NIWA
Taihoro Nukurangi

New Zealand Coastal Water Quality Assessment

Prepared for Ministry for the Environment

February 2017

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NIWA CLIENT REPORT No: 2016093CH
Report date: February 2017
NIWA Project: MFE165011

Quality Assurance Statement		
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Executive summary

As part of its national environmental reporting programme, the Ministry for the Environment (MfE) is investigating how to best measure and report coastal water quality variables. MfE commissioned NIWA to collate, review and analyse existing coastal water quality data gathered by the 16 regional and unitary authorities. Here we provide state and trend analyses of variables for monitoring and reporting on coastal water quality, based on the variables in current use, where these variable are those that best inform on eutrophication, sedimentation and climate related long-term change. Recommendations are made for future analysis and reporting of coastal water quality data, including thresholds to be used for reporting, communication of trends, data quality and uncertainty in water quality measurements. Recommendations are also made for improving monitoring networks at regional and national levels.

To our knowledge, this work represents the first national-scale compilation and analysis of New Zealand coastal water quality data.

Sections 2 to 4 of this report include detailed methods used for data processing, state and trends analysis and data presentation as a template for further public reporting. The statistical code used is provided as a tool for councils to replicate these methods. We also provide a concise summary of national-scale state and trends. Supplementary files include plots with site-specific 8- and 18-year trend data, and a spreadsheet with spatial data and results of the water quality state and trends analyses for every site that met our criteria for sampling duration and frequency.

Section 5 gives a detailed description of current sampling strategies in councils that contributed data, including an assessment of the national representativeness of regional council coastal water quality datasets. Based on this assessment we provide recommendations for the development of a national sampling strategy in Sections 6 and 7 guided by Tier 1 statistical principles and protocols (Statistics New Zealand 2007). We note that these recommendations were developed in conjunction with the inter-agency working group that is currently preparing the National Environmental Monitoring Standard (NEMS) for discrete water quality sampling and measurement and should be consulted in conjunction with that document (NEMS *in prep*) during development of a national monitoring strategy. Our recommendations are intended to supplement those in NEMS (*in prep*), and focus on core and supporting variables related to ecological health, sedimentation and climate change. Other variables informing contact recreation and shellfish gathering which are commonly collected in council monitoring are considered in the state and trends analysis but we do not focus on them in terms of recommendations on thresholds and monitoring protocols.

Our results show that water quality of coastal waters is strongly affected by land-derived contaminants delivered by rivers, and so correlates broadly with salinity. The summaries of coastal water quality state indicated that salinity was highest and least variable in open coastal waters and declined systematically through coastal hydrosystem types, in order: deep subtidal-dominated estuaries (DSDEs) shallow intertidal-dominated estuaries (SIDEs), shallow, short residence-time tidal river estuaries (SSRTREs), and intermittently closing and opening lagoons (ICOLLs). High freshwater influence coincided with high and variable nitrogen concentrations particularly at SSRTRE sites. Faecal bacteria were highest in the SIDE and SSRTRE classes, and lowest in marine-dominated open coast sites and DSDE class estuaries.

Trends in water quality were examined over two time scales: 2008-2015, that comprised the bulk of our sites (up to 170 sites considered) and 1998-2015 (up to 77 sites considered). Over 2008-2015, most sites where time trends could be confidently detected showed improving trends in nutrients and faecal pollution. In contrast, all sites at which trends were detectable for visual clarity were found to be degrading (declining visual clarity). Microbiological variables, including chlorophyll-*a* (CHLA) and bacterial variables enterococci (ENT) and faecal coliforms (FC), show the greatest improvement in classes with high freshwater influence (SIDE and SSRTRE). Over the 18-year time frame, faecal indicator bacterial concentrations trended downwards across all site classes. Trends in water quality from 1998-2015 were calculated from a smaller dataset than those from 2008-2015 because fewer sites were sampled as far back as 18 years and filtering rules excluded more sites from analyses.

We emphasise that site distribution maps included in this report should be consulted when interpreting state and trend at national scale. There are regional differences in the physical geography of New Zealand coastal hydrosystems and consequently regional differences in water quality. Also, there are large gaps in site coverage nationally and spatial coverage was further fragmented by data filtering rules applied to state and trend analyses. Lack of spatial representativeness in this dataset has likely created bias in national summaries (e.g., median values, percentiles) and trend analyses derived from this dataset.

There is a lack of national consistency in sampling methods across the data set which has created some regional bias in our analyses, such as inconsistent sampling with respect to tidal state at the time of sampling. Also, lack of national consistency in variable selection, as well as field and laboratory analytical methods, resulted in notable data losses.

The lack of spatial representativeness and consistency of sampling methods, arise because the monitoring networks that provided data for this report were designed to address regional and site-specific water quality issues, not for national-scale analyses. However, these issues provide useful lessons for design of a national coastal water quality sampling strategy.

Several time-consuming steps which resulted in discarding of data were employed in our compilation. Such steps would be unnecessary if coastal water quality sampling methods were standardised nationally. We recommend national uptake of the NEMS for coastal water quality sampling (NEMS *in prep*) regarding selection of core and supporting coastal water quality variables, sample collection methods, laboratory analytical methods and methods for reporting data quality and uncertainty to be employed in regional monitoring programmes.

The water quality variables recommended in (NEMS *in prep*) include all of those analysed for state and trends in this report, as well as four additional variables: light penetration, coloured dissolved organic matter (CDOM), colour matching, and *E. coli*. Variables selected for analysis in this report were decided in consultation with council scientists, and were generally those with the most available council data. Hence, overlap between NEMS recommendations and this report are likely to be advantageous for the continuation of long-term monitoring datasets, and future trend analyses. The 14 NEMS variables recommended for coastal waters also show considerable commonality with those (11) recommended for the National Environmental Monitoring and Reporting of rivers (NEMaR; Davies Colley et al. 2012), which may aid in source-to-sea modelling of contaminant flows. Three of the additional recommended variables are optical variables useful for addressing certain values of coastal waters and for developing algorithms to link sea-truth measurements to remote-sensed (i.e., satellite) data.

In summary, we recommend the following protocols for council sampling if the intention is to use council data to manage national coastal water quality:

1. Sites included in a national network should be replicated sufficiently with respect to environmental classes of catchment land use.
2. Sites included in a national network should be split proportionally across hydrosystem types, using the percentages provided in Table 5-4.
3. Nutrients affecting coastal hydrosystems should be assessed by monitoring water quality in terminal river reaches, within estuaries and on their adjacent coasts.
4. There should be unified use of the NEMS core water quality variables listed in Table 6-2.
5. An integrated index of hydrosystem ecological health should be included in future state and trend analysis to facilitate setting of water quality thresholds (i.e. boundaries between bands of environmental state) and increase the utility of monitoring.
6. There should be unified use of NEMS protocols with regard to metadata collection, reporting of measurement uncertainty and quality coding.
7. There should be unified use of NEMS protocols with regard to water sample collection and analytical methods.
8. Reporting uncensored data values by laboratories is strongly recommended.
9. The setting of water quality thresholds should account for characteristics of different hydrosystem types – some hydrosystem types are more sensitive to stressors than others.
10. We would not recommend using the current dataset for threshold setting using a percentile-based approach because 1) the dataset is not representative of water quality conditions in New Zealand coastal hydrosystems nationally, for the reasons laid out in Section 5, and 2) we currently do not fully understand how levels for each water quality variable relate to values (such as ecosystem health).
11. We recommend that thresholds for water quality and contaminant loads are set by comparing hydrosystem water quality with scores of ecosystem health and other values.
12. We recommend further development of relationships between contaminant loading rates, water quality, and hydrosystem ecological health to inform water quality threshold setting.
13. The state and trend results in this report are most appropriate as 'case study' indicators of coastal water quality for national reporting.

1 Introduction

As part of its national environmental reporting programme, the Ministry for the Environment (MfE) is investigating how to best measure and report coastal water quality variables[1]. MfE commissioned NIWA to collate, review and analyse existing coastal water quality data gathered by the 16 regional and unitary authorities to provide:

- The water quality variables and protocols that are currently used in coastal water quality monitoring programmes.
- The current state and recent temporal trends in these variables at sites across New Zealand using recommended data analysis and reporting methods.

MfE also commissioned NIWA to provide recommendations on future analyses of coastal water quality data and national reporting procedures, including:

- Improving monitoring networks at regional and national levels, in order to accurately assess water quality in three broad types of coastal waters: estuaries, harbours and open coasts.
- A list of core and supporting variables for monitoring and reporting on coastal water quality, based on the variables in current use. These variables will be those that best inform on eutrophication, sedimentation and climate related long-term change.
- Recommendations for other variables that would be useful to build more robust coastal water quality programmes.
- Thresholds to be used for reporting for each core variable.
- How to characterise and communicate trends for each of the core variables in a meaningful way for the general public.
- How to quantify and communicate data quality.
- How to quantify and communicate uncertainty in a meaningful way for the general public.

The report consists of detailed methods for data processing and analysis, a concise summary of state and trends for sites nationally where sufficient data exist, a review of current sampling network design and procedures, and recommendations for future national sampling and reporting strategies informed by analysis of the current data. The report is accompanied by several files: a file of all data compiled from councils, its associated metadata, site-specific state and trend results in excel format, trend analysis plots for each site, and the full R-script used in the analysis of data and production of figures herein.

The methods for data management and analysis used in the current study follow those used in recent national-scale freshwater quality reporting (Larned et al. 2015): 1) water quality measurements that were reported by the data suppliers to be "below detection limit" (left-censored) or "above reporting limit" (right-censored) were replaced with randomised imputed values; and 2) assessments of trends used confidence intervals to determine trend direction, and report trend magnitude for those cases where trend direction was identified. This second point facilitates

[1] http://www.stats.govt.nz/browse_for_stats/environment/environmental-reporting-series/environmental-indicators/Home/About.aspx#topics

subsequent assessment of trend importance when coastal water thresholds are established for the selected variables.

In this report we use a method modified from the procedure of Larned et al. (2015). The initial process of this method new procedure (used also in this study) tests the *direction* of a trend rather than the existence of a trend. If the direction of a trend cannot be confidently inferred, the result is stated as “insufficient data to reveal the trend direction”, rather than “not statistically significant”. The new procedure prevents the common misinterpretation of a trend test result that fails to attain statistical significance when testing the “nil hypothesis”—that conditions are “stable” or “being maintained”. If a trend direction can be inferred, we go on to report its magnitude. Subsequently the importance of a trend may be determined by estimating time to reach a recognised threshold toward which concentrations may be heading, such as a bottom line under the National Policy Statement for Freshwater Management (NPS-FM).

In section 5, we give a review of data collection by regional council monitoring programmes which informs on the limitations of data collated by this report for national reporting, and priorities for future monitoring. We also provide a summary of non-council data sources that may be suitable for future national analyses. The recommendations in Sections 6 and7 of this report are intended to help resource managers in two ways: 1) evaluating water quality state and trends with respect to national objectives; and 2) developing and operating coastal water quality monitoring programmes that produce data needed to evaluate local issues (e.g., impacts of land use on individual coastal hydrosystems. Our recommendations focus on metrics related to ecological health, sedimentation and climate change. Other metrics informing contact recreation and shellfish gathering which are commonly collected in council monitoring are considered in our state and trends analysis but we do not focus on them in terms of recommendations on thresholds and monitoring protocols. This report was written during the development of the National Environmental Monitoring Standards (NEMS) for discrete water quality sampling and measurement (NEMS, *in prep*), and we include recommendations for national sampling and reporting strategies that supplement those provided in the NEMS programme. We do this by:

- Basing our recommendations on deficiencies in the dataset compiled for this report when it is used for assessing national state and trends in coastal water quality.
- Comparing methods recommended by the NEMS programme to those used in existing monitoring programmes that provided data for this report.
- Focussing on aspects of a national monitoring strategy that are not covered in detail in the NEMS programme, including recommendations for network design for national state and trend analysis, and threshold development for national coastal water quality variables.

The recommendations in this report are intended to be read in conjunction with those of the NEMS (*in prep*) - as soon as that is available - during development of a national coastal water quality monitoring network.

2 Data acquisition, organisation and processing

New Zealand regional and unitary councils carry out water quality monitoring at > 300 open coastal and estuarine sites (Figure 2-1). For the monitoring sites used in this report, monthly or quarterly monitoring has been underway for 5 to 33 years. A variety of physical, chemical and biological variables are measured at these sites. In addition, water quality monitoring has been carried out by Invercargill City Council (ICC) since 1976 at sites in the New River Estuary. Sampling has been conducted monthly at high and low tide at nine sites since 1991. Recognising the high quality of the ICC data and the paucity of other information from Southland, we included the ICC data from 1991 to present in this study.

Council coastal and estuarine monitoring data have not been periodically acquired and analysed for national-scale state-of-environment reporting as is the case for New Zealand rivers and lakes (e.g., Sorrell et al. 2006, Ballantine et al. 2010, Larned and Unwin 2012, Larned et al. 2015). To our knowledge, the current project represents the first national-scale compilation and analysis of New Zealand coastal water quality data. In this section we describe the water quality variables, data sources and organisation of the coastal water quality data, and explain the data processing procedures used to derive datasets suitable for state and trend analyses.

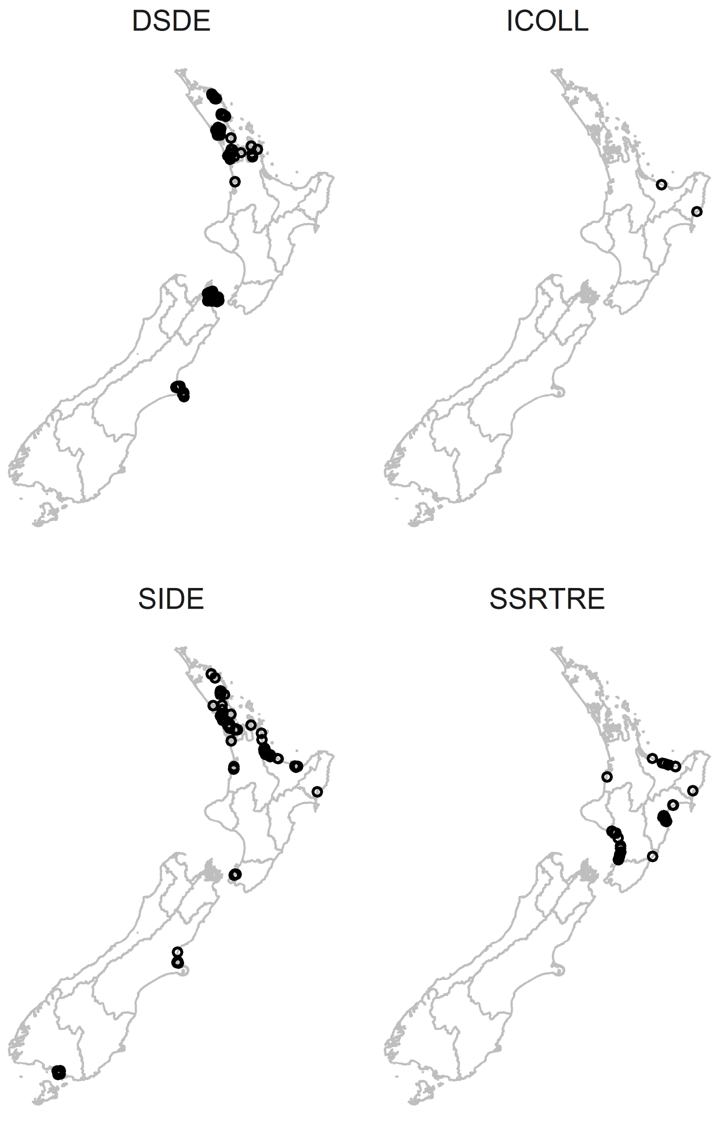

Figure 2-1: Locations of all coastal monitoring sites in water quality datasets provided by councils. The legend gives the New Zealand Estuarine Trophic Index (ETI) classification of the site. Classifications are: deep subtidal-dominated estuaries (DSDEs) shallow intertidal-dominated estuaries (SIDEs), shallow, short residence-time tidal river estuaries (SSRTREs), and intermittently closing and opening lagoons (ICOLLs). See section 3.2 for classification rationale.

2.1 Variable selection for analysis of state and trends in coastal water quality

We described coastal and estuarine water quality using twelve variables that correspond to physical, chemical and microbiological conditions (Table 2-1). In this report, we use "coastal water quality" as a general term to refer to some or all of the twelve variables. Unless otherwise stated, we made no distinction between data collected at regional council sites and ICC sites.

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

Variable type	Variable	Abbreviation	Units	Values addressed (rationale)
Physico-chemical	Dissolved oxygen	DO	mg/L	Ecosystem health
	pH	PH	pH units	Ecosystem health (local and global change)
	Salinity	SAL	parts per thousand	Ecosystem health ('master' variable measuring freshwater content)
	Temperature	TEMP	degrees Celsius	Ecosystem health (global change)
Optical	Visual clarity (Secchi)	CLAR	m	Ecosystem health; Recreation
	Turbidity	TURB	NTU	(Proxy for visual clarity or suspended particle concentration; continuously measurable)
	Suspended solids	SS	mg/L	Ecosystem health; Recreation
Nutrients	Ammoniacal nitrogen	NHXN	mg/L	Ecosystem health
	Nitrate-nitrite nitrogen	NOXN	mg/L	Ecosystem health
	Total nitrogen (unfiltered)	TN	mg/L	Ecosystem health
	Dissolved reactive phosphorus	DRP	mg/L	Ecosystem health
	Total phosphorus (unfiltered)	TP	mg/L	Ecosystem health
Microbiological	Faecal coliforms	FC	n/100 mL	Recreation; Shellfish aquaculture
	Enterococci	ENT	n/100 mL	Recreation; Shellfish aquaculture
	Chlorophyll-*a*	CHLA	mg/L	Ecosystem health

Dissolved oxygen (DO) is the oxygen concentration in water, and is influenced by oxygen supply and oxygen consumption taking place in water and sediments that are in contact with shallow water. High DO values can reflect high primary production or aeration relative to respiration. Low values can be indicative of high rates of decomposition of organic material in sediments and waters, and may result in reduced species diversity and faunal biomass (GESAMP 2001).

Salinity (SAL) was included because salinity data are needed to assess freshwater content of coastal waters. Water temperature (TEMP) was included because temperature controls rates of biochemical reactions plus equilibria (e.g., Dissolved oxygen saturation) and for assessing climate change. We have included pH data because decreases in pH result from sequestration of atmospheric CO_2, and may also reflect more local scale processes caused by eutrophication (Cai et al. 2011). However, in coastal waters interactions between DO, dissolved nitrogen, and dissolved inorganic carbon and their responsiveness to temperature, acidification and eutrophication make it difficult to assign a cause to observed changes in pH (Hewitt et al. 2014).

The optical variables provide information on the transmission of light through waters. Reductions in visual water clarity (CLAR) result from light attenuation due to absorption and scattering by dissolved and particulate material in water. Turbidity (TURB) measured with an optical sensor (nephelometer) is an index of side-scatter from a beam of light transmitting through the water sample. Visual clarity and turbidity are monitored because the attenuation of light in waters (and with depth in the water column) affects primary production, plant and animal distributions and ecological health, aesthetic quality and recreational values (Davies-Colley et al. 2003).

Suspended solids (SS) are a major cause of both reduced visual clarity in water and reduced light penetration with depth through the water column (Gall et al. *in review*). Suspended solids include organic matter (e.g., phytoplankton, or fine particles of decomposing plant matter), and inorganic matter (e.g., inorganic sediment from terrestrial erosion). High suspended sediment concentrations are associated with estuarine and coastal sedimentation, reduced light levels in benthic environments and reduced feeding rates and health of estuarine and coastal animals (Lowe et al. 2015).

The five nutrient species (NOXN, NHXN, DRP, TN and TP) were included because they influence aquatic primary production - the growth of benthic microalgae (periphyton), photosynthetic bacteria, phytoplankton, macroalgae, and aquatic vascular plants. This is because phosphorus (P) and particularly nitrogen (N) are the nutrients that are in shortest supply relative to demand by aquatic primary producers during spring and summer in temperate coastal waters, including in New Zealand (Hanisak, 1983). Hence, increases in the availability of these nutrients are associated with increased primary production. Estuaries and open coasts are mixing zones for nutrients that originate in fresh and marine water, which can increase the availability of multiple nutrients (Sharp 1983). In severe cases, nutrient loading in coastal mixing zones results in proliferations of aquatic primary producers that can, in turn, degrade estuarine and coastal habitat, cause water colour and odour problems, and may be toxic to consumers, including humans (GESAMP 2001, Karez et al. 2004). There are two or more methods in use to measure concentrations of some nutrient species, and not all methods give comparable results. Some data obtained by non-comparable analytical techniques/methods were excluded from the analyses (see Section 2.4).

Enterococci (ENT), and faecal coliform (FC) bacteria are included as their abundances indicate recent faecal pollution and the possible presence of human faecal pathogens in coastal waters. Hence, they represent the risk of infectious disease from waterborne pathogens; ENT is collected by councils as an indicator of the suitability of water for contact recreation and FC as an indicator of the suitability for gathering shellfish. CHLA is a measure of phytoplankton biomass. In coastal waters, high CHLA concentrations may occur during periods of high nutrient loading or upwelling of nutrients from deeper ocean waters, and CHLA is a primary indicator of eutrophication.

