River Water Quality Current State and Trends in Tāmaki Makaurau / Auckland to 2024

State of the Environment Reporting

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August 2025

Technical Report 2025/20

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Executive summary

This report presents the current state of river water quality in Tāmaki Makaurau, assesses it against the National Policy Statement for Freshwater Management (NPS-FM) 2020 National Objectives Framework (NOF) water quality attributes and additional draft regional water quality attributes, and explores how water quality has changed over the past seven years. It supports the *State of the environment 2025* report for the Auckland region, *Te oranga o te taiao o Tāmaki Makaurau – The health of Tāmaki Makaurau Auckland's Natural Environment in 2025: a synthesis of Auckland Council State of the Environment reporting*.

Water quality can be affected by land use activities, point and diffuse source discharges, land and stream channel erosion, as well as seasonal and climatic variability. River water quality is monitored monthly at 37 sites across Auckland, covering a range of parameters including water temperature, dissolved oxygen, nutrient concentrations (nitrogen and phosphorus), metal contaminants (copper and zinc), suspended sediment and water clarity, and faecal indicator bacteria (*Escherichia coli*).

The current state was assessed based on the five-year period from 1 July 2019 to 30 June 2024. Trend analysis followed methodology established for national water quality reporting which assesses the confidence in the trend direction and provides an estimate of the magnitude of change. Assessment focused on the seven-year period from 1 July 2017 to 30 June 2024 as multiple methodology changes in 2017 presented a limitation on long-term time series analysis.

Several historically significant climatic events occurred over these state and trend periods including a severe drought (2020) and several extreme rainfall and flooding events (2023). Water quality monitoring was representative of the flow conditions that occurred, although extreme flow events were not fully captured because of health and safety considerations.

Water quality was generally in the best condition in streams draining native forest catchments. Water quality was progressively worse in streams flowing through predominantly rural, and urban catchments, with urban streams in the poorest condition. Variation in some aspects of water quality including water clarity and total suspended solids was more clearly explained by climate and underlying geology than land cover.

All monitored rural and urban waterways were impacted by high *E. coli* levels that indicate faecal pollution. Most rural and urban waterways have elevated nutrient concentrations that can influence instream ecosystem health and function.

Multiple streams within the wider Pukekohe area failed the national bottom line for nitrate toxicity. This is a long-standing issue associated with nitrate contamination in several of the underlying shallow aquifers.

Multiple streams within urban areas failed the proposed regional bottom line for chronic zinc toxicity (i.e. consistently occurring exposure) and occasionally exceeded acute toxicity guidelines.

There were key changes to the assessment of chronic copper and zinc toxicity regional attributes. This change in assessment meant that the risk from chronic, long-term copper exposure on aquatic life was less than we had previously reported. New Zealand guideline values for acute (short-term) toxicity effects from copper and zinc on aquatic life have recently been developed and monthly water quality observations were also compared to these guidelines.

Several monitored streams failed the national bottom line for visual clarity (as a proxy for suspended sediment) including some rural sites and one urban site. This is likely due to a combination of catchment-specific factors, including environmental conditions such as climate and underlying geology.

Overall, there was wide variation in trend direction, and magnitude across river sites, land cover types, and among different measures of water quality. This highlights the complexity of interactions between land use activities, different contaminant pathways, and climatic influences.

Generally, there was a higher proportion of degrading trends where water quality was most impacted (i.e. bad and getting worse). In one catchment with increasing urban development (Vaughan Stream at Long Bay) there are indications that most measures of water quality are at least being maintained.

There were many site-specific issues, and multiple sites where a higher magnitude of change (improving or degrading) was observed across multiple measures of water quality. Potential further follow-on investigations have been identified for several of these locations.

The findings of this report can be used to inform current state and trends as required for NPS-FM implementation with a focus on ecosystem health water quality and human contact attributes. This also complements and informs understanding on other compulsory values under this policy. This report contributes to the regional evidence base to inform the effectiveness of policy initiatives. Long term, regional information provides important context for operational (and restoration) activities and other monitoring and research in the region by providing the expected range of various measures of water quality across different broadscale pressures, including seasonal variability.

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1 Introduction

Rivers form the interface between land and sea, conveying land-based contaminants to our estuaries, harbours, and coasts. In Tāmaki Makaurau, many waterways drain directly to the coast before they can merge with others to form larger river systems. Consequently, many streams are small in length, with most less than a few metres wide. The topography of Tāmaki Makaurau is predominantly gentle in comparison to other regions of New Zealand. This strongly influences the nature of Auckland's rivers, along with the underlying geology, typically resulting in slow flowing, low gradient, soft-bottomed rivers. High gradient rivers with hard stony substrates are mostly restricted to catchments that drain the Waitākere Ranges, Hunua Ranges and Aotea Great Barrier Island. The largest river systems in the region include the Hōteo River to the north, and the Wairoa River to the south.

1.1 Why do we monitor river water quality?

Auckland Council has obligations to report on the state of the environment in accordance with section 35 of the Resource Management Act (1991). This includes monitoring the state of water quality in our rivers and streams. The river water quality monitoring programme does not operate in isolation but forms part of the wider state of the environment monitoring programme network which contributes to understanding of the impacts of freshwater discharge to coastal and nearshore environments.

A key objective of the river water quality monitoring programme is to contribute to the evidence base to support management of growth and development and changing land use pressures in the region. State of the environment monitoring is best suited to identification of large scale and/or cumulative impacts of the mosaic of activities and land uses that occur over long time periods. State of the environment monitoring can also provide important context for shorter term monitoring undertaken for specific projects by providing understanding of climatic variation, seasonal patterns, and the expected range of variability in similar waterways.

1.1.1 National Policy Statement for Freshwater Management

The National Policy Statement for Freshwater Management 2020 (as amended October 2024) (NPS-FM) forms part of the Government's *Essential Freshwater* package which outlines the statutory direction for freshwater management across New Zealand. The main aims of this package, introduced in 2020, are to prevent further degradation of New Zealand's freshwater ecosystems, improve water quality, and address past damage to return freshwaters to a healthy state. The NPS-FM (2020) is underpinned by the fundamental concept of Te Mana o te Wai. Te Mana o te Wai prioritises the health and well-being of wider freshwater environments over provisions for human health needs and other forms of utilisation. Te Mana o te Wai, and other components of the NPS-FM, are managed through the National Objectives Framework (NOF) which requires Auckland Council to identify values associated with water bodies in the region and set ambitious but achievable desired environmental outcomes and objectives.

Under the NOF, the NPS-FM identifies four compulsory national values (Appendix 1A of the NPS-FM) and 22 accompanying compulsory national attributes (Appendix 2A and 2B of the NPS-FM) which must be managed to safeguard and promote the health of the region's lakes, rivers and streams. The four compulsory national values are:

Ecosystem health – refers to the management of freshwater ecosystems (including lakes, rivers and wetlands) to support diverse communities of native plants and animals. This value includes five biological and physical components that are supported by several compulsory attributes (see Table 1-1). Auckland Council has identified additional regional attributes which also influence ecosystem health in rivers.

Human contact – refers to the way people connect with the water through a range of recreational activities, such as swimming, boating, and fishing. This value includes two compulsory attributes for rivers.

Threatened species – refers to the maintenance of critical habitats and conditions required to sustain populations and enable the recovery of threatened native plants and animals.

Mahinga kai – refers to native freshwater species that are traditionally used for food, tools, and other customary practices, as well as the places where they are found and how they are gathered.

Our river water quality programme includes monitoring of attributes that contribute to the first two values: ecosystem health and human contact. Specifically, monitored information is required to:

- Assess the baseline state, and current state of defined water quality attributes to inform identification of targets for each attribute, and to measure the success of achieving those targets over time.
- Monitor trends over time to identify if water quality is being maintained, or improved, and to take action if environmentally meaningful degradation is identified.

1.2 Scope of this report

The purpose of this report is to assess the current state of river water quality across the Auckland region and identify changes in water quality over time.

This report:

- Describes the current state (based on data from 1 July 2019 to 30 June 2024) of water quality in the region including seasonal variability.

- Assesses the **current** grade for river water quality NPS-FM 2020 attributes for 'ecosystem health' and 'human contact'[1] values for the same period. Other ecosystem health attributes are also reported on in separate reports (see Table 1-1).
- Identifies temporal trends in water quality variables over time (period from 1 July 2017 to 30 June 2024) to understand where water quality is being maintained, improved, or degraded.

The state and trend analysis presented here forms a 'screening exercise' to identify where water quality is degraded or degrading, and to identify improvements in water quality. Further investigations including alternative trend analysis methods, modelling approaches (such as Auckland Councils Freshwater Management Tool) or targeted project based monitoring, are necessary to characterise drivers and inform management actions.

Table 1-1: Compulsory national and proposed regional values and attributes for rivers in the Auckland region (NPS-FM 2020).

Ecosystem type		Biophysical component	Attributes	Current state reporting
Rivers	Ecosystem health	Aquatic life	Macroinvertebrates (MCI/QMCI)	Surrey and Storey, 2025
			Macroinvertebrates (ASPM)	Surrey and Storey, 2025
			Fish (IBI)	Surrey and Storey, 2025
			Periphyton	Auckland Council (2023) (*interim*)
		Water quality	Ammonia: toxicity to aquatic life	*This report*
			Nitrate: toxicity to aquatic life	*This report*
			Dissolved reactive phosphorus (DRP)	*This report*
			*Dissolved inorganic nitrogen (DIN)	*This report*
			Dissolved oxygen	Young et al., 2025
			Dissolved oxygen (below point sources)	NA
			**Dissolved copper: toxicity to aquatic life	*This report*
			**Dissolved zinc: toxicity to aquatic life	*This report*
			**Temperature	Dikareva, 2025 (In prep)
			Suspended fine sediment	*This report*
		Physical habitat	Deposited sediment	Data requirements not yet met
		Ecosystem processes	Ecosystem metabolism (EM)	Young et al., 2025
		Water quantity	Not included to date	Lorrey et al., 2025
	Human contact		*Escherichia coli* (*E. coli*)	*This report*
			E. coli (primary contact sites)	Alternative reporting on primary contact sites. See Safeswim https://safeswim.org.nz/
	Threatened species		Not included to date	NA
	Mahinga kai		Not included to date	NA

* Draft attribute **not** included within the NPS-FM 2020. | ** Additional regional attributes

[1] The report does not include an assessment of primary contact sites (human contact value) which are covered by Auckland Council's Safeswim programme (www.safeswim.org.nz).

1.3 Supporting information

This report is one of a series of technical publications prepared in support of *Te oranga o te taiao o Tāmaki Makaurau – The health of Tāmaki Makaurau Auckland's Natural Environment in 2025: a synthesis of Auckland Council State of the Environment reporting*.

All related reports (past and present) are published on the Knowledge Auckland website.

All data supporting this report can be requested through our **Environment Auckland Data Portal**. Here you can also view live rainfall data and use several data explorer tools.

In 2025, the Water Quality and River Ecology Data Explorer [2] was launched to provide an interactive summary of water quality and freshwater ecology data across multiple domains. This dashboard provides summary statistics and interactive graphics for river water quality data collected from July 2009 to June 2024. Readers of this report can explore additional insights on the dashboard.

[2] https://environmentauckland.org.nz/Data/Dashboard/456

2 Methods

2.1 Monitoring network

Auckland Council's river water quality monitoring programme is primarily designed for detecting long-term changes in water quality across the region. The network aims to be regionally representative by including a range of river and catchment sizes, stream orders, catchment locations, and catchment land cover composition. This enables Auckland Council to present a region-wide perspective on water quality and infer the likely water quality of other rivers in the region that are not monitored.

The programme started in 1986 and originally included 17 sites (nine of which continue to be monitored) and has incrementally expanded over time with more substantial reviews and expansion of the network undertaken in 2009 and 2022 (e.g. Auckland Regional Council, 1995; Neale, 2010).

This report focuses data from 37 monitoring sites across the region (Figure 2-1). There were 12 additional sites added to the network in July 2022 however these sites do not yet meet minimum data requirements for assessing state and trends (at least three years of data – see section 2.4)

The river water quality monitoring network has always maintained a higher proportion of urban streams relative to the proportion of this land use type in the region (Table 2-1). This is considered to be appropriate due to the greater extent and diversity of pressures on streams in urban environments, and consequently greater variability in water quality and ecosystem health. This bias should be considered in any overall 'regional' summary of water quality.

Table 2-1: Number of monitoring sites in the regional network compared against the percentage cover of broadscale land cover categories (see section 2.1.1 below) across the region ()

Land cover category	No. of sites	% of sites		Regional coverage** (Griffiths and Lawrence, 2025)
Native forest	4	11%		27.1%
Exotic forest	2	6%		9.8%
Rural low	6	17%	47%	46.3%
Rural high	8	22%		
Pukekohe	3	8%		
Urban	13*	36%		12%

* Plus one urban 'project' site
** The remaining 4.8% includes water bodies and 'other' such as bare surfaces.

Figure 2-1: River water quality monitoring sites and the upstream catchments coloured by the broad land cover classification for each site (see section2.1.1). Note: 12 additional sites were established in July 2022 which did not meet minimum data requirements for reporting and are therefore not displayed here.

2.1.1 Land cover analysis

A geospatial assessment of land cover was undertaken for each catchment upstream of the monitoring location using an updated land cover database compiled for the Auckland region by Auckland Council to provide a contemporary snapshot of land cover based on aerial imagery from summer 2023/2024. This regional version of the land cover database is referred to here as LCDB (regional update 23/24) (Auckland Council, 2025) (see Appendix 1 for further details).

The upstream catchment areas for each river water quality monitoring site were defined using topography and the existing Auckland Council 'permanent streams network' layer. All polygons were manually reviewed by a trained GIS specialist, including reference to the Auckland Council 'underground services stormwater network in urban catchments' layer, as well as the 'overland flow path' layer and topographical contours in rural catchments.

The detailed land cover types identified in the regional LCDB update were aggregated into broader categories (see Appendix 1 for aggregation). The total area of each broad land cover category within each upstream catchment was then calculated and expressed as a percentage of the total catchment area. A dominant land cover class was then assigned to each site based on the rules outlined in Table 2-2. The dominant land cover categories that are used here are modified from the decision rules originally established by Snelder and Biggs (2002) (and reaffirmed by Fraser and Snelder (2021)) that were used in the national River Environment Classifications (REC). The categories applied here are more conservative than the national REC as we use a lower threshold (of >7% compared to >15%) for the area of urban land use in the catchment to categorise a site as 'urban'. These thresholds are relatively arbitrary and are not based on non-linear changes or thresholds in terms of ecological responses (Fraser and Snelder, 2021).

Streams within the wider Pukekohe area are reported separately from other 'rural high' streams throughout this report as these streams have been affected by intensive horticultural production and high connectivity between groundwater and surface water and may require different management considerations.

Pakuranga Stream is reported separately from other 'urban' streams throughout this report as a 'project' site as the site is not considered representative of diffuse urban land use pressures from a regional perspective.

Table 2-2: Dominant land cover class definitions.

Dominant Land Cover Class	Definition
Urban – Project	A single monitoring site reported separately from other urban streams due to point source discharges at this location.
Urban	More than 7% urban land cover in the upstream/surrounding catchment.
Pukekohe	Catchments within the southern Auckland wider Pukekohe area with a high proportion of horticultural production and a distinct surface water management issue associated with high connectivity with aquifers in this area.
Rural – High	Less than 50% exotic or native forestry cover remaining in the upstream/surrounding catchment.
Rural – Low	More than 50% of the upstream catchment retains indigenous or exotic forest cover.
Exotic	More than 80% of the upstream/surrounding catchment within exotic forestry.
Native	More than 95% native forest or scrub remaining within the upstream/surrounding catchment. These are intended to represent reference quality conditions that have a very low level of land use pressure influence though they are not necessarily 'pristine'.

2.1.2 Biophysical classes

The New Zealand River Environment Classification (REC2.4) groups segments of a river based on their upstream catchment characteristics. The REC classifications are expected to explain a degree of natural variation in water quality, nutrient and trophic responses, and sediment dynamics among streams (Stoffels et al., 2021, Canning, 2020). These classifications have been used extensively to inform freshwater policy development and implementation in New Zealand.

The third tier of the REC, based on combined climate, source of flow, and geology information, is utilised in spatial classifications and different attribute band ranges for the deposited and suspended sediment attributes and the periphyton attribute under the NPS-FM 2020.

There are two climate classes across Auckland: Warm Wet (WW) and Warm Dry (WD). All waterways in Tāmaki Makaurau are defined as low elevation (L) in terms of source of flow. Dominant geology types include soft sedimentary (SS), hard sedimentary (HS), volcanic acidic (VA), and Alluvium (Al) (Snelder et al., 2004). There are seven main third tier REC classes that occur across the region. The proportion of sites across the third-tier river environment classes (REC) closely aligns with regional coverage (Table 2-3), suggesting the monitoring network provides good representation at the regional scale.

- WW/L/VA streams within Auckland fall within three main areas, the Waitākere ranges, the southern Manukau / Pukekohe horticultural area, and Aotea/Great Barrier Island.
- WD/L/Al class primarily includes small streams that discharge directly to the west coast along the Kaipara South Head peninsula around Woodhill Forest. There are no river water quality monitoring sites in this class.
- WD/L/HS class waterways are primarily restricted to the Hauraki Gulf Islands of Waiheke Island, Ponui Island, and Motutapu Island.

- **WW/L/HS** class waterways are primarily located within the Hunua ranges and foothills. This includes Wairoa River, and Papakura Stream.
- **WD/L/SS** class is associated with the Kaipara South Head and Awhitu peninsulas, in the Whitford embayment, and urban areas of the North Shore south of Ōkura Estuary. The WD/L/SS class also aggregates WD/L/Misc areas which includes a high proportion of waterways within the central urban isthmus.
- **WD/L/VA** class waterways are located within the southern Manukau / Pukekohe horticultural area, smaller tributaries but are also found within Māngere and eastern Tāmaki urban areas. The waterways in this class in the southern Manukau are typically smaller tributaries near the coast while the main streams in this area are classified as WW/L/VA.
- **WW/L/SS** is the most common class across the region. This also aggregates WW/L/Misc areas within urban catchments.

Table 2-3: Long-term regional monitoring sites in this report compared against percentage (%) of stream length in each REC class across the region (Auckland Council, 2021; MFE, 2020).

Suspended Sediment Class	REC classes	No. sites	% of sites	Proportion of stream length across region
1	WW/L/VA	4	11%	14%
	WD/L/Al	0	0%	3.1%
2	WD/L/HS	2	5%	2.7%
	WW/L/HS	4	11%	9.8%
	WW/L/SS	18	49%	51%
	WD/L/SS	7	19%	14%
	WD/L/VA	2	5%	3.8%

* Not including REC classes with <2% coverage across region

2.2 Data collection

Water samples are collected from the surface (approximately the top 0.3 m of water) of each stream site by lowering bottles into the water directly either by hand or via a sampling pole. Water samples are analysed in the laboratory for forms of key nutrients – nitrogen and phosphorus, forms of copper and zinc, turbidity and total suspended solids, *Escherichia coli* bacteria, and other measures of water chemistry including water hardness, and dissolved organic carbon. Other aspects of water quality are measured directly in the stream using a handheld instrument including water temperature, dissolved oxygen, conductivity and salinity, pH, and other measures of turbidity. Refer to Appendix 2 for a detailed list of measures, and analytical methods.

Watercare Services Ltd. analysed samples prior to June 2017 and RJ Hill Laboratories analysed samples from July 2017 to the current date. In addition to the change in laboratories, there have been several changes to laboratory analytical methods and detection limits over time. These changes are outlined in Appendix 2.

Sites are grouped into sampling runs within a spatial area and all sites are sampled within a timespan of three weeks within each month. Sites are visited in the same order on each sampling occasion to ensure sampling at each site occurs at approximately the same time of day each month. The addition and removal of sites has led to changes of run order and sampling times over the years particularly in 2016 and 2022 associated with programme reviews.

NIWA previously monitored the Rangitōpuni River and Hōteo River sites and provided the data to Auckland Council[3]. NIWA discontinued monitoring in 2021 and 2023 respectively for these sites. Auckland Council started monitoring at Rangitōpuni River in 2016 and only Auckland Council results are provided in this report for this site. Auckland Council only started monitoring at Hōteo River in 2023 and so only NIWA results are provided in this report for this site. NIWA assessment did not include measures of salinity, total suspended solids, copper and zinc.

2.2.1 Quality assurance

All water quality data are stored in Auckland Council's water quality archiving database (KiWQM).

National Environmental Monitoring Standards (NEMS, 2019) are utilised for data verification and quality assurance to ensure consistency and accuracy across monitoring programmes. Auckland Council adopted the NEMS quality coding (QC) framework in January 2020 (NEMS, 2020). Data collected prior to this used the IANZ-certified Auckland Council Hydrological 10-151 QC system.

Data identified as poor quality by either QC standard were excluded from all analyses.

2.3 Flow regime

Flow adjustment can be undertaken to account for the influence of instantaneous flow on water quality observations. However, flow adjustments would not account for all the influence of flow regime variability (such as climatic oscillations) and would introduce additional unquantified uncertainty in relation to how adjustments are undertaken (Snelder and Kerr, 2022). For the purposes of regional state and trend assessments it was not recommended to adjust water quality results for differences in flow conditions and flow adjustment is not undertaken for this report. However, the representativeness of water quality sampling across the flow conditions that occurred over the full five year (state) and seven year (trend) periods was assessed.

2.3.1 Assessing representativeness of water quality sampling across flow conditions

Not all water quality monitoring sites are paired with flow monitoring sites. There were 13 water quality monitoring sites that had paired (or nearby) hydrology monitoring stations where at least 80% of the flow record was available over the assessment periods. We evaluated the similarity between the distributions of the continuous daily flow data, and the sample occasion flows for the five-year state and seven-year trend periods in two ways.

[3] Further information can be obtained from https://www.niwa.co.nz/freshwater/water-quality-monitoring-and-advice/national-river-water-quality-network-nrwqn.

Firstly, for each site and time period, we generated probability P-P plots of the continuous and sample distributions. A P-P plot compares the cumulative distributions of two datasets to assess how closely they agree. When the distributions are similar, the points will appear along the 1:1 line. Deviations from the 1:1 line, and the shape of the relationship can provide insight into the distributional differences between two datasets.

Secondly, we also quantitatively assessed whether there was bias in the representation of the flow distribution at each site by the observations. This assessment used the Kolmogorov-Smirnov (K-S) test (following Snelder and Kerr, 2022) to assess whether the distribution of flows on observation dates matched the distribution of all flows within each assessment period. We interpreted the p-values >0.05 for these tests as strong evidence that the distribution of observation-date flows was an unbiased sample of all flows within each five-year assessment period. Details of the use of the K-S test to determine similarity in flow distributions are provided in Snelder and Kerr (2022).

An additional analysis was carried out to assess how well instantaneous streamflow was represented by water quality observations over a five-year period (July 2019 - June 2024). For 17 sites with paired flow monitoring, low or unknown quality data points were removed. Mean and median discharge values and discharge exceedance probabilities were calculated for each site. Flow duration curves were generated and overlaid with water quality observations to visualise coverage across the flow range. The proportion of streamflow represented by water quality observations was calculated for each site, along with the average proportion of time represented across all sites.

2.3.2 Characterising flow regime

To provide some additional context, we evaluated the deviation of the full flow records over each assessment period from the long-term average flow regime. We evaluated this on a monthly basis to compare deviations from the average condition across different seasons. The long-term flow regime was characterised by taking the mean daily flow within each month of the entire record and then taking the mean of those values for each calendar month. Then, the mean monthly flow from the five and seven year periods assessed here was calculated and compared to the long term flow regime. Mean monthly flows for the assessment periods were only compared when there were at least 80% of flow observations available over the period.

We plotted these data to visualise the variability of flow regimes between assessment periods and the deviation of flows in each assessment period to the long-term flow regime. Following Snelder and Kerr (2022), we did not undertake formal statistical testing of the significance of the differences in flow regimes.

2.4 State assessment

All state and trend analyses were undertaken by Land and Water People (LWP) and outputs including graphics were provided to Auckland Council. Detailed methodology was described by Fraser (2025) with an abridged version outlined here. For all water quality state analyses, state was assessed based on five years of observations from 1 July 2019 to 30 June 2024

2.4.1 Data processing

2.4.1.1 Minimum data

For this study we applied the following minimum data rules for the calculation of summary statistics consistent with national analysis (Whitehead et al., 2022), recognising some missed sampling events (e.g. Covid-19 lock down periods or extreme weather conditions), and exclusion of any poor quality data is expected

- Final result: At least 90% of sampling occasions (54 out of 60 months).
- Interim result: At least 90% of sampling occasions over at least three out of five years (e.g. >32 observations).

This report covers all sites that met the 'final' standard and four sites[4] that only met the 'interim' standard for state assessment. Additional monitoring sites that were established in 2022 did not meet these minimum data requirements and are not reported here.

Occasionally, sampling events are delayed or early and the date of sampling may deviate from the intended 'month' resulting in two samples collected within a month and no observations in the preceding or following month.

2.4.1.2 Censored values

Censored values are those that are below a laboratory detection limit or above the reporting limit. These values were replaced by imputation for the purposes of calculating summary statistics. Values below the detection limit were replaced with values generated using regression on order statistics (ROS; Helsel, 2012) following the procedure described in Larned et al., (2015). Values above the detection limit were replaced with values estimated using a procedure based on "survival analysis" (Helsel, 2012). The supplementary file outputs provide details about whether and how imputation was conducted for each site.

2.4.2 Distribution of values

Summary statistics (minimum, maximum, 5th, 25th, 50th, 75th, 95th percentiles) were calculated for each site and water quality parameter for the five-year period of 1 July 2019 to 30 June 2024. Percentiles were calculated using the Hazen method.

River water quality state is also summarised as violin plots for each parameter demonstrating the distribution of observed values across sites grouped by the dominant land cover class assigned. Distributions are for raw values (without imputation) with the exception of pH adjusted ammonia and bioavailable zinc and copper as these adjustments are undertaken after imputation.

2.4.3 Seasonal variation

Previous analyses undertaken for AC indicated that many of the monitored water quality variables have a seasonal response. For the current assessment, we evaluated the seasonality of the water

[4] Hōteo River, Mangawheau Stream, Newmarket Stream, and Nukumea Stream

quality observations over the five-year period from July 2019 to June 2024 using the Kruskall-Wallis test where seasons were defined as each calendar month. We identified site/variable combinations as seasonal where the p-value from the Kruskall-Wallis test was 0.05 or lower. In addition, we visualised the site/variable seasonal response to explore patterns in terms of timing of peaks and amplitudes of seasonal variation per month. To do this, we evaluated the median of the observations for each season and standardised these by the median from the five year state period and displayed these as a colour gradient from bright yellow (higher than median) to dark purple (lower than median). Results are only reported for site/variable combinations where each season (month) had at least four observations.

2.4.4 National Objectives Framework Attributes

For each attribute, the NOF defines attribute bands as four (or five) categories, which are designated A (the best) to D (the worst), or A to E, in the case of the *E. coli* attribute. State is assessed by grading attributes into specific bands using various statistical metrics (e.g., median and 95^{th} percentile, see Tables 3 - 5). The lowest (worst) band of the contributing metrics determines the overall band for that attribute state assessment. The numerical ranges for each attribute applied here are outlined in Appendix 3. The 'National Bottom Line' refers to the minimum acceptable state for each attribute that councils must meet, or work towards meeting over time. For some attributes, the national bottom line is between bands C and D, whereas for some attributes, the national bottom line is between bands B and C (nitrate and ammonia toxicity). For some attributes, no bottom line is specified. This difference is displayed visually in this report for the ecosystem health value with different colours used for the state band C depending on whether the national bottom line is above or below the C band (Table 2-4).

Dissolved inorganic nitrogen (DIN) was not included as an attribute in the NPS-FM 2020. The assessment in this report is based on a draft attribute table presented in 2019 by the Freshwater Science and Technical Advisory Group. The draft DIN bottom line was set at the same level as the A band threshold for nitrate toxicity. Scientific consensus was not achieved on the use and application of these band thresholds. However, the Ministry for the Environment have indicated that these draft bands remain useful for councils to consider for nutrient management (MfE, 2023) and therefore an assessment of DIN is incorporated here.

Table 2-4: NPS-FM attribute bands generalised narratives used to describe state.

	NPS-FM attribute band					
Ecosystem health	A	B	C	C*	D	
	Conditions similar to natural reference state	Conditions slightly impacted, causing minor stress to sensitive fish and macroinvertebrate species.	Moderate impact on conditions and stress to sensitive fish and macroinvertebrate species. * Below the National Bottom Line.		Conditions are highly impacted, and ecological communities are altered, and sensitive species may be lost.	
Human contact	A	B	C		D	E
	Very low risk of infection*	Low risk of infection	Moderate risk of infection		High risk of infection	Very high risk of infection

*Risk of infection is the overall average based on a random exposure on a random day and not all infections will result in illness.

2.4.4.1 Regional attributes – copper and zinc

Auckland Council have proposed including copper and zinc toxicity attributes as additional indicators of the water quality component of ecosystem health recognising that metals can be key contaminants to aquatic life in urban streams (Auckland Council, 2023).

Work is still underway to develop attribute band tables for regional use. Draft strawman attribute tables were originally developed by Gadd et al., (2019) based on ANZECC 2000 chronic toxicity guidelines. These draft guidelines were revised in Gadd et al., (2023) based on the 2023 draft Australia and New Zealand Fresh Water Quality Guidelines for chronic copper and zinc toxicity in freshwater which incorporate a bioavailability adjustment to better account for the risk of toxicity effects on aquatic life. Those draft guidelines are under review and changes are expected before finalisation, including revisions to the implementation of bioavailability adjustments.

Currently, the attribute table presented in Gadd et al., (2023) provides the best framework available to assess the current state of this attribute (see Appendix 3). This aligns with assessment previously undertaken to define the baseline state for the region (Auckland Council, 2023).

To investigate potential acute impacts, we also compared measured soluble copper and zinc concentrations (July 2019 to June 2024) to interim tier 1 (not bioavailability adjusted) acute guideline values for protection of 95% of instream species following Gadd et al., (2024) (see Appendix 3). We then calculated tier 2 bioavailability-adjusted guideline values. This provides an example for the Auckland region demonstrating the influence of bioavailability adjustment on the indications of risk of toxicity effects on aquatic life.

2.4.4.2 Adjustments

2.4.4.2.1 Nitrate toxicity

To assess the Nitrate (toxicity) attribute we use a proxy measurement of total oxidised nitrogen (nitrate + nitrite nitrogen). The underlying assumption being that the proportion of nitrite is much smaller than nitrate and therefore total oxidised nitrogen is a reasonable proxy for grading against the NOF nitrate toxicity attribute. We explored the suitability of this assumption by evaluating the proportion of nitrite in nitrate-nitrite-nitrogen.

2.4.4.2.2 Ammonia toxicity

Ammonia is toxic to aquatic animals and is directly bioavailable. When in solution, ammonia occurs in two forms: the ammonium cation (NH_4^+) and unionised ammonia (NH_3); the relative proportions of the forms are strongly dependent on pH (and temperature).

We applied a pH correction to NH_4-N to adjust values to their equivalent value for a pH of 8, following the methodology outlined in Hickey (2014) as recommended in the NPS-FM (2020). For pH values outside the range of the correction relationship (pH 6 to 9), the maximum (pH<6) and minimum (pH>9) correction ratios were applied. The pH correction was applied to NH_4-N observations after censored values had been imputed. Field measurements of pH were preferentially used over lab measurements if both were available.

2.4.4.2.3 Visual clarity

Auckland Council did not historically monitor visual clarity in rivers. Visual clarity monitoring using the black disc or clarity tube methods was initiated in 2022. In this report, visual clarity was calculated based on turbidity (FNU) data[5] and grading is therefore considered to be provisional.

Visual clarity (VC) and turbidity measurements are strongly correlated. Regional, site specific relationships between visual clarity and field turbidity (FNU) were derived from two years of paired measurements. These relationships were used to convert the measured turbidity (FNU) to predictions of visual clarity (m) (see Appendix 3, Table A-5)). While the models developed for this purpose show a generally good fit, uncertainties remain – particularly at lower/poorer VC values. This supersedes previous analysis based on a single equation to convert turbidity (NTU) to visual clarity based on a national dataset (Franklin et al., 2020).

To assess the impact of the unprecedented storm events of 2023 on turbidity, an additional analysis was conducted using long-term field turbidity (FNU) data from 31 sites. Median annual turbidity values were calculated for each site, and a Mann-Whitney U test was used to evaluate differences in turbidity distributions between the 2013-2022 and 2023-2024 periods. A Shapiro-Wilk test for normality was also performed to assess whether the data followed a normal distribution. Finally, median annual turbidity values for each site were grouped by the two time periods and visualised using boxplots, faceted by site.

[5] Except for Hōteo River (NIWA) which is based on direct visual clarity monitoring.

2.4.4.2.4 Bioavailable metals (copper and zinc toxicity)

We applied a dissolved organic carbon (DOC) correction to the dissolved copper to adjust values to equivalent DOC 0.5 mg/L values as described in Gadd et al., (2023). Observations that were associated with pH values greater than 8.5 or less than 6, or with hardness values greater than 200 mg/L were excluded from the analysis.

Dissolved zinc observations were adjusted for pH, hardness, and dissolved organic carbon (DOC) following the method described in Australia and New Zealand Guidelines for fresh and marine water quality (ANZG, 2024) and using Burrlioz 2.0 software. Code to undertake the adjustment is available on github[6]. When the pH, hardness or DOC were outside the applicable range, the upper or lower limit of the applicable range was used. This approach leads to some adjustments being conservative and others potentially not being sufficiently protective. This included a small number of results in very soft waters <20 mg $CaCO_3$/L and/or at high pH (>8.3). All adjusted values were retained for subsequent analysis.

Additional assessment against tier 1 acute guideline values and calculation of tier 2 bioavailability adjusted acute guideline values followed Gadd et al., 2024 (see Appendix 3). Where the pH, DOC, or hardness values were outside of the applicable range[7] values were excluded from the analysis.

2.4.4.3 Uncertainty in assessment

All assessments of water quality state are uncertain as monthly monitoring provides only a snapshot of the full variability that exists over the five-year assessment period. Natural variability, in relation to NOF attributes, is variation in an attribute "state" due to natural processes such as climate cycles, or weather events (Snelder and Kerr, 2022).

Currently, no statistically robust method is available to account for natural variability and sample error[8] in the assessments of state for NOF attributes (Milne et al., 2023). Rolling yearly attribute states have been analysed and are provided in Appendix 10 for an indication of the environmental variability (i.e., both natural and anthropogenic) present at each site in accordance with recommendations in Milne et al. (2023).

2.5 Trend assessment

All state and trend analyses were undertaken by Land and Water People (LWP) and outputs including graphics were provided to Auckland Council. We evaluated trends using the LWP Trends functions (Fraser and Snelder 2025 – LWPTrends_v2502.r) that are implemented in the R statistical computing software (R Core Team, 2023). Trend assessment methods were described by Fraser (2025) following Fraser and Snelder (2025) and are briefly summarised here.

[6] https://github.com/niwa/CuZn_DGV_adjusters/tree/main

[7] The applicability range differs for each metal. For copper: pH ≥ 5 & ≤ 8.8, hardness ≥ 3.9 & ≤ 898 mg/L, DOC ≥ 0.1 & ≤ 30 mg/L. For zinc: pH ≥ 5.4 & ≤ 8.5, hardness ≥ 14 & ≤ 411 mg/L, DOC ≥ 0.1 & ≤ 20 mg/L.

[8] Sample error is a statistical term meaning the difference between the unknown, actual, population statistic and the sample statistic (Milne et al., 2023). It is not related to a lab or field-based error while taking a sample.

The purpose of trend assessment is to evaluate the direction and rate of change in the central tendency (the typical value) of water quality values over time. Trend analysis was undertaken over defined time periods for all site and parameter combinations.

Many water quality observations vary seasonally. Accounting for systematic seasonal variability should increase the statistical power of the trend assessment. Data is therefore first assessed to identify if seasons explain variation in water quality using the Kruskal-Wallis test ($p<0.05$). Seasons may be defined as each month, or bi-monthly, or quarterly. The best fit is identified automatically.

The trend direction and confidence in the trend direction were evaluated using the Mann Kendall assessment or the seasonal variation. The test looks at all possible pairs of observations and counts how often values increase or decrease over time. For the seasonal variant, pairs of observations between years are only compared when they are in the same month (or another seasonal period). A continuous measure of confidence in the assessed trend is determined based on the posterior probability distribution of the Kendall S statistic.

The continuous measure of confidence in the direction of the trend is interpreted based on the categories used by the Intergovernmental Panel on Climate Change and further aggregated to five categories for simplicity as per LAWA (Cawthron Institute 2019; Fraser and Snelder 2025) (Table 2-5).

Trends were classified as "not analysed" when a large proportion of the values were censored (i.e. <5 non-censored values and/or <3 unique non-censored values), or if there is no, or very little, variation, or poor precision in the data because this results in ties.

For most parameters, **a decreasing trend is interpreted as an improvement in water quality, and an increasing trend is a degradation in water quality.** Inversely, for visual clarity, a decreasing trend is interpreted as a degradation in water quality. For some physical variables we have referred to the confidence of the direction of the trend as increasing or decreasing and have not assigned this as either improving or degrading.

The proportion of sites in each category shown in Table 2-5 were calculated for each variable grouped by the dominant land cover class (as per 2.1.1) or biophysical class (as per 2.1.2) and these values were plotted as colour coded bar charts. These charts provide a graphical representation of the proportions of improving and degrading (or increasing and decreasing) trends at the levels of confidence indicated by the categories.

Table 2-5: Level of confidence categories used to convey trend confidence and direction.

Trend categories	Value of C_d*
Very likely decreasing	0.90-1.00
Likely decreasing	0.67-0.90
Low confidence	0.33-0.67
Likely increasing	0.10-0.33
Very likely increasing	0.0-0.10

*The confidence in direction (C) is transformed into a continuous scale of confidence the trend was decreasing (C_d). Where $S<0$, $C_d=C$; where $S>0$, $C_d=1-C$.

The rate or magnitude of the trend is characterised by Sen slope regression: the slope of the trend line using the Sen slope estimator (SSE) (or the seasonal version (SSSE)). The SSE is the median of all possible inter-observation slopes (i.e. the difference between measured observations divided by the time between sample dates). The seasonal variant calculates the median inter-observation slopes for each season and then takes the median across seasons. The 90 percent confidence intervals for the trend rate were also calculated.

2.5.1 Integration of state and trends

The importance and meaning of trends and identification of appropriate actions is dependent on many factors including the current state and the rate of change over time.

Graphical methods were used to explore relationships between state and trends by plotting the annual Sen slope against the current state median values and corresponding median attribute metric band ranges. The median state metric may differ from the overall state band however this was selected as trends focus on changes in the central tendency of the data. 90% confidence intervals are displayed for the Sen slope.

For some NOF attributes, for selected sites, we estimate the time for a very likely trend to result in a change in assessed state median attribute band (assuming a linear rate of change). These estimates are based on the difference between the current five-year state median concentration and the nearest relevant attribute band threshold, divided by the Sen slope rate of change. For example, if a site is currently in band 'B' and is very likely degrading, the difference between the current state and the B/C band threshold was used. Conversely if that site was very likely improving, we would refer to the difference between the current state and the A/B band threshold. These comparisons cannot be directly undertaken where state assessments are based on modified results (ammonia toxicity, and bioavailable metals) and trends are assessed based on unmodified observations.

This time estimate is only intended to provide context on the magnitude of change as trends are rarely linear and can vary in both magnitude and direction over time due to cyclical climatic patterns and other natural and anthropogenic causes of variability.

2.5.2 Time period selection

Because water quality is constantly varying through time, the evaluated direction, and magnitude of change, depends on the time-period over which it is assessed (Snelder and Kerr, 2022).

The shorter the time period over which a river water quality trend is assessed, the greater the level of influence of climatic variation on the assessed trend (Snelder and Kerr, 2022). Faced with these limitations, the most common choice of trend period is over 10 years.

However, a second important consideration is whether the data are being affected by processes other than environmental conditions. Confounding factors such as changes in methodology limit our ability to detect 'true' environmental change. Differences in analytical methodology can influence both the direction, and the magnitude of trends in the data (Wood, 2024). Differences between laboratories use of the same analytical methodology can also contribute to bias (Wood, 2024).

Auckland Council changed laboratory providers in July 2017 coinciding with a change in analytical methodology, and/or detection limits for several water quality parameters. Notable step changes were observed for several parameters over this period (including dissolved reactive phosphorus, soluble zinc and copper, and total suspended solids). At the time of the change in service provider, limited paired sampling was undertaken. Consequently, there was insufficient information available to calculate adjustment factors to align old and new methods. See Appendix 4 for more information.

In this situation, it is considered more appropriate to analyse state and trends before and after the methodology change (Wood, 2024). For the purposes of this report, trends were assessed in water quality data from the period after the laboratory change (July 2017 to June 2024) for all variables, including variables with no obvious step changes observed. This shorter, seven-year period for trend assessment is vulnerable to a greater level of influence of climatic variation than longer 10- or 20-year periods (Snelder and Kerr, 2022). In some instances, such as for parameters measured in the field, 15-year trend periods (July 2009 to June 2024) were also assessed.

2.5.3 Data processing

2.5.3.1 Minimum data requirements

It is general practice to define the acceptable proportion of data gaps and how these are distributed across sample occasions so that the reported trends are assessed from comparable data when comparing between sites or between different parameters.

Occasionally, sampling events are delayed or early and the date of sampling may deviate from the intended 'month' resulting in two samples collected within a month and no observations in the preceding or following month. Where this occurred, only one observation, closest to the midpoint of the calendar month, was retained for analysis. A median of the two observations was not taken as this would induce a trend in variance and invalidate distribution assumptions of the Mann Kendall S statistic (Fraser, 2025; Helsel et al., 2020).

Sites were retained for analysis where observations are available for at least 80% of the years within the trend time period and observations are available for at least 80% of months within the trend time period. If a site failed the second rule, then data were coarsened to bi-monthly, or quarterly, or biannually as required.

Trend assessment for the July 2017-June 2024 time period was not possible for the sites Mangawheau Stream and Newmarket Stream due to the more recent establishment of these sites.

2.5.3.2 Censored values

For evaluation of the confidence in trend direction (Mann Kendall test), censored values were treated according to Helsel (2005, 2012), using methods that are robust to changes in detection limits over time. When assessing trend direction increases or decreases in a water quality variable were identified wherever possible. A change from a censored value of <1 to a measured value of 10 was considered an increase. A change from a censored value of <1 to a measured value 0.5 was considered a tie, as is a change from <1 to a <5, because neither can definitively be called an increase or decrease (Fraser & Snelder, 2025). Similar logic applied to right censored values. The method is robust to changes in detection limit over time.

When assessing trend magnitude using Sen slopes, left censored values were substituted with their raw values (i.e., the numeric component of the detection limit) multiplied by a factor (0.5 for left-censored and 1.1 for right-censored values). This step ensures that any values measured exactly at a detection limit are treated as being larger than values less than the detection limit (which would be the case if the raw values were not multiplied by 0.5) (Fraser & Snelder, 2025). Similarly, measured values are treated as being smaller than the censored value if it is right-censored. The inter-observation slopes associated with censored values are therefore imprecise (because they are calculated from the substituted values). As the proportion of censored values increase, the probability that the Sen slope is affected by censoring increases. Supplementary files include an 'analysis note' to identify Sen Slopes where one or both of the observations associated with the median inter-observation slope is censored.

A new approach was applied to handle changes in detection limits over time on the Sen slope. Inter-observation slopes are considered to be ties and set to zero, regardless of their values, when: (1) both observations were either left or right censored, (2) when one observation is left censored and larger than the other non-censored observation; (3) when one observation is right censored and smaller than the other non-censored observation.

The estimate of the magnitudes (i.e., the SSE and SSSE values) and confidence intervals of individual site trends decreases in reliability as the proportion of censored values increases. When there are censored values, greater confidence should be placed in the statistics returned by the Kendall tests (the trend direction and the probability the trend was decreasing) (Fraser, 2025).

3 Upstream land cover

The dominant land cover within the upstream catchment of a river site is often used as a proxy for anthropogenic pressures impacting water quality. Across New Zealand, contaminant concentrations are frequently higher in rivers and streams with catchments dominated by urban and pastoral land cover types while ecological health indicators are lower (of poorer quality) (Snelder et al., 2017; Whitehead et al., 2018; Larned et al., 2019; Gadd et al., 2020; Whitehead et al., 2022). Land cover in the upstream catchment of a river site explains more variation in stream contaminant concentrations than land cover in the adjacent riparian zone of the sampling site (Larned et al., 2019). Land cover is however only one of many factors that influence water quality. Landscape variation, such as differences in underlying geology, geochemistry, topography, and climatic processes are also drivers of significant differences in water quality (Pearson and Rissman, 2021). While land cover presents a proxy for the generation of contaminants, the transformation of those contaminants and transport pathways across the catchment also contribute to water quality outcomes (Pearson and Rissman, 2021).

A summary of the proportions of current land cover within the upstream catchment for each site as of summer 2024/2025 are outlined in Figure 3-1. The ‘urban’ class includes some sites with combined influences of pastoral and urban land uses, such as Papakura Stream (lower), Onetangi Stream and Vaughan Stream (lower). In all cases, urban land uses are the most proximal pressure, closest to the water quality monitoring site with pastoral uses located in the upper catchment. We classified Mahurangi River (Right Branch) catchment as ‘Exotic forest’ for reporting although the 79% exotic forest cover in the catchment did not quite meet the definition of more than 80% (Table 2-2).

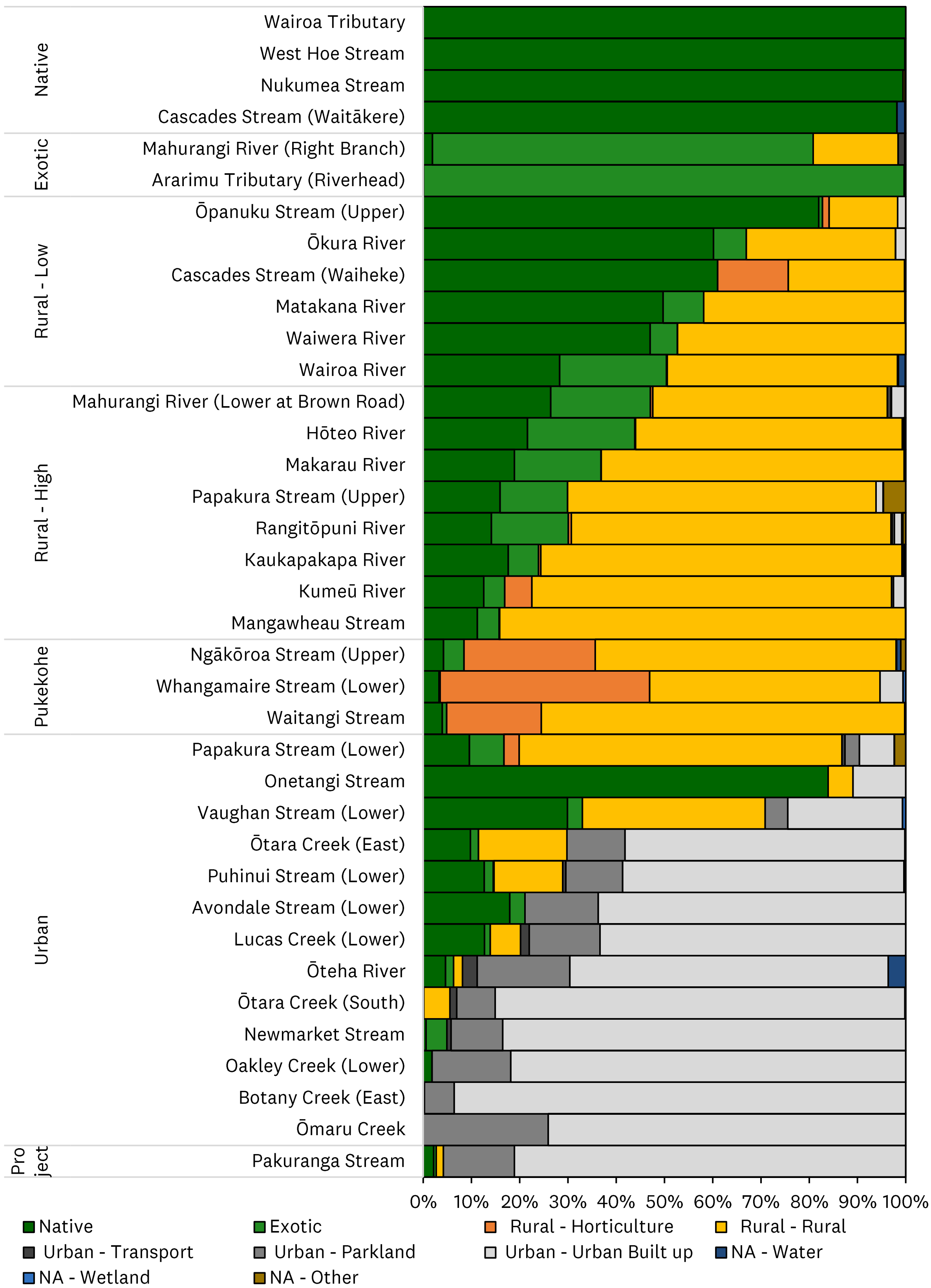


Figure 3-1: Proportion of landcover in the upstream catchment and dominant land cover class assigned (LCDB regional update 23/24 – provisional (Auckland Council, 2025)). Land cover classes are displayed at the second tier aggregation (see Appendix 1).

3.1 Land cover changes

Our assessment of change in land cover over time can identify broad-scale changes at a catchment level and at an interval relevant for river water quality trend analysis. This analysis does not necessarily capture differences in land use or management practices which may also affect river water quality.

In most catchments there were only minor changes in land cover from the last reporting period. For most sites there was less than three per cent change in total broad land cover classes between the 2018/2019 and 2023/2024 assessments of land cover. The greatest changes in land cover over this time period (>5% change) were associated with urban growth and development in greenfield areas within the upstream catchments of Vaughan Stream (Long Bay), Ōtara Creek East (Flat Bush), Lucas Creek (Albany), and a tributary of the Mahurangi River (Stage Highway 1 Puhoi to Warkworth) (Table 3-1). Intensification of urban development within existing urban areas (brownfield development) is not well captured by this analysis as these areas are typically already designated 'urban built up area'.

For all of these catchments, residential urban development commenced prior to July 2017. Therefore, the trend period assessed for water quality does not include a pre-development baseline. The greatest increase in urban land cover was within the Vaughan Stream catchment at Long Bay. The Long Bay Structure Plan became operative in 2011, and development and construction has continued over time. Collectively, an additional 20% of the total catchment upstream of the monitoring site is now (2023/2024) built up compared to 2008/2009, more than half of that change occurred since 2018/2019 (Table 3-1). The second greatest change was within the Ōtara Creek East catchment. Urban development commenced in this area in 2001 under the Flat Bush Structure Plan.

The Mahurangi River (Right Tributary) catchment was and remains dominated by exotic forestry. A large-scale harvest phase was undertaken predominantly between 2012-2019 with exotic forest (forested) cover decreasing from 97% of the catchment in 2012 to 35% of the catchment in 2023 (Table 3-2). In 2023, 44% of the catchment was categorised as 'Harvested' while 17% of the catchment was reclassified as 'Low producing grassland'. The completed motorway footprint accounted for 1.4% of the catchment area. During the construction period a larger area of approximately 10% of the catchment was visibly disturbed with earthworks disturbance, based on aerial imagery from 2019/2020. Riverhead Forest (Ararimu Tributary) also went through a harvest phase between 2018/2019-2023/2024 (Table 3-2). Almost 100% of the catchment was in 'Exotic forest' in 2018 reducing to 59% in 2023 with the remainder classified as 'Harvested'.

Increases in 'Native' land cover where more than 3% of the catchment changed between 2018/2019 to 2023/2024 was observed in five catchments. This appeared to be associated with riparian planting in the upper Puhinui Stream catchment within Tōtara Park, and in some headwater areas of Kaukapakapa River. Changes in 'Native' land cover within the Vaughan Stream catchment appeared to be associated with planting in new urban areas. The locations of 'Native' land cover change were more scattered across the Waiwera River, and Makarau River catchments.

Table 3-1: Percentage of total urban area within the upstream catchment at each time step for all 'urban' sites (LCDB regional update 23/24 – provisional (Auckland Council, 2025)).

Site	1996/97	2001/02	2008/09	2012/13	2018/19	2023/2024	Total Change % of catchment 2018-2023	*Total Change 1996-2023 % of catchment*
Papakura Stream (Lower)	5%	7%	8%	8%	9%	11%	2%	*6%*
Onetangi Stream	11%	11%	11%	11%	11%	11%	0%	0%
Vaughan Stream	6%	8%	9%	12%	16%	28%	12%	*22%*
Ōtara Creek (East)	10%	22%	43%	44%	60%	70%	10%	*60%*
Puhinui Stream (Lower)	62%	67%	68%	68%	69%	71%	2%	*9%*
Avondale Stream (Lower)	78%	78%	78%	79%	79%	79%	0%	1%
Lucas Creek	38%	57%	64%	66%	72%	80%	7%	*42%*
Ōteha River	69%	82%	84%	84%	84%	88%	4%	*20%*
Ōtara Creek (South)	81%	90%	93%	93%	94%	94%	0%	*14%*
Newmarket Stream	95%	95%	95%	95%	95%	95%	0%	0%
Pakuranga Stream	22%	95%	95%	96%	96%	96%	0%	*74%*
Oakley Creek (Lower)	98%	98%	98%	98%	98%	98%	0%	0%
Botany Creek (East)	96%	100%	100%	100%	100%	100%	0%	*4 %*
Ōmaru Creek	97%	97%	99%	100%	100%	100%	0%	3%

Table 3-2: Percentage of exotic forest and exotic forest harvested area within the upstream catchment at each time step for all 'exotic' sites (LCDB regional update 23/24 – provisional (Auckland Council, 2025)).

Site	Land cover	1996/97	2001/02	2008/09	2012/13	2018/19	2023/2024
Mahurangi River (Right Branch)	Exotic Forest	97%	97%	97%	97%	59%	35%
	Forest – Harvested	0%	0%	0%	0%	38%	44%
Ararimu Tributary (Riverhead)	Exotic Forest	59%	80%	95%	100%	100%	59%
	Forest – Harvested	41%	20%	5%	0%	0%	41%

4 Flow conditions over state and trend periods

4.1 Introduction

Variations in rainfall patterns influence river flows, contaminant runoff from land, erosion processes, and interactions between stormwater and wastewater networks. They also partially govern water use and abstraction. Consequently, water quality varies over different flow conditions in rivers and streams, but different parameters have different relationships. These relationships can be additive, compensatory, or synergistic with other natural and anthropogenic drivers. Temporal variation in water quality is, to some extent, driven by seasonal variability and periodic climatic cycles such as the El Niño – Southern Oscillation that drive variation in interannual rainfall and temperature over large spatial scales, though responses can vary by site and parameter. (Snelder et al., 2022).

It is important that water quality observations are unbiased with respect to flow to represent the state of water quality because values such as ecosystem health are not specific to certain flow states (Snelder and Kerr, 2022; Kennan et al., 2024). This is also important for comparability between sites. Water quality monitoring in this programme is not expected to capture the highest flow levels due to health and safety limitations on physical sampling in these conditions.

4.2 Representativeness of flow conditions

Probability-Probability (P-P) plots visualise deviations in the flow distribution over our monthly sampling occasions from the full flow distribution over the entire five-year (state) and seven-year (trend) assessment periods. When the coloured lines are above the black 1:1 line, the water quality samples for a given flow quantile are lower than those for the entire period and vice versa. These are shown (Figure 4-1) for all 13 sites with paired water quality and flow. At some sites (e.g. Kaukapakapa River), the coloured lines were slightly above the black line indicating that water quality samples were more often collected during lower flow conditions than the overall flow conditions observed throughout the entire period. Conversely, for some sites the coloured lines were slightly below the 1:1 black line indicating that water quality samples tended to be representative of higher flows than observed throughout the entire period. Where an ‘S’ shape curve is seen, such as for Ōteha River, this indicates that the samples are missing some representation of both the lowest and highest flows, with more samples obtained during moderate flow conditions.

However, for all sites, deviation from the full flow distribution was not found to be statistically significant (p>0.05) and we can conclude that water quality samples obtained at these sites were not biased with respect to flow.

