Title: Nitrate-nitrogen and Dissolved Reactive Phosphorus (Rivers and groundwater)

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Summary: This paper has been provided as a meeting document for the Science and Technical Advisory Group meeting on 26 February 2019, Agenda Item 6: Nutrients. It presents an alternative for managing nutrients in the NPS-FM: nationally applicable attributes for nitrate-nitrogen and dissolved reactive phosphorus.

Nitrate-nitrogen and Dissolved Reactive Phosphorus (Rivers and groundwater)

Value	Ecosystem health
Freshwater Body Type	Rivers and groundwater
Attribute	Nitrate-nitrogen (Ecosystem Health)
Attribute Unit	Milligrams of nitrate-nitrogen

Value	Ecosystem health
Freshwater Body Type	Rivers and groundwater
Attribute	Dissolved Reactive Phosphorus (Ecosystem Health)
Attribute Unit	Milligrams of dissolved reactive phosphorus (DRP) per litre

1. Nitrate-nitrogen and Dissolved Reactive Phosphorus (DRP) are key nutrients driving the eutrophication of rivers. Eutrophication, through excessive periphyton growth, can alter ecological communities through changes in food supply and hypoxic conditions. To prevent heavy eutrophication, attribute tables and national bottom-lines for both DRP and nitrate-nitrogen are proposed (appended below) at ecosystem health levels based on the paper:

   Death, R. G., Magierowski, R., Tonkin, J. & Canning, A. D. *Clean but not green: a weight-of-evidence approach for setting nutrient criteria in New Zealand rivers.*

2. The current NPS requires DIN and DRP to be set at concentrations to achieve periphyton objectives. A key difficulty with setting nutrient concentrations purely to achieve a periphyton outcome is the difficulty modelling nutrient-periphyton relationships. This arises because:
   a. Nutrients vary considerably with flow and diurnal fluctuations;
   b. Rivers vary in flood frequency and their stone movement;
   c. Large datasets across a range of environments and gradients are required, typically beyond what a single region can offer;
   d. Periphyton can be difficult to monitor, varying considerably between stones,

different river patches, can be influenced by differing levels of invertebrate grazing, and can change rapidly even between days;

e. A periphyton biomass that is suitable for providing invertebrate and fish health at one site may not be suitable at another. Ecological reality is that ecosystems are complex networks where indirect interactions are typically more dominant than direct interactions (Salas and Borrett, 2011). Given that we are seeking healthy ecosystems, to ignore *indirect* links is fraught with risk of setting nutrient attributes that are inadequate to safeguard overall ecosystem health.

3. As an alternative, the attribute tables presented here:
   a. Incorporate data across all trophic levels. Macroinvertebrates and fish tend to be more stable, long term integrators of environmental change than periphyton alone. By using all trophic levels, the compulsory Ecosystem Health value is better provided for.
   b. Combine multiple lines of evidence in a weight-of-evidence approach;
   c. Use data collected nationally

4. To explain further, we compiled several datasets and bodies 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. This included findings from New Zealand National Network Monitoring data (Unwin and Larned, 2013), published reports and papers (e.g., (Biggs, 2000, Matheson *et al.*, 2016, Joy, 2009), Professor Russell Death's data from 964 streams (Death *et al.*, 2015) and the ANZECC guidelines (Davies-Colley, 2000).

5. The multiple lines of evidence were combined in a weight-of-evidence approach . The weight of evidence approach involves transparent application of individual weights to individual results/lines of evidence. Weighted averaging was based on whether linkages between nutrients and a given ecosystem health metric 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.

6. The original paper provided numbers for a four-band system with a nutrient bottom-line in-line with a typical hard-bottomed stream that has an MCI of 80. The attribute tables presented here differ slightly in that they have been spliced to produce six bands and the suggested bottom-line in-line with an MCI of 90. This is following the suggestion in the original paper:

> *"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)."*

7. Table 1 shows that of the 792 SOE monitoring sites, approximately 20% and 17% are below the proposed bottom-lines for nitrate-nitrogen and DRP respectively.