The physical, chemical and microbiological variables described above are characterised by short-term variability. We note that nationally-used variables that give time-integrated measures of water quality are lacking from the dataset. These variables give information on the presence of nutrients or contaminants at a given point in space over longer timescales than the 'instantaneous' measures given by water sampling. Examples are algal bioindicators to measure nutrient availability (e.g., Barr et al. 2013), or the NOAA 'Mussel Watch' programme to measure water column contaminants. We note also that a nationally applicable 'integrated index' for assessing environmental condition is currently lacking from State of the Environment monitoring in coastal waters (see Borja et al. 2008 for examples). Integrated indices normally combine several biological elements, together with physico-chemical and pollution elements to quantify ecosystem status. A potential candidate for New Zealand coastal waters the 'ETI tool' (Robertson et al. 2016 a, b) is in its trial stages in council data collection programmes.

Several water quality variables that were initially considered for analysis were later excluded. These included concentrations of the indicator bacterium *Escherichia coli,* and dissolved zinc and copper, which may be highly relevant in some regions with highly urbanized estuaries and harbours. Few councils monitor these variables, and for those that do, the sampling frequencies and durations were inadequate for state and trend analyses.

2.2 Data acquisition and organisation

Data requests to regional councils and ICC were informed by the assessment of data availability in the report titled 'Development of a National Marine Environment Monitoring Programme (MEMP) for New Zealand' (Hewitt et al. 2014). We requested data from the beginning of systematic coastal water quality monitoring to the present day. Water quality data were supplied by 10 of the 16 regional councils and unitary authorities and by ICC. Abbreviations of council names used in the report are as follows: NRC - Northland Regional Council, AC – Auckland Council, WRC - Waikato Regional Council, BOPRC - Bay of Plenty Regional Council, GDC - Gisborne District Council, HBRC - Hawke's Bay Regional Council, HRC - Horizons Regional Council, GWRC - Greater Wellington Regional Council, MDC, Marlborough District Council, CRC –Canterbury Regional Council, ICC – Invercargill City Council . For most variables and councils, the ending dates for data ranged from mid-2015 to early 2016.

An important decision made with the consultation of council staff during data acquisition was the limitation of the dataset to variables measured in the water column. Measurements made in sediment, benthic ecology, and bioindicator data were omitted. While these variables are often reflective of water quality, this decision was made due to wide range of sediment, ecology and indicator variables measured, and the lack of a coastal trophic index with which to convert variables to a standardised index of coastal water quality.

2.3 Data processing

The raw coastal water quality data provided by councils varied widely in reporting formats, reporting conventions for variable names, site identifiers, date and time formats, units of measurement, and other data structure elements. We imported the datasets into the statistical software 'R', and applied a consistent set of reporting conventions. We manually inspected the datasets and used time-series plots and other diagnostics to identify and correct errors. The errors included mislabelled site-names, georeferencing errors, incorrect units and data transcription errors. Analysing and formatting the database in R allowed us to attach information to individual data points. This information included flags for censored data, unit conversions (e.g., from µg/L to mg/L), and quality codes. Our final database had 347 sites, consisting of 338 regional council sites plus 9 ICC sites in the New River Estuary.

In addition to water quality data, the following spatial data were associated with each monitoring site: Regional Council ID, regional council site identification code, site names (if available), NZTM grid reference, and site notes. After compiling the site data, each site was assigned a unique identifier.

Water quality data were processed in several steps to ensure that the data were accurate and the datasets used for analyses were internally consistent.

Step 1. Comparable field and laboratory methods. The first data processing step was to assess methodological differences for all variables. For many variables, two or more measurement procedures were represented in the datasets. We grouped data by procedure, then pooled data for which different procedures gave comparable results, based on assessments in Davies-Colley et al. (2012), Larned and Unwin (2012), and Larned et al. (2015). Data measured using the less-common and non-comparable methods were omitted. Table 2-2 lists the most common procedures used for each variable, and the procedures corresponding to data retained for analysis.

The data produced by multiple procedures used to measure ENT, FC, NOXN, TN and DO were pooled, based on the assumption that the different procedures gave comparable results. In addition, turbidity measurements may be highly instrument-specific, even for nephelometers of apparently similar design (Davies-Colley et al. 2015; Hicks et al. 2016), and our inclusion of datasets across councils assumes that differences due to instrumentation are minor in comparison to measured environmental differences. Around half of the councils measure TN using the alkaline persulfate digestion method and half use a sulphuric acid digestion procedure to measure total Kjeldahl nitrogen (TKN) and calculates TN as TKN + NOXN. Data received during consultation with the NEMS group (Pers. Com. Peter Robinson) suggest that these two procedures give comparable results across the range of concentrations common for council water quality sampling. However, this may not extend to samples with high suspended particulate matter loads (Rus et al. 2012, Davies-Colley and McBride 2016), and further research is required to establish comparability of these two methods in marine waters. For this study, we retained data derived from the two methods for statistical analyses.

In contrast, some procedures used to measure TN and TP are unlikely to give comparable results. At least one council uses filtered samples for the data labelled TN and TP, although the filtered samples are more correctly labelled as ‘total dissolved nitrogen’ and ‘total dissolved phosphorus’. Also, alternative chemical analysis methods for TP could generate substantial differences in reported concentrations. Therefore, only TP measured by the persulfate digestion method with unfiltered samples were retained for analysis.

Step 2. Error correction and adjustment. The second data processing step was to manually inspect the data, and correct identifiable errors. We used quantile plots to identify and remove gross outliers for each variable. These were restricted to pH values above 14, a single temperature value of 80 °C and a single DO concentration over 30 mg/L. Where necessary, values were adjusted to ensure consistent units of measurement across all datasets. Site location information was used to group sites and to correct mismatched site location and site names.

Step 3. Censored and substituted values. The final data processing step concerned censored and substituted values. For several water quality variables, some values were too low (or, occasionally, too high) for laboratories to measure with precision, and these are traditionally reported as less than a “detection limit”, even though this amounts to ‘censoring’ (of information) because the laboratories do have an (imprecise) estimate. Cases where values of variables are below the detection limit or above the reporting limit are often indicated by the data entries “<DL” and “>RL”, where DL and RL are the laboratory detection limit and reporting limit, respectively. In some cases, the censored values had been replaced (by the monitoring agency) with substituted values to facilitate statistical analyses. Common substituted values are 0.5×detection limit and 1.1×reporting limit. Water quality datasets from New Zealand often include DRP and NHXN measurements that are below detection limits, and occasional ECOLI and CLAR measurements that are above reporting limits. Although common, replacement of censored values with constant multiples of the detection and reporting limits can result in misleading results when statistical tests are subsequently applied to those data (Helsel 2012). Data that we received that were composed of censored and substituted values were replaced with imputed values using the procedures in section 3.1.

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

Variable type	Variable	Measurement procedures	Procedures retained
Physico-chemical	DO	In situ, automatic profilers, surface water grab-samples, DO measured on boat or in helicopter from surface water	Both procedures (presumed to give comparable results)
	PH	APHA 4500-H B. Surface water pH measurement using handheld meter	APHA 4500-H B. Surface water measurement using handheld meter
	SAL	Handheld digital salinometer in surface water. Method APHA 2520 B.	Handheld digital salinometer in surface water. Method APHA 2520 B.
	TEMP	Glass mercury/alcohol thermometer Handheld digital water quality meter (e.g., YSI)	Both procedures (presumed to give comparable results)
Optical	CLAR	Black-disk Secchi-disk	Secchi-disk
	TURB	Hach turbidity meter. Method APHA 2130 B	Hach turbidity meter. Method APHA 2130 B
	SS	Gravimetric determination of total suspended solids	Gravimetric determination of total suspended solids
Nutrients	NHXN	Filtered. Phenyl/hypochlorite colorimetry	Filtered. Phenyl/hypochlorite colorimetry
	NOXN	Nitrate-N, filtered, Ion chromatography Nitrate-N + nitrite-N (or "NNN"), filtered, cadmium reduction. Nitrate + Nitrite-N – Nitrite-N (filtered, Azo dye colourimetry)	All procedures (NNN used when NO_3^- unavailable; nitrite presumed to be negligible in unpolluted water)
	TN	Unfiltered, persulfate digestion Filtered, measured as dissolved inorganic+organic nitrogen Sample filtered, filtrate N measured as dissolved inorganic+organic	Unfiltered, persulfate digestion Sample filtered, filtrate N measured as dissolved inorganic+organic nitrogen, added to mass of N in filtered solids.

Variable type	Variable	Measurement procedures	Procedures retained
		nitrogen, added to mass of N in filtered solids. Unfiltered, by Kjeldahl digestion (TKN + NNN)	Unfiltered, by Kjeldahl digestion (TKN + NNN)
	DRP	Filtered, molybdenum blue colourimetry	Filtered, molybdenum blue colourimetry
	TP	Unfiltered, persulfate digestion Unfiltered, nitric acid/hydrogen peroxide digestion. Filtered, measured as dissolved inorganic+organic phosphorus	Unfiltered, persulfate digestion
Microbiological	FC	Membrane filtration (APHA 9222D) Multiple tube (APHA 9221E)	Both procedures (presumed to give comparable results)
	ENT	Multiple tube (APHA 9230B) Membrane filtration (APHA 9230C) Fluorogenic Substrate Enterococcus Test 'Enterolert' (APHA 9230D)	All procedures (presumed to give comparable results)
	CHLA	Acetone pigment extraction, spectrofluorometric measurement. In situ and laboratory fluorometry	Acetone pigment extraction, spectrofluorometric measurement.

3 Analysis methods

3.1 Censored values

In this study, we used a three-step process to impute replacements for censored values. For comparative purposes we also performed equivalent analyses using the traditional substitution rules (i.e., left censored values substituted with values corresponding to one half the reported laboratory detection limit and right censored values increased by 10%).

Step 1. Left-censored data. We manipulated “less than” data using ROS (Regression on Order Statistics) to impute replacement values (Helsel 2012). The ROS procedure produces a separate replacement value for each censored datum. This procedure accommodates multiple censoring limits, which typically occurs when detection limits change over time. Briefly, the ROS method develops probability plotting positions for each data point (censored and uncensored) based on the ordering of the data. A relationship between data values and the uncensored probability plotting positions is fitted by least-squares regression, and this relationship is then used to predict the concentrations for the censored values based on their plotting positions. The ROS procedure produces estimated values for the censored data that are consistent with the distribution of the uncensored values, when distribution of these values in time is unknown. We randomised the predicted values to avoid inducing trends that would be associated with sequential plotting positions, which for the censored values is their order of appearance in time-series.

Step 2. Right-censored data. The right-censored data in our datasets were limited to field CLAR (Secchi depth) measurements limited by shallow water, and ENT and FC measurements that exceeded the value which laboratories could measure on their chosen dilutions (they should have retained sufficient sample for re-testing at a higher dilution.) All right-censored data were replaced with values estimated using a procedure based on “survival analysis” (Helsel 2012). These models are routinely used to estimate the survival time of samples beyond the period of observation or experiment. In this approach a parametric distribution is fitted to the uncensored values using maximum likelihood. The values for the censored observations are then estimated by randomly sampling values larger than the censored values from this distribution.

Step 3. Striping. In some cases, laboratory results for low nutrient concentrations were reported on a semi-discrete scale (e.g., 1-2 decimal places), resulting in horizontal lines on plots of water quality variable versus time, or “striping”. These stripes correspond to tied data, which can pose problems for trend analyses, such as producing trends with slopes of exactly zero. Replacement of these tied values by imputation of randomised ROS values is inappropriate, because the striped concentrations are not the result of censoring. Instead, we “jittered” these results about their reported values to minimise the occurrence of ties. The jittering procedure is not applied to any previously imputed values and only considers duplicated values, i.e. where more than one instance of the same number is reported for each variable at each site. For these duplicated numbers a small (<2% of value), randomly selected number is either added to or subtracted from the reported value.

3.2 Grouping sites

Open coastal, fjord, and estuarine monitoring sites were grouped into classes to aid the explanatory power of state and trend analyses. Classifications were made according to the typology used in the New Zealand Estuary Trophic Index (ETI) (Robertson et al. 2016a). The definitions of the classes are included as they appear in Robertson et al. 2016a, in Table 3-1. These classifications are designed to reflect the susceptibility of hydrosystems to eutrophication resulting from nutrient loading, and may account for some variation in water quality associated with environmental heterogeneity.

The ETI rationale for hydrosystem classification is based on dilution, retention and loss of inflowing nutrients. For a given rate of nutrient loading, eutrophication is more likely to occur when dilution is low, and retention and uptake of nutrients within the hydrosystem are high.

Because the ETI typology is informed by depth, water residence time, inflow/estuary volume and intertidal area, we used hydrological and geographical information from the Coastal Explorer database (Hume et al. 2007) to inform our decisions when classifying sites. For classification of borderline/transitional sites we compared hydrological and geographical data and Estuarine Environment Classification (EEC) class (Hume et al. 2007) to the draft "New Zealand Hydrosystems Classification" (NZCH) class descriptions (Hume 2016). We then compared NZCH class descriptions to the corresponding ETI class according to Hume (2016). As the ETI typology is focussed on estuarine systems it does not cover open coastal locations included in council monitoring programmes. In this report we have grouped sites that did not conform to an ETI class (those sites with mean salinity > 30, indicating that freshwater content was low, <u>and</u> on exposed coastlines with an angle between head of estuary and two outer headlands > 150°, indicating little or no shelter from oceanic swell) in a further class designated as 'Open Coast'.

Both the ETI and NZCH projects recognise that many coastal hydrosystems, particularly the large ones, contain areas that are more suitably described as subtypes of the larger system (Hume 2016). An example of this are the shallow inner arms of the Waitemata Harbour; while the Waitemata harbour system meets the ETI classification of a Deep Subtidal Dominated Estuary (DSDE) based on mean depth and intertidal area, the northern inner arms contain extensive tidal flats more suitably classified as Shallow Intertidal Dominated Estuaries (SIDEs). Based on recommendations in the ETI, we grouped sites within large hydrosystems that fitted different ETI class descriptions according the classification appropriate at the finer scale.

Table 3-1: Main hydrosystem classifications used in New Zealand Estuary Trophic Index eutrophication susceptibility analysis. Reproduced from Robertson et al. 2016

1. Intermittently Closed/Open Lake and Lagoon Estuaries (ICOLLs)
Shallow tidal lagoon and tidal river type estuaries (<3m deep) that experience periodical mouth closure or constriction (called ICOLLs) have the highest susceptibility to nutrient retention and eutrophication, with the most susceptible being those with closure periods of months (e.g., Waituna Lagoon) rather than days (e.g., Lake Onoke). In general, the tidal river ICOLLs have shorter periods of mouth closure (unless they are very small) than the more buffered tidal lagoon ICOLLs. The high susceptibility arises from reduced dilution (absence of tidal exchange at times) and increased retention (through both enhanced plant uptake and sediment deposition). Excessive phytoplankton and macroalgal growths and reduced macrophyte growth are characteristic symptoms of ICOLL eutrophication. In ICOLLs, which vary between marine and close to freshwater salinities, a co-limiting situation between N and P is expected, and as a consequence nutrient load/estuary response relationships should consider both N and P. Susceptibility to Nutrient Loads: Very High Major Primary Producers: Both Macroalgae and Phytoplankton

Example: Te Waihora / Lake Ellesmere
2. Shallow, Intertidal Dominated Estuaries (SIDEs)
For New Zaland’s dominant estuary types (i.e. shallow, short residence time (<3 days), and predominantly intertidal, tidal lagoon estuaries and parts of other estuary types where extensive tidal flats exist e.g., Firth of Thames, Kaipara Harbour, Freshwater Estuary), flushing is too strong for significant retention of dissolved nutrients. Nevertheless, retention time can still be sufficient to allow for retention of fine sediment and nutrients (particularly if these are excessive), deleterious for healthy growths of seagrass and saltmarsh, and nuisance growths of macroalgae in at-risk habitat. In these latter estuary types, assessment of the susceptibility to eutrophication must focus on the quantification of at-risk habitat (generally upper estuary tidal flats), based on the assumption that the risk of eutrophication symptoms increases as the habitat that is vulnerable to eutrophication symptoms expands. Nitrogen has been identified as the element most limiting to algal production in most estuaries in the temperate zone and is therefore the preferred target for eutrophication management in these estuaries (Howarth and Marino 2006). Susceptibility to Nutrient Loads: Moderate to High Major Primary Producers: Macroalgae Examples: Tauranga Harbour, Estuary of the Heathcote and Avon Rivers/Ihutai
3. Shallow, Short Residence Time Tidal River, and Tidal River with Adjoining Lagoon, Estuaries (SSRTREs)
New Zealand also has a number of shallow, short residence time (<3 days) tidal river estuaries (including those that exit via a very well-flushed small lagoon) that have such a large flushing potential (freshwater inflow/estuary volume ratio >0.16) that the majority of fine sediment and nutrients are exported to the sea. Tidal River ICOLLs with closure periods of days rather than months and high freshwater inflows (e.g., Lake Onoke) can also fit in this category. In general, these estuary types have extremely low susceptibilities and can often tolerate nutrient loads an order of magnitude greater than shallow, intertidal dominated estuaries. These shallow estuary types are generally N limited. Susceptibility to Nutrient Loads: Low to Very Low Major Primary Producers: Macroalgae, but low production, especially if freshwater inflow high. Example: Piha Lagoon, Whanganui River mouth
4. Deeper, Subtidal Dominated, Estuaries (DSDEs)
Mainly subtidal, moderately deep (>3m to 15m mean depth) coastal embayments (e.g., Firth of Thames) and tidal lagoon estuaries (e.g., Otago Harbour), with moderate residence times >7 to 60 days) can exhibit both sustained phytoplankton blooms, and nuisance growths of opportunistic macroalgae (especially *Ulva sp.* and *Gracilaria sp.*) if nutrient loads are excessive. The latter are usually evident particularly on muddy intertidal flats near river mouths and in the water column where water clarity allows. Deeper, long residence time embayments and fiords are primarily phytoplankton dominated if nutrient loads are excessive. Outer reaches of such systems which sustain vertical density stratification can be susceptible to oxygen depletion and low pH effects

(Sunda and Cai 2012, Zeldis et al. 2015). In both cases, it is expected that the US ASSETS approach will adequately predict their trophic state susceptibility. These deeper estuary types are generally N limited.

Susceptibility to Nutrient Loads: Moderate to Low

Major Primary Producers: Macroalgae (moderately deep) and phytoplankton (deeper sections).

Examples: Firth of Thames, Queen Charlotte Sound

3.3 Water quality state

For each site, we characterised the current water quality state as percentiles (5th, 20th, 50th, 80th, 95th) of the distribution of measured values of the variables listed in Table 2-1 for the period 2011 to 2015 (inclusive). These percentiles were calculated using the Hazen method.[2]

The confidence with which we can describe water quality state at a site depends on variability in the measurements between sampling dates and on sample size (i.e., the number of sampling dates). There are diminishing returns on increasing confidence with increasing sample size and as a general rule the rate at which confidence increases for estimates of population statistics levels off with sample sizes greater than 30 (McBride 2005) (2-3 years of monthly data). Temporal trends can affect estimates of water quality state if state is assessed over long periods. We assessed state over a period of five years because it represented a reasonable trade-off between sample size and resistance to the effects of trends. For monthly sampling a period of 5 years will yield up to 60 data points, however a large proportion of sites within our dataset were sampled quarterly for the last five years. In order to include these sites, we set the minimum number of measurements at 18 (allowing for some missing data). For all data, we applied three filtering rules to ensure that site median values were reliable: 1) less than 50% of the values for a variable were censored; 2) values for at least 80% of monthly or quarterly sampling dates were available, including imputed values; 3) the datapoints were distributed over four of the five years from 2011 to 2015. Site by variable combinations that did not comply with these rules were excluded from the state analysis.

3.4 Trend analyses

3.4.1 Sampling dates and time periods for trend analyses

Trend analysis is only meaningful for a specified time period over which the dataset being analysed has few missing values. The datasets provided by the regional councils had variable starting and ending dates, variable sampling frequencies (monthly or quarterly), and variable numbers of missing values. We selected time periods for trend analyses by examining the trade-off between the number of qualifying sites (i.e., sites that met our filtering rules concerning missing and censored values) and the duration of monitoring. Variation in site numbers with duration for each variable is presented graphically in Section 4. We assessed trends using monthly data preferentially, and quarterly data when monthly data were not available, provided the filtering rules were met.[3] We applied two

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

[3] Note that as in Larned et al. (2015), quarterly sampling will more commonly give rise to the finding of "insufficient data to detect trend direction".

filtering rules to identify the sites to be included in trend analyses for each water quality variable: 1) 80% of the sampling dates in each of 80% of the years in a trend period had to have observations. For all variables, the rule about 80% of sampling dates applied to monthly or quarterly samples. 2) The number of censored values in a trend period had to be < 15% of the total number of observations. We note that relaxing and tightening filtering rules about sampling dates results in a trade-off between the yield of sites included in the analysis and the number of sites for which there was insufficient data to establish trend direction with confidence. We employed the more relaxed cut-off option (80% of dates in each of 80% of the years in a trend period) used in Larned et al. (2015), due to the relatively few sites available in our dataset.

3.4.2 Statistical trend analyses

We used the approach of Larned et al. (2015) to draw inferences about trend direction; if a symmetric confidence interval around the trend (estimated using the Seasonal Sen Slope Estimator SSSE) did not contain zero, then the trend direction was established with confidence. If it did contain zero, we concluded that there were insufficient data to determine the trend direction. For significant trends, in Larned et al. (2015), the “equivalence testing” procedure advanced by McBride et al. (2014) was extended to trend analyses to define trend importance using threshold-values of different water quality variables and critical time spans. This method used published guidelines, including attribute bands in the NPS-FM, as threshold-values for different water quality variables. In the absence of widely recognised thresholds or baseline conditions for New Zealand coastal water quality, when a trend direction was established with confidence our approach necessarily stopped short of assessing trend importance. In this study we present counts of sites at which positive and negative trend directions were established with confidence for each variable within each ETI class, and group these results according to trend magnitude. Our assessment method presents general change for each variable but leaves interpretation of the importance of these trends to later consideration.