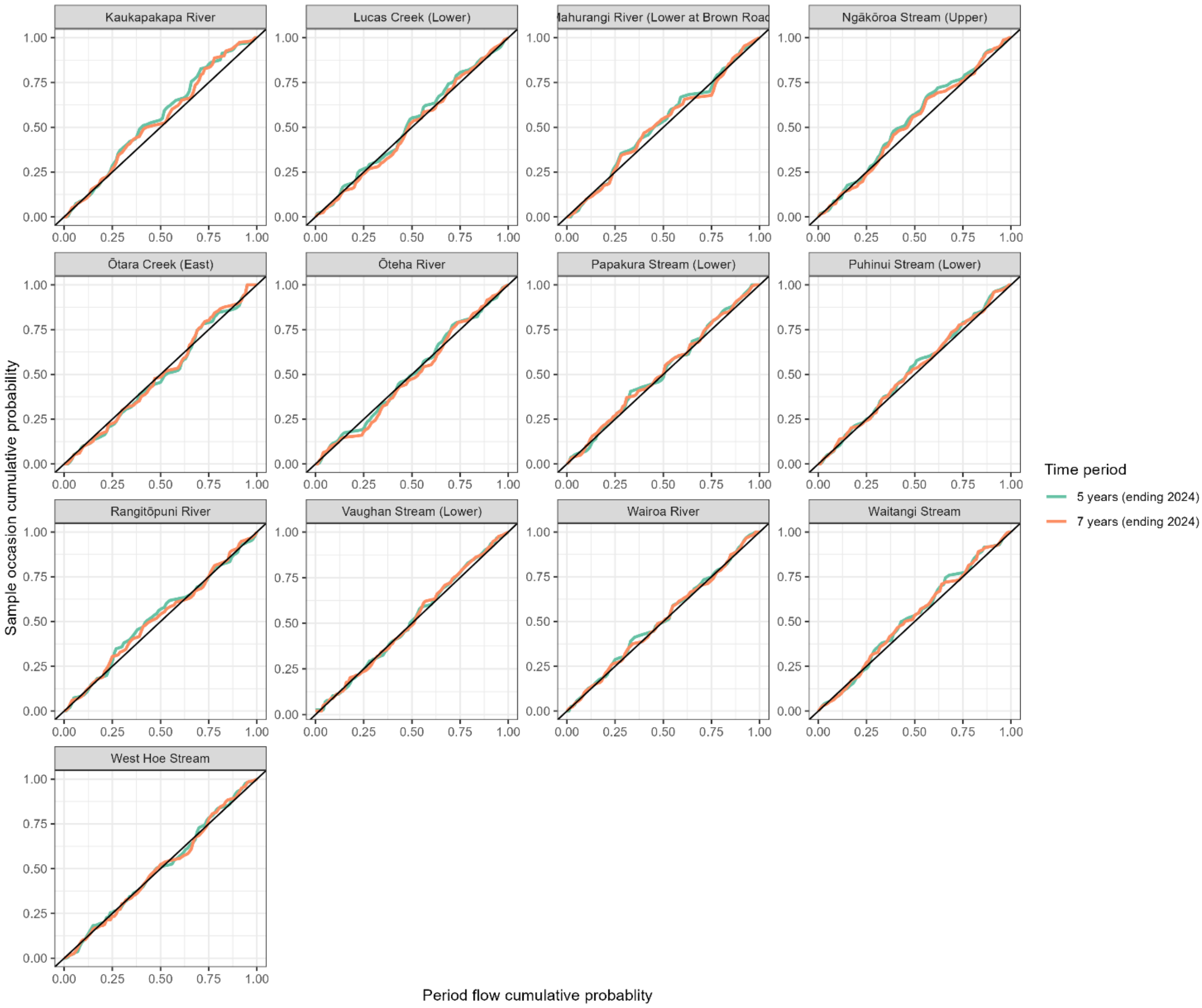


Figure 4-1: P-P plots comparing the cumulative probabilities of the full assessment period flows and the flows on water quality sampling occasions. Coloured lines above the black 1:1 line indicate some bias towards lower flows on sampling occasions.

An additional analysis of 17 paired sites was undertaken to evaluate how well water quality sampling represents the full distribution of stream flows. The proportion of flow conditions captured ranged from 92.6% at Ōtara (East) to 99% at Lucas Creek. Several sites – such as Mangawheau Stream, Papakura Stream, Waitangi Stream, Newmarket Stream, Ōteha River, Ōtara Creek, Ngākōroa Stream, and Wairoa River – had limited water quality observations during high-flow events. In contrast, other sites (Puhinui Stream, Lucas Creek, Vaughan Stream, and Mahurangi River) showed good representation across the full flow spectrum (Figure 4-2).

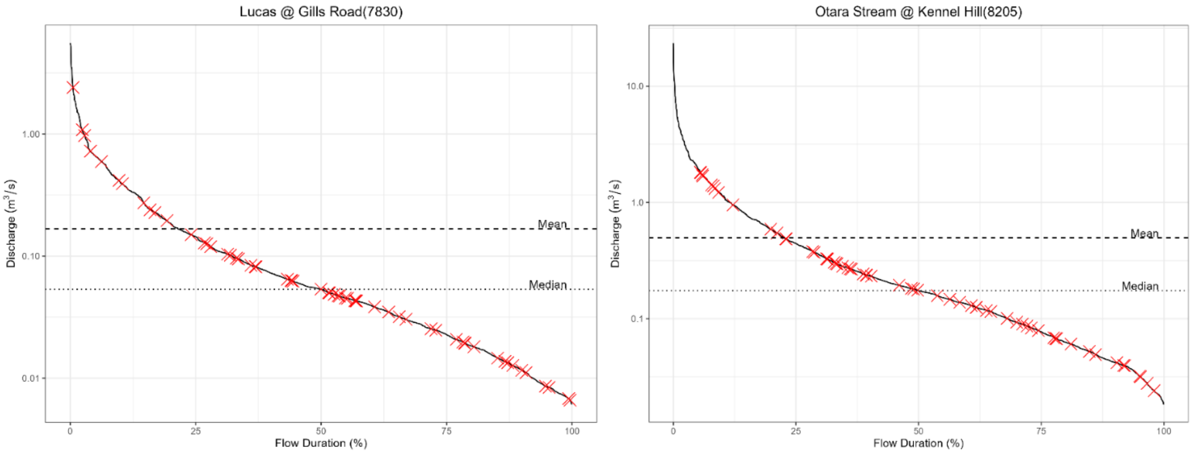


Figure 4-2. Flow distribution curves for five-year time period between July 2019 and June 2024 for two selected paired river water quality and flow monitoring sites: Lucas Creek and Ōtara Stream. Red crosses indicate flows when water quality samples were collected.

4.3 Climatic context of state and trend periods

La Niña conditions prevailed over spring 2020 through to winter 2022 (NIWA, n.d.). This is typically associated with higher rainfall in summer in the Auckland region. No significant La Niño periods were identified over the state and trend assessment periods with the last notable event occurring in 2015-2016 (NIWA n.d.). Several historically significant climatic events occurred over the state and trend assessment periods including a severe drought in January and February 2020 and the extreme rainfall and floods that occurred in January 2023 and Cyclone Gabriele in February 2023, and in May 2023 (Johnson, 2023). For the 13 paired water quality and flow sites assessed, the long-term records indicate lower flows around summer (Dec-Mar) and much higher flows in winter (July) (Figure 4-3). Extreme rainfall events during the five and seven year assessment periods, in conjunction with expected higher summer rainfall during La Niña periods is demonstrated by much higher summer flows than the long-term average for most sites. Lower flows than the long term average were observed across March-April, and higher flows in winter for some sites such as Ōteha River and West Hoe Stream. Peak flows typically occurred in July across sites. The extreme, and opposite, conditions that occurred during summer months may influence seasonal patterns observed.

Previous analysis of the influence of flow on water quality observations within Auckland found that concentrations of nutrients, particularly forms of nitrogen, increase with increasing flow conditions, and visual clarity generally reduces with increasing flow conditions (Snelder and Kerr, 2022). There is higher variability in water quality at very high flows, and concentrations can also change over the course of a peak flood event (e.g. 'first-flush'). Few long-term monitoring programme water quality samples are obtained in these conditions (Snelder and Kerr, 2022).

For the trend period assessed, over July 2017 to June 2024, there was a severe drought in the middle of this time period (summer 2020) followed by several extreme rain events later in the time period. From this pattern of flow conditions, we may therefore expect to see higher concentrations in the later part of the time series. We may then anticipate a higher proportion of degrading trends that is partly attributable to climatic variation, rather than changes in pollution sources, land cover or management actions.

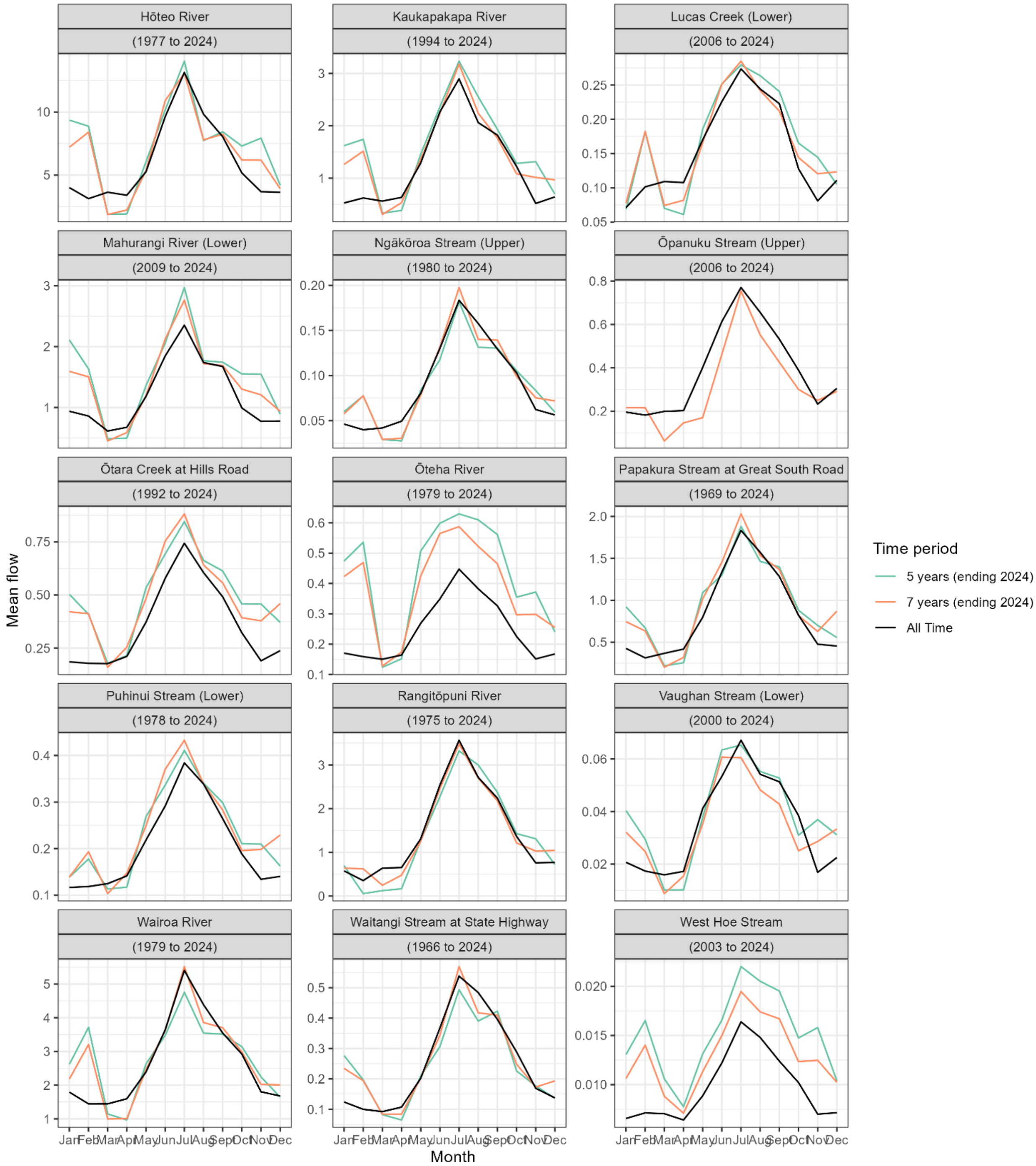


Figure 4-3: Flow regimes in assessment periods and the long-term flow regime at paired river water quality and flow monitoring sites. The period for the ‘All time’ flow record for each site is indicated in the plot headings.

5 Physico-chemical characteristics

Elevated water temperatures in streams is a significant environmental issue in Auckland rural and urban areas and is fundamental in regulating the solubility of oxygen and other chemical constituents. Temperature has been included as an additional NPSFM ecosystem health attribute for Auckland as unshaded rural streams and urban stormwater are particularly prone to high temperatures and this can have considerable negative impacts on aquatic ecology.

Dissolved oxygen (DO) is essential for most forms of life. Low levels can impact the growth and reproductive success of fish and macroinvertebrates, and very low levels can be lethal. Minimum instream DO concentrations typically occur in summer and are associated with high plant and algae growth, decaying organic matter, low flows, and increased water temperature. The lowest DO levels usually occur near dawn resulting from plant respiration overnight.

Conductivity is a useful general indicator of nutrient/mineral enrichment although the nutrient-conductivity relationship may break down in situations where there is salt spray influence or geologies enriched in certain compounds (Biggs, 2000).

pH is a measure of how acidic or alkaline the river water is. New Zealand river waters are usually slightly alkaline (Davies-Colley et al., 2013). pH also varies over daily cycles, particularly where there is abundant instream vegetation or algae, and is typically highest in the afternoon (Davies-Colley et al., 2013). High, or low pH can have adverse effects on the growth or survival of aquatic animals, and it is also an important modifier of other water quality parameters. For example, the toxicity of ammoniacal N to instream life increases with higher pH.

Water hardness refers to the concentration of calcium and magnesium ions, which can affect the survival of aquatic organisms, influence nutrient availability, and buffer against toxic metals. Alkalinity measures the water’s capacity to neutralise acids, providing essential buffering that helps maintain stable pH levels and protect aquatic life from harmful pH swings. Dissolved organic carbon consists of organic compounds from decomposing plants, soil, or human activities and plays a key role in nutrient cycling, energy supply for microorganisms, and light penetration in streams; however, excessive DOC can reduce oxygen levels, potentially harming sensitive species.

5.1 State

5.1.1 Distributions across land cover

In this section, data from sites within each land cover category were pooled to reveal broader patterns across the region. Concentrations also varied both between and within sites. This variation is illustrated in box plots in Appendix 5.

5.1.1.1 Temperature and dissolved oxygen

Median temperatures across all monitored streams varied from 12.5°C in Wairoa Tributary to 20.1°C in Botany Creek (East) (Appendix 5, Figure A5-1). There was a general pattern of cooler temperatures in native forest streams increasing to warmer temperatures in urban streams (Figure 5-1). However, streams within the wider Pukekohe area were slightly cooler than other rural streams. Temperatures varied by about threefold within each land cover class except for urban streams where variation was from 5.4°C to 31.1°C, indicating occasional temperature spikes in those streams. Notably, the two hottest urban streams, Pakuranga Stream and Botany Creek (East), are channelised streams with open-concrete channel beds. The highest median temperature was at Botany Creek (East) at 20.3°C, with summer temperatures often exceeding 25°C.

Streams in the wider Pukekohe area had the lowest median DO concentrations, while native forest streams had the highest DO levels. Median dissolved oxygen (DO) concentrations varied from 7.2 mg/L in Papakura (Upper) and Waitangi Streams to 11.4 mg/L in Botany Creek (East). Dissolved oxygen concentrations generally had high variability within each land cover class except for native forest sites and Pakuranga Stream (Urban – Project). Within site variability also differed, being the lowest for Wairoa tributary and the highest for Papakura Stream (Upper) (Appendix 5).

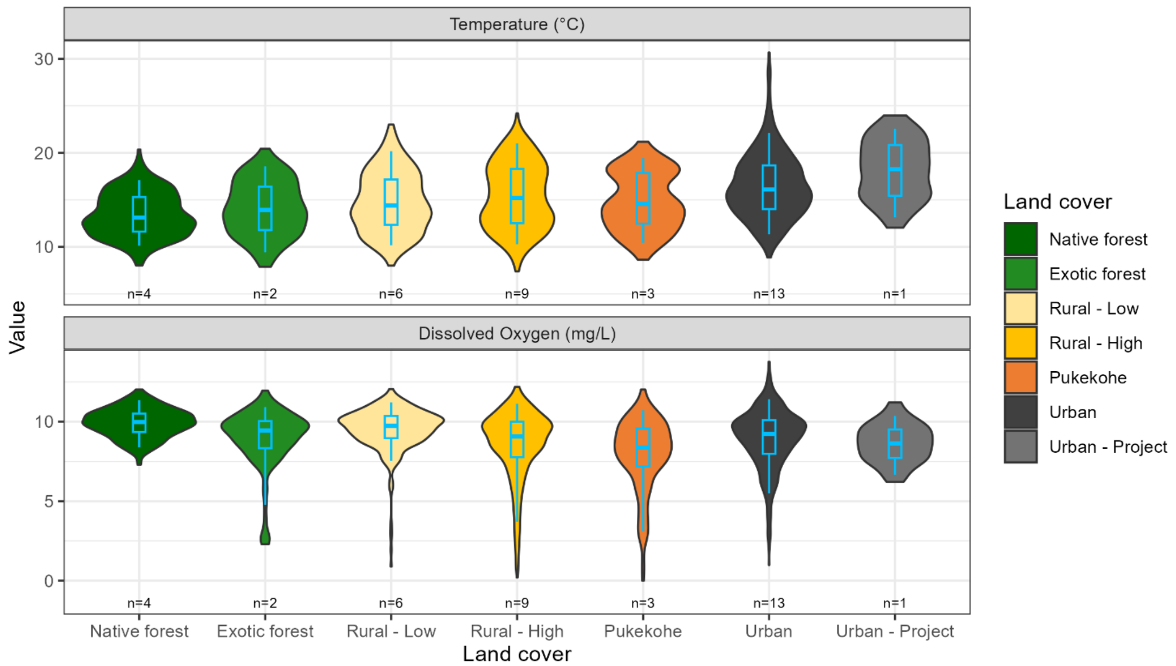


Figure 5-1: Distribution of observed temperature (°C) and dissolved oxygen (mg/L) across sites within each dominant land cover class based on five years of observations (01 July 2019 - 30 June 2024). The blue box indicates the interquartile range, the central line is the median, and the whiskers are the 5th and 95th quantiles. n indicates the number of sites represented in each violin.

5.1.1.2 Conductivity

The highest median conductivity among land cover categories was for the urban – project site, which is most probably due to high ammoniacal-N concentrations at this site (see section 6.1). Streams draining native forest catchments exhibited the lowest median conductivity among all land cover types. Median conductivity was the lowest for Mangawheau Stream in the rural (high) land cover and Wairoa tributary in native forest and the highest in the Onetangi (Urban) and Pakuranga (Urban-Project) streams (Appendix 5). Conductivity varied considerably for urban sites with the span over three orders of magnitude, while the variation within other land cover classes was notably lower (Figure 5-2). A notably high and unusual conductivity of 24.6 mS/cm was recorded in Newmarket Stream in January 2023, when a strong sewage odour was detected at the site during sampling. Within-site variability was also the highest in Newmarket Stream indicating contaminant inflow (Appendix 5).

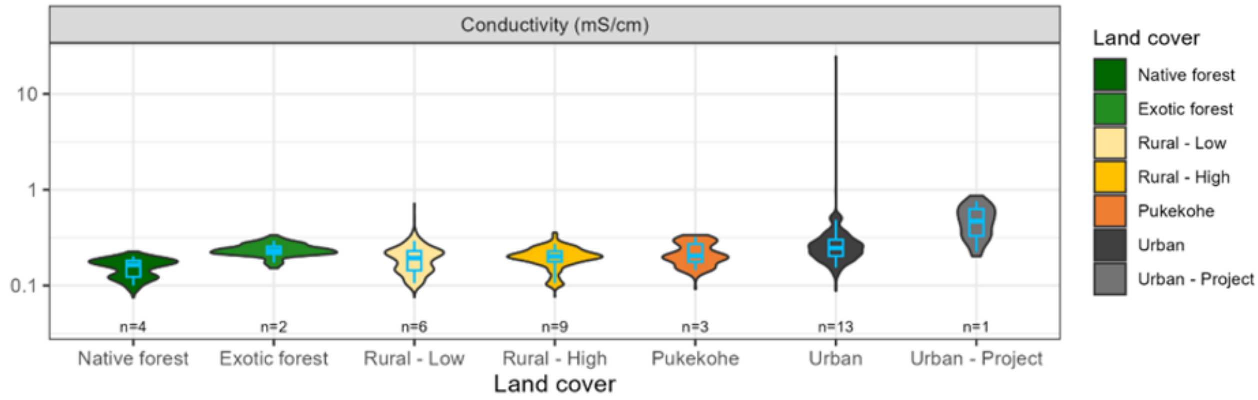


Figure 5-2: Distribution of observed conductivity (mS/cm) across sites within each dominant land cover class based on five years of observations (01 July 2019 - 30 June 2024). The blue box indicates the interquartile range, the central line is the median, and the whiskers are the 5th and 95th quantiles. n indicates the number of sites represented in each violin.

5.1.1.3 pH, hardness, alkalinity and dissolved organic carbon

Unlike other parameters, there were no major differences in pH between land cover groups. Pukekohe area and exotic forest sites had the lowest pH among land cover classes, while the urban – project site had the highest. There was a high variation in pH distribution across streams within native and exotic forest land cover groups which was assumed to be driven by site-specific underlying geological conditions. West Hoe and Nukumea native forest streams had slightly acidic waters while Cascades Stream (Waitakere) and Wairoa tributary – slightly alkaline. Ararimu Tributary, within the exotic forest land cover class, had the most acidic waters among all sites and the highest within-site variability, while Mahurangi River had slightly alkaline pH (Appendix 5, Figure A5-3). pH in rural and Pukekohe area streams varied less than in urban sites.

There were no major differences between land cover groups in total hardness and alkalinity with only Pakuranga Stream (Urban – Project) displaying notably higher median values. Streams draining native forest catchment had slightly lower median total hardness and alkalinity values than other land cover groups. There was relatively high variability within urban and rural (low) land cover classes and especially high – for the urban – project site. Exotic forest sites displayed a bimodal distribution driven by the differences between the two sites: Ararimu Tributary characterised by the low hardness and low alkalinity, and Mahurangi River characterised by elevated hardness and alkalinity. Newmarket Stream, Vaughan Stream and Ōmaru Creek had the highest total hardness and alkalinity levels out of urban sites and the concentrations were the most variable at those sites, potentially indicating occasional wastewater overflows and stormwater runoff from impervious surfaces.

Elevated dissolved organic carbon (DOC) concentrations indicate high organic matter inputs, leaching from poorly drained soils and reducing shallow groundwater systems, and may also be associated with runoff containing organic waste, fertilizers, or wastewater discharges in agricultural and urban areas. DOC did not differ much between land cover groups, with streams in the Pukekohe area and native forests exhibiting slightly lower median DOC concentrations, while urban and rural (high) streams exhibited slightly higher concentrations (Figure 5-3). DOC concentrations varied across streams even within the same land cover group. For example, within rural (high) land cover group Rangitōpuni had elevated median DOC of 7.2 mg/L, while median DOC in Mangawheau Stream was 3.1 mg/L. The highest within-site variability in DOC was in Pakuranga Stream (Urban – Project) and Ararimu Tributary (Appendix 5). The urban sites Papakura (Lower) and Vaughan streams had the highest median DOC of 6.8 mg/L and 6 mg/L respectively.

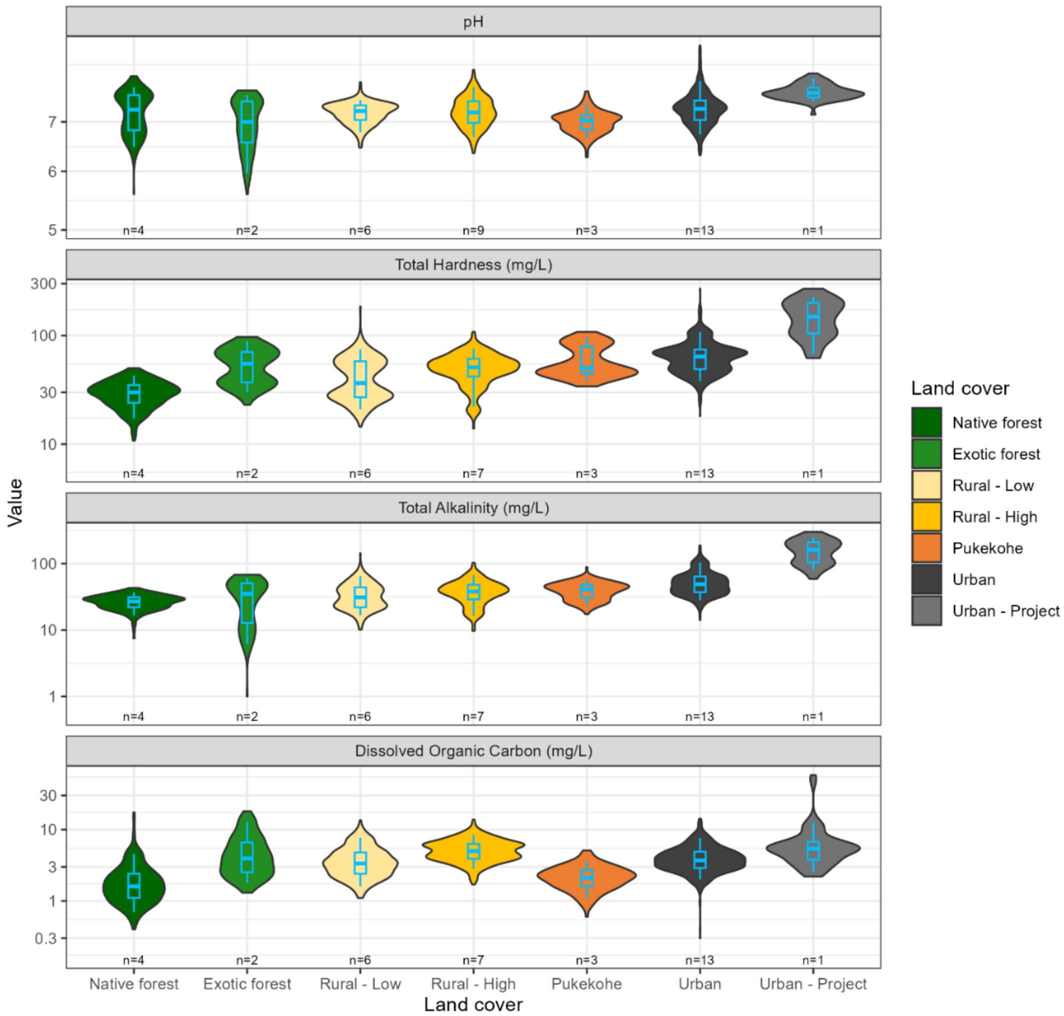


Figure 5-3: Distribution of observed pH, total hardness and alkalinity, and dissolved organic carbon across sites within each dominant land cover class based on five years of observations (01 July 2019 - 30 June 2024). The blue box indicates the interquartile range, the central line is the median, and the whiskers are the 5th and 95th quantiles. n indicates the number of sites represented in each violin land cover class.

5.1.2. Seasonality

Obvious seasonal patterns for temperature and DO were observed for all streams with higher temperatures and lower DO during summer months (Figure 5-4). Seasonality for other attributes was variable across different land cover groups. For example, Rural and Pukekohe area sites displayed seasonal patterns for conductivity, total hardness and alkalinity with lower values during winter months. Some streams, such as Ararimu Tributary, Vaughan Stream, Kaukapakapa River and Hōteo River displayed specifically pronounced seasonal trends in total alkalinity with higher values in summer. Out of all sites, Ararimu Tributary had the most significant seasonal trends in total hardness, total alkalinity and DOC with peaks occurring in late summer and lower concentrations in winter. These summer peaks coincided with lower zinc levels and higher pH values in Ararimu Tributary during those months (see section 7.1.2).

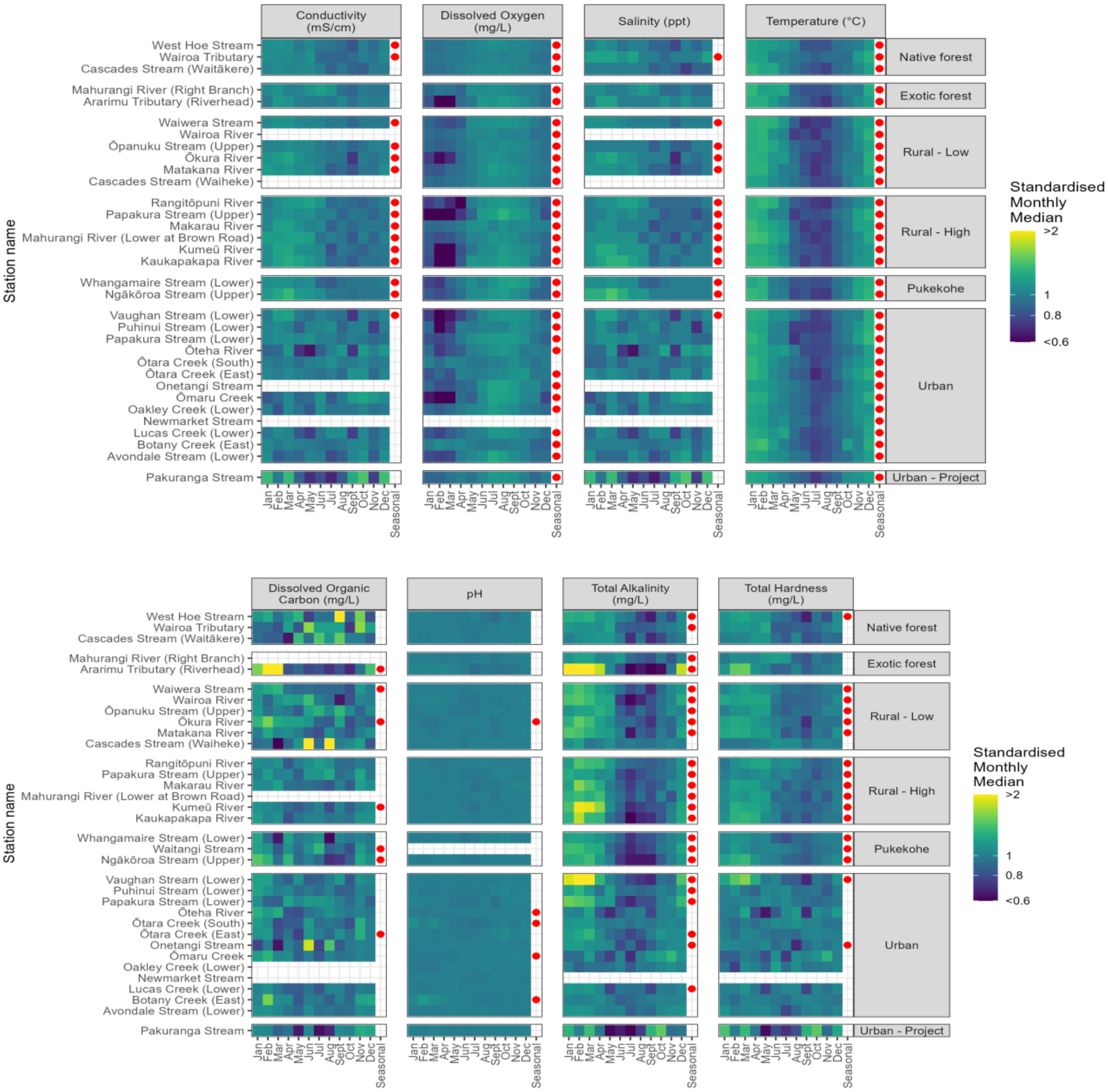


Figure 5-4: Water quality monthly medians standardised by overall median for physio-chemical water quality variables at each site based on five years of observations (01 July 2019 - 30 June 2024).

5.2 Trends

Trends for physico-chemical parameters in this section were analysed over the most recent seven years to align with the nutrient and metal trend analyses in this report. However, longer-term trends are generally more robust and less affected by climatic influences. Unlike metals and nutrients, physico-chemical parameters are measured in the field and were therefore not affected by laboratory changes, making it valid to assess trends over the past 15 years where data are available. However, methodological changes over the 15-year period – such as changes to sampling times and updates to calibration procedures – should be considered when interpreting the results (see sections 2.1.1 and 5.3 for more details and discussion). To provide more insight, 7-year trends were compared to 15-year trends for some parameters in this section.

Temperature was likely or very likely decreasing at more sites than likely or very likely increasing when assessing the most recent seven-year trend. This was the case for all land cover groups – except for the Pukekohe area and exotic forest streams (Figure 5.5). In contrast, the 15-year trends showed temperature was either likely or very likely increasing at more than 80% of the sites (Figure 5.6). Two exotic forest sites showed opposite trends: at Ararimu Tributary, water temperature was very likely increasing in both time periods, whereas at Mahurangi River (Right), it was likely decreasing over 15 years and showed a low-confidence decreasing trend over 7 years. Mahurangi River (Lower), in the Rural – High land cover group, was the only site with a very likely decreasing temperature trend over 15 years. Temperature in Pukekohe streams was likely or very likely increasing in both periods. While seven-year decreasing trends in stream water temperature suggest short-term improvements, 15-year increasing trends point to a long-term decline.

Trends in DO also differed between the 7- and 15-year analyses (Figure 5-5 and Figure 5-6), though to a lesser extent. Overall, more sites showed decreasing trends in DO in the 15-year period than in the 7-year period. While low DO levels are commonly used as an indicator of ecological degradation, in some cases, very high daytime DO concentrations may also signal ecosystem stress. This can occur in streams with excessive growth of aquatic plants and algae, which produce oxygen through photosynthesis during the day – leading to oversaturation – followed by oxygen depletion (hypoxia) at night when respiration dominates. For example, Botany Creek (East) is currently oversaturated (high DO concentrations during daytime); however, both the 7-year and 15-year trends indicate a very likely decline in DO concentrations.

Trends in DO were very likely decreasing at all Pukekohe sites for both trend periods. Among native forest streams, DO in West Hoe was likely increasing, while other streams either showed likely decreasing trends or low confidence in trend direction over the 7-year period. However, the 15-year trends showed decreasing DO at all native forest sites. For urban and rural (High and Low) land cover groups, DO concentrations were likely or very likely increasing at most sites for both periods, with more confidence in the trend direction in the 15-year analysis. For exotic forest sites and the Pakuranga Stream (Urban-Project), DO trends were in opposite directions between the two time periods: DO was increasing in exotic forest sites and decreasing in Pakuranga Stream over the 7-year period, but the reverse was observed in the 15-year period, except for the Ararimu Tributary which had low confidence in trend direction. The differences in trend direction between 7-year and 15-year time periods may be associated with the influence of climatic processes on some water quality parameters, such as temperature and DO.

Two significant changes in sampling logistics have occurred in recent years. In 2016, nine sites experienced a shift in sampling time of more than three hours, resulting in apparent temperature decreases at sites sampled earlier and increases at sites sampled later. In contrast, sites with sampling time shifts of less than two hours showed likely or very likely increasing temperature trends over 2010-2019 (Ingley, 2021). The second change occurred in 2022 following a program review, which resulted in the addition of 13 new sites and a general shift to earlier sampling times. For 10 sites, the median sampling time changed by more than two hours (but typically less than three). Within-site sampling time variation from 2020-2024 was generally under three hours but reached up to six hours in some cases. These time differences may have contributed to the observed seven-year decreasing temperature trends and to the mixed DO results between the two time periods.

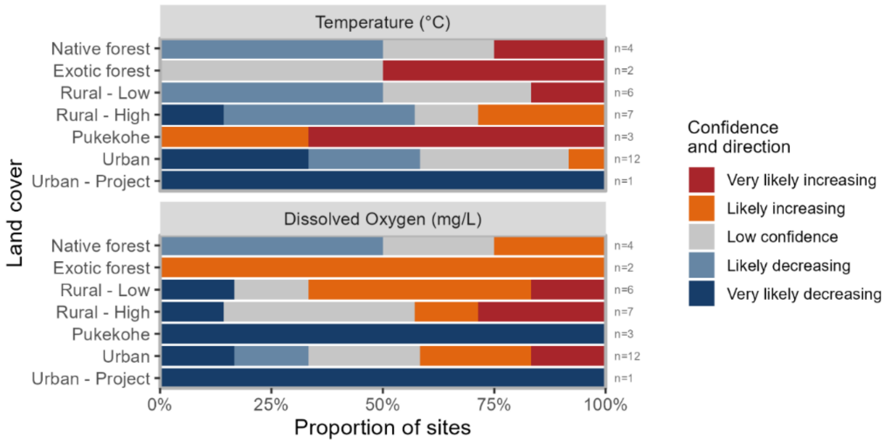


Figure 5-5: Proportion of sites in each trend category for temperature or dissolved oxygen concentrations grouped by dominant land cover class for the 7-year period (01 July 2017 - 30 June 2024).

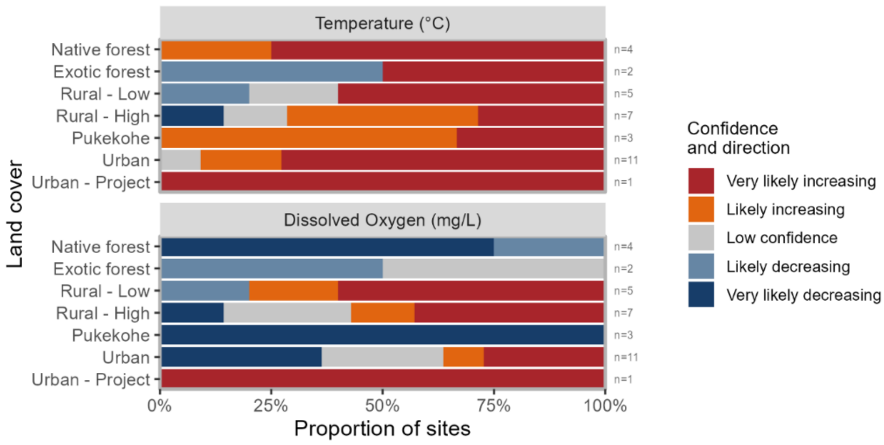


Figure 5-6: Proportion of sites in each trend category for temperature and dissolved oxygen concentrations grouped by dominant land cover class for the 15-year period (01 July 2010 - 30 June 2024).

Salinity and conductivity parameters had low confidence in trend direction for most sites across different land categories for 7-year time period (Figure 5-7), while for the 15-year time period there was more confidence in trend direction (Figure 5-8). This likely reflects the increased power of the statistical analysis based on a longer period and more samples. For most Pukekohe area and rural (high) sites salinity and conductivity were likely or very likely increasing. For native forest and exotic forest 50% of sites had likely or very likely decreasing trends in salinity and conductivity. For rural (low) and urban areas in the 7-year trend period approximately 30% of streams had likely or very likely decreasing trends, and 30% had likely or very likely increasing trends in conductivity and salinity. Over the 15-year period, more sites showed likely or very likely increasing trends in salinity and conductivity than sites showing likely or very likely decreasing trends.

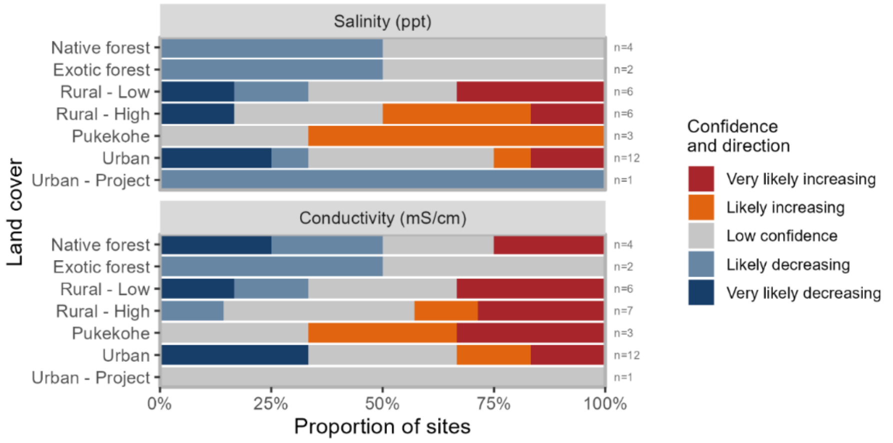


Figure 5-7: Proportion of sites in each trend category for salinity and conductivity grouped by dominant land cover class for the 7-year period (01 July 2017 - 30 June 2024)

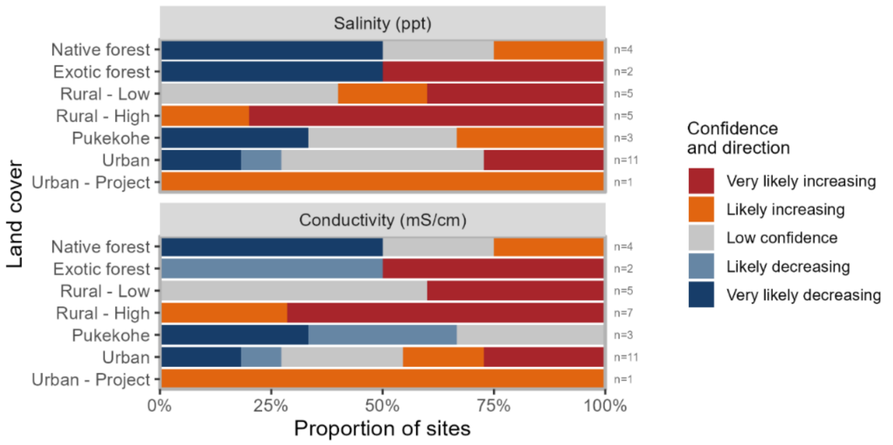


Figure 5-8: Proportion of sites in each trend category for salinity and conductivity grouped by dominant land cover class for the 15-year period (01 July 2010 - 30 June 2024).

pH displayed opposing trends for the 7- and 15-year time period assessment (Figure 5-9 and Figure 5-10). For the 7-year period pH was decreasing at most sites across all land cover groups, while for the 15-year period pH was increasing in most streams. The only stream that had increasing trends in pH for the 7-year period was Kaukapakapa River, while Ōmaru Creek, Puhinui Stream (Lower) and Waitangi streams were the only three that had decreasing trends in pH for the 15-year trend period.

Trends for total hardness, total alkalinity and dissolved organic carbon were assessed only for the most recent 7-year period, because measurements for those metrics started in 2018 (Figure 5-9). Total hardness and total alkalinity decreased in most sites across all land category types with a few exceptions. Half of the sites in rural (high) and urban categories showed increasing trends in total hardness. Total alkalinity likely or very likely increased at most sites in the Pukekohe area. Dissolved organic carbon increased in most sites across all land category types except for exotic forest where DOC decreased in both streams.

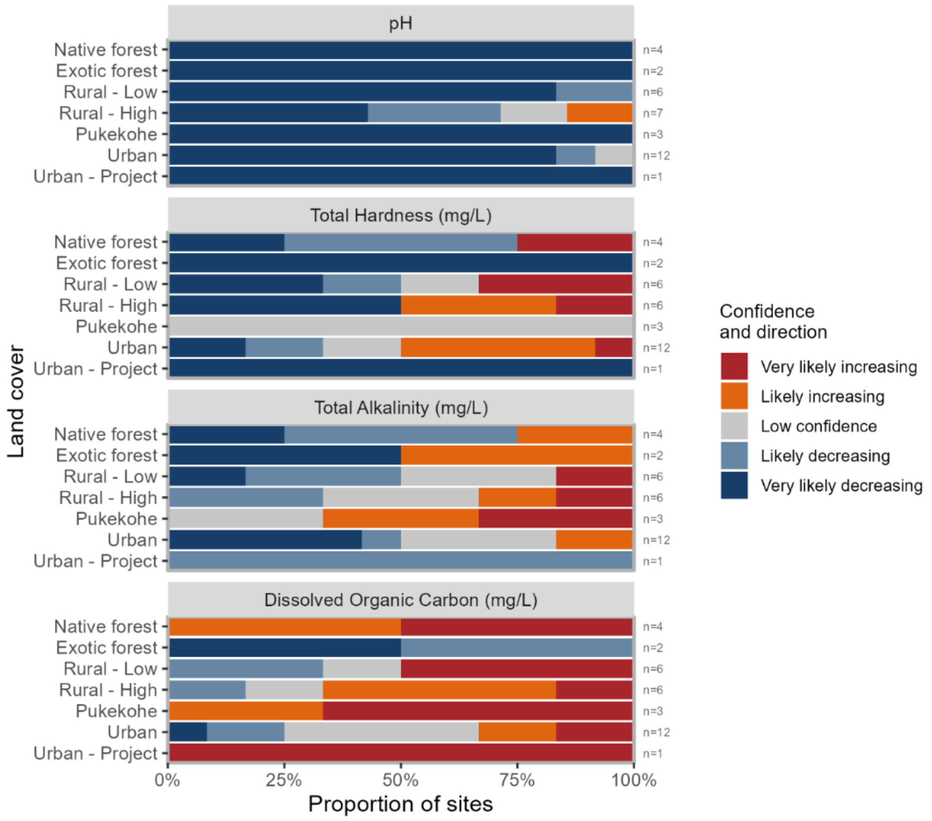


Figure 5-9: Proportion of sites in each trend category for pH, total hardness, total alkalinity and dissolved organic carbon grouped by dominant land cover class for the 7-year period (01 July 2017 - 30 June 2024).

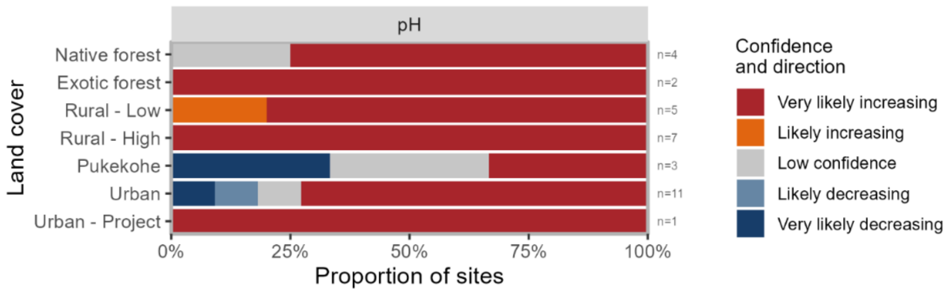


Figure 5-10: Proportion of sites in each trend category for pH grouped by dominant land cover class for the 15-year period (01 July 2010 - 30 June 2024).

5.2.1 Trend magnitude + state

Current state relative to trend direction and magnitude was assessed in this section for those parameters that are relatively stable during the day to eliminate the influence of the sampling time of the day changes.

Rate of change in total hardness generally increased with increasing total hardness and conductivity (Figure 5-11). Pakuranga Stream (Urban -Project) had the highest very likely decreasing trend of 4.2 mg/L per annum in total hardness. Out of urban sites, Ōmaru Creek displayed the largest decreasing rate of change in both total hardness and conductivity. Ōkura River (rural low) was changing in the opposite direction for both attributes with the highest increasing trend magnitude in total hardness and conductivity. Total hardness in exotic forest Mahurangi River (Right), with a relatively high median total hardness concentration of 69 mg/L, was very likely decreasing with an annual rate of change of 2.5 mg/L. Conductivity in Botany Creek was very likely decreasing with the annual rate of change of 0.004 mS/cm.

Streams with low DOC concentrations displayed predominantly increasing trends in DOC (Figure 5-11). All native forest sites and Whangamaire Stream displayed likely or very likely decreasing trends in DOC. DOC in Pakuranga Stream, with a relatively high median DOC concentration of 5.1 mg/L, was very likely increasing, with the highest annual rate of change of 0.26 mg/L. Papakura (Lower), with one of the highest median DOC concentrations of 6.8 mg/L, displayed likely increasing trends in DOC, while Rangitōpuni River, with the highest median DOC of 7.2 mg/L, had indeterminate trends in DOC. Onetangi Stream had the highest magnitude of increasing trends in DOC among urban sites, with an annual rate of change of 0.13 mg/L. DOC was decreasing in both exotic forest streams, with Ararimu Stream having higher median concentrations and displaying stronger decreasing trends in DOC than the Mahurangi River (Right).

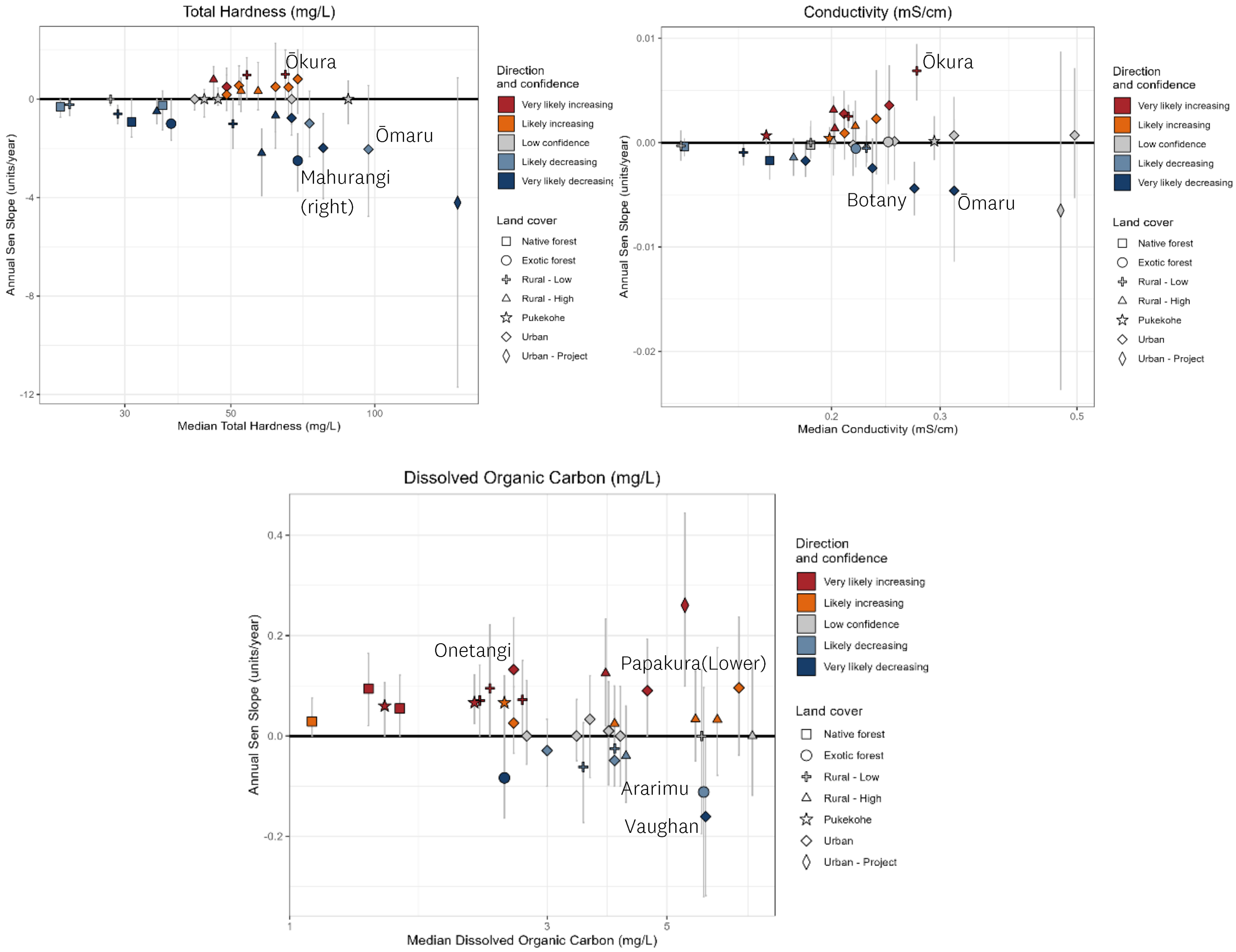


Figure 5-11: State (median) vs 7-year trend rate (Sen slope) for total hardness, conductivity and dissolved organic carbon. Note differences in Y axes. Whiskers indicate the 90% confidence interval of the Sen Slope.

5.3 Discussion

This and the preceding state and trends analysis of physico-chemical characteristics in streams demonstrate the challenges in providing a clear regional picture of water quality. In many cases, the underlying causes of specific issues are highly site-specific and cannot be meaningfully assessed at a regional or even land-cover scale. The river water quality monitoring program operates on a monthly basis, which limits its ability to detect short-term or daily fluctuations. Therefore, trend results for parameters like temperature and dissolved oxygen – which vary diurnally – should be interpreted with caution. Monthly water quality monitoring may underestimate impacts of low dissolved oxygen in streams particularly for sites sampled later in the day. Similarly, it may also underestimate the impacts of high water temperatures particularly for sites sampled earlier in the day. Continuous monitoring is essential to capture the seasonal and daily cycles of instream dissolved oxygen and water temperatures. Additional continuous monitoring sensors are deployed at a subset of water quality and hydrology monitoring sites across the region and the result of this work is reported separately in Young et al., 2025, and Dikareva, 2025 (in prep).

Current regional assessments of monthly discrete water quality monitoring shows that urban streams are the warmest across all land cover groups, with the highest temperatures observed in heavily modified streams lined with concrete channels. This is consistent with earlier assessments, which found that urban streams with modified beds often had elevated temperatures – exceeding 18°C for most of the year – based on regional water quality data from 2017-2019 (Ingley and Groom, 2021). Among all urban sites, Vaughan Stream was one of the coolest and exhibited the fastest decreasing temperature (Appendix 7) trends over the recent 7-year period. Additional analysis of continuous temperature measurements also showed decreasing maximum daily temperatures at this site over the past 20 years, while most other urban streams displayed the opposite pattern (Appendix 10).

Both temperature and dissolved oxygen (DO) vary not only seasonally but also diurnally. Typically, the lowest values occur early in the morning before dawn, while the highest are recorded in the late afternoon. For robust trend assessments, sampling should ideally occur at the same time of day. However, this is logistically challenging. Shifts in sampling times in 2016 and 2022 – some exceeding three hours – likely influenced observed temperature trends, with earlier sampling generally associated with lower recorded temperatures. Additionally, the extremely high flow events in 2023 (Lorrey et al., 2025) may have further skewed short-term trends downward.

The consistent increase in temperature over the 15-year trends across all land cover types – including native forests – suggests the influence of climate change and rising atmospheric temperatures. The contrast between the 7- and 15-year trends likely reflects this climatic signal: 15-year trends are long enough to capture long-term changes, whereas 7-year trends may be too short to do so. However, this interpretation requires confirmation through analysis of continuous temperature data, as the observed changes may still be influenced by the shift to later sampling times in 2016.

Warmer stream water holds less oxygen and also increases biological oxygen demand, leading to lower DO levels. This creates an expected inverse relationship between temperature and DO. The current state confirms this pattern, with higher DO concentrations in cooler, native forest streams and lower concentrations in warmer urban streams. However, this relationship is not straightforward at individual sites due to confounding influences such as organic matter inputs, wastewater discharges, and nutrient levels. For example, the Waitangi Stream in Pukekohe, although relatively cool, has the lowest median DO concentrations – associated with high nutrient enrichment in the Pukekohe area (see section 6). Low DO levels were observed in all Pukekohe streams, with all sites graded in Band D (NPS-FM) based on summer 1-day and 7-day minimum DO attributes (Young et al., 2025).

Conversely, Botany Creek (East), the warmest stream with high nutrient levels, had the highest DO concentrations. This is likely due to elevated photosynthetic activity from algae and macrophytes during daylight sampling hours, which can oversaturate the water with oxygen. However, this oxygen level could decrease substantially at night, leading to hypoxic conditions.

Mahurangi River (Right Tributary) and Mahurangi River (Lower) were the only two sites showing decreasing temperature trends for the 15-year trend period. This is surprising given that the Right Tributary underwent large-scale exotic forest harvesting between 2012 and 2019, with forest cover in the catchment dropping from 97% to 35%. The cause of this cooling trend is unclear. A potential further investigation could look at effects of the long-term restoration and planting efforts in the Mahurangi Regional Park area on stream water quality parameters including temperature.

It is unclear why pH trends across all but two sites showed a decrease over the 7-year period, while over the 15-year period pH was increasing at most sites. Contributing factors to this discrepancy might include the implementation of daily calibration procedures in 2019, following the release of the NEMS protocol, and the unprecedented rainfall in 2023 (Lorrey et al., 2025), which may have contributed to lower pH levels. Similarly, trends in coastal waters showed decreasing pH levels at the majority of sites over the 7-year period, while pH increased over the 15-year period (Kamke and Gadd, 2025).

Dissolved organic carbon (DOC) exhibited spatial patterns across the Auckland region (see the online Data Explorer for an interactive map). Higher DOC levels were recorded in the Rangitōpuni, Kaukapakapa, and Kumeū Rivers and the Ararimu Tributary in the northwestern area. These are rural, or exotic forest sites within pine plantations (e.g., Riverhead Forest). Elevated DOC levels here likely result from high organic matter inputs and limited microbial decomposition. In contrast, Pukekohe-area streams in the south had lower DOC levels, likely due to volcanic soils that bind organic matter and promote subsurface flow paths, which filter DOC.

