Water Research 246 (2023) 120731

Contents lists available at ScienceDirect

Water Research

journal homepage: www.elsevier.com/locate/watres

Establishing riverine nutrient criteria using individual taxa thresholds

A.D. Canning[a,b,*], R.G. Death[c]

[a] Centre for Tropical Water and Aquatic Ecosystem Research (TropWATER), James Cook University, 1 James Cook Drive, Townsville 4811, Australia
[b] Faculty of Science and Engineering, Southern Cross University, 1 Military Drive, Lismore 2480, Australia
[c] School of Natural Sciences, Massey University, Private Bag 11 222, Palmerston North 4442, New Zealand

ARTICLE INFO

Keywords:
Ecological health
Eutrophication
Macroinvertebrates
Nutrient criteria
Nutrients
River management

ABSTRACT

Nutrient enrichment is one of the most pervasive impacts on aquatic ecosystems globally. Approaches to establish nutrient criteria that safeguard aquatic ecosystem health are highly variable and, in many instances, criteria are derived from correlations between in-situ nutrient concentrations and biological indices. Summarising entire assemblages with a single index can result in a substantial loss of information and potentially weaker relationships. In this study, we compared the derivation of nutrient criteria using biological indices and those from individual taxa for rivers and streams in New Zealand. Random forest models, including nutrient concentrations, were built to predict two biological indices and individual taxa across New Zealand's river monitoring network. For all acceptable models, the response of the biological indices and individual taxa to increasing Dissolved Inorganic Nitrogen (DIN) and Dissolved Reactive Phosphorus (DRP) were then predicted for every river reach across the nation, and nutrient concentrations that protected 80% of taxa were then identified. Models for the biological indices were poor but were good for most of the taxa, with nutrient concentrations almost always being the most influential factor. To ensure persistence of at least 80% of the taxa within a river reach, we estimated that DIN (Dissolved Inorganic Nitrogen) concentrations would need to be below 0.57–1.32 mg/L, and DRP (Dissolved Reactive Phosphorus) concentrations below 0.019–0.033 mg/L, depending on the river type. In general, high order, low slope rivers and streams required more stringent nutrient criteria than steep, low order streams. The link between nutrient concentrations and biological indices were weak and likely suffer from the loss of information from summarising an entire assemblage into a single numeric. We consider that the derivation of nutrient criteria for waterways should also examine the individual relationships with the taxa in a river system to establish protection for a desired proportion of taxa.

1. Introduction

Managing nutrients to protect aquatic species and ecosystem health is one of the biggest challenges currently facing environmental managers (Carpenter and Bennett, 2011; Carrizo et al., 2017; Vorosmarty et al., 2010). Nitrogen and phosphorus inputs to land are increasing dramatically as native vegetation continues to be replaced by agriculture in many parts of the world, and the use of existing farmland is intensified in more developed countries (Anon., 2005; Carpenter et al., 1998; Scanlon et al., 2007). Much of this nutrient addition ends up in groundwater, lakes, rivers and ultimately the ocean with often detrimental impacts on those ecosystems. Nutrient enrichment can drive excessive algal and microbial growth, which then alters biological communities, and can even result in large die-off events (e.g., mass fish kills) if dissolved oxygen is driven to lethal levels (Ferreira et al., 2015; Le Moal et al., 2019; Wurtsbaugh et al., 2019).

Not surprisingly, there has been considerable research to investigate nutrient ecosystem health relationships and to establish nutrient criteria to protect freshwater life (Dodds, 2007; Evans-White et al., 2009; Herlihy and Sifneos, 2008). Despite this plethora of information, and often well-founded legislation to protect waterways from excess nutrients, the problem of how to limit nutrient impacts remains and, may even be getting worse (Seitzinger et al., 2010). Many countries, including those with generally good environmental protections, extensive data monitoring and strong supporting science, such as New Zealand, are still hesitating to develop rigorous guidelines for managers to use in managing eutrophication (Foote et al., 2015; Joy and Canning, 2020; Weeks et al., 2016).

The limitations of scientifically robust nutrient guidelines or criteria, even when there is extensive data and ongoing monitoring, may rest on

* Corresponding author.
E-mail address: adam.canning@jcu.edu.au (A.D. Canning).

https://doi.org/10.1016/j.watres.2023.120731
Received 12 May 2023; Received in revised form 29 September 2023; Accepted 10 October 2023
Available online 12 October 2023

the multidimensionality of how nutrients impact ecosystem health (Canning and Death, 2021; Wagenhoff et al., 2017b, 2017a). Nutrient inputs can increase algal and microbial production directly, which can cause a range of cascading impacts that are difficult to reliably predict. Increased algal and microbial growth can promote the growth of primary consumers, such as invertebrates, either by increasing food supply for algivores, or through conditioning nutrient-poor organic matter for detritivores (Dodds and Smith, 2016; Elser et al., 2000; Ferreira et al., 2015). Given that smaller, fast-growing invertebrates have a high nutrient turnover and demand for nitrogen and phosphorus, nutrient enrichment may increase the abundance of small invertebrates (Elser et al., 2000; Gillooly et al., 2005; Hessen et al., 2013), and result in assemblages that are less energetically rewarding for fish (Schindler and Eby, 1997; Shearer and Hayes, 2019; Vinson and Baker, 2008). The increased metabolic activity and decomposition can also increase hypoxic conditions; alter the availability of course and fine particulate organic matter for microbes, invertebrates, and downstream ecosystems (Benstead et al., 2009; Davis et al., 2010; Stelzer et al., 2003); and reduce the abatement of greenhouse gas by aquatic environments (Macreadie et al., 2017). At very high concentrations, nutrients can be directly toxic to invertebrates and fish by disrupting numerous metabolic pathways (Camargo and Alonso, 2006; Romano and Zeng, 2013). Nutrient enrichment can also alter the intensity and incidence of pathogenic infections, often by exacerbating infections of generalist parasites with direct or simple lifecycles (Frost et al., 2008; Johnson et al., 2010). As a result, nutrients can adversely impact, directly or indirectly entire ecological communities, and limits on nutrient enrichment of rivers need to be adopted to reduce the prevalence of adverse impacts.

Limits on nutrient enrichment in rivers are typically derived from relationships between biological metrics of ecological health and nutrient concentrations, and to a lesser extent from laboratory experiments on toxicity responses. Given the wide array of potential ecological impacts, using observational data of the linkages between biological assemblages and in-situ nutrient concentrations to derive nutrient criteria can be problematic (Evans-White et al., 2013; Huo et al., 2018; Poikane et al., 2019; Wagenhoff et al., 2017a). Biological community composition data is often summarised into biological metrics, such as the proportion of EPT (Ephemeroptera, Plecoptera and Trichoptera) taxa, community sensitivity and diversity indices, or even Observed/Expected ratios (Friberg et al., 2011; Norris and Thoms, 1999; Rosenberg and Resh, 1993), which are then modelled against nutrient concentrations for meaningful thresholds or change-points (King and Richardson, 2003; Poikane et al., 2019; Yuan, 2010). Clearly, not all the species that comprise an index will be responding to the potential stressor in the same way, or at the same threshold. Any index by its very nature will summarise information from the multiple responses of taxa (Dale and Beyeler, 2001; Landres, 1992). Using laboratory experiments to derive nutrient criteria are also problematic, particularly given that laboratory environments often do not reflect the range and variability of stressors that are observed in the real world, such as flooding, and the food web and dissolved oxygen responses to nutrient enrichment (Camargo and Alonso, 2006). It would make more sense to examine the responses of each individual taxa within the community (rather than biological indices) for significant changes with increasing nutrient enrichment, in the context of the localised environment (rather than an artificial laboratory environment) (Baker and King, 2010; Sundermann et al., 2015). This should provide the most biologically realistic analysis for establishing thresholds that represent real community effects. By linking community changes analytically with a stress gradient, it also reduces the influence of other potential stressors, that may be concurrently affecting biological communities.

Here we used data from across New Zealand to derive river-specific nutrient criteria that are built on the protection of individual species, and then compare these to nutrient criteria derived to support the achievement of biological index thresholds. Biological indices examined are those with legislated national minimum standards (termed 'national bottom lines' or 'NBL') in the *National Policy Statement for Freshwater Management 2020* (NPS-FM 2020), specifically the Macroinvertebrate Community Index (MCI) and the Average Score Per Metric (ASPM). New Zealand provides an ideal case study given the recent, widespread decline in water quality due to the intensification of agriculture, large environmental variability within small geographical distances, and an extensive national monitoring data on riverine macroinvertebrates and nutrients (Joy and Canning, 2020; Julian et al., 2017; Snelder et al., 2021). We focus on identifying nutrient criteria for invertebrates as they are well established biological indicators of river water quality because of their sensitivity to pollutants, prolonged exposure to water conditions due to their sedentary nature and long lifespans, the ecological insights their diversity and functional composition offers, and their role as food higher trophic species such as fish (Abbasi and Abbasi, 2011; Parmar et al., 2016; Sumudumali and Jayawardana, 2021). Setting nutrient criteria for macroinvertebrates, amongst other components such as periphyton and oxygen, is also a requirement within the NPS-FM 2020. While nutrient criteria for periphyton-based outcomes have been well-examined in New Zealand (Biggs, 2000; Kilroy et al., 2019; Snelder et al., 2019), macroinvertebrates have not received comparable attention.