We have interpreted decreasing concentrations of nutrients, ENT, FC, SS, CHLA, TURB and increases in CLAR, and DO as improving water quality. We have stopped short of classing trends in PH, SAL and TEMP as ‘improving’ or ‘degrading’ as we cannot say with confidence that trends in these variables reflect changes in ecosystem health. For example, eutrophication can cause both increases (e.g., in surface waters when photosynthesis increases) and decreases in pH of water (Cai et al. 2011). Salinity and temperature changes in estuaries may be caused by natural changes in flow patterns and movements in river mouth position and trends may be affected by long-term hydrological cycles (e.g., Interdecadal Pacific Oscillation (IPO)).

Trend assessments for all water quality variables that were measured monthly or quarterly were based on slopes estimated with the SSSE, where seasons were either months or quarters. Estimated slopes were calculated with the SSSE using a modification of the “zyp” package in R (http://www.r-project.org). The symmetric 95% confidence intervals around each slope were estimated using the method of Sen (1968).[4] The estimated slopes were then standardised by dividing by the corresponding median value and expressed as percentage changes; standardisation facilitates comparisons between groups of sites. These “relativised” trends are denoted as the Relative Sen Slope Estimator (RSSE).

[4] A summary of the method is provided at this website: (http://vsp.pnnl.gov/help/Vsample/Nonparametric_Estimate_of_Trend.htm).

4 Results

4.1 Coastal water quality state

Between 60 and 252 monitoring sites met the filtering rules for the state analysis of different coastal water quality variables; the number of qualifying sites varied by water quality variable and by ETI class (Table 4-1). The geographic distribution of sites is shown in Figure 4-1. The sites are reasonably well-distributed around the North Island, but there are large gaps in the south and west of the South Island.

The distributions of site-median values of the water quality variables for the 2001-2015 period are summarized with box-and-whisker plots for the ETI classes for which there were sufficient sites (Figure 4-2). The plots in Figure 4-2 indicate that ETI classes explain some of the variation in water quality state. Sites in the different ETI classes had different water quality characteristics both in terms of their central tendencies (indicated by the median of the median site values) and their variation. Salinity was highest and least variable in Open Coast and DSDE sites, and lower and more variable in SIDE and SSRTRE sites, reflecting greater influence of freshwater flows in the latter two classes. Freshwater influence coincided with higher median and more variable TN and NOXN concentrations particularly at SSRTRE sites. NHXN, PH and DRP had larger ranges at SIDE and SSRTRE sites than the higher salinity ETI classes – as expected since tidal prism is the highest proportion of total volume in SIDEs and the variation reflects tidal phase. The lowest ETI class median for CLAR and the greatest variability occurred in the SIDE class. ENT and FC concentrations appeared highest in the SIDE and SSRTRE classes, and lowest in marine-dominated open coast sites and DSDE class hydrosystems. ETI class explained little of the variability seen in TEMP, DO, TURB, CHLA, or SS. The complete set of state analysis results is provided in the supplementary file “all_results_by_site.csv”.

Table 4-1: Number of monitoring sites by hydrosystem class and water quality variable in the analysis of water quality state. The site numbers shown refer to sites where less than 50% of the values for a variable were censored, and ≥ 18 values were available, distributed over at least four of the five years from 2011 to 2015.

			ETI class					
Variable type	**Variable**	**Total**	**ICOLL**	**SSTRE**	**SIDE**	**DSDE**	**Open Coast**
Physico-chemical	DO	196	1	34	65	57	39
	pH	180	1	35	59	22	63
	SAL	191	1	6	75	67	42
	TEMP	252	1	35	78	64	74
Optical	CLAR	60	0	0	13	43	4
	TURB	208	1	27	69	57	54
	SS	173	1	25	57	36	54
Nutrients	NHXN	168	0	21	69	52	26
	NOXN	131	0	30	42	35	24
	TN	139	0	30	51	23	35
	DRP	226	0	30	78	70	48
	TP	199	0	30	69	47	53
Microbiological	FC	91	1	14	46	3	27
	ENT	110	1	16	56	0	37
	CHLA	214	0	23	72	66	53

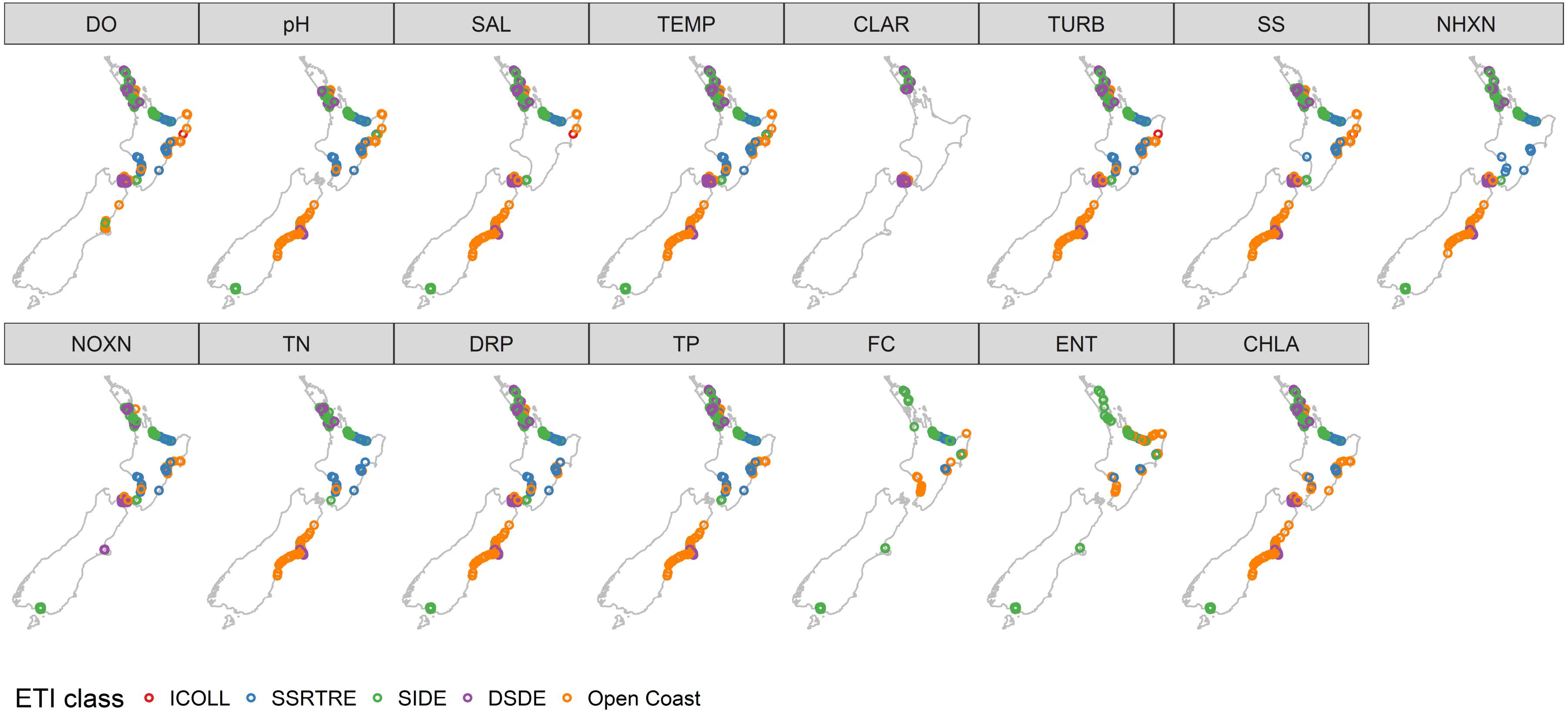

Figure 4-1: Locations of coastal monitoring sites used for water quality state analyses.

The legend gives the ETI classification of the site. Classifications are: deep subtidal-dominated estuaries (DSDEs) shallow intertidal-dominated estuaries (SIDEs), shallow, short residence-time tidal river estuaries (SSRTREs), and intermittently closing and opening lagoons (ICOLLs). See section 3.2 for classification rationale. See section 3.2 for classification rationale.

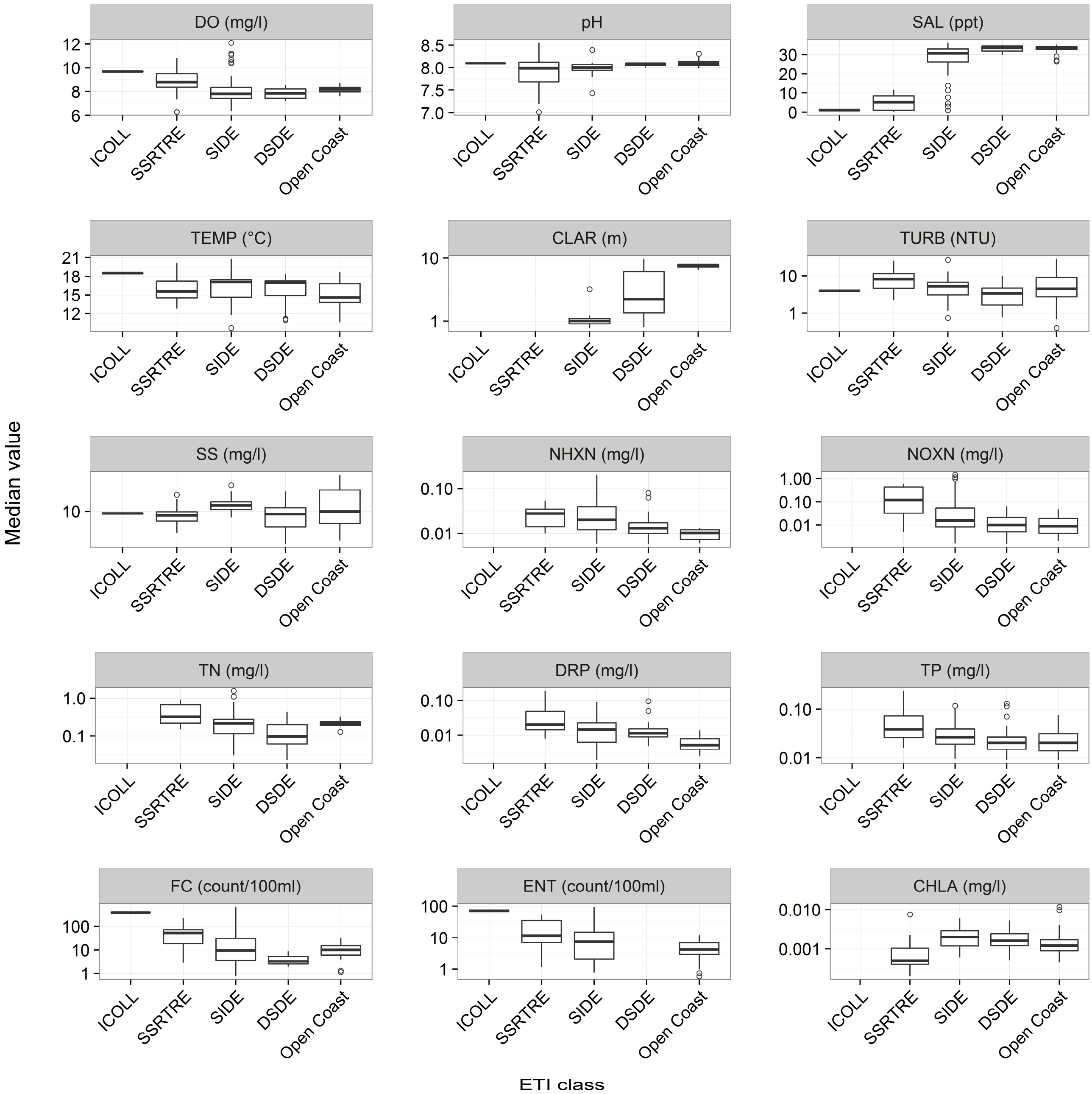

Figure 4-2: Coastal water quality state in ETI classes, and open coastal sites. Box-and-whisker plots show the distributions of monitoring site medians within ETI classes and open coast sites. The line within in each box indicates the median value, the box indicates the inter-quartile range and the whiskers extend from the box to the largest value within 1.5 x the inter-quartile range. Outliers (any data beyond the whiskers) are indicated by open circles. Note log scaled y-axes on all plots except DO, SAL, pH and TEMP.

4.2 Coastal water quality trends

4.2.1 Trade off analysis

The trade-offs between the number of qualifying monitoring sites (i.e., sites that met our filtering rules) and the time period represented by those sites are shown for each water quality variable in Figure 4-1. Trend periods of eight years (2008-2015) and 18 years (1998-2015) were used to make relatively robust (in terms of site number) estimates of monotonic short- and long-term trends. The eight and 18- year periods coincided with the start of regular coastal water quality monitoring by some councils; for example, regular coastal water quality monitoring for a number of variables began at CRC in 2007, and at BOPRC between 1996 and 1999. The two trend periods were selected to coincide with these abrupt increases in site numbers. Multiple trends with different magnitudes and directions may be nested within the eight and 18-year trend periods. Site-specific time-series plots are supplied as supplementary files to this report (Appendix A).

4.2.2 Eight-year trends (2008-2015)

Between 27 and 170 monitoring sites met the filtering rules for the eight-year trend analyses of the 15 water quality variables (Table 4-2). The qualifying sites were reasonably well-distributed geographically for some variables (such as TEMP), with gaps in the south of the North Island, and the South Island west coast (Figure 4-4). For other variables, such as CLAR, FC and ENT, sites that met the filtering rules were restricted to a small number of regions and ETI classes; these trends cannot be expected to be representative of national-scale trends. All site locations, ETI classes, state and trend data are included in the supplementary file "all_results_by_site.csv".

Across the improving and degrading categories, almost all sites where trends could be confidently detected showed improving trends in TP, ENT, FC, NHXN, PH and DRP over the past eight years. There were also more sites with improving trends in TN, SS, CHLA and TURB than degrading trends. In contrast, there were four times as many sites with degrading trends in DO as improving trends, and all sites at which trends were detectable for visual clarity were found to be degrading. There were a large number of sites and variables for which we could not confidently determine a trend direction; this the case for SAL, TEMP, and TURB at the majority of sites. No trend could be detected for PH, DO and CHLA for around half the sites that met filtering rules. These data are summarised in Table 4-2. Trends grouped by ETI class are shown in Figure 4-5. Large variability in salinity trends in the SSRTRE class are apparent; changes in freshwater flows to these hydrosystems may also explain trends in other variables. Salinity changes across other classes, and in DO, PH and TEMP in all classes are very slight if present. For nutrient variables that showed improvement at most sites (TP, NHXN and DRP) these improvements appear relatively consistent in magnitude across ETI classes. Microbiological variables (CHLA, ENT, FC) show the greatest improvement in classes with higher freshwater influence (SIDE and SSRTRE). Trend results partitioned by council are presented in Figure 4-6.

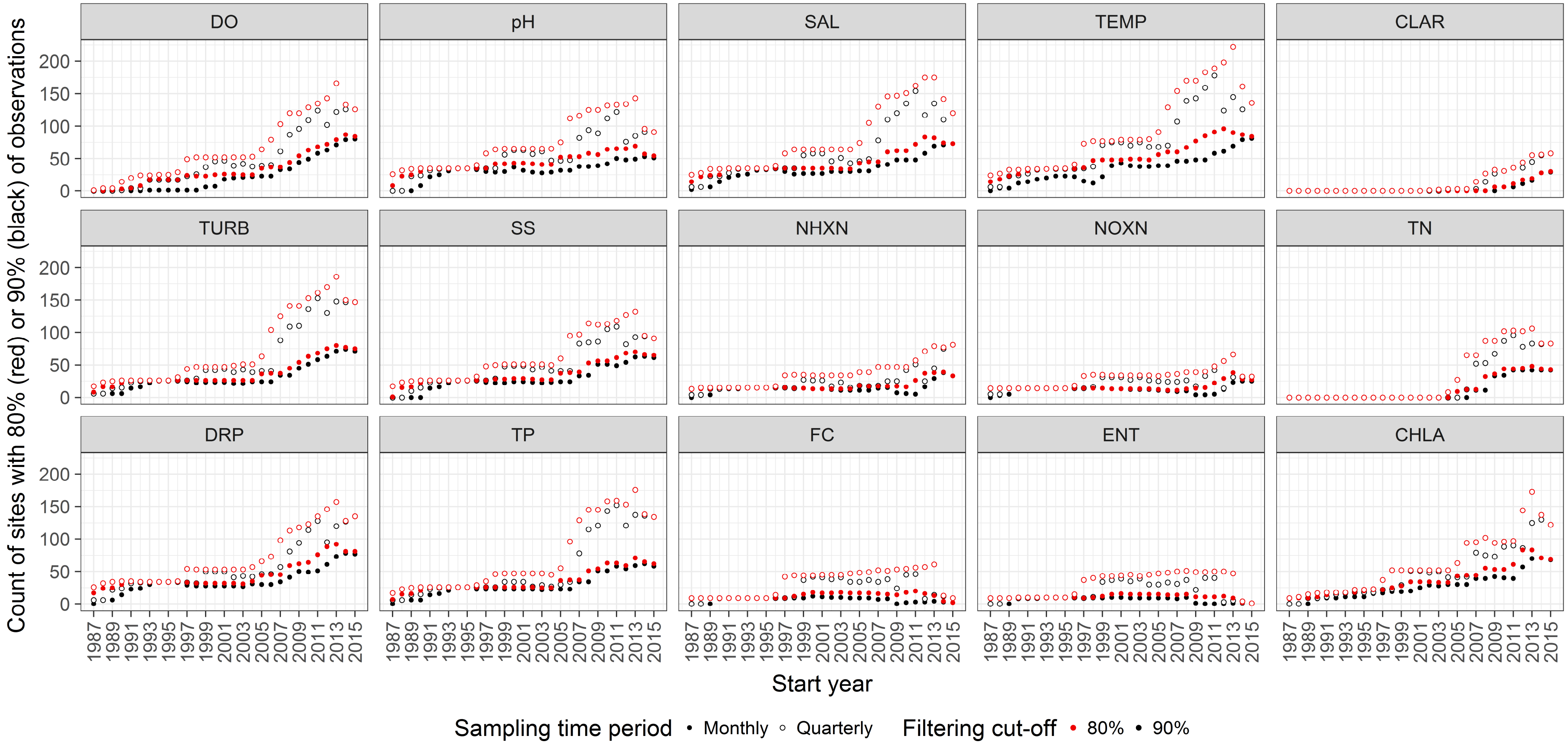

Figure 4-3: Changes in the number of monitoring sites that met the filtering rules for each water quality variable versus the period of site operation. Durations of periods are shown in parentheses. Open circles: monthly data, filled circles: quarterly data. The plots were used to select time periods for trend analyses.

Table 4-2: Number of monitoring sites by ETI class and variable that were included in the 8-year trend analyses of water quality.The site numbers shown refer to sites where 80% of the sampling dates and seven of the years in the 2008-2015 period had observations, and less than 15% of the data for each variable consisted of censored values.