Low DOC levels at native forest sites – despite high organic matter availability – suggest high microbial decomposition and mineralisation rates (Findlay et al., 2001). These low DOC levels, combined with low hardness and relatively high pH, particularly in Cascades Stream (Waitākere) and Wairoa Tributary, mean heavy metals like zinc and copper, will be more bioavailable, which can threaten stream health.

In contrast, Ararimu Tributary – characterised by relatively high DOC and low pH – creates low-metal-bioavailability conditions. However, as pH is rising and DOC is declining at this site, metal bioavailability may increase in the future.

6 Nutrients

Excessive nutrients can have direct and indirect effects on stream ecosystems by fuelling excessive growth of plants and algae which can then affect dissolved oxygen levels and communities of aquatic fauna, though these effects are complex and variable. Dissolved forms of nutrients are the most readily available for uptake by algae, but nutrients bound to sediments in the stream bed can also support the growth of plants. Some forms of nitrogen can be toxic and have direct effects on the health and survival of fish and macroinvertebrates.

Inorganic nitrogen occurs in three forms nitrate, nitrite, and ammoniacal nitrogen. Nitrate is the preferred form of nitrogen for plant uptake. It is highly soluble and easily transported through soils, particularly after heavy rainfall, to streams through lateral flow, or via deep drainage to underlying aquifers (Pearson and Rissman, 2021). Nitrite nitrogen is an intermediary form between ammoniacal nitrogen and nitrate, and it is usually short lived in the environment. Ammoniacal nitrogen is less soluble and is more likely to enter waterways through more direct pathways such as wastewater discharges. Most phosphorus enters waterways attached to sediment via overland runoff (Pearson and Rissman, 2021).

6.1 State

In this section, the distributions of water quality observations are first described to reveal broader patterns across the region, seasonal variability is also described. Secondly, grading of water quality attributes under the National Objectives Framework (NOF) is undertaken.

6.1.1 Distributions across land cover

Data from sites within each land cover category were pooled to reveal broader patterns across the region. Concentrations also varied both between and within sites. This variation is illustrated in box plots in Appendix 5 and online in the Data Explorer.

The violin plots[9] below show that there were considerable differences in the concentration and spread of both ammoniacal N and values adjusted for pH across land cover categories (Figure 6-1). After undertaking adjustment for pH, ammonia concentrations were lower than for ammoniacal N. Adjusted values were based on imputed values when ammoniacal N concentrations were lower than detection limits resulting in a broader apparent distribution than for ammoniacal N. A long upper tail is evident for ammonia concentrations among urban sites, indicating occasional high concentrations were more common in these streams than in rural streams. Ammonia concentrations were an order of magnitude higher at the Pakuranga Stream (Urban – Project) than at other urban sites. A synoptic survey conducted in 2024 traced the likely source of this ammonia to the boundary of the Greenmount closed landfill where the underground drainage connects to the stormwater network.

[9] Nutrient concentrations span a wide range, and plots are displayed using a log scale to better visualise the distribution. The online Data Explorer also provides alternative options to display results on log scale or numerical scale to visualise absolute differences.

Further monitoring and investigations of the impacts of the landfill on the receiving environment are currently underway through the resource consenting process.

Dissolved inorganic nitrogen is the sum of total oxidised nitrogen and ammoniacal nitrogen. The shape and distribution were very similar to total oxidised nitrogen which reflects the dominant proportion of DIN (typically >90% of DIN). There was a general pattern of lower DIN concentrations at native forest sites, moderate concentrations at exotic forest and rural sites (High and Low), and higher concentrations at urban sites. Streams in the wider Pukekohe area have the highest inorganic nitrogen levels. Bimodality is evident for native forest, exotic forest, and 'Pukekohe' streams reflecting the smaller number of sites in these categories and some differences among sites within these groups. Total nitrogen concentrations showed a similar pattern across land cover categories.

There was a general pattern of increasing phosphorus concentrations from forested streams through to urban streams (Figure 6-2). Phosphorus concentrations were high at the 'Urban – Project' site compared to other urban streams. Streams in the wider Pukekohe area had considerably lower phosphorus levels than other rural sites (High or Low) (Figure 6-2). As with DIN, the 'Rural-High' and 'Rural-Low' sites shared a similar distribution although a long upper tail was evident for the Rural-Low group. This was driven by higher concentrations at a single site – Okura River (see Appendix 5). A long upper tail was also evident for urban sites, also largely driven by a single site – Newmarket Stream (see Appendix 5). Bimodality was evident for monitored streams within native forest catchments reflecting differences in reference conditions for phosphorus within the region.

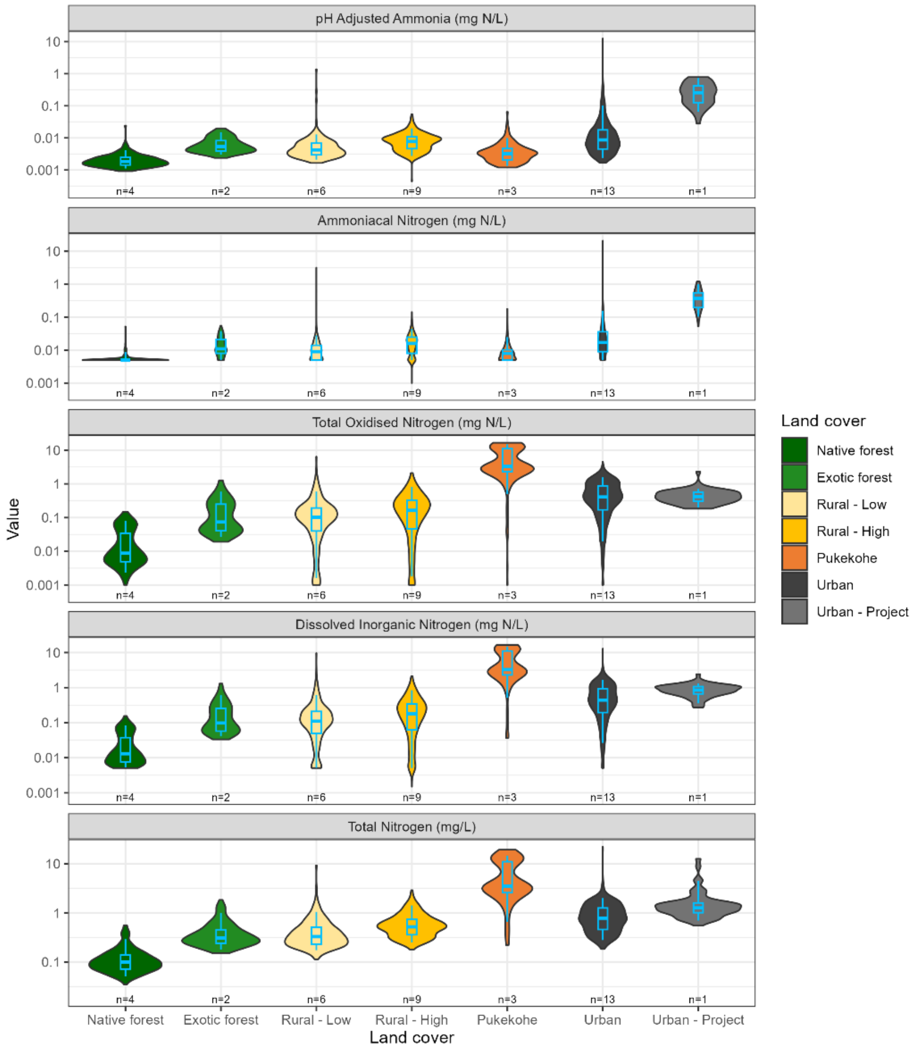


Figure 6-1: Violin plots showing the distribution of observed nitrogen concentrations across sites within each dominant land cover class (01 July 2019 - 30 June 2024). The blue box indicates the interquartile range, the central line is the median, and the whiskers are the 5^{th} and 95^{th} quantiles. n indicates the number of sites represented in each violin. Note log-scale axis in these plots may visually reduce apparent differences.

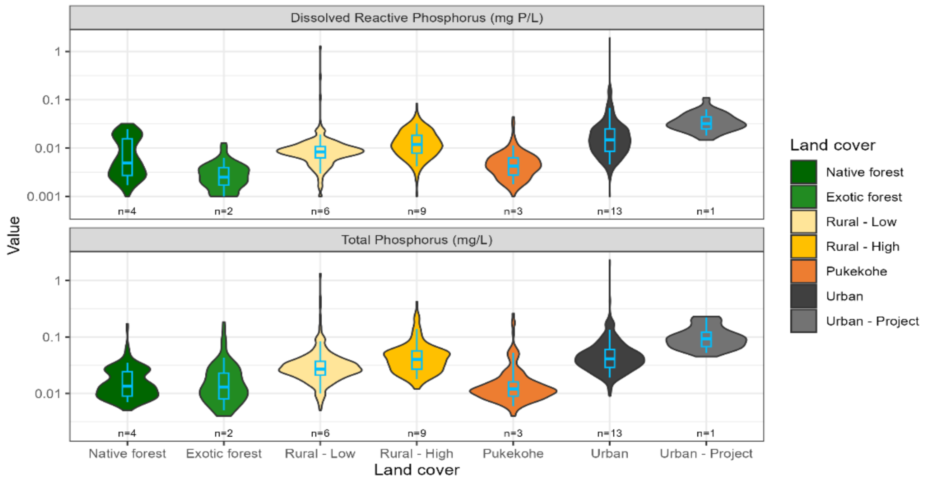


Figure 6-2: Violin plots showing the distribution of observed phosphorus concentrations across sites within each dominant land cover class (01 July 2019 - 30 June 2024). The blue box indicates the interquartile range, the central line is the median, and the whiskers are the 5th and 95th quantiles. n indicates the number of sites represented in each violin.

6.1.2 Seasonality

There were clear seasonal (monthly) patterns for total oxidised nitrogen with76% of sites displaying significant seasonal patterns with large amplitude changes indicated by changes from bright yellow to dark purple squares (Figure 6-3). Concentrations were higher in winter months (Jun-Aug) and lower in summer (Dec - Feb) across almost all rural (High and Low) sites. Although two of the three streams in the 'Pukekohe' area showed statistically significant seasonal patterns, the amplitude of seasonal variability was small relative to other rural streams. Statistically significant seasonal patterns were also identified for >83% of urban streams. The amplitude of seasonal changes in these streams was larger at the four sites that have a high proportion of rural land use within the catchment (Papakura Stream (Lower), Vaughan Stream, Puhinui Stream, and Ōtara Creek (East)) (Figure 6-3). Seasonal patterns were also observed at two native forest sites, and one exotic forest site though with a lower amplitude.

Seasonal patterns were less evident for ammoniacal nitrogen with only a third of sites showing statistically significant seasonal (monthly) changes (Figure 6-3). Statistically significant seasonal patterns (higher concentrations in winter) were identified for all 'Rural – High' sites (except Rangitōpuni River), and 50% of 'Rural – Low' sites. Seasonal patterns were not identified among 'Urban' streams except for two sites with a high proportion of rural land use in the catchment (Papakura – Lower, Puhinui Stream) and Ōtara Creek (East)). No patterns were identified at native reference sites where concentrations were consistently low.

Only 27% of sites were found to have statistically significant seasonal patterns in dissolved reactive phosphorus concentrations (Figure 6-4). Where significant seasonal patterns were identified, these also varied among sites, with winter concentrations higher at some sites, but lower at others (e.g. Puhinui Stream compared to Omaru Creek) (Figure 6-4). The clearest seasonal signature was observed at Ararimu Tributary (Riverhead) with peak concentrations occurring in the summer (Figure 6-4).

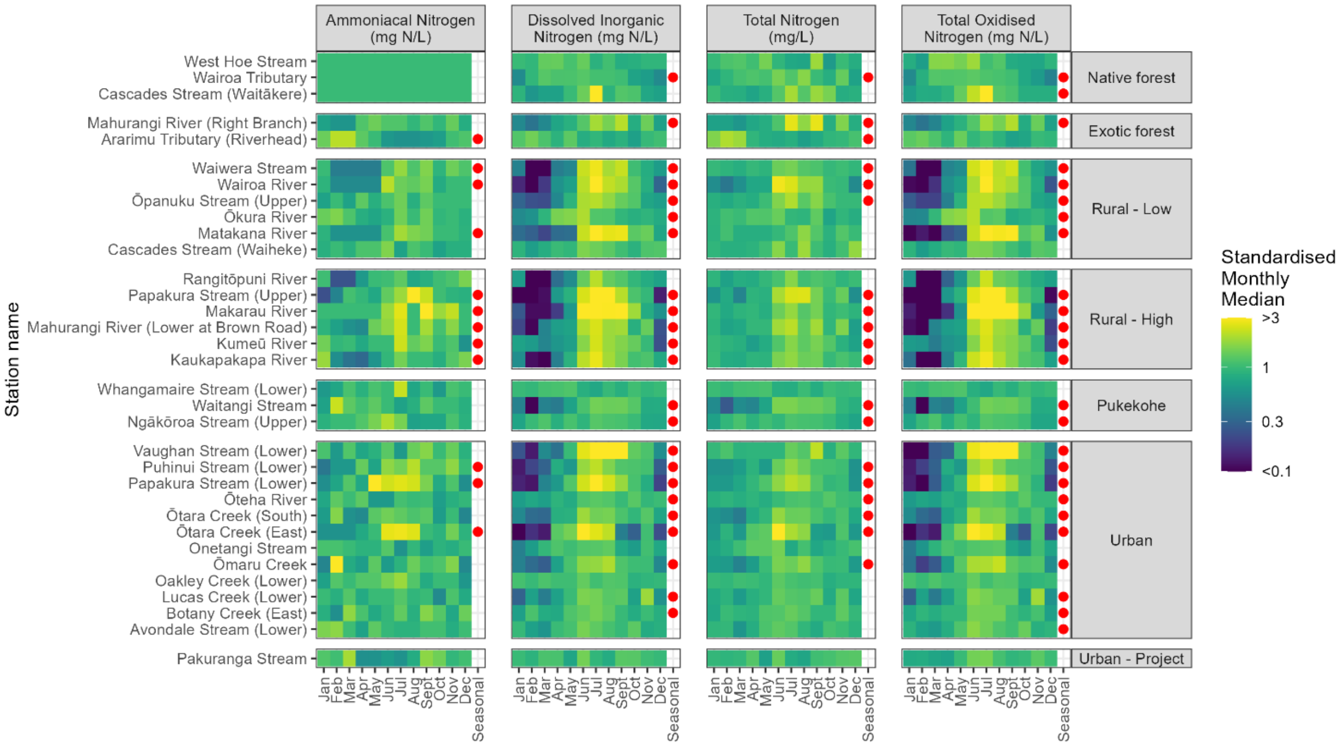


Figure 6-3: Water quality monthly medians standardised by overall median for the current state period (01 July 2019 - June 2024) for nitrogen water quality variables at each site. Red dots indicate significant seasonal patterns (p<0.05).

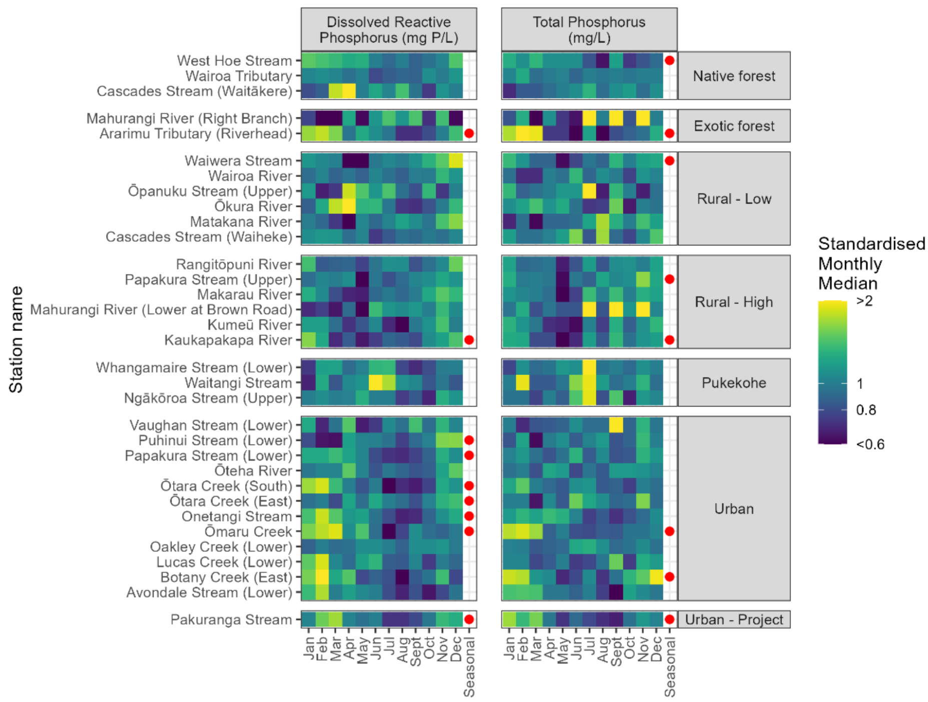


Figure 6-4: Water quality monthly medians standardised by overall median for the current state period (01 July 2019 - June 2024 for phosphorus water quality variables at each site. Red dots indicate significant seasonal patterns (p<0.05).

6.1.3 NOF – ammonia and nitrate toxicity

National bottom lines are defined for nitrate and ammonia forms of nitrogen. Bands C and D indicate that there is a risk that the growth and survival of some sensitive species may be affected by chronic, or long term exposure.

Regionally there is a low risk of nitrate or ammonia toxicity to aquatic fauna even for the most sensitive species. Most monitored streams across the region have nitrate and ammonia levels that are above the national bottom line (in bands A and B) (Figure 6-5 and Figure 6-6). This indicates that 95% of aquatic species would be expected to be protected from chronic toxicity effects in these areas.

In the wider Pukekohe area of south Auckland, some rural streams failed the nitrate toxicity national bottom line (Figure 6-6). Our previous monitoring (Ingley et al., 2023) showed all three streams in this area (Ngākōroa Stream, Whangamaire Stream, and Waitangi Stream) were below the national bottom line for nitrate toxicity (band C or D, see Appendix 10 rolling state assessment). However, this latest assessment has graded Waitangi Stream above the national bottom line, in band B. The most impacted site was Whangamaire Stream which was graded in band D. See box insert 'Pukekohe Nitrate Contamination' below for more information.

Several urban sites were graded in band B for nitrate toxicity including Newmarket Stream, Oakley Creek, Ōtara Creek (South), Botany Creek, and Ōmaru Creek (see Appendix 9). These sites are the most intensively developed catchments with >80% built up area in the catchment. This grading was driven by the 95th percentile metric except at Newmarket Stream which was also graded in band B for the median state. One 'Rural' (High) stream was also graded in band B, Papakura Stream (Upper). This was also driven by the 95th percentile metric and appears to be associated with seasonal peaks in June-August. While the 95th percentile metric is more likely to be influenced by differences in climatic conditions, particularly stormflow conditions, further review of the rolling state over time indicates consistency in this grading assessment within each site (Appendix 10). All other sites across the region were graded in band A for nitrate toxicity.

A single location failed the national bottom line for ammonia toxicity, for both metrics, Pakuranga Stream (Urban – Project). Several 'Urban' sites were graded in band B including Newmarket Stream, Botany Creek, Ōmaru Creek, Ōtara Creek (South), and Avondale Stream, and one 'Rural' stream – Ōkura River. This band assessment was driven by the 95th percentile metric for all but Newmarket Stream which was also graded in band B for the median metric (Figure 6-5). Over the past five years, the highest concentrations of ammoniacal N were recorded at Newmarket Stream (21 mg/L) and Botany Creek (12.4 mg/L) coinciding with wastewater observed in the stream at the time of sampling. All other sites across the region were graded in band A for ammonia toxicity.

Review of the rolling state assessment over time (Appendix 10) indicates consistency in grading within each site over time (including over the method change period). Only two sites changed band over time, Avondale Stream and Ōkura River where the grades changed from A to B, and at Newmarket Stream where the interim grade changed from C to B. These changes coincided with short term increases in ammoniacal N concentrations were observed over several months at Avondale Stream (September 2021 - January 2022) and at Ōkura River (February 2024 - April 2024). There were no obvious signs of pollution observed at the time and there were no records of pollution response incidents in either area (pers. comm. Pollution Compliance team).

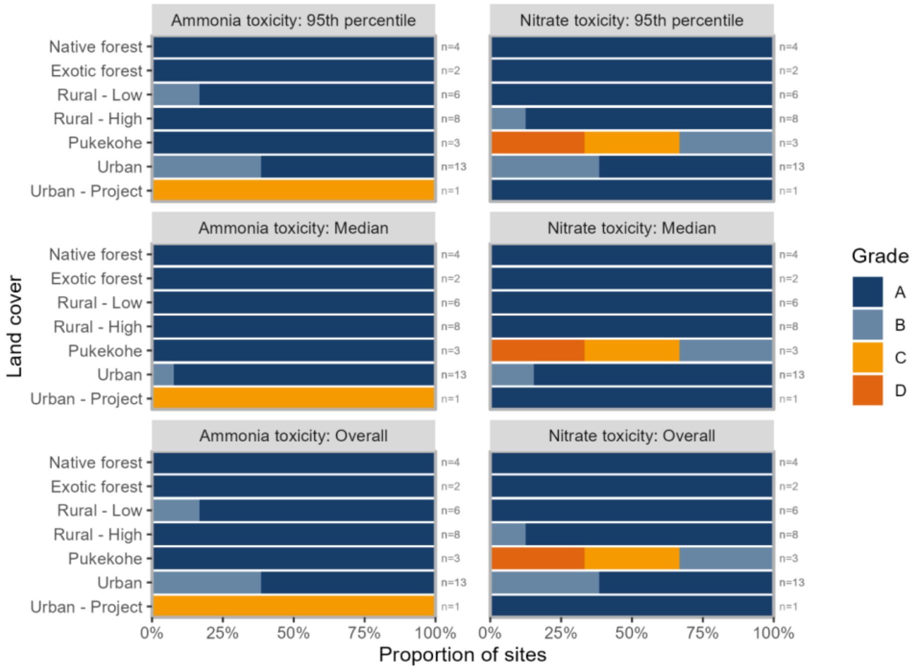


Figure 6-5: The proportion of river sites in each attribute band metric for the current state (01 July 2019 to 30 June 2024) for Ammonia (as pH adjusted NH4) and Nitrate toxicity (as TON) grouped by dominant land cover class. Sites with interim grades are included in this plot.

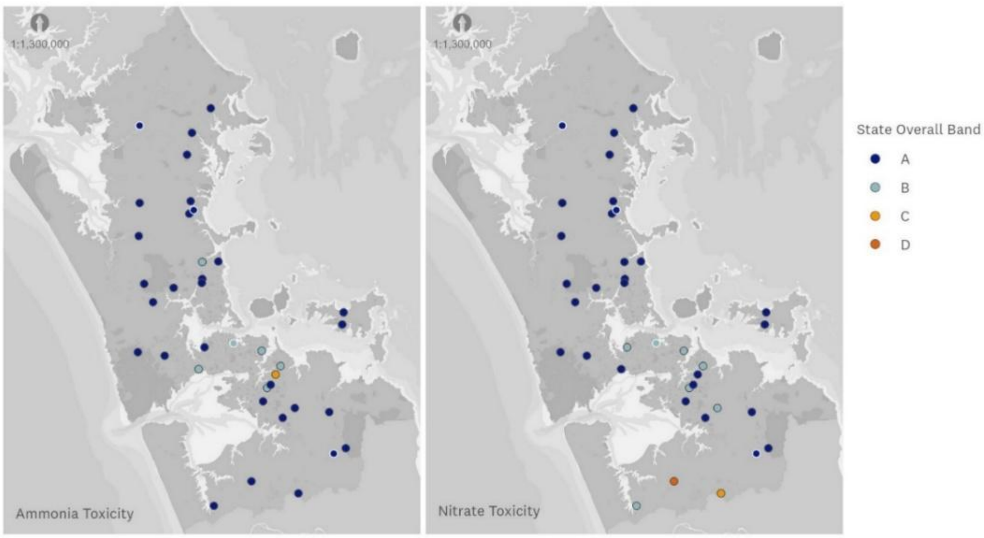


Figure 6-6: Regional map of site locations displaying overall grade for the current state (01 July 2019 to 30 June 2024) for ammonia and nitrate toxicity (as TON). Dots with a white border are ‘interim’ grades. Refer to Figure 2-1 for site names.

6.1.4 NOF – nutrient enrichment (DIN and DRP)

National bottom lines are not defined for dissolved inorganic nitrogen or for dissolved reactive phosphorus.

All native forest reference streams were graded within band A for DIN however only half of these reference sites were graded in band A for DRP (Figure 6-7). One native forest reference site was graded in band D for DRP.

Most rural streams had moderate nutrient enrichment with 43% of all rural (High and Low) sites graded in bands B and C for DIN and 79% of sites graded in bands B and C for DRP. Eight ‘Rural’ sites were graded in band A for DIN, whereas only one site was graded in band A for DRP (Ōpanuku Stream (Upper)). Two ‘Rural’ sites were graded in band D for DRP, Ōkura River (Rural – Low) and Papakura Stream (Upper) (Rural – High). At Ōkura River, this was driven by the 95th percentile metric associated with high nutrient concentrations observed over February 2024 -April 2024. At Papakura Stream (Upper) this was driven by the median metric indicating concentrations are elevated most of the time at this site. Both sites were also graded in band C for DIN following similar patterns.

As outlined in section 6.1.1, phosphorus concentrations were much lower, and nitrate concentrations were much higher, within streams in the wider Pukekohe area than other ‘Rural’ streams. This is reflected by all sites in this southern area graded in band D for DIN and in band A for DRP (Figure 6-8).

Urban streams were the most impacted by nutrient enrichment across both nitrogen and phosphorus. Only three of 13 urban sites were assessed in band A for DIN (Onetangi Stream, Vaughan Stream, and Lucas Creek). Three urban sites were graded in band D for DIN (Newmarket Stream, Ōmaru Creek and Te Auaunga/Oakley Creek). The remaining sites were in bands B and C. Over 90% of ‘Urban’ streams were graded in bands C and D for DRP and no sites were graded in band A.

Review of the rolling state assessment over time (Appendix 10) indicates that the step change observed for DRP from 01 July 2017 (see Appendix 4) coincides with in a change in band assessment by at least one band for many sites spanning this change, particularly at the 2020 and 2021 periods (01 July 2015 – 30 June 2020, 01 July 2016 – 30 June 2021). These changes in band should not be interpreted as an improvement over time. Trend analysis is provided in section 6.2 below.

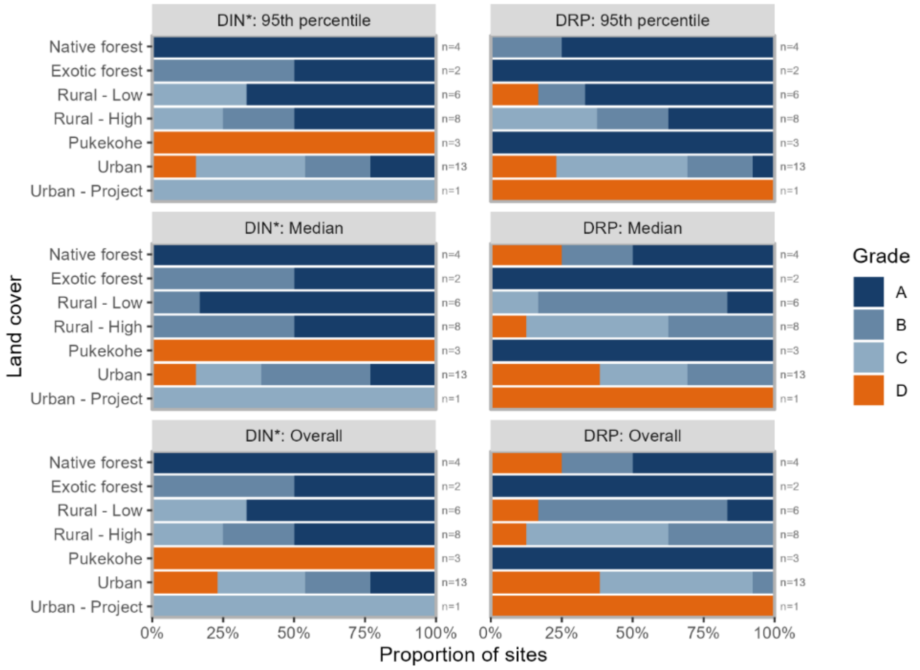


Figure 6-7: The proportion of river sites in each attribute band metric for the current state (01 July 2019 to 30 June 2024) for DIN and DRP grouped by dominant land cover class. Sites with interim grades are included in this plot. *DIN based on a draft attribute table (STAG, 2019).

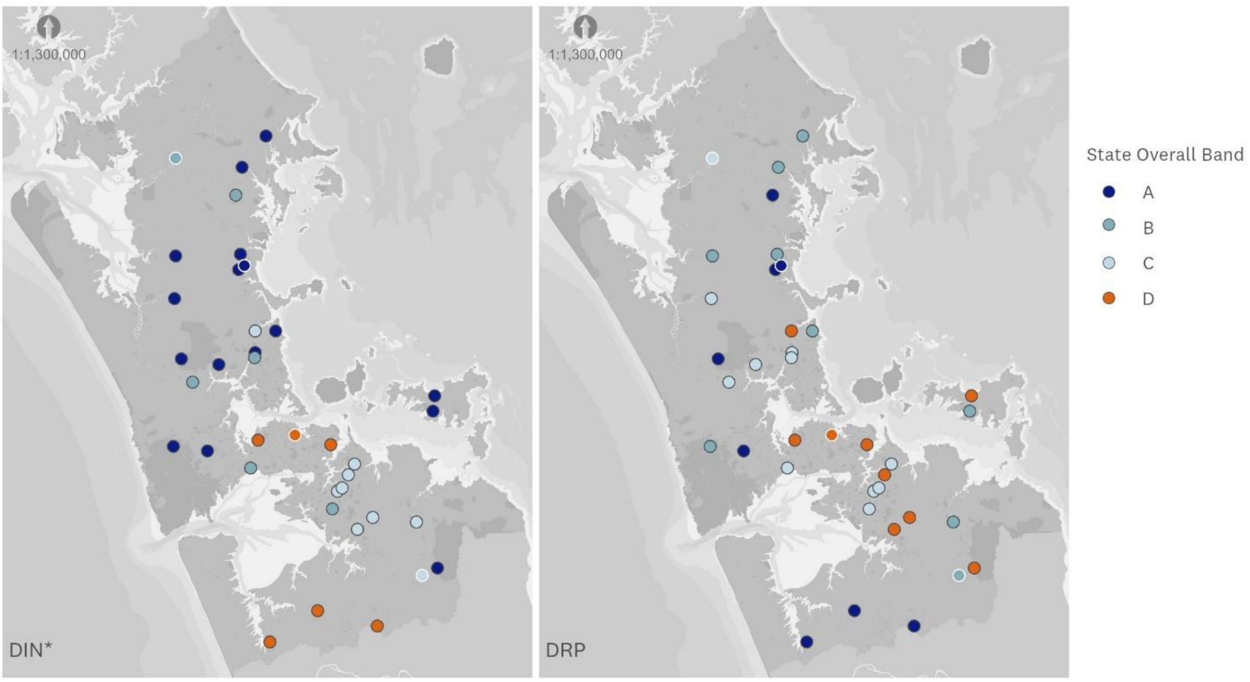


Figure 6-8: Regional map of site locations displaying overall grade for the current state (01 July 2019 to 30 June 2024) for DIN and DRP. Dots with a white border are 'interim' grades. Refer to Figure 2-1 for site names. *DIN based on a draft attribute table (STAG, 2019).

6.2 Trends

Trends were analysed over the seven-year period from 1 July 2017 to 30 June 2024 for all forms of nutrients.

Ammoniacal nitrogen concentrations were found to be very likely degrading at the majority of sites (60% across the region) with a further 14% likely degrading. Only one site (Ōpanuku Stream (Rural – Low) was found to be likely improving. Trends in total oxidised nitrogen were mixed and there was a relatively high proportion (31% of all sites) with low confidence in the trend direction. Across other rural and forested streams there were some degrading and some improving trends. No urban streams were found to be improving for soluble nitrogen forms however total nitrogen concentrations were found to be improving at Vaughan Stream and Lucas Creek (urban). Trends also varied among native forest sites suggesting that climatic or natural processes contributing to these observed trends may vary spatially. Both Nukumea Stream and West Hoe Stream were found to have very likely degrading concentrations of total oxidised nitrogen, which appeared to be associated with sustained higher concentrations from April 2023 onwards coinciding with multiple large storm events occurring over January to May 2023. However, there was a likely improving trend at Cascades Stream (Waitākere) and trends in nitrogen forms could not be identified with confidence at Wairoa Tributary.

As expected, trends were generally consistent (the same or varying by one trend category e.g. likely to very likely degrading) between total nitrogen and total oxidised nitrogen (the greatest proportion of soluble nitrogen in total nitrogen) at each site.

All exotic forest and most native forest sites were found to be likely to very likely improving for both dissolved reactive phosphorus (DRP) and total phosphorus. One reference site (Wairoa Tributary) was found to have a very likely degrading trend in DRP. Streams in the wider Pukekohe area were also found to be improving. Conversely, all urban sites were found to have very likely degrading trends in DRP except for Papakura Stream (Lower) which was also likely degrading. Trends in the rural environment were mixed but spatially clustered (see Figure 6-10 and Figure 6-12). Trends in phosphorus concentrations tended to be improving in the north, and degrading in the southeast.

There were several sites that displayed very likely degrading trends in DRP which contrasted with very likely improving trends in total phosphorus. It is unclear what may be driving these differences, no obvious differences were identified in relation to the proportion of dissolved and total forms of phosphorus. Further evaluation, such as adjusting for flow, may be useful as this included several catchments with a high proportion of urban development occurring, Ōteha River, Ōtara Creek (East), and Vaughan Stream.

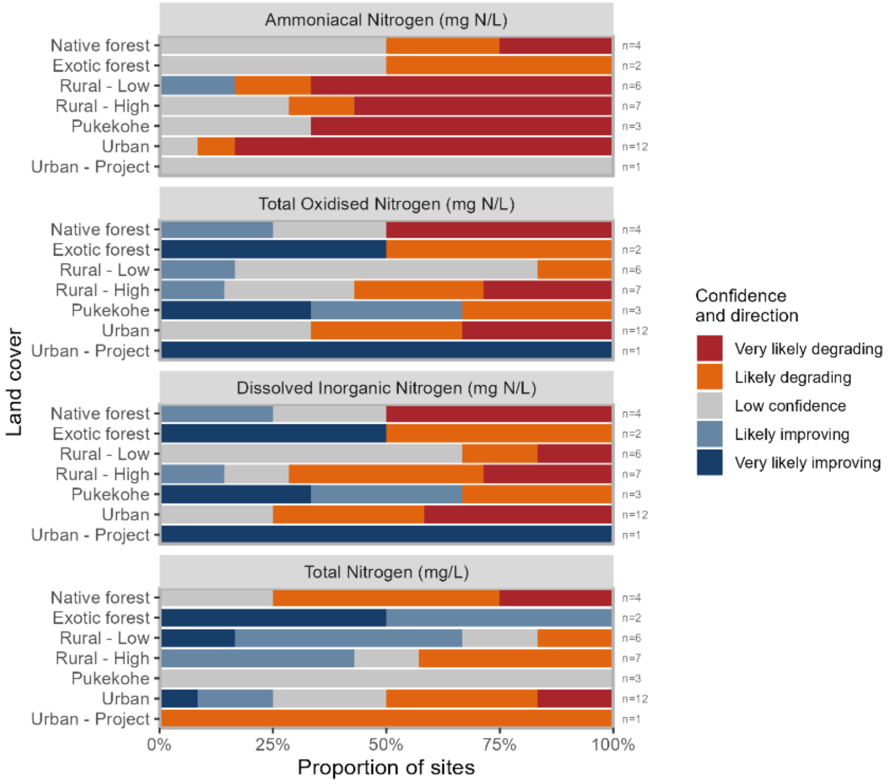


Figure 6-9: The proportion of river sites in each trend category (01 July 2017 - 30 June 2024) forms of nitrogen grouped by dominant land cover class.

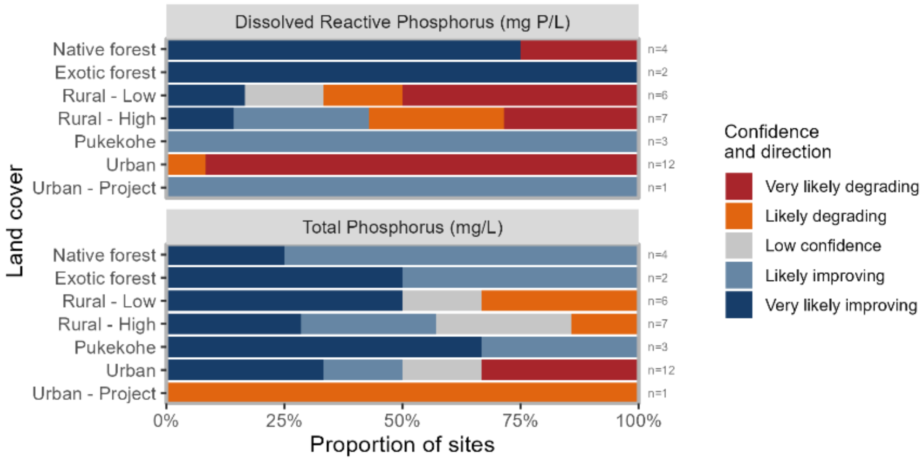


Figure 6-10: The proportion of river sites in each trend category (01 July 2017 - 30 June 2024) forms of phosphorus grouped by dominant land cover class.

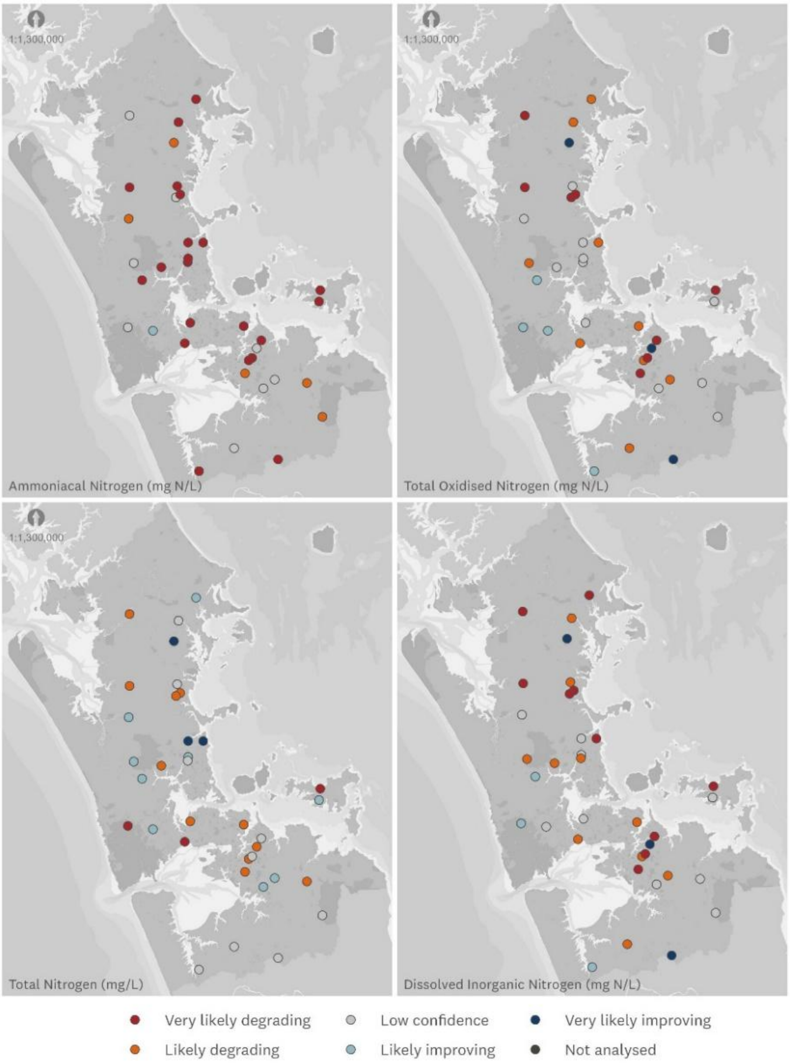


Figure 6-11: Regional map of site locations displaying trend direction for nitrogen parameters (1 July 2017 - 30 June 2024). Refer to Figure 2-1 for site names.

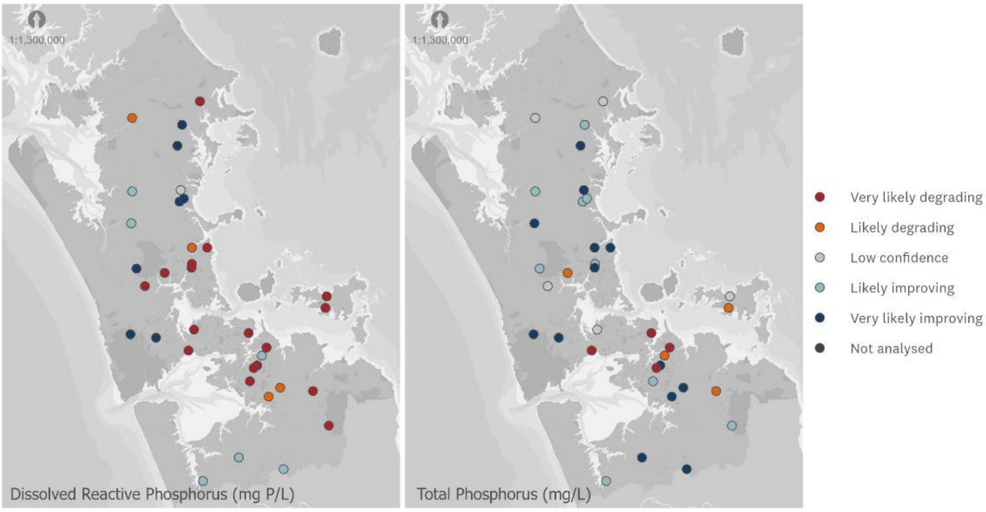


Figure 6-12: Regional map of site locations displaying trend direction for phosphorus parameters (1 July 2017 - 30 June 2024). Refer to Figure 2-1 for site names.

6.2.1 Trend magnitude + state

The annual Sen slope for each nutrient was compared to the five-year median state for that nutrient to provide information on the rate of change relative to state. The overall state can differ from the median metric, particularly for ammonia toxicity as outlined in section 6.1.3. The rate of change in ammoniacal N is not directly comparable to pH adjusted state band thresholds and therefore change cannot be expressed as an estimate of time to cross a band threshold as the band thresholds related to pH adjusted values.

At many sites, the rate of change in ammoniacal N was negligible or could not be estimated because of a high proportion of values below the detection limit (Figure 6-13). In such cases, reporting a very likely degrading trend serves as an early warning signal. There was low confidence in trend direction for the most impacted site, Pakuranga Stream, with very wide confidence intervals reflecting high variability in the data. High magnitude, very likely degrading trends were observed at sites with high median concentrations of ammoniacal N, most notably Botany Creek and Ōmaru Creek (Figure 6-13). The rate of change at these two sites was estimated at 0.0023 and 0.0036 mg/L per annum respectively. While both sites are currently within the A band for the median metric for ammonia toxicity, they were both graded in the B band overall due to the 95th percentile metric. As median concentrations at these sites are currently close to the threshold between A and B it is likely that further degradation, consistent with the current trend direction, would result in these sites being graded band B across both metrics in the future. Several other 'Urban' and one 'Rural' site had very likely degrading trends with Sen slopes ranging from 0.0017 to 0.002 mg/L per annum.

For nitrate toxicity, the highest magnitude trends (in either direction) were observed for the three most impacted sites, all located in the wider Pukekohe area. The most impacted site (graded in band D), Whangamaire Stream, was likely degrading further. However total oxidised nitrogen concentrations were likely to very likely improving at the other two sites (Waitangi Stream and Ngākōroa Stream). Ngākōroa Stream, with a very likely improving trend, is currently graded below the national bottom line in band C (median concentration 3.1 mg/L). An improving trend at an estimated rate of 0.05 mg/L per annum at this site could translate to an improvement above the national bottom line in approximately 14 years (with confidence intervals ranging from approximately seven years to no change). Furthermore, trends in the underlying shallow aquifers did not show strong evidence of change, and ongoing inputs of nitrate enriched groundwater will continue to influence these streams (see Buckthought, 2025 and box insert 'Pukekohe Nitrate Contamination' below for more information).

Likely and very likely degrading trends were observed in total oxidised nitrogen at several urban sites. For Botany Creek, currently graded band A (median concentration of 0.79 mg/L), degradation at a rate of 0.026 mg/L per annum could translate to crossing the A/B band threshold for nitrate toxicity within approximately eight years. However, wide confidence intervals indicate high variability in the data and this time estimate could range from less than five to more than 90 years

Similar patterns and sites of interest were identified for DIN. The highest rate of change was observed at Botany Creek (Figure 6-15). The site is currently graded in band C (median concentration of 0.93

mg/L) and degradation at a rate of 0.036 mg/L per annum could translate to crossing the C/D band threshold for DIN nutrient enrichment within approximately two years (with confidence intervals ranging from 1.2 to 10.5 years) (Figure 6-15).

A very likely improving trend in DIN concentration was observed at Mahurangi River (Right) tributary, currently graded in band B (median concentration of 0.25 mg/L). Early in the trend assessment period there were ‘spikes’ in total oxidised nitrogen, over April to October 2019 and May to August 2020 (coinciding with large scale land cover changes) but these spikes have not been observed since, resulting in an overall improving trend.

A very likely degrading trend, and the highest rate of change in DRP was observed at the most impacted site (band D, median concentration of 0.026 mg/L), Ōmaru Creek. DRP concentrations were estimated to be further increasing at 0.0024 mg/L per annum. The next highest rate of degradation was at Rangitōpuni River at 0.0009 mg/L per annum. The site is currently graded in band C (median concentration of 0.016 mg/L) and degradation at this rate could translate to crossing the C/D band threshold for DRP nutrient enrichment within approximately 2.1 years (with confidence intervals ranging from 1.3 to 7.5 years).

The highest rate of improvement in dissolved reactive phosphorus was observed at Cascades Stream (Waitākere) at 0.0013 mg/L per annum. Currently graded in band B, (median concentration of 0.0065 mg/L) this could translate to crossing the A/B band threshold for DRP nutrient enrichment in less than a year.

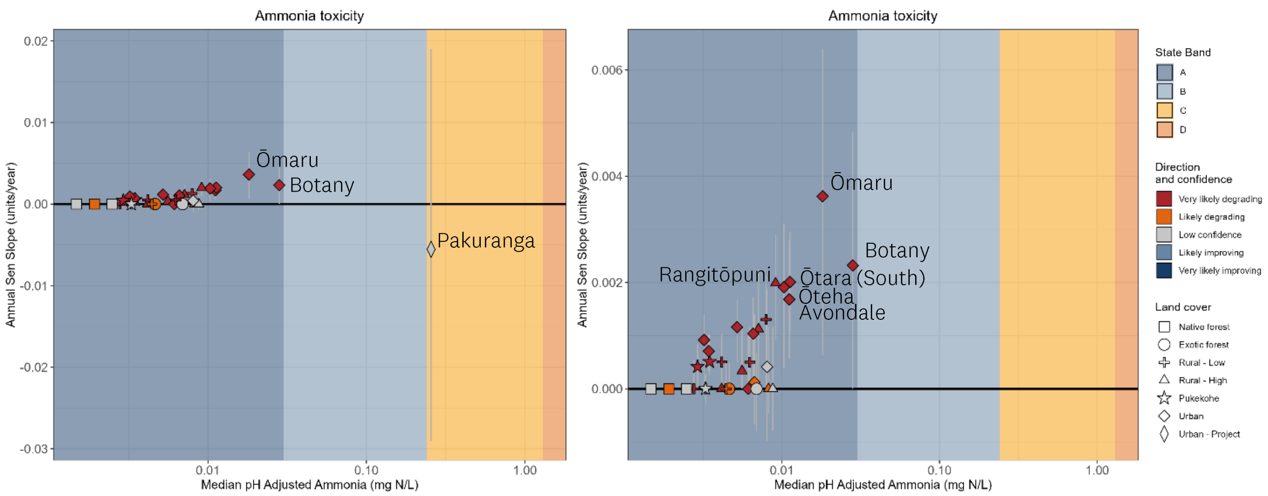


Figure 6-13: Median state (July 2019-June 2024) (median pH adjusted) vs trend rate (Sen slope ammoniacal N, not pH adjusted) (July 2017 - June 2024) all sites (left), excluding Pakuranga Stream (right). Note differences in Y axes. Error bars indicate 90% confidence intervals. Plots are coloured by the median NOF attribute band. Points are coloured by the confidence in trend direction (ammoniacal N, not pH adjusted).

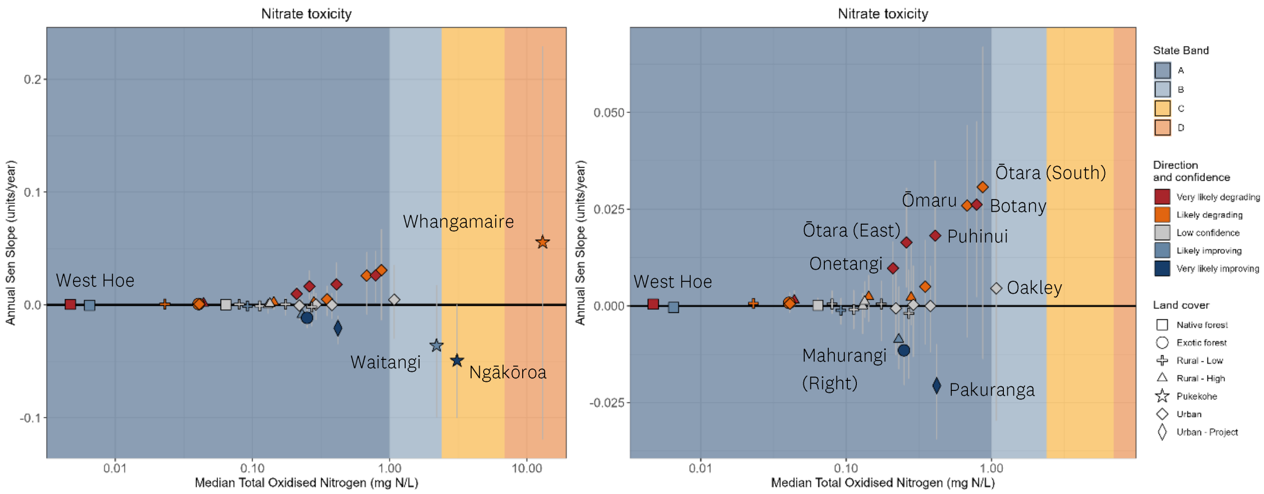


Figure 6-14: Median state of nitrate toxicity (based on total oxidised nitrogen) (July 2019 - June 2024) vs trend rate (Sen slope) total oxidised nitrogen (July 2017 - June 2024). All sites (left), excluding Pukekohe sites (right). Note differences in Y axes. Error bars indicate 90% confidence intervals. Plots are coloured by the median NOF attribute band. Points are coloured by the confidence in trend direction (TON).

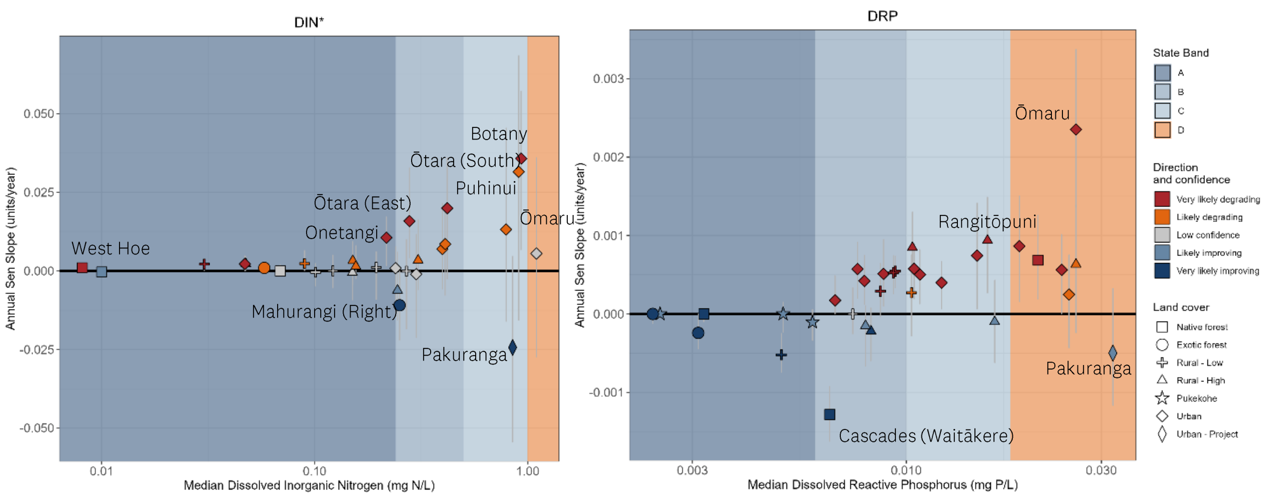


Figure 6-15: Median state of DIN (July 2019 - June 2024) vs trend rate (Sen slope) DIN (July 2017 - June 2024). (excluding Pukekohe sites) (left), Median state of DRP (July 2019 - June 2024) vs trend rate (Sen slope) DRP (July 2017 - June 2024) all sites (right). Note differences in Y axes. Error bars indicate 90% confidence intervals. Plots are coloured by the median NOF attribute band. Points are coloured by the confidence in trend direction.

6.3 Discussion

The greatest risk of toxicity impacts from nitrate on instream aquatic life were identified in the wider Pukekohe area associated with high instream nitrate concentrations (see 'Pukekohe Nitrate Contamination' below). There are indications of improvement in nitrate concentrations in two of the three monitored streams in the area with one site now graded above the national bottom line (band B).

Other risks of toxicity impacts from ammonia were identified in one urban stream. Ammoniacal nitrogen contamination at Pakuranga Stream has been traced to an ongoing long-term discharge from the Greenmount landfill to the stormwater network. This situation is unique to this location and further investigation is underway through the resource consenting process. All other sites remained above the national bottom line for ammonia toxicity however emerging risks were also identified in several highly developed urban catchments, most commonly associated with wastewater discharges to these waterways. Ōmaru Creek and Botany Creek, both in the wider Tamaki watershed, were also particularly notable for multiple nutrient enrichment issues and strong evidence of further degradation.

Areas that were identified as having nitrogen toxicity risks were also identified with the highest risks for nitrogenous nutrient enrichment based on DIN grading (as DIN is the sum of ammoniacal N, nitrate N and nitrite N). Almost all urban streams and most rural streams were found to have a moderate risk of nutrient enrichment across both DIN and DRP. The overall state for DRP was generally poorer than for DIN.

Guidance suggests that dissolved oxygen may be a relevant indicator of nutrient impacts in soft bottomed streams, which are the most common stream type in Auckland (MfE, 2023). However, the relationships between dissolved oxygen and instream nutrient concentrations are complex. Elevated nutrient concentrations in rivers can stimulate excessive primary production including the proliferation of periphyton (algae), and instream macrophytes (plants). Increased primary production can lead to changes in dissolved oxygen levels in streams leading to hypoxic or anoxic conditions occurring. However, multiple external mitigating factors can either exacerbate or moderate the effects of nutrient inputs on oxic responses including water temperature, stream hydraulics, algal and macrophyte growth, other organic inputs from plant detritus or fine sediment runoff, and groundwater inputs (MfE, 2021).

Dissolved oxygen levels are monitored continuously over the summer period at 23 of the 36 sites reported here including nine 'Rural' streams, all three streams in the Pukekohe area, seven 'Urban' streams, three 'Native forest' sites, and one 'Exotic forest' site (Young et al., 2025). The state of dissolved oxygen conditions in streams followed similar patterns to the state of nutrient enrichment (Young et al., 2025). The lowest dissolved oxygen levels were observed among the streams in the Pukekohe area, urban streams, and two other large rural waterways (Kumeū River and Rangitōpuni Stream) (Young et al., 2025). Sites that were graded in band D for summer dissolved oxygen levels were also typically graded in band C or D for DRP and/or band D for DIN.

There were exceptions to this dominant pattern. One site, Vaughan Stream, failed the national bottom line for indicators of dissolved oxygen and indicators of macroinvertebrate health (band D) (Young et al., 2025; Surrey and Storey, 2025) however the site was graded in bands A and B for DIN and DRP respectively. In contrast, one reference site (Wairoa Tributary) that was graded in band D for DRP maintained high dissolved oxygen levels, and healthy macroinvertebrate communities and was graded in band A across these metrics (Young et al., 2025; Surrey and Storey, 2025). These streams respectively exemplify exacerbating and mitigating influences of excess nutrients on primary production and dissolved oxygen in streams. Vaughan Stream is poorly shaded and macrophytes proliferate at this site, while Wairoa Tributary is a hard bottomed, shallow, stream with high shading from the surrounding native forest.

