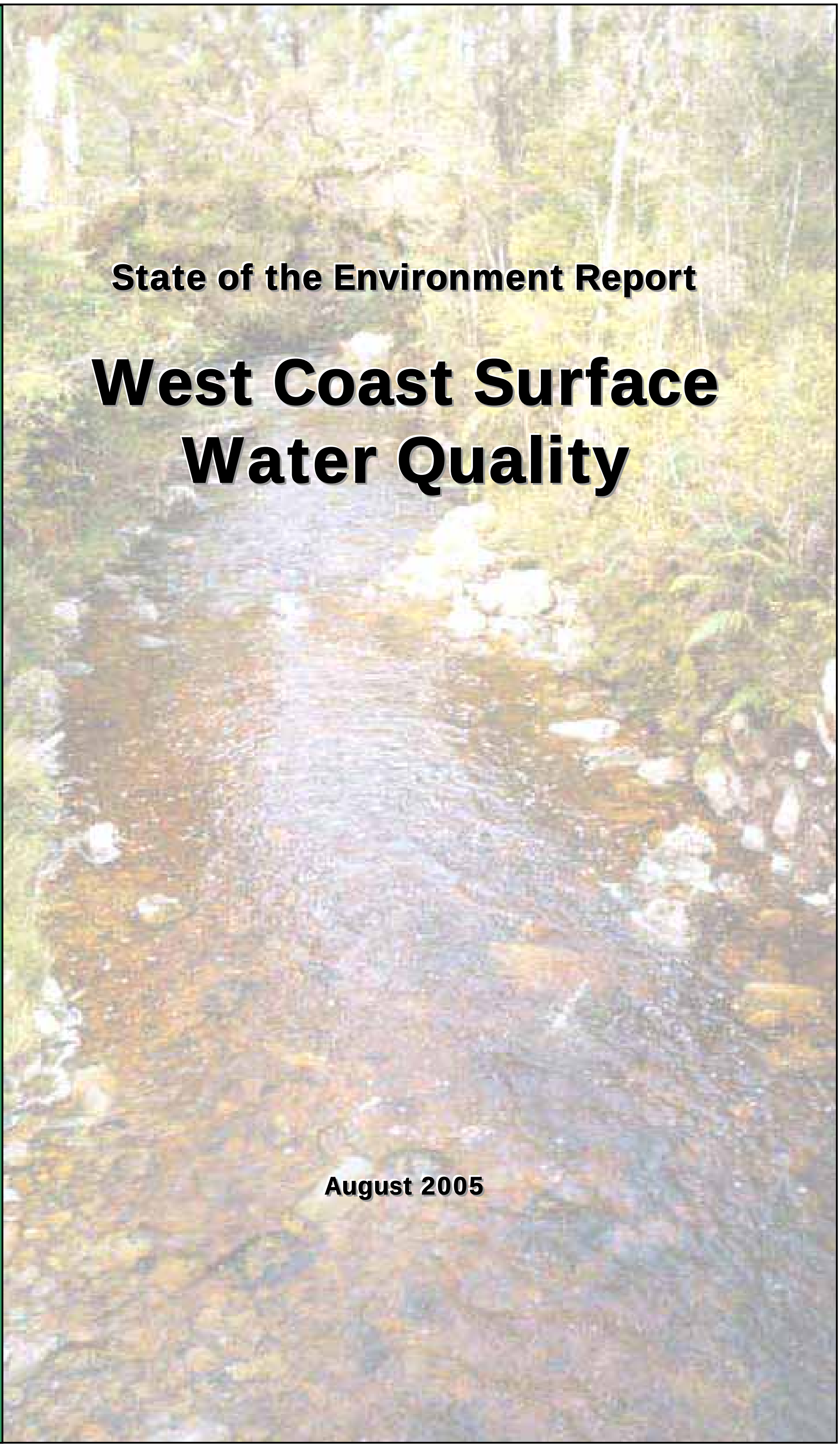

State of the Environment Report

West Coast Surface Water Quality

August 2005

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State of the Environment Technical Report 05001

August 2005

A technical report presenting results of the West Coast Regional Council's Surface Water Quality Monitoring Programme from 1996 to 2003 and incorporating monitoring data collected by other organisations from 1989 to 2003.

Prepared by:

J. Horrox — *West Coast Regional Council*

With assistance from:

Trevor James — *Tasman District Council*

Dr Mike Scarsbrook — *National Institute of Water and Atmospheric Research*

Dr Brian Stewart — *Ryder Consulting Ltd*

Reviewed by:

Chris Ingle

Approved for release by:

David Horn

Photo – front cover: Un-named Stream, Okarito Forest

Executive Summary

This report summarises results from the West Coast Regional Council (WCRC) State of the Environment (SOE) monitoring of surface waters. This program assessed state and trends for surface water quality in areas where human impacts/pressures were considered to exist. Therefore conclusions made in this report refer in particular to these types of waters.

The WCRC SOE monitoring program sampled a number of sites for physical, chemical, and bacteriological water quality parameters, as well as periphyton and macroinvertebrate communities, from 1996 to 2003. A group of 39 sites were monitored four times per year. Ten of these were reference sites, eight of which had a corresponding 'impact' site(s) down stream. Thirteen sites in the group, located at the far north and far south of the region, were sampled on a bi-annual basis. An additional five sites were sampled as a part of the National River Water Quality Network (NRWQN). The River Environment Classification (REC) was used to group these sites by climate, source of flow, geology, land cover, and stream order. Other data included in this report comes from WCRC freshwater fish surveys, the WCRC Contact Recreation water quality program, and the Lake Brunner water quality program.

This report is intended to identify differences in water quality state, and changes in water quality over time. Individual water quality parameters were compared with guidelines, and levels found in other parts of New Zealand.

State of water quality in the West Coast Region

Water quality patterns from REC

Many significant differences in physical, chemical, and biological water quality parameters were observed between the REC classes of flow source, geology, landcover, and stream size. Patterns observed for these parameters amongst REC classes suggested that streams could be characterised as:

- Streams with a hill source of flow; hard sedimentary or plutonic geology, often incorporating larger rivers; with higher, less variable water quality; brown trout more abundant.

- Lowland streams (low elevation source of flow); higher turbidity, nutrients and temperature (which may not solely have been a response of anthropogenic disturbance); smaller, more variable and susceptible to impact; potentially higher fish and invertebrate diversity.

- Streams draining predominantly agricultural catchments (sub-set of lowland streams); comparatively poorer water quality with fewer sensitive macroinvertebrate taxa. Nutrients may be elevated where there is significant groundwater recharge (i.e. spring source).

- Stream catchments with soft sedimentary geology; higher turbidity; distinctive physically and chemically from other geology classes; smaller size; lower source of flow.

Comparisons from reference to downstream impact sites showed significant differences in faecal coliforms (increase), ammoniacal nitrogen (increase), and pollution sensitive macroinvertebrate taxa (decrease). To a lesser extent, nutrients increased and clarity decreased; the later exacerbated where soft sedimentary geology occurred. The main cause of this was agricultural landuse. Past and present opencast and underground mining caused significant lowering of pH in areas where sufficient quantities of exposed pyrite generated acid rock drainage. This did not happen where mining occurred in low pyrite geology's, or where highly calcareous rock buffered pH.

West Coast waterways: guidelines and standards

The Proposed Water Plan for the West Coast (as of August 2005) sets two water quality objectives, in accordance with Schedule 3 of the RMA. These are: Contact Recreation for specific bathing and secondary contact sites; and an Aquatic Ecosystem standard, which applies to all West Coast water bodies. Other additional guidelines from various literature have been used for assessment in this document.

Summarising site comparisons to guidelines for aquatic ecosystems: no guidelines applied to sulphate or conductivity, but high levels where observed at some sites. Elevated sulphate was found in catchments with major mining operations, as were high conductivities. Limestone geology and saltwater intrusion also influenced the latter variable at some sites. Values of pH lower than our designated guideline occurred in catchments with current and historic mining operations. Mining did not cause low pH if it occurred in areas where suitably calcareous geology provided a buffer.

Dissolved oxygen, temperature, and nitrogenous nutrients were within respective guidelines for all sites on the West Coast. Twelve percent of sites failed the clarity guideline relating to protection of aquatic life, which is based on un-disturbed to slightly disturbed ecosystems. Water quality as determined by macroinvertebrate communities ranged from moderately impacted to pristine. The median within this range fell into the doubtful quality (mildly impacted) category.

Approximately a third of SOE sites were above the faecal coliform and clarity guideline used for bathing, but of this third only one site was a scheduled swimming area in the Water Plan. No site medians were above the faecal coliform guideline for secondary contact recreation.

Water quality for contact recreation

Water quality observations for contact recreation at WCRC contact recreation monitoring sites were generally poor for sites sampled in the Buller district, with the possibility that water quality had deteriorated slightly at some of these.

Grey District had more sites, and a greater range of water quality. Lagoons had poor water quality, although the beaches near them were generally good, with only occasional exceedences. River sites were also predominantly good, but occasionally exceeded. All sites in Lake Brunner were safe, other than occasional spikes at Moana, although because these poor results are only periodic, chance events cannot be ruled out as the cause. Hence there was not sufficient evidence to suggest a consistent problem at Moana. Lake Kaniere had the highest water quality for contact recreation with no problems evident.

Water quality of Lake Brunner

Epilimnetic nitrogen stabilized in 2003-04. However, trends in phosphorus and phytoplankton biomass were up, and clarity down. In Cashmere Bay anoxic conditions near the sediment caused nutrient releases from the sediment. Nutrients in the tributaries (Crooked, Orangipuku, and Hohonu) have stabilised and this may lead to more stable nutrient levels in the lake. While faecal coliforms were higher in the rivers with higher agricultural activity (Crooked and Orangipuku), they were within contact recreation guidelines.

Are things getting better or worse?

Replication was not sufficient to assess if any synchronous impact/reference changes over time were driven by climatic factors. In some cases, the determination of trends was hindered by insufficient data and high variation. Moderate to high correlations between most paired impact and reference data indicated that variation induced by natural factors such as rainfall was an important driver of many water quality patterns observed over time. Tentative evidence suggested a reduction in faecal coliforms at Karamea, Orowaiti, and Baker Creek, with a possible increase in ammoniacal nitrogen at the Orowaiti River reference site.

How does the West Coast compare to elsewhere in NZ

West Coast waterways are unique in that nearly all have a cool, extremely wet climate, and this climate type had the highest water quality out of a total of six climate types observed nationally - a situation helped by the high proportion of natural vegetation cover in these catchments. This climate type, combined with a high elevation source of flow, had some of the highest water quality among all climate/source flow types in New Zealand, with all median water quality parameters within their respective guidelines. This was also observed on the West Coast.

Half of West Coast SOE sites fell into the low elevation source of flow class. In a study of national rivers no low elevation source sites were among those with high water quality, although low elevation source of flow sites on the West Coast were some of the best nationally.

The mean West Coast level of the faecal coliform *Escherichia coli* (528 *E. coli* /100 ml) was situated in the middle of the national range of 140 to 1026 *E. coli* /100 ml. Mean water clarity (using black disk) in other regions ranged between 0.7 to 3.5 m. Water clarity on the West Coast, with a mean of 2.8 m, was in the upper clarity range, and higher than most other regions in New Zealand. For ammoniacal nitrogen, the West Coast was close to the best in New Zealand with a mean of 0.02 mg/L. This compared to the regional range of 0.01 to 0.112 mg/L.

Nationally, pastoral landcover classes had lower water quality than natural ones, which was also observed for the West Coast Region. At a national scale, negative trends were observed for ammoniacal nitrogen, but the opposite occurred in at least one part of the West Coast. It should be noted that West Coast ammoniacal nitrogen levels remained some of the lowest when compared to the national range.

Can the WCRC adequately assess state and trends using current monitoring network?

There were difficulties measuring trends with the quarterly monitoring data contained in this dataset. From this recent work, much valuable knowledge has been gained on the state of water quality in the West Coast region. Looking toward future monitoring, improved ability to detect trends is now likely to become a greater priority. One means of making this economically viable for the council would be to reduce the number of core monitoring sites, and increase monitoring frequency. From the use of REC, and the subsequent information gained from this report, it may be possible to determine where there are significantly homogeneous groups of sites that can be scaled down to allow for greater resource investment into higher sampling frequency. Improved trend analysis would allow the WCRC to better assess the effectiveness of its management systems and activities like resource planning, consenting, and compliance.

Statement of data verification and liability

West Coast Regional Council recognises the importance of good quality data. This first comprehensive surface water quality technical report provides interpretation of results from the West Coast Regional Council Surface Water Quality Monitoring Programme and a summary of relevant information available at time of producing the report.

Data collection and management systems follow systematic quality control procedures (see West Coast Regional Council Surface Water Quality Monitoring Programme). International Accreditation New Zealand (IANZ) laboratories carried out sample analysis excluding field analysis. When possible expert staff have been involved in each stage of the monitoring process. A process of internal and external review of this report has been implemented.

While every attempt has been made to ensure the accuracy of the data and information presented, West Coast Regional Council does not accept any liability for the accuracy of the information. It is the responsibility of the user to ensure the appropriate use of any data or information from the text, tables or figures. Not all available data or information is presented in the report. Only information considered reliable, of good quality and of most importance to the readers has been included.

Acknowledgements

The West Coast Regional Council wishes to acknowledge the following people and organisations for their contributions to this report:

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1 Introduction

1.1 Rationale

The West Coast Region is renowned for its natural and physical attributes, including its lakes, rivers, and coastal areas. It is also renowned for its wet climate - something that has played an important role over time to help form the unique features we see today. These attributes, or resources, must be sustainably managed in support of their many uses that include recreation, industry, energy, and agriculture, not to mention maintaining the integrity of ecological and cultural values.

Under the Resource Management Act 1991 the West Coast Regional Council (WCRC) is responsible for the management of: water, air and land (soil conservation, natural hazards, and hazardous substances); activities in the Coastal Marine Area; the discharge of contaminants; and the use of river and lakebeds. As a result of these responsibilities the WCRC is required to monitor the overall state of the regions environment. This monitoring is important because it helps the Regional Council and the West Coast community to gauge the state of environmental quality and how it changes over time. The WCRC does this by regularly monitoring the quality of the Coast's key natural and physical resources using a range of scientific techniques - surface water being one of the main ones and the focus of discussion in this report. This monitoring allows us to make better decisions on how we manage the West Coast's water resources. It also provides feedback as to how effective our policies are i.e. if water quality is improving, stable, or deteriorating.

The WCRC will prepare a State of the Environment (SOE) report every five years to provide information about the quality of the West Coast's water resources. This technical report synthesises information from the WCRC surface water quality monitoring program, and includes some information from territorial authorities, and other resource management agencies and institutes. Separate technical reports will be produced to discuss the state of the West Coast's groundwater, and air quality.

1.2 Objectives and scope

The aim of this report is to analyse surface water quality data from the WCRC SOE monitoring and other records, to assess patterns in both state and recent trends in water quality within the region. The WCRC surface water quality-monitoring program can be broken into three main areas.

The first involves regular monitoring of a wide range of stream and river sites across the region. These sites vary greatly in terms of their environmental characteristics like their topography, geology, land use, and size. To make comparisons between them, a spatial framework was required that would group and standardise them according to these factors. The GIS-based River Environment Classification (REC) was used to organise sites according to these characteristics (Refer to 2.1 for an overview of REC).

Seasonal monitoring of waters popular for contact recreation that are located in marine areas, rivers and lakes is another important task for the WCRC.

The third involves quarterly monitoring of water quality in Lake Brunner, and three of its main tributaries. There are other studies that complement much of the information gained from the main monitoring program, which are periodically carried out to address individual issues.

This report aims to:

- Identify differences in water quality state, and compare these to water quality guidelines and other regions of New Zealand. This will be done in some applications using REC site groupings based on climate, source of flow, geology, landcover, and stream order.

- Investigate changes in water quality over time using REC site groupings, between reference/impact site pairs, and among individual sites.

These aims target the following areas:

- WCRC SOE monitoring sites, past and present, located across the West Coast region, which are representative the range of conditions found in the West coast Region.

- Lake Brunner.

- Contact recreational areas.

An outline of the methods, followed by explanations of some of the measurements, or parameters, used to assess water quality are provided as a reference in Sections 2.0 and 3.0.

2 Methods

2.1 Nature of data

The West Coast Regional Council (WCRC) has sampled a number of sites for water quality, periphyton and macroinvertebrate communities. The time period of data used spanned from 1996 and late 1990's, to late 2003. The WCRC maintains 39 sites throughout the region for the purpose of State of the Environment (SOE) monitoring with water quality, macroinvertebrate, and periphyton data collected from them. Ten were used for the purpose of reference sites, eight of which had a corresponding 'impact' site(s) down stream. Sites were visited four times per year with thirteen sites in the group, located at the far north and far south of the region, rotated on a bi-annual basis. An additional five sites are sampled as a part of the National River Water Quality Network (NRWQN).

A number of data have been collected by the WCRC for a range of water quality parameters. The main ones used in this report include turbidity, percentage dissolved oxygen, pH, temperature, ammoniacal nitrogen, and faecal coliforms. Total nitrogen and phosphorus, dissolved inorganic nitrogen, nitrate, dissolved reactive phosphorus, and sulphate data existed for many sites, albeit in less abundance than the main parameters. A range of attributes and indices have been presented for macroinvertebrate and periphyton communities data.

The River Environment Classification

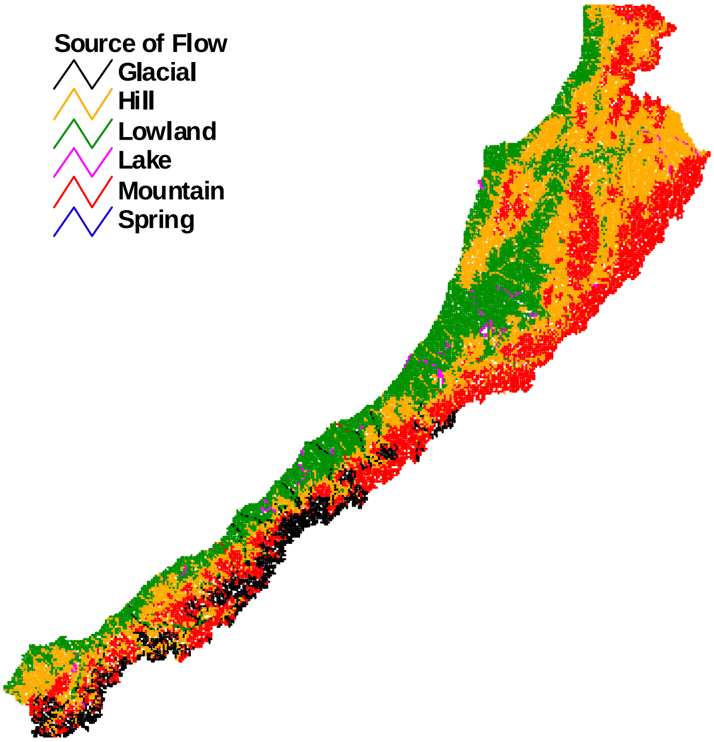
Water quality patterns in the West Coast Region were investigated using the framework of the River Environment Classification (REC) (Snelder et al. 2003).

The REC characterises river environments at six hierarchical levels, each corresponding to a controlling environmental factor. The factors, in order from largest spatial scale to smallest, are climate, source-of-flow, geology, landcover, network position and valley landform. Each factor is associated with a suite of physical processes that influence water quality, and vary at approximately the same scale. For example, the climate level of the REC is associated with precipitation and thermal regimes that vary at scales of 10^3 – 10^4 km2. Each REC factor is composed of 4 – 8 categories that differentiate all New Zealand rivers. Categories at each classification level and their abbreviations [relevant to the West Coast] are shown in Table 2.1.1 The number of possible classes at any level is equal to the number of categories at that level multiplied by the number of classes at the preceding level. For example, the source of flow level has 24 possible classes (6 climate classes × 4 source-of-flow classes). At the geology level there are 144 possible classes, and 1152 at the land-cover level (from Larned et al. 2005).

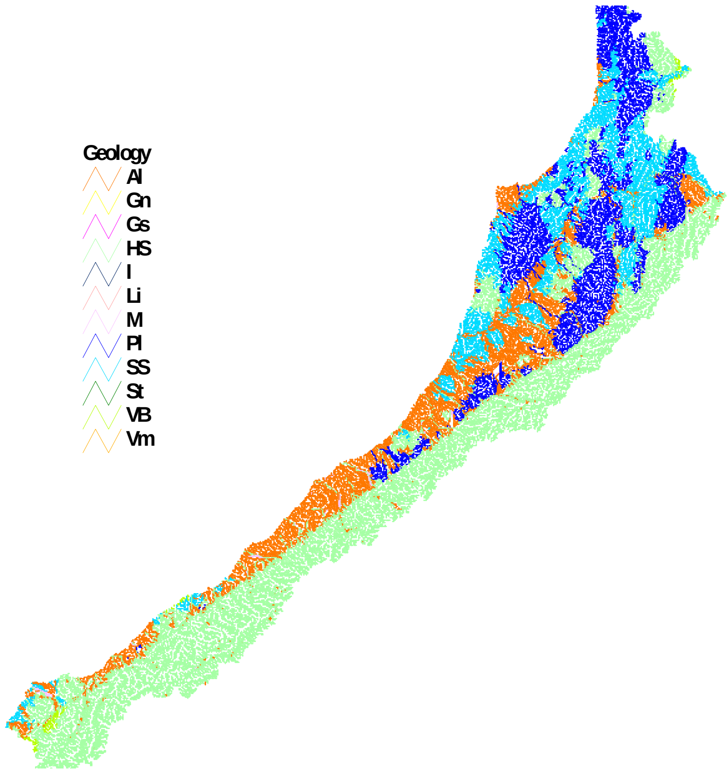
Typical use of the REC involves grouping REC classes from each level e.g. climate/source-of-flow/geology/land-cover/network position/valley landform. However, WCRC sites were analysed mainly via individual controlling environmental factors because there was not in most cases sufficient replication for sites to be compared based on combined REC levels. Not all classes occuring in New Zealand are represented in the WCRC dataset, and those that are, are listed in Table 2.1.1. Map distributions of source of flow, geology, and landcover are shown in Figure 2.1.1 to 2.1.3.

Table 2.1.1 REC classes found in the West Coast region. Classes are hierarchical starting at the top in the order of: climate/source-of-flow/geology/land-cover/network position/valley landform. Those in bold are represented in the WCRC SOE monitoring dataset.

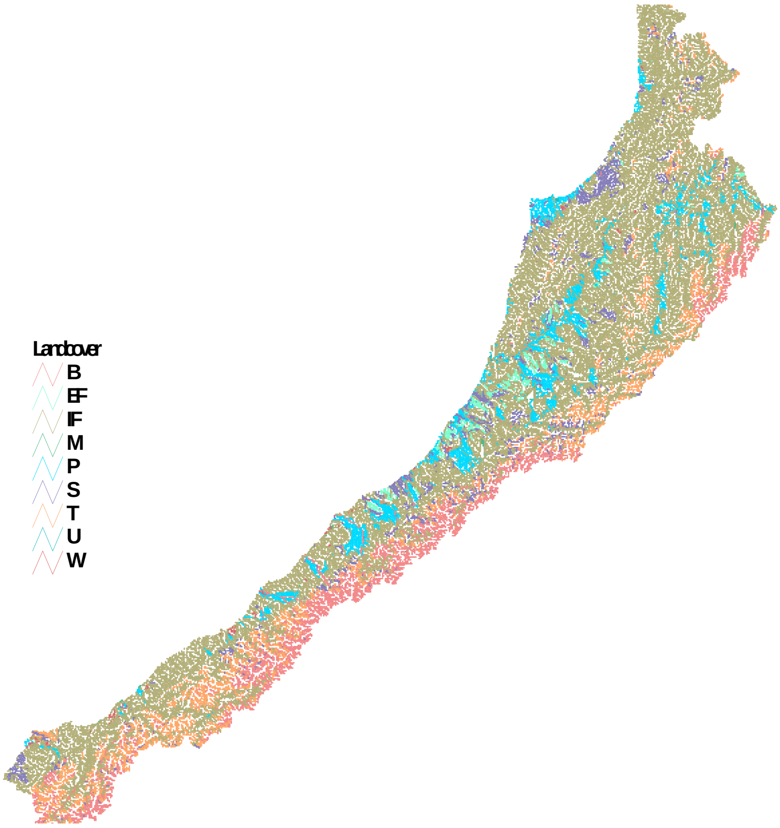
Class	Definition
Climate:	
CX	Cool, extremely wet (mean annual temp. < 12, rainfall > 1000mm)
CW	Cool, wet (mean annual temp. < 12, rainfall > 500, < 1000 mm)
CD	Cool, dry (mean annual temp. < 12, rainfall < 500mm)
WW	Warm, wet (mean annual temp. > 12, rainfall > 500, < 1000 mm)
Source of flow:	
L	Low elevation (> 50 % of annual precipitation occurs < 400m ASL)
H	Hill (> 50 % of annual precipitation occurs between 400 and 1000m ASL)
M	Mountain (> 50 % of annual precipitation occurs > 1000m ASL)
Lk	Lake sourced
S	Spring
Gl	Glacial
Geology:	
Al	Alluvial and sand
HS	Hard sedimentary
SS	Soft sedimentary
Pl	Plutonic
St	Schist
M	Miscellaneous
Landcover	
IF	Indigenous forest
P	Pastoral
T	Tussock
S	Scrub
EF	Exotic forest
W	Wetland
U	Urban
Stream order:	
HO	High order (> 4)
MO	Mid order (3-4)
LO	Low Order (< 3)
Valley landform:	
HG	High gradient (slope > 0.04)
MG	Medium gradient (slope 0.02-0.04)
LG	Low gradient (slope < 0.02)



Figures 2.1.1 Map of the West Coast region showing source of flow according to REC.



Figures 2.1.2 Map of the West Coast region showing geology class according to REC. From the key, only Al (alluvial), HS (hard sedimentary), M (metamorphic), Pl (plutonic), and SS (soft sedmentary), are present on the map.



Figures 2.1.3 Map of the West Coast region showing landcover type according to REC.

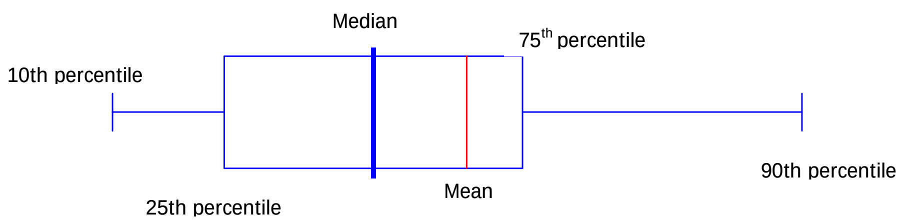
2.1 Analysis of water quality state

2.2.1 Comparison of differences in water quality parameters within REC levels and between individual sites

Ideally the REC is used to categorise sites according to combined levels with one class from each level, repeated for all permutations. REC levels are hierarchical spatially and in importance e.g. climate is at the top of the hierarchy (Table 2.1.1). Climate controls many physical and biological processes; therefore a REC level is dependent on those above it. Using combined REC classes requires many sites - far more than the number contained in the WCRC dataset, hence data investigation has been carried out for one level at a time. However, common combinations of classes from different levels were investigated, as well as investigating what classes were most common among sites.

Analysis of Variance (ANOVA) was used to compare differences between water quality parameters (physical, bacteriological, and macroinvertebrate) based on single REC level. Data was transformed prior to analysis (log [x+1]) to satisfy the assumptions of the test. Total nitrogen, nitrate, total phosphorus, dissolved reactive phosphorus, and sulfate have numerous data points, but these come from fewer sites compared with the other parameters. There were only two sites, on the same river, which were lake fed. These lake fed sites were excluded from ANOVA analysis between sources of flow due to insufficient replication for the lake class, but they have been included in the box and whisker plots. Valley landform has not been included in the analysis as it was decided that the other REC levels provided were sufficient.

Box and whisker plots have been used to compare water quality parameters between classes within an REC level. Only parameters that were statistically significant were plotted. Figure 2.2.1 below describes what statistical criteria they display. Plots have also been presented with separate sites for each parameter.



Figures 2.2.1 Box and whisker plots for water quality and macroinvertebrate parameters for core monitoring sites. Outliers are not shown. The mean (red line) is only shown on plots with individual sites.

The percentage of time that a water quality parameter went beyond its respective guideline was plotted using bar graphs (refer Section 3 for discussion of water quality guidelines). Sites were colour coded and ordered according to their landuse. Note that the terms landuse and landcover, the latter a REC level, are interchangeable in this report.

2.2.2 Multivariate analyses

Multivariate analyses were carried out using the program Primer (Version 4, Plymouth Marine Laboratory, Clarke & Warwick 1994). The data was normalised using the internal procedure within Primer. Euclidean distances were used to construct a matrix for physical and chemical data, and Bray-Curtis distance were used for biological data. Multi-dimensional scaling (MDS) ordination was used to represent the data. The resulting ordinations are considered to be the most accurate way to express multivariate data (Clarke and Warwick 1994) and include a measure of 'stress' that expresses the degree to which a two-dimensional representation can summarise the data. Stress values of < 0.2 are considered good, while stress value of < 0.5 are considered acceptable but difficult to interpret. To interpret analysis of similarities, look at the 'R' value. A value of 0 means the groups are indistinguishable; a value of 1 means that all similarities within groups are less than any similarity between groups i.e. the closer the number is to 1, the more different groups are.

Relationships between REC classes and macroinvertebrate data

Invertebrate data was collated for all sites and plotted using multi-dimensional scaling (MDS) with REC types used to differentiate sites. Coded abundance invertebrate data were changed to numeric data by substituting the value at the base of the SQMCI rank class (Stark 1998) for the rank class itself (Table 2.2.2). The data was transformed (Log [x+1]) in order to compensate for the large number of zeros in the data, and normalised using the internal procedure within Primer. Hierarchical agglomerative clustering was applied to generate a similarity matrix of Bray-Curtis similarities (Clarke and Warwick 1994). Multi-dimensional scaling (MDS) ordinations were used to represent the data graphically. Sites coded according to the different REC classes to ascertain whether the different classes clustered together. To test whether the REC classes formed significantly different groupings within the data, analysis of similarities (ANOSIM) was used (Clarke and Warwick 1994).

Table 2.2.2 Values used for conversion of ranked abundances to numeric abundances for macroinvertebrate data. Ranks based on Stark (1998).

Rank class	Abundance range	Value used
Rare (R)	1-5	1
Common (C)	5-19	5
Abundant (A)	20-99	20
Very abundant (VA)	100-499	100
Very very abundant (VVA)	> 500	500

Relationships between REC classes and periphyton data

Periphyton data was analysed based on the presence or absence of taxa, with following analysis conducted in the same manner as for the invertebrate data.

2.2.3 Freshwater fish diversity and abundance

Studies have been conducted by the WCRC in 2001 and 2003 to compare the diversity and abundance of freshwater fish in farmland streams of varying habitat condition, caused by various activities. In these studies, 41 streams were assessed using the following physical and chemical factors: channel and bank form, bank stability, bed substrate composition, water quality (conductivity, temperature and pH), and amount and type of riparian vegetation. Electric fishing was used to collect fish except where waterways were too deep or unsuitable in other ways e.g. extensive macrophyte beds. Spotlighting was used as a substitute in these situations. The area of wetted bed surveyed was 150-200 m^2. Sites were assessed by dividing them into four disturbance classes (1 being the most highly modified), based on the following criteria: stream canalisation and straightening, amount of riparian vegetation that has been disturbed or removed and restriction of stock access.

Based on the habitat requirements of freshwater fish species, as well as their sensitivity to habitat disturbance, fish were placed into groups for analysis for varying levels of disturbance (Table 2.2.3).

Table 2.2.3 Composition of fish groups used to analyse electro-fishing-data

Trout	Least sensitive native	Moderately sensitive native	Most sensitive native
Brown trout	Lamprey	Koaro	Giant Kokopu
	Mudfish	Dwarf Galaxias	Banded Kokopu
	Common Bully	Torrentfish	Short-Jawed Kokopu
	Redfin Bully		
	Bluegill Bully		
	Inanga		
	Longfin Eel		
	Shortfin Eel		

2.2.4 Water quality comparisons of paired reference-impact sites

Water quality data was displayed for eight paired reference/impact sites, using box and whisker plots. ANOVA on two levels was used to compare values between reference/impact sites, which included Karamea River, Sawyers Creek, Seven Mile Creek, Baker Creek, Orowaiti River, Grey River, Crooked River, and Garvey/Lawyer Creeks. Faecal coliform data was highly skewed and log transformed to improve data normality.

2.3 Trends in water quality

2.3.4 Trend analysis

Trends in surface water quality over the last five years (1998-2003) were investigated based on the relationship between paired/impact sites over time. Sites used for this analysis were Karamea River, Sawyers Creek, Seven Mile Creek, Baker Creek, Orowaiti River, Grey River, and the Crooked River. Firstly, levels of each parameter were plotted over time for each site pair, and fitted with a regression line. Then the difference between reference/impact parameters was calculated, plotted over time, and fitted with a regression line. Finally, the data was smoothed using LOWESS (LOcally WEighted Scatterplot Smoothing) to eliminate seasonal differences, and the resulting final regressions overlain on the original plots.

Kendall's tau (a nonparametric test of correlation) was calculated to examine the correlation between reference and impact water quality parameters at paired sites. Tau ranges between –1 and 1. To interpret Kendall's tau, the closer to 0 tau becomes, the less similar the trends between sites are. If tau is close to 1 or –1, both reference and impact sites displayed a similar pattern over time. Both Kendall's test, and comparison of LOWESS smoothed regressions can help determine what patterns are caused by anthropogenic causes, and what are related to natural environmental changes.

2.3.5 Changes in macroinvertebrate data and indices according to REC class

Changes in macroinvertebrate community composition (taxonomic richness, number and percent EPT), and two macroinvertebrate indices (MCI and SQMCI) were plotted sequentially for each year according to the REC levels: climate, source of flow (spring vs. non-spring and hill vs. lowland treated separately), geology, landcover, stream order, and valley landform (gradient).

2.4 Contact recreation

Analysis is based on 21 sites located between Hokitika and Westport that include marine, estuarine and fresh waters. The sampling season runs from the beginning of November through to the end of March. The Ministry for the Environment recommends a minimum of 20 samples per season as a basis for calculation of medians. WCRC data and sampling has been less frequent than 20, and this should be borne in mind when drawing conclusions from analysis presented in Section 7.

The length of data record varies between sites. For most of them, monitoring began in the summer of 1999 – 2000, except for the Lake Kaniere @ Sunny Bight and Orowaiti Creek @ Excelsior Rd sites, where monitoring began more recently in the summer of 2001 – 2002. Two other exceptions are Blaketown Beach @ South Tiphead and Blaketown Lagoon @ Slipway Beach, which were initiated in the summer of 1998 – 1999. Refer to 2.1.8 for a description of the various bacteriological guidelines.

2.5 Lake Brunner

Monitoring of water quality, in various forms, for Lake Brunner and its catchment began in the early 1990's. Concerns over water quality lead to a more intensive study initiated around 2000. This looked at water quality parameters in both the lake, and in some of the key waterways feeding the lake.

Three sites were monitored in the lake: at Iveagh Bay, Cashmere Bay, and a point in the middle of the lake. Profiling of conductivity, dissolved oxygen and temperature was carried out by lowering a water quality probe slowly through the water column, which regularly recorded values of these parameters as it descended. This was done six times spread throughout each year. Other data collected at these sites were secchi and black disk clarity, ammonia, nitrate, and dissolved reactive phosphorus. Further data was collected from the mid-lake site including chlorophyll a, total phosphorus, and total nitrogen. Nitrate and dissolved reactive phosphorus, and faecal coliforms were measured at eight tributaries in 2001-2003, and three tributaries in 2004-05.

3 Common freshwater ecological parameters: significance and guidelines

This section provides an overview of common parameters that are monitored by the West Coast Regional Council, their significance in freshwaters, and guidelines commonly used in New Zealand for assessing water quality in rivers and lakes.

3.1 Physical and chemical parameters

3.1.1 pH

At a given temperature, the intensity of the acidic or basic character of a solution is indicated by the pH or hydrogen ion activity (APHA 1992). Most natural waters fall within the pH range of 6.5 to 8.0 (ANZECC 2000), and in the absence of contaminants, most waters maintain a pH value that varies only a few tenths of a pH unit. Recommended trigger limits for pH of New Zealand upland and lowland rivers are in the pH range of 7.2 to 8.0. A more appropriate means of setting pH limits involves using the 20^{th} and 80^{th} percentiles, calculated from seasonal medians in a reference site (ANZECC 2000). It is recommended that changes of more than 0.5 units from the natural seasonal maximum and minimum be investigated (ANZECC 1992). However, there are many streams and rivers on the West Coast that have naturally low pH (as low as pH 4), which may originate from humic acids or come from young sedimentary geologies with a pyrite component. Some plants and animals are adapted to naturally lower pH (refer Collier et al. 1990). The key difference between streams with naturally low pH and those that are such as a result of acid rock drainage is the higher level of metals in the latter. The toxicity alone of these metals may prove detrimental to a streams ecological health and be exacerbated further when combined with low pH, but evidence of increased toxicity is not conclusive from New Zealand studies. As well as toxicity, high concentrations of metal can give rise to precipitates that negatively effect macroinvertebrate habitat and food quality, and subsequently, food webs. Overall, it seems clear that invertebrate diversity is negatively impacted by pH and elevated metal concentrations below pH 4.5. We have chosen a minimum level of pH 5.5, based on studies of West Coast streams (e.g. Collier et al. 1990; Rowe 1991), as a general criterion for measuring exceedences in section 3.2, applicable to sites with anthropogenic acid generation, as a buffer to allow for more sensitive taxa and potential chronic effects of metal toxicity on certain species. It also considers that while many West Coast streams have lower pH, many others are within the range specified by ANZECC (2000) guidelines. Higher than 'average' pH can occur where a catchment contains limestone geology, although not common, parts of the West Coast have elevated pH for this reason. These higher pH's are not toxic, although higher pH will increase the ratio of toxic un-ionised to ionised ammonium ions.

3.1.2 Temperature

Temperature is fundamental to the rate of biological processes in a water body. For many micro-organisms, metabolism doubles with each rise of 10 °C, but tolerance of temperature extremes between different species is generally quite specific.

Aquatic biota are strongly influenced by water temperature in terms of their growth, reproduction, and survival. The biota of Westland streams and rivers contain elements that are valued for their recreational opportunities (brown trout, whitebait) and national endemism (various native fish). Increased water temperatures may affect these taxa directly, for example via oxygen removal, and indirectly via aquatic food chains. The key components of river ecosystems (algae, plant, macroinvertebrates and fish) are all affected by temperature. Introduced sport fish (trout and salmon species) are very susceptible to high temperatures and their success in New Zealand has largely been attributed to cool summer water temperatures, and winter temperatures generally high enough to allow for some food (i.e. invertebrate prey) production (Viner 1987).

As temperature varies widely both spatially and temporally in aquatic systems, it is difficult to assign low risk trigger values for temperature. It is, however, recommended that temperatures should not be varied beyond the 20th and 80th percentiles of natural ecosystem temperature distribution (ANZECC 2000).

Algae and plant growth in New Zealand rivers are most strongly affected by a combination of nutrient supply and disturbance regime, however temperature has also been identified as an important factor in determining periphyton biomass and community structure. Higher temperatures favour high biomass accrual and the dominance of erect, stalked and filamentous algae (often synonymous with nuisance algal growths). Such effects are also strongly influenced by disturbance (i.e., floods), with low disturbance favouring increased biomass of algae and plants.

In general, algae and plants are much more resilient to high temperatures than invertebrates, and some elements of the algal community exhibit high growth rates at temperatures as high as 45 °C. Lethal temperatures for algae and plants are likely to be much higher than would occur in lowland rivers. The effects of increases in water temperature on algae and plant growth are likely to be predominantly positive, presuming that nutrients are not limiting and the system is not subject to major disturbance. Therefore, no standards are recommended for protecting plants and algae.

There is relatively detailed information available on the effects of water temperature on aquatic macroinvertebrates. Water temperature can affect abundance, growth, metabolism, reproduction, and activity levels of aquatic insects. A detailed analysis of 88 New Zealand rivers (Quinn and Hickey 1990) identified water temperature as one of the important variables affecting species distribution. Stoneflies (Plecoptera) were largely confined to rivers between 13 and 19 °C, and mayflies (Ephemeroptera) were less common in rivers with maximum temperatures of > 21.5 °C (Quinn and Hickey 1990).

Laboratory studies of the effects of water temperature on invertebrate taxa have also identified Ephemeroptera and especially Plecoptera and as being particularly sensitive to high water temperatures. The common mayfly (*Deleatidium* spp.) is a common invertebrate species in many West Coast Rivers with a LT_{50} (the temperature at which 50 % of individuals will die) of 22.6 °C. There is the potential at high temperatures for *Deleatidium* to be replaced by the grazing snail *Potamopyrgus antipodarum*, which has a much higher LT_{50} (31.0 °C). Potamopyrgus can be considered a less desirable taxa, as it is a less attractive prey item for trout and native fish. Some recent research has suggested that *Deleatidium* may be able to survive short periods of high temperatures, providing they have experienced a summer acclimation period.

Fish are often strongly affected by temperature, with effects of temperature on mortality, growth and reproductive behaviour all described from New Zealand or elsewhere. Some of these effects are direct, with water temperature affecting behaviour, egg maturation, growth and mortality. Other effects are subtler; increased water temperatures can increase rates of disease, reduce resistance to pollutants, and reduce competitive abilities. Approximate preferred temperatures of some main New Zealand fish groups, in degrees celsius, include: just above 25 for short fin eels and just below 25 for long fins; around 20 for many bully species; and below 20 for trout and galaxid species. Greater detail is provided in Richardson et al. (1994).

3.1.3 Dissolved Oxygen and Biochemical Oxygen Demand

In order to characterise the potential for a body of water to lose oxygen, Biochemical Oxygen Demand (BOD) is often measured. The BOD of water may be defined as the amount of oxygen required for aerobic microorganisms to oxidise organic matter to a stable inorganic form.

Unpolluted waters typically have BOD_5 (5 day biochemical oxygen demand) values of 2 mg/L or less, whereas receiving waters of waste may have values up to 10 mg/L or more, particularly near a point of a wastewater discharge. Raw sewage has a BOD_5 of about 600 mg/L, whereas treated sewage effluents have BOD_5 values ranging from 20 to 100 mg/L depending on the level of treatment applied.

An adequate supply of dissolved oxygen (DO) is essential to the metabolism of all aerobic organisms and for the maintenance of purification processes in aquatic systems. DO levels are most often reduced in aquatic ecosystems directly by the addition of organic material and indirectly through the addition of plant nutrients (ANZECC 2000).

The total amount of oxygen that can be dissolved in a water body is dependent upon temperature and salinity. By measuring the DO content, the effects of oxidisable wastes (e.g., human and animal faeces, dead algae) on receiving waters may be assessed. DO levels also indicate the capacity of a natural body of water for maintaining aquatic life. The DO depletion in nutrient enriched waters may be offset during the day by algal photosynthesis. As photosynthesis requires light, a high DO concentration may build up during the day but depletion will occur during the night due to respiration of the aquatic plants.

Concentrations of dissolved oxygen below 5 mg/L may adversely affect the functioning and survival of biological communities and below 2 mg/L may lead to the death of most fish.

Aquatic heterotrophic bacteria and fungi (the main components of undesirable feathery, cotton-wool-like growths commonly referred to as “sewage fungus”) grow in response to readily degradable organic compounds, such as short-chain organic acids, sugars, and alcohol, which are sometimes found in wastewater discharges (e.g., dairy shed, piggery, meat works, and cheese factory effluents). In doing so, they consume oxygen from the water and can detract from the aesthetic appeal of a water. To prevent undesirable growths of sewage fungus, concentrations of BOD_5 in the receiving water should be less than 1-2 mg/l depending on analytical methods (MfE, 2000). Sewage fungus should not be visible to the naked eye as obvious plumes or mats.

Water quality criteria for dissolved oxygen generally state that daytime DO concentrations should not be permitted to fall below 6 mg/L (80 % saturation) for water quality classes AE (aquatic ecosystems), F (fisheries), FS (fish spawning), and SG (gathering or cultivation of shellfish for human consumption), as specified in the Third Schedule of the RMA 1991. The West Coast Water Management Plan classifies all freshwater bodies as AE (Aquatic Ecosystem) except those identified for bathing.

3.1.4 Suspended Sediment, Turbidity & Clarity

Sediments suspended in the water column are often referred to as suspended solids. “Turbidity” is an optical property of water where suspended and some dissolved materials cause light to be scattered and absorbed rather than be transmitted in straight lines. Clarity refers to the “transparency” of water.

Turbidity and suspended solid sampling have been used traditionally as methods for determining the degree of impact and sediment loading in waters. Assessing ‘visual water clarity’, measured using either ‘Secchi’ (for vertical water clarity) or ‘black’ disks (for horizontal water clarity) is now recommended for determining the visual and ecological effects of turbidity (MfE 1994). The greater the viewing distance, the greater the water clarity.

For most rivers, suspended solid concentration is positively correlated with turbidity, and both suspended solids concentrations and turbidity are inversely correlated with visual clarity. In other words, as the visual clarity decreases, suspended solid concentration and associated turbidity increase.

In rivers, excessive concentrations of suspended sediment can affect chemical and physical water characteristics, plants, algae, invertebrates, and fish, as well as human aesthetic, recreational, and spiritual values, as described below.

Sediment influxes can physically alter rivers and lakes by creating excessive turbidity and changing the nature of the bed. Coarser graded particles fill in the interstices between stones and cobbles, while finer graded particles smother or “blanket” the bed.

Sediment-laden water affects benthic macroinvertebrates by five primary mechanisms. These are:

- reduction of light penetration;
- abrasion;
- absorbed toxicants;
- changes in substrate character; and

- reduction in food quality.

Increased water turbidity, caused by suspended sediments, can affect benthic algae and macrophyte growth by reducing light penetration through the water column. This can reduce the “euphotic depth” of water (the depth at which irradiance, the penetration of diffuse light from the sun into water, is reduced to 1 % of the surface value, a point below which most aquatic plants can not grow for the lack of light). Altering the natural euphotic depth of a river or lake can result in a shift in plant and algal communities, that in turn, can affect the composition of the benthic invertebrate and fish communities. As well as reducing algal growth by reducing light penetration, fine sediments can smother algae and plants when they settle out.

Reduction of light penetration reduces periphyton production, which may result in a limiting food supply for the invertebrates (as stated above). Abrasion can act directly on benthic invertebrates by physical contact and, indirectly, by abrading periphyton.

Elevated levels of sediment in rivers and lakes affect fish, both directly and indirectly. Direct effects usually occur when concentrations of suspended solids are high. These include avoidance of turbid water by some fish, lower growth rates, impairment of growth in fish that use vision during feeding, and clogging of gills resulting in death. Indirect effects include reduction in the invertebrate food source (by mechanisms discussed above), avoidance by adult fish of silted gravels for spawning, and high egg mortality due to reduced oxygen levels in gravel fouled by silt deposition.

Turbidity, caused by suspended solids affecting the colour and clarity of water, may also have special significance to humans. Under New Zealand law, discharges of contaminants to water are not supposed to cause conspicuous changes in water colour and clarity (Resource Management Act 1991, section 70). Most people accept that the visual clarity of running water decreases as the flow increases (Davis-Colley 1990). However, increases in turbidity that occur during low or normal flows are generally regarded as unacceptable.

Studies on bathing preference in New Zealand waters revealed that an average minimum water clarity of 1.2 m is required before water is perceived as suitable for bathing (Smith et al. 1991).

As discussed above, decreased water quality, due to increased concentrations of suspended solids, can affect freshwater aquatic organisms and human values in a number of ways. In order to protect these attributes, guidelines have been developed by the Ministry for the Environment (MfE 1994). These numerical guidelines were developed to aid the interpretation of the narrative guideline found within the RMA (1991) that implies that discharges should not cause conspicuous changes in colour or clarity (section 107). MfE guidelines of relevance to water clarity are:

Visual clarity change

For Class A waters where visual clarity is an important characteristic of the water body, the visual clarity should not be changed by more than 20 % (visual clarity is measured with a black disk). For more general waters the visual clarity should not be changed by more than 33 % to 50 % depending on the site conditions.

Significant adverse effects on aquatic life

The protection of visual clarity (as recommended above) will usually also protect aquatic life. Settlement of solids onto the beds of water bodies should be minimised, but guidelines have not been recommended. For lowland New Zealand rivers ANZECC (2000) recommends clarity trigger levels of 0.8m. In this document we will use 10 NTU as a turbidity guideline (Quinn et al. 1992), opposed to 5.6 NTU (ANZECC 2000), which is difficult to apply.

Water managed for contact recreation.

