Urban river and stream water quality state and trends 2008-2017

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Executive summary

Ministry for the Environment (MfE) and Statistics New Zealand (Stats NZ) use analyses of urban river and stream water quality state and trends to inform policy development and meet their requirements for environmental reporting on the freshwater domain under the Environmental Reporting Act 2015. In a previous report for MfE (Gadd 2016), we provided water quality state and trends based on state-of-environment (SoE) monitoring data from 17-54 urban stream monitoring sites (depending on the variable); all located in either the Auckland or Wellington regions or in Christchurch City. Those data were assessed for the time period up to and including December 2015.

The current report provides an update and advancement on the 2016 report as described below:

- This report includes a new data compilation in order to report updated states and trends with data assessed for the time period up to and including December 2017.
- Four additional water quality variables and MCI scores are included, as well as additional metrics for assessing *E. coli*.
- We have included sites outside the three geographical areas covered in the 2016 report by incorporating data compiled for the national river water quality state and trends report (Larned et al. 2018).
- Statistical methods used in the current study include several advances on previous urban stream water quality state and trend analyses: 1) an updated statistical procedure was used to determine the directions of trends (and associated confidence) and the magnitudes of trends; 2) new statistical procedures were used to assess aggregations of trends at multiple sites.
- Quantitative relationships were investigated between water quality state and both urban land cover and the proportion of impervious surface in the catchment (for a subset of sites where data were available.)

There was a wide variation between sites in the median concentrations of dissolved zinc, DRP, nitrate-N, turbidity and *E. coli*, whereas the range was much smaller for dissolved copper, TN, clarity and turbidity.

For *E. coli*, 56 out of 75 sites were in the E (Red) attribute state based on median *E. coli* concentrations. There were no sites where nitrate-N or ammoniacal-N were below the National Bottom Line, with either 95% or 100% of sites respectively in A or B attribute state. Around 42% of sites where dissolved zinc was measured, and about 25% of sites where dissolved copper was measured had median concentrations that were greater than the ANZECC (2000) default guideline for 95% protection. For many of the sites where median concentrations were below the guidelines, the 75th percentiles exceeded the guideline values.

The water quality varied considerably between regions but quality also varied within regions and there were few regions where there were more than 3 sites monitored, restricting comparisons. There were differences in the water quality state between REC classes for nitrate-N, dissolved zinc, turbidity, MCI and some metrics of *E. coli*. For DRP there was very little difference between REC classes.

For dissolved zinc, ammoniacal-N, DRP, TP, turbidity and nitrate-N, median concentrations tended to be higher at sites with more than 30% urban land cover in the upstream catchment. There was a significant relationship between urban land cover in the catchment and both dissolved zinc and ammoniacal-N and, based on the subset of sites where data were available, between impervious area and the same two water quality variables and also DRP.

For trends assessed over 10-years (or 7-years for metals at some sites), there were more improving trends than degrading trends, except for *E. coli* and MCI. For these two variables, close to 50% of sites were unlikely (or very unlikely, extremely unlikely etc) to be improving and less than 30% of sites were likely to be improving. There were no clear differences in the trends (direction and magnitude, or percent improving trends) between regions, but there were some differences between REC classes for DRP and *E. coli*; and some differences between urban land cover categories for DRP and TN.

Recommendations for future reporting and monitoring include analysis across the state and trends results to assess whether the improving trends are at those sites with very poor water quality or with acceptable water quality; testing the effect of flow-adjustment to determine trends; and acquiring impervious surface data where absent and updating where >10 years old.

1 Introduction

The New Zealand Ministry for the Environment (MfE) and Statistics New Zealand (Stats NZ) use analyses of urban river and stream water quality state and trends to inform policy development and meet their requirements for environmental reporting on the freshwater domain under the Environmental Reporting Act 2015. In this report, we use “urban stream water quality” as a general term to refer to the physical, chemical and biological variables that are included in these river state-of-environment (SoE) monitoring programmes. In a previous report for MfE (Gadd 2016), we provided water quality state and trends based on monitoring data from 17-54 urban stream monitoring sites (depending on the variable); all located in either the Auckland or Wellington regions or in Christchurch City. Those data were assessed for the time period up to and including December 2015.

In the current report, we have undertaken a new data compilation in order to report updated states and trends with data assessed for the time period up to and including December 2017. In addition, we have included sites outside these three geographical areas by incorporating data compiled for the national river water quality state and trends report (Larned et al. 2018). The data assessed are from monitoring that is usually carried out at scheduled monthly visits regardless of stream flow levels and therefore the resulting data includes both samples from low flows and high flows, if a high flow occurred on the scheduled monitoring date. It does not include sampling that was targeted at storm events, or sampling of piped stormwater.

The methods used in the current study include several advances on previous urban stream water quality state and trend analyses: 1) a new statistical procedure was used to determine the directions of trends (and associated confidence) and the magnitudes of trends; 2) new statistical procedures were used to assess aggregations of trends at multiple sites. These methods were also used in a report on national water quality state and trends (Larned et al. 2018).

This report on urban stream water quality state and trends consists of the following:

- detailed methods for data processing and analysis;
- analyses of water quality state assessed for each region, REC class and by urban land characteristics (urban land cover and impervious area);
- analyses of water quality trends, including the new trend aggregation methods, assessed for each region, REC class and by urban land characteristics (urban land cover);
- a summary of the findings and recommendations for future work.

This report is supplemented by files provided to MfE with site-specific results and associated metadata. The tabulated, site-specific results will enable MfE and other users to use the results for a wide range of purposes (e.g., grouping by environmental classes, mapping) that are all based on a single, comprehensive explanation of methods.

The analyses in this report were aligned, where possible, with attributes incorporated in the National Policy Statement for Freshwater Management of 2017 (NPS-FM, (New Zealand Government 2017)) The NPS-FM requires regional councils, through their regional plans, to set freshwater objectives that provide for freshwater values, and to set limits and develop management actions to achieve those objectives. The NPS-FM identifies multiple attributes to assist regional councils in developing numeric objectives for rivers and lakes, and policies (including limits) for achieving those objectives.

2 Data Methods

2.1 Water quality variables

We assessed river water quality using ten variables that correspond to physical, chemical and microbiological conditions, and macroinvertebrate community composition (Table 2-1). Data corresponding to the physical, chemical and microbiological variables came from monthly (or occasionally quarterly) samples; macroinvertebrate data came from annual samples.

Table 2-1: Water quality variables included in this study.

Variable type	Variable	Abbreviation	Units
Physical	Clarity	None	m
	Dissolved copper	None	mg/L
	Dissolved zinc	None	mg/L
	Ammoniacal nitrogen	None	mg/L
Chemical	Nitrate nitrogen	None	mg/L
	Total nitrogen (unfiltered)	TN	mg/L
	Dissolved reactive phosphorus	DRP	mg/L
	Total phosphorus (unfiltered)	TP	mg/L
Microbiological	*Escherichia coli*	*E. coli*	n/100 mL
Biological	Macroinvertebrate community index	MCI	unitless

Visual water clarity (clarity) is a measure of light attenuation due to absorption and scattering by dissolved and particulate material in the water column. Clarity is monitored because it affects primary production, plant distributions, animal behaviour, aesthetic quality and recreational values, and because it is correlated with suspended solids, which can impede feeding in fish and cause riverbed sedimentation. Visual clarity in rivers is generally measured *in situ* as the horizontal sighting range of a black disc (Ministry for the Environment 1994). At a small number of sites, clarity is measured adjacent to the river with water samples in clarity tubes.

Copper and zinc are important pollutants in urban streams, transported via stormwater from roads (zinc from tyre wear, copper from brake pad wear), roofs (zinc from galvanised roofing) and other impervious surfaces (including paved areas around industrial sites). The dissolved forms of the metals are more routinely monitored as they are of more importance for toxicity than the 'total' forms, which includes metals attached to particulate material and are less bioavailable (ANZECC & ARMCANZ 2000). Dissolved metals are typically defined as those that pass through a 0.45 µm filter (USEPA 1996).

The five nutrient species (nitrate-N, ammoniacal-N, DRP, TN and TP) were included because they influence the growth of benthic river algae (periphyton) and vascular plants (macrophytes), and because nitrate and ammonia can be toxic to aquatic organisms at high concentrations. Nutrient enrichment from point and non-point source discharges is strongly associated with intensive land use in New Zealand (Larned et al. 2016; Snelder et al. 2018). Nutrient enrichment can promote excessive, 'nuisance' growth of periphyton and macrophytes that can, in turn, degrade river habitat, increase daily fluctuations in dissolved oxygen and pH, impede flows, block water intakes, and cause water colour and odour problems. At elevated concentrations, nitrate and ammonia can be toxic to river fish and invertebrates (Hickey 2013; Hickey 2014). Mechanisms of nitrate and ammonia toxicity include reduced oxygen transport by haemoglobin, carcinogenic nitrosamine formation, and disruption of ion transport across cell membranes (Camargo et al. 2005).

The concentration of the bacterium *E. coli* is used as an indicator of human or animal faecal contamination and the risk of infectious human disease from waterborne pathogens in contact-recreation and drinking water. There is an NPS-FM attribute based on *E. coli* concentrations that is related to the management of human health. The national bottom line for *E. coli* concentrations for secondary contact recreation is 1000/100 ml (as a median). For primary contact recreation sites, the *E. coli* concentration at the "minimum acceptable state" is 540/100 ml, as a 95th percentile.

We used the New Zealand Macroinvertebrate Community index (MCI) as a biotic indicator of general river health. MCI scores are calculated using tolerance values for the macroinvertebrate taxa that are present in benthic samples. Tolerance values are weighting factors that correspond to the relative abundance of taxa along stressor gradients. We used the non-quantitative MCI in lieu of the quantitative or semi-quantitative forms of MCI because some council datasets do not include invertebrate abundance data (Stark and Maxted 2007) and non-quantitative MCI scores (based on presence/absence) are more widely available. We did not calculate the MCI scores but used the scores as supplied by each council or obtained from LAWA. For some sites, the scores were specified as either MCI-hb (hard-bottomed) or MCI-sb (soft-bottomed), which indicates the tolerance values used in the calculation of the MCI score, as appropriate for either hard-bottomed (stony) or soft-bottomed (fine sediment dominated) streams. However, there were also many sites where this was not specified and therefore in this report all scores are presented simply as MCI. In contrast to the monitoring frequencies for physical and chemical variables and *E. coli*, which are measured monthly or quarterly, the invertebrate samples used to calculate MCI scores are generally collected once each summer. Due to the difference in sampling frequency, trend analyses of MCI scores were carried out using a different procedure that that used for the other variables (see Sections 3.2.1 and 3.2.5).

Five additional river water quality variables were considered for analysis: suspended sediment concentration (SSC), deposited fine sediment areal cover, periphyton biomass, and *Phormidium* areal cover. Several regional councils had no corresponding data and most of the remaining council datasets comprised few sites or did not meet the sampling frequency and duration criteria we applied (Sections 3.1.1 and 3.2.1). After assessing the number and geographic distribution of measurements for these variables, and in consultation with MfE, these variables were omitted from further analysis.

2.2 Data acquisition and compilation

2.2.1 Council data

New Zealand regional councils, district/city councils and unitary authorities carry out state-of-environment monitoring at numerous sites in rivers, however only a sub-set of these can be considered urban. Furthermore, few councils monitor metals, a key variable for reporting on urban streams. Metal data were supplied directly to the authors by three councils: Auckland Council (AC), Greater Wellington Regional Council (GWRC) and Christchurch City Council (CCC), who do monitor metals in their urban streams, along with other water quality variables. These councils also monitor metals in streams with catchments that are not dominated by urban landcover, either as reference sites, to assess rural metal sources or as the catchments are expected to undergo urbanisation in the future. In addition, metal data were sought and acquired from Taranaki Regional Council, however none of the data were suitable for analysis of state or trends due to the infrequency of sampling (see rules in Section 3.1.1).

For the other water quality variables, the data were obtained from the database (an Rdata file) of national river water quality monitoring data that has been compiled for the national river water quality state and trends report (Larned et al. 2018). Those data were compiled from previous reporting projects, interrogation of data servers operated by individual regional councils, Land Air Water Aotearoa (LAWA), NIWA (for National River Water Quality Network (NRWQN) data) requests to LAWA data managers for the most recent (2017) data, and direct requests to councils for data that were unavailable through data servers or LAWA.

Data for all streams that were classified as having "urban" landcover in that database were acquired. Because this classification was based on the Land Cover Database version 3 (LCDB3), which was based on satellite data acquired during 2008/09 (Landcare Research 2012) and is now nearly 10 years out of date, several additional sites were added where Councils classified these sites as urban in more recent reports (Keenan and Morar 2015; Margetts and Marshall 2015; Holland et al. 2018).

2.2.2 Data processing

River water-quality data were processed in several steps to ensure that the datasets acquired from different sources were internally consistent, that site information was complete and accurate, that consistent measurement procedures were used, and that data were as error-free as possible. These steps were largely carried out during the data compilation and processing steps for the national river water quality state and trends database, and were replicated for the metals data set.

Step 1. Reporting conventions. The water-quality data received from councils and LAWA varied widely in reporting formats, reporting conventions for variable names, site identifiers, date and time formats, units of measurement, and other data structure elements. We first applied a consistent set of reporting conventions. Common errors included mislabelled site-names, incorrect units and data transcription errors. We applied a flagging system developed in the previous project that attaches metadata to individual data points. Flags include censored data, unit conversions (e.g., from mg/L to µg/L), and values that were synthesised from other data (e.g., MCI).

Step 2. Monitoring site spatial information. The following spatial data were associated with each river monitoring site: site name, location and regional council identifier (if available), NZMS260 grid reference (converted from NZTM as necessary), and NZReach number. NZReach numbers are defined in the River Environment Classification (REC) geodatabase. Sites were mapped to reveal and correct georeferencing errors.

Step 3. Comparable field and laboratory methods. The first data processing step was to assess methodological differences for all variables. For most of the variables, two or more measurement procedures were represented in the datasets. We grouped data by procedure, then pooled data for which different procedures gave comparable results, based on assessments set out in Larned et al. (2016) and Gadd (2016). Data measured using the less common and non-comparable methods were eliminated. Table 2-2 lists the most common procedures used for each variable, and the procedures corresponding to data retained for analysis.

Table 2-2: Measurement procedures for water quality variables. Procedures retained: data generated by the procedures in this column, and corresponding monitoring sites, were retained for analysis in this study.

Variable	Measurement procedures	Procedures retained
Dissolved copper	Filtration through 0.45 µm membrane filters, ICP-MS (USEPA 200.8); Filtration through GF/F filters (0.7 µm), GFAA; Filtration through 0.45 µm membrane filters, ICP-MS (APHA 3125 B, 21st ed. 2005)	All procedures retained, difference in filter pore size expected to result in negligible differences
Dissolved zinc	Filtration through 0.45 µm membrane filters, ICP-MS (USEPA 200.8); Filtration through GF/F filters (0.7 µm), ICP-OES; Filtration through 0.45 µm membrane filters, ICP-MS (APHA 3125 B, 21st ed. 2005)	All procedures retained, difference in filter pore size expected to result in negligible differences
Nitrate-N	Ion chromatography, filtered samples Cadmium reduction, filtered samples Azo dye colourimetry, filtered samples	All procedures retained (nitrite in cadmium-reduction and Azo-dye measurements is assumed to have minimal influence on nitrate-N for purposes of this study [1])
Ammoniacal-N	Phenol/hypochlorite colorimetry, filtered samples	Phenol/hypochlorite colorimetry, filtered samples
Total nitrogen	Persulfate digestion, unfiltered samples Dissolved inorganic+organic nitrogen, filtered samples Kjeldahl digestion (TKN + NNN)	Persulfate digestion, unfiltered samples
Total phosphorus	Persulfate digestion, unfiltered samples Dissolved inorganic+organic phosphorus, filtered samples	Persulfate digestion, unfiltered samples
DRP	Molybdenum blue colourimetry, filtered samples	Molybdenum blue colourimetry, filtered samples
Clarity	Black-disk Horizontal clarity tube	Both procedures retained (presumed to give comparable results)
Turbidity	Measurement in Formazin Nephelometric units Measurement in Nephelometric turbidity units Measurement in Jackson turbidity units	All procedures retained (presumed to give comparable results)
E. coli	Colilert QuantiTray 2000 Membrane filtration	Both procedures retained (presumed to give comparable results)
MCI	Collection procedures C1, C2, C3, C4 Processing procedures P1, P2, P3	All procedures retained (presumed to give comparable presence/absence data for calculating non-quantitative MCI scores)

[1] Note that while this is considered acceptable for this report, which provides a broad scale assessment of stream water quality, this assumption may not be appropriate for more detailed analyses, including statistical comparisons between sites.

At the completion of the data processing steps, our dataset comprised 87 sites where water quality variables were measured with values for some or all of the variables listed in Table 2-1. There were fewer sites for clarity and metals as these were monitored at only a subset of the sites (number of sites for each variable listed in The data produced by multiple procedures used to measure *E. coli*, nitrate-N, clarity and MCI were pooled, based on the assumption that the different procedures gave comparable results. The cadmium reduction and azo dye methods for nitrate-N include nitrite-N, which while typically negligible in unpolluted waters, may be present at substantial concentrations (compared to nitrate-N) in urban streams at times (Gadd 2016). However, as these incidents are likely to be relative rare, and there are much larger differences in nitrate-N between sites, methods that include nitrite-N are considered acceptable.

In contrast, some procedures used to measure TN and TP are unlikely to give comparable results. Most councils and the NRWQN use the alkaline persulfate digestion method and unfiltered water samples. A smaller group of councils uses a sulphuric acid digestion procedure to measure total Kjeldahl nitrogen (TKN) and calculates TN as TKN + nitrate-N. At least one council uses filtered samples for the data labelled TN and TP, although the filtered samples are more correctly labelled total dissolved nitrogen and phosphorus. The alternative methods could generate substantial differences in reported TN and TP concentrations (Patton and Kryskalla 2003; Horowitz 2013). Therefore, only TN and TP measured by the persulfate digestion method with unfiltered samples were retained for analysis.

Step 4. Error correction and adjustment. We manually inspected the data to correct identifiable errors (e.g., transcription errors), and rescale data. We used time-series plots and quantile plots to identify and remove gross outliers for each variable. Where necessary, values were adjusted to ensure consistent units of measurement across all datasets.

Table 2-3). There were 53 sites for MCI, only some of which were the same sites as the water quality sites. In total there were 98 distinct river monitoring sites.

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identify and remove gross outliers for each variable. Where necessary, values were adjusted to ensure consistent units of measurement across all datasets.

Table 2-3: Number of sites in the processed data set.

Variable	Number of sites in processed data set
Dissolved copper	57
Dissolved zinc	57
Nitrate-N	80
Ammoniacal-N	80
TN	80
TP	74
DRP	80
Clarity	33
Turbidity	76
E. coli	80
MCI	41

2.2.3 REC Classes, urban land cover and imperviousness

Information on the climate and source-of-flow was obtained for each site from version 2 of the digital river network (REC2), the database of catchment spatial attributes for New Zealand's river network [2]. This digital river network, representing New Zealand's rivers as ~594,000 segments with associated catchment boundaries, was derived from a digital elevation model (DEM). REC2 is a recent update of the network to correct errors and to improve representation of rivers. The network has an associated geodatabase that contains attributes representing the climate, topography, geology, vegetation, infrastructure and hydrology for the catchments of all segments. These attributes were used to classify streams according to climate and source-of-flow as defined in Snelder and Biggs (2002). Landcover was not included in the classification for this report as all sites included in the analysis of urban river and stream state had > 15% urban land use in the catchment, the classification used in Snelder and Biggs (2002) to indicate an urban stream.

The proportion of urban land cover in the upstream catchment was calculated using data from the Land Cover Database version 4.1 (LCDB4.1), which is based on satellite data acquired during summer 2012/13 (Landcare Research 2015). Urban land use was defined as the sum of proportional land cover in three LCDB4 classes (built-up areas, urban parks, transport infrastructure). Note that this definition differs slightly from that used by the REC and by the national rivers water quality report, which also includes mines and dumps as urban land use.

Impervious surface data were acquired for three geographical areas: Auckland, Wellington and Christchurch. Data for the Auckland Region were obtained from the Auckland Council GIS team and included two layers: building footprints and impervious surfaces (2008 data). Data for Wellington was obtained from Wellington Water Ltd and was only available for sites within the Wellington City

[2] https://www.niwa.co.nz/freshwater-and-estuaries/management-tools/river-environment-classification-0

Council boundary. These data represent the impervious surfaces in 2013/2014 based on 2013/2014 satellite imagery. Data for Christchurch was obtained from Christchurch City Council and was available at the meshblock resolution. These data represent the extent of impervious surfaces present in January 2007 satellite imagery. In total, impervious surface data were available for 53 of the sites in the urban stream database.

Upstream catchments were defined as in the REC2 database, except for catchments for the Avon and Styx Rivers in Christchurch. For these, catchment boundaries and sub-catchments were supplied as GIS shape files by Christchurch City Council and the sub-catchment boundaries were further split as needed based on sampling site location. These catchments were used in preference to the REC catchments which have incorrect upstream boundaries and incorrectly include sizeable areas of rural land. Calculations of the area of upstream land cover classes and of impervious surface cover were undertaken in ArcGIS, exported into excel for post-processing and then summary data were imported into the R statistical computing environment for analysis (R Core Team 2018).

3 Analysis Methods

3.1 Methods for state assessment

3.1.1 Sampling dates and time-periods

The previous report used a three-year time period to assess state, as a reasonable trade-off between minimising the effects of temporal trends, avoiding the effects of temporary wastewater discharges to streams in Christchurch and maximising the number of sampling sites that would be included. For consistency, a three-year time period has been used again in this report, from January 2015 to December 2017. For the monthly monitoring data used in this assessment, a period of at least 3 years should yield at least 30 samples (maximum 36 samples) depending on the amount of missing data. As a general rule, 30 is considered a sufficient number of samples for providing a reasonable level of confidence in statistical estimates, and there are diminishing returns on increasing confidence with increasing sample size above this (McBride 2005). For MCI which is usually sampled annually over summer, we used the time period 1 July 2014– 30 June 2017 where available, in order to capture complete summer sampling seasons. This three-year period typically yields only 3 samples, so there is lower confidence in these estimates.

Because concentrations of water quality variables tend to vary seasonally, it is also important that each season is well-represented over the period of record. In New Zealand, it is common to sample either monthly or quarterly, and in these cases, seasons are defined by months or quarters. We therefore applied a rule that restricted site × variable combinations in the state analyses to those with measurements for at least 90% of the sampling intervals in that period (at least 32 of 36 months or 11 of 12 quarters) and at least 2 out of 3 of the years for MCI to include more sites. If this was restricted to all years, this would have resulted in only 28 sites being included for analysis of state. Site × variable combinations that did not comply with these rules were excluded from the state analysis.

This yielded 14 to 76 sites for analysis of state, depending on variable (Table 3-1) with most from the Auckland, Wellington and Canterbury Regions. There were more acceptable sites for dissolved nutrients and *E. coli* (71-76 sites) than for metals and total nutrients (51-55 sites). This is due to dissolved metals being monitored only in Auckland, Christchurch and Wellington; and total nitrogen and phosphorus being monitored in only selected catchments (rather than all sites) in Christchurch.

Table 3-1: Number of sites meeting data acceptability rules for 3 year periods for assessing state.

	Total	Auckland	Wellington	Canterbury	Other Regions
Dissolved copper	55	11	5	39	0
Dissolved zinc	55	11	5	39	0
Nitrate-N	76	11	5	42	18
Ammoniacal-N	76	11	5	42	18
Ammoniacal-N adjusted to pH 8.0	71	11	NA	42	18
DRP	74	11	5	42	16
Total nitrogen	53	11	5	19	18
Total phosphorus	51	11	5	19	16
Clarity	14	0	4	2	8
Turbidity	71	11	5	37	18
E. coli	75	11	5	42	17
MCI	39	7	6	6	20

The locations of sites included in the state assessment are shown in Figure 3-1 and a full list of the sites is in Appendix A.

