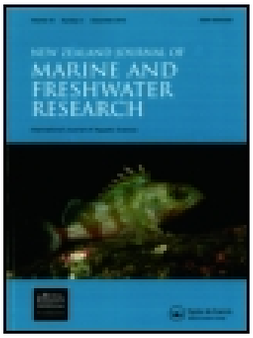

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RESEARCH ARTICLE

Water quality in New Zealand rivers: current state and trends

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ABSTRACT
River water quality, particularly in lowland catchments, is a matter of concern to the New Zealand public. We assessed river water quality and biological state and trends using data from more than 900 monitoring sites. Parallel state and trend analyses were carried out using all sites and a subset of lowland river sites. Median water-quality state in urban and pastoral land-cover classes was poorer than in exotic forest and natural land-cover classes, and lowland sites in the urban and pastoral classes had the poorest water quality. Nutrient and *Escherichia coli* concentrations increased and visual clarity and Macroinvertebrate Community Index scores decreased as proportions of catchments in high-intensity agricultural and urban land cover increased. Ten-year trends (2004–2013) indicated recent improvements in ammoniacal nitrogen, dissolved reactive phosphorus and total phosphorus in the pastoral and urban classes, possibly reflecting improved land management. In contrast, trends in nitrate-nitrogen in the exotic forest and cool-dry/pastoral classes indicated worsening conditions.

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Escherichia coli; land cover; lowland; macroinvertebrates; New Zealand nitrogen; phosphorus; rivers; trends; visual clarity; water quality

Introduction

River water quality is an issue of long-standing concern in New Zealand. In repeated public-perception studies, few respondents have considered the state of New Zealand's rivers to be 'good' or 'very good'; the majority consider river state to range from 'adequate' to 'very bad' (Cullen et al. 2006; Hughey et al. 2013). The same surveys indicate widespread dissatisfaction with river management. The high level of public concern is based in part on New Zealand's reliance on rivers for economic, social and cultural well-being, and on the government mandate to protect and improve river health and water supplies (Tipa 2009; Fisher & Russell 2011; Hughey & Booth 2012). Preventing and reversing degradation in river water quality are stated aims of the freshwater reforms recently enacted by the New Zealand government (New Zealand Government 2014).

Results from previous regional and national monitoring programmes indicated that water quality and biological conditions were poor in some New Zealand rivers, particularly those in urban and agricultural catchments (e.g. Collier et al. 2009; Ballantine & Davies-Colley 2014). Negative relationships between water quality and the prevalence of high-intensity agricultural and urban land cover have been identified at catchment, regional and national scales (Larned et al. 2004; McDowell et al. 2009, 2013; Roygard et al. 2012;

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Wilcock et al. 2013a). These observations suggest that the risk of water-quality degradation is greatest in low-elevation catchments where high-intensity agriculture and urban development are the dominant land uses (Cullen et al. 2006; Monaghan et al. 2007; Collier & Clements 2011).

Rivers in New Zealand's low-elevation catchments are generally groundwater dominated, i.e. more flow is derived from underlying aquifers than from surface runoff and interflow (White 2009; Guggenmos et al. 2011; Larned et al. 2011; Baalousha 2012). Contaminant concentrations in some of these rivers are high due to contaminated groundwater input, and their contaminant loads may increase downstream due to flow accretion (Larned et al. 2004, 2015). In addition to high contaminant concentrations, many lowland rivers have algal and macrophyte proliferations, large diurnal pH fluctuations and periodic hypoxia (Wilcock et al. 2007; Wilding et al. 2012). These conditions pose multiple ecological risks, including physiological stress in fish and loss of pollution-sensitive invertebrates (Collier et al. 1998; Franklin 2014). Water-quality degradation in lowland rivers also has negative effects on downstream-receiving waters (Schallenberg et al. 2010; Drake et al. 2011).

In this study, we assessed the current state and temporal trends in New Zealand rivers in terms of physical-chemical water quality and benthic invertebrates. The state analyses are based on data for the 2009–2013 period and the trend analyses on data for the 2004–2013 period. The first part of the study used data from rivers distributed across New Zealand. The second part used a subset of the data from sites on lowland rivers. The only previous national-scale assessment of lowland river water quality was carried out in 2003, using data for the 1996–2002 period (Larned et al. 2004). During the 11 year interval between the 2004 study and the current study, agricultural intensification and urbanisation continued: national dairy cow numbers increased by 26%, annual urea fertiliser use increased by 63%, and the human population in urban centres (populations >30,000) increased by 13% (Statistics New Zealand 2015). Other land-use indicators declined over the same period: sheep numbers decreased by 22% and annual phosphorus fertiliser use decreased by 37% (Millner et al. 2013; Statistics New Zealand 2015; Fertiliser Association of New Zealand, unpubl. data). During the same period, agricultural industry groups introduced voluntary nutrient-management and stock-exclusion schemes, with the intention of improving water quality or preventing further degradation (Wilcock et al. 2013a; Swaffield 2014). In light of the recent changes in land use and continuing public concern about conditions in New Zealand's rivers, an assessment of river water-quality state and trends is timely.

Methods

Data acquisition

We assessed river water quality and biological state and trends using eight variables: visual clarity, ammoniacal nitrogen (NH_4-N); nitrate nitrogen (NO_3-N); total nitrogen (TN); dissolved reactive phosphorus (DRP); total phosphorus (TP); the faecal indicator bacterium *Escherichia coli*; and the Macroinvertebrate Community Index (MCI) (Table 1). MCI scores provide an indication of ecosystem health based on the presence of macroinvertebrate taxa and their tolerance to nutrient enrichment and organic pollution (Stark & Maxted 2007). Separate versions of the MCI are available for hard-bottom and soft-

Table 1. River water-quality variables and the measurement methods most frequently used in New Zealand river monitoring programmes.

Variable (abbreviation)	Comparable methods	Non-comparable methods	Reference
Water clarity*	Black-disk*	Horizontal clarity tube	Kilroy & Biggs 2002
Ammoniacal-nitrogen (NH_4-N)*	Filtered samples, phenol/ hypochlorite colorimetry	Colorimetric test kit	Zhou & Boyd 2015
Nitrate-nitrogen (NO_3-N)	Nitrate-N in filtered samples, ion chromatography Or nitrate-N + nitrite-N ("NNN") in filtered samples, cadmium reduction Or nitrate + nitrite-N − nitrite-N in filtered samples, azo dye colorimetry Or optical sensor	Colorimetric test kits	Ormaza-González & Villalba-Flor 1994
Total nitrogen (TN)*	Unfiltered samples, persulphate digestion to nitrate, nitrate-N by cadmium reduction	Filtered samples Kjeldahl digestion	Patton & Kryskalla 2003; Horowitz 2013
Dissolved reactive phosphorus (DRP)*	Filtered samples, molybdenum blue colorimetry	Colorimetric test kit	Ormaza-González and Villalba-Flor 1994
Total phosphorus (TP)*	Unfiltered samples, persulphate digestion to orthophosphate, molybdenum blue colorimetry	Filtered samples	No published comparisons of filtered and unfiltered samples
Escherichia coli (*E. coli*)	Defined substrate (e.g. Colilert QuantiTray) or membrane filtration	Molecular methods (e.g. immunofluorescence)	Rompré et al. 2002; Hamilton et al. 2005
Macroinvertebrate community index (MCI)	Collection procedures C1, C2, C3, or C4 Processing procedures P1, P2, or P3 MCI hard bottom calculation procedure	MCI soft bottom calculation procedure	Stark et al. 2001; Stark and Maxted 2007.

Comparability among methods for each variable was based on published inter-method comparisons.
References are for reports with inter-method comparisons.
*Variables for which there was a single acceptable method.

bottom channels; scores from the two versions cannot be aggregated into single data sets due to differences in taxa and tolerance values. However, more than 95% of the MCI data that we acquired consisted of hard-bottom scores, and we used those scores in all analyses to ensure comparability.

Water quality and MCI data for the period January 2004 to December 2013 were acquired from each of New Zealand's 16 regional councils and unitary authorities and the National Institute of Water and Atmospheric Research (NIWA). Each council operates a river monitoring network and NIWA operates the National River Water Quality Network (NRWQN). These networks generate monthly or quarterly water-quality data and annual invertebrate data (used to calculate MCI scores) for sites distributed across New Zealand. The most recent data for council monitoring sites were downloaded from the Land and Water Aotearoa database (www.lawa.org.nz). The aggregated data set represented 7956 MCI scores at 1212 sites, and 70,188 nutrient, visual clarity and *E. coli* values at 785 sites.

Data processing

The individual regional council and NIWA data sets varied in structure, reporting conventions, measurement units and laboratory detection and reporting limits. Each data set

included erroneous and censored data (data entries that represent unknown measurement values). For most water-quality variables, two or more measurement methods were reported. We used a multi-step process to produce a corrected and internally consistent data set. First, we compiled the raw data in a Microsoft Access database, and applied a consistent set of reporting conventions and measurement units. We then inspected the data using time-series plots and other diagnostics to identify and correct obvious errors. We identified the field or laboratory methods used to produce each data point and pooled the data for which different methods gave comparable results (Table 1). Comparability of methods was based on published reports of the comparative accuracy, precision and bias of two or more methods. For each variable, the data corresponding to the most widely used and comparable procedures were retained and the remaining data omitted.

The next data-processing steps concerned censored values in the raw data sets. Most censored values were data entries indicating that the true values were too low (i.e. below the analytical detection limit) or too high (i.e. above the reporting limit) for precise measurement. Water-quality data sets from New Zealand rivers often include DRP, TP and NH_4-N measurements below detection limits, and *E. coli* and visual clarity measurements above reporting limits. In the raw data sets, censored data were generally flagged as such or replaced with constants such as 0.5 × detection limit and 1.1 × reporting limit. We removed all identifiable censored and substituted values, then used two statistical procedures to impute replacement values. These procedures produce a replacement value for each censored data point that is consistent with the distribution of the non-censored values. Imputation methods were used to replace censored data in lieu of other methods that can generate tied data and adversely affect distributions and trend estimates (Helsel 2006). For below-detection-limit data, we used regression on order statistics (ROS) to impute replacement values (Helsel 2012). Briefly, the ROS method develops probability plotting positions for each data point (censored and non-censored) based on the ordering of the non-censored data. A relationship between the observations and the non-censored probability plotting positions is fitted by least squares regression, and this relationship is used to predict the concentrations for the censored values based on their plotting positions. For above-reporting-limit data, we used a survival analysis procedure (Helsel 2012). In this procedure, a parametric distribution is fitted to the non-censored values using maximum likelihood methods. The values for the censored values are then estimated by randomly sampling larger values from this distribution. Both the ROS and survival analysis procedures are applicable to data sets with multiple detection and reporting limits, which were common in our aggregated data set.