Nationally, extensive research has been undertaken relating instream nutrient concentrations to periphyton communities in hard bottomed streams (Snelder et al., 2022). Hard bottomed streams are relatively uncommon in the Auckland Region, largely restricted to the Waitākere and Hunua ranges, and Great Barrier Island. The potential for developing more comprehensive statistical relationships between nutrient concentrations and other indicators of nutrient enrichment in soft bottomed streams, such as oxic stress, is worth further consideration.

It is likely that elevated DRP levels in the upper Wairoa River Tributary are naturally occurring associated with the underlying geology in the Hunua ranges (Rissman et al., 2024). The downstream Wairoa River site was graded in band B for DRP and band C for instream dissolved oxygen levels. The upper tributaries of Papakura Stream are also located within the Hunua foothills and both upper and lower monitoring sites within Papakura Stream had elevated DRP levels (band D). Additional monitoring sites within this area, with lower intensity land uses, have been added to the monitoring network in 2022. Additional information from these new sites will improve our understanding of spatial patterns of elevated dissolved reactive phosphorus in this area.

The broader patterns in trends in forms of nutrients differed from previous national analysis (Whitehead et al. 2022). For example, national analysis found that the majority (66%) of monitored sites across New Zealand had improving (likely to very likely) trends in ammoniacal N over the 10-year period 2011 to 2020. This analysis also found there was a greater proportion of improving trends at urban monitoring sites (Whitehead et al. 2022), a group dominated by sites located in Auckland. In contrast, over July 2017 to June 2024 there was a greater proportion (60%) of degrading trends in ammoniacal N within Auckland, particularly at urban sites. Some urban sites with higher median concentrations of ammoniacal N also exhibited very likely degrading trends at a high rate of change. Trends can oscillate in direction and magnitude over different time periods of analysis (Snelder and Kerr, 2022; Whitehead et al. 2022). However, the national analysis also spans a change in analytical methodology for Auckland regional data (see Appendix 4). There was a clear discontinuity in the data set for dissolved reactive phosphorus and potential discontinuities in several other parameters. The trend analysis undertaken in this report explicitly avoided the period before July 2017 to avoid the influence of 'step changes' in the data that may induce a trend. Any previous state or trend assessments spanning the 1st of July 2017 where this has not been accounted for should be disregarded.

Pukekohe Nitrate Contamination in Rivers

Nitrate contamination in the waterways in the wider Pukekohe area has been reported for decades (Rogers and Buckthought, 2022). Nitrate contamination in these streams is largely associated with nitrate contamination in the underlying shallow aquifers which discharge to these streams, via springs, providing much of their baseflow (Morgenstern et al., 2023). High nitrate loads from farming (predominantly horticultural) activities that enter the groundwater system will eventually discharge back to the streams with little attenuation occurring overtime (Rogers and Buckthought, 2022; Morgenstern et al., 2023). This groundwater connection means there are time lags in the order of decades between the activities occurring on land and influences on water quality in streams. Whangamaire Stream is connected to actively recharging groundwater from the Pukekohe basalt lava aquifer, with mean transit times of 18 years between the recharge area and the spring outlet (Morgenstern et al., 2023). The sources of groundwater influx to Waitangi Stream are less certain, but similar mean transit times were identified (17 years) (Morgenstern et al., 2023). The mean transit time for Ngākōroa Stream was estimated at 11 years,, significantly shorter than the mean residence time of the underlying Bombay basalt aquifer (37 years) indicating that baseflow in this stream is connected to localised groundwater systems separated from the older Bombay basalt groundwater system (Morgenstern et al., 2023). Further information on water quality in these underlying groundwater systems is provided in Buckthought (2025).

While the direct effects of nitrate toxicity on instream biota are a concern in the wider Pukekohe area, nutrient enrichment in general is also a concern. All sites had high dissolved inorganic nitrogen concentrations; however dissolved reactive phosphorus concentrations were much lower than other rural streams. The observed low concentrations in surface waters (median 0.0025 mg/L to 0.0059 mg/L) are inconsistent with higher phosphorus concentrations observed in the underlying shallow aquifers (median 0.012-0.107 mg/L) indicating uptake occurring instream (Buckthought, 2025). The other main transport pathway of phosphorus to streams is typically via overland flow and transport of sediments. Excess soil phosphorus concentrations are observed in the surrounding area (high Olsen-P) which can increase the risk of P loss in runoff, particularly in compacted soils, however this risk is reduced where soils have a high retention capacity for phosphorus such as the volcanic soils that occur here (Guinto, 2025). It is possible that there is lower supply of phosphorus to streams via runoff in this area. However, extensive macrophyte production is also observed in these streams and it is possible that the low phosphorus concentrations in streams are indicative of nutrient uptake and P limitation.

All three sites in the wider Pukekohe area failed the national bottom line for summer dissolved oxygen concentrations in streams, and all indicators of macroinvertebrate health indicating the life-supporting capacity of the stream was compromised (Storey and Surrey 2025; Young et al., 2025). The lowest oxygen levels, approaching anoxic conditions, occurred at Waitangi Stream (Young et al., 2025). It is possible this is related to the discharge of anoxic groundwater in the Waitangi catchment (Morgenstern et al., 2023).

Measures of ecosystem metabolism functions varied, with no indications of impact at Ngākōroa Stream, but severe impairment was noted at Waitangi Stream (Young et al., 2025). Collectively, these attributes demonstrate that ecosystem health functions are degraded in these streams. Further investigations could look at the indirect effects on nutrient enrichment and interactions between nitrogen and phosphorus in these waterways.

Extensive macrophytes at Waitangi Stream

7 Metals – copper and zinc

Copper and zinc are naturally occurring elements found in all freshwater environments, but these metals can have toxic effects on instream communities at high concentrations. Copper and zinc are recognised as common pollutants in urban environments associated with stormwater runoff, and industrial processes and can also be associated with some agricultural practices such as copper based fungicides (Gadd et al., 2024).

The toxicity of a contaminant is influenced by its bioavailability – how readily it is absorbed by organisms. Soluble forms of copper and zinc are the most readily available for uptake. Dissolved organic carbon (DOC) binds copper and zinc reducing the amount of metal able to be absorbed by aquatic organisms (Gadd et al., 2024). Water hardness affects how metals are absorbed, particularly for zinc, as calcium and magnesium in stream water block channels of metal uptake (like gills) in animals (Gadd et al., 2024). pH impacts metal solubility and speciation in different ways (Gadd et al., 2024). While zinc is more soluble (more readily dissolved) in more acidic conditions, it is less bioavailable. High metal bioavailability conditions therefore occur in streams with low DOC, soft water, and slightly alkaline pH. Low bioavailability conditions occur in streams with high DOC, hard water, and slightly acidic pH. Refer to section 5 for further information on these water chemistry conditions across monitored sites.

Particulate forms (included in total metals) are not readily available for uptake by instream biota but may also be important for some species that are exposed via their diet (such as filter or deposit feeding organisms) and this aspect is not considered by these water quality guidelines (Gadd et al., 2024). Total metals better reflect the load transported via streams to the coastal environment, where metals can accumulate in estuarine and harbour sediments. More information on metal contamination in coastal sediments in Auckland can be found in Allen (2025).

7.1 State

In this section, the distributions of water quality observations are first described to reveal broader patterns across the region, seasonal variability is also described. Secondly, grading under the proposed regional attribute framework is undertaken.

7.1.1 Distributions across land cover

Metal concentrations vary both between and within sites as illustrated in box plots (Appendix 5, Figures A-10 and A-11) and through the online Data Explorer. In this section, data from sites within each broad land cover category were pooled to identify broader patterns across the Auckland region.

The violin plots below demonstrate that concentrations of metals are much higher in urban areas than other dominant land cover groups (Figure 7-1, Figure 7-2). For example, the median soluble zinc concentration at urban sites is approximately eight times higher than that at rural sites while the median soluble copper concentration is around double. Concentrations of metals were comparable between the ‘Urban- Project’ site and other urban sites. The log-scale axis in the zinc plots may visually reduce these apparent differences.

Concentrations are typically low across all other land use types with violin plots displaying similar shapes and distributions between land cover classes. A greater spread is observed for zinc concentrations at exotic forest sites. Indicating site specific differences between the two sites within this category (Figure 7-2). Bimodality is also observed for copper concentrations among native forest sites, indicating site-specific differences (Figure 7-1; Appendix 5).

Once values are adjusted for toxicity modifying factors, bioavailable metal concentrations are substantially lower than soluble concentrations. The median bioavailable zinc concentration for urban sites was nearly an order of magnitude greater than rural sites – with distributions spanning nearly three orders of magnitude.

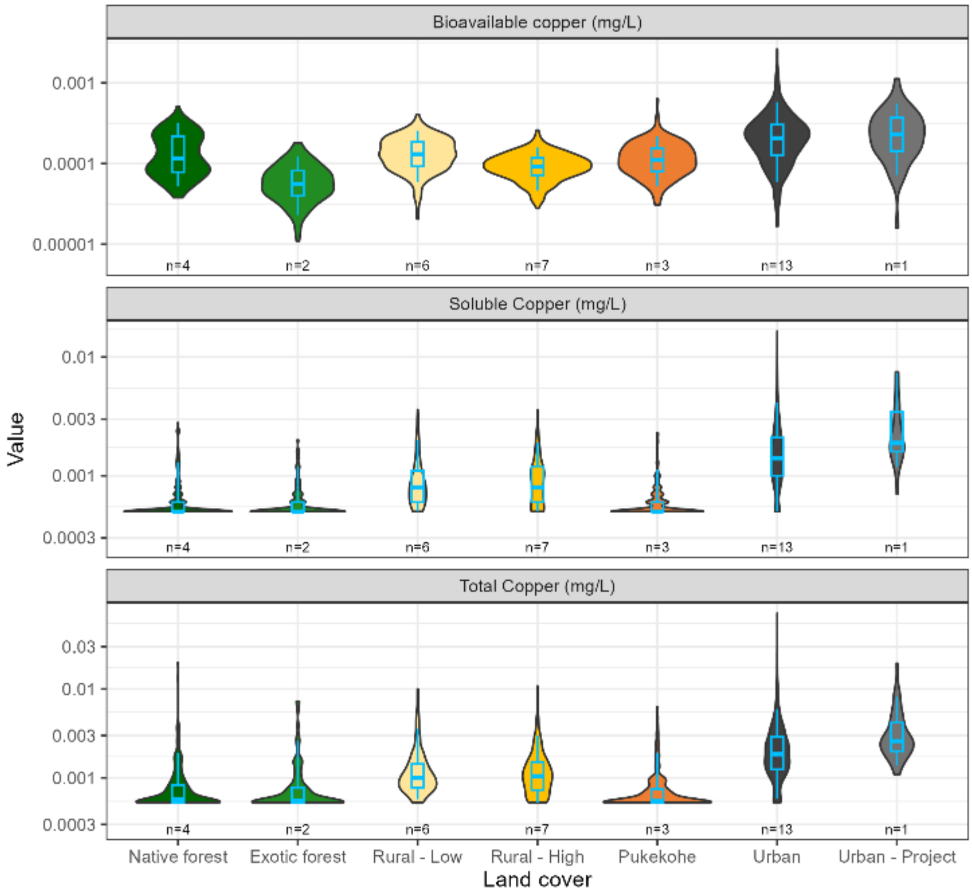


Figure 7-1: Violin plot – distribution of copper aggregated among land cover classes (01 July 2019 - 30 June 2024). The blue box indicates the interquartile range, the central line is the median, and the whiskers are the 5th and 95th quantiles. n indicates the number of sites represented in each violin. Note different Y axis among sub-plots.

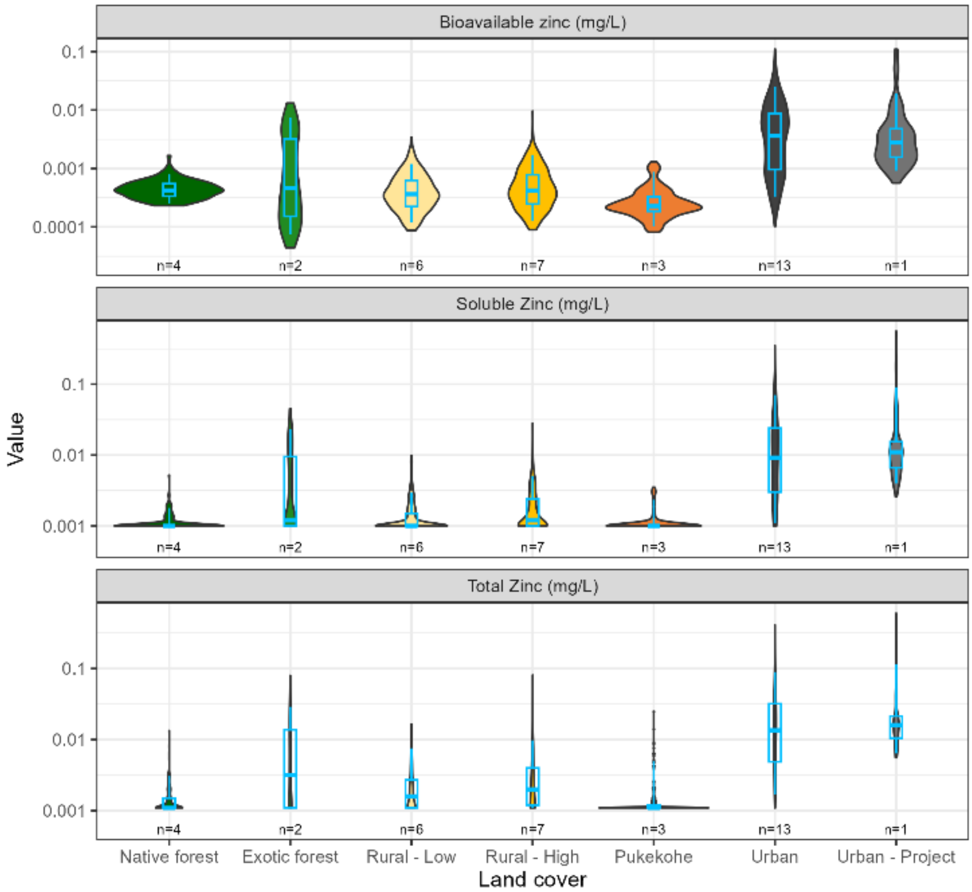


Figure 7-2: Violin plot – distribution of zinc aggregated among land cover classes (01 July 2019 - 30 June 2024). The blue box indicates the interquartile range, the central line is the median, and the whiskers are the 5th and 95th quantiles. n indicates the number of sites represented in each violin. Note different Y axis among sub-plots.

7.1.2 Seasonality

Significant seasonal (monthly) patterns were identified in soluble copper concentrations across all monitored rural-high streams and one urban stream (that maintains a high proportion of rural land use in the catchment (Figure 7-3).

Significant seasonal (monthly) patterns were identified in soluble zinc concentrations for the majority of monitored sites including at Riverhead exotic forest, most rural – high sites (83%), half of the rural – low sites and more than half (67%) of urban sites. Peak concentrations typically occurred in the winter (June – August), with lower concentrations in late summer to autumn (January to April).

No significant seasonal patterns were identified in metal concentrations at native forest or Pukekohe sites. Seasonal changes were similar for total metals with a higher proportion of significant patterns.

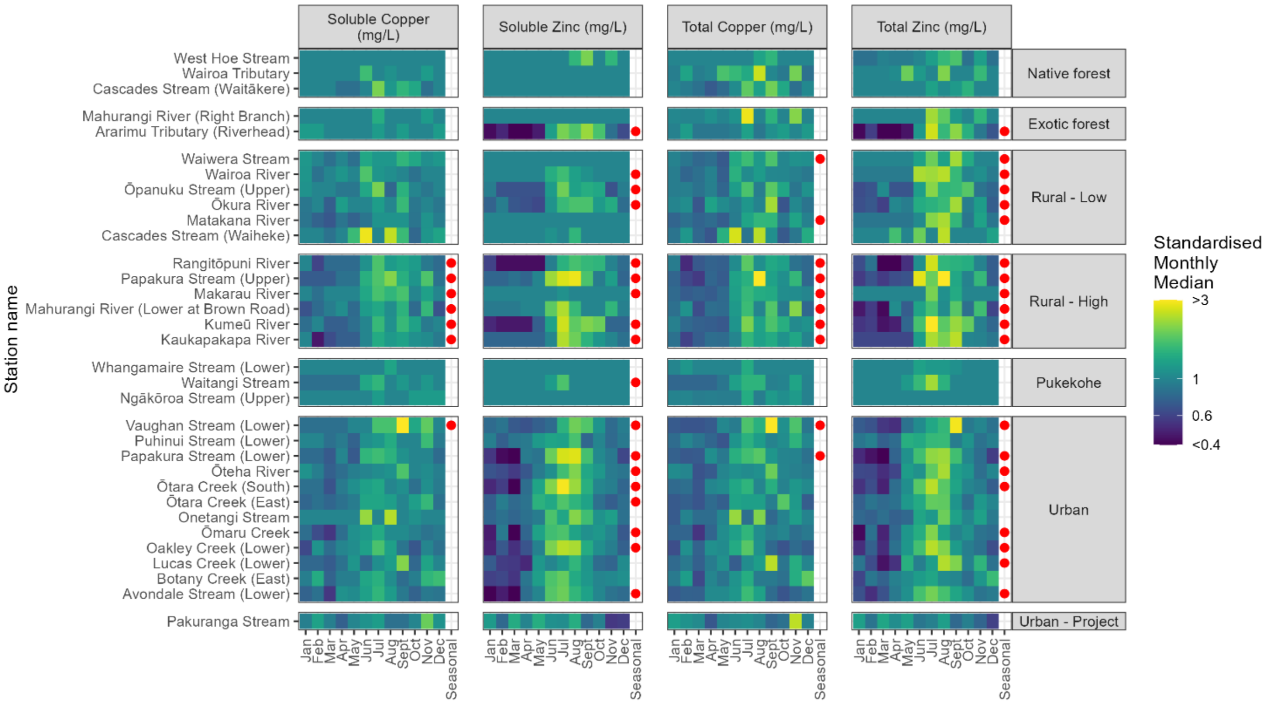


Figure 7-3: Water quality monthly medians standardised by overall median of the current state period (01 July - 30 June 2024) for copper and zinc water quality variables at each site. Red dots indicate significant seasonal patterns ($p<0.05$).

7.1.3 Copper and zinc toxicity

Copper and zinc were not included as national attributes in the NPS-FM but have been included as additional attributes for Auckland recognising the potential importance of metals in influencing stream ecosystem health (Auckland Council, 2023). Work is still underway to finalise attribute band tables including revisions to the bioavailability adjustments (see section 2.4.4.1). All grading of bioavailable copper and zinc based on the draft attribute bands is considered to be provisional.

Regionally there is a low risk of copper toxicity to aquatic fauna even for the most sensitive species. All monitored streams were above the proposed regional bottom line (band D) for chronic copper toxicity (Figure 7-4). One native forest site (Cascades – Waitākere), and one rural-low site (Ōpanuku Stream) fell within band B for copper toxicity. This was associated with high bioavailability conditions (low dissolved organic carbon) and likely to be from natural sources such as weathering from the underlying volcanic geology within the Waitākere ranges (see Appendix 5 – Site specific box plots). Urban streams had higher copper concentrations. Nearly half (46%) of the monitored urban sites were graded in band B and one site was graded in band C (Figure 7-4). This was driven by the median attribute metric indicating that concentrations are elevated most of the time, not only during peak events such as storm flows. Four urban sites[10] were in band B for both the median and 95th percentile metrics (Figure 7-4). Newmarket Stream, in the central urban area was in band C for both metrics. This reflects a moderate likelihood of chronic effects (80% species protection 50% of the time).

[10] Including the urban project site – Pakuranga Stream.

The majority of monitored urban streams were found to be impacted by zinc contamination (bands C or D). Three monitored urban stream sites failed the proposed regional bottom line (band D) for zinc toxicity, two of which failed both the median and 95th percentile metrics (Newmarket Stream and Ōmaru Stream). This reflects a high likelihood of chronic adverse effects on multiple species. A further five urban sites were graded in band C (Figure 7-4). The four urban streams that were graded in band A included the three sites with the lowest proportion of urban land use within the upstream catchment (<25% built up area) and Lucas Creek which is more extensively developed (>60% built up area).

In contrast, all monitored streams in native forest and rural catchments were graded in band A for zinc toxicity (Figure 7-4). One exotic forestry site, Ararimu Stream (Riverhead forest), was graded in band C for zinc toxicity, driven by the 95th percentile metric (see box inset “Zinc at Ararimu Tributary”) for further information).

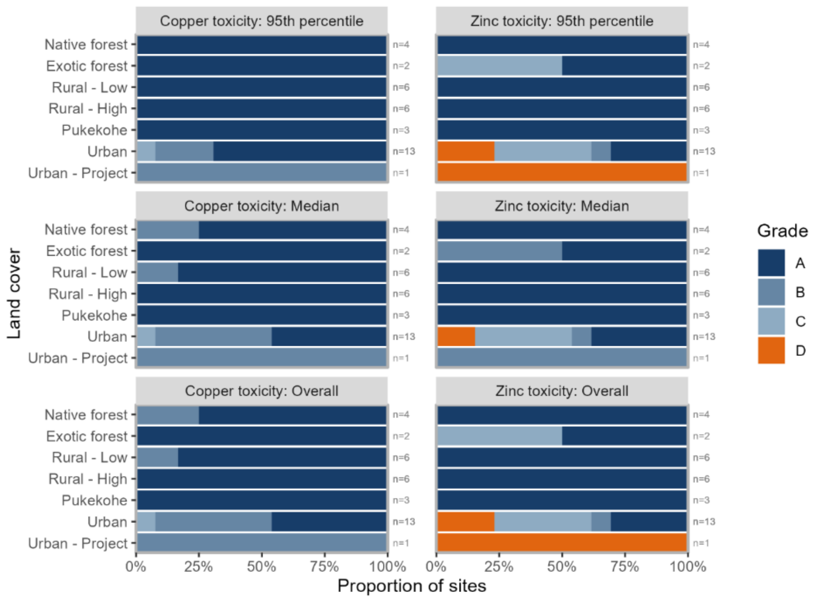


Figure 7-4: Proportion of sites within each regional attribute metric and overall band per dominant land cover class for draft regional water quality attributes for copper and zinc (1 July 2019 - 30 June 2024).

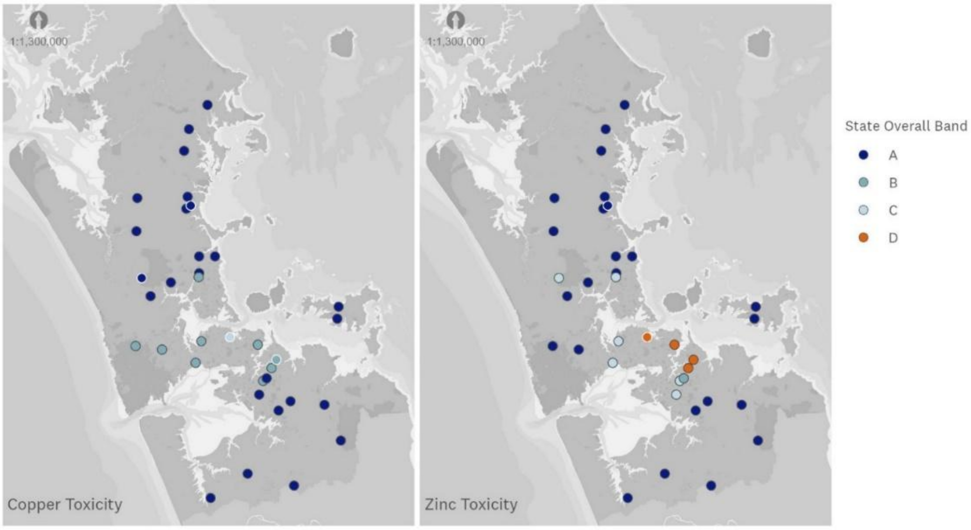


Figure 7-5: Regional map of site locations displaying overall grade for the current state (1 July 2019 - 30 June 2024) for copper and zinc toxicity. Dots with a white border are ‘interim’ grades. Refer to Figure 2-1 for site names.

7.1.4 Acute guidelines

The copper and zinc regional attributes are based on chronic toxicity – ongoing exposure over the long-term. However, toxicity from metals can also occur when organisms are exposed to higher concentrations for a short time period (acute toxicity). Recently acute guideline values have been developed for New Zealand to assist in assessing potential risks of acute toxicity. Monthly state of the environment monitoring is **not** expected to accurately assess the risk of acute toxicity of metals in streams. Higher metal concentrations are more likely to occur during storm events (if driven by stormwater inputs) which may or may not be intercepted during this routine monitoring. Even if intercepted, the single sample may not capture the peak concentration over the duration of that event.

Where we do intercept events that exceed acute guidelines, we can identify that there is a risk for these location/activity combinations and further monitoring could be prioritised to evaluate the magnitude and duration of concentrations to understand the risk of acute toxicity to freshwater biota.

All but one site exceeded the dissolved copper Tier 1 acute guideline values to protect 95% species (1.3 μg/L) at least on one occasion. This includes a reference stream within a native forest catchment (Cascades Stream (Waitākere) (Figure 7-6). However, after adjusting GVs for bioavailability (i.e. tier 2 acute), only a single site was found to have occasional exceedances of the adjusted GV (Newmarket Stream).

Seven urban sites (Avondale Stream (Lower), Botany Creek (East), Newmarket Stream, Ōmaru Creek, Ōtara Creek (South), Ōteha River, Puhinui Stream (Lower)), Pakuranga Stream, and the exotic forest site, Ararimu Tributary (Riverhead) were found to regularly exceed Tier 1 acute guideline values for 95% species protection for dissolved zinc (24 μg/L, Figure 7-7). These streams were also found to occasionally exceed bioavailability-adjusted acute Tier 2 guideline values except for Ararimu Tributary (Riverhead Forest), and Avondale Stream. Zinc concentrations were less bioavailable at Ararimu Tributary due to higher dissolved organic carbon. Further evaluation of the risk of acute toxicity effects in these urban streams including exceedance of 90% and 80% species protection guideline values could also be undertaken.

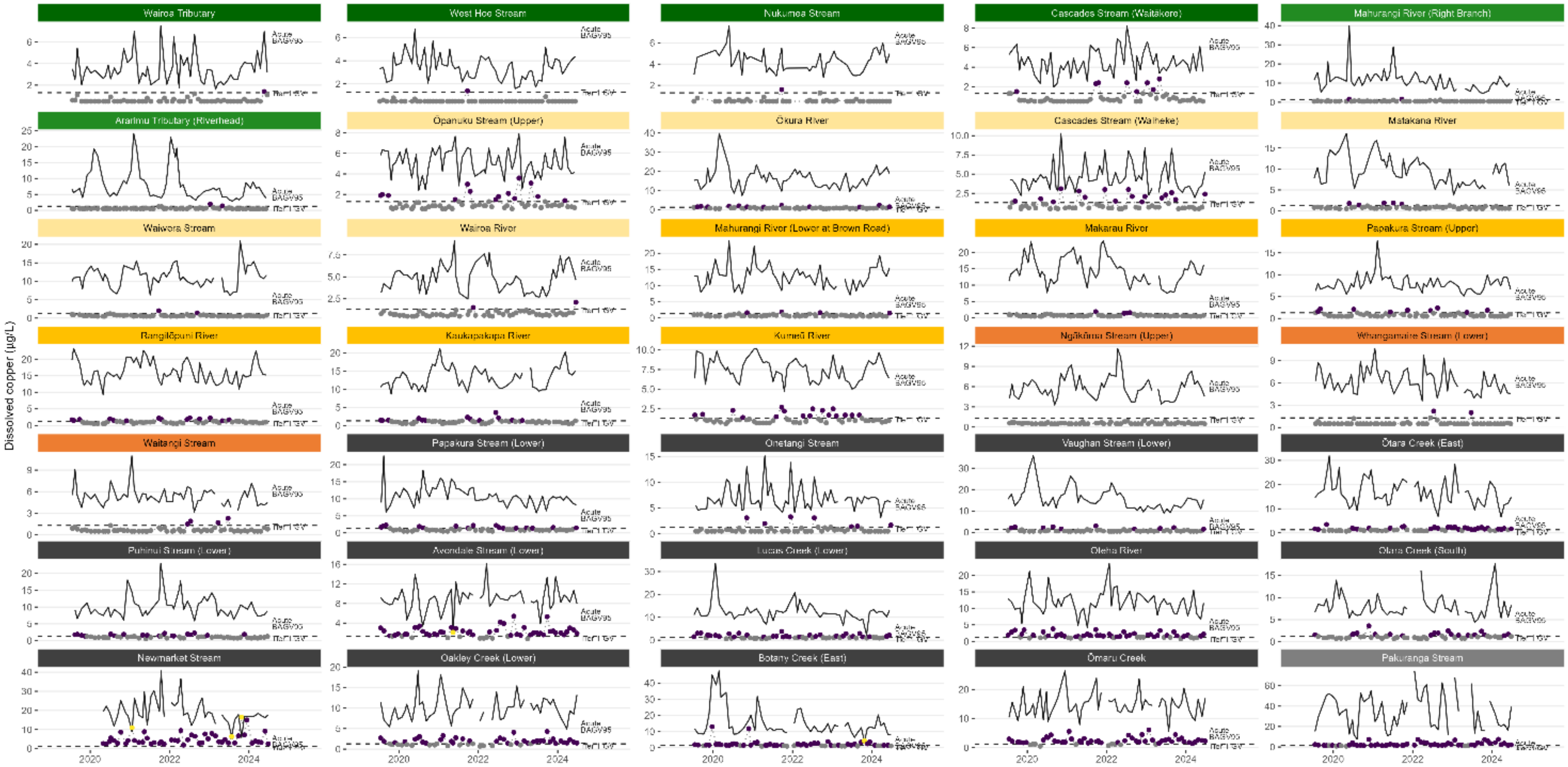


Figure 7-6: Dissolved copper concentrations compared to acute guideline values to protect 95% of freshwater species over the five year state period (July 2019 - June 2024). The dashed line shows the Tier 1 acute toxicity default guideline values (1.3 µg/L) and the solid black line shows the bioavailability adjusted guidelines. Purple dots exceed the Tier 1 GV, while yellow dots exceed the bioavailability adjusted Tier 2 guideline. Site names are coloured by the dominant land cover class (Native forest (Dark Green), Exotic forest (Light Green), Rural – Low (Light Yellow), Rural-High (Yellow), Pukekohe (Orange), Urban (Grey), Urban – Project (Light Grey)). Note the Y axis varies among plots.

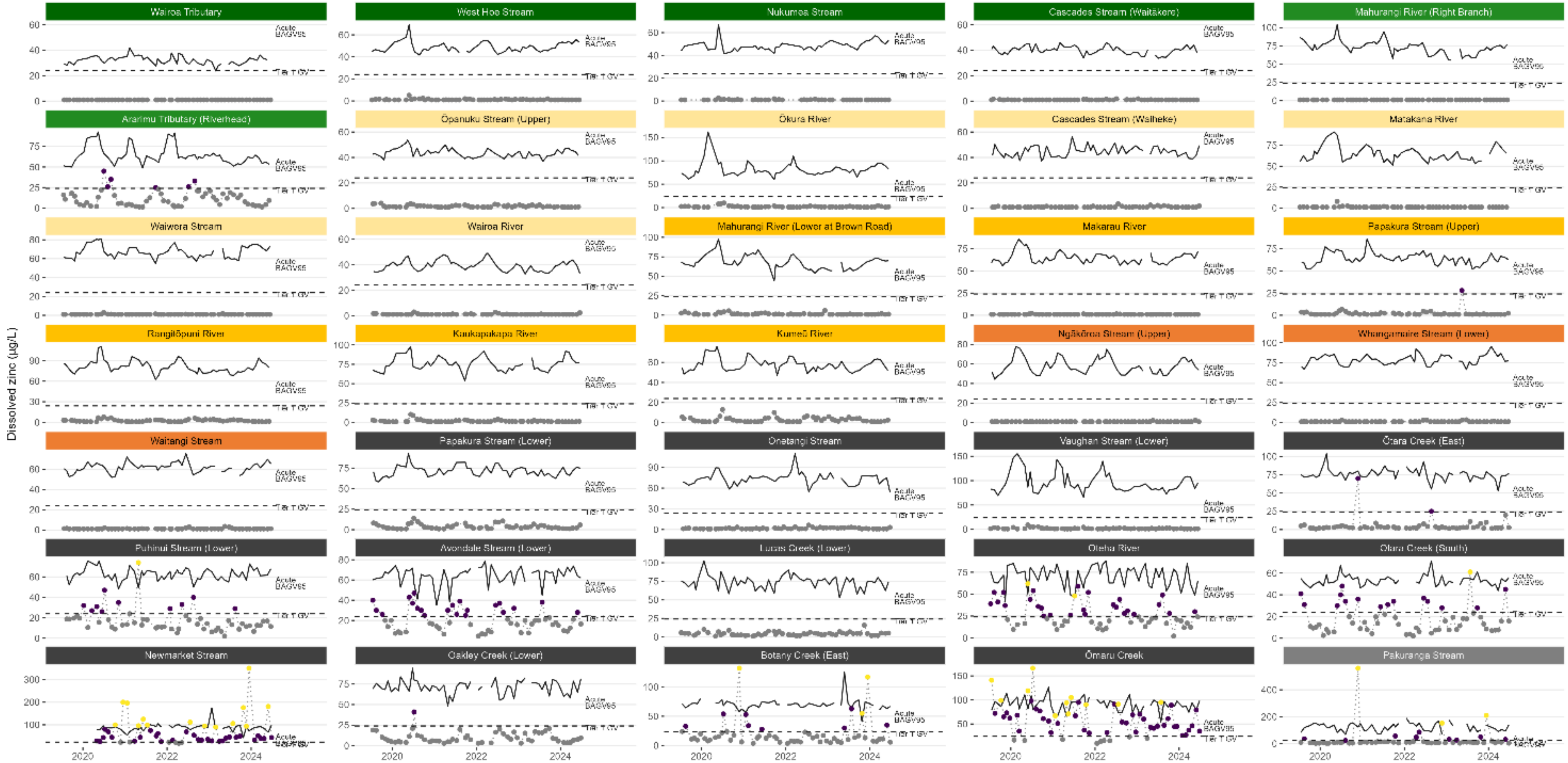
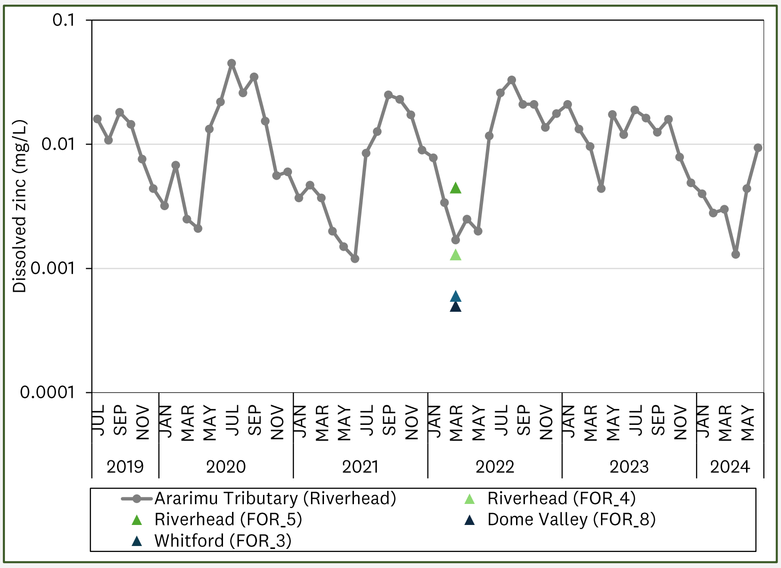


Figure 7-7: Dissolved zinc concentrations compared to acute guideline values to protect 95% of freshwater species over the five year state period (July 2019 - June 2024). The dashed line shows the Tier 1 default guideline values (24 mg/L) and the solid black line shows the bioavailability adjusted guidelines. Purple dots exceed the Tier 1 GV, while yellow dots exceed the bioavailability adjusted Tier 2 guideline. Site names are coloured by the dominant land cover class (Native forest (Dark Green), Exotic forest (Light Green), Rural – Low (Light Yellow), Rural-High (Yellow), Pukekohe (Orange), Urban (Grey), Urban – Project (Light Grey)).

Zinc at Ararimu Tributary (Riverhead Forest)

Elevated concentrations of zinc compared to other monitored forested or rural streams have been recorded at Ararimu Tributary since monitoring of metals began at this site in 2010. Concentrations at this site are indicative of moderate chronic toxicity effects on freshwater species. The site is located within a small tributary flowing through a pine plantation forest at Riverhead with harvest phases occurring within the last five years (section 3.1). A synoptic survey focussing on rural and forested catchments across Auckland was undertaken in March 2022 by NIWA using ultra trace analytical methods (Borne, 2022). Whilst investigating this zinc source was not the primary objective of that study, its findings are useful in understanding results at Riverhead.

Samples obtained from other tributaries draining Riverhead Forest also had elevated concentrations of zinc, particularly the site located in the northern part of the forest (Borne, 2022). The results from the synoptic survey from locations with exotic forestry upstream are displayed in the figure below alongside the long-term time series for Ararimu Tributary (Riverhead).



Timeseries of dissolved zinc concentrations at Ararimu Tributary (Riverhead) for the current state period (Jul 2019 to Jun 2024) with additional results from NIWA synoptic survey of exotic forest locations from March 2022 (FOR_3-8) (Borne, 2022)

These additional results indicate that the issue is common to the wider Riverhead forest area including multiple, unconnected tributaries. Further, the results show this issue is localised to Riverhead and is not consistent across all exotic forest locations. Historic fertiliser trials were conducted at Riverhead forest in the 1950s including top dressing of zinc sulphate (Weston, 1956). It is unknown if zinc is included in current forestry fertiliser applications or foliar dressing in this area (Davis et al. 2010). It is possible that this could be a source of elevated zinc in these streams.

Recent studies from Europe have identified zinc and copper as the most important stressor of river macroinvertebrates compared to other water quality, and habitat stressors (Johnson et al., 2025). Indicators of macroinvertebrate community health at Ararimu Tributary were also moderate to poor (B-C band) despite habitat values being maintained (Stream Ecological Valuation = ‘Good’) (Storey and Surrey, 2025).

Additional monitoring could investigate potential sources of zinc and transport pathways and further characterise the risk of possible acute toxicity effects occurring within this area. Additional monitoring undertaken within these watersheds should target seasonal peaks during winter.

7.2 Trends

Trends were analysed over the seven year period from 1 July 2017 to 30 June 2024 for both soluble and total metals. Trends were not analysed for adjusted bioavailable concentrations due to the uncertainty added when adjusting the values for water chemistry. Trends were also not assessed for the most impacted urban site, Newmarket Stream, due to insufficient data.

Overall, 50% of monitoring sites were found to be likely to very likely degrading for soluble copper with 29% of sites likely to very likely improving. There were no obvious differences among dominant land cover classes or clear spatial patterns in trends (Figure 7-8 and Figure 7-9).

There was generally lower confidence in trend direction for zinc concentrations, particularly for soluble zinc. There were no obvious differences among dominant land cover classes or clear spatial patterns in trends (Figure 7-8). A quarter of urban sites were very likely improving, and a quarter of urban sites were likely to very likely degrading.

The trend direction was generally consistent (the same or varying by one trend category e.g. likely to very likely degrading) between total and soluble metals at each site. There was often higher confidence in trends for total concentrations.

For the three sites with the greatest urban development occurring withing the upstream catchments (Ōtara Creek (East), Vaughan Stream, and Lucas Creek), trends in soluble and total copper were variable. Ōtara Creek (East) was found to be very likely degrading while Vaughan Stream and Lucas Creek were found to be likely, to very likely improving respectively. Across these three sites, trends in soluble and total zinc ranged from low confidence to very likely improving.

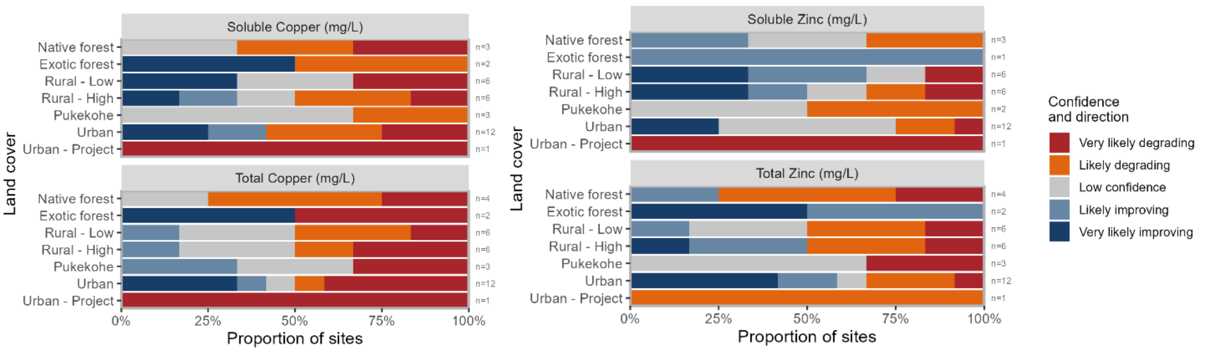


Figure 7-8: The proportion of river sites in each trend category (1 July 2017 - 30 June 2024) for copper and zinc grouped by dominant land cover class.

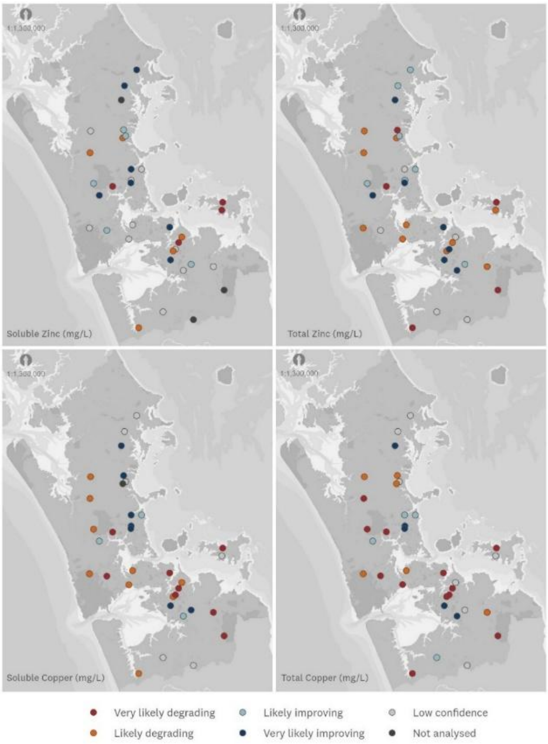


Figure 7-9: Regional map of site locations displaying trend direction for soluble and total copper and zinc (1 July 2017 - 30 June 2024). Refer to Figure 2-1 for site names.

7.2.1 Trend magnitude + state

The annual Sen slope for soluble copper and zinc (unadjusted) was compared to the five-year median chronic bioavailability adjusted state bands to provide information on the rate of change relative to state. The rate of change cannot be conveyed in terms of changes in state bands as bioavailable adjusted concentrations are lower than unadjusted values (see 7.1.1).

At many sites, the rate of change was negligible or could not be estimated because of a high proportion of values below the detection limit. In such cases, reporting a likely degrading trend serves as an early warning signal.

Most sites that were in band B for the median bioavailable state for copper toxicity had likely to very likely degrading trends in soluble copper (Figure 7-10). Trends were not assessed for the most impacted site (graded in band C, median concentration of 0.00062 mg/L), Newmarket Stream, due to insufficient data. The site with the second highest median bioavailable copper (0.0003 mg/L), Ōmaru Creek, was found to have very likely degrading trends in soluble copper at a rate of 0.00008 mg/L per annum. The greatest rate of degradation was at Pakuranga Stream at 0.00013 mg/L, currently graded in band B. While median concentrations of bioavailable copper remained in the A band (0.00018 mg/L) at Ōtara Creek (East), this site also exhibited a high rate of degradation in soluble copper at 0.00008 mg/L per annum, approximately 4.5 times greater than the rate of change within the adjacent tributary at Ōtara Creek (South) (0.000018 mg/L per annum). Among rural sites, the greatest rate of degradation was at Rangitōpuni River (0.00005 mg/L per annum). Conversely, soluble copper concentrations improved most notably at Ōteha River (0.0001 mg/L per annum), Puhinui Stream (0.00007 mg/L per annum), and Lucas Creek (0.00005 mg/L per annum).

Where median bioavailable zinc concentrations are currently very low (within band A), the rate of change in soluble zinc concentrations was also very low, or poorly estimated due to a high proportion of values below the limit of detection (Figure 7-11). The greatest rate of degradation was at Pakuranga Stream (Urban – Project) at 0.0007 mg/L per annum, currently graded in band B. The most impacted urban site (Ōmaru Creek – currently graded in band D) had the highest rate of improvement in soluble zinc at 0.0032 mg/L per annum, although wide confidence intervals indicate high variance in observations (Figure 7-11). Moderately impacted sites, Ōteha River and Puhinui Stream, were also found to have very likely improving trends in soluble zinc with a higher rate of change at 0.0013 and 0.0006 mg/L per annum respectively. For Puhinui Stream, this improving trend appears to have led to a change in the overall zinc state assessment from band D to band C, over rolling time periods (Appendix 10, Figure A10-5).

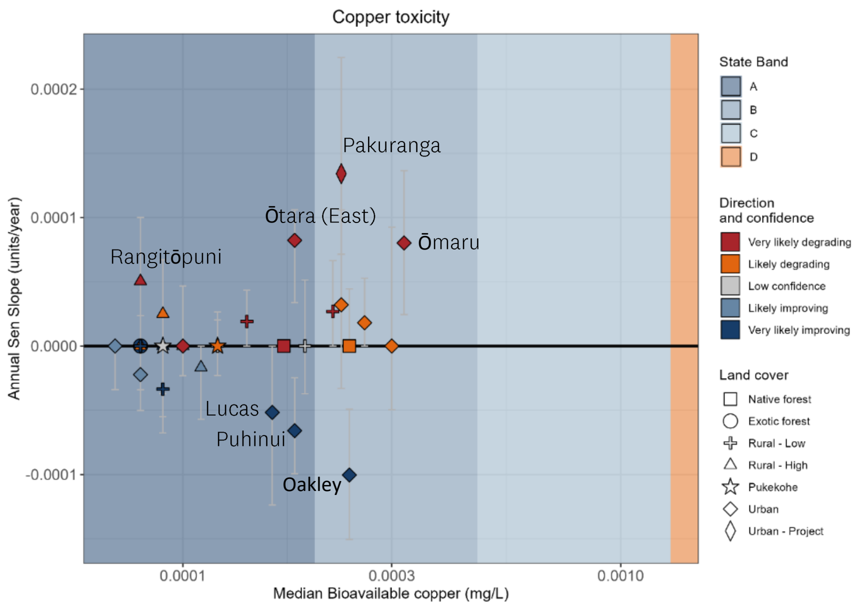


Figure 7-10: Median state (bioavailable copper) (July 2019 - June 2024) vs trend rate (Sen slope soluble copper (mg/L) – not adjusted) (July 2017 - June 2024). Error bars indicate 90% confidence intervals. Plots are coloured by the median NOF attribute band. Points are coloured by the confidence in trend direction.

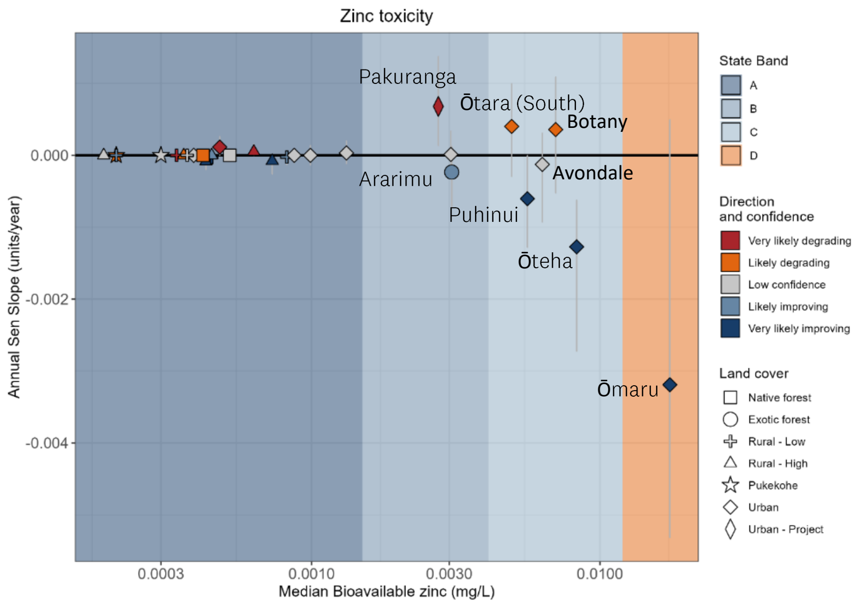


Figure 7-11: Median state (bioavailable zinc) (July 2019 - June 2024) vs trend rate (Sen slope soluble zinc (mg/L)). Error bars indicate 90% confidence intervals. Plots are coloured by the median NOF attribute band. Points are coloured by the confidence in trend direction.

7.3 Discussion

The current state assessment of copper and zinc toxicity differs from previous regional reports as no adjustment for bioavailability was previously undertaken (Ingley et al., 2023). In the last river water quality assessment of current state, all urban sites, nearly half of monitored rural sites, and one native forest site were found to be in band C for copper toxicity (Ingley et al., 2023). The updated analysis undertaken here demonstrates that the level of risk of chronic toxic effects of copper on aquatic instream life is less than we had previously identified[11]. This demonstrates the importance of accounting for metal bioavailability to instream life when considering risks of copper toxicity in streams. These results exemplify the importance of including dissolved organic carbon[12] in test suites to provide a more informed assessment of effects where there are likely risks of metal toxicity (e.g. pollution events, compliance monitoring, discharge consents). The risk of toxic effects of zinc on aquatic life assessed here, accounting for bioavailability, was found to be similar to our previous analysis[9], with several urban streams failing the proposed regional bottom line (Ingley et al., 2023). Recent studies from Europe have identified zinc and copper as the most important stressor of river macroinvertebrates (and/or zinc and copper are indicators of other correlated factors) compared to other water quality, and habitat stressors (Johnson et al., 2025). State of the Environment monitoring in Tāmaki Makaurau demonstrates that macroinvertebrate community health is the poorest in urban streams (Surrey and Storey, 2025) and chronic and/or acute metal contamination may be a contributing factor.

[11] A reassessment of the previously reported time period (to the end of 2022) with the latest bioavailability adjustments is provided in Appendix 10 showing no difference between the current state (end year 2024) and the 2022 state.
[12] As dissolved non-purgeable organic carbon (Gadd et al., 2024).

The most recent national assessment of copper and zinc contamination within New Zealand did undertake adjustment for bioavailability and compared observations to ANZG draft chronic guideline values (Gadd et al., 2023). The national assessment did not include grading against bands[13]. However similar patterns in the state of water quality were observed with higher concentrations in urban areas, and the highest percentage of exceedance of guideline values occurred in urban areas (Gadd et al., 2024). Additional regression analysis undertaken nationally demonstrated that dissolved zinc concentrations increased with increasing proportion of urban land cover and increasing impervious surface cover, but copper concentrations did not follow this same pattern (Gadd et al., 2024).

Overall, the current state assessment here identified the greatest risk of toxicity impacts to instream life from metal contamination within monitored streams was at the urban sites Newmarket Stream and Ōmaru Stream. Trends could not be assessed at Newmarket Stream as monitoring at this site commenced in 2021. Strong trends in metal concentrations with a high rate of change were observed at Ōmaru Creek however these were in opposite directions. Copper levels were found to be very likely degrading at Ōmaru Stream however zinc concentrations were very likely improving.

Overall, the trend assessment here did not identify clear patterns in trend direction for copper or zinc and changes over time appeared to be site specific. Previous national assessments of trends in copper and zinc contamination were assessed across a different time period, 2013 to 2022 which spanned the change in laboratory and analytical methods. The national assessment found that trends in dissolved copper were consistently degrading within the Auckland region (Gadd et al., 2023). Conversely, for the current trend period, 2017 to 2024, while a higher proportion of sites were found to be degrading than improving, there was more variability in trend direction. The previous national trend assessment identified a higher proportion of sites with improving dissolved zinc concentrations (Gadd et al., 2023) while this assessment found there was generally a lower confidence in trend direction for zinc.

Strong evidence of degradation was observed for both copper and zinc concentrations at Pakuranga Stream (Urban- Project). This site is influenced by a closed landfill discharge and also includes a small industrial area and larger residential areas within the wider catchment.

The current state for the three catchments monitored with the greatest rate of urban development, Vaughan Stream, Ōtara Creek (East) and Lucas Creek, remained in band A for copper toxicity. Vaughan Stream, and Lucas Stream also remained in band A for zinc toxicity. Ōtara Creek (East) remained in band B. High rates of change were observed in copper concentrations for two of these three catchments but in opposite directions, with very likely degrading trends observed at Ōtara Creek (East) and very likely improving trends observed at Lucas Creek. There was low confidence in trends in soluble zinc but likely to very likely improving trends in total zinc across these three catchments over 2017-2024. This trend period assessed does not encompass a pre-development baseline. Urban development started

[13] The regional framework applied in this report will require further revision to refine the attribute table when chronic guideline values are finalised through the ANZG process (Gadd et al., 2024).

within these catchments prior to 2017. The preceding trend period (1 July 2009 to 30 June 2017) was also reviewed for these catchments. This encompasses the predevelopment period for Vaughan Stream but not Ōtara Creek (East) and Lucas Creek (see section 3.1). However, within all three catchments, the trend direction was generally consistent between time periods, with the level of confidence in trends varying from low confidence to very likely improving across all forms of soluble and total metals. The exception to this was the very likely (and rapidly) degrading trends in copper identified at Ōtara Creek (East) in the most recent time period. Trends in copper in the preceding time period were likely improving within this catchment.

Very likely improving metal concentrations for both copper and zinc were observed within the urban Puhinui Stream and Ōteha Stream. At Puhinui Stream, this appears to be associated with lower concentrations observed from late 2022/early 2023 which is also demonstrated by fewer exceedances of the Tier 1 acute guideline values for copper and zinc.

8 Sediment and water clarity

Sediment enters rivers through natural catchment processes like erosion, particle transport, and landslides, as well as human activities such as urban runoff, industrial discharges, and land-use change. Storm events can trigger soil erosion and landslides, leading to pulses of sediment and increases in suspended sediment and turbidity.

Instream sediment-related impacts on aquatic life are typically measured using suspended sediment concentration and turbidity. While the two are often correlated, they are not interchangeable; their relationship can vary between watersheds, so separate assessments are recommended (Henley et al., 2000). Turbidity refers to reduced optical clarity due to particles that scatter or absorb light (Matos et al., 2024).

Excess delivery of fine sediments (mud, clay, silt) is a major driver of aquatic biodiversity loss. Suspended fine sediment reduces light availability, clogs fish gills and macroinvertebrate feeding structures, and impairs photosynthesis in aquatic plants and algae (McKenzie, 2000; Matos et al., 2024). Deposited sediments smother instream habitats and disrupt food webs by reducing primary production (algae and macrophytes) and food availability across trophic levels (Henley et al., 2000).

Turbidity in Auckland streams is measured in two complementary ways: in the field (FNU) and in the laboratory (NTU). Field measurements (FNU) are often more representative of true, in-situ conditions at the time of sampling. However, we have a longer historical record of laboratory analysis, compared to field measurements.

Turbidity provides a prediction of visual clarity, which is a water quality attribute for rivers under the National Policy Statement for Freshwater Management (NPS-FM, 2020). As visual clarity (VC) was not routinely monitored by Auckland Council until 2022, we converted measured turbidity values to VC for assessing this attribute. This conversion used site-specific models as outlined in section 2.3. Current state assessment is based on the rolling median calculated from the turbidity time series based on this site-specific regression model, while the previous state assessment of visual clarity was based on a different national regression model, and turbidity-NTU instead of FNU was used for the conversion.

8.1.2. Visual clarity, turbidity and total suspended solids distribution

Stream water clarity, turbidity, and total suspended solids concentrations vary both between and within sites as illustrated in box plots (Appendix 5) and through the online Data Explorer. In this section, data from sites within each biophysical and land cover class were pooled to identify broader patterns across the Auckland region.

Sites were grouped according to the River Environment Classification (REC), which included the catchment spatial attributes climate, source of flow, and underlying geology (see full description of REC classes in section 2.1.2). Most sites fell within the warm wet, low elevation, soft sedimentary (WW/L/SS) class, and the variability of VC, turbidity, and TSS metrics across sites within this class was the highest (Figure 8-1). Median concentrations were similar across all groups, except for the warm wet, low elevation, volcanic acidic (WW/L/VA) class, where median VC was higher and turbidity and TSS were lower compared to other REC classes. TSS showed a bimodal distribution within the WW/L/VA class, indicating differences between sites in this class.

