al. 2015; Hughey et al. 2010; Weeks et al. 2016), it is important that they are objectively determined from as wide a range of data and in as robust a manner as possible.

This is the first example we are aware of where fish have been included with periphyton and macroinvertebrates in such an assessment, despite their obvious public interest. Interestingly, the derived nutrient criteria for fish (IBI) were very similar to those for the other taxa. Perhaps one of the impediments has been that a range of variables, besides nutrients, will also affect river health and thus it is not always easy to determine rigorous relationships between nutrients and indices of ecological health. This is clear in the large amount of data scatter in the relationships used in this study. It may also explain why some of the national datasets used, such as that collected by NIWA (Supplementary Material) did not yield significant relationships between the biological indices and nutrient levels. These NIWA sites are predominantly on larger rivers that are more likely to be influenced by multiple stressors than those from a wider range of stream sizes and more limited land uses (e.g., Death 2013, Death *et al.* 2015a). However, it is reassuring that all the data sets yielded numerics within the same small range. Furthermore, in a Boosted Regression Tree analysis of the Death *et al.* (2015a) data, nutrients explained 51% (n=962, cross-validated correlation coefficient = 0.65) and 50% (cross-validated correlation coefficient = 0.76), of the modelled MCI and QMCI, respectively, from 15 potential geographic, geomorphological and catchment predictor variables.

As with any freshwater resource management, adhering to these nutrient limits will not provide a panacea for maintaining good ecological health. Many other factors may interact with, or override the effects of nutrients on river health. However, as a well-established determinant of river food web structure, managing below these nutrient concentrations will certainly be a step in the right direction (Clapcott et al. 2012; Matthaei et al. 2010; Wagenhoff et al. 2012; Wagenhoff et al. 2011). Similarly, establishing limits for only nitrate-nitrogen or dissolved reactive phosphorus will not serve to limit adverse environmental effects, as when and where the respective nutrients become limiting changes and is thus often hard to establish (Death et al. 2007; Dodds and Welch 2000; Jarvie et al. 2013; Keck and Lepori 2012).

Previous studies using the weight-of-evidence approach to establish nutrient thresholds have applied nonparametric changepoint analysis to identify significant biological transition thresholds (e.g., King et al. 2005; King and Richardson 2003; Smith and Tran 2010). However, there was weak evidence for thresholds in our ecological metric nutrient relationships examined in the compiled data. Rather than any particular threshold response there seemed to be an almost continuous, although log-linear change in declining ecological condition with increasing stressor concentration. Therefore, in line with the approach adopted in Government policy, criteria were determined *a priori* for each of the four attribute states using pre-established biological index criteria (e.g., MCI, QMCI). Although, somewhat subjective these thresholds have been in use for a long time in river management (Stark 1985;

Stark 1993; Wright-Stow and Winterbourn 2003), are familiar to all river managers and fit the model of four category attribute states adopted by government policy (Ministry for the Environment 2014).

Perhaps the only concern we have in using this approach is that the established bottom line for MCI/QMCI of 80/4 appears to be very low. Once ecological health reached that point the long flat tail of the relationship (e.g. Fig. 2) along the right of the nutrient axis meant there could be large increases in nutrient levels with only a very small decline in health. In other words, once the ecological health is at the bottom line, condition is relatively unaffected no matter how many more nutrients are added. This suggests the bottom line for the MCI/QMCI may be better at a slightly higher level (e.g., 90 or 4.5 for the MCI and QMCI, respectively).

It is extremely difficult to put the nutrient criteria established in this study for New Zealand in a global context, as differing countries and regions use different chemical species (e.g., total nitrogen and total phosphorus vs nitrate and DRP), they have differing numbers of classes (e.g., the USA has two and Europe five) and many also divide criteria between upland and lowland sites (ANZECC 2000; European Commission 2000; Smith and Tran 2010; USEPA 2000). Table 4 provides a cross-section of those criteria for Australia, USA, England and Wales. Although ranges of nutrient criteria for most of these countries are much larger, reflecting their greater area and geological variability, they do not suggest those developed for New Zealand are incorrect. Those for South Eastern Australia, perhaps the most similar to New Zealand geologically, are very similar.