For each river monitoring site in the aggregated data set, the NZMS260 grid reference (converted from other coordinate systems as necessary) was used to position the site on a river segment (i.e. a length of river bounded by upstream and downstream confluences). Additional site information (e.g. stream and road names, distances from grid references to adjacent reaches) was used to check the accuracy of reach assignments and correct georeferencing errors.

Spatial data for the catchment upstream of every river segment in New Zealand are stored in the River Environment Classification (REC) geodatabase (Snelder et al. 2010). We extracted the following spatial data for the segment corresponding to each site in our database: land-cover data from the Land Cover Database-3 (LCDB3), catchment area, mean catchment slope and elevation, and REC class, as defined by Snelder &

Biggs (2002). LCDB3 data for each site consisted of proportions of the upstream catchment in 33 land-cover classes, based on satellite imagery from 2008 (lris.scinfo.org.nz). Sites were classified using the climate and land-cover categories of the REC, with land-cover categories modified following Larned et al. (2004): pastoral (P), exotic forest (EF), urban (U) and a natural (N) category composed of the aggregated indigenous forest, tussock, scrub and bare-land categories. A river segment is classified as exotic forest or natural if those categories account for the largest proportion of the upstream catchment area, unless pastoral land exceeds 25% of the catchment, in which case the segment is classified as pastoral, or urban land exceeds 15% of the catchment, in which case the segment is classified as urban. The six REC climate classes are based on mean annual air temperature and precipitation: warm extremely wet (WX), warm wet (WW), warm dry (WD), cool extremely wet (CX), cool wet (CW) and cool dry (CD).

We defined lowland monitoring sites as those for which the mean elevation of the upstream catchment was below 350 m above sea level (ASL) and the mean catchment slope was less than 15°. These criteria differ from those used in the REC to classify 'low-elevation' river segments (>50% of the annual rainfall in the upstream catchment occurs below 400 m ASL; Snelder & Biggs 2002) and from the criterion used in the Australian and New Zealand Guidelines for Fresh and Marine Water Quality to classify 'lowland' river monitoring sites (below 150 m ASL; ANZECC 2000). Our intention was to identify those monitoring sites on low-elevation coastal plains that were likely to be groundwater dominated, and separate them from the low-elevation downstream sections of runoff-dominated rivers that originate at high elevations. Most sites that met our elevation and slope criteria are located on alluvial plains within 25 km of the coast, and most coastal alluvial plains in New Zealand are underlain by porous gravel aquifers (Moreau & Bekele 2015). High-intensity agricultural and urban land cover currently accounts for 60% of the land below 350 m ASL (based on LCDB3). Parallel water-quality state and trend analyses were carried out using the entire set of monitoring sites and the subset of lowland sites.

Data analysis: water-quality state

We used data for the 5 year period 2009–2013 for analyses of water-quality state, as 5 years represented a reasonable trade-off between site numbers, precision of summary statistics and resistance to the effects of temporal trends on those statistics. For nutrients, visual clarity and *E. coli*, we used three rules to identify monitoring sites with sufficient durations and frequencies of observations for inclusion in state analyses: (1) less than 50% of the values for a variable were censored, to ensure that medians were based on uncensored values; (2) there were 30 or more values for a variable, including imputed values; (3) values were available for at least 4 of the 5 years. MCI scores were based on annual samples and there were no censored values; the sole inclusion rule was that data were available for at least 4 of the 5 years. Site by variable combinations that did not meet these criteria were excluded prior to analysis.

Site medians were calculated from individual sampling dates at each site, and sites were used as replicates within REC classes. For each variable, the state in each REC class was characterised using percentiles of the distributions of site medians. Categorical comparisons of water quality and MCI scores were based primarily on REC land-cover classes.

Each of the four land-cover classes encompasses a wide range of climatic, topographic and geological conditions, which increases within-class variation. To reduce some of that variation, we subdivided the two largest land-cover classes, pastoral and natural, into REC climate classes. These subdivisions were limited to climate by land-cover combinations for which there were at least 10 site medians. There were too few lowland sites to subdivide land-cover classes into climate classes and the assessment of water-quality state in lowland rivers was based on land-cover classes alone.

For each variable, REC-class medians were compared with guideline values and numeric objectives for New Zealand rivers (Table 2). Guidelines and objectives came from three sources: (1) trigger values for physical and chemical stressors in New Zealand rivers in the ANZECC guidelines (ANZECC 2000, Table 3.3.10); (2) the national 'bottom-line' median NO_3-N and pH-adjusted NH_4-N concentrations for toxicity, the bottom-line median *E. coli* concentration for secondary contact, and the *E. coli* concentration corresponding to the minimum acceptable state for primary contact (95th percentile) from the New Zealand National Policy Statement for Freshwater Management 2014 (NPS-FM; New Zealand Government 2014); and (3) MCI quality classes from the MCI user's guide (Stark & Maxted 2007). The NPS-FM bottom-line for NH_4-N toxicity is 1300 mg m^{-3} at pH 8. In order to compare NH_4-N concentrations at monitoring sites to the NPS-FM bottom-line, we adjusted the measured NH_4-N concentrations to pH 8 using the conversion ratios in the Draft Guide to Attributes in the NPS-FM (http://www.mfe.govt.nz/publications/fresh-water/draft-guide-attributes-appendix-2-national-policy-statement-freshwater). The adjustments were limited to monitoring site–date combinations for which both pH and NH_4-N were measured.

Table 2. Water-quality and biotic guidelines and numeric objectives for protection of ecological values and human health in New Zealand rivers.

Variable (unit)	Guideline value or numeric objective
Water clarity[a] (m)	Upland: 0.8 Lowland: 0.6
NH_4-N[a] (mg m^{-3})	Upland: 10 Lowland: 21
NH_4-N toxicity[b] (mg m^{-3})	Annual median: 1300
NO_3-N[a] (mg m^{-3})	Upland: 167 Lowland: 444
NO_3-N toxicity[b] (mg m^{-3})	Annual median: 6900
TN[a] (mg m^{-3})	Upland: 295 Lowland: 614
DRP[a] (mg m^{-3})	Upland: 9 Lowland: 10
TP[a] (mg m^{-3})	Upland: 26 Lowland: 33
E. coli secondary contact[b] (cfu 100 mL^{-1})	Annual median: 1000
E. coli primary contact[b] (cfu 100 mL^{-1})	95th percentile: 540
Macroinvertebrate Community Index[c]	119 (Excellent) 100–119 (Good) 80–99 (Fair) <80 (Poor)

See text for summaries of the guidelines and objectives, and criteria for upland and lowland sites.
[a]ANZECC and ARMCANZ (2000).
[b]New Zealand Government (2014).
[c]Stark and Maxted (2007).

Table 3. Percentages of monitoring sites at which median values for water-quality variables exceed guideline and numeric objective values across all sites in the state data set, and within the subsets of lowland and upland sites.

			All sites		
NPS-FM objective	Natural	Exotic forest	Pastoral	Urban	All classes
NH_4-N toxicity	0% (0, 53)	0% (0, 9)	0% (0, 283)	0% (0, 20)	0% (0, 365)
NO_3-N toxicity	0% (0, 138)	0% (0, 18)	1% (4, 408)	0% (0, 23)	0.7% (4, 587)
E. coli secondary contact	0% (0, 126)	0% (0, 13)	2% (8, 327)	20% (4, 20)	2% (12, 486)
E. coli primary contact	29% (37, 126)	46% (6, 13)	91% (296, 327)	100% (20, 20)	74% (359, 486)
NPS-FM objective	Lowland sites				
NH_4-N toxicity	0% (0, 5)	0% (0, 5)	0% (0, 182)	0% (0, 20)	0% (0, 215)
NO_3-N toxicity	0% (0, 13)	0% (0, 7)	2% (4, 238)	0% (0, 23)	1% (4, 281)
E. coli secondary contact	0% (0, 13)	0% (0, 6)	3% (6, 184)	20% (4, 20)	4% (10, 223)
E. coli primary contact	77% (10, 13)	83% (5, 6)	99% (182, 184)	100% (20, 20)	97% (217, 223)
ANZECC trigger value	Upland sites				
Clarity	4% (4, 97)	100% (8, 8)	12% (13, 112)	No sites	8% (25, 309)
NH_4-N	5% (2, 43)	25% (1, 4)	43% (40, 92)	No sites	31% (43, 139)
NO_3-N	14% (15, 111)	13% (1. 8)	65% (99, 153)	No sites	42% (115, 272)
TN	10% (7, 71)	100% (3, 3)	65% (64, 98)	No sites	43% (74, 172)
DRP	28% (23, 81)	88% (7, 8)	57% (79, 138)	No sites	48% (109, 227)
TP	11% (11, 104)	75% (6, 8)	47% (72, 152)	No sites	34% (89, 264)
ANZECC trigger value	Lowland sites				
Clarity	20% (2, 10)	0% (0, 5)	20% (36, 183)	30% (3, 10)	20% (41, 208)
NH_4-N	25% (2, 8)	20% (1, 5)	36% (66, 182)	60% (12, 20)	38% (81, 215)
NO_3-N	0% (0, 13)	14% (1, 7)	56% (133, 238)	96% (22, 23)	54% (152, 281)
TN	0% (0, 6)	17% (1, 6)	65% (86, 132)	88% (15, 17)	63% (102, 161)
DRP	33% (3, 9)	50% (3, 6)	73% (164, 229)	87% (20, 23)	72% (191, 267)
TP	23% (3, 13)	43% (3, 7)	58% (137, 237)	65% (15, 23)	56% (157, 280)

Paired values in parentheses: number of sites exceeding and total number of sites. See text for summaries of the guidelines and objectives, and criteria for upland and lowland sites.