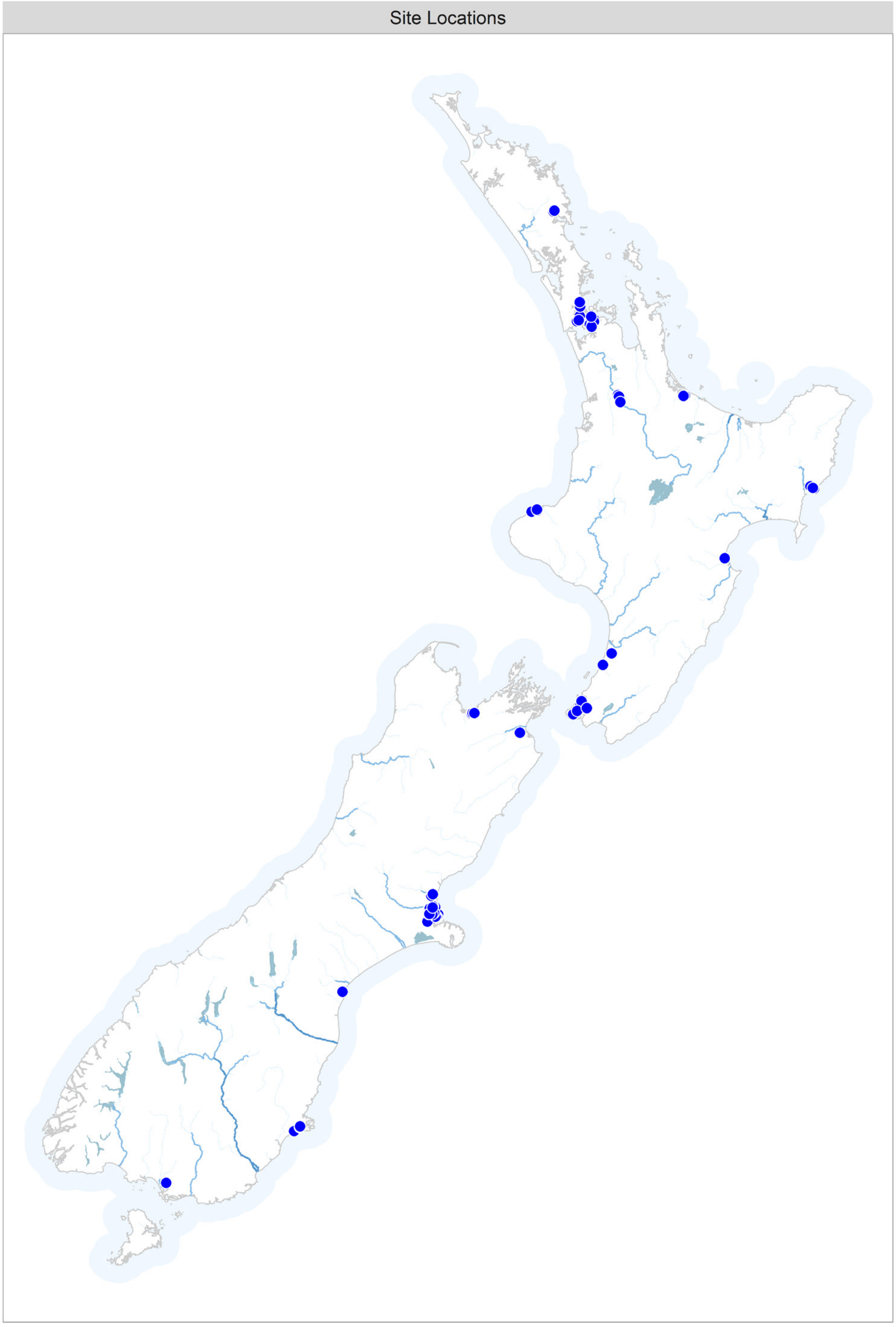

Figure 3-1: Location of sites for assessing the state of urban river and stream water quality.

3.1.2 Censored value methods to calculate summary statistics

Censored values are those above or below a detection limit (e.g., >2.5 or <0.001). Values below the detection limit(s) are described as left censored and values above the detection or quantitation limit are described as right censored. Censored values were replaced by imputation for the purposes of calculating the state statistics. Left censored values were replaced with imputed values generated using ROS (Regression on Order Statistics, Helsel 2012) following the procedure described in Larned et al. (2015). The ROS procedure produces estimated values for the censored data that are consistent with the distribution of the uncensored values and can accommodate multiple censoring limits.

Censored values above the detection limit were replaced with values estimated using a procedure based on "survival analysis" (Helsel 2012). A parametric distribution is fitted to the uncensored observations and then values for the censored observations are estimated by randomly sampling values larger than the censored values from the distribution. The survival analysis requires a minimum number of observations for the distribution to be fitted; hence in the case that there were fewer than 24 total observations, censored values above the detection limit were replaced with 1.1 times the detection limit. There was only one site and variable combination where that rule applied, for clarity of > 8 m and this was replaced with 8.8 m.

For dissolved copper, there were 18 sites where all data were censored and 19 sites where the proportion of censored values was between 72 and 97%, and for which the ROS procedure failed. These sites were all from Christchurch City and were based on a detection limit of 0.002g/m^3. This detection limit was higher than that used in Auckland and Wellington and above many of the measured concentrations in the Auckland and Wellington streams. These37 sites were removed from the data set for further analyses. There were five sites with censored data (2.8-61%) where the ROS procedure was used as there were 9-10 non-censored data points, sufficient for developing the ROS and estimating a median using imputation, and these sites were retained in the analyses. There were 13 sites with no censored values. The proportion of censoring for copper resulted in 20 sites being acceptable for analysis for dissolved copper compared to 55 sites for dissolved zinc.

Ammoniacal-N was the only other water quality variable where there were sites with greater than 50% censoring (5 sites, 53 to 92% censored). Imputation was used to calculate median values (and other percentiles) for three of these sites as described above For the remaining two (86-92% censored data) the ROS procedure failed, summary statistics were based on replacement of censored data with half of the detection limit (0.01 g/m^3), resulting in quantile estimates of 0.005 g/m^3 for quantiles from the 5th percentile to the 80th. The 95th percentile estimates were based on non-censored data. Although the procedure to calculate these quantile estimates was not as robust as the ROS procedure, the two sites were retained for further analysis, as the detection limit (and consequently the quantile estimates) was below measured concentrations at many locations (unlike that for copper).

Median (and other percentiles) that were calculated using imputation (or substitution, as for ammoniacal-N) are included in the analyses in Section 4 and on each plot these are indicated by a different colour. The database associated with this report indicates when the estimated concentrations at any quantile (e.g., 5th, 50th percentiles) are censored.

For each river site and variable, we characterised the current state using percentiles (5th, 20th, 25th, 50th, 75th, 80th, 95th) of the distribution of measured values for the period 2015 to 2017 (inclusive) calculated using the Hazen method[3]. For MCI we used the time period 1 July 2015 – 30 June 2017 to prevent splitting samples collected over summer into two calendar years. For *E. coli*, percentage exceedances over 260 and 540 cfu/100ml were also calculated.

3.1.3 Data analyses

We plotted the calculated percentile values to compare water-quality state against water quality guidelines, between sites, regions, REC classes and with varying levels of urban land cover. The state of urban stream water quality at each site is presented as median concentrations in most plots. For *E. coli* additional statistics were used to describe the state: 95th percentile and percentage exceedance of 260 and 540 cfu/100mL. A number of different data presentation methods are used for the state statistics including:

- point-range plots that indicate the median by a marker and percentile ranges by vertical lines;
- box plots that indicate the median of site medians (or other state statistics) by a horizontal line, the inter-quartile range (25-75%iles) of site medians by a box, the 1.5x interquartile range in site medians (or other state statistics) by whiskers and individual site medians as points;
- scatterplots that show the relationship between individual site medians (or other state statistics) and a continuous explanatory variable (urban land cover or impervious area).

Point range plots were used for the comparison between regions as many regions had too few sites (e.g., 1-3) for appropriate use of a box plot. All plots were constructed in R (R Core Team, 2018).

We used linear regression to relate water quality variables to proportions of urban land cover and proportions of impervious surfaces in the catchments of monitoring sites. All variable values with the exception of MCI were log-transformed to improve the normality of residuals. Regression analyses were also undertaken in R.

3.1.4 Guidelines used

In this assessment, we compared the state of water quality variables with relevant guidelines to provide context. Where possible, NPS-FM numeric attribute states for rivers were used as these are nationally relevant. The NPS-FM provides attributes for *E. coli* and two forms of nitrogen included in this assessment (nitrate-N and ammoniacal-N). The nitrate-N and ammoniacal-N thresholds are based on their toxicity and as such are generally much higher than concentrations associated with proliferations of periphyton and macrophytes. Thresholds relating to nutrient enrichment are not included in the NPS-FM at present.

The NPS-FM numeric attribute states are presented as four bands: A, B, C, and D, ranging from excellent water quality to poor. For this report, water quality in urban streams was compared with the national bottom line (threshold between C and D states) and thresholds between bands A, B and C. These thresholds are applicable to all streams in New Zealand. For assessment of *E. coli*, median *E. coli* concentration at each urban monitoring site was compared to the thresholds for secondary contact recreation (Table 3-2). For nitrate-N and ammoniacal-N, median concentrations at each

[3] (http://www.mfe.govt.nz/publications/water/microbiological-quality-jun03/hazen-calculator.html) Note that there are many possible ways to calculate percentiles. The Hazen method produces middle-of-the-road results, whereas the method used in Excel does not (McBride 2005, chapter 8).

urban monitoring site were compared to the national bottom line and thresholds between bands A, B and C based on median data (other thresholds are provided for the annual maxima (ammoniacal-N) or 95th percentiles (nitrate-N)).

The NPS-FM does not include attributes for dissolved copper or zinc. However, the ANZECC (2000) guidelines do provide trigger values for dissolved copper and zinc based on toxicity. These trigger values are provided at varying levels of protection (as % species protected) and are hardness-dependent. For this assessment, a 95% level of protection was used, as recommended for slightly to moderately disturbed systems. The default trigger values were used based on hardness concentrations of 30 g/m^3 as $CaCO_3$. A site median that exceeds the trigger value based on hardness of 30 g/m^3 as $CaCO_3$ does not signify that effects will occur –application of the ANZECC (2000) guidelines involves undertaking a further analysis to check whether this default guideline is appropriate.

Classifications for MCI scores are provided by Stark and Maxted (2007) and are appropriate for both hard-bottomed and soft-bottomed streams.

Table 3-2: Guideline values used in this assessment.

Water quality variable	Threshold type	Guideline value (units g/m^3, except *E. coli* cfu/100mL)	Source
Nitrate-N	Attribute state A	Annual median < 1.0	(New Zealand Government 2017)
	Attribute state B	Annual median < 2.4	
	Attribute state C	Annual median < 6.9	
	Attribute state D (below national bottom line)	Annual median > 6.9	
Ammoniacal-N	Attribute state A	Annual median < 0.03 [a]	(New Zealand Government 2017)
	Attribute state B	Annual median < 0.24 [a]	
	Attribute state C	Annual median < 1.3 [a]	
	Attribute state D (below national bottom line)	Annual median > 1.3 [a]	
Dissolved copper	95% protection, hardness 30 g/m^3 as $CaCO_3$	< 0.0014	(ANZECC & ARMCANZ 2000)
	80% protection, hardness 30 g/m^3 as $CaCO_3$	< 0.0025	
Dissolved zinc	95% protection, hardness 30 g/m^3 as $CaCO_3$	< 0.008	(ANZECC & ARMCANZ 2000)
	80% protection, hardness 30 g/m^3 as $CaCO_3$	< 0.031	
E. coli	Attribute state A, B, C (blue, green, yellow)	Annual median < 130	(New Zealand Government 2017)
	Attribute state D (orange)	Annual median < 260	
	Attribute state E (red)	Annual median > 260	
MCI	Excellent Good Fair Poor	119 100–119 80–99 <80	(Stark and Maxted 2007)

Notes: [a] Based on pH 8.0 and temperature of 20°C.

In order to compare against the thresholds for ammoniacal-N, data were converted to the equivalent concentration at pH 8.0, using the equation outlined in conversion ratios in the Draft Guide to Attributes in the NPS-FM (New Zealand Government 2014). These pH-adjustments were restricted to dates when both pH and ammoniacal-N were measured. These adjusted data are used only in the plot that compares data to guidelines (Figure 4-2) and raw ammoniacal-N data are used in all other plots to enable inclusion of data from Wellington (see Table 3-1).

3.2 Methods for trends assessment

3.2.1 Sampling dates, seasons and time-periods for analyses

In this study, trend analyses were carried out for each water quality variable × site combination that met the inclusion rules set out below. Trends in most water quality variables were analysed for 10 years, from January 2008 to December 2017. For MCI, we used the time period 1 July 2008 – 30 June 2017 where available, in order to capture complete summer sampling seasons. For metals, monitoring only commenced at many sites in 2011, so trends were analysed for 7 years (January 2011 to December 2017) at these sites (predominantly sites in Christchurch). Where monitoring commenced earlier, the 10 year time period was used. In addition, 22-year trends were assessed at five sites with long-term monitoring records, from January 1995 to December 2017.

The processed dataset had variable starting and ending dates, variable sampling frequencies, and variable numbers of missing values. Site inclusion rules (i.e., filtering rules) were used to ensure that for each variable, the data for each site would provide a robust assessment of the trend. We used the filtering rules suggested by Helsel and Hirsch (1992), which restricted site and variable combinations for trends in a given time period such that there were measurements for at least 90% of the years and at least 90% of seasons. All site by variable combinations that did not comply with these filtering rules were excluded from the analysis.

For assessments of trends in water quality variables other than MCI, we used seasons defined by months preferentially, and quarters when there were insufficient monthly observations. The trend analysis procedure accounted for seasonal variability in these monthly and quarterly data. Macroinvertebrates are generally sampled annually at SoE monitoring sites, so there is no seasonal variability in these data. For some sites and water quality variables, there was more than one sample within some seasons, and for some sites, MCI scores were available for more than one invertebrate sample within some years. In these cases, we used the median of the values for the season (or the year for the invertebrate samples) to ensure consistent statistical power across sites. We note that when there is more than one sample in a season, all samples can be used in a trend analysis resulting in increased statistical power and potentially different results. However, because our analyses are used to make regional comparisons and to contribute to spatial models, we elected to ensure that the site-specific analyses had consistent statistical power.

3.2.2 Analyses of site-specific trends

<u>Trend magnitude and confidence in trend direction</u>

The statistical analyses of water quality trends were performed using the LWP-Trends library, which comprises functions coded in the R statistical programming language. The statistical analyses of trends involve the evaluation of (1) the magnitude of the trend and (2) the confidence in the trend direction.

Trend magnitude was characterised by the Sen slope estimator (SSE; Hirsch et al. 1982). The SSE is the slope parameter of a non-parametric regression, which is calculated as the median of all possible inter-observation slopes (i.e., the difference in the measured observations divided by the time between sample dates; see Figure 3-2).

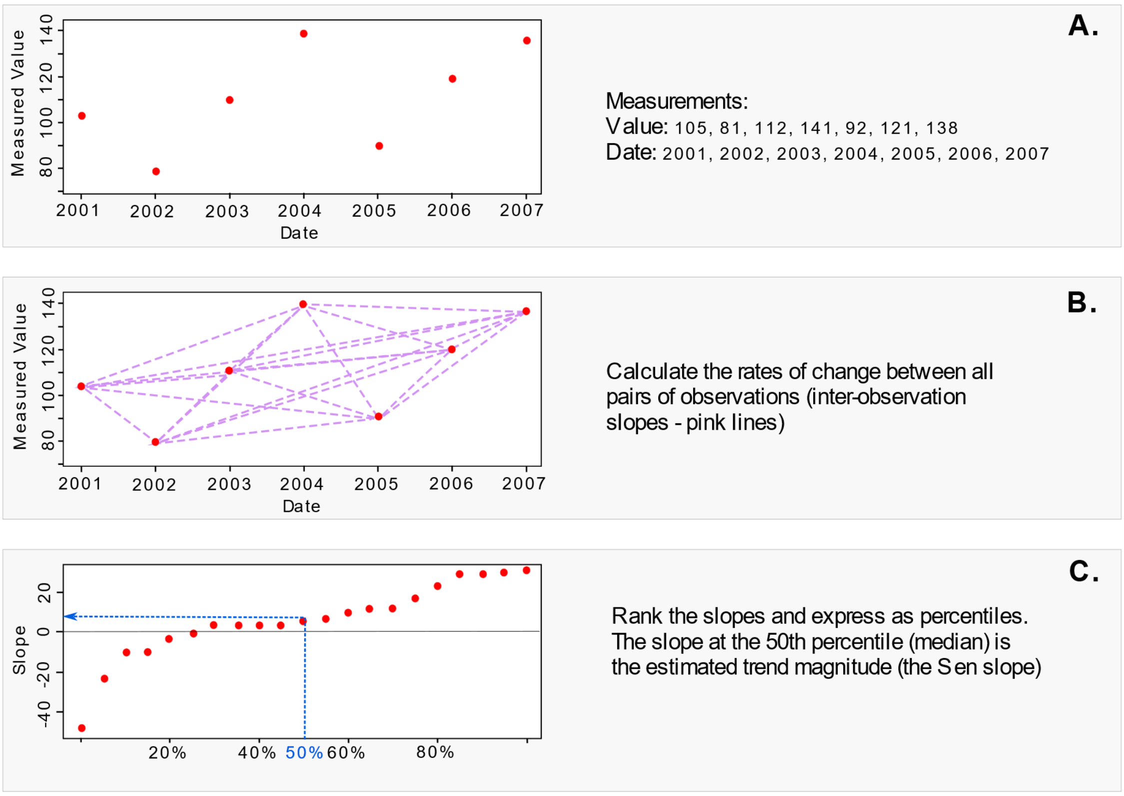

Figure 3-2: Pictogram of the steps taken in the trend analysis to calculate the Sen slope, which is used to characterise trend magnitude in the time-series of data for each site × variable combination.

The seasonal version of the SSE is used in situations where there are significant ($p \leq 0.05$, as evaluated using a Kruskall Wallis test) differences in water quality measurements between 'seasons'. As noted above, seasons are defined primarily by sampling intervals, which were monthly or quarterly for all variables except MCI. The seasonal Sen slope estimator (SSSE) is the median of all inter-observation slopes within each season. Trend magnitudes for the sites and variables that demonstrated significant seasonality were estimated with SSSE. Trend magnitudes in annual MCI scores and for sites and variables that did not demonstrate significant seasonality were analysed with SSE. For example, at sites where DRP showed seasonality, the SSSE was estimated, but at sites where seasonality was not significant for DRP, SSE was estimated.

The Kendall test S and *p*-values are used by the LWP-Trends library to establish confidence in the trend direction (rather than using the Sen slope and its confidence intervals as used by Larned et al. 2015; the reasons for which are related to treatment of censored values and discussed in the following section). The Kendall test measures the rank correlation, which is a nonparametric correlation coefficient measuring the monotonic association between two variables, x and y. In water quality trend analysis, y is a sample of water quality measurements and x is the corresponding sample dates. Traditionally, the Kendall test is used to determine whether trends are statistically "significant" or "insignificant" (see Figure 3-3).

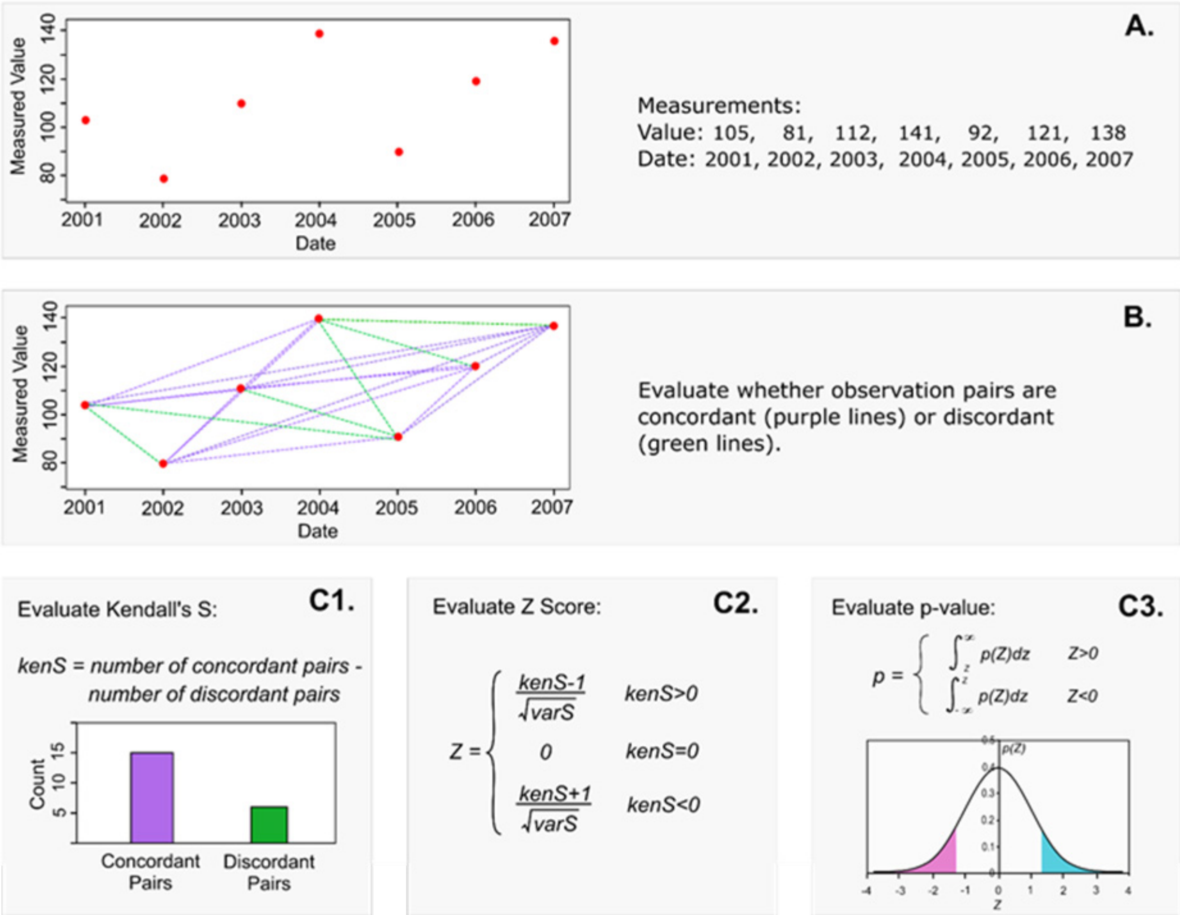

Figure 3-3: Pictogram of the steps taken in the trend analysis to calculate the Kendal S statistic and its p-value, which is used to characterise the confidence in trend direction.

In the LWP-Trends library and in the current report, confidence in the direction of each trend was evaluated by interpreting the Kendall *p*-value as a **probability** that the trend was decreasing as follows:

$$P(S < 0) = 1 - 0.5 \times pvalue$$

$$P(S > 0) = 0.5 \times pvalue,$$

where $pvalue$ is the *p*-value returned by Kendall test (either seasonal or non-seasonal), *S* is the S statistic returned by Kendall test (either seasonal or non-seasonal) and *P* is the probability that the trend was decreasing. The trend direction is interpreted as decreasing when P > 0.5 and increasing when P < 0.5. Note that if data are seasonal (i.e., Kruskall Wallis test p ≤ 0.05), a seasonal version of the Kendall test is used to evaluate the $pvalue$ and P.

The trend direction is established with a 95% level of confidence if the probability associated with S < 0 (i.e., a decreasing trend) is ≥ 95%, or the probability associated with S > 0 (i.e., an increasing trend) s ≤ 5%. In both, these cases the trend is categorised as 'established with confidence' and when the probability the trend is decreasing is between the 90% confidence limits (i.e., is ≥5% and ≤95%), the trend is categorised as 'insufficient data".

For some site × variable combinations, trends were not analysed for one of two reasons:

1) When a large proportion of the values were censored (data has <5 non-censored values and/or <3 unique non-censored values). This situation results in so many ties that there is little information content in the data and a meaningful statistic cannot be calculated.

2) When there is no, or very little variation in the data (<3 unique, non-censored values), because this also results in ties. This can occur because laboratory analysis of some variables has low precision (i.e., values have few or no significant figures). In this case, many samples have the same value resulting in ties.

These conditions occurred for dissolved copper at 15 sites.

3.2.3 Handling censored values

Censored values in the data used to calculate Kendall's S and its *p*-value were robustly handled in the manner recommended by Helsel (2005; 2012). Briefly, for left-censored data, increases and decreases in a water quality variable were identified whenever possible. Thus, a change from a censored data entry of <1 to a measured value of 1 was considered an increase. A change from a censored data entry of <1 to a measured value 0.5 was considered a tie, as was a change from <1 to a <5, because neither can definitively be called an increase or decrease. Similar logic applied to right censored values. The information about ties was used in the calculation of the Kendall S statistic and its variance and this provided for a robust calculation of the *p*-value associated with the Kendall test.

Note that as the proportion of censored values increases, the proportion of ties increases and the confidence in the trend direction decreases. Therefore, site and variable combinations with high proportions of censored observations tend to be categorised as 'insufficient data".

The inter-observation slope cannot be definitively calculated between any combination of observations in which either one or both are censored. Therefore, when SSE and SSSE (i.e., Sen slopes) are calculated by the LWP-Trends library, the censored data entries are replaced by their corresponding raw values (i.e., the numeric component of a censored data entry) multiplied by a factor (0.5 for left-censored and 1.1 for right-censored values). This ensures that any measured value that is equal to a raw value is treated as being larger than the censored value if it is left-censored value and smaller than the censored value if it is right-censored. The inter-observation slopes associated with the censored values are therefore imprecise (because they are calculated from the replacements). However, because the Sen slope is the median of all the inter-observation slopes, the Sen slope is unaffected by censoring when a small proportion of observations are censored. As the proportion of censored values increase, the probability that the Sen slope is affected by censoring increases.

Helsel (1990) estimated that the impact of censored values on the Sen slope is negligible when fewer than 15% of the values are censored. However, this is a rule of thumb and is not always true. Depending on the arrangement of the data, a small proportion of censored values (e.g., 15% or less) could affect the computation of a Sen slope (Helsel 2012). To provide information about the robustness of the SSE and SSSE values, the supplementary output for every trend analysis includes the proportion of observations that were censored and whether the Sen slope (i.e., the median of all inter-observation slopes) was calculated from observations that were censored. The magnitudes (i.e., the SSE and SSSE values) of individual site trends calculated from observations with many censored values should be considered less precise than those with fewer censored values. In addition, when there are censored values, greater confidence should be placed in the statistics returned by the Kendall tests (including the trend direction and the probability the trend was decreasing).