2. Materials and methods

River-specific nutrient criteria were derived with the following steps: (1) collation of macroinvertebrate and environmental data; (2) using random forests to establish relationships between environmental variables (including nutrient concentrations) and either the probability of occurrence for each taxon or the probability of biological indices meeting the NBL; (3) using cross-validation to assess the performance of each model; (4) for each well-performing model, predicting across a nutrient concentration gradient at every river reach in New Zealand either the change in the probability of occurrence for each taxon or change in probability of biological indices meeting the NBL; (5) at every river reach, identifying the nutrient concentrations that yield a 20% change in the probability of occurrence for each taxon or change in probability of biological indices meeting the NBL from that predicted in low nutrient conditions; and (6) at every river reach, identify the nutrient concentrations that protect 80% of taxa from experiencing more than a 20% change in occurrence probability from that predicted in low nutrient conditions.

2.1. Macroinvertebrate and environmental data

Macroinvertebrate data used in this study was sourced from New Zealand's regional environmental monitoring network of rivers and streams (Ministry for the Environment et al., 2019). Benthic macroinvertebrates have been surveyed annually by regional environment agencies between 1990 and 2016, are typically sampled in riffles using either kick nets or Surber samplers, stored in either ethanol or formalin, and identified using common keys (Moore, 1998; Winterbourn et al., 1989). A total of 396 taxa were identified from 15,508 surveys collected from 1966 sites nationwide, and the MCI and ASPM indicators calculated (J.). Only samples that had concurrent monthly assessments of dissolved inorganic nitrogen (DIN) and Dissolved Reactive Phosphorus (DRP) concentrations at those sites for the preceding 12 months were used, with the monthly data collected over the 12 months summarised as medians for analysis. As different water quality laboratories are used across the country, laboratories used are accredited by International Accreditation New Zealand (IANZ), which requires adherence to quality controls, including participation in an Inter-Laboratory Comparison Programme, where they analyse external water samples and benchmark their results against other labs. This yielded a collection of 1784 samples across 312 sites (Fig. 1, Table 1). In keeping with the recommendations made by the Science and Technical Advisory Group that informed the development of New Zealand's recent freshwater policy (Essential

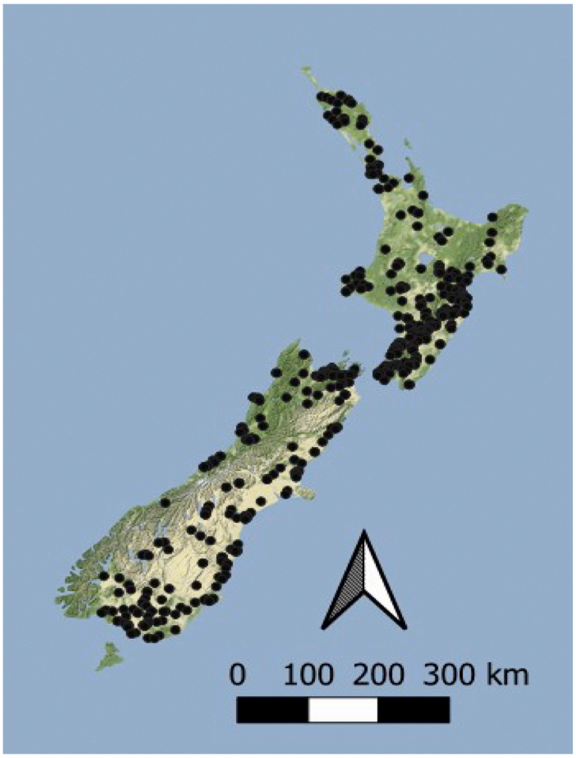

Fig. 1. The location of sites sampled by Environment Agencies annually for macroinvertebrates and monthly for nutrients across New Zealand (Ministry for the Environment et al., 2019).

Table 1
The minimum, median, mean, maximum, 25th and 75th percentiles of the median of 12 consecutive monthly samples, and the MCI and ASPM scores, across New Zealand's state of environment monitoring network data used in this analysis.

Metric	Min	25th percentile	Median	Mean	75th percentile	Max
DIN	<0.001	0.09	0.32	0.65	0.75	7.35
DRP	0.0003	0.005	0.010	0.024	0.020	0.682
MCI	38	88	102	101	115	153
ASPM	0.04	0.26	0.39	0.38	0.51	0.80

Freshwater Science and Technical Advisory Group 2019a, 2019b) nutrients were measured in their dissolved form for this study as this is the bioavailable form. Using dissolved forms reduced the potentially confounding effects of having areas with high nutrient loads but low ecological impact in situations where a high proportion of nutrients is either bound to suspended sediment or stable in organic matter. New Zealand, with its high tectonic activity, has highly varied geology with soils differing substantially in their erodibility and affinity to bind nutrients (Davies-Colley, 2013; Elliott et al., 2005; McDowell and Condron, 2000; Quinn and Stroud, 2002). Regional authorities can determine localised relationships between dissolved and total forms of nutrients if desired.

Most sites do not have consistent physical habitat assessments. Instead, most environmental variables used in the analysis were extracted from the Freshwater Environments New Zealand (FENZ) geodatabase (Leathwick et al., 2010), except hydrological characteristics which were sourced from Booker and Woods (2014), and fine sediment cover from Clapcott et al. (2011) (this resulted in 25 variables, Table 2). FENZ is a geodatabase that contains a range of modelled and measured habitat, land use and climatic characteristics for every river reach in New Zealand. Booker and Woods (2014) modelled hydrological

Table 2
Environmental predictors used in the random forest models to predict the biological indices and each taxa (D.J. Booker and Woods, 2014; Leathwick et al., 2010).

SegJanAirT	Average January Air Temperature (°C)
SegMinTNorm	Average minimum daily air temperature (°C) normalised with respect to SegJanAirT
SegSlope	Slope of segment (°)
SegRipShade	The likely proportion of stream shaded from riparian
SegRipNative	Proportion of native riparian vegetation within a 100 m buffer of the river
USCalcium	Average calcium concentration of rocks in the catchment, 1 = very low to 4 = very high
USHardness	Average hardness of rocks in the catchment, 1 = very low to 5 = very high
USPhosporus	Average phosphorus concentration of rocks in the catchment, 1 = very low to 5 = very high
USPeat	Proportion of upstream catchment covered by peat
USWetland	Proportion of upstream catchment covered by wetland
USGlacier	Proportion of upstream catchment covered by glacier
ReachSed	Weighted average of proportional cover of bed sediment using categories of: 1 = mud; 2 = sand; 3 = fine gravel; 4 = coarse gravel; 5 = cobble; 6 = boulder; 7 = bedrock
ReachHab	Weighted average of proportional cover of local habitat using categories of: 1 = still; 2 = backwater; 3 = pool; 4 = run; 5 = riffle; 6 = rapid; 7 = cascade
DSDist2Coast	Distance to coast (km)
USDaysRain	Days/year with rainfall greater than 25 mm in the upstream catchment
USLake	Proportion of upstream catchment covered by lake
USWetland	Proportion of upstream catchment covered by wetland
USIndigFor	Proportion of upstream catchment covered by indigenous forest
USNative	Proportion of upstream catchment covered by native vegetation
USPasture	Proportion of upstream catchment covered by pasture
USGlacier	Proportion of upstream catchment covered by glaciers
MALF	Mean annual 7-day low flow (m^3/sec)
MeanF	Mean of all daily flows (m^3/sec)
Feb	Mean daily February flow divided by the overall mean daily flow
WidthQ5	Predicted wetted width (m) at 5th percentile of flow
FRE3	Predicted annual frequency of flows exceeding three times the annual median flow
SedAdded	The difference between the current and human-absent predicted fine sediment cover (%)

statistics of flow volume, flow variability and stream width for all river reaches. There are 567,299 river reaches, averaging ~700 m in length, covering the entire surface water network across New Zealand. There was very little collinearity between variables (Table S1).

2.2. Data analysis

2.2.1. Modelling macroinvertebrate occurrence

Random forests is a machine learning method that uses a collection of regression trees, whereby each tree is fitted to a bootstrapped sample (with replacement) and then validated on the out-of-bag sample (Breiman, 2001). Random forest predictions are the average of the predictions of each tree. Regression trees, and consequently random forests, work by partitioning observations at splits of predictors that minimise the sum of squares error. They have a high level of flexibility and can handle non-linear relationships and complex interactions (Cutler et al., 2007; Ellis et al., 2012; Hastie et al., 2009).

Random forests modelling was used to yield the probability of occurrence for all macroinvertebrate taxa, or the probability of passing the NBL for MCI and ASPM (90 and 0.3, respectively), using the potential predictor variables in Table 2 and median DIN or DRP concentrations from the preceding 12 months. As both nutrients are often correlated (Canning, 2020), separate models were constructed for each nutrient. Taxa were only included in the study if they were present in at least 20 surveys (n = 181 taxa included). Each model was made using the 'randomForest' function (trees=500) from the randomForest package in R (Liaw and Wiener, 2002; R Core Team, 2016). The rfUtilities package (Evans and Murphy, 2019) was then used to assess the performance (interrater reliability) of each model by calculating the average Cohen's Kappa (McHugh, 2012a) for the probability of occurrence for each taxon, or probability of NBL for indices, using 99 cross-validations, each with 20% data withheld. According to McHugh (McHugh, 2012b), models that have strong levels of agreement have Cohen's Kappa values of at least 0.8 – a threshold we adopted for considering a model's performance as acceptable. The globally important variables were also identified using the 'importance' function, which measures the decrease in Gini index from splitting on each variable, averaged over all trees.