As noted above, the definition of lowland sites used here (average catchment elevation <350 m ASL, average catchment slope <15°) differs from the ANZECC definition (site elevation <150 m ASL). However, there was substantial overlap between sites: 95% of the sites that met our criteria were also located below 150 m ASL. Therefore, we compared water-quality state at the sites that met our lowland criteria with the ANZECC trigger values for lowland sites, and higher elevation sites with the ANZECC trigger values for upland sites.

We used multiple linear regression to relate water-quality variables to proportions of urban and high-intensity agricultural land cover in the catchment. Multiple regression was used in lieu of stepwise regression because there was no a priori reason to predict that urban and agricultural land cover differ in explanatory power, and because the primary aim was to relate water quality to land cover, not to predict water quality at a given site. The proportion of urban land cover was defined as the sum of proportional land cover in four LCDB3 classes (built-up areas, urban parks, mines and dumps, transport infrastructure). The proportion of high-intensity agricultural land cover was defined as the sum of proportional land cover in three LCDB3 classes (high-producing exotic grassland, short-rotation crops, orchards and vineyards). Proportions of urban and high-intensity agricultural land cover upstream of monitoring sites were weakly and negatively correlated (linear correlation coefficients: −0.13 for the 928 sites used for state analyses, −0.42 for the 447 lowland sites). All variable values were log-transformed to improve the normality of residuals. We assessed the effects of collinearity between the land-cover classes on each regression using the variance inflation factor (VIF) (Zuur et al. 2010).

Data analysis: water-quality trends

For trend analyses, we used the 10 year period from 2004 to 2013 as it represented a reasonable trade-off between site numbers and monitoring duration. We applied two inclusion rules to each site by variable combination: (1) data were available for 90% of the sampling dates and 9 out of the 10 years; (2) imputed values comprised <15% of the observations. For nutrients, visual clarity and *E. coli*, the first rule applied to monthly or quarterly samples. For MCI scores, the first rule applied to annual samples.

Trend analyses were carried out in three steps: flow adjustment, estimation of trend magnitude and direction, and confidence interval calculation. Nutrient, visual clarity and *E. coli* data were flow-adjusted to remove variation associated with fluctuating river flow. MCI scores were not flow-adjusted because invertebrate sampling is generally carried out during low-flow periods that do not represent the range of naturally occurring flows. Detailed procedures for flow adjustments are given below.

The Sen slope estimator (SSE) and its variant, the seasonal Sen slope estimator (SSSE) were used to estimate trend magnitude and direction for each site by variable combination that met the inclusion rules (Hirsch et al. 1982). The SSSE was used to estimate flow-adjusted trends in the water-quality variables measured at monthly or quarterly intervals and the SSE to estimate trends in annual MCI scores. SSSE and SSE calculations were made with a modified version of the zyp package in R (www.r-project.org). Each SSSE and SSE was standardised by dividing by the corresponding median value to give a 'relativised' trend (denoted RSSSE or RSSE) with units of % yr^{-1}.

Confidence intervals around each trend were used to infer trend direction, based on the outcome of 'two one-sided tests' (TOSTs) of the dual hypotheses that a trend is positive or it is negative; the TOST procedure requires two one-sided confidence intervals for each trend (Dixon & Pechmann 2005; McBride et al. 2014). Symmetric 90% confidence intervals for the SSSEs and SSEs were calculated using the method of Sen (1968). In the TOST approach, two symmetric, one-sided 90% confidence intervals are used to achieve 95% confidence (McBride et al. 2014). For each trend, if the confidence intervals did not contain zero, we considered the trend direction to be established with confidence. If the confidence intervals contained zero, we considered there to be insufficient data to determine trend direction. In a departure from previous trend analyses, we did not use statistical significance tests to test the null hypothesis of no trend for two reasons: (1) conclusions about the statistical significance of trends are influenced by sample size as well as trend magnitude; and (2) as observed by McBride et al. (2014), the failure to reject the null hypothesis is frequently and incorrectly treated as evidence for no trend (e.g. that water quality is 'stable').

Flow adjustments were based on measured or modelled daily average flows on sampling dates. For monitoring sites with flow recorders on the same segment, daily average flows were based on flow measurements. However, most river monitoring sites lack flow recorders and daily average flows for these sites were estimated by hydrological modelling. We used predicted flows from the TopNet hydrological model, corrected using flow-duration curves, which were in turn estimated with random forest models (Booker & Snelder 2012; Booker & Woods 2014). TopNet is a spatially distributed time-stepping model that combines water-balance models with a kinematic wave channel-routing algorithm (McMillan et al. 2013). Flow adjustments were made using second-order generalised additive models

(GAMs). For each variable and site, a GAM was fitted to the log(variable) versus log(daily average flow) relationship. Flow-adjusted values were calculated by subtracting the modelled value from each raw value, and adding the mean (Smith et al. 1996). Flow adjustments were made for all monitoring sites irrespective of the strengths of the water quality–flow relationships.

Results

Water-quality state: all sites

After applying the inclusion rules described above, 928 river monitoring sites were retained for state analyses of one or more variables (Figure 1). For each of the seven water-quality variables measured at monthly or quarterly intervals, there were 30–124 sampling dates per site in the 2009–2013 period; for MCI scores, there were four to five annual sampling dates per site.

Figure 1. Locations of 447 lowland and 481 upland river monitoring sites in the water-quality state data set. Trend analyses were based on data from 376 of the lowland sites and 180 of the upland sites.

Several patterns were apparent in river water-quality conditions between and within the four REC land-cover classes (Figure 2). Median concentrations of NO_3-N, NH_4-N, TN, TP and *E. coli* increased and median MCI scores decreased across land-cover classes in the following order: natural, exotic forest, pastoral, urban. Median visual clarity was highest in the natural class (2.7 m) and uniformly low (1.4–1.7 m) in the other land-cover classes. Within the pastoral land-cover class, sites in warm-climate areas tended to have higher median nutrient and *E. coli* concentrations than in cool-climate areas. Within the natural land-cover class, median nutrient concentrations were consistently low across climate classes, but median *E. coli* concentrations in the warm-wet climate class were elevated compared with the other climate classes. The limited distribution of urban and exotic forest monitoring sites across climate classes precluded assessments of water-quality variation within climate classes.

Median concentrations of all nutrients in the urban land-cover class and median DRP concentrations in the pastoral and exotic forest classes exceeded the ANZECC trigger values for both upland and lowland sites (Figure 2; Table 3). Median MCI scores corresponded to 'poor' water quality for the urban class; 'good' water quality for the pastoral and exotic forest classes; and 'excellent' water quality for the natural class (Figure 2; Table 2). The median value for 95th percentile *E. coli* concentrations across all land-cover classes was 1861 cfu 100 mL^{-1}, more than three times higher than the 95th percentile concentration corresponding to the NPS-FM 'minimum acceptable state' for primary contact. The NPS-FM minimum acceptable state for primary contact was exceeded at all urban sites, 91% of pastoral sites, 46% of exotic forest sites and 29% of natural sites (Table 3). In contrast, only 2% of monitoring sites had median *E. coli* concentrations that exceeded the NPS-FM bottom-line for secondary contact (Table 3). Median NO_3-N concentrations exceeded the NPS-FM bottom-line for toxicity at less than 1% of sites. Median pH-adjusted NH_4-N concentrations did not exceed the NPS-FM bottom-line for toxicity at any of the 715 sites with synoptic NH_4-N and pH data (Table 3).

Mean concentrations of all nutrients and *E. coli* were likely to be higher, and mean visual clarity and MCI scores lower in the urban and pastoral classes compared with the natural class, based on non-overlapping 95% confidence intervals (Table 4). Mean NO_3-N, TN, DRP and TP concentrations were likely to be higher and visual clarity lower in the exotic forest class compared with the natural class. Mean NO_3-N and TN concentrations were likely to be higher and MCI scores lower in the urban and pastoral classes than in the exotic forest class. The 95% confidence intervals for mean concentrations of all nutrients and visual clarity in the urban and pastoral classes overlapped. However, mean *E. coli* concentrations were likely to be higher and MCI scores lower in the urban class compared with the pastoral class.

Regression results indicated that the concentrations of each nutrient and *E. coli* increased, and MCI scores and visual clarity decreased, with increasing proportions of high-intensity agricultural and urban land cover in the upstream catchment (Table 5). Both agricultural and urban land cover were retained in the regression model for each variable, but agricultural land cover explained more variability than urban land cover in all cases, as indicated by partial R^2 values (Table 5). Proportions of high-intensity agricultural and urban land cover jointly explained 20%–65% of the variation in log-transformed water-quality variables. The regression models were minimally affected by multi-collinearity, as indicated by VIF <2 for each model. The Y-intercepts of the regression models can