			ETI class				
Variable type	**Variable**	**Total**	**ICOLL**	**SSTRE**	**SIDE**	**DSDE**	**Open Coast**
Physico-chemical	DO	120	1	7	54	38	20
	PH	125	1	8	54	18	44
	SAL	146	0	5	66	43	32
	TEMP	170	1	8	66	44	51
Optical	CLAR	27	0	0	9	18	0
	TURB	141	1	7	56	40	37
	SS	114	1	7	48	22	36
Nutrients	NHXN	47	0	5	38	4	0
	NOXN	39	0	7	28	2	2
	TN	87	0	6	39	15	27
	DRP	113	0	7	65	36	5
	TP	145	0	7	57	44	37
Microbiological	FC	51	1	7	30	0	13
	ENT	51	1	6	32	0	12
	CHLA	102	0	6	53	22	21

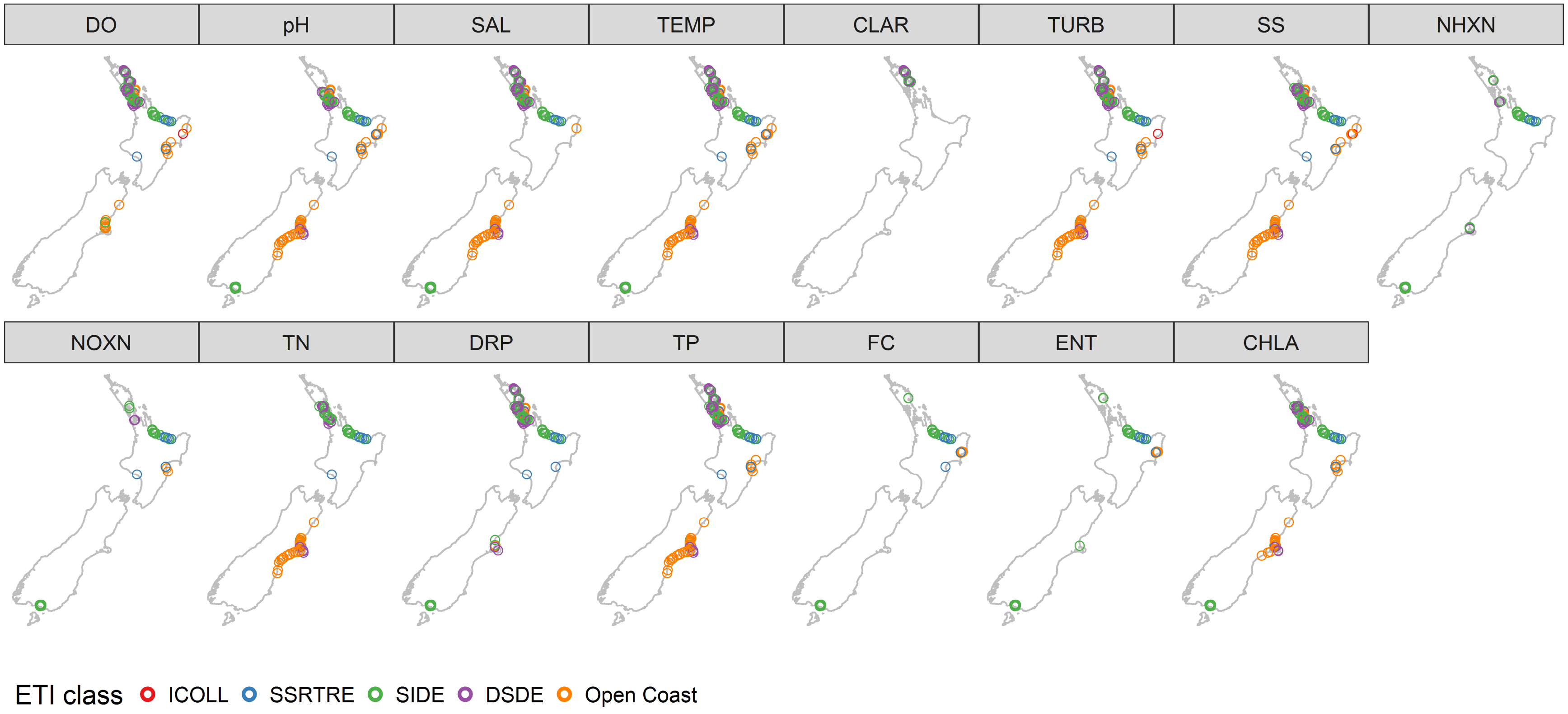

Figure 4-4: Locations of monitoring sites used for eight-year trend analyses of water quality variables. Legend gives ETI class of each site. Classifications are: deep subtidal-dominated estuaries (DSDEs) shallow intertidal-dominated estuaries (SIDEs), shallow, short residence-time tidal river estuaries (SSRTREs), and intermittently closing and opening lagoons (ICOLLs). See section 3.2 for classification rationale.

Table 4-3: Numbers of sites in trend categories for 8-year trends across ETI classes. Decreasing concentrations of nutrients, ENT, FC, SS, CHLA, TURB and increases in CLAR, and DO can be interpreted as improving trends. Environmental degradation/improvement is not implied by trends in PH, SAL and TEMP (see methods). Insufficient data implies not enough data to reveal a trend direction (see Section 1 above).

Variable type	Variable	Magnitude of 8-year trend								Totals		
		Decreasing > 5% p.a	Decreasing 3 - 5% p.a	Decreasing 1 - 3% p.a	Decreasing 0 - 1% p.a	Increasing 0 - 1% p.a	Increasing 1 - 3% p.a	Increasing 3 - 5% p.a	Increasing > 5% p.a	Decreasing	Increasing	Insufficient data
Physico-chemical	DO	0	1	24	22	7	4	0	0	47	11	62
	pH	0	0	0	5	48	0	0	0	5	48	72
	SAL	2	0	2	8	12	16	0	3	12	31	103
	TEMP	0	0	11	6	27	12	0	0	17	39	114
Optical	CLAR	6	3	1	0	0	0	0	0	10	0	17
	TURB	26	7	1	0	0	1	5	2	34	8	99
	SS	35	3	1	0	0	4	5	3	39	12	63
Nutrients	NHXN	23	2	0	0	0	0	0	0	25	0	22
	NOXN	4	1	1	0	0	0	2	2	6	4	29
	TN	22	4	2	0	0	2	3	4	28	9	50
	DRP	42	15	7	0	0	0	1	0	64	1	48
	TP	69	21	5	0	0	0	0	0	95	0	50
Microbiological	FC	17	0	0	0	0	0	0	0	17	0	34
	ENT	13	0	0	0	0	0	0	1	13	1	37
	CHLA	30	6	2	0	0	0	2	4	38	6	58

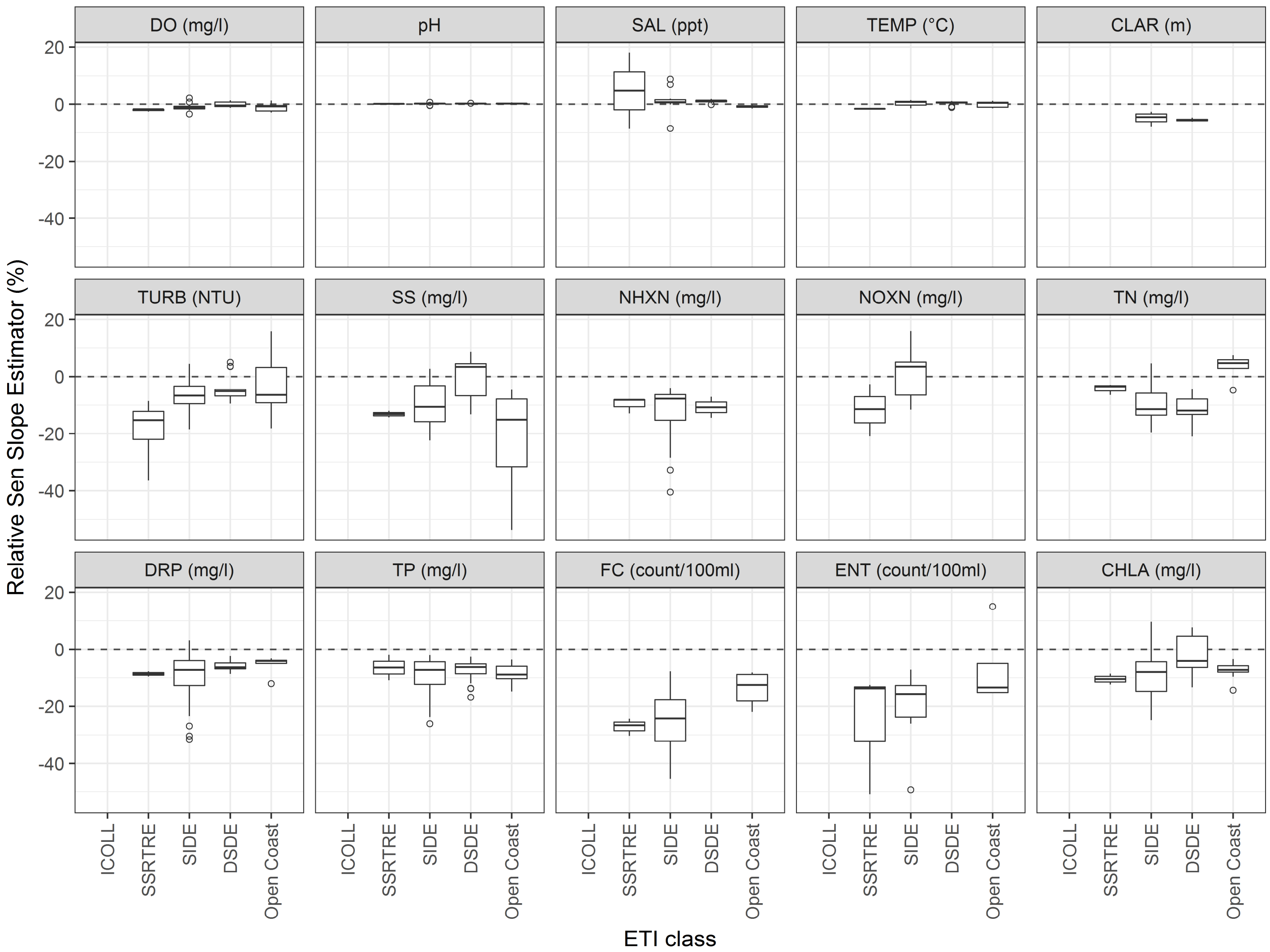

Figure 4-5: **Summary of 8-year trends.** Box-and-whisker plots show the distributions of site trends within ETI classes. The line within in each box indicates the median of site trends, the box indicates the inter-quartile range and the whiskers extend from the box to the largest value within 1.5 x the inter-quartile range. Outliers (any data beyond the whiskers) are indicated by open circles.

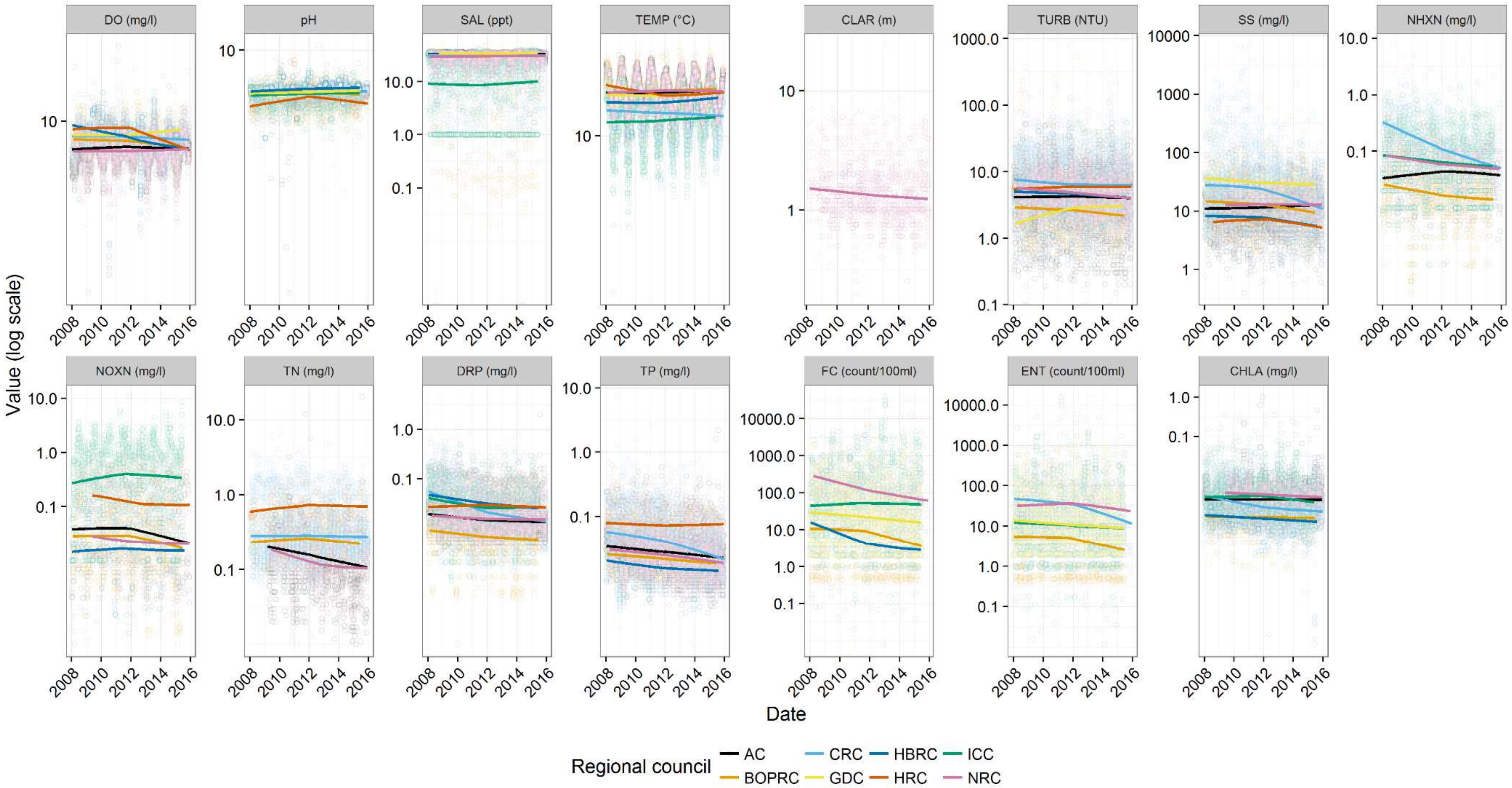

Figure 4-6: Trends in water quality variables over the 8 year period 2008-2015 partitioned by council. Note that the trendlines in each panel correspond to locally weighted (LOWESS) regressions, not seasonally adjusted trends. We suggest that care needs to be taken when interpreting differences in trends between regions due to inherent geographic variability, variation in numbers of sites between regions and differing site selection criteria (This topic is covered in detail in Section 5).

4.2.3 Eighteen year trends (1998 – 2015)

Between 0 and 77 monitoring sites met the filtering rules for the 18-year trend analysis of water quality variables (Table 4-4). No sites met the filtering rules for 18-year trends in CLAR or TN. The numbers of sites also varied substantially by ETI class, and there were few or no qualifying sites for some variables in some ETI classes. For most variables, qualifying sites were clustered around Auckland area, Bay of Plenty, or in the New River Estuary (Figure 4-7). All site locations, ETI classes and trend data are included in the supplementary file in Appendix B "all_results_by_site.csv".

The analysis of 18-year trend categories is shown in Table 4-5. As in the 8-year analysis, there were improving trends in DO, TP and NHXN at most sites. There were also considerable numbers of sites that showed improvement in microbiological variables – FC, ENT and CHLA as well as TURB. Only PH and NOXN were degrading at more sites than improving. Figure 4-8 shows the average trend for each site class. Salinity in the SSRTRE class increased slightly on average; which may explain trends in some other variables. Salinity change across other classes appeared very slight if present. DO, PH and TEMP change in all classes was slight if present. SS and TURB likewise showed small changes, or no average change across site classes. For nutrient variables, NOXN showed increases in hydrosystem classes with lower salinities (SIDE and SSRTRE), but decreased in DSDE hydrosystems. NHXN showed a mean decrease for each of the site classes. DRP showed slight decreases in DSDE and open coastal sites, but little change in SIDEs and slight increases in the five SSRTREs that met filtering rules. TP similarly decreased in DSDE, SIDE and open coastal sites, but showed slight increases in the five SSRTRE sites. For microbiological variables, CHLA showed a slight average decrease in DSDE and open coast sites, and little average change in SIDEs. ENT and FC concentrations trended downwards across all site classes, suggesting improving recreational quality and for bivalve shellfish harvest.

Given the relatively sparse distribution of sites in the 18-year dataset, and the different locations of data that contributed to the 8- and 18-year datasets, a comparison of water quality trends between the 8- and 18-year time periods is probably not useful at a national scale. To illustrate this we have presented the non-monotonic 18-year trends for each council in Figure 4-9, and the relative proportions of data derived from each council in Figure 4-10. These plots show the relative dominance of the AC, BOPRC and CRC datasets in the 8-year DO, DRP, CHLA, SS, TP and TURB trends, and the particular dominance of AC and BOPRC for these variables over the 18-year time period. NHXN and NOXN measured in the New River Estuary (the ICC dataset from southland), and BOPRC sites contribute strongly to the total dataset for those variables. An exception to the regional bias in the datasets is the bacterial data, ENT and FC, which are relatively consistent in their origin between the 8- and 18-year time periods. For these bacterial data, trends appeared consistent showing mostly improvement through the 8- and 18-year time periods.

Table 4-4: Number of monitoring sites by ETI class and variable that were included in the 18-year trend analyses of water quality. The site numbers shown refer to sites where 80% of the sampling dates and seven of the years in the 1998-2015 period had observations, and less than 15% of the data for each variable consisted of censored values.

		ETI class					
Variable type	**Variable**	**Total**	**DSDE**	**ICOLL**	**Open Coast**	**SIDE**	**SSTRE**
Physico-chemical	DO	52	12	1	4	30	5
	pH	64	8	1	10	39	6
	SAL	64	12	1	4	42	5
	TEMP	77	12	1	15	43	6
Optical	CLAR	0	0	0	0	0	0
	TURB	47	8	0	4	30	5
	SS	50	8	1	6	30	5
Nutrients	NHXN	35	4	0	0	26	5
	NOXN	35	5	0	0	25	5
	TN	0	0	0	0	0	0
	DRP	53	7	0	2	39	5
	TP	46	8	0	4	29	5
Microbiological	FC	44	0	1	11	26	6
	ENT	40	0	1	9	24	6
	CHLA	52	8	0	3	36	5

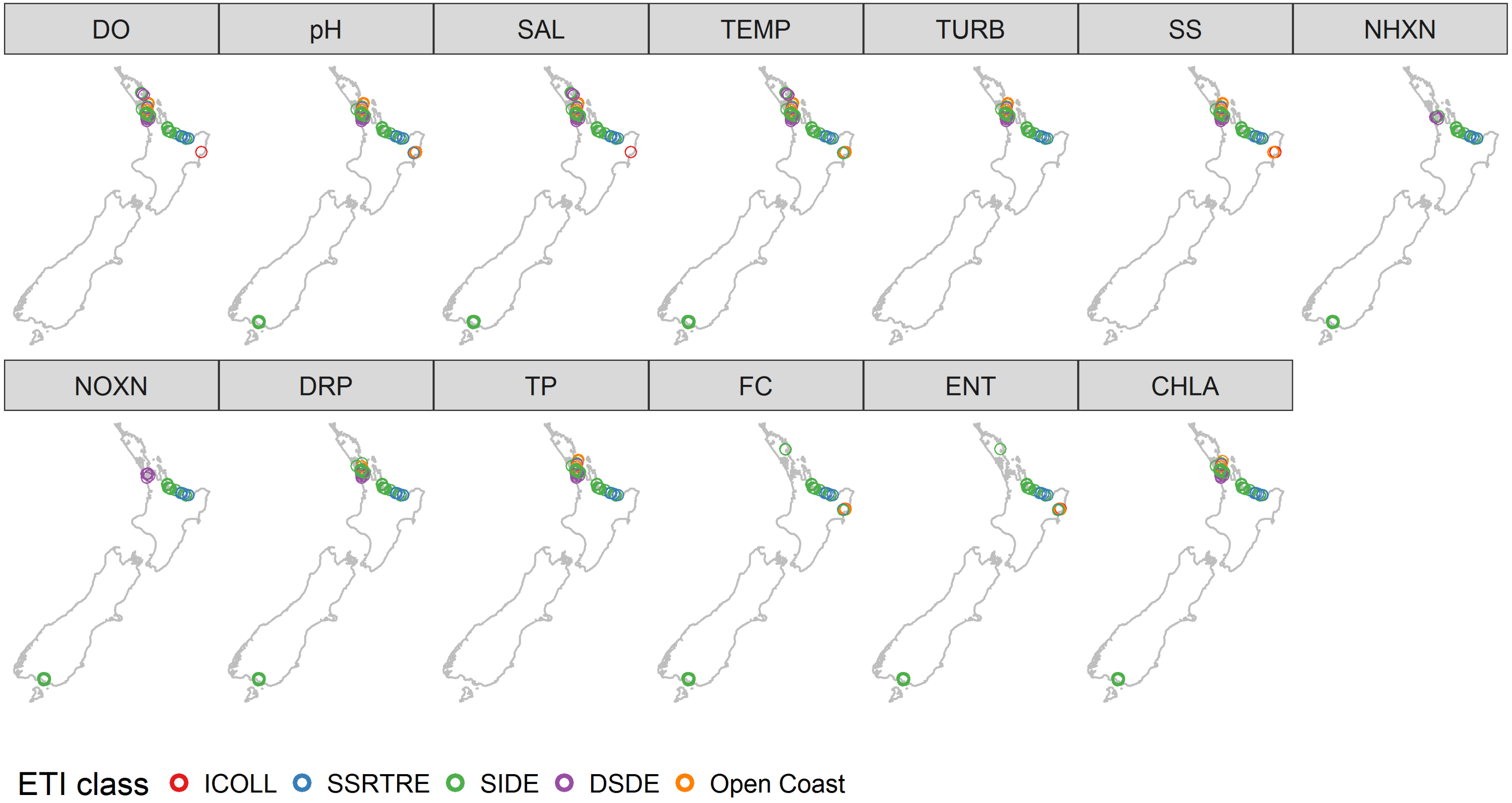

Figure 4-7: Locations of monitoring sites used for 18-year trend analyses of coastal water quality variables. Legend gives ETI class of each site. Classifications are: deep subtidal-dominated estuaries (DSDEs) shallow intertidal-dominated estuaries (SIDEs), shallow, short residence-time tidal river estuaries (SSRTREs), and intermittently closing and opening lagoons (ICOLLs). See section 3.2 for classification rationale.

Table 4-5: Numbers of sites in trend categories for 18-year trends across ETI classes. Decreasing concentrations of nutrients, ENT, FC, SS, CHLA, TURB and increases in CLAR, and DO can be interpreted as improving trends. Environmental degradation/improvement is not implied by trends in PH, SAL and TEMP (see methods). Insufficient data implies not enough data to reveal a trend direction (see Section 3.4 above).