There were also differences in VC, turbidity, and TSS distributions across land cover types (Figure 8-2). The greatest variation across sites for all metrics was observed within the rural-high land cover class, and there was a higher frequency of occasional extreme values in streams within this cover type. VC was generally higher, while turbidity and TSS were lower, at the Pukekohe sites – likely due to the VA underlying geology in this area. In contrast, the median VC in Pakuranga Stream (Urban-Project) was the lowest among all land cover types, while turbidity and TSS were the highest. Native forest sites showed a multimodal distribution, indicating differences in underlying conditions among the four sites within this land cover class.

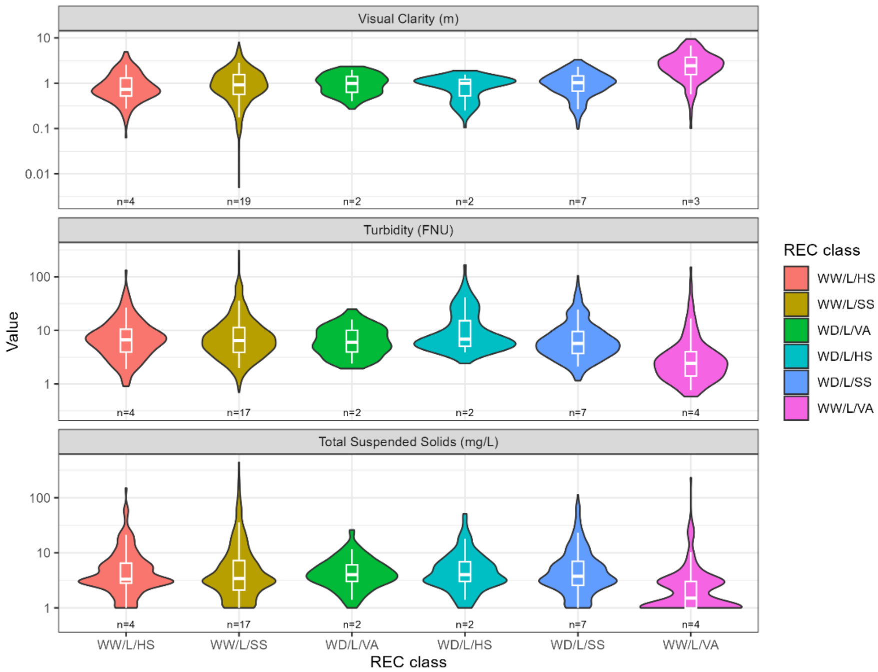


Figure 8-1: Distribution of visual clarity, turbidity and TSS across sites within each REC class (01 July 2019 - 30 June 2024). The white box indicates the interquartile range, the central line is the median, and the whiskers are the 5th and 95th quantiles. n indicates the number of sites represented in each violin.

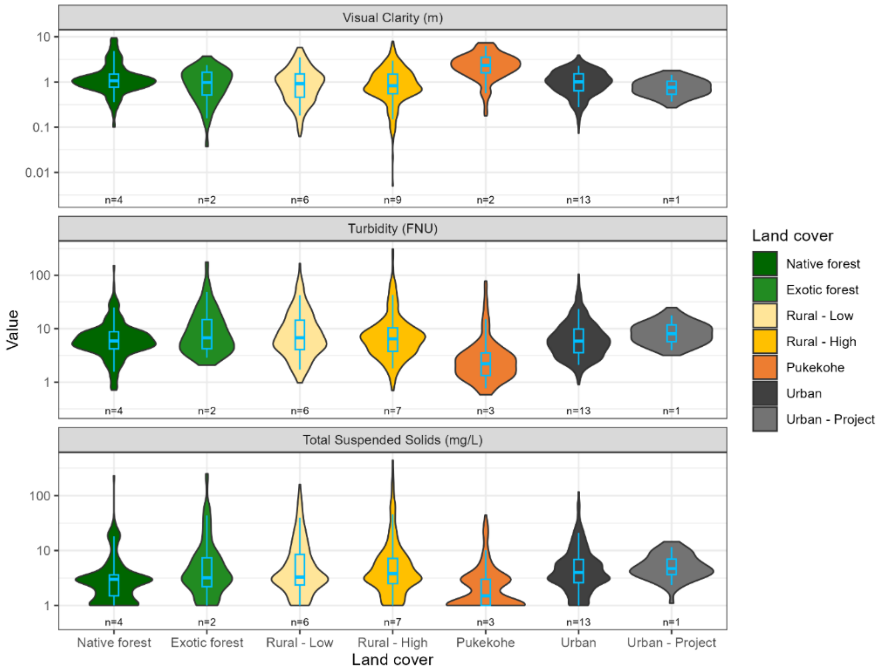


Figure 8-2: Distribution of visual clarity, turbidity and TSS across sites within each dominant land cover class (01 July 2019 - 30 June 2024). The blue box indicates the interquartile range, the central line is the median, and the whiskers are the 5th and 95th quantiles. n indicates the number of sites represented in each violin.

8.1.3. Seasonality

Seasonality was evaluated for turbidity measured in NTU and FNU, as well as for total suspended solids (TSS). Visual clarity was excluded from the seasonality analysis due to the absence of long-term measured data, with only one site (Hōteo NIWA) possessing a sufficient record to support statistical assessment. For all other sites, visual clarity was converted from turbidity (FNU); therefore, seasonal patterns in turbidity (FNU) are assumed to adequately reflect those of visual clarity.

There were more distinct seasonal patterns for turbidity (as measured in field and lab) than TSS. Seasonal patterns for both metrics were most common in rural and urban streams, with peaks occurring in August-September (Figure 8-3). Among rural streams, the highest peaks during the winter months were observed in Cascades Stream (Waiheke) and the Mahurangi River (Lower at Brown Rd). Among urban sites, the Vaughan, Lucas, and Ōteha streams showed the strongest seasonality. These streams are located near each other and have experienced significant urban development in their catchments over the past five years. Turbidity varied considerably during the winter months across most streams (Appendix 6), consistent with the high flows in winter (see section 4.3).

Interestingly, streams in the Pukekohe area showed higher turbidity peaks in winter, but little seasonal variation in TSS. This suggests that elevated turbidity at these sites may not be sediment related. The area is known for high nutrient levels (see section 6.1.1), which can promote algal blooms that may, in turn, increase turbidity.

Of the two exotic forest sites, the Mahurangi River (Right Branch) had higher turbidity and TSS during the winter months, while the Ararimu Tributary showed the opposite pattern, with higher median values during the summer season. This is consistent with elevated phosphorus concentrations measured during the summer season for Ararimu Tributary (section 6.1.2)

Native forest sites showed no seasonal variation in TSS, and only a slight increase in turbidity during winter. However, these seasonal differences were minor compared to those observed in other land cover classes. This likely reflects the resilience of streams in native forest catchments to winter rainfall and high flows, owing to intact vegetation, stable soils, better infiltration, low human disturbance, and less deposited sediment available for resuspension during flushing flows.

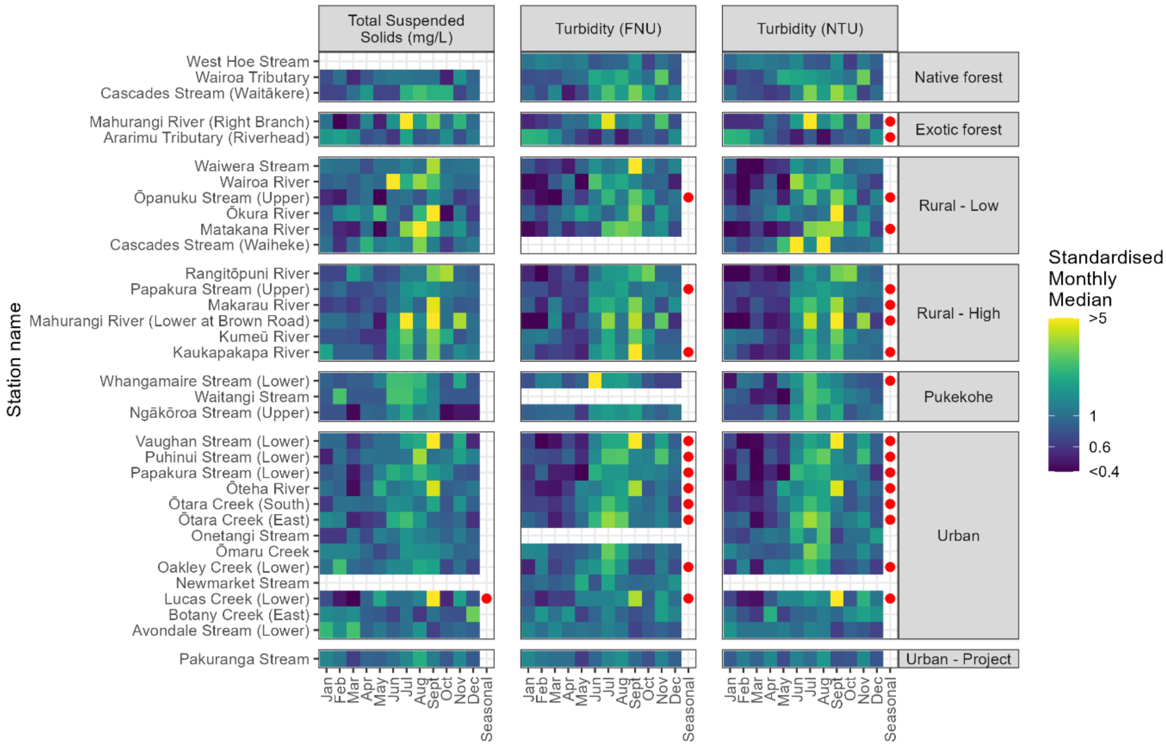


Figure 8-3: Water quality monthly medians standardised by overall median for total suspended solids, turbidity (FNU) and turbidity (NTU) at each site (01 July 2019 - 30 June 2024). Red dots indicate a statistically significant seasonal response of the site/variable combination based on the *p*-value from the Kruskall-Wallis test with α=0.05. Results are only reported for site/variable combinations where each season had at least four observations.

8.1.1. Visual clarity within an NPS-FM context

Median visual clarity varied by an order of magnitude among streams and across different land cover types, with the lowest median values observed in the Ōkura River (Rural-High) and the highest in the Cascades Stream (Native forest) (Appendix 5).

Within-site variability was noticeable, particularly at all rural (low) sites. Lower river VC generally coincided with high-flow events (Lorrey et al., 2025), which reduces visual clarity due to elevated sediment loads.

Most sites were classified in band A and B according to the NOF indicating minimal to moderate impact of suspended sediment on instream biota for a particular suspended sediment class. Only two sites were classified in band C and three sites were attributed to band D, failing the national bottom line (Figure 8-4). According to a previous state assessment, for an earlier time period, only two sites failed the national bottom line (Ingley, et al., 2023)

Sites within the Rural-High land cover class were in the poorest condition, with only one site classified in band A (Figure 8-4). Papakura Stream (Upper) was classified in band D (below the national bottom line) and has remained there for the past seven years (Appendix 10), indicating persistent high levels of suspended sediment impacting aquatic life. Kumeu River is currently classified in band C but was previously in band D within the past five years (Appendix 10). However, according to the previous state assessment, that was based on turbidity-NTU to visual clarity conversion, those two sites did not fail the national bottom line (Ingley et al., 2023)[14].

In the Rural-Low land cover class, only the Ōkura River was classified in band D, and it has remained in this band for the past seven years (Appendix 10), which is consistent with the previous state assessment (Ingley et al., 2023). The Wairoa River moved from band A to band B in the most recent years assessment. The remaining streams in this category were in band A, indicating minimal impact of visual clarity on aquatic biota.

Among urban sites, 75% were classified in band A, with only one stream assigned to each of the other bands. Avondale Stream was in the poorest condition, having shifted from band C to band D in the past two years[14], suggesting recent changes in its upstream catchment. All remaining sites within other land cover types (Native forest, Exotic forest, Pukekohe, and Urban-Project) were in good condition in terms of visual clarity, being classified in bands A or B.

[14] A reassessment of the previously reported time period (to the end of 2022) following the updated conversion relationships is provided in Appendix 10.

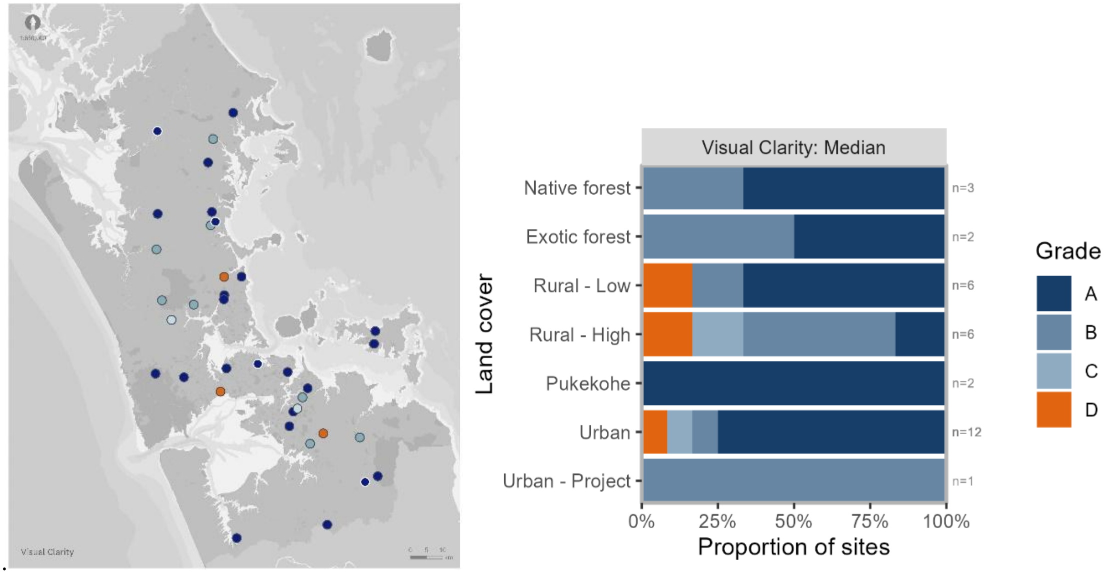


Figure 8-4: Left: Map showing spatial distribution of sites colour coded by the band metric for visual clarity attribute. Right: the proportion of river sites in each attribute band metric for the current state for visual clarity grouped by dominant land cover (01 July 2019 - 30 June 2024).

8.1 Trends

Trends in visual clarity, turbidity (FNU) and TSS were analysed over the seven-year period from 1 July 2017 to 30 June 2024. For all sites except the Hōteo River (NIWA), visual clarity was calculated from turbidity (FNU), resulting in nearly identical trends for the two parameters. Therefore, while results are presented for both, the discussion will focus on turbidity (FNU).

For all three parameters there were more sites where trends were degrading than improving within the streams classed as rural, native forest and the Pakuranga Stream (Urban – Project) (Figure 8-5). Of the two exotic forest sites, one site (Mahurangi River Right Branch) was likely improving in visual clarity, turbidity and TSS. However, at the other exotic forest site (Ararimu Tributary) TSS was very likely degrading, while there was low confidence in trend direction for VC/turbidity (Appendix 11). All sites in Pukekohe area had either improving trends or there was low confidence in the trend direction in all three parameters (Figure 8-5).

The trend direction was generally consistent between the three parameters with some minor deviations (Figure 8-5). For example, three native forest streams (Nukumea, West Hoe and Cascades) were very likely or likely degrading and Wairoa tributary was likely improving in terms of TSS. For turbidity and visual clarity Nukumea and Cascades were likely degrading, while West Hoe and Wairoa had indeterminate trends.

Of the three sites with the greatest change in urban land cover over the trend period, Ōtara Creek (East) showed indeterminate trends, Lucas Creek (Lower) was likely improving, and Vaughan Stream showed the strongest probability of improvement across all three parameters.

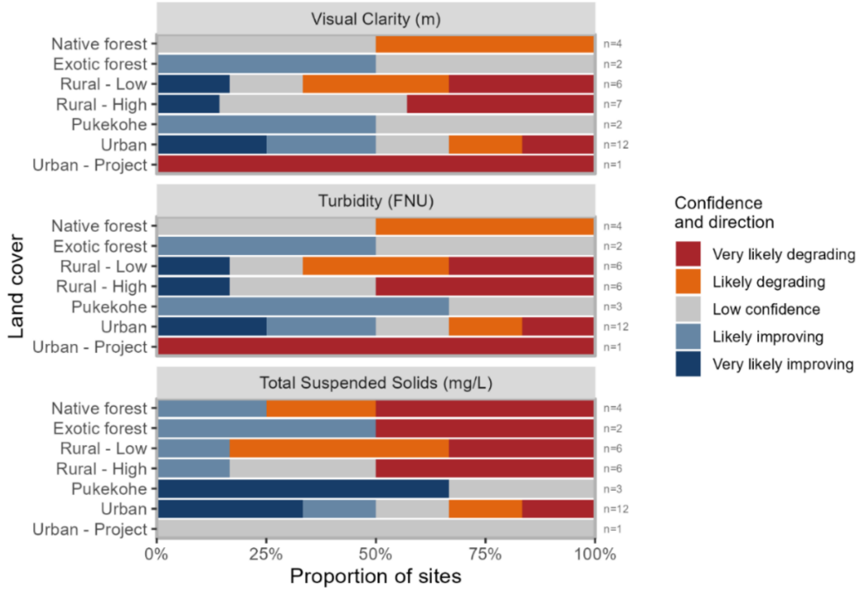


Figure 8-5: The proportion of river sites in each trend category (01 July 2017 - 30 June 2024) for visual clarity (m), turbidity (FNU) and total suspended solids (mg/l) grouped by dominant land cover class. Sites with interim grades are included in this plot.

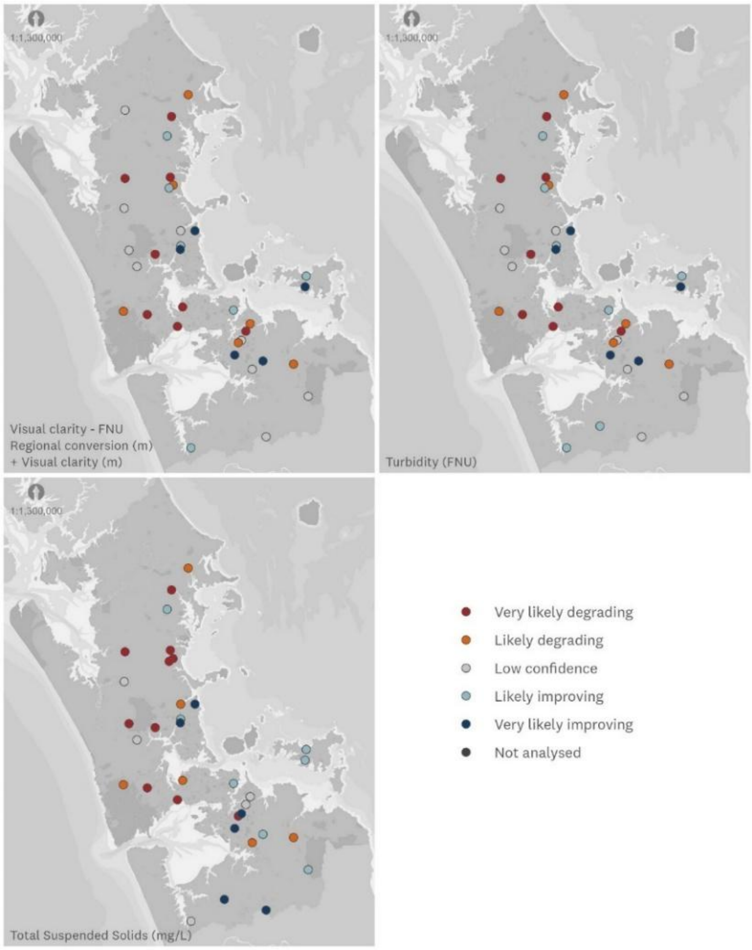


Figure 8-6: Regional map of site locations displaying trend direction for visual clarity, turbidity (FNU) and total suspended solids (1 July 2017 - 30 June 2024). Refer to Figure 2-1 for site names.

8.2.1. Trend magnitude + state

To gain better insight into how the magnitude and direction of trends relate to the current state, the annual Sen slope was compared to the five-year median value for each parameter (Figure 8-7; Figure 8-8). There were many sites where there were measurable increases or decreases in VC or turbidity.

As median VC increased across sites there was an increasing rate of change, either positive or negative. The two sites with the highest clarity showed the highest rates of change – but in opposite directions. For Cascades Stream (Waitākere), in the native forest land cover class, VC was likely degrading at 0.1 m per annum, while in Waitangi Stream, in the Pukekohe area, VC was likely improving at 0.066 m per annum (Figure 8-7). Both streams showed high uncertainty in the rate of change, as indicated by wide confidence intervals. VC in Ōpanuku Stream, Makarau River, and Waiwera Stream in rural areas was very likely degrading at relatively high rates (0.083-0.095 per annum), potentially crossing the A/B band threshold in 4-5 years assuming the same rate of change. However, the wide confidence intervals indicate high variability in the data, and the estimated time could range from 1 year to potentially never crossing the A/B band threshold. In Puhinui Stream (Lower) VC was very likely improving, with a rate of change of 0.06 m per annum. The current state of Avondale Stream, in terms of visual clarity, is in band D and continues to degrade. Papakura Stream (Upper), which was also in band D, was very likely improving, with an annual rate of change of 0.02 m. At this rate, it could cross the D/C threshold within three years. In contrast, visual clarity trends in the Ōkura River – classified in the worst condition – showed no significant improvement or decline. Urban-Project site is currently at the lower threshold of band B and continues to degrade at a rate of 0.03 m per annum. This could translate to failing the B/C threshold next year and reach band D within approximately five years.

For turbidity (NTU), the highest rates of change were observed at sites with median turbidity values around 6-7 NTU (Figure 8-8). In the urban Vaughan Stream and the exotic forest Mahurangi River (Right Branch) turbidity was very likely improving, with rates of change of 0.63 NTU per annum for each site. In the Ōkura River, with the highest median turbidity (15.5 NTU), turbidity was likely degrading at a rate of 0.53 NTU per annum – the highest rate of degradation. Avondale Stream, with the highest median turbidity among urban streams (11.3 NTU), also had likely degrading trends in turbidity, whereas most other urban sites were likely or very likely improving. The majority of rural (low) sites were very likely degrading at varying rates (0.36-0.53 NTU per annum), with the exception of Cascades Stream (Waiheke), which was very likely improving at 0.55 NTU per annum. Notably, at the Pukekohe area sites, median turbidity was the lowest, and very likely improving, albeit at relatively slow rates (0.07-0.25 NTU per annum).

Streams with the highest median TSS values generally displayed the highest rates of change (Figure 8-8). Consistent with the findings for turbidity, median TSS was highest in Ōkura River (8.65 mg/L) and very likely degrading at 0.47 mg/L per annum. Similarly, in Avondale Stream median TSS was the second highest of all sites (6.45 mg/L) and was very likely degrading at

0.54 mg/L per annum. Ōtara (East) creek had a similarly high median TSS (6.05 mg/L) however TSS was very likely improving at a 0.54 mg/L per annum. Three urban streams (Vaughan, Ōteha and Puhinui) were very likely improving with the rate of change ranging from 0.11 to 0.3 mg/L per annum. Rangitōpuni River in the rural (high) land cover category was likely improving, while Papakura (Upper) with the same land cover was very likely degrading at 0.28 mg/L per annum. As already stated, for many of the remaining sites, Sen slope was below detection limit of the trend rate.

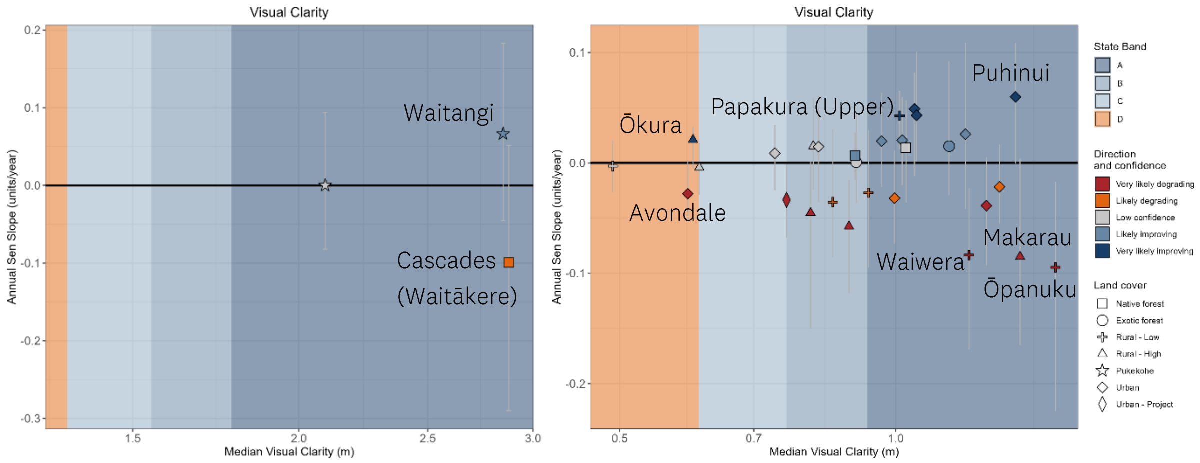


Figure 8-7: Median state (July 2019-June 2024) vs trend rate (visual clarity converted from FNU) (July 2017 - June 2024). Left – Suspended sediment class 1, Right – Suspended sediment class 2. Note differences in Y and X axes. Error bars indicate 90% confidence intervals. Plots are coloured by the median NOF attribute band. Points are coloured by the confidence in trend direction.

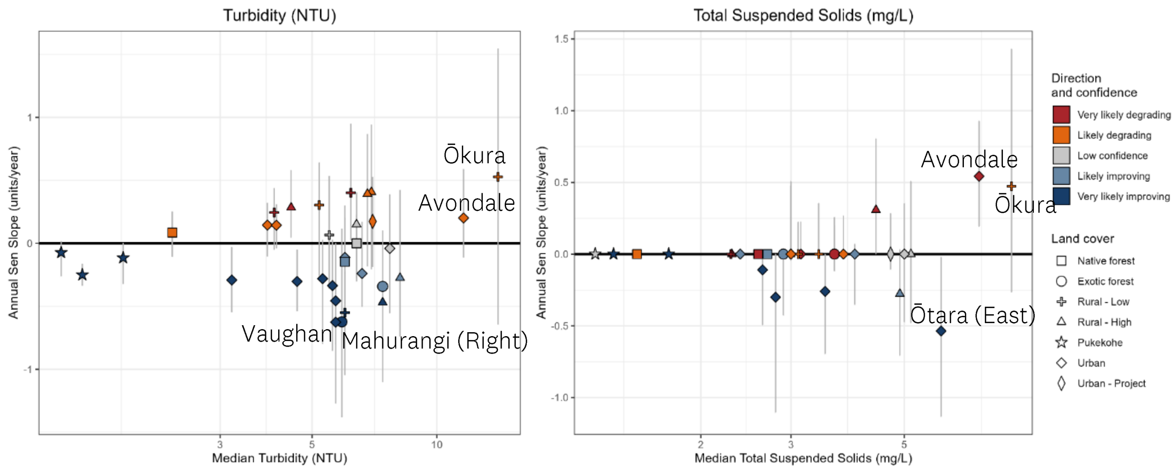


Figure 8-8: Median state (July 2019-June 2024) VS trend rate (Sen slope) (July 2017 - June 2024) for turbidity (NTU) and total suspended solids (mg/L). Error bars indicate 90% confidence intervals. Points are coloured by the confidence in trend direction.

8.2 Discussion

Ōkura and Avondale streams are currently in the poorest condition (band D) for visual clarity (VC) and have high turbidity, with both sites showing ongoing degradation. Papakura (Upper) Stream is also classified in band D; however, trends suggest very likely improvement in VC. If this trend continues at the current rate, the stream may cross the D/C threshold as early as next year.

None of these three streams have experienced substantial land cover changes in their upstream catchments over the past five years (Section 3.1), suggesting that the observed degradation is likely driven by other factors.

At Ōkura, the monitoring site is located at the lower end of the catchment within a lens of mudstone (Tauranga Group), interbedded with gravel and peat. Geologically, tributaries from the northwest originate in the Northland Allochthon, primarily composed of undifferentiated Mangakahia complex mudstone. Mudstone is typically highly erodible. Northland Regional Council has shown a strong correlation between weak sedimentary rocks of the Northland Allochthon, as well as depositional landforms such as peat and lacustrine sediments of the Tauranga Group and elevated sediment-water quality metrics (turbidity, clarity, and total suspended sediment) (Rissman & Pearson, 2020).

The causes of continuous decline in turbidity, visual clarity and total suspended solids in Avondale stream are unknown. Increased activity at the golf course located immediately upstream of the sampling site may be a contributing factor; however, it is more likely that the observed changes result from a combination of factors and broader changes within the upstream catchment.

8.3.1. Effect of flow representation and flood events on sediment characteristics.

A few streams had limited water quality observations during high-flows (see section 4.2), as a result the effects of extreme flow events on water quality have not been fully captured in the state and trend assessment. For example, Ōtara (East) Stream showed the strongest improving trends in TSS, but this may not reflect the influence of storm events on TSS concentrations. By contrast, Lucas Creek and Vaughan Stream had better representation of flow extremes, including major recent events. This can be explained by the fact that sampling occurs only once a month. As a result, some sites may have been sampled shortly after a major rain event, while others were sampled before or well after such events. Additionally, the impact of storms on turbidity varies over time – typically causing short-term spikes in smaller urban streams (such as Ōtara Creek) and more prolonged effects in larger rural rivers (such as Hōteo River).

To assess the impact of the 2023 Auckland floods on river turbidity measured in the field, long-term records of the past 10 years were analysed for 31 sites. Median turbidity (FNU) was slightly higher in 2023-2024, and there was a greater spread compared to the 2013-2022 period ($p = 2.2e\text{-}16$). At the Hōteo River site – where long-term records of measured visual clarity are available – 2023 had the lowest annual median visual clarity in the past 24 years (Figure 8-9), reflecting the impact of major rainfall events that year, including the Auckland Anniversary storm, Cyclone Gabrielle, and the 9 May storm.

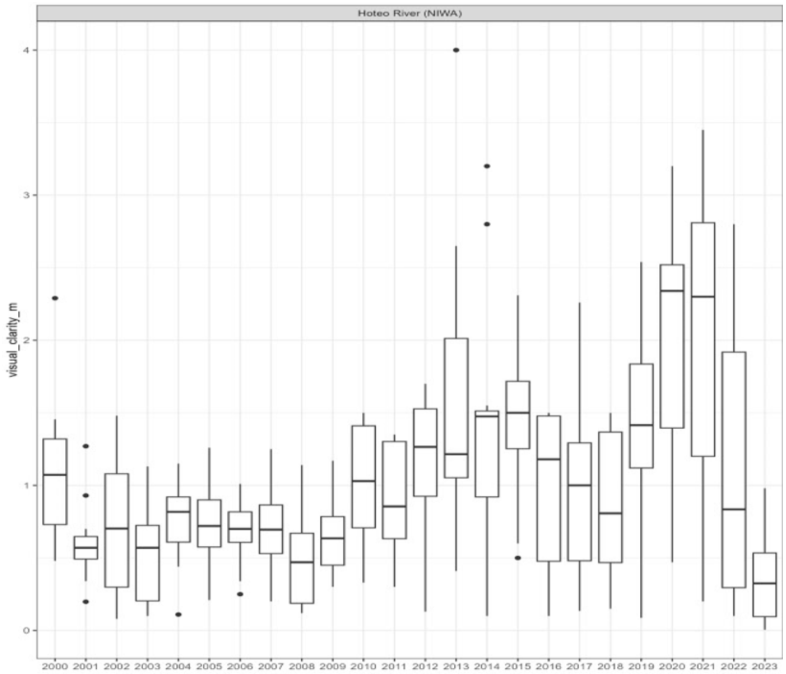


Figure 8-9: Annual distribution of visual clarity (m) measurements at the Hōteo river during the 2000-2023 period. The boxes show the data range between 25th and 75th quantiles. The centre line represents the median, and the whiskers show the 5th and the 95th percentile. Values beyond that range are plotted as outliers.

Urban Development at Long Bay – Vaughan Stream

The Vaughan Stream catchment has experienced the most significant land cover change of all monitored streams over the past five years. The Long Bay development within the lower Vaughan catchment was a master planned area and the structure plan became operative in 2011.

Water sensitive design was applied at the catchment scale through this plan to limit impervious areas, protect natural areas, and minimise the effects of stormwater discharges throughout this development (Ira, 2022; Lewis et al., 2015). Collectively, an additional 20% of the total catchment upstream of the monitoring site is now urban (2023/2024) compared to 2008/2009, and more than half of that change has occurred since 2018/2019.

Urban streams across the region, generally had poor water quality with multiple attributes in a degraded state. Vaughan Stream, in the lower part of the catchment and Long Bay area, remained in the best water quality state attribute band (A) across indicators of metal contamination, ammonia and nitrate toxicity, dissolved inorganic nitrogen, and water clarity despite the large-scale land use change. Some ecological functions were compromised at this site as indicated by poor dissolved oxygen levels (Young et al., 2025), and poor macroinvertebrate community health (Surrey and Storey, 2025). *E. coli* levels were also high (band E), indicative of faecal contamination.

Metal contamination is associated with increasing urban land cover and impervious surface area within the upstream catchment (Gadd, 2023). However, metal concentrations were found to be likely to very likely improving, or there was low confidence in the trend direction across all forms of copper and zinc across both the 2009-2017 and 2018-2024 time periods assessed, encompassing the development period. This suggests that the current state is at least being maintained at this site.

If not effectively mitigated, construction earthworks are expected to contribute to increased sediment inputs into nearby streams. Despite this, Vaughan Stream showed the strongest improving trends in turbidity over 2018-2024. Recent analyses of both event-based and discrete sediment yields found no significant changes in the sediment rating curve for Vaughan Stream (Tsyplenkov, 2025). Annual suspended sediment yields showed a consistent – though not statistically significant – decline of approximately 5% per year since 2012. This suggests that the impacts of earthworks may have been minimal, or alternatively, not captured during storm event sampling (Tsyplenkov, 2025).

Increasing urbanisation is also expected to influence instream water temperatures through heated stormwater runoff from impervious surfaces (Young et al., 2013). Vaughan Stream generally had cooler instream temperatures than most urban streams and exhibited very likely decreasing trends over 2018-2024 (Appendix 7). Additional analysis of continuous temperature measurements showed decreasing maximum daily temperatures at this site over the past 20 years, in contrast to other urban streams with increasing temperatures (Appendix 10).

These findings may reflect the effectiveness of water sensitive design approaches implemented in the Long Bay area. However, these conclusions remain preliminary, as land use change impacts may not have been fully captured during our routine monitoring or may still become apparent with additional development.

9 Human health risk – *E. coli*

In freshwater environments, the presence of *Escherichia coli* is indicative of faecal pollution and associated harmful bacteria or viruses that represent risks for human health. The NPS-FM includes an attribute for all rivers and lakes associated with the value of human contact with freshwater environments that is based on *E. coli* levels. The attribute bands are based on human health risk of bacterial *Campylobacter* infection from swimming in and ingesting water. This report does not assess grades for primary contact sites (Attribute Table 22 – Appendix 2B of the NPS-FM), or for rivers that are fourth-order or larger in relation to the Ministry for the Environment's national target for primary contact (Appendix 3 of the NPS-FM).

9.1 State

9.1.1 Distributions

E.coli levels vary both between and within sites as illustrated in box plots (Appendix 5) and through the online Data Explorer. In this section, data from sites within each broad land cover category were pooled to identify broader patterns across the Auckland region.

E. coli levels followed a general pattern of lower concentrations in native forest reference sites, increasing levels across rural sites, and highest in urban environments. The median *E. coli* level among native forest sites was approximately an order of magnitude lower than that of rural and urban streams (Figure 9-1). The median *E. coli* level for urban streams (800 CFU/100 mL) was more than double the median levels observed in rural (High and Low) streams (<300 CFU/100 mL). More extreme levels were also observed in some urban waterways with tails of the distribution spanning four orders of magnitude. For example, the most impacted site was Newmarket Stream. The median *E. coli* level at this site (at 10,500 CFU/100 mL) was more than an order of magnitude higher than the median of all other urban streams and the 95th percentile at this site (at 300,000 CFU/100 mL) was also an order of magnitude higher that the 95th percentile for other urban sites (see Appendix 5).

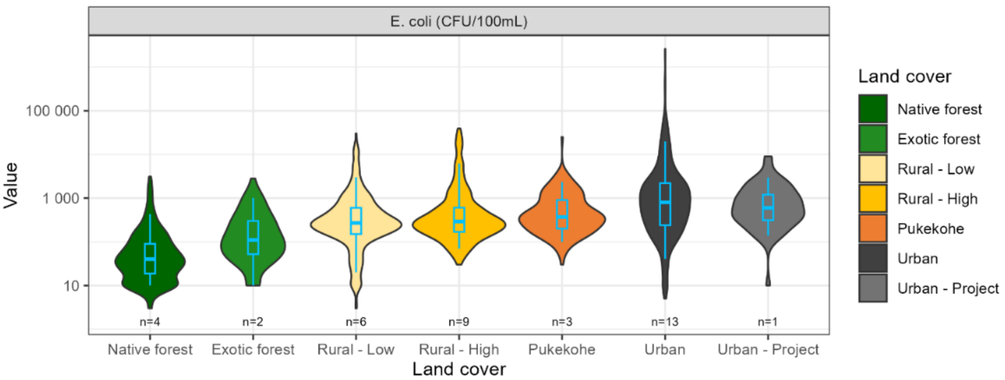


Figure 9-1: Violin plot – distribution of *Escherichia coli* levels aggregated among land cover classes (01 July 2019 - 30 June 2024). The blue box indicates the interquartile range, the central line is the

median, and the whiskers are the 5th and 95th quantiles. The log-scale axis may visually reduce apparent differences.

9.1.2 Seasonality

E. coli levels were highly variable over time both between and within sites with no clear seasonal pattern. Levels were sporadically much higher and lower than the median state within each site with large amplitude changes as indicated by changes from bright yellow to dark purple squares (Figure 9-2).

Only three sites were found to have significant seasonal patterns over the past five years. Even where seasonality was identified, there was variation in these patterns between sites.

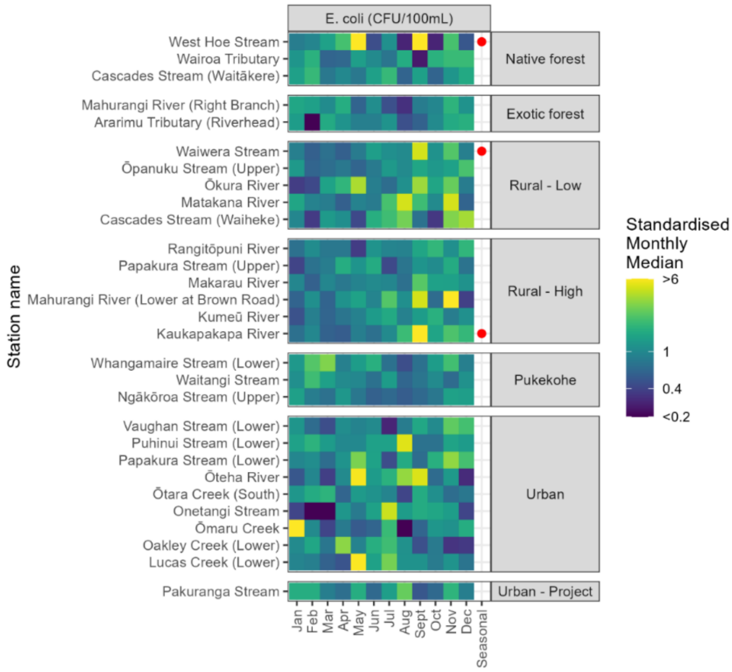


Figure 9-2: Water quality monthly medians standardised by overall median over the current state period (01 July 2019 - 30 June 2024) for *E. coli* water quality variables at each site. Red dots indicate significant seasonal patterns (p<0.05).

9.1.3 NOF – *E. coli*

The NPS-FM *E. coli* attribute is based on human health risk of bacterial *Campylobacter* infection from swimming in water. There are four underlying metrics with the poorest metric determining the overall attribute state: the median concentration (*E. coli*/100mL), 95th percentile of *E. coli* concentrations, the percentage of samples exceeding 260 *E. coli* per 100mL, and percentage of samples exceeding 540 *E. coli* per 100mL.. There is no national bottom line for the *E. coli* attribute.

Almost all urban and rural waterways were found to be impacted by faecal pollution (Figure 9-3). Collectively, 83% of sites (25 out of 30) across urban, and rural (High and Low) and 'Pukekohe' areas were graded in band E which indicates the predicted average infection risk to swimmers is >7%[15]. A further four sites were graded in band D with only one 'Rural- Low' site graded in band C. This site was Cascades Stream (Waiheke) where rural land uses are dominated by horticulture (vineyards).

Urban streams were graded in the poorest band (E) across all underlying metrics except for two sites, Onetangi Stream and Puhinui Stream (Figure 9-4). Onetangi Stream was graded in band A for the median metric and band D for the 95th percentile metric indicating that only occasional exceedances occur at this site. The upstream land cover for this site is largely native forest, with urban land covering 11% of the catchment area[16]. Puhinui Stream was graded in band D across all underlying metrics. Most rural (High and Low) streams were also graded in the poorest band (E) across all underlying metrics. Several sites were graded in bands C and D for the %>540 metric. Collectively these metrics indicate that *E. coli* is elevated most of the time in urban and rural streams, not only associated with short term wastewater overflow or storm events and runoff.

Forested catchments were graded in bands A and B except for Mahurangi River (Right Branch) which was graded in band D; this was driven by the median metric (Figure 9-4).

[15] Based on a random exposure on a random day. Actual risk will generally be less if a person does not swim during high flows (NPS-FM 2020 Table 9).
[16] This exceeds our 7% threshold for urban land cover and so is categorised as 'urban'.

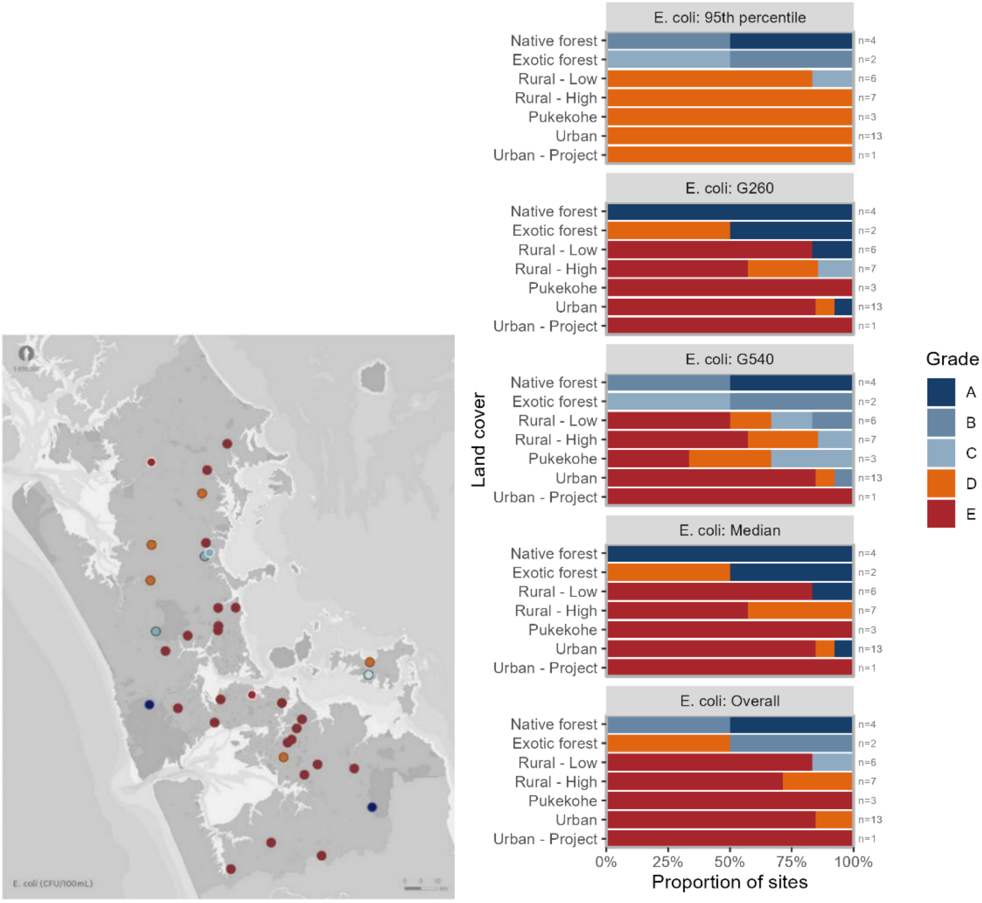


Figure 9-3: Left: Regional map of site locations displaying overall grade for the current state (01 July 2019- 30 June 2024) for *E. coli*. Refer to Figure 2-1 for site names. Right: The proportion of river sites in each attribute band metric for the current state (1 July 2019- 30 June 2024) for *E. coli* (Human Health) grouped by dominant land cover class. Sites with interim grades are included in this plot.

9.2 Trends

Trends were analysed over the seven-year period from 1 July 2017 to 30 June 2024. Trends were not assessed for the most impacted urban site, Newmarket Stream, as monitoring here only began in 2019.

There was no clear direction in trends across the region or across each land cover category. For example, for the native forest streams, one site (Nukumea Stream) was found to have very likely improving trends in *E. coli* while one site (Cascades Stream (Waitākere)) was found to have very likely degrading trends.

More rural stream sites showed degrading trends in *E. coli* than urban stream sites (Figure 9-4). Approximately 50% of rural (High and Low) stream sites had very likely degrading trends in *E. coli*. While 25% of urban stream sites were likely to very likely degrading only one urban site was very likely degrading (Ōtara Creek (East).

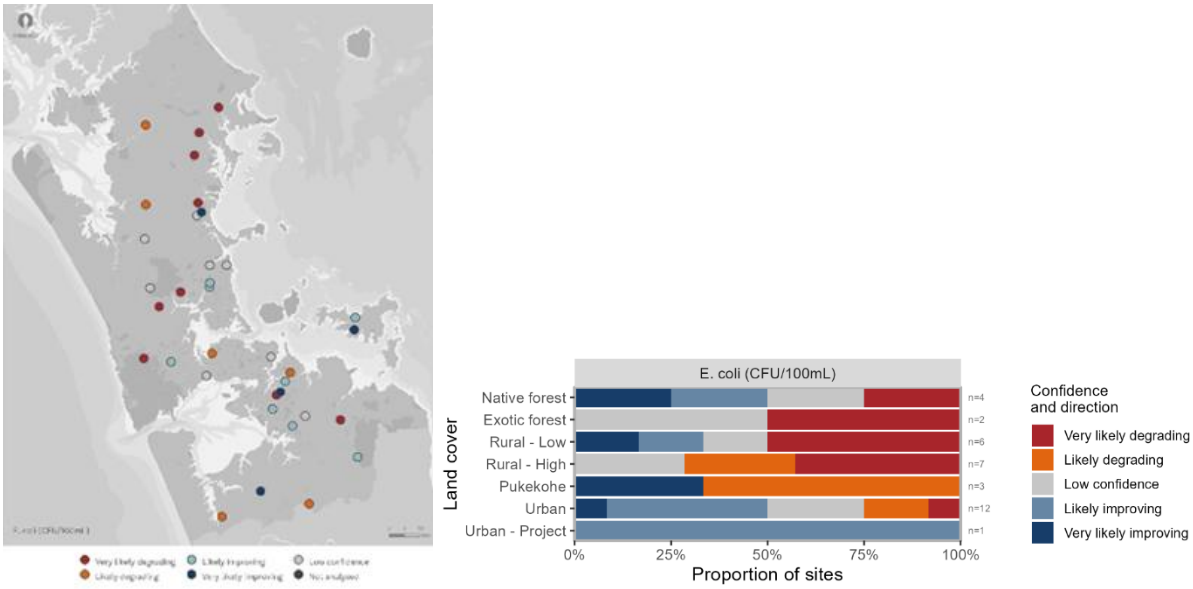


Figure 9-4: The proportion of river sites in each trend category (01 July 2017 - 30 June 2024) for *E. coli* grouped by dominant land cover class. Sites with interim grades are included in this plot. Regional map Regional map of site locations displaying trend direction for *E. coli* (01 July 2017 - 30 June 2024). Refer to Figure 2-1 for site names.

9.2.1 Trend magnitude + state

The annual Sen slope was compared to the five-year median state to provide information on the rate of change relative to state.

Half of the six most impacted urban sites[17], where median *E. coli* levels were >1000 CFU/100 mL had degrading trends, there was low confidence in in change at two sites, and one site (Papakura Lower) was likely improving (Figure 9-5). The highest estimated rate of degradation was at Botany Creek at 106 CFU/100 mL per annum. The second highest rate of change was at Ōtara Creek (South) where *E. coli* levels were very likely degrading at an estimated rate of 88 CFU/100 ml per annum. Conversely, the adjacent tributary, Ōtara Creek (East) was the only urban site that had a very likely improving trend in *E. coli*. Currently graded in band E (median concentration 700 CFU/100 ml), improvement at a rate of 64 CFU/100mL per annum for this site could translate to crossing the E/D median state threshold (<260 CFU/100 mL) within approximately seven years assuming a linear rate of change. However, the wide confidence interval for the trend Sen slope indicates high variability in the observations (translating to a range from three years (138 CFU/100mL) to indefinite (0 CFU /100 mL)) .

The greatest rate of improvement was identified at Whangamaire Creek at 85 CFU/100 ml per annum. With a current median state in the E band (800 CFU/100 mL) improvement at this rate could translate to crossing the D/E threshold within approximately 6.5 years assuming a linear rate of change (confidence intervals translating to 4 to 18 years). The trend at this site appears

[17] Not including Newmarket Stream, which was the most impacted site as trends were not assessed as monitoring at this site started in 2019.

to be driven by an abrupt decrease in *E. coli* levels from June 2023 onwards. The highest rate of degradation across rural (High and Low) sites was at Kumeu River (34 CFU/100 mL per annum). This appears to be associated with an increased frequency of high observations (>1200 CFU/100 mL) occurring over 2021-2023 (11 samples) compared to earlier in the period (only 2 samples >1200 CFU 100mL over 2017-2021).

The estimated rate of change at all native forest sites was low at ≤5 CFU/100 mL per annum (increasing or decreasing).

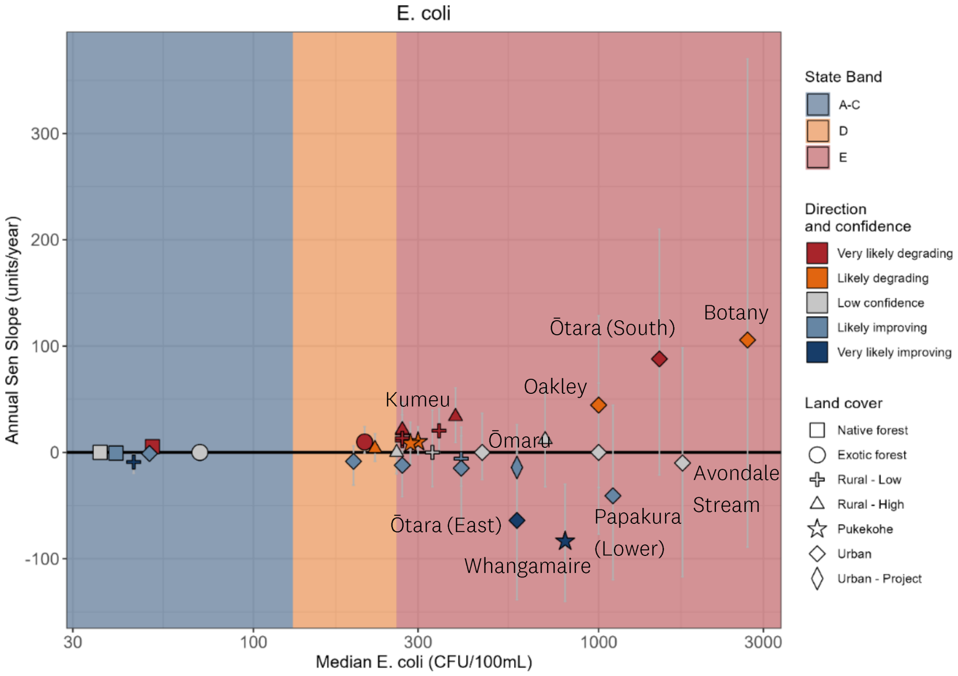


Figure 9-5: Median state (July 2019 - June 2024) *E. coli* vs trend rate (Sen slope) (July 2017 - June 2024). Error bars indicate 90% confidence intervals. Plots are coloured by the median NOF attribute band. Points are coloured by the confidence in trend direction.

9.3 Discussion

Widespread issues with high levels of *E. coli* across the region have been reported across previous state and trend reporting (Ingley, 2021; Ingley et al., 2023) and the results presented here demonstrate that this continues to be a key issue. The *E. coli* attribute is one of the worst scoring attributes under the NPS-FM 2020 across Tāmaki Makaurau, and nationally (Whitehead et al., 2022; LAWA, 2024).

Regional trend analysis indicates that more sites were found to be degrading in rural areas, however the greatest rate of degradation was occurring in some of the most impacted urban environments.

In urban environments, overflows of untreated wastewater can enter streams from the wastewater network. This can occur through engineered overflow points that are designed to overflow during wet weather conditions, or to relieve excess pressure in the network. Overflows can also occur from other locations in wet weather and dry weather conditions due to blockages. Wastewater can also enter the stormwater network (and then streams) via illegal cross-connections and pipe leakage via cracks or breaks. Microbial dynamics in streams are further influenced by remobilisation within stream channels (Wilkinson et al., 2011)

As of June 2024, the highest annual volume of discharge, and frequency of discharges from engineered overflow points upstream of all urban stream state of the environment monitoring sites was recorded at Te Aunanga / Oakley Creek, and Newmarket Stream, followed by Ōtara Creek (South) and Ōmaru Stream (Morphum, 2024). Occasional overflows were also recorded at Ōteha Stream and Ōtara Creek (East) (Morphum, 2024). As outlined above, Auckland Council monitoring showed that all six of these sites had high levels of *E. coli* faecal indicator bacteria (all graded in band E). Conversely, for two urban sites, Botany Creek and Avondale Stream, where Auckland Council monitoring demonstrated very high *E. coli* levels (and elevated ammonia levels, see section 6.2.1), no engineered overflow discharges were identified in the upstream catchment, suggesting that contamination in these areas may be more likely to be associated with dry weather discharges.

A very likely degrading trend, at a high rate of change, in *E. coli* levels (and ammonia, see section 6.2.1) was identified at the monitoring site in the southern Ōtara Creek catchment. Wastewater network capacity issues have been identified in the Ōtara catchment associated with overflows during heavy rainfall and further investigations are scheduled for late 2025 to inform infrastructure upgrades (Watercare, 2025). State of the environment monitoring demonstrates that *E. coli* levels were elevated at this site most of the time, therefore investigations targeting dry weather events may also support identification of other issues influencing water quality in this area.

In rural environments, faecal contamination in streams is most commonly associated with livestock, including direct stock access to streams and wash off from pastoral areas. Other potential sources also include dairy shed effluent, septic tanks and disposal fields, and waterfowl and feral animals. Evidence is currently lacking on the potential sources of faecal contamination influencing the state of water quality in monitored streams. Identifying whether the sources of *E. coli* are dominated by human inputs, livestock, or other animals such as ducks can help to inform management actions, additionally the source of faecal contamination influencing the risk of infection to humans (Devane et al., 2021).

While changes in *E. coli* levels were relatively small at sites monitored within forested catchments, it is unclear what may be driving increases and decreases in *E. coli* in these streams. Potential sources of *E. coli* in these areas include naturalised bacteria, feral animals such as pigs, possums, or avifauna.

Further investigations were initiated in 2024 by Auckland Council's Environmental Evaluation and Monitoring Unit to track the sources of *E. coli* on a regional scale. Microbial source tracking (MST) is a molecular technique that identifies host-specific bacterial groups presence in faecal material which allows us to identify the key sources of faecal contamination. Key objectives of this work include identifying sources of faecal contamination across different land cover classes, particularly in more rural and forested environments; identifying differences between dry weather and wet weather conditions; and also improving our understanding of resampling requirements to identify the range of potential sources.