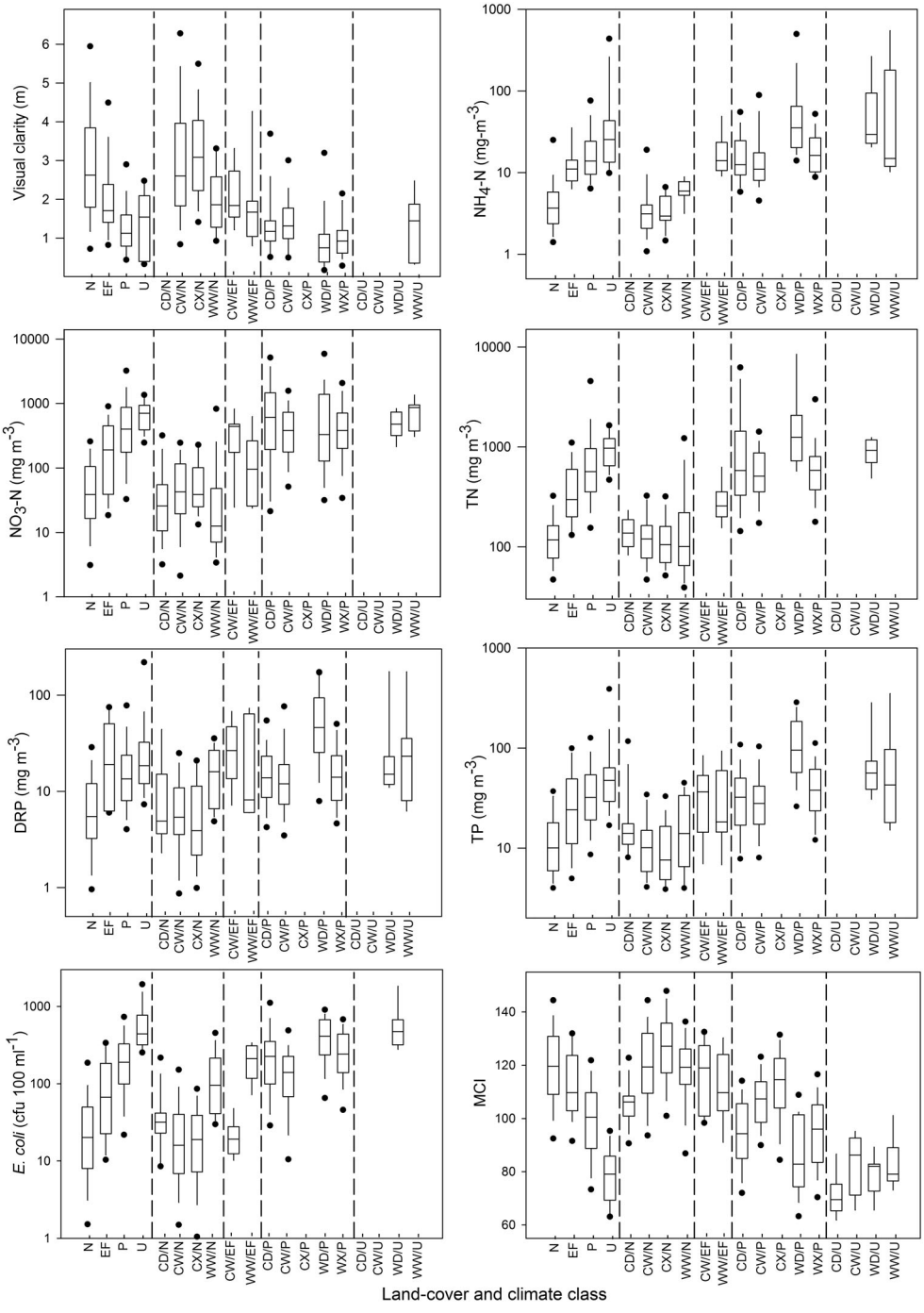

Figure 2. All sites. Distributions of median values of water-quality variables in the state data set, grouped by REC classes. Each panel is subdivided into compartments by dashed lines. Far left compartment: sites grouped by REC natural (N), exotic forest (EF), pastoral (P) and urban (U) land-cover class. Remaining compartments: sites grouped by climate classes within each land-cover class. See text for land-cover and climate-class definitions. Nutrient and *E. coli* concentrations are on log-scales and visual clarity and MCI scores are on linear scales. Percentiles: boxes = 25% and 75%; horizontal bars = medians; whiskers = 10% and 90%; closed circles = 5% and 95% (for classes with >10 sites). Classes with fewer than five sites not shown.

Table 4. All sites. Means and normal 95% confidence intervals (CI) for water-quality variables at sites in the state data set, subdivided into land-cover classes.

	Land-cover class							
	Natural		Exotic forest		Pastoral		Urban	
Variable (unit)	Mean	CI	Mean	CI	Mean	CI	Mean	CI
Clarity (m)	3.0	0.3	2.0	0.5	1.3	0.1	1.5	0.5
NH_4-N (mg m^{-3})	6.4	2.9	14.5	8.7	29.5	8.4	82.8	62.8
NO_3-N (mg m^{-3})	83.0	22.4	284.9	130.1	826.9	138.7	704.9	149.9
TN (mg m^{-3})	147.8	30.7	403.3	186.8	1063.0	226.7	1002.3	175.8
DRP (mg m^{-3})	9.2	1.8	30.1	13.8	22.5	3.0	37.5	25.5
TP (mg m^{-3})	14.3	2.4	36.4	14.2	45.7	4.4	76.3	42.1
E. coli (cfu 100 mL^{-1})	44.1	11.8	114.2	64.2	258.4	28.3	671.5	236.5
MCI	119.7	2.4	112.7	5.7	99.1	1.7	78.8	3.8

Means for each land-cover class are based on site medians.

Table 5. All sites. Multiple linear regression models.

Variable (N)	Regression equation	VIF	Partial and total R^2	*P* value
Clarity (454)	$\log_{10}$CLAR = 0.381 - (0.005 × Agri) - (0.005 × Urban)	1.00	Agri: 0.260 Urban: 0.009 Total: 0.269	<0.001
NH_4-N (365)	$\text{Log}_{10}NH_4$-N = 0.646 + (0.008 × Agri) + (0.011 × Urban)	1.051	Agri: 0.224 Urban: 0.129 Total: 0.353	<0.001
NO_3-N (587)	$\text{Log}_{10}NO_3$-N = 1.580 + (0.015 × Agri) + (0.014 × Urban)	1.014	Agri: 0.425 Urban: 0.0486 Total: 0.474	<0.001
TN (354)	Log_{10}TN = 2.040 + (0.012 × Agri) + (0.011 × Urban)	1.028	Agri: 0.548 Urban: 0.123 Total: 0.651	<0.001
DRP (519)	Log_{10}DRP = 0.777 + (0.006 × Agri) + (0.007 × Urban)	1.026	Agri: 0.166 Urban: 0.035 Total: 0.201	<0.001
TP (577)	Log_{10}TP = 1.026 + (0.008 × Agri) + (0.008 × Urban)	1.016	Agric: 0.326 Urban: 0.060 Total: 0.386	<0.001
E. coli (486)	Log_{10}*E. coli* = 1.341 + (0.013 × Agri) + (0.018 × Urban)	1.018	Agri: 0.353 Urban: 0.108 Total: 0.461	<0.001
MCI (511)	Log_{10}MCI = 2.079 - (0.001 × Agri) - (0.003 × Urban)	1.014	Agri: 0.257 Urban: 0.225 Total: 0.482	<0.001

N, number of monitoring sites; VIF, variance inflation factor; Agri, high-intensity agricultural land cover; Urban, urban land cover.

be used to estimate water-quality baseline values for each variable (i.e. variable values in catchments with natural or exotic forest land cover alone or in combination). For each water-quality variable, the estimated intercept and the median for all sites in the natural land-cover class differed by <10%.

Proportions of high-intensity agricultural land cover alone explained 17%–55% of the variation in water-quality variables, as indicated by simple linear regressions (Figure 3); these relationships were strongest for median TN, NO_3-N, TP and *E. coli* concentrations and MCI scores. Urban land cover alone explained 1%–23% of the variation in water quality (data not shown).

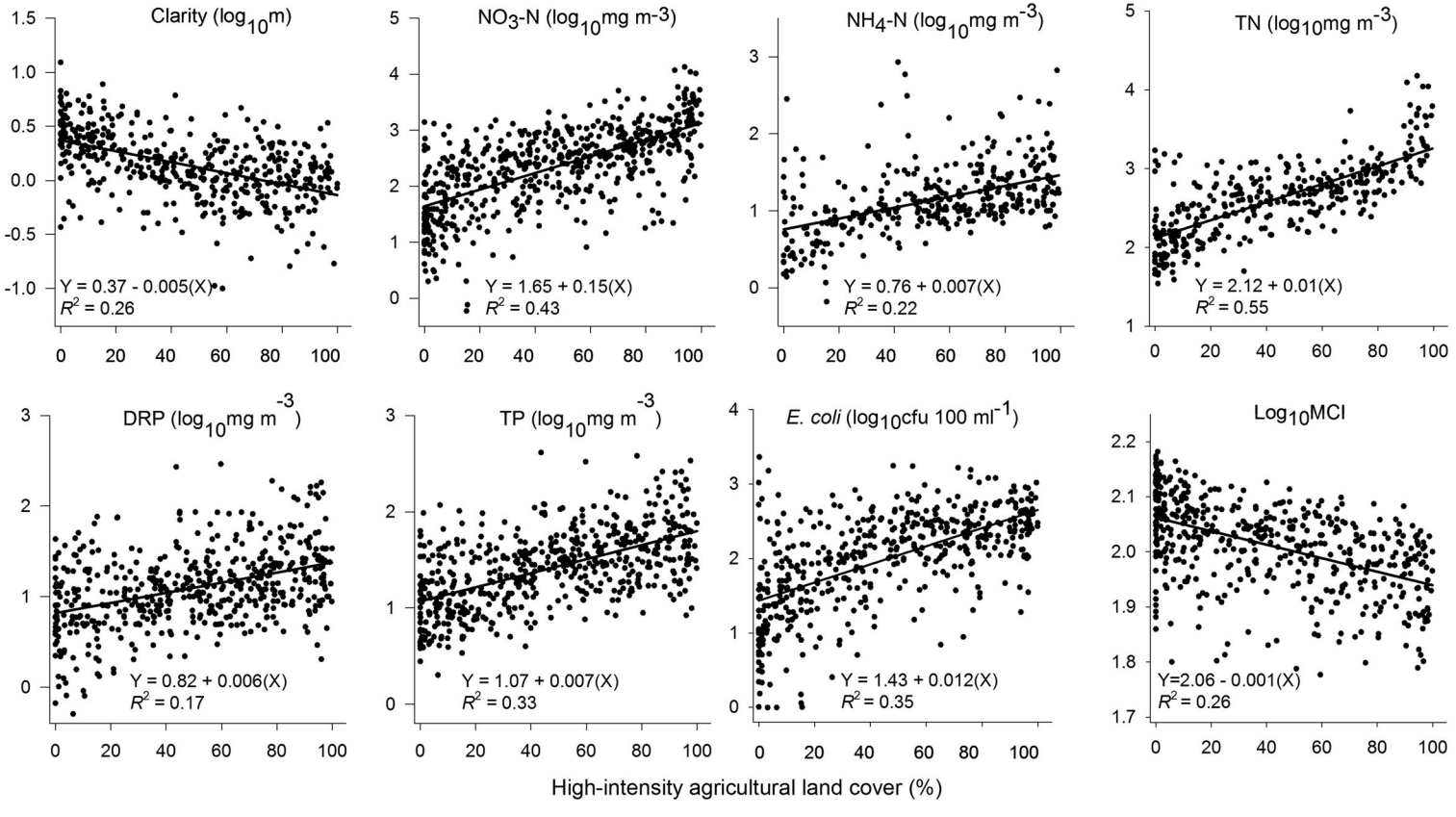

Figure 3. All sites. Relationships between median water-quality state and high-intensity agricultural land cover in the catchments above monitoring sites in the state data set. Solid lines indicate least squares linear regression models.