2.2.2. Deriving nutrient criteria for every river reach and river type

For every model with a Cohen's Kappa of at least 0.8, the probability of taxon occurrence and probability of index NBL passing was predicted in response to a DIN (at 0.05 mg/L increments) or DRP (at 0.001 mg/L increments) gradient for every river reach, using the same environmental conditions in the initial models. We then determined the DIN and DRP concentrations predicted to yield a 20% change in the probability of taxon occurrence or probability of biological index passing NBLs, relative to that predicted at low nutrient levels (DIN=0.10 mg/L, DRP=0.008 mg/L) that are broadly comparable to concentrations observed in native cover reference conditions (Julian et al., 2017; Larned et al., 2016; McDowell et al., 2013). We considered these the observable effects concentrations. This resulted in a database of observable effect DIN and DRP concentrations for every taxon and biological index with good models at every river reach.

According to the Australia and New Zealand Guidelines for Fresh and Marine Water Quality (ANZG, 2018), when setting pollutant limits based on species protection, 99% species protection thresholds are applicable to ecosystems with high ecological/conservation value, 95% species protection thresholds are applicable for slightly to moderately disturbed ecosystems, and 80–90% species protections is applicable for highly disturbed ecosystems. As we set to compare species protection thresholds with biological indicators set at the national bottom-line (NBL) scores (broadly similar to scores set for highly disturbed systems), we calculated DIN and DRP criteria that protect 80% of taxa from a 20% change in occurrence probability or protect the biological indices from a 20% change in meeting the NBLs, relative to that predicted in low nutrient conditions. Percentiles were calculated using the quantile function in R.

2.2.3. Summarising nutrient criteria by river classifications

The ANZG (2018) also presents water quality guidelines for New Zealand based on river-type, rather than individual river reaches, using the New Zealand River Environment Classification (REC) system (Snelder et al., 2010). The REC classifies each river reach into a grouping with similar characteristics, based on landform, stream order, climate and geology. Classification trees were used to group and summarise streams with similar DIN or DRP criteria (for protecting 80% of taxa) by the REC climate, geology and source of flow classifications. Trees were calculated with a Poisson response distribution using the rpart package in R (R Core Team, 2016; Therneau and Atkinson, 2022). Within each of the resulting tree groupings, the median nutrient criteria and median absolute deviation (MAD) was determined for all taxa protection levels.

3. Results

3.1. Random forest models

For the taxon-specific DIN-based random forest models, 100 taxa (of 181 assessed) had Cohen Kappa values of at least 0.8. On average across all taxa, DIN was the most important predictor (based on mean decrease in Gini index), followed by MALF (mean annual low flow) and MeanF (mean flow), though there was substantial variation in magnitude of variable importance between species (Fig. 2, Table S2). For the DRP-based random forest models, 104 taxa (of 181 assessed) had Cohen Kappa values of at least 0.8 and, on average across all taxa, DRP was the most important predictor, followed by MALF and USDaysRain (average number of days of rain annual in upstream catchment). Although again there was considerable variation in magnitude of variable influence between species (Fig. 2). The invertebrates whose distribution models were most influenced by DIN or DRP are presented in Table 3.

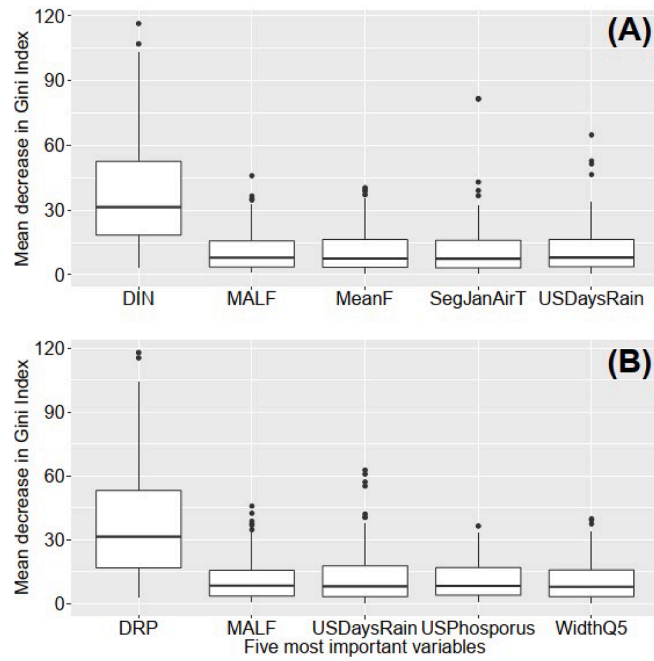

Fig. 2. The reduction in the Gini Index for the five most important variables (on average) in predicting the distribution of New Zealand riverine macroinvertebrate species using random forests. The higher the Gini Index is, the greater the variance and misclassification there is. Variables with the largest reduction in Gini Index are more influential in yielding models with a better classification rate. Panels (A) and (B) show DIN and DRP based models respectively. Boxplots indicate the 5th, 25th, 50th' 75th and 95th percentiles, while dots indicate extreme values.

All models for the biological indices (MCI and ASPM) had Cohen Kappa values below 0.8, with values between 0.63 and 0.68 (Table 4, Table S2). For all of the biological indicator models, the nutrient variable was the most important predictor (based on mean decrease in Gini index), followed by USHardness (upstream hardness), SedAdded (anthropogenic sedimentation), and USNative (upstream native cover).

Table 3
Taxa whose random forest distribution models were most influenced by nutrient concentrations. FFG indicates the functional feeding group. Scores indicate the decrease in Gini index by inclusion of DIN or DRP as a predictor. Direction indicates whether, on average, the taxa's probability of occurrence increases or decreases with nutrient enrichment.

Taxa	FFG	DIN	DRP	Direction
Aoteapsyche spp.	Filter feeder	116.5	115.6	Increase
Hydrobiosis spp.	Predator	107.1	101.4	Increase
Pycnocentrodes spp.	Collector-gatherer	103.2	104.1	Increase
Psilochorema spp.	Predator	100.1	118.1	Decrease
Austrosimulium spp.	Filter feeder	92.0	88.2	Increase
Physella acuta	Grazer	90.2	75.0	Increase
Orthocladiinae	Collector-gather	88.7	88.0	Increase
Oligochaeta	Deposit feeder	85.3	88.5	Increase
Ostracoda	Scavenger	79.0	70.7	Increase
Pycnocentria	Collector-gatherer	76.4	76.8	Increase
Platyhelminthes	Predator	74.0	66.7	Increase
Archichauliodes diversus	Predator	73.8	73.1	Decreases
Oxyethira spp.	Filamentous algae	69.9	65.1	Increase

Table 4
Confusion matrices from the predicted versus observed MCI (Stark and Maxted, 2007) and ASPM (Collier, 2008) macroinvertebrate indicator scores at riverine monitoring sites across New Zealand. For both indicators, predictions were made using two random forest models which differed in the nutrient used as a predictor, one model used dissolved inorganic nitrogen (DIN) while the other used dissolved reactive phosphorus (DRP).

Indicator	Nutrient	Classification	Fail	Pass
MCI	DIN	Fail	349	144
		Pass	75	1201
	DRP	Fail	345	148
		Pass	74	1202
ASPM	DIN	Fail	412	176
		Pass	104	1077
	DRP	Fail	416	172
		Pass	101	1080

3.2. Nutrient criteria within REC classes

The classification trees used to group REC river classes with similar DIN concentrations that protect 80% of taxa, identified the larger rivers (high order) as requiring the most stringent criteria (0.57 mg/L) and smaller streams in cool climates and non-plutonic geology as requiring the least stringent criteria (1.32 mg/L; Table 5). While the classification trees used to group river classes with similar DRP concentrations that protect 80% of taxa, identified high order rivers in low and mid gradient valleys as requiring the most stringent criteria (0.019 mg/L), and rivers in high gradient valleys as requiring the least stringent criteria (0.029–0.033 mg/L; Table 5).

Table 5
The mean DIN and DRP concentrations (mg/L) predicted to protect a given proportion of macroinvertebrate taxa from a 20% change in occurrence probability across New Zealand's rivers and streams, as classified (with regression trees) into river environment classes following Snelder et al., 2010.