3.2.4 Differences in trend analysis methods to previous reports

The general approach to trend analyses in this study is consistent with the approach used by Larned et al. (2015 and 2016) and described by McBride (in press). Each of these studies has assessed whether there are monotonic changes in the central tendencies of water quality values through time and have used the Sen slope estimator as to characterise the magnitude of these changes. In addition, statistical significance tests were replaced in each study with evaluations of the confidence in the trend direction; this advancement distinguished the studies by Larned et al. (2015, 2016) from previous national-scale water-quality trend analyses (e.g.,Ballantine et al. 2010). However, some steps in the trend analysis procedures used in this study differ from all of the previous studies; most of these differences concern improved methods for handling censored values.

In the studies by Larned et al. (2015, 2016), confidence in trend directions were evaluated using the Sen slope confidence intervals. If the symmetric confidence intervals around a Sen slope did not contain zero, the trend direction (either positive of negative) was classified as 'established with confidence'. If the symmetric confidence intervals around did contain zero, it was concluded that there were "insufficient data" to determine the trend direction at the nominated level of confidence. Note that if two symmetric, one-sided 90% confidence intervals do not contain zero, the trend direction is established with 95% confidence. The reasons for this are explained by Larned et al. (2015) and McBride (in press). For the same reason, the analysis used in the current study categorises a trend as 'established with confidence' at 95% confidence when the probability that the trend is decreasing is ≤5% or ≥95%, and as 'insufficient data" when the probability lies between these 90% confidence limits.

We recently identified a problem with the use of the Sen slope confidence intervals to make inferences about trend directions, which is based on inadequate treatment of censored values. The problem includes both the potentially imprecise estimate of the Sen slope (discussed above) and its confidence intervals. Analytically the difference between a pair of censored values is not measurable and must be treated as zero, which is referred to as a 'tie'. Equally, the difference between a measured value that is less than the raw value of a censored value and that censored value is not measurable, and is also considered a tie. Replacement of censored values with imputed values can affect the identification of tied values, which reduces the robustness of the calculations of the confidence interval. While the imputation of censored values by Larned et al. (2015) was not strictly correct, the rule in that study that restricted site × variable combinations to those with < 15% censored values ensured that imputation *per se* had minimal effects on estimates of trend magnitude or confidence intervals.

The approach used with censored values in this study has two advantages compared with the previous studies. First, evaluations of confidence in trend directions for individual sites are reliable, irrespective of the proportion of censored observations. In turn, the methods used to aggregate site trends are robust, because these procedures are based on levels of the confidence in the trend directions at individual sites (discussed in detail in Section 3.2.7). Second, censored values can represent a large proportion of observations for some variables (e.g., ammoniacal-N). The analyses in this study did not need to restrict site and variable combinations based on the proportion of censored observations (i.e., sites with >15% censored values were not discarded). This has the advantage of preserving a larger number of sites in the analysis and maximising their spatial coverage.

3.2.5 Interpretation of trend directions

Results of the trend analyses were used to classify the trends for all site × variable combinations into four trend direction categories: improving, degrading, indeterminant and not analysed. An increasing or decreasing trend category was assigned when the when probability ≥95% or ≤5% (i.e., the trend direction is established with confidence; Larned *et al.*, 2016). An "indeterminant" trend category was assigned when the when probability ≤95% and ≥5%; (the trend direction was not defined with confidence; Larned *et al.*, 2016).

3.2.6 Presentation of trend results

Trend results are presented in this report for individual sites using the Relative Sen slope estimate (RSSE) or the Relative seasonal Sen slope estimate (RSSSE), depending on whether the data demonstrated seasonal trends or not. Both measures are calculated from the SSE or SSSE divided by the median concentration and therefore provide a measure of the percent annual change.

3.2.7 Aggregation of site trends

The aggregated results of analysis of water-quality trends are intended to provide an overview of recent water quality changes over a spatial domain of interest (e.g., an environmental class). Traditionally, tables enumerating site trends by trend-direction categories (i.e., increasing, decreasing, and indeterminant) have been used. In this study, two new approaches to aggregating sites trends have been used to provide overviews of recent water quality changes (Snelder and Fraser 2018).

The first approach utilises the probability that the true trend was decreasing, which is derived from the Kendall test statistics (see Section 3.2.2). This probability facilitates a more nuanced inference rather than the 'yes/no' output corresponding to the trend-direction categories (i.e., increasing, decreasing, and indeterminant (McBride in press)). Confidence categories can be used to express the probability that a trend is improving (or its complement; degrading). Note that the conversion of the probability that a trend is decreasing to the probability it is improving (and its complement, degrading) depends on whether decreasing values represent improvement or degradation and differs between variables.

This study has used the approach to presenting categorical levels of confidence recommended by the Intergovernmental Panel on Climate Change (IPCC; Stocker et al. 2014). The narrative descriptions of confidence associated with different probabilities are shown in Table 3-3. Note that descriptions of the probabilities of degrading trends are the complement of the confidence categories in Table 3-3, i.e. an "exceptionally unlikely" degrading trend is the same as a "virtually certain" improving trend.

Table 3-3: Level of confidence categories used to convey the probability that water quality was improving. The confidence categories are those used by the Intergovernmental Panel on Climate Change (IPCC; Stocker et al. 2014).

Categorical level of confidence	Probability (%)
Virtually certain	99–100
Extremely likely	95–99
Very likely	90–95
Likely	67–90
About as likely as not	33–67
Unlikely	10–33
Very unlikely	5–10
Extremely unlikely	1–5
Exceptionally unlikely	0–1

The categorical levels of confidence presented in Table 3-3 were used to aggregate the site data describing the likelihood that water quality was improving for each variable. Each site trend was assigned a categorical level of confidence that the trend was improving according to its evaluated probability and the categories shown in Table 3-3. For the chemical and microbiological water quality measures (Table 2-1), improvement is indicated by decreasing trends (i.e. decreasing concentrations). For MCI scores and visual clarity, improvement is indicated by increasing trends.

We calculated the proportion of sites in each level of confidence category for each variable and present these values as colour coded bar charts. These charts were produced using all available sites (i.e., national-scale aggregation). Graphical presentations were not produced for other site groupings in this study because we considered that the proportion of improving trends (PIT) statistics were a simpler way to represent grouped aggregate trends.

The second approach also utilises the probability that the true trend was decreasing to provide a probabilistic estimate of the proportion of improving trends (PIT) within a domain of interest. For a given water quality variable, the trends at several monitoring sites distributed across a domain of interest can be assumed to represent independent samples of the population of trends, at all sites within that domain. Let the sampled sites within this domain be indexed by s, so that $s \in \{1, \dots, S\}$ and let I be a random Bernoulli distributed variable that takes the value 1 with probability p and the value 0 with probability $q = 1 - p$. Therefore, $I_s = 1$ denotes an improving trend at site $s \in \{1, \dots, S\}$ when the estimated $p_s \geq 0.5$ and a degrading trend as 0 when $p_s < 0.5$. Then, the estimated proportion of sites with improving trends in the domain is:

$$PIT = \sum_{s=1}^{s=S} I_s / S$$

Because the variance of a random Bernoulli distributed variable is $Var(I) = p(1-p)$, and assuming the site trends are independent, the estimated variance of PIT is:

$$Var(PIT) = \frac{1}{S^2} \sum_{s=1}^{s=S} Var(I_s) = \frac{1}{S^2} \sum_{s=1}^{s=S} p_s(1-p_s)$$

PIT and its variance represent an estimate of the population proportion of improving trends and the uncertainty of that estimate. It is noted that the proportion of degrading trends is the complement of the result (i.e., 1 – PIT). The estimated variance of PIT can be used to construct 95% confidence intervals[4] around the PIT statistics as follows:

$$CI_{95} = PIT \pm 1.96 \times \sqrt{Var(PIT)}$$

We calculated PIT and its confidence interval for all water quality variables and for domains of interest defined by the entire country, by region, by REC classes as defined in Section 2.2.3 and by the proportion of urban land use in the catchment.

[4] Note that +/- 1.96 are approximately the 2.5th and 97.5th percentile of a standard normal distribution.

4 Water Quality State

4.1 State of urban river and stream water quality

The state of urban stream water quality at each site is summarised in Figure 4-1 to Figure 4-4 by the median concentrations (ordered from lowest to highest) and range (from 5th to 95th percentiles) to show the variation in concentrations of each water quality variable within and between sites. Where nationally-accepted water quality standards or guidelines are available, these are indicated on the plots. For copper there were fewer site medians as many sites had insufficient data above the detection limit to enable calculation of a site median, while for clarity there were few sites where this variable was measured.

For some of the water quality variables (dissolved zinc, DRP, nitrate-N, turbidity and *E. coli*), there was a two order-of-magnitude range between the lowest and highest site medians. For ammoniacal-N the range covers nearly three orders-of-magnitude, from 0.0009 mg N/L to 0.56 mg N/L. There was less variation between sites for dissolved copper, TP, TN, and clarity. For DRP, most site medians were between 0.005 and 0.05 mg P/L whereas 5 sites formed a distinctive cluster with concentrations measuring 0.20-0.35 mg P/L. There was a similar pattern for TP with most site medians between 0.012 and 0.1 mg P/L and 6 sites above this, spread between 0.15-0.42 mg P/L. The sites with higher concentrations are not all from the same region or REC class (see Figure 4-5 and Figure 4-8).

Within site variation was also high for dissolved zinc, ammoniacal-N and *E. coli* with up to an order of magnitude differences between the lower (25th) and upper (75th) quartiles for most sites. High within site variation was also shown for nitrate-N and clarity at some sites but not at the majority of sites. The wide range in site medians demonstrates that although water quality can be poor in some urban streams, there are also urban streams where water quality is not poor. The within-site ranges were substantially smaller for dissolved copper, DRP and nitrate-N.

56 out of 75 sites (~75% of sites) were in the "Red" attribute state for *E. coli*, where site median exceeded 260 *E. coli* /100 mL. A further 14 sites (19% of sites) were in the "Orange" attribute site and only 5 sites (7%) were in either the Blue to Yellow attribute states, with median concentrations <130 /100mL (shown as green in the plot). Note that categorisation to Blue, Green or Yellow attribute state also depends on the 95th percentile and percentage exceedances of 260 and 540 /100mL. No sites exceeded the "National Bottom Line" thresholds for ammoniacal-N or nitrate-N. More than 50% of sites for these variables were in attribute state "A", and 100% and 95% of sites for ammoniacal-N and nitrate-N, respectively, were in either "A" or "B" state. Around 42% of sites had median zinc concentrations greater than the ANZECC (2000) default guideline for protection of 95% of species (yellow or orange shading); and 22% of sites had median copper above this guideline. However, for both metals, at most sites the 75th percentile concentrations exceeded this guideline. The median concentrations at one site for copper and four sites for zinc exceeded the guideline for protection of 80% of species, shown by the orange shading. Almost all MCI scores for the streams were rated as either "Poor" (19 sites, 48%) or "Fair" (18 sites, 46%). Only one site was rated as "Good" and none were rated as "Excellent" quality.

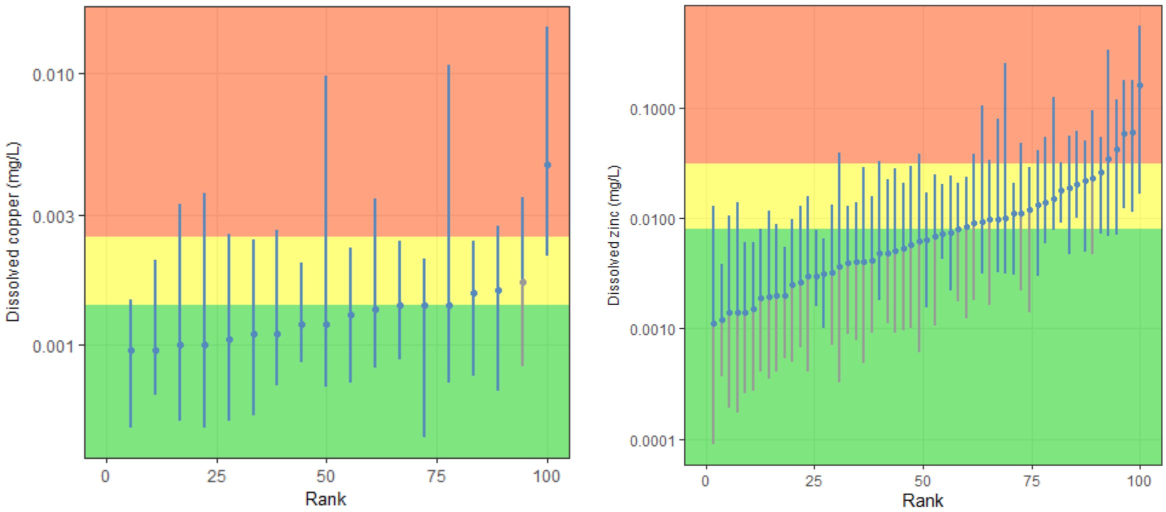

Figure 4-1: Dissolved metal concentrations for each urban river and stream site ordered by median values and compared to water quality guidelines. Marker indicates median value for each site and bars represent the range from 5th to 95th percentiles. Markers and bars in grey indicate imputed data (i.e., grey marker indicates median value is based on imputation and grey bar indicates 5th percentile is based on imputation). Background shading represent bands based on ANZECC (2000) guideline values for dissolved copper and zinc for protection of 95% and 80% of species at hardness of 30 g/m^3 as $CaCO_3$: green is less than the 95% guideline value; yellow is between the 95% and 80% values; red is greater than the 80% value. Note log-scale on y-axes.

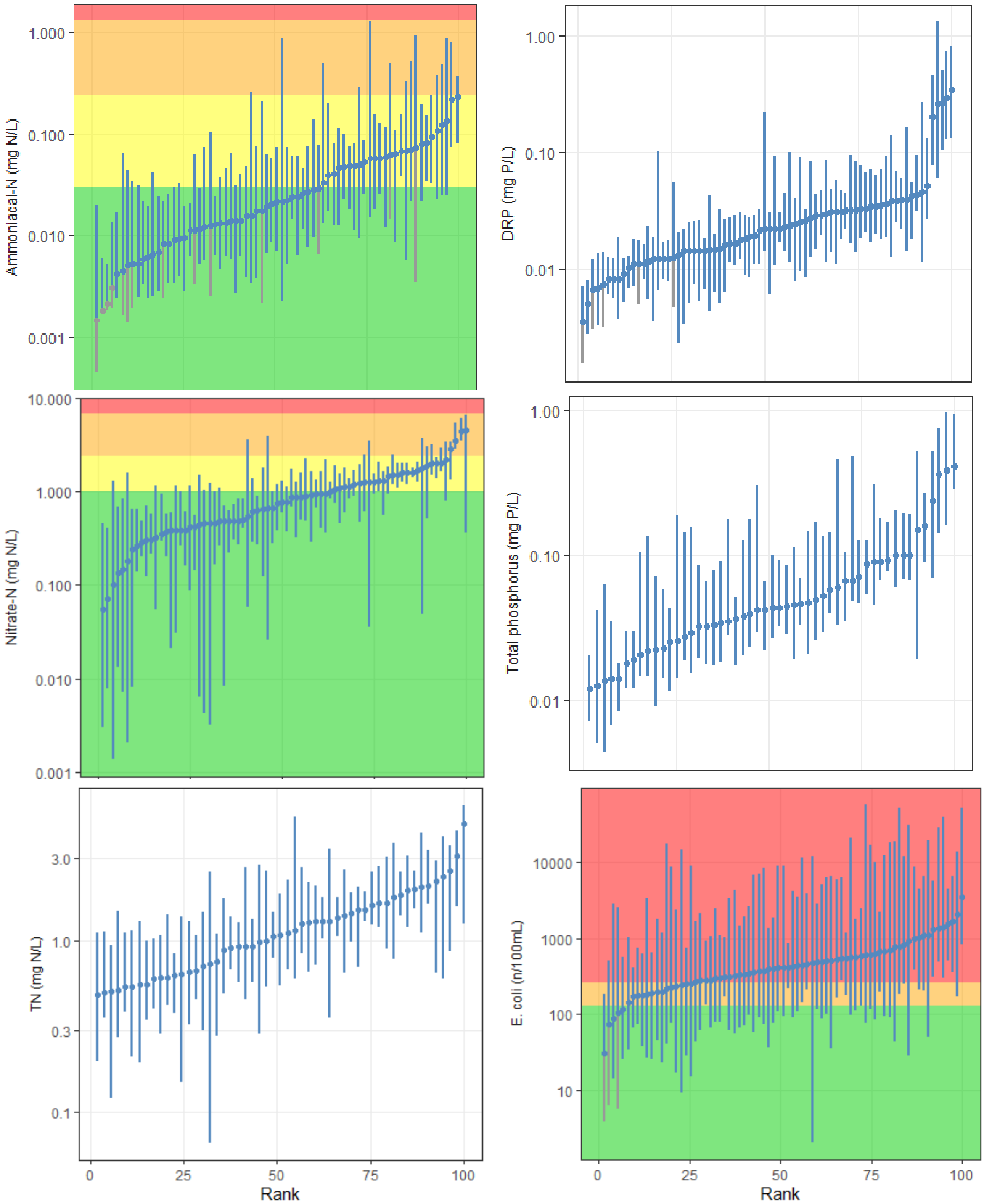

Figure 4-2: Water quality in urban river and stream sites ordered by median values and compared to water quality guidelines where national guidelines apply. Marker indicates median value for each site and bars represent the range from 5th to 95th percentiles. Markers and bars in grey indicate imputed data (i.e., grey marker indicates median value is based on imputation and grey bar indicates 5th percentile is based on imputation). Background shading for nitrate-N, ammoniacal-N and *E. coli* represents NPS-FM bands: green is A, yellow is B, orange is C and red is D, except for *E. coli* where green is C or better, orange is D and red is E. Note log-scale on y-axes. Ammoniacal-N is adjusted to pH 8.0 for comparison to NOF attributes.

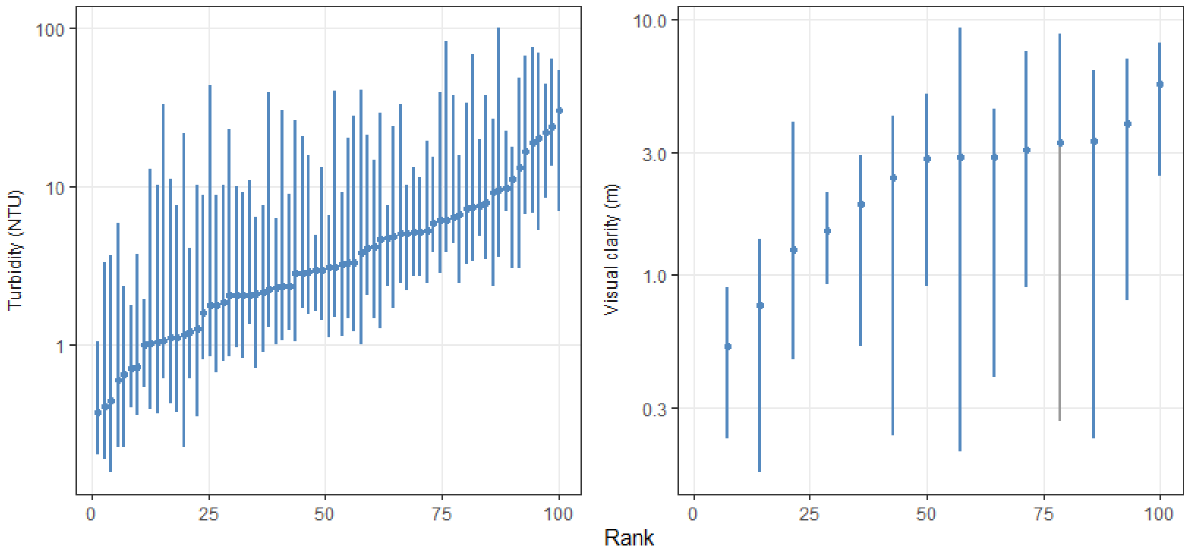

Figure 4-3: Turbidity and clarity for each urban river and stream site ordered by median values. Marker indicates median value for each site and bars represent the range from 5th to 95th percentiles. Markers and bars in grey indicate imputed data (i.e., grey marker indicates median value is based on imputation and grey bar indicates 5th percentile is based on imputation). Note log-scale on y-axes.

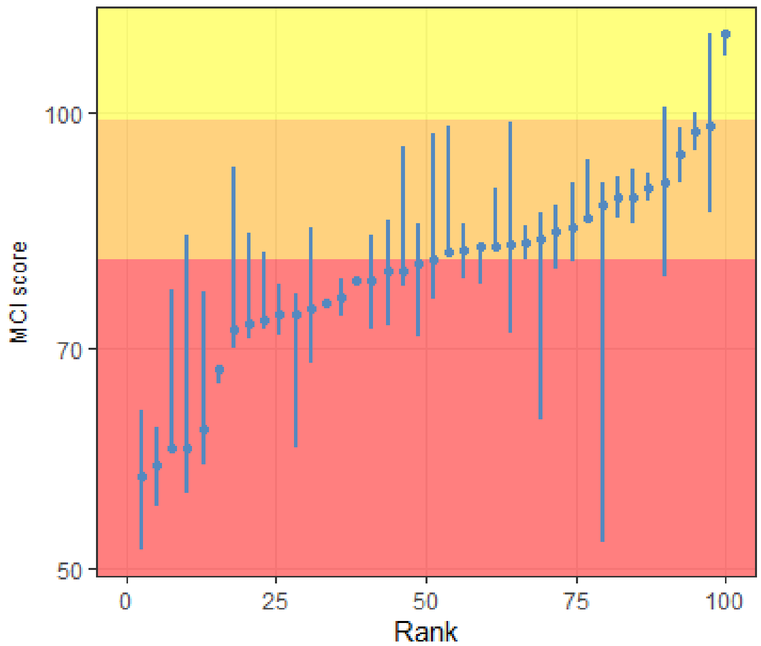

Figure 4-4: MCI score for each urban river and stream site ordered by median values. Marker indicates median value for each site and bars represent the range from 5th to 95th percentiles. Background shading indicates stream condition ranges: yellow is good; orange is fair; and red is poor.

4.2 State categorised by region

Water quality varied considerably between regions and also within regions, where there are multiple sites in a region (

Figure 4-5 to Figure 4-7). For the majority of parameters, this within-region variation was most pronounced for the Canterbury Region (ECAN), possibly due to the greater number of sites included for this region (e.g., 41 sites for ammoniacal-N, see Table 3-1). Sites in Marlborough (MDC), Nelson (NCC), Northland (NRC) and Otago (ORC) appeared to have low concentrations of ammoniacal-N, DRP, TP, TN and turbidity compared to other regions (
Figure 4-5). Sites in the Waikato Region (WRC) had higher ammoniacal-N and turbidity than almost all sites in other regions.

The median *E. coli* concentrations were highest in Auckland, Canterbury, Southland, Waikato and Hawkes Bay regions, over 1000 /100mL. These regions, and Nelson, also had sites with the highest 95^{th} percentiles, and sites in Auckland, Canterbury, Waikato and Hawkes Bay had the highest percent exceedances. Manawatu-Whanganui (HRC) and Northland (NRC) had sites with the highest median MCI scores, while the sites with lowest median scores were found in Canterbury (ECAN), Southland (ES), Nelson, and Otago (Figure 4-7). There were too few sites for dissolved metals or visual clarity to comment on any differences between regions.