Water-quality state: lowland sites

Of the 928 river monitoring sites used in the preceding analysis of water-quality state, 447 were classified as lowland sites. Catchment land cover corresponding to the lowland sites was predominately pastoral: 84% of the sites were in the REC pastoral land-cover class, 9% in the urban class, 5% in the natural class and 3% in the exotic forest class. Lowland sites were unevenly distributed among REC climate classes: 85% were in the WW, CW and CD classes, and the remainder were in the WD, WX and CX classes. The number of lowland sites available for state analysis of each variable ranged from 161 for TN to 281 for NO_3-N.

Lowland site medians for most water-quality variables were strongly related to land-cover class (Figure 4). Median concentrations of NO_3-N, NH_4-N, TN, TP and *E. coli* at the lowland pastoral and urban sites were elevated and MCI scores were low compared with sites in the natural land-cover class. In particular, median NO_3-N and TN concentrations at lowland pastoral and urban sites were c. 10 times higher than at lowland natural sites. Median NO_3-N, TN and *E. coli* concentrations for all lowland sites combined were about twice as high as the nationwide medians. Median concentrations of the other nutrients at lowland sites were 25%–40% higher and visual clarity 25% lower than the nationwide medians.

The 95th percentile *E. coli* concentration corresponding to the NPS-FM minimum acceptable state for primary contact was exceeded at 97% of the lowland sites, including all urban sites, 99% of pastoral sites, 83% of exotic forest sites and 77% of natural sites (Table 3). The NPS-FM *E. coli* bottom-line for secondary contact was exceeded at 4% of lowland sites. Median NO_3-N concentrations at 1% of lowland sites exceeded the NPS-FM bottom-line for toxicity, and median pH-adjusted NH_4-N concentrations did not exceed the NPS-FM bottom-line for toxicity at any lowland sites. Median NO_3-N, TN, DRP and TP concentrations at most lowland pastoral and urban sites exceeded the ANZECC trigger values, and median NH_4-N concentrations at most lowland urban sites exceeded the ANZECC trigger values. Table 3 also shows the proportions of upland monitoring sites where median nutrient concentrations exceeded the ANZECC upland trigger values; the upland trigger values for NO_3-N, TN and DRP concentrations were exceeded at more than half of the upland pastoral sites. There were no upland urban sites, and too few upland exotic forest sites to assess.

Mean NO_3-N, TN and *E. coli* concentrations were likely to be higher and MCI scores lower at lowland pastoral and urban sites compared with natural sites, based on non-overlapping 95% confidence intervals (Table 6). Mean NO_3-N and *E. coli* concentrations were likely to be higher and MCI scores lower at lowland urban sites compared with pastoral sites. The limited number of lowland exotic forest sites and the limited number of lowland natural sites with sufficient DRP, NH_4-N and TN data precluded comparisons with other land-cover classes.

Regression analyses for the lowland sites indicated that median nutrient and *E. coli* concentrations increased and MCI scores decreased with increasing proportions of high-intensity agricultural and urban land cover (Table 7). Visual clarity decreased with increasing proportions of agricultural land cover, but urban land cover explained little variation in visual clarity. The regression models were minimally affected by multi-collinearity, as indicated by VIF <2 for each model. Proportions of high-intensity agricultural

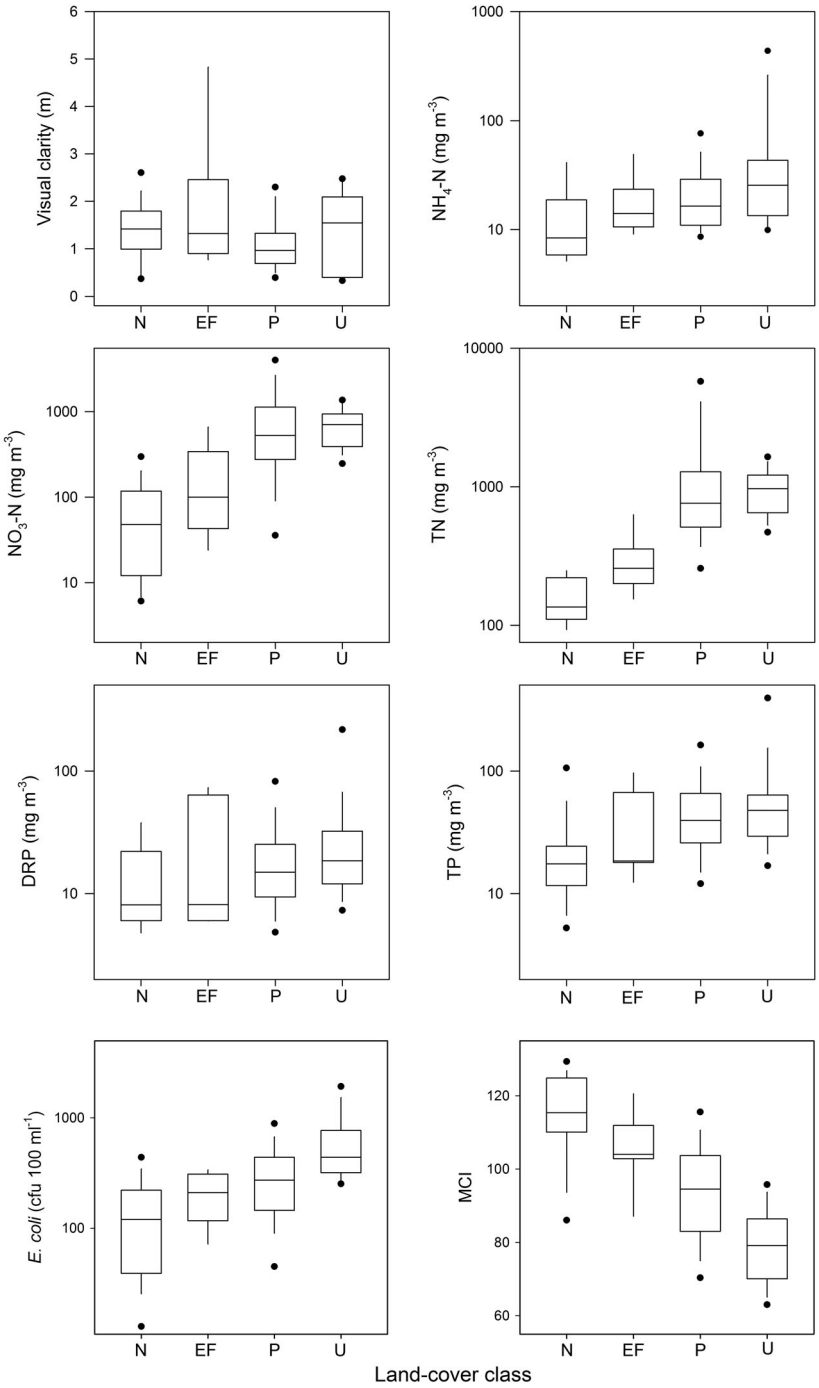

Figure 4. Lowland sites. Distributions of median values of water-quality variables, grouped by REC natural (N), exotic forest (EF), pastoral (P) and urban (U) land-cover class. Scales and percentiles as in Figure 2. Nutrient and *E. coli* concentrations are on log-scales and visual clarity and MCI scores are on linear scales. Percentiles: boxes = 25% and 75%; horizontal bars = medians; whiskers = 10% and 90%; closed circles = 5% and 95% (for classes with >10 sites).

Table 6. Lowland sites. Means and normal 95% confidence intervals (CI) for water-quality variables at lowland sites in the state data set, subdivided into land-cover classes.

	Land-cover class							
	Natural		Exotic forest		Pastoral		Urban	
Variable (unit)	Mean	CI	Mean	CI	Mean	CI	Mean	CI
Clarity (m)	1.4	0.5	1.9	–	1.1	0.1	1.4	0.6
NH_4-N (mg m^{-3})	14.7	–	19.6	–	30.5	8.7	82.8	–
NO_3-N (mg m^{-3})	77.7	55.0	212.7	–	1102.1	224.0	704.9	158.7
TN (mg m^{-3})	156.9	–	312.9	–	1565.3	396.9	1002.3	190.1
DRP (mg m^{-3})	14.8	–	27.7	–	24.6	3.7	37.5	27.0
TP (mg m^{-3})	25.4	17.7	40.9	–	54.3	6.2	76.3	44.6
E. coli (cfu 100 mL^{-1})	156.5	79.3	209.1	–	334.5	37.9	671.5	252.6
MCI	114.4	6.0	105.4	–	93.4	2.1	79.2	4.1

Means for each land-cover class are based on site medians. Dashes indicate variables with too few sites (<10) to calculate confidence intervals.

Table 7. Lowland sites. Multiple linear regression models.