Visual clarity affects bather preferences. Potential hazards should be visible in bathing waters and thus it is recommended that in such waters the horizontal sighting range of a 200 mm black disk should exceed 1.6 m (MfE 1992). Smith et al. (1991) recommend that total suspended solids should not exceed 4 mg/l, and turbidity should not exceed 2 NTU, but these figures are not used as a guide here as it should be applied to base flow samples only. This also applies to the ANZECC (2000) default trigger value for lowland river water clarity of 0.8 m, which is referenced from unmodified or slightly disturbed ecosystems. Many WCRC samples were collected during periods when flows may have been insufficiently low for effective use of these latter guidelines i.e. higher flows normally correspond with increased mobilisation of suspended sediment and a subsequent decrease in visual clarity.

3.1.5 Conductivity

The concentration of dissolved solids in solution is generally determined by salinity or conductivity measurements. Conductivity is a numerical expression of the ability of an aqueous solution to carry an electric current. This ability depends on the presence of ions, their total concentration, mobility, valence, and relative concentrations, and on the temperature during measurement (APHA 1985).

Anions (including bicarbonates, carbonates, chlorides, sulfates, phosphates, and often nitrates) occur in combination with such metallic cations as calcium, sodium, potassium, magnesium, and iron, to form ionisable salts. Because of the high availability and solubility of carbon dioxide, carbonates are usually the most abundant salts in fresh water.

Total dissolved solids (in mg/l) may be obtained by multiplying the conductance (in m*S*/m) by a factor, which is commonly between 0.55 and 0.75. The lower these measurement are, the more pure the water.

Certain dissolved mineral salts serve as nutrients for plants, while other salts may limit metabolism through osmotic effects. The conductivity of a liquid increases in relation to the concentration of dissolved ionised substances and, therefore, provides an indirect measure of the concentration of dissolved salts in a water sample. Conductivity monitoring is often used as a surrogate measure of nutrient enrichment in rivers.

Conductivity can be greatly affected by geology with streams in limestone catchments often having conductivities > 300 µ*S*/cm. There are no guidelines for conductivity levels in water (ANZECC 2000) but it is suggested that guidelines for southeastern Australian coastal rivers may be applicable where geology is not a significant factor (i.e. 125-300 µ*S*/cm).

3.1.6 Plant Nutrients: Nitrogen and Phosphorus

Nutrient monitoring in relation to nuisance aquatic plant and algal growths usually focuses on nitrogen and phosphorus. Dissolved inorganic nutrient concentrations are most relevant for predicting periphyton and macrophyte biomass in flowing waters. However, total nutrient concentrations are also relevant in rivers because particulate material can settle out in calm areas and become biologically available to plants via mineralisation (MfE 1992).

Aquatic plant and algal growths are important in rivers and streams as they provide food for both invertebrate and vertebrate life forms that live in, or are associated with, the water. However, if algal growth becomes excessive, due to an oversupply of nutrients (particularly nitrogen and phosphorus), the quality of the river or lake ecosystem deteriorates.

In most catchments where human impacts have been minimised, phosphorus and sometimes nitrogen are generally in short supply. As human activities intensify, the supply of both elements increases, leading to over-enrichment with the associated threat of eutrophication. The severity of eutrophication in a water body is also strongly controlled by the flushing rate. Rapidly flushed areas can tolerate higher levels of nutrient inflows than stagnant areas. Careful monitoring of phosphorus and nitrogen levels, along with flushing rates will, therefore, give a good indication of the susceptibility to eutrophication of a particular water body.

For New Zealand lowland rivers the trigger value for total nitrogen (TN) is 0.614 mg/L, and for total phosphorus (TP) 0.033 mg/L (ANZECC 2000).

In some circumstances it be more useful to consider dissolved inorganic nitrogen (DIN) and dissolved reactive phosphorus (DRP), as these are the forms that are readily assimilated by living organisms. DIN is made up of a combination of soluble oxides of nitrogen (nitrites/nitrates (NO_x) and ammoniacal nitrogen (NH_4^+). Trigger values for NO_x are 0.444 mg/L, and 0.010 mg/L for DRP (ANZECC 2000). The upper limit for DIN (MfE 1992) is 0.10 mg/L.

3.1.7 Ammoniacal nitrogen, ammonia, and ammonium

Ammonia is a common constituent of aquatic environments. It is present both as a natural breakdown product of nitrogenous organic matter and as a contaminant from wastewater discharges and run-off.

Ammoniacal nitrogen is the combination of ammonium ions (or ionised ammonia) (NH_4^+), and [un-ionised] ammonia (NH_3). The prevalence of these two forms is dependent on the pH, temperature, and salinity of the water. Concentrations are usually expressed either as total ammonia (or ammoniacal nitrogen, the sum of NH_3 and NH_4^+), or as concentration of the un-ionised NH_3 only. NH_3 is the main poisonous component for aquatic organisms, so when ammoniacal nitrogen is quoted, the pH and temperature are also relevant in determining toxicity (Figure 3.1.1).

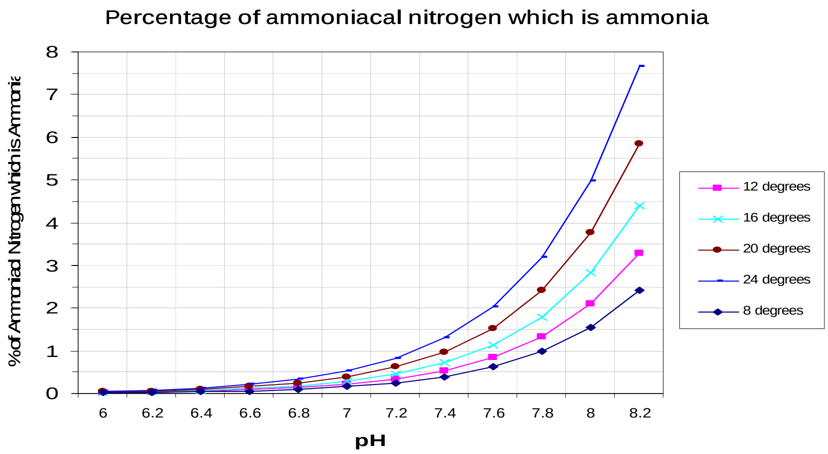


Figure 3.1.1: Percentage of ammoniacal nitrogen which is ammonia depending on the water pH and temperature.

Most of the trigger values for toxicants in the 2000 ANZECC guidelines have been derived using data from single-species toxicity tests on a range of test species, because these formed the bulk of the concentration–response information. 'High reliability' trigger values were calculated from chronic 'no observable effect concentration' (NOEC) data. However the majority of trigger values were 'moderate reliability' trigger values, derived from short-term acute toxicity data (from tests ≤ 96 hour duration) by applying acute-to-chronic conversion factors.

An ammoniacal nitrogen value of 0.9 mg/L (at pH 8, 20 °C), has been suggested as a high reliability (95 %) trigger value for freshwater (ANZECC 2000). This has been used as a general worst case benchmark as some West Coast streams can achieve this level. However, this trigger value varies with pH and temperature, and most temperatures and pHs are lower than this meaning the trigger value for most streams will be higher (refer Table 3.1.1).

Table 3.1.1 2000 ANZECC freshwater trigger values for total ammonia-N at different pH (temperature not taken into account).

pH	Freshwater trigger value (mg/L ammoniacal nitrogen-N)	pH	Freshwater trigger value (mg/L ammoniacal nitrogen-N)
6.5	2.46	7.8	1.18
6.6	2.43	7.9	1.03
6.7	2.38	8.0	0.90
6.8	2.33	8.1	0.78
6.9	2.26	8.2	0.66
7.0	2.18	8.3	0.56
7.1	2.09	8.4	0.48
7.2	1.99	8.5	0.40
7.3	1.88	8.6	0.34
7.4	1.75	8.7	0.29
7.5	1.61	8.8	0.24
7.6	1.47	8.9	0.21
7.7	1.32	9.0	0.18

3.1.8 Faecal microbiological indicators

Microbiological criteria are important because humans (particularly children) can contact various diseases from microbes in water: from drinking it, swimming in it, or eating shellfish harvested from it. The categories of microbes that can cause disease (pathogens) are well documented (e.g. McNeill 1985). Examples of water-borne diseases include: salmonella, gastroenteritis, hepatitis, and giardia.

To contain the risk of contracting such water-borne diseases various criteria have been derived from studies in which the density of suitable “indicator” organisms is correlated with disease risk. An acceptable value of this risk is then selected. Unfortunately, the relationship of the disease risk to the density of the “indicator” organisms is not clear.

Numerical standards are applied to New Zealand waters to protect them for recreational water use and for the gathering of shellfish for consumption. Typically, faecal coliforms and Enterococci are the groups of bacteria used as indicators of public health concern.

The main water quality parameters used for monitoring WCRC sites are faecal coliforms/*Escherichia coli* and Enterococci. The later is used only at sites that have tidal influence or are located in marine waters. *E. coli* and Enterococci seasonal medians were plotted for all WCRC contact recreation sites (Section 7). Individual values have been plotted for *E. coli* with values separated by the following criteria: circle =

acceptable (< 260 *E. coli*/100 ml), triangle = alert (260 – 550 *E. coli* 100 ml), and square = action (> 550 *E. coli* 100ml) values in accordance with MfE (2003) contact recreation guidelines for individual values. For medians, the Department of Health (1992) guidelines for contact recreation waters recommend a season median of 126 E.coli/100 ml, with ANZECC (2000) stipulating a median of 150 faecal coliform cfu/100 ml and 35 Enterococci/100 ml as a safe limit.

The older MfE (1999) secondary contact guideline was used as a benchmark for comparing faecal coliforms among SOE monitoring sites (1000 cfu/100 ml median from a minimum of five samples taken at regular intervals not exceeding one month has been use). This was easier to apply to the SOE monitoring site data than the 2003 MfE contact recreation guidelines, and was the same figure as that used for 1999 ANZECC stock drinking water quality guidelines also applied here. The ANZECC 1992 guidelines specify for stock drinking a faecal coliform limit of 1000 cfu/100 ml, where as the limit for stock drinking in the ANZECC 2000 guidelines is 1000 cfu/100 ml.

Guidelines for shellfish gathering recommend that the mean faecal coliform content shall not exceed 14 cfu/100 ml and not more than 10 % should exceed 43 cfu/100 ml (MfE 2003).

3.1.9 Sulfate

Sulfate is found in most natural waters as a result of the dissolution of sulfate-bearing minerals in soils and rocks. Mine wastewaters, tannery wastes and other industrial discharges often contain high concentrations of sulfate. Under anoxic conditions bacteria in water can reduce sulfate to sulfide, which results in the release of hydrogen sulfide, causing an unpleasant taste and odour and increasing the potential for corrosion of pipes and fittings.

Sulfate is used as an indicator by the WCRC to monitor the effects of mining, current and historic, on water chemistry. The main environmental implications associated with sulfate are for consumption by stock and humans. Sulfate is an essential element for animal nutrition. It is not a highly toxic substance, but excessive concentrations of sulfate in water typically cause diarrhoea, especially if a change from low to high sulfate water occurs quickly. Animals generally avoid water containing high sulfate concentrations in favour of water containing lower concentrations. No adverse effects to stock are expected if the concentration of sulfate in drinking water does not exceed 1000 mg/L. Adverse effects may occur at sulfate concentrations between 1000 and 2000 mg/L, especially in young or lactating animals or in dry, hot weather when water intake is high. These effects may be temporary and may cease once stock become accustomed to the water. Levels of sulfate greater than 2000 mg/L may cause chronic or acute health problems in stock. The USEPA recommended maximum guideline for sulfate as a secondary contaminant in human drinking water is currently 250 mg/L, based on aesthetic effects (i.e., taste and odour).

3.2 Biological Parameters

3.2.1 Periphyton

Periphyton is the slime coating stones, wood, weeds or any other stable surface in streams and rivers. The community is composed predominantly of algae, cyanobacteria (formerly “blue-green algae”) and diatoms (Biggs 2000). Periphyton occurs in a variety of thicknesses and forms depending on conditions.

Periphyton is the “foodstuff” of aquatic grazing animals, mainly macroinvertebrates, which are, in turn, fed upon by fish. Without periphyton many waterways would be barren of life. Periphyton also plays a role in the maintenance of water quality, the community removing nitrogen, phosphorous and unwanted organic contaminants (Biggs 2000). During periods of low flows and high nutrient levels, however, periphyton communities may proliferate to the extent that aesthetics, biodiversity and other in stream parameters are compromised.

Periphyton is assessed by the WCRC once during autumn and once during spring using the Rapid Assessment Method 2 (RAM 2), which uses four transects, each with five points where a stone is selected with percentage cover of various category of periphyton visually estimated for each stone (Biggs & Kilroy 2000). Categories are differentiated by colour and thickness, and are likely to represent certain groups of periphyton. Categories have an assigned score, and the combination of these can be used to calculate an enrichment indicator.

3.2.2 Macroinvertebrates

Freshwater benthic macroinvertebrates are bottom-dwelling animals that have no backbone and are, simply speaking, large enough to be seen with the naked eye. In the case of macroinvertebrates collected by the WCRC for monitoring, they are of a size at least as large as 500 microns (0.5 mm) as this is the mesh size of the net used to collect them. Macroinvertebrates include insect larvae (e.g. caddisflies, mayflies, stoneflies), aquatic worms (oligochaetes), aquatic snails, and crustaceans (e.g., amphipods, isopods and freshwater crayfish). Macroinvertebrates utilise a variety of food sources depending on the species.

Aquatic macroinvertebrates are good indicators of ecological change in freshwater environments. Changes in density (numbers) can indicate changes in productivity of algae (e.g. periphyton), which may suggest increased nutrient inflows. Because different macroinvertebrate species have different tolerances to environmental factors, such as dissolved oxygen, chemical pollutants and fine sediment, the presence or absence of different species can also indicate changes in water quality.

Taxonomic richness (number of different types of animals); Ephemeroptera, Plecoptera, Trichoptera number and percentage (Lenat 1988); the Macroinvertebrate Community Index (MCI) (Stark 1985); and the Quantitative Macroinvertebrate Community Index (QMCI) (Stark 1993), are typical indices that are used to assess macroinvertebrate community health. The MCI uses the occurrence of specific macroinvertebrate taxa to determine the level of organic enrichment in a stream, using the following formula:

$$MCI = \left(\frac{\Sigma \text{ of taxa scores}}{\text{Number of scoring taxa}} \right) X\ 20$$

Taxa are scored between 1 and 10, with low scores indicating high tolerance to organic pollution and high scores indicating taxa that will only be found in "pristine rivers" (Stark 1985). A site score is obtained by summing the scores of individual taxa and dividing this total by the number of taxa present at the site, then multiplying by 20. Scores can range from 0 (no species present) to 200, with different scores indicating different pollution status (Table 3.3).

The QMCI (Stark 1993) uses the same approach as the MCI but weights each taxa score on the abundance of the taxa within the community. As for MCI, QMCI scores can be interpreted in the context of national guidelines (Table 3.2.1).

$$QMCI = \frac{\text{Taxa Score X No. present in that taxa}}{\text{Total No. present}}$$

Table 3.2.1 Interpretation of macroinvertebrate community index values from stony riffles (after Boothroyd & Stark 2000).

Interpretation	MCI	QMCI
Clean water	> 120	> 6.00
Doubtful quality	100-119	5.00-5.99
Probable moderate pollution	80-99	4.00-4.99

MCI and QMCI scores may be affected by a number of factors other than pollution (e.g. bed stability, recent flow conditions and regimes, water temperature, habitat type). Consequently, a useful approach is to compare MCI and QMCI scores upstream and downstream of an impact. In such a situation the differences between scores for the index are much more important than the actual scores.

4 Surface water quality based on REC

The WCRC maintains thirty-nine sites for characterising State of the Environment with respect to water quality. Replication varied with few cool wet and exotic forestry sites, but replication on the whole was adequate. There were many significant differences in water quality parameters between many of the REC classes. Many of these REC classes, were related between different REC levels, which contributed to some of the distinctive groupings charactering streams and rivers into types. Below are some examples of stream characteristics that were often found together, bearing in mind there was some gradient between them. Stream types could often be characterised into:

- Streams with a hill (high) source of flow; hard sedimentary or plutonic geology, often incorporating larger rivers; higher, less variable water quality; brown trout more abundant.
- Lowland streams (low elevation source of flow); higher turbidity, nutrients and temperature (which may not solely have been a response of anthropogenic disturbance); smaller, more variable water quality and susceptible to impact; potentially higher fish and invertebrate diversity.
- Streams draining predominantly agricultural catchments (sub-set of lowland streams); comparatively poorer water quality with fewer sensitive macroinvertebrate taxa. Nutrients may be elevated where there is significant groundwater recharge (i.e. spring source).
- Stream catchments with soft sedimentary geology; higher turbidity; distinctive physically and chemically from other geology classes; smaller size; lower altitudinal source of flow.

West Coast streams and rivers often have at least some indigenous vegetation in their catchments, and this - combined with high, regular rainfall – provides mitigation of anthropogenic impacts. Nearly all of the sites in this study were to varying extents impacted in some way. A moderate trend was observed showing higher water quality in larger streams and rivers compared to that of smaller ones, although the latter did not generally fall below guidelines. It is possible that slight changes in water quality are of greater significance in these larger rivers and might be borne in mind when making water quality comparisons between large and small waterways.

Summarising site comparisons to guidelines: approximately a third of sites were above the faecal coliform guideline used for bathing seasonal medians. No site medians were above the guideline for secondary contact recreation. Twelve percent of sites failed the clarity guideline relating to protection of aquatic life, which is based on un-disturbed to slightly disturbed ecosystems.

No guidelines applied to sulphate or conductivity, but some sites had high levels compared to others. High sulphate was found in catchments with major mining operations, as were high conductivities. Limestone geology and saltwater intrusion also influenced the latter variable at some sites. pH's lower than our designated guideline occurred in catchments with current and historic mining operations. Mining did not cause low pH where it occurred in areas where suitably calcareous geology provided a buffer.

Dissolved oxygen, temperature, and nitrogenous nutrients were within respective guideline levels. Phosphorus nutrients exceeded by 19 %, but this only constituted 4 sites, hence low replication.

Water quality as determined by macroinvertebrate communities ranged from moderately polluted to pristine. The middle of this range fell in the doubtful quality (mild pollution) category, erring toward pristine. Macroinvertebrate communities differed less between REC classes compared to other parameters (excluding pastoral landcover), but ordination indicated more distinct responses by them based on REC differentiation, and these supported water quality observations based on physicochemical parameters among REC classes.

4.1 Comparison of physical and biological parameters based on REC classification

Most sites in this study were categorised as having cool, extremely wet climates (mean annual temp. < 12 rainfall > 1000 mm)(Table 4.1.1). Cool wet (mean annual temp. < 12, rainfall > 500, < 1000 mm) represented only 6 % of sites, thus had far less replication compared with extremely wet.

Table 4.1.1 Table showing the proportion of sites in a class within their respective REC level.

Level	Percentage of class in that level
Climate	
Cool extremely wet	94
Cool wet	4
Warm wet	2
Source of Flow	
Hill	37
Low elevation	47
Spring	8
Geology	
Alluvial	38
Soft sedimentary	32
Hard sedimentary	19
Plutonic	11
Landcover	
Indigenous forest	67
Exotic forest	5
Pasture	23
Scrub	5
Order (ie size)	
High order	35
Mid order	42
Low order	13

Nearly half (47 %) of sites had a low elevation source of flow (> 50 % of annual precipitation < 400 m a.s.l.). Low elevation sites were split evenly between soft sedimentary and alluvial geological classes.

Hill source sites (> 50 % of annual precipitation between 400 and 1000 m a.s.l.) were less numerous, but well represented. The majority of them (77 %) had either plutonic or hard sedimentary geology. Landcover at seventy percent of high source sites had predominantly indigenous forest landcover, with scrub cover for the remaining. Spring sourced streams accounted for only a few sites, all of which were situated in alluvial sand.

All four geology classes were well represented. Of the softer types, the alluvial and soft sedimentary classes were most common. The soft sedimentary class had few developed landcovers, with pasture and exotic forestry dominating in only 9 % and 6 % of catchments, respectively, with the remaining either indigenous forest or scrub (47 % and 38 %, respectively). Fifty percent of alluvial sites had pasture landcover, and 85 % of pasture sites had alluvial geology. Pastoral land on the West Coast is in most cases confined to alluvial lowlands and plains. A third of sites had catchments dominated with harder geology types, all of which with indigenous forest catchments.

As mentioned, catchment landcover was predominantly indigenous forest, with a range of geological classes occurring within it. Of the remaining landcovers, pasture was the next largest accounting for almost a quarter of sites, most of which having alluvial geology (85 %). Scrub and exotic forest split the remaining 10 percent.

Most sites were mid order in size (order 3-4). High order sites (order > 4) comprised just over a third, with low order sites (order < 3) less common at 13 %. All high order sites also had indigenous forest landcover. Mid order and low order sites had a mixture of landcover and geologies, although they were more likely to be low in elevation.

Statistical comparisons of REC land classes discussed in the following section are presented in Table 4.1.2 and 4.1.3.

Table 4.1.2 Table of one-way ANOVA results between REC classes for respective macroinvertebrate parameters. F – ratios (1 decimal place) are listed for ANOVA between all classes within a level. Probabilities (four decimal places) for comparisons between class pairs are included (there is no F – ratio relating to these). Yellow and orange cells denote probabilities below or equal to 0.01, and 0.05, respectively. The lowest REC class is listed for all comparisons.

Parameter → REC level ↓		SQMCI			MCI			Taxonomic richness			Percent EPT		
		F	*P*	*Low*	*F*	*P*	*Low*	*F*	*P*	*Low*	*F*	*P*	*Low*
Climate	CW vs. CX	8.5	0.003	CX	0.2	0.675	CW	4.7	0.032	CX	0.4	0.536	CX
Source of flow	All	2.3	0.099	L	2.6	0.074	S	12.9	0.000	H	0.4	0.648	S
	H vs. L	-	0.102	L	-	1.000	L	-	0.000	H	-	1.000	L
	H vs. S	-	0.478	S	-	0.087	S	-	0.000	H	-	1.000	S
	L vs. S	-	1.000	L	-	0.146	S	-	1.000	L	-	1.000	L
Geology	All	1.1	0.338	Al	1.8	0.145	Al	2.8	0.045	HS	1.3	0.265	HS
	SS vs. HS	-	1.000	SS	-	1.000	HS	-	0.250	HS	-	1.000	HS
	SS vs. PL	-	0.479	SS	-	1.000	SS	-	0.903	SS	-	0.706	SS
	SS vs. Al	-	1.000	Al	-	0.282	Al	-	1.000	SS	-	1.000	Al
	Al vs. HS	-	1.000	Al	-	1.000	Al	-	1.000	HS	-	1.000	HS
	Al vs. PL	-	0.421	Al	-	0.805	Al	-	1.000	Al	-	0.466	Al
	PL vs. HS	-	0.913	HS	-	1.000	HS	-	0.109	HS	-	0.353	HS
Landcover	All	14.4	0.000	P	13.9	0.000	P	8.6	0.000	S	5.0	0.003	P
	EF vs. IF	-	0.130	IF	-	0.023	IF	-	0.071	IF	-	1.000	IF
	EF vs. P	-	0.000	P	-	0.000	P	-	0.562	P	-	0.015	P
	EF vs. S	-	1.000	S	-	1.000	S	-	0.000	S	-	1.000	S
	IF vs. P	-	0.000	P	-	0.000	P	-	1.000	IF	-	0.010	P
	IF vs. S	-	1.000	IF	-	0.844	IF	-	0.000	S	-	1.000	S
	P vs. S	-	0.051	P	-	0.013	P	-	0.000	S	-	0.873	P
River order	All	7.2	0.001	LO	8.0	0.000	LO	6.3	0.002	HO	12.7	0.000	LO
	HO vs. MO	-	0.045	MO	-	0.150	MO	-	0.023	HO	-	0.085	MO
	HO vs. LO	-	0.001	LO	-	0.000	LO	-	1.000	HO	-	0.000	LO
	MO vs. LO	-	0.065	LO	-	0.010	LO	-	0.017	LO	-	0.000	LO

Table 4.1.3 Table of one-way ANOVA results between REC classes for respective water quality parameters. F – ratios (1 decimal place) are listed for ANOVA between all classes within a level. Probabilities (four decimal places) for comparisons between class pairs are included (there is no F – ratio relating to these). Yellow and orange cells denote probabilities below or equal to 0.01, and 0.05, respectively. The highest or lowest (depending on the parameter being compared) REC class is listed for all comparisons, except where there is no difference, denoted by two hyphens. Insufficient data is denoted by *

Parameter →		Turbidity			Dissolved oxygen %			pH			Conductivity			Temperature			Ammonia		
REC level ↓		*F*	*P*	*High*	*F*	*P*	*Low*	*F*	*P*	*Low*	*F*	*P*	*High*	*F*	*P*	*High*	*F*	*P*	*High*
Climate	CW vs. CX	28.1	0.000	CW	0.09	0.755	CW	0.1	0.827	CX	0.04	0.827	CX	8.2	0.004	CW	54.5	0.000	CW
Source of flow	All	18.0	0.000	L	1.2	0.290	S	37.3	0.000	L	0.7	0.510	L	59.4	0.000	S	51.0	0.000	S
	H vs. L	-	0.000	L	-	1.000	- -	-	0.000	L	-	0.740	L	-	0.000	L	-	0.000	L
	H vs. S	-	0.000	H	-	0.350	S	-	0.000	S	-	1.000	S	-	0.080	S	-	0.000	S
	L vs. S	-	0.000	L	-	0.380	S		1.000	- -	-	1.000	L	-	1.000	S	-	0.621	S
Geology	All	19	0.000	SS	1.4	0.243	AI	75.0	0.000	SS	5.1	0.002	SS	43.3	0.000	SS	40.0	0.000	SS
	SS vs. HS	-	0.000	SS	-	0.919	HS	-	0.000	SS	-	0.001	SS	-	0.000	SS	-	0.000	SS
	SS vs. PL	-	0.000	SS	-	1.000	PL	-	0.000	SS	-	1.000	SS	-	0.000	SS	-	0.000	SS
	SS vs. AI	-	0.000	SS	-	0.252	AI	-	0.000	SS	-	0.056	SS	-	1.000	SS	-	0.033	SS
	AI vs. HS	-	1.000	AI	-	1.000	AI	-	0.000	AI	-	0.801	AI	-	0.000	AI	-	0.000	AI
	AI vs. PL	-	1.000	AI	-	1.000	AI	-	0.000	AI	-	1.000	PL	-	0.000	AI	-	0.000	AI
	PL vs. HS	-	1.000	HS	-	1.000	PL	-	1.000	- -	-	1.000	PL	-	0.006	PL	-	1.000	- -
Landcover	All	5.6	0.001	EF	33.5	0.016	P	80.4	0.000	S	2.5	0.060	S	18.5	0.000	P	79.0	0.000	P
	EF vs. IF	-	0.004	EF	-	1.000	IF	-	0.002	EF	-	0.244	IF	-	0.330	EF	-	1.000	- -
	EF vs. P	-	0.005	EF	-	0.857	P	-	1.000	- -	-	0.057	P	-	1.000	-	-	0.045	P
	EF vs. S	-	0.000	EF	-	1.000	EF	-	0.000	S	-	0.678	S	-	0.367	EF	-	1.000	EF
	IF vs. P	-	1.000	IF	-	0.010	P	-	0.000	P	-	0.920	P	-	0.000	P	-	0.000	P
	IF vs. S	-	0.178	IF	-	1.000	IF	-	0.000	S	-	1.000	S	-	1.000	IF	-	1.000	IF
	P vs. S	-	0.401	P	-	1.000	P	-	0.000	S	-	1.000	S	-	0.001	P	-	0.000	P
River order	All	18	0.000	MO	10.1	0.000	LO	230.6	0.000	LO	14.7	0.000	LO	32.0	0.000	LO	86.3	0.000	LO
	HO vs. MO	-	0.000	MO	-	0.144	HO	-	0.000	MO	-	0.000	MO	-	0.000	MO	-	0.000	MO
	HO vs. LO	-	0.000	HO	-	0.000	LO	-	0.000	LO	-	0.000	LO	-	0.000	LO	-	0.000	LO
	MO vs. LO	-	0.000	MO	-	0.000	LO	-	0.000	LO	-	0.066	LO	-	1.000	LO	-	0.272	LO

Parameter →		Faecals			Total nitrogen			Nitrate			Total phosphorus			Dissolved reactive phosphorus			Sulfate		
REC level ↓		*F*	*P*	*High*	*F*	*P*	*High*	*F*	*P*	*High*	*F*	*P*	*High*	*F*	*P*	*High*	*F*	*P*	*High*
Climate	CW vs. CX	13.8	0.000	CW	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Source of flow	All	131.0	0.000	S	100.0	0.000	S	296.0	0.000	S	3.7	0.024	S	100.0	0.000	S	16.0	0.000	L
	H vs. L	-	0.000	L	-	0.000	L	-	0.000	L	-	1.000	L	-	0.002	L	-	0.000	L
	H vs. S	-	0.000	S	-	0.000	S	-	0.000	S	-	0.022	S	-	0.000	S	-	0.027	S
	L vs. S	-	0.000	S	-	0.000	S	-	0.000	S	-	0.015	S	-	0.000	S	-	0.000	L
Geology	All	69.9	0.000	AI	50.5	0.000	SS	119	0.000	SS	1.2	0.296	SS	1.2	0.296	SS	19.7	0.000	SS
	SS vs. HS	-	0.000	SS	-	0.000	SS	-	0.000	SS	-	1.000	SS	-	1.000	SS	-	0.001	SS
	SS vs. PL	-	0.000	SS	-	0.030	SS	-	0.000	SS	-	1.000	SS	-	1.000	SS	-	0.000	SS
	SS vs. AI	-	1.000	AI	-	0.36	SS	-	0.000	SS	-	1.000	SS	-	1.000	SS	-	0.000	SS
	AI vs. HS	-	0.000	AI	-	0.000	AI	-	0.000	AI	-	1.000	AI	-	1.000	AI	-	1.000	HS
	AI vs. PL	-	0.000	AI	-	0.000	AI	-	0.000	AI	-	0.440	AI	-	0.446	AI	-	0.535	AI
	PL vs. HS	-	1.000	PL	-	0.000	PL	-	1.000	- -	-	1.000	HS	-	1.000	HS	-	0.014	HS
Landcover	All	79.5	0.000	P	*	*	*	170	0.000	P	*	*	*	152	0.000	P	1.2	0.315	S
	EF vs. IF	-	1.000	IF	*	*	*	*	*	*	*	*	*	*	*	*	-	1.000	IF
	EF vs. P	-	0.002	P	*	*	*	*	*	*	*	*	*	*	*	*	-	1.000	P
	EF vs. S	*	*	*	*	*	*	*	*	*	*	*	*	-	0.000	P	-	0.700	S
	IF vs. P	-	0.000	P	*	*	*	-	0.000	P	*	*	*	*	*	*	-	1.000	IF
	IF vs. S	*	*	-*	*	*	*	-	0.016	IF	*	*	*	-	1.000	IF	-	0.661	S
	P vs. S	*	-*	*	*	*	*	-	0.000	P	*	*	*	-	0.000	P	-	0.803	S
River order	All	79.3	0.000	LO	152	0.000	MO	70.3	0.000	LO	1.9	0.149	MO	87.8	0.000	LO/MO	56.1	0.000	LO
	HO vs. MO	-	0.000	MO	-	0.000	MO	-	0.000	MO	-	0.343	MO	-	0.000	MO	-	0.000	MO
	HO vs. LO	-	0.000	LO	-	0.806	HO	-	0.000	LO	-	0.670	LO	-	0.000	LO	-	0.000	LO
	MO vs. LO	-	0.270	LO	-	0.000	MO	-	0.000	LO	-	1.000	MO	-	0.000	- -	-	0.000	LO

4.1.1 Climate

As mentioned, nearly all sites had CX climates. There is insufficient replication (3 sites) within the CW class to consider it well-represented class the West Coast region. CW sites are specific to the Karamea area, so can be used to compare this area to other areas on the West Coast. These Karamea/CW sites are located near the coast, normally draining indigenous forested hill country in their headwaters, and moving through dairy farmland in their lower reaches. As might be expected, temperatures were higher than the West Coast norm (Figure 4.1.1). So was turbidity, which might relate to the prevalence of erodable geology in this region. Poorer water quality was indicated by higher turbidity, ammoniacal nitrogen, faecal coliforms, and lower SQMCI's, although these sites had a more diverse fauna as suggested by a higher taxonomic richness.

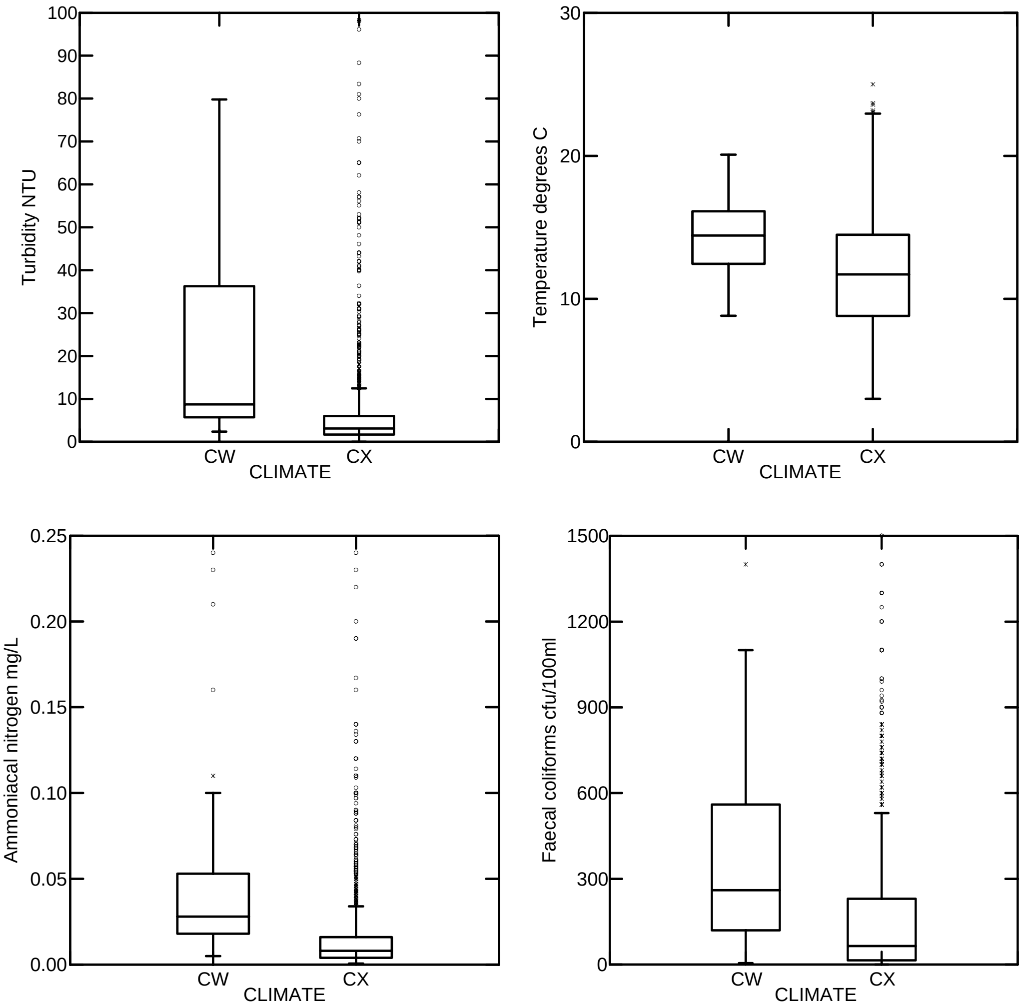


Figure 4.1.1 Distributions of turbidity, temperature, ammoniacal nitrogen, and faecal coliforms between climate classes CW (cool, wet), and CX (cool, extremely wet).

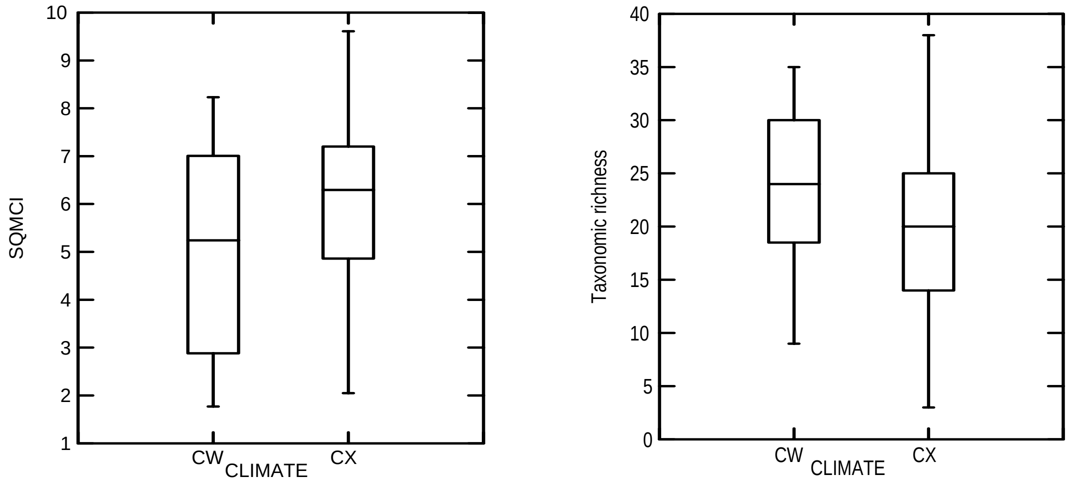


Figure 4.1.1 continued: Distributions of SQMCI and MCI between climate classes CW (cool, wet), and CX (cool, extremely wet).

4.1.2 Source of flow

The mountain source of flow class, which had few replicates, was incorporated into the hill source of flow class. Turbidity was highest and most variable in the low elevation class (Figure 4.1.2), all of which occurred in areas with soft sedimentary or alluvial geology. Hill source of flow sites were less turbid, with either indigenous forest or scrub landcover - both of which are indicative of a lack of development. Turbidity was lower in spring source sites, despite them all having pasture landcover. Spring streams are fed largely by groundwater that is very clear. They have little or no tributaries draining steeper gradient country, hence erosive forces are minimal.

The hill source of flow class had a pH around pH 7.5, while median pH for low elevation and spring source were lower - nearer pH 7. Lowland sites had greater variability, which probably reflects their more erodable geologies, and greater pastoral development. Lack of riparian shade associated with pastoral development, and lower altitude probably account for higher temperatures in low elevation sites.

Faecal coliform, and levels of all forms of nitrogen and phosphorus, were lowest at hill source of flow sites. Higher levels were found at low elevation sites, with the most at spring source sites – all of which were surrounded by intensive pastoral development. Median coliform levels in the pasture class exceeded the season median guideline for bathing, although this is an arbitrary comparison because these sites are unlikely to be used for bathing purposes. Spring source sites had a collective median on the guideline for DRP, well above other classes. Nutrients were evidently higher in spring source streams, probably originating from nutrient enriched groundwater. Pastoral agriculture is the dominant land use surrounding these spring fed streams. Sulfate was higher in streams with low elevation. Pyrite and sulphatic rocks are more abundant at lower elevations, as is the mining activity that often exposes them. Source of flow had less significance on macroinvertebrate communities, although hill source of flow sites had lower taxonomic diversity.

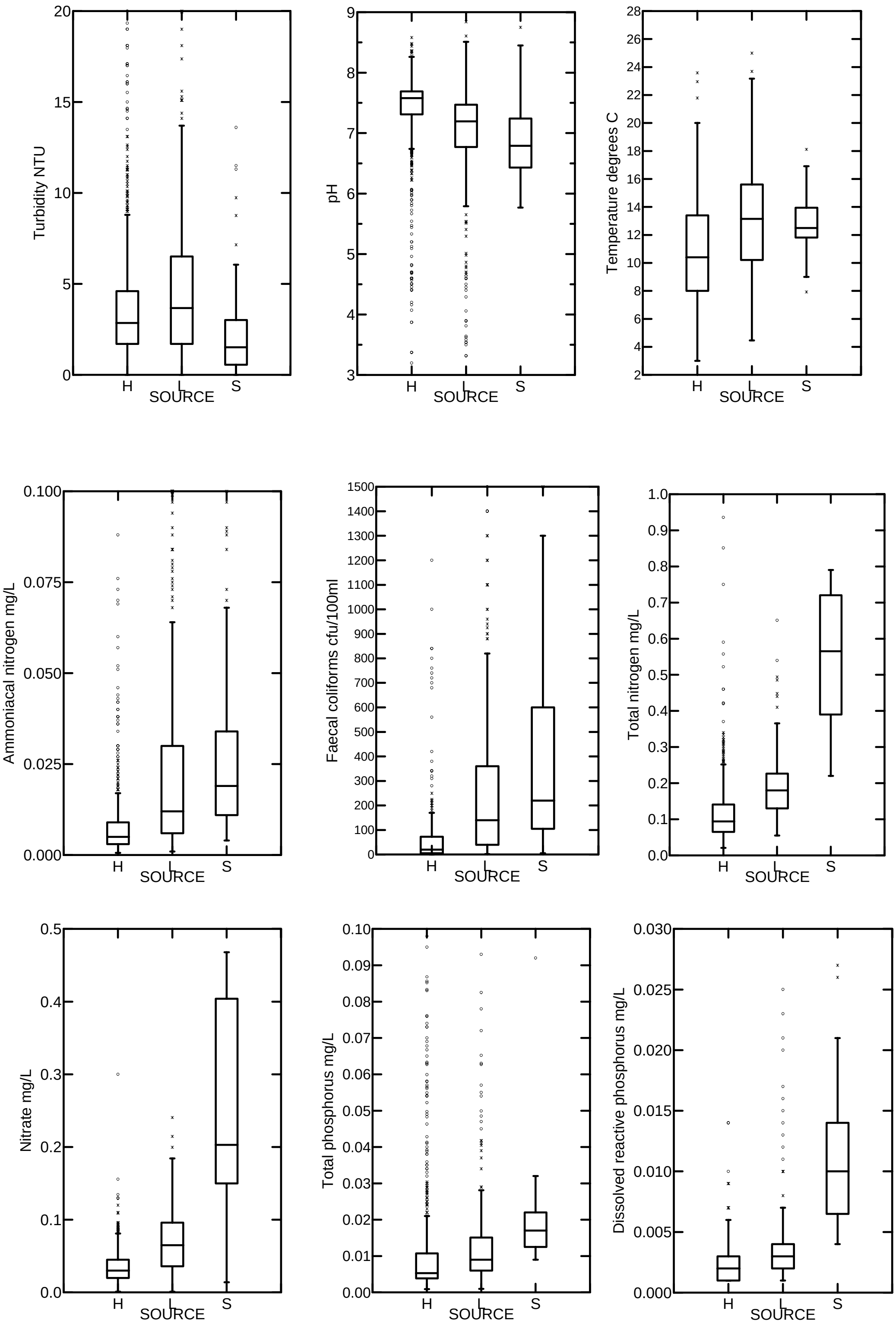


Figure 4.1.2 Distributions of turbidity, pH, temperature, ammoniacal nitrogen, faecal coliforms, total nitrogen, nitrate, total phosphorus, and dissolved reactive phosphorus between source classes H (hill country), L (low elevation), and S (spring).

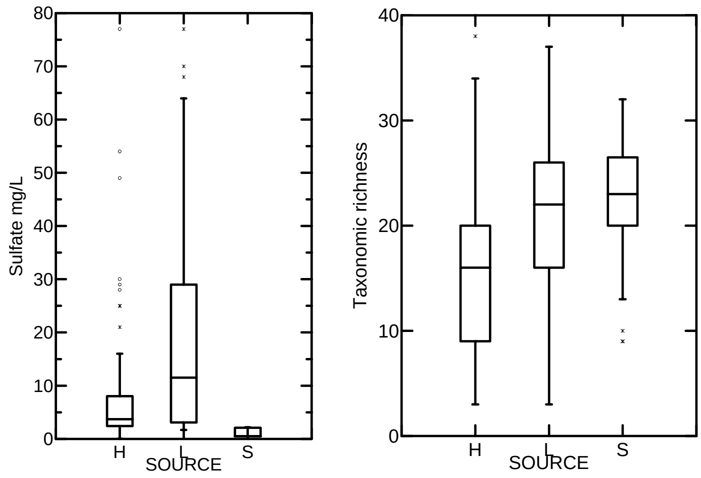


Figure 4.1.2 continued: Distributions of sulfate and taxonomic richness between source classes H (hill country), L (low elevation), and S (spring).

4.1.3 Geology

The soft sedimentary geology class differed significantly from the other geology classes for most parameters (Figure 4.1.3). It had higher turbidity than harder, and alluvial classes. Being more erodable, soft sedimentary class catchments can have continual sources of fresh material exposed to weathering (Maasdam and Smith 1994). That soft sedimentary was not higher yet probably relates to 85 % of soft sedimentary sites covered with indigenous or revegetating cover. Levels of pH were lowest and most variable among the soft sedimentary class, with the alluvial class also lower than the harder geologies. The hard sedimentary and PL classes varied little. Most mine sites occurred in catchments with soft sedimentary geology. Certainly, soft sedimentary catchments had higher conductivity than other classes, probably resulting from both mining and natural weathering.

Temperature was highest in the soft sedimentary class, followed by alluvial – both of which had exclusively low elevation, with all soft sedimentary sites either low to mid order in size. All geology classes had a similar degree of temperature variation.

The alluvial class had the highest faecal coliform concentrations (proven by ANOVA, despite the similarity with soft sedimentary when plotted), and nitrate levels. Soft sedimentary had the highest ammoniacal nitrogen and sulfate, caused by a low number of soft sedimentary sites that had consistently high levels. Lack of replication for the soft sedimentary class means high observed values for total nitrogen and total phosphorus (TP above guideline) should be interpreted with caution.

Macroinvertebrate community structure did not vary much between geologies, although taxonomic richness was higher in the PL class.

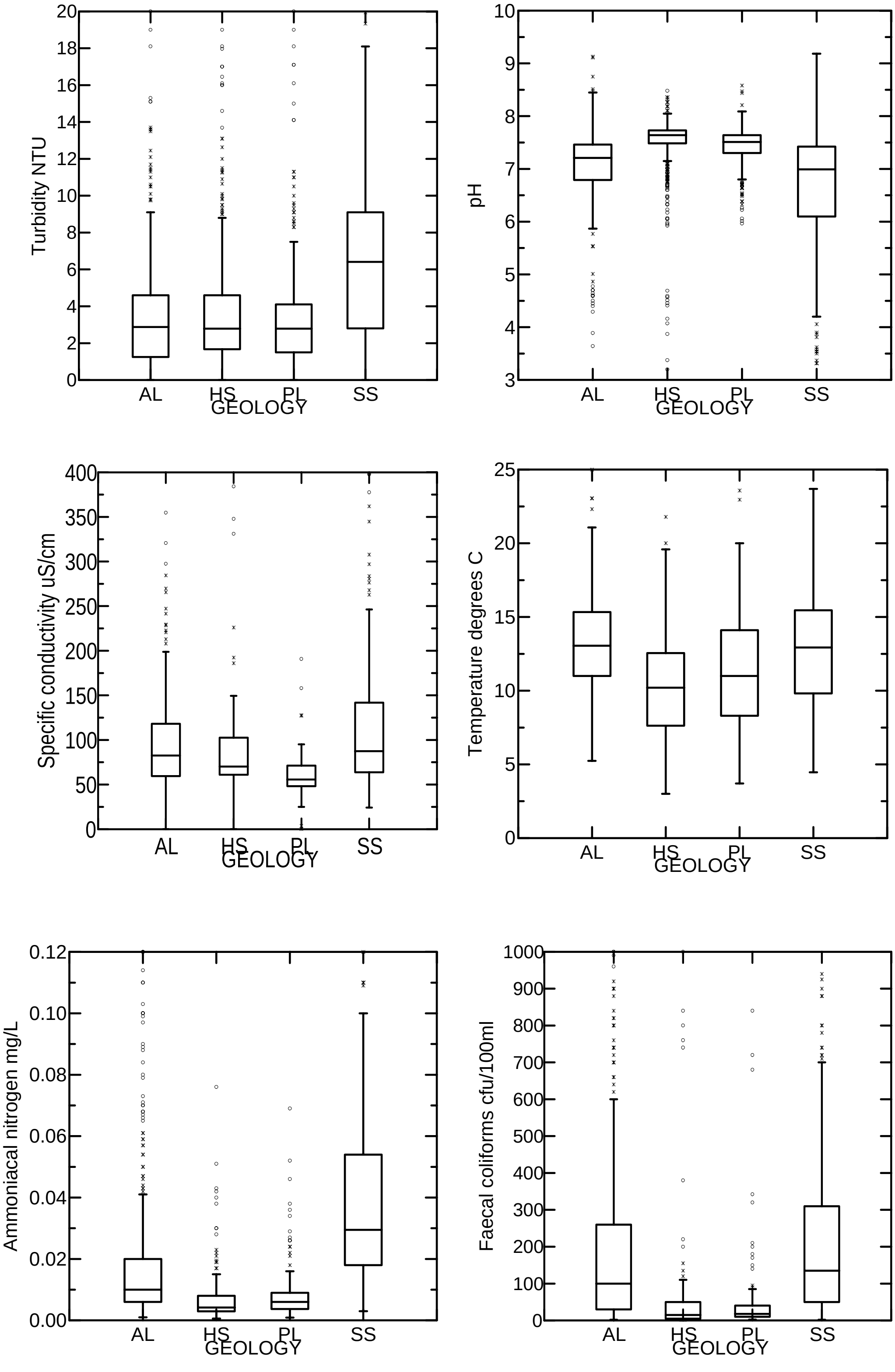


Figure 4.1.3 Distributions of turbidity, pH, specific conductivity, temperature, ammoniacal nitrogen, faecal coliforms between geology classes Al (alluvial), HS (hard sedimentary), PL (plutonic), and SS (soft sedimentary).