10 Regional overview

Seven water quality indicators of river ecosystem health and one indicator of human health risk were assessed for the current state (01 July 2019 to 30 June 2024) in accordance with the National Objectives Framework[18] (NOF), and draft regional objectives[19]. Grading for each of the eight attributes was assessed in this report, which was provided within each chapter above including description of broad land cover and spatial patterns, and evaluation of the underlying attribute metrics. This section provides an overview of the overall current state across these eight attributes. This section also provides an overview of the confidence and direction of trends for these attributes.

10.1 Overview of state

There are widespread, regional issues with potential human health risks from faecal pollution, with over 70% of monitored streams in band E (Figure 10-1). There are also widespread, regional issues from nutrient enrichment that may impact ecosystem health and function. These pressures influence streams in most rural and urban environments. Streams with predominantly urban land cover in the upstream catchment generally had the poorest water quality (Figure 10-2). These findings are consistent with the general picture of water quality nationally (Whitehead et al., 2022).

In addition to nutrient enrichment and high *E. coli* levels, rural streams in Auckland generally exhibited lower visual clarity and a greater proportion of degrading trends than urban streams (Figure 10-2). Three sites across rural (High and Low) and urban land cover classes failed the national bottom line for visual clarity (Figure 10-2). All three sites were in REC categories with a warm-wet climate, two of these river sites had soft sediment beds, while one had a hard sediment bed (see section 8). Avondale Stream, an urban stream graded in band D for visual clarity, was degrading across all measures of visual clarity, turbidity and total suspended sediments. Higher visual clarity (and lower suspended sediment concentrations) was measured in streams underlain by volcanic-acidic geology (including a reference site in the Waitākere ranges and streams in the wider Pukekohe area).

Urban waterways with the highest *E. coli* levels also demonstrated elevated levels of ammonia and nitrite indicating that wastewater discharges are influencing these streams. Metal contamination, particularly zinc, was also common in urban streams (Figure 10-2). A risk of zinc toxicity (both chronic and acute) was identified at several urban streams, particularly in monitored streams discharging to the Tāmaki Estuary, where three sites failed the proposed regional bottom line for chronic toxicity. Some risks of ammonia toxicity, and copper toxicity are also emerging in urban streams, as several urban sites were graded close to these thresholds and there was evidence of further degradation. The sites with the highest

[18] As per the National Policy Statement for Freshwater Management 2020 (amended October 2024).
[19] For bioavailable copper and zinc.

concentrations, and degrading further, were again predominantly in streams discharging to the Tāmaki Estuary. Copper concentrations reached levels above acute water quality guidelines at one urban site, Newmarket Stream.

Rural streams in the wider Pukekohe area differed from other rural waterways, associated with different geology, land use pressures, and different interactions with groundwater. Nitrate nitrogen was at toxic concentrations in some streams in the southern Pukekohe area associated with nitrate contamination in the underlying shallow aquifers (Buckthought, 2025). This contamination pathway means that time lags between actions occurring on land and what is then observed in streams are expected, in the order of decades. Encouragingly, one of the three streams monitored in this area demonstrated evidence of improving trends in nitrate[20]. This has also translated to the site being graded in band B, above the National Bottom Line, an improvement on previous reporting where all sites failed the National Bottom Line (bands C and D). Observed concentrations are close to the B/C threshold and some state switching (i.e., between B and C bands) may be observed in the future.

One urban site was evaluated separately as a 'project' site due to an identified point source discharge influencing this stream. This is the only site that failed the national bottom line for ammonia toxicity while other issues observed as this site were common to other urban streams including high dissolved reactive phosphorus, zinc, and *E. coli* levels (all in the poorest bands) (Figure 10-2).

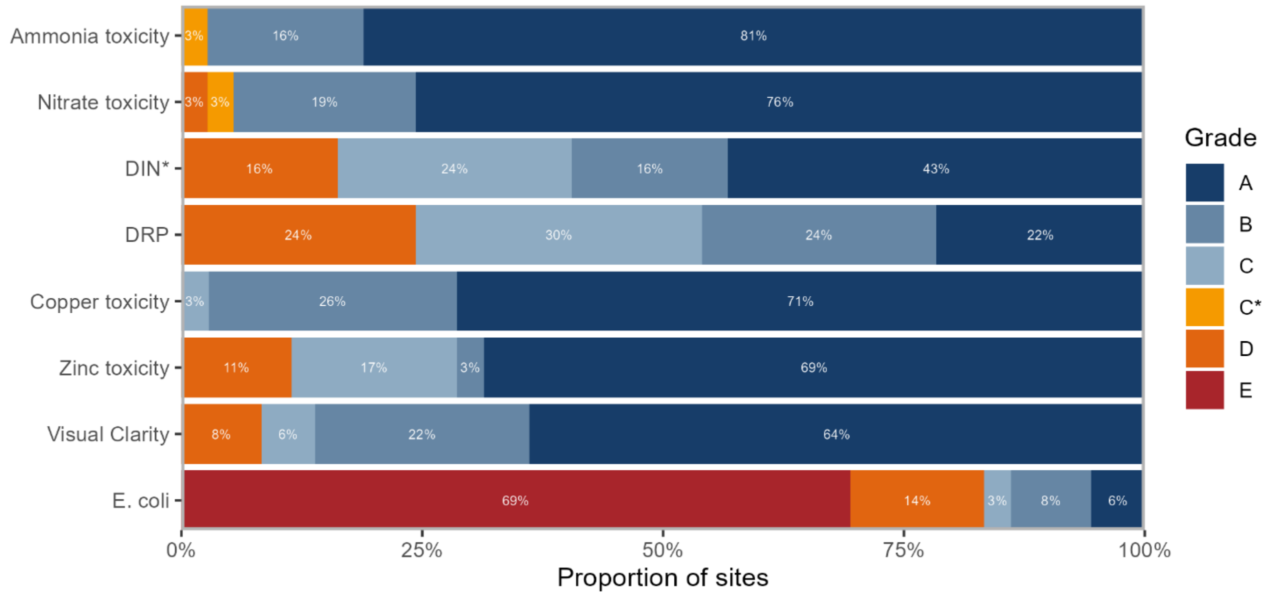


Figure 10-1: The proportion of river sites in each attribute band metric for the current state (1 July 2019 - 30 June 2024). Sites with interim grades are included in this plot. NOF state summary (n=36). C* where C is below the National Bottom Line, and C where C is above the National Bottom Line or no NBL.

[20] Assessed as total oxidised nitrogen (nitrate + nitrite) nitrogen as nitrite is typically a negligible proportion.

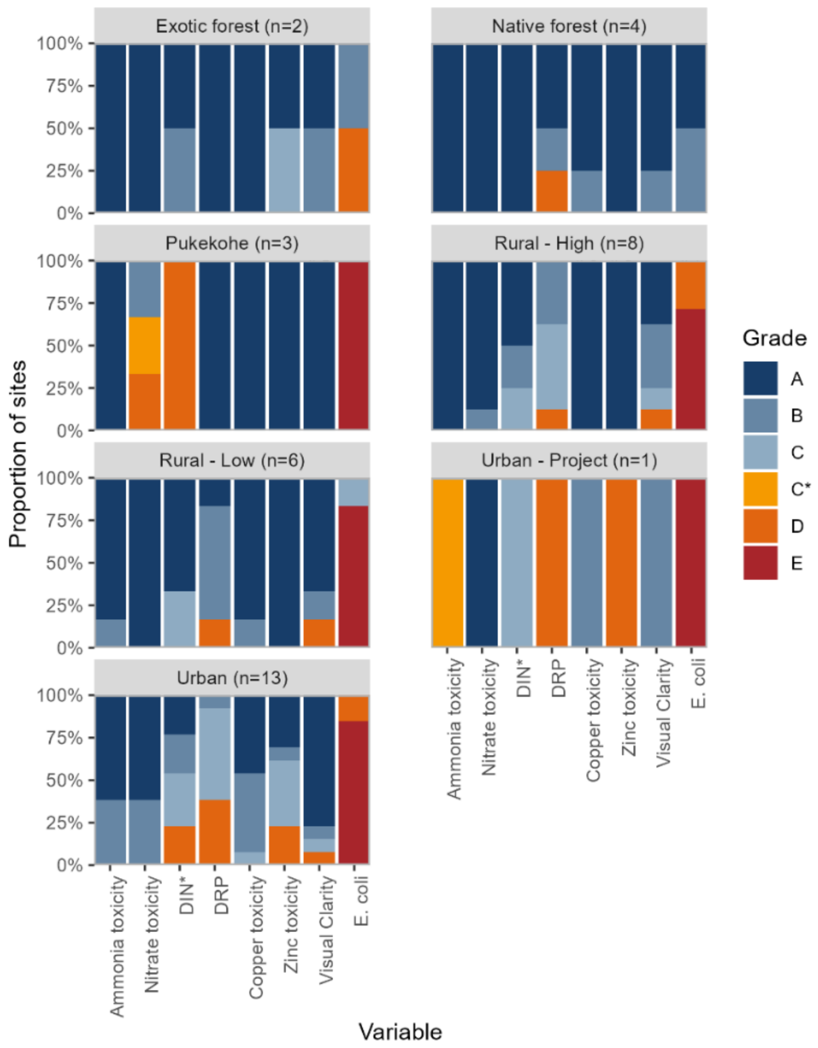


Figure 10-2: The proportion of river sites in each attribute band metric for the current state (1 July 2019 - 30 June 2024) grouped by dominant land cover class. Sites with interim grades are included in this plot. C* where C is below the National Bottom Line, and C where C is above the National Bottom Line or no NBL.

10.2 Overview of changes over time

Patterns in trends in water quality parameters over the past seven years (01 July 2017 to 30 June 2024) were much more variable than patterns in current state.

Some measures of water quality improved and some measures degraded within each site, and among different dominant land cover classes (Appendix 11). The parameters with the greatest proportion of sites with degrading trends were ammoniacal nitrogen and dissolved reactive phosphorus (Figure 10-3). The parameter with the greatest proportion of sites with improving trends was Total P; it is unclear why measures of dissolved and total phosphorus are demonstrating different patterns. The regional findings for ammoniacal nitrogen and dissolved reactive phosphorus are not consistent with the general picture of water quality trends nationally, though the findings for Total P are consistent. The different findings may be associated with differences in the time periods analysed (2011 to 2020 in the national study) or slight differences in trend assessment methods (Whitehead et al., 2022).

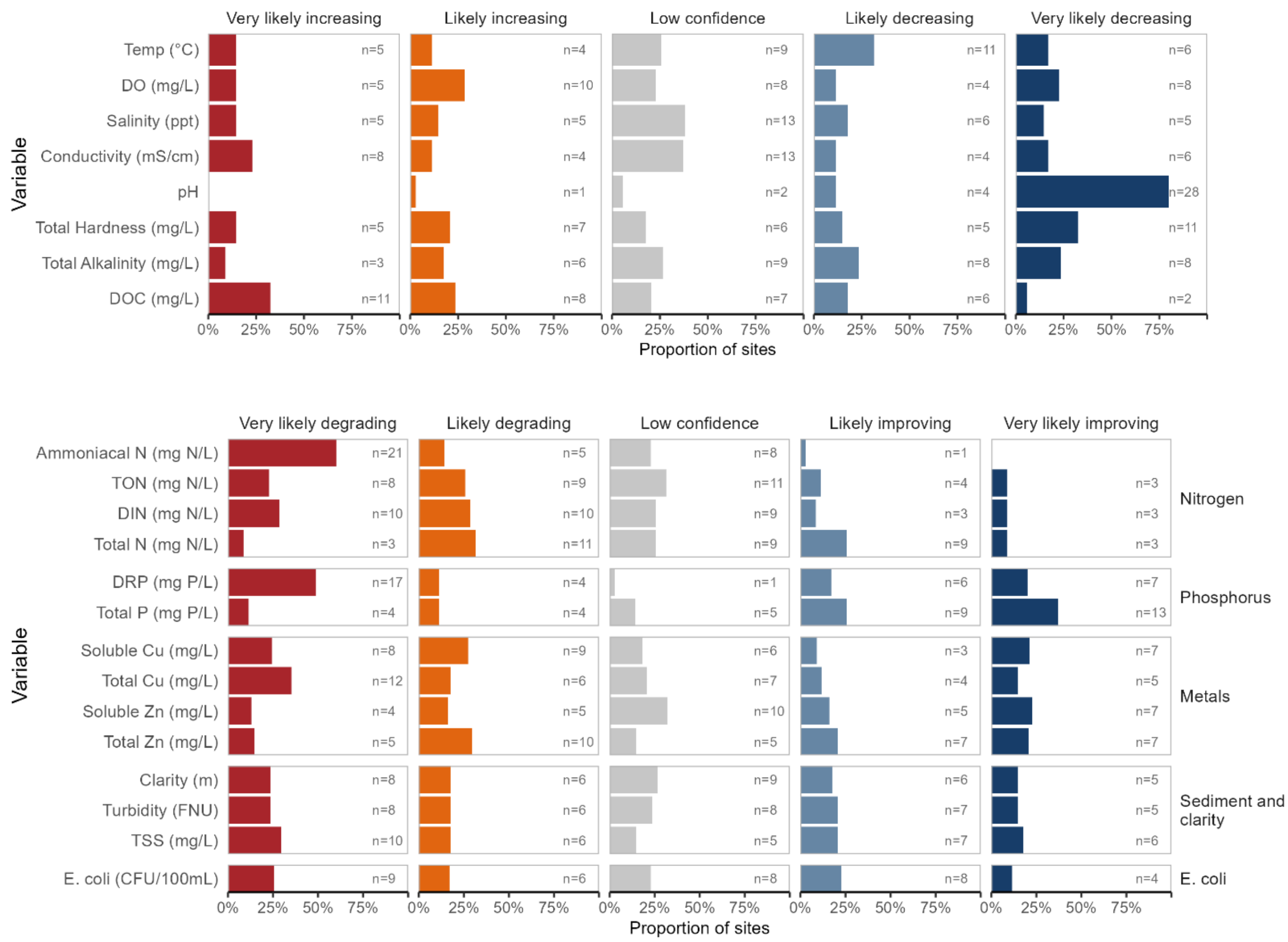


Figure 10-3: The proportion of river sites in each trend category for the period 1 July 2017 - 30 June 2024 (n=35, except for metals and sediment and clarity).

Figure 10-4 show the distribution of the confidence in trend direction across all sites grouped by the NOF attribute state grade. This provides an indication of where water quality is 'good and being maintained' through to 'bad and getting worse'.

Figure 10-4 demonstrates wide variation in the direction of trends across sites and different conditions of state for most water quality indicators. Some sites were improving and others degrading within each state assessment band for most parameters, but particularly for visual clarity and *E. coli*. For some parameters, the sites with poorer state bands tended to have a greater proportion of degrading trends, e.g. ammoniacal N for sites in band B, soluble copper for sites in band B, soluble zinc in band D, and DRP in bands C and D (Figure 10-4). Conversely there was a greater proportion of sites with improving trends were DRP levels were in band A (Figure 10-4). DRP was the only water quality parameter that showed a clearer pattern in trends among different dominant land cover classes with a greater proportion of degrading trends at urban sites and improving trends at forested sites (see section 6.2).

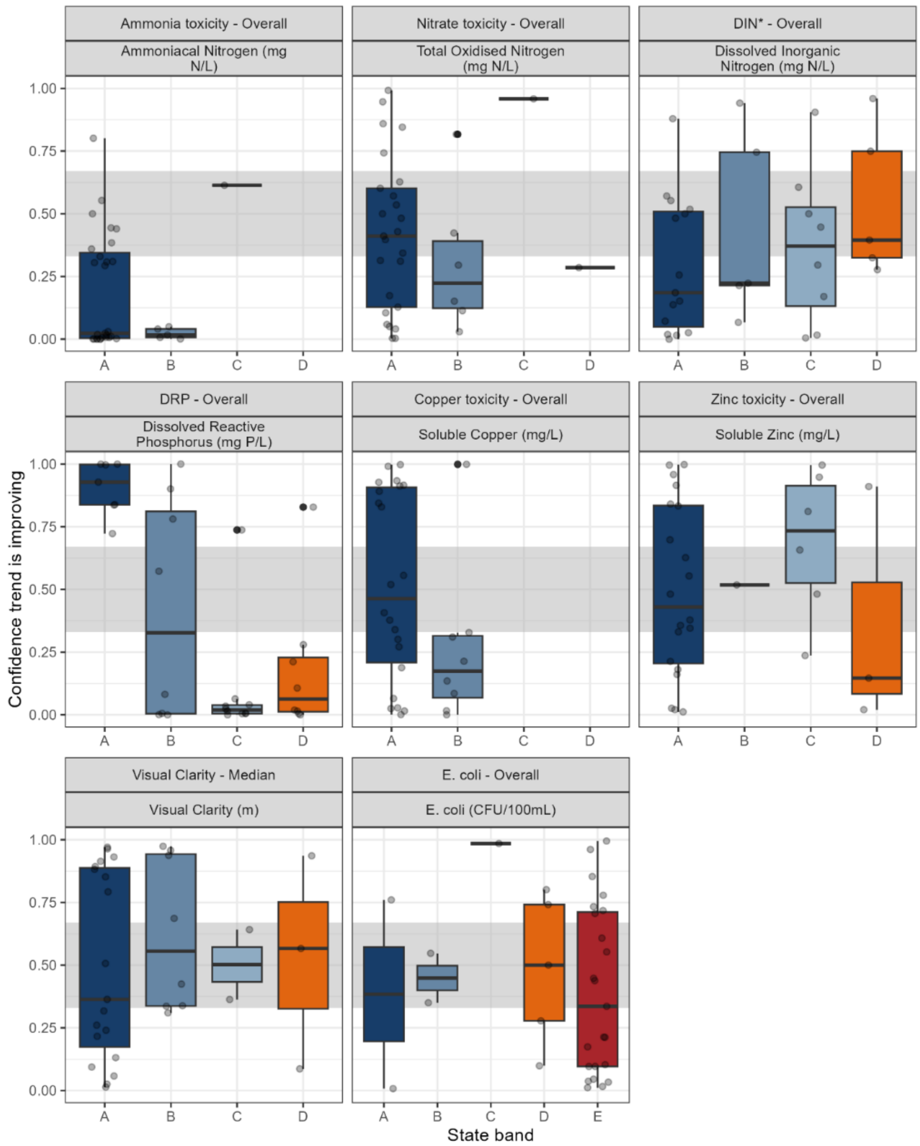


Figure 10-4: Confidence in improving trends (1 July 2017 - 30 June 2024) by overall state grade for multiple parameters. For each plot the first line in the title represents the numeric attribute state name and, and the second line in the title is the trend water quality variable and associated measurement units. The colour of the box indicates the water quality overall attribute state band. Grey shaded area shows the range of ‘low confidence’ in direction of trend.

11 Summary and recommendations

11.1 Summary of key findings

This report has provided information on the current state of river water quality in Tāmaki Makaurau, assessed water quality against national and regional guidelines, and explored how water quality has changed over the past seven years.

Monthly water quality monitoring was conducted at 37 sites, covering a range of parameters including water temperature, dissolved oxygen, nutrient concentrations (nitrogen and phosphorus), metal contaminants (copper and zinc), suspended sediment and water clarity, and faecal indicator bacteria (*E. coli*).

Water quality was generally in the best condition in streams draining native forest catchments. Water quality was progressively worse in streams flowing through predominantly rural, and urban catchments, with urban streams in the poorest condition with typically higher nutrient concentrations, higher metal concentrations, and higher water temperatures, and lower dissolved oxygen concentrations in urban waterways. Many water quality parameters vary widely across sites, sometimes over several orders of magnitude, particularly in urban environments. Variation in some aspects of water quality including water clarity and total suspended solids was more clearly explained by climate and underlying geology than land cover.

Widespread issues persist: these include risks to human health from faecal contamination, and potential impacts on ecosystem health and function from nutrient enrichment influencing most of the rural and urban waterways. Previously identified localised issues also remain, including nitrate contamination in the wider Pukekohe area (linked to aquifer contamination), and zinc contamination in urban areas. In both of these areas, several streams failed the national and proposed regional bottom lines respectively indicating risks of chronic adverse toxicity effects on multiple freshwater species occur, negatively impacting ecosystem health values.

Regionally there is a low risk of water quality issues in relation to ammonia toxicity, or copper toxicity impacts on aquatic fauna for even the most sensitive species.

In contrast to the relatively consistent picture of current state, trends in water quality varied in both direction and magnitude across sites and land cover types. Trends also varied among native forest sites suggesting that climatic or natural processes contributing to observed trends may vary spatially. Ammoniacal nitrogen and dissolved reactive phosphorus were the parameters that showed the greatest proportion of sites with very likely degrading trends. There was a tendency for sites with poorer water quality to have a greater likelihood of degrading trends, i.e. degraded and degrading.

For one key issue, there were indications of improvement. Two of the three monitored streams in the wider Pukekohe area had improving trends in total oxidised nitrogen and at one site this was also accompanied by a change in NOF grade for the nitrate toxicity attribute to above the national bottom line.

Land cover changes within the upstream catchments of most monitoring sites were minimal. There were several monitoring sites where changes were identified associated with urban development in greenfield areas (rural to urban conversion). Trends in water quality measures were also variable within these areas with some parameters improving and some degrading within each site. In one of these catchments (Vaughan Stream at Long Bay) there are indications that most measures of water quality are at least being maintained which may reflect water sensitive urban design practices however land use change impacts may not have been fully captured or may still become apparent over time.

Long-term trend analysis for dissolved oxygen and temperature was complicated by variation in the time of day samples were collected. Despite this limitation, most streams across all land cover classes – including those in native forest catchments – were found to be exhibiting long-term warming trends (according to 15-year trend assessment), consistent with broader climatic changes and rising atmospheric temperatures.

The wide variation in trend directions across water quality parameters and sites highlights the complex interplay between land use activities, differences in contaminant pathways from land to streams, and climatic influences that can either amplify or mitigate land-based impacts. Many water quality issues and their underlying causes are highly site-specific.

The next section outlines some suggestions for further work, including gathering further information on potential drivers of change, based on sites with specific issues and/or a high magnitude of change (improving or degrading).

11.2 Recommendations

The findings of this report can be used to inform current state and trends as required for NPS-FM implementation, to inform the effectiveness of policy initiatives, strategies and operation and to support other monitoring and research in the region. This state and trend analysis provides a broad screening tool to guide future research and help prioritise further investigations.

The trend period assessed here was relatively short due to limitations associated with changing methodologies. Shorter trend periods are more vulnerable to the influence of climatic variation. Because of the potential for trend direction and magnitude to change, a single trend period should not be regarded as the sole basis for making decisions to act (Snelder and Kerr, 2022). Further analysis, including adjusting water quality observations for flow variability, exploring alternative trend methods, and modelling, may support further evaluation of the drivers of water quality change.

However, there were many site-specific issues, and sites where a high magnitude of change (improving or degrading) was observed across multiple measures of water quality. We recommend investigations for several of these locations, as summarised in Table 11-1.

Other recommendations for future research include:

- Further evaluation of landscape pressures in urban environments, particularly within catchments with observed higher rates of change in water quality measures could be undertaken. Brownfield intensification within existing urban areas was not well captured by the land cover analysis in this report.
- The monitoring network would benefit from further review in relation to the Auckland Future Development Strategy (Auckland Council 2023(c)), and representation at the scale of Freshwater Management Units for reporting under the NPS-FM 2020 (or future iteration).
- Consideration of options for alternative/additional indicators of ecosystem health relevant to soft-bottomed streams. Because there are few hard-bottomed streams in the Auckland Region, there is limited use for nutrient targets based on managing periphyton growth
- Short term monitoring projects would benefit from utilising the long term state of the environment monitoring results to contextualise any seasonal variation or to compare to relevant similar locations or reference sites.

Recommendations in relation to future assessment of water quality state include:

- Directly measured visual clarity should be used in future state assessments once minimum data requirements are met, as the NPS-FM sets limits and national bottom lines based on visual clarity, and direct measurements provide greater accuracy than turbidity-derived values.

- Both visual clarity and field turbidity should continue to be monitored in parallel to improve predictive model performance, as models will still be needed for trend assessment.
- Future reporting and assessments of copper and zinc should consider any updates to revised guideline values and attribute bands. The attribute bands used in this report were based on the 2023 draft Australia and New Zealand Fresh Water Quality Guidelines which are currently under review and some changes are expected before finalisation, later in 2025. Following that, some work will be required to develop attribute band tables for both metals for ongoing regional use.

Table 11-1: Summary of selected site-specific issues and potential future steps.

Land Cover	Site(s)	Issue	Potential future steps
Exotic forest	Ararimu Tributary	Elevated zinc issues have been identified in this catchment for many years. A synoptic survey has indicated that this is not isolated to this single monitoring site but that the issue is not common to exotic forestry areas in the wider region.	Further investigation may support the identification of drivers of zinc contamination in the area.
Exotic forest	Mahurangi River (Upper)	Very likely improving trends in total oxidised nitrogen appear to be driven by peaks in 2018-2019. Large scale land cover changes occurred within the Mahurangi upstream catchment over the trend period (exotic forest harvest and SH1 motorway construction).	Further investigation including collation of motorway project construction monitoring information at the localised scale may support attribution of change in water quality to specific actions and drivers over this period.
Rural – Low	Ōkura River	Poor visual clarity has been consistently identified at this site. A synoptic survey did not identify any specific point source issues. Short term high concentrations of nutrients were also observed indicative of potential pollution events.	An additional monitoring site was implemented in an adjacent catchment, Orewa River, which shares similar underlying geology to Okura River. Recommend comparing the results for these streams, along with information on geology and predicted sediment yields to understand whether the poor clarity in Okura River is due to natural sources, land management or other human activities.
Pukekohe	Ngākōroa Stream Waitangi Stream Whangamaire Stream	Long term nitrate toxicity issues linked with nitrate contamination in underlying shallow aquifers. Potential other nutrient related issues.	Further investigations are warranted on the indirect effects of nutrient enrichment and interactions between nitrogen and phosphorus. Additional monitoring sites have been implemented at the lower Ngākōroa Stream, and at Whangapouri Stream which will provide additional data for these investigations.

Land Cover	Site(s)	Issue	Potential future steps
Rural – High	Rangitōpuni Stream	Higher magnitude degrading trends in ammoniacal N and DRP.	Additional surveys or evaluation could support management options to improve water quality.
Rural – High	Kumeu River	Higher magnitude degrading trends in *E. coli* indicative of faecal contamination.	Additional surveys or evaluation, particularly evaluation of wet weather/dry weather events could support management options to improve water quality.
Urban	Omaru Creek Botany Creek Ōtara Creek (South) Ōtara Creek (East) Avondale Stream.	Sites that are consistently noted as degraded, and there are indications of further degradation. Higher magnitude trends were identified across multiple measures of nutrients, metals, and in some cases *E. coli*. Avondale is the only urban stream that failed the national bottom for visual clarity and keeps degrading	Additional surveys or evaluation could support management options to improve water quality. Investigate the possible causes of visual clarity degradation in Avondale Stream by implementing additional synoptic survey upstream of the current monitoring site.
Urban	Puhinui Stream Ōteha Stream	Very likely improving, higher magnitude trends were observed across multiple measures of water quality. Puhinui Stream has been a target of ongoing interventions and stream restoration with extensive further works planned that are intended to contribute towards ongoing improvement of this waterway. Urban development has been occurring within the upstream catchment of Ōteha Stream.	Further research could be undertaken, including project scale intervention monitoring to improve ability to attribute change to actions.

12 Acknowledgements

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14 Appendices

Appendix 1: Land cover analysis

The provisional 2023/24 land cover update for the Auckland region was undertaken by Auckland Council using a desktop-based manual mapping approach. Land cover changes since the previous 2018/19 dataset (LCDBv5) were systematically identified and delineated following a 5x5km grid overlaid on the region in ArcPro. This involved visual interpretation of multiple optical imagery sources, including recent Sentinel-2 (2023/24), Maxar (2023), and high-resolution aerial imagery (2023-2024), alongside a Sentinel-based NDVI difference model to highlight potential areas of change. Identified changes were then classified directly into a copy of the national LCDBv5 dataset, ensuring backward compatibility and adherence to the national LCDB classification schema and mapping standards (minimum mapping unit of 1 hectare, minimum polygon width of 30m). The detailed classes applied in the LCDB update are outlined in Table A1-1 and these classes were aggregated for the purposes of this report.

Table A1-1: Detailed land cover classes (Auckland Council, 2025) and aggregations applied in this report.

Detailed Class	Aggregated Land Cover Classes	Dominant Land Cover Class
Deciduous Hardwoods	Exotic forest	Exotic
Exotic Forest	Exotic forest	Exotic
Forest - Harvested	Exotic forest	Exotic
Sand or Gravel	Other	NA
Surface Mine or Dump	Other	NA
Gravel or Rock	Other	NA
Landslide	Other	NA
Not Land	Other	NA
Lake or Pond	Water	NA
Estuarine Open Water	Water	NA
River	Water	NA
Mangrove	Wetland	NA
Flaxland	Wetland	NA
Herbaceous Freshwater Vegetation	Wetland	NA
Herbaceous Saline Vegetation	Wetland	NA
Broadleaved Indigenous Hardwoods	Native forest	Native
Indigenous Forest	Native forest	Native
Manuka and/or Kanuka	Native forest	Native
Matagouri or Grey Scrub	Native forest	Native
Fernland	Native forest	Native
Orchard, Vineyard or Other Perennial Crop	Horticulture	Rural
Short-rotation Cropland	Horticulture	Rural
Mixed Exotic Shrubland	Rural	Rural
Gorse and/or Broom	Rural	Rural
High Producing Exotic Grassland	Rural	Rural
Low Producing Grassland	Rural	Rural
Built-up Area (settlement)	Urban Built-up Area	Urban
Transport Infrastructure	Urban Transport	Urban
Urban Parkland/Open Space	Urban Parkland	Urban

Appendix 2: Analytical Methods

Table A2-1: River water quality analytical methods and detection limits. Parameters measured in the field

Group	Parameter	Units	Field Equipment/ 2010-2014	Detection Limit	Equipment 2014*-current	Detection Limit
Physical	Dissolved oxygen saturation	% sat	YSI 556	0	EXO sonde, optical method	0
Physical	Dissolved oxygen	mg/L	YSI 556	0	EXO sonde, optical method	0
Physical	Temperature	°C	YSI 556	-5	EXO sonde, thermistor	-5
Physical	Conductivity	mS/cm	YSI 556	0	EXO sonde, 4-electrode nickel cell	0
Physical	Salinity	ppt	YSI 556	0	EXO sonde, 4-electrode nickel cell	0
Physical	pH	pH units	YSI 556	0	EXO sonde, glass combination electrode	0
Clarity	Turbidity	FNU	NA	NA	EXO sonde, optical 90° scatter	0

Table A2-2: River water quality analytical methods and detection limits. Parameters measured in the laboratory.

Group	Parameter	Units	Watercare Lab 2009-June 2017		Hill Lab July 2017-current	
			Methods	Detection Limit	Methods	Detection Limit
Clarity	Total suspended solids	mg/L	APHA (2005/2012) 2540 D	0.2	APHA (2017) 2540 D 23rd ed (modified)	3 (2017-Oct 2020) 1 (October 2020-Current)
Clarity	Turbidity	NTU	APHA (2005/2012) 2130 B (modified)	0.1 (2010-August 2015) 0.05 (from August 2015)	APHA (2017) 2130 B 23rd ed (modified)	0.05
Nutrients	Ammoniacal nitrogen	mg N/L	APHA (2005/2012) 4500-NH3 G (Modified) APHA (online edition) 4500-NH3 H (modified) (from July 2016)	0.005	APHA (2017) 4500-NH3 H 23rd ed	0.005
Nutrients	Nitrite nitrogen	mg N/L	NA	NA	APHA (2017) 4500-NO3- I 23rd ed (modified)	0.001
Nutrients	Nitrate nitrogen	mg N/L	NA	NA	Calculation ((NO3N+NO2N) – NO2N)	0.001
Nutrients	Dissolved inorganic nitrogen	mg N/L	AC Calculation (NH4-N + NO3-N + NO2-N)	0.007	Calculation (NH4-N + NO3-N + NO2-N)	0.005
Nutrients	Total oxidised nitrogen	mg N/L	APHA (2005/2012) 4500-NO3 F (modified)	0.002	APHA (2017) 4500-NO3- I. Flow injection	0.001

Group	Parameter	Units	Watercare Lab 2009-June 2017		Hill Lab July 2017-current	
			Methods	Detection Limit	Methods	Detection Limit
			APHA (online edition) 4500-NO3 I (from July 2016)			
Nutrients	Total Kjeldahl nitrogen	mg N/L	Calculation	0.02	Calculation (TN – (NO3N+NO2N))	0.01
Nutrients	Total nitrogen	mg N/L	APHA (2005/2012) 4500-P J & 4500-NO3 F (modified), APHA (online edition) 4500-P J & 4500-NO3 I (modified) (from July 2016)	0.02, 0.01 (from September 2014)	APHA (2017) 4500-N C & 4500-NO_3^- I 23rd ed (modified)	0.01
Nutrients	Dissolved reactive phosphorus	mg P/L	APHA (2005/2012) 4500-P B, F (modified), APHA (online edition) 4500-P F (from October 2015)	0.005, 0.002 (from September 2014)	APHA (2017) 4500-P G 23rd ed (modified) Flow injection	0.004, 0.001 (from May 2019)
Nutrients	Total phosphorus	mg P/L	APHA (2005/2012) 4500-P B, J (modified)	0.005, 0.004 (from August 2014)	APHA (2017) 4500-P B, E (modified), APHA (2017) 4500-P H (modified) (from December 2020)	0.004, 0.002 (from December 2020)
Metals	Soluble copper	µg/L	USEPA 200.8 (modified)	0.00001	APHA (2017) 3125 B 23rd ed	0.0005
Metals	Total copper	µg/L	USEPA 200.8 (modified)	0.00001	APHA (2017) 3125 B 23rd ed / USEPA 200.8	0.00053
Metals	Soluble zinc	µg/L	USEPA 200.8 (modified)	0.0003	APHA (2017) 3125 B 23rd ed	0.001
Metals	Total zinc	µg/L	USEPA 200.8 (modified)	0.0003	APHA (2017) 3125 B 23rd ed / USEPA 200.8	0.0011
Bacteria	E.coli	cfu/100mL	USEPA (2002) Method 1603	2	APHA (2017) 9222 G, APHA (2017) 9222 I 23rd ed (From March 2020)	1
Modifiers	Dissolved organic carbon	mg/L	NA	NA	APHA (2012/2017) 5310 C (modified) 23rd ed	0.3
Modifiers	Total hardness	mg/L	NA	NA	Calculation APHA (2017) 2340 B 23rd ed.	1.0
Modifiers	Soluble calcium	mg/L	NA	NA	APHA (2017) 3125 B 23rd ed	0.05
Modifiers	Soluble magnesium	mg/L	NA	NA	APHA (2017) 3125 B 23rd ed	0.02
Physical	Total alkalinity	mg/L	NA	NA	APHA (2017) 2320 B 23rd ed (modified)	1.0
Physical	pH		APHA 4500-H B	0.1	NA	NA

Appendix 3: Additional NOF methodology details

The NPS-FM NOF specifies a minimum of 60 observations over five years for some attributes. For monthly monitoring, this would require all observations with no missing data (or data omitted for quality standards). All AC records include at least some missing data including due to missed sampling during national and regional COVID-19 lockdown periods so the specified minimum could not be met. Therefore, alternative minimum data standards were applied here.

Some compliance statistics are specified as “Annual” (minimum, maximum, median, 95th percentile) in the NPS-FM Appendices. However, we calculated all compliance statistics for all attributes over the entire five year state period, following advice given by McBride (2016).

It was recommended in Gadd et al. (2019) and Gadd et al. (2023) that the US EPA CMC and Canadian acute guideline values could be replaced in the strawman table if acute guidelines were developed for Australia and New Zealand. The derivation of acute guidelines was detailed in Gadd et al. (2024) and interim Tier 1 acute guideline values were provided. However, Gadd et al. (2024) noted inconsistencies between chronic and acute guideline values and further work was recommended to develop a robust attribute table once the chronic default guideline values are finalised.

Table A3-1: Appendix 2A River water quality attributes. The shaded boxes indicate values below the national bottom line.

NOF River Attribute	Ammonia (Ecosystem Health – toxicity)		Nitrate (Ecosystem Health – toxicity)		Suspended fine sediment (Ecosystem Health)	
Unit	mg NH_4-N/L pH adjusted		mg NO_3-N/L		Visual clarity (m)	
Metric	Median	95^{th} %ile	Annual Median	95^{th} %ile	Median Class 1	Median Class 2
A	≤ 0.03	≤ 0.05	≤ 1.0	≤ 1.5	≥ 1.78	≥ 0.93
B	> 0.03 and ≤ 0.24	> 0.05 and ≤ 0.40	> 1.0 and ≤ 2.4	> 1.5 and ≤ 3.5	< 1.78 and ≥ 1.55	< 0.93 and ≥ 0.76
C	> 0.24 and ≤ 1.30	> 0.40 and ≤ 2.20	> 2.4 and ≤ 6.9	> 3.5 and ≤ 9.8	< 1.55 and > 1.34	< 0.76 and > 0.61
D	> 1.30	> 2.20	> 6.9	> 9.8	< 1.34	< 0.61

Table A3-2: NPS-FM (2020) DRP Action Plan Attribute Bands and draft proposed DIN band

NOF River Attribute	DRP		DIN	
Metric	Median	95th %ile	Median	95th %ile
Unit	mg/L			
A	≤ 0.006	≤ 0.021	≤ 0.24	≤ 0.56
B	> 0.006 and ≤ 0.01	> 0.021 and ≤ 0.030	>0.24 and ≤ 0.50	> 0.53 and ≤ 0.01.10
C	>0.01 and ≤ 0.018	> 0.030 and ≤ 0.054	>0.50 and ≤ 1.0	>1.10 and ≤2.05
D	> 0.018	> 0.054	>1.0	>2.05

Table A3-3: *Appendix* 2A River human health attribute (modified)

NOF River Attribute	*Escherichia coli* (Human Contact)			
	% > 540	% > 260	Median	95th %ile
	cfu/100mL			
A	≤ 5%	≤ 20%	≤ 130	≤ 540
B	>5- ≤10%	>20- ≤30%	≤ 130	>540 - ≤ 1000
C	>10-≤20%	>30- ≤34%	≤ 130	>1000 - ≤ 1200
D	>20-≤30%	> 34% - ≤50%	> 130 - ≤260	> 1200
E	>30%	> 50%	> 260	> 1200

Table A3-4: Recommended attribute tables for use by NZ regional councils for chronic copper and zinc toxicity (Gadd et al. 2023). The shaded boxes indicate values below the proposed regional bottom line.

Attribute Band	Attribute State	Bioavailable Copper		Bioavailable Zinc	
		Median (ug/L)	95th %ile (ug/)	Median (ug/L)	95th %ile (ug/L)
A	Low likelihood of toxic effects on even the most sensitive species. 50% of time protect 99% species from chronic toxicity, 95% of time protect 95% species from chronic toxicity.	≤0.2	≤0.47	≤1.5	≤4.1
B	Possible toxic effects on the most sensitive species, but low likelihood of toxic effects on most (~90%) species 50% of time protect 95% species from chronic toxicity, 95% of time protect 90% species from chronic toxicity.	>0.2 - ≤0.47	>0.47 - ≤0.73	>1.5 - ≤4.1	>4.1 - ≤6.8

Attribute Band	Attribute State	Bioavailable Copper		Bioavailable Zinc	
C	Moderate likelihood of effects. Possible toxic effects (chronic) on sensitive species (20% most sensitive), but low likelihood of toxic effects (chronic) on most species 50% of time protect 80% species from chronic toxicity. Acute toxicity possible for the most sensitive species.	>0.47 - ≤1.3	>0.73 – ≤1.8	>4.1 - ≤12	>6.8 - ≤24
D	High likelihood of adverse effects on multiple species. Toxic effects (chronic or acute) are possible on sensitive and insensitive species	>1.3	>1.8	>12	>24

* Bioavailable copper = Dissolved copper ÷ $(DOC/0.5)^{0.9777}$

* Bioavailable zinc = Adjusted based on DOC, hardness, and pH utilising script available on github https://github.com/niwa/CuZn_DGV_adjusters/tree/main

Interim Tier 1 acute guideline values are outlined below in Table A-5. These guidelines are based on high bioavailability conditions (based on New Zealand waters). Tier 2 acute guideline values provide a sample specific assessment based on the equations outlined in Table A-6 (Gadd, 2024). The 95% level of protection was applied for analysis in this report applicable to most modified environments. Higher guideline values such as at 90% species protection may be more applicable for highly modified urban environments. Refer to Gadd et al. 2024 for further information on the use and application of these guidelines.

Table A3-5: Interim Tier 1 dissolved copper and zinc acute guideline values for the protection of aquatic life in freshwater. Modified from Gadd et al., 2024.

Level of species protection	Dissolved copper (ug/L)	Dissolved zinc (ug/L)
99% (most protective)	0.7	11
95% (applicable to most human – modified environments)	1.3	24
90% (applicable to highly disturbed systems	1.7	36
80% (least protective)	2.9	59

Table A-6. Formulae for the calculation of Tier 2 bioavailability adjusted acute guideline values. Modified from Gadd et al., 2024. Level of species protection	Dissolved copper (ug/L)	Dissolved zinc (ug/L)
99% (most protective)	=exp (-7.2 + 0.78 *pH* + 0.58 ln(*hard.*) + 0.7 X ln(*DOC*)	=exp (1.75-0.12 x *pH* + 0.6 X ln(*hard.*) + 0.13 X ln(*DOC)*
95% (applicable to most human – modified environments)	=exp (-6.6 + 0.78 *pH* + 0.58 ln(*hard.*) + 0.7 X ln(*DOC*)	=exp (2.5-0.12 x *pH* + 0.6 X ln(*hard.*) + 0.13 X ln(*DOC)*
90% (applicable to highly disturbed systems	=exp (-6.3 + 0.78 *pH* + 0.58 ln(*hard.*) + 0.7 X ln(*DOC*)	=exp (2.9-0.12 x *pH* + 0.6 X ln(*hard.*) + 0.13 X ln(*DOC)*
80% (least protective)	=exp (-5.8 + 0.78 *pH* + 0.58 ln(*hard.*) + 0.7 X ln(*DOC*)	=exp (3.4-0.12 x *pH* + 0.6 X ln(*hard.*) + 0.13 X ln(*DOC)*

Table A3-7: Site-specific linear regressions for conversions from turbidity (FNU) (x) to visual clarity (m) (y). Note, Whangamaire Stream (Lower) was excluded from the analysis because of measurement procedure inconsistency.

Station name	Conversion formula	Dominant land cover class
Wairoa Tributary	log(y) = 0.785 + -0.963log(x)	Native forest
West Hoe Stream	log(y) = 0.543 + -0.755log(x)	Native forest
Nukumea Stream	log(y) = 0.682 + -0.819log(x)	Native forest
Cascades Stream (Waitākere)	log(y) = 0.852 + -0.848log(x)	Native forest
Mahurangi River (Right Branch)	log(y) = 0.898 + -1.035log(x)	Exotic forest
Ararimu Tributary (Riverhead)	log(y) = 0.672 + -0.82log(x)	Exotic forest
Ōpanuku Stream (Upper)	log(y) = 0.787 + -0.922log(x)	Rural - Low
Ōkura River	log(y) = 0.743 + -0.975log(x)	Rural - Low
Cascades Stream (Waiheke)	log(y) = 0.536 + -0.682log(x)	Rural - Low
Matakana River	log(y) = 0.622 + -0.847log(x)	Rural - Low
Waiwera Stream	log(y) = 0.903 + -1.104log(x)	Rural - Low
Wairoa River	log(y) = 0.679 + -0.886log(x)	Rural - Low
Mahurangi River (Lower at Brown Road)	log(y) = 0.751 + -0.968log(x)	Rural - High
Makarau River	log(y) = 0.835 + -1.022log(x)	Rural - High
Papakura Stream (Upper)	log(y) = 0.256 + -0.555log(x)	Rural - High
Rangitōpuni River	log(y) = 0.702 + -0.962log(x)	Rural - High
Kaukapakapa River	log(y) = 0.596 + -0.874log(x)	Rural - High
Kumeū River	log(y) = 0.369 + -0.646log(x)	Rural - High
Mangawheau Stream	log(y) = 0.733 + -0.858log(x)	Rural - High
Ngākōroa Stream (Upper)	log(y) = 0.625 + -0.738log(x)	Pukekohe
Waitangi Stream	log(y) = 0.664 + -0.861log(x)	Pukekohe
Papakura Stream (Lower)	log(y) = 0.556 + -0.822log(x)	Urban
Onetangi Stream	log(y) = 0.734 + -0.857log(x)	Urban
Vaughn Stream (Lower)	log(y) = 0.563 + -0.735log(x)	Urban
Ōtara Creek (East)	log(y) = 0.593 + -0.771log(x)	Urban
Puhinui Stream (Lower)	log(y) = 0.574 + -0.852log(x)	Urban
Avondale Stream (Lower)	log(y) = 0.85 + -1.021log(x)	Urban
Lucas Creek (Lower)	log(y) = 0.729 + -0.885log(x)	Urban
Ōteha River	log(y) = 0.775 + -0.973log(x)	Urban
Ōtara Creek (South)	log(y) = 0.466 + -0.762log(x)	Urban
Newmarket Stream	log(y) = 0.655 + -0.882log(x)	Urban
Oakley Creek (Lower)	log(y) = 0.704 + -0.977log(x)	Urban
Botany Creek (East)	log(y) = 0.833 + -0.963log(x)	Urban
Ōmaru Creek	log(y) = 0.637 + -0.875log(x)	Urban
Pakuranga Stream	log(y) = 0.711 + -0.918log(x)	Urban - Project

Appendix 4: Trend Period Selection

Introduction

In July 2017 Auckland Council changed laboratory providers resulting in a change in analytical methodology, and/or detection limits for several water quality parameters. A limited amount of paired sampling was undertaken during this laboratory change, whereby samples were collected and sent to both laboratories, and analysed with the old and new methods. This appendix presents the results of that paired sampling and statistical analyses to assess differences between the laboratory methods.

Analytical methodology changes are inevitable and also occur within laboratory providers due to advances in equipment and other technology that improve the accuracy of analysis. Changes of laboratory provider may also be motivated by other financial or service provision requirements.

Additional guidance now available recommends that paired testing is critical to manage such methodology changes (NEMS, 2019; Wood, 2024) to maintain the value of long term data collection. The objective of paired sampling is ideally to enable the calculation of adjustment factors to align historic datasets and enable trend analysis to be undertaken and enable comparison between state periods. Alternatively, the objective is to be able to adequately describe expected bias and take this into consideration in the interpretation of state and trend analysis.

Paired testing of environmental samples should preferably be undertaken at all water quality sites, to assess the degree of systemic bias between methods, and any site specific interactions and variation in random differences (Wood, 2024). Paired sampling should cover an adequate range of the historic data set which is expected to require a minimum of 12 months of monthly sampling or additional resource investment into more frequent or targeted sampling (such as wet weather events) (Wood, 2024). Poorly accounting for such systemic changes can confound results and risk falsely attributing observed changes to environmental changes and/or anthropogenic actions.

Methods

Paired samples were collected at three sites: one reference site (Nukumea Stream), with generally high water quality and low contaminant concentrations; a rural[21] site (Papakura Stream – Lower) with moderate contaminant concentrations; and an urban site (Pakuranga Stream) with high contaminant concentrations. For two of the sites, these samples were collected for five consecutive months (from August to December 2017) whereas for the rural site, samples were only collected for four months (August, October to December 2017).

[21] Papakura Stream – Lower is reported in the main body of this report as an 'urban' site however in 2017/2018 it was regarded as a rural site with only 7% urban land cover in the upstream catchment.

There are limited records available regarding the sampling methodology employed for the collection of duplicate samples at the time. It is assumed that samples were collected and evenly divided between bottles to be sent to each respective laboratory following standard protocols. No quality codes were assigned to duplicate samples from Watercare Services Ltd. One notable outlier for total copper at Papakura Stream was omitted from analysis.

Paired analysis was undertaken for 13 water quality parameters. The laboratory analytical methods and detection limits utilised by each laboratory are outlined in Appendix 1. Some parameters reported in the main body of this report were added to the monitoring network after 2017 and therefore paired testing was not undertaken (e.g. metal toxicity modifying factors, separate measurements of nitrite and nitrate).

Representation of the paired sampling undertaken.

It was recommended by Wood (2024) that the range of pairwise sample results should cover the interquartile range of the historical data. If it does not, then the paired data may not be adequately representative of the preceding period and may poorly account for differences between methods.

The representativeness of the paired sampling available was evaluated on a site specific basis. The range (min-max values) of the original laboratory (Watercare Services) over the paired sampling period was compared to the interquartile range for that site for the historical data available from the preceding five years (July 2012 to June 2017).

Level of agreement between parallel testing

The primary objective of parallel testing is to assess the level of agreement between the two methods. Assessing the level of agreement involves estimating the systemic differences (bias) and random differences (Wood, 2024). Wood (2024) recommend the Bland and Altman (1986) procedure which provides a simple summary of the differences and similarities between methods.

The Bland and Altman method uses two plots to visually assess bias and precision, supported by simple calculations to estimate the level of agreement (Wood, 2024). Where one or both of the paired samples were censored, they were omitted from analysis. As measured values approach the detection limit estimates of bias are less precise (Wood, 2024).

The first plot displays the paired samples in a simple scatter plot indicating deviation from the 1:1 line of agreement. In this analysis this visualisation is supported by Lin's concordance correlation (Lin 1989), this estimates how close two methods are to a 1:1 line of perfect numeric agreement. Lin's concordance correlation was undertaken utilising the *epi.ccc.* function from the epiR package (Stevenson and Sergeant 2023). The interpretation applied here is outlined in Table A4-1. Lin's CCC is sensitive to sample size.

Table A4-1: Interpretation of Lin's Concordance Correlation Coefficient (CCC). Recommended by Wood (2024), after Davies-Colley and McBride (2016).

Lin's CCC	Strength of agreement
>0.99	Almost perfect
0.96-0.99	Substantial
0.90-0.95	Moderate
<0.90	Poor

The second plots the difference between the paired results against the average value of each pair. From this, the mean difference between the paired results indicates the bias or systemic difference between the methods. As water quality results are typically highly skewed, results are log transformed prior to analysis, this estimates relative rather than absolute differences. Following calculation of the mean differences the inverse is calculated to present the level of agreement as a ratio rather than a log of the ratios. A value greater than 1 implies that method B (Hills) generally gives higher results than method A (Watercare) and vice versa. The standard deviation estimates the extent of random difference between the two methods.

Difference d = ($log1_0$Hills-log_{10}Watercare)
Mean bias (ratio) = (Σ*d)/n*
Upper level of agreement = d +2s
Lower level of agreement = d = 2s
Deming regression lines were also fitted to these plots to further identify if there is a trend in the bias e.g. the bias changes from lower to higher concentrations. This was undertaken using the *mcreg* (Deming method comparison) function from the *R* package MCR (Potapov et al. 2023) after Wood (2024).

Results

Representation of the paired sampling undertaken in 2017.

The paired sampling undertaken did not cover the interquartile range for most site x parameter combinations (Table A4-2). The interquartile range was covered for two out of three sites for Ammoniacal N, Total Oxidised N, Total P, soluble copper, and TSS.

Table A4-2: Comparison between the min/max values of the paired Watercare samples (August-September 2017) to the interquartile range from the preceding five years (01 July 2012 - 30 June 2017).

	Nukumea Stream		Papakura Stream (Lower)		Pakuranga Stream	
Parameter	Min≤Long term lower quartile	Max≥Long term upper quartile	Min≤Long term lower quartile	Max≥Long term upper quartile	Min≤Long term lower quartile	Max≥Long term upper quartile
Ammoniacal N	TRUE	TRUE	TRUE	TRUE	FALSE	TRUE
Total oxidised nitrogen	TRUE	FALSE	TRUE	TRUE	TRUE	TRUE
Total Kjeldahl Nitrogen (calculated)	FALSE	TRUE	TRUE	TRUE	FALSE	TRUE
Total Nitrogen	TRUE	TRUE	TRUE	FALSE	FALSE	TRUE
Dissolved Reactive Phosphorus	TRUE	TRUE	FALSE	FALSE	FALSE	TRUE
Total Phosphorus	FALSE	TRUE	TRUE	TRUE	TRUE	TRUE
Soluble copper	TRUE	TRUE	FALSE	TRUE	TRUE	TRUE
Total copper	FALSE	TRUE	FALSE	TRUE	TRUE	FALSE
Soluble zinc	TRUE	TRUE	TRUE	FALSE	TRUE	FALSE
Total zinc	TRUE	TRUE	TRUE	FALSE	TRUE	FALSE
Total Suspended Solids	TRUE	FALSE	TRUE	TRUE	TRUE	TRUE
Turbidity (NTU)	FALSE	TRUE	FALSE	TRUE	TRUE	TRUE
E. coli	FALSE	FALSE	TRUE	TRUE	TRUE	FALSE

Level of agreement between parallel testing

All analytical comparisons are limited by the sample size available (maximum pooled data set n=8 to 14) and poor representation of the range of expected results as outlined above.

Figure A-0-1 provides a summary of the systemic bias, and random differences (error bars indicating upper and lower levels of agreement) between paired results for each parameter assessed. The numeric values and interpretation for each parameter are also summarised in Table A4-3. The greater the deviation from 1, the greater the degree of estimated systemic bias. The larger the random difference, the harder it is to estimate systemic bias accurately. The underlying Bland Altman and Deming regression plots are shown for parameters that are assessed under the National Objectives Framework and or additional regional attributes in Figure A-0-2 and Figure A-0-3.

For all parameters assessed, the upper and lower levels of agreement spanned greater than and less than 1 indicating that within each parameter, there were some individual samples where the Watercare results were higher than the Hills results and vice versa.

For six parameters measured in the paired sampling programme, there was a poor (<0.90) level of concordance. Several of these parameters (metals, TSS) had a tendency for the Hills results to be higher than the Watercare results (>1). Conversely, Hills results tended to be lower than the Watercare results (<1) for DRP and *E. coli*. The parameters with the greatest deviation from 1 were TSS (LoA=1.71) and DRP (LoA=0.49). The upper and lower levels of agreement spanned a wide range, indicating a high level random difference between the two laboratories. In some instances, there was also a significant trend in the direction of bias with a tendency to greater deviation at lower concentrations.

The remaining seven parameters, including all measures of nitrogen, demonstrated moderate or better level of concordance between the two methods. There was almost perfect level of concordance (>0.99) for total oxidised nitrogen and turbidity and minimal bias (LoA= 1.02). Total copper showed substantial concordance and no indication of bias with low random deviation.