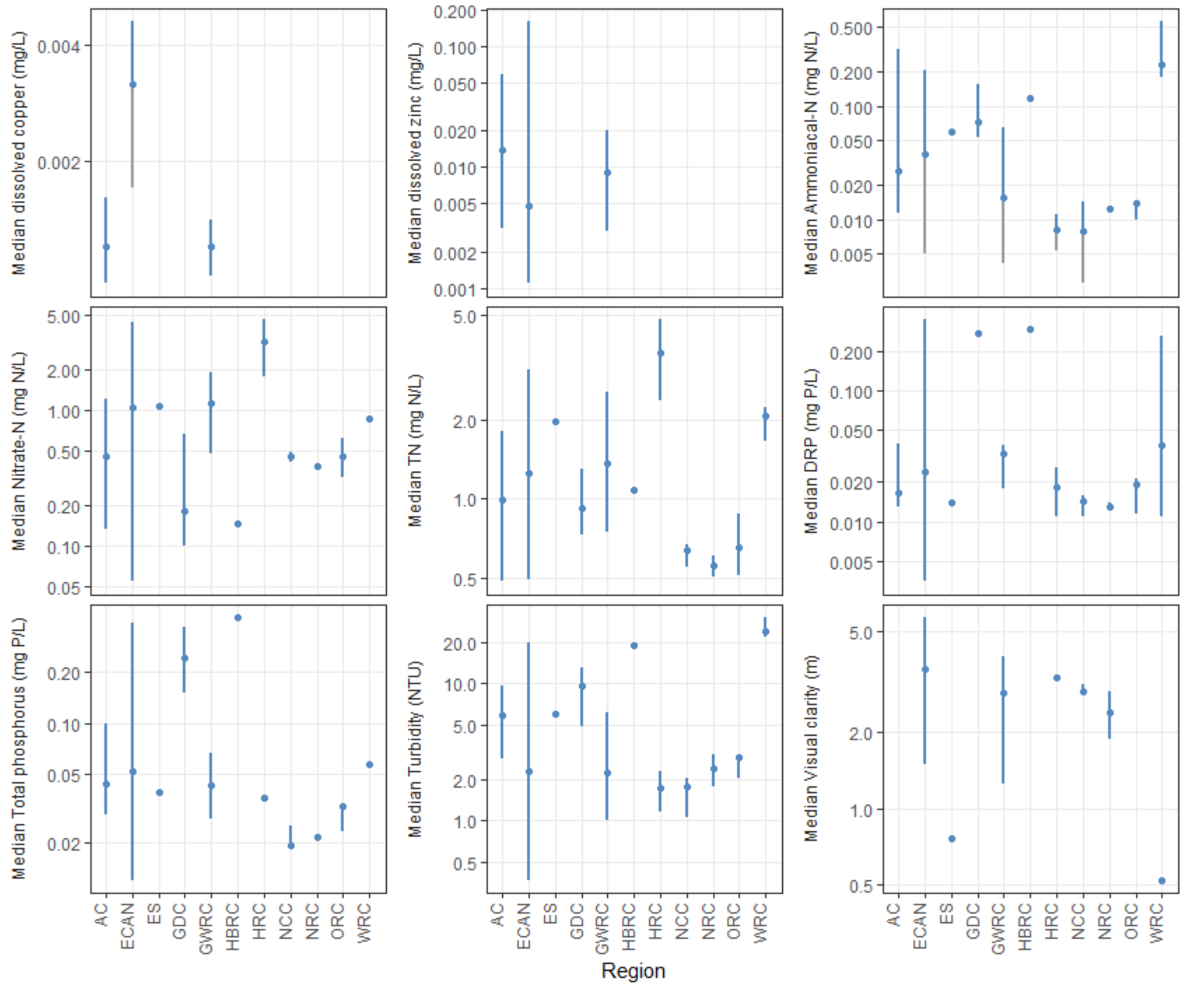

Figure 4-5: Median and range of site medians in water quality variables in urban river and streams categorised by region. Marker indicates median of site medians and bars represent range (minimum and maximum of site medians). Markers and bars in grey indicate imputed data (i.e., grey marker indicates the median site median was based on imputation and grey bar indicates the minimum site median was based on imputation). Note log-scale on y-axes.

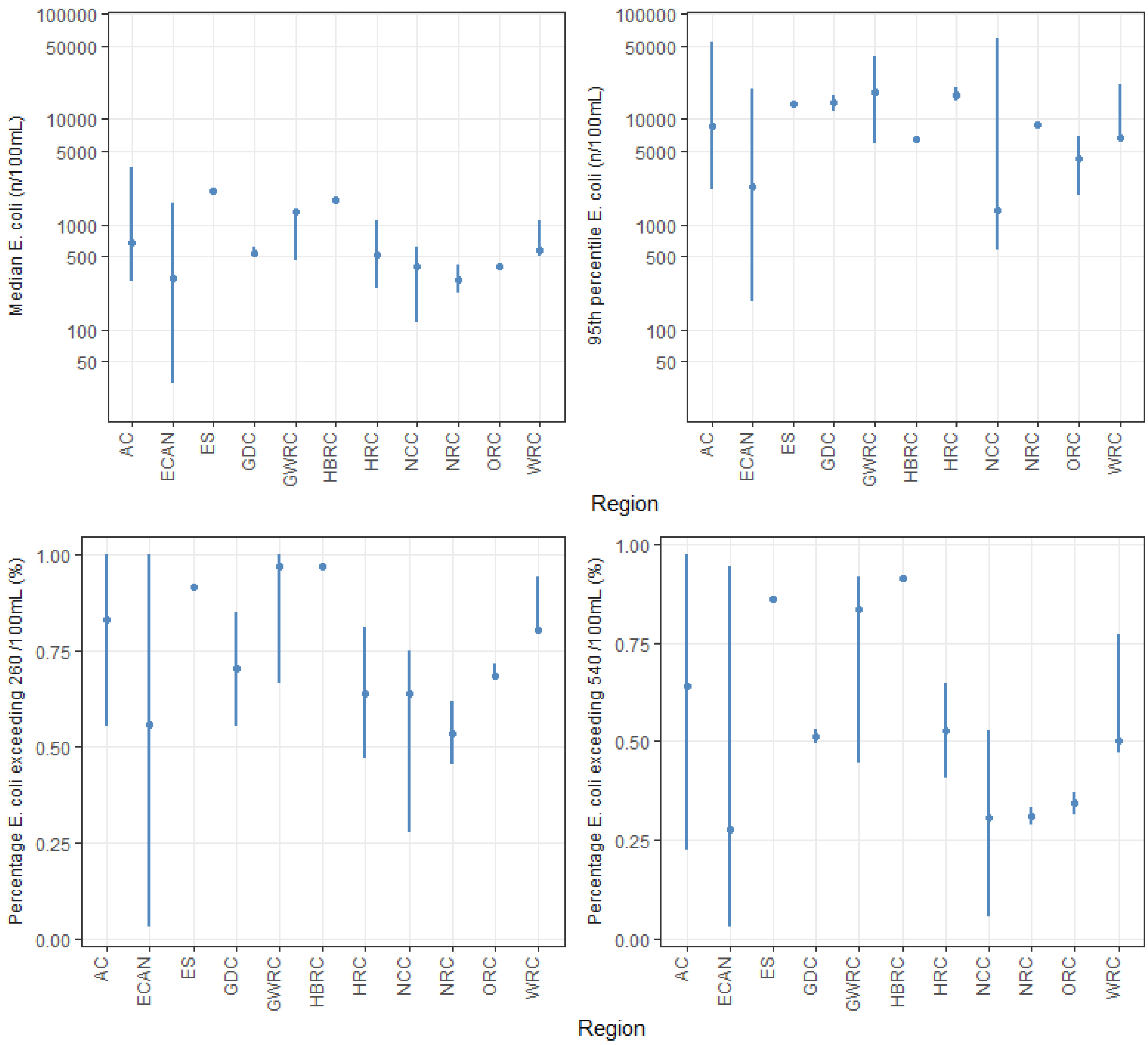

Figure 4-6: ***E. coli* metrics in urban rivers and streams categorised by region.** Marker indicates median of the relevant metric and bars represent range (minimum and maximum of site metric). No site medians for *E. coli* were based on imputation. Note log-scale on y-axes for median and 95th percentiles.

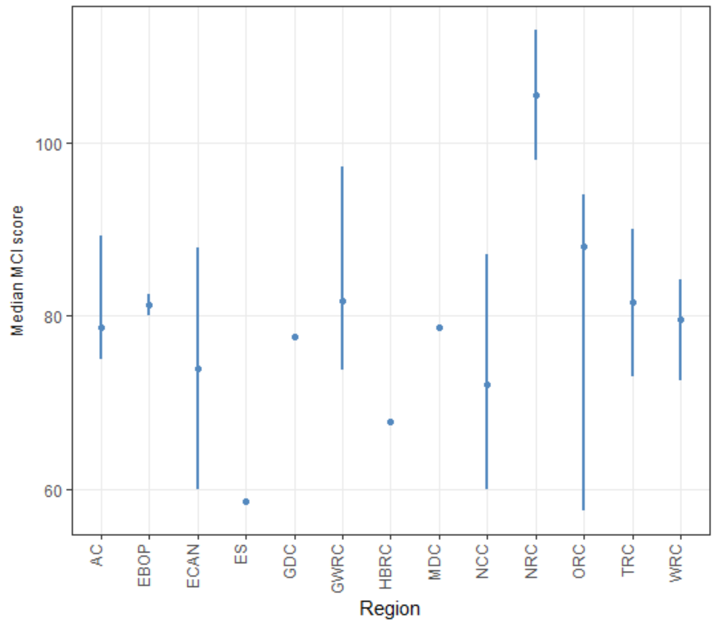

Figure 4-7: Median MCI in urban river and streams categorised by region. Marker indicates median of site medians and bars represent range (minimum and maximum of site medians).

4.3 State categorised by REC class

The following boxplots present the state of water quality variables categorised by the REC class, based on the REC climate and source of flow classification (Figure 4-8 to 4-11). Each plot shows the distribution of median concentrations of all sites in the given REC class as follows: median (horizontal line), the inter-quartile range (25-75%iles: upper and lower bounds of box), the 1.5x interquartile range (whiskers) and individual data (points).

There appear to be some differences in the water quality state between REC classes for some variables but not others (Figure 4-8). For example, median nitrate-N was more frequently higher and dissolved zinc was more frequently lower in the CD_L class than in other classes. Turbidity was highest in the WD_L class (warm dry lowland) and lowest in the CW_L class. In contrast, for DRP there was very little difference between REC classes, with most classes having a very similar overall median and inter-quartile range. There were too few sites for visual clarity to comment on any differences between REC classes.

Median *E. coli* concentrations were more frequently lower in the REC class of CD_L for all four metrics and more frequently higher in the WD_L REC class. MCI scores were lowest in the CD_L REC class and highest in the WW_L.

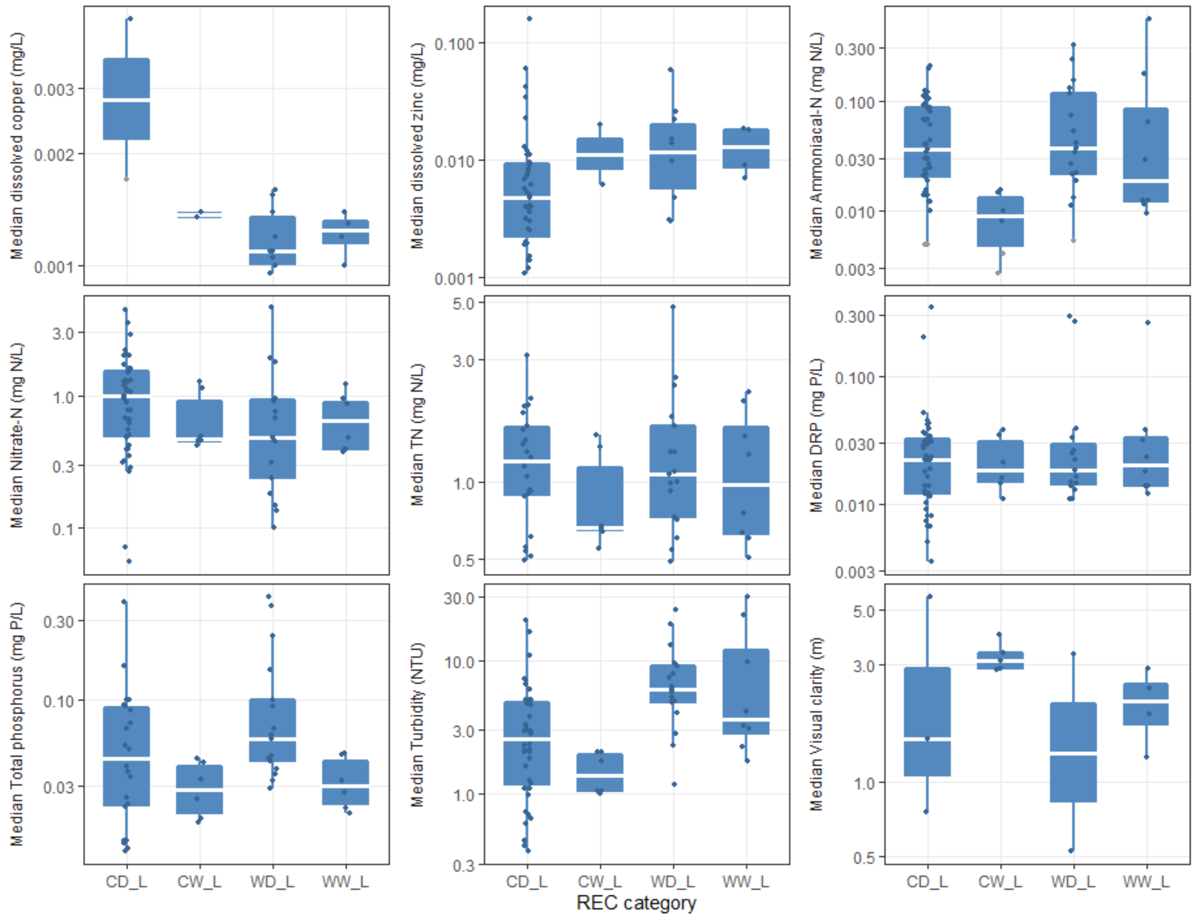

Figure 4-8: Distribution of site median concentrations for water quality variables in urban river and streams categorised by REC class. White horizontal line in each box indicates the median of site medians, box indicates the inter-quartile range, whiskers indicate the 5th and 95th percentiles. Individual site medians indicated by dots. Grey marker indicates that site median was based on imputation. Note log-scale on Y-axes. CD_L = cool dry climate, lowland source; CW_L = cool wet climate, lowland source; WD_L = warm dry climate, lowland source; WW_L = warm wet climate, lowland source.

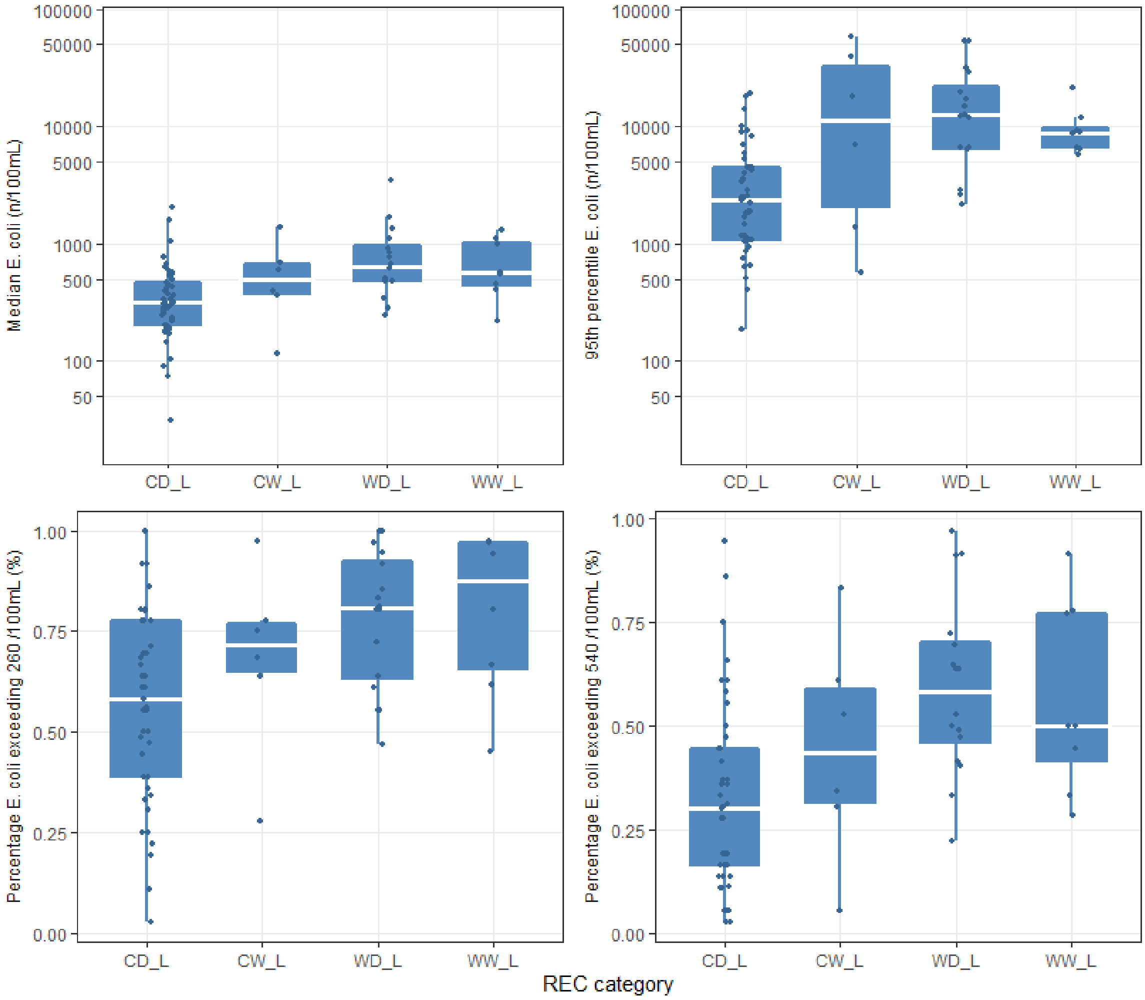

Figure 4-9: Distribution of *E. coli* metrics in urban river and streams categorised by REC class. White horizontal line in each box indicates the median of site medians, box indicates the inter-quartile range, whiskers indicate the 5th and 95th percentiles. Individual site medians indicated by blue dots (no site medians were based on imputation). Note log-scale on Y-axes. CD_L = cool dry climate, lowland source; CW_L = cool wet climate, lowland source; WD_L = warm dry climate, lowland source; WW_L = warm wet climate, lowland source.

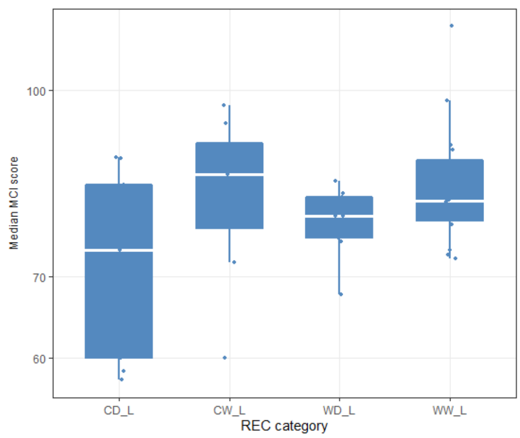

Figure 4-10: Distribution of median MCI in urban river and stream sites categorised by REC class. White horizontal line in each box indicates the median of site medians, box indicates the inter-quartile range, whiskers indicate the 5th and 95th percentiles. Individual site medians indicated by blue dots. Note log-scale on Y-axes. CD_L = cool dry climate, lowland source; CW_L = cool wet climate, lowland source; WD_L = warm dry climate, lowland source; WW_L = warm wet climate, lowland source.

4.4 State categorised by urban land cover

The following plots (Figure 4-11 to Figure 4-13) present the state of water quality variables categorised by the percentage of urban land cover in the upstream catchment. The data suggest some relationships of increasing urban land cover and declining water quality state, for example for dissolved zinc and ammoniacal-N. For all variables, the category with the lowest proportion of urban land cover (15-30%) had the poorest water quality (lowest median of the site median concentrations for most variables, highest median for clarity and MCI).

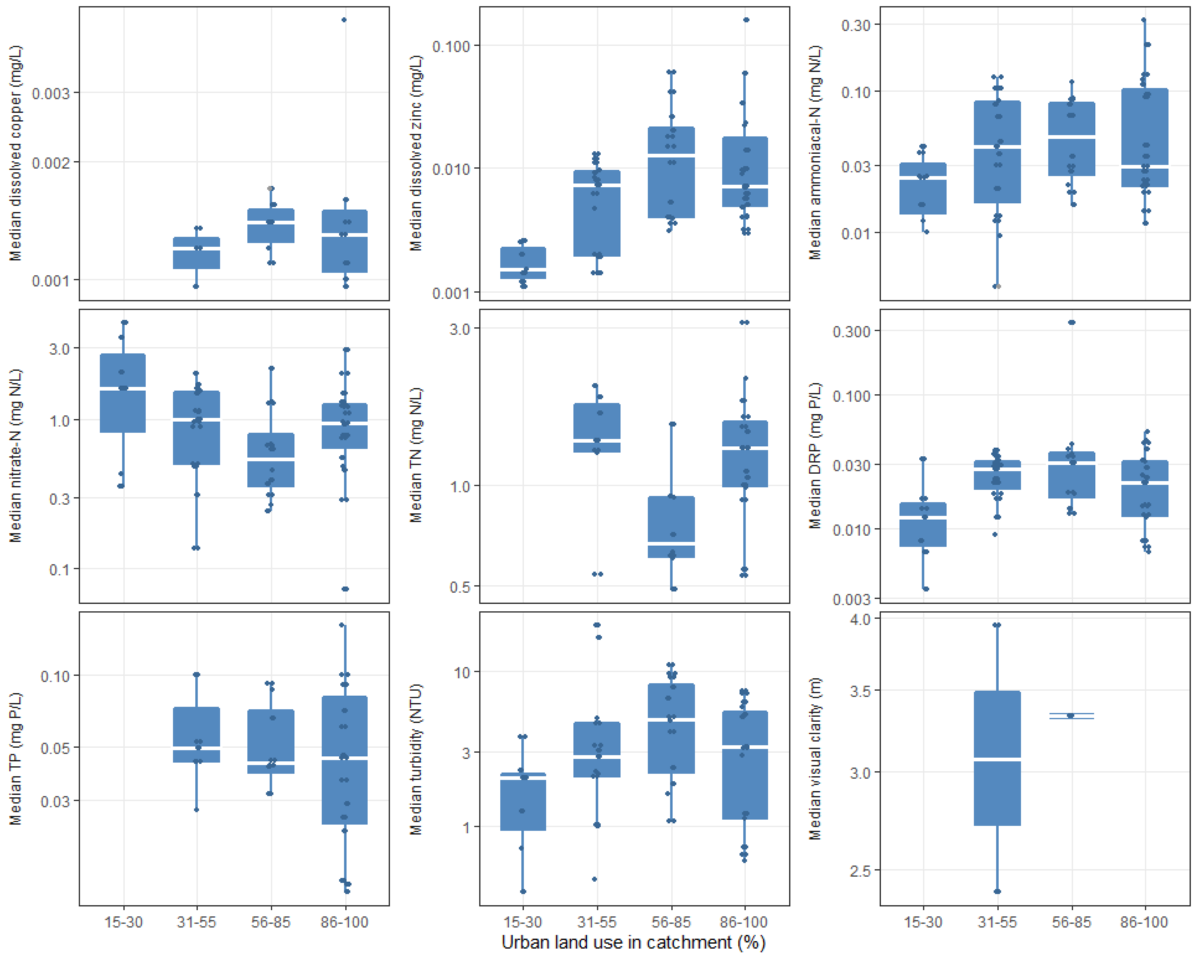

Figure 4-11: Distribution of site median water quality concentrations in urban river and stream sites categorised by percent urban land cover in upstream catchment. White horizontal line in each box indicates the median of site medians, box indicates the inter-quartile range of medians, whiskers indicate the 5th and 95th percentiles of medians. Individual site medians indicated by blue dots. Grey marker indicates that site median was based on imputation. Note log-scale on Y-axes.

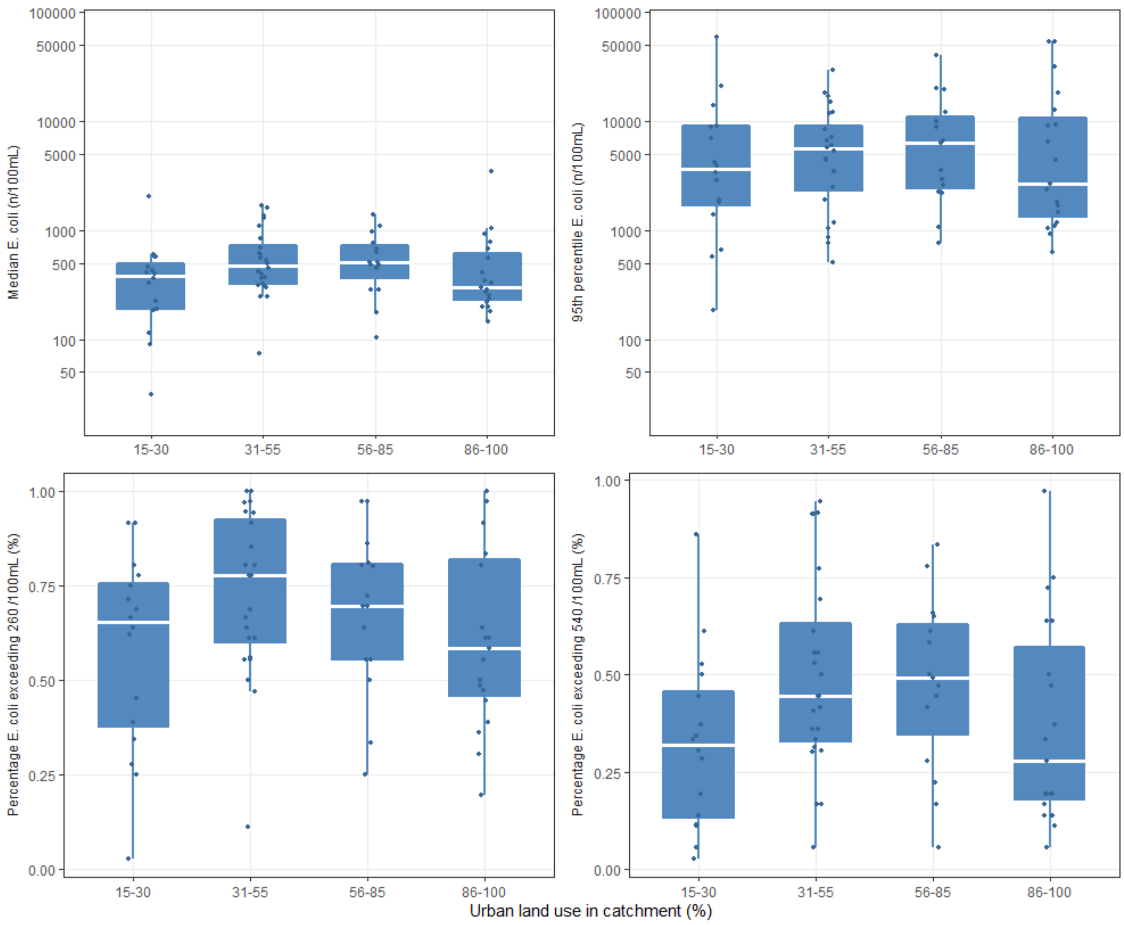

Figure 4-12: Distribution of site *E. coli* metrics in urban river and stream sites categorised by percent urban land cover in upstream catchment. White horizontal line in each box indicates the median of site medians, box indicates the inter-quartile range of medians, whiskers indicate the 5th and 95th percentiles of medians. Individual site medians indicated by blue dots (no site medians were based on imputation). Note log-scale on Y-axes.