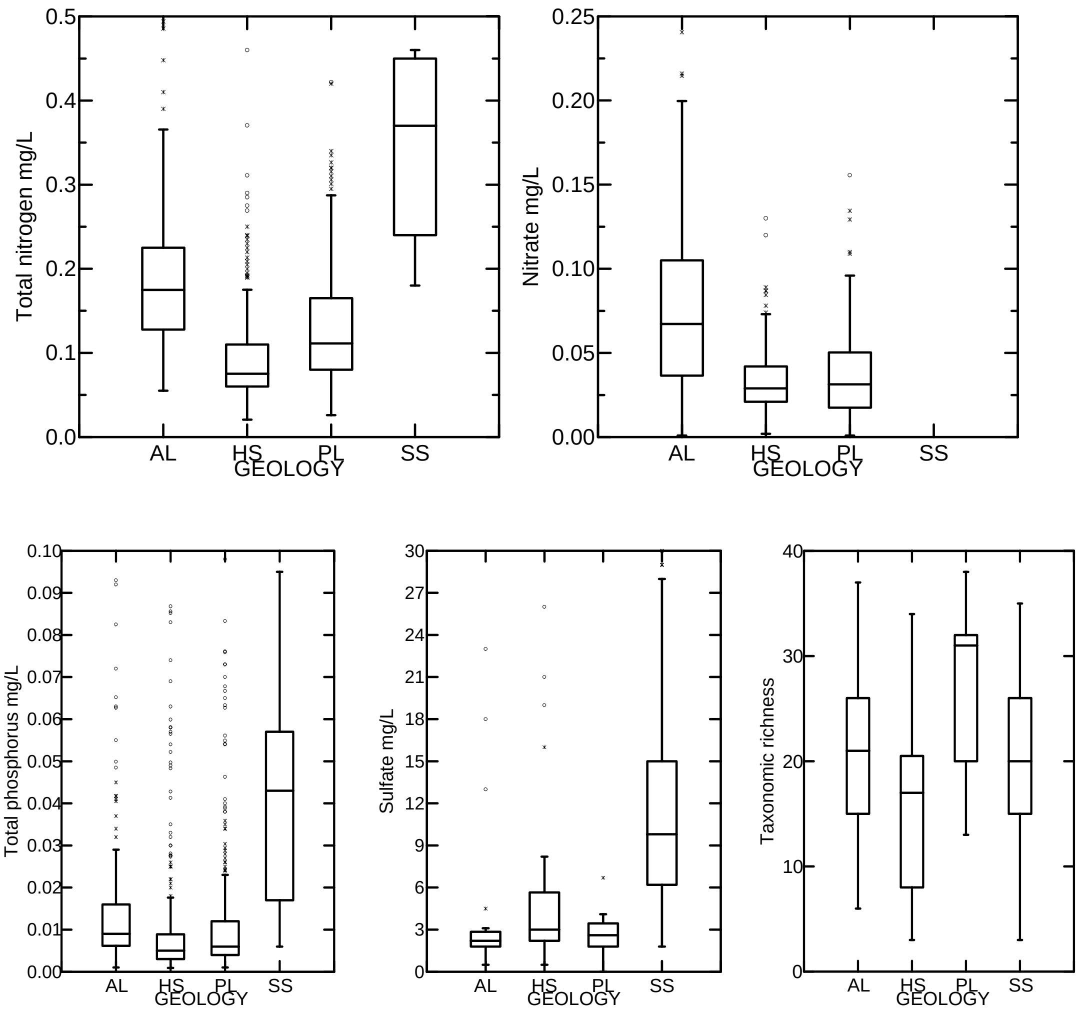


Figure 4.1.3 continued: Distributions of total nitrogen, nitrate, total phosphorus, sulfate and taxonomic between geology classes Al (alluvial), HS (hard sedimentary), PL (plutonic), and SS (soft sedimentary).

4.1.4 Landcover

Exotic forest sites had higher turbidity than others - exotic forestry only consists of three sites (Figure 4.1.4). Other classes were not significantly different. The indigenous forest upper-quartile for turbidity was nearly within 10 NTU, but there were many outliers above this figure. Indigenous forest was the most common landcover class with the most data, and while indigenous forest catchments were predominantly in indigenous forest, they often have disturbed areas resulting from development, often in their lower reaches. Poorer dissolved oxygen and temperature conditions were apparent in the pasture class. Wide variations in temperature, again for indigenous forest sites, were reflective of the large dataset and array of sites in this group.

The scrub class had few sites with pH data and those included were effected but acid mine drainage, hence the low pH.

Pasture sites had higher faecal coliforms, ammoniacal nitrogen, nitrate, and dissolved reactive phosphorus than any other class. Faecal coliforms were above the season median guideline for bathing, and dissolved reactive phosphorus was above its respective guideline (see 4.1.6). Presence of 'pollution sensitive' taxa, measured by the SQMCI, MCI, and percent EPT, was least for pasture sites. The lower quartile of both MCI and QMCI macroinvertebrate indice scores for this class fell into the 'poor' water quality range, although the median was indicative of mild pollution.

Scrub sites had the lowest taxonomic richness, bearing in mind poor replication within the spring source class. Of note was that turbidity did not differ significantly between classes, other than exotic forest, which had a higher upper quartile range than other land classes.

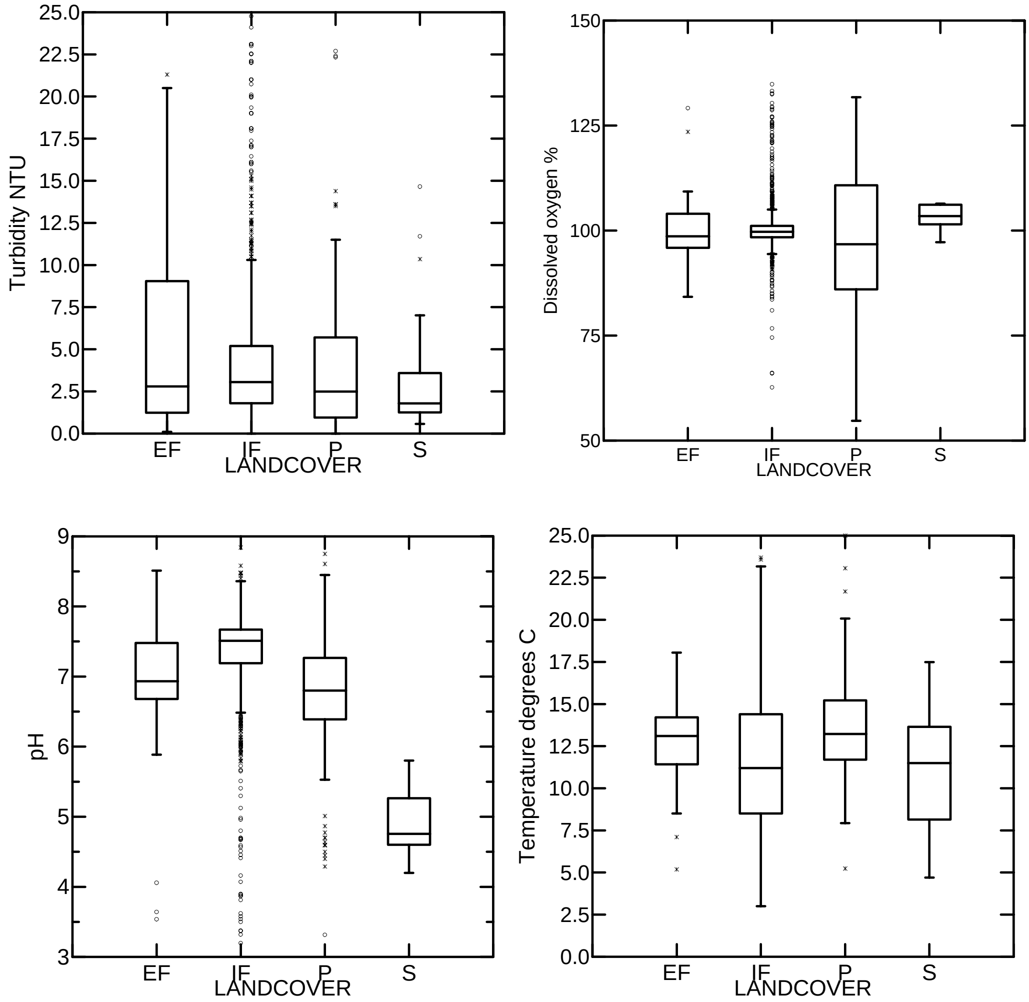


Figure 4.1.4 Distributions of turbidity, dissolved oxygen %, pH, and temperature between landcover classes EF (exotic forest), IF (indigenous forest), P (pasture), and S (scrub).

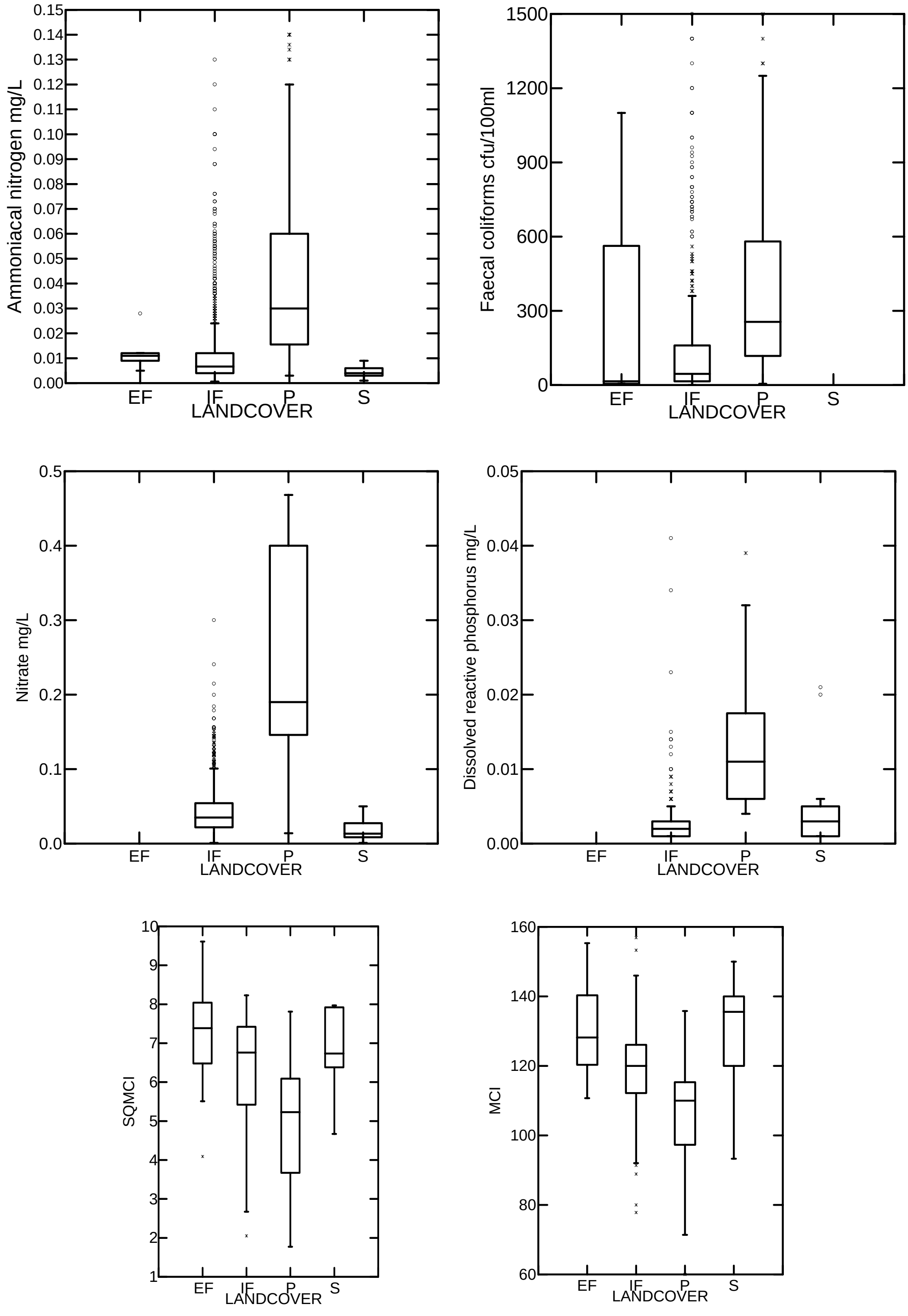


Figure 4.1.4 continued: Distributions of ammoniacal nitrogen, faecal coliforms, nitrate, dissolved reactive phosphorus, SQMCI, and MCI between landcover classes EF (exotic forest), IF (indigenous forest), P (pasture), and S (scrub).

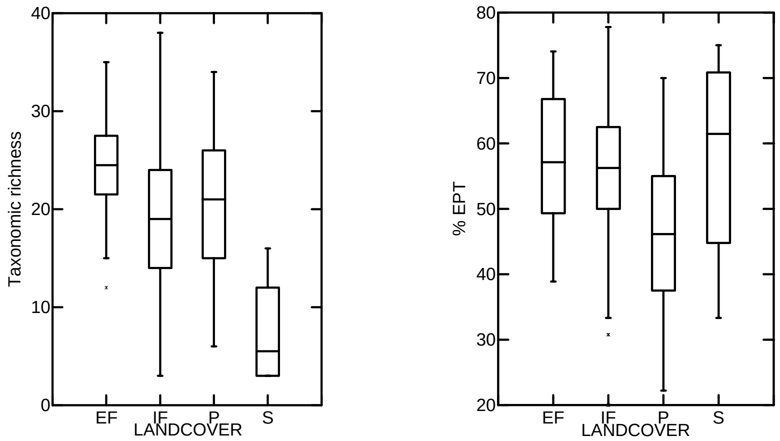


Figure 4.1.4 continued: Distributions of taxonomic richness and % EPT between landcover classes EF (exotic forest), IF (indigenous forest), P (pasture), and S (scrub).

4.1.5 Order

There were many significant differences between the three different stream orders (Figure 4.1.5). Low order sites had the lowest dissolved oxygen, pH, and highest conductivity, temperatures, ammoniacal nitrogen, faecal coliforms, nitrate, and sulphate. Nutrient differences were not pronounced between low order and mid order classes, and the mid order class had the highest turbidity. High order sites had higher water quality compared to their smaller counterparts; although mid order class dissolved oxygen was higher. This trend of better water quality with size continued to be consistent with the number of pollution sensitive taxa, although high order sites had the lowest taxonomic richness, on a par with the low order class.

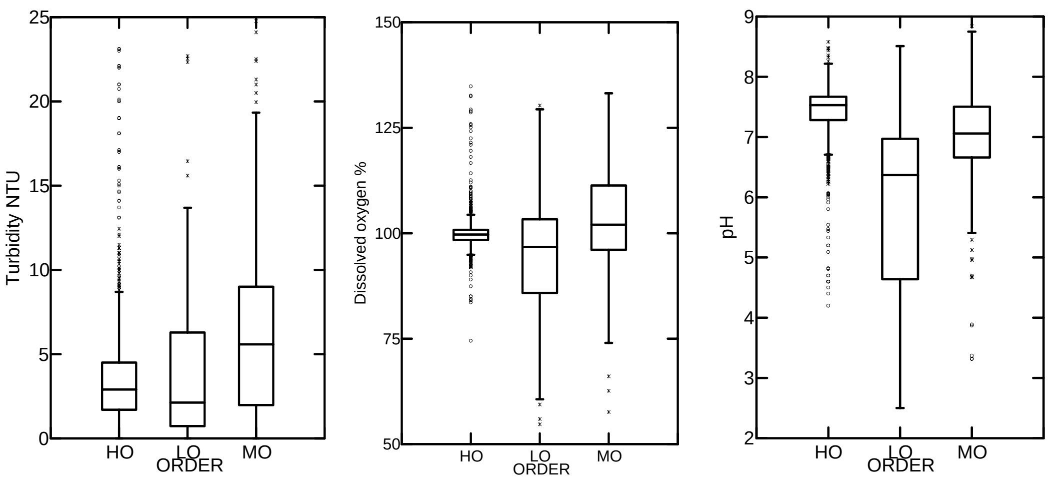


Figure 4.1.5 Distributions of turbidity, dissolved oxygen %, pH between order classes HO (high order), MO (mid order), and LO (low order).

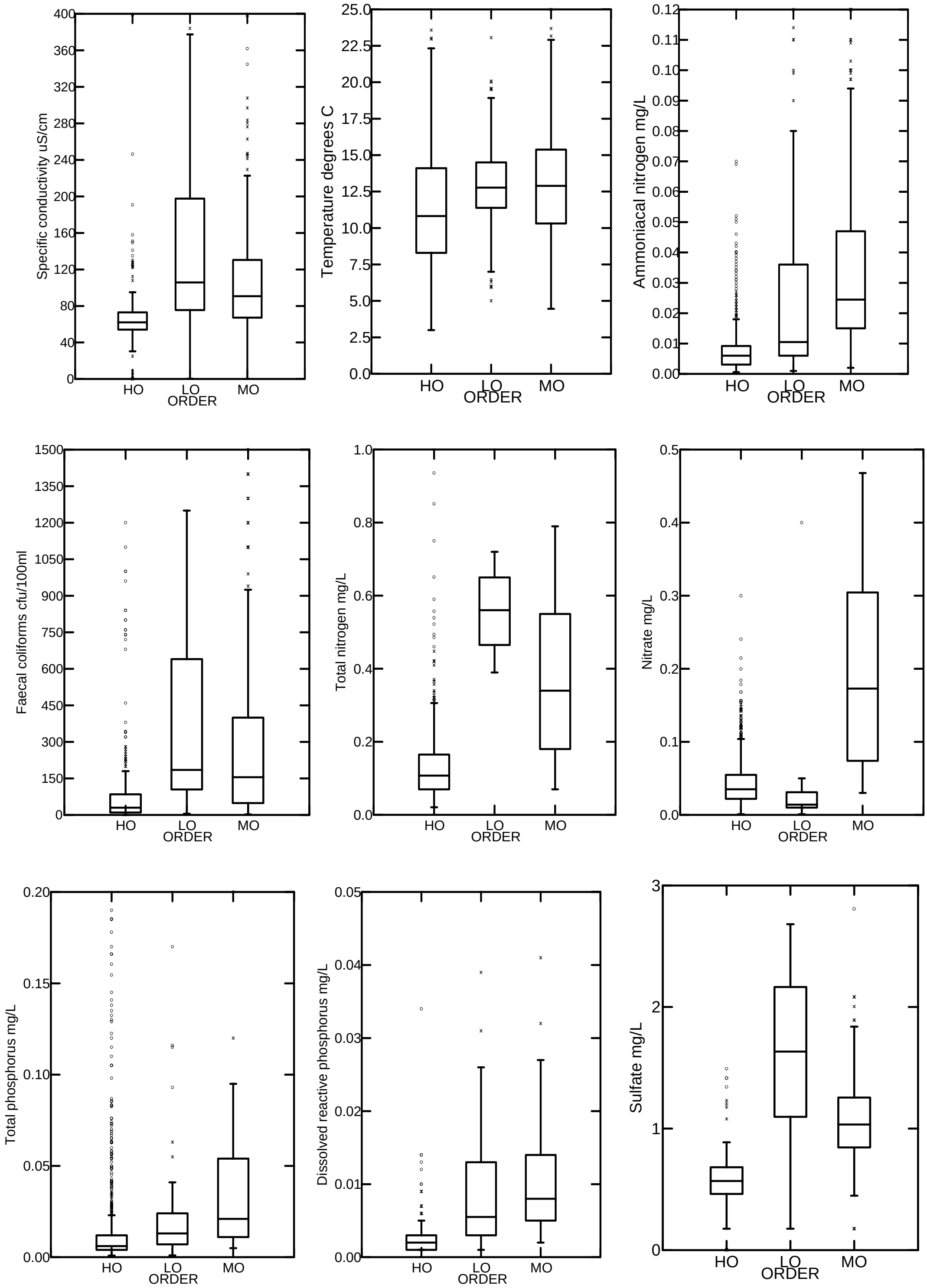


Figure 4.1.5 continued: Distributions of specific conductivity, temperature, ammoniacal nitrogen, faecal coliforms, total nitrogen, nitrate, total phosphorus, dissolved reactive phosphorus, and sulphate between order classes HO (high order), MO (mid order), and LO (low order).

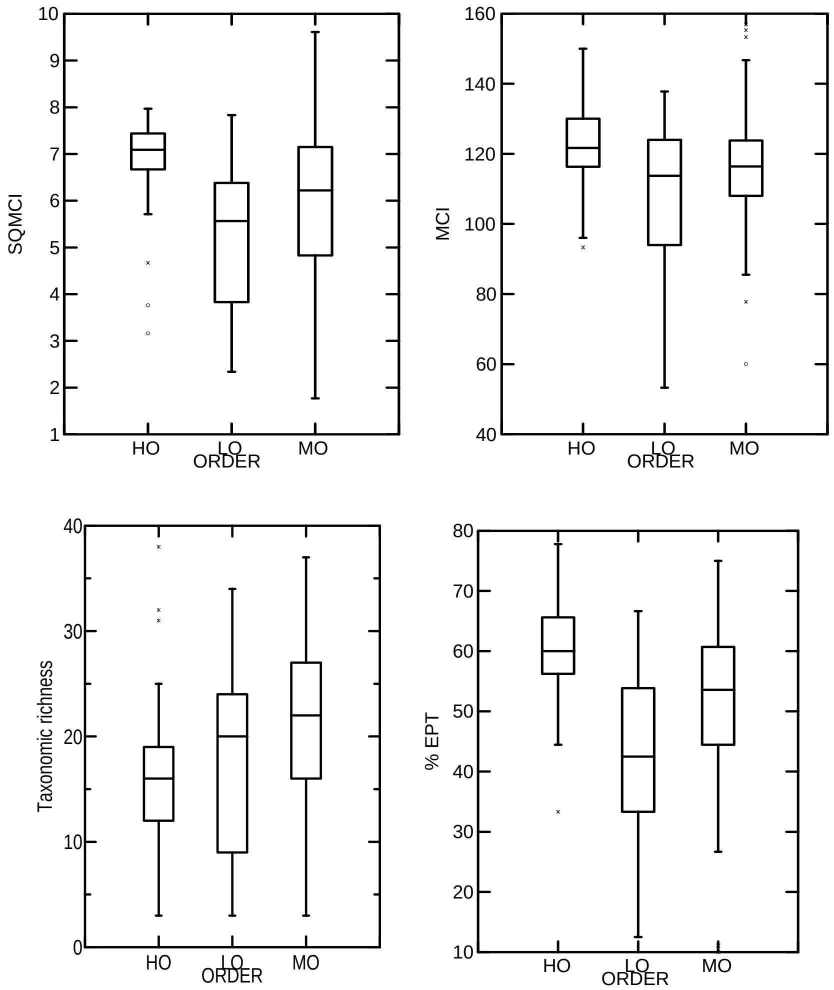


Figure 4.1.5 continued: Distributions of SQMCI, MCI, taxonomic richness, and % EPT between classes HO (high order), MO (mid order), and LO (low order).

4.2 Comparisons of individual sites, and comparison to guidelines

The range of all individual sites have been plotted per parameter so these individual sites can be compared. Keys to the graphs are presented in Tables 4.2.1 and 4.2.2. Please note that these keys are different. Table 4.2.1 is for all figures of physical, chemical, and bacteriological data, while Table 4.2.2 refers to those figures with macroinvertebrate indices and attributes.

4.2.1 Turbidity and clarity

Turbidity and clarity were closely correlated (Pearson $p < 0.001$). Averaged median turbidity across all classes was below turbidity guidelines (Figure 4.2.1). Most upper quartiles were beneath 10 NTU, with the exception of the soft sedimentary class median (6.4 NTU). Only 6 % of sites had medians beyond the guideline. Guideline figures used in the ANZECC (2000) guidelines are for unmodified or slightly disturbed ecosystems. It should be noted that some sites included in this analysis fall outside this description.

Ford at Taylorville-Blackball Rd, Garvey, Blackwater, Stillwater, Soldiers and Molloy Creek had turbidity medians above 10 NTU. These figures were indicative of elevated levels, but they lack the replication evident in those other sites with more outliers, therefore these medians would be more reliable with more data. It is worthwhile to compare individual clarity with turbidity here.

Clarity reciprocated turbidity at many sites, including some poorer ones e.g. Ford at Blackball-Taylorville Rd, Garvey, Soldiers and Blackwater Creeks. Other sites with reduced clarity were: Goat Ridge, Sawyers at Dixon Park, Baker, Seven Mile at Rapahoe, Bradshaw's, and both Burke Creek sites. Some sites were not correlated, reflecting possible changes in techniques used in the past. Figure 4.2.2, for clarity, has a trigger value guideline (protection for aquatic life) placed on it. There were 12 % sites that exceeded the clarity guideline. This value has been determined using un-modified to slightly modified systems, which many of these WCRC sites are not.

Natural tannins can also reduce water clarity, a characteristic typically of lowland native forest catchments. This was unlikely to play a significant contributing role in water clarity at levels around or below the guideline value, where suspended sediment is the main mechanism. Most catchments in this study that had reduced water clarity did not have major tannin colouration.

4.2.2 Percent dissolved oxygen saturation

Median values of dissolved oxygen are generally good, or at least acceptable. Blackwater Creek at farm 846, however, shows consistently low levels of dissolved oxygen with a median below 80 % saturation (Figure 4.2.3). Other sites that have lower quartile ranges dropping below 80 % saturation are Maori, Murray, Seven Mile (Rapahoe), Baker, and Bradshaw Creeks. These sites have slower flow, often abundant macrophyte and/or periphyton growth, and higher nutrient inputs, with few rapids to re-aerate the water.

4.2.3 Conductivity

As can be expected, sites where there was a marine influence, like Ngakawau and lower Karamea, had higher levels of conductivity than purely freshwater sites (Figure 4.2.4). Sites where mining activity occurred upstream also had noticeably higher conductivity e.g. Canal, Ford at T-B Rd, Garvey, Blackwater, and Soldiers Creeks, and Page Stream.

4.2.4 pH

pH for most sites generally lay within the accepted guidelines for natural waterways (Figure 4.2.5.). However, Canal, Garvey, Ngakawau, and Page Creeks showed consistently acidic conditions, particularly so in the case of Canal and Garvey Creeks. It is likely that discharges from coal mining areas, particularly for these sites, contributed to this acidity (James 2003).

4.2.5 Temperature

As a result of a wide variety of source and altitudes, median temperatures varied considerably (Figures 4.2.6). It remains that large streams and those that originate at higher elevations are colder than small, low elevation streams. Also, for streams with equivalent proportions, those with pasture catchments are warmer.

4.2.6 Ammoniacal nitrogen

Levels of ammoniacal nitrogen were generally low and below the ANZECC (2000) trigger value. But some pasture sites were noticeably higher (Figure 4.2.7), including Blackwater, Burke, Murray, Bradshaw's, and Baker Creeks, and the Orowaiti and Mawheraiti Rivers. Seven Mile Creek at Rapahoe received discharges from leaking septic tanks, and sewage treatment plants. Note that the Orowaiti River and Seven Mile catchments have soft sedimentary geology, helping raise ammonia for the soft sedimentary class.

Table 4.2.1 Site key for all individual site plots of physical, chemical, and bacteriological data.

Site code	Site	Site code	Site
1CXHHSIFHO	Ahaura Rv @ SH7 Br Ahaura	30CXHPLIFHO	Grey Rv @ Waipuna
2CXLKALIFHO	Arnold Rv @ Blairs Rd No. 2 Br	31CXMHSIFHO	Haast Rv @ Roaring Billy
3CXLKALIFHO	Arnold Rv @ Kotuku Fishing Access	32CXSALPMO	Harris Ck @ Mulvaney Rd
4CWLSSIFMO	Baker Ck @ Baker Ck Rd	33CXMHSIFHO	Hokitika Rv @ Gorge
5CWLSSPMO	Baker Ck @ Oparara Rd	34CXHHSIFHO	Hokitika Rv @ Kaniere Br
6CXSALPMO	Berry Ck @ N Brch Wanganui Flat Rd	35CXHPLIFHO	Inangahua Rv @ Landing
7CWLALPLO	Blackwater Ck @ Farm 846	36CXHPLIFHO	Karamea Rv @ Nat Pk boundary
8CXLALPMO	Bradshaws Ck @ Bradshaw Rd	37CXHPLIFHO	Karamea Rv @ Wharf Rd
10CXLALEFLO	Brennans Ck @ Blue Spur Rd	38CXHHSIFLO	Lankey Ck @ SH7
11CXHHSIFHO	Buller Rv @ Longford	39CXLALIFMO	Maori Gully Ck @ Chesterfield Rd
12CXHPLIFHO	Buller Rv @ Te Kuha	40CXLSSIFMO	Mawheraiti Rv @ SH7 Maimai
13CXLHSIFLO	Burke Ck @ Blairs Rd Br	41CXLALIFMO	Molloy Ck @ Rail Line
14CXLALPMO	Burkes Ck @ SH69	42CXSALPLO	Murray Ck @ Ford Rd S
15CXLSSIFLO	Cannel Ck @ Magazine Rd	43CXLALIFHO	Nelson Ck @ Swimming Hole Reserve
16CXLALPLO	Chinn Ck @ 500m d/s Douglas Rd Waitaha	44CXHSSSHO	Ngakawau Rv @ Ngakawau
17CXHHSIFMO	Crooked Rv @ Rotomanu-Bell Hill Rd	45CXLSSPMO	Orowaiti Rv @ Excelsior Rd
18CXHHSIFHO	Crooked Rv @ Te Kinga	46CXLSSIFMO	Orowaiti Rv @ Keoghans Rd
19CXSALPMO	Duck Ck @ Kokatahi-Kowhitirangi Rd Br	47CXLALPLO	Page Stm @ Chasm Ck Walkway
20CXLALSLO	Dunlop Ck @ SH6	48CXLSSIFMO	Sawyers Ck @ Dixon Pk
21CXLALSLO	Fergusson Ck @ Bold Head Rd	49CXLSSIFMO	Sawyers Ck @ u/s Boddytown No.1 Br
22CXLSSIFMO	Ford Ck @ Blackball-Taylorville Rd	50CXHSSIFMO	Seven Mile Ck @ 300m d/s Raleigh Ck
23CXHSSIFMO	Ford Ck @ u/s Confl Soldiers Ck	51CXHSSIFMO	Seven Mile Ck @ Dunollie 400m u/s Ox Pd
24CXHHSIFLO	Garvey Ck @ SH7	53CXLSSIFMO	Seven Mile Ck @ SH6 Rapahoe
25CXLSSEFLO	Goat Ridge Ck @ Waimea Forest	54CXHSSIFMO	Seven Mile Ck @ u/s Tillers Mine Ck
26CXLALIFHO	Grey Rv @ Dobson	55CXLSSIFMO	Soldiers Ck @ u/s Confl Ford Ck
27CXLALIFHO	Grey Rv @ Kiwi Pt	56CXHALPMO	Springlands Ck @ 250m u/s confl Rahu Rv
28CXHPLIFHO	Grey Rv @ SH7 Br Ikamatua	57CXLSSEFMO	Stillwater Ck @ Stillwater Rd
29CXLALIFHO	Grey Rv @ Taylorville Swimming Hole		

Table 4.2.2 Site key for all individual site plots of macroinvertebrate attributes and indices.

Site code	Site name	Site code	Site name
1CXLALEFLO	Brennans Ck@Blue Spur Rd	20CXHSSIFMO	Seven @ 400m u/sDunollie Ox Pd
2CXLSSEFLO	Goat Ridge Ck@Waimea Forest	21CXHSSIFMO	Seven @ Tillers Mine Ck
3CXLSSEFMO	Stillwater Ck@Stillwater Rd.	22CXHSSIFMO	Seven @Dunollie d/s Ox Pd
4CXLALIFMO	Molloy Ck@50m d/s Railway	23CXLSSIFMO	Soldiers Ck.@u/s Ford Ck.
5CXLALIFHO	Nelson Ck.@swimming hole	24CXLALPMO	Burkes Ck@SH69
6CXLHSIFLO	Burke Ck.@Blairs Rd Br.	25CXLALPMO	Duck Ck@Kokatahi
7CXLHSIFLO	Burke Ck.@d/s Blair Rd.	26CXLALPMO	Harris Ck.@Kaniere
8CXHHSIFMO	Crooked R.@Rotomanu-	27CXLALPMO	Harris Ck.@Mulvaney Rd.
9CXHHSIFHO	Crooked Rv@Te Kinga	28CXLALPMO	Lawyer Ck.@Municipal Rd.
10CXHHSIFLO	Garvey Ck@SH7	29CXLALPLO	Murray Ck.@Ford Rd. S
11CXHHSIFLO	Lankey Ck@SH7	30CXLALPLO	Murray Ck.@Municipal Rd.
12CXHPLIFHO	Grey Rv@Ngahere	31CXLALPMO	Orangapuku R.@300 m d/s
13CXHPLIFHO	Inanguhua Rv.@landing	32CXLALPLO	Page Stm.@Chasm Ck.
14CWLSSIFMO	Baker Ck.@Baker Ck Rd.	33CXHALPMO	Springlands Ck.@Hunters Rd.
15aCXLSSIFMO	Ford Ck@Blackball-Taylorville Rd.	34CXSALPMO	Vickers Ck.@Whataroa N base
15bCXHSSIFMO	Ford Ck@u/s Soldiers Ck.	35CWLSSPMO	Baker Ck.@Oparara Rd.
16CXLSSIFMO	Mawheraiti Rv.@Maimai Valley Rd Ford	36CWLSSPMO	Baker Ck@Quinlans Br SH67
17CXLSSIFMO	Mawheraiti Rv@SH7 Maimai	37CXLSSPMO	Orowaiti @Excelsior Rd
18CXLSSIFMO	Orowaiti @Keoghans Rd	38CXHSSSHO	Ngakawau Rv@Ngakawau
19CXLSSIFMO	Sawyers Ck.@Dixon Park	39CXHSSSLO	Rapid Ck.@SH67

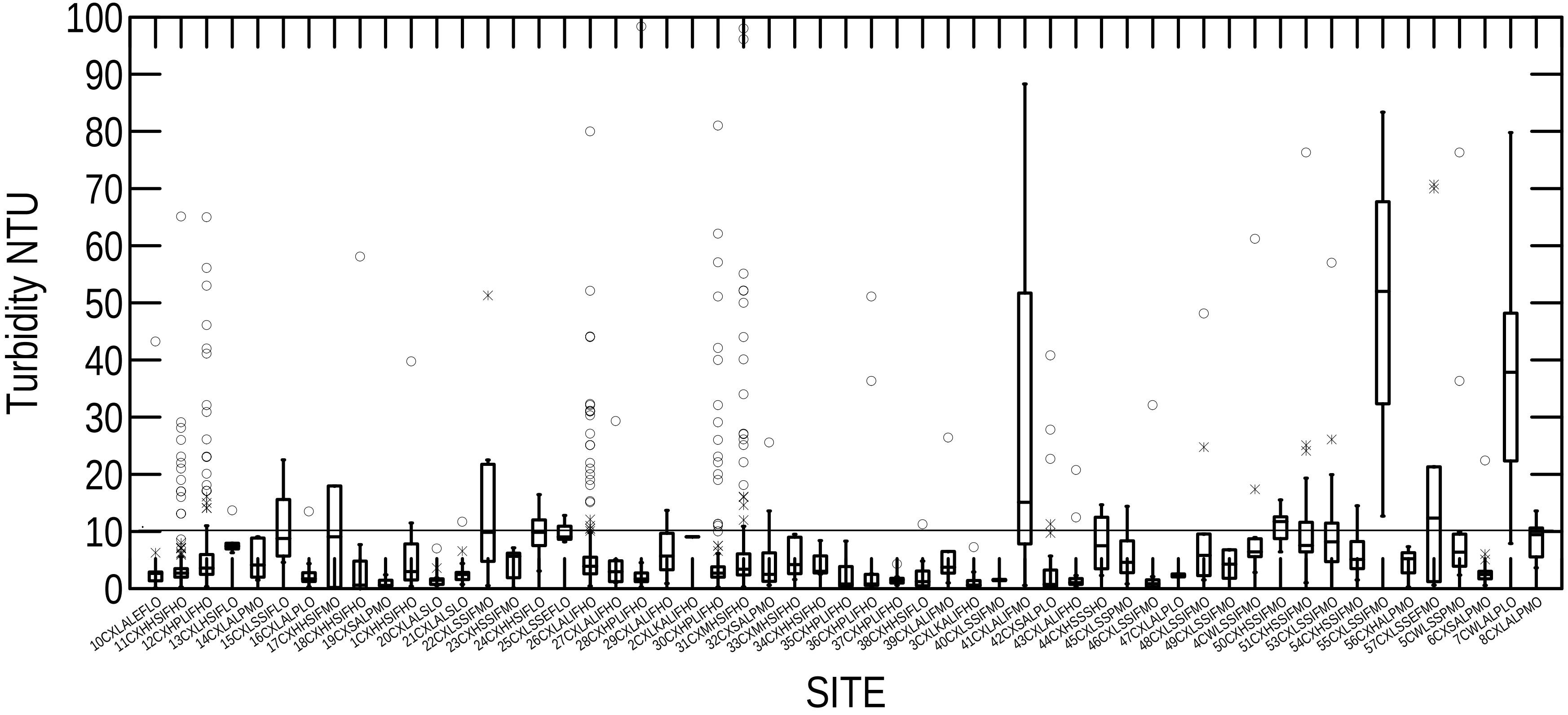


Figure 4.2.1 Turbidity for individual sites. For site identification, the first number refers to the site, with the following letters signifying the combined REC as far down to river order, for that site. Refer to Table 4.4 for a site key. The guideline is shown (Quinn et al. 1992).

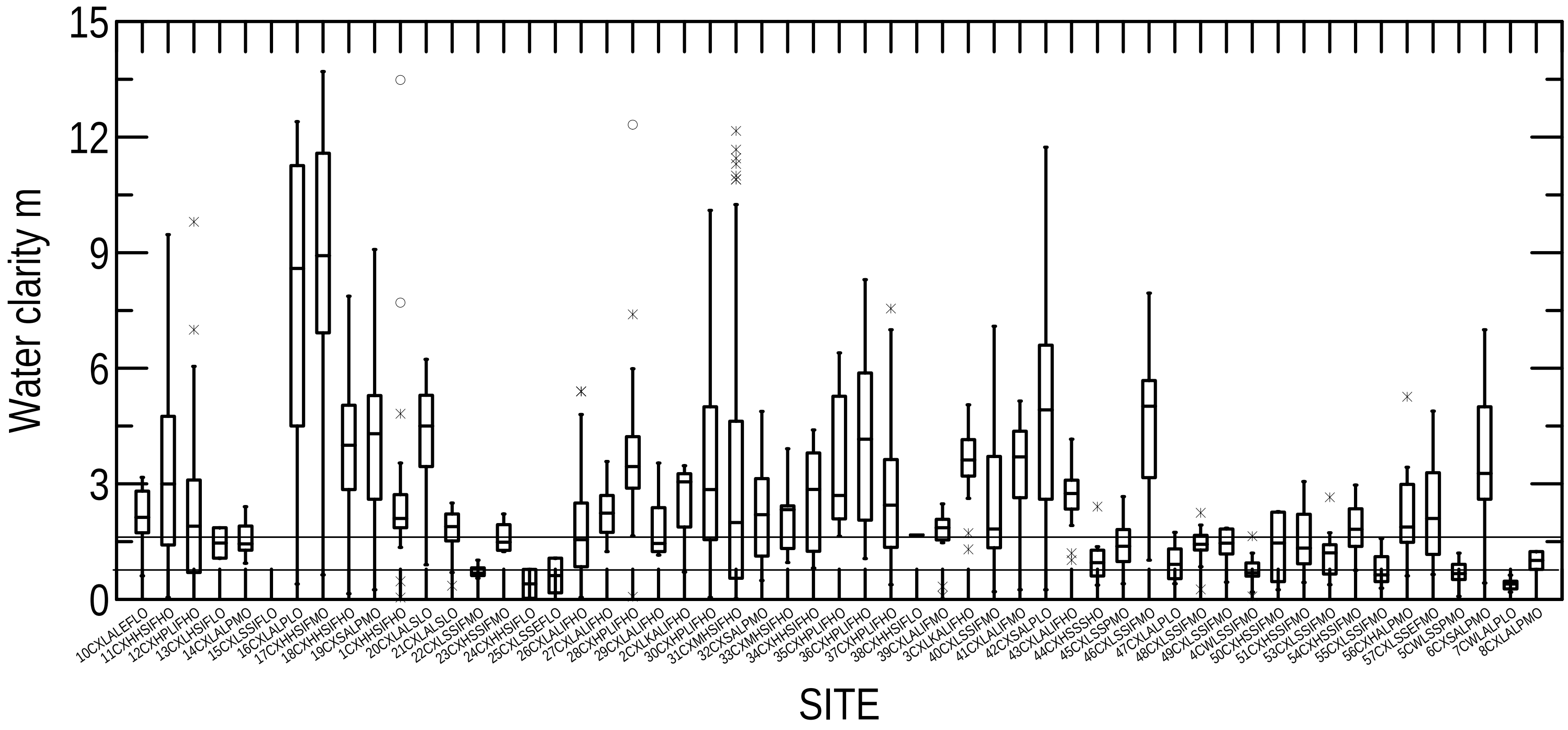


Figure 4.2.2 *Water clarity for individual sites. For site identification, the first number refers to the site, with the following letters signifying the combined REC as far down to river order, for that site. Refer to Table 4.4 for a site key. The upper line represents the MfE (2000) guideline for contact recreation (1.6 m), while the lower represents the ANZECC (2000) trigger value for protection of lowland aquatic ecosystems (0.8 m).*

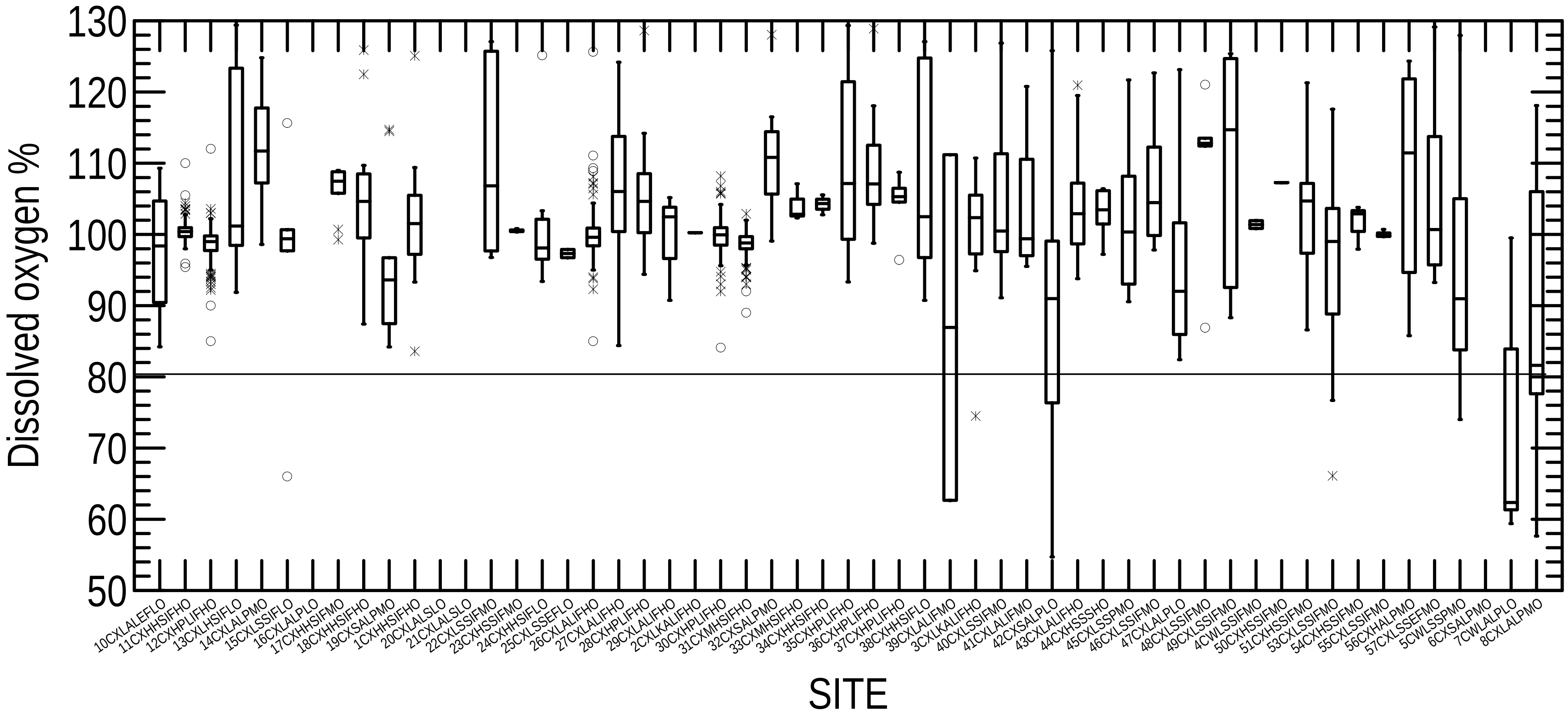


Figure 4.2.3 Percentage dissolved oxygen saturation for individual sites. For site identification, the first number refers to the site, with the following letters signifying the combined REC as far down to river order, for that site. Refer to Table 4.4 for a site key. The line represents the percentage dissolved oxygen saturation limit stipulated by the RMA.