Variable (N)	Regression equation	VIF	Partial and total R^2	*P* value
Clarity (208)	$\log_{10}$CLAR = 0.102 - (0.002 × Agri)	1.066	Agri: 0.029 Total: 0.029	0.015
NH_4-N (215)	$\text{Log}_{10}NH_4$-*N* = 0.931 + (0.005 × Agri) + (0.008 × Urban)	1.322	Agri: 0.0911 Urban: 0.0543 Total: 0.145	<0.001
NO_3-N (281)	$\text{Log}_{10}NO_3$-*N* = 1.684 + (0.014 × Agri) + (0.012 × Urban)	1.218	Agri: 0.239 Urban: 0.093 Total: 0.332	<0.001
TN (161)	Log_{10}TN = 2.117 + (0.012 × Agri) + (0.010 × Urban)	1.342	Agri: 0.326 Urban: 0.176 Total: 0.503	<0.001
DRP (267)	Log_{10}DRP = 0.929 + (0.004 × Agri) + (0.005 × Urban)	1.274	Agri: 0.0375 Urban: 0.0336 Total: 0.0710	<0.001
TP (280)	Log_{10}TP = 1.231 + (0.005 × Agri) + (0.006 × Urban)	1.219	Agri: 0.0864 Urban: 0.0650 Total: 0.151	<0.001
E. coli (223)	Log_{10}*E. coli* = 2.146 + (0.003 × Agri) + (0.009 × Urban)	1.249	Agri: 0.0433 Urban: 0.0834 Total: 0.127	<0.001
MCI (232)	Log_{10}MCI = 2.060 - (0.001 × Agri) - (0.003 × Urban)	1.211	Agri: 0.237 Urban: 0.165 Total: 0.402	<0.001

N, number of monitoring sites; VIF, variance inflation factor; Agri, high-intensity agricultural land cover; Urban, urban land cover.

land cover alone explained 2%–33% of the variation in nutrient concentrations, visual clarity and MCI scores (Figure 5).

Water-quality trends: all sites

After applying the inclusion rules for trend analyses, 556 river monitoring sites were retained for analyses of one or more variables, and trends were estimated for 3015 variable by site combinations. Symmetric confidence intervals around 1559 of those trends did not contain zero, and we considered the directions of those trends to be established with confidence. Trend directions were classed as degrading if they indicated increasing nutrient or

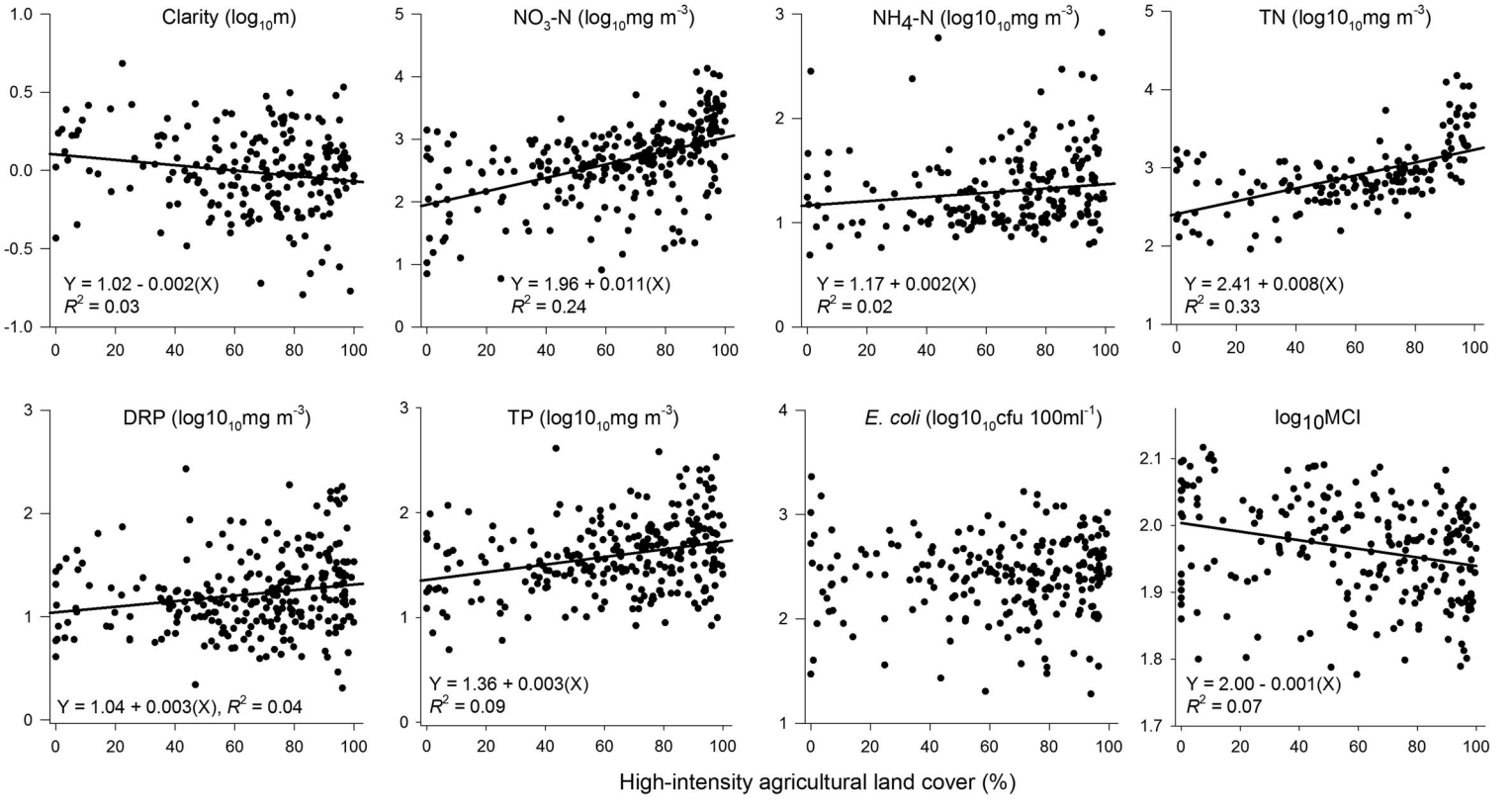

Figure 5. Lowland sites. Relationships between median water-quality state and high-intensity agricultural land cover in the catchments above lowland monitoring sites. Solid lines indicate least squares linear regression models.

E. coli concentrations or decreasing MCI scores or visual clarity, and improving if they indicated decreasing nutrient or *E. coli* concentrations or increasing MCI scores or visual clarity.

Pooling across the four land-cover classes, there were 10 times as many improving trends as degrading trends in TP, three times as many improving trends as degrading trends in DRP, approximately twice as many improving trends in NH_4-N and visual clarity, and 50% more improving trends in *E. coli* (Table 8). In contrast, there were three times as many degrading trends as improving trends in MCI scores and 60% more degrading trends in NO_3-N. There were approximately equal numbers of improving and degrading trends in TN.

The distributions of water-quality trend magnitudes at monitoring sites are shown in Figure 6, grouped by land-cover class and by climate class within the natural and pastoral land-cover classes. No consistently ordered variation in trend magnitude across land-cover classes was apparent, in contrast to the results for water-quality state. Within land-cover classes, the strongest improving trends were for TP and NH_4-N in the pastoral, urban and exotic forest classes, for DRP in the pastoral and urban classes, for *E. coli* in the exotic forest class, and for visual clarity in the urban and exotic forest classes; median magnitudes for all of these trends exceeded 1.5% yr^{-1}. The largest degrading trends were for NO_3-N in the exotic forest class (median magnitude 1.9% yr^{-1}) and NH_4-N in the natural class (median magnitude 2.7% yr^{-1}); no other degrading trends exceeded 1% yr^{-1}. Pooling across land-cover classes, the strongest trends were improving trends in TP (2.9% yr^{-1}), NH_4-N (2.1% yr^{-1}) and DRP (1.6% yr^{-1}). Median trend magnitudes for MCI scores indicated small degrading trends (<1% yr^{-1}) in each land-cover class. Median trend direction (i.e. improving or degrading) for some variables differed between climate classes within

Table 8. Numbers of monitoring sites with degrading and improving 10 year, flow-adjusted trends and numbers of sites with insufficient data to infer trend direction. Degrading trends indicate increasing nutrient and *E. coli* concentrations, and decreasing clarity and MCI scores.

	All sites											
	Degrading trends				Improving trends				Insufficient data			
Variable	N	EF	P	U	N	EF	P	U	N	EF	P	U
Clarity	23	5	42	0	40	10	75	5	50	7	124	5
NH_4-N	20	0	31	2	7	3	70	9	17	2	44	1
NO_3-N	44	13	131	1	35	4	77	6	46	7	132	15
TN	11	3	49	5	25	2	44	6	20	2	72	4
DRP	22	3	41	3	25	7	157	12	29	6	79	7
TP	5	1	17	2	37	9	196	13	42	4	91	4
E. coli	13	2	38	1	17	5	55	4	60	13	175	13
MCI	21	0	31	4	4	2	14	0	60	17	233	22
	Lowland sites											
	Degrading trends				Improving trends				Insufficient data			
Variable	N	EF	P	U	N	EF	P	U	N	EF	P	U
Clarity	0	1	23	0	2	1	45	4	5	0	73	5
NH_4-N	0	0	11	2	1	2	55	9	0	0	22	1
NO_3-N	3	3	61	0	5	0	50	6	2	1	86	14
TN	0	1	26	5	2	1	31	6	1	1	38	4
DRP	1	0	26	2	2	2	96	12	0	0	47	6
TP	0	0	10	2	6	4	132	13	3	0	49	4
E. coli	1	0	23	1	1	0	33	4	5	2	103	11
MCI	1	0	15	4	1	1	11	0	13	4	130	20

Land-cover classes: N, natural; EF, exotic forest; P, pastoral; U, urban.