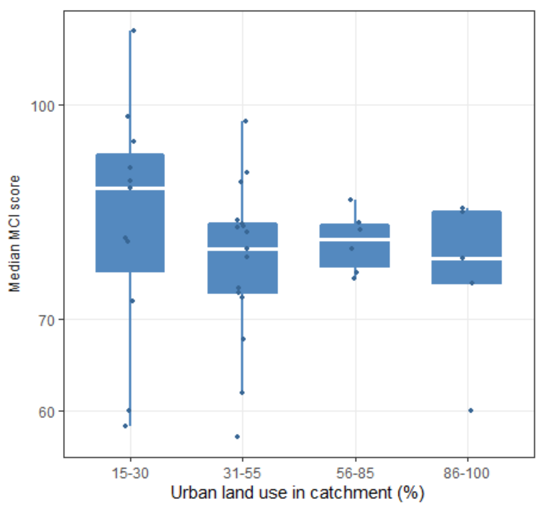

Figure 4-13: Distribution of site median MCI in urban river and stream sites categorised by percent urban land cover in upstream catchment. White horizontal line in each box indicates the median of site medians, box indicates the inter-quartile range of medians, whiskers indicate the 5th and 95th percentiles of medians. Individual site medians indicated by blue dots. Note log-scale on Y-axes.

For dissolved copper and zinc data for non-urban streams (native forest, exotic forest and pastoral land uses) are also shown (Figure 4-14) as, unlike other variables, comparison across the full range of land uses for dissolved metals is not included in the report on national river water quality state and trends (Larned et al. 2018). These data show lower concentrations of both copper and zinc compared to sites with 15% or more urban land cover. The difference is clear for dissolved copper. All but one of the sites with <15% urban land cover had a median copper concentration of less than 0.001 mg/L whereas all but two of the sites with >15% urban land cover exceeded this concentration. Similarly, for dissolved zinc, at sites with >15% urban land cover, all site median concentrations were above 0.001 mg/L, whereas for sites with <15% urban land cover only ~50% of sites exceeded this threshold.

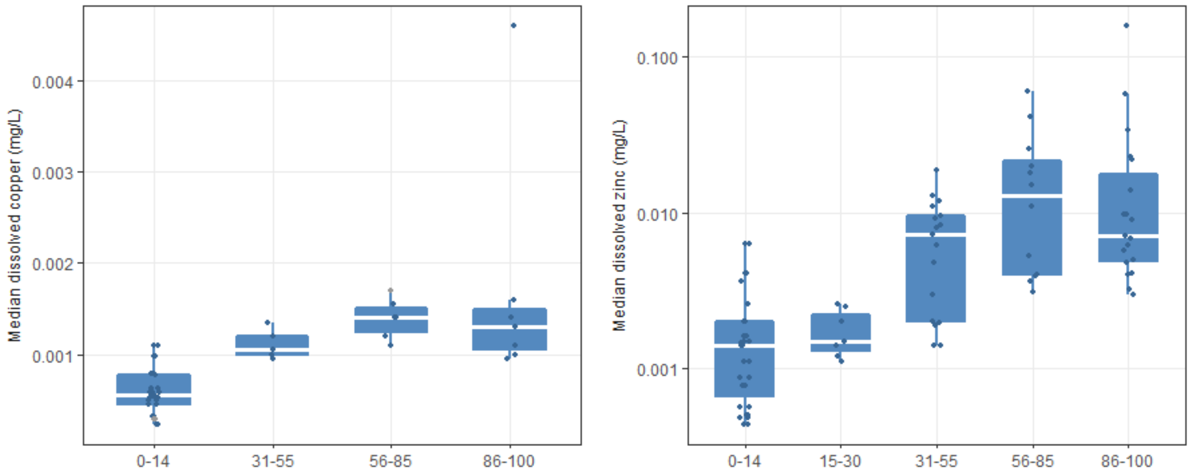

Figure 4-14: Distribution of median dissolved copper and zinc in all river and stream sites categorised by percent urban land cover in upstream catchment. White horizontal line in each box indicates the median of site medians, box indicates the inter-quartile range of medians, whiskers indicate the 5th and 95th percentiles of medians. Individual site medians indicated by blue dots (no site medians were based on imputation). Note log-scale on Y-axes.

The relationships between individual site medians and the percent urban land use are shown in Figure 4-15 to Figure 4-17. Five outliers for DRP and three outliers for TP were removed prior to this analysis. The scatter on these plots show the considerable variation in site medians and the indistinct nature of relationships with percent urban land cover. Dissolved zinc and ammoniacal-N were the only variables that had a significant relationship with urban land cover (p-value <0.05, Table 4-1), as determined through linear regression, though there was a lot of scatter, as shown by the low correlation coefficient R^2 of 0.25 and 0.08 respectively. When non-urban streams were included (Figure 4-18) The relationship for zinc was strengthened and with the inclusion of these data, there was also a statistically significant relationship between dissolved copper and urban land cover (p-value <0.001, Table 4-1).

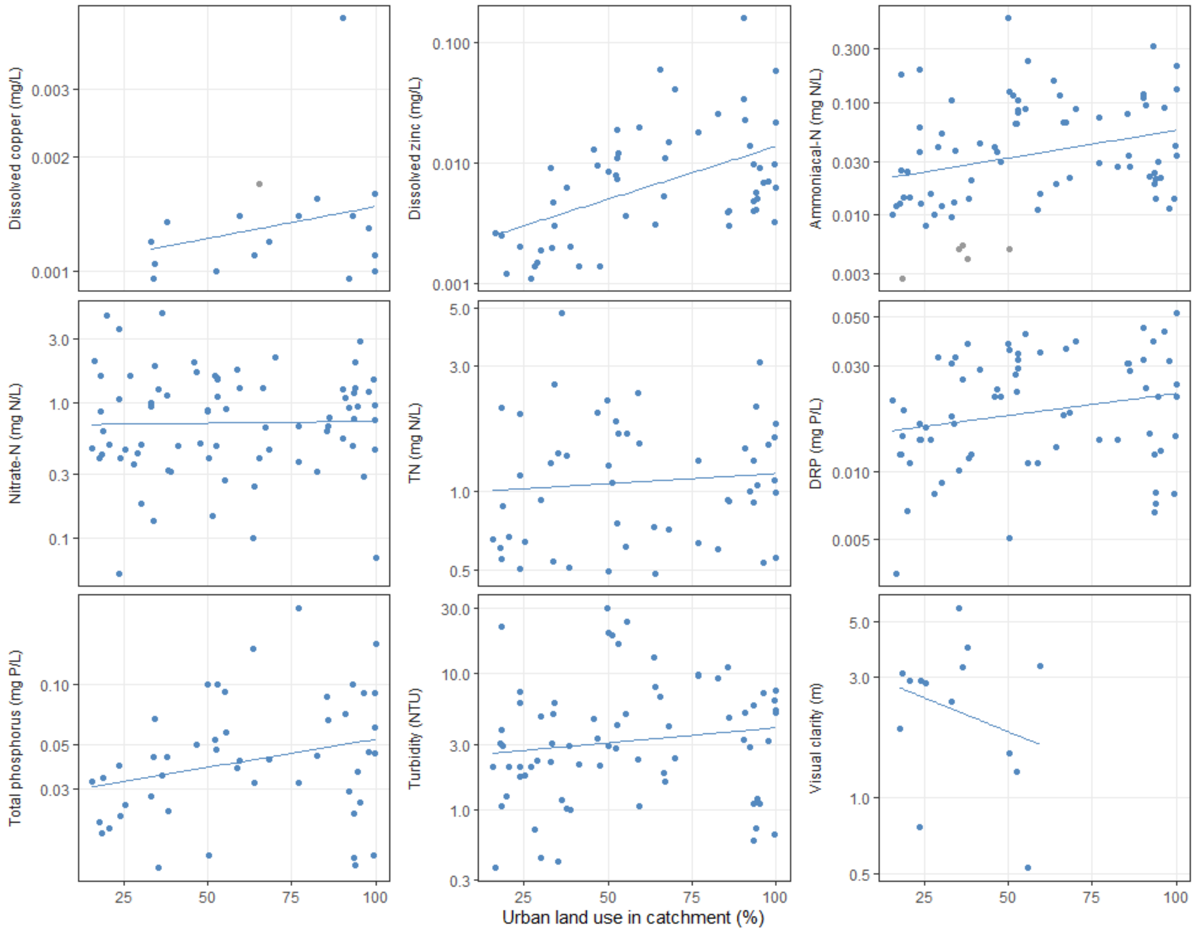

Figure 4-15: Median water quality in urban river and stream sites versus percent urban land cover in upstream catchment. Grey dots indicate site medians based on imputation. Note log-scale on y-axes.

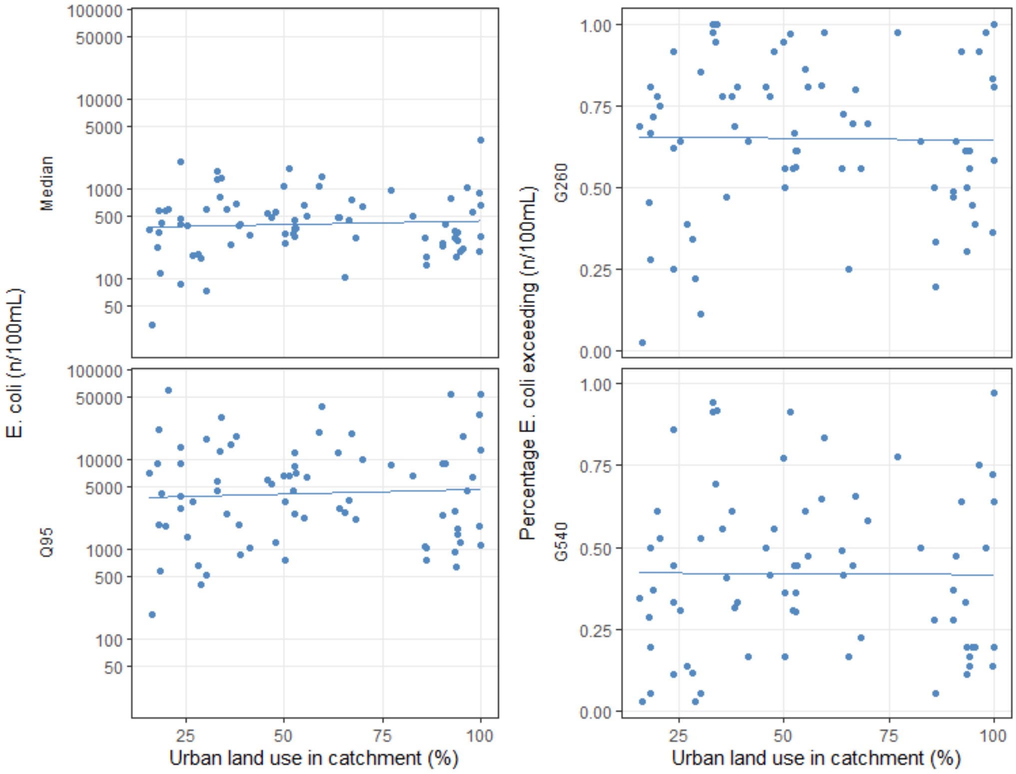

Figure 4-16: *E. coli* metrics in urban river and stream sites versus percent urban land cover in upstream catchment.

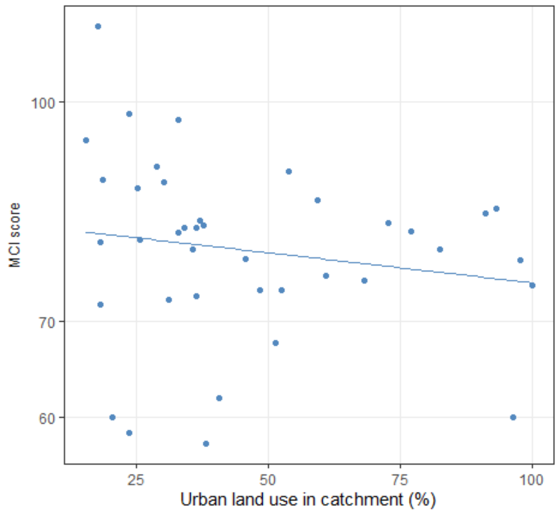

Figure 4-17: Median MCI score in urban river and stream sites versus percent urban land cover in upstream catchment.

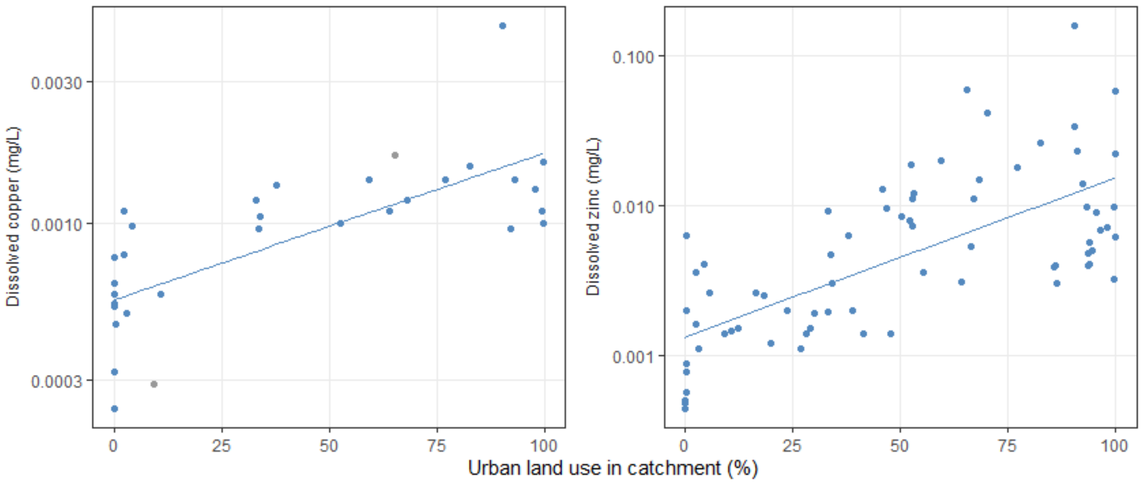

Figure 4-18: Median dissolved copper and zinc in all river and stream sites versus percent urban land cover in upstream catchment. Grey dots indicate site medians based on imputation. Note log-scale on y-axes.

Table 4-1: Quantitative relationships between water quality and urban land cover. Lines in bold represent statistically significant relationships based on p-value <0.05.

Water quality variable	Regression equation for median concentrations	Total R^2	P-value
Dissolved copper (urban streams only)	$\log_{10} dCu = 0.16 \times Urban - 3.0$	0.07	0.28
Dissolved copper (including rural streams)	$\boldsymbol{\log_{10} dCu = 0.49 \times Urban - 3.3}$	**0.50**	**< 0.001**
Dissolved zinc (urban streams only)	$\boldsymbol{\log_{10} dZn = 0.88 \times Urban - 2.7}$	**0.25**	**< 0.001**
Dissolved zinc (including rural streams)	$\boldsymbol{\log_{10} dZn = 1.1 \times Urban - 2.8}$	**0.51**	**< 0.001**
Nitrate-N	$\log_{10} NO3N = 0.03 \times Urban - 0.15$	0.0004	0.86
Ammoniacal-N	$\boldsymbol{\log_{10} NH4N = 0.50 \times Urban - 1.7}$	**0.08**	**0.01**
TN	$\log_{10} TN = 0.08 \times Urban - 0.01$	0.009	0.50
DRP	$\log_{10} DRP = 0.19 \times Urban - 1.8$	0.05	0.07
TP	$\log_{10} TP = 0.28 \times Urban - 1.6$	0.07	0.07
Water clarity	$\log_{10} CLAR = -0.54 \times Urban + 0.53$	0.07	0.34
Turbidity	$\log_{10} TURB = 0.22 \times Urban + 0.38$	0.02	0.24
E. coli	$\log_{10} ECOLI = 0.08 \times Urban + 2.6$	0.004	0.58
MCI	$MCI = -9.1 \times Urban + 84$	0.015	0.21

4.5 State categorised by impervious surface in catchment

Data were available on the area of impervious surfaces for 51 sites in Auckland, Christchurch and Wellington cities. Water quality variables are compared to the proportion of impervious surface in the upstream catchment in the plots below (Figure 4-19 and Figure 4-20) and quantitative relationships are explored in Table 4-2 and Figure 4-20. MCI is not included in this section as there were too few monitored site where impervious surface data were available. Higher median dissolved zinc, ammoniacal-N, DRP and turbidity were related to higher impervious surface proportions and relationships for the first three were statistically significant, albeit with considerable scatter, as shown in Figure 4-20 and demonstrated by the low R^2 values (Table 4-2). Impervious surface area explained dissolved zinc concentration slightly better than urban land cover, as demonstrated by the R^2 values of 0.12 for urban land cover and 0.16 for impervious area. The impervious surfaces data were over 10 years old, and it is likely that there have been changes in most catchments, and potentially large changes (for example, in Christchurch City following the earthquakes). This is likely to contribute to the variability in the relationships between imperviousness and water quality state.

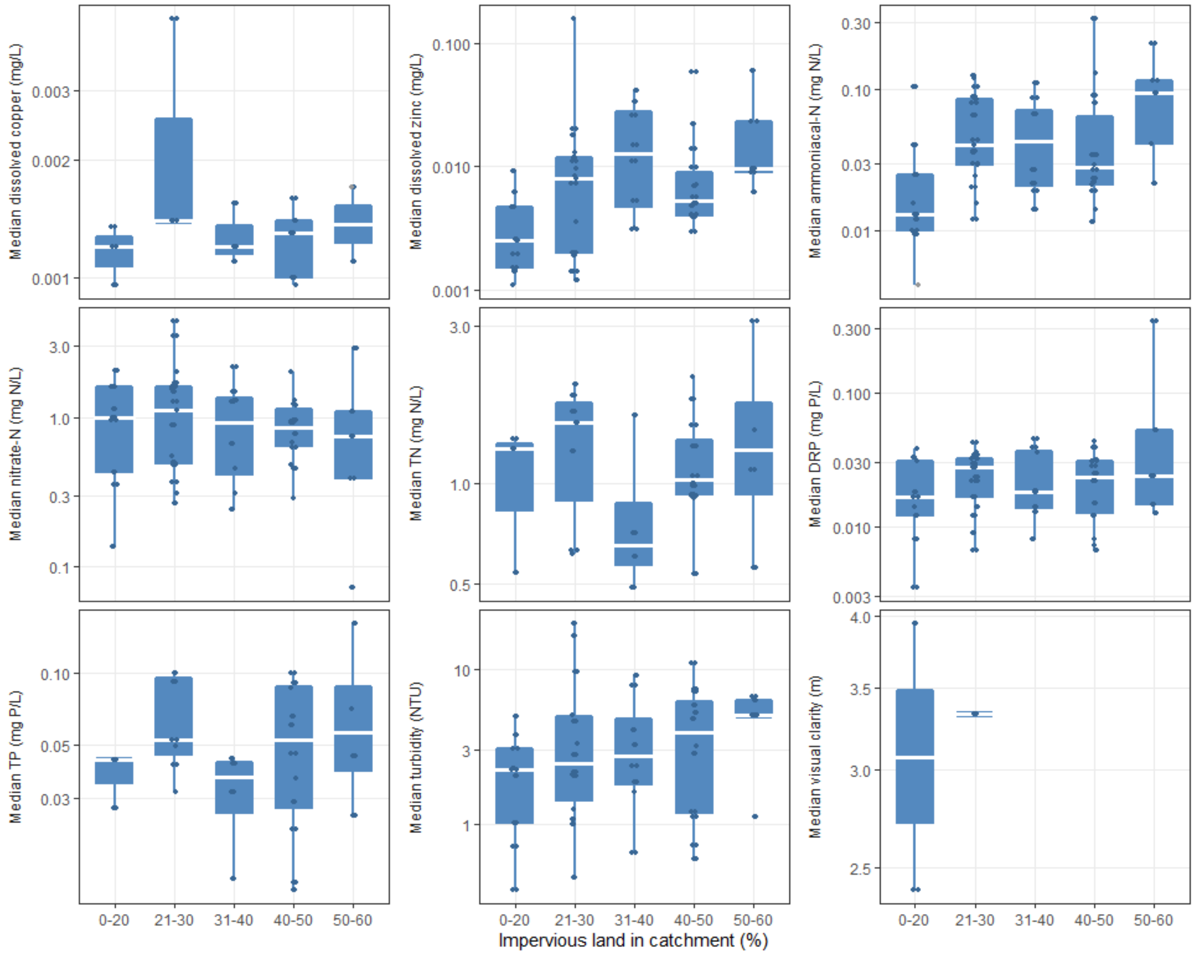

Figure 4-19: Distribution of median water quality concentrations in urban river and stream sites categorised by percent impervious land cover in upstream catchment. White horizontal line in each box indicates the median of site medians, box indicates the inter-quartile range of medians, whiskers indicate the 5th and 95th percentiles of medians. Individual site medians indicated by blue dots. Grey marker indicates that site median was based on imputation. Note log-scale on Y-axes.

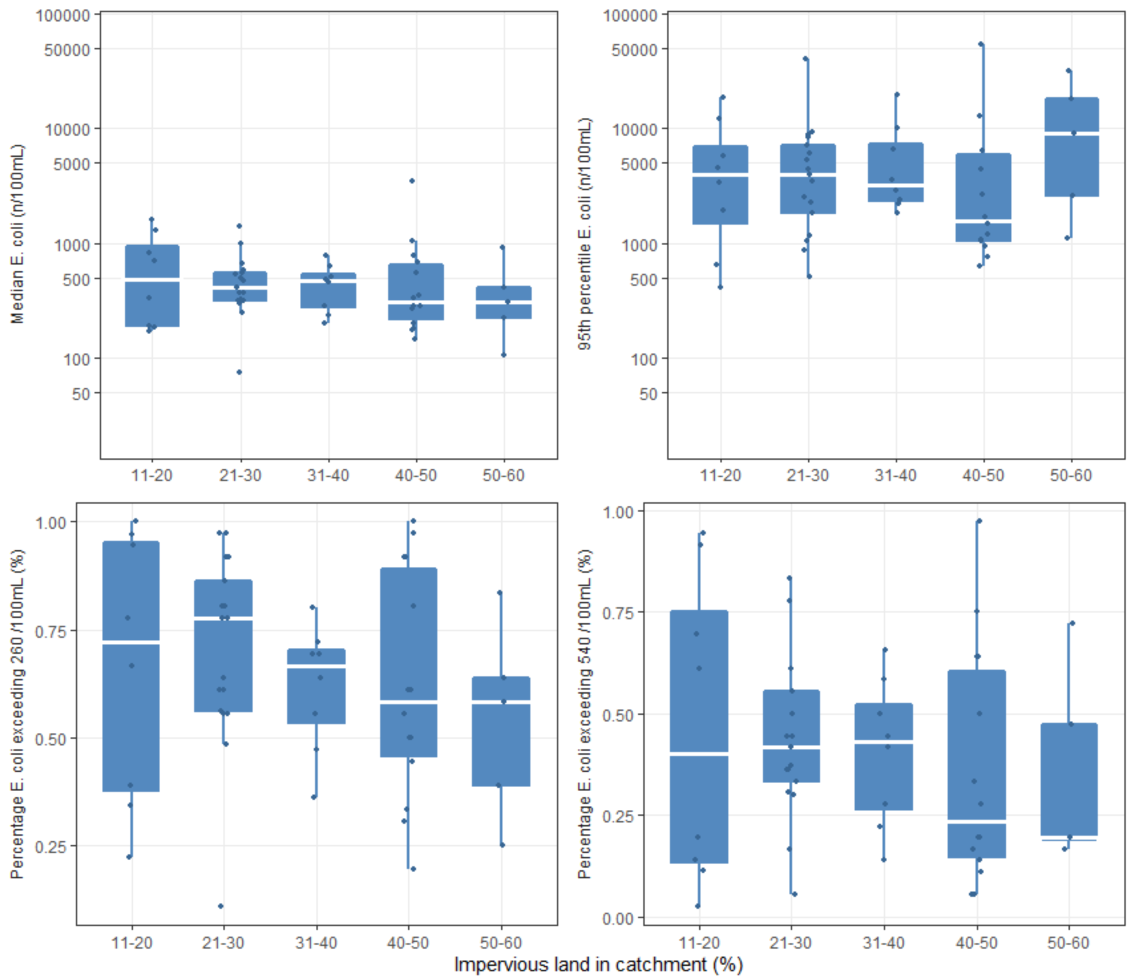

Figure 4-20: *E. coli* metrics in urban river and stream sites categorised by percent impervious land cover in upstream catchment. White horizontal line in each box indicates the median of site medians, box indicates the inter-quartile range of medians, whiskers indicate the 5th and 95th percentiles of medians. Individual site medians indicated by blue dots (no site medians were based on imputation). Note log-scale on Y-axes.