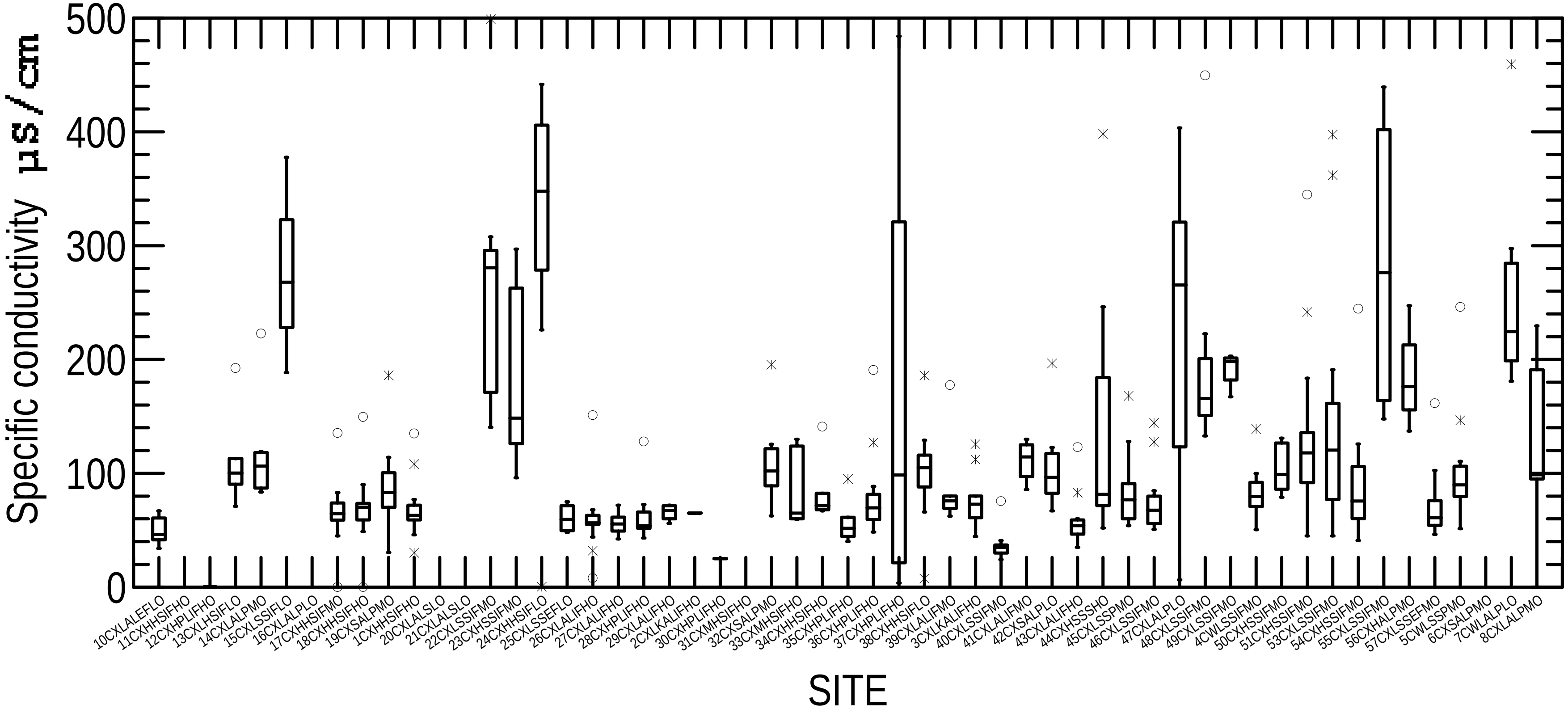


Figure 4.2.4 *Specific conductivity for individual sites. For site identification, the first number refers to the site, with the following letters signifying the combined REC as far down to river order, for that site. Refer to Table 4.4 for a site key.*

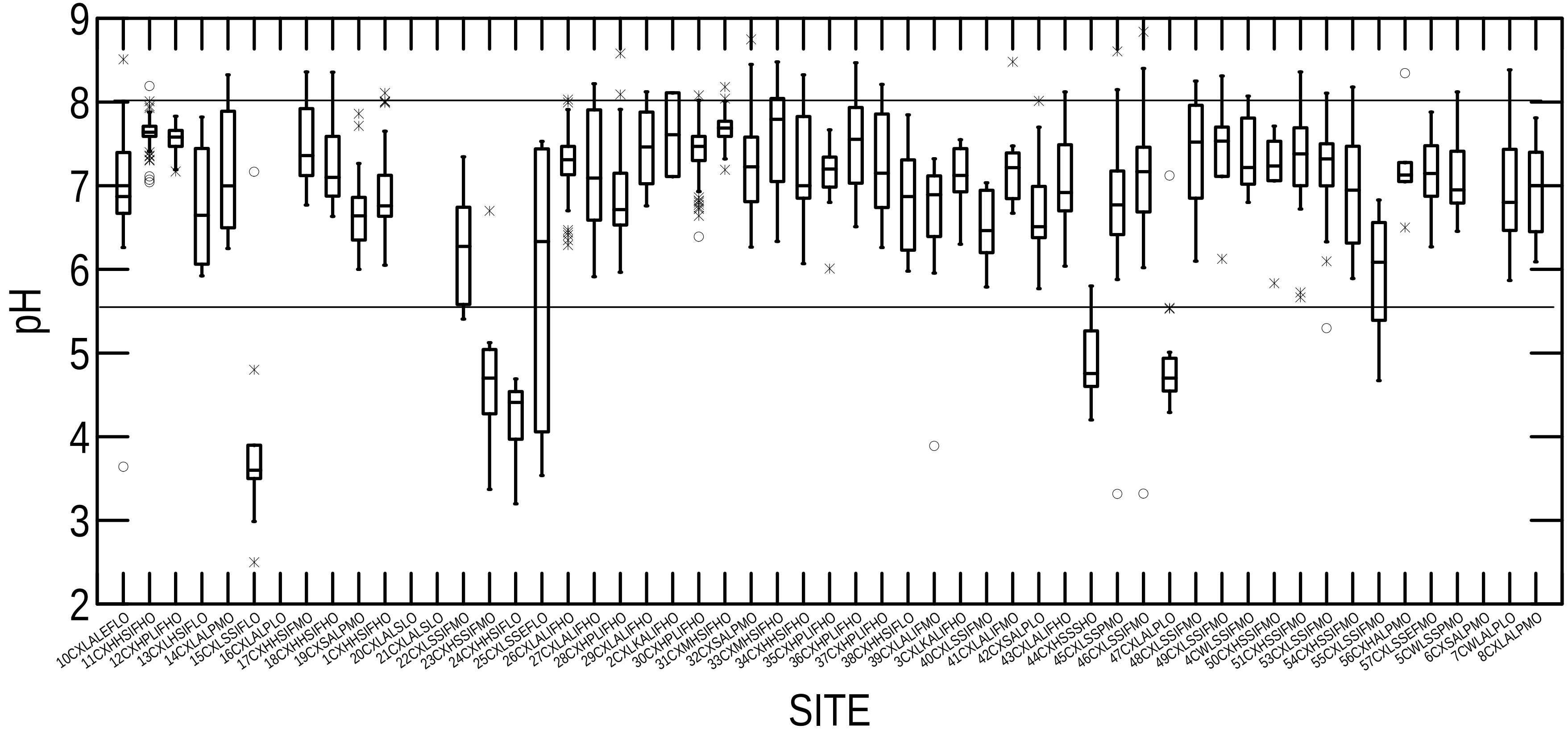


Figure 4.2.5 *pH for individual sites. For site identification, the first number refers to the site, with the following letters signifying the combined REC as far down to river order, for that site. Refer to Table 4.4 for a site key. The guideline range of acceptable pH's suggested by this document is located within the two horizontal lines.*

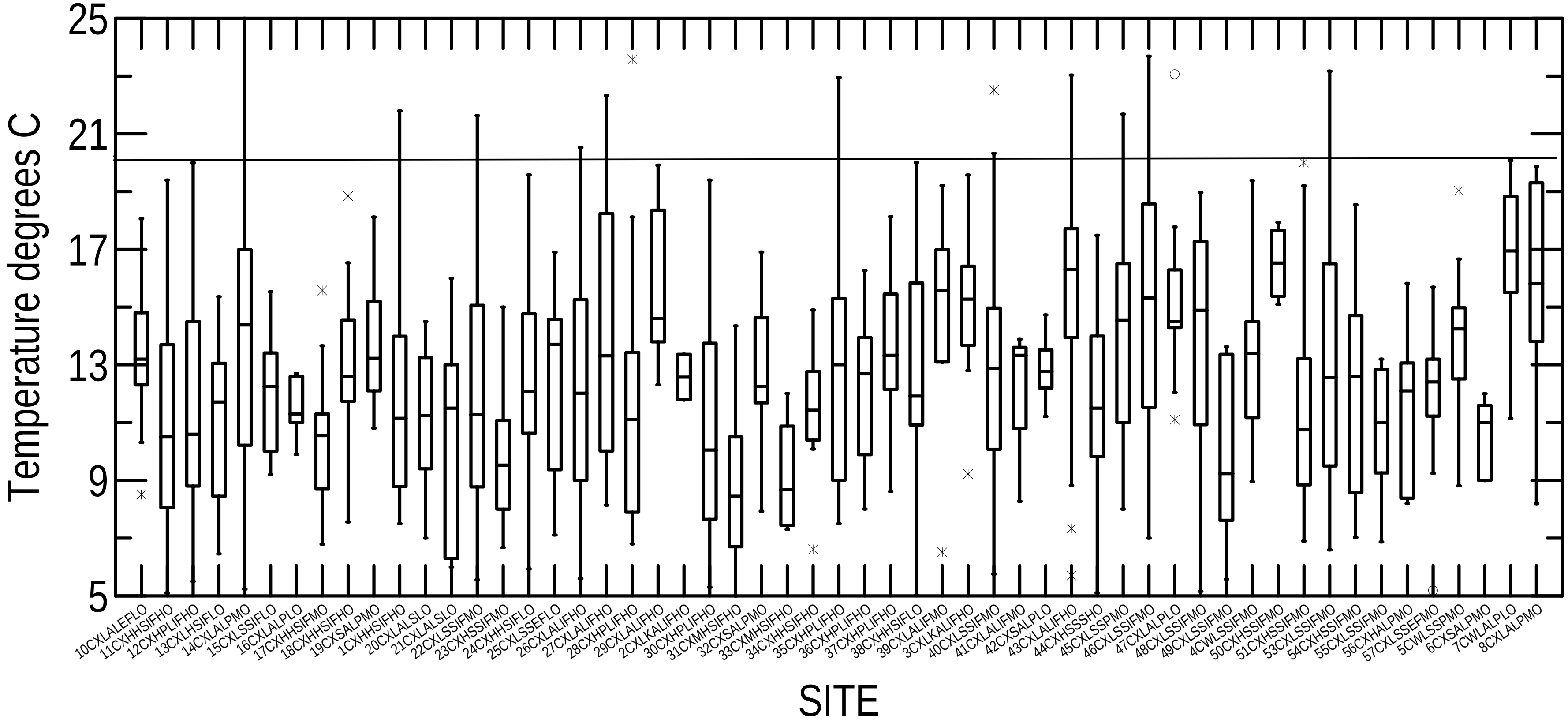


Figure 4.2.6 *Temperature for individual sites. For site identification, the first number refers to the site, with the following letters signifying the combined REC as far down to river order, for that site. Refer to Table 4.4 for a site key. The upper limit for trout spawning is shown.*

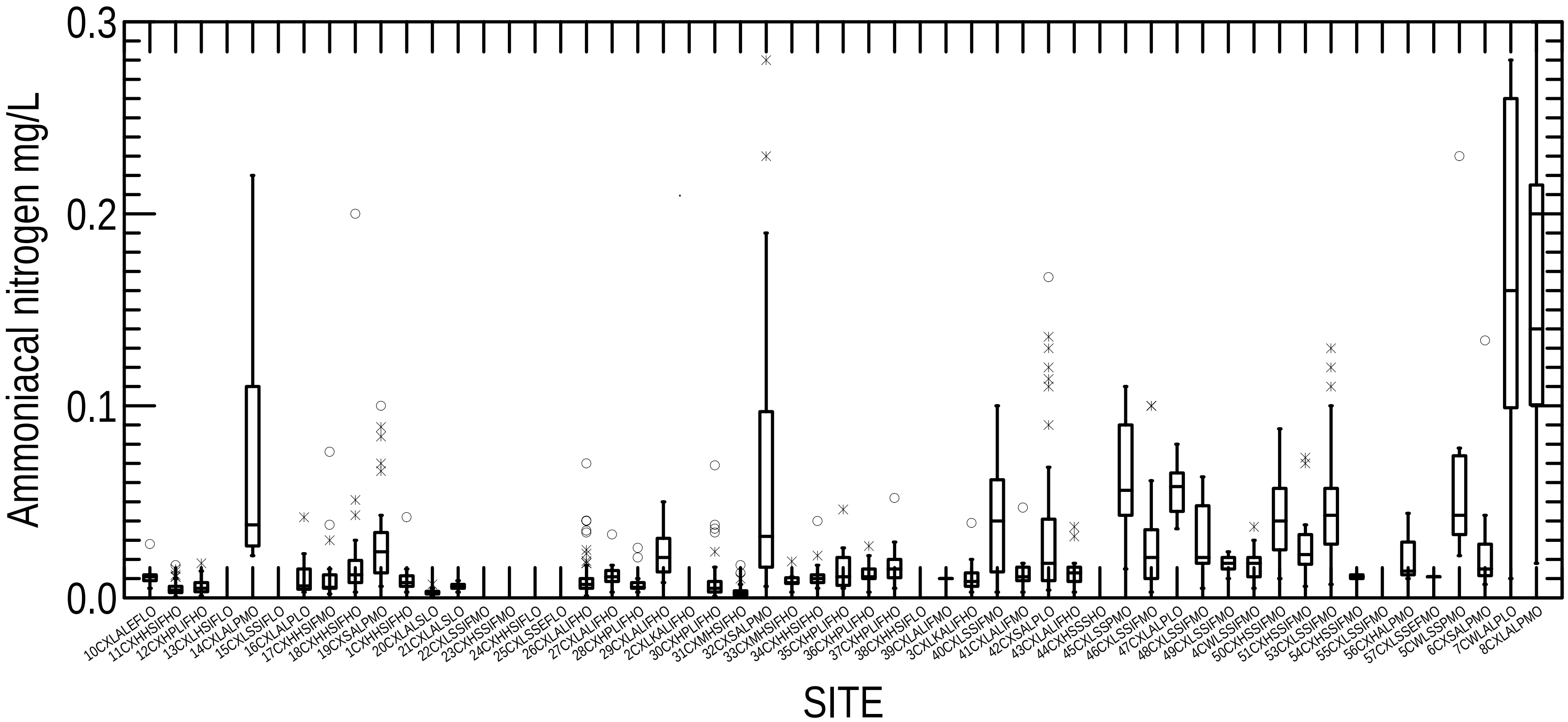


Figure 4.2.7 *Ammoniacal nitrogen for individual sites. For site identification, the first number refers to the site, with the following letters signifying the combined REC as far down to river order, for that site. Refer to Table 4.4 for a site key.*

3.2.7 Faecal coliforms

Guidelines for shellfish gathering recommend that the mean faecal coliform content shall not exceed 14 cfu/100 ml and not more than 10 % should exceed 43 cfu/100 ml (James 2003). These figures are exceeded on many occasions at all sites where shellfish are likely to be collected (Figure 4.2.8a).

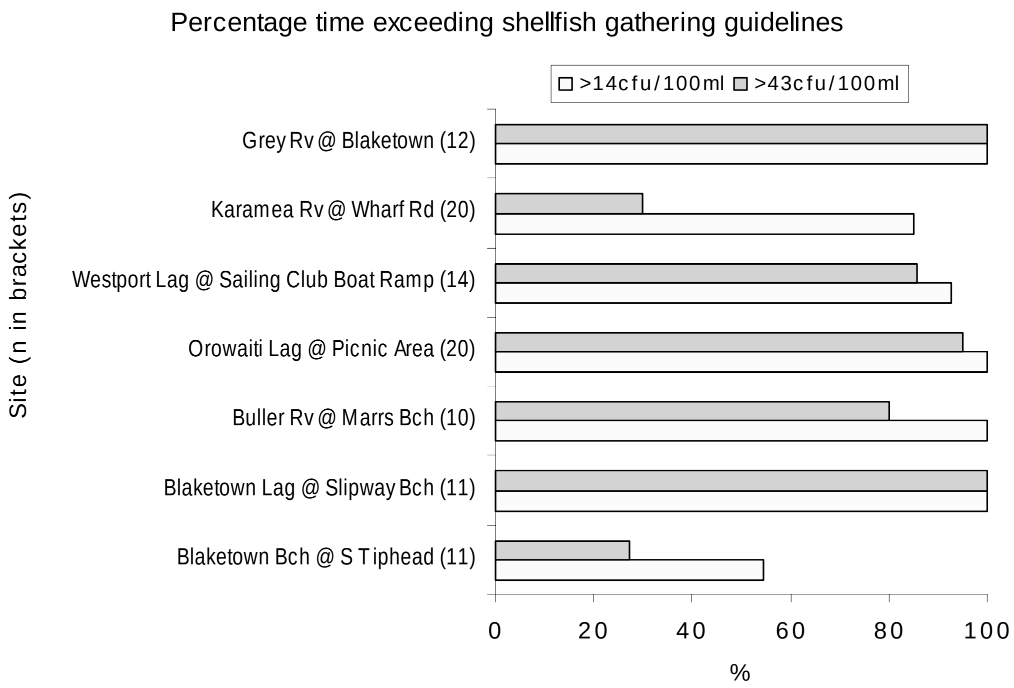


Figure 4.2.8a Percentage time faecal coliforms exceeded the recommended guideline for shellfish. Guidelines for shellfish gathering recommend that the mean faecal coliform content shall not exceed 14 cfu/100 ml and not more than 10 % should exceed 43 cfu/100 ml (James 2003).

Wide ranges of faecal coliform levels were evident amongst sites with intensive agriculture in their catchment (Figure 4.2.8b). Pastoral areas had some very high levels on occasion. None of these sites had median levels that were beyond stock (ANZECC 1992) and secondary contact recreation (MfE1999) guidelines, although a number were above the ANZECC (2000) recommended level for contact recreation and stock drinking, including the following sites: Baker, Brennan's, Burke's, Bradshaw's Duck, Blackwater, Molloy, Murray at Ford Rd South, Seven Mile (Rapahoe and 300 m d/s Raleigh), and Harris Creeks, Mawheraiti and Orowaiti Rivers, and Sawyers Creek. The lower site at Dixon Park probably has more faecal coliforms due to reticulated sewage pipes leaking into the stormwater drainage system.

4.2.8 Total nitrogen and nitrate

All site medians were well below ANZECC (2000) guidelines, with nearly all 80^{th} percentiles below it (Figures 4.2.9 and 4.2.10). The majority of sites with nitrogen measurements had alluvial or hard (hard sedimentary and PL) geologies. Spring creeks (Murray, Harris, and Duck) seemed consistently high, all of which have smaller, intensively farmed catchments. The Crooked River at Te Kinga, and the Inungahua at Landing were higher than expected for indigenous forest class catchments, although the Crooked River has intensively farmed land in its lower reaches. Again, treated sewage effluent elevated Seven Mile Creek (Rapahoe and 300 m d/s Raleigh Ck).

4.2.9 Total phosphorus and dissolved reactive phosphorus

Fewer sites had sufficient data to plot meaningful graphs for total (TP) or dissolved reactive (DRP) phosphorus (Figures 4.2.11 and 4.2.12). It must be noted that data replication between and within sites was limited for nutrient data for all except Haast, and the Grey River sites. Interestingly, these sites had outliers beyond the guideline of 0.033 mg/L for TP, despite being indigenous forest rivers. This may result from flood samples.

Seven mile Creek had noticeably high levels of both phosphorus forms, attributable to sewage treatment pond discharges. Harris, Murray at Ford Rd South, and Chinn Creeks had DRP levels above the guideline. The Crooked River at Te Kinga had high DRP despite having a predominantly indigenous forest catchment.

4.2.10 Sulfate

Sites with high sulfate typically had mining activity in their catchments. This included Garvey, Seven Mile (all), Ford at T-B Rd, and Canal Creeks, Page Stream, and the Ngakawau River – all with soft sedimentary geology (Figure 4.2.13). There was no concern of sulfate negatively impacting water potability at these sites.

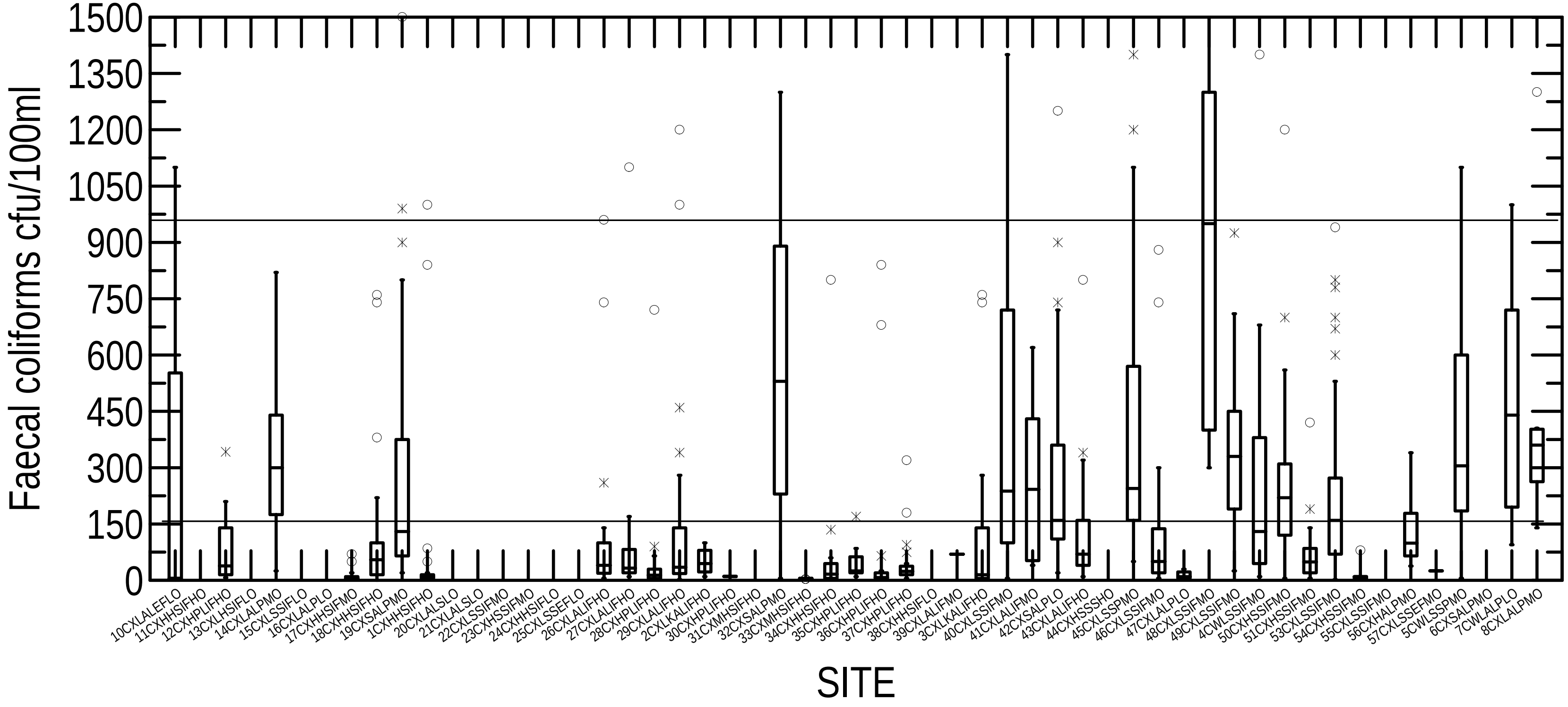


Figure 4.2.8b Faecal coliforms for individual sites. For site identification, the first number refers to the site, with the following letters signifying the combined REC as far down to river order, for that site. Refer to Table 4.4 for a site key. The lower line represents the median faecal coliform limit for bathing, and the upper represents the guideline for secondary contact recreation (MfE 1999) and stock consumption (ANZECC 1992).

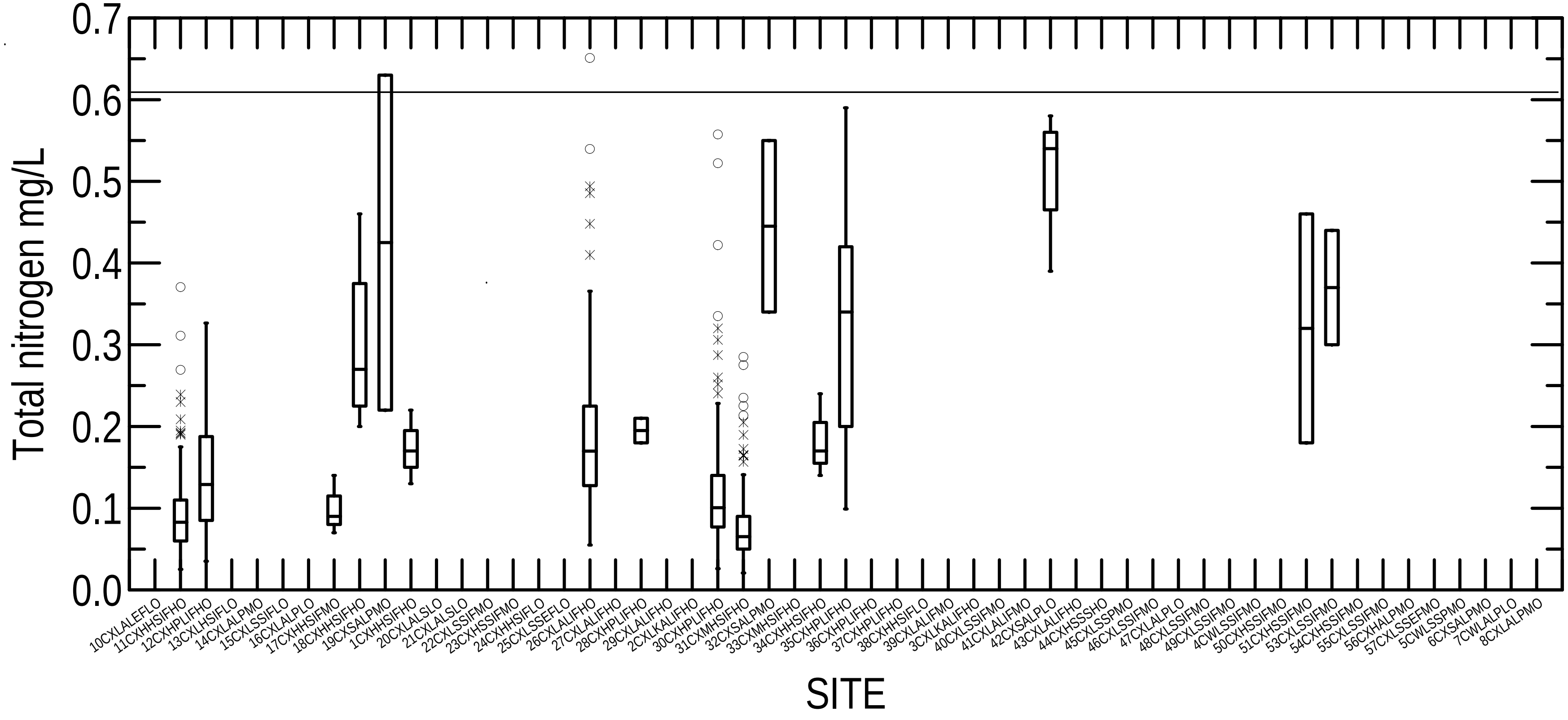


Figure 4.2.9 *Total nitrogen for individual sites. For site identification, the first number refers to the site, with the following letters signifying the combined REC as far down to river order, for that site. Refer to Table 4.4 for a site key. The Total Nitrogen guideline (ANZECC 2000) is shown.*

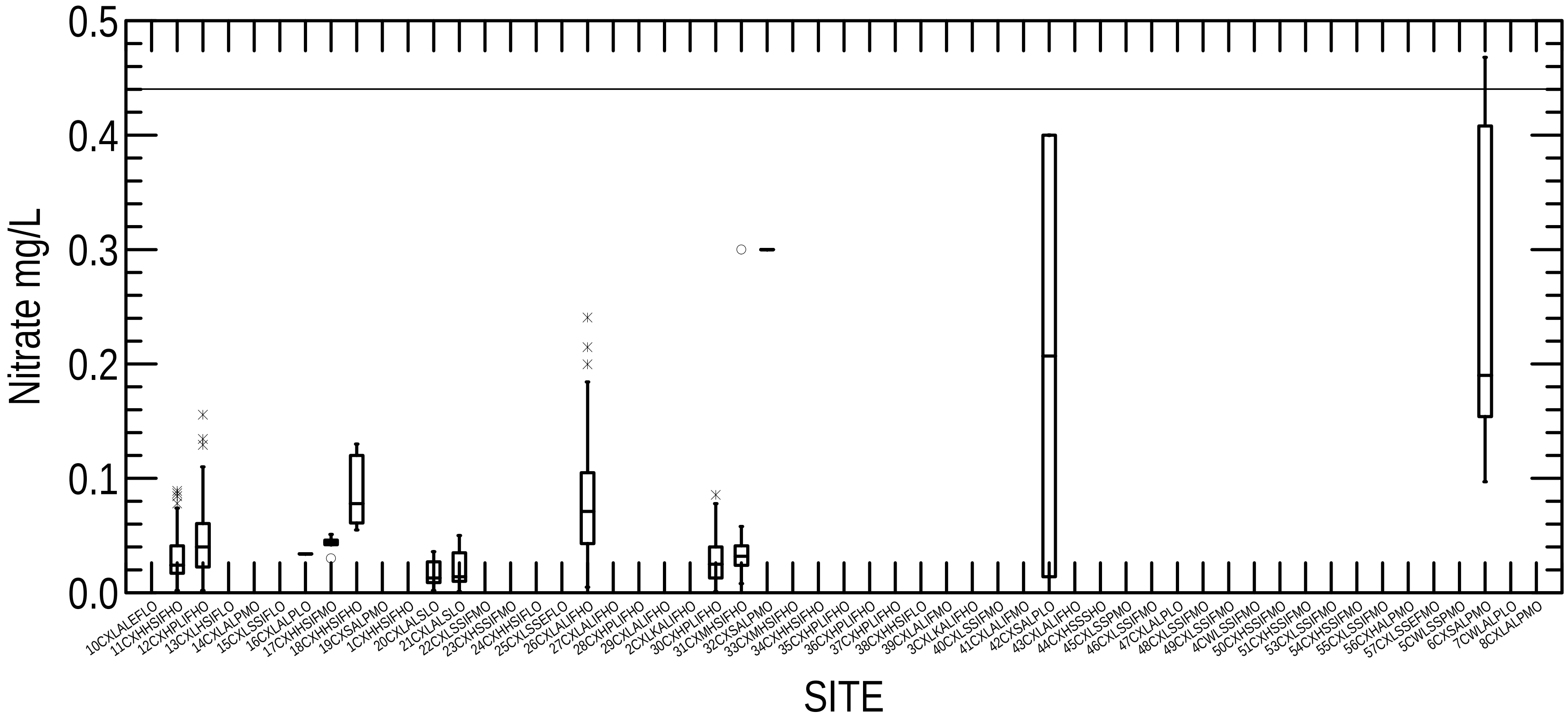


Figure 4.2.10 Nitrate for individual sites. For site identification, the first number refers to the site, with the following letters signifying the combined REC as far down to river order, for that site. Refer to Table 4.4 for a site key. The Nitrate guideline (ANZECC 2000) is shown.

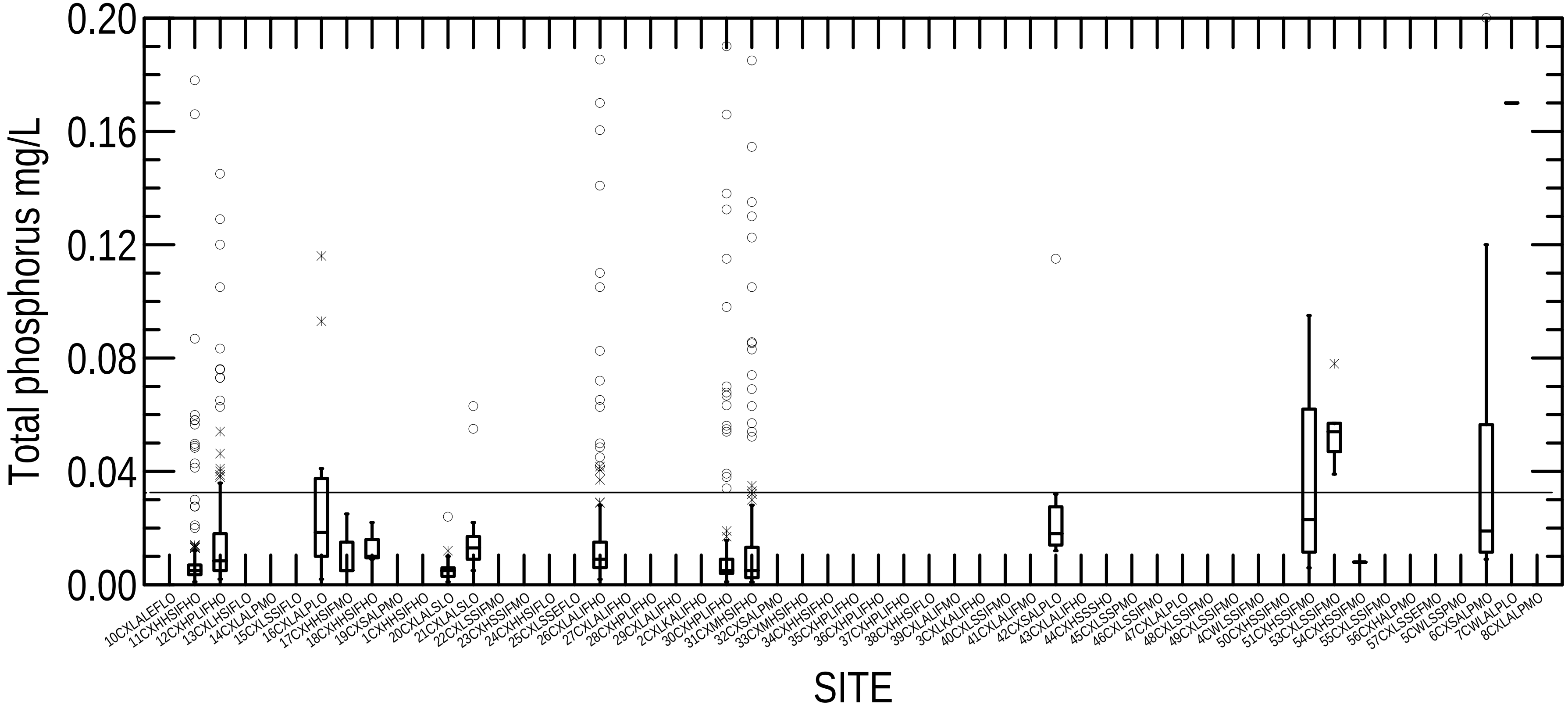


Figure 4.2.11 Total phosphorus for individual sites. For site identification, the first number refers to the site, with the following letters signifying the combined REC as far down to river order, for that site. Refer to Table 4.4 for a site key. The Total Phosphorus guideline (ANZECC 2000) is shown.

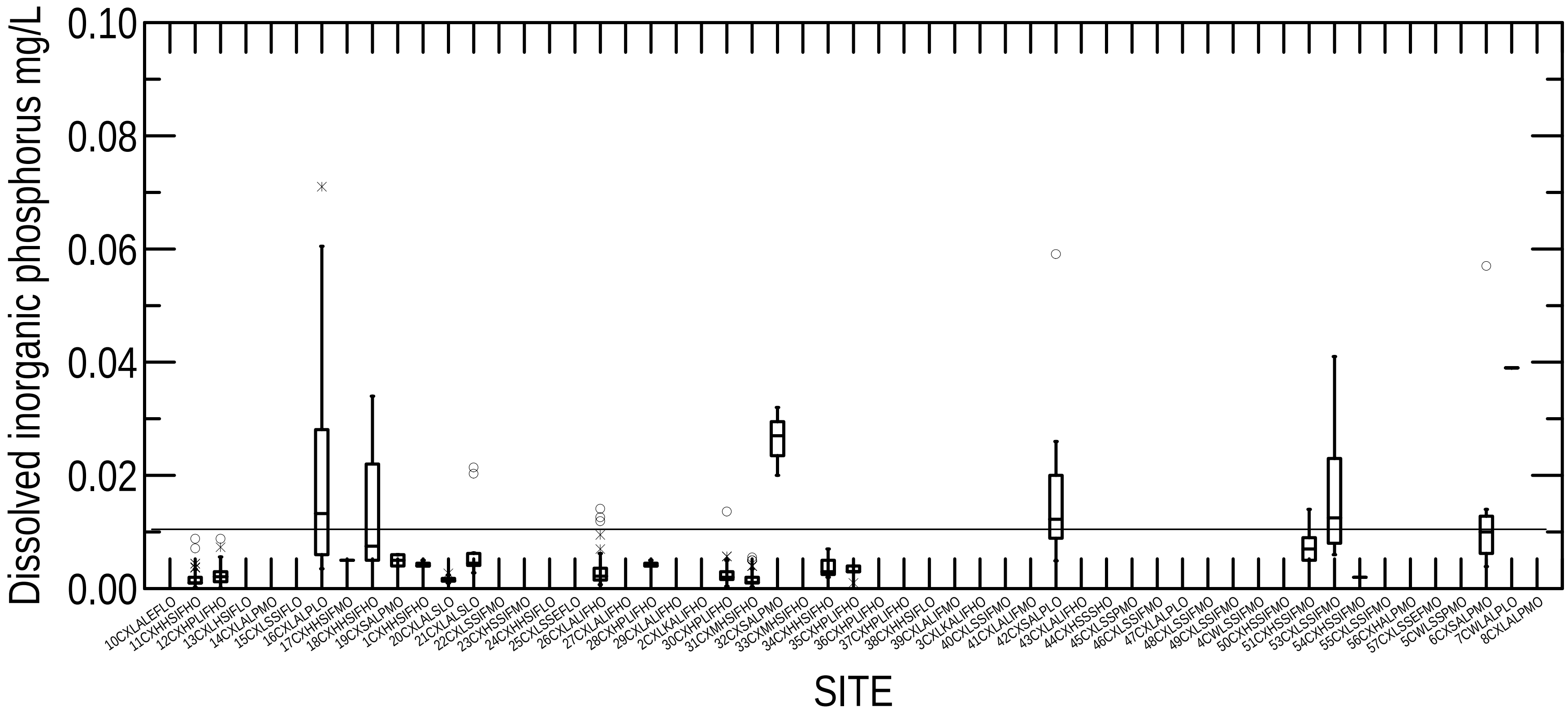


Figure 4.2.12 Dissolved reactive phosphorus for individual sites. For site identification, the first number refers to the site, with the following letters signifying the combined REC as far down to river order, for that site. Refer to Table 4.4 for a site key. The Dissolved Reactive Phosphorus guideline (ANZECC 2000) is shown.

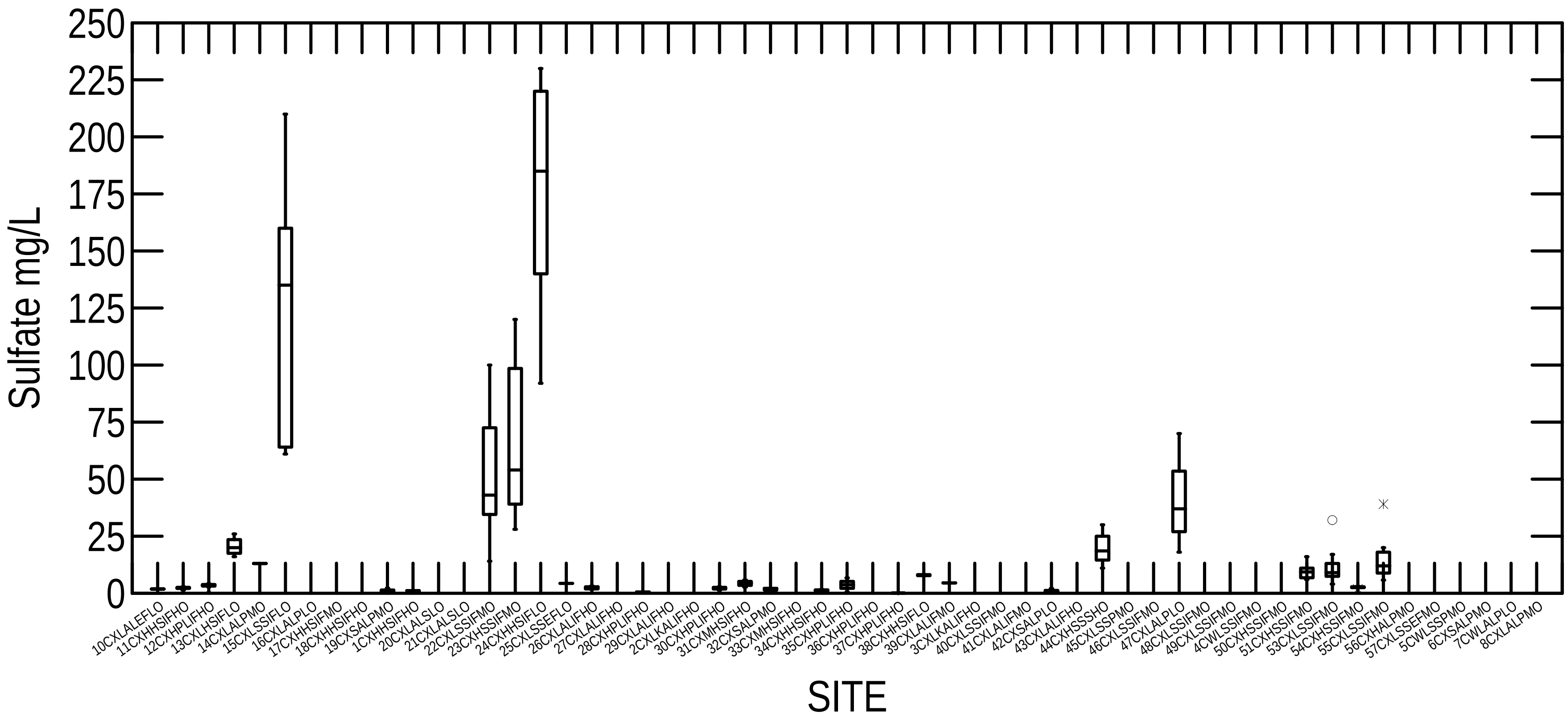


Figure 4.2.13 Sulfate for individual sites. For site identification, the first number refers to the site, with the following letters signifying the combined REC as far down to river order, for that site. Refer to Table 4.4 for a site key.

4.2.11 Macroinvertebrate community characteristics

Macroinvertebrate community indices and semi-quantitative macroinvertebrate indices are commonly used to assess water quality, categorising waters as poor, fair, good or excellent according to the scores achieved. Based on these the quality of the waters surveyed varied, but the majority were either considered to be subject to 'mild pollution', or 'clean', with very few instances of probable severe pollution (Figures 4.2.14 to 4.2.17).

Baker Creek at Oparara Rd joins Garvey Creek and Page stream in the poor water quality SQMCI scores (Figure 4.2.14 and 4.2.16), probably as a result of agricultural runoff. Sites with probable moderate pollution were Seven Mile 300m d/s Raleigh Ck, Burke's Creek, Murray Creek at Municipal Rd, Orowaiti at Excelsior Rd, and both Bradshaw Creek sites. Sites with 'possible mild pollution' included Seven Mile at Dunollie 400m u/s ox ponds, Lawyer Creek, Murray Creek at Ford Rd South, and Vickers Creek. Habitat and salinity may affect the lower Bradshaw Creek site, and Orowaiti River at Excelsior Road, and this means their impoverished macroinvertebrate communities are not necessarily attributable to poor water quality.

The overall standard of water quality using MCI compared with SQMCI was higher (Figures 4.2.15 and 4.2.17). The MCI works only on the presence/absence of species, so it was likely that the abundances of pollution sensitive taxa to those more tolerant were lower. Only Page Stream scored in the 'poor' MCI water quality category, with this site also having low taxonomic richness (Figure 4.2.18) and percentage EPT taxa (Figure 4.2.19). Garvey Creek remained impoverished by standards of taxonomic richness and percent EPT. The following sites increased their water quality rating with MCI: Orowaiti River at Excelsior Rd, Garvey, Sawyer's, Baker, Burke's, and both Burke Creeks. It is possible that certain habitat or environmental factors suit 'pollution tolerant' taxa, and their abundance may be because of this, rather than only poor water quality. This situation is likely where, despite significant diversity and abundance, there are also many pollution tolerant taxa. This may also occur in mildly impacted sites that have some of the conditions favourable for pollution tolerant taxa, but insufficiently poor water quality to exclude more sensitive species. The Ngakawau River had low taxonomic richness, but what species were present must have been sensitive. More data would be needed to test this theory.

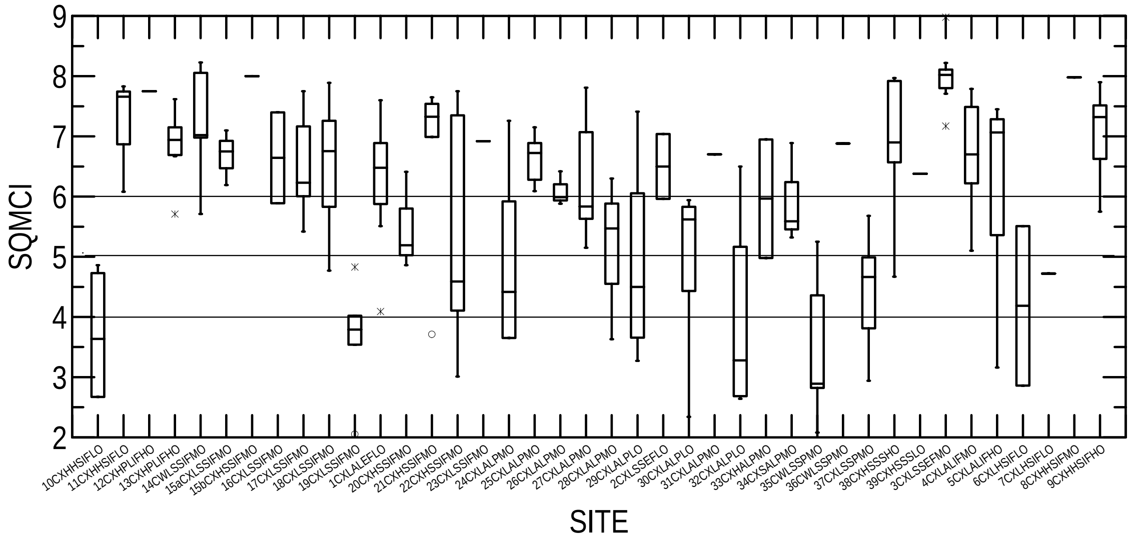


Figure 4.2.14 SQMCI for individual sites. For site identification, the first number refers to the site, with the following letters signifying the combined REC as far down to river order, for that site. Refer to Table 4.5 for a site key. Values below 4 indicate poor water quality; 4-5 is probable moderate pollution; 5-6 is doubtful quality; and over 6 is considered equivalent to pristine.

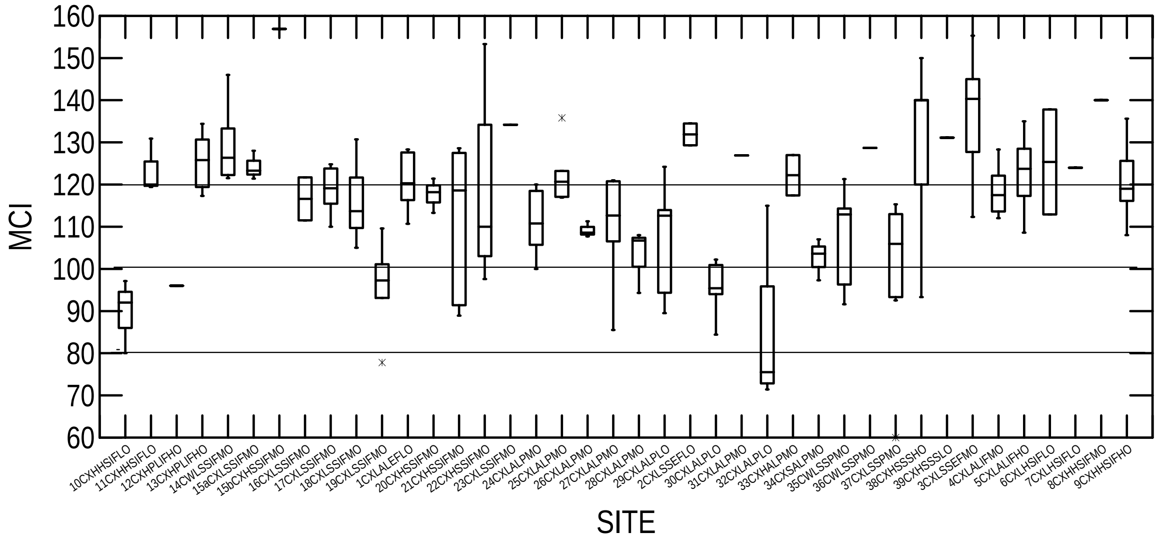


Figure 4.2.15 MCI for individual sites. For site identification, the first number refers to the site, with the following letters signifying the combined REC as far down to river order, for that site. Refer to Table 4.5 for a site key. Values below 80 indicate poor water quality; 80-90 is probable moderate pollution; 90-100 is doubtful quality; and over 100 is considered equivalent to pristine.

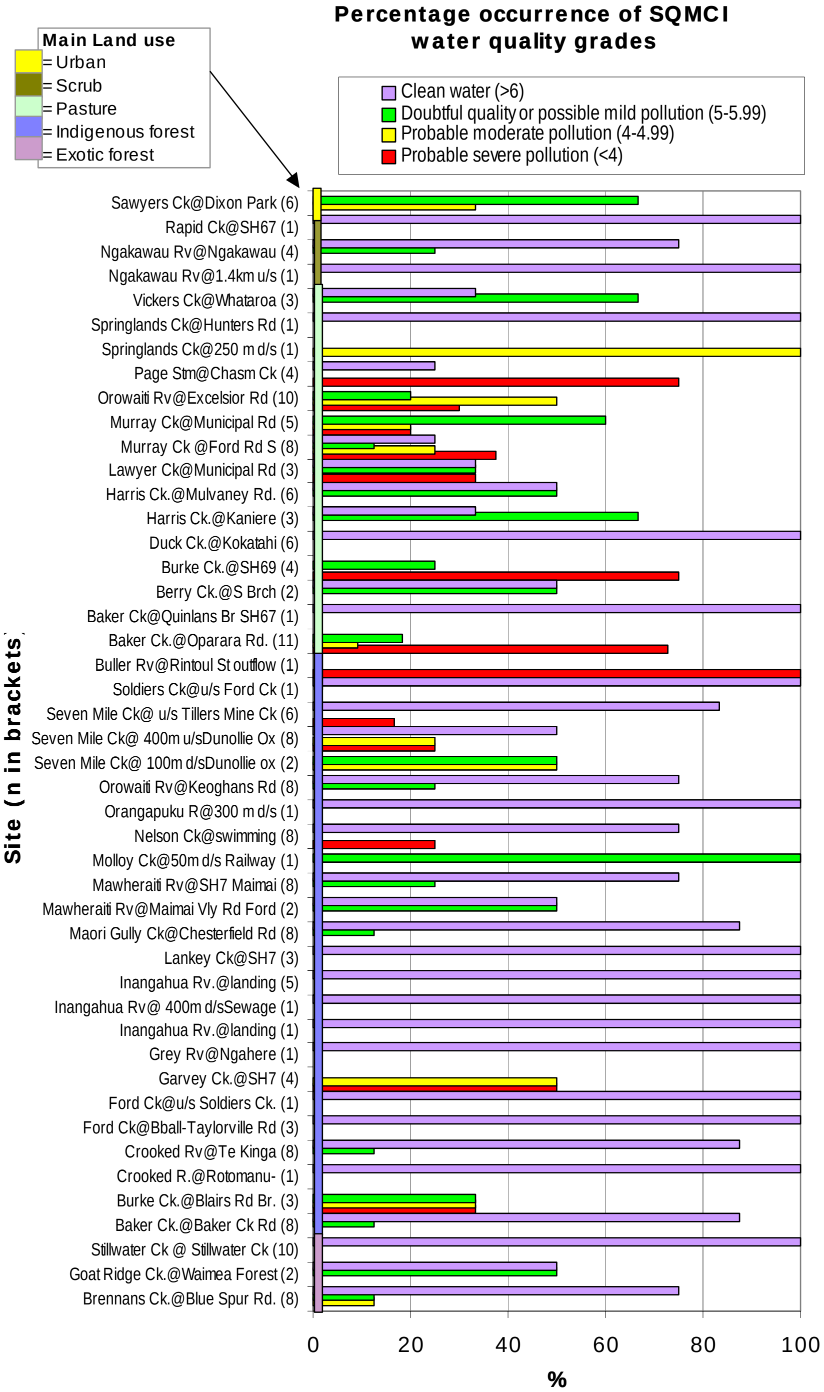


Figure 4.2.16 Percentage of time water quality poor, fair, good or excellent based on semi-quantitative macroinvertebrate community index (SQMCI).