In conclusion, we derived the nitrate concentrations of 0.10, 0.46 and 1.32 mg/l, and DRP concentrations of 0.006, 0.019 and 0.057 mg/l, which correspond with numerical threshold states A to D (high to low ecological health). We believe these provide rigorous and objective levels at which to set instream nutrient concentrations to protect New Zealand river ecological health. These have been compiled across a range of studies over the full length of New Zealand without any indication of regional differences that might affect the efficacy of these limits in protecting and maintaining the desired ecological state of rivers or streams. Given the pervasive and ever-increasing eutrophication of waterbodies worldwide, we hope these limits will be adopted by New Zealand freshwater managers as one more tool in the arsenal of techniques to better protect and manage freshwater.

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Table 1. Data sources compiled and/or used for analysis. Reference numbers are used to link with Table 2.

Data	Number of reaches	Weight of evidence category	Time interval	Variables used	Reference
Modelled data for National Environmental Monitoring and Reporting	All river reaches in NZ	Percentile analysis	2006-2011	Nitrate, DRP	1 (Unwin and Larned 2013)
Modelled data for National Environmental Monitoring and Reporting	All river reaches in NZ	Metric relationship	2007-2011	MCI, QMCI^A	2 (Clapcott et al. 2103)
Russell Death private data collection	962 streams and rivers in lower half North Island	Metric relationship	1994-2007	MCI, QMCI, EPT(animals), EPT(taxa)	3 (Death et al. 2015a)
Russell Death Freshwater Animal Targets (FAT) model^B	24 Manawatu streams multiple temporal measures (inverts yearly, nutrients monthly)	Metric relationship	1999-2011	Nitrate, DRP, MCI, QMCI	4 (Death 2013)

NIWA data	64 rivers multiple temporal measures (inverts yearly, nutrients monthly)	Metric relationship	1989-2014	Nitrate, DRP, MCI, QMCI	5 (Unwin and Larned 2013)
Mike Joy IBI fish model	All river reaches in NZ	Metric relationship	1970-2007	IBI	6 (Joy 2009)
Biggs (2000) model	30 rivers throughout New Zealand	Regression equations	1995-1998	Periphyton measured as chlorophyll *a*	7 (Biggs 2000a)
Matheson et al. 2016	64+ rivers NRWQN and Regional Council data from throughout New Zealand	Summary table 1-3 from regression analysis.	Not stated	Periphyton measured as chlorophyll *a*	8 (Matheson et al. 2016)
ANZEC guidelines	Table 3.3.10		Not stated	Nutrient measures	9, 10 (Davies-Colley 2000)

[A] QMCI was calculated for the Clapcott et al (2013) MCI predictions by deriving a regression equation between measured MCI and QMCI from 962 North Island sites (Death et al. 2015a) ($F_{1,961}$ =1761 p<0.001; r^2=0.65).

[B] Median values of all temporal replicates were used (i.e. one value per site).

Table 2 Numerical nutrient thresholds (mg/l) for each freshwater state (A-D) derived from multiple lines of evidence (weighted according to whether it is a direct (2) or indirect (1) relationship). See Table 1 for details on source data. See Supplementary material for derivation of evidence for multiple metrics from the same data source. Columns shaded in grey involve at least some data derived from models. PCL = public conservation land.

Source or Source nutrient dataset	1	1 PCL only	1	1	4	5	1	9,10	7	8				
Source ecological dataset	n/a	n/a	2	3	4	5	6	n/a	7	8				
Ecological metric	n/a	n/a	MCI/QMCI/ OE	MCI/QMCI/ EPT	MCI/QMCI	MCI	IBI	n/a	Chl a	Chl a	Weight. Mean	Std. Err.	Min.	Max.
Nitrate														
Weight of evidence	1	1	2	2	2	2	1	2	2	2				
A/B threshold	0.03	0.08	0.02	0.28	0.08	0.00	0.00	0.17	0.12	0.10	0.10	0.03	0.00	0.28
B/C threshold	0.06	0.12	0.29	0.84	0.43	0.60	0.21		0.43	0.63	0.46	0.09	0.06	0.84
C/D threshold	0.28	0.20	0.79	2.58	2.78	1.60	1.10	0.44	0.90	1.10	1.32	0.29	0.20	2.78
DRP														
A/B threshold	0.004	0.011	0.004	0.011	0.007		0.002	0.009	0.002		0.006	0.001	0.002	0.011
B/C threshold	0.008	0.014	0.014	0.021	0.031		0.007		0.007	0.110	0.019	0.003	0.007	0.031
C/D threshold	0.012	0.021	0.025	0.039	0.177		0.014	0.100	0.014	0.018	0.057	0.019	0.012	0.177

Table 3. Percentage of river reaches in each nutrient attribute state. NPS state = New Zealand National Policy Statement for freshwater state. Nutrient data for all New Zealand river reaches are derived from the modelling of Unwin & Larned (2013).