Variable type	Variable	Magnitude of 18-year trend								Totals		
		Decreasing > 5% p.a	Decreasing 3 - 5% p.a	Decreasing 1 - 3% p.a	Decreasing 0 - 1% p.a	Increasing 0 - 1% p.a	Increasing 1 - 3% p.a	Increasing 2 - 3% p.a	Increasing > 3% p.a	Decreasing	Increasing	Insufficient data
Physico-chemical	DO	0	0	0	2	31	2	0	0	2	33	17
	pH	0	0	0	39	11	0	0	0	39	11	14
	SAL	0	0	4	10	22	1	2	2	14	27	23
	TEMP	0	0	0	3	16	1	0	0	3	17	57
Optical	TURB	3	6	13	2	0	2	0	0	24	2	21
	SS	1	4	6	0	2	7	1	0	11	10	29
Nutrients	NHXN	5	4	8	1	1	1	0	0	18	2	15
	NOXN	3	1	1	0	1	9	4	3	5	17	13
	DRP	4	6	22	2	0	7	2	0	34	9	10
	TP	9	12	9	0	0	1	0	0	30	1	15
Microbiological	FC	6	12	4	1	0	0	0	0	23	0	21
	ENT	4	7	3	0	0	0	0	1	14	1	25
	CHLA	1	6	14	0	0	7	3	1	21	11	20

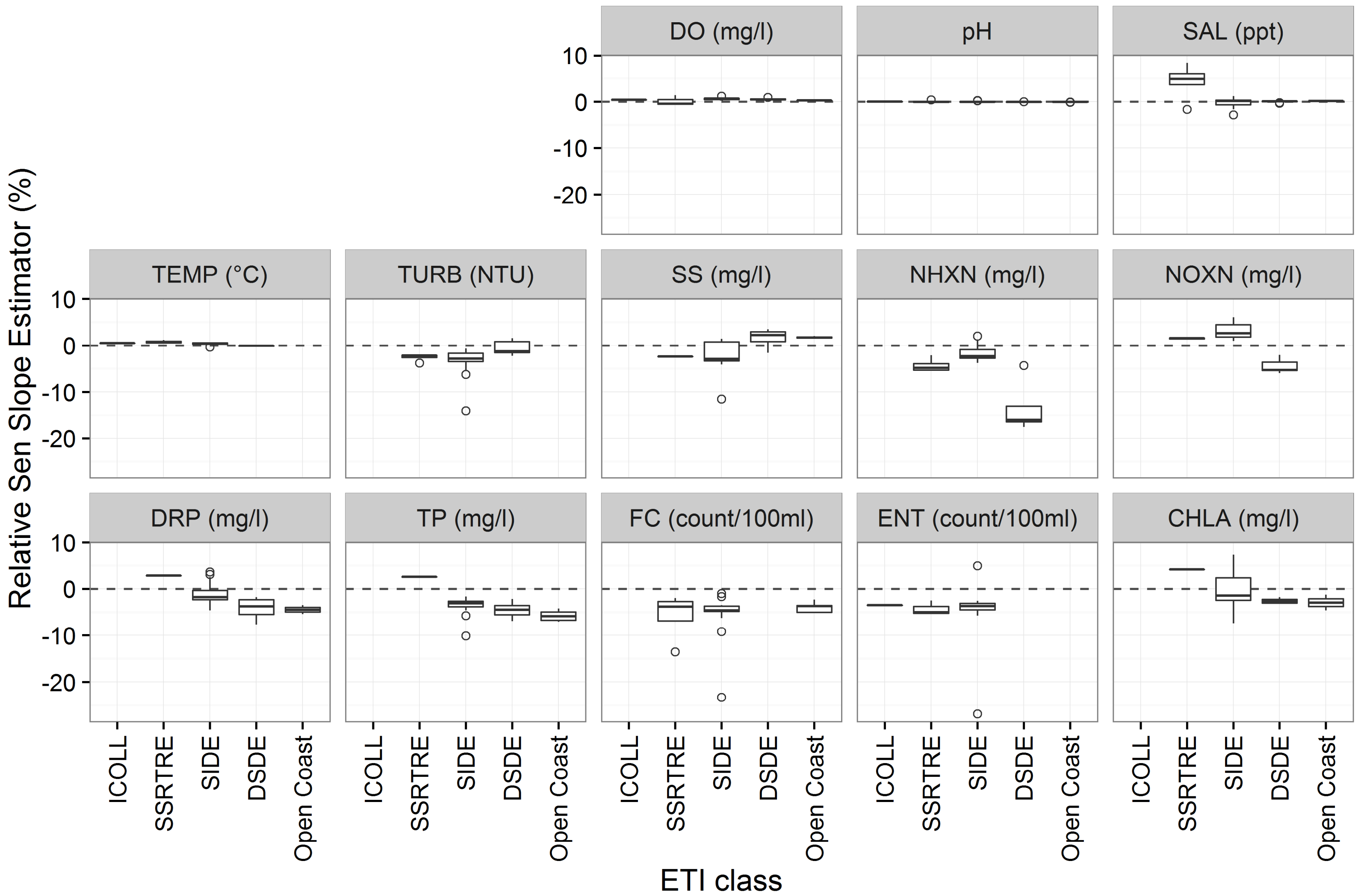

Figure 4-8: Summary of 18-year trends. Box-and-whisker plots show the distributions of site trends within ETI classes. The line within in each box indicates the median of site trends, the box indicates the inter-quartile range and the whiskers extend from the box to the largest value within 1.5 x the inter-quartile range. Outliers (any data beyond the whiskers) are indicated by open circles.

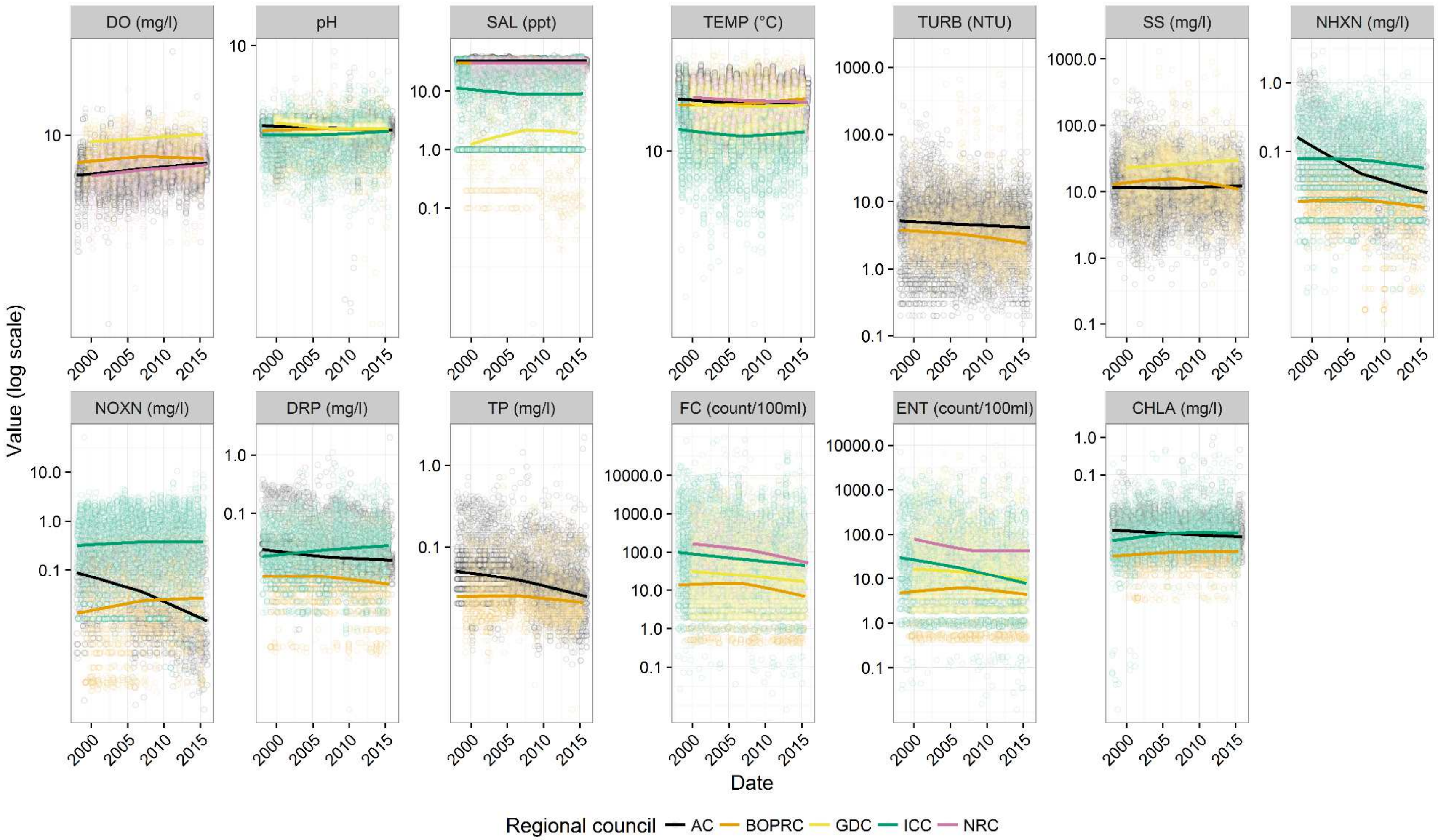

Figure 4-9: Trends in water quality variables over the 18 year period 1998-2015, partitioned by council. Note that the trendlines in each panel correspond to locally weighted (LOWESS) regressions, not seasonally adjusted trends. We suggest that care needs to be taken when interpreting differences in trends between regions due to inherent geographic variability, variation in numbers of sites between regions and differing site selection criteria. (This topic is covered in detail in section 5).

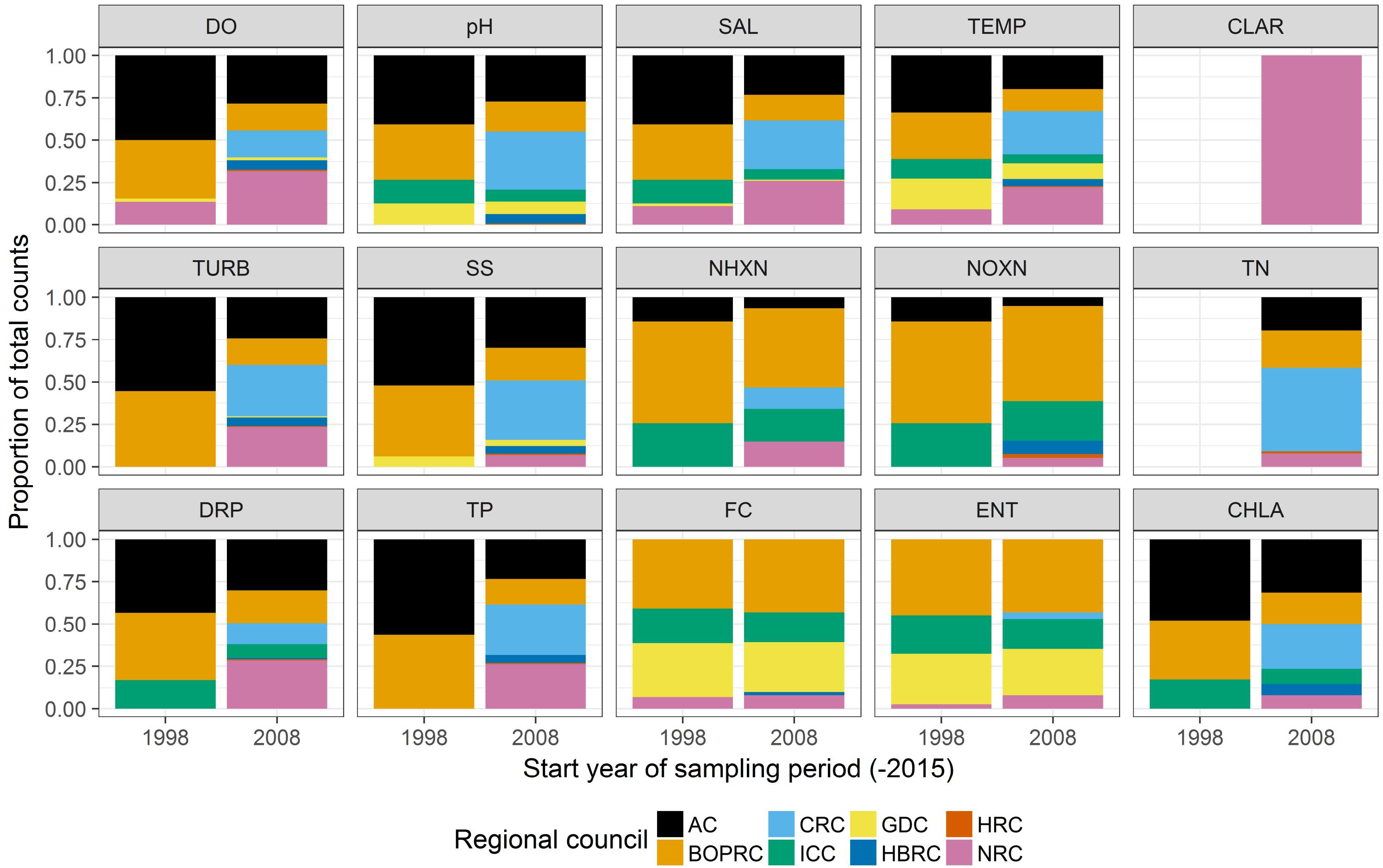

Figure 4-10: Origin of sites included in 8- and 18-year trend analyses. Bars show proportion of total sites for each variable derived from each council.

4.3 Coastal water quality state and trend summary

As well as the summary statistics and plots above, detailed information for each coastal water quality monitoring site is contained in the supplementary files that accompany this report. The sites and corresponding water quality conditions can be aggregated in different ways to suit further reporting (e.g., by region, environmental class, nation-wide).

In general, nutrient and bacterial (FC and ENT) concentrations were elevated, and showed higher between-site variability in the low-salinity ETI hydrosystem types, these same variables were lowest and most stable between sites in open coastal and DSDE site classes. These results suggest that at sites with substantial freshwater input a large proportion of nutrients and faecal indicator bacteria are land-derived. This is consistent with international understanding of the susceptibility of coastal zones to land-based activities (e.g., Vitousek et al. 1997, GESAMP 2001). Poorer water quality at sites with higher freshwater influence may reflect the relatively high proportions of agricultural and urban land in low-elevation areas of New Zealand (Larned et al. 2004), and subsequent effects on water quality of lowland rivers that feed estuaries (Larned et al. 2015).

The 8- and 18-year trend analyses indicated that with the exception of NOXN more monitoring sites have improving trends in nutrients, FC and ENT than degrading trends. Trends for TP and DRP showed particularly strong declines. These patterns are consistent with freshwater nutrient concentrations over the last 20 years (Larned et al. 2015), and may in part reflect reductions in freshwater phosphorus enrichment and increases in freshwater NOXN concentrations. However, it appeared that nutrient reductions in coastal waters were strongest in high salinity site classes. Nutrient reductions in these waters may also reflect improvement in point-source (sewage) discharges from urban areas.

The maps above (Figures 4-1, 4-4 and 4-7) and the origin of data used in trend analyses (Figure 4-10), show large disparities in the spread of sites around New Zealand's coastline, and numbers of sites from each council contributing to analyses. These patterns are in stark contrast to the network of water quality sampling sites available for state and trend analysis of New Zealand's fresh waters. For example, 20 year trends for nutrients NOXN and DRP can be conducted on over 200 sites in New Zealand's rivers with a comparatively even spread around the country (Larned et al. 2012, Larned et al. 2015). In comparison, 18-year trend analysis for DRP, NOXN and CHLA in this study of coastal waters were carried out almost exclusively on data from the Auckland and BOPRC regions and the New River Estuary (Southland). Contrasting patterns in the high-quality datasets from these regions provide an example of why assessment of national representativeness may be important when interpreting the results of the preceding sections. Many sites from the Auckland region show large decreases in nutrients (DRP, TP, NHXN, and NOXN) as well as CHLA and TURB. Possible reasons for these changes include the expansion of Auckland's urban areas at the expense of agriculture, (leading to a switch from diffuse nutrient loading to discharge from treated wastewater) and an improvement in wastewater treatment plant performance, although many of the sites showing improving trends in this dataset still show degraded water quality (Pers. Com. Jarrod Walker, Senior Marine Scientist, Auckland Council). Sites within the New River Estuary, Southland, showed increases in nutrients (especially DRP and NOXN), and increases in CHLA. The catchments of the New River Estuary have seen an intensification in agriculture over the period of this trend analysis, and the estuary itself is highly eutrophic and consistently rates poorly in terms of ecological condition (Robertson and Stevens 2013). The Auckland (40 sites), BOPRC (55 sites), and New River Estuary (9 sites) datasets provide excellent information on trends in water quality at the local scale, but represent only a very small portion of New Zealand's coastline. We therefore have little understanding on how well data from these areas convey trends at a national scale. It is not within the scope of this report to make a detailed analysis or comparison of regional water quality trends; these may be affected by (for example) land use changes, changes in site selection and climatic factors such as ESNO cycles. Furthermore, trends derived from all sites within a regional dataset may not convey trends within subsets of those sites. For detailed information on regional trends we direct the reader to environmental monitoring sections of council websites.

5 Review of council water quality programmes

5.1 Representative sampling site selection and distribution

There are regional differences in the physical geography of New Zealand coastal hydrosystems, resulting in regional variation in riverine and coastal hydrology, nutrient, sediment and other contaminant flows, and seasonal primary production (Hume 2007). If an aim of sampling is to give an average condition of water quality, then sampling effort needs to takes this predictable variation into account; we use the term 'representativeness' to refer to the degree to which a monitoring programme accomplishes this. In this section we examine whether the current coastal water quality sampling network is representative with respect to variation 1) at the national scale, 2) in hydrosystem typology and 3) within individual hydrosystems.

5.1.1 National scale

Not all regional councils regularly collect coastal water quality data, and among those councils that do, there have been large differences in sampling effort. This results in significant spatial gaps and differing lengths of record at the national scale, creating bias in state (e.g., median values, percentiles) and trend analyses presented in this report. For example, in 18-year trend analysis for DRP and TP the majority of sites analysed were in the Auckland region. Trends in DRP in TP in Auckland have been influenced by land use and wastewater treatment changes which may not have taken place equally over the rest of the country. Therefore we do not know that the Auckland site-specific state and trend analyses in this report represent national-scale state and trends of coastal water quality.

5.1.2 Coastal hydrosystem typology

Different types of coastal hydrosystems vary in residence times and are likely to vary in their susceptibility to land use change and associated changes in water quality (Section 3.2). Table 5-1 gives the distribution of sites from which data was provided for this report across ETI classes. For this table we include sites that were removed from state and trend analysis by our filtering rules. Many of these sites were only recently established, and were omitted from state and trend analyses based on their relatively short time series of data, but are likely to contribute to future analyses.

Table 5-1: Number of monitoring sites by coastal hydrosystem and council that were represented in the full dataset.The numbers shown include sites excluded by filtering rules used for state and trend analyses.

Council	Total sites	Coastal hydrosystem				
		ICOLL	SSTRE	SIDE	DSDE	Open Coast
AC	40	0	0	21	15	4
BOPRC	55	1	7	32	0	15
CRC	55	0	0	10	15	30
ES*	9	0	0	8	0	1
WRC	25	0	1	6	6	12
GDC	24	1	1	1	0	21
GWRC	8	0	0	8	0	0
HBRC	43	0	28	0	0	15
HRC	24	0	12	0	0	12
MDC	46	0	0	0	38	8
NRC	42	0	0	15	27	0
SUM OF COUNCILS	371	2	49	101	101	118

*Data in this dataset provided by Invercargill City Council

We also present the sampling effort of councils in terms of numbers of coastal hydrosystems of each type sampled, relative to the number in each region. We separate this analysis across ETI classes, and include councils that did not contribute to the dataset presented in this report. For this analysis, counts of the coastal hydrosystems in each class in each region were taken from the Coastal Explorer database (Hume et al. 2007), and counts of coastal hydrosystems sampled were taken from data provided to us for this project. Coastal hydrosystem classes contained in the Coastal Explorer database were converted to ETI class according to Table 5-2. Note that the descriptive names we provide in Table 5-2 are based on the narrative given in Hume et al. (2007) and Hume et al. (2016).

Table 5-2: Mapping of coastal hydrosystems contained in Coastal Explorer (Hume et al. 2007) to ETI class (Robertson et al. 2016a).We note that ETI class ICOLL is currently being revised as a subclass of SIDE and SSRTRE but for this analysis (Table 5-2) we retained the definition given in Robertson et al. (2016a) and the state and trend analyses above. Accordingly, we have mapped the ETI class ICOLL to the Hume et al. category A (Coastal Lake).

Hume et al. (2007) Category	Descriptive name	ETI Class
A	Coastal Lake	ICOLL
B	Tidal River Mouth	SSRTRE
C	Tidal River Lagoon	SSRTRE
D	Coastal Embayment	DSDE
E	Tidal Lagoon	SIDE
F	Barrier-enclosed lagoon	SIDE
G	Fjord/Sound	DSDE
H	Sound	DSDE

The coastal hydrosystems listed in Table 5-3 were found in the council data provided for this project but are not in the Coastal Explorer database. For example, BOPRC monitors the ICOLL Matata Lagoon, but no ICOLLs in the Bay of Plenty region are included in the Coastal Explorer. We also note that Kaipara Harbour System is on the boundary between NRC and AC. Both councils sample in Kaipara Harbour, and it is included in the counts for both councils.

Coastal hydrosystem name	ETI class	Council
Matata Lagoon	ICOLL	BOPRC
Oamaru Bay	DSDE	WRC
Hamanatua Stream	SSRTRE	GRC
Tukituki River	SSRTRE	HBRC
Mohaka River	SSRTRE	HBRC
Hokio River	SSRTRE	HRC
Kaikokopu River	SSRTRE	HRC
Mowhanau River	SSRTRE	HRC
Wairarawa River	SSRTRE	HRC

Table 5-3: Coastal hydrosystems found in council data not in the Coastal Explorer database (Hume et al. 2007).