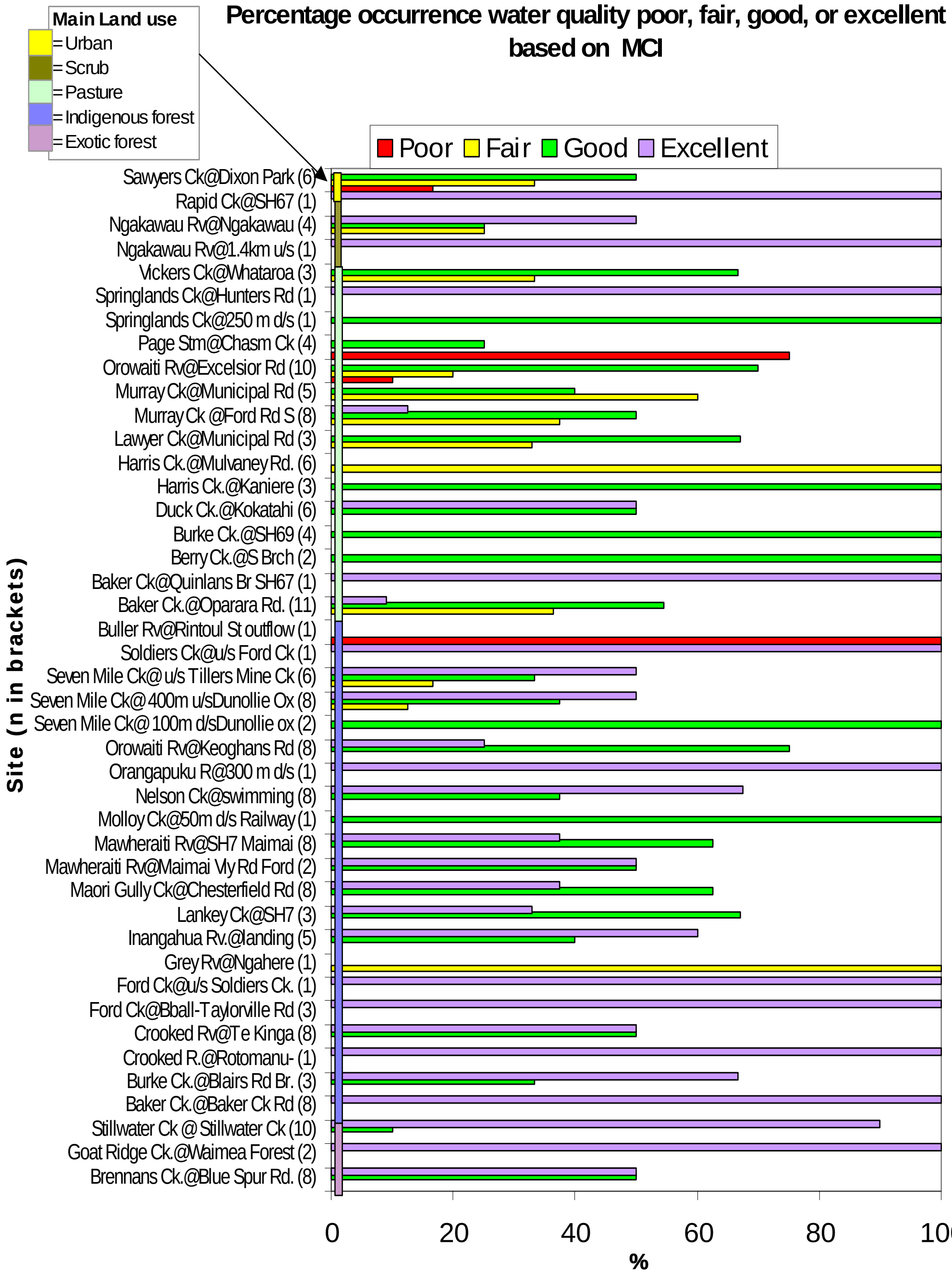


Figure 4.2.17 Percentage of time water quality poor, fair, good or excellent based on macroinvertebrate community index (MCI).

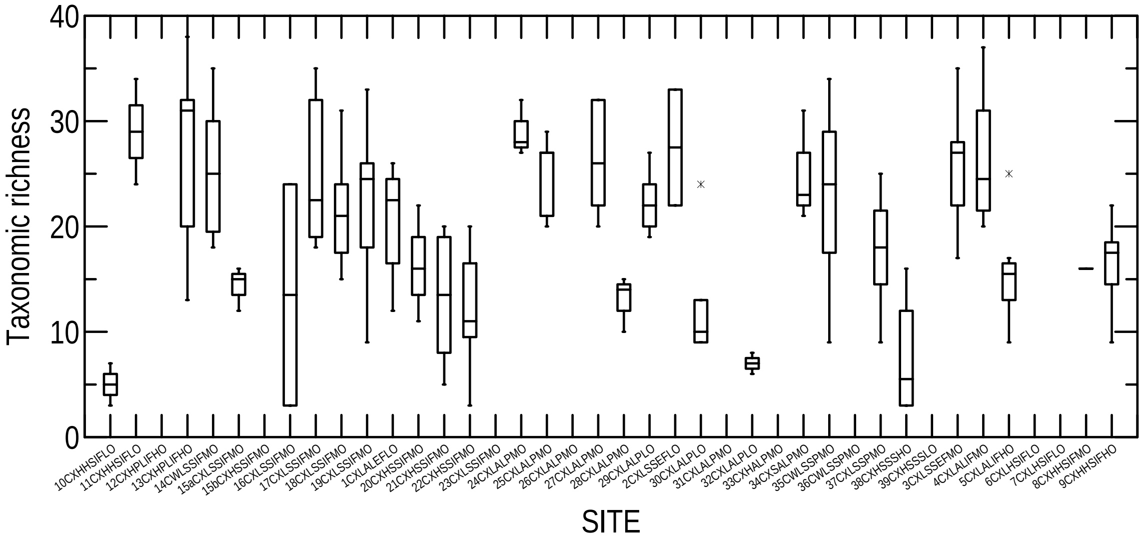


Figure 4.2.18 taxonomic richness for individual sites. For site identification, the first number refers to the site, with the following letters signifying the combined REC as far down to river order, for that site. Refer to Table 4.5 for a site key.

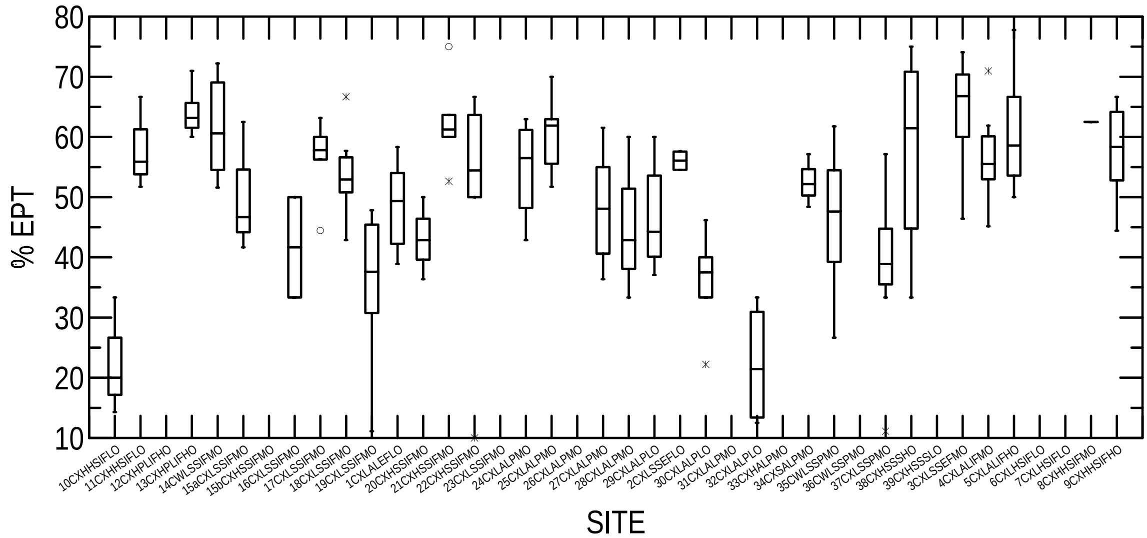


Figure 4.2.19 Percentage EPT for individual sites. For site identification, the first number refers to the site, with the following letters signifying the combined REC as far down to river order, for that site. Refer to Table 4.5 for a site key.

4.3 Spatial representation of sites according to REC class

4.3.1 Macroinvertebrate community structure

Invertebrate data was collated for all sites and plotted using multi-dimensional scaling (MDS). The resulting ordination plot grouped sites according to their REC levels (Figure 4.3.1). The stress level for the ordination was marginal at 0.24. Analysis of similarities yielded an R-value of 0.447, suggesting that the sites are of moderate similarity. This is reflected in the amount of scatter in the ordination plot. Most sites are grouped nearer the centre, but there appears to be some separation of sites on the basis of combined REC type. While groupings are not overly distinct, there is a vertical trend in REC characteristics along the y-axis of the ordination. Larger rivers, and those with predominantly native and hill country catchments are higher on the y-axis, and based on previous analysis in this section, means higher water quality. There was no obvious influence from geology on macroinvertebrate communities when it is examined in combination with land class.

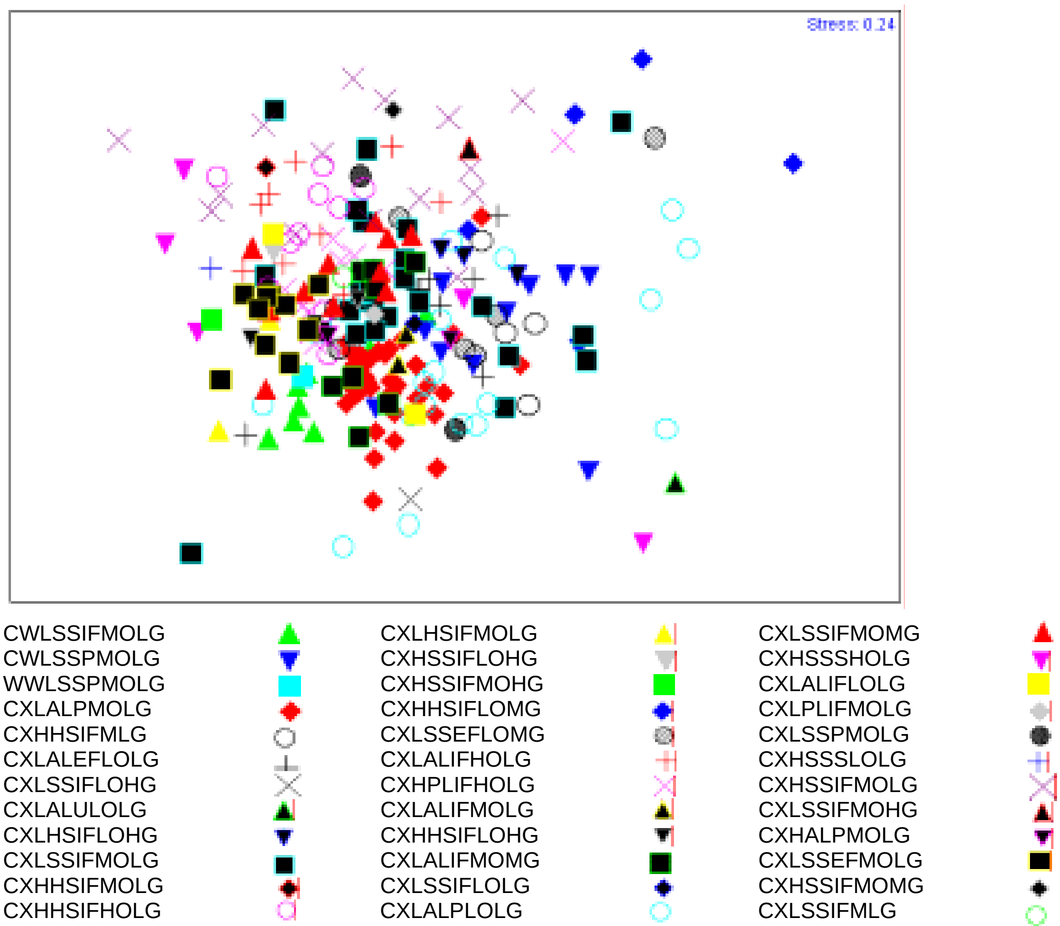


Figure 4.3.1 Ordination comparing invertebrate data among all sites and using REC type to define sites. R = 0.447

Combined REC types were then separated into respective classes for each level, and the ordinations re-plotted. When climate is considered as a factor, any differences in the sites are almost indistinguishable (R=0.056). This is borne out in the ordination (Figure 4.3.2) with the positions of the CW sites being scattered throughout the CX sites. CX had the most sites and the greater variation in macroinvertebrate structure is probably an artefact of its greater sample size.

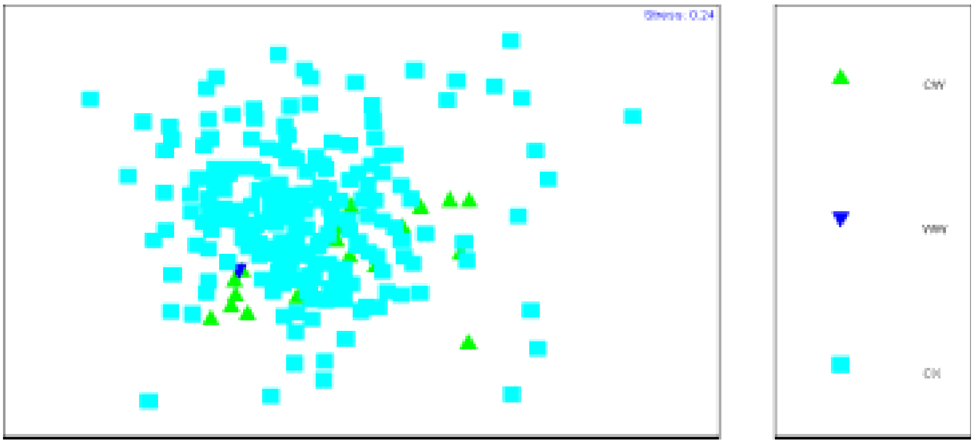


Figure 4.3.2 Ordination comparing climate at all sites. CW = cool wet; WW = warm wet; CX = cool extremely wet. R = 0.056.

When source is used as the separating factor we found that sites were quite scattered, but there is a degree of difference between classes (R=0.208). Sites with a hill source of flow source consistently tended to be positioned higher on the y-axis (Figure 4.3.3). The Ngakawau River was one exception, indicated by the lone blue triangle at the lower right of the ordination plot.

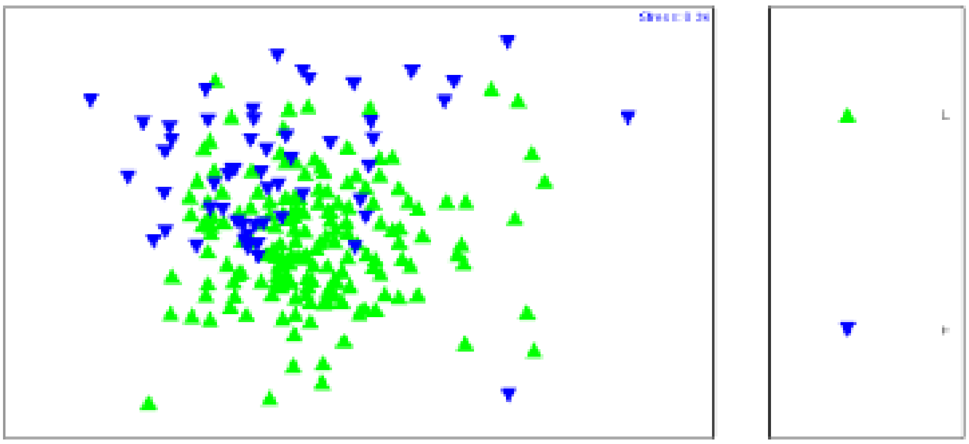


Figure 4.3.3 Ordination comparing source for all sites. L = low elevation (> 50 % of rain occurs < 400 masl); H = hill (> 50 % of rain occurs between 400 masl and 1000 masl). R=0.208.

Sites classified as having soft sedimentary geology were uniformly spread across the ordination plot (Figure 4.3.4), with low differentiation between classes (R=0.054). Sites with hard sedimentary and PL appeared higher and to the left of the ordination compared to sites with alluvial geology. The PL class appeared to form a tighter group, although this may have resulted from insufficient replication of macroinvertebrate data.

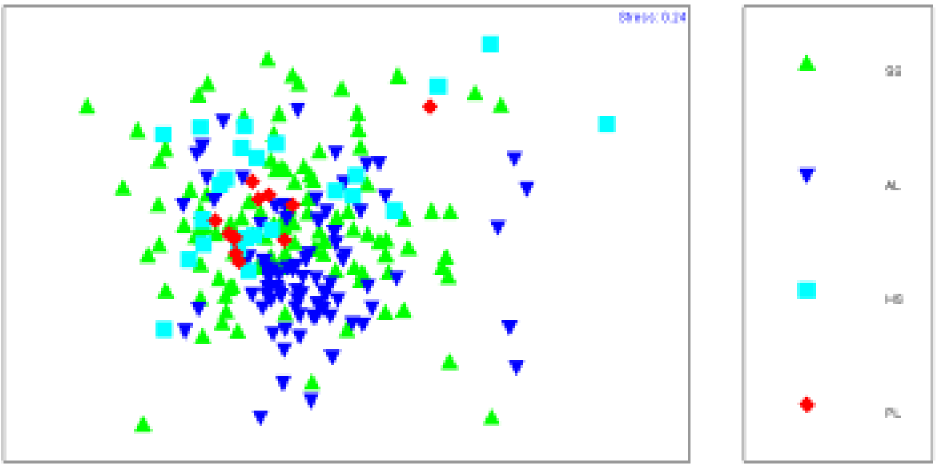


Figure 4.3.4. Ordination comparing geology among all sites. SS = soft sedimentary; Al = alluvial and sand; HS = hard sedimentary; PL = plutonic. R=0.054.

Representation of macroinvertebrate communities based on land class groupings (Figure 4.3.5) displayed a similar pattern to geology due to the relationships between them. Pasture sites are more to the bottom-right of the ordination. Exotic forest sites are in the middle – these all have a low elevation source of flow. Scrub sites are few in number and highly variable within this class. The one urban site was peripheral in relation to the majority of other sites.

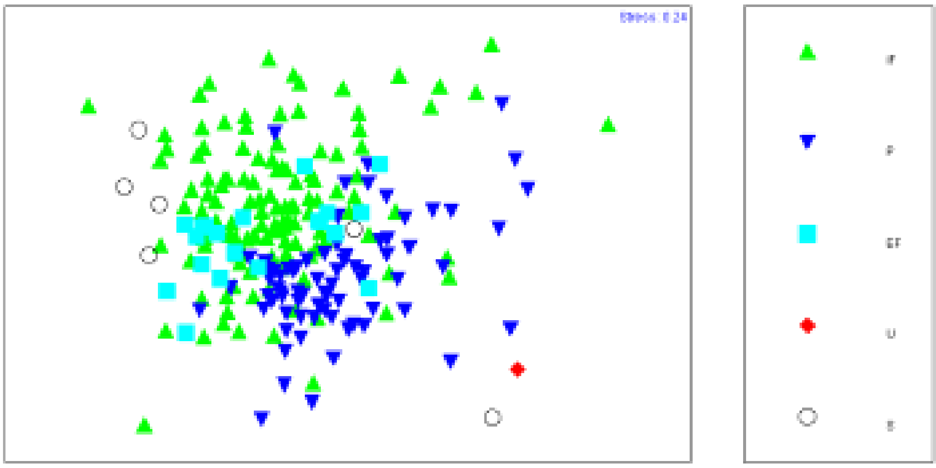


Figure 4.3.5 Ordination comparing landuse among all sites. IF = indigenous forest; P = pasture; EF = exotic forest; U = urban; S = scrub. R = 0.178.

When stream order is considered we find a similar degree of separation among sites as for landuse (R=0.138). High order sites are generally clumped together with the exception, once again, of the Ngakawau River (red diamond at lower right, Figure 4.3.6). Separation of low and middle order sites is less well defined.

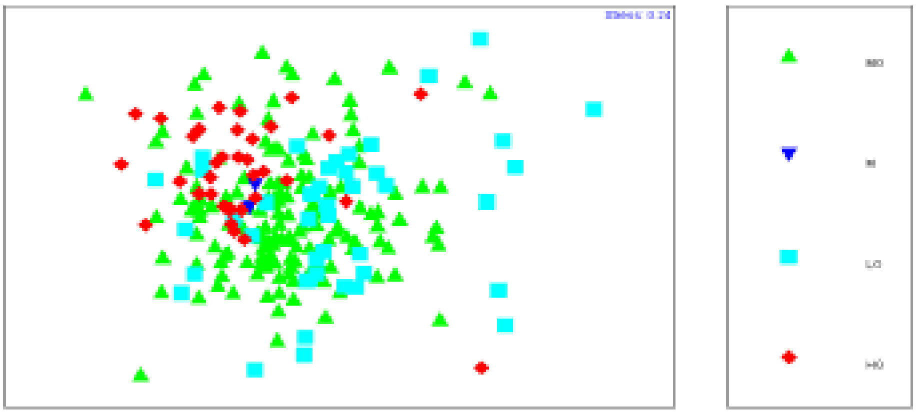


Figure 4.3.6 Ordination comparing river order among all sites. HO = high order; MO = mid order; LO = low order; M = modified. R=0.138.

Gradient appears to have little bearing on invertebrate community structure with all sites being reasonably similar (R=0.004). This is proven by a high degree of scattering among different gradient classifications (Figure 4.3.7).

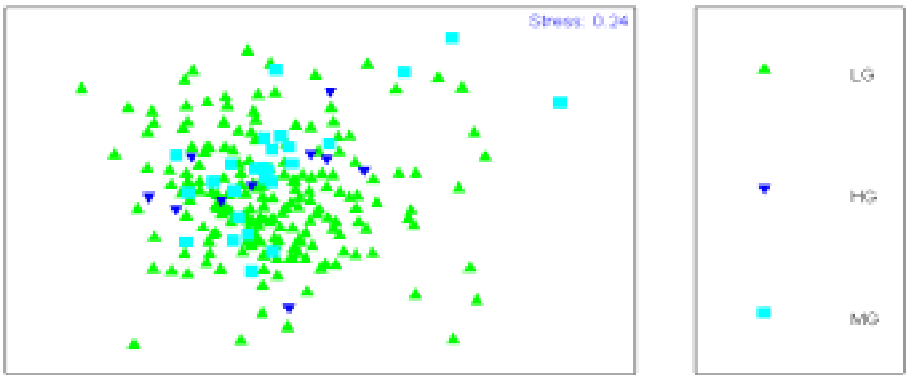


Figure 4.3.7 Ordination comparing valley landform among all sites. LG = Low gradient (slope = < 0.02); MG = medium gradient (slope =0.02-0.04); HG = high gradient (slope > 0.04).

4.3.2 Periphyton community structure

Similarities among sites were plotted for sites where there was sufficient periphyton data available, with combined REC type and landuse being used to group sites. Unfortunately the number of sites with sufficient periphyton data was limited and sites showed few similarities, as seen from the high degree of scatter. For combined REC type there was very little replication. CXLALPMOLG is represented three times, CXLSSIFMOLG is represented twice and the remaining sites represented just once each, and there were no obvious similarities between sites (Figure 4.3.8).

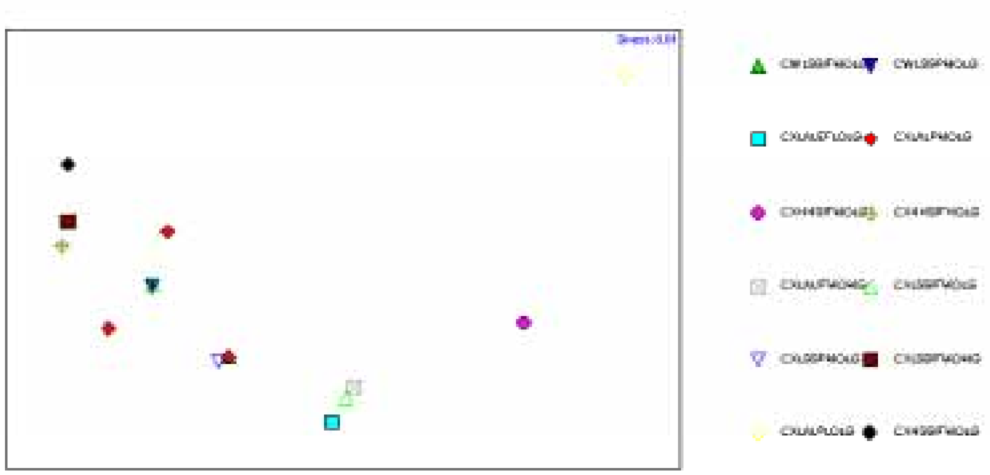


Figure 4.3.8 Ordination comparing periphyton data among sites. Sites classified by REC codes.

Grouping the sites by landuse (Figure 4.3.9) provided slightly more replication, but there is little evidence that landuse has an influence on periphyton cover from this ordination. It could be argued that the sites where the main landuse is pastoral are grouped together, but the existence of Page stream at Chasm Creek walkway far from the remainder confounds this.

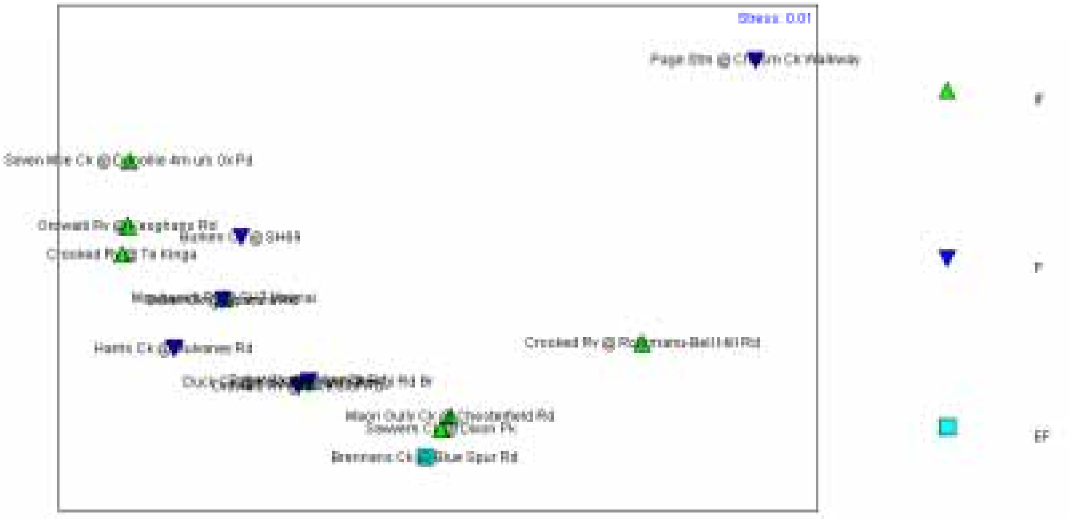


Figure 4.3.9 Ordination comparing periphyton data among sites. Sites classifiad by landuse. P = pasture; IF = indigenous forest; EF = exotic forest.

4.4 Freshwater fish diversity and abundance

Of 36 sites surveyed in 2003, Galaxiids other than Dwarf Galaxias were found at 11 sites. Dwarf Galaxias were found as the only Galaxiid species at six sites. Galaxiids in all 11 sites except Arthur's Creek were found where there was reasonable in-stream cover. The greatest number of galaxiids were found in Sunday Ck (97 Galaxiids; Feb 2003).

Individual species distributions

Dwarf Galaxias and Inanga seem to have little habitat preference and were found in the highly modified creeks of Mosquito Ck (Dwarfs), Findlay Ck (Dwarfs) and Paddy Ck (Inanga). Eels were found at all sites except Brown River, Rotomanu. Giant Kokopu were present only where there was flax or wood in the stream. A small Giant Kokopu was found in a reasonably modified mid section of Puzzle Ck, Rotomanu. Banded Kokopu were only found in tannin-stained streams with good cover (in-stream and overhead). Some were found where there was a silty bed (probably deposited weeks-months prior to the survey). Koaro were found in reasonably modified habitat e.g. Hatters Ck at SH7. Torentfish seem to tolerate reasonable habitat modification e.g. Brandy-Jack Ck at SH7. Brown Trout were found at all but four sites (Waimea Ck, German Gully Ck, Lawyer Ck and Paddy Ck). No Shortjaw Kokopu were found in the survey sites.

Changes in fish distribution and abundance from 2001 survey

Differences at a site between the two surveys are probably attributable to seasonal differences in distribution and highlight the importance of carrying out multiple surveys over different seasons. No mature lamprey were found in the 2003 survey compared to reasonable numbers found at five sites in 2001. This may be due to seasonal migration patterns. More Dwarf Galaxiids were found in 2003 than in 2001. It is suspected that this could be due to misidentification in the 2001 survey. The only major stream habitat changes at the sites compared to the 2001 survey was replacement of TLB stopbanks and rip-rap on Raft Creek (but with very little bed material removed). And Palmers Creek, which had humping and hollowing in its catchment 120m from the creek.

Relationship between Brown Trout and sensitive native fish

Where Brown Trout were found in high numbers, particularly in the Grey River catchment, few or no sensitive native fish (Galaxiids) were found (Figure 4.4.1). Similar relationships were found when Brown Trout under 120 mm in length were compared with the number of sensitive native fish. Larger size classes of trout are known to be piscivorous (fish-eating). This relationship was similar to that found in a study carried out for Department of Conservation in 2001-02 (Harding et al 2002) and the WCRC 2001 survey.

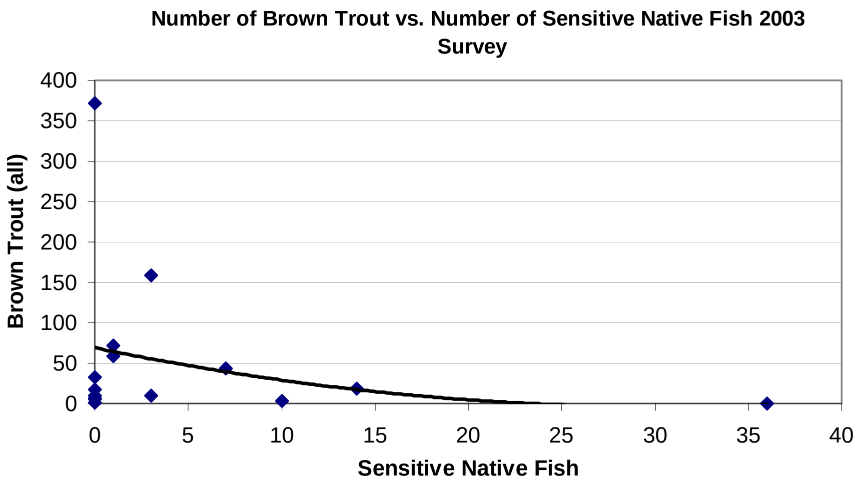


Figure 4.4.1 Number of Brown Trout vs. number of sensitive native fish, 2003 survey.

However, there were several sites where Brown Trout and Sensitive Native Fish coexisted. These sites were: Duck Ck reference site and Upper Kokatahi (Banded Kokopu); Orangipuku River, Inchbonnie, Pidgeon Ck and Rotomanu (Giant Kokopu); Hatters Ck downstream SH7, Totara Flat, Arthurs (Doughboy) River and Kowhitirangi (Koaro).

It has been suggested that this co-existence could be related to the wetted width of the stream, with wider streams affording greater area for fish to escape their predators. With the exception of Pigeon Ck, the wetted width of all these waterways is greater than two metres. However, little relationship can be drawn between waterway width and co-existence as several larger waterways such as Brandy-Jack Creek (Totara Flat), Souters Creek (Nelson Creek), Brown River (Rotomanu), Nicholas-Clear Creek (Taramakau Settlement) and Vickers Creek (Whataroa) did not contain any sensitive native fish.

Trout appeared less sensitive to habitat modifications, although still seem to require some habitat characteristics such as limited suspended sediments and lower temperatures.

Relationship with fish distribution and REC

REC levels were used to compare geology, source of flow and stream order, with species distributions. There appeared to be significantly more species of fish found in low elevation waterways compared to hill or spring source of flow (Figure 4.4.2). Galaxiid species, including sensitive native fish, were only found at low elevation sites. Sites in the hill source of flow class had more torrent fish and trout, and fewer eels. Hill source of flow sites may have steeper gradients and more turbulent flow suited to the first two species, and not to the third. In the 2001 survey, no clear relationships between fish distribution and river environment classification were observed.

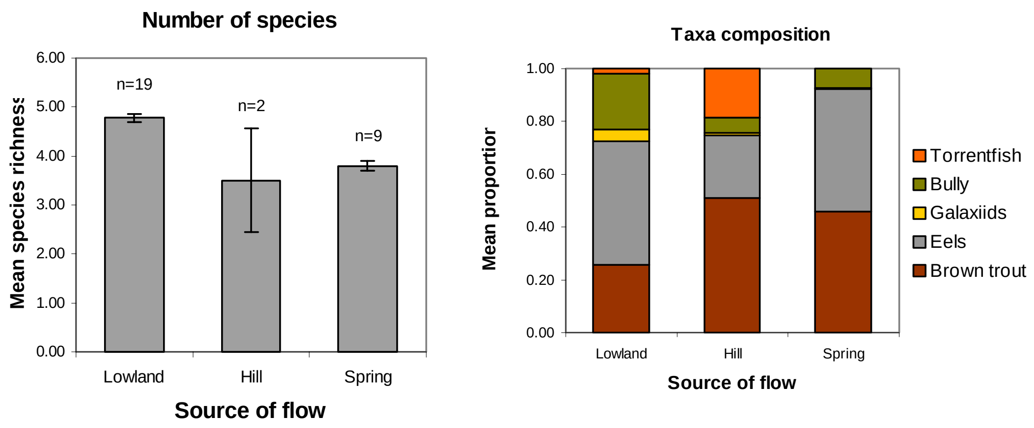


Figure 4.4.2 Number of species and taxonomic composition among REC source of flow classes, 2003 survey.

There were significantly more fish species in waterways dominated by soft sedimentary geology compared to alluvial or hard sedimentary geology (Figure 4.4.3). Torrentfish were not found at sites with alluvial geology. Galaxiids, including sensitive native fish, were found across all three geology classes assessed.

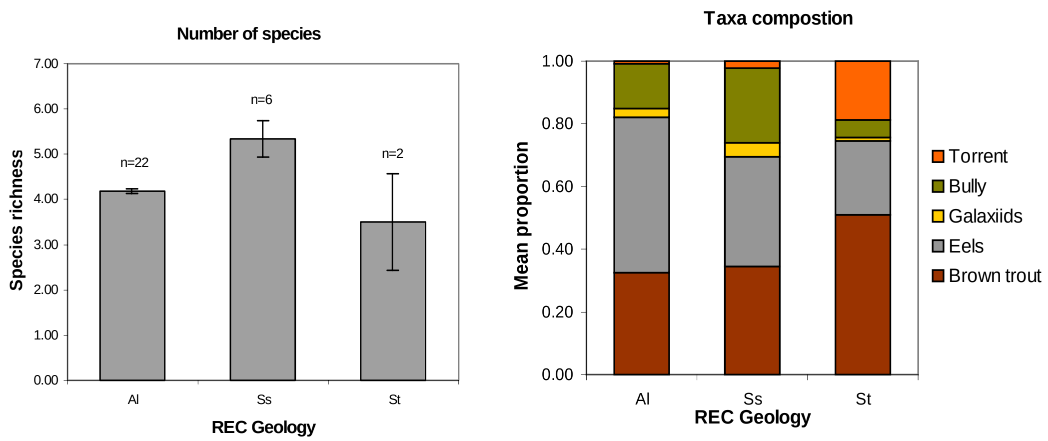


Figure 4.4.3 Number of species and taxonomic composition among REC geology classes, 2003 survey. Al = alluvial, SS = soft sedimentary; St = schist

It appears that there is very little difference in the number of species between first, second and third order streams (Figure 4.4.4). With only one sample of a fourth order stream, it is not possible to make a comparison between fourth and smaller order sites. There did appear to be an inverse relationship with higher ratios of Bullies to Brown Trout in larger streams, possibly resulting from predation pressure on the Bullies in smaller streams.

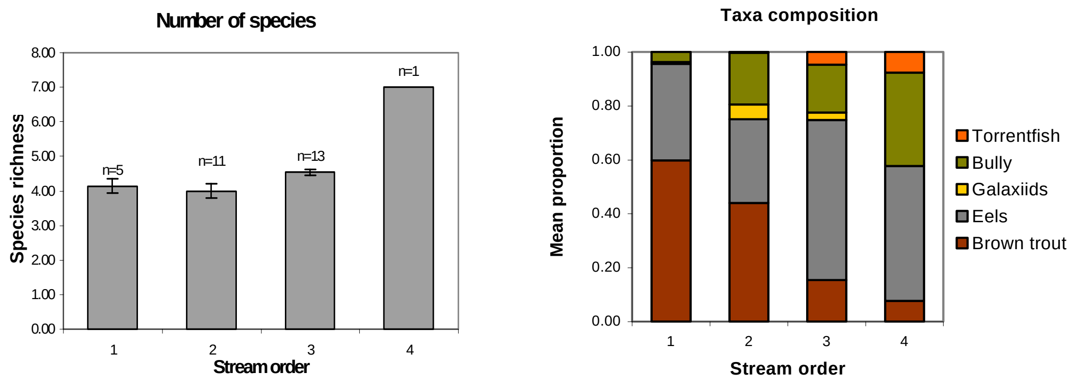


Figure 4.4.4 Number of species and taxonomic composition among different stream orders, 2003 survey.

The total number of fish species was slightly greater at sites in the least disturbed classes (Class 3 and 4) unlike the 2001 survey where there was no significant difference in total fish species observed between the disturbance classes (4.4.5).

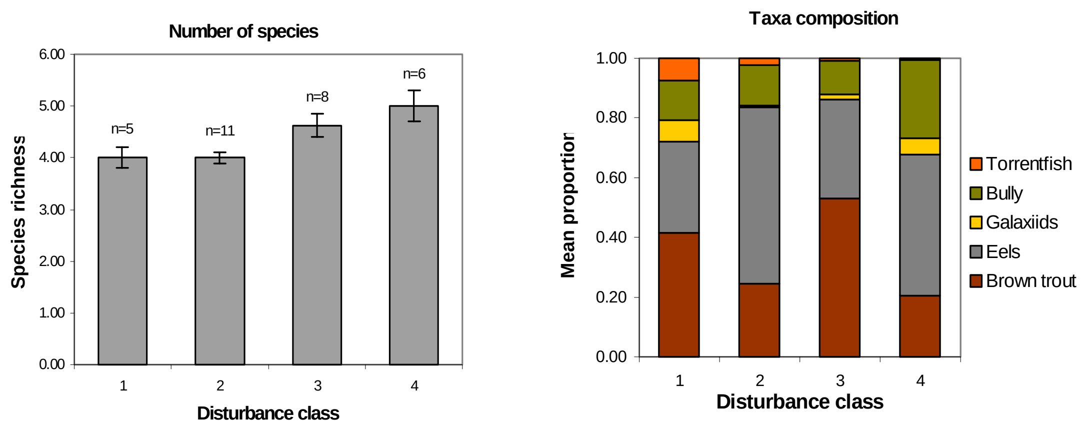


Figure 4.4.5 Number of species and taxonomic composition among disturbance categories, 2003.

The density of sensitive native fish species was highest in class four sites with little difference between Class two and three sites. The 2003 graph has been redrawn from that plotted in James et al 2001 on the basis that Koaro are no longer considered sensitive native fish (Figure 4.6).

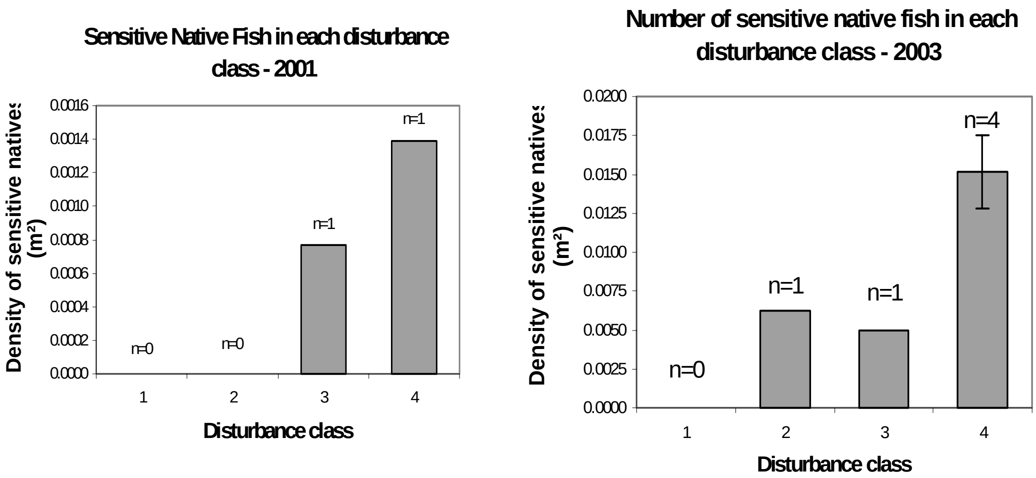


Figure 4.4.6 Comparison of sensitive native fish densities between 2001 and 2003 surveys.

4.5 How does the Coast compare to conditions in other regions of New Zealand?

Nearly all West Coast sites in this study were categorised as having cool, extremely wet (CX) climates. From a NIWA study of national river data (Larned et al. 2005), the CX climate had the highest water quality out of a total of six climate types. The fact that 87 % of CX sites were covered with mainly natural vegetation will have contributed to higher water quality.

This study found that West Coast sites with high elevation source of flow had typically higher water quality, compared to those sites from lowland sources of flow. The main reason for this difference stems from the predominantly undisturbed native vegetation and hard, un-erodable geology possessed by hill source of flow sites. The national study analysed six climate types, and four sources of flow classes. Of these 24 combinations they found that CXH consistently had some of the highest water quality amongst all climate/source flow types with all median water quality parameters within respective guidelines. Approximately half of the sites in this study were CXH, and it could be considered that roughly half of the rivers in the WCRC SOE monitoring dataset have overall water quality (excluding faecal coliforms) as high as anywhere in New Zealand.

The other half of West Coast sites fell into the low elevation source of flow class. Of the CXL sites in the national study, most came from the West Coast Region. No low elevation sites were among those with high water quality in the study of national rivers, although low elevation sites on the West Coast were some of the best nationally in terms of water quality. In general, lowland areas have a greater frequency of development, which often impacts negatively on water quality. The CXL sites on the West Coast all had either soft sedimentary or alluvial geology. Soft sedimentary geology is more predisposed to erosion – something exacerbated by development – with increased sediment in this case being the primary mechanism of poor water quality. As a result of lower channel gradient, and through groundwater/spring water source West Coast Rivers in alluvial catchments had high background water clarity. However, much of this low gradient alluvial land has been developed for pastoral agriculture, with subsequent water quality issues arising, particularly faecal contamination from agricultural sources and domestic sewage.

Mean *E. coli* levels in other regions ranged from 140 to 1026 *E. coli*/100 ml. The average of all site means for the West Coast core monitoring sites was 528 *E. coli*/100 ml, situated in the middle of the national range. Means are used here to compare West Coast faecal indicators with other regions, but medians have been used elsewhere in this report as they give a more accurate representation of normal stream conditions.

Mean water clarity (using black disk) in other regions ranged between 0.7 to 3.5 m. Water clarity on the West Coast, with a mean of 2.8 m, was in the upper clarity range. Ammoniacal nitrogen was close to the best with a mean of 0.02 mg/L. This compared to the regional range of 0.01 to 0.112 mg/L.

Nationally, pastoral landcover classes had lower water quality than natural ones, which was also apparent in the West Coast Region.

At a national scale, negative trends were observed for ammoniacal nitrogen, but the opposite has occurred in at least one part of the West Coast, remembering that ammoniacal nitrogen levels were some of the lowest when compared to the national range. Positive national trends in Nitrate and temperature have not been observed in the West Coast data, with high variation and a lack of replication, possibly contributing to the result.

5 Comparison between reference and impact sites

5.1 Statistical Comparisons at Paired Sites

Where data availability allowed, water quality parameters were compared between paired reference and impact sites using one-way analysis of variance (ANOVA)(Table 5.1.1). Graphical comparisons have also been presented for this data. The parameters that differed significantly from reference to impact sites were faecal coliforms (increase), ammoniacal nitrogen (increase), and pollution sensitive macroinvertebrate taxa (decrease). To a lesser extent nutrients and suspended particles (inferred from clarity) increased; the later was exacerbated by soft sedimentary geology. Dissolved oxygen and conductivity did not show any consistent pattern. Neither did pH, except for those sites with acid rock drainage, which caused a significant drop in pH where buffering from highly calcareous rock did not occur.

5.1.1 Water clarity

There was more data for water clarity than turbidity, so clarity was used for comparison of suspended material between reference and impact sites. Many but not all reference sites had significantly higher clarity with Baker and Sawyers Creeks, and the Karamea River having similar levels (Figure 5.1.1). It is unclear whether this was the result of minimal downstream impacts, or inputs from natural processes – note that none of these sites have development upstream of their reference sites. Both Baker and Sawyers Creeks have soft sedimentary geology in their catchments, which may have contributed to reducing clarity at these two reference sites. The Karamea River is a large river and while the reference site had lower clarity, its size provided a certain amount of buffering from impacts associated with agricultural landuse.

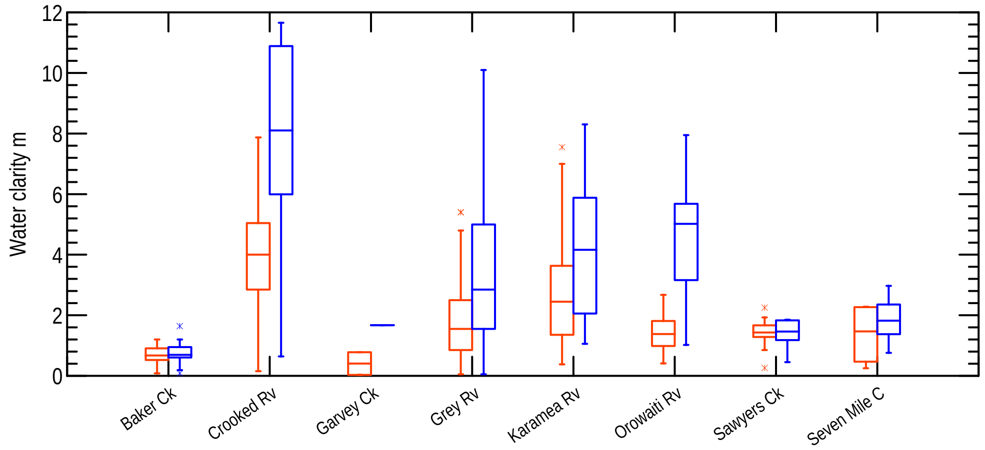


Figure 5.1.1 Water clarity at paired impact (red) and reference (blue) sites.

5.1.2 Dissolved oxygen

Baker Creek had significantly lower dissolved oxygen at its reference site, as did the Grey River, although the Grey had low variation and a large dataset, possibly contributing to this result. Overall, dissolved oxygen did not vary greatly between impact and reference conditions (Figure 5.1.2).

