Hon Dr Nick Smith
Freshwater Consultation 2017
Ministry for the Environment
PO Box 10362
Wellington 6143

28 April 2017

Dear Minister,

FEEDBACK ON THE 'CLEAN WATER' DISCUSSION DOCUMENT AND THE PROPOSED FRESHWATER REFORMS 2017

CHOOSE CLEAN WATER NZ

Introduction

1. Choose Clean Water NZ is a student-led group that has been campaigning for strong protection for fresh water, rivers, streams, and lakes for the past two years. We call ourselves Choose Clean Water because we believe that the continued degradation and pollution of New Zealand's waterway[1] is not an inevitability. With political and public will, we can choose clean, safe fresh water for this country. New Zealanders widely choose clean water above short-term economic growth and identify that our country has a serious freshwater contamination, as has been repeatedly demonstrated by not only the public support of our petition to raise the bottom line from 'wadeable' to swimmable[2], but also submissions to the Ministry for the Environment on previous iterations of the National Policy Statement for Freshwater Management (NPS-FM)[3], and informal and formal polling carried out on the subject[4],[5]. The public are clear that clean safe fresh water must be the priority to protect the health of our people, our wildlife and our environment. They are desperate for the Government to choose clean water too and commit to substantial changes to the NPS-

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[1] http://www.mfe.govt.nz/publications/environmental-reporting/our-fresh-water-2017

[2] http://www.newshub.co.nz/home/politics/2016/05/petition-hopes-to-improve-govt-aims-on-water.html

[3] http://www.mfe.govt.nz/node/21977/

[4] http://www.stuff.co.nz/environment/83738717/two-meetings-two-visions-for-canterburys-freshwater

[5] http://www.lincoln.ac.nz/Documents/LEaP/perceptions2016_feb17_LowRes.pdf

FM and its associated projects.

2. While we are pleased to see the shift in focus of management of waterways from a 'wadeable' (secondary contact) standard to swimmable (primary contact) standards, it appears clear that the Government still does not choose clean water for our country or our people. Instead, the Government has taken the extraordinary good will of ordinary Kiwis around the country from all walks of life and chosen to confuse, undermine and dishonour them and their goals. They have done so by redefining swimming standards and covering only 10% of the country's waterways in the NPS-FM by only applying standards to fourth order and above rivers and lakes with perimeters longer than 1500 metres[6].

3. Because young New Zealanders face unprecedented environmental problems[7], we believe that protecting our natural environment, particularly clean safe fresh water, must be our priority and that the NPS-FM is a crucial part of establishing this priority for both central and local government. For many years, we have been told that the Government is "balancing" environmental and economic needs but we have ended up with widespread contamination of freshwater and a GDP that is declining if we take into consideration the cost of pollution to the economy (which currently goes largely unaccounted for)[8]. This strategy is clearly not working. It is delusional and is risking our environmental and economic stability as well as our social and cultural well-being.

4. We must be clear about what the NPS-FM is really for and what it must be aiming to achieve. The NPS-FM must be a document that aims to protect and enhance Te Hauora o te Taiao (the health of the environment), Te Hauora of te Wai (the health of the water body) and Te Hauora o te Tangata (the health of the people) and that these take precedence over economic goals. It must acknowledge that the health of people, the environment and waterways are linked and have the goal of integrated management, taking into account the relationship between the land and water, from source to sea. It cannot be a tool for driving economic growth that erodes and exploits our natural heritage and cultural identities.

5. Our team has a background in ecology, environmental management and agricultural science and, as we are in our twenties and thirties, we have a significant proportion of our lives ahead of us. We are determined to improve New Zealand's freshwater management to ensure present and future New Zealanders, our wildlife and this country are respected and

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[6]http://www.mfe.govt.nz/fresh-water/freshwater-management-reforms/clean-water-package-2017

[7]http://royalsociety.org.nz/assets/Uploads/Climate-change-implications-for-NZ-2016-report-web.pdf

[8] http://www.oecd.org/newzealand/environmental-pressures-rising-in-new-zealand.htm

protected.

6. If the NPS-FM remains as it is, a justification for ongoing pollution of fresh waterways and a vehicle to push economic growth far beyond environmental limits, it will condemn younger New Zealanders and future generations to unsafe and extremely difficult lives as poor freshwater management will be amplified by the increasing effects of climate change.

7. If the NPS-FM remains as it is currently, without significant overhaul (outlined below), Choose Clean Water, its supporters and the New Zealand public will continue to fight the Government and the Ministry for valuable policy and meaningful change to the country's land and water management. We are saddened that our interactions with the Government and the Ministry can be described as a fight as, we and other environmental groups, along with many in the scientific community and the public have spent an enormous amount of time and resource trying to engage and develop better freshwater policy and management but have been largely ignored, belittled or our words manipulated. This fight has been created by the response from Government and the Ministry as we can all agree that it is reasonable for the public to call for and expect clean safe swimmable fresh water to be the responsibility of the New Zealand Government to deliver to all its citizens and we can also agree that its response has been inadequate at best and actively undermining and destructive at worst.

Attributes in National Objectives Framework

8. We support the inclusion of a genuine swimmable bottom line, returning to the Ministry of Health's guidelines for *E. coli* (where the acceptable standard is a 95th percentile E. coli count of 260/100mL)[9].

   It has been attempted to justify the redefinition of swimming standards to bands based on the proportion of time a site is below 540/100mL by arguing that councils will be required to provide better information on the contamination levels of a waterbody at a given time because sometimes rivers will exceed safe swimming standards. This justification does not preclude having better standards. Regardless of what our standards are, councils have an important role in communicating the level of contamination of rivers. Strong standards require communication as do weak ones. Better communication is not an argument for weakening standards. New Zealanders should not have to have an internet connection to be able to swim safely in rivers. We should protect the quality of our water and hold ourselves to high standards. That *E.coli*

[9] http://www.mfe.govt.nz/publications/fresh-water/microbiological-water-quality-guidelines-marine-and-freshwater-0

counts may become elevated at high flows is documented and taken into account within the 95th percentile target (i.e. it is not a 100% target).

9. We support the inclusion of ecosystem health bottom lines for major nutrient contaminants, phosphorous and nitrogen (represented by Dissolved Reactive Phosphorous and Nitrate limits).

   Because these nutrients are the drivers of algal blooms that choke life in rivers and lakes, they must be controlled for ecosystem health not toxicity. The Parliamentary Commissioner for the Environment has identified nutrients as one of three major pollutants of freshwater in New Zealand[10], along with pathogens and sediment. Dr Russell Death, Professor of Freshwater Ecology at Massey University, has established DRP and Nitrate limits for ecosystem health. These are provided in full in Appendix A, with support material in Appendix B.

10. We support the inclusion of deposited sediment limits, as currently this major contributor to the pollution of waterways is ignored in the NPS-FM.

    Erosion is a natural process but human activity greatly accelerates erosive processes and leads to smothered habitats, water that is unsafe for humans and aquatic wildlife, and can affect drinking water supplies as recently witnessed in Auckland.[11]

11. We support the inclusion of a Habitat Quality Index, to ensure that our approach to freshwater management protects the character and beauty of natural rivers and does not turn them into soulless drains.

    It is possible to strive for good water quality while ignoring the importance of natural character to the health of fresh waterbodies. Rivers and lakes in a natural state (including meanders, vegetative cover, stony bottoms, etc.) provide vital habitat for wildlife and also preserve the natural beauty of our country for our people and for our tourism industry that relies on this for its prosperity.

    Dr Russell Death's Habitat Quality Index attribute table has been provided in Appendix A.

12. We support the inclusion of an actual measure of ecological health (the Macroinvertebrate Community Index - MCI) as an attribute to ensure an environmental bottom line and the requirement that councils take action to improve MCI scores in waterways where they are low or declining.

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[10] http://www.pce.parliament.nz/publications/water-quality-in-new-zealand-understanding-the-science

[11] http://www.nzherald.co.nz/nz/news/article.cfm?c_id=1&objectid=11823425

Simply measuring MCI scores is not sufficient nor is it a step forward as many councils already do this as a matter of course. MCI must be a tool for measuring the health of a waterway but also as a tool for action to improve the health of waterways. The NPS-FM must include the requirement for councils to take action where scores are low or declining.

13. It appears that the Government and the Ministry for the Environment is reluctant to have waterways fail standards and so have sought to manipulate standards in order to achieve the best possible statistics (the redefinition of swimming are an example of this where sites that met swimming standards jumped from 38% nationally to 72% overnight). To have truly meaningful standards, we must be prepared to acknowledge the damage that has already been done. Then, rather than seek to establish standards on a new degraded scale, we must stick to the values we have as New Zealanders and aim for excellent water quality by established standards that reflect and respect our people's' deep connection to their waterways and their desire to have safe clean freshwater for present and future generations.

14. We do not support the new "swimming" standards, as they have redefined what is safe for swimming and made swimming standards worse.

The table on the Ministry's website on the redefined swimming standards does not appear in the consultation document (except as a link that was added the day after the policy was released), we therefore have to conclude that it is not part of the policy. We are also reasonably sure this is the case due to the Minister's comments where he is quoted as saying, "the Government **intends** to implement its commitment to swimmability of waterways includes **consideration** of the range of values included in Table 1 on the Ministry for the Environment's website."[12] There has also been no written confirmation from the Minister or the Minister's office that this table is an official part of the policy.

Even if this table were part of the policy, it still appears to be a worsening of standards (from the Ministry of Health's guidelines where an 'A' grade was a 96th percentile of <130/100mL)[13].

What is the purpose of standards that are so confusing that even New Zealand's top freshwater scientists cannot interpret them? How can the Ministry expect council staff to be able to use a standard that is incomprehensible to University Professors?[14]

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12 http://www.voxy.co.nz/politics/5/276607

13 http://www.mfe.govt.nz/publications/fresh-water/microbiological-water-quality-guidelines-marine-and-freshwater-0

14 http://www.radionz.co.nz/news/national/329349/swimmable-waterways-standard-

The Minister and the Ministry have also not provided information requested by the President of the New Zealand Freshwater Sciences Society to the show if the new standards were in line with the old guidelines, and the overseas standard by comparing which of New Zealand's rivers would meet the swimmability standard under each set of guidelines despite this being raised with Environment Minister Nick Smith after the clean package was announced and Dr Smith saying the Ministry would do this analysis[15].

15. We support the application of E.coli guidelines and criteria for primary contact recreation to lakes as well as rivers.

16. We do not support use of nitrate toxicity as a measure for ecosystem health and call for this attribute to be removed and replaced with definitions for nitrate for ecosystem health in point 8 above

    Fresh water that is high in nitrates is an extreme, cruel and reckless burden to place on younger generations and those to come. Climate change will amplify the effects of high nutrient concentrations in waterbodies (i.e. increased frequency and severity of algal blooms). To retain nitrate toxicity as a bottom line for waterways is to knowingly condemn young and future New Zealanders to very difficult lives for the sake of short-term economic growth.

17. We oppose the proposal to define 'maintain' as staying within the same attribute band for the reasons provided by the New Zealand Freshwater Sciences Society in their submission to this consultation.

Preamble

18. We support establishing in the preamble that Te Hauora o te Taiao (the health of the environment), Te Hauora of te Wai (the health of the water body) and Te Hauora o te Tangata (the health of the people) are the focus of the policy and that they are all linked, with the goal of integrated management taking into account the relationship between the land and water, from source to sea.

19. We see the preamble as establishing the context for the interpretation of the policy. It must therefore reflect the values of New Zealanders and responsibility of the Ministry to responsibly protect the environment rather than the economic goals of the current Government.

20. We do not support the existing preamble as it prioritises economic

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'confusing'-scientist

15 http://www.radionz.co.nz/news/national/329349/swimmable-waterways-standard-'confusing'-scientist

considerations and establishes economic considerations clearly as a priority for New Zealand's primary document on freshwater protection from the Ministry for the Environment.

Freshwater Improvement Fund

21. We do not support the use of public money, provided through the Freshwater Improvement Fund, to finance irrigation schemes.

    Large-scale irrigation schemes have led to the increase of pressure on aquatic ecosystems and groundwater, particularly in Canterbury. There appears to be little evidence to suggest that they have net positive environmental outcomes and a body of evidence to suggest they contribute negatively to the the degradation of freshwater systems. This is surely contrary to the purpose of the NPS-FM or, at least, is contrary to what the New Zealand public expect is the purpose of the NPS-FM.

22. We support instead the reallocation of the $400 million earmarked for irrigation schemes to the establishment and work of an Agricultural Transition Fund to be used to research, innovation and implementation of strategies to transition New Zealand away from damaging models of agriculture and their proliferation towards sustainable models of agriculture.

Economic considerations

23. We do not support the inclusion of any new test that requires a balancing of environmental considerations against economic ones as in practice this will be used to set freshwater objectives or quantity limits that do not achieve ecosystem health, which is unacceptable.

24. We do not support and request the removal of the addition of "while providing for economic well-being, including productive economic opportunities."

Targets

25. Inclusion of a requirement for regional councils to meet specific water quality targets within realistic timeframes; this means councils must have a plan and begin implementation of waterway protection and improvement across their region by 2020 with specific and genuinely ambitious programme of targets for improvement in the next 5, 10 and 20 years.

26. Targets should not be national but rather regional so that all regional

councils are required to make improvements.

27. 2022 deadlines for fencing and stock exclusion being brought forward to 2020; 2025 deadlines should be brought forward to 2022; 2030 deadlines should be brought forward to 2025.