Nutrient	River classification	80%	85%	90%	95%	Number of reaches
DIN	High order rivers (HO)	0.57	0.44	0.3	0.25	30,485
	Mid order rivers (MO)	0.76	0.56	0.35	0.25	111,733
	Low order rivers in warm climates (LO/W)	0.86	0.63	0.39	0.27	104,509
	Low order rivers in cool climates with plutonic geology (LO/C/Pl)	1.09	0.77	0.45	0.3	287,780
	Low order rivers in cool climates with all other geology (LO/C/Al, HS, SS, M, VA, VB)	1.32	0.89	0.45	0.3	27,048
DRP	High order rivers in low and mid gradient valleys (HO/LG, MG)	0.019	0.016	0.013	0.012	29,683
	Low and mid order rivers in low and mid gradient valleys (LO, MO/LG, MG)	0.024	0.018	0.014	0.012	232,367
	Rivers in high gradient valleys with cool wet and extra wet climates (HG/CW, CX)	0.029	0.02	0.016	0.012	94,813
	Rivers in high gradient valleys with all other climates (HG/CD, WD, WW, WX)	0.033	0.022	0.016	0.014	204,692

4. Discussion

Nutrient enrichment is one of the most pervasive and rapidly increasing detrimental impacts on aquatic ecosystems. Determining the numerical criteria for nutrients to avoid these adverse impacts on biological communities is therefore critical for managing and mitigating these effects. Historically these criteria have been determined using correlations between biological indices and instream nutrient measures; however, biological indices by their very nature summarise the state of biological communities. Such summarisation potentially ignores idiosyncratic responses of individual taxa to the multidimensional impacts of nutrients. In this study, we compared the nutrient criteria determined from modelling the response of individual taxa to nutrient increase and those determined from the more traditional index nutrient correlations. We established the DIN and DRP concentrations to prevent a 20% change in the occurrence probability for 100 and 104 macroinvertebrate taxa, respectively. We compared those nutrient criteria with those modelled in an identical way for two commonly used biological indices in New Zealand – the macroinvertebrate community index (MCI) and the average score per metric (ASPM). However, models of the biological indices did not reach our acceptable performance criteria, thus nutrient criteria comparisons could not be made.

The poor performance of the biological index nutrient models may result from species replacement and the generally depauperate macroinvertebrate fauna in New Zealand (Thompson and Townsend, 2000). Previous studies have found the species richness of New Zealand's macroinvertebrates can be insensitive to anthropogenic disturbances such as flow reduction and/or land use change, there is simply a change in the species present and/or their relative abundance rather than a reduction in diversity (Death and Zimmermann, 2005; Dewson et al., 2007; Townsend et al., 1997). Changes in species identity can be easily overlooked if one is just focusing on biological indices and this may explain why the models for the biological indices were less accurate than those for individual taxa. However, the data used only allowed us to examine indices that use taxa presence absence; indices that use changes in absolute or relative abundance may prove more effective and warrant further investigation.

Most species distribution models fitted the data extremely well. The approach used here to model each individual taxa using random forests and identify changes in probability of occurrence across nutrient gradients is conceptually very similar to the gradient forest methodology (Ellis et al., 2012), which has been previously applied to examining nutrient gradients (Roubeix et al., 2017, 2016; Wagenhoff et al., 2012). This study differed from those gradient forest approaches by using Cohen's Kappa to filter poor performing models and only create nutrient criteria from taxa with well-performing models. For all those species with good models, nutrient concentrations were usually the most important variable determining the probability of occurrence. Approximately 70% of rivers by length are estimated to have nutrient concentrations above that expected if they were in a natural state (Ministry for the Environment and Stats NZ, 2020), it is perhaps not surprising then that nutrient enrichment is such a dominant driver of invertebrate community structure in New Zealand rivers and streams. Aside from the direct impacts of nutrient enrichment, such as hypoxia and toxicity (Camargo and Alonso, 2006; Dodds and Smith, 2016), the effect of nutrients on food quality and quantity are likely to be one of the strongest pathways through which nutrient increases impact species. Other land use changes, such as decreases in riparian vegetation or increases in sedimentation, may not have been so important in the models because their impacts are more indirect and/or localised.

The river classes with similar nutrient criteria, as identified by the classification trees, were largely distinguished from each other by stream order, slope, and to a lesser extent climate. In general, high order, low slope streams required more stringent nutrient criteria than their low order, steeper counterparts. There are numerous differences between small and large streams, including forest cover, aeration, degree of human impact, altitude, and flood frequency, all of which have the potential to influence invertebrates. The random forest models, however, generally found stream size (as indicated by mean annual low flow, mean flow, catchment rainfall, and stream width) as influential secondarily to nutrient concentrations, but ahead of flood frequency, upstream land use, and riparian vegetation cover. One plausible explanation is that there is greater aeration over riffles in low order, high slope streams, which could reduce the likelihood of hypoxia from occurring. Another potential explanation is that the food base of higher order streams comprises fine particulate organic matter (FPOM) more than lower order streams (Doretto et al., 2020; Vannote et al., 2011). FPOM will have greater cumulative surface area than CPOM, providing a greater area for microbial attachment, increasing decomposition rates and microbial conditioning of detritus with nutrients for detritivores. Further investigation would be required into the frequency and severity of hypoxic events and the size of particulate organic matter with stream size in New Zealand to better understand these potential causal pathways.

Detritivores dominated the taxa with models that were most influenced by nutrient concentrations. As detritus is typically nutrient poor, detritivores would benefit from nutrient enrichment as microbes attached to detritus are able to take up dissolve nutrients and condition detritus, improving the nutritional value and palatability, in turn, relieving nutrient constraints to growth (Elser et al., 2000; Hessen et al., 2013). This is an often-unrecognised pathway by which nutrient enrichment can affect aquatic ecosystems even if algal growth remains unaltered. A global meta-analysis by Ardón et al. (2021), synthesised the findings from 184 studies, 885 experiments, and 3497 biotic responses to nutrient enrichment, to examine how nutrient enrichment can alter ecosystem structure and function. They found that both autotrophic and heterotrophic pathways responded with similar magnitude to nutrient enrichment. Furthermore, they did not find convincing evidence of trophic buffering with either pathway. That is, the magnitude of the response to nutrient enrichment did not decline with increasing trophic level under either pathway. Thus, even in rivers without observable changes in algal biomass, alterations in the heterotrophic pathway from nutrient enrichment can have profound impacts at higher trophic levels.

4.1. New Zealand's freshwater management framework

New Zealand's freshwater ecosystems are primarily managed by regional authorities (16 regional and district councils), which establish, enact and enforce regional plans. Regional plans outline the values and goals held by local communities with respect to waterways (amongst other environmental matters), and stipulate the rules, limits, targets, approval frameworks used in managing land use and activities within the jurisdiction.

At the national level, the *National Policy Statement for Freshwater Management (NPS-FM 2020)* stipulates components and processes that must be adopted when regional authorities develop regional plans.

The NPS-FM 2020 is guided by the Māori (NZ First Nations People) concept of 'Te Mana o te Wai,' which refers to the fundamental importance of water and recognises that protecting the health of freshwater protects the health and well-being of the wider environment. Within Te Mana o te Wai, there is a hierarchy of obligations that need to be followed when making freshwater management decisions:

1. The health and well-being of water bodies and freshwater ecosystems.
2. The health needs of people (such as drinking water).
3. The ability of people and communities to provide for their social, economic, and cultural well-being, now and in the future.

Therefore, when making freshwater management decisions, social and economic needs cannot take precedence over the essential needs for healthy freshwater ecosystems. In the context of these obligations, the NPS-FM (2020) contains four values that are compulsory for regional authorities to maintain and improve (where degraded) at all natural freshwater ecosystems:

1. Ecosystem Health. This value stipulates five biophysical components (water quality, water quantity, habitat, aquatic life, and ecological processes) that must all be suitably managed to sustain the indigenous aquatic life expected in the absence of human disturbance or alteration.
2. Human Contact. This value stipulates the management of factors, such as pathogens, clarity, flow, and toxicants, to support human recreation, such as swimming, fishing, and boating.
3. Threatened Species. In addition to the requirement for Ecosystem Health, this value requires additional management measures be adopted to support the recovery of threatened species where needed.
4. Mahinga kai. This value requires management to uphold traditional Māori uses, harvests and customary practices.

When maintaining and improving freshwater ecosystem health, the NPS-FM (2020) also stipulates minimum environmental standards (termed 'National Bottom Lines' - NBL) for a suite of physical, chemical, and biological indicators. If a natural freshwater ecosystem fails any relevant NBL, then regional authorities must improve the waterway to at least the NBL within a generation. NBLs of relevance to rivers include: periphyton cover, ammonium and nitrate toxicity, dissolved oxygen, suspended and deposited fine sediment, the Macroinvertebrate Community Index (MCI), the Average Score Per Metric (a macroinvertebrate index; ASPM), and *Escherichia coli*. In addition to these indicators and standards, regional authorities are required to implement any other indicators and standards necessary to manage freshwater ecosystem health and meet the expectations of local communities.

While there are NBLs for nitrate and ammonium concentrations based on laboratory toxicity studies, these concentrations have not been developed to support ecological health. The NPS-FM (2020) requires that regional authorities set instream standards for nitrogen and phosphorus to support the achievement of any indicator affected by nutrients and meet the requirements of nutrient-sensitive downstream environments. This direction is outlined in section 3.13 of New Zealand's *National Policy Statement for Freshwater Management 2020*, which states:

(1) "To achieve a target attribute state for any nutrient attribute, and any attribute affected by nutrients, every regional council must, at a minimum, set appropriate instream concentrations and exceedance criteria, or instream loads, for nitrogen and phosphorus.

(2) Where there are nutrient-sensitive downstream receiving environments, the instream concentrations and exceedance criteria, or the instream loads, for nitrogen and phosphorus for the upstream contributing water bodies must be set so as to achieve the environmental outcomes sought for the nutrient-sensitive downstream receiving environments.