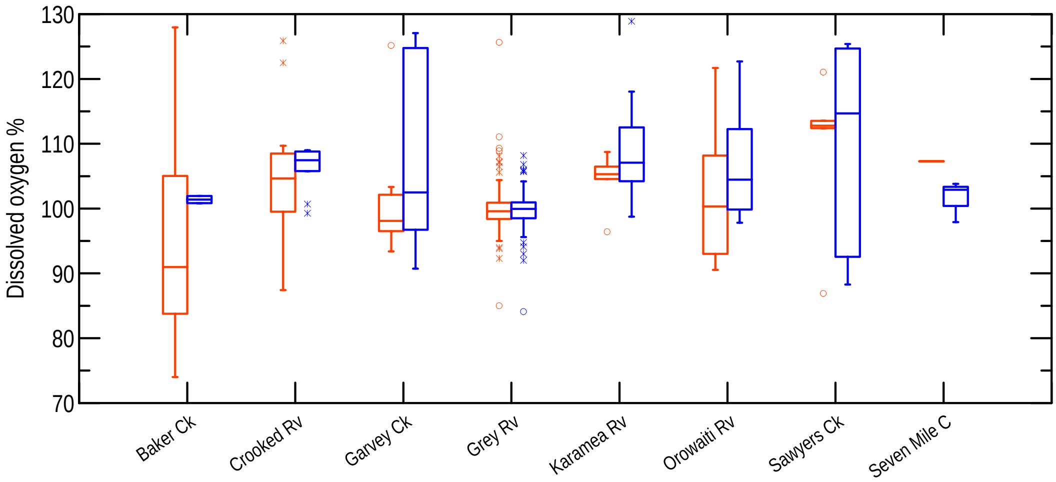


Figure 5.1.2 Percentage dissolved oxygen saturation at paired impact (red) and reference (blue) sites.

5.1.3 pH

Levels of pH were lower at all impact sites except for Seven Mile Creek, with significant differences observed at Baker and Garvey Creeks, and the Grey River (Figure 5.1.3). Seven Mile has historic acid mine drainage above the reference site, while additional acidic inputs are ameliorated by geological areas high in calcium carbonate downstream. Acid mine drainage above the Garvey Creek impact site was responsible for the low pH at this site (note that the reference site for Garvey Creek is located nearby in Lankey Creek).

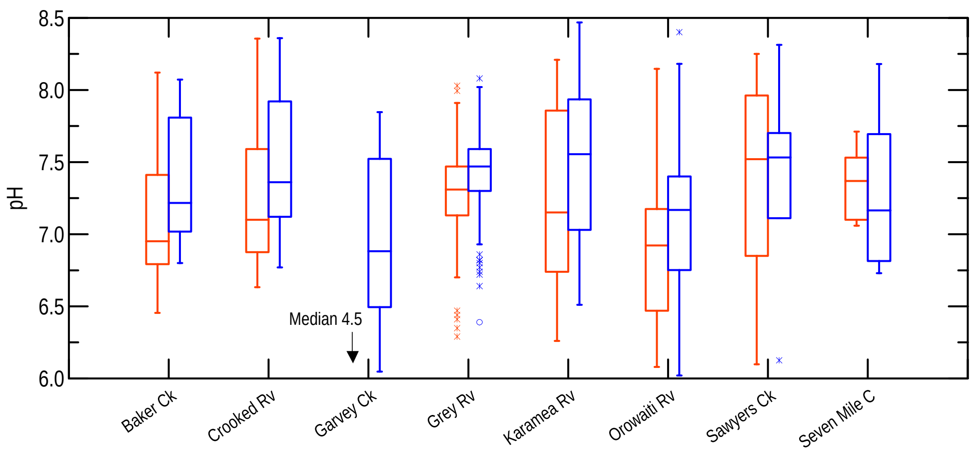


Figure 5.1.3 pH at paired impact (red) and reference (blue) sites.

5.1.4 Specific conductivity

There was no clear 'impact – reference' pattern for specific conductivity. It was higher at some impact sites, Grey and Garvey the significant ones, with the exception of Sawyers and Karamea, which had higher reference conductivities (Figure 5.1.4). Increases at Grey and Garvey are likely to relate to the effects of agriculture and mining. Why the reference sites at Karamea and Sawyers Creek are higher in unclear.

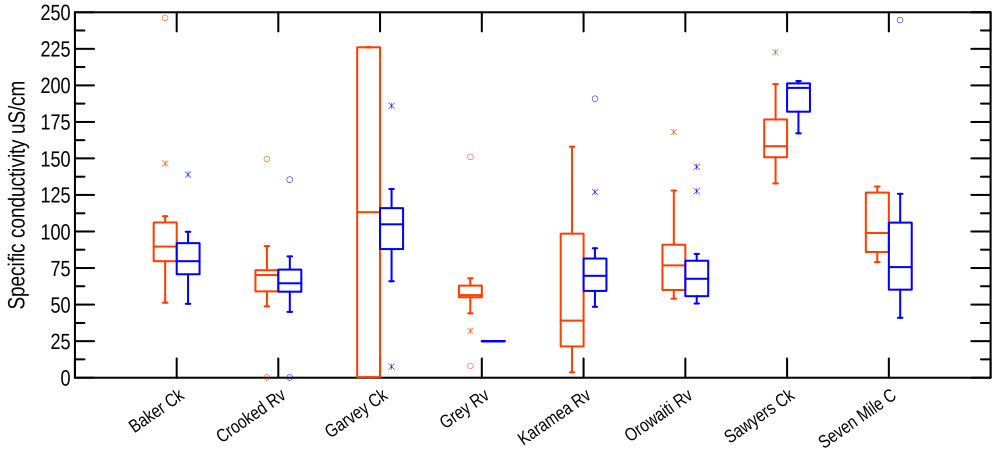


Figure 5.1.4 Specific conductivity at paired impact (red) and reference (blue) sites.

5.1.5 Temperature

While cooler temperatures occurred at reference sites, this was only significant for the Crooked River, with variation between both reference and impact conditions similar (Figure 5.1.5).

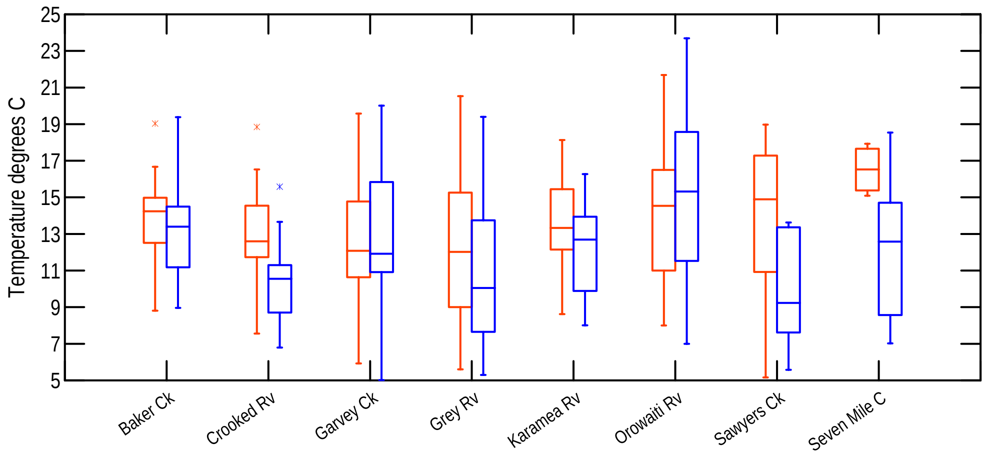


Figure 5.1.5 Temperature at paired impact (red) and reference (blue) sites.

5.1.6 Ammoniacal nitrogen and faecal coliforms

Ammoniacal nitrogen and faecal coliforms (Figures 5.1.6 and 5.1.7) were higher at impact sites. Agricultural development is the likely cause for this pattern, with lesser contributions from urban sources like septic tank leakage and sewerage. Ammoniacal nitrogen varied less among reference sites as compared with the variation observed for impact sites.

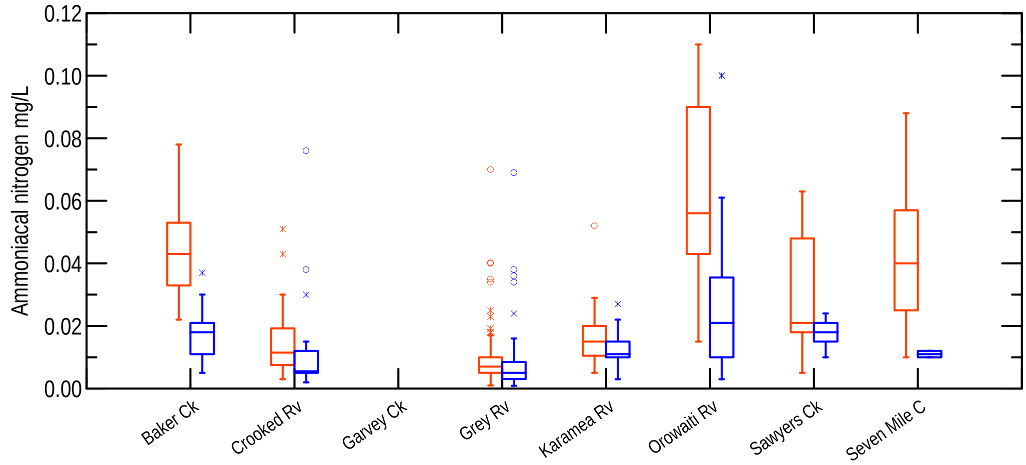


Figure 5.1.6 Ammoniacal nitrogen at paired impact (red) and reference (blue) sites.

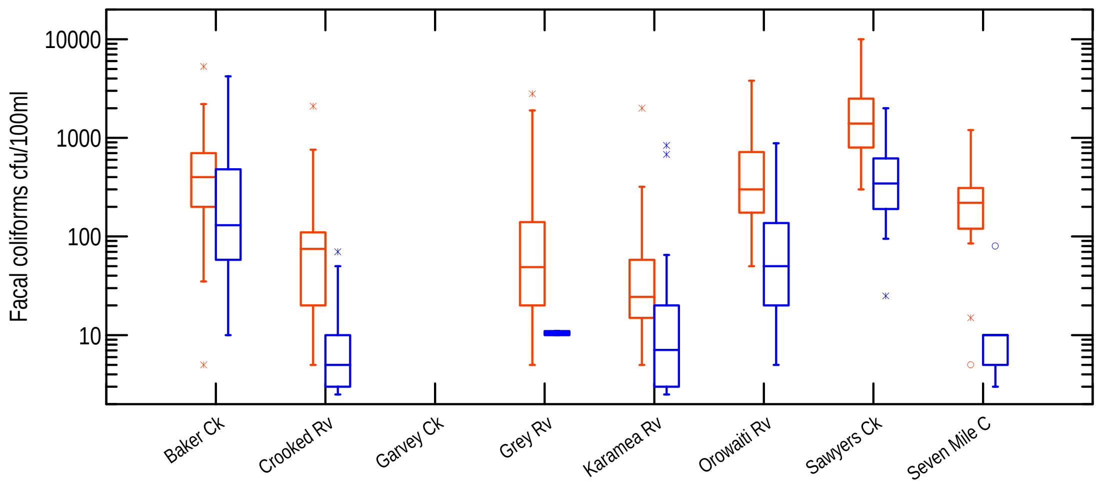


Figure 5.1.7 Faecal coliforms at paired impact (red) and reference (blue) sites. Note the log scale.

5.1.7 Nutrients and sulfate

There were only two sites with sufficient nutrient data to justify comparison, both of which were the large rivers Grey and Crooked (Figure 5.1.8). Reference sites had lower nutrient levels, but none were overly marked, possibly reflecting the fact that these sites are predominantly fed by indigenous forested headwaters.

Likewise, Sulfate only had two sites with data. Both had coal mining activity in their catchments - Garvey Creek had a highly significant difference between impact and reference conditions (Figure 5.1.8).

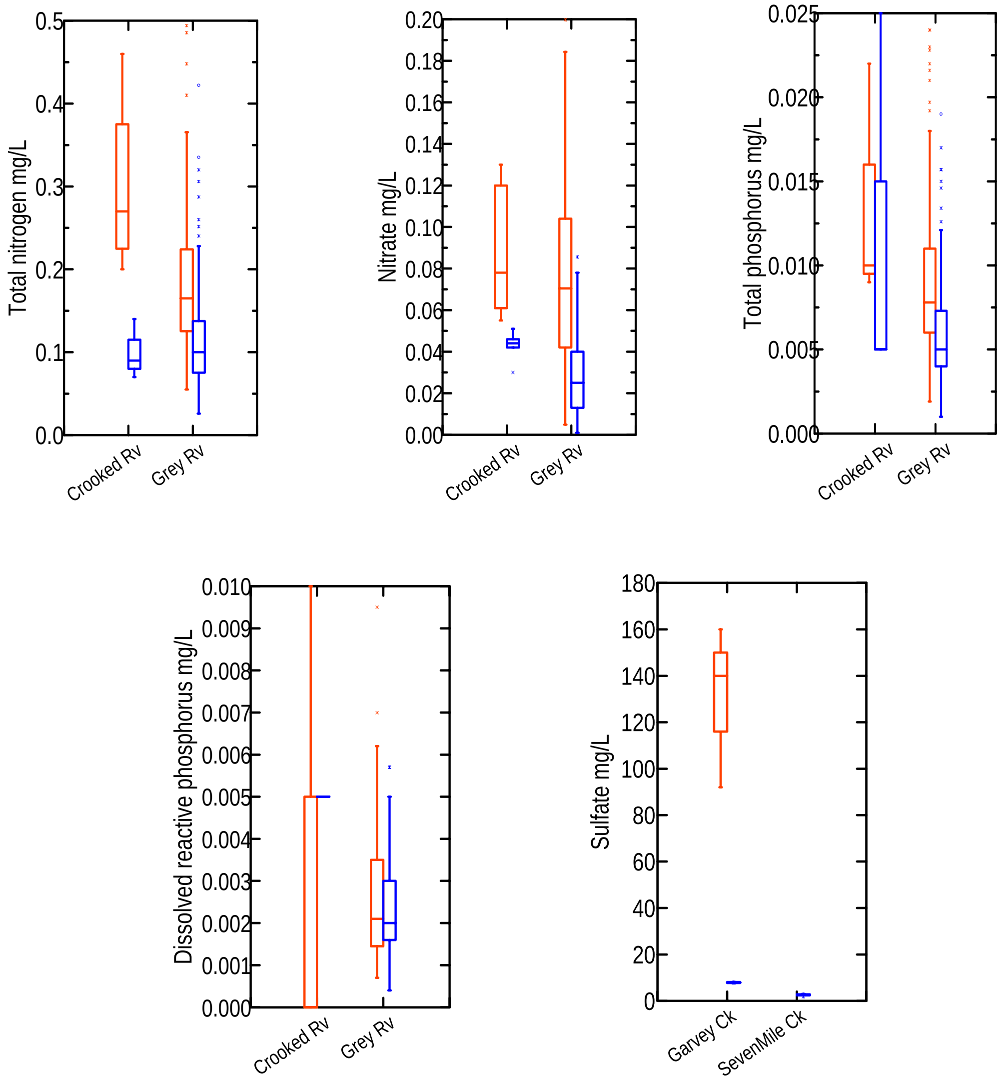


Figure 5.1.8 Nutrients (total nitrogen, nitrate, total phosphorus, and dissolved inorganic phosphorus), and sulfate at paired impact (red) and reference (blue) sites.

5.1.8 Macroinvertebrate parameters

Numbers of pollution sensitive taxa clearly decreased from reference to impact sites, most pronounced at Baker and Garvey Creeks. The SQMCI seemed to display the most significant differences between impact and reference sites. The reference site at Seven Mile Creek had a greater proportion and abundance of pollution sensitive taxa, but taxonomic richness was greatest downstream at its impact site (Figure 5.1.9). Overall, taxonomic richness alone was not particularly good at discriminating between impact and reference sites.

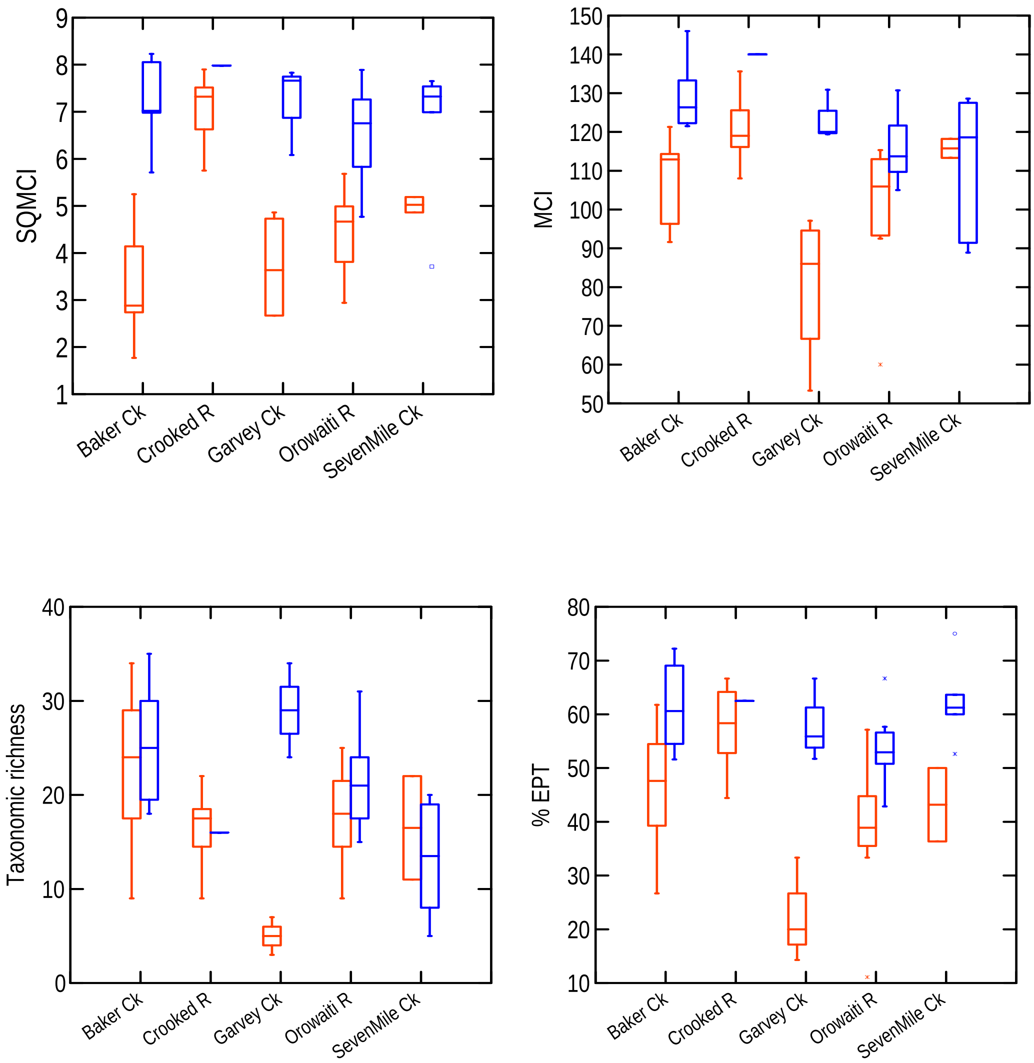


Figure 5.1.9 SQMCI, MCI, taxonomic richness, and percent EPT at paired impact (red) and reference (blue) sites.

Table 5.1.1 Table of one-way ANOVA results between paired impact and reference sites for respective water quality parameters. Yellow and orange cells denote probabilities below or equal to 0.01, and 0.05, respectively. Insufficient data is denoted by *

	Baker Creek		Crooked River		Garvey Creek		Grey River		Karamea River		Orowaiti River		Sawyers River		Seven Mile Creek	
Parameter	*F*	*P*	*F*	*P*	*F*	*P*	*F*	*P*	*F*	*P*	*F*	*P*	*F*	*P*	*F*	*P*
% Dissolved oxygen saturation	530.363	0.000	*	*	0.871	0.371	22.923	0.000	*	*	0.475	0.497	0.001	0.978	0.057	0.944
Water clarity	0.017	0.896	22.047	0.000	*	*	39.200	0.000	0.517	0.479	47.832	0.000	0.128	0.724	3.428	0.043
pH	5.105	0.031	1.758	0.193	105.916	0.000	21.436	0.000	*	*	0.258	0.615	0.236	0.635	0.817	0.448
Conductivity	0.004	0.951	3.097	0.086	16.726	0.001	9.697	0.000	2.795	0.118	0.557	0.461	0.020	0.889	1.586	0.217
Temperature	0.194	0.663	6.726	0.015	0.337	0.571	1.160	0.283	*	*	0.341	0.562	0.934	0.347	0.117	0.889
Total ammonia	6.369	0.017	1.368	0.247	*	*	4.972	0.026	18.840	0.000	4.176	0.047	1.960	0.178	*	*
Faecal coliforms	4.175	0.047	35.506	0.000	*	*	7.657	0.008	4.300	0.045	5.798	0.021	3.040	0.097	3.424	0.036
Nitrate	*	*	0.121	0.736	*	*	*	*	1.310	0.262	*	*	*	*	*	*
Total phosphorus	*	*	*	*	*	*	*	*	1.976	0.171	*	*	*	*	*	*
Dissolved reactive phosphorus	*	*	*	*	*	*	*	*	1.339	0.299	*	*	*	*	*	*
Sulfate	*	*	*	*	36.821	0.000	*	*	*	*	*	*	*	*	1.291	0.285
SQMCI	21.120	0.000	*	*	21.204	0.019	*	*	*	*	23.917	0.000	*	*	2.841	0.126
MCI	17.678	0.001	*	*	1.436	0.317	*	*	*	*	3.906	0.068	*	*	0.555	0.475
Taxonomic richness	0.418	0.528	*	*	81.176	0.003	*	*	*	*	2.147	0.165	*	*	0.039	0.847
% EPT	8.771	0.011	*	*	10.735	0.047	*	*	*	*	3.701	0.075	*	*	0.144	0.713
Periphyton richness	136.219	0.000	*	*	*	*	*	*	*	*	1.597	0.242	*	*	*	*

6 Water quality trends in West Coast surface waters

6.1 Trends Analysis

Investigation of trends between impact and reference sites was carried out using linear regression and LOWESS smoothed regression for physicochemical and bacteriological parameters. Kendall's tau was used to determine levels of significance for the correlations between reference and impact site parameters. The closer Kendall's tau 'b statistic' was to 0, the less similar the trends between sites were. If b was closer to 1 or –1, trends between sites were similar. Note that ammoniacal nitrogen is referred to as ammonia on the following plot titles of this section.

When both impact and reference sites varied in the same way, it was assumed that natural variation - caused mainly by differing rainfall and flow regimes - was influential. Where their responses differed, anthropogenic influences were the likely mechanism. For parameters that can relate closely to landuse like clarity, faecal coliforms, ammoniacal nitrogen and conductivity, a drop in levels for both sites would probably mean either lower rainfall and run-off (e.g. turbidity and faecal coliforms), or higher rainfall and dilution (e.g. conductivity and ammoniacal nitrogen), at the time of sampling. The magnitude of rainfall, and corresponding stream flow may also have had an influence. For example, the difference in faecal coliforms between reference and impact sites may have been accentuated as run-off increased. Hence changes in difference over time between impact and reference sites were not necessarily solely related to land management practice.

Replication was not sufficient to assess if any synchronous impact/reference changes over time were driven by climatic factors. In some cases, determination of trends was hindered by insufficient data and high variation. Moderate to high correlations between most paired impact and reference data indicated that natural mechanisms were important drivers of many patterns. Tentative evidence suggested a reduction in faecal coliforms at Karamea, Orowaiti, and Baker Creek, with a possible increase in ammoniacal nitrogen at the Orowaiti River reference site.

There were difficulties measuring trends with the quarterly monitoring data contained in this dataset. From this recent work, much valuable knowledge has been gained on the state of water quality in the West Coast region. Looking toward future monitoring, improved ability to detect trends is now likely to become a greater priority. One means of doing this would be to reduce the number of core monitoring sites, and increase their monitoring frequency. From the use of REC, and the subsequent information the technique has provided, it may be possible to determine where there are significantly homogeneous groups of sites that can be culled to allow for more resources to be invested into higher sampling frequency. Better trend analysis would allow the WCRC to better assess the effectiveness of its management systems and activities like resource planning, consenting, and compliance.

6.1.2 Crooked River

There was a gradual trend in Crooked River pH to decrease towards neutrality (pH 7). The difference in pH between sites appeared to get smaller, although differences varied highly over time (Figure 6.1.1). Trends at both sites were quite similar (b=0.806), with this decrease in pH possibly a result of natural mechanisms.

Some high outliers in May/June 2003 raised levels of ammoniacal nitrogen that otherwise were relatively stable over time (Figure 6.1.2). Differences between sites appeared to be static, with the high May/June 2003 figure causing the slight increase shown in the regression. There was no trending in temperature over time (Figure 6.1.3), with impact and reference sites responding in similar fashion (b=0.735).

There was a slight reduction in clarity at the reference site as suggested by a minor increase in the difference and between LOWESS regressions. But both sites were quite similar hence rainfall may have been the main determinant of these patterns, rather than activities associated with surrounding agricultural landuse (Figure 6.1.4).

There was high variability and little in the way of correlation between reference and impact faecal coliforms. Correspondingly no clear trend was present, although there was a possible improvement at the reference site (Figure 6.1.5).

Conductivity at both sites was very similar (b=0.881), so it is difficult to determine if increasing difference between impact and reference sites was due to rainfall variation during sampling, or increased pressure from agricultural development (Figure 6.1.6).

6.1.3 Seven Mile Creek

Trends in pH, temperature, and clarity at Seven Mile Creek were difficult to interpret, and there was insufficient data to investigate trending. LOWESS smoothed regression lines for these parameters followed a similar path, indicating that any patterns over time were unlikely to be anthropogenic in origin (Figure 6.1.7, 6.1.8 and 6.1.9). The prognosis for conductivity is similar - Rapahoe had few data points, with possible tidal influence (Figure 6.1.10).

6.1.4 Karamea River

Differences of pH between sites over time appeared constant, as did the relationship between LOWESS regressions (Figure 6.1.11). Along with a fair correlation (b=0.615) between data at both sites, most variation in water quality was attributable to differing rainfall and flow regimes during sample collection. Ammoniacal nitrogen levels in the Karamea River appeared to drop slightly over time, most significantly up to 2001 (Figure 6.1.12). The main support for this was a reduction in difference between impact and reference sites. There was high correlation of temperatures between sites (b=0.822)(Figure 6.1.13).

Difference between faecal coliforms at impact and reference sites was very slight, with no apparent trending. Despite significant dilution, changes at the impact site were not highly correlated to those at the reference site (b=0.491)(Figure 6.1.14).

6.1.5 Orowaiti River

Despite variability, trends in pH for the Orowaiti River were similar at both impact and reference sites (b=0.708). There may have been a gradual increase in pH at the reference site, based on a minor increase in the differences between sites (Figure 6.1.15).

Differences in ammoniacal nitrogen between sites were highly variable at the start and near the end of the record, flattening the regression (Figure 6.1.16). LOWESS regression of ammoniacal nitrogen at the impact site showing a drop in levels around 2000, followed by an increase – but this pattern was not reciprocated at the reference site (b=0.157), suggesting an increase in ammoniacal nitrogen after 2000. Temperature in the Orowaiti River was closely correlated between sites (b=0.874), with no trends (Figure 6.1.17).

Diminishing differences and closer non-adjusted regression indicates improving water claity, yet LOWESS regression from 2002 onwards suggests otherwise (Figure 6.1.18). Between them we can conclude that there was insufficient evidence to indicate any clarity trend. A minor reduction of faecal coliforms occurred at the Orowaiti River impact site. Differences between sites clearly appeared to decrease both for difference regression, and between LOWESS regressions for each site (Figure 6.1.19).

Levels of dissolved oxygen at both the reference and the impact site on the Orowaiti River are high, and changed little (Figure 6.1.20). High initial levels at both sites may relate to different sampling instruments used between early on, and later in the record. Conductivity in the Orowaiti River was variable, but both sites were closely related (b=0.778)(Figure 19.21), hence no real change in conductivity occurred over time.

6.1.6 Sawyers Creek

pH in Sawyers Creek seems to have gradually dropped towards neutral (pH7), but the number of data points for this time series was very low, therefore results should be viewed with some caution (Figure 6.1.22).

While trends in ammoniacal nitrogen at the reference site in Sawyers Creek appeared to have remained static, levels at the impact site showed a marked trend towards increase, supported by differences between sites (Figure 6.1.23). Temperature in Sawyers Creek showed a high degree of variability, but as there are so few data points, this is likely due to seasonality (Figure 6.1.24). Clarity in Sawyers creek too, is quite variable, and due to the small number of data points any trends should, once again, be viewed with caution (Figure 6.1.25).

Faecal coliform levels were quite stable at the reference site, and highly variable at the impact site. A trend towards reduced levels of faecal coliforms at the impact site was encouraging, but variability and few data points make this conclusion uncertain (Figure 6.1.26). Too few data points for dissolved oxygen and conductivity make it difficult to assume any reasonable trends exist for these parameters (Figure 6.1.27 and 6.1.28).

6.1.7 Baker Creek

Levels in pH for Baker creek appeared reasonably constant, with little evidence of any trending (Figure 6.1.29). There was a moderate correlation between data from both sites (b=0.604), and high values may have contributed to decreasing the regression for impact ammoniacal nitrogen, although it remains that impact levels dropped closer to those at the reference site in later years (Figure 6.1.30).

As with most other monitoring sites, temperatures in Baker Creek showed little difference between impact and reference sites (b=0.806)(Figure 6.1.31).

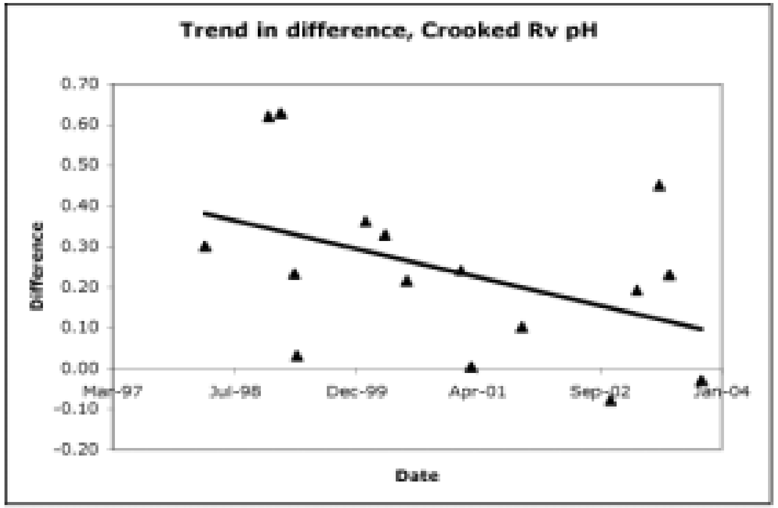
Clarity was highly variable (b=0.326) with the impact site having both higher and lower clarity compared to the reference site (Figure 6.1.32). Differences in faecal coliform levels were also highly variable (b=0.127), but displayed a possible trend toward improvement (Figure 6.1.33), as differences between sites declined. Conductivity was variable and showed no consistent pattern through time (Figure 6.1.34).

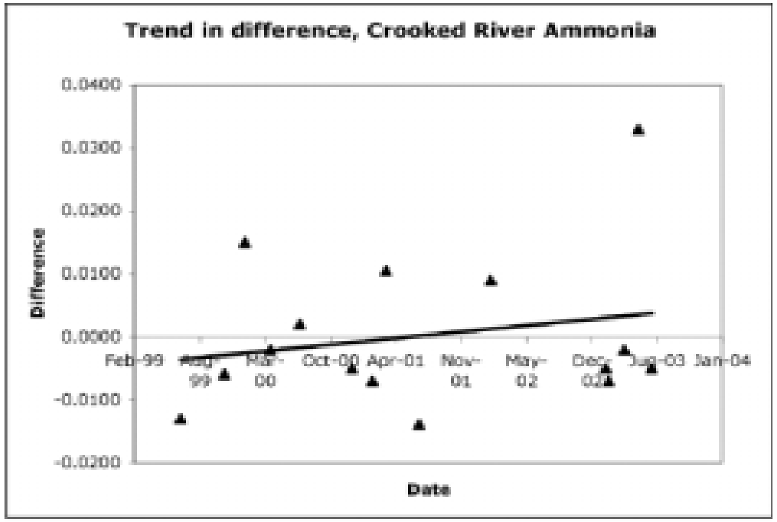
6.1.8 Grey River

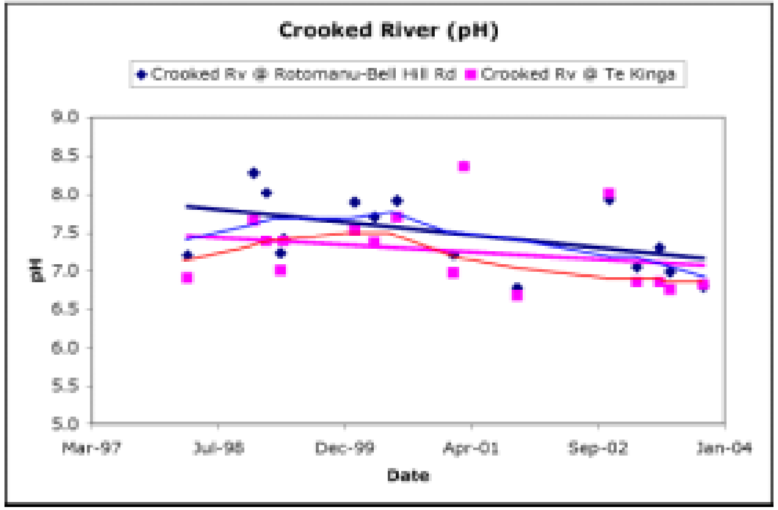
The Grey River had varied pH levels for a large river (Figure 6.1.35). The relationship between sites was close (b=0.786). Ammoniacal nitrogen levels in the Grey River remained reasonably constant over time (except for a very high reading in 2000), with no trends discernible at either impact or reference site (Figure 6.1.36).

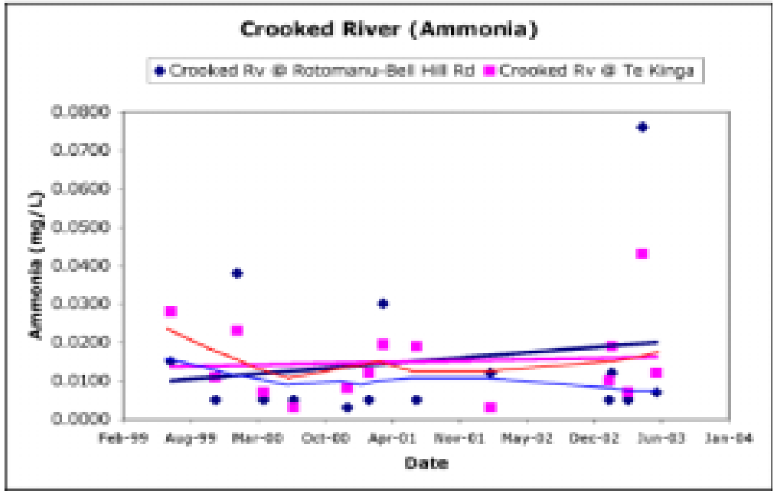
As for other paired sites, temperature was variable, within a reasonably narrow seasonal range, but closely related between reference and impact conditions (b=0.981)(Figure 6.1.37).

Faecal coliforms at the impact site came closer to those at the reference site. Coliform variations at this site were not well accounted for by natural processes, implied by the lack of impact/reference correlation (b=0.453)(Figure 6.1.38), but no trend was apparent. Neither was a trend apparent for dissolved oxygen or conductivity (Figure 6.1.39 and 6.1.40).

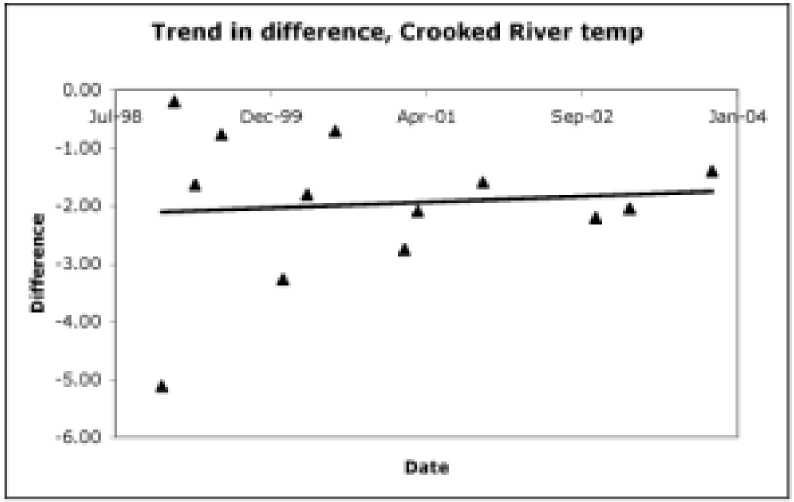


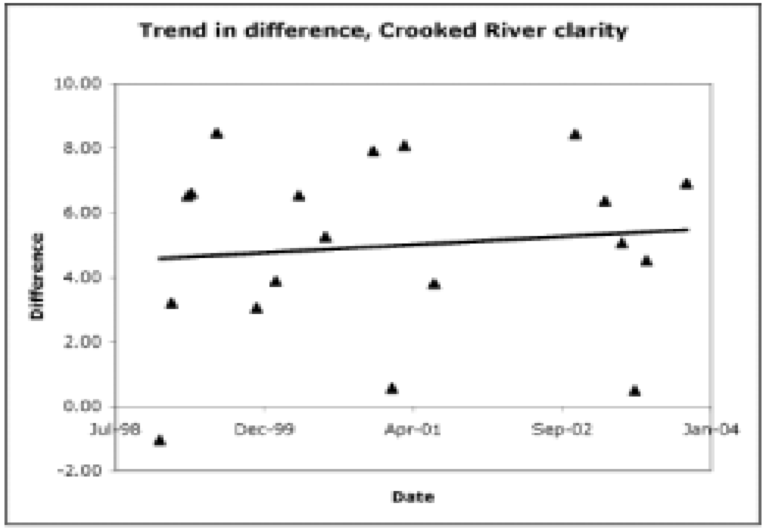


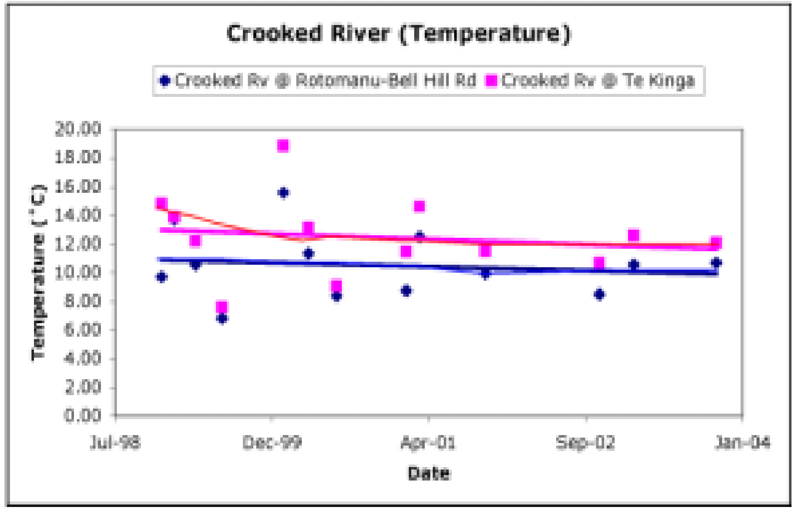


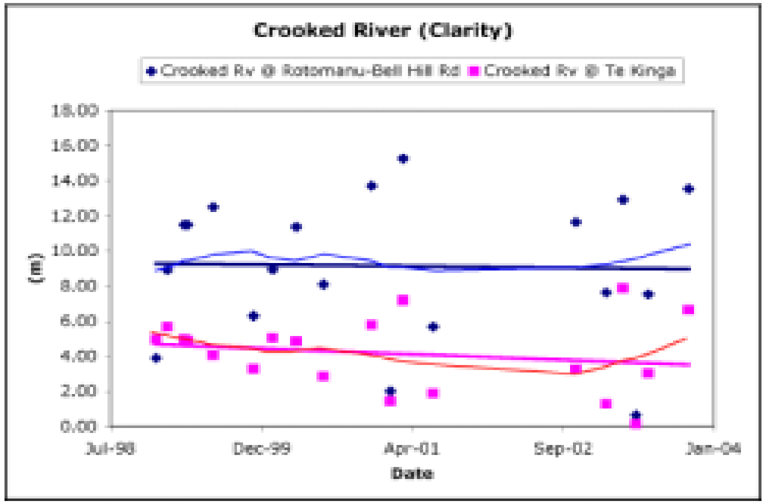


Figures 6.1.1 & 6.1.2 (left to right) Figure 6.1.1 - pH. Crooked River trend in differences between reference and impact sites and trends through time. Straight line regression and after LOWESS smoothing (crooked lines) are shown. Kendall's tau=0.806. Figure 6.1.2 - Ammoniacal nitrogen. Crooked River trend in differences between reference and impact sites and trends through time. Straight line regression and after LOWESS smoothing (crooked lines) are shown. Kendall's tau=0.568

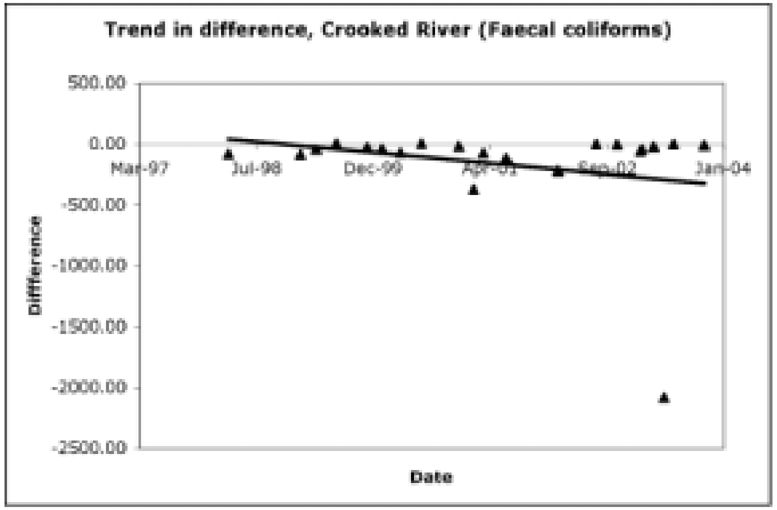


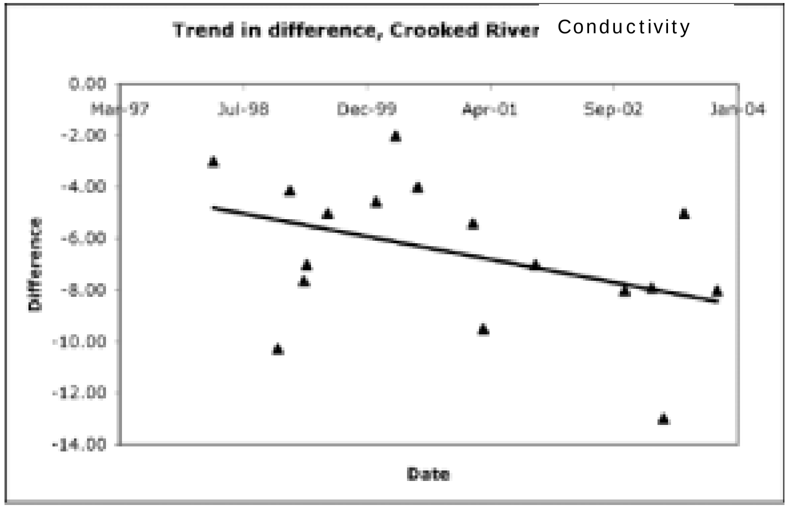


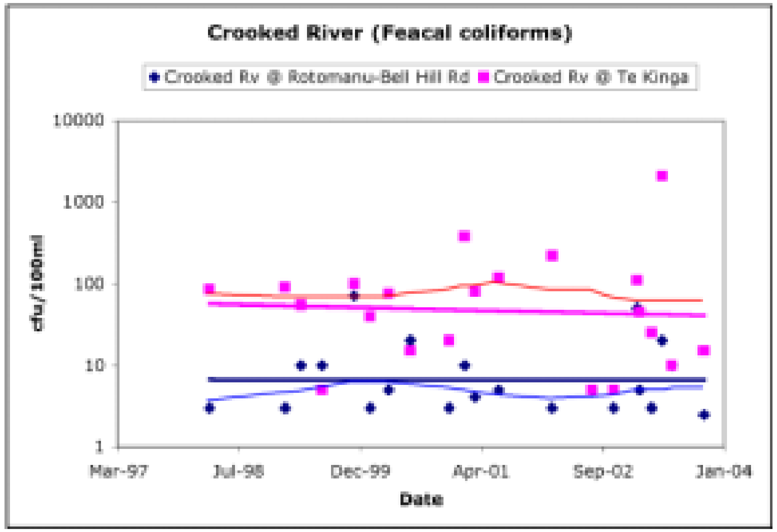


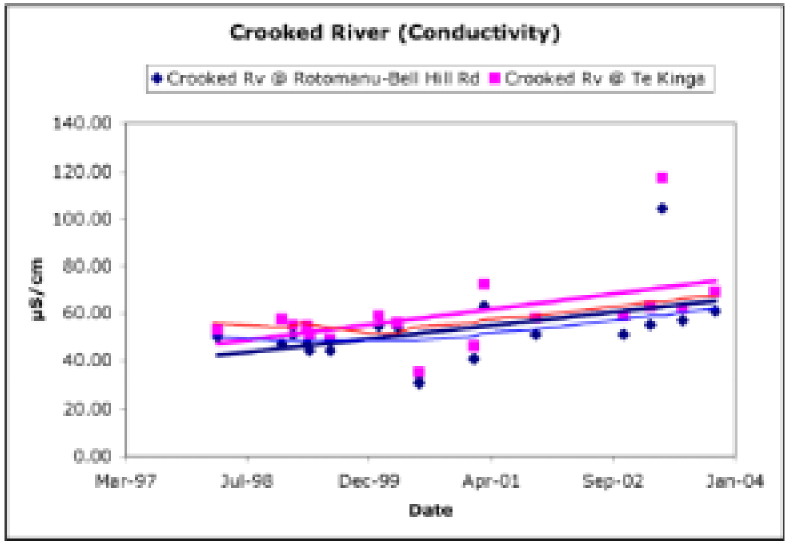


Figures 6.1.3 & 6.1.4 (left to right) Figure 6.1.3 - Temperature. Crooked River trend in differences between reference and impact sites and trends through time. Straight line regression and after LOWESS smoothing (crooked lines) are shown. Kendall's tau=0.736. Figure 6.1.4 - Clarity. Crooked River trend in differences between reference and impact sites and trends through time. Straight line regression and after LOWESS smoothing (crooked lines) are shown. Kendall's tau=0.551