Table 4-2: Quantitative relationships between water quality and the proportion of impervious surfaces the upstream catchment. Lines in bold represent statistically significant relationships.

Water quality variable [1]	Regression equation for median concentrations	Total R^2	P-value
Dissolved copper	$\log_{10} dCu = -0.02 \times Imperviousness - 2.9$	0.0002	0.95
Dissolved zinc	$\boldsymbol{\log_{10} dZn = 1.53 \times Imperviousness - 2.7}$	**0.17**	**0.002**
Nitrate-N	$\log_{10} NO3N = -0.34 \times Imperviousness + 0.025$	0.016	0.37
Ammoniacal-N	$\boldsymbol{\log_{10} NH4N = 1.1 \times Imperviousness - 1.8}$	**0.13**	**0.008**
TN	$\log_{10} TN = 0.19 \times Imperviousness - 0.03$	0.012	0.57
DRP	$\boldsymbol{\log_{10} DRP = -0.65 \times Imperviousness - 1.9}$	**0.07**	**0.05**
TP	$\log_{10} TP = 0.24 \times Imperviousness - 1.43$	0.011	0.58
Turbidity	$\log_{10} TURB = 0.86 \times Imperviousness + 0.16$	0.079	0.054
E. coli	$\log_{10} ECOLI = 0.1 \times Imperviousness + 2.6$	0.001	0.79

Note: [1] Regression not undertaken for visual clarity or MCI due to insufficient number of sites.

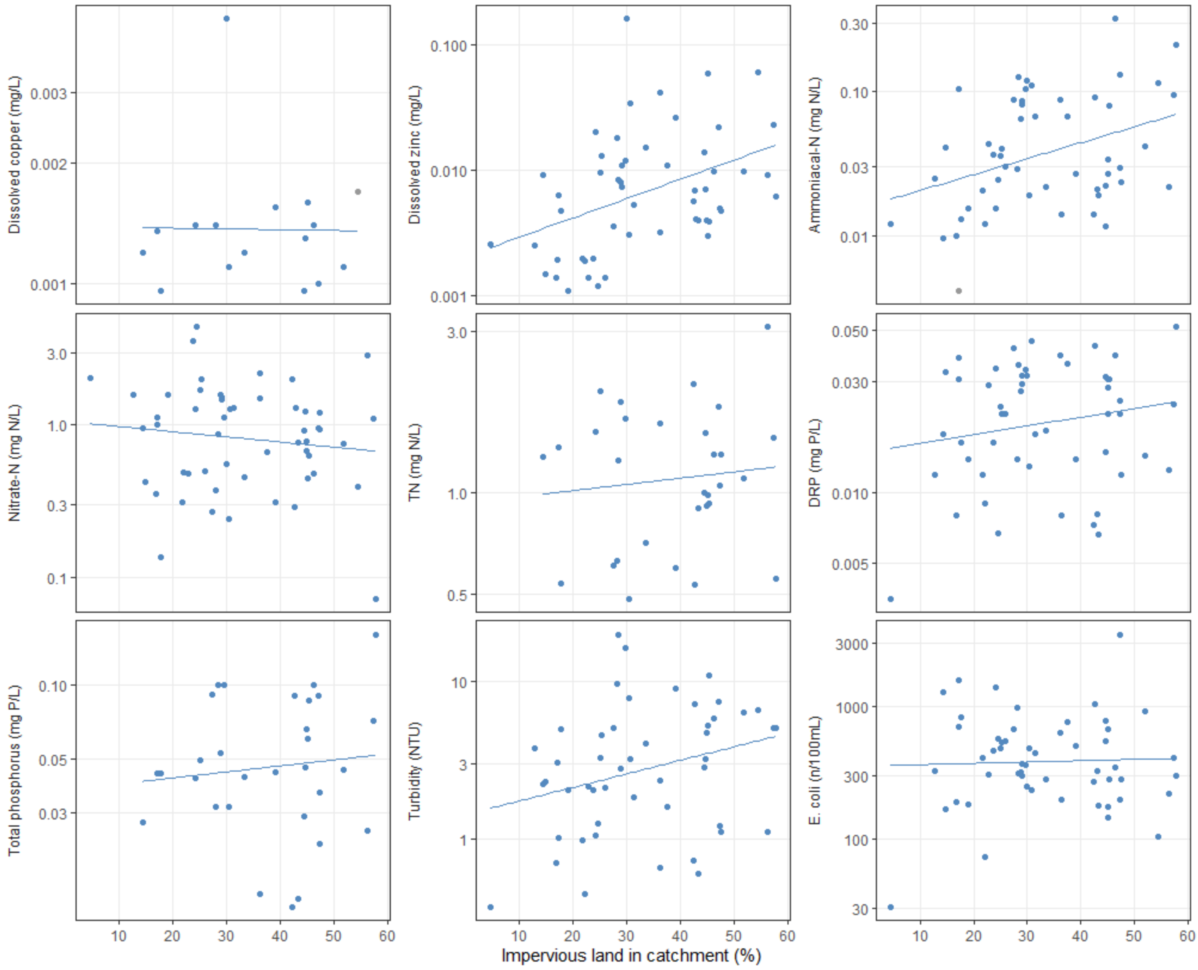

Figure 4-21: Median water quality in urban river and stream sites versus percent impervious surface in upstream catchment. Grey dots indicate site medians based on imputation. Note log-scale on y-axes.

5 Water Quality Trends

5.1 Trends over 10 years

For most water quality variables trends were analysed over the 10-year period from 2008-2017 but for metals at some sites trends were analysed over the 7-year period from 2011 to 2017, reflecting data availability (Section 3.2.1). Trends from 10-year and 7-year datasets were combined where relevant to investigate overall trends on a variable by variable basis, and to assess relationships between explanatory variables such as region, REC class and urban land cover. Time-series plots for each site and variable are provided in Appendix B.

There were 15 sites where trends could not be assessed for dissolved copper due to excess censored values (i.e., either <5 non-censored values and/or <3 unique non-censored values); but trends were assessed at all other sites and for other variables. It should be noted that even where trends can be assessed, if there is a high percentage of censored data there is less certainty in the trend results. There were several sites where the level of censoring exceeded 15% of the data record (17 sites for ammoniacal-N, 16 for dissolved copper, 12 for dissolved zinc, 6 for TP and 1 for visual clarity). However, trends for all sites are shown in this section, regardless of the level of censoring and including trends categorised as indeterminant.

Where trends could be assessed, trend direction and magnitude is shown in Figure 5-1 and the numbers of sites with improving, degrading and indeterminate trends for each variable are listed in Table 5-1. All water quality variables except *E. coli* and MCI showed improving trends at more sites than they showed degrading trends (Table 5-1). For all variables except DRP, nitrate-N and total nitrogen, there were as many or more sites with indeterminate trends than either improving or degrading trends.

The magnitude of decreasing trends (Figure 5-1) was frequently larger than the magnitude of increasing trends, as shown by the distance between the markers and the line at 0 (no trend).

Table 5-1: Summary of trend directions for each water quality variable.

	No. sites in each trend category			Total no. sites assessed	No. sites not assessed
	Improving	Degrading	Indeterminant (insufficient data)		
Dissolved Copper	9 (29%)	0	22 (71%)	31	15
Dissolved Zinc	18 (38%)	3 (6%)	26 (55%)	47	0
Ammoniacal-N	12 (18%)	7 (10%)	48 (72%)	67	2
Nitrate-N	32 (48%)	3 (4%)	31 (47%)	66	0
Total nitrogen	22 (52%)	3 (7%)	17 (40%)	42	0
Dissolved Reactive Phosphorus	33 (49%)	5 (7%)	29 (43%)	67	0
Total phosphorus	15 (37%)	5 (12%)	21 (51%)	41	0
Turbidity	22 (36%)	3 (5%)	36 (59%)	61	0
Visual clarity	5 (50%)	0	5 (50%)	10	0
E. coli	7 (11%)	14 (21%)	45 (68%)	66	0
MCI	0	2 (6%)	29 (94%)	31	0

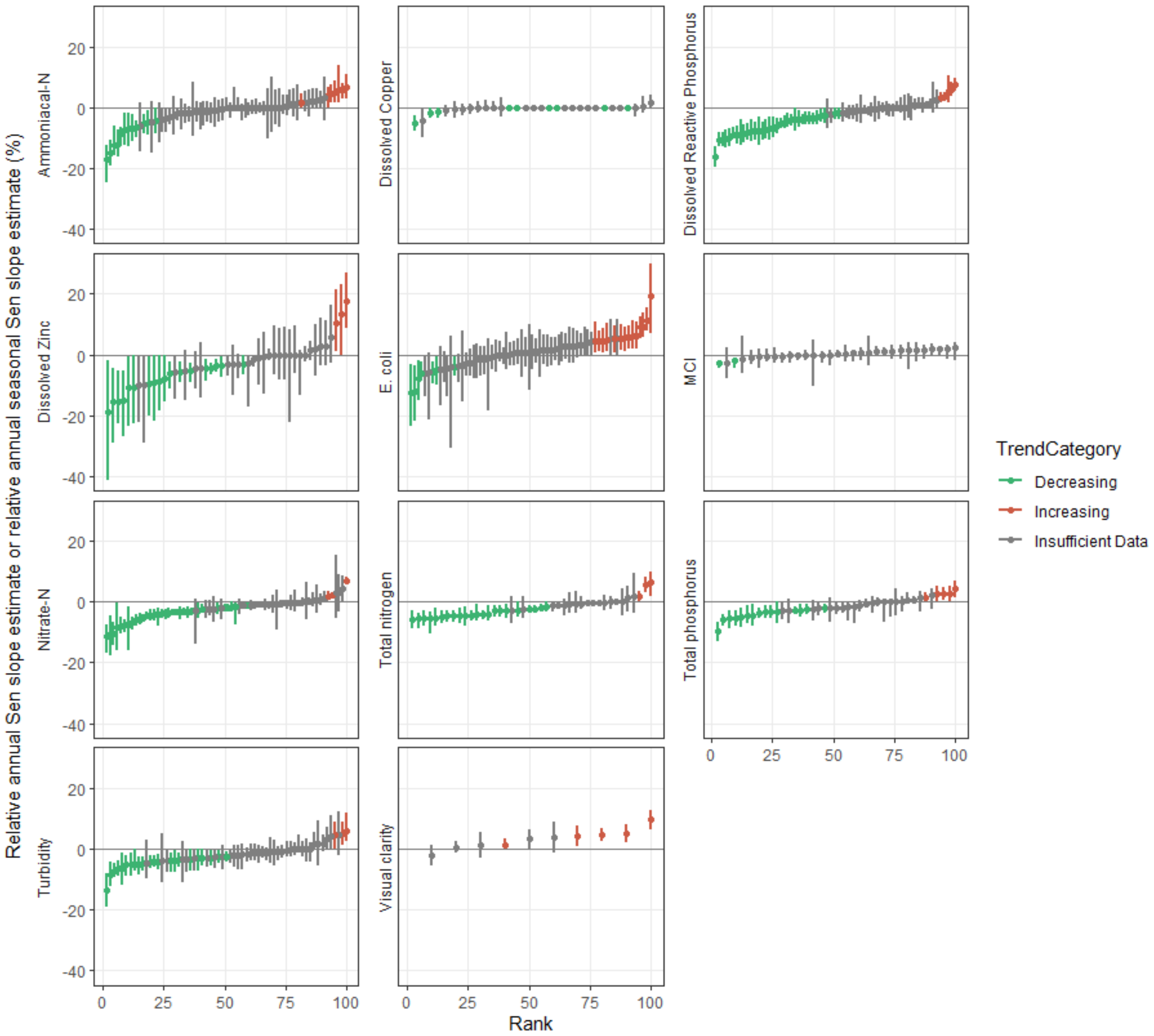

Figure 5-1: Trends in stream water quality for each site, ordered by direction and magnitude. Markers indicate the estimated trend slope for each variable and site, and bars represent 90% confidence interval. Positive slopes (marker and confidence interval above black horizontal line at zero, coloured in orange) indicates increasing concentrations/values, whereas negative slopes (marker and confidence interval below black line at zero, coloured in green) indicates decreasing concentrations. Grey markers and confidence intervals correspond to slope whose directions could not be established with confidence. Note increases in MCI and visual clarity are improving trends, not degrading.

Most regions had few urban stream monitoring sites, and it is therefore difficult to make any conclusions about overall regional trends. Nitrate-N concentrations decreased in most regions (Figure 5-2), but there were sites in the Auckland and Canterbury regions where nitrate-N increased.

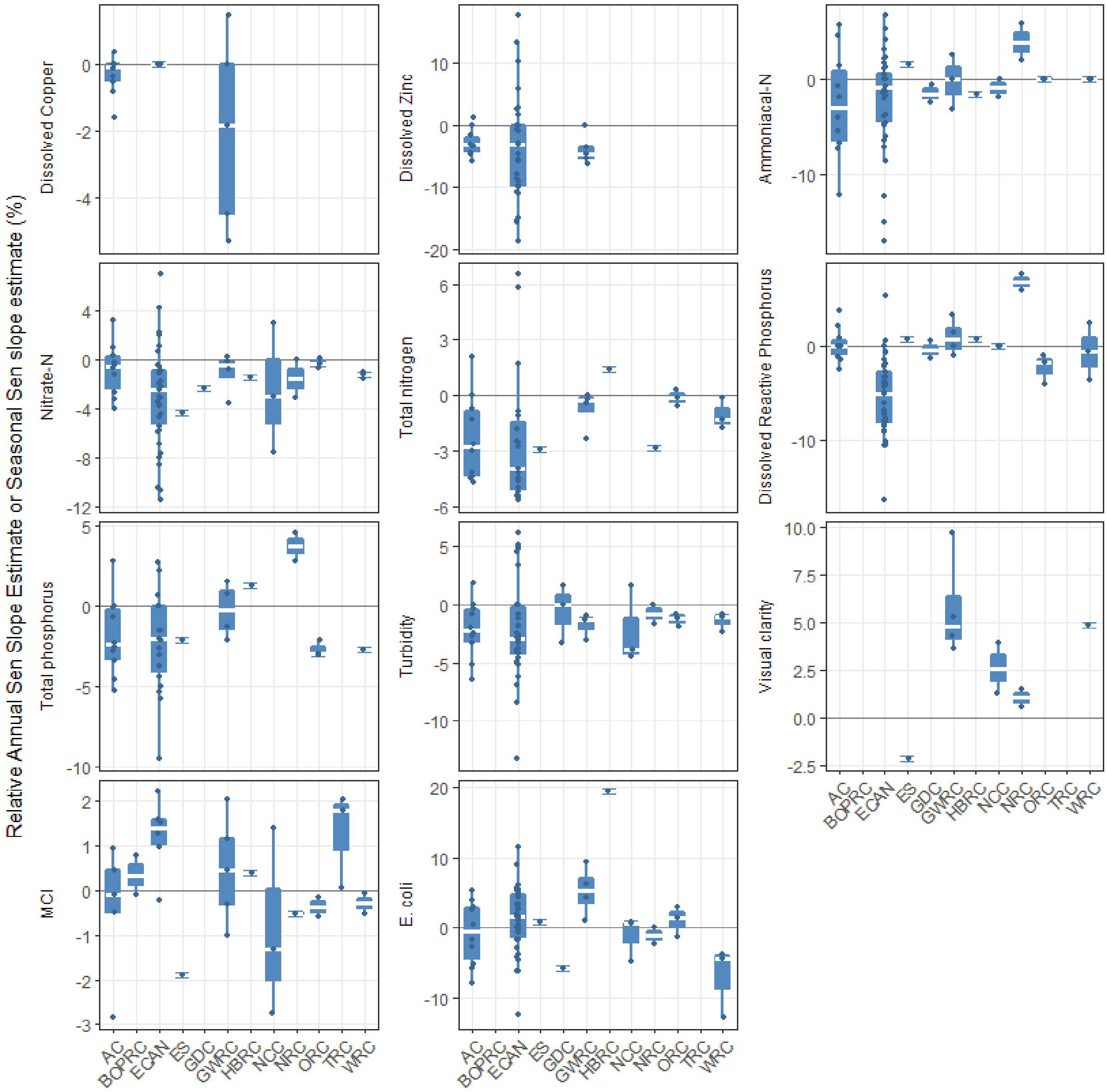

Figure 5-2: Comparison of trends in stream water quality between regions. Sites where trend category was "indeterminant" are also included in this plot.

When sites are categorised by REC class (Figure 5-3), there appears to be some difference in the direction of trends in DRP, with most sites in class CD_L showed decreasing trends and sites in WW_L showing increasing trends. For *E. coli*, there were a higher proportion of sites with increasing trends in REC class CD_L than in other classes.

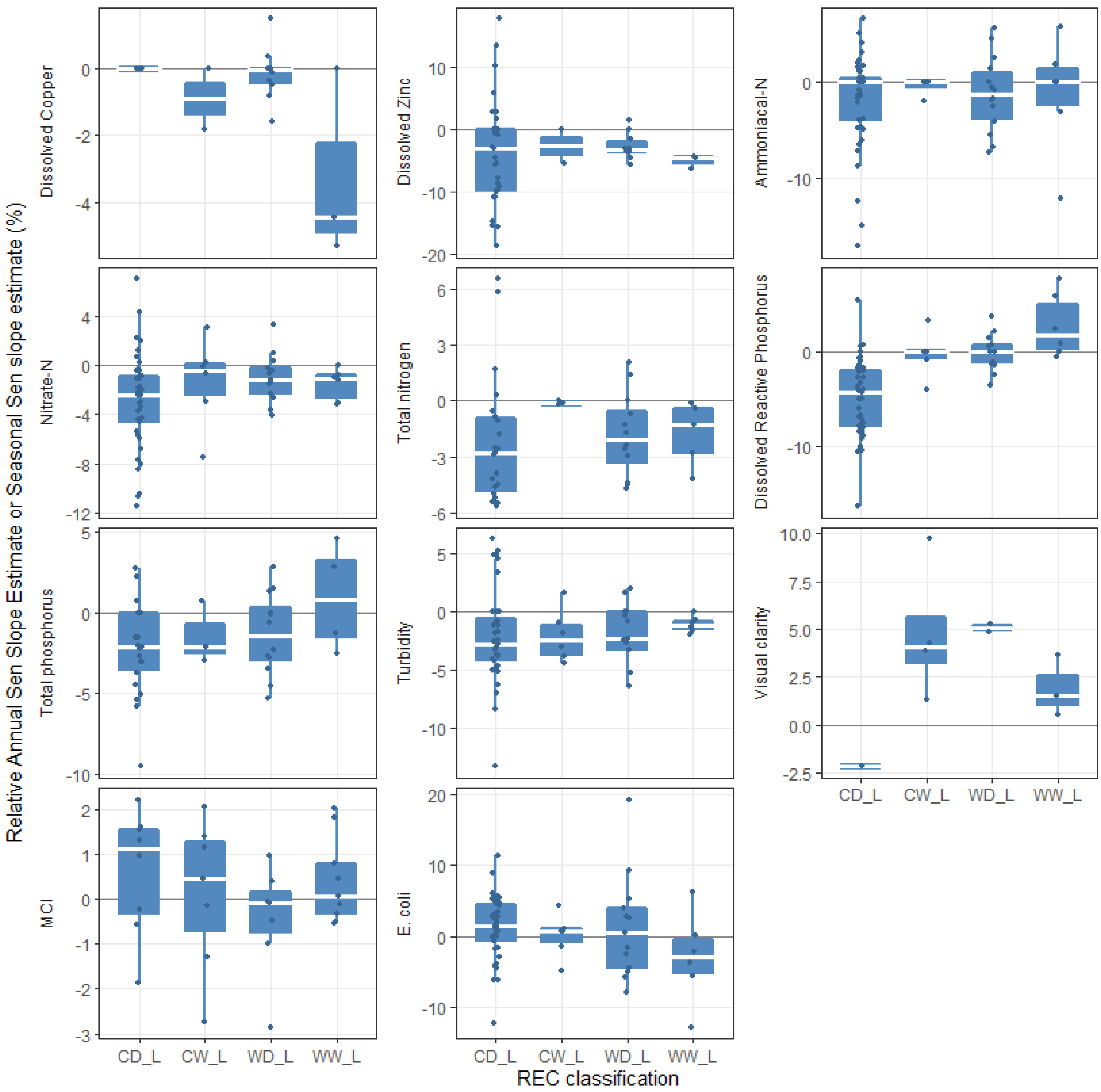

Figure 5-3: Comparison of trends in stream water quality between REC classes. Sites where trend category was "indeterminant" are also included in this plot. CD_H = cool dry climate, hill source; CD_L = cool dry climate, lowland source; CW_L = cool wet climate, lowland source; WD_L = warm dry climate, lowland source; WW_L = warm wet climate, lowland source.

Trend directions and magnitudes are also plotted in Figure 5-4 by the proportion of urban land cover in the upstream catchment. This plot suggests that sites with higher proportions of urban land cover have decreasing concentrations of DRP and TN. For other water quality variables, there are no clear differences in the trend directions and magnitude between urban land cover categories.

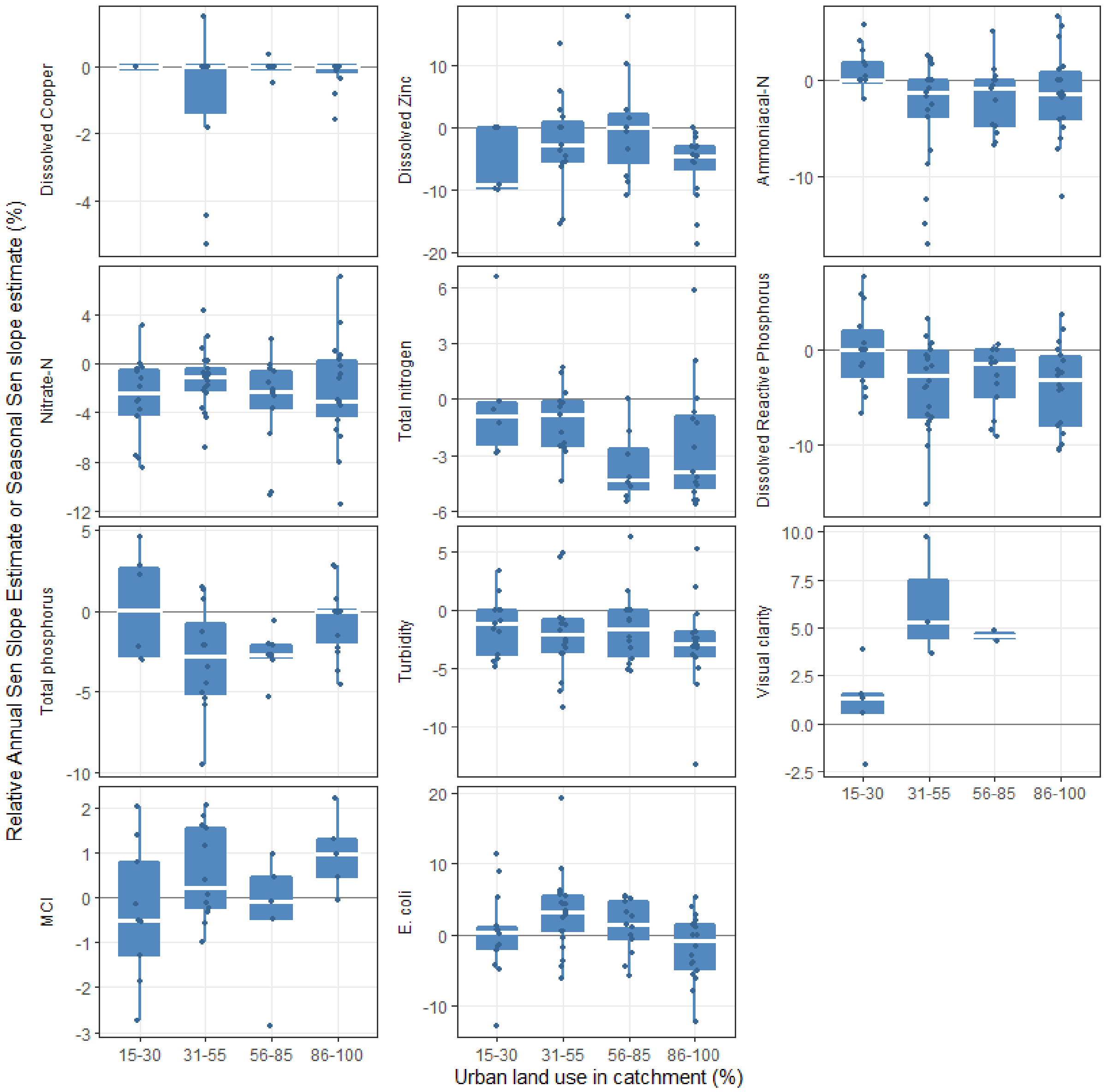

Figure 5-4: Comparison of trends in stream water quality between categories of urban land cover in the upstream catchment. Sites where trend category was "indeterminant" are also included in this plot.