(3) In setting instream concentrations and exceedance criteria, or instream loads, for nitrogen and phosphorus under this clause, the regional council must determine the most appropriate form(s) of nitrogen and phosphorus to be managed for the receiving environment.

(4) Every regional council must adopt the instream concentrations and exceedance criteria, or instream loads, set under subclauses (1) and (2) as nutrient outcomes needed to achieve target attribute states.

(5) Examples of attributes affected by nutrients include periphyton, dissolved oxygen (Appendix 2A, Tables 2 and 7 and Appendix 2B, Tables 17, 18, and 19), submerged plants (invasive species)

> (Appendix 2B, Table 12), fish (rivers) (Appendix 2B, Table 13), macroinvertebrates (Appendix 2B, Tables 14 and 15), and ecosystem metabolism (Appendix 2B, Table 21)."

In practice, this directive requires regional authorities to identify suitable nutrient criteria that support achieving multiple indicator targets, including the MCI (Stark and Maxted, 2007) and ASPM (Collier, 2008) bioindicators examined here, and adopt the most protective criteria to support the achievement of all ecosystem health indicator targets. While previous relationships have been established in academic literature between nutrient concentrations and epilithon cover (benthic algae growing on rocks) (Biggs, 2000; Kilroy et al., 2019; Snelder et al., 2019), further work, beyond nutrient-macroinvertebrate relationships, is required to inform relationships between nutrients and other freshwater ecosystem components, such as dissolved oxygen, ecosystem metabolism, forms of periphyton other than epilithon, aquatic plants, and fish (Canning, 2020).

The present study seeks to inform the nutrient criteria adoption process by establishing relationships between instream nutrient concentrations and macroinvertebrate outcomes. Although the models of the MCI and ASPM had unsatisfactory performance, using the present findings, regional authorities can identify nutrient criteria that protect a desired proportion of macroinvertebrate taxa (i.e., 80%, 90% or 95%) from a large change in occurrence probability. To exemplify how nutrient criteria derived here may be applied, Table 6 shows a selection of river monitoring sites in lowland rural areas across New Zealand's Manawatu River catchment. This shows the current 5-year median (monthly sampling) DIN and DRP concentrations (from www.lawa.co.nz), and the target concentrations and reduction (%) required to meet the target concentrations, depending on the level of protection desired.

The level of protection adopted would be a normative management decision, informed by community aspirations and any potential consequential impacts on ecosystem health and other values. Decision makers need to be cognisant that the impacts of disrupting sensitive taxa may not necessarily be linear to the proportion of taxa disrupted, particularly if disturbed species have keystone ecological positions or are valued by the community more so than others (Rooney and McCann, 2012; Saint-Béat et al., 2015; Zhao et al., 2016). For example, Canning and Death (2021) demonstrated, using food webs rivers across the Manawatu region (NZ), non-linear responses in total system throughflow of energy, respiration, indirect flow intensity, relative ascendency, and food web synergism, across a nutrient enrichment gradient. Further research is needed to understand the ecosystem-wide impacts of altering species most sensitive to change from nutrient enrichment in different river types across New Zealand.

Although the models of biological indices had unsatisfactory performance, biological indicators are still valuable as they can indicate a combination of pressures (Al-Zankana et al., 2020; Kath et al., 2018; Nõges et al., 2016; Poikane et al., 2020). Potential pressures that were not directly included as model predictors include changes in: observed oxygenation, detritus composition, extreme temperature events, pesticide and herbicide runoff, pharmaceuticals in wastewater, microplastics (Mora-Teddy and Matthaei, 2020), physical habitat, salinity, climate, hydrology, and hyporheic interactions (Beermann et al., 2018; Blöcher et al., 2020; Matthaei et al., 2010; Mouton et al., 2022). The NPS-FM (2020) mandates the use of the examined biological indices and requires they be managed through an adaptive framework. Adaptive management refers to a systematic process for improving policies and

Table 6
The current dissolved inorganic nitrogen (DIN) and dissolved reactive phosphorus (DRP) concentrations (mg/L) at six sites across the Manawatu River catchment, along with potential target concentrations and reductions (%) estimated to protect either 80%, 85%, 90%, or 95% (threshold), of macroinvertebrate taxa from a substantial change in occurrence probability. Site river environment classifications (REC) are also provided following Snelder et al., 2010, along with coordinates using NZGD2000 (EPSG: 4167). NR indicates 'no reduction' required to comply with criteria.

	Site name	Manawatū at Hopelands	Manawatu at Ōpiki Bridge	Ōroua at Awahuri Bridge	Mangatainoka at Brewery	Pohangina at Mais Reach	Mangapapa at Troup Rd
Coordinate	Northing	−40.359787	−40.435014	−40.275647	−40.422678	−40.224579	−40.341929
	Easting	175.961725	175.468598	175.521583	175.861913	175.782998	175.847880
	REC	CW/L/SS/P/HO/LG	CW/L/M/P/HO/LG	CD/H/M/P/HO/LG	CW/L/Al/P/HO/LG	CW/H/HS/P/HO/LG	CW/L/M/P/MO/LG
	Current DIN concentration (mg/L)	0.64	0.60	0.82	0.74	0.05	0.28
Threshold	Current DRP concentration (mg/L)	0.028	0.020	0.019	0.016	0.014	0.013
80%	Target DIN concentration (mg/L)	0.57	0.57	0.57	0.57	0.57	0.76
	Reduction required (%)	−11	−5	−30	−23	NR	NR
	Target DRP concentration (mg/L)	0.190	0.190	0.190	0.190	0.190	0.024
	Reduction required (%)	NR	NR	NR	NR	NR	NR
85%	Target DIN concentration (mg/L)	0.44	0.44	0.44	0.44	0.44	0.56
	Reduction required (%)	−31	−27	−46	−41	NR	NR
	Target DRP concentration (mg/L)	0.016	0.016	0.016	0.016	0.016	0.016
	Reduction required (%)	−43	−20	−16	NR	NR	NR
90%	Target DIN concentration (mg/L)	0.30	0.30	0.30	0.30	0.30	0.35
	Reduction required (%)	−53	−50	−63	−59	NR	NR
	Target DRP concentration (mg/L)	0.013	0.013	0.013	0.013	0.013	0.014
	Reduction required (%)	−54	−35	−32	−19	−7	NR
95%	Target DIN concentration (mg/L)	0.25	0.25	0.25	0.25	0.25	0.25
	Reduction required (%)	−61	−58	−70	−66	NR	−11
	Target DRP concentration (mg/L)	0.012	0.012	0.012	0.012	0.012	0.012
	Reduction required (%)	−57	−40	−37	−25	−14	−8

practices by learning from the monitoring outcomes of previously implemented decisions and adjusting strategies based on that feedback (Eberhard et al., 2009; Keith et al., 2011; Kingsford et al., 2011). Given that the biological indices can respond to a range of pressures, continuing with an adaptive management framework for these indices would still be prudent, with the individual taxa based nutrient criteria derived in the present study used as a complement to, rather than a replacement of, adaptively managed biological indices. The derived nutrient criteria could serve as a guide as to whether nutrients are likely to be affecting invertebrate assemblages and if nutrient reductions may be required. If current nutrient concentrations are well within the criteria but the biological indices indicators poor health, then the adaptive management response should examine and reduce other potential stressors further until the desired level of health is achieved.

5. Conclusion

Our modelling suggests that DIN and DRP are amongst the most important factors determining the distributions of New Zealand macroinvertebrates and thus the associated biological indices. However, the loss of information caused by summarising an entire assemblage into the biological indicators examined resulted in inferior models to those for individual taxa. We consider that examining all species individually, rather than a general biological indicator, will be more informative and accurate for deriving nutrient criteria to safeguard freshwater ecosystem health. Although ecosystem health goals have historically been set with reference to biological assemblage indices, setting nutrient criteria to achieve the desired outcomes would be better achieved by using taxa individually and then summarising the results.

CRediT authorship contribution statement

A.D. Canning: Conceptualization, Methodology, Data curation, Visualization, Writing – original draft. **R.G. Death:** Conceptualization, Methodology, Writing – original draft.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

Supplementary materials

Supplementary material associated with this article can be found, in the online version, at doi:10.1016/j.watres.2023.120731.

References

Abbasi, T., Abbasi, S.A., 2011. Water quality indices based on bioassessment: the biotic indices. J. Water Health 9, 330–348. https://doi.org/10.2166/wh.2011.133.

Al-Zankana, A.F.A., Matheson, T., Harper, D.M., 2020. How strong is the evidence – based on macroinvertebrate community responses – that river restoration works? Ecohydrol. Hydrobiol. 20, 196–214. https://doi.org/10.1016/j.ecohyd.2019.11.001.

Anon, 2005. Millennium Ecosystem Assessment, 2005. Ecosystems and human well-being: synthesis.

ANZG, 2018. Australian and New Zealand Guidelines for Fresh and Marine Water Quality. Canberra, Australia.