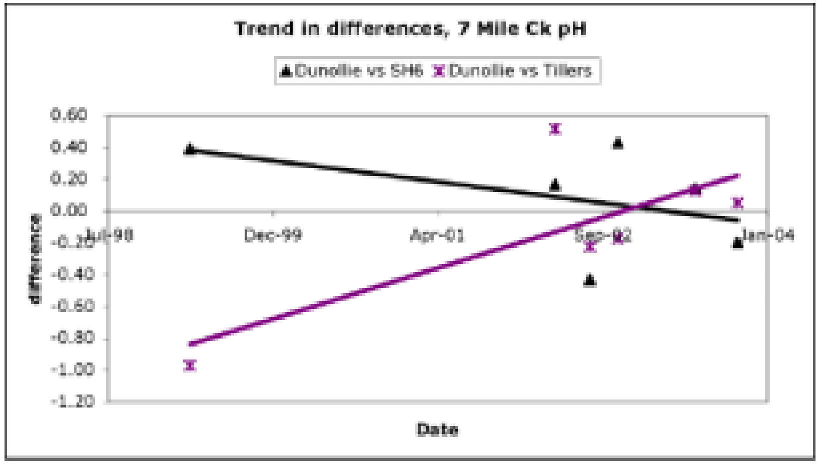


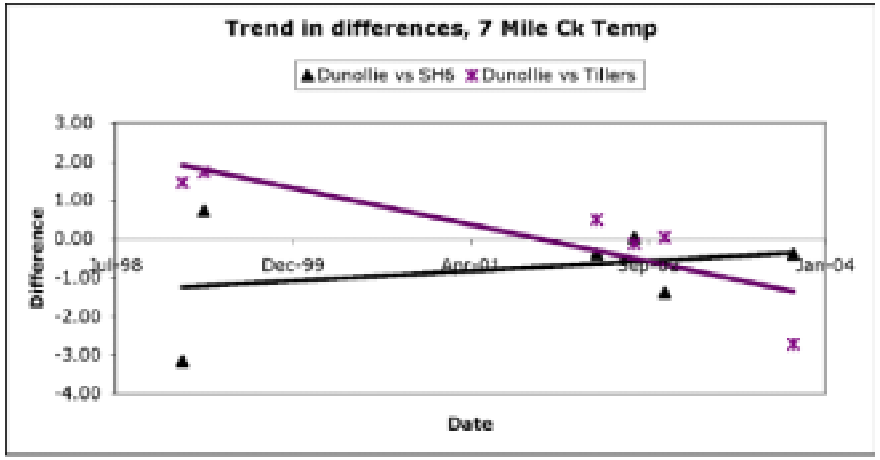


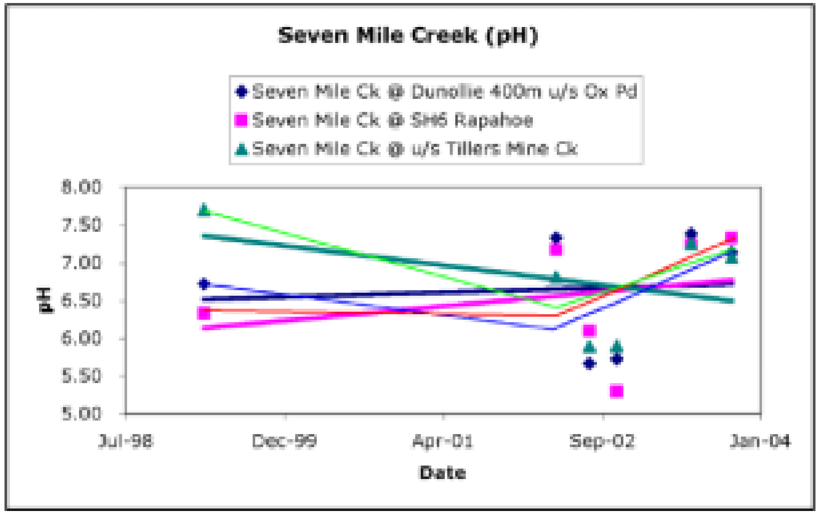


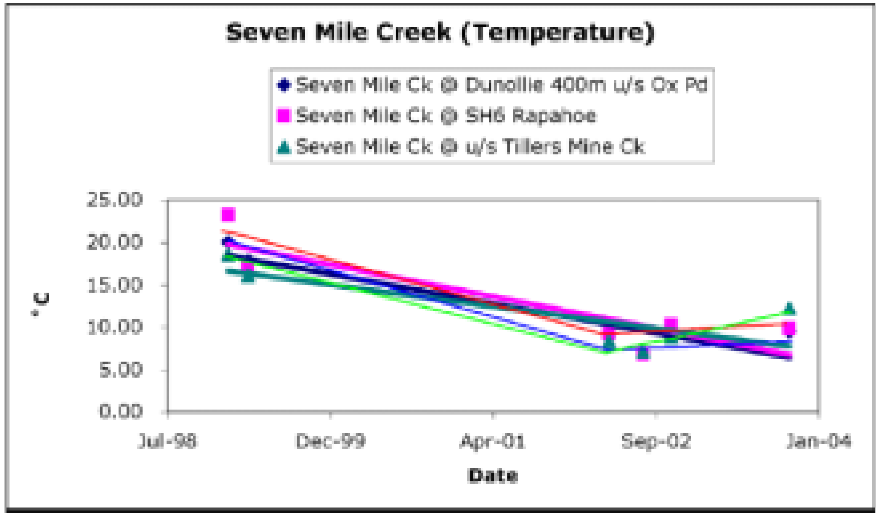


Figures 6.1.5 & 6.1.6 (left to right) Figure 6.1.5 - Faecal coliforms. Crooked River trend in differences between reference and impact sites and trends through time. Straight line regression and after LOWESS smoothing (crooked lines) are shown. Kendall's tau=0.145. Figure 6.1.6 - Conductivity. Crooked River trend in differences between reference and impact sites and trends through time. Straight line regression and after LOWESS smoothing (crooked lines) are shown. Kendall's tau=0.881

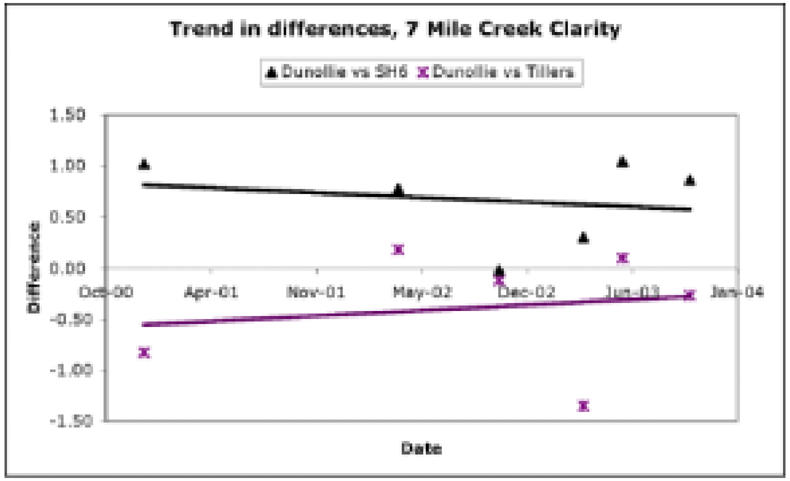


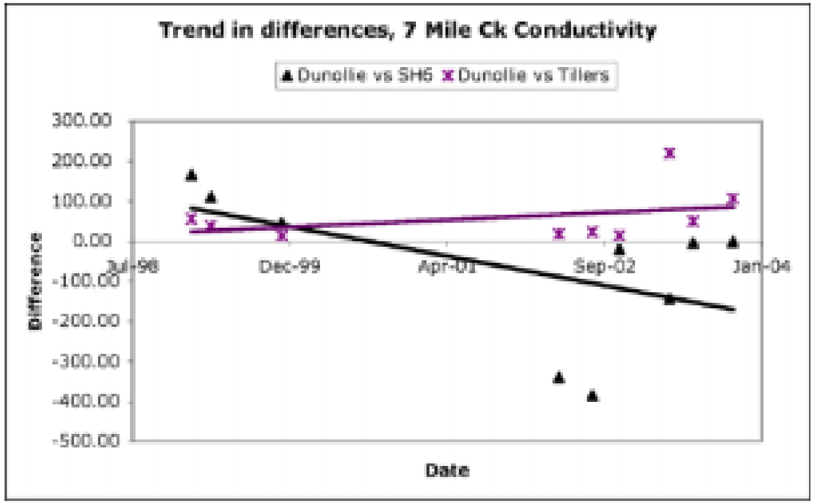


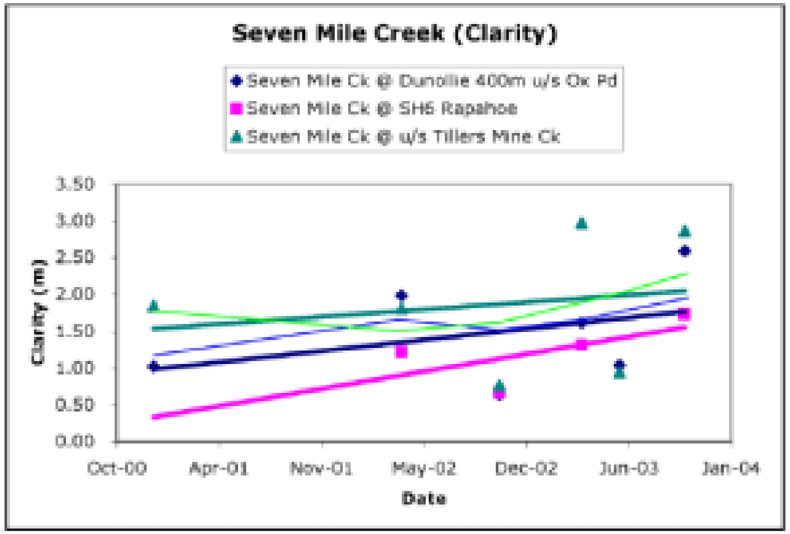


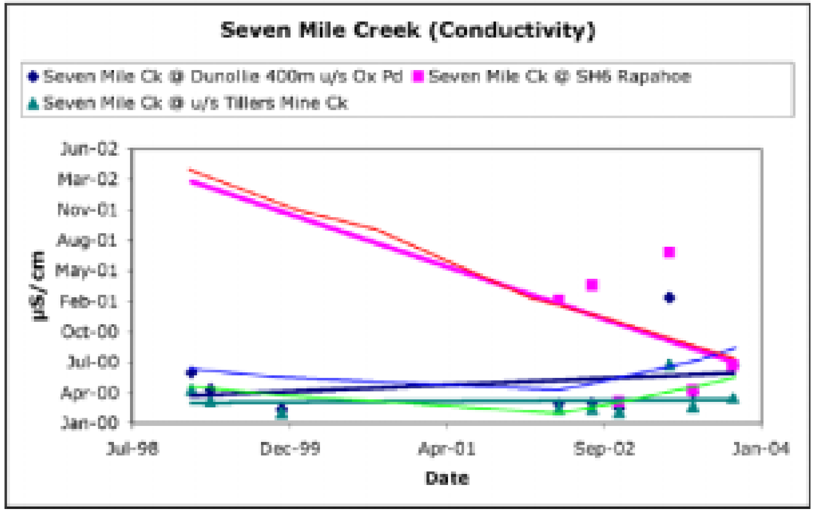


Figures 6.1.7 & 6.1.8 (left to right) Figure 6.1.7 - pH. Seven Mile Creek trend in differences between reference and impact sites and trends through time. Straight line regression and after LOWESS smoothing (crooked lines) are shown. Kendall's tau for Dunollie vs Rapahoe = 0.600, Dunollie vs Tillers Mine = 0.467, Rapahoe vs Tillers Mine = 0.333. Figure 6.1.8 - Temperature. Seven Mile Creek trend in differences between reference and impact sites and trends through time. Straight line regression and after LOWESS smoothing (crooked lines) are shown. Kendall's tau for Dunollie vs Rapahoe = 0.867, Dunollie vs Tillers Mine = 1.00, Rapahoe vs Tillers Mine = 0.867.

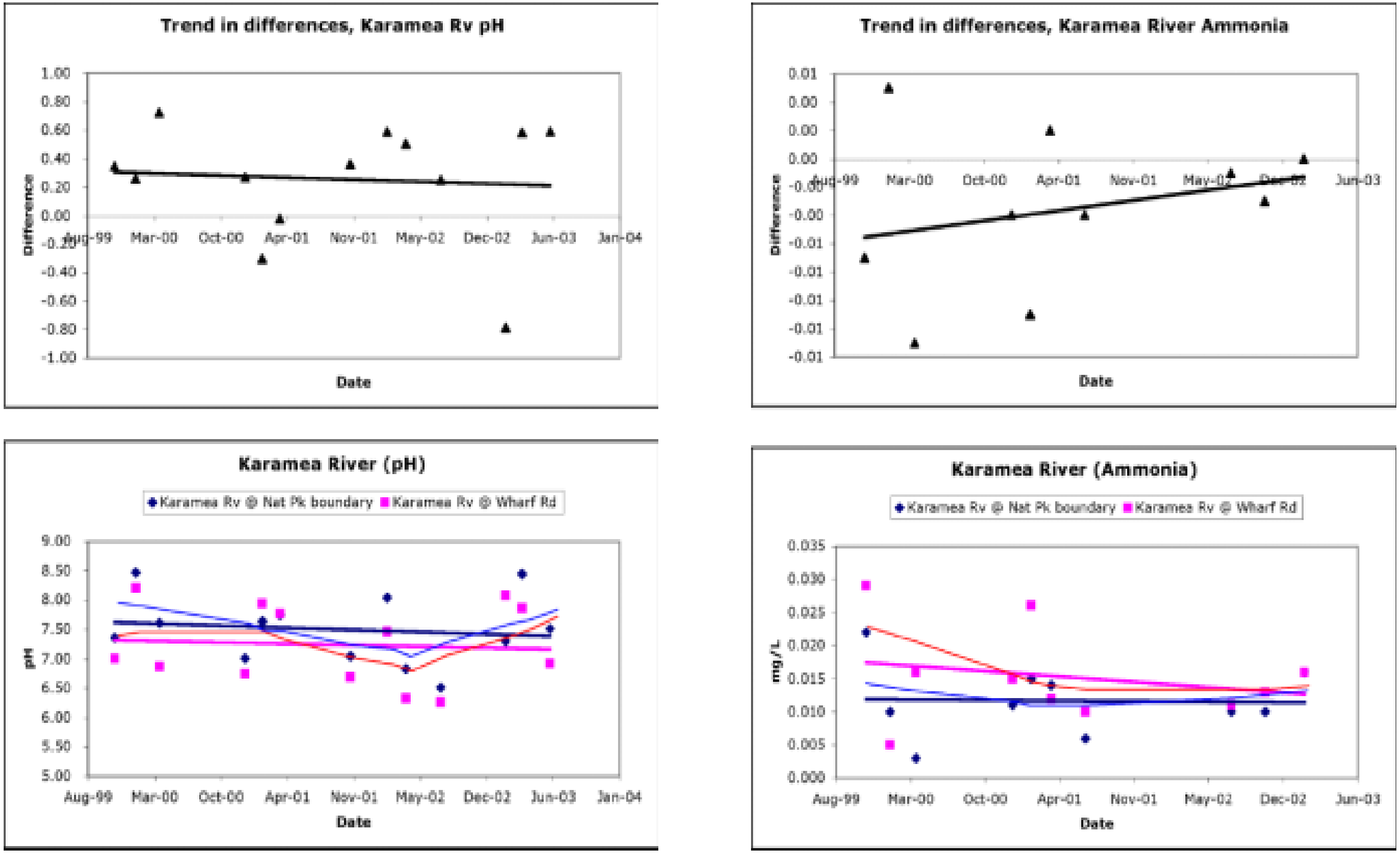




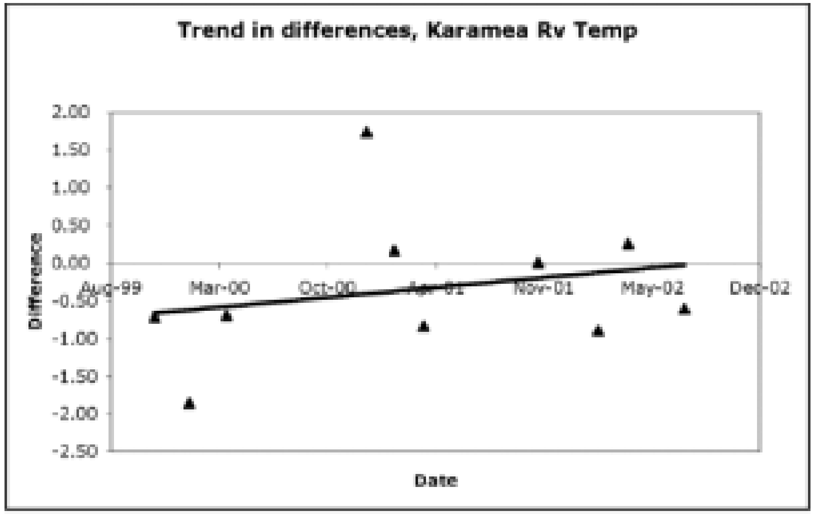


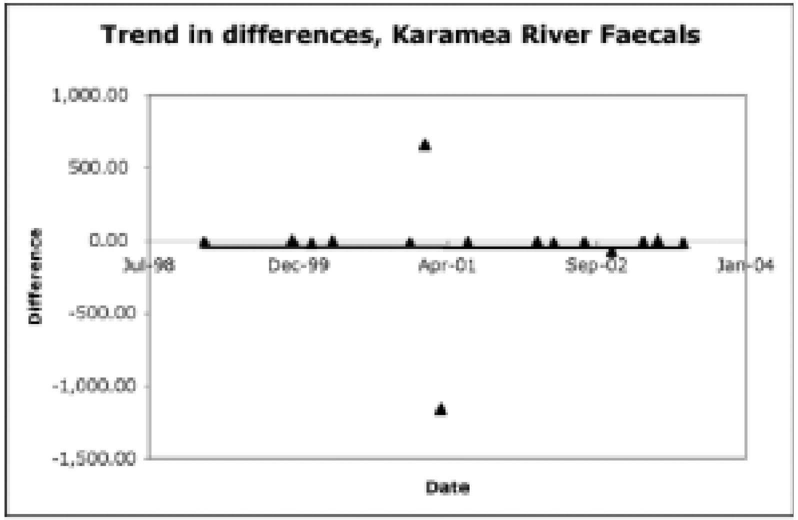


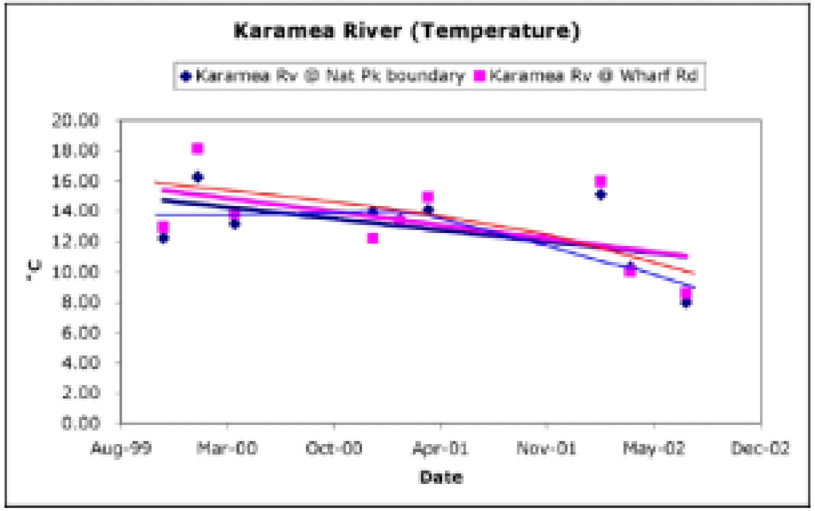
Figures 6.1.9 & 6.1.10 (left to right) Figure 6.1.9 - Clarity. Seven Mile Creek trend in differences between reference and impact sites and trends through time. Straight line regression and after LOWESS smoothing (crooked lines) are shown. Kendall's tau for Dunollie vs Rapahoe = 0.667, Dunollie vs Tillers Mine = 0.467, Rapahoe vs Tillers Mine = 0.667.Figure 6.1.10 - Conductivity. Seven Mile Creek trend in differences between reference and impact sites and trends through time. Straight line regression and after LOWESS smoothing (crooked lines) are shown. Kendall's tau for Dunollie vs Rapahoe = 0.467, Dunollie vs Tillers Mine = 0.889, Rapahoe vs Tillers Mine = 0.333.

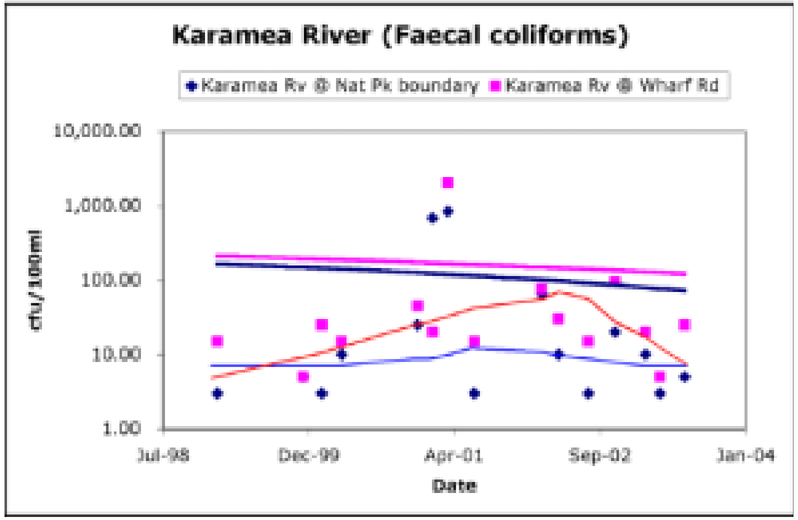


Figures 6.1.11 & 6.1.12 (left to right) Figure 6.1.11 - pH. Karamea River trend in differences between reference and impact sites and trends through time. Straight line regression and after LOWESS smoothing (crooked lines) are shown. Kendall's tau=0.615. Figure 6.1.12 - Ammoniacal nitrogen. Karamea River trend in differences between reference and impact sites and trends through time. Straight line regression and after LOWESS smoothing (crooked lines) are shown. Kendall's tau=0.489.

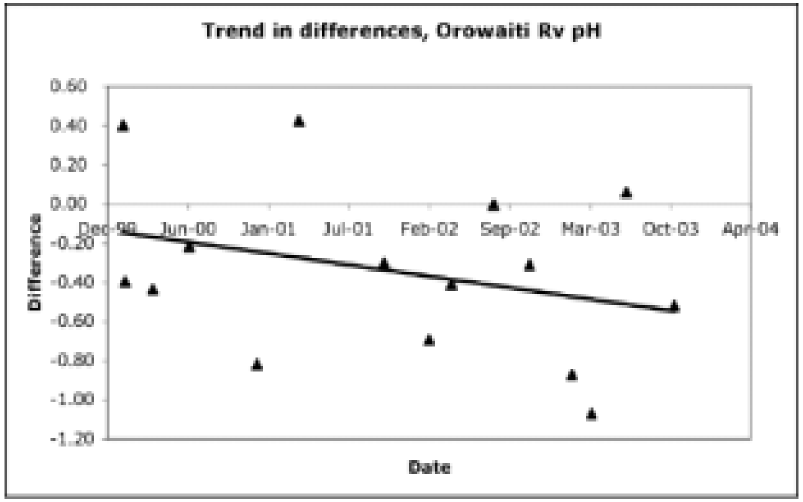


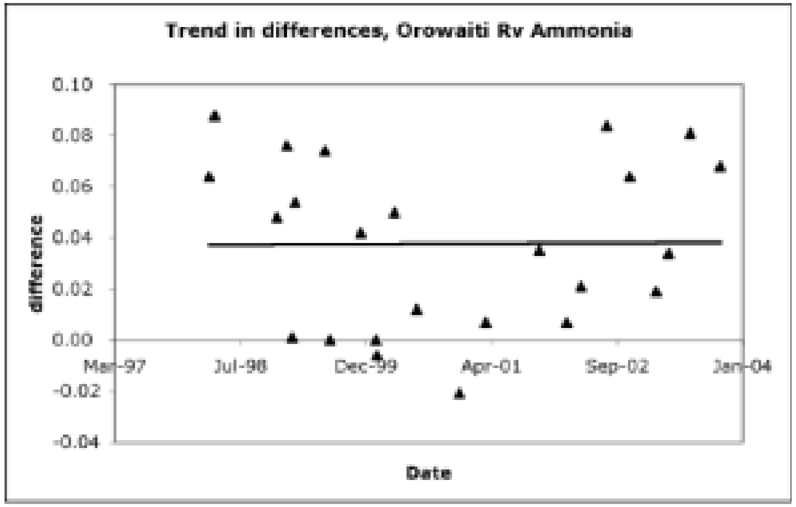


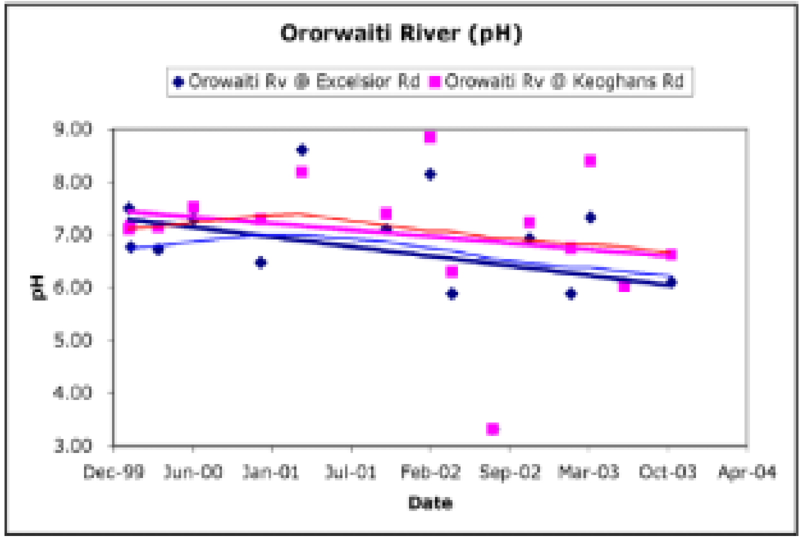


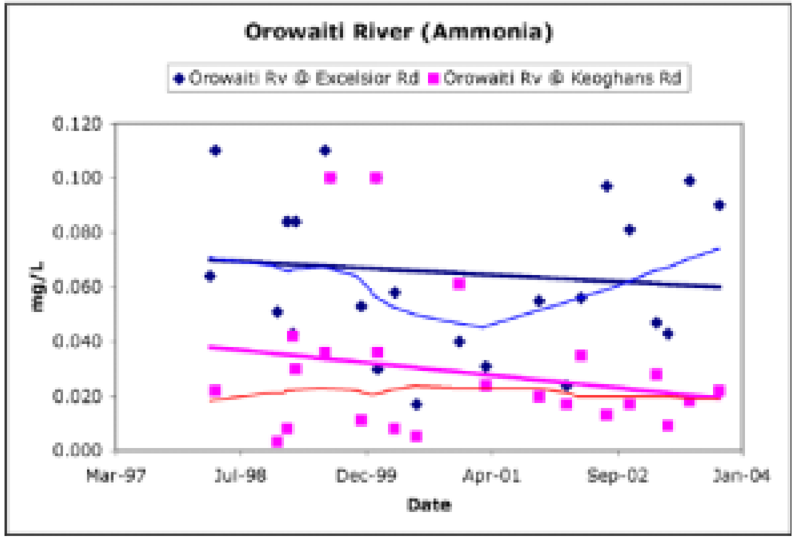


Figures 6.1.13 & 6.1.14 (left to right) Figure 6.1.13 Temperature. Karamea River trend in differences between reference and impact sites and trends through time. Straight line regression and after LOWESS smoothing (crooked lines) are shown. Kendall's tau=0.822. Figure 6.1.14 - Faecal coliforms. Karamea River trend in differences between reference and impact sites and trends through time. Straight line regression and after LOWESS smoothing (crooked lines) are shown. Kendall's tau=0.491.

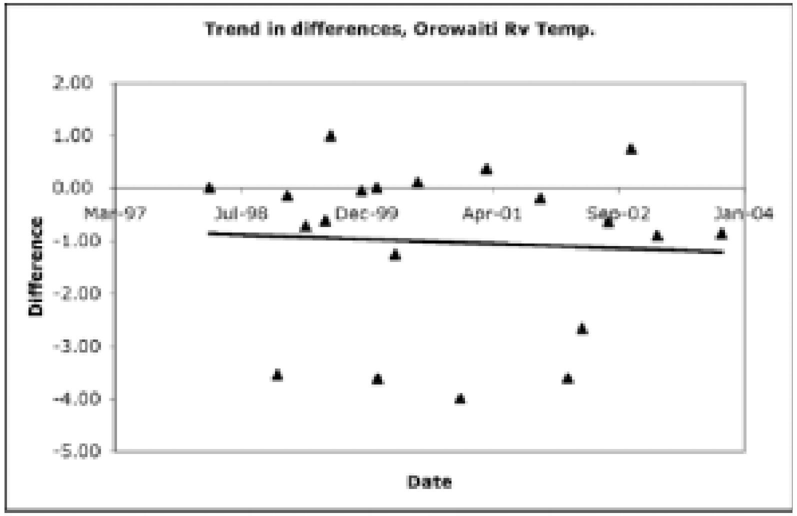


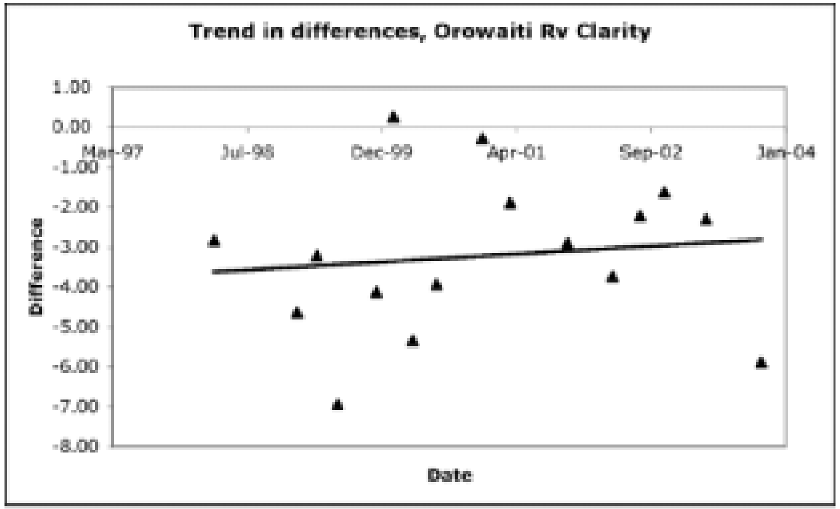


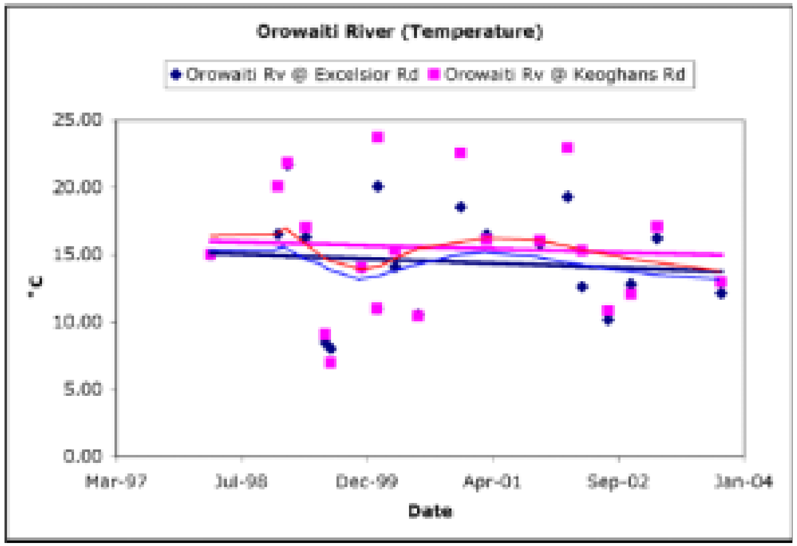


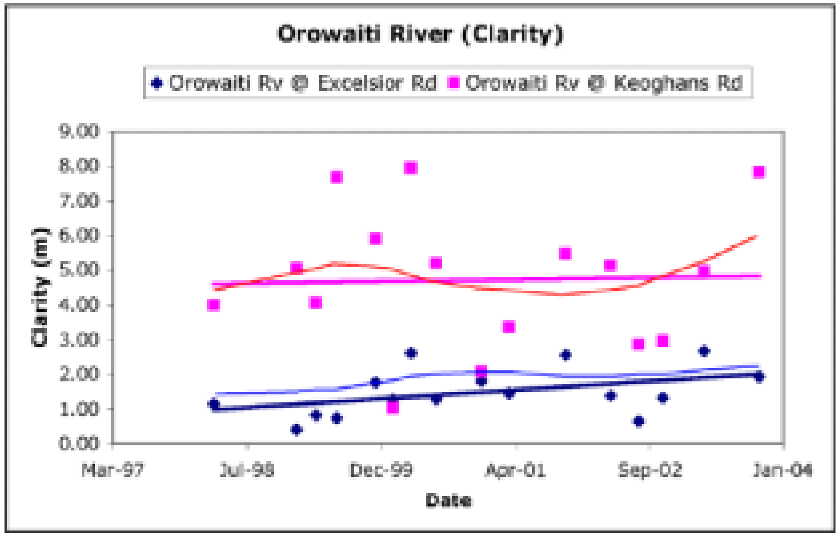


Figures 6.1.15 & 6.1.16 (left to right) Figure 6.1.15 - pH. Orowaiti River trend in differences between reference and impact sites and trends through time. Straight line regression and after LOWESS smoothing (crooked lines) are shown. Kendall's tau=0.708. Figure 6.1.16 - Ammoniacal nitrogen. Orowaiti River trend in differences between reference and impact sites and trends through time. Straight line regression and after LOWESS smoothing (crooked lines) are shown. Kendall's tau=0.157.

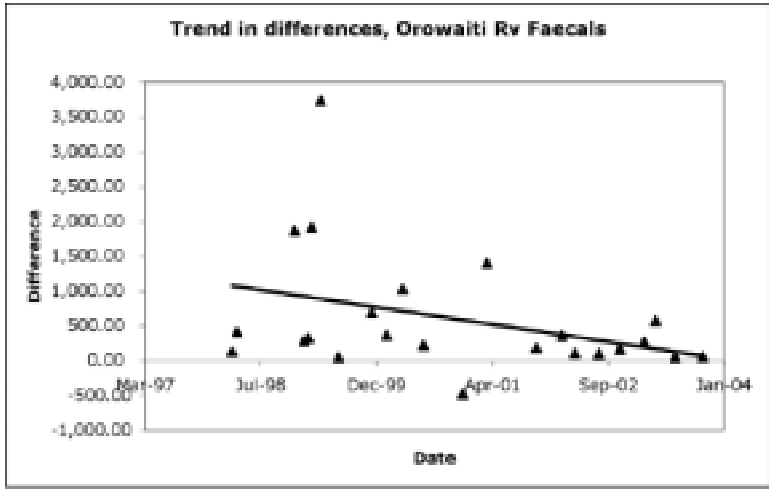


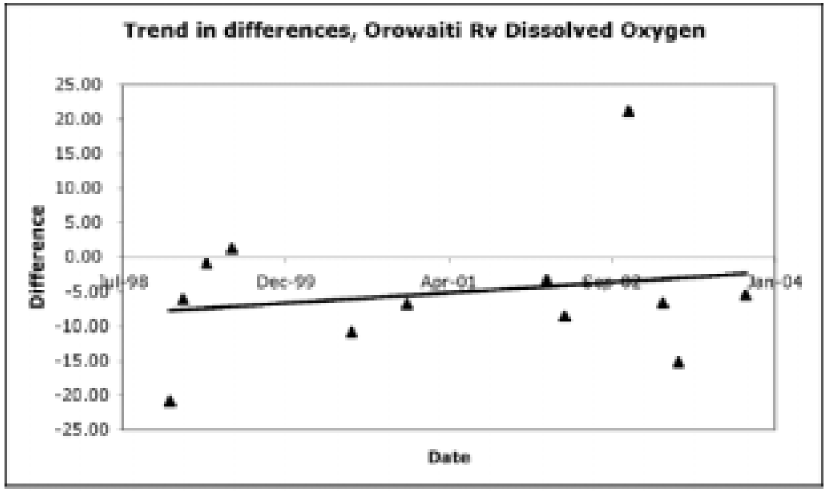


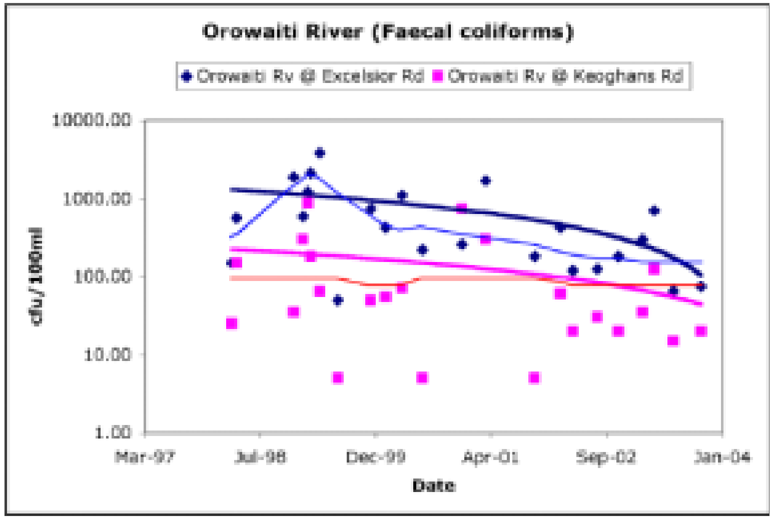


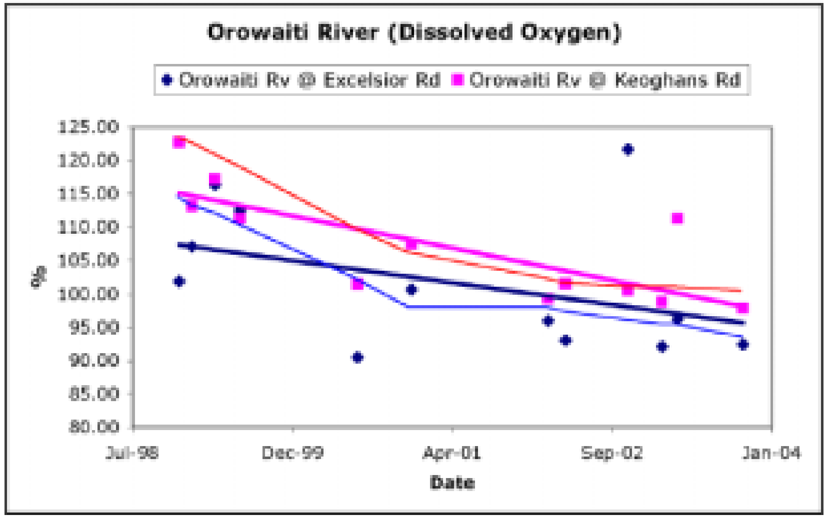


Figures 6.1.17 & 6.1.18 (left to right) Figure 6.1.17 - Temperature. Orowaiti River trend in differences between reference and impact sites and trends through time. Straight line regression and after LOWESS smoothing (crooked lines) are shown. Kendall's tau=0.875. Figure 6.1.18 - Clarity. Orowaiti River trend in differences between reference and impact sites and trends through time. Straight line regression and after LOWESS smoothing (crooked lines) are shown. Kendall's tau=0.200.

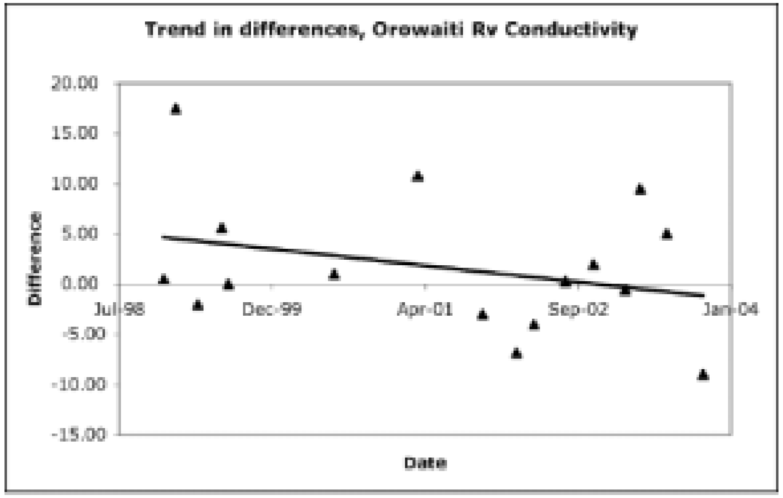


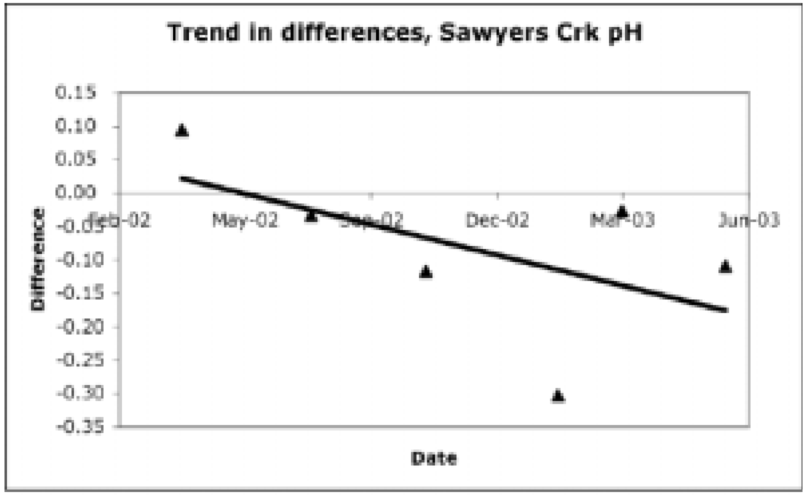


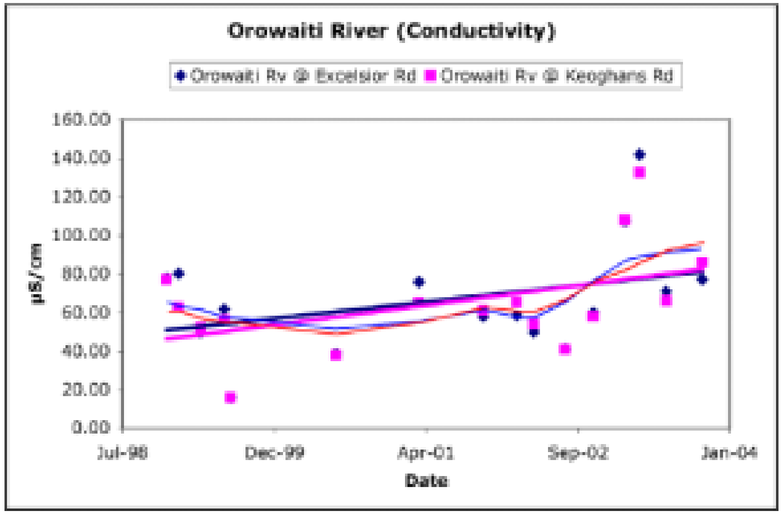


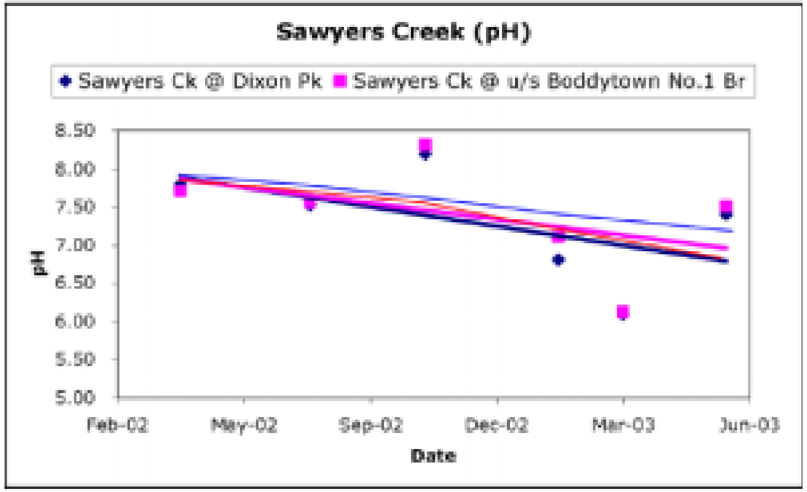


Figures 6.1.19 & 6.1.20 (left to right) Figure 6.1.19 - Faecal coliforms. Orowaiti River trend in differences between reference and impact sites and trends through time. Straight line regression and after LOWESS smoothing (crooked lines) are shown. Kendall's tau=0.528. Figure 6.1.20 - Dissolved oxygen. Orowaiti River trend in differences between reference and impact sites and trends through time. Straight line regression and after LOWESS smoothing (crooked lines) are shown. Kendall's tau=0.455.

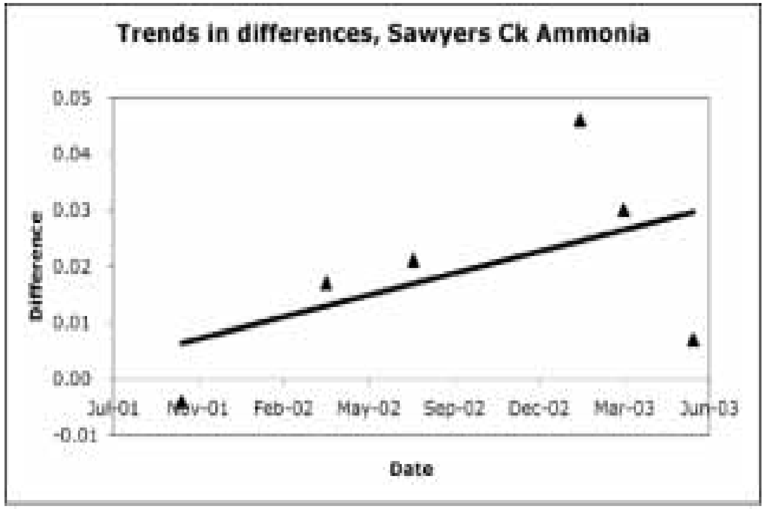


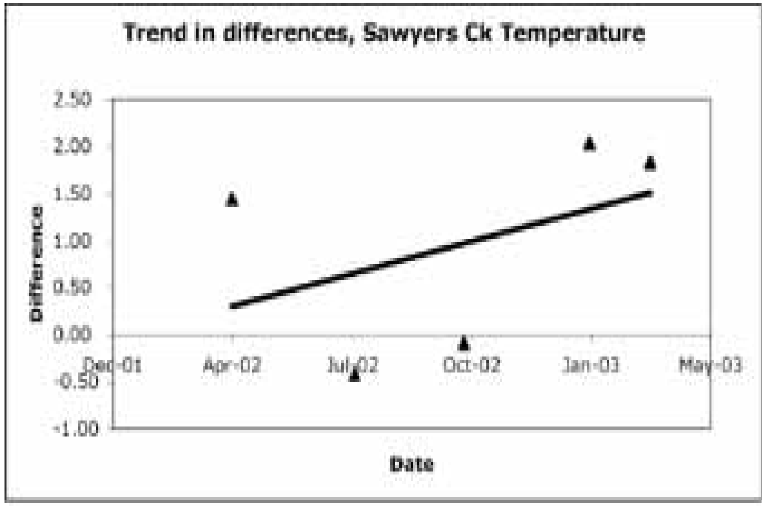


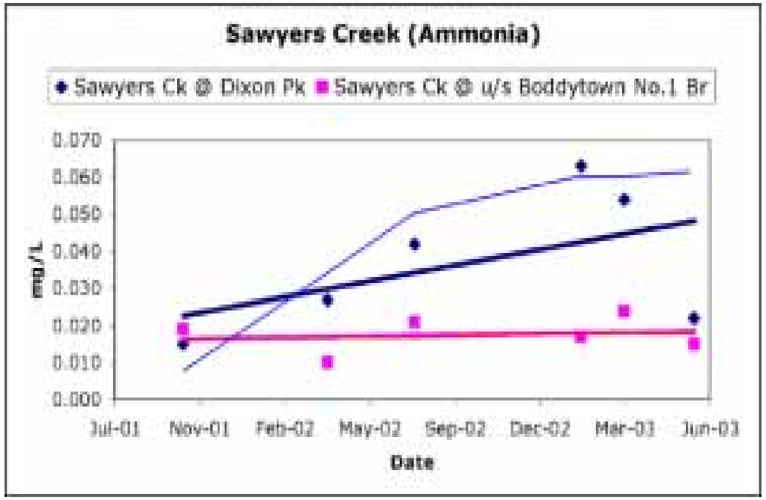


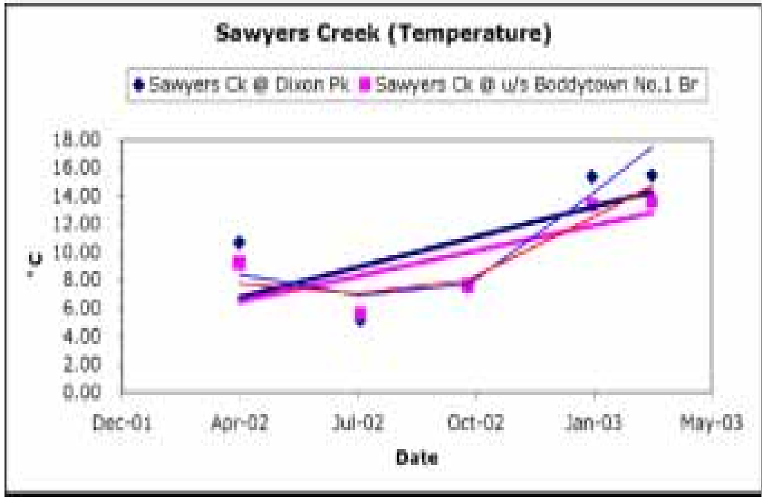


Figures 6.1.21 & 422 (left to right) Figure 6.1.21 - Conductivity. Orowaiti River trend in differences between reference and impact sites and trends through time. Straight line regression and after LOWESS smoothing (crooked lines) are shown. Kendall's tau=0.778. Figure 6.1.22 - pH. Sawyers Creek trend in differences between reference and impact sites and trends through time. Straight line regression and after LOWESS smoothing (crooked lines) are shown. Kendall's tau=1.00.