In Table 5-4 we show the Coastal Explorer database counts of estuaries for each region, compared to their occurrence in the datasets provided for this report**.**

Table 5-4: Number of coastal hydrosystems that were represented in the full dataset, relative to the number of coastal hydrosystems of each ETI class in each region, and sampling effort required to give representative coverage to coastal hydrosystem types in each region. We note that sites counted as present in each region are derived from the Coastal Explorer database that informs Hume et al. (2007), and these counts are conservative; 9 coastal hydrosystems were included in council sampling nationwide that were not in the Coastal Explorer database. Abbreviations of councils not included previously in this report are: CIDC = Chatham Islands District Council, NCC = Nelson City Council, ORC = Otago Regional Council, TDC = Tasman District Council, TRC = Taranaki Regional Council, WCRC = West Coast Regional Council.

Council	All hydrosystems		Hydrosystem								Representative hydrosystem percentages			
			ICOLL		SSRTRE		SIDE		DSDE		ICOLL	SSRTRE	SIDE	DSDE
	All†	Data††	All	Data	All	Data	All	Data	All	Data				
AC	47	6	0	0	2	0	24	6	21	0	0	4	51	45
BOPRC	11	10	0	1	7	5	4	4	0	0	0	64	36	0
CIDC	1	0	0	0	0	0	1	0	0	0	0	0	100	0
CRC	41	4	6	0	10	1	3	3	22	0	15	24	7	54
WRC	50	10	0	0	10	1	29	7	11	2	0	20	58	22
GRC	10	3	0	0	8	1	2	2	0	0	0	80	20	0
GWRC	23	2	6	0	8	1	1	1	8	0	26	35	4	35
HBRC	16	4	7	0	7	4	2	0	0	0	44	44	13	0
HRC	8	10	0	0	8	10	0	0	0	0	0	100	0	0
MDC	14	2	1	0	1	0	4	0	8	2	7	7	29	57
NCC	4	0	0	0	0	0	4	0	0	0	0	0	100	0
NRC	59	5	2	0	3	0	32	5	22	0	3	5	54	37
ORC	22	0	3	0	6	0	13	0	0	0	14	27	59	0
ES*	29	1	2	0	4	0	7	1	16	0	7	14	24	55
TDC	33	0	0	0	8	0	18	0	7	0	0	24	55	21
TRC	11	0	0	0	11	0	0	0	0	0	0	100	0	0
WCRC	37	0	6	0	28	0	3	0	0	0	16	76	8	0
Total	**416**	**57**	**33**	**1**	**121**	**23**	**147**	**29**	**115**	**4**	**8**	**29**	**35**	**28**

* Data provided by ICC, † All hydrosystems in database informing Hume et al. 2007, †† All hydrosytems in council data informing this report.

From Table 5-4 we can see that:

1. With one exception (HRC), the number of coastal hydrosystems recorded in each ETI class in each region is greater than the number sampled.
2. The number of coastal hydrosystems in each class sampled was rarely in proportion to the number in each class that exist within each region. For example ICOLLs and DSDEs were often severely under-sampled whereas SIDES were often oversampled, relative to their numbers within regions.

The unbalanced and disproportionate sampling efforts noted above may have arisen in many cases because sampling sites were selected to monitor issues within specific estuaries (see Section 5.1.3, below). Furthermore, data from low-salinity hapua/ICOLL sites was not provided by some councils where those sites were categorized within council freshwater monitoring programmes (Pers. Com. Lesley Bolton-Richie, CRC). While we believe that the proportions of hydrosystem types provided in the coastal explorer database are reasonably close to their actual proportions, we recommend that each council confirms proportions within their region if allocating monitoring effort based on hydrosystem typology.

5.1.3 Individual hydrosystem scale

Physical conditions at prospective monitoring sites within hydrosystems affect water quality conditions and the degree to which a given site is representative of a larger area. For example, water depth, proximity of the benthos, and whether the site can be sampled at all states of tide are likely to vary at different sites within a single coastal hydrosystem. If an objective is to give an unbiased picture of an entire hydrosystem, sampling needs to be distributed spatially in a representative fashion. There has been no national guidance for site selection during the period that the data described in this report were collected. Currently sites within individual hydrosystems are selected using a variety of criteria, including ease of access, monitoring of likely points of water quality change, and community interest. To examine how sites from which the dataset used in this study were selected, site selection criteria were requested from coastal scientists at contributing councils, and are detailed below.

Auckland Council

Most of the sites in the AC dataset are in major estuaries within the region, and a subset of sites were selected to be spread evenly within those estuaries. However, additional sites were selected to monitor likely points of water quality change within estuaries. For example, sampling sites were selected at the entry point of major rivers to Kaipara Harbour, as well as a central point in the harbour. In the Manukau Harbour, some sites are located adjacent to the Mangere Wastewater Treatment Plant to monitor potential impacts from effluent.

Bay of Plenty Regional Council

Bay of Plenty coastal water quality monitoring sites were selected to be representative of particular water bodies, i.e. site locations were intended to minimise the influence of single streams, drains or other discharge sources, if the impact of those discharges did not represent the larger part of that hydrosystem or area. Site selection was limited by practical considerations such as access to water of the required sampling depth from the shore. Additional sites were selected to monitor sensitive waters (i.e. enclosed hydrosystems as opposed to open coastal ocean sites).

Canterbury Regional Council

Canterbury Regional Council monitors 30 sites along the Canterbury coastline with the aim of comparing baseline water quality (sites at 3km offshore) with water quality close to potential influence of river plumes, wastewater and other discharges and land runoff. The CRC dataset also includes short-term investigations in Akaroa and Lyttleton Harbours, and regular sampling to monitor wastewater influence and river impacts on the Avon-Heathcote Estuary. The Avon-Heathcote data only includes sites fringing the estuary that can be sampled on foot.

Invercargill City Council

The ICC dataset is derived from of 8 sites spread around the New River Estuary, Southland, and a single site on the open coast at Oreti Beach, 8 km north of the harbour entrance. Within the estuary, sites are placed at the entrance points of the major rivers, and adjacent to the former tip site at Pleasure Bay.

Waikato Regional Council

No information received on site selection criteria.

Gisborne District Council

Water quality monitoring sites at the terminal reaches of major rivers in the Gisborne region were selected from existing catchment board flood warning network sites. Coastal sampling sites were selected as part of a larger water quality and quantity sampling network; coastal open water sites between large rivers were selected to monitor changes in coastal water quality outside the extent of river plumes.

Greater Wellington Regional Council

GWRC’s water quality monitoring sites are mostly limited to Porirua Harbour. The original sites, located near the estuary fringes for cost and practicality reasons, were chosen to obtain information on spatial variation in water quality within the harbour’s two arms, with an additional site included at the mouth of the harbour. The two current sites are also located near the estuary fringes where data are collected to support a seagrass restoration project. GWRC also monitors water quality at one site in Lake Onoke (an ICOLL). This site is located at the mouth of the Ruamahanga River as it is the only easily sampling location that can be easily accessed without a boat.

Hawke’s Bay Regional Council

HBRC water quality monitoring sites were selected to monitor the effects of land use change on coastal waters. Sites were located off major river mouths, and at locations along the coast that would enable representation of coastal water quality with a limited number of sites. Estuarine sites have recently been added to the HBRC network to inform regional plan changes.

Horizons Regional Council

No information received on site selection criteria.

Marlborough District Council

Coastal water quality monitoring sites are located along the main axes of Pelorus and Queen Charlotte Sounds. Water quality monitoring in these sounds was initially driven by shellfish health monitoring requirements, and subsequently expanded to form the current State of the Environment (SoE) monitoring network.

Northland Regional Council

The Northland coastal water quality network was designed for SoE monitoring. Site selection was made to give the best possible spatial representation of harbours by evenly distributing sites throughout the harbours. While point source discharges were not specifically targeted, some sites are affected by point source discharges (e.g., one site in Whangarei is downstream from the city's wastewater treatment plant discharge). Monitoring is confined to major harbours, and there are currently no sites in the NRC network on the open coast, except Mair's Bank at the entrance to Whangarei Harbour.

5.2 Variables measured

All water quality variables included in the datasets provided by councils are listed in Table 5-5. In general, the variables that we used for state and trend analysis were also the most commonly collected variables by Councils. We note however that issues in interpretation of these data still emerge because of regional differences in monitoring variables. For example, our national analysis of 8-year trends (Table 4-3) show general improvement in visual clarity, but declines in TURB and SS. These patterns are counterintuitive at the national scale because TURB and SS are strongly inversely correlated with visibility (Hicks et al. 2016). It is likely that this issue emerges because the visual clarity data that meet our filtering rules originate only from NRC, while SS and TURB data are derived primarily from AC and CRC datasets. Over time, consensus on variables included in regional sampling programmes would alleviate this issue. The NEMS Discrete Water Quality *(in prep)* should assist with this through inclusion of recommended core and supporting variables for monitoring coastal waters.

Table 5-5: Variables in coastal water quality datasets provided by Councils. The site numbers and numbers of data points listed refer to data prior to the application of filtering rules used for state and trend analyses.

Variable	Number of sites in database	Oldest record	Newest record	Total number of data points	Councils with data
Water temperature (TEMP)	335	3/08/1973	1/03/2016	34,893	AC, BOPRC, CRC, ICC, WRC, GDC, GWRC, HBRC, HRC, MDC, NRC
Enterococci (ENT)	259	13/01/1991	25/02/2016	30,478	AC, BOPRC, CRC, ICC, WRC, GDC, HBRC, HRC, NRC
pH (PH)	220	8/01/1976	1/03/2016	27,796	AC, BOPRC, CRC, ICC, WRC, GDC, HBRC, HRC, NRC
Salinity (SAL)	239	27/06/1978	19/02/2016	26,970	AC, BOPRC, CRC, ICC, WRC, GDC, GWRC, MDC, NRC
Ammoniacal-N (NHXN)	283	25/10/1983	1/03/2016	25,635	AC, BOPRC, CRC, ICC, WRC, GDC, GWRC, HBRC, HRC, MDC, NRC
Dissolved reactive phosphorus (DRP)	278	14/12/1977	1/03/2016	25,517	AC, BOPRC, CRC, ICC, WRC, GDC, GWRC, HBRC, HRC, MDC, NRC

Variable	Number of sites in database	Oldest record	Newest record	Total number of data points	Councils with data
Faecal coliforms (FC)	234	3/08/1973	24/02/2016	25,069	AC, BOPRC, CRC, ICC, WRC, GDC, HBRC, HRC, NRC
Dissolved oxygen concentration (DO)	243	3/08/1973	1/03/2016	24,245	AC, BOPRC, CRC, ICC, WRC, GDC, GWRC, HBRC, HRC, MDC, NRC
Chlorophyll-*a* (CHLA)	259	13/01/1991	1/03/2016	20,890	AC, BOPRC, CRC, ICC, WRC, GWRC, HBRC, HRC, MDC, NRC
Turbidity (TURB)	262	5/10/1987	1/03/2016	20,569	AC, BOPRC, CRC, WRC, GDC, GWRC, HBRC, HRC, MDC, NRC
Conductivity	206	18/04/1977	1/03/2016	20,365	AC, BOPRC, ICC, WRC, GDC, GWRC, HBRC, HRC, MDC
Nitrate-N	159	19/04/1988	1/03/2016	18,998	AC, CRC, ICC, GDC, GWRC, HBRC, HRC, MDC, NRC
Total phosphorus (TP)	251	25/10/1983	1/03/2016	18,958	AC, BOPRC, CRC, WRC, GDC, GWRC, HBRC, HRC, NRC
Total suspended solids (SS)	275	21/07/1976	1/03/2016	18,820	AC, BOPRC, CRC, WRC, GDC, GWRC, HBRC, HRC, MDC, NRC
Dissolved oxygen saturation	228	5/10/1987	1/03/2016	16,845	AC, BOPRC, CRC, WRC, GDC, GWRC, HBRC, HRC, MDC, NRC
Nitrate + nitrite (NOXN)	221	14/12/1977	19/02/2016	15,932	AC, BOPRC, CRC, WRC, GDC, GWRC, HBRC, NRC
Total coliforms	82	3/08/1973	25/02/2016	13,863	AC, ICC, WRC, GDC, HRC
Total nitrogen (TN)	273	12/04/1989	1/03/2016	10,769	AC, BOPRC, CRC, WRC, GDC, GWRC, HBRC, HRC, MDC, NRC
Nitrite-N	113	22/09/1992	1/03/2016	8,711	AC, GDC, GWRC, HBRC, HRC, NRC
Visual clarity (CLAR)	133	5/10/1987	19/02/2016	6,841	AC, BOPRC, WRC, HBRC, HRC, MDC, NRC
E. coli	140	7/01/1995	1/03/2016	6,754	AC, BOPRC, CRC, WRC, GDC, HBRC, HRC
Total dissolved phosphorus	36	13/01/1991	1/03/2016	6,461	ICC, HRC, MDC
Chloride	62	18/04/1977	9/02/2016	6,399	AC, ICC, WRC, GDC, HRC, NRC
Total Kjeldahl Nitrogen	161	19/04/1988	19/02/2016	5,825	AC, BOPRC, CRC, WRC, GWRC, HBRC, HRC, NRC
Dissolved inorganic nitrogen	13	13/01/1991	8/12/2015	5,491	ICC, GWRC, HBRC
Biological oxygen demand	37	3/08/1973	24/09/2014	5,192	AC, WRC, GDC, HRC

Variable	Number of sites in database	Oldest record	Newest record	Total number of data points	Councils with data
Depth of water at sampling site	60	4/10/1999	19/02/2016	2,427	WRC, GDC, HBRC, NRC
Volatile solids	25	18/08/1997	13/01/2016	1,869	CRC, HRC, MDC
Chlorophyll to pheophytin ratio	26	3/07/2001	22/06/2005	1,426	AC
Total dissolved nitrogen	24	20/07/2011	16/12/2015	1,092	MDC
Dissolved silica	26	14/12/1999	9/02/2016	987	WRC, HRC, MDC
Total organic nitrogen	31	5/11/2007	1/03/2016	949	HBRC, HRC
Dissolved organic carbon	11	24/01/2007	5/11/2012	836	CRC
Particulate organic carbon	24	20/07/2011	19/06/2014	635	MDC
Particulate organic nitrogen	24	20/07/2011	19/06/2014	635	MDC
Inorganic suspended solids	12	20/07/2011	16/12/2015	557	MDC
Total organic carbon	12	21/01/2009	13/01/2016	444	BOPRC, CRC
Particulate Carbon	24	23/07/2014	16/12/2015	385	MDC
Particulate nitrogen	24	23/07/2014	16/12/2015	385	MDC
Total copper	33	16/07/2015	21/01/2016	132	NRC
Total lead	33	16/07/2015	21/01/2016	132	NRC
Total zinc	33	16/07/2015	21/01/2016	132	NRC
Dissolved sodium	2	8/12/1999	9/02/2016	119	GDC, HRC
Potassium	2	8/12/1999	9/02/2016	119	GDC, HRC
Sulphate	2	8/12/1999	9/02/2016	119	GDC, HRC
Calcium hardness	1	8/12/1999	24/09/2014	106	GDC
Total hardness (CaCo3 + MgCo3)	1	8/12/1999	24/09/2014	106	GDC
Magnesium hardness (MgCO3)	1	8/12/1999	19/03/2003	40	GDC
Percent C (SS)	6	8/08/2006	9/11/2011	35	BOPRC

Variable	Number of sites in database	Oldest record	Newest record	Total number of data points	Councils with data
C:N ratio of particulate solids	3	8/08/2006	9/11/2011	29	BOPRC
Alkalinity	1	4/08/2015	9/02/2016	13	HRC
Dissolved calcium	1	4/08/2015	9/02/2016	13	HRC
Dissolved iron	1	4/08/2015	9/02/2016	13	HRC
Dissolved magnesium	1	4/08/2015	9/02/2016	13	HRC
Dissolved manganese	1	4/08/2015	9/02/2016	13	HRC
Fluoride	1	4/08/2015	9/02/2016	13	HRC
Total bromide	1	4/08/2015	9/02/2016	13	HRC
N:P ratio in water	3	21/09/2010	9/11/2011	12	BOPRC
Percent N (SS)	3	8/08/2006	12/03/2009	8	BOPRC
Total petroleum hydrocarbons	2	4/11/2005	26/11/2007	4	GDC
Dissolved organic phosphorus	2	19/03/2015	8/06/2015	3	GDC
Total dissolved salts	1	28/12/2005	4/01/2006	2	GDC
Hydrogen oxygen ratio of water	1	22/05/2006	22/05/2006	1	WRC
Oxygen isotope ratio of water	1	22/05/2006	22/05/2006	1	WRC
Sediment particle size	1	21/01/2009	21/01/2009	1	BOPRC
Total dissolved solids	1	13/10/2010	13/10/2010	1	BOPRC

5.3 Collection methods

5.3.1 Collection platform

Collection platform (e.g., boat, wading) can influence the comparability of water quality datasets, for three general reasons. First, accurate measurement of some variables may not be possible from some platforms. Examples include visual clarity and light penetration, which are difficult to measure accurately from a helicopter due to large distances between the observer or operator and the surface of the water. Second, more mobile platforms (boat, helicopter) provide access to deep areas of water bodies not accessible on foot, which are likely to differ biologically, physically and chemically from shallow waters. Third, more mobile platforms enable rapid movement between sampling sites, so that many sites can be sampled at similar, comparable, tidal state. Samples represented in this report were collected on foot, from boats and from helicopters. Data on sampling platform was not provided by all councils, and the proportion of data collected from each platform varied between councils. Furthermore, councils often collected data using several platforms. For example, CRC collects offshore data from helicopters while data from the Avon Heathcote Estuary time series are collected from the shore by foot.

5.3.2 Collection depth

Sample collection or measurement depth affects water quality data, particularly at sites in deep estuaries and open coasts. Values of variables vary with depth because of factors including physical stratification, variable uptake of nutrient by primary producers with depth (and light environment), and proximity of sediments, which may influence turbidity and release entrained nutrients.

With the exception of 12 sites in the Marlborough Sounds within the MDC dataset, all of the data in this report were from surface-water samples and measurements. However, surface-water sampling methods are not currently standardised between councils and the surface water sampling depths provided by councils varies between a 'surface grab sample' and an integrated tube sample taken from the top 15 m. A list of sampling methodologies is available in the metadata associated with this dataset, 'metadata.xls'.

5.3.3 Collection timing

Seasonal timing is important because coastal water quality and ecosystem health typically vary seasonally. Expressions of ecosystem deterioration could be most pronounced in some seasons (e.g., dissolved oxygen decreases in bottom waters and acidification in the whole water column in the Firth of Thames occurs most strongly in late summer and autumn (Zeldis et al. 2015)). Water-column nutrient concentrations may vary seasonally, even in eutrophic environments, due to depletion of nutrients during summer growth maxima among primary producers.

Timing with respect to tidal state also affects coastal water quality datasets; in many coastal hydrosystems, sampling at high tide can bias the results toward oceanic (high salinity) conditions, while sampling at low tide will bias the results towards riverine (low salinity) conditions. The differences between tidal states vary with oceanic dilution of estuary water (i.e. tidal state is more important in estuaries with a high ratio of tidal prism to volume (e.g., Tauranga Harbour) than when this ratio is low (e.g., Pelorus Sound).

For some water quality variables (e.g., DO, TEMP), there is substantial diurnal variation in measurement values. Time-of-day of sampling is also relevant for visual clarity and light penetration measurements. Both optical attributes must be measured during daylight hours and both are weakly dependent on incident lighting.

While the data filtering rules employed in our methods ensure that seasonal variation is accounted for in state and trend analyses, no adjustments have been made for tidal state or sampling times. There is as yet no national consensus on tide or time standardisation. Some councils collect across all tidal states (e.g., NRC) other councils standardise to a single tidal state (e.g., AC), and others repeat collections at high and low tide for a given sampling date (ICC sampling in the New River Estuary, Southland). While each of these approaches is suitable for regional monitoring and reporting, the aggregated council data used for national reporting may be affected by differences among councils in sample timing. For example, if AC collections are performed routinely at high tide they may be more representative of oceanic waters than those of NRC that are collected across all tidal states.

5.4 Analytical methods

Consistency of laboratory analytical methods is an essential part of a national monitoring strategy, as it reduces regional bias arising from differences in methods (Davies-Colley et al. 2011). The dataset we analysed for state and trends initially contained data from a wide variety of laboratory analytical methods of unknown comparability. Steps involved in data processing to allow analysis of presumably comparable data are described in Section 2.3, although we note that this process results in the exclusion of a significant portion of the sampling effort represented in the full dataset. Furthermore, there is scope for further work on defining comparable methods for measuring each variable. This work should include parallel analyses of split samples using alternate methods, followed by statistical comparisons of the measurement data (e.g., Hamilton et al. 2005, Horowitz 2013). A full list of analytical methods used in data provided for this report is provided in the Microsoft Excel file accompanying this report 'Metadata.xls'.

5.5 Data quality

Because data quality codes were not included in all council datasets, all of the data provided to us were included in data processing regardless of the quality. Ignoring data quality codes prevented the introduction of biases associated with removing low-quality data from some but not all regions. However, it may have caused an overall reduction in data quality. All data quality information that we received is included in the full dataset accompanying this report.