Ardón, M., Zeglin, L.H., Utz, R.M., Cooper, S.D., Dodds, W.K., Bixby, R.J., Burdett, A.S., Follstad Shah, J., Griffiths, N.A., Harms, T.K., Johnson, S.L., Jones, J.B., Kominoski, J.S., McDowell, W.H., Rosemond, A.D., Trentman, M.T., Van Horn, D., Ward, A., 2021. Experimental nitrogen and phosphorus enrichment stimulates multiple trophic levels of algal and detrital-based food webs: a global meta-analysis from streams and rivers. Biol. Rev. 96, 692–715. https://doi.org/10.1111/BRV.12673.

Baker, M.E., King, R.S., 2010. A new method for detecting and interpreting biodiversity and ecological community thresholds. Methods Ecol. Evol. 1, 25–37. https://doi.org/10.1111/j.2041-210X.2009.00007.x.

Beermann, A.J., Elbrecht, V., Karnatz, S., Ma, L., Matthaei, C.D., Piggott, J.J., Leese, F., 2018. Multiple-stressor effects on stream macroinvertebrate communities: a mesocosm experiment manipulating salinity, fine sediment and flow velocity. Sci. Total Environ. 610–611, 961–971. https://doi.org/10.1016/j.scitotenv.2017.08.084.

Benstead, J.P., Rosemond, A.D., Cross, W.F., Wallace, J.B., Eggert, S.L., Suberkropp, K., Gulis, V., Greenwood, J.L., Tant, C.J., 2009. Nutrient enrichment alters storage and fluxes of detritus in a headwater stream ecosystem. Ecology 90, 2556–2566. https://doi.org/10.1890/08-0862.1.

Biggs, B., 2000. Eutrophication of streams and rivers: dissolved nutrient-chlorophyll relationships for benthic algae. J. North Am. Benthol. Soc. 19, 17–31.

Blöcher, J.R., Ward, M.R., Matthaei, C.D., Piggott, J.J., 2020. Multiple stressors and stream macroinvertebrate community dynamics: interactions between fine sediment grain size and flow velocity. Sci. Total Environ. 717, 137070 https://doi.org/10.1016/j.scitotenv.2020.137070.

Booker, D.J., Woods, R.A., 2014. Comparing and combining physically-based and empirically-based approaches for estimating the hydrology of ungauged catchments. J. Hydrol. 508, 227–239. https://doi.org/10.1016/j.jhydrol.2013.11.007.

Breiman, L., 2001. Random forests. Mach Learn. 45, 5–32.

Camargo, J., Alonso, Á., 2006. Ecological and toxicological effects of inorganic nitrogen pollution in aquatic ecosystems: a global assessment. Environ. Int. 32, 831–849. https://doi.org/10.1016/j.envint.2006.05.002.

Canning, A., 2020. Nutrients in New Zealand Rivers and Streams: An exploration and Derivation of National Nutrient Criteria. Essential Freshwater Science and Technical Advisory Group, Wellington, New Zealand. https://doi.org/10.6084/m9.figshare.12116460.

Canning, A., Death, R., 2021. The influence of nutrient enrichment on riverine food web function and stability. Ecol. Evol. 11, 942–954. https://doi.org/10.1002/ece3.7107.

Carpenter, S.R., Bennett, E.M., 2011. Reconsideration of the planetary boundary for phosphorus. Environ. Res. Lett. 6, 14009.

Carpenter, S.R., Caraco, N.F., Correll, D.L., Howarth, R.W., Sharpley, A.N., Smith, V.H., 1998. Nonpoint pollution of surface waters with phosphorus and nitrogen, Issues in Ecology No. 3. Ecological Society of America.

Carrizo, S.F., Jahnig, S.C., Bremerich, V., Freyhof, J., Harrison, I., He, F.Z., Langhans, S. D., Tockner, K., Zarfl, C., Darwall, W., 2017. Freshwater megafauna: flagships for freshwater biodiversity under threat. Bioscience 67, 919–927. https://doi.org/10.1093/biosci/bix099.

Clapcott, J., Wagenhoff, A., Neale, M., Storey, R., Smith, B., Death, R., Harding, J., Matthaei, C., Quinn, J., Collier, K., Atalah, J., Goodwin, E., Rabel, H., Mackman, J., Young, R., 2017. Macroinvertebrate Metrics For the National Policy Statement for Freshwater Management. Nelson, New Zealand, New Zealand.

Clapcott, J.E., Young, R.G., Harding, J.S., Matthaei, C.D., Quinn, J.M., Death, R.G., 2011. Sediment Assessment Methods: protocols and guidelines for assessing the effects of deposited fine sediment on in-stream values. Cawthron Institute, Nelson, New Zealand 108.

Collier, K., 2008. Average score per metric: an alternative metric aggregation method for assessing wadeable stream health. N Z J. Mar. Freshwater Res. 42, 367–378. https://doi.org/10.1080/00288330809509965.

Cutler, D.R., Edwards Jr, T.C., Beard, K.H., Cutler, A., Hess, K.T., Gibson, J., Lawler, J.J., 2007. Random forests for classification in ecology. Ecology 88, 2783–2792.

Dale, V.H., Beyeler, S.C., 2001. Challenges in the development and use of ecological indicators. Ecol. Indic. 1, 3–10. https://doi.org/10.1016/S1470-160X(01)00003-6.

Davies-Colley, R., 2013. River water quality in New Zealand: an introduction and overview. In: Dymond, J. (Ed.), Ecosystem Services in New Zealand - Conditions and Trends. Manaaki Whenua Press, Lincoln, New Zealand, pp. 432–447.

Davis, J.M., Rosemond, A.D., Eggert, S.L., Cross, W.F., Wallace, J.B., 2010. Long-term nutrient enrichment decouples predator and prey production. Proc. Natl. Acad. Sci. 107 https://doi.org/10.1073/pnas.0908497107, 121 LP–126.

Death, R.G., Zimmermann, E.M., 2005. Interaction between disturbance and primary productivity in determining stream invertebrate diversity. Oikos 111, 392–402.

Dewson, Z., James, A., Death, R., 2007. Invertebrate responses to short-term water abstraction in small New Zealand streams. Freshw. Biol. 52, 357–369. https://doi.org/10.1111/j.1365-2427.2006.01682.x.

Dodds, W., 2007. Trophic state, eutrophication and nutrient criteria in streams. Trends Ecol. Evol. 22, 669–676. https://doi.org/10.1016/j.tree.2007.07.010.

Dodds, W., Smith, V., 2016. Nitrogen, phosphorus, and eutrophication in streams. Inland Waters 6, 155–164. https://doi.org/10.5268/IW-6.2.909.

Doretto, A., Piano, E., Larson, C.E., 2020. The river continuum concept: lessons from the past and perspectives for the future. Can. J. Fish. Aquat.Sci. 77, 1856–1864. https://doi.org/10.1139/CJFAS-2020-0039/ASSET/IMAGES/LARGE/CJFAS-2020-0039F3.JPEG.

Eberhard, R., Robinson, C.J., Waterhouse, J., Parslow, J., Hart, B., Grayson, R., Taylor, B., 2009. Adaptive management for water quality planning from theory to practice. Mar. Freshw. Res. 60, 1189–1195.

Elliott, A.H., Alexander, R.B., Schwarz, G.E., Shankar, U., Sukias, J.P.S., McBride, G.B., 2005. Estimation of nutrient sources and transport for New Zealand using the hybrid mechanistic-statistical model SPARROW. J. Hydrol. 44, 1–27.

Ellis, N., Smith, S.J., Pitcher, C.R., Roland Pitcher, C., Pitcher, C.R., 2012. Gradient forests: calculating importance gradients on physical predictors. Ecology 93, 156–168. https://doi.org/10.1890/11-0252.1.

Elser, J., Fagan, W.F., Denno, R.F., Dobberfuhl, D.R., Folarin, A., Huberty, A., Interlandi, S., Kilham, S.S., McCauley, E., Schulz, K.L., Siemann, E.H., Sterner, R.W., 2000a. Nutritional constraints in terrestrial and freshwater food webs. Nature 408, 578. https://doi.org/10.1038/35046058.

Elser, J.J., Sterner, R.W., Gorokhova, E., Fagan, W.F., Markow, T.A., Cotner, J.B., Harrison, J.F., Hobbie, S.E., Odell, G.M., Weider, L.W., 2000b. Biological stoichiometry from genes to ecosystems. Ecol. Lett. 3, 540–550. https://doi.org/10.1111/j.1461-0248.2000.00185.x.

Essential Freshwater Science and Technical Advisory Group, 2019a. Essential Freshwater Science and Technical Advisory Group report to the Minister for the Environment. Wellington, New Zealand.

Essential Freshwater Science and Technical Advisory Group, 2019b. Essential Freshwater Science and Technical Advisory Group Supplementary Report to the Minister for the Environment. Wellington, New Zealand.

Evans, J.S., Murphy, M.A., 2019. Package 'rfUtilities'.

Evans-White, M.A., Dodds, W.K., Huggins, D.G., Baker, D.S., 2009. Thresholds in macroinvertebrate biodiversity and stoichiometry across water-quality gradients in Central Plains (USA) streams. J. North Am. Benthol. Soc. 28, 855–868. https://doi.org/10.1899/08-113.1.

Evans-White, M.A., Haggard, B.E., Scott, J.T., 2013. A review of stream nutrient criteria development in the United States. J. Environ. Qual. 42, 1002–1014. https://doi.org/10.2134/jeq2012.0491.