Other

28. We suggest that Managed Aquifer Recharge should be banned in the NPS-FM as it is an untested and potentially damaging practice that does not address the root cause of groundwater degradation.
29. We support the OECD recommendations in relation to freshwater management and policy, including an independent review of the NPS-FM to establish if it can meet its own goals of safeguarding human and ecosystem health as well as meeting public expectation.
30. We support the OECD recommendations that freshwater pollution charges are implemented.
31. We support the urgent address of overallocation nationally with a view to prompt central Government action to address this problem.

Conclusion

32. This has been a long process of consultation. What New Zealanders really want is reliable and accurate information informing accurate and uncompromised ecosystem and human health standards that drive meaningful action to save our rivers, lakes, streams, wetlands, aquifers and estuaries from ongoing pollution.

33. We ask that the Minister and the Ministry remembers that they work for the public not private interests and that the public good requires the support of Government and government.

34. We ask also that the Ministers and the Ministry staff reading this submission remember that their children, nieces, nephews, grandchildren are relying on them to stand up for clean, safe freshwater and swimmable, healthy rivers and lakes.

Regards,

Marnie Prickett

Organiser & Spokesperson
on behalf of Choose Clean Water

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

Additional keywords: Ecological health, eutrophication, New Zealand, multiple lines of evidence, nutrient criteria, nutrients, river management,

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 ecological health (states A, B, C and D). We compiled a large range of data sources and used a weight-of-evidence approach to objectively determine nitrate and dissolved reactive phosphorus (DRP) limits to manage rivers and streams into the four ecological states. This established that the critical nutrient concentrations differentiating rivers in each of the states are 0.08, 0.39 and 1.33 mg/l for nitrate and 0.006, 0.024 and 0.068 mg/l for DRP. While ecological health of rivers is affected by a range of interacting stressors we believe the evidence supports the view that managing to these nutrient thresholds will provide for better ecological condition in New Zealand's rivers and streams.

Introduction

Globally freshwater biodiversity is under considerable threat from a wide variety of anthropogenic stresses (Dudgeon *et al.* 2006; Dudgeon 2010; Vorosmarty *et al.* 2010). This decline in biodiversity has resulted from multiple interacting stressors (Matthaei *et al.* 2010; Wagenhoff *et al.* 2011; Piggott *et al.* 2012; Leps *et al.* 2015) including water abstraction for consumptive and agricultural needs (Dewson *et al.* 2007; Poff and Zimmerman 2010; McDowell *et al.* 2011), invasive species (Olden *et al.* 2010; Collier and Grainger 2015), channelization, sedimentation, eutrophication (Carpenter *et al.* 1998b; Allan 2004) and changing climate regimes (Palmer *et al.* 2008; Death *et al.* 2015b). 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 (Pretty *et al.* 2003; Dodds *et al.* 2009; Jarvie *et al.* 2013). 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; Verburg *et al.* 2010; Parlimentary Commisioner for the Environment 2013; Joy and Death 2014; Foote *et al.* 2015; Joy 2015). 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 (Davies-Colley and Nagels 2002; Ballantine and Davies-Colley 2010; Unwin and Larned 2013; Foote *et al.* 2015; Ministry for the Environment and Statistics New Zealand 2015). Most sites in lowland pastoral catchments and all sites in urban catchments exceed safe swimming standards for pathogens and 60% of sites have increasing nitrogen levels (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 and 84% of lakes in pastoral catchments are the same (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 and funding initiatives in an attempt to improve water quality (Ministry for the Environment 2004; Cullen *et al.* 2006; Hughey *et al.* 2010; Ministry for the Environment 2014; Joy 2015). However, the 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 (Wilcock *et al.* 2007; Rutherford 2013; Chisholm *et al.* 2014). 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 phosphorus criteria for lakes, but it only establishes nutrient criteria (i.e. nitrate) for rivers at toxic levels, not to manage ecological health (Ministry for the Environment 2010, 2014). Despite the obvious and extensively documented links between high nutrient levels in rivers and declines in ecological health (Biggs 1996, 2000a; Death *et al.* 2007; Clapcott *et al.* 2012; Collier *et al.* 2013; Death *et al.* 2015a), current government policy does not provide mechanisms to manage nutrients to safeguard ecological health.

Many countries have established nutrient criteria or thresholds to protect aquatic life in their waterways (Dodds and Welch 2000; Camargo and Alonso 2006; Smith and Tran 2010; Jarvie *et al.* 2013; Heiskary and Bouchard 2015). There are three 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. 25^{th}, 50^{th} 75^{th} percentile). Expert-opinion uses the knowledge of a range of experts to determine the critical levels of a stressor where 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.* (2013)). 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. Smith and McBride (1990); Smith and Maasdam (1994); Scarsbrook *et al.* (2000); Larned *et al.* (2004); Clapcott *et al.* (2012); Unwin and Larned (2013)).

In this study we adopt the weight-of-evidence approach 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’ where 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 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 characterise water bodies as belonging to one of five states of ecological status from bad to high (European Commission 2000; Birk *et al.* 2012; Poikane *et al.* 2014). Our work improves on the existing nutrient limits for New Zealand's rivers through the use of multiple lines of evidence from real data rather than expert opinion and by defining states to safeguard for ecological health rather than for toxicity.

Materials and methods

Methods for nutrient identifying criteria

There are four established methods for identifying nutrient limits (USEPA 2000; Smith and Tran 2010). These are 1) division of known nutrient measures into equal classes (percentile analysis); 2) identification of significant change points in the relationship between nutrient values and ecosystem health metrics (King and Richardson 2003; Baker and King 2010; Smith and Tran 2010); 3) identification of signification relationships between nutrient values and ecosystem health metrics at predetermined points; 4) experimental manipulation of the effect of nutrient values on ecosystem health metrics. For this study approaches 1 and 3 have been used to set thresholds for both nitrate and dissolved reactive phosphorus (DRP). A combination of real and modelled data were sourced from a variety of publications and agencies (Table 1) and threshold limits were determined by weighting each line of evidence based on whether the effects 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 (Larned et al. 2004; McDowell et al. 2009; Ballantine and Davies-Colley 2010; Larned and Unwin 2012; Unwin and Larned 2013). Unwin and Larned (2013) have compiled data, from 786 water quality sites, collected 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 correlated with actual measures (r=0.64 and r=0.83, for nitrate and DRP, respectively) made at 22 Manawatu rivers and streams (R Death, pers. comm.) and at 77 National River Water Quality Network (NRWQN) sites (r=0.86 and r=0.73, for nitrate and DRP, respectively). As this modelled data gave a more extensive, consistent and spatially unbiased measure of nitrate and DRP the data was used to assess percentiles.

To assign sites into percentile groups, based on their nitrate and DRP values, we used the percentile analysis approach of (USEPA 2000; Smith and Tran 2010). This method splits sites into their respective percentiles based on pre-defined values. For data from all sites we used the 25^{th}, 50^{th} and 75^{th} percentiles for A, B, C and D thresholds for nitrate and DRP following the USEPA (2000) approach. For sites in the Conservation Estate (n = 242,521) that are relatively pristine we used the 95^{th}, 99^{th} and 99.9^{th} percentiles. These sites will reflect natural geographic and geological variation in nitrate and DRP levels but have little or no anthropogenic nutrient influences; 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 ecosystem health (Table 1: datasets 2-8). We could not identify any breaks or change-points in the relationship suitable for changepoint analysis (King and Richardson 2003; Smith and Tran 2010). However, New Zealand has well established biological indicator criteria for the Macroinvertebrate Community Index (MCI) and its quantitative variant (QMCI) (Quantitative Macroinvertebrate Community Index) (Stark 1985; Boothroyd and Stark 2000; Stark and Maxted 2007). These have been in place since 1985 and are now widely used in all environmental monitoring 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 eutrophication (Stark 1985) and have been shown to be insensitive to heavy metals, acid mine drainage and deposited sediment (Hickey and Clements 1998; Gray and Harding 2012; Death and Death 2014). The 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 thresholds 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 from their relationship with the QMCI criteria. For the metrics EPT(taxa) (percent of EPT taxa in all taxa) and EPT(animals) (percent of EPT individuals in all collected animals) thresholds were determined by regressing the metric against QMCI and using the values of the equation at QMCI = 4, 5 and 6 (EPT(animals) = 23%, 39% and 55% (r^2=0.81) and EPT(taxa) = 30%, 38% and 46% (r^2=0.55).

The datasets utilized 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.

Modelled nutrient and ecosystem health metric relationships (Table 1: datasets 1 vs. 2)- Clapcott *et al.* (2103) 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 predicted MCI values against the modelled nutrient values from Unwin & 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 & Larned (2013).

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 & 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 & 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= 55%, 39% and 23% for EPT(animals), and y = 46%, 38% and 30% for EPT(taxa).

Previously published numerics and ecosystem health metric relationships

Several previous publications have investigated nitrate 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 & 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

Each regression of the datasets was used to determine the numerical nutrient limits for each ecological state (Table 2). The final nutrient limits were determined by calculating a weighted average of those 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 a weighted value of 2 in the analysis and purely statistical or less direct linkages were allocated a weighted value of 1 (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.

Results

Ecosystem health metric relationships

The relationships between the health metrics and nutrient concentrations were predominantly exponential (Table 2) with health declining more rapidly for increasing nutrient concentrations at low levels and plateauing as ecological health approached poor condition (e.g. Fig. 1). 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. This yielded nitrate concentrations of 0.08, 0.39 and 1.33 mg/l, and DRP concentrations of 0.006, 0.024 and 0.068 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 (Standard Error = 0.02, 0.05 and 0.29 for the three nitrate criteria and 0.001, 0.003 and 0.020 for the three DRP criteria). 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 0.07- 0.08, 0.33-0.39 and 1.09-1.42 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.

The percentage of New Zealand river reaches with median nitrate or DRP levels from Unwin and Larned (2013) in each of these attribute states is given in Table 3.

Discussion

Although the ecological health of rivers and streams is determined by a wide range of potentially interacting stressors, it is clear that nutrients are one of the most pervasive and detrimental stressors for the fauna and flora of rivers globally (Carpenter *et al.* 1998a; Allan 2004; 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, rather than the heavy industry or manufacturing that dominates in many other places (Foote *et al.* 2015; Joy 2015; Weeks *et al.* 2016). The developed nitrate concentrations of 0.08, 0.39 and 1.33 mg/l, and DRP concentrations of 0.006, 0.024 and 0.068 mg/l will therefore be very valuable policy tools to maintain or improve the ecological health of rivers in good, moderate or poor condition.

Although there can be many situations where expert opinion rather than data are necessary to establish management objectives 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 (Hughey *et al.* 2010; Foote *et al.* 2015; 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 impact on 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 (Table 2) 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=963, cross-validated correlation coefficient = 0.65) and 50%

(cross-validated correlation coefficient = 0.76), of the modelled MCI and QMCI, respectively, from 15 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 (Matthaei *et al.* 2010; Wagenhoff *et al.* 2011; Clapcott *et al.* 2012; Wagenhoff *et al.* 2012). Similarly, establishing limits for only nitrate 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 (Dodds and Welch 2000; Death *et al.* 2007; Keck and Lepori 2012; Jarvie *et al.* 2013).

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 and Richardson (2003); King *et al.* (2005); Smith and Tran (2010)). However, there was no evidence of thresholds in any of the ecological metric nutrient relationships examined in the compiled data. Rather than any 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 indices (e.g. MCI, QMCI). Although, somewhat subjective these thresholds have been in use for a long time in river management (Stark 1985, 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 too low. Once ecological health reached that point the long flat tail of the relationship (e.g. Fig. 1) 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; USEPA 2000; Smith and Tran 2010). 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 0.08, 0.39 and 1.33 mg/l, and DRP concentrations of 0.006, 0.024 and 0.068 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 every 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	No. sites	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	24 Manawatu	Metric relationship	1999-2011	Nitrate, DRP, MCI, QMCI	4 Death (2013)

Animal Targets (FAT) model[B]	streams multiple temporal measures (inverts yearly, nutrients monthly)				
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	Summary table 1-3	Not stated	Periphyton measured as	8 Matheson *et al.* (2016)

	and Regional Council data from throughout New Zealand	from regression analysis.		chlorophyll *a*	
ANZEC guidelines	Table 3.3.10			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 963 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 New Zealand National Policy Statement for freshwater state (A-D) derived from multiple lines of evidence. Weighting of each piece of evidence is provided along with regression statistics (F statistic, degrees of freedom, probability value and r^2) when relevant. See Table 1 for details on source data. PCI = public conservation land.