Table 2. The number and proportion of 792 SOE monitoring sites meeting each of the proposed grades

	Nitrate-nitrogen (mg/L)		DRP (mg/L)	
Grade	# of sites	%	# of sites	%
A	180	23	205	26
B	172	22	234	30
C	127	16	105	13
D	132	17	143	18
E	64	8	82	10
F	92	12	52	7

Potential question and answers:

The Redfield ratios for a given river suggest that DRP is limiting. Can I set nitrate-nitrogen at a lower grade than DRP?

This is not recommended. Neither nutrient should be set a level lower than is consistent with overall desired ecosystem health. Both nutrients need to be managed to prevent excessive periphyton as flow, temperature, pH and nutrient fluxes can easily switch a DRP limited stream to a nitrogen limited stream, and vice versa (Briand 1983; Wilcock et al. 2007); different algae species thrive in and are composed of different N:P ratios (Biggs 1990; Biggs and Price 1987; Milner 1953); and two recent reviews of an extensive array of studies (237 and 382 studies, respectively) have found Redfield ratios (the molar N:P ratio) are inaccurate for determining nutrient limitation (Francoeur 2001; Keck and Lepori 2012).

There is a lot of local variability - surely, we can't set nutrient concentrations nationally?

Firstly, the bands approach does not set limits nationally for all rivers. The only binding limit is the national bottom-line (of which most rivers will already be above) that prevents objectives with excessive concentrations from being set. If nutrients are not set at levels worse than the bottom-line or current state (as per recent STAG recommendations) then regional councils are still free to exercise local flexibility in deciding nutrient concentrations should they deem this appropriate – as they have to date with the lake nutrient limits.

Secondly, there was not substantial variation in natural nutrient concentrations between regions. Reference state concentrations for all river classes as derived by McDowell *et al.* (2013) are also well within the proposed bottom-lines.

Thirdly, relationships between ecosystem health metrics and nutrients provided similar nutrient ranges between the national datasets used.

Whilst it is only really the bottom-line (or current state if above) that would be legally binding, the advantage of providing all bands is to assist communities in understanding nutrient concentrations, as with all other attributes.

What about the existing requirement to set limits that also provide for sensitive downstream environments?

This should be retained. As explained above, Councils are still left with sufficient flexibility to set more stringent concentrations if they are required to provide for sensitive downstream environments.

Why are the nutrient attributes also applied to groundwater?

The flow in rivers is made up of runoff (overland flows) and baseflows (upwelling from subsurface water). As rainfall is largely out of our control, if groundwater concentrations are too high then river concentrations may never meet their objectives. This is particularly important when considering climate change as many rivers may have less runoff and have relatively higher contributions of groundwater. In Summer, many rivers are also almost entirely reliant on baseflow. Therefore, it is proposed that the nutrient criteria for rivers also be applied to groundwater.

Groundwaters also have their own ecological communities, though very little is known about them. Managing nutrients in groundwater is also a precautionary approach to managing potential impacts on stygofauna. One of the few studies examining the impact of nitrate on groundwater communities found that increasing nitrate concentrations were associated with reduce biofilm biomass and activity (Williamson *et al.*, 2012).

Are fish really impacted by nutrients?

As explained above, ecosystems, almost by definition, are dominated by indirect interactions. Nutrients can have cascading effects that can reduce fish food (invertebrate) quality and drive hypoxic conditions by promoting excessive algal growth. One of the notorious difficulties of elucidating the impacts of nutrients on fish assemblages is accounting for the multiple drivers, such as habitat, location, sediment, riparian, flows, introduced fish, disease etc. However, with sufficiently large and long datasets (such as the NZFFD), modelling can begin to control for the multiple influences. Recently, Canning (2018) found that nitrate-nitrogen and DRP predicted approximately half of the departures in contemporary fish assemblages from reference state, with the remainder being largely downstream dams, riparian loss and sedimentation. Joy *et al.* (2019) also found significant declining trends in Fish IBI across pastured sites (but not at native forest sites) nationally, of which an increasing impact from pastured sites has been increasing nutrients. When datasets are large enough, a simple but highly effective way to remove the noise from multiple drivers is through application of quantile regressions (Cade and Noon, 2003). In the figure below, Cade and Noon (2003) hypothetically exemplify how quantile regressions can capture responses to a given limiting factor (when it is actually limiting of course).