5.6 Non-council datasets that may supplement national analyses

Metadata for known supplementary non-council New Zealand coastal water quality data sources are shown in Table 5-6 (discrete water quality sampling data) and Table 5-7 (continuous water quality sampling data). The tables are not exhaustive (e.g., very little consent monitoring data has been accessed). We restricted the inclusion of datasets to those that include core and supporting variables recommended by NEMS for future council monitoring (see Section 6.3), that have at least 5 years of accumulated quarterly or monthly data, and for which the collection and analysis methods are presumably compatible with the data included in the present study. Therefore the data sources in Tables 5-6 and 5-7 may supplement regional sampling networks in future analyses. We note that recreational water quality monitoring has not been included in Table 5-6 on the understanding that these data are commonly only collected during the bathing season; the seasonal restriction means the data are not comparable with the other council datasets analysed in this report.

Table 5-6: Non-council datasets that may supplement future national analyses of core and supporting variables.

Name of Database	Owner	Location type (location)	Variables measured	Length of monitoring (years)	Access constraints
Seawater Nutrients	Hurd, UoO	2 coastal	NOXN, NHXN, DRP	>10	restricted
Lyttleton Dredge Spoil Monitoring	Lyttelton Port Company (LPC)	1 estuary	SAL, TEMP, DO, CLAR	>10	restricted
Physical and Biological Monitoring in Doubtful Sound	Meridian Energy Limited	2 estuaries	SAL, TEMP	>10	restricted
CTD (Fisheries Oceanography)	MPI Research Data Manager	EEZ	COND, TEMP	>10	restricted
Bell Island Receiving Waters Survey	Nelson Regional Sewerage Business Unit (NRSBU)	1 estuary	NHXN, NOXN, TN, DON, DRP, TP, SAL, TEMP, CHLA, TURB, FC, ENT	>10	restricted
Sea Temperature	NIWA	9 estuaries, 3 coastal	TEMP	>10	public

Name of Database	Owner	Location type (location)	Variables measured	Length of monitoring (years)	Access constraints
Sea Temperature	UoA	1 coastal	TEMP	>10	public
Sea Temperature	UoO	1 estuary	TEMP	>10	public
Cross-Shelf Exchange (C-SEX) CTD series	NIWA	1 estuary (Hauraki Gulf/Firth of Thames)	SAL, DO, COND, CLAR, TEMP, DRP, TP, TN, NOXN, NHXN, CHLA, PH, light penetration, particulate carbon, particulate organic carbon, particulate nitrogen.	>10	restricted
Marlborough Shellfish Quality Programme (MSQP)	NIWA/Sanford Ltd	Pelorus and Queen Charlotte Sound	CHLA, SAL, TEMP, DRP, TP, TN, NOXN, NHXN, CHLA, CLAR, particulate carbon, particulate organic carbon, particulate nitrogen.	>10 years	restricted

Table 5-7: **Non-council datasets that include variables measured using continuous sampling methods.**

Name of Database	Owner	Location type	Variables measured	Length of monitoring (years)	Access constraints
Sea surface temperature and ocean colour using satellite remote sensing	NASA	EEZ	TEMP, CHLA, SS, dissolved organic matter.	>10	Public. Proper use is constrained by considerable processing challenges to derive useful products in coastal regions.
Cross-Shelf Exchange (C-SEX) mooring series	NIWA	1 estuary (Hauraki Gulf/Firth of Thames)	SAL, DO, COND, TEMP, CHLA, PH, light penetration	>10 years	Restricted.

6 Recommendations for variables and sampling protocols

6.1 Purposes of coastal water quality and ecology monitoring

Some overarching principles in national coastal water quality and ecology monitoring are:

- Central government agencies want to gain a nationwide view of state and trends of water quality, whereas councils have a regional focus and may wish to also carry out monitoring that addresses local issues (e.g., impacts from a catchment dominated by a particular development or land use).
- The national and regional monitoring objectives both lie within the ambit of regional council activities. While regional councils do not have a mandate to provide for national monitoring, their monitoring programmes need to achieve national standards, while being able to address regional issues. The current drive toward consistency between council monitoring that is being promoted by NEMS is an important way forward.
- The monitoring toward these objectives by councils will overlap, i.e. a subset of sites that address regional monitoring objectives will likely fit within a nationally representative network. In national-scale reporting, subdivision by regions may not be appropriate and it is only the individual sites that are important.

The role of this section (and the NEMS) is to provide guidance for councils on monitoring programmes that can assist with achieving regionally relevant monitoring while also contributing to robust national state and trends analyses.

6.2 Site selection

There has never been a national sampling strategy put in place in New Zealand to monitor coastal water quality. Section 5 demonstrates the lack of representativeness in sampling to date when council data is viewed at a national scale. The following recommendations are made regarding site selection for council sampling if an intention is to use council sites to monitor and report on state and trends in both regional and national coastal water quality:

- Sites included in a national network should be replicated sufficiently with respect to environmental classes of catchment land use. This is important if an aim of national reporting is to make comparisons between impacted environmental classes and reference classes, and among impacted environmental classes (Larned and Unwin 2012).
- Sites included in a national network should be proportional across hydrosystem types, using the percentages provided in Table 5-4. This is important because there are differences in the physical geography of New Zealand coastal hydrosystems that create differences in the effects of stressors on water quality. Proportional sampling across hydrosystem types would be aided by maintenance of coastal water quality sampling programmes by as many councils as possible, because some hydrosystem types are more common in some regions.
- Sampling at hydrosystems included in a national network should include at least one site within each selected hydrosystem, as well as a site on the terminal river reach and in the open coast. Terminal reach and open coastal sampling (the two nutrient-contributing end-members) aid assessment of land based effects on water quality within the estuary, by allowing an understanding of the relative importance of nutrient loading from these two

sources. This provides information for integrated management of the effects of use and development of land and freshwater on coastal water.

The numbers of sampling sites monitored is an important sampling network design consideration because site numbers will influence the level of precision with which statements about water quality in coastal hydrosystems can be made. Sampling across regions, and proportionally across hydrosystem types will provide a representative monitoring network, but if the number of monitoring sites is low, estimated states and trends may be accurate but imprecise (i.e., there will be large uncertainties (e.g., standard error, confidence intervals) around means, medians, or trend lines). In most cases precision increases as the number of monitoring sites increases, and the number of sites required to achieve a desired minimum level of precision for a monitoring variable can be estimated using existing datasets (see methods of Larned and Unwin 2012).

Further considerations for site selection for water quality monitoring networks, or selecting sites from existing networks for analysis depend on monitoring purposes. These are summarised in Table 6-1. The reader is referred to the coastal chapter of the NEMS for Discrete Water Quality (*in prep*) for more site-specific sampling location considerations, including methods for representative sampling of individual hydrosystems.

Table 6-1: Purposes and design criteria of national and regional coastal hydrosystem water quality monitoring networks

Purpose	Primary design criterion	Secondary design criteria
Assessment of state of coastal hydrosystem health	Representativeness (hydrosystem types well represented, good spatial coverage nationwide)	Good spatial coverage within individual hydrosystems, temporal representativeness (e.g., replication across tidal state and seasons).
Assessment of trends in coastal hydrosystem health	Representativeness (hydrosystem types well represented, good spatial coverage nationwide)	Length of time series, consistency (e.g., tide state, seasons) of replication in time.
Comparisons of impacted environmental classes *versus* reference classes	Representativeness (hydrosystem types well represented, good spatial coverage nationwide, good coverage of coastal hydrosystem states). Power sufficient to resolve differences between reference and impacted coastal hydrosystems.	Considerations for both state and trend analysis, above, apply.
Threshold development	Availability of indicator data suitable for generating scores for ecosystem health or other values (e.g., recreational suitability).	WQ monitoring variables that inform regarding values are important. Information on hydrology and mass flows at terminal reaches of rivers are important.

6.3 Core and supporting water quality variables

Core variables (those thought fundamental to assessing coastal water quality) and supporting variables (those that provide valuable supporting information on coastal water quality) selected in NEMS (*in prep.*) for future monitoring efforts are listed in Table 6-2. We note that most of these variables are also recommended for assessment of water quality in New Zealand's fresh waters (Davies-Colley et al. 2012).

Table 6-2: Recommended core and supporting water quality variables. Table modified from NEMS *in prep*.

Variable	Core	Supporting	Abbreviation/s in this report	Values addressed (rationale)
Major physico-chemical variables				
Salinity	✓		SAL	Ecosystem health ('master' variable measuring freshwater content)
Water temperature	✓		TEMP	Ecosystem health (global change)
Dissolved oxygen	✓		DO	Ecosystem health
pH	✓		PH	Ecosystem health (local and global change)
Optical variables				
Visual clarity	✓		CLAR	Ecosystem health; Recreation
Turbidity		✓	TURB	(Proxy for visual clarity or suspended particle concentration; continuously measurable)
Total Suspended Solids	✓		SS	Ecosystem health; Recreation
Light penetration		✓	Not reviewed in this report	Ecosystem health
CDOM		✓	Not reviewed in this report	Ecosystem health
Munsell Colour		✓	Not reviewed in this report	(QA for water quality)
Nutrients				
Total nutrients	✓		TN, TP	Ecosystem health
Dissolved nutrients	✓		NOXN, NHXN, DRP*	Ecosystem health
Microbiological indicators				
Enterococci	✓		ENT	Recreation; Shellfish aquaculture
Faecal coliforms		✓	FC	Recreation; Shellfish aquaculture
E. coli		✓	Not reviewed in this report	Recreation; Shellfish aquaculture
Chlorophyll-*a*	✓		CHLA	Ecosystem health
Phytoplankton assemblage		✓	Not reviewed in this report	Ecosystem health

* Deemed supporting variable in fully marine (oceanic) waters

6.3.1 Rationale for variable selection

All except five of the variables selected in the development of the NEMS as core or supporting variables (Table 6-2) were analysed for state and trends in this report, and rationale for measurement of those variables is provided in Section 2.1. Rationale for measurement of the five additional variables selected in Table 6-2 is provided below, although we note that the NEMS variables are not yet finalised.

6.3.2 Additional optical variables

Three optical variables (Light penetration, CDOM and Munsell colour) are recommended as supporting variables by NEMS (*in prep*) in addition to those optical variables analysed for state and trends in this report (CLAR, SS and TURB). The first additional optical variable, light penetration, is important because together with water depth it controls the light available to benthic plants (e.g., seagrasses) which contribute strongly to the ecological functioning of estuaries. We note that light penetration is often not well represented by visual clarity (CLAR; measured by Secchi depth or black disc), and must be measured explicitly by profiling with appropriate light sensors (Hicks et al. 2016).

Munsell colour is a valuable observation on water optical character that can be useful in QA of water quality. CDOM is a useful index of freshwater content of water that correlates inversely with salinity (e.g., Gall et al. *in review*). These two variables are recommended because of their potential application in remote sensing of estuarine and coastal water quality as well as relationship to water values. CDOM is also of interest in the context of export of organic carbon from land to the ocean.

We note that no records of light penetration, CDOM or Munsell colour were included in the data provided for this report, i.e. they currently do not appear to be a significant component of council sampling programmes.

6.3.3 Additional microbiological indicators

In addition to monitoring of ENT and FC, *E. coli* is recommended as a secondary variable where modelling the fate of river plumes/freshwater contamination is of interest. *E. coli* is a variable of primary importance in fresh waters but is often absent from coastal monitoring due to its poor tolerance to salt water. Nevertheless it is seen as a valuable link between monitoring of freshwaters, where *E. coli* is the primary microbiological indicator of faecal pollution, and saline waters, because most of the faecal contamination that affects coastal sites is typically delivered by rivers. In this study *E. coli* data were provided by 7 of the 11 contributing councils, the complete dataset comprising 6,754 samples from 140 sites.

Phytoplankton assemblages are included as a secondary variable to provide information on trophic state in coastal waters. An example is cyanobacterial outbreaks associated with excess phosphorus availability. Phytoplankton assemblages are not a current component of council sampling programmes.

6.3.4 Current data availability of recommended variables

In table 6.3 we provide a list of all core and supporting variables recommended for council sampling by NEMS (*in prep*) along with information on how the state and trends filtering alters coverage across councils. We expect that this will aid assessment of which variables are currently suitable for national reporting, and the extent of increased coverage required for variables to be used in national reporting.

Table 6-3: Current data availability for recommended core and supporting water quality variables.

Variable	Core	Supporting	Councils that provided data			
			Total dataset	State dataset	**8-year trends dataset**	**18-year trends dataset**
Salinity (SAL)	✓		AC, BOPRC, CRC, ICC, WRC, GDC, GWRC, MDC, NRC	AC, BOPRC, CRC, ICC, WRC, GDC, GWRC, MDC, NRC	AC, BOPRC, CRC, ICC, GDC, NRC	AC, BOPRC, ICC, GDC, NRC
Water temperature (TEMP)	✓		AC, BOPRC, CRC, ICC, WRC, GDC, GWRC, HBRC, HRC, MDC, NRC	AC, BOPRC, CRC, ICC, WRC, GDC, GWRC, HBRC, HRC, MDC, NRC	AC, BOPRC, CRC, ICC, GDC, HBRC, HRC, NRC	AC, BOPRC, ICC, GDC, NRC
Dissolved oxygen (DO)	✓		AC, BOPRC, CRC, ICC, WRC, GDC, GWRC, HBRC, HRC, MDC, NRC	AC, BOPRC, CRC, WRC, GDC, GWRC, HBRC, HRC, MDC, NRC	AC, BOPRC, CRC, GDC, HBRC, HRC, NRC	AC, BOPRC, GDC, NRC
pH (PH)	✓		AC, BOPRC, CRC, ICC, WRC, GDC, HBRC, HRC, NRC	AC, BOPRC, CRC, ICC, GDC, HBRC, HRC, NRC	AC, BOPRC, CRC, ICC, GDC, HBRC, HRC	AC, BOPRC, ICC, GDC
Visual clarity (CLAR)	✓		AC, BOPRC, WRC, HBRC, HRC, MDC, NRC	AC, BOPRC, MDC, NRC	NRC	X
Turbidity (TURB)		✓	AC, BOPRC, CRC, WRC, GDC, GWRC, HBRC, HRC, MDC, NRC	AC, BOPRC, CRC, WRC, GDC, GWRC, HBRC, HRC, MDC, NRC	AC, BOPRC, CRC, GDC, HBRC, HRC, NRC	AC, BOPRC
Total Suspended Solids (SS)	✓		AC, BOPRC, CRC, WRC, GDC, GWRC, HBRC, HRC, MDC, NRC	AC, BOPRC, CRC, WRC, GDC, GWRC, HBRC, HRC, MDC, NRC	AC, BOPRC, CRC, GDC, HBRC, HRC, NRC	AC, BOPRC, GDC
Light penetration		✓	X	X	X	X
CDOM		✓	X	X	X	X
Munsell Colour		✓	X	X	X	X
TN	✓		AC, BOPRC, CRC, WRC, GDC, GWRC, HBRC, HRC, MDC, NRC	AC, BOPRC, CRC, HBRC, HRC, NRC	AC, BOPRC, CRC, HRC, NRC	X
TP	✓		AC, BOPRC, CRC, WRC, GDC, GWRC, HBRC, HRC, NRC	AC, BOPRC, CRC, WRC, GWRC, HBRC, HRC, NRC	AC, BOPRC, CRC, HBRC, HRC, NRC	AC, BOPRC
NOXN	✓		AC, CRC, ICC, GDC, GWRC, HBRC, HRC, MDC, NRC	AC, BOPRC, ICC, WRC, GWRC, HBRC, HRC, MDC, NRC	AC, BOPRC, ICC, HBRC, HRC, NRC	AC, BOPRC, ICC

NHXN	✓			AC, BOPRC, CRC, ICC, WRC, GDC, GWRC, HBRC, HRC, MDC, NRC	AC, BOPRC, CRC, ICC, WRC, GDC, HBRC, HRC, MDC, NRC	AC, BOPRC, CRC, ICC, NRC	AC, BOPRC, ICC
DRP*	✓			AC, BOPRC, CRC, ICC, WRC, GDC, GWRC, HBRC, HRC, MDC, NRC	AC, BOPRC, CRC, ICC, WRC, GDC, GWRC, HBRC, HRC, MDC, NRC	AC, BOPRC, CRC, ICC, HBRC, HRC, NRC	AC, BOPRC, ICC
Enterococci (ENT)	✓			AC, BOPRC, CRC, ICC, WRC, GDC, HBRC, HRC, NRC	AC, BOPRC, CRC, ICC, WRC, GDC, HBRC, HRC, NRC	BOPRC, CRC, ICC, GDC, NRC	BOPRC, ICC, GDC, NRC
Faecal coliforms (FC)		✓		AC, BOPRC, CRC, ICC, WRC, GDC, HBRC, HRC, NRC	AC, BOPRC, CRC, ICC, WRC, GDC, HBRC, HRC, NRC	BOPRC, ICC, GDC, HBRC, NRC	BOPRC, ICC, GDC, NRC
E. coli		✓		AC, BOPRC, CRC, WRC, GDC, HBRC, HRC	X	X	X
Chlorophyll-*a* (CHLA)	✓			AC, BOPRC, CRC, ICC, WRC, GWRC, HBRC, HRC, MDC, NRC	AC, BOPRC, CRC, ICC, HBRC, HRC, MDC, NRC	AC, BOPRC, CRC, ICC, HBRC, NRC	AC, BOPRC, ICC
Phytoplankton assemblage			✓	X	X	X	X

6.3.5 Use of integrated indices in regional council SoE reporting

The last 15-20 years have seen an international trend towards multi-variable indices to describe trophic state and ecosystem health in coastal waters following earlier approaches for freshwater systems (e.g., Carlson 1977; see review by Borja et al. 2008 for discussion of this approach). The indices generally use variables which indicate ecological outcomes (i.e. ecological 'attributes') of changed water quality. A summary of these indices is presented in Table 6-3. A general advantage of multi-variable indices over single indices is that degradation in ecosystem health in coastal waters tends to alter more than one ecological/water quality variable at a time, so examining several variables that are likely to be impacted provides a more robust estimate of ecological health and impact than single variables considered in isolation. Some multi-variable methods provide a ranking system for ecosystem health that may be appropriate as a framework for limit setting when considered with respect to ecological thresholds (e.g., Bricker et al. 2003, Robertson et al. 2016b). Quantification and threshold setting of ecological health for coastal hydrosystems also offers a pathway to developing thresholds for core water quality variables (see Section 7.2).

A disadvantage is that very few of the variables included in standard water quality sampling described the present report are used in these indices. For example the ETI that is currently being developed to assess sedimentation and eutrophication effects on New Zealand coastal hydrosystems (Robertson et al. 2016b) includes macroalgal biomass/cover, sediment organic content and muddiness, impacted invertebrate communities, and phytoplankton blooms. With the exception of phytoplankton chlorophyll, these constituent variables are not included in current water quality sampling, although some have been collected by councils as part of their wider coastal monitoring programmes.

Given the advantages of integrated index approaches, absence of sampling for components of such indices for assessment of trophic status is, in our opinion, a deficiency of current coastal monitoring water sampling networks. We note, however, that none of the available multi-variable indices are appropriate for all of the coastal hydrosystem classes included in this report (Table 6-4), and all require further development to be appropriate as national indicators of ecosystem health for New Zealand coastal waters.

Table 6-4: Summary of integrated indices for coastal ecosystem health.

Index	Applicable coastal hydrosystem(s)	Variables integrated	Development required for use as New Zealand national indicator	Reference for recommended method
Trophic Index of Marine Systems (TRIX)	Coastal open water and DSDE	TN, TP, Chlorophyll-*a*, Secchi depth	Yes – see Morrisey et al. 2015	Vollenweider et al. 1998
OSPAR Eutrophication Objectives	Estuarine and coastal	Dissolved N & P, chlorophyll-*a*, DO; indicator species, algal blooms and fish kills	Yes	OSPAR 2005

Index	Applicable coastal hydrosystem(s)	Variables integrated	Development required for use as New Zealand national indicator	Reference for recommended method
Assessment of Estuarine Trophic Status (ASSETS)	Estuarine	Nutrient inputs (loading), chlorophyll-*a*, nuisance and toxic algal blooms, macrophytes, epiphytes, submerged aquatic vegetation, DO.	Yes – ETI (below) is a New Zealand development of this approach	Bricker et al. (2003)
Traits based macroinvertebrate index (TBI)	Coastal soft-sediment habitats, developed for Auckland area intertidal estuarine habitats	Benthic macroinvertebrate community data, mud percent, metal concentration	Designed for soft sediments in northern New Zealand estuaries. Generalisation possibly needed across New Zealand estuaries	Hewitt et al. 2012, Rodil et al. 2013
NZ-AMBI	SSRTREs, SIDES	Benthic macroinvertebrate community data, sediment organic carbon (SOC), sediment mud content	Based on a reasonable spread of estuaries, central areas of N&S Islands under-represented.	Robertson et al., (2015, 2016).
ETI	Estuarine - SSRTRE's, SIDES, DSDE, ICOLL	NZ-AMBI, Macroalgae, Phytoplankton and others. See ETI tool 2 pp 31-34 for full list.	Further development required.	Robertson et al. 2016a, b (ETI tool 1 and 2)

Of the approaches listed in Table 6-3, the Traits Based macroinvertebrate Index (TBI), NZ-AMBI and ETI are in development for New Zealand coastal systems. The TBI (Hewitt et al. 2012), and NZ-AMBI index (Robertson et al. 2015, 2016) were developed specifically to examine benthic ecosystem health using sediment characteristics and New Zealand benthic macroinvertebrate community data. Both use diversity and abundances of taxonomic groupings to assess benthic ecosystem health. The ETI tool (Robertson et al. 2016 a, b) is a derivation of the US-ASSETS estuarine monitoring approach modified for New Zealand estuaries. The first part of the ETI provides a method to prioritize estuaries for more rigorous monitoring and management (ETI tool 1, Robertson et al. 2016a). The tool produces a physical susceptibility score (i.e. very high, high, moderate, low susceptibility), and can be combined with nutrient load data to produce a combined physical and nutrient load susceptibility score. A second part of the ETI (ETI tool 2; Robertson et al. 2016b) provides a monitoring method that categorizes estuary trophic condition based on ecological response indicators (e.g., macroalgal biomass, dissolved oxygen, and NZ-AMBI scores). It provides guidance on which response indicators are most appropriate for monitoring each of the ETI coastal hydrosystem classes.