5.2 Improving trends over 10 years

This section examines improving trends in water quality, which equates to decreasing concentrations for all variables except clarity and MCI, where improving trends equates to increasing clarity and MCI score. Figure 5-5 maps for each water quality variable the likelihood of improving trends at each categorical level of confidence defined in Table 3-3.

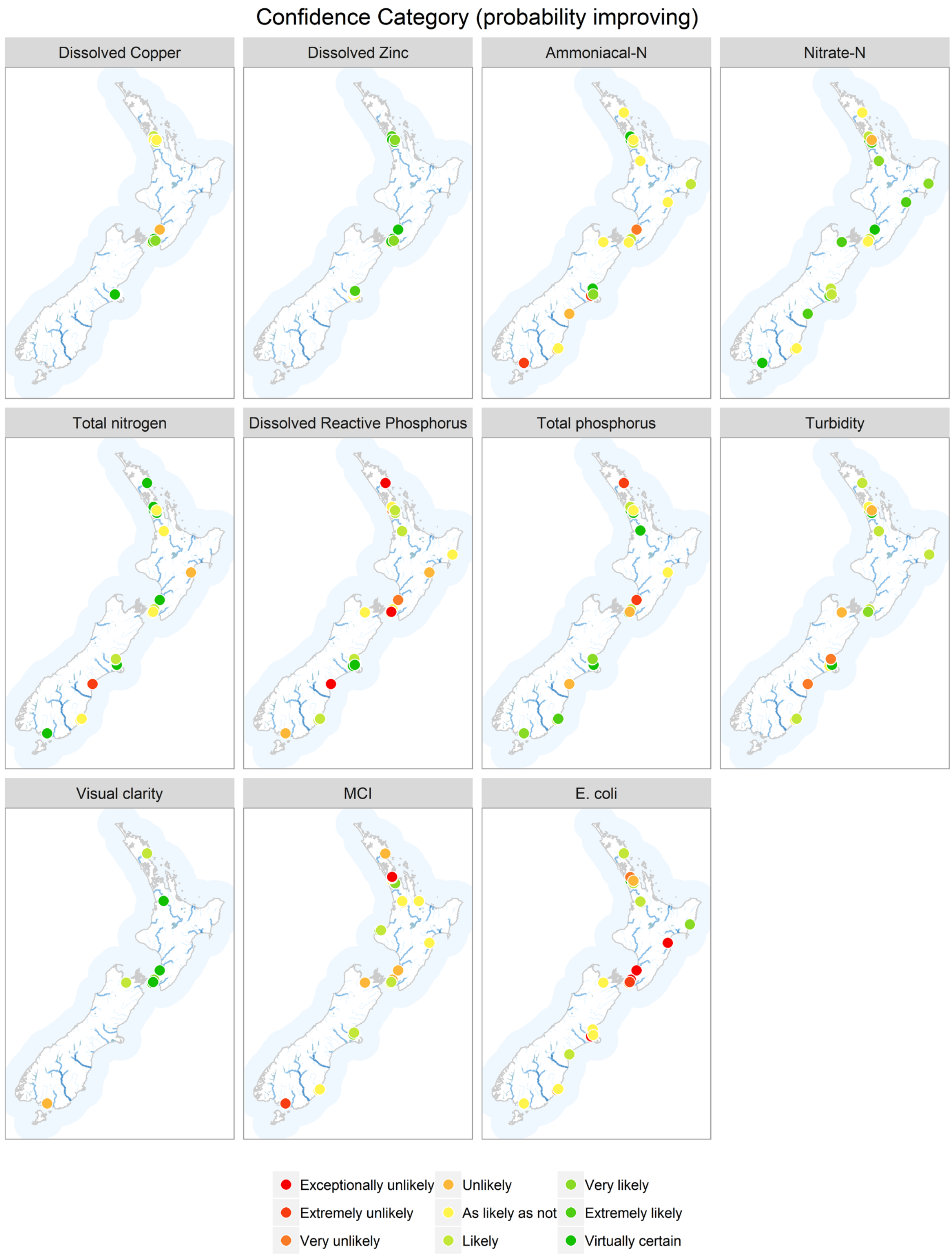

Figure 5-5: Map indicating location of sites and likelihood of improving trends at each categorical level of confidence. Trends assessed over 7- or 10-year time period. Levels of confidence defined in Table 3-3.

Figure 5-6 shows the proportion of urban stream sites by variable, for which 10-year (or 7-year for Cu and Zn at some sites) water quality trends indicated improvement at the nine categorical levels of confidence. The plot indicates that ~50% or more of sites were at least likely to be improving for all water quality variables except MCI and *E. coli*. For MCI, 39% of sites were at least likely to be improving, and 23% of sites were unlikely (or very unlikely, extremely unlikely etc) to be improving (34% of sites as likely as not). For *E. coli*, close to 44% of sites were unlikely (or very unlikely, extremely unlikely etc) to be improving and only 32% likely to be improving. Note that as the probability of improvement is the complement of probability of degradation, for these variables, close to 50% of sites are at least likely to be degrading.

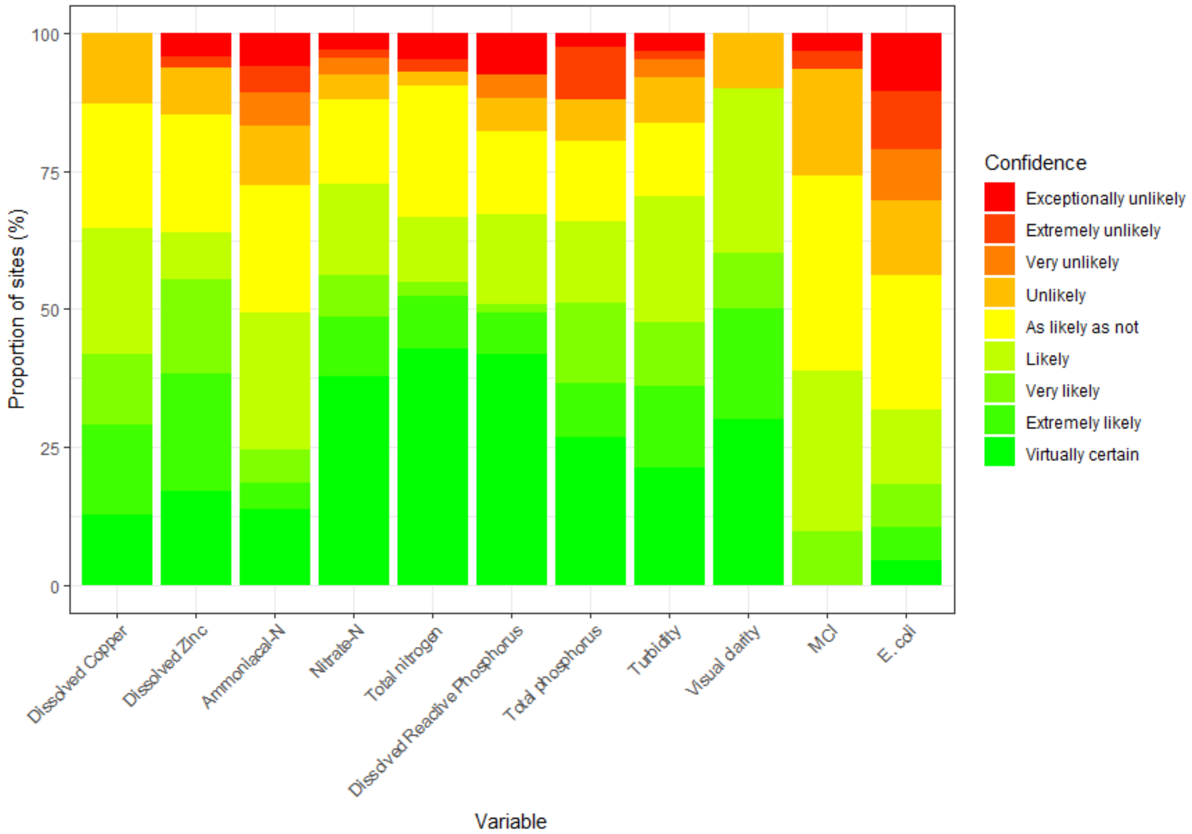

Figure 5-6: Summary plot representing the proportion of sites with improving trends at each categorical level of confidence. Trends assessed over 7- or 10-year time period, depending on data availability. The plot shows the proportion of sites with improving trends at levels of confidence defined in Table 3-3.

The proportion of improving trends (PIT) statistics are shown for all of New Zealand (Figure 5-7) and aggregations by region, REC class and percentage urban land cover in the catchment (Figure 5-8 to Figure 5-10). For many regions, REC classes and urban land cover categories there were too few sites to accurately calculate PIT statistics and these are only shown for regions with at least 4 sites.

The national-scale PIT statistics indicate a majority of improving trends at the 95% confidence interval for all variables except MCI and *E. coli*. *E. coli* had a majority (i.e., PIT <50%) of degrading trends, at the 95% confidence level. For MCI, the 95% confidence intervals for the PIT included 50%, and we cannot infer national-scale degradation or improvement for this variable. Visual clarity had the highest PIT statistic (i.e., the greatest proportions of improving trends), however this is based on fewer sites than other variables and therefore may not be representative of urban rivers and streams nationwide.

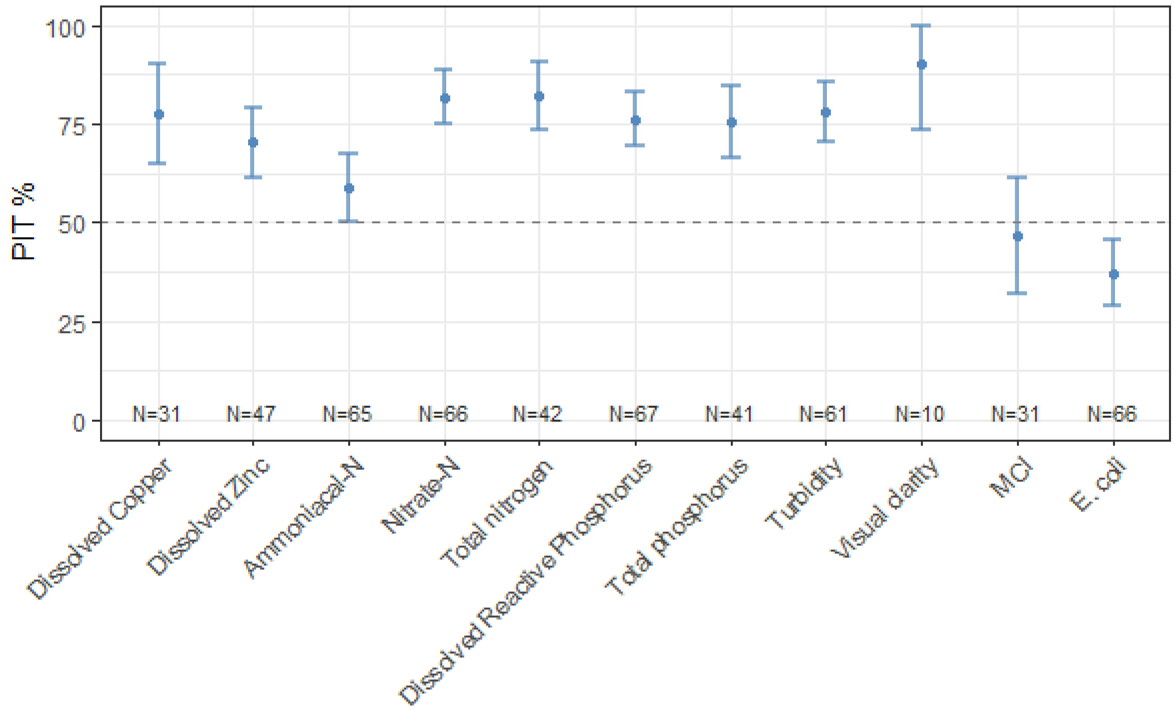

Figure 5-7: Proportion of improving 10-year trends at all urban river and stream sites. The error bars indicate the 95% confidence interval for the proportion of improving trends. The dashed grey line indicates where 50% of sites have improving trends. N indicates number of sites where trends were assessed for each variable.

The PIT statistics and 95% confidence intervals for each water quality variable and region are shown only for regions with more than 3 sites (Figure 5-8). Dissolved zinc was the only water quality variable which had a had a majority of improving (i.e., >50%) trends, at the 95% confidence level for all three regions with measurements. Dissolved copper had a majority of improving trends at sites in Canterbury and Greater Wellington, but not at sites Auckland (the 95% confidence intervals for the PIT included 50%). For Canterbury, nine of the 10 measured variables had a majority of improving (i.e., >50%) trends, at the 95% confidence level, whilst *E. coli* had a majority of degrading trends, at the 95% confidence level.

For Auckland, five of the 10 measured variables (ammoniacal-N, dissolved zinc, TN, TP, turbidity) had a majority of improving (i.e., >50%) trends, at the 95% confidence level; and for the remaining five water quality variables (dissolved copper, DRP, *E. coli*, MCI, nitrate-N) the 95% confidence intervals for the PIT included 50% and we cannot infer regional degradation or improvement for these variables. For Greater Wellington (GW), three of the nine measured variables (dissolved copper, dissolved zinc, visual clarity) had a majority of improving trends, at the 95% confidence level; one (*E. coli*) had a majority of degrading trends and for the remaining five water quality variables (DRP, MCI, nitrate-N, TN, TP) the 95% confidence intervals for the PIT included 50%.

Canterbury was the region with the greatest number of sites for which trends were assessed. A larger number of sites typically results in more narrow confidence intervals whereas regions with fewer sites had broader confidence intervals, making them more likely to include 50%.

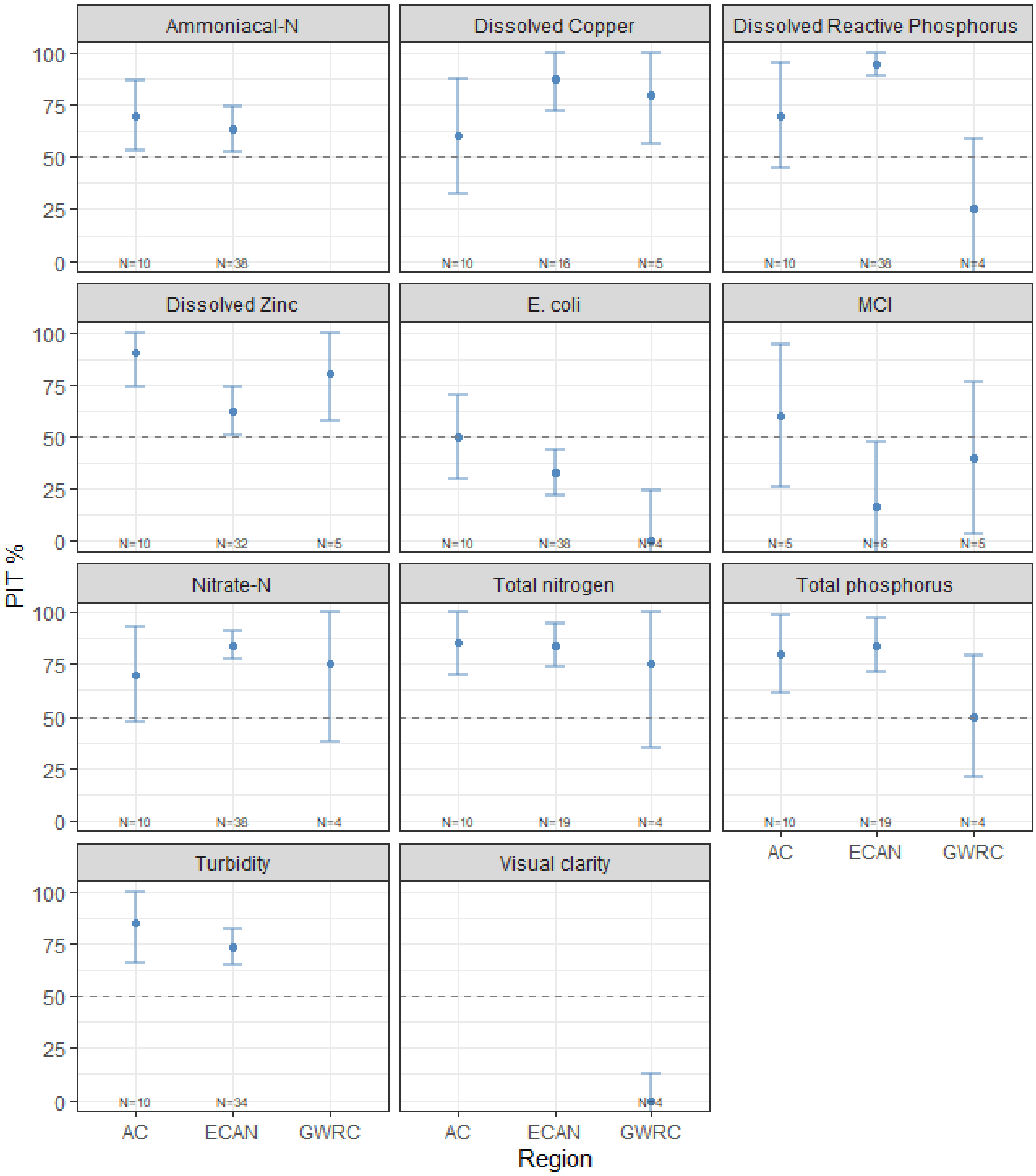

Figure 5-8: Proportion of improving 10-year trends by region for regions with > 3 sites only. The error bars indicate the 95% confidence interval for the proportion of improving trends. The dashed grey line indicates where 50% of sites have improving trends. N indicates number of sites where trends were assessed for each variable and category.

Dissolved zinc, TN and turbidity had a majority of improving trends, at the 95% confidence level, for all REC classes (Figure 5-9). The CD_L class, which includes the Canterbury region, showed a majority of improving trends for seven out of ten variables (dissolved copper, DRP, dissolved zinc, nitrate-N, TN, TP, turbidity), a majority of degrading trends for *E. coli* and the 95% confidence intervals for the PIT for ammoniacal-N and MCI included 50%. All other variables differed according to REC class and in many cases the 95% confidence intervals for the PIT included 50%, so no inference can be made as to degradation or improvement on those REC classes and variables.

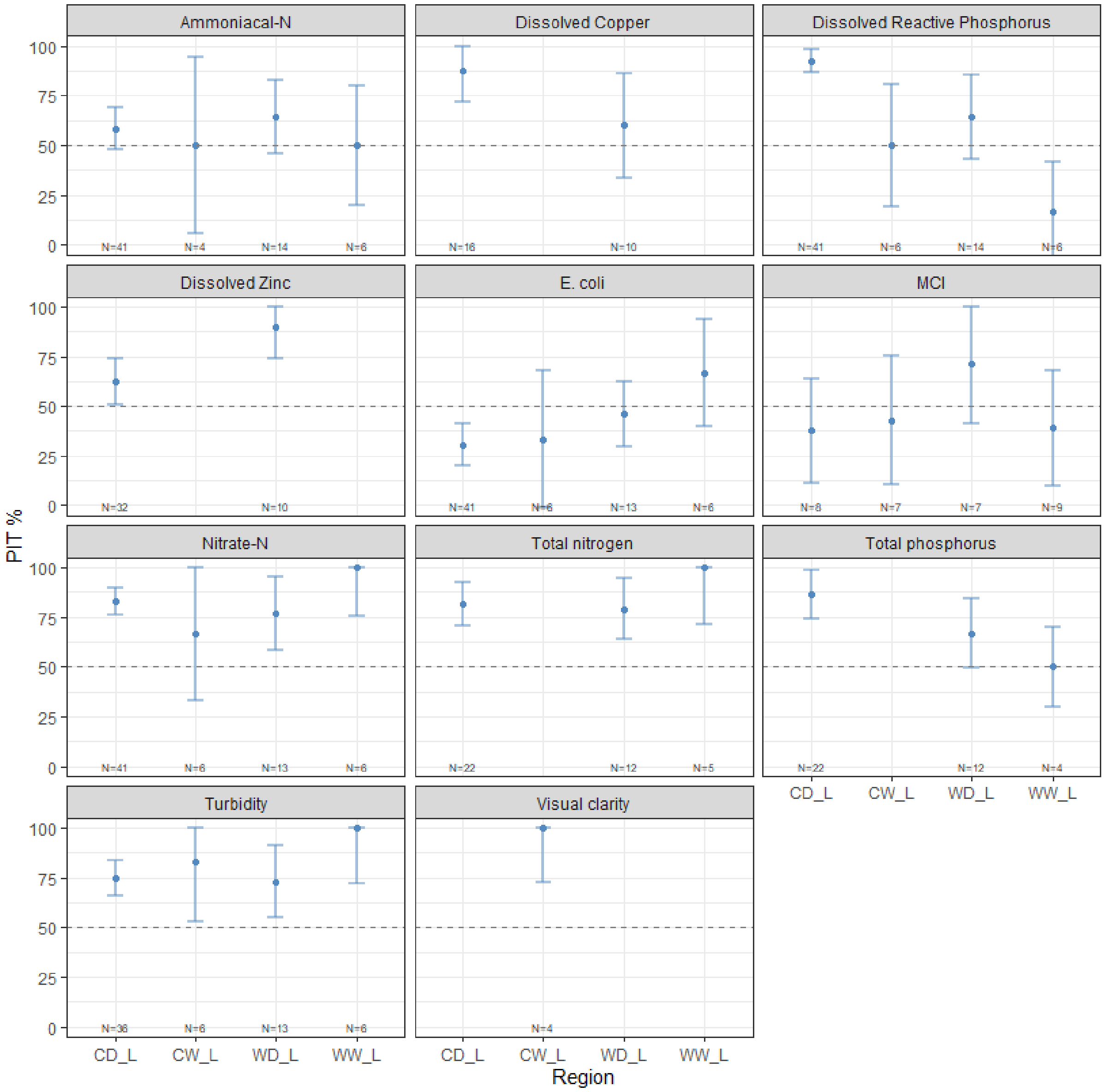

Figure 5-9: Proportion of improving 10-year trends by REC class for classes with > 3 sites only. The error bars indicate the 95% confidence interval for the proportion of improving trends. The dashed grey line indicates where 50% of sites have improving trends. N indicates number of sites where trends were assessed for each variable and category.

When grouped by proportion of urban land cover in the upstream catchment (Figure 5-10), nitrate-N, TN and turbidity showed a majority of improving trends across all categories of urban land cover. However, for the remaining variables there were one or more categories where the 95% confidence intervals for the PIT included 50%.

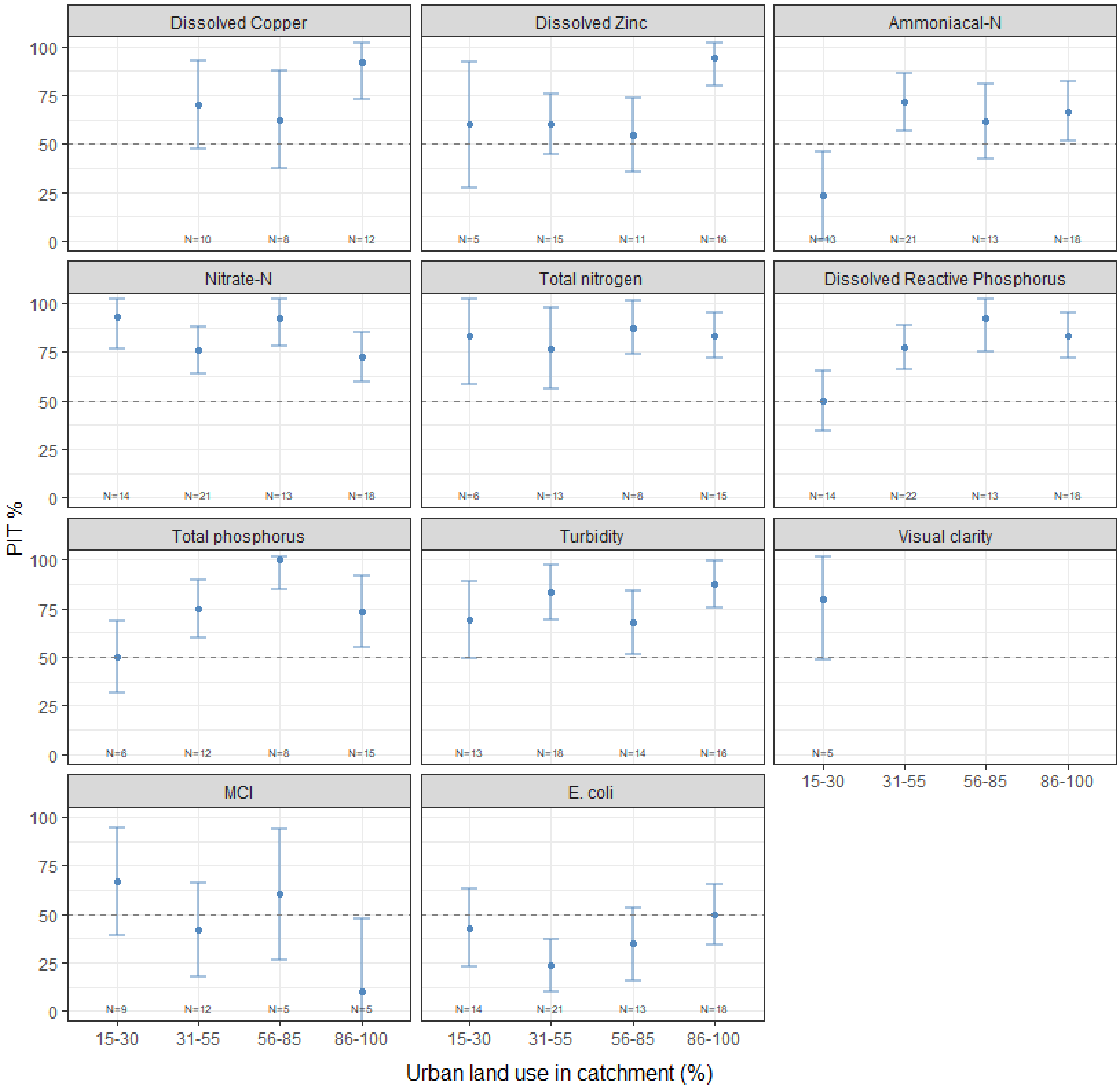

Figure 5-10: Proportion of improving 10-year trends by percentage of urban land cover in the upstream catchment for categories with >3 sites only. The error bars indicate the 95% confidence interval for the proportion of improving trends. The dashed grey line indicates where 50% of sites have improving trends. N indicates number of sites where trends were assessed for each variable and category.