Source nutrient dataset	1	1 PCI only	1	1	1	1	1	1	4	4	5	5	1	9,10	7	8	
Source ecological dataset	n/a	n/a	2	2	3	3	3	3	4	4	5	5	6	n/a	7	8	
Ecological metric	n/a	n/a	MCI	QMCI	MCI	QMCI	EPT animals	EPT taxa	MCI	QMCI	MCI	QMCI	IBI	n/a	Chl a	Chl a	
NO_3																	
Equation	n/a	n/a	ln y = ln (x+1)	ln y = ln (x+1)	y = ln x	y = ln x	y = ln x	y = ln x	y=lnx	y=lnx	y=x	y=lnx	y=lnx	n/a	log10(max Chl a) = x	See Matheson et al 2016	Weighted mean
Weight of	1	1	2	2	2	2	2	2	2	2	2	0	1	2	2	2	

evidence																	
A/B threshold	0.03	0.08	0.02	0.00	0.11	0.10	0.11	0.20	0.06	0.09	0.00	0.00	0.00	0.17	0.03	0.10	0.08
B/C threshold	0.06	0.12	0.45	0.29	0.58	0.34	0.30	0.47	0.53	0.33	0.60	0.13	0.21		0.10	0.63	0.39
C/D threshold	0.28	0.20	1.22	0.77	3.01	1.09	0.87	1.09	4.36	1.20	1.60	9.10	1.54	0.44	0.20	1.10	1.33
r^2			0.53	0.54	0.35	0.27	0.28	0.29	0.37	0.27	0.08	0.04	0.09		0.3		
F			632224	653084	513	363	377.6	390.6	51.72	32.66	6.78	3.85	3775				
df			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	0.01	0.05	<0.0001				
DRP																	
Equation	n/a	n/a	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	n/a	$\log_{10}$(max chla) = x	See Matheson etal 2016	Weighted mean
Weight of evidence	1	1	2	2	2	2	2	2	2	2	0	0	1	2	2	2	

A/B threshold	0.004	0.011	0.004	0.003	0.008	0.006	0.005	0.015	0.005	0.008	0.000	0.000	0.002	0.009	0.002		0.006
B/C threshold	0.008	0.014	0.016	0.012	0.022	0.015	0.009	0.021	0.038	0.025	0.023	0.008	0.007		0.007	0.120	0.024
C/D threshold	0.012	0.021	0.032	0.024	0.040	0.027	0.016	0.028	0.275	0.079	0.066	0.024	0.014	0.100	0.014	0.200	0.068
r^2			0.38	0.39	0.18	0.15	0.18	0.18	0.54	0.420	0.02	0.04	0.04		0.3		
F			349187	357979	210.3	165	217.80	211.10	99.83	63.89	2.160	3.610	15770				
df			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	<0.001	0.15	0.06	<0.0001				

Table 3. Percentage of river reaches in each nutrient attribute state. NPS state = New Zealand National Policy Statement for freshwater state.

NPS state	NO_3-N (mg/l)	Percent	DRP (mg/l)	Percent
A	< 0.08	54.4	< 0.006	37.4
B	0.08 ≤ x < 0.39	26.2	0.006 ≤ x < 0.024	58.4
C	0.39 ≤ x < 1.33	17.0	0.024 ≤ x < 0.068	4.2
D	> 1.33	2.4	> 0.068	0.02

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

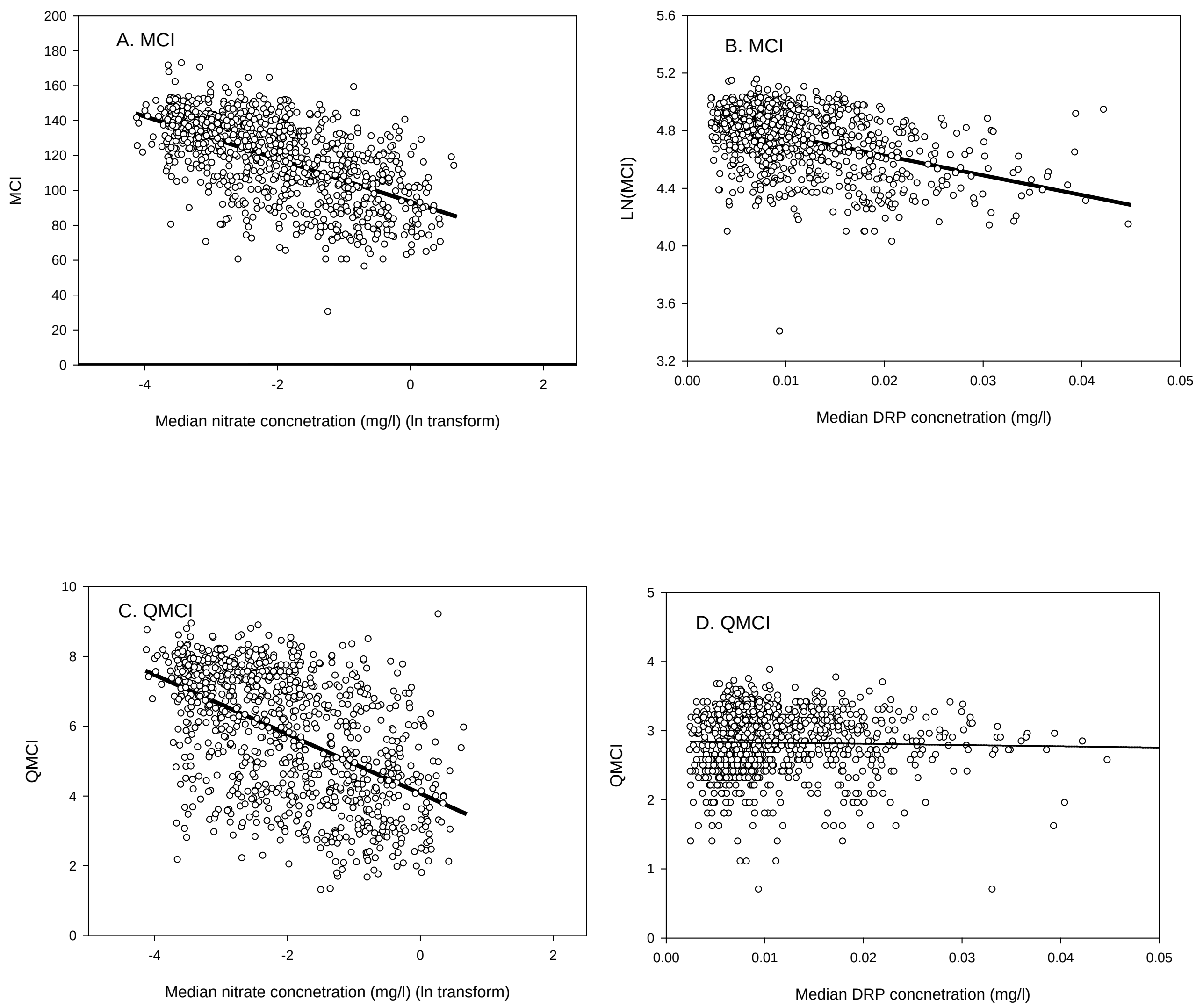
[1] https://www.epa.gov/sites/production/files/2014-08/documents/criteria-nutrient-ecoregions-sumtable.pdf

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



National Environmental Objectives Framework (NEOF)

(Defining ecologically relevant limits for rivers and streams in New Zealand)

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Executive Summary

1. The current National Objectives Framework (NOF) fails to provide adequate attributes to protect or maintain ecological health of New Zealand rivers and streams.
2. The MCI (Macroinvertebrate Community Index), nitrate, DRP (Dissolved Reactive Phosphorus) and NCI (Habitat Quality Index) need to be included in the NOF to adequately protect riverine ecological health.
3. MCI attribute states should be set at 120, 100 and 80 for the A/B, B/C, C/D thresholds.
4. Nitrate (annual median) attribute states should be set at 0.08, 0.39 and 1.33 for the A/B, B/C, C/D thresholds.
5. DRP (annual median) attribute states should be set at 0.006, 0.024 and 0.057 for the A/B, B/C, C/D thresholds.
6. NCI attribute states should be set at 0.8, 0.50 and 0.20 for the A/B, B/C, C/D thresholds.

Introduction

There has been concern for some time that water quality, ecological health and biodiversity of many of New Zealand's freshwater bodies has been declining (Verburg *et al.*, 2010; Parlimentary Commisioner for the Environment, 2013; Joy & Death, 2014; Foote, Joy & Death, 2015). This decline is a result of multiple interacting stressors (Matthaei, Piggott & Townsend, 2010; Wagenhoff *et al.*, 2011; Piggott *et al.*, 2012) including water abstraction for consumptive and agricultural needs (Dewson, James & Death, 2007b; Poff & Zimmerman, 2010), invasive species (Olden *et al.*, 2010) channelization, sedimentation, eutrophication (Carpenter *et al.*, 1998; Allan, 2004) and changing climate regimes (Palmer *et al.*, 2008; Death, Fuller & Macklin, 2015). 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 & Davies-Colley, 2010; Unwin & Larned, 2013; Foote, Joy & Death, 2015; Ministry for the Environment and Statistics New Zealand, 2015). Most sites in lowland pastoral catchments and all of the sites in urban catchments do not have pathogen standards that are safe for swimming and 60% of sites have increasing nitrogen levels (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 and 84% of lakes in pastoral catchments are the same (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 & Wall, 2007).

The condition of New Zealand's freshwater has become such an issue of national significance that the New Zealand Government has responded to this concern with a 'National Policy Statement for Freshwater Management 2014' to direct local government in the management of freshwater (Ministry for the Environment, 2010; Ministry for the Environment, 2014). It details in the 'National Objectives Framework' a number of measures (termed attributes) and their states (from A to D) identified by numerical thresholds, that are related to achieving the management value of Ecosystem Health. State D is the 'National Bottom Line' or minimum acceptable state with the intention that waterbodies will need to be improved to at least the national bottom lines over time. The minimum acceptable state is a confusing term as it actually represents an unacceptable or poor state, as referred to in earlier discussion documents (Ministry for the Environment, 2012). For 'Rivers' the attributes are Periphyton (Trophic state) (mg chl-a/m^2), Nitrate (Toxicity) (mg NO_3-N/L), Ammonia

(Toxicity) (mg NH_4-N/L) and Dissolved Oxygen (mg/L). The numerical thresholds for Periphyton attribute states were determined by expert opinion (Snelder *et al.*, 2013)
Many countries have established nutrient criteria or thresholds to protect aquatic life in their waterways (Dodds and Welch 2000; Camargo and Alonso 2006; Smith and Tran 2010; Jarvie et al. 2013; Heiskary and Bouchard 2015). There are three broad approaches; ecological, statistical and expert-opinion, that can be used alone or in combination (Birk et al. 2012). In setting the current numerical thresholds for ecological health the New Zealand Ministry for the Environment appears to have relied predominantly on expert opinion (e.g.(Snelder et al. 2013). While this approach can be useful when there is insufficient data to make more objective decisions this does not seem to the case in New Zealand where multiple parameters of water quality and ecological health have been monitored in New Zealand rivers for nearly 3 decades (e.g. (Smith and McBride 1990; Smith and Maasdam 1994; Scarsbrook et al. 2000; Larned et al. 2004; Clapcott et al. 2012; Unwin and Larned 2013).

We adopt the weight-of-evidence approach (Smith and Tran 2010) to develop several ecological health criteria for New Zealand rivers and streams to protect ecosystem health that we believe are missing from current policy. We adopt the Ministry for the Environment approach detailed in the 'National Objectives Framework' where 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 the national bottom lines over time (Ministry for the Environment 2014). This approach differs from that in the USA where nutrient criteria are derived for impaired / not-impaired waterways (Dodds and Welch 2000; USEPA 2000), but is similar to that of the European Water Framework Directive that also characterise water bodies as belonging to one of five states of ecological status from bad to high (European Commission 2000; Birk et al. 2012; Poikane et al. 2014).

Ecosystem/ ecological health?

The adoption of the term "life supporting capacity" in the Resource Management Act (1991) has provided some challenges for scientists attempting to manage freshwater resources as the term is not used in ecological science. To an ecologist crude oil has life supporting capacity for the dipteran larvae that thrive in it. The term used by ecological scientists that aligns most closely with the sentiment of the planners and policy makers that developed the RMA is probably “ecosystem or ecological health”. The concept of ‘ecosystem health’ is used widely in policy and science for the management of freshwaters (Costanza, Norton & Haskell, 1992; Reynoldson & Metcalfe-Smith, 1992; Scrimgeour & Wicklum, 1996; Rapport *et al.*, 1998; Boulton, 1999; Fairweather, 1999; Norris & Thoms, 1999; Davies *et al.*, 2010). The European Union Water Framework Directive for example seeks to attain “good ecological status” in freshwater bodies (European Commission, 2000). In New Zealand, the National Policy Statement for Freshwater 2014 recognises the importance of values relating to “safeguarding the life-supporting capacity of water and associated ecosystems” which include the value of “healthy ecosystem processes functioning naturally” (Ministry for the Environment, 2014). This contrasts with ecosystem integrity (e.g. (Schallenberg *et al.*, 2011) which is closer to a state where ecosystems might be in the absence of human impacts. Comparing it with a human example; an individual who has had his legs amputated can still be healthy but lacks integrity.

Ecosystem health has been variously defined as a combined measure of the vigour, organisation and resilience of an ecosystem (Rapport et al., 1998). In this context, ‘vigour’ can be related to the ability of an ecosystem to sustain life. In freshwaters, this may, for example, become impaired by the presence of a toxic pollutant. ‘Organisation’ relates to the extent of integration between ecosystem components. In freshwaters, this may, for example, become impaired by the extirpation of native species due to a change in habitat quality. ‘Resilience’ has been identified as an important feature of a ‘healthy’ ecosystem. In freshwaters, increasing pollutant levels or habitat loss could decrease the ability of the community to recover from natural disturbances such as floods and thus impair resilience (Death, Fuller & Macklin, 2015).

All these components in essence correspond to a situation where the composition, diversity and abundance of all species in a community are present for the given habitat characteristics (e.g. spring, braided river, headwater forest stream) in the absence of any external stress. This has been adopted by most developed countries as the goal and process for biomonitoring using RIVPAC type predictive modelling to assess impairment (Wright *et al.*, 2000; Wright, Sutcliffe & Furse, 2000; Reynoldson, Rosenberg & Resh, 2001; Bailey, Norris & Reynoldson, 2004; Linke *et al.*, 2005; Clarke & Murphy, 2006; Van Sickle, 2008). New Zealand freshwater managers have failed to adopt this approach and still rely on single metrics (e.g. MCI, QMCI, IBI) to assess ecosystem health. Thus the MCI, QMCI, IBI, Periphyton composition index are the closest measures of ecosystem health in use by freshwater managers in New Zealand.