Ferreira, V., Castagneyrol, B., Koricheva, J., Gulis, V., Chauvet, E., Graça, M.A.S.S., 2015. A meta-analysis of the effects of nutrient enrichment on litter decomposition in streams. Biol. Rev. 90, 669–688. https://doi.org/10.1111/brv.12125.

Foote, K.J., Joy, M.K., Death, R.G., 2015. New Zealand dairy farming: milking our environment for all it's worth. Environ. Manage. 56, 709–720. https://doi.org/10.1007/s00267-015-0517-x.

Friberg, N., Bonada, N., Bradley, D.C., Dunbar, M.J., Edwards, F.K., Grey, J., Hayes, R.B., Hildrew, A.G., Lamouroux, N., Trimmer, M., 2011. Biomonitoring of human impacts in freshwater ecosystems: the good, the bad and the ugly. Adv. Ecol. Res. 44, 1–68.

Frost, P.C., Ebert, D., Smith, V.H., 2008. Responses of a bacterial pathogen to phosphorus limitation of its aquatic invertebrate host. Ecology 89, 313–318. https://doi.org/10.1890/07-0389.1.

Gillooly, J.F., Allen, A.P., Brown, J.H., Elser, J.J., del Rio, C.M., Savage, V.M., West, G.B., Woodruff, W.H., Woods, H.A., 2005. The metabolic basis of whole-organism RNA and phosphorus content. Proc. Natl. Acad. Sci. USA 102. https://doi.org/10.1073/pnas.0504756102, 11923 LP–11927.

Hastie, T., Tibshirani, R., Friedman, J., 2009. Random forests. The Elements of Statistical Learning. Springer, pp. 587–604.

Herlihy, A.T., Sifneos, J.C., 2008. Developing nutrient criteria and classification schemes for wadeable streams in the conterminous US. J. North Am. Benthol. Soc. 27, 932–948. https://doi.org/10.1899/08-041.1.

Hessen, D.O., Elser, J.J., Sterner, R.W., Urabe, J., 2013. Ecological stoichiometry: an elementary approach using basic principles. Limnol. Oceanogr. 58, 2219–2236. https://doi.org/10.4319/lo.2013.58.6.2219.

Huo, S., Ma, C., Xi, B., Zhang, Y., Wu, F., Liu, H., 2018. Development of methods for establishing nutrient criteria in lakes and reservoirs: a review. J. Environ. Sci. 67, 54–66. https://doi.org/10.1016/j.jes.2017.07.013.

Johnson, P.T.J.J., Townsend, A.R., Cleveland, C.C., Glibert, P.M., Howarth, R.W., McKenzie, V.J., Rejmankova, E., Ward, M.H., 2010. Linking environmental nutrient enrichment and disease emergence in humans and wildlife. Ecol. Appl. 20, 16–29. https://doi.org/10.1890/08-0633.1.

Joy, M.K., Canning, A.D., 2020. Shifting baselines and political expediency in New Zealand's freshwater management. Mar. Freshw. Res. https://doi.org/10.1071/MF20210.

Julian, J.P., de Beurs, K.M., Owsley, B., Davies-Colley, R.J., Ausseil, A.G.E., 2017a. River water quality changes in New Zealand over 26 years: response to land use intensity. Hydrol. Earth Syst. Sci. 21, 1149–1171. https://doi.org/10.5194/hess-21-1149-2017.

Julian, J., de Beurs, K., Owsley, B., Davies-Colley, R., Ausseil, A., 2017b. River water quality changes in New Zealand over 26 years: response to land use intensity. Hydrol. Earth Syst. Sci. 21, 1149–1171. https://doi.org/10.5194/hess-21-1149-2017.

Kath, J., Thomson, J.R., Thompson, R.M., Kefford, B.J., Dyer, F.J., Mac Nally, R., 2018. Interactions among stressors may be weak: implications for management of freshwater macroinvertebrate communities. Divers. Distrib. 24, 939–950. https://doi.org/10.1111/ddi.12737.

Keith, D.A., Martin, T.G., McDonald-Madden, E., Walters, C., 2011. Uncertainty and adaptive management for biodiversity conservation. Biol. Conserv. 144, 1175–1178. https://doi.org/10.1016/j.biocon.2010.11.022.

Kilroy, C., Whitehead, A., Howard, S., Greenwood, M., 2019. Modelling periphyton in New Zealand rivers. 1. An Analysis of Current Data and Development of National Predictions. Christchurch, New Zealand.

King, R.S., Richardson, C.J., 2003. Integrating bioassessment and ecological risk assessment: an approach to developing numerical water-quality criteria. Environ. Manage. 31, 795–809. https://doi.org/10.1007/s00267-002-0036-4.

Kingsford, R.T., Biggs, H.C., Pollard, S.R., 2011. Strategic adaptive management in freshwater protected areas and their rivers. Biol. Conserv. 144, 1194–1203. https://doi.org/10.1016/j.biocon.2010.09.022.

Landres, P.B., 1992. Ecological indicators: panacea or liability. Ecol. Indic. 2.

Larned, S.T., Snelder, T., Unwin, M.J., McBride, G.B., 2016. Water quality in New Zealand rivers: current state and trends. N Z J. Mar. Freshwater Res. 50, 389–417. https://doi.org/10.1080/00288330.2016.1150309.

Le Moal, M., Gascuel-Odoux, C., Ménesguen, A., Souchon, Y., Étrillard, C., Levain, A., Moatar, F., Pannard, A., Souchu, P., Lefebvre, A., Pinay, G., 2019. Eutrophication: a new wine in an old bottle? Sci. Total Environ. 651, 1–11. https://doi.org/10.1016/j.scitotenv.2018.09.139.

Leathwick, J.R., West, D., Gerbeaux, P., Kelly, D., Robertson, H., Brown, D., Chadderton, W., Ausseil, A.G.E., 2010. Freshwater Ecosystems of New Zealand (FENZ) Geodatabase. Department of Conservation, Wellington.

Liaw, A., Wiener, M., 2002. Classification and regression by randomForest. R news 2, 18–22.

Macreadie, P.I., Nielsen, D.A., Kelleway, J.J., Atwood, T.B., Seymour, J.R., Petrou, K., Connolly, R.M., Thomson, A.C.G.G., Trevathan-Tackett, S.M., Ralph, P.J., 2017. Can we manage coastal ecosystems to sequester more blue carbon? Front. Ecol. Environ. 15, 206–213. https://doi.org/10.1002/fee.1484.

Matthaei, C.D., Piggott, J.J., Townsend, C.R., 2010. Multiple stressors in agricultural streams: interactions among sediment addition, nutrient enrichment and water abstraction. J. Appl. Ecol. 47, 639–649. https://doi.org/10.1111/j.1365-2664.2010.01809.x.

McDowell, R.W., Condron, L.M., 2000. Chemical nature and potential mobility of phosphorus in fertilized grassland soils. Nutr. Cycl. Agroecosyst. 57, 225–233. https://doi.org/10.1023/A:1009838424935.

McDowell, R.W., Snelder, T.H., Cox, N., Booker, D.J., Wilcock, R.J., 2013. Establishment of reference or baseline conditions of chemical indicators in New Zealand streams and rivers relative to present conditions. Mar. Freshw. Res. 64, 387–400. https://doi.org/10.1071/MF12153.

McHugh, M.L., 2012a. Interrater reliability: the kappa statistic. Biochemia medica 22, 276–282.

McHugh, M.L., 2012b. Interrater reliability: the kappa statistic. Biochemia medica 22, 276–282.

Ministry for the Environment, Statistics New Zealand, Environment, M. for the, Zealand, S.N., 2019. New Zealand's Environmental Reporting Series: environment Aotearoa 2015, New Zealand's Environmental Reporting Series. Wellington, New Zealand.

Ministry for the Environment, Stats NZ, 2020. New Zealand's Environmental Reporting Series: our freshwater 2020. Wellington, New Zealand.

Moore, S.C., 1998. A Photographic Guide to Freshwater Invertebrates of Taranaki's Rivers and Streams. Taranaki Regional Council.

Mora-Teddy, A.K., Matthaei, C.D., 2020. Microplastic pollution in urban streams across New Zealand: concentrations, composition and implications. N Z J. Mar. Freshwater Res. 54, 233–250. https://doi.org/10.1080/00288330.2019.1703015.

Mouton, T.L., Leprieur, F., Floury, M., Stephenson, F., Verburg, P., Tonkin, J.D., 2022. Climate and land-use driven reorganisation of structure and function in river macroinvertebrate communities. Ecography 2022, e06148. https://doi.org/10.1111/ecog.06148.

Nõges, P., Argillier, C., Borja, Á., Garmendia, J.M., Hanganu, J., Kodeš, V., Pletterbauer, F., Sagouis, A., Birk, S., 2016. Quantified biotic and abiotic responses to multiple stress in freshwater, marine and ground waters. Sci. Total Environ. 540, 43–52. https://doi.org/10.1016/J.SCITOTENV.2015.06.045.

Norris, R.H., Thoms, M.C., 1999. What is river health? Freshw. Biol. 41, 197–209. https://doi.org/10.1046/j.1365-2427.1999.00425.x.

Parmar, T.K., Rawtani, D., Agrawal, Y.K., 2016. Bioindicators: the natural indicator of environmental pollution. Front. Life Sci. 9, 110–118. https://doi.org/10.1080/21553769.2016.1162753.