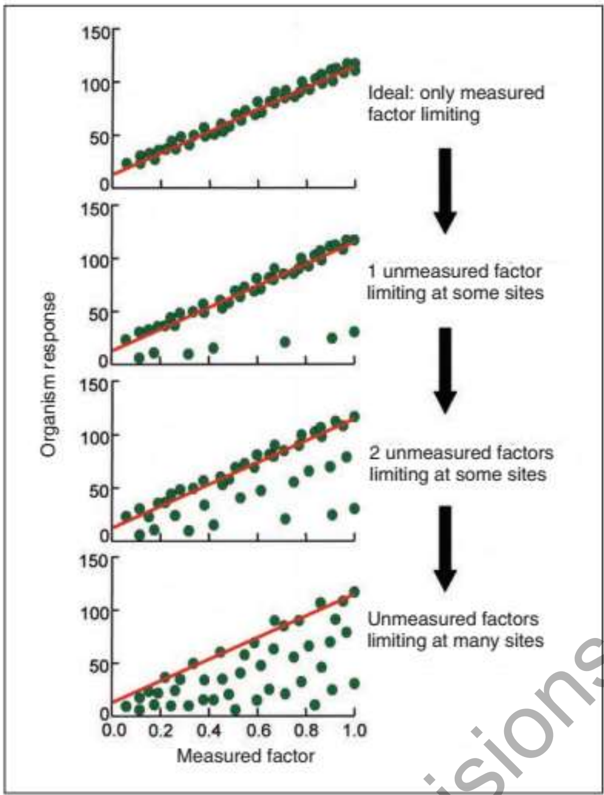

Figure 2. The top graph represents the ideal statistical situation where an organism response is driven primarily by the measured factor(s) included in the linear regression model; ie all other potential limiting factors are at permissive levels. As we proceed from top to bottom, an increasing number of factors that were not measured become limiting at some sample locations and times, increasing the heterogeneity of organism response with respect to the measured factor(s) included in the regression model.

Reproduced from: Cade, B. S. & Noon, B. R. 2003. A gentle introduction to quantile regression for ecologists. *Frontiers in Ecology and the Environment*, 1 (8), 412-420. Available: DOI doi:10.1890/1540-9295(2003)001[0412:AGITQR]2.0.CO;2

Quantile Regressions have been applied in New Zealand to relate periphyton and macroinvertebrate communities to nutrient concentrations (Matheson *et al.*, 2016, Matheson *et al.*, 2012). Here I related Fish IBI with nitrate-nitrogen. When the Fish IBI (Joy and Death, 2004) is calculated for sites from the NZFFD and correlated against predicted nitrate-nitrogen concentrations (Larned *et al.*, 2015) an upper limiting quantile is clear. The blue line represents the 85th percentile, red the 90th percentile and green the 95th percentile. As a guide, Joy and Death (2004) suggest that an IBI between 0-20 represents poor health, 20-40 represents moderate health and 40-60 represents good health. Examining Table 2 suggests that to safeguard IBI at a minimum of 20, then bottom-line nitrate-nitrogen concentrations should be between 0.76-1.29 mg/L. The proposed bottom-line of 0.89mg/L sits comfortably within this range at the slightly more conservative end, though a highly precautionary approach may adopt for the more stringent 0.76mg/L.