Not surprisingly a considerable effort has been put into research assessing how, when and why a variety of anthropogenic stressors effect ecological health in rivers and streams (e.g. (Matthaei, Piggott & Townsend, 2010; Wagenhoff *et al.*, 2011; Piggott *et al.*, 2012) (Dewson, James & Death, 2007b; Poff & Zimmerman, 2010), (Olden *et al.*, 2010) (Carpenter *et al.*, 1998; Allan, 2004) (Palmer *et al.*, 2008; Death, Fuller & Macklin, 2015). We have captured these potential drivers of ecological health in Figure 1.

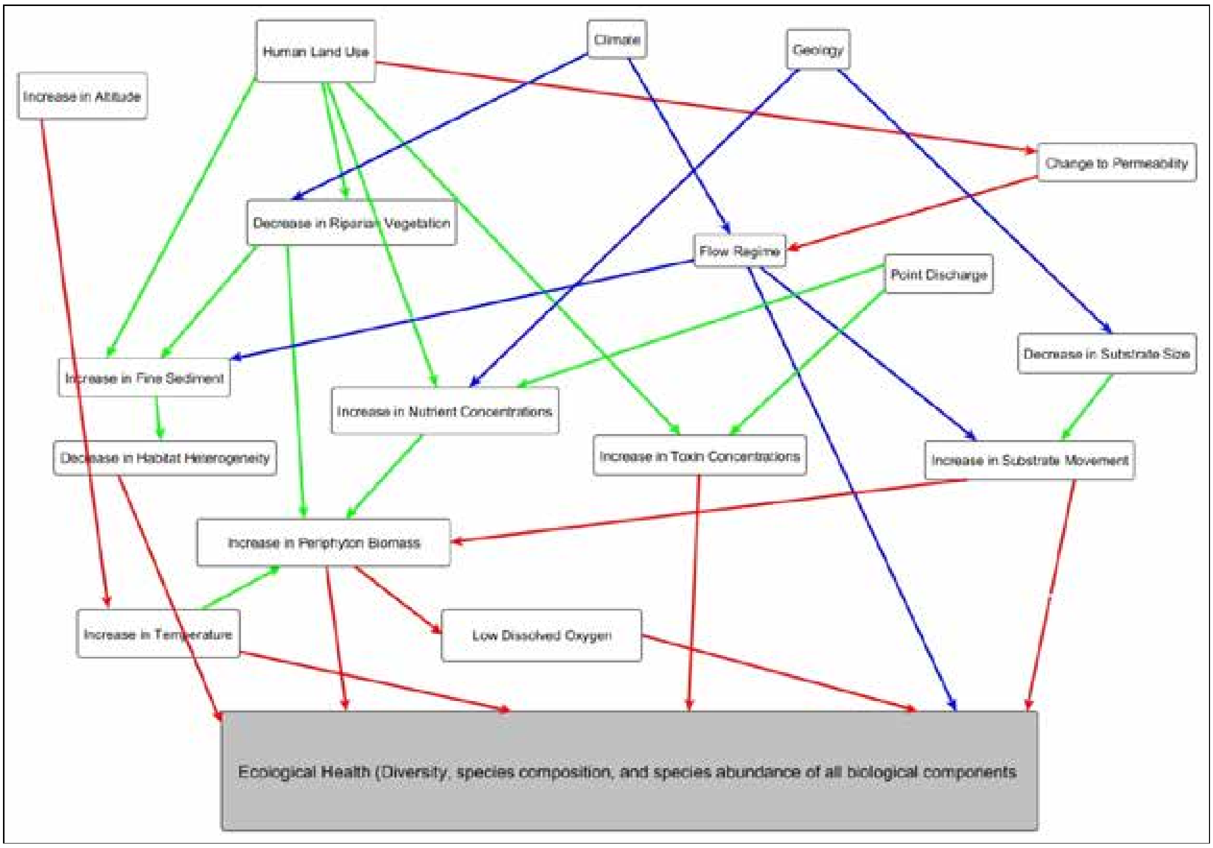

Figure 1 Variables that can affect ecological health in New Zealand rivers and streams.

The missing variables for ecological health from the National Objectives Framework (NOF)

The NOF has a number of attributes selected by the Ministry of the Environment through consultation with 60 freshwater scientists from throughout New Zealand. The attributes identified to protect ecosystem health of rivers include periphyton biomass, nitrate toxicity, ammonia toxicity and dissolved oxygen (for point source only). However, a quick scan of Figure 1 above shows that only 3 components of the many that can affect ecological health are addressed.

Nitrogen and phosphorus are attributes listed for lakes but not for rivers; which seems strange given the eutrophication of running waters is just as dependent on N and P as lakes. Periphyton is present as an attribute, which some would argue is why we would want to manage N and P (Fig. 1). But periphyton is extremely variable in space and time and extremely difficult to link with instream nutrient levels, let alone leaching of those nutrients from the land. As such, a number of attempts to manage periphyton by relating nutrient levels to periphyton biomass or cover have not been successful (Rutherford, 2011; Snelder *et al.*, 2013).

Nitrate toxicity is present as an attribute in the NOF for rivers, however, the toxicity of nitrate occurs at much higher levels of nitrate (6.9 – 9.8 mg/l) than where the nitrate has an adverse effect on ecosystem health (Appendix 1). These changes occur because of excess periphyton growth that in turn alters water chemistry, macroinvertebrate community composition and eventually fish abundance and biomass. This occurs at nitrate levels an order of magnitude lower than those for toxicity. A useful analogy is alcohol intake by humans, very high levels in the blood can be fatal but most people get very sick before fatally toxic levels of alcohol are ever reached.

The other unusual absence from the NOF is any measure of bioassessment of ecological health. In New Zealand the Macroinvertebrate Community Index (MCI) (Stark, 1985) has been used widely for 25 – 30 years by Regional Councils and is now used throughout New Zealand. Furthermore, although less widely used in New Zealand there is an index for the health of fish communities the Index of Biotic Integrity that could also be used as a direct assessment of ecological health (Joy & Death, 2004).

Finally, one of the biggest oversights for ecological health is a lack of any measure of habitat quality. A river may have plenty of clean water, but if there is no habitat for ecological communities there will not be healthy ecological communities. Again there has been some recent work in establishing deposited sediment limits for ecological health (Clapcott *et al.*, 2011; Burdon, McIntosh & Harding, 2013) and a coverage of 20% deposited fine sediment is present in the One Plan as an environmental limit. A Habitat Quality Index (NCI) has also been developed to assess the level of geomorphological / habitat diversity at the reach level (Death, Fuller & Death, unpublished) (Appendix 2).

It should be noted that the criteria assessed in this document are judged for their ability to protect ecosystem health, without any consideration of economic, social or cultural considerations. Based on our assessment of the current NOF attributes and the potential detrimental drivers of ecological health we believe there is a strong case for including nitrate, dissolved reactive phosphorus, MCI and the Habitat Quality Index as a mechanism of assessing habitat quality.

Attributes

These values are based on the current state of knowledge at the time of writing and may be modified as more information becomes available.

Macroinvertebrate Community Index

The Macroinvertebrate Community Index MCI was developed in 1985 (Stark, 1985). There are a variety of variants QMCI, SQMCI, Soft-bottom MCI. However, the MCI is the most widely used amongst environmental agencies and even many of the general public. There are well established numerical thresholds for the four attribute states (A, B, C and D) that have again been in widespread use for quarter of a century.

Nitrate

Nitrogen is a well-established stressor of ecological health that at high levels can result in waterbody eutrophication, before the chemical has toxic effects on aquatic organisms. We used a wide range of evidence sources to derive the numerical thresholds for four attribute states. The procedure is detailed in Appendix 1. We chose nitrate as the target chemical species for nitrogen as it is widely measured by Regional Councils, is usually the dominate chemical species of nitrogen in waterways, and is often closely linked with the other widely measured chemical species Dissolved Inorganic Nitrogen (DIN).

Phosphorus

Phosphorus, as with nitrogen, is a well-established stressor of ecological health that at high levels can result in waterbody eutrophication. We used a wide range of evidence sources to derive the numerical thresholds for four attribute states. The procedure is detailed in Appendix 1. We used dissolved reactive phosphorus (DRP), again, as it is the chemical form of phosphorus most widely measured by Regional Councils.

Habitat Quality Index (HQI)

Habitat structure and/or habitat quality is one of the most critical determinates of healthy ecological communities (i.e. you can have plenty of clean water but if there is nowhere for species to live they will be absent) that has been overlooked completely in the very water quality focused NOF. In part this may have been a result of the lack of suitable metrics to use in New Zealand for this purpose, but we have developed an index the HQI that can be used to assess this component of ecological health (Appendix 2).

These four attributes are placed in the NOF four state framework in the following pages.

Macroinvertebrate Community Index (MCI and MCI-sb)

Value	Ecosystem health		
Freshwater Body Type	Rivers		
Attribute	Macroinvertebrate Community Index	Soft-bottom Macroinvertebrate Community Index	
Attribute units	dimensionless	dimensionless	
Attribute State	Numeric Attribute State	Numeric Attribute State	Narrative Attribute State
	Three year median	Three year median	
A	≥120	≥120	Clean water. River ecosystem health high, similar to natural reference condition.
B	≥ 100 and < 120	≥ 100 and < 120	Doubtful quality or possible mild pollution. River ecosystem health good. Some degradation of life supporting capacity but ecosystem still functioning well.
C	≥ 80 and < 100	≥ 80 and < 100	Moderately polluted. River ecosystem health moderate to poor. Life supporting capacity reduced.
National Bottom Line	80	80	
D	< 80	< 80	River ecosystem health bad. Severely polluted.

Nitrate (NO_3)

Value	Ecosystem health	
Freshwater Body Type	Rivers	
Attribute	Nitrate	
Attribute units	mg/l (milligrams per litre)	
Attribute State	Numeric Attribute State	Narrative Attribute State
	Annual median	
A	≤ 0.08	River ecosystem health high, similar to natural reference condition.
B	> 0.08 and ≤ 0.39	River ecosystem health good. Some degradation of life supporting capacity but ecosystem still functioning well.
C	> 0.39 and ≤ 1.33	River ecosystem health moderate to poor. Life supporting capacity degraded but acceptable.
National Bottom Line	1.33	
D	> 1.33	River ecosystem health bad. Severely polluted.

Dissolved Reactive Phosphorous (DRP)

Value	Ecosystem health	
Freshwater Body Type	Rivers	
Attribute	Dissolved Reactive Phosphorus	
Attribute units	mg/l (milligrams per litre)	
Attribute State	Numeric Attribute State	Narrative Attribute State
	Annual median	
A	≤ 0.006	River ecosystem health high, similar to natural reference condition.
B	> 0.006 and ≤ 0.024	River ecosystem health good. Some degradation of life supporting capacity but ecosystem still functioning well.
C	> 0.024 and ≤ 0.068	River ecosystem health moderate to poor. Life supporting capacity degraded but acceptable.
National Bottom Line	0.057	
D	> 0.057	River ecosystem health bad. Severely polluted.

Habitat Quality Index HQI

Value	Ecosystem health	
Freshwater Body Type	Rivers	
Attribute	HQI	
Attribute units	dimensionless	
Attribute State	Numeric Attribute State	Narrative Attribute State
	Annual median	
A	≤ 0.80	River habitat condition similar to natural reference condition.
B	> 0.80 and ≤ 0.50	River habitat condition good. Some degradation of life supporting capacity but ecosystem still functioning well.
C	> 0.50 and ≤ 0.20	River habitat condition moderate to poor. Life supporting capacity degraded but acceptable.
National Bottom Line	0.20	
D	>0.20	River habitat condition bad. Habitat degraded to the point it Severely polluted.

How to use to use these and the NOF to manage freshwater ecological health

Managing ecological health in rivers, streams and lakes cannot be done by balancing the attributes presented here and in the current NOF. Ecological theory stresses that organisms and the ecosystems that they live within are affected by the attribute which is at the most lowest and thus most stressful – Liebig's law of the minimum (Harpole *et al.*, 2011). For example, a river can have plenty of good quality habitat, low nutrient levels, appropriate flows and no excess periphyton or sediment but if there are high levels of arsenic all the animals will die, thus yielding low health. Just like with our health, we can have low blood pressure, a healthy weight, a 37°C body temperature but if our brains don't get oxygen we will die. Similarly we can make sure there is plenty of petrol in our car but it will soon stop functioning if we fail to keep the oil topped up.

It is not possible to add and subtract and try and balance these attributes; for example keeping nitrate very low but letting DRP go high. Whatever attribute is most stressful for the ecosystem when corrected will just move the limiting factor to the next most stressing attribute. Thus to achieve an A all of the attributes will need to be in the A attribute state: an A for nitrate and a D for MCI makes no ecological sense and will not result in a healthy ecosystem. Similarly focusing on one nutrient or the other, e.g. DRP will not result in low periphyton biomass, as limiting nutrients fluctuate seasonally, spatially and in interaction with other stressors (Dodds & Welch, 2000; Death, Death & Ausseil, 2007; Wilcock *et al.*, 2007; Keck & Lepori, 2012; Jarvie *et al.*, 2013).

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Appendix 1

Keeping the green out of ‘clean green’ New Zealand rivers: a weight-of-evidence approach for setting nutrient criteria

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Right running head: Nutrient criteria for New Zealand rivers

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Abstract

Eutrophication of waterbodies is a major stress on freshwater ecosystems the world over and New Zealand is no exception, with increasing agricultural intensification increasing nutrient levels in rivers throughout the country. As a result the New Zealand Government has established a policy of freshwater management where waterbodies are managed within four states (creatively termed A, B, C and D, from high to low ecological health). We compile multiple lines of evidence from a large range of data sources using a weight-of-evidence approach to objectively determine nitrate and dissolved reactive phosphorus limits to manage rivers and streams into those four states. This established that nitrate concentrations of 0.08, 0.39 and 1.33 mg/l, and DRP concentrations of 0.006, 0.024 and 0.068 mg/l would result in rivers having ecological health in the A, B, C and D attribute states. Although ecological health of rivers is affected by a range of interacting stressors we believe the evidence supports the view that managing to these nutrient thresholds will provide for better ecological condition in the rivers and streams of New Zealand.