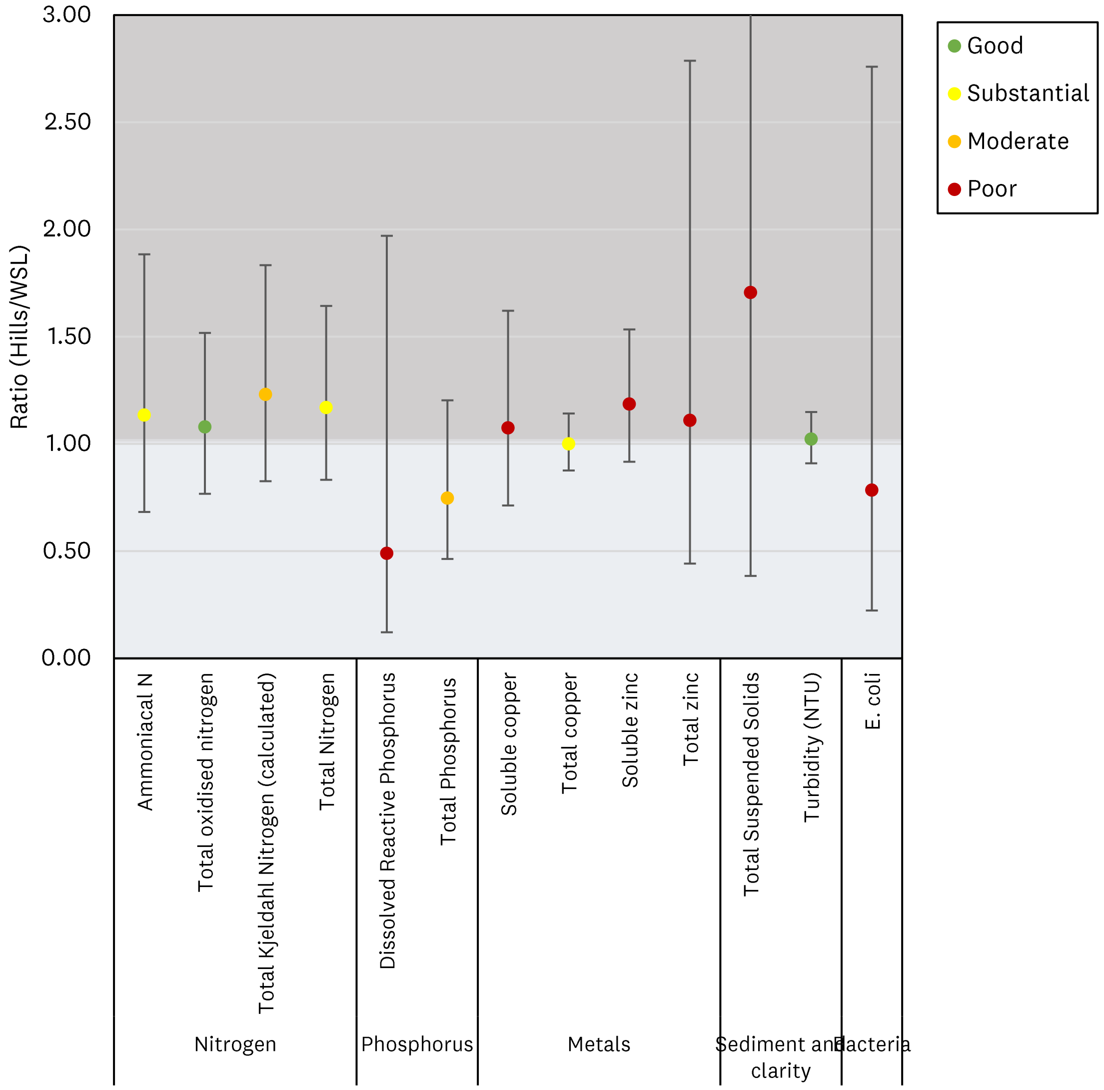
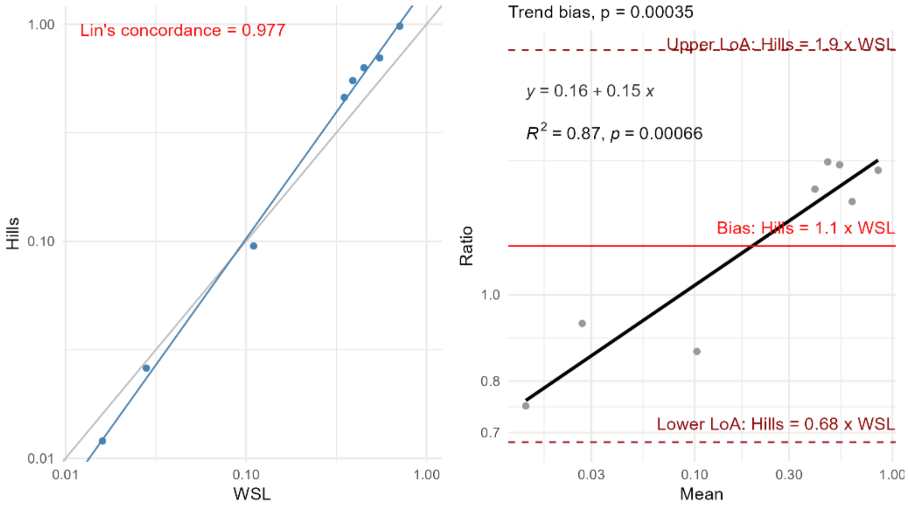


Figure A-0-1. The level of agreement between paired samples for each parameter (n=11 to14). Values >1 (grey background) indicate results from Hills are higher than results from Watercare and values <1 (blue background) indicate results from Watercare are higher than results from Hills. Bars display the upper and lower levels of agreement. Points are coloured by strength of concordance (Lin's CCC). Note upper limit for TSS of 7.57 not displayed.

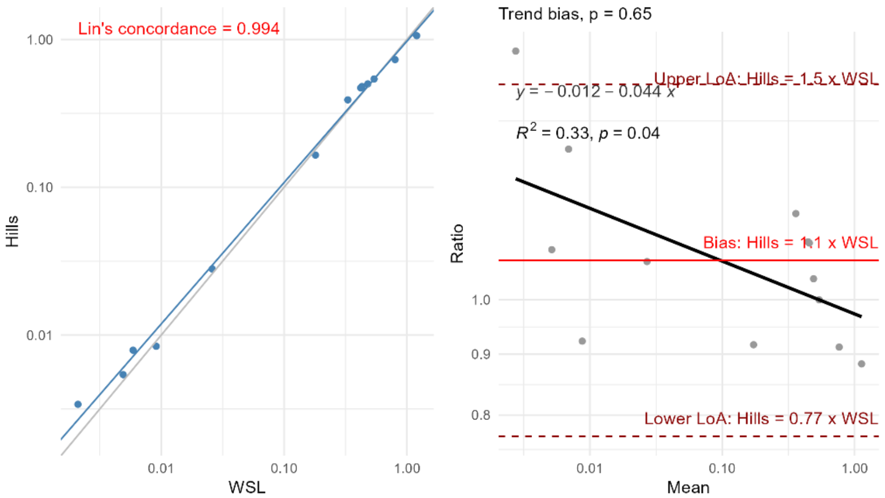
Table A4-3: Summary of level of agreement (LoA) between paired samples. Significant trend in bias (<0.05) in bold.

Group	Parameter	*n*	Lin's CCC	Ratio (Hills/ WSL)	LoA- Lower	LoA - Upper	Intercept	Slope	P value	Interpretation
Nitrogen	Ammoniacal N	8	0.977	1.13	0.68	1.88	0.177	1.165	**0.000**	Substantial concordance. Hills lower than Watercare. Significant trend in bias.
Nitrogen	Total oxidised nitrogen	13	0.994	1.08	0.77	1.52	-0.012	0.956	0.654	Near perfect concordance. Strong agreement between methods.
Nitrogen	Total Kjeldahl Nitrogen (calc.)	14	0.956	1.23	0.83	1.83	0.043	0.900	0.152	Moderate concordance. Hills higher than Watercare
Nitrogen	Total Nitrogen	14	0.980	1.17	0.83	1.64	0.040	0.911	**0.040**	Substantial concordance. Hills higher than Watercare. Low range of random difference.
Phosphorus	Dissolved Reactive Phosphorus	13	0.623	0.49	0.12	1.97	1.179	1.895	**0.003**	Poor concordance. Hills lower than Watercare. Wide range of random difference.
Phosphorus	Total Phosphorus	14	0.922	0.75	0.46	1.20	-0.025	1.077	0.805	Moderate concordance. Hills lower than Watercare. Low range of random difference.
Metals	Soluble copper	10	0.782	1.08	0.71	1.62	-0.828	0.711	0.160	Poor concordance. No indication of bias. Low range of random difference.
Metals	Total copper	11	0.986	1.00	0.88	1.14	0.034	1.012	0.838	Substantial concordance. Strong agreement. wide range of random difference.
Metals	Soluble zinc	12	0.888	1.19	0.92	1.53	-0.066	0.946	0.647	Poor concordance. Hills higher than Watercare.
Metals	Total zinc	12	0.897	1.11	0.44	2.79	0.563	1.225	0.199	Poor concordance. Hills higher than Watercare. Wide range of random difference.
Sediment and clarity	Total Suspended Solids	8	0.404	1.71	0.38	7.57	0.628	0.357	0.078	Poor concordance. Hills higher than Watercare. Very wide range of random difference.
Sediment	Turbidity (NTU)	14	0.995	1.02	0.91	1.15	0.023	0.986	0.399	Near perfect concordance. Strong agreement.
Bacteria	*E. coli*	14	0.879	0.78	0.22	2.76	-0.787	1.301	**0.035**	Poor concordance. Hills lower than Watercare. Wide range of random difference.

Ammoniacal N



Total oxidised N



Dissolved reactive phosphorus

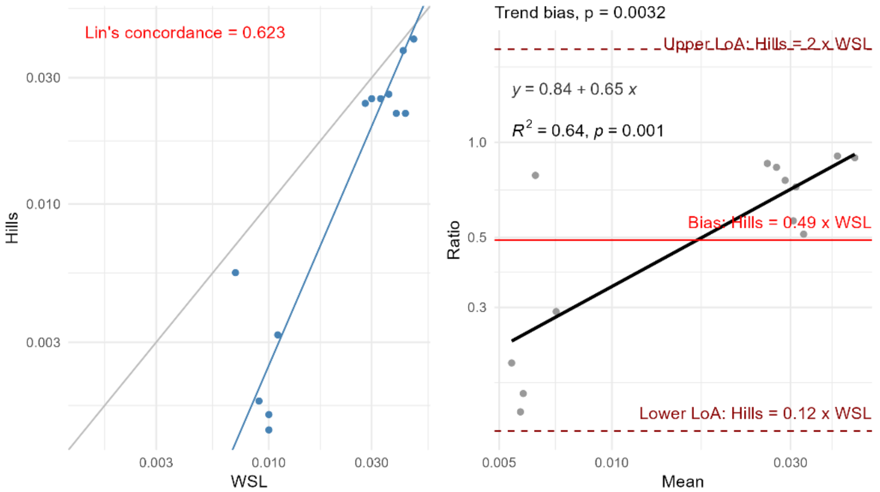
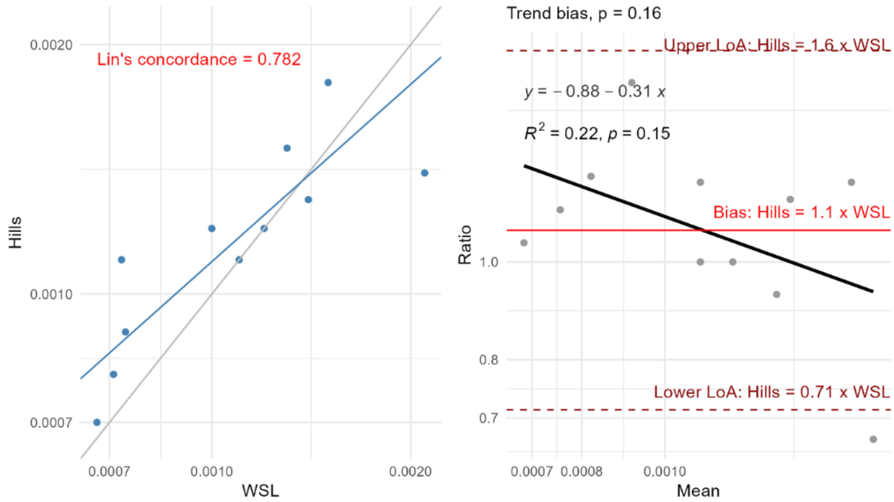
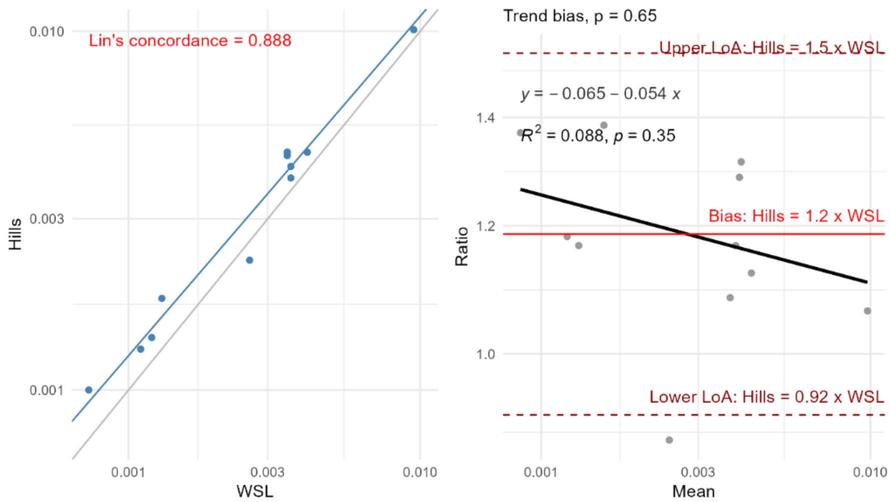


Figure A-0-2. Bland Altman plots for ammoniacal N (top), total oxidised nitrogen (middle) and dissolved reactive phosphorus (bottom).
Left Shows the relationship between the two methods and deviation from the 1:1 line of agreement, including Lin's CCC. Right Shows the difference between methods with their average value. Line and text in red show the mean bias and the upper and lower levels of agreement. The line and text in black shows the fitted Deming regression and p value.
Soluble copper



Soluble zinc



E. coli

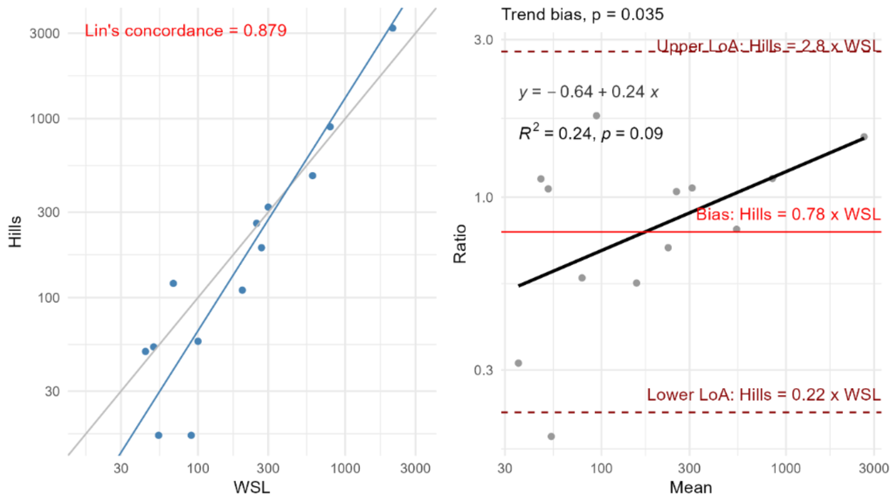


Figure A-0-3. Bland Altman plots for soluble copper (top), soluble zinc (middle) and *E.coli* (bottom).
Left Shows the relationship between the two methods and deviation from the 1:1 line of agreement, including Lin's CCC. Right Shows the difference between methods with their average value. Line and text in red show the mean bias and the upper and lower levels of agreement. The line and text in black shows the fitted Deming regression and p value.

The strong indication of bias from the paired sampling for dissolved reactive phosphorus is clearly visible in the time series for many monitoring sites as a 'step change'. For example, Figure A-0-4 shows an abrupt shift to lower concentrations from July 2017. The influence of this change on trend analysis is also exemplified. Failure to consider the change in methodology would result in a 'very likely improving' trend with apparent decreasing concentrations over the past 15 years. Conversely, if the time period is partitioned before and after this change, both periods would result in 'very likely degrading' trends being identified.

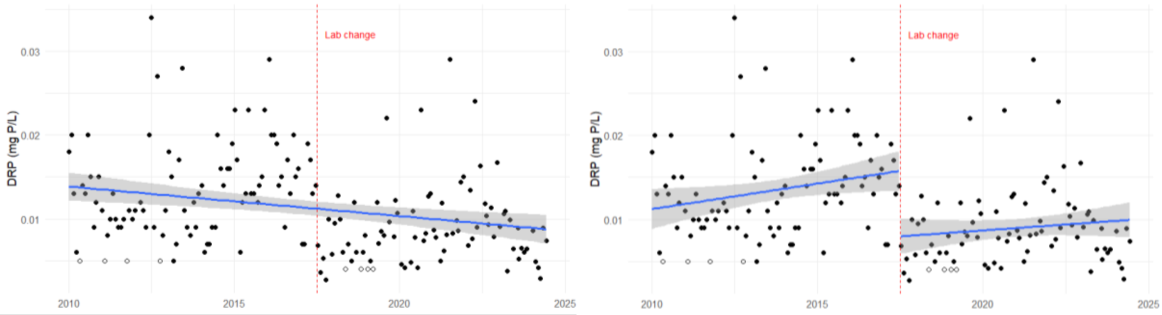


Figure A-0-4. Dissolved reactive phosphorus concentrations observed at Matakana River (Rural-Low) showing the 'step change' to lower concentrations after July 2017 following the change in laboratory provider and analytical method. Left displays the Sen Slope for the entire period 2010-2025 compared to Right the Sen slopes for trend analysis partitioned before and after this change.

Conclusions

The available evidence from paired sampling undertaken in 2017 is limited by the small sample size that does not adequately represent the range of environmental variation expected within the paired sites tested, and by extension the river water quality state of the environment monitoring network in general.

However, the information available suggests that for total oxidised nitrogen, turbidity (NTU) and total copper, no adjustment is necessary to align the old and new methods.

For six parameters, most notably DRP and TSS, there are clear indications of systemic bias between the old (Watercare) and new (Hills) methods. These changes are also clearly visible in time series as 'step changes' from the 1st of July 2017 (see the online Data Explorer). The remaining three parameters showed moderate agreement. However, for all of these parameters, there was also a high degree of random difference between paired samples that cannot be accounted for. Therefore, it is not possible to systematically adjust the data from one laboratory to better match the results from the other.

These issues influenced multiple groups of parameters, across some measures of nutrients, metals, suspended sediment, and bacteria. Consequently, any trend analysis undertaken utilising Auckland Council data bridging this period should be treated with caution. Caution should also be exercised when comparing state assessments before and after this period. This has implications for comparisons between baseline state, which are targeted to the time period before September 2017, and current state.

Appendix 5: Site specific box plot

Site specific variation in water quality parameters can also be viewed through the online interactive Data Explorer https://rimu.shinyapps.io/WQRE_DataExplorer/

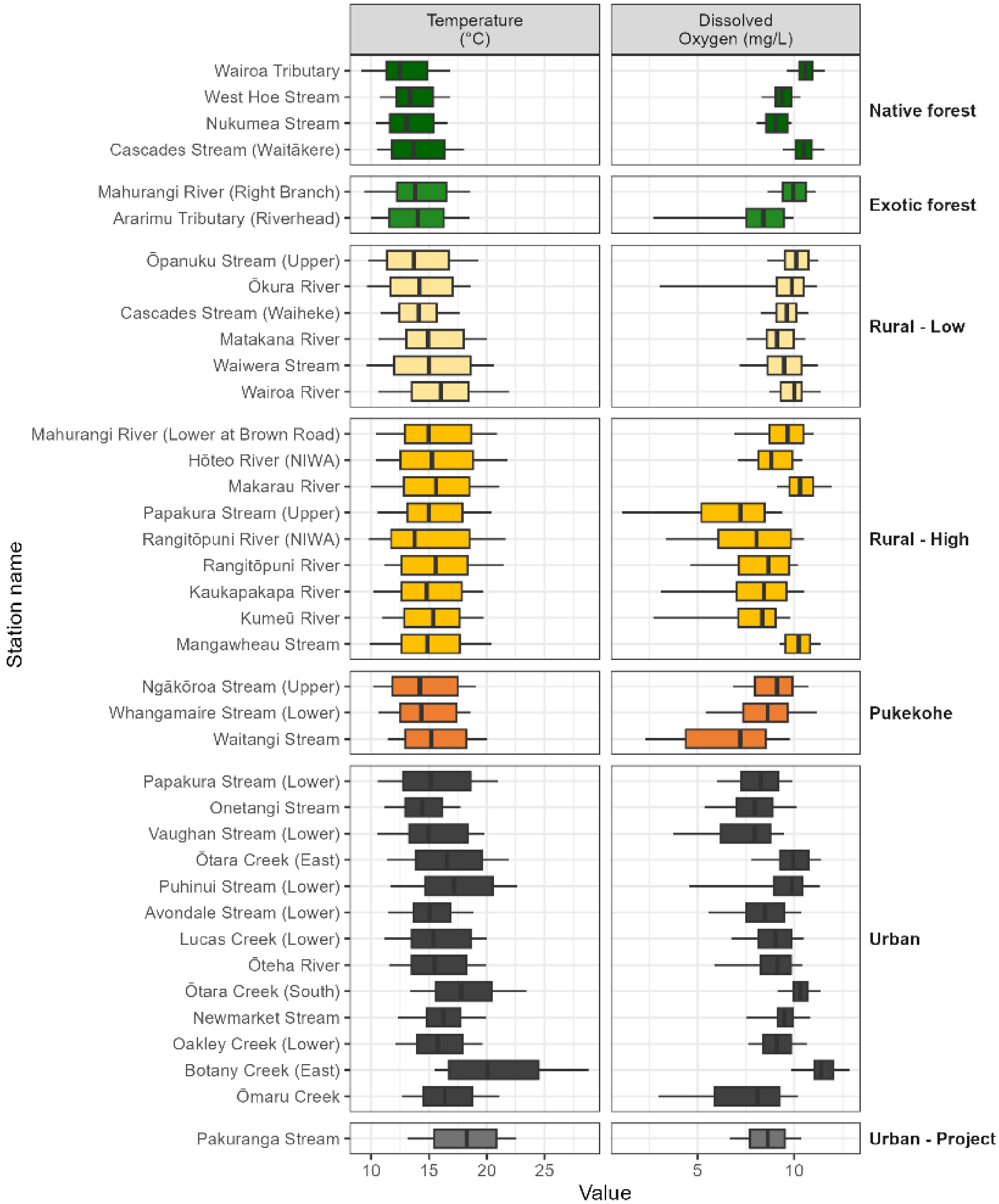


Figure A5-1: Box plots displaying the variation in water temperature and dissolved oxygen at each site for river water quality data collected from 01 July 2019 to 30 June 2024. Whiskers show 5th and 95th percentiles.

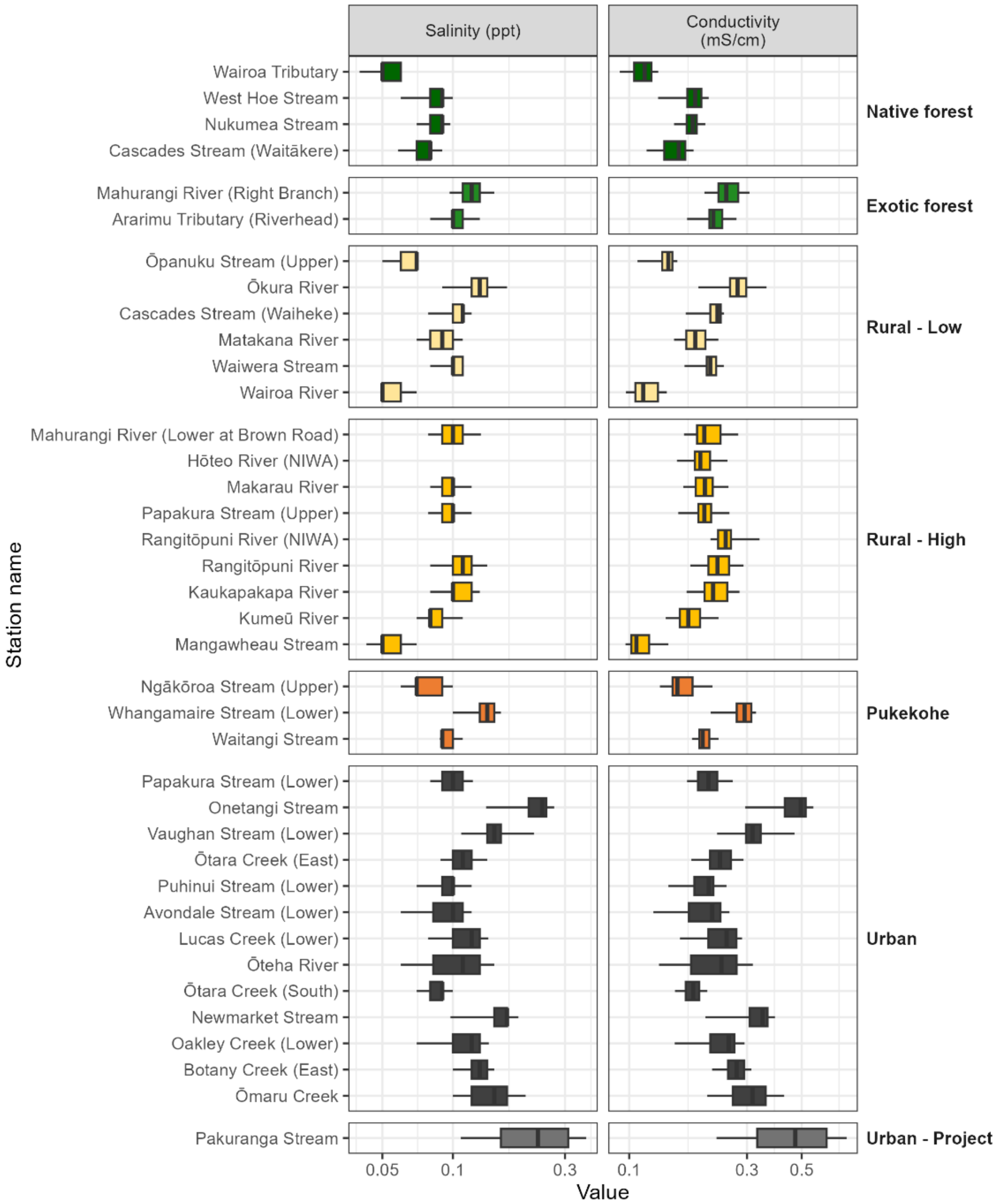


Figure A5-2: Box plots displaying the variation in salinity and conductivity at each site for river water quality data collected from 01 July 2019 to 30 June 2024. Whiskers show 5th and 95th percentiles.

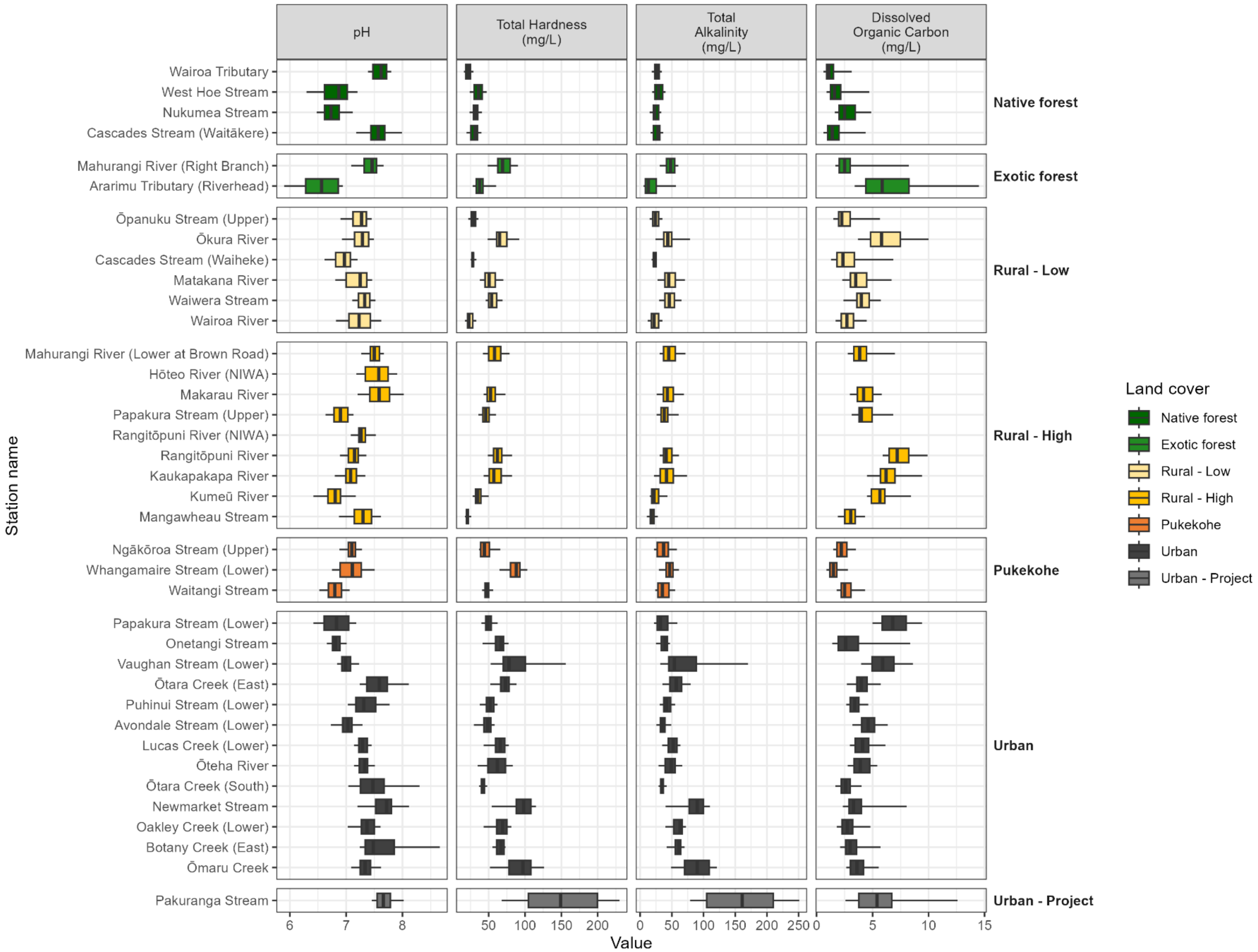


Figure A5-3: Box plots displaying the variation in pH, total hardness, total alkalinity, and dissolved organic carbon at each site for river water quality data collected from 01 July 2019 to 30 June 2024. Whiskers show 5th and 95th percentiles.

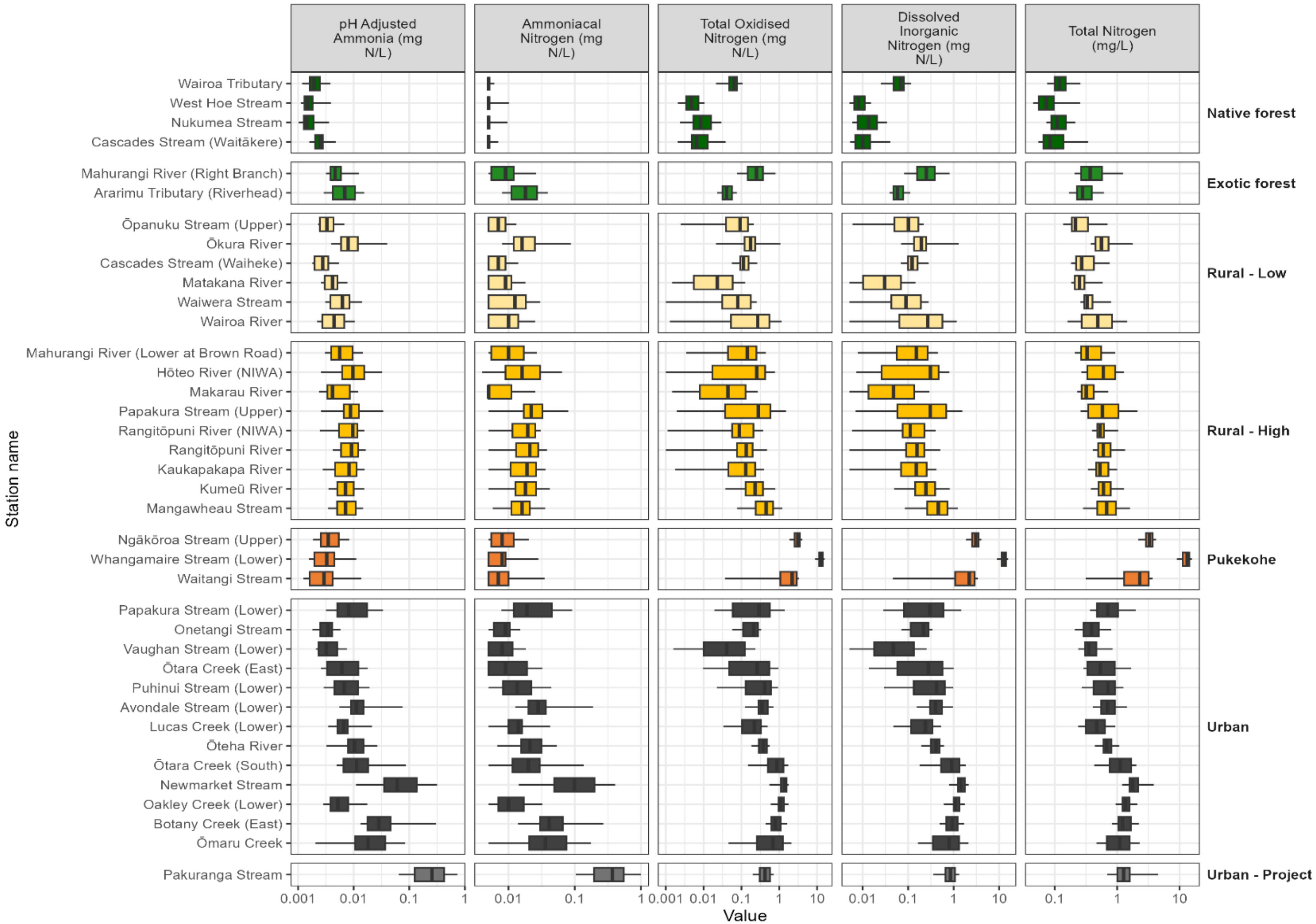


Figure A5-4: Box plots displaying the variation in forms of nitrogen at each site for river water quality data collected from 01 July 2019 to 30 June 2024. Whiskers show 5th and 95th percentiles.

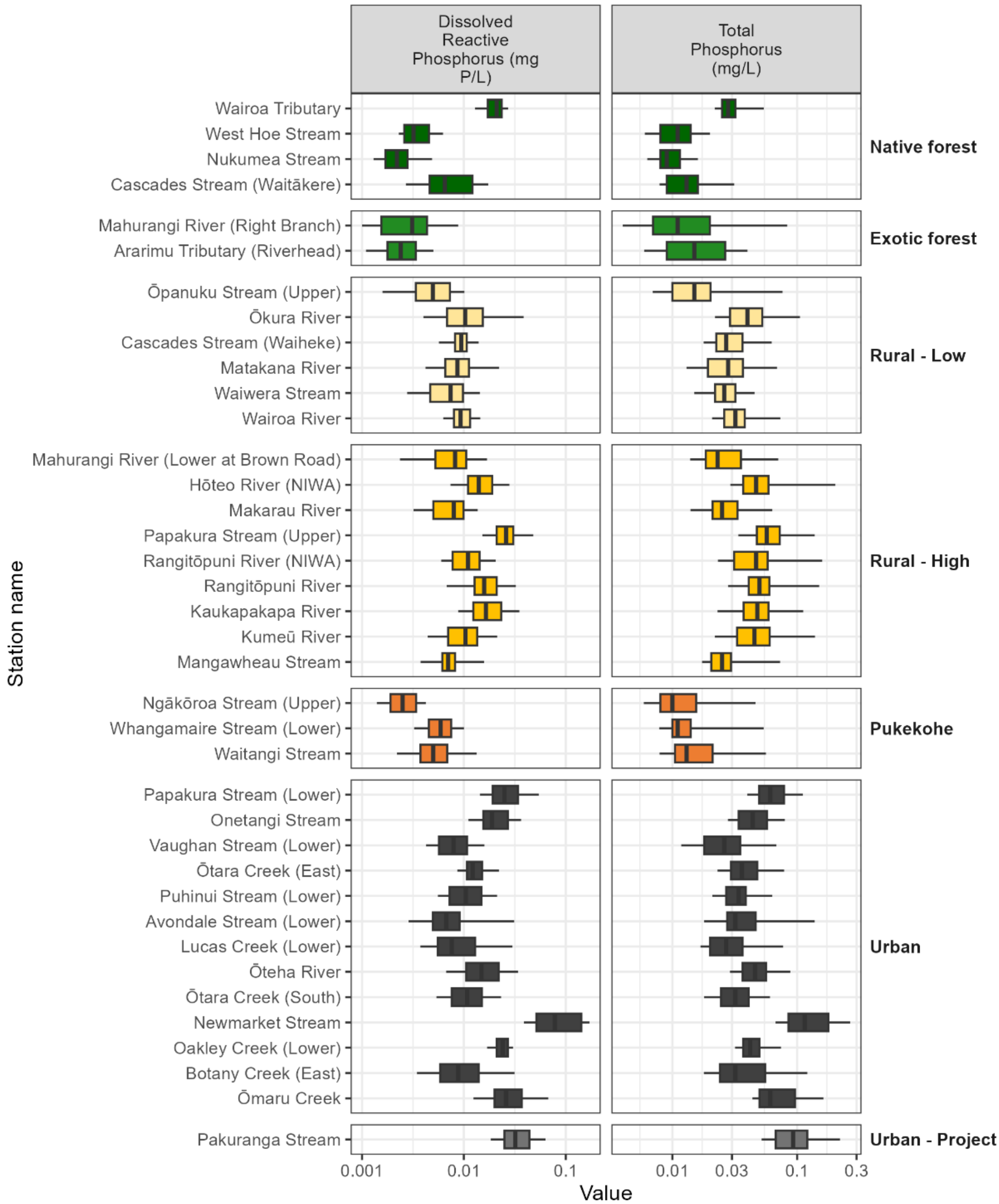


Figure A5-5: Box plots displaying the variation in forms of phosphorus at each site for river water quality data collected from 01 July 2019 to 30 June 2024. Whiskers show 5th and 95th percentiles.

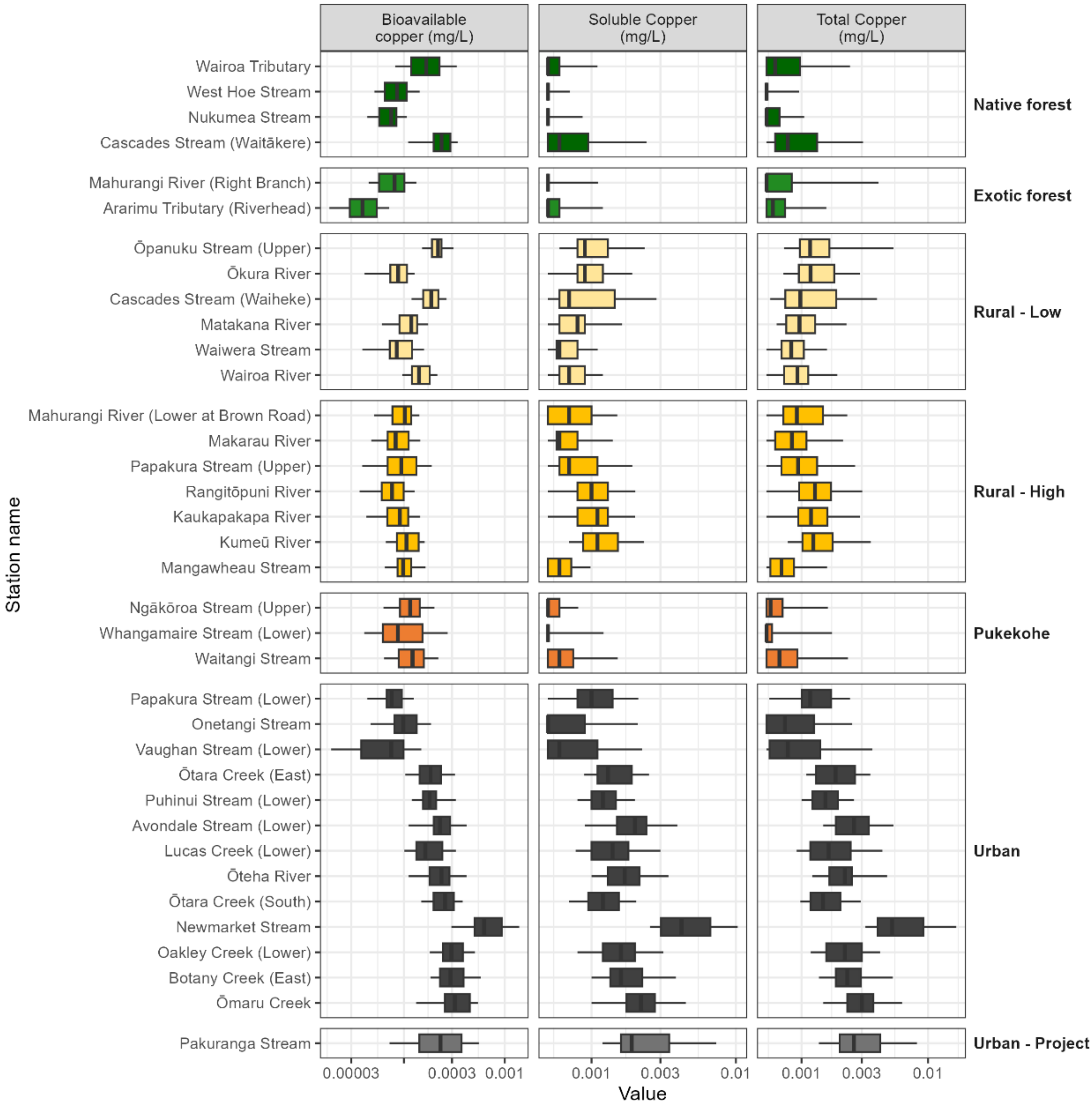


Figure A5-6: Box plots displaying the variation in forms of copper at each site for river water quality data collected from 01 July 2019 to 30 June 2024. Whiskers show 5th and 95th percentiles.

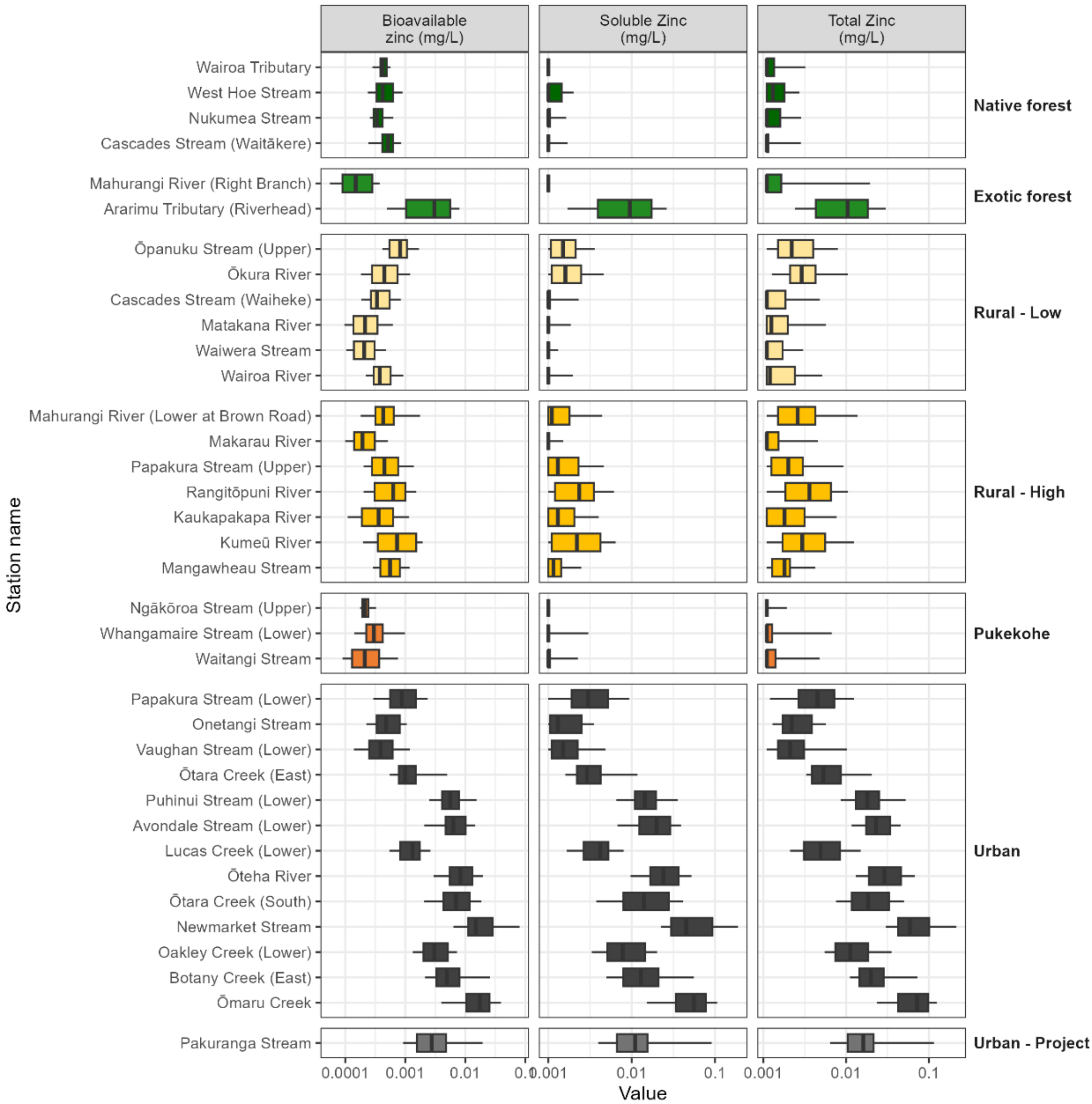


Figure A5-7: Box plots displaying the variation in forms of zinc at each site for river water quality data collected from 01 July 2019 to 30 June 2024. Whiskers show 5th and 95th percentiles.

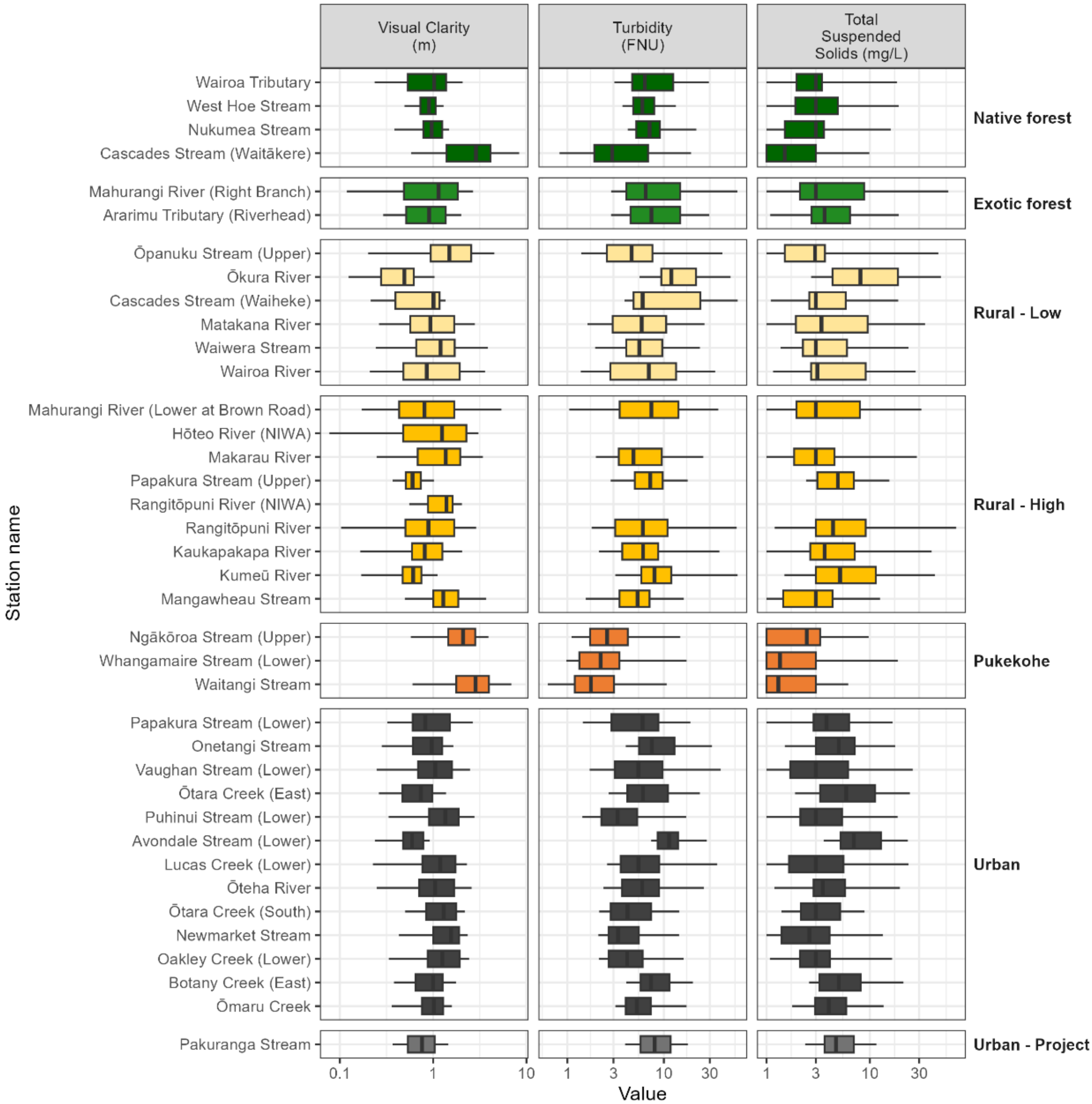


Figure A5-8. Box plots displaying the variation in visual clarity (converted from turbidity FNU), turbidity FNU, and total suspended solids at each site for river water quality data collected from 01 July 2019 to 30 June 2024. Whiskers show 5th and 95th percentiles.

Figure A5-9: Box plots displaying the variation in *E. coli* at each site for river water quality data collected from 01 July 2019 to 30 June 2024. Whiskers show 5th and 95th percentiles. * Note for Hoteo River (NIIWA) *E.coli* is measured in MPN/100 mL not CFU.

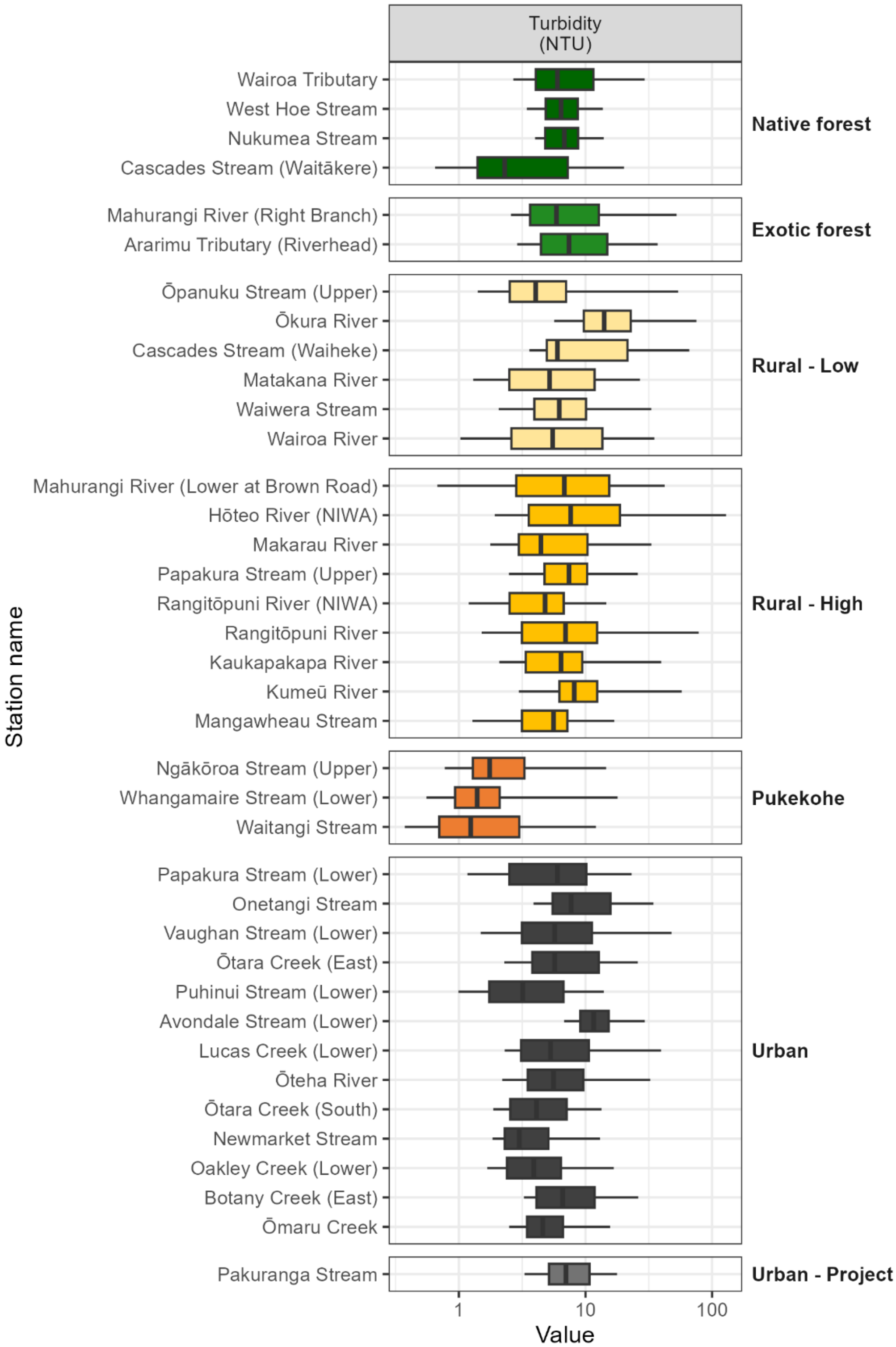


Figure A5-10: Box plots displaying the variation in turbidity (NTU) at each site for river water quality data collected from 01 July 2019 to 30 June 2024. Whiskers show 5th and 95th percentiles.

Appendix 6: Seasonality

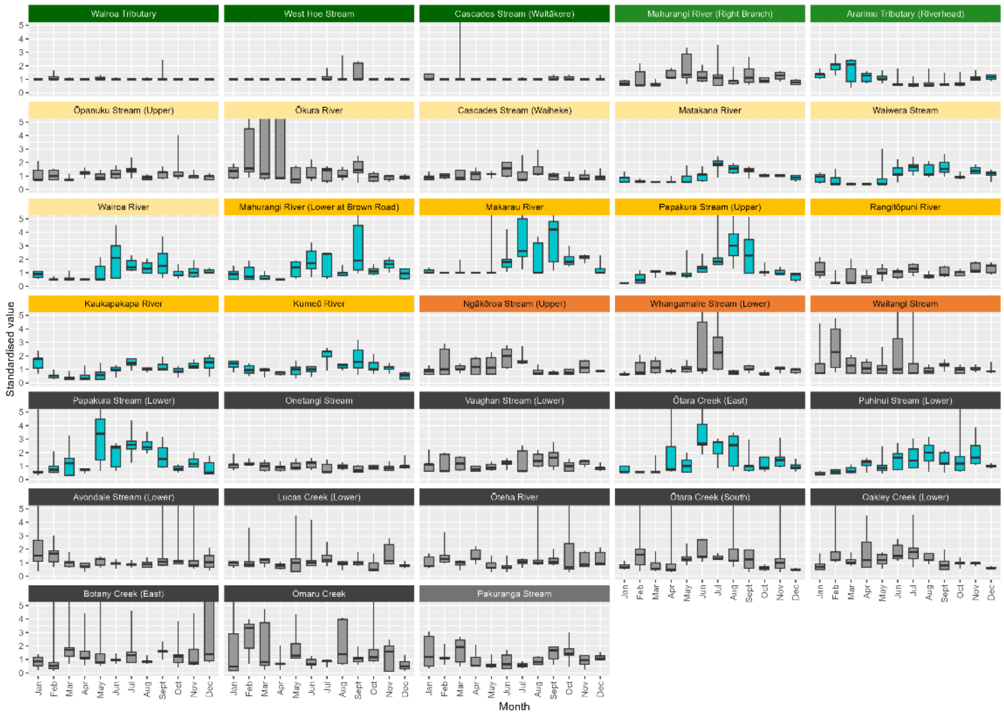


Figure A6-1: Seasonal boxplots of monthly medians standardised by overall median for ammoniacal N for each site based on the past five years (01 July 2019 - 30 June 2024). Site names are coloured by the dominant land cover class (Native forest (Dark Green), Exotic forest (Light Green), Rural – Low (Light Yellow), Rural-High (Yellow), Pukekohe (Orange), Urban (Grey)). Blue boxplots indicate a statistically significant seasonality trend as calculated by the Kruskall-Wallis test (p<0.05).

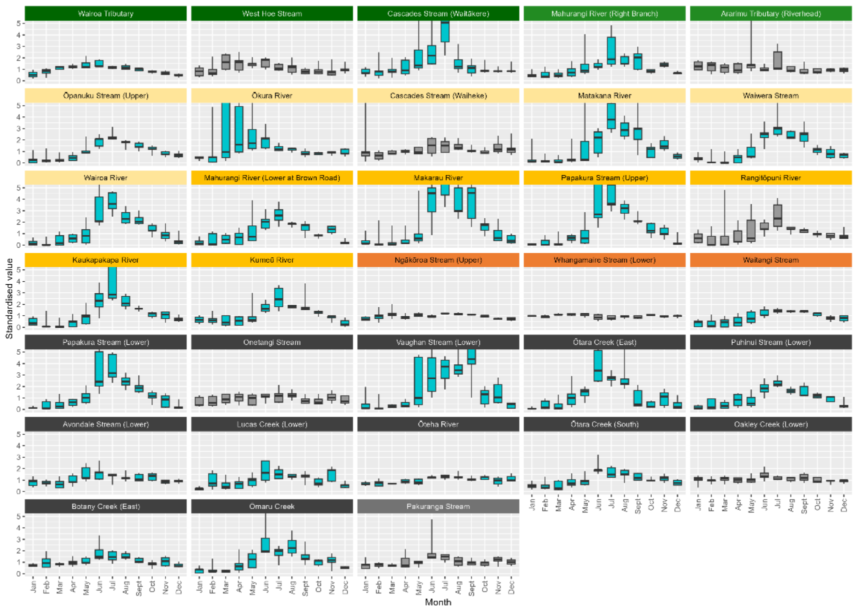


Figure A6-2: Seasonal boxplots of monthly medians standardised by overall median for total oxidised N for each site based on the past five years (01 July 2019 to 30 June 2024). Site names are coloured by the dominant land cover class (Native forest (Dark Green), Exotic forest (Light Green), Rural – Low (Light Yellow), Rural-High (Yellow), Pukekohe (Orange), Urban (Grey)). Blue boxplots indicate a statistically significant seasonality trend as calculated by the Kruskall-Wallis test (p<0.05).