5.3 Longer-term trends

Trends were assessed for the ~22-year time period of monitoring from January 1995 to December 2017 at 5 Auckland urban stream sites. These analyses indicate that there is considerably more certainty in the trend direction for the longer time frame as demonstrated by the smaller confidence intervals (Figure 5-11) compared to trends assessed over the shorter period of 10 years. In most cases the 22-year trend directions were the same as the corresponding 10-year trend directions, but the magnitudes changed. In a few cases the trend direction was different for the 10-year and 22-year trend periods, such as nitrate-N at site ARC-00019. These findings are the same as noted in the previous analysis of urban water quality trends (Gadd 2016) when comparing trends assessed over shorter and longer time frames.

In comparing the trend results for the 22-year data sets assessed according to the updated method, with the 20-year data sets assessed using the previous method (Larned et al. 2015; Gadd 2016), the trend direction has not changed with one exception. DRP at site ARC-00019 showed an increasing trend in the 22-year data set and in the 10-year data set but an indeterminant trend for the 20-year data set. For other sites and variables, the magnitude of the trends changed only by a percentage or two between the 20-year and 22-year analyses. As there were very few censored data for nitrate-N (0-1%) and ammoniacal-N (0-4%), differences here unlikely to be due to the different trend assessment method and more likely due to the different time frame assessed.

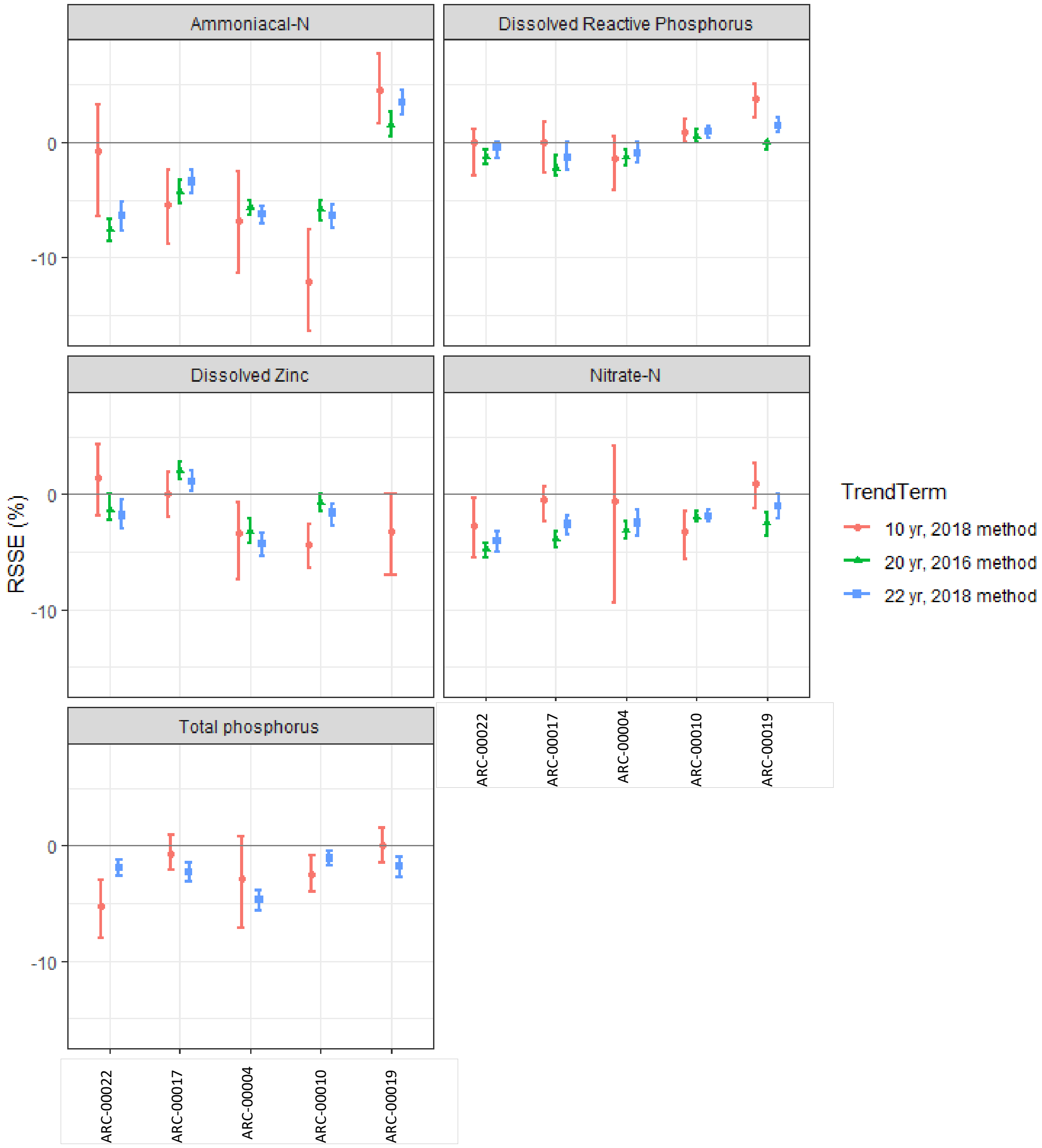

Figure 5-11: Comparison of short-term (red) and long-term (green and blue) trends in urban stream water quality for five Auckland sites with monitoring records greater than 15 years. The error bars indicate the 95% confidence interval for the Relative Seasonal Sen Slope. Trends in total phosphorus were not assessed in the 2016 report.

6 Summary and Recommendations

6.1 Summary of Findings

Median concentrations of water quality state variables ranged over two- to three-orders of magnitude between sites for ammoniacal-N, dissolved zinc, DRP, nitrate-N, turbidity and *E. coli*, whereas the range was much smaller for dissolved copper, TP, TN and clarity. The range in median concentrations demonstrates that although water quality can be poor in some urban streams, there are also some urban streams where water quality is not poor. Variation within a particular site was high for dissolved zinc, ammoniacal-N and *E. coli* across all sites. For nitrate-N and visual clarity there was a wide range in concentrations within some sites but not all sites. Within-site ranges were substantially smaller for dissolved copper, DRP and nitrate-N.

For *E. coli*, 56 out of 75 sites were in the E (Red) attribute state based on exceedance of the threshold for annual median *E. coli* concentrations. There were no sites where nitrate-N or ammoniacal-N were below the National Bottom Line (i.e., D attribute state), with either 95% or 100% of sites respectively in A or B attribute state. Around 42% of sites where dissolved zinc was measured, and about 25% of sites were dissolved copper was measured, had median concentrations that were greater than the ANZECC (2000) default guideline for 95% protection. For many of the sites where median concentrations were below the guidelines, the 75th percentiles exceeded the guideline values.

Urban water quality varied considerably between regions, but quality also varied within regions and there were only three regions (Auckland, Wellington, Canterbury) where there were more than 3 sites monitored. There were differences in the water quality state between REC classes for nitrate-N, dissolved zinc, turbidity, MCI and some metrics of *E. coli*. For DRP there was very little difference between REC classes.

For dissolved zinc, ammoniacal-N, DRP, TP, turbidity and nitrate-N, median concentrations tended to be higher at sites with more than 30% urban land cover in the upstream catchment. Linear regressions showed there were significant relationships between urban land cover in the upstream catchment and median concentrations for ammoniacal-N, dissolved zinc and copper (and for copper only when rural streams were included). Where data were available, impervious area in the catchment was a slightly better predictor of dissolved zinc and ammoniacal-N concentration than urban land cover. There was also a statistically significant relationship between impervious area and DRP.

This report used updated methods to interpret trend directions and to aggregate site trends by different categories. For trends assessed over 10-years (or 7-years for metals at some sites), there were more improving trends than degrading trends for urban streams nationwide, except in the case of *E. coli*. For *E. coli*, close to 50% of sites were unlikely (or very unlikely, extremely unlikely, etc.) to be improving and less than 30% of sites were likely to be improving. There were no clear differences in the trends (direction and magnitude, or percent improving trends) between regions with the exception of the Canterbury region for which most variables indicated a majority of improving trends. There were some differences between REC classes for DRP and *E. coli*; and some differences between urban land cover categories for DRP (based on PIT statistics) and TN (based on Relative Annual Sen Slope Estimates).

6.2 Recommendations

This is the second report for MfE concerning state and trends in urban stream water quality. The first report made several recommendations for future reporting, including expanding the reporting to other geographic areas, which was implemented in the current report. In this section we suggest further recommendations for future reporting and monitoring.

The main findings of the state and trends assessment were that a) water quality was typically poor in urban streams (e.g., ranking in attribute state C or D; exceeding water quality guidelines; poor-fair MCI scores) but that b) water quality was improving at more sites than it was degrading at (with the exceptions of *E. coli* and MCI). There was no breakdown of the trends according to state, for example to understand whether water quality is improving at the sites with the poorest water quality or at the sites where water quality is already acceptable. This analysis would assist in predicting the future state of New Zealand's urban streams and in understanding whether management actions to improve water quality should be targeted at particular types of streams. Such analysis could be undertaken in future reports of urban streams, and for other state and trend reports.

This report assessed state and trends for a time-period that was 2 years subsequent to the previous state and trends report (i.e., for state 2015-2017 rather than 2013-2015; for trends 2008-2017 rather than 2008-2015). The overall findings with respect to both state and trends are similar in both reports, although more sites (from additional regions) and additional variables were included in this updated report (TN, TP, MCI, turbidity, visual clarity and additional *E. coli* metrics). The addition of quantitative analysis of water quality state versus urban land cover or impervious cover demonstrated that these were useful predictors for dissolved zinc and copper (when rural streams were included) and ammoniacal-N (against impervious cover only). Further assessments of urban stream water quality should also include these predictors, and if possible, impervious surface data should be acquired for the locations where not currently available, and updated in locations where over 10 years old.

This report included water quality of urban streams in 12 regions, but there were data for dissolved copper and zinc from only 3 regions and concentrations at many sites repeatedly exceeded water quality guidelines. The assessment of state for other water quality variables did not suggest that these three regions had substantially poorer water quality than other regions, and therefore it is likely that urban streams in other regions will also have concentrations of copper and zinc that exceed water quality guidelines. Councils in these other regions should consider also monitoring copper and zinc in their urban streams. When doing so, councils should use trace-level analyses to ensure that trends can be assessed when sufficient data has been collected. In Canterbury streams, the use of a higher detection limit for dissolved copper has resulted in trends not being assessed at most sites due to very high percentages of censored data, despite the use of updated trend assessment methods that enable assessment in datasets with greater than 15% censoring.

Relatedly, many councils have recently begun obtaining reports from laboratories that include both "raw" measurements and the censored values. The "raw" data was not included in the current assessment of state and trends as the effect of using this "raw" data within such assessments has not been investigated. Statistical methods to deal with such data need to be investigated and in particular for cases where the detection limit has changed over time.

For many sites, trends were assessed as "indeterminant" due to insufficient data (and/or high variability). Flow-adjustment was not used in this report (or in the previous report, Gadd 2016) due to a lack of flow monitoring data at urban stream water quality monitoring sites. The effect of flow-adjustment on the trend results could be assessed in a future report, for the subset of sites where flow data are available. This would provide evidence for deciding whether flow adjustment should be undertaken in the future, where it is possible.

For dissolved copper and zinc in Christchurch, monitoring data were only available for seven years and a substantial proportion of those data were censored. The detection limit for these was decreased around March 2018 and we expect fewer censored values from that time forward. We do not recommend repeating these trend analyses for dissolved copper until there is sufficient data to enable assessment at most sites. Furthermore, the assessment of longer-term trends (~22 years) showed the same trends as the previous assessment for all but one variable x site combination. We do not recommend analysis of long-term trends for future reports on urban stream water quality until there are more than 5 sites with long-term (20-year) monitoring records.

7 Acknowledgements

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Annette Semadeni-Davies, NIWA calculated areas of impervious surfaces and urban land cover in Auckland, Wellington and Christchurch catchments. Simon Howard, NIWA, is thanked for assistance with R coding for plots in draft versions of this report.

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Appendix A List of Sites

Region	LAWA site ID	Site Name	Easting (NZTM)	Northing (NZTM)	REC class	Urban land cover (%)	Imper-vious surface area (%)
AC	ARC-00004	Lucas Creek at Gills Rd Bridge	1751448	5934310	WD_L	64%	30%
AC	ARC-00010	Oakley Creek at Carrington Creek	1751942	5917436	WW_L	98%	45%
AC	ARC-00014	Otaki Creek at Middlemore Crescent	1764285	5907017	WD_L	100%	47%
AC	ARC-00015	Otara Creek at East Tamaki Rd	1767401	5907336	WD_L	92%	45%
AC	ARC-00016	Otara Creek at Kennel Hill	1768314	5908177	WD_L	34%	18%
AC	ARC-00017	Oteha Stream at Days Bridge	1751305	5933319	WD_L	83%	39%
AC	ARC-00018	Pakuranga Creek at Botany Rd	1769952	5912814	WD_L	100%	52%
AC	ARC-00019	Pakuranga Creek at Greenmount Drive	1769452	5910614	WD_L	93%	46%
AC	ARC-00022	Puhinui Creek at Drop Structure	1766419	5904096	WD_L	68%	33%
AC	ARC-00034	Omaru Creek at Maybury St	1765998	5916564	WD_L	100%	45%
AC	ARC-00036	Avondale Stream at Shadbolt Park	1750666	5912101	WW_L	77%	28%
AC	ARC-00123	Lignite	1752318	5929264	WD_L	73%	NA
AC	ARC-00124	Lucas LTB @ Tennis	1751750	5934493	WD_L	61%	NA
AC	LAWA-102236	Avondale @ Reserve	1748383	5910930	WW_L	54%	NA
AC	LAWA-102237	Avondale @ Shadbolt Park	1750666	5912102	WW_L	77%	NA
BOPRC	EBOP-00183	Waioraka	1881741	5819389	WW_L	26%	NA
BOPRC	EBOP-00184	Otumanga	1880008	5819101	WW_L	37%	NA
ECAN	CCC-00001	Wairarapa Stream	1568232	5180932	CD_L	94%	43%
ECAN	CCC-00002	Waimairi Stream	1568214	5180801	CD_L	100%	36%
ECAN	CCC-00003	Avon River at Mona Vale	1568316	5180675	CD_L	93%	42%
ECAN	CCC-00004	Avon River at Carlton Mill Corner	1569718	5180888	CD_L	94%	43%
ECAN	CCC-00005	Riccarton Main Drain	1569000	5179666	CD_L	95%	56%
ECAN	CCC-00006	Addington Brook	1569408	5179456	CD_L	91%	57%
ECAN	CCC-00007	Avon River at Manchester Street	1570870	5180111	CD_L	93%	47%
ECAN	CCC-00008	Dudley Creek	1572554	5181780	CD_L	96%	43%
ECAN	CCC-00009	Avon River at Dallington Terrace/Gayhurst Road	1573541	5180839	CD_L	95%	47%
ECAN	CCC-00010	Horseshoe Lake Discharge	1574323	5182923	CD_L	55%	27%
ECAN	CCC-00011	Avon River at Avondale Road Bridge	1574732	5183186	CD_L	86%	45%
ECAN	CCC-00012	Avon River at Pages/Seaview Bridge	1577464	5182219	CD_L	86%	45%
ECAN	CCC-00013	Avon River at Bridge Street	1577671	5180443	CD_L	86%	45%
ECAN	CCC-00014	Heathcote River at Templetons Road	1565896	5176527	CD_L	71%	39%
ECAN	CCC-00015	Haytons Stream at Retention Basin	1566002	5177226	CD_L	65%	54%
ECAN	CCC-00016	Curletts Road Stream Upstream of Heathcote River	1566909	5177341	CD_L	90%	31%

ECAN	CCC-00017	Curletts Road Stream at Motorway	1566387	5177987	CD_L	90%	30%
ECAN	CCC-00018	Heathcote River at Rose Street	1568682	5175547	CD_L	70%	36%
ECAN	CCC-00019	Cashmere Stream at Sutherlands Road	1566067	5173618	CD_L	16%	5%
ECAN	CCC-00020	Cashmere Stream at Worsleys Road	1569011	5174785	CD_L	18%	13%
ECAN	CCC-00021	Heathcote River at Ferniehurst Street	1569138	5175242	CD_L	46%	25%
ECAN	CCC-00022	Heathcote River at Bowenvale Avenue	1571178	5175410	CD_L	47%	25%
ECAN	CCC-00023	Heathcote River at Opawa Road/Clarendon Terrace	1573051	5177245	CD_L	52%	29%
ECAN	CCC-00024	Heathcote River at MacKenzie Avenue	1573500	5177547	CD_L	53%	29%
ECAN	CCC-00026	Heathcote River at Tunnel Road	1575054	5177173	CD_L	53%	30%
ECAN	CCC-00027	Heathcote River at Ferrymead Bridge	1576472	5176779	CD_L	50%	28%
ECAN	CCC-00028	Smacks Creek at Gardiners Road	1566785	5187585	CD_L	30%	22%
ECAN	CCC-00029	Styx River at Gardiners Road	1566771	5186855	CD_L	28%	17%
ECAN	CCC-00030	Styx River at Main North Road	1569047	5186848	CD_L	39%	22%
ECAN	CCC-00031	Kaputone Creek at Blakes Road	1570381	5187659	CD_L	66%	31%
ECAN	CCC-00032	Kaputone Creek at Belfast Road	1572174	5187896	CD_L	33%	17%
ECAN	CCC-00033	Styx River at Marshland Road Bridge	1572338	5187407	CD_L	48%	26%
ECAN	CCC-00034	Styx River at Richards Bridge	1573965	5189454	CD_L	41%	23%
ECAN	CCC-00035	Styx River at Harbour Road Bridge	1574978	5194377	CD_L	29%	15%
ECAN	CCC-00036	Nottingham Stream at Candys Road	1564514	5172709	CD_L	67%	38%
ECAN	CCC-00037	Knights Stream at Sabys Road	1563704	5172481	CD_L	20%	25%
ECAN	CCC-00038	Halswell River at Akaroa Highway (Tai Tapu Road)	1564428	5171351	CD_L	24%	24%
ECAN	CCC-00040	Wilsons Stream	1571222	5190422	CD_L	27%	19%
ECAN	CCC-00041	Linwood Canal/City Outfall Drain	1575932	5177656	CD_L	100%	58%
ECAN	ECAN-00127	Taitarakihi Creek SH1	1459888	5084972	CD_L	24%	NA
ECAN	ECAN-00141	North Brook Upstream side of Bridge Marsh Rd	1569435	5203128	CD_L	50%	NA
ECAN	ECAN-00175	Taranaki Creek Gressons Rd	1570979	5205066	CD_L	35%	NA
ECAN	ECAN-10005	Heathcote River at Catherine St	1574394	5177513	CD_L	53%	29%
ECAN	LAWA-100405	Styx River-Styx Mill Reserve	1567934	5187735	CD_L	30%	NA
ECAN	LAWA-100410	Dudley Creek	1572809	5182447	CD_L	96%	NA
ECAN	LAWA-100420	Avon-Victoria Square	1570498	5180476	CD_L	49%	NA
ECAN	LAWA-100421	Avon at USCA	1566173	5180854	CD_L	100%	NA
ECAN	LAWA-100422	Waimari	1567036	5181167	CD_L	91%	NA
ECAN	LAWA-102235	Kaputone Ck	1570849	5188905	CD_L	41%	NA
ES	ES-00019	Otepuni Creek at Nith Street	1242697	4849257	CD_L	24%	NA
GDC	GDC-00007	Kopuawhakapata Stream at Hirini St	2037810	5707539	WD_L	77%	NA
GDC	GDC-00020	Waikanae Creek at Grey St Bridge	2036799	5707865	WD_L	64%	NA
GDC	GDC-00026	Wainui Stream at Pare Street	2041143	5705007	WD_L	30%	NA
GDC	LAWA-100719	Wainui Stream at Heath Johnston Park	2039784	5705953	WD_L	46%	NA

GWRC	GW-00002	Mangapouri Stream at Bennetts Rd	1780881	5487452	WD_L	34%	NA
GWRC	GW-00015	Porirua Stream at Glenside Overhead Cables	1753269	5438172	CW_L	33%	14%
GWRC	GW-00016	Porirua Stream at Milk Depot	1754366	5443031	WW_L	33%	14%
GWRC	GW-00018	Karori Stream at Makara Peak Mountain Bike Park	1744193	5426683	CW_L	59%	24%
GWRC	GW-00019	Kaiwharawhara Stream at Ngaio Gorge	1749069	5431077	CW_L	38%	17%
GWRC	GW-00264	Waiwhetu Stream at Whites Line East	1760959	5434320	WW_L	53%	NA
HBRC	HBRC-00063	Taipo Stream at Church Road	1930939	5618935	WD_L	51%	NA
HRC	LAWA-101921	Lake Horowhenua Inflow at culv d/s Queen St	1791699	5501801	WD_L	59%	NA
HRC	LAWA-101927	Makomako Road Drain at L Horowhenua	1790903	5500828	WD_L	41%	NA
HRC	LAWA-101955	Queen Street Drain at L Horowhenua	1791541	5501612	WD_L	36%	NA
MDC	MDC-00014	Murphys Creek at Taylor confluence	1678587	5404337	WD_L	36%	NA
NCC	NCC-00004	Poorman Valley Stream at Seaview Rd	1618958	5427353	CW_L	18%	NA
NCC	NCC-00006	Jenkins Creek at Pascoe St	1620147	5428267	CW_L	25%	NA
NCC	NCC-00009	York at Waimea Rd	1622314	5428182	CW_L	21%	NA
NRC	NRC-00004	Waiarohia Stream @ Second Ave Footbridge	1719028	6045812	WW_L	24%	NA
NRC	NRC-00019	Hatea River @ Mair Park Footbridge	1720265	6047089	WW_L	18%	NA
ORC	ORC-00021	Kaikorai Stream at Brighton Road	1400312	4913362	CD_L	38%	NA
ORC	ORC-00095	Water of Leith at Dundas Street Bridge	1407404	4918173	CW_L	16%	NA
ORC	ORC-00119	Lindsays Creek at North Road Bridge	1407703	4919273	CD_L	19%	NA
TDC	TDC-00042	RW Watercress @ u-s Dairy Factory	1584319	5477128	WW_L	20%	NA
TRC	TRC-00063	Huatoki Stream at Molesworth Street	1692797	5676423	WW_L	29%	NA
TRC	TRC-00079	Mangati Stream 200m d/s railbridge	1700096	5678042	WW_L	36%	NA
TRC	TRC-00080	Mangati Stream adj Te Rima Plce footbridge	1699382	5679102	WW_L	36%	NA
WRC	EW-00011	Kirikiriroa Stm at Tauhara Dr	1799226	5819872	WD_L	56%	NA
WRC	EW-00020	Mangakotukutuku Stm (Rukuhia) at Peacockes Rd	1802438	5812475	WW_L	18%	NA
WRC	EW-00098	Waitawhiriwhiri Stm at Edgecumbe Street	1799931	5816873	WW_L	50%	NA
WRC	LAWA-100272	Bankwood Stream @ Emerald Tce	1800357	5818318	WD_L	93%	NA
WRC	LAWA-100280	Mangakotukutuku Stream (Rukuhia) @ Pelorus Street	1802003	5811494	WW_L	31%	NA

Notes: NA = Not available

Appendix B Time series plots of the data

Appendix C Time series plots of the 22-year data sets