Introduction

As in most developed countries there has been considerable concern on the declining water quality, ecological health and biodiversity of many of New Zealand's freshwater bodies (Ballantine & Davies-Colley, 2010; Verburg *et al.*, 2010; Parlimentary Commisioner for the Environment, 2013; Joy & Death, 2014; Foote, Joy & Death, 2015; Joy, 2015). This global decline is a result of multiple interacting stressors (Matthaei, Piggott & Townsend, 2010; Wagenhoff *et al.*, 2011; Piggott *et al.*, 2012) including water abstraction for consumptive and agricultural needs (Dewson, James & Death, 2007b; Poff & Zimmerman, 2010; McDowell, van der Weerden & Campbell, 2011), invasive species (Olden *et al.*, 2010; Collier & Grainger, 2015) channelization, sedimentation, eutrophication (Carpenter *et al.*, 1998; Allan, 2004) and changing climate regimes (Palmer *et al.*, 2008; Death, Fuller & Macklin, 2015). 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 & Davies-Colley, 2010; Unwin & Larned, 2013; Foote, Joy & Death, 2015; Ministry for the Environment and Statistics New Zealand, 2015; Davies-Colley & Nagels, 2022). Most sites in lowland pastoral catchments and all of the sites in urban catchments do not have pathogen standards that are safe for swimming and 60% of sites have increasing nitrogen levels (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 and 84% of lakes in pastoral catchments are the same (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 & 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 and funding initiatives in an attempt to improve water quality (Ministry for the Environment, 2004; Cullen, Hughey & Kerr, 2006; Hughey, Kerr & Cullen, 2010; Ministry for the Environment, 2014; Joy, 2015). However, the regulation and/or limit setting with respect to waterbody nutrient levels has become one of the most contentious issues (Wilcock *et al.*, 2007; Rutherford, 2013; Chisholm *et al.*, 2014). 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 considerable less than trying to remove them once they are there (Foote, Joy & Death, 2015; Joy, 2015). The government has established total nitrogen and total phosphorus 'environmental bottom lines' to direct local government in managing lakes, but it only outlines nutrient criteria (i.e. nitrate) for rivers at toxicity not ecological health (Ministry for the Environment, 2010; Ministry for the Environment, 2014). Given the obvious, and extensively documented links between high nutrient levels in rivers and declines in ecological health this is a surprising oversite (Biggs, 1996; Biggs, 2000a; Death, Death & Ausseil, 2007; Clapcott *et al.*, 2012; Collier *et al.*, 2013; Death *et al.*, 2015)

Many countries have established nutrient criteria or thresholds to protect aquatic life in their waterways (Dodds & Welch, 2000; Camargo & Alonso, 2006; Smith & Tran, 2010; Jarvie *et al.*, 2013; Heiskary & Bouchard, 2015). There are three broad approaches; ecological, statistical and expert-opinion, that can be used alone or in combination (Birk *et al.*, 2012). In setting the current numerical thresholds for ecological health the New Zealand Ministry for the Environment appears to have relied predominantly on expert opinion (e.g.(Snelder *et al.*, 2013). While this approach can be useful when there is insufficient data to make more objective decisions this does not seem to the case in New Zealand where multiple parameters of water quality and ecological health have been monitored in New Zealand rivers for nearly 3 decades (e.g. (Smith & McBride, 1990; Smith & Maasdam, 1994; Scarsbrook, Boothroyd & Quinn, 2000; Larned *et al.*, 2004; Clapcott *et al.*, 2012; Unwin & Larned, 2013).

In this study we adopt the weight-of-evidence approach (Smith & Tran, 2010) to develop nitrogen and phosphorus nutrient criteria for New Zealand rivers and streams to protect ecosystem health. We adopt the Ministry for the Environment approach detailed in the 'National Objectives Framework' where 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 the national bottom lines over time (Ministry for the Environment, 2014). This approach differs from that in the USA where nutrient criteria are derived for impaired / not-impaired waterways (Dodds & Welch, 2000; USEPA, 2000), but is similar to that of the European Water Framework Directive that also characterise water bodies as belonging to one of five states of ecological status from bad to high (European Commission, 2000; Birk *et al.*, 2012; Poikane *et al.*, 2014).

Materials and methods

Data sets

A range of real and modelled data was examined as lines of evidence for nitrate and dissolved reactive phosphorus thresholds (Table 1).

Percentile analysis

Collection of data on water chemistry in New Zealand rivers is relatively good, but highly variable in space and time, with proportionally more sites in lowland areas than higher altitude conservation land (Larned *et al.*, 2004; McDowell, Larned & Houlbroke, 2009; Ballantine & Davies-Colley, 2010; Larned & Unwin, 2012; Unwin & Larned, 2013). Unwin and Larned (2013) have compiled data from 786 water quality sites collected between 2006-2011 by Regional Councils and NIWA (National Institute of Water and Atmospheric Research) around New Zealand. 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 correlated with actual measures (r=0.64 and r=0.68, for nitrate and DRP, respectively) made at 22 Manawatu rivers and streams (R Death, pers. comm.) and at 77 National River Water Quality Network (NRWQN) sites (r=0.86 and r=0.73, for nitrate and DRP, respectively). As this modelled data gave a more extensive, consistent and spatially unbiased measure of nitrate and DRP the data was used to assess percentiles.

We used the percentile analysis approach of (USEPA, 2000; Smith & Tran, 2010). For data from all sites we used the 25th, 50th and 75th percentiles for A, B, C and D thresholds for nitrate and DRP. For sites in the Conservation Estate (n = 242,521) we used the 95th, 99th and 99.9th percentiles.

Ecosystem health metric relationships

The datasets examined for ecosystem health - biological metric relationships did not indicate any breaks in the relationship suitable for changepoint analysis (King & Richardson, 2003; Smith & Tran, 2010). However, New Zealand has well established biological indicator criteria for the Macroinvertebrate Community Index (MCI) and its quantitative variant (QMCI) (Quantitative Macroinvertebrate Community Index) (Stark, 1985; Boothroyd & Stark, 2000; Stark & Maxted, 2007). These have been in place since 1985 and are now widely used in all environmental monitoring in New Zealand (Boothroyd & Stark, 2000; Ministry for the Environment and Statistics New Zealand, 2015). Furthermore, although there is some suggestion they may respond to a variety of stressors in New Zealand waterways they were specifically developed to assess eutrophication (Stark, 1985) and have been shown to be insensitive to heavy metals, acid mine drainage and deposited sediment (Hickey & Clements, 1998; Gray & Harding, 2012; Death & Death, 2014). The MCI and QMCI thresholds (120, 100 and 80, and 6, 5 and 4, for MCI and QMCI respectively) therefore provide ideal criteria against which to assess A, B, C and D thresholds for nutrients. For the metrics EPT(taxa) and EPT(animals) thresholds were determined by regressing the metric against QMCI and using the values of the equation at QMCI = 4, 5 and 6 (EPT(animals) = 23%, 39% and 55% (r^2=0.81) and EPT(taxa) = 30%, 38% and 46% (r^2=0.55).

The data sources used to examine invertebrate water quality metric (MCI, QMCI, EPT(taxa), and EPT(animals) nutrient relationships are listed in Table 1. Data ranged from modelled metric and nutrient values, measured metric and modelled nutrient values for a reach, to measured metric and nutrient values. (Clapcott, Goodwin & Snelder, 2103) modelled MCI calculated from invertebrate collections in 1033 unique stream segments collected by NIWA and Regional Councils between 2007 and 2011 using Random Forests to yield predictions of MCI for all river reaches in New Zealand (r=0.83 between observed and predicted) MCI. These were regressed against the modelled nutrient values from Unwin & 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.*, 2015) ($F_{1,961}$ =1761 $p<0.001$; r^2=0.65). These QMCI values for each river reach in New Zealand were regressed against the modelled nutrient values from Unwin & Larned (2013).

Metrics calculated for invertebrate data collected from 962 sites sampled throughout the southern North Island during studies conducted at Massey University between 1994 and 2007 (Death *et al.*, 2015). Most of these sampling occasions involved 5 replicate 0.1 m^2 Surber samples, although some collections comprised a single 1-minute kick-net sample (see (Death & Joy, 2004) for more details). Samples were filtered through a 500 µm mesh sieve and identified to the lowest possible taxonomic level (usually genera) using (Winterbourn, Gregson & Dolphin, 2006). Where samples were collected from a site in multiple years, only the most recent was used in the analysis. The MCI and QMCI is relatively independent of sampling effort and season (Duggan, Collier & Lambert, 2002), and we are therefore 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 & Larned (2013).

Median metrics calculated from invertebrates and nutrients collected at 24 Manawatu streams and rivers (Death, 2013) and 64 nationwide NIWA monitoring rivers (Larned & Unwin, 2012; Unwin & Larned, 2013) on multiple occasions (monthly for nutrients, yearly for invertebrates) between 1999-2011 and 1989-2014, for Death (2013) and NIWA, respectively were regressed against each other.

The Index of Biotic Integrity (IBI) (Joy & Death, 2004) a bioassessment metric used for fish assemblages in New Zealand was calculated for data collected irregularly and nationally (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 & Larned (2013) for the corresponding reach. IBI thresholds for A, B, C and D were set at 42, 32 and 24 following Joy (2009).

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 but in all cases y=ln(x) proved the best fit to the data. 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= 55%, 39% and 23% for EPT(animals), and y = 46%, 38% and 30% for EPT(taxa).

Previously published numerics and ecosystem health metric relationships

Several previous publications have investigated nitrate and DRP thresholds for water management. 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 NIWA (Davies-Colley, 2000). 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. This information has also been used in management recommendations on water quality in New Zealand (Biggs, 2000b; Biggs & Kilroy, 2000). The current National Policy Statement for Freshwater Management 2014 currently 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 Bigg's equations to derive nitrate and DRP numerics (Ministry for the Environment, 2014). (Matheson, Quinn & Unwin, 2016) have also used quantile regression on data from NRWQN and Reginal Council (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 were also included.

Weighting lines of evidence

Following (Smith & Tran, 2010) direct linkage relationships between ecosystem health measures and nutrients were weighted 2 in the assessment and less direct linkages were weighted 1 (e.g. percentage analysis and Fish IBI). Where relationships were not significant they were not included as a line of evidence i.e. they were weighted 0.

Results

Ecosystem health metric relationships

The relationships between the health metrics and nutrient concentrations were predominantly exponential with health declining more rapidly for increasing nutrient concentrations at low levels and plateauing as ecological health approached poor condition (e.g. Fig. 1). 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. However, 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 NPS states derived from each line of evidence. For the ecosystem health metric regressions the F test of significance, significance and r^2 are also provided. This yielded nitrate concentrations of 0.08, 0.39 and 1.33 mg/l, and DRP concentrations of 0.006, 0.024 and 0.068 mg/l for the A, B, C and D states (Table 2). The percentage of New Zealand river reaches with median nitrate or DRP levels from (Unwin & Larned, 2013) in each of these attribute states is given in Table 3.

Discussion

There is a large amount of raw and modelled data available in New Zealand from which to objectively assess nutrient limits for maintaining ecological health in New Zealand rivers and streams. Given the considerable public concern over water quality in New Zealand (Hughey, Kerr & Cullen, 2010; Foote, Joy & Death, 2015; Weeks *et al.*, 2016) and Government policy (e.g. National Policy Statement of Freshwater Management (Ministry for the Environment, 2014)) it is surprising that this information has not been compiled before now to develop evidence based nutrient limits to protect ecological health. Perhaps one of the impediments has been that a range of variables, besides nutrients, will also impact on 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; however, it is reassuring that almost all the data sets yielded numerics within the same small range.

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 determinate of river food web structure managing below these nutrient concentrations will certainly be a step in the right direction (Matthaei, Piggott & Townsend, 2010; Wagenhoff *et* *al.*, 2011; Clapcott *et al.*, 2012; Wagenhoff, Townsend & Matthaei, 2012). Similarly, establishing limits for only nitrate or dissolved reactive phosphorus will not serve to limit adverse environmental effects, as when and where the respective nutrients become limiting changes and is often hard to establish (Dodds & Welch, 2000; Death, Death & Ausseil, 2007; Keck & Lepori, 2012; Jarvie *et al.*, 2013).

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 & Richardson, 2003; King *et al.*, 2005; Smith & Tran, 2010), however, there was no evidence of thresholds in any of the ecological metric nutrient relationships examined in the compiled data. Rather than any 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 (e.g. MCI, QMCI) thresholds that have been in use in New Zealand freshwater management since the mid 1980's (Stark, 1985; Stark, 1993; Wright-Stow & Winterbourn, 2003). Although, somewhat subjective these thresholds have been in use for a long time in river management, are familiar to all river managers and fit the four category attribute states model 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 too low. Once ecological health reached that point the long flat tail of the relationship (e.g. Fig. 1) 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 it does not seem to matter how many more nutrients are added, health is relatively unaffected. 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).

Although there can be many situations were expert opinion rather than data are necessary to establish management objectives this does not seem to be 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 a 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 it is important that they are objectively determined from as wide a range of data as possible (e.g. this is the first example we know of where fish data have been included in such an assessment). Despite this we find it reassuring that the derived numerics are all around the same general set of numbers irrespective of where the data came from, whether it was compiled data, index health nutrient relationships or previously published research.