6.4 Water sample collection

6.4.1 Sampling platform

Recommendations for sampling platform will require council agreement during the establishment of any national sampling strategy, and may vary between councils based on the regional distribution of hydrosystem types (Table 5-4). Different sampling platforms confer a range of advantages, making some more suited for particular coastal hydrosystem types and sampling goals. For example, in water bodies prone to stratification (e.g., DSDEs) it is particularly important to sample over depth profiles, which is most easily performed using boat-deployed equipment. Furthermore, boats are essential as platforms for measuring visual clarity and light penetration in deep waters particularly where the bottom is not visible from the surface. In contrast, helicopter sampling enables many sites to be sampled during a similar tidal state, which is likely to be important for monitoring water quality trends in estuaries with a large tidal prism and intertidal area (e.g., SIDES). Structures such as jetties or wharves may provide for convenient sampling access to deep water, and wading on foot may provide access to otherwise inaccessible shallow waters but may risk disturbance of bottom sediments that would affect water quality measurements.

We think that boats should be recognised as generally the preferred platform for water quality measurements in coastal waters. Helicopter sampling, if adopted, should probably be supported by boat sampling on some occasions in order to locally calibrate turbidity to measures of visual clarity and light penetration, and provide reliable depth profiles. Further information and advice on sampling platform selection will be available in the NEMS for Discrete Water Quality, *in prep*.

6.4.2 Sample timing

Most of the variables currently sampled (Table 5-5) and recommended for future sampling in Table 6-2 would be expected to have seasonal, tidal and/or daily cycles; timing is an important consideration in the design of a national monitoring network. In Section 5, we noted that current sampling regimes typically account for seasonal variation by sampling repeatedly through annual cycles. We recommend that this is maintained. We also observed that there is little standardisation between regional authorities with respect to tidal state, or time of day. Given that standardization of tide or time of day are mutually exclusive for regular sampling, we recommend that standardised sampling with respect to tidal state takes priority because it is the most likely source of regional bias in water quality measurements. However, consultation with regional council scientists has led to the understanding that sampling at a single tidal state is impractical for some regions, particularly due to travel times between sampling sites. Furthermore, sampling at a single tidal state may generate water quality values that depart from the 'average condition' for a coastal hydrosystem, particularly where those coastal hydrosystems have a large tidal influence (i.e. tidal standardisation may not be desired for assessment of national 'state'). Water sampling that is representative of all tidal conditions could be aided by high frequency sampling techniques (e.g., moored sondes or CTDs) or, for example, stratifying or randomizing times for discrete sampling with respect to tidal state. Sampling using a stratified approach would have the advantage that a subset of sampling points collected at a single tidal state would be appropriate for analysis for national trends. However, a sampling approach stratified with respect to tidal state would take a greater sampling effort (more sampling points) to be able to confidently detect trends than if tidal state is kept the same in repeat sampling. Hence, decisions around timing of sampling will likely depend on objectives of the sampling, particularly whether it is intended to describe state or trends in water quality.

6.5 Field and laboratory analytical methods

In Section 5.4 we observed that filtering datasets provided by councils to only include the most commonly used, comparable field and laboratory analytical methods resulted in a substantial rejection of data from the datasets originally provided by councils. This likely resulted in reduced spatial representativeness in state analyses, and reduced power to detect trends, relative to sampling effort. For a national sampling strategy we therefore recommend consensus, so far as possible, on the measurement procedures used to gather data.

The NEMS Discrete Water Quality (NEMS *in prep*) will provide a list of preferred analytical methods for the suite of water quality variables listed in Table 6-2. More than one laboratory analytical method exists for some variables (e.g., ENT and TN). Similarly, alternative field protocols exist for some core and supporting variables. For example, NEMS recommends visual clarity (CLAR) as a core variable. Secchi depth provides an easy measurement of visual clarity in clear water under choppy sea conditions (Pers. Com. Mark Gall, NIWA). However, Secchi depth may give a poor representation of visual clarity in stratified water columns, or very turbid waters (Mitchell 2013). NEMS (*in prep*) suggests black disc (horizontal) visibility as an alternative measure of visual water clarity in some situations. This is likely to make black disk measurements more appropriate in SSRTRE and turbid SIDE hydrosystems, as well as coastal water bodies that stratify (e.g., some DSDEs). In cases where more than one method may be used, consistency through time should be a priority to avoid potential for creating artificial trends in the long term data record through changes in method. Where a methodological change is unavoidable, NEMS (*in prep*) recommend a minimum period of 12 months of monthly paired monthly analysis by *both* the existing and new analytical methods is recommended. This will enable the results of the two methods to be compared to determine if an 'adjustment' (i.e. correction) needs to be applied to one data set prior to trend analysis.

6.6 Key recommendations for variables and sampling protocols

Key recommendations in this section

- Sites included in a national network should be replicated sufficiently with respect to environmental classes of catchment land use.
- Sites included in a national network should be split proportionally across hydrosystem types, using the percentages provided in Table 5-4.
- Nutrients affecting coastal hydrosystems should be assessed by monitoring water quality in terminal river reaches, within estuaries and on their adjacent coasts.
- There should be unified use of the NEMS core water quality variables listed in Table 6-2.
- An integrated index of hydrosystem ecological health should be included in future state and trend analysis to facilitate setting of water quality thresholds (i.e. boundaries between bands of environmental state) and increase the utility of monitoring.
- There should be unified use of NEMS protocols with regard to metadata collection, reporting of measurement uncertainty and quality coding.

7 Recommendations for data analysis and reporting protocols

7.1 Reporting data quality and uncertainty

As discussed in Section 5.5, reporting and accounting for data quality in national state and trend analysis requires that information to be available for all contributing datasets. We recommend a unified implementation of the following guidance contained within the NEMS Discrete Water Quality (*in prep*):

- Standardised nomenclature (to be ratified by NEMS) should be used for exchanging and reporting water quality data.
- Uncensored data values are reported by the laboratory for all measurements together with the analytical method and Uncertainty of Measurement (UoM) to maximise the information captured and available for later data assessment. This recommendation echoes that of Davies-Colley et al. (2012b).
- Laboratory measurements are made by a laboratory that is IANZ accredited for the tests performed.
- To minimise UoM, laboratory measurements are made at the optimum resolution of the selected test, guided by the typical range found in the hydrosystem at the sampling location.
- Checks are made of measurements reported by the laboratory within two weeks of receipt to enable sample re-testing if necessary (e.g., an unusually high value is reported).
- All field and laboratory data are quality coded as per the NEMS quality coda schema (ranging from QC 100 for incorrect data/missing record to QC 600 for good quality data), in accordance with a Quality Codes Flowchart. This flowchart addresses all aspects of the water measurement, including field calibration and measurement and/or water sample collection, handling and laboratory analysis.

Reporting uncensored data values by laboratories would reduce the need for the methods used in this report for dealing with future incidences of censored data, and striping. This is advantageous because our recommended methods for assessing water quality state discards sites where more than 50% of the values for a variable were censored. These filtering rules are necessary to prevent assessment of state on datasets made up predominantly of imputed data. However, these methods may also lead to bias in state analysis towards sites where levels of variables are regularly high enough to be ‘confidently’ measured. Similarly, reporting of uncensored data values is likely to aid our ability to monitor water quality trends at relatively pristine sites. For analysis of trends, our method requires that the number of censored values in a trend period is < 15% of the total number of observations. Reporting uncensored data values is likely to reduce the number of sites with (for example) low nutrient concentrations being excluded from trend analysis, and may reduce the influence of ‘striping’ caused by premature rounding of data (Davies-Colley et al. 2012).

7.2 Metadata

We refer the reader to requirements for collection of metadata described in NEMS *(in prep)* which includes making records of:

- Sample depth(s) and bottom depth
- Depth profiles from field instruments
- Tidal state and height (all estuarine waters)
- State of sea during sampling
- Wind direction and strength (Beaufort scale), and
- Solar altitude/lighting conditions (clear sun, sun-occluded, octets of cloud cover).
- Any other features possibly relevant to water quality, including (but not limited to): fronts, river plumes, recent rainfall, vessel activity, dredging, and concentrations of birdlife (which may affect microbial data).

7.3 Thresholds for variables

7.3.1 Approaches to limit setting and thresholds

For setting limits and thresholds in coastal hydrosystems it is important to consider the difference between 'drivers' and 'attributes' of condition of estuarine and coastal waters. Nutrients are considered key 'drivers' of eutrophication 'attributes', the latter including phytoplankton and macroalgal blooms and oxygen depression (Hughes et al. 2015). Thresholds for nutrients and sediments (drivers) are likely to be set depending on the eutrophication and sedimentation susceptibility of a coastal water body based on its physiographic type (e.g., ETI or NZCHT class; (Robertson and Stevens 2016). In contrast, thresholds for dissolved oxygen, pH and contact and recreational microbial standards (attributes) are unlikely to vary with respect to estuary type and are more likely to be set to avoid unacceptable physiological stress to biota or hazard to human health.

However, nutrient *concentrations* are not often ascribed thresholds or used in a limit-setting context (Sutula 2011) because concentrations often do not reflect nutrients available to primary producers, during nutrient-limited phases of the annual cycle (Bricker et al. 2003). For example phytoplankton or macroalgal blooms can potentially reduce nutrient concentrations to negligible levels as they are incorporated into eutrophic levels of algal biomass. In contrast, measures of nutrient *loads* measure nutrients available to the primary producers prior to uptake, and therefore more faithfully record potential for trophic impact (NRC 2000). This concept underlies a GIS-based tool recently developed for New Zealand waters, Catchment Land Use for Environmental Sustainability-Estuary (CLUES-Estuary) which mixes loads entering estuaries using simple hydraulic models. Because managers have found it challenging to use concentrations of nutrients in coastal systems in a limit-setting context with a high level of confidence (Sutula 2011) metrics of nutrient loading are commonly used e.g., the Total Maximum Daily Load (TMDL) approach (NRC 2001).

Notwithstanding the above considerations, below we give a brief literature review of some ways water quality thresholds have been applied in New Zealand coastal hydrosystems.

- Waikato RC give guidelines and standards for estuarine water quality for ecological health, contact recreation and shellfish gathering: http://www.waikatoregion.govt.nz/Environment/Environmental-information/Environmental-indicators/Coasts/Coastal-water-quality/Estuarine-water-quality-techinfo/. For ecological health, these followed the ANZECC (2000) guidelines for nutrient concentrations and New Zealand's microbiological water quality guidelines for marine and freshwater recreational areas (Ministry for the Environment and Ministry of Health 2003) for contact recreation and for shellfish-gathering. Notably, the ANZECC (2000) guideline trigger levels are based on oligotrophic (low-nutrient) waters of southeast Australia. It has been noted that in terms of application in New Zealand estuarine and coastal waters the ANZECC guidelines for nutrient concentrations have been considered too conservative (Bolton-Ritchie and Main 2005).

- Auckland Council are currently investigating limit setting options for coastal water quality, and initial review of available guidelines has resulted in the recommendation to use the ANZECC guidelines approach for water quality variables where possible (Williamson et al. *in review*). However, Williamson et al. (*in review*) point out that the uncritical use of the default numerical trigger values is actually in contradiction to the ANZECC guidelines approach, and that local numerical values have to be derived for some variables. Nutrients in the water column of open coastal and open estuary waters (requiring development based on cause and effect relationships) and clarity in estuaries (requiring development based on local reference measurements) are examples of variables requiring development at the local scale.

- Horizons Regional Council have set estuarine water quality limits in their One Plan [Schedule I] https://www.horizons.govt.nz/CMSPages/GetFile.aspx?guid=2e9546e4-b224-433a-8728-2f6fbcbe9fe8. Notably, the nutrient concentration standards included in the One Plan are considerably above the ANZECC guidelines; these higher nutrient concentrations are based on the typology of estuaries in the Manawatu-Wanganui region which are uniformly SSRTRE type (Table 5-1) with high flushing rates and relatively low susceptibility to eutrophication (Zeldis 2009).

Overseas, nutrient concentration criteria have been developed for estuaries of southeast USA (Sheldon and Alber 2010) derived from the USA National Estuarine Eutrophication Assessment (NEEA) programme (NRC 2000, Bricker et al. 2003). A relevant approach to limit setting to manage eutrophication in Europe is conducted to meet the OSPAR Nutrient Management Objectives (OSPAR 2005). These objectives set area-specific guidelines for dissolved inorganic nitrogen (DIN), dissolved inorganic phosphorus (DIP), CHLA, phytoplankton eutrophication indicator species, and DO. With the exception of DO, guidelines are set relative to baseline values for each area, while DO guidelines are set to between 4 and 6 mg oxygen per litre. Because both the NEEA and OSPAR guidelines are largely based on deep estuaries and open coastal waters for which phytoplankton-based eutrophication is the major concern, they are applicable to only a minor subset of New Zealand estuaries (large, deep river estuaries and coastal embayments). For many New Zealand estuaries which are dominated by shallow mudflats (i.e. SIDEs), macro-algal eutrophication is the major concern (Robertson et al. 2016a). Therefore, we consider that while overseas systems provide a useful starting point for thresholds setting, they need to be modified or adjusted for the New Zealand context.

Estuarine typology aids the development of thresholds in the New Zealand context (Hume et al. 2007, Robertson et al. 2016a). For example, in the Manawatu/Wanganui region, the major estuaries of the region do not show overt signs of eutrophication, in terms of depressed oxygen or excessive algal outgrowths, because of their type. They are almost exclusively river-type estuaries of various sizes, in general with very little intertidal area capable of supporting attached macroalgae. They are turbid (hence probably light-limited) and have low residence time of water, so that phytoplankton blooms do not have time to form before they are flushed from the estuary. These estuaries essentially act as 'pipelines to the sea' for their high dissolved and particulate nutrient loads, with negligible biological processing by water column or benthic algae during transit. As a consequence, they can sustain high nutrient concentrations without showing eutrophication and as such have been ascribed relatively high nutrient-based thresholds (Zeldis 2009).

In contrast, shallow intertidal dominated estuaries such as the Avon-Heathcote Estuary and other New Zealand mudflat-dominated estuaries can sustain macroalgal bloom problems under considerably lower nutrient concentrations because of their physiographic features (large areas of well-lit habitat, sediment based eutrophication; see Barr et al. (2013)). These issues have been discussed in some detail in Robertson et al. (2016 a) with respect to the ETI project.

In the case of sediments, impacts of sedimentation can be assessed by comparing current rates of sedimentation to natural (e.g., pre-human) rates (Green 2013; Robertson and Stevens 2016). Similar to the case for nutrients, this reflects the fact that susceptibility of estuaries to the accumulation of fine sediments is estuary type-dependent, related both to the suspended sediment input load and the physical (flushing) characteristics of each estuary. Hence, responses to sediment loads in terms of water quality and ecosystem health are also likely to be estuary type-dependent (Robertson and Stevens 2016). One possible approach to limit setting that extends from this is to set thresholds on the current state sediment load (CSSL)/natural state sediment load (NSSL) ratio as a means of identifying catchments with excessive sediment loads. In this approach, both natural and current loads are calculated based on data derived from NIWA's CLUES model. Thresholds for the CSSL/NSSL ratio are assigned based on flushing capacity of the loads by different estuary types. It should be noted that this approach is currently used as a screening tool to identify estuaries requiring field study (Robertson and Stevens 2016); further research is required to derive robust sedimentation susceptibility thresholds for New Zealand estuaries.

7.3.2 Using historical water quality data for threshold setting

We would not recommend using the current dataset for threshold setting using a percentile-based approach in which water quality data is divided into percentiles to represent, for example, 'good', 'fair' or 'poor' conditions without reference to values or suitability-for-use in hydrosystems sampled. We feel this would not be advisable for the following reasons:

1. The dataset is unlikely to be representative of the spectrum of water quality conditions in New Zealand coastal hydrosystems for the reasons laid out in Section 5, considering network coverage and current council sampling protocols.
2. We currently do not fully understand how levels for each water quality variable relate to values (including ecosystem health) for each New Zealand coastal hydrosystem type.

A current initiative to address the second point above, focussed on nutrients and eutrophication, is the ETI approach which uses a database of condition attributes across estuaries, and from these generates scores of ecosystem health, for example the Manawatu-Wanganui estuaries (Robertson and Stevens 2016). The health scores are then be compared with the drivers (e.g., nutrient loading, and water quality variables described in this report) to arrive at thresholds for the drivers. When considered with respect to hydrosystem type, the health scores indicate appropriate threshold levels for the water quality variables. Development of this approach would allow use of the data presented in this report, and future datasets, to derive water quality thresholds for New Zealand coastal hydrosystems.

This approach is already being implemented in the case of nutrient *loads* within the ETI for New Zealand (Robertson et al. 2016a, b). It will be critical in future to examine the relationships between concentration thresholds and load thresholds using these respective datasets. This would allow a useful combination of two approaches – load and concentration monitoring with impact monitoring (e.g., ETI indices) to benefit from the best aspects of each (NRC 2000). A logical extension of this approach would be to derive thresholds for sediment loading, and water quality attributes reflecting sedimentation impacts, by building relationships between sedimentation and ecosystem health attributes across estuary types (e.g., Robertson et al. 2016c, Hewitt et al. 2012, Rodil et al. 2013).

We note that the approach we recommend requires supporting data be collected in coastal hydrosystems, for example ecological health, assessed using indices of macroalgal biomass and benthic animals (Robertson et al. 2016b). Within a given hydrosystem type, we would expect ecosystem health to respond to areal loadings of contaminants. Therefore, during guideline development, in addition to sampling representatively across types within regions**,** we recommend assessment across a range of annual loading rates. This could be set up using *a priori* judgement based on modelled loading information (Plew et al. 2015).

7.4 Using data from regional monitoring for national reporting

The data used in this report were compiled from accurate, high quality sampling programmes designed to monitor regional state and trends in water quality. There is a lack of national consistency in sampling methods across the data set which has created some regional bias in our analyses, such as inconsistent sampling with respect to tidal state at the time of sampling. However, data processing, organisation, analysis and presentation methods detailed in this report add to the accessibility, coherence, and interpretability of the data. Our primary concern regarding the suitability of our state and trend results for national reporting lie in the national relevance of the data. Because data were collected to address regional monitoring objectives, there are tendencies toward monitoring of higher-susceptibility hydrosystem types that may cause biases in national state and trend analyses. There are large gaps in site coverage nationally, and spatial coverage was further fragmented by filtering rules applied to state and trend analyses. Because of these representativeness issues, the state and trend results in this report appear to be most appropriate as 'case study' indicators of coastal water quality (as defined by Statistics New Zealand[5]). As shown in table 6.3, council coverage is generally better for state analyses than trend analyses, and differs between variables. In time, we expect uptake of protocols provided in NEMS (*in prep*) to significantly improve national representativeness of water quality datasets, as more sites from regional monitoring programmes will provide data that meets rules for inclusion in national analysis and reporting.

[5] http://www.stats.govt.nz/browse_for_stats/environment/environmental-reporting-series/environmental-indicators/Home/About.aspx#topics (accessed 31/10/16).

7.5 Key recommendations for data analysis and reporting protocols

Key recommendations in this section

- There should be unified use of NEMS protocols with regard to water sample collection and analytical methods.
- Reporting uncensored data values by laboratories is strongly recommended.
- The setting of water quality thresholds should account for characteristics of different hydrosystem types – some hydrosystem types are more sensitive to stressors than others.
- We would not recommend using the current dataset for threshold setting using a percentile-based approach because 1) the dataset is not representative of water quality conditions in New Zealand coastal hydrosystems nationally, for the reasons laid out in Section 5, and 2) we currently do not fully understand how levels for each water quality variable relate to values (such as ecosystem health).
- We recommend that thresholds for water quality and contaminant loads are set by comparing hydrosystem water quality with scores of ecosystem health and other values.
- We recommend further development of relationships between contaminant loading rates, water quality, and hydrosystem ecological health to inform water quality threshold setting.
- The state and trend results in this report are most appropriate as 'case study' indicators of coastal water quality for national reporting.

8 Acknowledgements

We thank Pierre Tellier (Ministry for the Environment) and the regional sector’s Coastal Special Interest Group for project support, feedback, enthusiasm, and patience. We thank the many council staff who provided additional data and information about monitoring programmes, and Juliet Milne, Jarrod Walker and other members of the NEMS coastal water quality subgroup. Martin Unwin and Doug Booker helped greatly with data preparation and analysis. Rob Davies-Colley, Scott Larned, Juliet Milne and Clive Howard-Williams reviewed the report.

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Appendix A Trend plots by site

Figure A-1: Eight year trends at each site, grouped by variable. Trendline is a lowess smoother, not seasonally adjusted. See 'CvsT_trends_eightyears.pdf' for these figures.

Figure A-2: Eighteen year trends at each site, grouped by variable. Trendline is a lowess smoother, not seasonally adjusted. See 'CvsT_trends_eighteenyears.pdf' for these figures.

Appendix B All state and trends data by site

See ‘all_results_by_site.csv’ for these data.