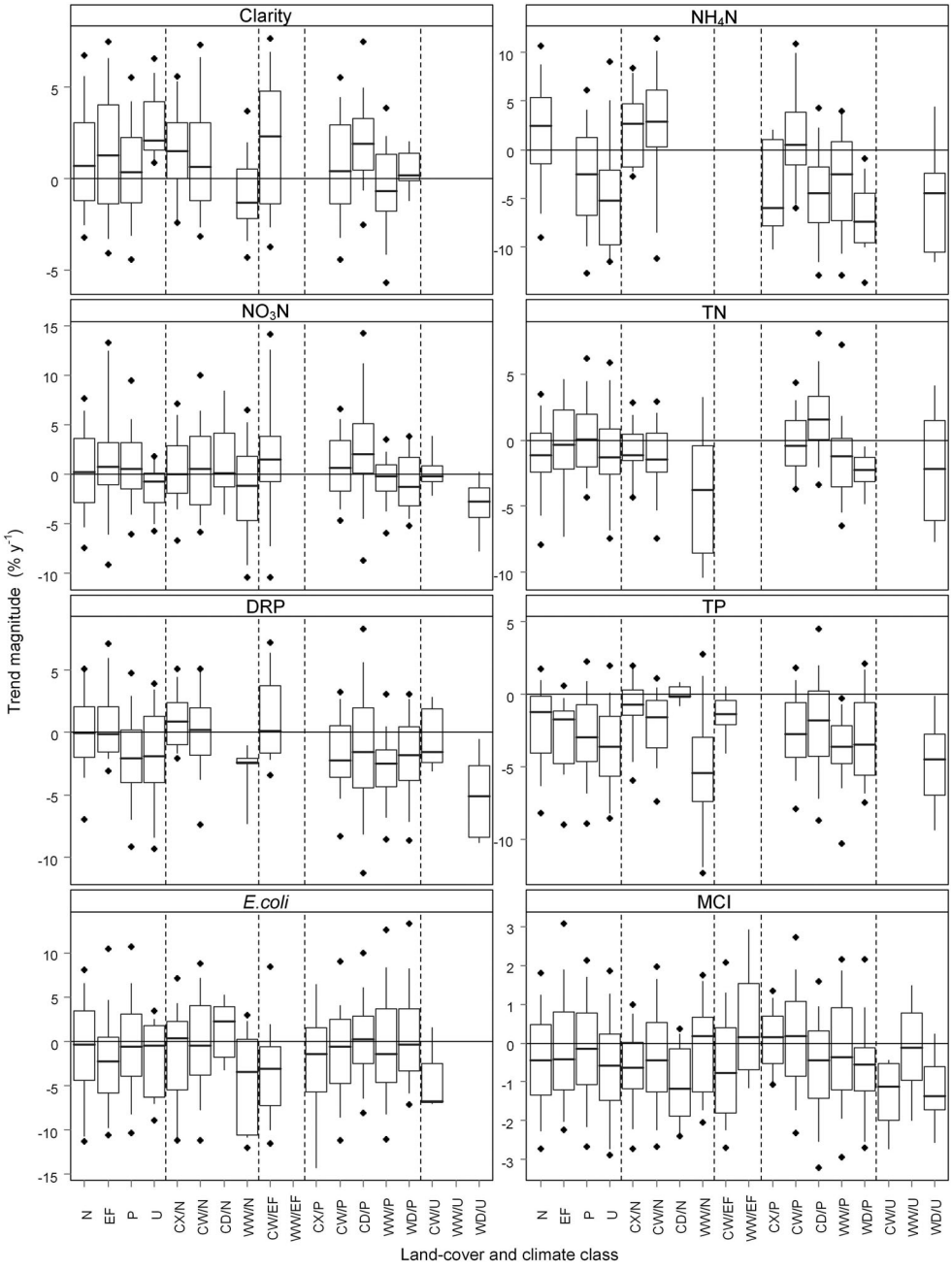

Figure 6. All sites. Temporal trends in water-quality variables, grouped by REC land-cover class. Each panel is subdivided into five compartments by dashed lines. Far left compartment: sites grouped by REC land-cover class. Remaining compartments: sites grouped by climate classes within each land cover class. Percentiles as in Figure 2. Classes with fewer than five sites not shown.

the natural and pastoral land-cover classes. For example, TN and NO_3-N trends in the cool-dry/pastoral class were strongly degrading (>2% yr^{-1}), but were improving in other climate classes.

Water-quality trends: lowland sites

A total of 376 lowland sites met the inclusion rules for trend analyses, and trends were estimated for 1410 variable by site combinations. For 761 trends, the symmetric confidence intervals did not contain zero. Patterns of degrading and improving trends at lowland sites were similar to those listed above for all monitoring sites: across land-cover classes, there was a predominance of improving trends in visual clarity, NH_4-N, DRP, TP and *E. coli*, and degrading trends in MCI scores (Table 8). However, the numbers of lowland sites with improving and degrading trends in NO_3-N and TN were nearly equal.

The distributions of water-quality trend magnitudes at lowland sites are shown in Figure 7. The strongest improving trends were for TP and DRP in the natural, urban and pastoral classes, NH_4-N in the urban and pastoral classes, NO_3-N in the natural class, and visual clarity in the natural class; the median magnitudes of these trends were >1.5% yr^{-1}. The strongest degrading trends were for MCI scores in the pastoral, urban and natural classes and NO_3-N in the pastoral class; the median magnitudes of these trends were < 1% yr^{-1}.

Discussion

Water-quality state: all sites

In the past 12 years, three successive river water-quality analyses have been carried out using combined data from regional council and NIWA monitoring sites: Larned et al. (2003) used data for the 1996–2002 period, Ballantine et al. (2010) used data for the 1998–2007 period and the current study used data for the 2004–2013 period. All three studies used the same general procedures for analysing water-quality state, although specific sites and inclusion rules varied. Qualitative results of these studies were consistent: nitrogen, phosphorus and *E. coli* concentrations were elevated and visual clarity low in the pastoral and urban cover classes compared with the natural class, and nitrogen, phosphorus and *E. coli* concentrations and visual clarity in the exotic forest class were intermediate.

In addition to the well-established categorical differences in water quality between land-cover classes, results of the current study indicate that national-scale water quality declines continuously as proportions of high-intensity agricultural and urban land cover in upstream catchments increase, based on responses of eight different water-quality variables. Some residual variation in these water quality–land cover relationships is attributable to spatial heterogeneity in climate and other environmental drivers. The differences in water quality that we observed between climate classes within the natural and pastoral land-cover classes provide an indication of the effects of climate heterogeneity. Despite the effects of other sources of variation, the water quality–land cover relationships reported here are moderately strong, particularly for TN, NO_3-N, MCI and *E. coli* ($R^2 \geq 0.5$). These relationships are useful for predicting effects of large-scale changes in land use, and for defining national reference conditions (McDowell et al. 2013).

Water-quality trends: all sites

The results of trend analyses in the three national-scale river water-quality studies (Larned et al. 2003; Ballantine et al. 2010; current study) are less consistent than the results of state

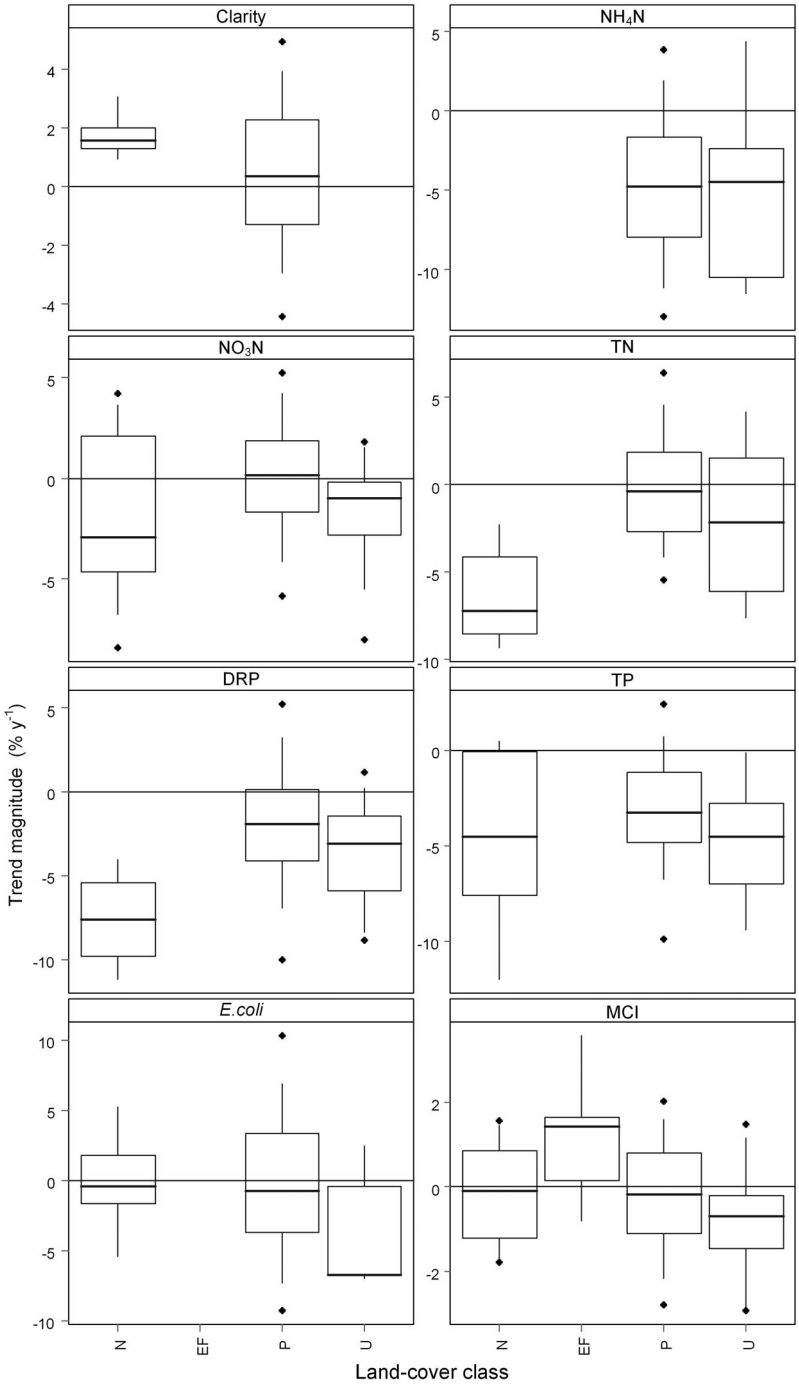

Figure 7. Lowland sites. Temporal trends in water-quality variables at lowland sites, grouped by REC land-cover class. Percentiles as in Figure 2. Classes with fewer than five sites not shown.

analyses. Here we summarise the predominant directions of trends at sites in the REC pastoral and natural land-cover classes, as these two classes accounted for most of the sites

used in each study. Larned et al. (2003) reported that improving 5–7 year trends outnumbered degrading trends in visual clarity at pastoral and natural sites, in NO_3-N at natural sites and in NH_4-N at pastoral sites. Ballantine et al. (2010) reported that improving 10 year trends outnumbered degrading trends in NH_4-N at pastoral and natural sites, and in *E. coli* and DRP at natural sites, whereas degrading trends outnumbered improving trends in visual clarity, NO_3-N, TN and TP at pastoral sites. In the current study, improving 10 year trends outnumbered degrading trends in visual clarity and TP at natural and pastoral sites, and in DRP and NH_4-N at pastoral sites; degrading trends outnumbered improving trends in MCI scores at natural and pastoral sites, in NH_4-N at natural sites, and in NO_3-N at pastoral sites.