In conclusion we believe the nitrate concentrations of 0.08, 0.39 and 1.33 mg/l, and DRP concentrations of 0.006, 0.024 and 0.068 mg/l 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 every increasing eutrophication of waterbodies worldwide we hope they 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 in for analysis.

Data	No. sites	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 & Larned, 2013)
Modelled data for National Environmental Monitoring and Reporting	All river reaches in NZ	Metric relationship	2007-2011	MCI, QMCI[A]	2 (Clapcott, Goodwin & Snelder, 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.*, 2015)
Russell Death Freshwater Animal Targets (FAT) model	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 & 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, Quinn & Unwin, 2016)
ANZEC guidelines	Table 3.3.10			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 963 North Island sites (Death *et al.*, 2015) ($F_{1,961}$ =1761 p<0.001; r^2=0.65).

Table 2 Numerical nutrient thresholds (mg/l) for the A, B, C and D NPS states derived multiple lines of evidence. Weighting of each piece of evidence is provided along with statistics (F statistic, degrees of freedom, probability and r^2) on any regression analysis that was used. See Table 1 for details on data sources (e.g. 1-9) used.

NO_3																	
Data x	1	1	1	1	1	1	1	1	4	4	5	5	1	9	7		
y			2	2	3	3	3	3	4	4	5	5	6	9	7		
	Mod NO3	Mod NO3 PCI only	MfE MCI	MfE QMCI	BBN MCI	BBN QMCI	BBN EPT animals	BBN EPT taxa	FAT MCI	FAT QMCI	NIWA MCI	NIWA QMCI	IBI	ANZECC 2000	Biggs 2000	Matheson et al. 2016	
Equation			ln y = ln (x+1)	ln y = ln (x+1)	y = ln x	y = ln x	y = ln x	y = ln x	y=lnx	y=lnx	y=x	y=lnx	y=lnx		log10 (max chla) = x		Final
Weight of evidence	1	1	2	2	2	2	2	2	2	2	2	0	1	2	2	2	
A/B	0.03	0.08	0.02	0.00	0.11	0.10	0.11	0.20	0.06	0.09	0.00	0.00	0.00	0.17	0.03	0.10	0.08
B/C	0.06	0.12	0.45	0.29	0.58	0.34	0.30	0.47	0.53	0.33	0.60	0.13	0.21		0.10	0.63	0.39
C/D	0.28	0.20	1.22	0.77	3.01	1.09	0.87	1.09	4.36	1.20	1.60	9.10	1.54	0.44	0.20	1.10	1.33

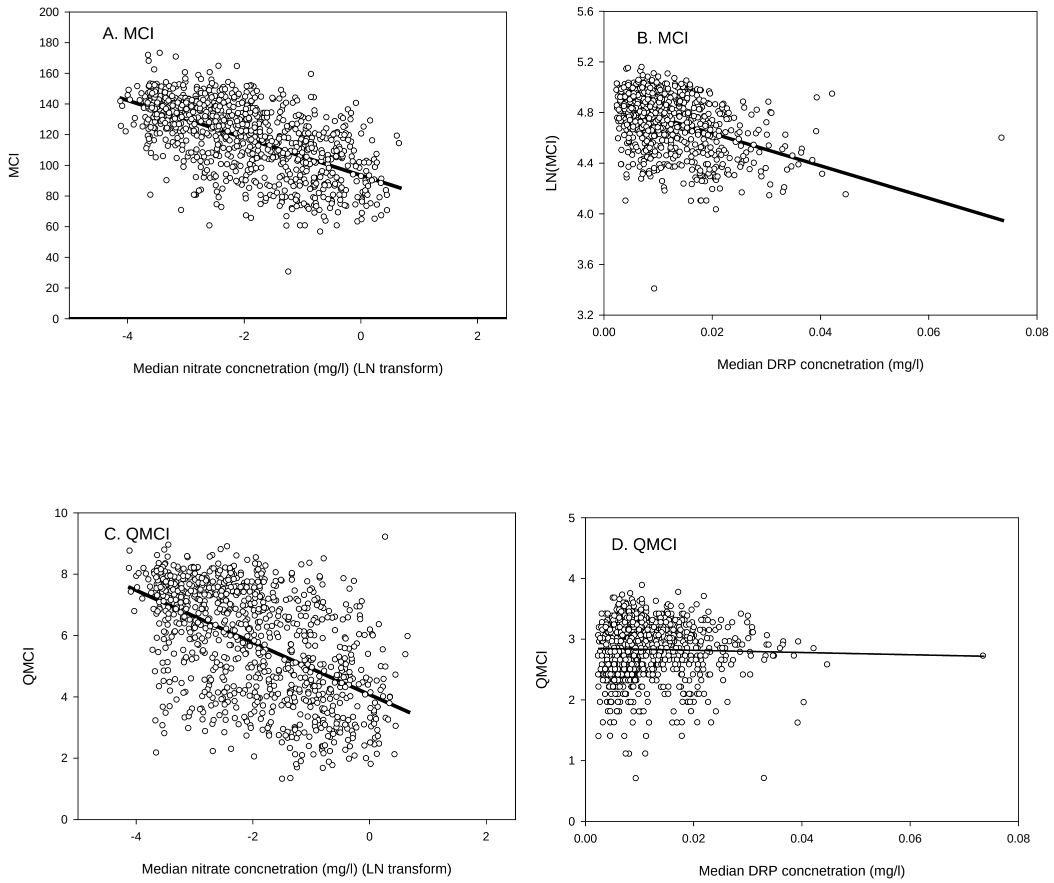
r^2			0.53	0.54	0.35	0.27	0.28	0.29	0.37	0.27	0.08	0.04	0.09		0.3		
F			632224	653084	513	363	377.60	390.60	51.72	32.66	6.78	3.85	3775.00				
df			1,566 548	1,566 548	1,961	1,961	1,961	1,961	1,86	1,86	1,62	1,62	1,392,543				
p			<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	0.01	0.05	<0.0001				
DRP																	
Equation			ln y =x	ln y =x	ln y = x	ln y = x	ln y = x	y=x	y=lnx	y = lnx	y=x	Y=lnx	lny=ln x		$\log_{10}$(max chla) = x		Final
Weight of evidence	1	1	2	2	2	2	2	2	2	2	0	0	1	2	2	2	
A/B	0.004	0.011	0.004	0.003	0.008	0.006	0.005	0.015	0.005	0.008	0.000	0.000	0.002	0.009	0.002		0.006
B/C	0.008	0.014	0.016	0.012	0.022	0.015	0.009	0.021	0.038	0.025	0.023	0.008	0.007		0.007	0.120	0.024
C/D	0.012	0.021	0.032	0.024	0.040	0.027	0.016	0.028	0.275	0.079	0.066	0.024	0.014	0.100	0.014	0.200	0.068
r^2			0.38	0.39	0.18	0.15	0.18	0.18	0.54	0.420	0.02	0.04	0.04		0.3		
F			349187	357979	210.30	165.000	217.80	211.10	99.83	63.89	2.160	3.610	15770.00				

df			1,566 548	1,566 548	1,961	1,961	1,961	1,961	1.86	1,86	1,62	1,62	1,392 ,543				
P			<0.00 01	<0.00 01	<0.00 01	<0.00 01	<0.00 01	<0.00 01	<0.00 01	<0.00 1	0.15	0.06	<0.00 01				

Table 3. Percentage of river reaches in each nutrient attribute state

	NO_3-N	Percent	DRP	Percent
A	< 0.08	54.4	< 0.006	37.4
B	0.08 ≤ x < 0.39	26.2	0.006 ≤ x < 0.024	58.4
C	0.39 ≤ x < 1.33	17.0	0.024 ≤ x < 0.068	4.2
D	> 1.33	2.4	> 0.068	0.02

Figure 1 MCI and QMCI measured at 963 North Island rivers and streams as a function of median modelled nitrate and DRP from Unwin & Larned (2013).



Appendix 2

Quantifying habitat quality – the missing dimension for water resource management

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SUMMARY

1. Concern about the ecological health and biodiversity status of the fauna in the world's rivers and streams has led to many national governments and international bodies instigating policies, laws, education programs and action plans to combat declining water quality and quantity.
2. However, if the purpose of these programs is to halt, or even enhance the ecological condition and biodiversity values of those waterbodies then we must also ensure adequate quality habitat is available for those organisms. An abundance of clean water alone does not improve ecological condition if there is nowhere for the organisms to live or no way for them to reach those habitats.
3. The use of habitat quality in resource management activities has been hampered by the lack of a simple and effective method to quantify the quality of that habitat.
4. Here we use an assessment of river geomorphology to develop an index of natural character (Habitat Quality Index or HQI) which provides a geomorphic framework to quantify habitat condition. This approach assesses the degree to which reach geomorphology has changed from a 'natural benchmark' derived from the historical (pre-modification) condition at that reach.

5. We provide examples deriving HQI in rivers subject to on-going flood control management in the lower North Island of New Zealand.

Keywords: ecological health, habitat, hydromorphology, natural character, HQI, New Zealand, river geomorphology

Introduction

The ecological health, diversity of biological communities, and viability of species in rivers and streams throughout the globe are in dramatic decline (Dudgeon *et al.*, 2006; Vorosmarty *et al.*, 2010; Wohl, 2010; Feld *et al.*, 2011). This is the result of a wide range of potentially interacting pressures including increasing nutrient levels, greater deposited and suspended sediment, water abstraction, changes in flow regime, overexploitation of species and changing climate regimes (e.g., (Carpenter *et al.*, 1998; Allan, 2004; Arthington *et al.*, 2006; Dewson, James & Death, 2007a; Palmer *et al.*, 2008). These declines have motivated environmental agencies and governments in many countries to develop a wide range of mechanisms to quantify and manage the amount and quality of water in rivers to halt or reverse some of these declines (e.g., (Resh & McElravy, 1993; Wright, Sutcliffe & Furse, 2000; Friberg *et al.*, 2011). This in turn has been associated with more ecologically linked planning, policy and legal frameworks all with the aim of halting the decline in riverine ecological condition (e.g., (Fore *et al.*, 2008; Acreman & Ferguson, 2010; Hering *et al.*, 2010).

Biological communities will certainly be affected by declines in the quantity and quality of water in a river. However, if there is limited or low quality habitat, irrespective of how much and how clean the water is, then there is unlikely to be healthy diverse biological communities (Elosegi, Díez & Mutz, 2010; Elosegi & Sabater, 2013). The importance of both instream and flood plain habitat in affecting riverine biological communities has always been widely understood, and integral to interpreting most stream ecology research (Hynes, 1970; Hynes, 1975). The influence of geomorphological structure in determining river ecosystem structure and function often forms the underlying framework for many hypotheses in stream ecology such as the River Continuum Concept (Vannote *et al.*, 1980), Network Dynamics Hypothesis (Benda *et al.*, 2004) and the Riverine Ecosystem Synthesis (Thorp, Thoms & Delong, 2006; Thorp, 2008). At the individual species level the habitat and geomorphological requirements of salmonids have probably received the most attention (e.g., (Kondolf, 2000; Fukushima, 2001; Sear & DeVries, 2008; Wheaton *et al.*, 2010). The still widely applied technique of Instream Flow Incremental Method (IFIM) setting of minimum flows uses habitat suitability curves to link potential effects on habitat loss of differing levels of flow alteration (Richter *et al.*, 1997; Tharme, 2003; Lamouroux & Jowett, 2005), although it has been widely criticised for its simplistic assessment of habitat (Maddock, 1999; Gore, Layzer & Mead, 2001; Hudson, Byron & Chadderton, 2003). The habitat requirements for other freshwater fish and invertebrates have also been considered, although not to the same degree as for salmonids (Death, 2000; Helfman, 2007; Barnes, Vaughan & Ormerod, 2013; Gorski *et al.*, 2013a; Koehn & Kennard, 2013).

The link between geomorphological changes in river morphology and how they impact on the associated river biota has had considerable discussion in the literature (Newson & Newson, 2000; Stallins, 2006; Vaughan *et al.*, 2009; Brierley *et al.*, 2010; Darby, 2010; Poole, 2010; Rice, Lancaster & Kemp, 2010). However, much of the focus was on developing a geomorphological typology for understanding the relative roles of channel-shaping processes, both spatially and over time. In turn, there was an assumption that if river morphology was altered then so too is the habitat for the animals and plants living within that river. But there have been few, if any, attempts to quantify the actual linkages between these river typologies and the organisms that are believed to be affected (Vaughan & Ormerod, 2010; Gorski *et al.*, 2013a; Gostner *et al.*, 2013). Elosegi et al. (2010) in their review of the effects of human modification of riverine geomorphological character on biodiversity and ecosystem function concluded this lack of research was a result of the highly variable nature of geomorphology depending on climate, geology, catchment position, land use and flow regulation.

There are a number of physical habitat survey methodologies that have been used in association with biological assessments of river condition (e.g., (Raven *et al.*, 1997; Raven *et al.*, 1998; Harding *et al.*, 2009). These approaches developed by ecologists have been criticised, by geomorphologists, because of a lack of the correct temporal and spatial perspective in the techniques to link stressors and response (Fryirs, Arthington & Grove, 2008; Rinaldi *et al.*, 2013). In essence there appears to be a disjunct between the assessment approach of river habitat by geomorphologists and ecologists; the latter deal with responses over very short time scales (days to years) whereas the former deal with responses over much longer time scales (decades to millennia) (Death, Fuller & Macklin, in prep). Furthermore, the approaches used by ecologists to assess habitat are more useful for simple description, than assessment of whether they provide suitable quantity and quality of habitat and/or if habitat differs significantly from some minimum ‘healthy ecosystem’ requirement.