NPS state	NO_3-N (mg/l)	Percent	DRP (mg/l)	Percent
A	< 0.10	58.2	< 0.006	37.4
B	0.10 ≤ x < 0.46	25.2	0.006 ≤ x < 0.019	52.0
C	0.46 ≤ x < 1.32	14.1	0.019 ≤ x < 0.057	10.5
D	> 1.32	2.5	> 0.057	0.03

Table 4. Nutrient criteria developed for other countries

	USA[1]		South Eastern Australia[2]		Rest of Australia[2]		England and Wales[3]		
			Upland	Lowland	Upland	Lowland	DRP (mg/l)	Upland	Lowland
Total phosphorus (mg/l)	0.01-0.076*	Filterable reactive phosphorus (mg/l)	0.015	0.02	0.005-0.01	0.01-0.04	High	0.013-0.024	0.019-0.036
Total nitrogen (mg/l)	0.12-2.18	NOx (mg/l)	0.015	0.4	0.15-0.20	0.15-1.00	Good	0.028-0.048	0.040-0.069
							Moderate	0.087-0.132	0.114-0.173
							Poor	0.752-0.898	0.842-1.003

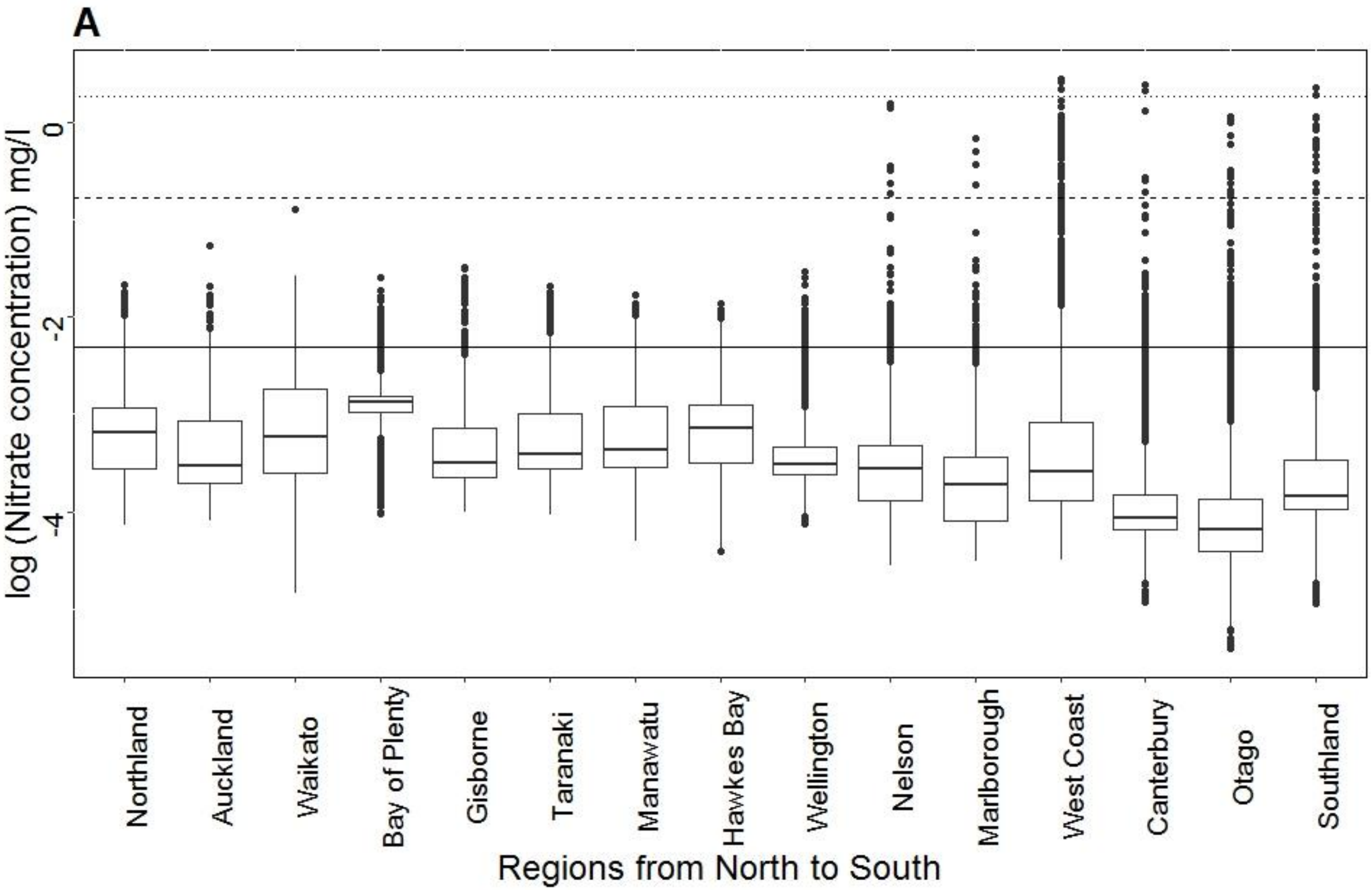
[1] https://www.epa.gov/sites/production/files/2014-08/documents/criteria-nutrient-ecoregions-sumtable.pdf