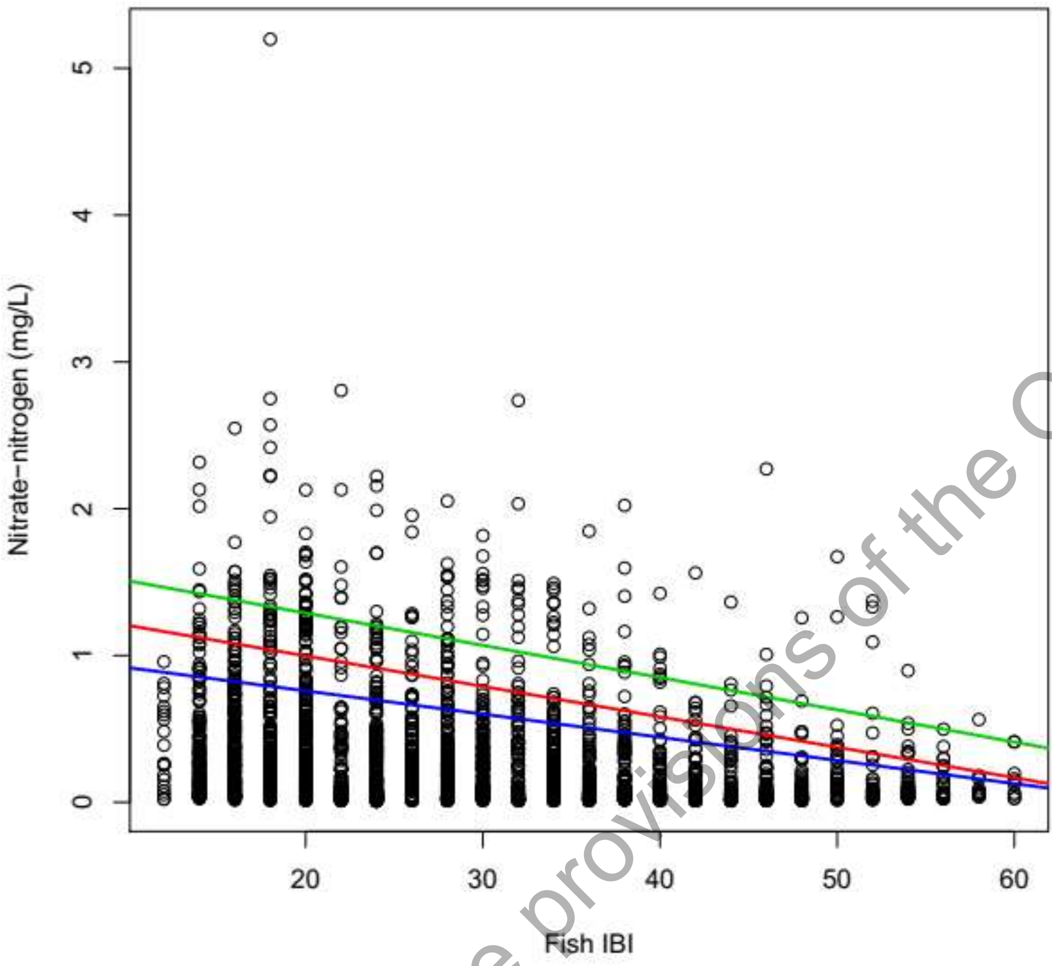

Table 2. Nitrate-nitrogen concentrations derived from three quantile regressions against the Fish IBI			
IBI	85th %ile	90th %ile	95th %ile
50	0.28	0.37	0.63
40	0.44	0.58	0.85
30	0.60	0.79	1.07
20	0.76	1.00	1.29
10	0.92	1.20	1.51

Proposed nutrient attribute tables

Value	Ecosystem health	
Freshwater Body Type	Rivers and groundwater	
Attribute	Nitrate-nitrogen (Ecosystem Health)	
Attribute Unit	Milligrams of nitrate-nitrogen	
Attribute State	**Numeric Attribute State**	**Narrative Attribute State**
	Nitrate-nitrogen (NO_3-N) – Annual median[1]	**Description**
A	≤ 0.10	Minimal nitrate-nitrogen enrichment
B	> 0.10 and ≤0.28	Mild nitrate-nitrogen enrichment
C	> 0.28 and ≤ 0.46	Moderate nitrate-nitrogen enrichment
D	> 0.46 and ≤ 0.89	Substantial nitrate-nitrogen enrichment
National Bottom Line	0.89	
E	> 0.89 and ≤ 1.32	Severe nitrate-nitrogen enrichment
F	>1.32	Highly severe nitrate-nitrogen enrichment

Value	Ecosystem health	
Freshwater Body Type	Rivers and groundwater	
Attribute	Phosphorus (Ecosystem Health)	
Attribute Unit	Milligrams of dissolved reactive phosphorus (DRP) per litre	
Attribute State	**Numeric Attribute State**	**Narrative Attribute State**
	Dissolved reactive phosphorus (DRP) - Annual median[1]	**Description**
A	≤ 0.006	Minimal DRP enrichment
B	> 0.006 and ≤0.013	Mild DRP enrichment
C	> 0.013 and ≤ 0.019	Moderate DRP enrichment
D	> 0.019 and ≤ 0.038	Substantial DRP enrichment
National Bottom Line	0.038	
E	> 0.038 and ≤ 0.057	Severe DRP enrichment
F	>0.057	Highly severe DRP enrichment

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