The need to maintain and improve river hydromorphology is gradually becoming as important to river management as preserving water quality and quantity, to protect healthy ecosystems and also to mitigate flood effects (European Commission, 2000; Sear & Arnell, 2006; Feld *et al.*, 2011). However, although there are many techniques to describe habitats, there are limited mechanisms or protocols to evaluate those measures against what is required for a ‘healthy’ hydromorphology. (Davies, Norris & Thoms, 2000; Parsons, Thoms & Norris, 2004) adopted an approach from biomonitoring to develop a predictive model of habitat condition using the RIVPACs modelling approach (Wright, Sutcliffe & Furse, 2000) that we believe offers considerable potential for both describing and quantitatively assessing habitat quality. In this paper we present a simplification of their approach; using historical records for a given site to determine the reference condition for assessment, rather than using multiple sites for that benchmark as in Davies et al. (2000) and Parsons et al. (2004). We demonstrate this approach on three New Zealand rivers that have had their hydromorphology altered for flood control of nearby urban areas. We believe the technique is quick and simple to apply but provides a robust and quantified measure of habitat change. If hydromorphology must be altered by flood engineering to prevent damage to people and their infrastructure then quantifying the loss with our index will allow mitigation of that same quanta at a more suitable location, hopefully with the end result that there is no net loss of habitat.

Methods

Study sites

We assessed the geomorphological change in the lower reaches of three major rivers in the lower North Island of New Zealand (Fig. 1). All three rivers arise in native vegetation in the Tararua Ranges and pass alongside or through major urban areas in their lower reaches before emptying into the ocean. As a result of their proximity to these urban areas all three rivers have varying amounts of flood engineering to prevent inundation of housing and other infrastructure during high flows.

The Otaki River rises in the western Tararua Range. In the lower catchment, native forest has been cleared for agriculture and has resulted in substantial bank erosion as the steep-gradient river re-worked its floodplain. This has led to the development of a braided river planform prior to management. The Otaki is the only significant west coast river in the lower North Island to deliver gravel to the coast, and accordingly the channel is characterized by a high degree of change and activity and exemplifies a typical wandering planform. The town of Otaki is sited on the river floodplain within an area subject to substantial inundation during a 100 year flood (Otaki River FMP, 1998).

The smaller Waikanae River also rises in the western Tararua Range. Like the Otaki, the upper Waikanae comprises steep, forest-covered slopes in the upper catchment, but a larger proportion of the catchment was cleared, with 40% now in pasture (Waikanae River FMP, 2013). The river is less dynamic than the Otaki and although gravelly for the most part, the lower part of the river, where it runs adjacent to the town of Waikanae (sited on the floodplain) is a substantial depositional zone, with gravel deposited short of the coastline due to insufficient stream power related to channel slope reduction. Channel planform is typically wandering.

The Hutt River drains the southern Tararua Range and northern Rimutaka Range. While large parts of the upper catchment comprise steep, forested terrain, the main floodplains have been cleared for agriculture and urban development. The floodplain in the lower catchment is highly urbanized and the river is restricted to a narrow corridor towards the true right of the valley for much of its length in this part of the catchment. The floodplain was inundated entirely by c.2000 m^3s^{-1} floods that occurred in 1858, 1878, and 1898, prior to flood protection construction (Hutt River FMP, 2001). Under the present rainfall-runoff regime, the river in the lower Hutt valley would naturally occupy a much larger swathe of floodplain and be characterized by a more extensive and dynamic wandering and braided channel than current management allows.

The reference condition approach

The reference condition approach in freshwater ecology was first adopted for assessing biological water quality by Wright and colleagues, and is now a widely used technique for bioassessment (Wright *et al.*, 1984; Reynoldson *et al.*, 1997; Wright, Sutcliffe & Furse, 2000). Assessment is achieved by comparing taxa collected at a test site with those predicted from a model of unimpacted condition generated from sampling multiple reference sites. Davies et al. (2000) and Parsons et al. (2004) adopted a similar approach for use with the geomorphological assessment of river habitat. However, the reference condition approach, at least for bioassessment, suffers from the issue that reference or unimpacted condition for some river types (e.g., large lowland rivers) may longer exist (Scarsbrook *et al.*, 2003; Bailey, Norris & Reynoldson, 2004; Stoddard *et al.*, 2006; Hawkins, Olson & Hill, 2010). In geomorphology the large variability in river typology will also make identifying appropriate sites for assessment of reference condition difficult (Elosegi, Díez & Mutz, 2010).

However, historical records of geomorphological characteristics at a given river segment such as aerial photographs mean that, unlike the biological reference condition, it is possible to assess what the condition at an actual site was historically. As the reference condition is assessed from an historical (pre-impact condition) assessment for the same site it guarantees that the reference state is the most appropriate. This differs fundamentally from the approach of (Davies, Norris & Thoms, 2000; Parsons, Thoms & Norris, 2004) where a range of sites were sampled in the present day to develop the predicted reference state.

Our approach of using the historical condition of a site as the reference point also simplifies the calculation of the Observed/Expected (O/E) index. When multiple sites are used, to establish the reference condition the sampled rivers are grouped into types, usually with a classification procedure. The O/E index is then calculated using the probabilities of the test site being a member of that (or other) river types and the associated probabilities of each taxon's (or geomorphological characteristic) presence in that river type(s) summed. Further detail on calculation of the O/E index using multiple reference sites can be found in (Furse *et al.*, 1984; Wright, Sutcliffe & Furse, 2000). In contrast, in our approach we simply compare the test condition with the historical reference condition at the same point in space for each characteristic as a ratio; the closer the value is to unity the less it has altered. Values smaller than 1 represent a decline in the historical condition, and those greater than 1 an increase. The further from 1 the greater the proportionate change. We have called the ratio the Habitat Quality Index, as this is the term used in planning, management and legal documents for describing the geomorphological characteristics of rivers in New Zealand (Froude, Rennie & Bornman, 2010).

Measured parameters and data sources

As the reference condition we choose for assessment was the historical condition of our study sites this limited the number and type of geomorphological characteristics we could measure. We included a range of measurable and biologically relevant characteristics that we had previously assessed alongside potential alternatives that could not be assessed historically (e.g., substrate composition, percent deposited sediment) (Death, 2012). This work identified useful and measurable parameters to characterise geomorphological condition to be sinuosity, extent of braiding, percent of pools, active, bankfull and floodplain channel widths.

We used geo-referenced aerial photographs from the Greater Wellington Regional Council (GWRC) dating from 1939 (Otaki), 1951 (Hutt), 1952 (Waikanae) and 2010 (all) to measure sinuosity and the percent of pools. Airborne laser mapping (Light Detection and Ranging, LiDAR) from 2003 was used to precisely define floodplain and channel parameters and identify previous channel courses. Each river is managed by the environmental agencies as a series of reaches, so parameters were assessed for each of these reaches in the three rivers. In the Otaki River reaches were on average 1.39 km (total length = 12.5), in the Waikanae River 0.83 km (total length = 5) and in the Hutt River reaches were 2.53 km (total length = 27.8).

Channel and floodplain widths

Active channel width, bankfull channel width, natural floodplain and permitted floodplain width were measured within ArcMap® 10.0 GIS, using LiDAR imagery. Active channel width is defined as the width of the wetted channel and the active gravel bars (devoid of vegetation) combined. Bankfull width is defined as the distance from banktop to banktop, encompassing wetted channel, active gravel bars and older semi-vegetated bars. Natural floodplain width is the area of valley floor that would be naturally inundated during a flood, which is usually between the youngest two river terraces at the valley margins in this area of New Zealand, while permitted floodplain width is the floodable valley floor usually confined between two stopbanks (levees / flood walls). Widths were measured at approximately 50 m intervals along the entirety of each reach in each river using the ruler tool in ArcMap®. Values were then averaged for each management reach to simplify subsequent analysis. Since river management is most likely to result in constraining the river channel, the extent of this constriction was assessed by deriving a ratio of parameter width in 2010 to width from historic imagery (active and bankfull channel widths, plus permitted floodplain width). In addition, ratios showing the changes in active channel to natural floodplain width, which expresses the proportion of the floodplain occupied by the active channel; bankfull width to natural floodplain width, which expresses the size of the flood channel as a proportion of the natural floodplain; and natural to permitted floodplain width, which highlights any reduction in floodable area; were derived.

Channel planform

River management also tends to straighten channels, the extent to which this occurred can be assessed using sinuosity. The distance between two end points of a reach was measured following the midpoint of the wetted channel, which was then divided by the linear length between the two end points, providing a measure of sinuosity for each reach.

Many of New Zealand's rivers are locally braided. Management approaches tend to seek to simplify channel pattern. The extent of braiding was calculated using Brice's index (Brice, 1974), which states that the extent of braiding is 'twice the total length of the bars within the reach divided by the mid-channel reach'. Mid channel bars were identified and the length of each was measured in ArcMap® using the ruler tool. The total was calculated, multiplied by two and then divided by the mid-channel length used in the sinuosity calculation. However, this index was only feasible to generate in the Hutt River, which has clearly defined mid-channel bars. The Waikanae River is, and has been, primarily single thread. The Otaki River also proved difficult to assess in terms of braiding, since properly developed medial bars have not been a consistent characteristic of overall river morphology. However, anabranching channels were evident in 1939. Total thalweg length in each management reach was therefore measured in preference to a braiding index in the Otaki. Channel and floodplain widths were averaged for each management reach. Channel sinuosity and braiding index (or thalweg length) were calculated for each management reach as a whole. The number of pools was also counted in each management reach, as a measure of geomorphic and habitat diversity.

Data analysis

HQI values were compared between rivers with a Kruskal-Wallis one-way ANOVA in Statistix© 9.0.

Results

The ratio of each geomorphological characteristic in 2010 is expressed over the appropriate historical condition. For example, the permitted floodplain width component of reach XS370-XS220 in the Otaki River is:

$$HQI_{pf} = \frac{Width\ in\ 2010}{Width\ from\ LIDAR} = \frac{223}{2001} = 0.11$$

The ratios for each component of each reach are presented in Table 1, 2 and 3 for the Otaki, Waikanae, and Hutt Rivers, respectively. The summary HQI value for each management reach is derived as a median of the component ratios for that reach. The use of the median in preference to the mean avoids skewing the assessment by large values. There is a tendency in the Hutt and Otaki Rivers for modification to increase downstream (Fig. 2). Modification of the Waikanae River is restricted to an area around a road bridge at the most upstream reaches of our study. The Otaki River was significantly more modified than the Waikanae River, however neither was different from the Hutt River (Kruskal-Wallis One-Way ANOVA = 5.89, P=0.05) (Fig. 3).

Discussion

Geomorphological condition

The extent to which geomorphological condition had been modified varied between and within rivers. For example, although the Otaki River as a whole was the most modified, the upper reaches of the Otaki, situated at the outlet of a natural gorge, have been relatively untouched. Overall, the least modified, was the Waikanae River. However, modification of the lower reaches in the Hutt River has occurred for over 100 years, further back than the earliest aerial photos available for the reaches, so we may have underestimated modification for this river. The most modified reaches in all the rivers not surprisingly were generally towards the lower extents of the river (i.e. towards the river mouth) where population density was highest.

Consistently, the least changed parameter in the rivers was sinuosity (Tables 1-3). This may reflect the fact that these wandering rivers are not classic meandering systems. It may therefore not be the most appropriate measure of natural character for these river types. However, changes to the thalweg length in the Otaki and braiding index in the Hutt have been altered, so there have clearly been changes to the natural planform. In the Hutt River, the dramatic decline in braiding is a product of the simplification of channel form engineered as part of flood protection in these reaches, through both direct channel modification and gravel extraction. In multi-threaded systems, these parameters appear to be the most sensitive measures of habitat quality and geomorphic diversity.

Active channel and bankfull channel widths, indicate variable degrees of channel modification, from very little in the Waikanae River to substantial in the Hutt and Otaki Rivers. The extent of modification of these parameters in the rivers ranges from a median of 0.45 (Otaki bankfull) to 0.85 (Hutt active), indicating that both river channels have been narrowed, the Otaki to a greater degree. This has a significant impact on habitat diversity, because the channel form has necessarily become simplified and high flows have become more confined, producing a greater hydraulic uniformity in the channel, relative to a more natural diversity. This is consistent with the observed reduction in pools in the rivers, most marked in the Otaki River (median = 0.16). In more hydraulically uniform, narrowed channels, sediment transport tends to be more efficient and pool-riffle sequences become less distinct as sediment is forced downstream.

All three rivers demonstrate a marked reduction in permitted floodplain width. Settlements in New Zealand were often established alongside suitable water and transport bases such as rivers, however as urban areas have grown and expanded river floodwaters have had to be increasingly confined to avoid damage to people, housing, industry and infrastructure. This has resulted in a loss of floodplain habitat for many riparian organisms and removal of the connection between the river and floodplain that is critical for many ecological functions (Tockner *et al.*, 2010; Gorski *et al.*, 2013b; Reckendorfer *et al.*, 2013; Death, Fuller & Macklin, in prep). Flood flows that once dissipated across broad floodplains are now confined within the channel. This, together with channel narrowing exacerbates increased flow energy and increases sediment transport, contributing to further habitat reduction.