* there is one value higher in the report but document implies it is likely to be incorrect

[2] (ANZECC 2000)

[3] https://www.gov.uk/government/uploads/system/uploads/attachment_data/file/307788/river-basin-planning-standards.pdf

Figure 1 Boxplots of modelled median A) nitrate and B) DRP concentrations from river reaches in the conservation estate in each region of New Zealand from Unwin and Larned (2013). Nutrient thresholds are plotted as solid, dashed and dotted straight lines for nitrate concentrations of 0.10, 0.46 and 1.32 mg/l, respectively and for DRP concentrations of 0.006, 0.019 and 0.057 mg/l, respectively.



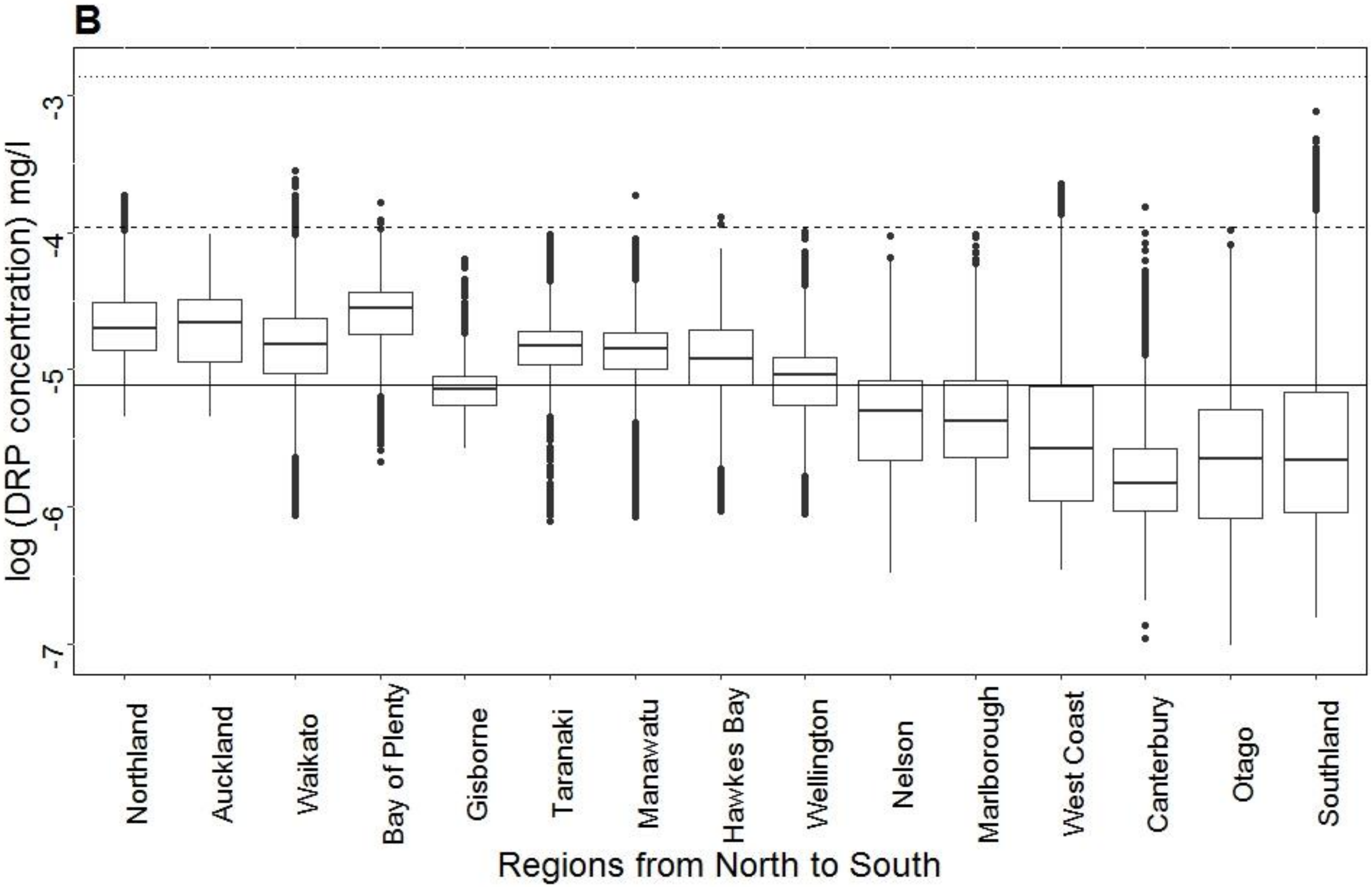
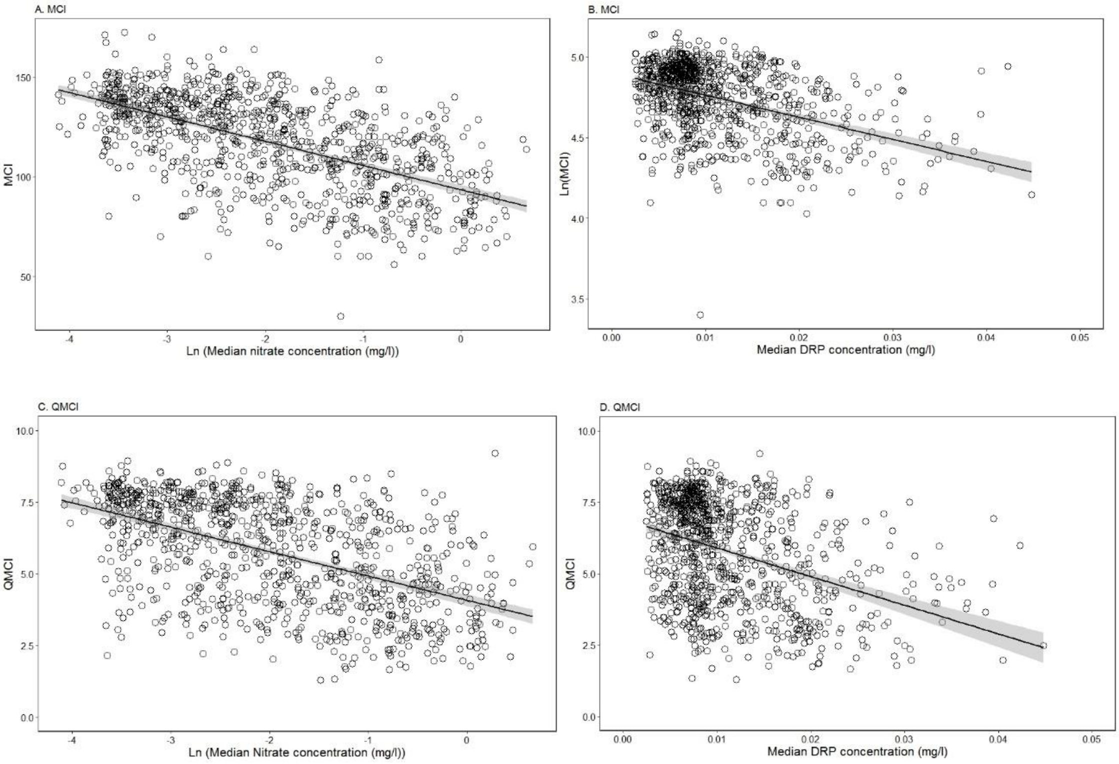