Poikane, S., Kelly, M.G., Salas Herrero, F., Pitt, J.A., Jarvie, H.P., Claussen, U., Leujak, W., Lyche Solheim, A., Teixeira, H., Phillips, G., 2019. Nutrient criteria for surface waters under the European water framework directive: current state-of-the-art, challenges and future outlook. Sci. Total Environ. 695, 133888 https://doi.org/10.1016/j.scitotenv.2019.133888.

Poikane, S., Salas Herrero, F., Kelly, M.G., Borja, A., Birk, S., van de Bund, W., 2020. European aquatic ecological assessment methods: a critical review of their sensitivity to key pressures. Sci. Total Environ. 740, 140075 https://doi.org/10.1016/J.SCITOTENV.2020.140075.

Quinn, J.M., Stroud, M.J., 2002. Water quality and sediment and nutrient export from New Zealand hill-land catchments of contrasting land use. N Z J. Mar. Freshwater Res. 36, 409–429. https://doi.org/10.1080/00288330.2002.9517097.

R Core Team, 2016. R: a language and environment for statistical computing.

Romano, N., Zeng, C., 2013. Toxic effects of ammonia, nitrite, and nitrate to decapod crustaceans: a review on factors influencing their toxicity, physiological consequences, and coping mechanisms. Rev. Fisheries Sci. 21, 1–21. https://doi.org/10.1080/10641262.2012.753404.

Rooney, N., McCann, K.S., 2012. Integrating food web diversity, structure and stability. Trends Ecol. Evol. 27, 40–46.

Rosenberg, D.M., Resh, V.H., 1993. Freshwater Biomonitoring and Benthic Macroinvertebrates.

Roubeix, V., Danis, P.A., Feret, T., Baudoin, J.M., 2016. Identification of ecological thresholds from variations in phytoplankton communities among lakes: contribution to the definition of environmental standards. Environ. Monit. Assess. 188, 246. https://doi.org/10.1007/s10661-016-5238-y.

Roubeix, V., Daufresne, M., Argillier, C., Dublon, J., Maire, A., Nicolas, D., Raymond, J. C., Danis, P.A., 2017. Physico-chemical thresholds in the distribution of fish species among French lakes. Knowl. Manag. Aquat. Ecosyst.

Saint-Béat, B., Baird, D., Asmus, H., Asmus, R., Bacher, C., Pacella, S.R., Johnson, G.A., David, V., Vézina, A.F., Niquil, N., 2015. Trophic networks: how do theories link ecosystem structure and functioning to stability properties? A review. Ecol. Indic. 52, 458–471. https://doi.org/10.1016/j.ecolind.2014.12.017.

Scanlon, B.R., Jolly, I., Sophocleous, M., Zhang, L., 2007. Global impacts of conversions from natural to agricultural ecosystems on water resources: quantity versus quality. Water Resour. Res. 43 https://doi.org/10.1029/2006wr005486 n/a-n/a.

Schindler, D.E., Eby, L.A., 1997. Stoichiometry of fishes and their prey: implications for nutrient recycling. Ecology 78, 1816–1831. https://doi.org/10.1890/0012-9658(1997)078[1816:SOFATP]2.0.CO;2.

Seitzinger, S.P., Mayorga, E., Bouwman, A.F., Kroeze, C., Beusen, A.H.W., Billen, G., Van Drecht, G., Dumont, E., Fekete, B.M., Garnier, J., Harrison, J.A., 2010. Global river nutrient export: a scenario analysis of past and future trends. Global Biogeochem. Cycles 24. https://doi.org/10.1029/2009GB003587.

Shearer, K.A., Hayes, J., 2019. Assessing aquatic invertebrate prey indices for fish: trout. Prepared For Ministry for the Environment. Nelson, New Zealand.

Snelder, T., Biggs, B.J., Weatherhead, M., 2010. New Zealand river Environment Classification User Guide. Ministry for the Environment.

Snelder, T.H., Fraser, C., Larned, S.T., Monaghan, R., Malmanche, S.De, Whitehead, A.L., Snelder, T.H., Fraser, C., Larned, S.T., Monaghan, R., Malmanche, S.De, Whitehead, A.L., 2021. Attribution of river water-quality trends to agricultural land use and climate variability in New Zealand. Mar. Freshw. Res. 73, 1–19. https://doi.org/10.1071/MF21086.

Snelder, T.H., Moore, C., Kilroy, C., 2019. Nutrient concentration targets to achieve periphyton biomass objectives incorporating uncertainties. JAWRA. https://doi.org/10.1111/1752-1688.12794, 0.

Stark, J.D., Maxted, J.R., 2007. A user guide for the Macroinvertebrate Community Index, Cawthron Report. Nelson, New Zealand.

Stelzer, R.S., Heffernan, J., Likens, G.E., 2003. The influence of dissolved nutrients and particulate organic matter quality on microbial respiration and biomass in a forest stream. Freshw. Biol. 48, 1925–1937. https://doi.org/10.1046/j.1365-2427.2003.01141.x.

Sumudumali, R.G.I., Jayawardana, J.M.C.K., 2021. A review of biological monitoring of aquatic ecosystems approaches: with special reference to macroinvertebrates and pesticide pollution. Environ. Manage. 67, 263–276. https://doi.org/10.1007/s00267-020-01423-0.

Sundermann, A., Leps, M., Leisner, S., Haase, P., 2015. Taxon-specific physico-chemical change points for stream benthic invertebrates. Ecol. Indic. 57, 314–323. https://doi.org/10.1016/j.ecolind.2015.04.043.

Therneau, T., Atkinson, B., 2022. rpart: recursive Partitioning and Regression Trees.

Thompson, R.M., Townsend, C.R., 2000. New Zealand's stream invertebrate communities: an international perspective. In: New Zealand stream invertebrates: Ecology and Implications For Management, 1. New Zealand Limnological Society, Christchurch, New Zealand, pp. 53–74.

Townsend, C.R., Scarsbrook, M.R., Dolédec, S., 1997. Quantifying disturbance in streams: alternative measures of disturbance in relation to macroinvertebrate species traits and species richness. J. North Am. Benthol. Soc. 16, 531–544. https://doi.org/10.2307/1468142.

Vannote, R.L., Minshall, G.W., Cummins, K.W., Sedell, J.R., Cushing, C.E., 2011. The River Continuum Concept. https://doi.org/10.1139/f80-017 37, 130–137.

Vinson, M.R., Baker, M.A., 2008. Poor growth of rainbow trout fed New Zealand mud snails Potamopyrgus antipodarum. N Am. J. Fish Manag. 28, 701–709. https://doi.org/10.1577/M06-039.1.

Vorosmarty, C.J., McIntyre, P.B., Gessner, M.O., Dudgeon, D., Prusevich, A., Green, P., Glidden, S., Bunn, S.E., Sullivan, C.A., Liermann, C.R., Davies, P.M., 2010. Global threats to human water security and river biodiversity. Nature 467, 555–561. http://www.nature.com/nature/journal/v467/n7315/abs/nature09440.html#supplementary-information.

Wagenhoff, A., Clapcott, J.E., Lau, K.E.M.M., Lewis, G.D., Young, R.G., 2017a. Identifying congruence in stream assemblage thresholds in response to nutrient and sediment gradients for limit setting. Ecol. Appl. 27, 469–484. https://doi.org/10.1002/eap.1457.

Wagenhoff, A., Liess, A., Pastor, A., Clapcott, J.E., Goodwin, E.O., Young, R.G., 2017b. Thresholds in ecosystem structural and functional responses to agricultural stressors can inform limit setting in streams. Freshw. Sc. 36, 178–194. https://doi.org/10.1086/690233.

Wagenhoff, A., Townsend, C.R., Matthaei, C.D., 2012. Macroinvertebrate responses along broad stressor gradients of deposited fine sediment and dissolved nutrients: a stream mesocosm experiment. J. Appl. Ecol. 49, 892–902. https://doi.org/10.1111/j.1365-2664.2012.02162.x.

Weeks, E.S., Death, R.G., Foote, K., Anderson-Lederer, R., Joy, M.K., Boyce, P., 2016. Conservation science statement. The demise of New Zealand's freshwater flora and fauna: a forgotten treasure. Pacific Conserv. Biol. 22, 110–115.

Winterbourn, M., Gregson, K., Dolphin, C., 1989. Guide to the Aquatic Insects of New Zealand. Entomological Society of New Zealand Auckland.

Wurtsbaugh, W.A., Paerl, H.W., Dodds, W.K., 2019. Nutrients, eutrophication and harmful algal blooms along the freshwater to marine continuum. Wiley Interdisciplinary Rev. Water 6, 1–27. https://doi.org/10.1002/wat2.1373.

Yuan, L.L., 2010. Estimating the effects of excess nutrients on stream invertebrates from observational data. Ecol. Appl. 20, 110–125. https://doi.org/10.1890/08-1750.1.

Zhao, L., Zhang, H., O'Gorman, E.J., Tian, W., Ma, A., Moore, J.C., Borrett, S.R., Woodward, G., 2016. Weighting and indirect effects identify keystone species in food webs. Ecol. Lett. 19, 1032–1040. https://doi.org/10.1111/ele.12638.