Use and limitations of HQI to evaluate habitat quality

The HQI is not a geomorphological characterisation or absolute measure of habitat diversity and quality, but is a quantified assessment of the relative change in selected geomorphic characteristics and habitat quality between successive phases of human intervention in river channels. Furthermore, appropriate parameters must be measured: there is little to be gained in quantifying a braiding index if the river is only locally divided or lacks proper medial bar development, or assessing sinuosity if the river lacks a meandering planform. In the illustrated examples we compared the geomorphic condition in 2010 with an historical reference condition. This is akin to the reference condition approach for hydromorphology advocated by organisations like the EU, that is already in widespread use for assessment of river biological communities (European Commission, 2000). It also provides a mechanism for developing flood management practises more reflective of natural processes advocated by the Foresight report and others to improve both flood management and biodiversity conservation (Evans *et al.*, 2004; Sear & Arnell, 2006; Evans *et al.*, 2008). HQI values for proposed alternative flood management options can be measured and the one with the least reduction in HQI used.

The same approach could also be adopted for assessing more short term changes in geomorphological structure from river engineering activities. The HQI could be calculated before and after an engineering activity and if there has been a significant change in the index then engineers could provide mitigation, either on site or elsewhere, to that same quanta of change. Furthermore, the component HQI scores would indicate where mitigation was most appropriate; for example if the HQI component for percent pools had declined the most new pools could be created. In this way flood engineering could still be conducted to protect people's lives and infrastructure without any net loss in habitat diversity of a river. It could also be used for state of the environment monitoring; the index could be re-measured as part of ongoing river monitoring. Deterioration of the score could then indicate more appropriate river management strategies, and/or rehabilitation schemes may be necessary to improve degraded reaches.

Conclusion

The use of a geomorphic reference condition approach to quantifying change in river habitat has been used before (Davies, Norris & Thoms, 2000; Parsons, Thoms & Norris, 2004), however we present a much simpler and potentially more appropriate protocol. Rather than sampling a range of sites to establish the reference state we simply use historical records of the site of interest to establish the actual reference condition (at least back to a certain point in time). We believe this offers an easy way for geomorpholgists, ecologists and river engineers to interact more effectively to determine management options that will facilitate flood protection but also avoid any future loss of habitat and biodiversity as a consequence. It allows the quanta of any potential or actual habitat loss to be measured, and will even isolate the particular geomorphological characteristic in need of remediation if habitat diversity is deemed to have declined too much. We hope that this will help move river management further from flood engineering science to a more holistic environmental science (Sear & Arnell, 2006).

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Table 1. HQI parameter ratios and overall median HQI per management reach defined by cross-section distance upstream from the river mouth (km), Otaki River.

	XS220–XS120	XS370–XS220	XS490-XS370	XS600-XS490	XS720–XS600	XS860–XS720	XS1020–XS860	XS1180–XS1020	**Median**
Sinuosity	0.92	0.91	0.99	0.83	0.86	1.21	0.97	0.97	**0.95**
Active channel	0.47	1.13	1.01	0.75	0.68	0.94	0.62	0.84	**0.79**
Bankfull channel	0.24	0.26	0.34	0.62	0.25	0.57	0.71	0.92	**0.45**
Permitted Floodplain	0.10	0.11	0.23	0.34	0.84	0.36	0.82	0.97	**0.35**
Thalweg length	0.41	0.59	0.79	0.47	0.94	1.53	1.06	1.01	**0.87**
Pools	0	0.32	0.64	0	0	0.75	1.26	0	**0.16**
OVERALL HQI	**0.41**	**0.32**	**0.64**	**0.62**	**0.68**	**0.75**	**0.82**	**0.92**	**0.71**

Table 2. HQI parameter ratios and overall median HQI per management reach defined by cross-section distance upstream from the river mouth (km), Waikanae River.

	XS80 –XS20	XS175 – XS80	XS240 – XS175	XS310 – XS240	XS350 – XS310	XS430 – XS350	**Median**
Sinuosity	0.96	0.97	0.79	0.98	0.98	0.74	**0.97**
Active channel	1.10	0.94	1.27	0.79	0.76	1.03	**0.99**
Bankfull channel	1.00	1.00	0.99	1.00	0.99	1.00	**1.00**
Permitted Floodplain	1.02	0.99	0.73	0.41	0.52	0.82	**0.78**
Pools	1.00	0.80	1.50	0.00	0.26	0.83	**0.82**
OVERALL HQI	**1.00**	**0.97**	**0.99**	**0.79**	**0.76**	**0.83**	**0.97**

Table 3. HQI parameter ratios and overall median HQI per management reach defined by cross-section distance upstream from the river mouth (km), Hutt River.

	XS210–XS100	XS360–XS210	XS510-XS360	XS850-XS510	XS1090-XS850	XS1350–XS1090	XS1630-XS1350	XS1920-XS1630	XS2270-XS1920	XS2550-XS2270	XS2780-XS2550	XS2830-XS2780	**Median**
Sinuosity	1.0	1.0	0.87	0.98	1.00	0.99	0.98	0.98	0.89	0.99	1.00	0.95	**0.99**
Active channel	0.98	1.13	0.78	0.64	0.65	0.79	0.89	0.72	0.95	1.21	0.95	0.70	**0.84**
Bankfull channel	0.73	1.03	0.50	0.47	0.56	0.72	0.59	0.81	0.57	1.06	0.91	0.90	**0.73**
Permitted Floodplain	0.22	1.00	0.98	0.28	0.59	0.54	0.28	0.09	0.17	0.44	0.98	0.98	**0.49**
Braiding Index	0.00	0.00	0.00	0.26	0.00	0.11	0.44	0.14	0.49	0.00	0.16	0.00	**0.06**
Pools	0.00	0.00	1.00	0.31	0.76	0.40	0.00	0.78	1.00	2.00	1.00	2.10	**0.77**
OVERALL HQI	**0.73**	**1.0**	**0.78**	**0.47**	**0.59**	**0.72**	**0.59**	**0.72**	**0.57**	**1.06**	**0.95**	**0.90**	**0.73**

Figure Legends

Figure 1. A. Catchment and study reach locations: 1. Otaki River (11.8 km reach length), 2. Waikanae River (4.3 km reach length), 3. Hutt River (28.3 km reach length). B. Otaki River at XS600-490 (cf Table 1 and * in Figure 1A), typical of laterally constrained wandering rivers in this region, photo is looking upstream, bridges are State Highway 1 and railway respectively. Photo: ICF 31 March 2013.

Figure 2. Summary of reach HQI scores in the Otaki, Waikanae, and Hutt Rivers, per management reach defined by cross-section distance upstream from the river mouth (km). Grey indicates agreement with reference condition and black deviation.

Figure 3. Summary of median HQI scores for the Otaki, Waikanae, and Hutt Rivers. Grey indicates agreement with reference condition and black deviation.

Fig. 1.

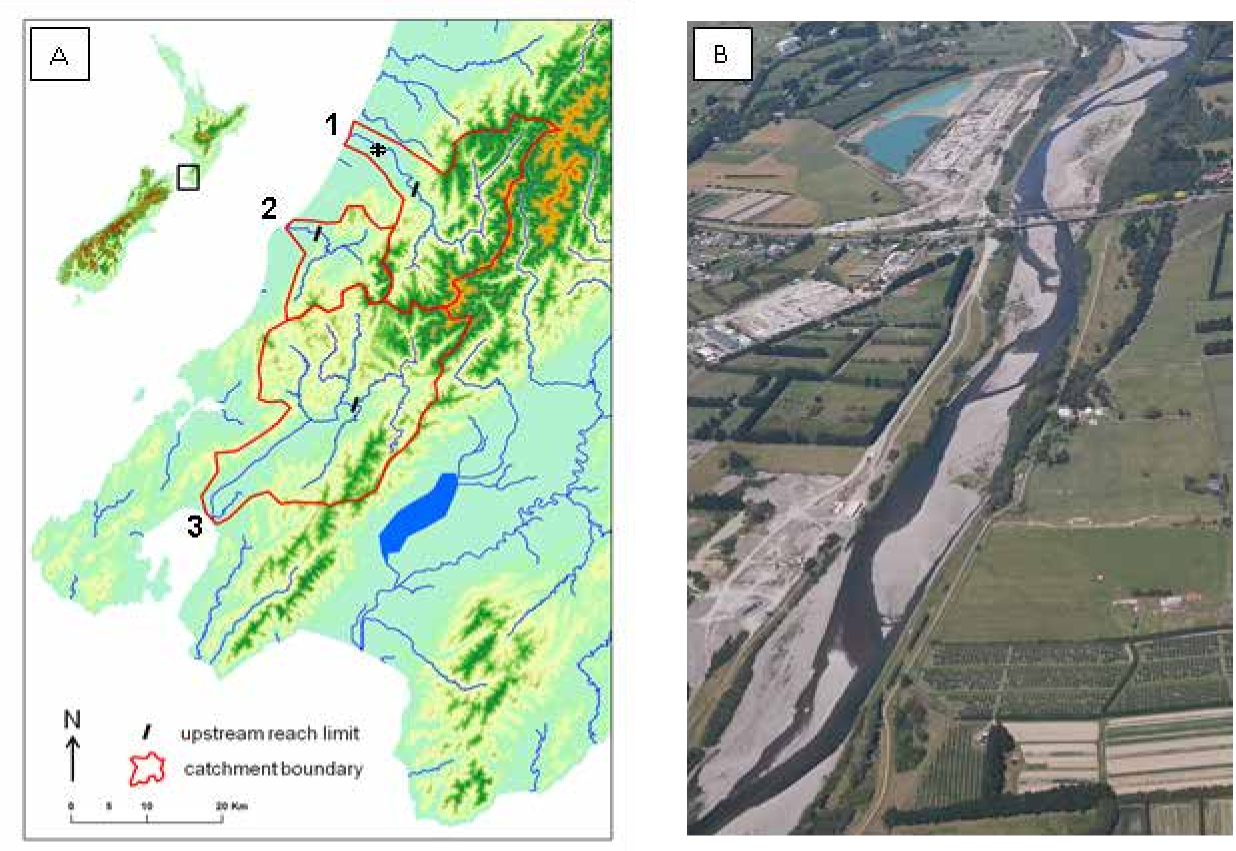

Fig. 2

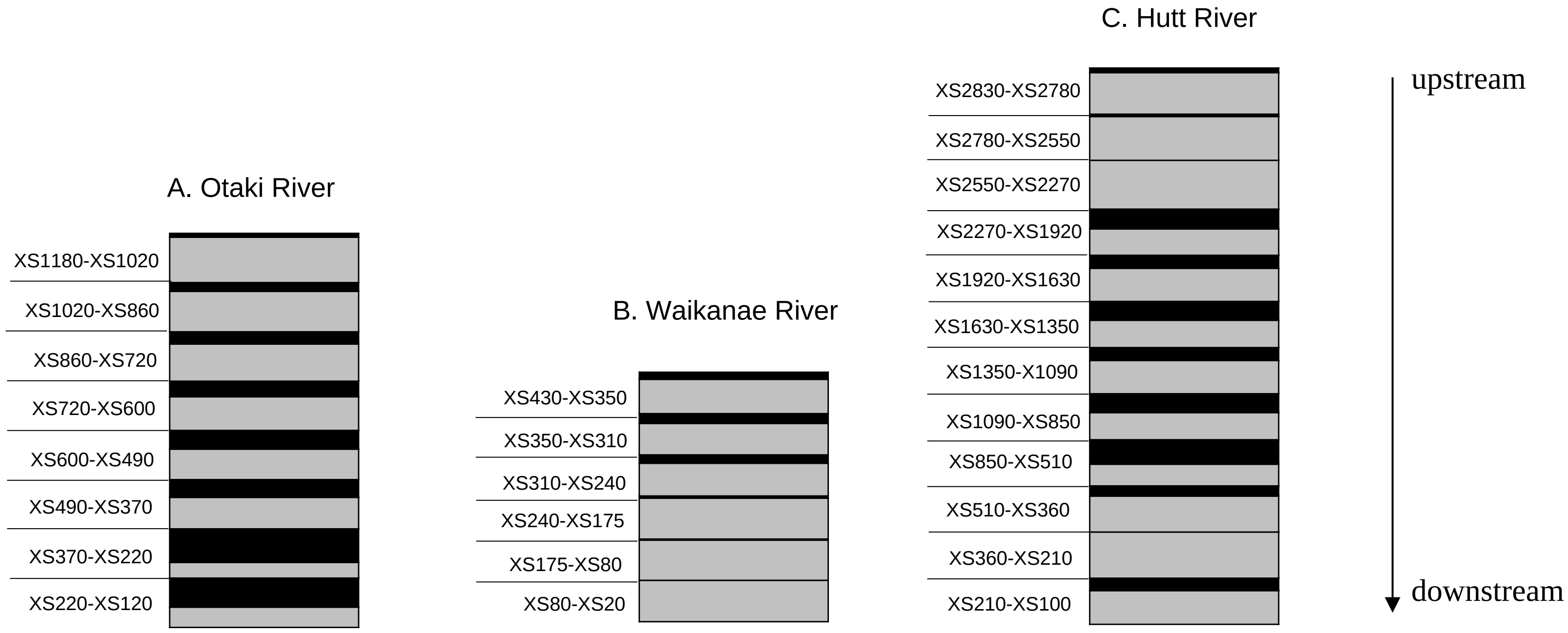
A. Otaki River
XS1180-XS1020
XS1020-XS860
XS860-XS720
XS720-XS600
XS600-XS490
XS490-XS370
XS370-XS220
XS220-XS120
B. Waikanae River
XS430-XS350
XS350-XS310
XS310-XS240
XS240-XS175
XS175-XS80
XS80-XS20
C. Hutt River
XS2830-XS2780
XS2780-XS2550
XS2550-XS2270
XS2270-XS1920
XS1920-XS1630
XS1630-XS1350
XS1350-X1090
XS1090-XS850
XS850-XS510
XS510-XS360
XS360-XS210
XS210-XS100
upstream
downstream

Fig. 3