B
log (DRP concentration) mg/l
-3
-4
-5
-6
-7
Northland
Auckland
Waikato
Bay of Plenty
Gisborne
Taranaki
Manawatu
Hawkes Bay
Wellington
Nelson
Marlborough
West Coast
Canterbury
Otago
Southland
Regions from North to South

Figure 2 MCI and QMCI measured at 962 North Island rivers and streams as a function of median modelled nitrate and DRP from Unwin and Larned (2013).



Supplementary Material

Numerical nutrient thresholds (mg/l) for each freshwater state (A-D) derived from multiple metrics from a single data source. Average nutrient thresholds (in bold) were used in Table 1. Regression statistics (F statistic, degrees of freedom, probability and r^2) for relationships are provided along with the data source used from Table 1.

Source nutrient dataset	1	1	1		1	1	1	1		4	4		5	5	1
Source ecological dataset	2	2	2		3	3	3	3		4	4		5	5	6
Ecological metric	MCI	QMCI	O/E		MCI	QMCI	EPT animals	EPT taxa		MCI	QMCI		MCI	QMCI	IBI
Nitrate															
Equation	ln y = ln (x+1)	ln y = ln (x+1)	y = ln x		y = ln x	y = ln x	y = ln x	y = ln x		y=lnx	y=lnx		y=x	y=lnx	y=lnx
A/B threshold	0.02	0.00	0.05	**0.02**	0.11	0.10	0.20	0.71	**0.28**	0.06	0.09	**0.08**	0.00	0.00	0.00
B/C threshold	0.45	0.29	0.14	**0.29**	0.58	0.34	0.52	1.92	**0.84**	0.53	0.33	**0.43**	0.60	0.13	0.21

C/D threshold	1.22	0.77	0.39	**0.79**	3.01	1.09	2.51	3.71	**2.58**	4.36	1.20	**2.78**	1.60	9.10	1.54
r^2	0.53	0.54	0.51		0.35	0.27	0.28	0.29		0.37	0.27		0.08	0.04	0.09
F	632224	653084	588600		513	363	377.6	390.6		51.72	32.66		6.78	3.85	3775
df	1,566548	1,566548	1,566548		1,961	1,961	1,961	1,961		1,86	1,86		1,62	1,62	1,392543
p	<0.0001	<0.0001	<0.0001		<0.0001	<0.0001	<0.0001	<0.0001		<0.0001	<0.0001		0.01	0.05	<0.0001
DRP															
Equation	ln y =x	ln y =x	ln y =x		ln y = x	ln y = x	ln y = x	y=x		y=lnx	y = lnx		y=x	Y=lnx	lny=lnx
A/B threshold	0.004	0.003	0.006	**0.004**	0.008	0.006	0.014	0.015	**0.011**	0.005	0.008	**0.007**	0.000	0.000	0.002
B/C threshold	0.016	0.012	0.013	**0.014**	0.022	0.015	0.025	0.022	**0.021**	0.038	0.025	**0.031**	0.023	0.008	0.007
C/D threshold	0.032	0.024	0.019	**0.025**	0.040	0.027	0.055	0.035	**0.039**	0.275	0.079	**0.177**	0.066	0.024	0.014
r^2	0.38	0.39	0.32		0.18	0.15	0.18	0.18		0.54	0.420		0.02	0.04	0.04
F	349187	357979	265000		210.3	165	217.80	211.10		99.83	63.89		2.16	3.61	15770
df	1,566548	1,566548	1,566548		1,961	1,961	1,961	1,961		1.86	1,86		1,62	1,62	1,392543
P	<0.0001	<0.0001	<0.0001		<0.0001	<0.0001	<0.0001	<0.0001		<0.0001	<0.001		0.15	0.06	<0.0001