The single consistent pattern across the three studies was the predominance of improving trends in NH_4-N at pastoral sites. This pattern suggests that a long-term improvement is under way. However, median NH_4-N concentrations at pastoral sites were <5% of median NO_3-N concentrations in these studies, which indicates that NH_4-N generally comprises a small proportion of the dissolved inorganic nitrogen pool. In view of the predominance of degrading (i.e. increasing) trends in NO_3-N at pastoral sites in both Ballantine et al. (2010) and the current study, it is unlikely that reductions in NH_4-N will substantially reduce nitrogen enrichment at these sites in the near future.

Trend analyses in the current study indicated that NH_4-N, TP and DRP concentrations at sites in the pastoral land-cover class have decreased over the 2004–2013 period at median rates >1.5% yr^{-1}. Causes of these improving trends have not been identified, but stock exclusion from waterways, improved farm effluent treatment, improved fertiliser management and reductions in phosphorus fertiliser use may all be contributing factors. Stock exclusion reduces direct excretion in waterways, bank erosion and sediment resuspension (McDowell & Nash 2012; Wilcock et al. 2013a, 2013b). The number and length of New Zealand rivers with exclusion fencing has increased in the past decade, although verification is incomplete (Sanson & Baxter 2011; Shaw & Wimmers 2014). Farm effluent treatment in New Zealand, particularly the North Island, has gradually shifted from effluent pond discharge to waterways to low-rate land application, which reduces nutrient loading to surface water (Monaghan et al. 2007, 2010). Increased compliance with fertiliser best-management practices may be reducing phosphorus loading to waterways through surface runoff (McDowell & Nash 2012; NZFA 2013). Finally, a recent decline in phosphorus fertiliser use may contribute to the improving trends in DRP and TP in some areas. National phosphorus fertiliser use declined by approximately 6% yr^{-1} over the 2004–2013 period (Fertiliser Association of New Zealand, unpubl. data).

Despite improvements in effluent treatment, fertiliser management and stock exclusion, degrading trends in NO_3-N occurred at many sites in the pastoral land-cover class; the median trend indicated that NO_3-N concentrations have increased by 0.6% yr^{-1} for the past decade. In contrast to this gradual trend, the annual rate of nitrogen input to New Zealand pastoral land has increased rapidly; estimated input (predominately through urea application and biological nitrogen fixation) increased at c. 4% yr^{-1} from 2001 to 2010 (Parfitt et al. 2012), and nitrogen fertiliser input continued to rise from 2010 to 2014 (Statistics NZ 2015). Annual nitrogen input to agricultural land has exceeded 900 Gg yr^{-1} for more than a decade (Parfitt et al. 2012).

There are at least three possible explanations for the difference between large trends in nitrogen input and much smaller trends in river NO_3-N concentrations: (1) a large

proportion of the nitrogen input over the past decade is accumulating in agricultural soils and has not yet leached to groundwater or reached rivers (Parfitt et al. 2012); (2) a large proportion of the nitrogen input to land leaches rapidly and is accumulating in groundwater, but has not yet reached rivers (Daughney & Reeves 2006; Morgenstern & Daughney 2012); (3) NO_3-N leaching and river enrichment have both increased sharply in response to nitrogen input in some regions, but these increases are offset by decreased leaching and declining river concentrations in other regions (Dymond et al. 2013). Some support for the third explanation comes from the differences between climate classes in river NO_3-N trends (Figure 6). The largest increasing trend in river NO_3-N concentrations (median rate: 2.3% yr^{-1}) was in the cool-dry/pastoral class. This class is dominated by sites in the eastern South Island, where NO_3-N leaching has also increased rapidly over the past decade (Dymond et al. 2013). In contrast, river NO_3-N concentrations were decreasing in the warm-dry/pastoral and warm-wet/pastoral classes (median rates: 1.3 and 0.3% yr^{-1}, respectively); these classes are dominated by sites in the central and northern North Island, where NO_3-N leaching has decreased or changed little over the past decade (Dymond et al. 2013).

Macroinvertebrates are a recent addition to the river monitoring programmes of most New Zealand councils, and data required for national-scale trend analyses were unavailable until recently. The only previous national-scale analysis of MCI trends was based on data from 66 NRWQN sites for the 1989–1995 period (Scarsbrook et al. 2000). In that study, median MCI scores at minimally impacted baseline sites increased by 18% between 1989 and 1995. In the current study, the median MCI score at natural land-cover sites (the closest analogues to NRWQN baseline sites) decreased by 4% between 2004 and 2013. Scarsbrook et al. (2000) suggested that temporal changes in environmental factors independent of land use caused the improving trend at NRWQN baseline sites, and the same may be true for the recent degrading trend at natural land-cover sites. In both cases, controlling factors may include components of flow and thermal regimes that affect invertebrate immigration and emigration, thereby affecting MCI scores (Booker et al. 2015).

Lowland river water quality

Differences in classification criteria preclude quantitative comparisons between this study and the previous national study of lowland river water quality (Larned et al. 2004). We altered the classification criteria in the current study to focus the analysis on groundwater-dominated lowland rivers, which are considered to be at risk of ecological degradation and unsafe for contact recreation (Hancock 2002). Here we restrict our comparisons between the two lowland river studies to qualitative patterns. In both studies, median NO_3-N, DRP, NH_4-N and *E. coli* concentrations at pastoral and urban sites, and TN and TP concentrations at pastoral sites exceeded the ANZECC trigger values. In the earlier study, visual clarity at lowland pastoral and urban sites was low compared with natural sites, but these differences were not evident in the current study, due in part to increased variability at natural sites. Flow-adjusted trends in median nutrient concentrations were uniformly small (<0.2% yr^{-1}) in the earlier study. The trends in that study for which directions could be inferred with confidence were limited to river flow (decreasing in the pastoral and urban classes) and water temperature and visual clarity

(increasing in the pastoral class). In the current study, we observed improving trends of substantially larger magnitude (median >1.5% yr^{-1}) for DRP, TP and NH_4-N concentrations in the lowland pastoral and urban land-cover classes and for DRP and NO_3-N in the natural class. The possible causes of the recent improving trends discussed above apply to the lowland sites as well.

Despite some improving trends, the current state of water quality in New Zealand lowland rivers is generally poor, as indicated by elevated median nitrogen, phosphorus and *E. coli* concentrations. The negative relationships between water quality and high-intensity agricultural and urban land cover, and the predominance of pastoral and urban land cover in low-elevation catchments suggests that lowland river water-quality state is strongly influenced by agricultural and urban land use. The association between water-quality degradation and agricultural and urban land use in lowland catchments is not unique to New Zealand; similar associations have been reported from many other regions (e.g. Neal & Jarvie 2005; Morgan & Kline 2011). Water-quality degradation in lowland rivers in New Zealand is not limited to catchments dominated by pastoral and/or urban land cover, as indicated by the elevated median *E. coli* concentrations at some lowland sites in the natural land-cover class. Elevated *E. coli* concentrations at these sites may be associated with point sources (e.g. septic tanks, livestock fords) rather than large-scale land use, or to small pastoral or urban areas within otherwise natural land-cover-dominated catchments (Line et al. 2008; McDonald et al. 2008).

In many groundwater-dominated lowland rivers, poor water quality is partly due to the 'legacy effects' of land use extending back several decades (Jarvie et al. 2013, Tesoriero et al. 2013). Legacy effects occur when contaminant leaching from the land surface to groundwater and discharge from groundwater into surface waters are separated by long groundwater residence times. In New Zealand, consideration of the legacy effects of contaminated groundwater have generally focused on NO_3-N, which leaches rapidly to groundwater from a wide range of soils (Howard-Williams et al. 2010; McDowell et al. 2015; Morgenstern et al. 2015). However, there is circumstantial evidence suggesting that phosphorus from historical land use is also transported to surface waters via groundwater, and poses a risk of adverse legacy effects. Results of a New Zealand-wide assessment of DRP in groundwater and rivers (1997–2007 data) and Olsen phosphorus in soils (1988–2003 data) indicated that: (1) phosphorus levels are elevated in soils used for dairying and industry; (2) groundwater under dairy catchments, particularly in alluvial aquifers, is DRP-enriched compared with other land-use classes; and (3) degrading trends in river and groundwater DRP concentrations within agricultural catchments are correlated (McDowell et al. 2015). These observations suggest that phosphorus leaches from agricultural soils and accumulates in groundwater when soil retention capacities are exceeded, as has been observed in small-scale lysimeter studies (McDowell 2008). Large stores of soil and groundwater phosphorus with long lag times are land-use legacies that may affect rivers in the future, and reduce the long-term benefits of improved fertiliser and effluent management (Dodd et al. 2012).

The following picture is emerging from repeated analyses of New Zealand river water-quality data: (1) water-quality state at sites in the pastoral and urban land-cover classes is generally poor, as indicated by elevated median nutrient and *E. coli* concentrations and low MCI scores relative to natural sites; (2) water-quality state at lowland sites is generally poorer than that at upland sites, and lowland urban and pastoral sites have the poorest

overall water-quality state; (3) median trend magnitudes in the urban and pastoral land-cover classes indicate recent, substantial decreases in DRP, TP and NH_4-N concentrations, but not in NO_3-N, TN or *E. coli* concentrations. Elevated NO_3-N, TN and *E. coli* levels at sites in the urban and pastoral classes have persisted for two decades, based on analyses extending back to 1996. Legacy effects, continued agricultural intensification and urban growth, and projections of future intensification (Anastasiadis & Kerr 2013; Keller et al. 2014) all highlight the need for continual improvements in land-use management, to limit future water-quality degradation.

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