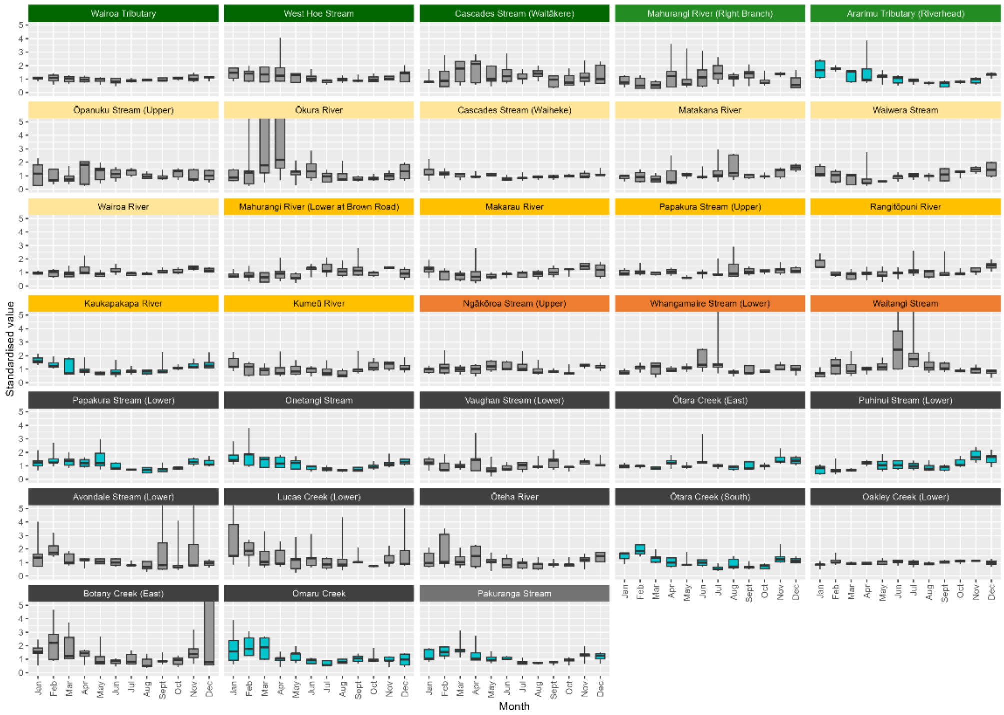


Figure A6-3: Seasonal boxplots of monthly medians standardised by overall median for dissolved reactive phosphorus (DRP) for each site based on the past five years (01 July 2019 to 30 June 2024). Site names are coloured by the dominant land cover class (Native forest (Dark Green), Exotic forest (Light Green), Rural – Low (Light Yellow), Rural-High (Yellow), Pukekohe (Orange), Urban (Grey)). Blue boxplots indicate a statistically significant seasonality trend as calculated by the Kruskall-Wallis test (p<0.05).

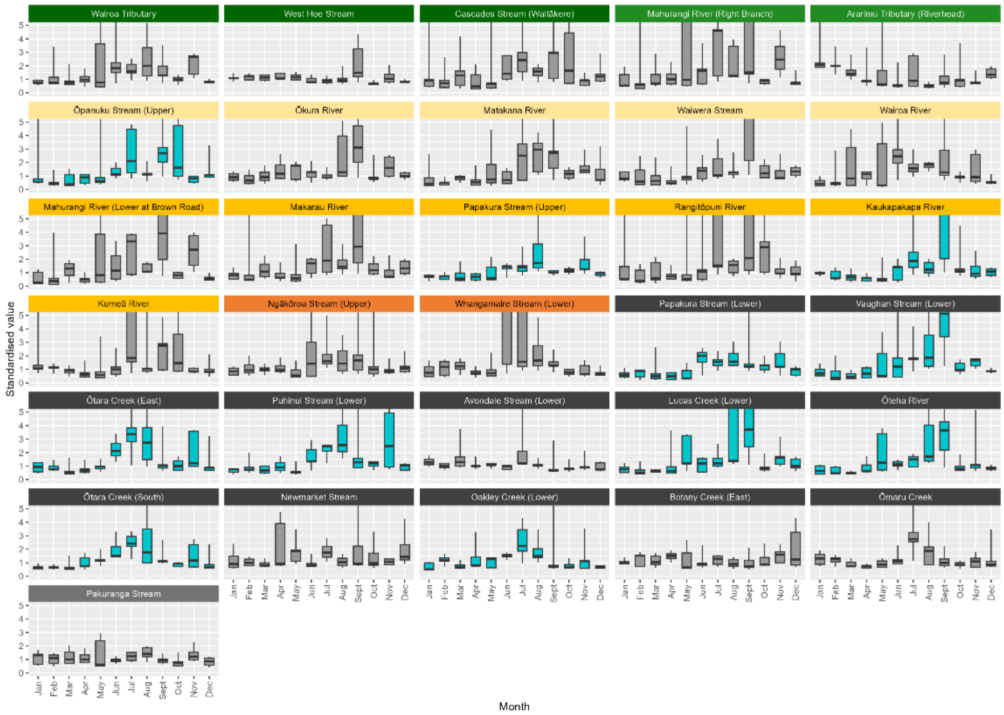


Figure A6-4: Seasonal boxplots of monthly medians standardised by overall median for turbidity (FNU) for each site based on the past five years (01 July 2019 to 30 June 2024). Site names are coloured by the dominant land cover class (Native forest (Dark Green), Exotic forest (Light Green), Rural – Low (Light Yellow), Rural-High (Yellow), Pukekohe (Orange), Urban (Grey)). Blue boxplots indicate a statistically significant seasonality trend as calculated by the Kruskall-Wallis test (p<0.05).

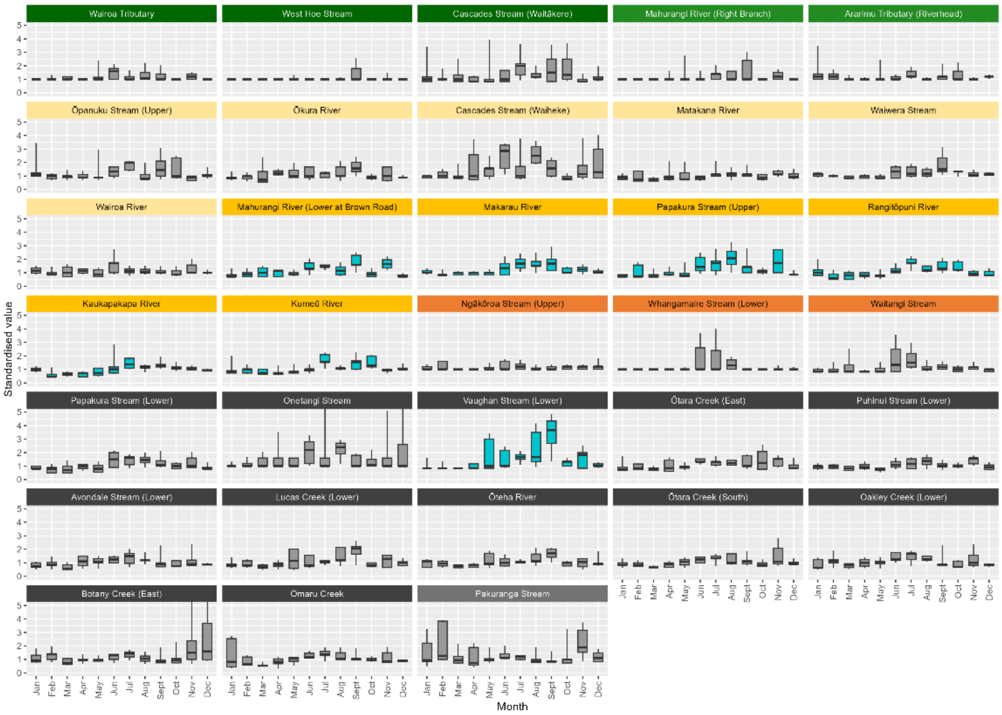


Figure A6-5: Seasonal boxplots of monthly medians standardised by overall median for soluble copper for each site based on the past five years (01 July 2019 to 30 June 2024). The background colours of each stream title indicate land cover type. Blue boxplots indicate a statistically significant seasonality trend as calculated by the Kruskall-Wallis test (p<0.05).

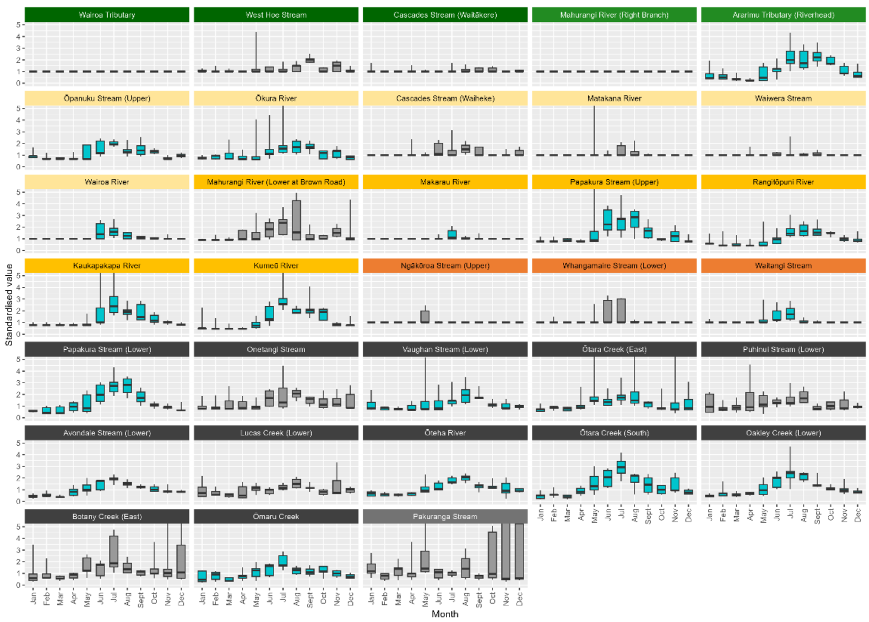


Figure A6-6: Seasonal boxplots of monthly medians standardised by overall median for soluble zinc for each site based on the past five years (01 July 2019 to 30 June 2024). Site names are coloured by the dominant land cover class (Native forest (Dark Green), Exotic forest (Light Green), Rural – Low (Light Yellow), Rural-High (Yellow), Pukekohe (Orange), Urban (Grey)). Blue boxplots indicate a statistically significant seasonality trend as calculated by the Kruskall-Wallis test (p<0.05).

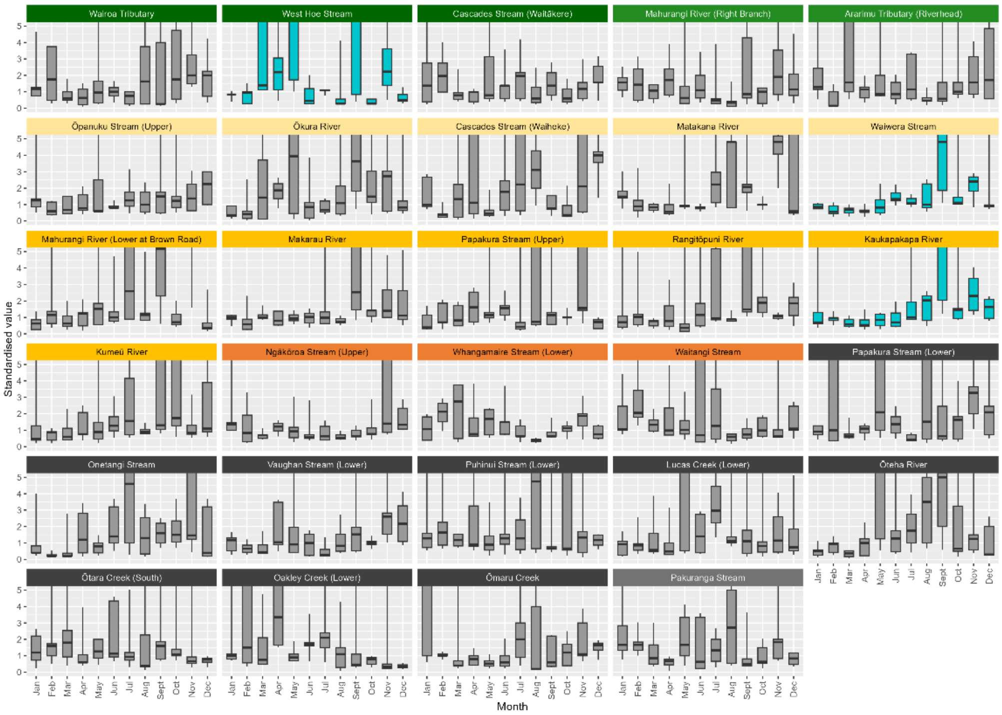


Figure A6-7: Seasonal boxplots of monthly medians standardised by overall median for *E. coli* for each site based on the past five years (01 July 2019 to 30 June 2024). Site names are coloured by the dominant land cover class (Native forest (Dark Green), Exotic forest (Light Green), Rural – Low (Light Yellow), Rural-High (Yellow), Pukekohe (Orange), Urban (Grey)). Blue boxplots indicate a statistically significant seasonality trend as calculated by the Kruskall-Wallis test (p<0.05).

Appendix 7: Temperature state + trend magnitude for the 7-year period

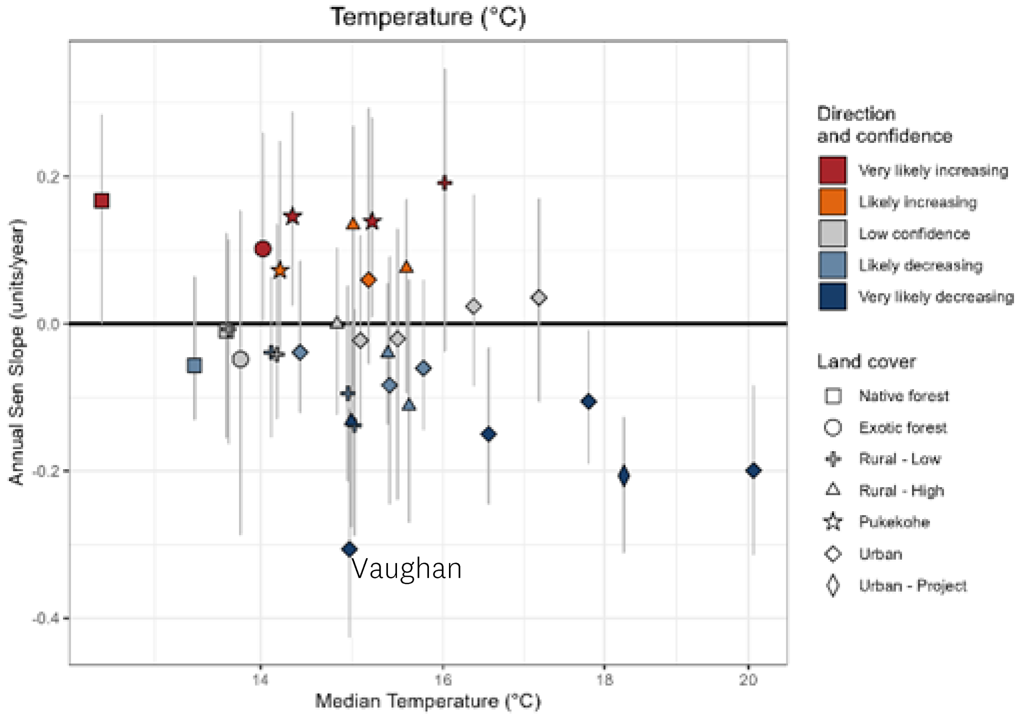


Figure A7-1: State (median) vs 7-year trend rate (Sen slope) for temperature. Whiskers indicate the 90% confidence interval of the Sen Slope.

Appendix 8: Maximum Daily Temperature

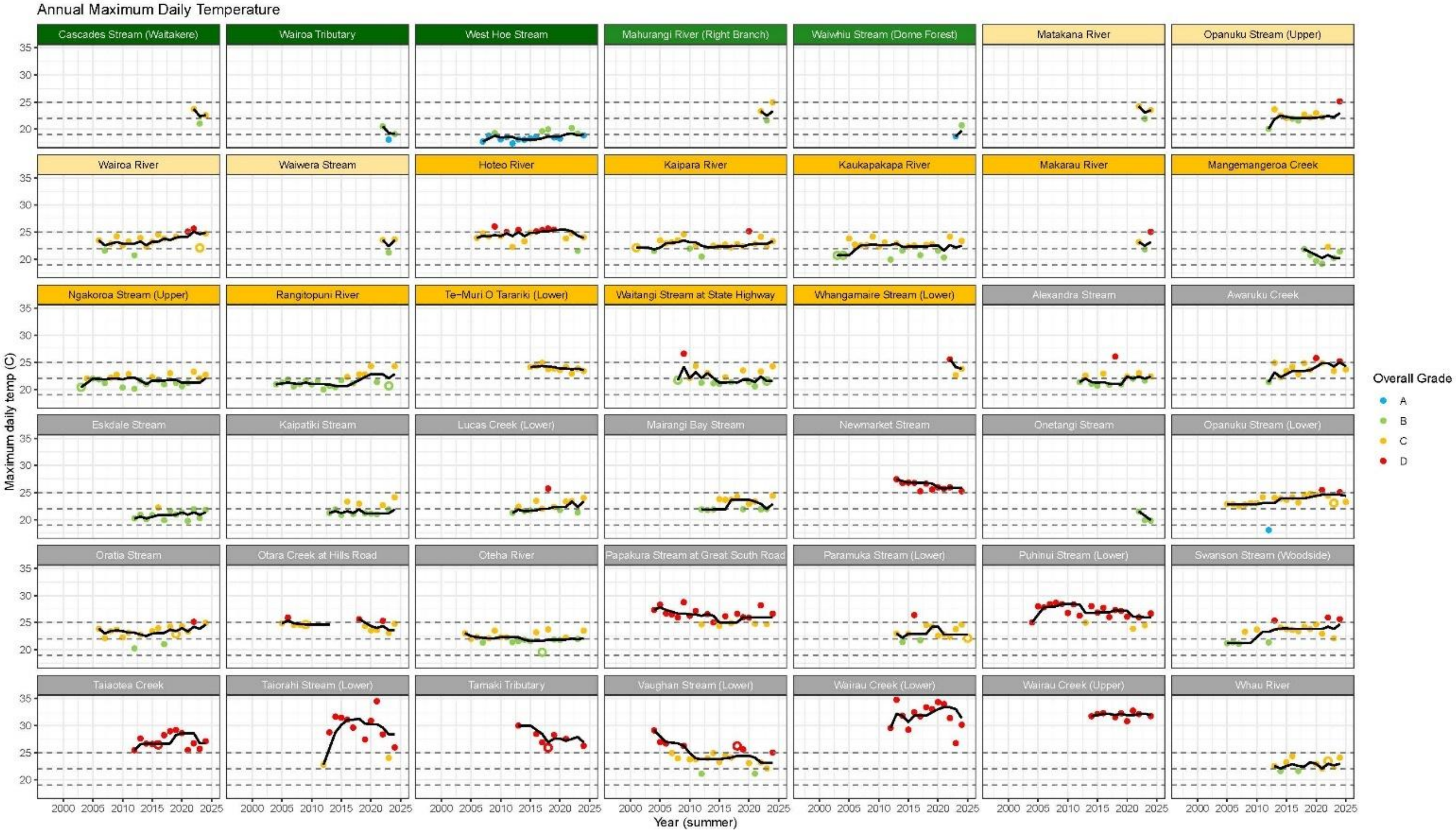


Figure A8-1: Draft from Dikareva (in prep): Mean annual maximum daily temperatures for sites that have continuous temperature records. Black solid lines show the 5-year rolling medians, while dots show the actual values per hydrological year (summer) color-coded by draft regional attribute band for the daily maximum temperature metric based on Table 1 in Clapcott et al. 2015, these thresholds are also marked by dashed lines (A≤19ºC, B≤22ºC, C≤25ºC, D>25 ºC). Note that the regional attribute framework includes assessment of average temperature conditions based on the Cox-Rutherford Index not presented here. Further analysis of continuous temperature information will be available in Dikareva (2025) (IN PREP). Site names are coloured by the dominant land cover class (Native forest (Dark Green), Exotic forest (Light Green), Rural – Low (Light Yellow), Rural-High (Yellow), Urban (Grey)).

Appendix 9: Site specific current state assessment 2024

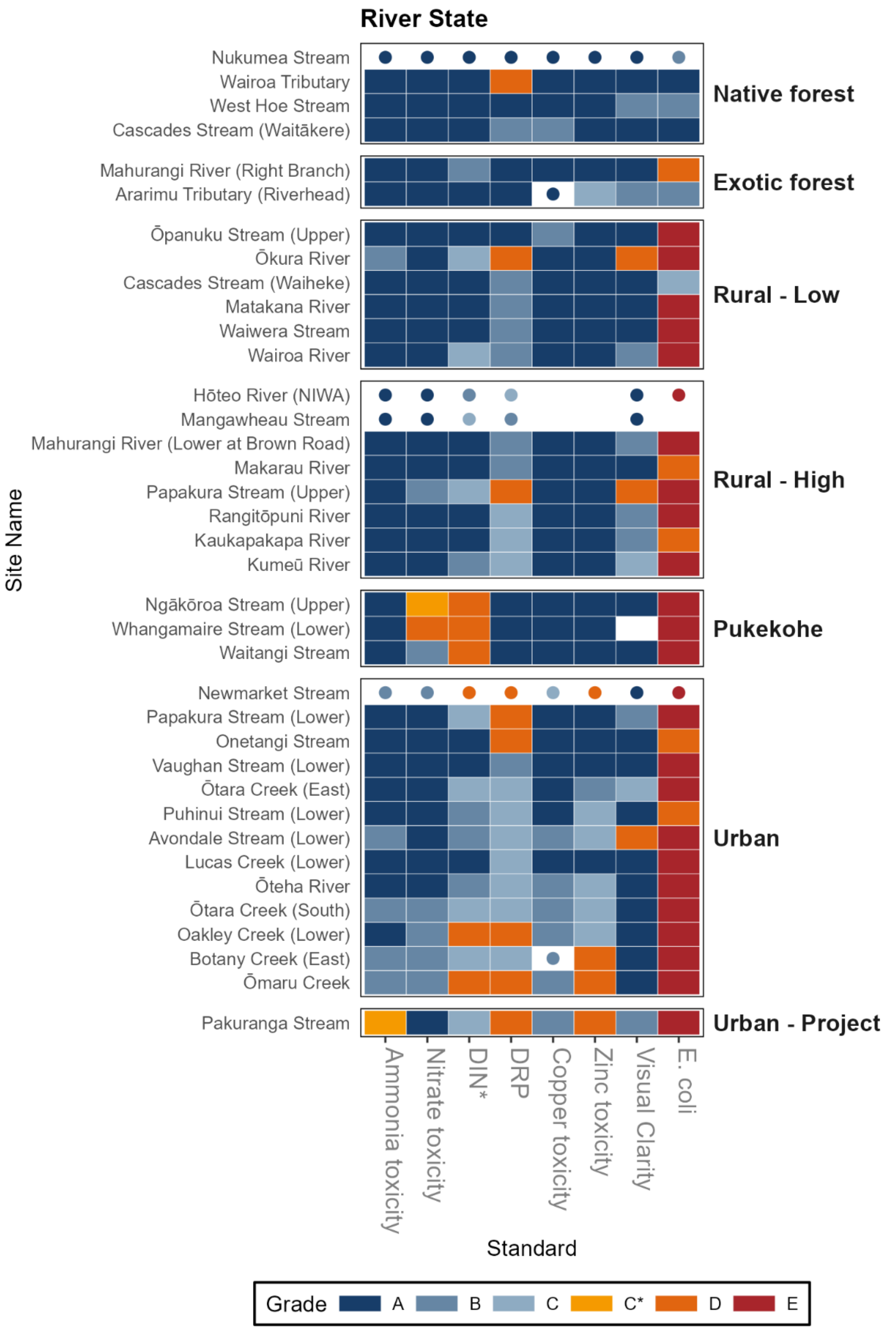


Figure A9-1: Overall band grade for each river site for each attribute for the current state (01 July 2019 - 30 June 2024). Sites are grouped by dominant land cover class and ordered by increasing proportion of anthropogenic land uses. Sites with interim grades are displayed with a circle. Alternative C* colour used for ammonia and nitrate toxicity. E band is only applicable to the *E. coli* attribute.

Appendix 10: Rolling state assessment

Note that rolling overall attribute state is presented for periods bridging the laboratory change. Pre 2017 inclusive is based on Watercare results. 2022 onwards is based on Hills results. 2018 to 2021 assessment includes results from both.

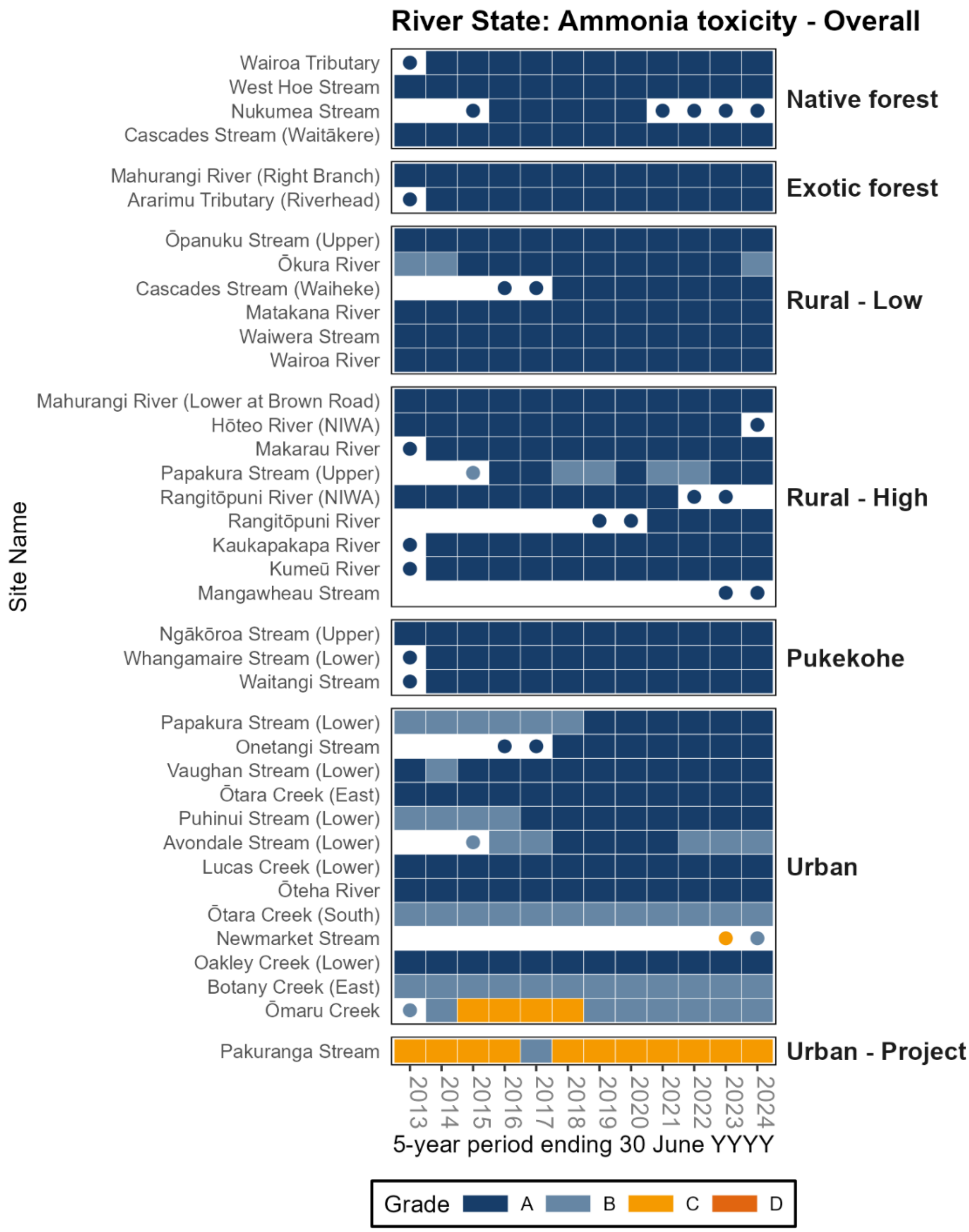


Figure A10-1: Overall band grade for each river site for ammonia toxicity rolling five year periods. Sites are grouped by dominant land cover class and ordered by increasing proportion of anthropogenic land uses. Sites with interim grades are displayed with a circle.

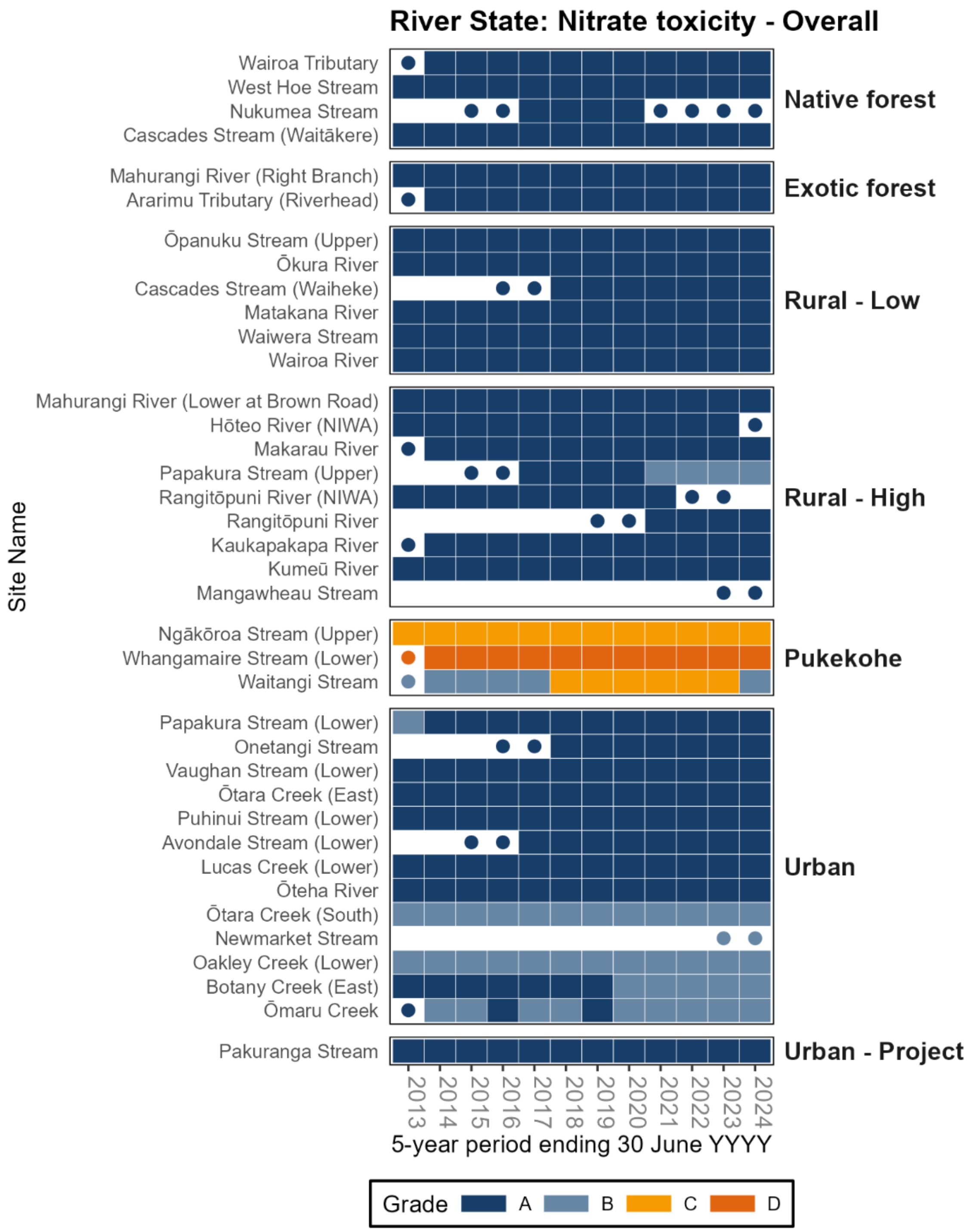


Figure A10-2: Overall band grade for each river site for nitrate toxicity rolling five year periods. Sites are grouped by dominant land cover class and ordered by increasing proportion of anthropogenic land uses. Sites with interim grades are displayed with a circle.

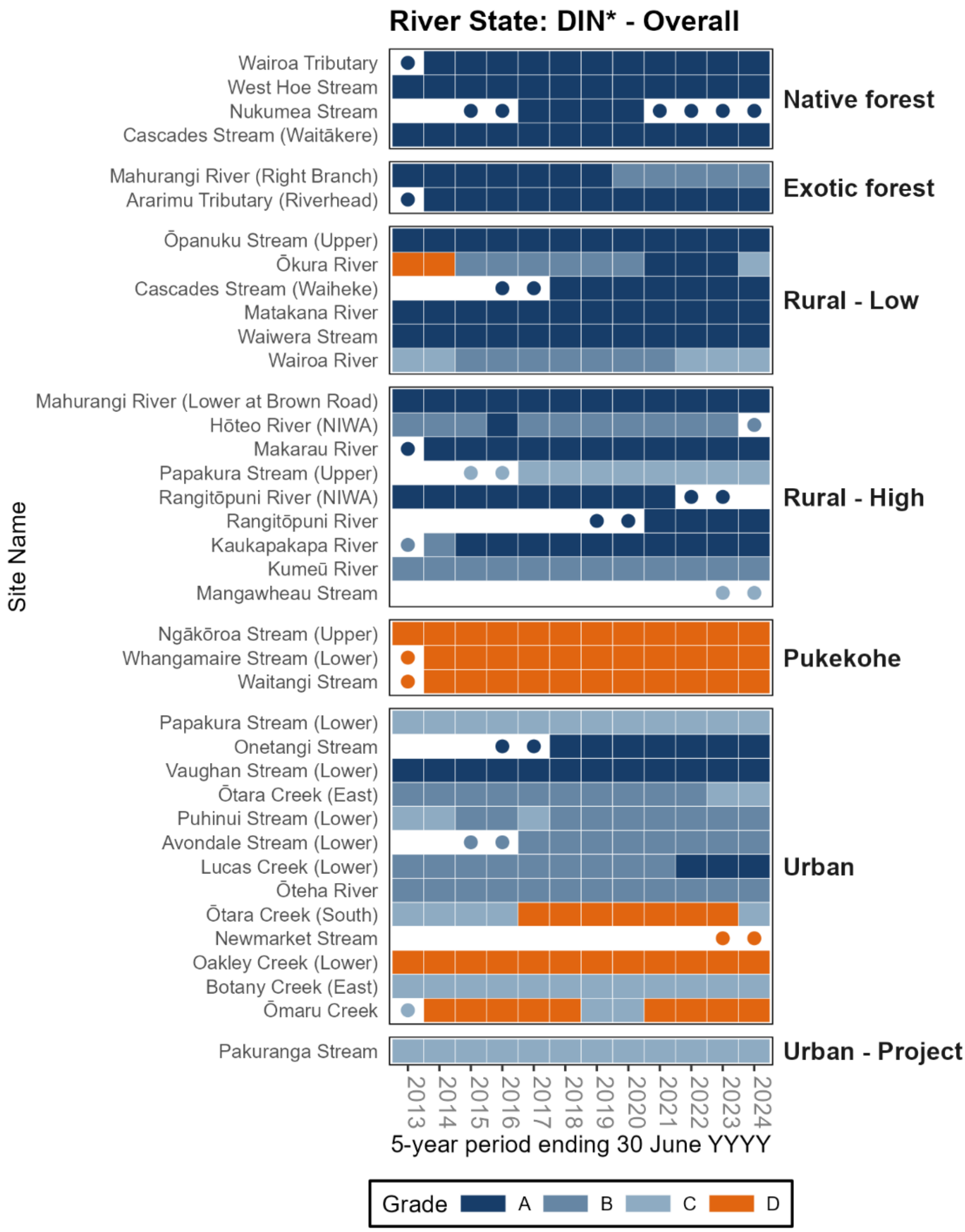


Figure A10-3: Overall band grade for each river site for dissolved inorganic nitrogen rolling five year periods. Sites are grouped by dominant land cover class and ordered by increasing proportion of anthropogenic land uses. Sites with interim grades are displayed with a circle. *DIN draft attribute table (STAG, 2019).

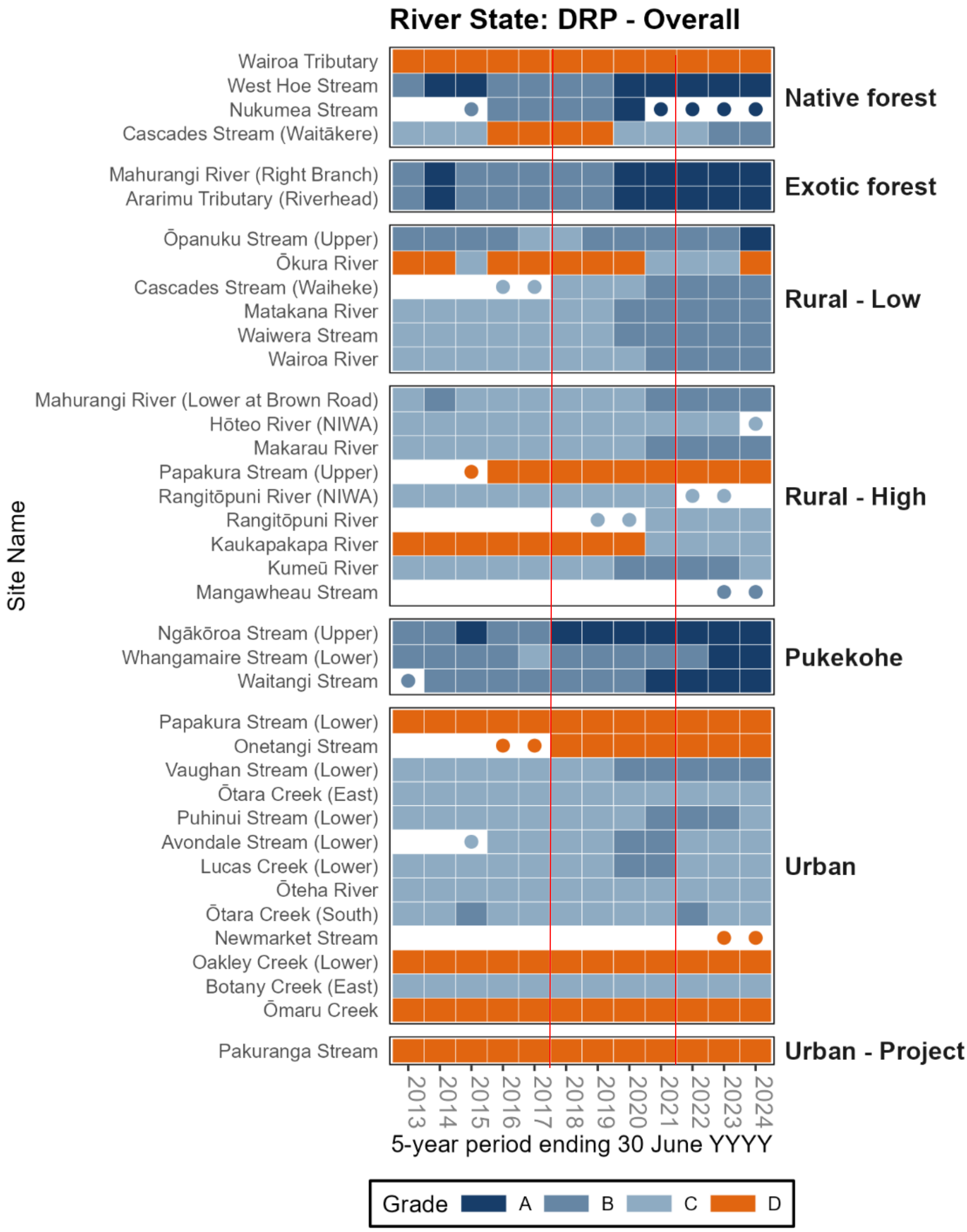


Figure A10-4: Overall band grade for each river site for dissolved reactive phosphorus (DRP) rolling five year periods. Sites are grouped by dominant land cover class and ordered by increasing proportion of anthropogenic land uses. Sites with interim grades are displayed with a circle. Red lines show the period between 2017 – 2022 where rolling five year windows span the 01 July 2017 step change.

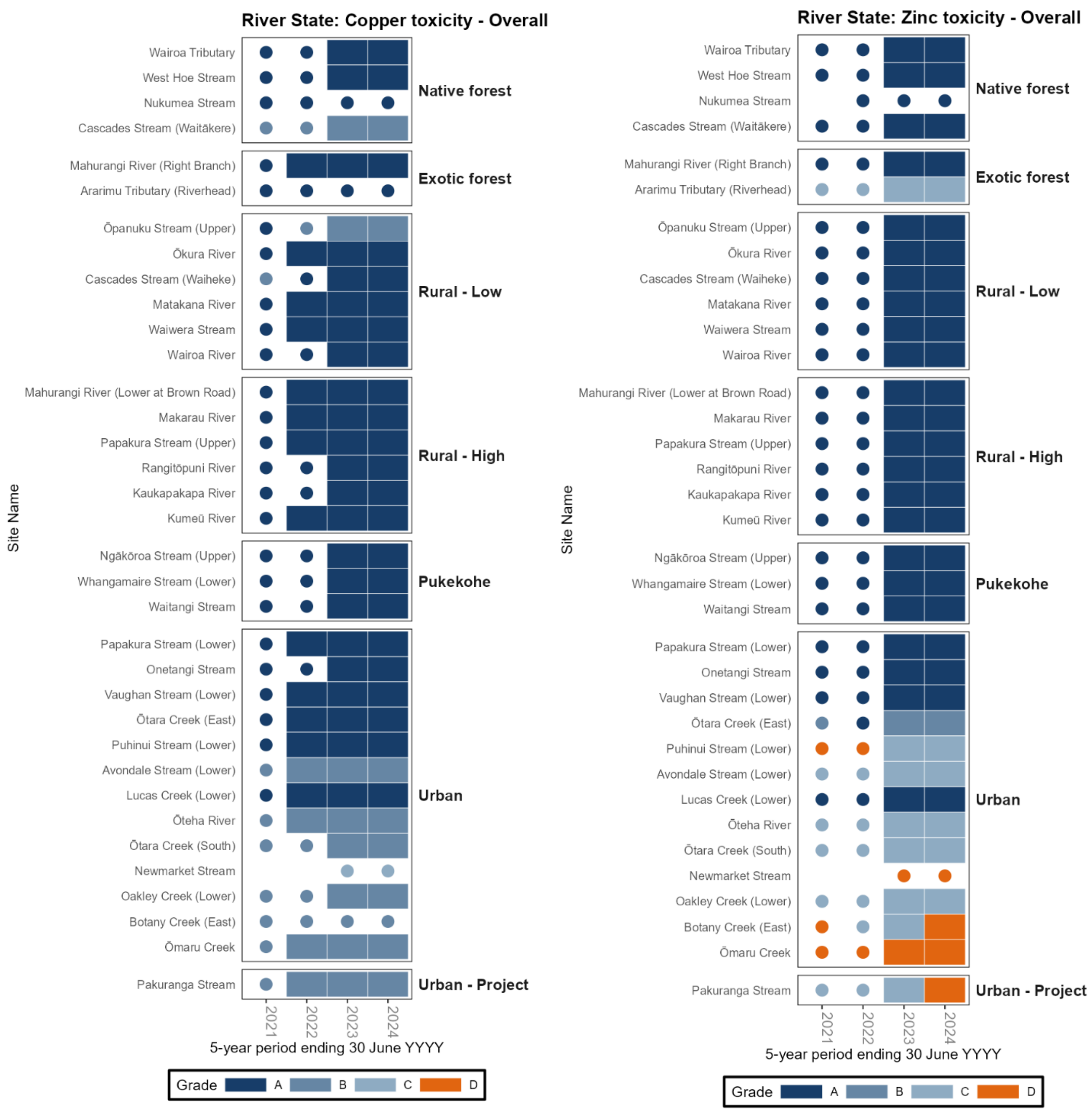


Figure A10-5: Overall band grade for each river site for Copper and Zinc chronic toxicity rolling five year periods. Sites are grouped by dominant land cover class and ordered by increasing proportion of anthropogenic land uses. Sites with interim grades are displayed with a circle.

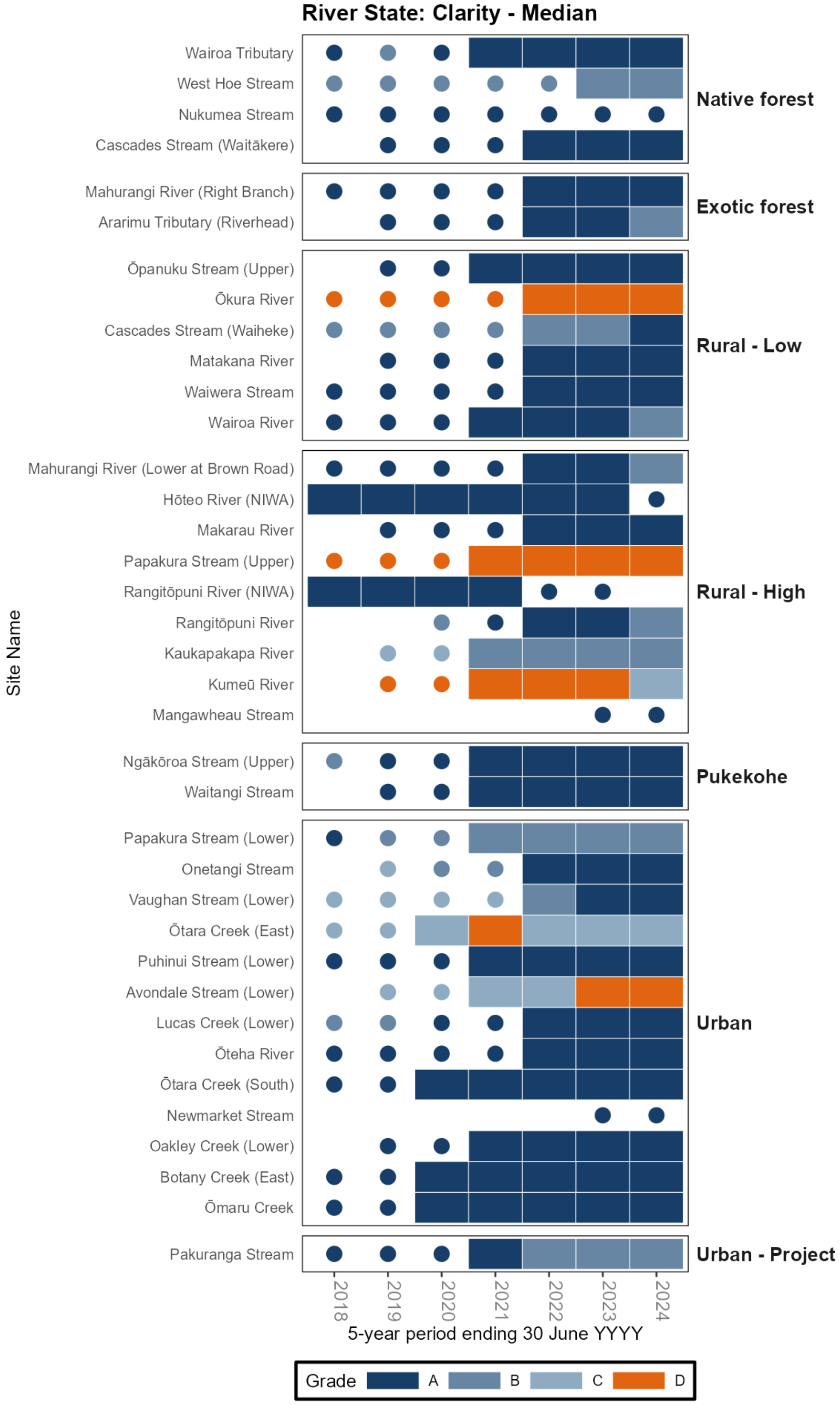


Figure A10-6: Overall band grade for each river site for visual clarity (converted from turbidity (FNU)) over rolling five year periods. Sites are grouped by dominant land cover class and ordered by increasing proportion of anthropogenic land uses. Sites with interim grades are displayed with a circle.

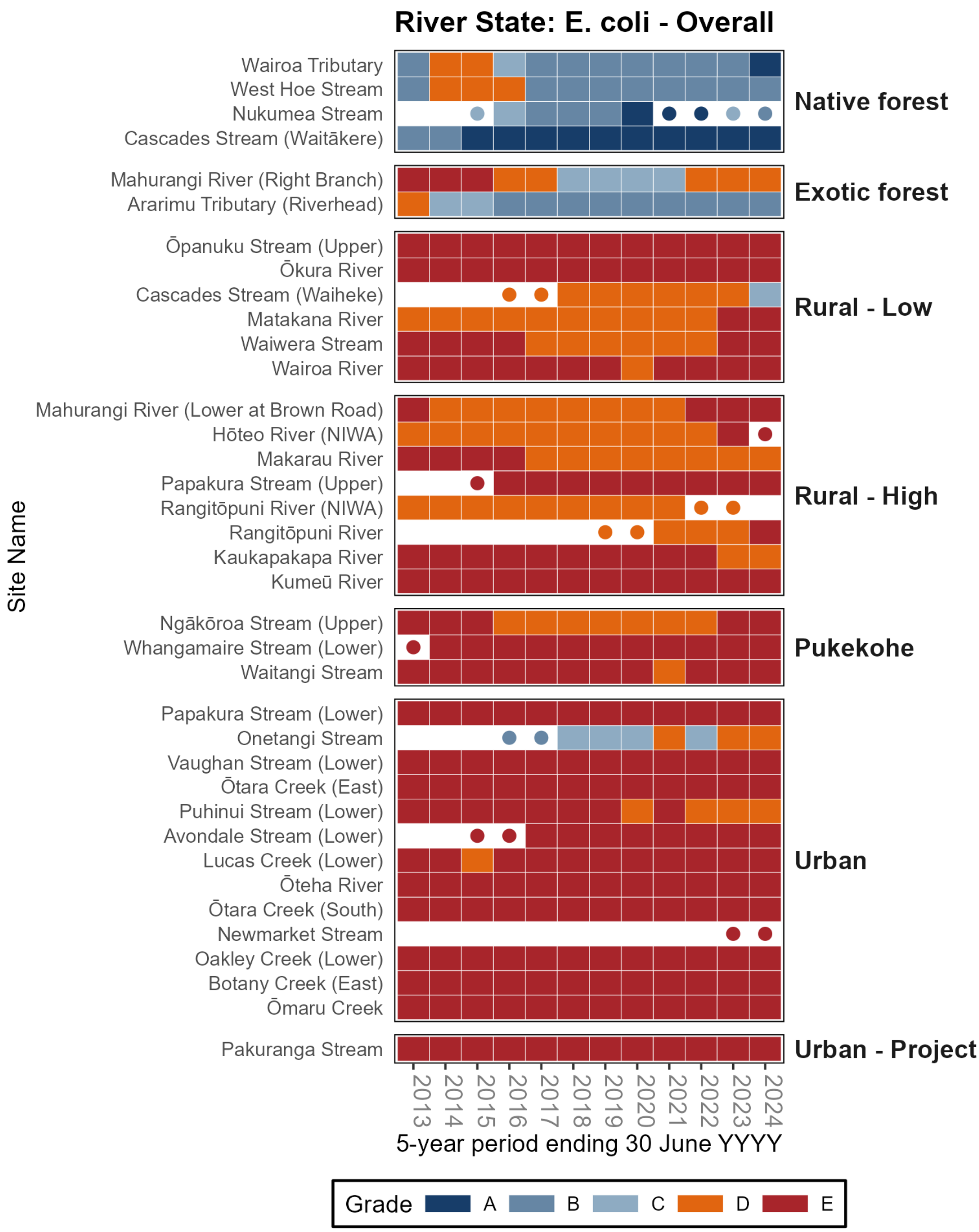


Figure A10-7: Overall band grade for each river site for *E. coli* over rolling five year periods. Sites are grouped by dominant land cover class and ordered by increasing proportion of anthropogenic land uses. Sites with interim grades displayed with a circle.

Appendix 11: Site specific trend assessment (01 July 2017-30 June 2024)

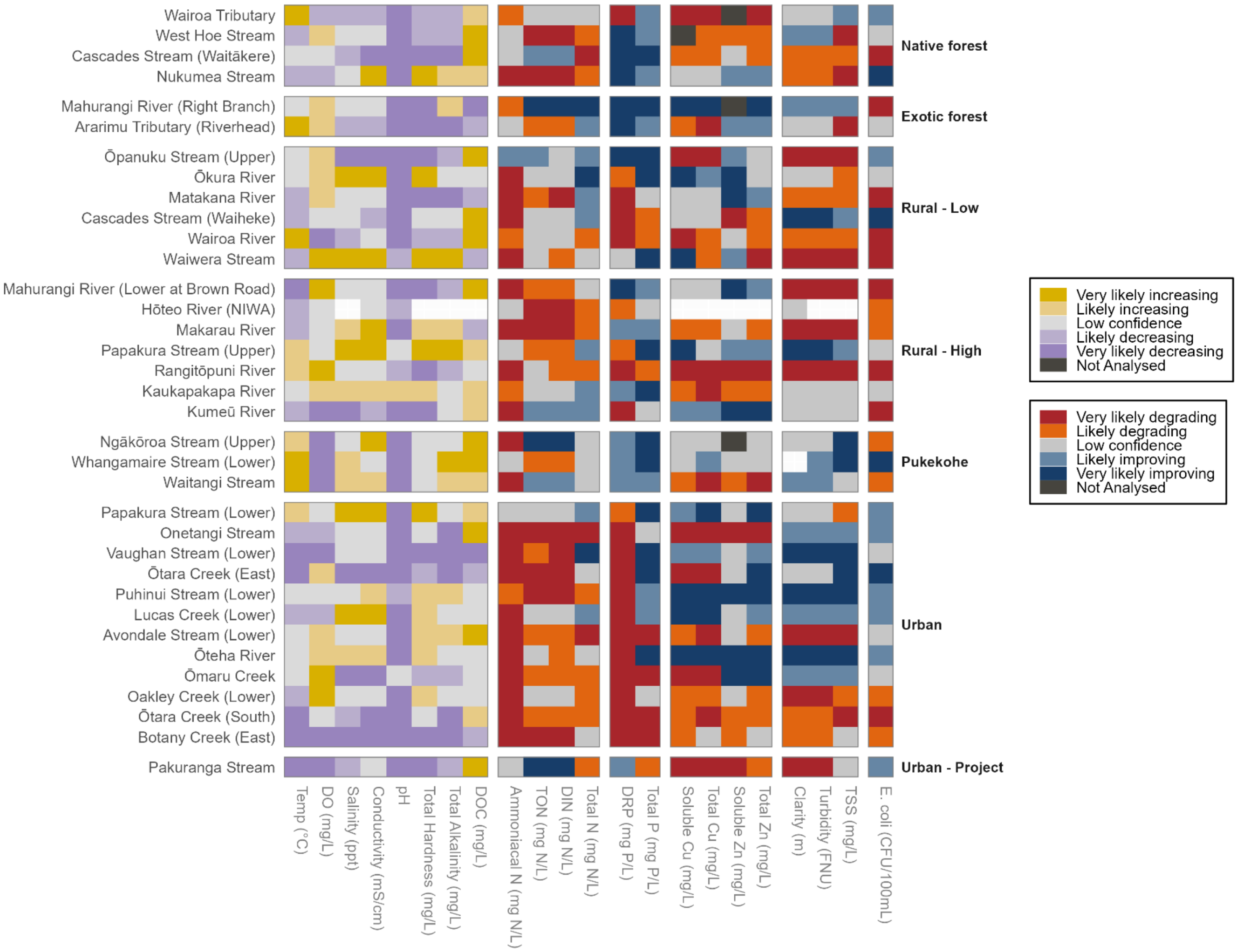


Figure A11-1: Confidence in trend direction for each water quality parameter and each site for the period 01 July 2017 - 30 June 2024. Sites are grouped by dominant land cover class and ordered by increasing proportion of anthropogenic land us.

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