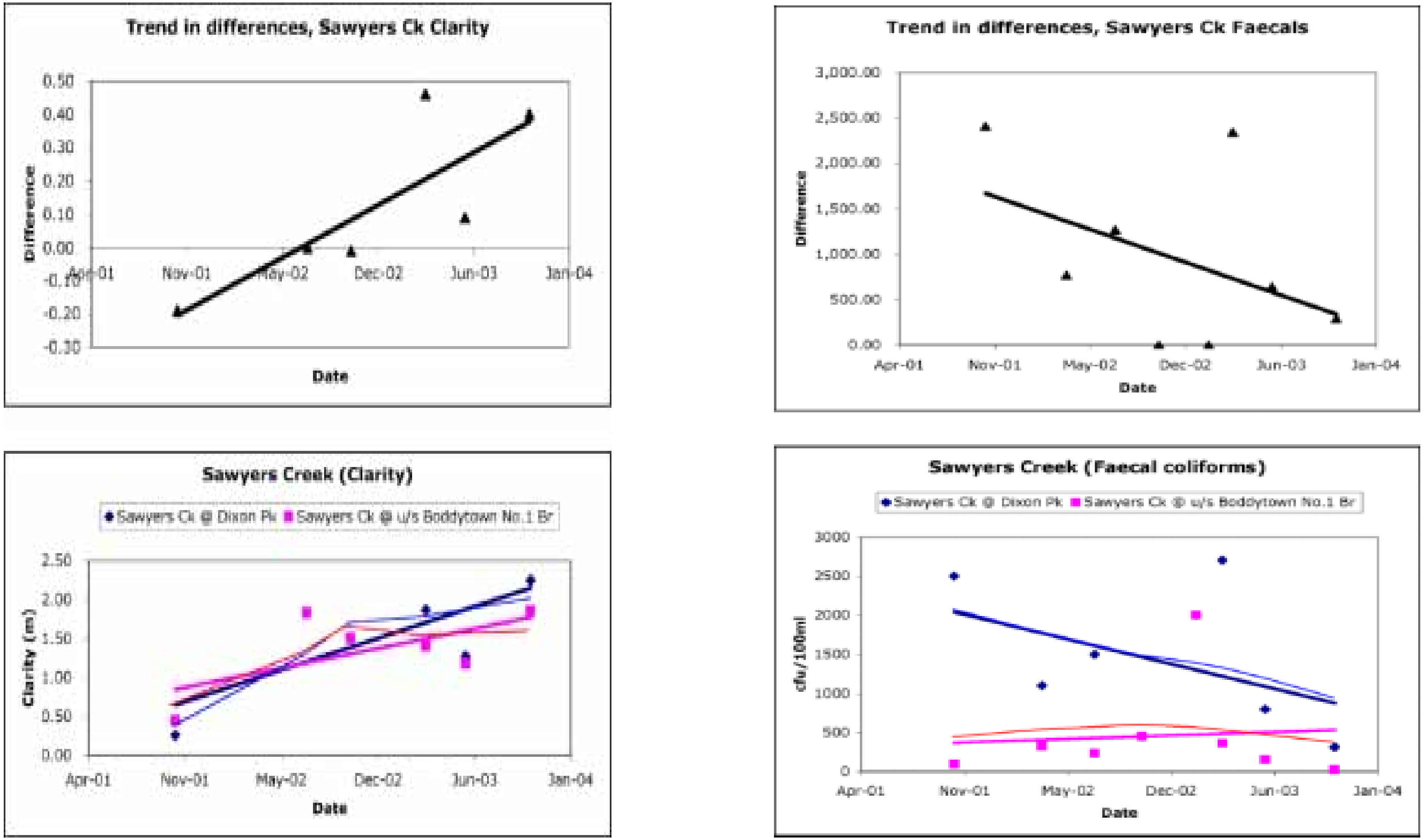




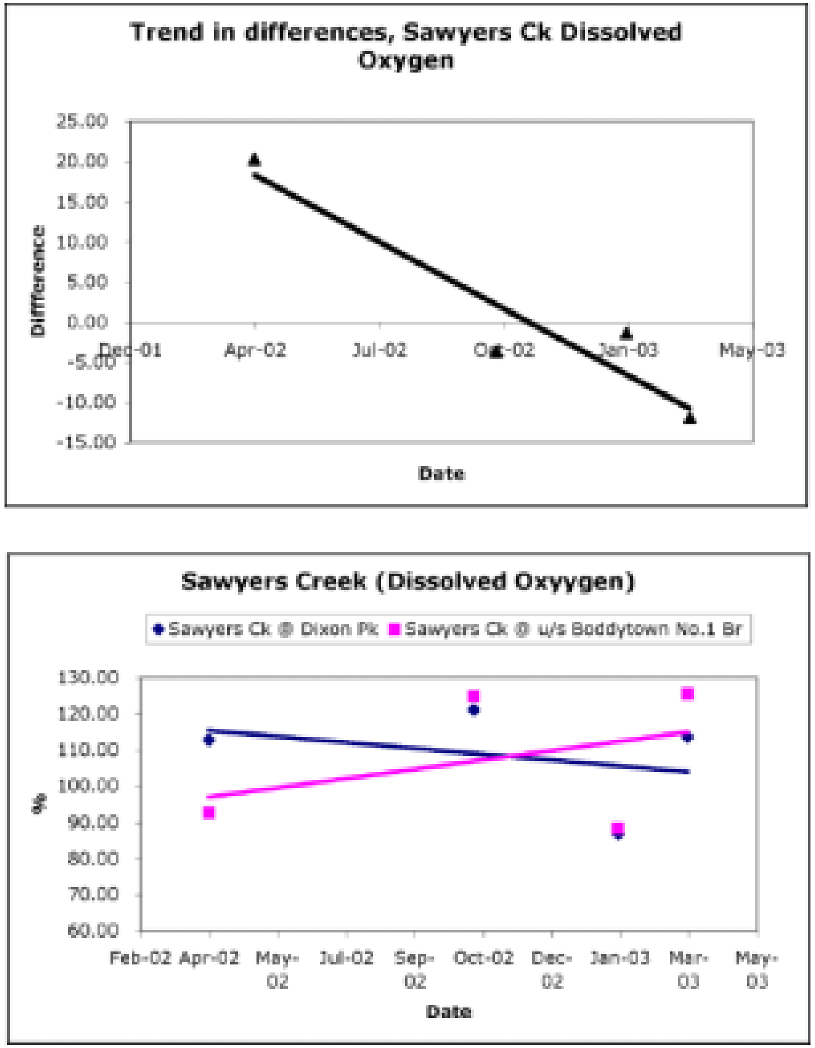


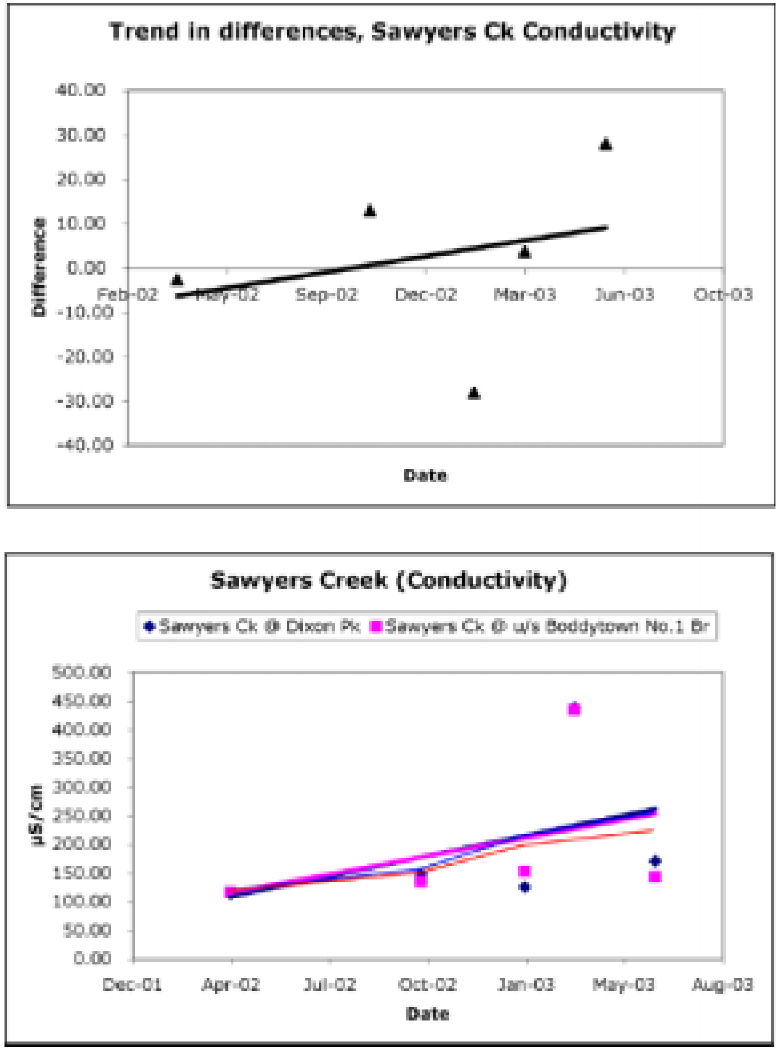


Figures 6.1.23 & 6.1.24 (left to right) Figure 6.1.23 - Ammoniacal nitrogen. Sawyers Creek trend in differences between reference and impact sites and trends through time. Straight line regression and after LOWESS smoothing (crooked lines) are shown. Kendall's tau=0.200. Figure 6.1.24 - Temperature. Sawyers Creek trend in differences between reference and impact sites and trends through time. Straight line regression and after LOWESS smoothing (crooked lines) are shown. Kendall's tau=1.00.

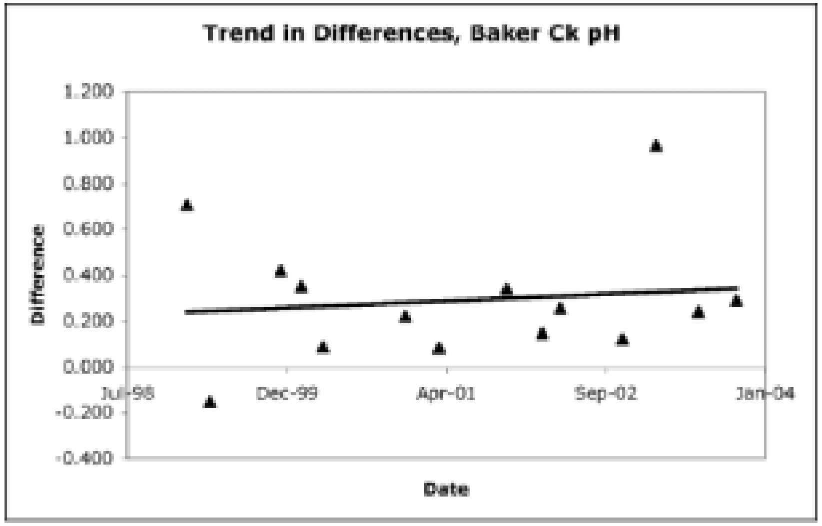


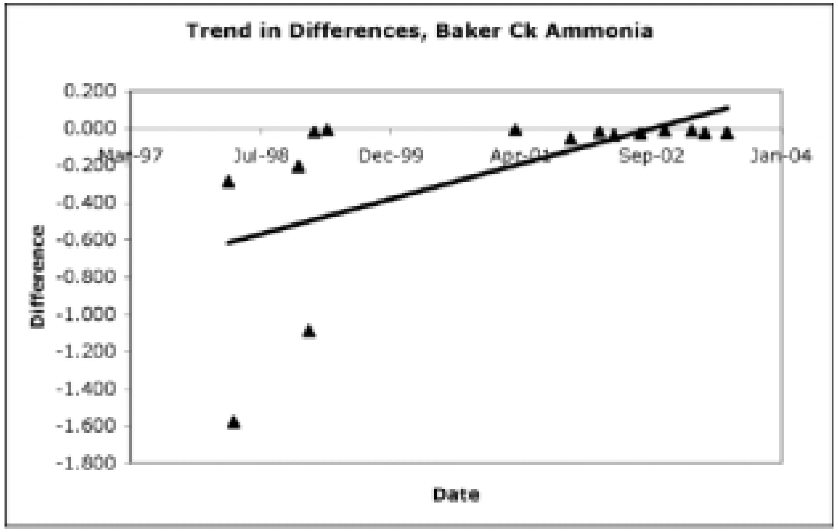
Figures 6.1.25 & 6.1.26 (left to right) Figure 6.1.25 - Clarity. Sawyers Creek trend in differences between reference and impact sites and trends through time. Straight line regression and after LOWESS smoothing (crooked lines) are shown. Kendall's tau=0.733. Figure 6.1.26 - Faecal coliforms. Sawyers Creek trend in differences between reference and impact sites and trends through time. Straight line regression and after LOWESS smoothing (crooked lines) are shown. Kendall's tau=0.215

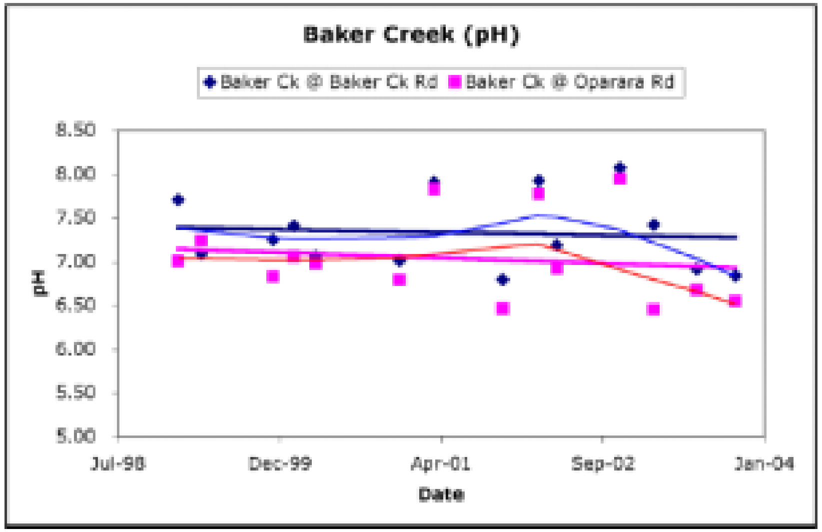


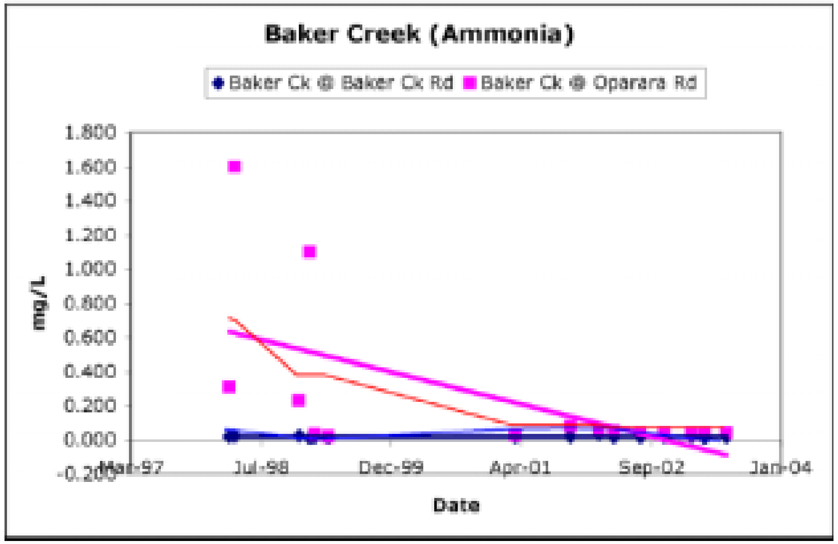


Figures 6.1.27 & 6.1.28 (left to right) Figure 6.1.27 - Dissolved oxygen. Sawyers Creek trend in differences between reference and impact sites and trends through time. Straight line regressions and are shown. Insufficient data for LOWESS smoothing. Kendall's tau=0.667. Figure 6.1.28 - Conductivity. Sawyers Creek trend in differences between reference and impact sites and trends through time. Straight line regression and after LOWESS smoothing (crooked lines) are shown. Kendall's tau=0.600.

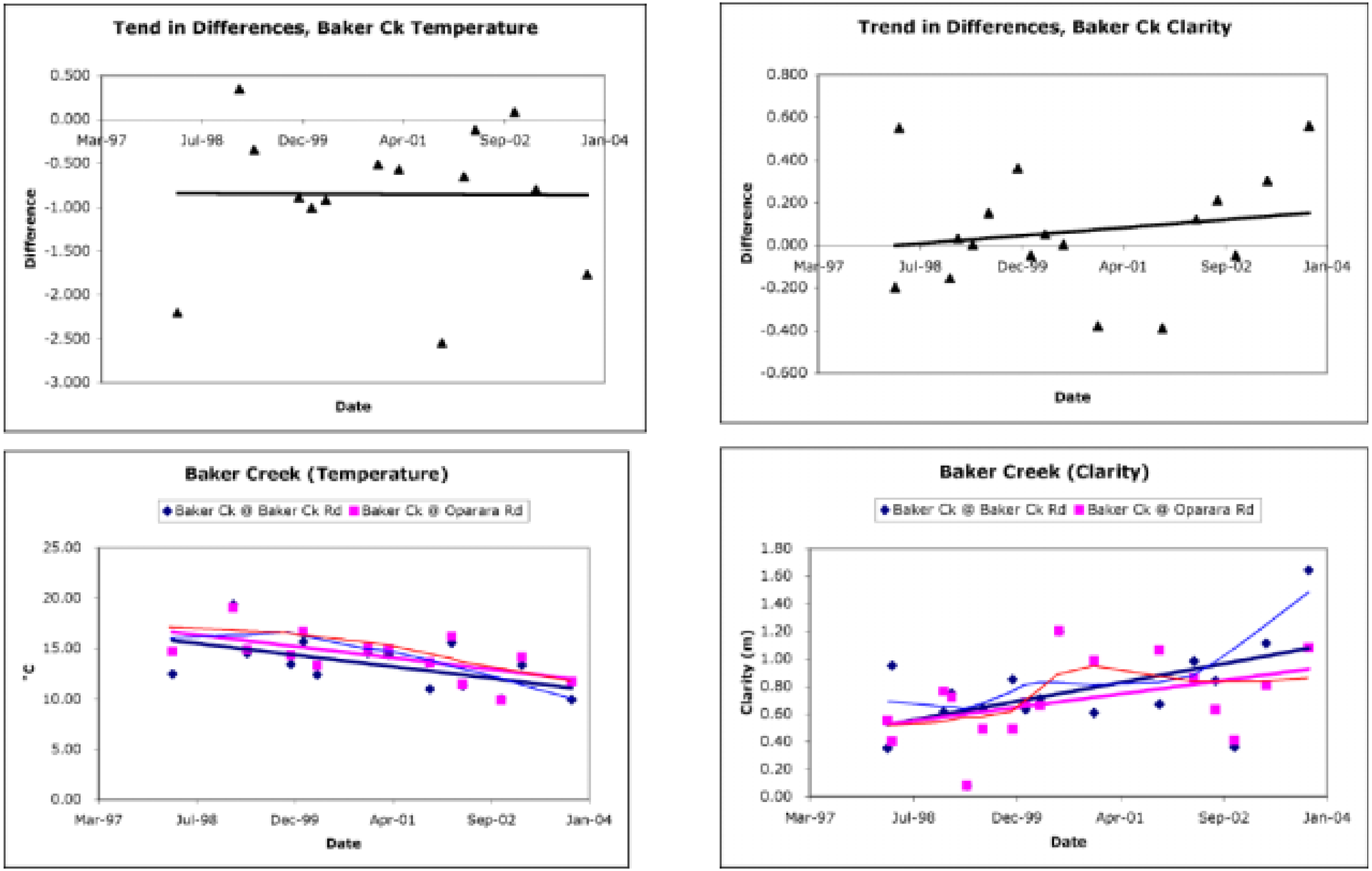




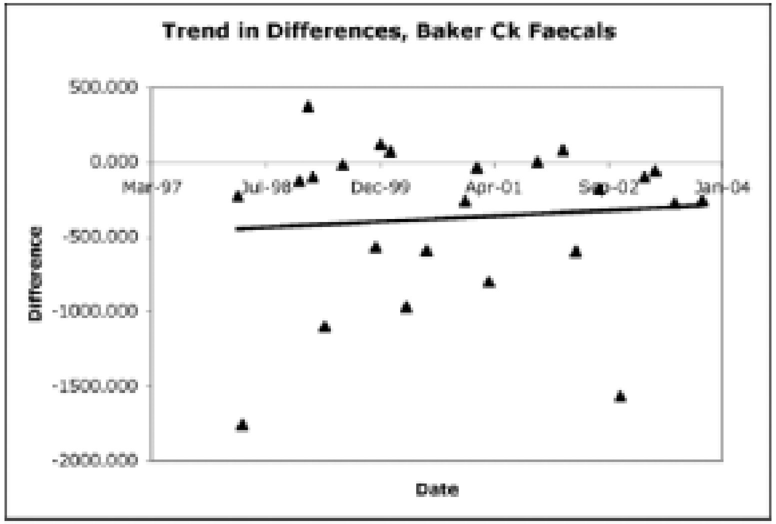


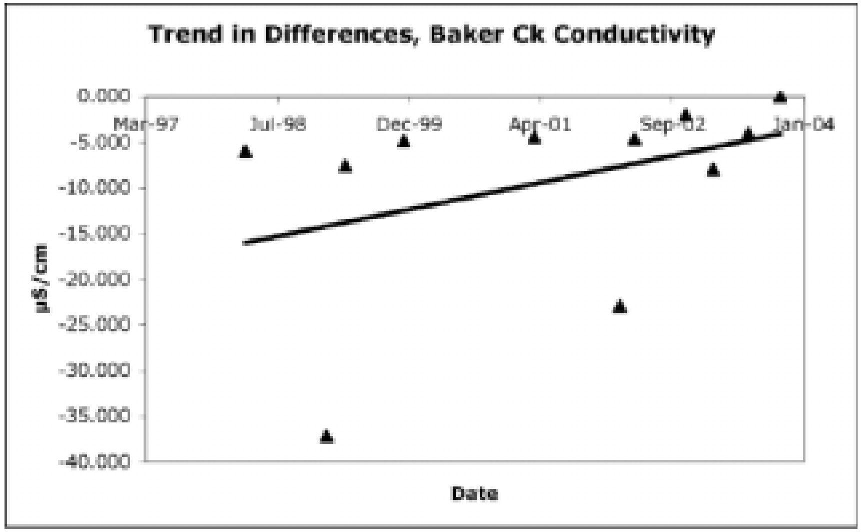


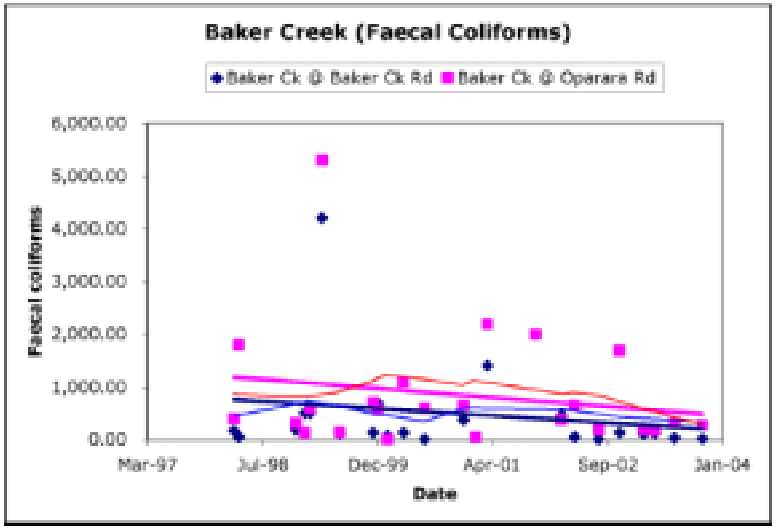
Figures 6.1.29 & 6.1.30 (left to right) Figure 6.1.29 - pH. Baker Creek trend in differences between reference and impact sites and trends through time. Straight line regression and after LOWESS smoothing (crooked lines) are shown. Kendall's tau=0.605. Figure 6.1.30 - Ammoniacal nitrogen. Baker Creek trend in differences between reference and impact sites and trends through time. Straight line regression and after LOWESS smoothing (crooked lines) are shown. Kendall's tau=0.312.

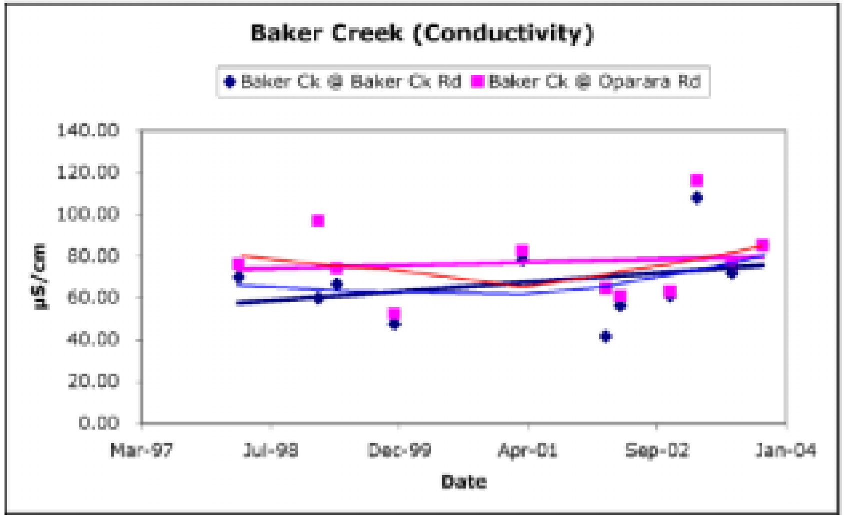


Figures 6.1.31 & 6.1.32 (left to right) Figure 6.1.31 - Temperature. Baker Creek trend in differences between reference and impact sites and trends through time. Straight line regression and after LOWESS smoothing (crooked lines) are shown. Kendall's tau=0.846. Figure 6.1.32 - Clarity. Baker Creek trend in differences between reference and impact sites and trends through time. Straight line regression and after LOWESS smoothing (crooked lines) are shown. Kendall's tau=0.326.

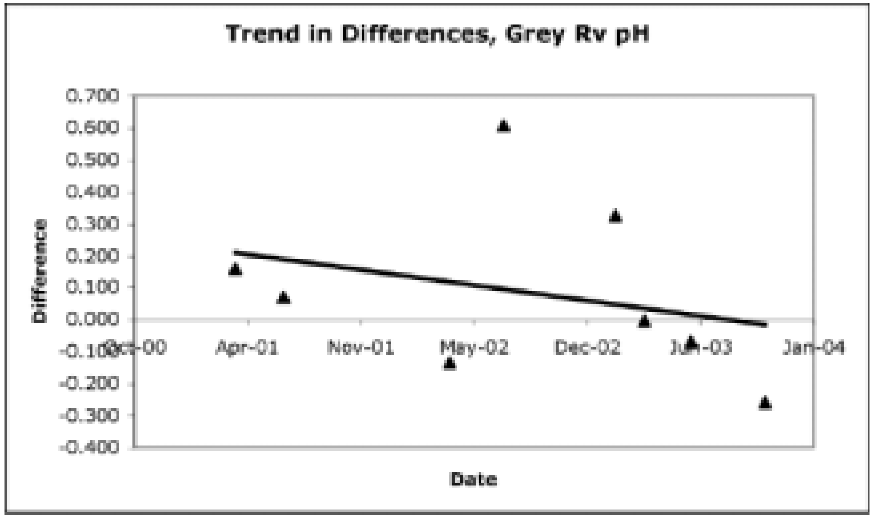


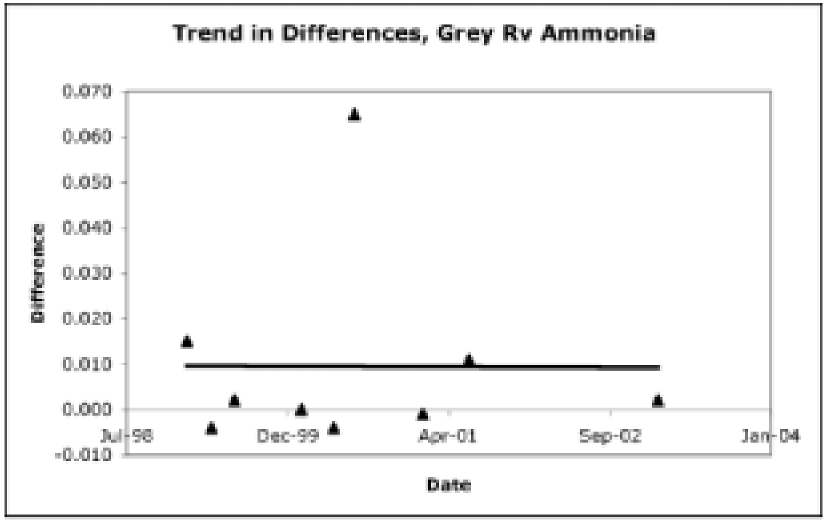


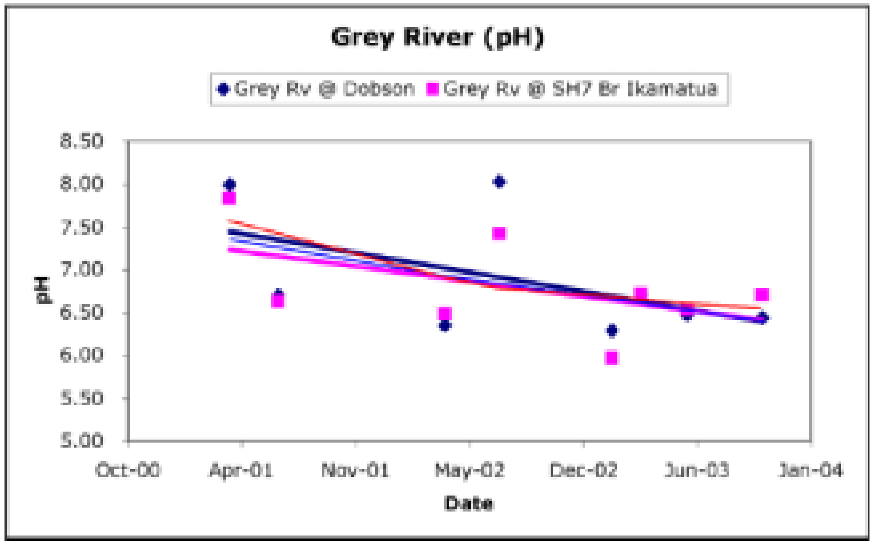


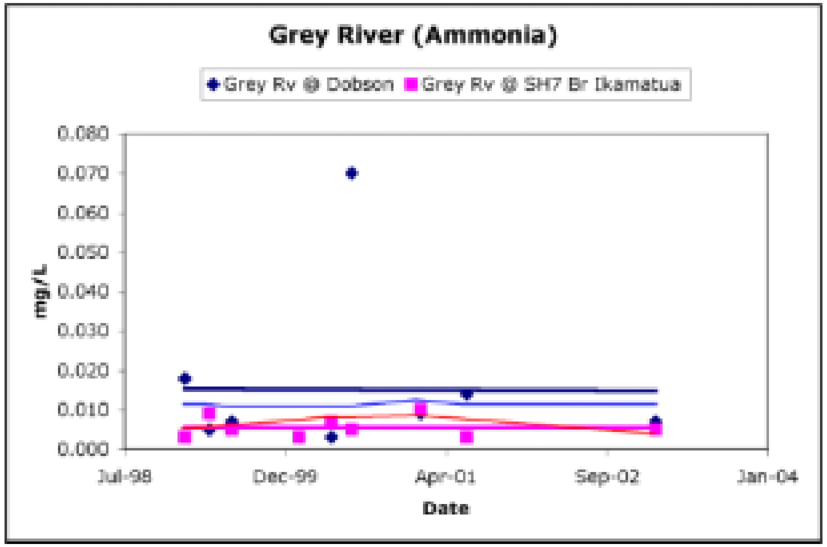


Figures 6.1.33 & 6.1.34 (left to right) Figure 6.1.33 - Faecal coliforms. Baker Creek trend in differences between reference and impact sites and trends through time. Straight line regression and after LOWESS smoothing (crooked lines) are shown. Kendall's tau=0.127. Figure 6.1.34 - Conductivity. Baker Creek trend in differences between reference and impact sites and trends through time. Straight line regression and after LOWESS smoothing (crooked lines) are shown. Kendall's tau=0.661.

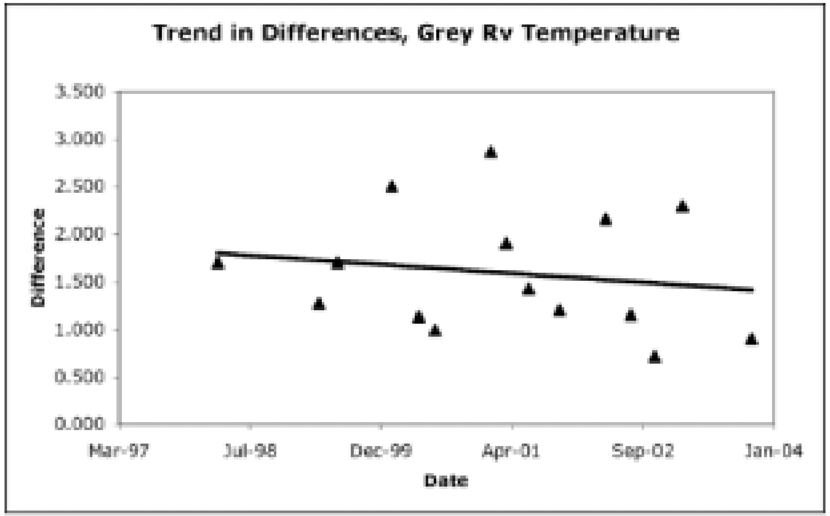


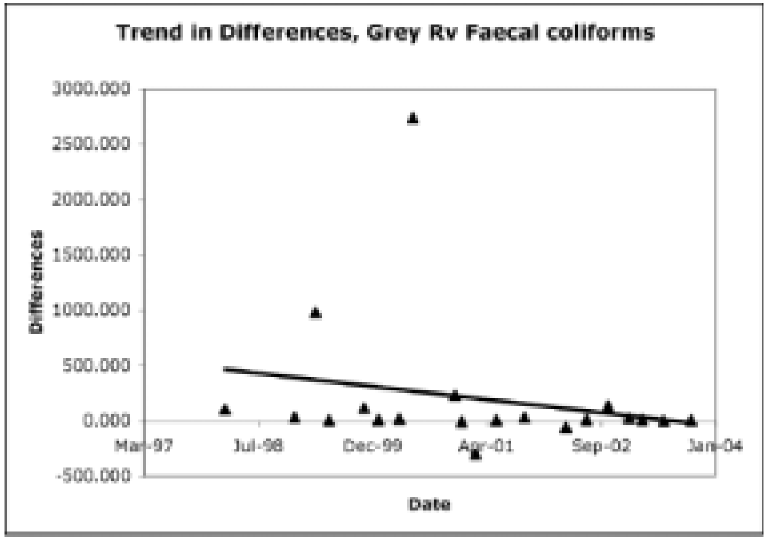


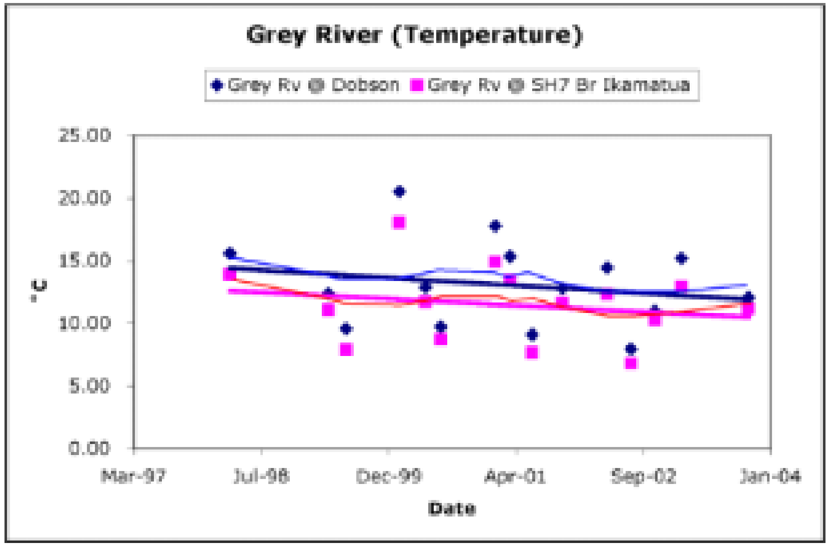


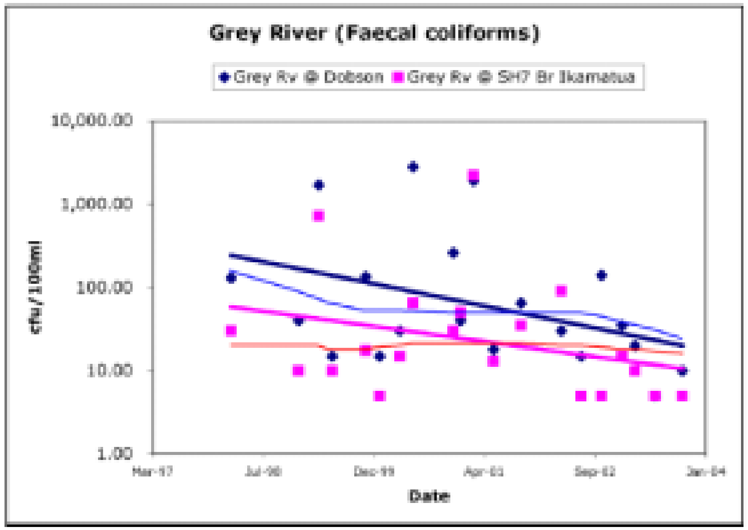


Figures 6.1.35 & 6.1.36 (left to right) Figure 6.1.35 - pH. Grey River trend in differences between reference and impact sites and trends through time. Straight line regression and after LOWESS smoothing (crooked lines) are shown. Kendall's tau=0.786. Figure 6.1.36 - Ammoniacal nitrogen. Grey River trend in differences between reference and impact sites and trends through time. Straight line regression and after LOWESS smoothing (crooked lines) are shown. Kendall's tau=-0.157.

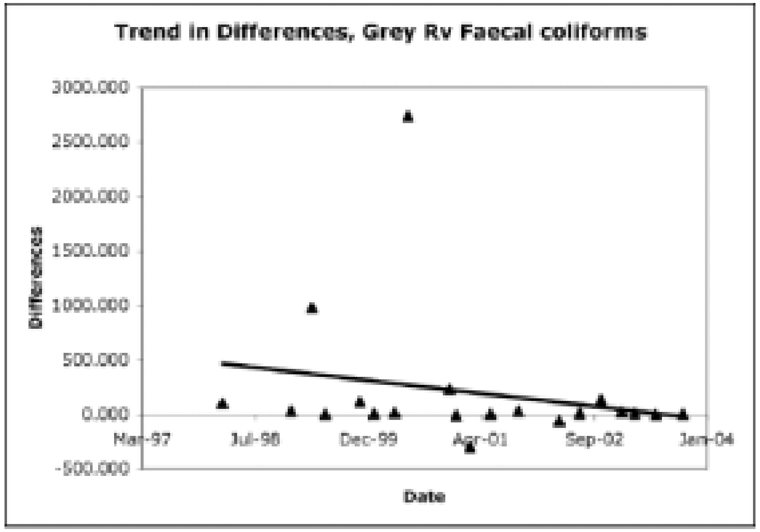


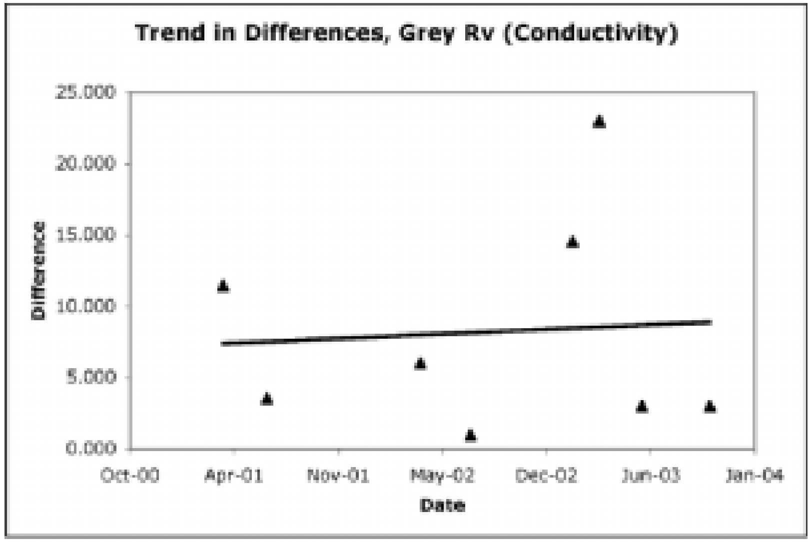


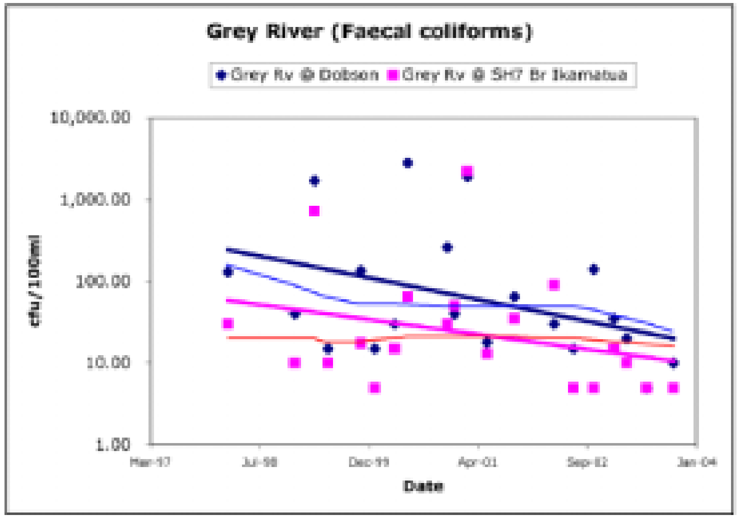


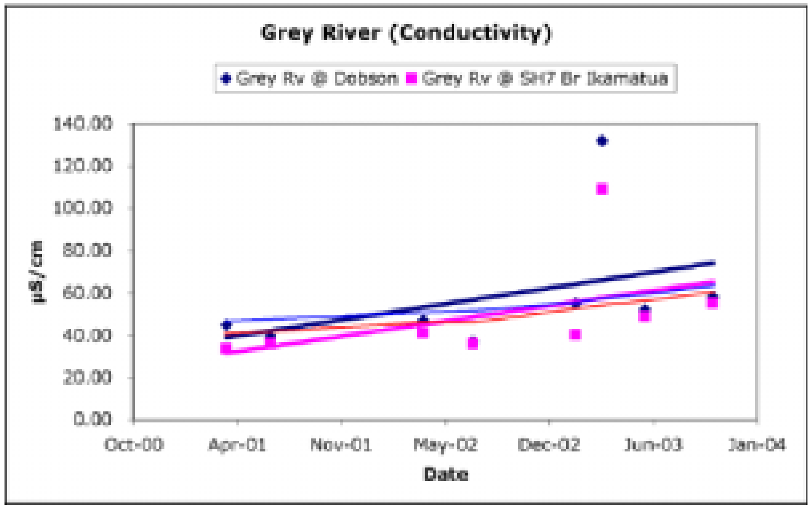


Figures 6.1.37 & 6.1.38 (left to right) Figure 6.1.37 - Temperature. Grey River trend in differences between reference and impact sites and trends through time. Straight line regression and after LOWESS smoothing (crooked lines) are shown. Kendall's tau=0.981. Figure 6.1.38 - Faecal coliforms. Grey River trend in differences between reference and impact sites and trends through time. Straight line regression and after LOWESS smoothing (crooked lines) are shown. Kendall's tau=0.453.









Figures 6.1.39 & 6.1.40 (left to right) Figure 6.1.39 - Dissolved oxygen. Grey River trend in differences between reference and impact sites and trends through time. Straight line regression and after LOWESS smoothing (crooked lines) are shown. Kendall's tau=0.867. Figure 6.1.40 - Conductivity. Grey River trend in differences between reference and impact sites and trends through time. Straight line regression and after LOWESS smoothing (crooked lines) are shown. Kendall's tau=0.691.

6.2 Ecosystem change through time

Indices and features of macroinvertebrate communities (taxonomic richness, % EPT, number of EPT taxa, MCI and SQMCI) were plotted as an indicator of ecosystem health over time. These were grouped into respective REC classes for climate, source of flow (excluding the spring class), geology, landclass, stream order and spring vs. non-spring fed sites. Note that taxonomic richness and 'number of invertebrate taxa' are identical.

Diversity according to taxonomic richness and EPT numbers seemed to dip in 2002 – evident for all REC classes and most sites. This including sites with little or no development activity in their catchment, therefore some climatic event may have been influential. It is unclear what mechanism(s) might have been responsible. Macroinvertebrate samples were collected in the autumn of each year, with climatic conditions during the 2001 autumn not unusually dry or wet compared to autumns of other years.

Macroinvertebrate communities varied among sites that constituted each REC class, contributing to much of the variation observed within a class. This, combined with a certain amount of flow-induced variability, made trend detection difficult. There were no *obvious* trends among REC classes over time for the macroinvertebrate data analysed. Taxonomic and EPT richness seemed to increase, but this may have resulted from increased taxonomic resolution used for sample identification.

Trends over time were more consistent with measures that had a proportional and/or weighted aspect, like % EPT, MCI, and SQMCI are used. As mentioned, taxonomic richness, and also EPT were less stable.

Pasture sites had both pollution tolerant and intolerant taxa, while indigenous forested sites were limited mainly to pollution tolerant species. SQMCI scores for the pasture class dropped to their lowest in 2002, but increased in 2003 back to a levels as high as those pre 2000. Spring sites showed a similar pattern, with their macroinvertebrate parameters dropping a year earlier in 2001 to levels well below non-spring fed sites.

6.2.1 Climate

The cool, extremely wet class had low replication (Figures 6.2.1). Cool wet encompassed nearly all sites within this analysis, and thus basically represents the whole dataset, which is discussed in the following section. EPT and taxonomic richness increased over time, but this was not matched by increases in MCI, SQMCI, or % EPT, which remained level. Water quality according to the MCI varied less than that indicated by the SQMCI, with median MCI consistently placing sites in the doubtful quality (mild pollution) category, opposed to the SQMCI's 'pristine' rating. Considering medians and variation for both indices, we can state that most sites stayed over time within the 'doubtful' water quality margin, indicative of mild pollution, varying between moderate to pristine on the periphery.

Increased taxonomic resolution during invertebrate identification of samples may be the cause of higher trending taxonomic diversity into the more recent samples. Taxonomic and EPT diversity seemed to dip in 2002 between the years of 2001 and 2003. This was most pronounced at lowland sites and in part matched by SQMCI for this year.

6.2.2 Source

Hill source of flow sites tended to have higher water quality based on physicochemical and bacteriological data, but this was not clear from macroinvertebrate data (refer 3.1.2). Inversely, low elevation sites had the highest diversity, again showing an increasing trend, related maybe to increased taxonomic resolution used when identifying macroinvertebrate samples (Figure 6.2.2). Hill source of flow catchments were either clad in indigenous forest or scrub. We would expect these to be more stable than low elevation sites over time, and display less variation, but they weren't being comparable to patterns shown by their low elevation counterparts.

6.2.3 Geology

Physicochemical parameters for the hard sedimentary class varied less than other geology classes (refer 3.1.3), but hard sedimentary macroinvertebrate data varied more, in part due to reduced replication. class and AL classes varied highly within each year, with the highest median value alternating between them in different years. Overall, no trend was apparent among geology classes from this data.

5.2.4 Landuse

Exotic forestry sites were sampled between 1999–2001 and maintained higher water quality than either pasture or indigenous forest throughout their period of sampling. Caution should be used in making assumptions for all exotic forestry streams based on this data as replication is not high, but it can be taken as evidence that exotic forest streams can have macroinvertebrate community's characteristic of a pristine nature.

Pasture sites had consistently high taxonomic richness and total numbers of EPT taxa to rival indigenous forest sites. However, water quality indices were consistently lower in the pasture class, particularly when analysed by SQMCI. Both indices placed HF in all years as 'pristine', while the pasture class was either 'doubtful' (MCI), or 'moderately polluted' (SQMCI). This raises the question of what macroinvertebrate community characteristics are the best indicator of higher water quality and an 'ecologically' healthy stream – the two are always identical. Taxa considered indicative of pristine waters used in the MCI/SQMCI are normally associated with fast flowing, stony streams. Hence habitat type, as well as water quality, can effect the presence/absence of these 'tolerant' species, independent of water quality. That the pasture class was lower according to SQMCI indicated that high abundances of less tolerant taxa contributed to a lower SQMCI (SQMCI utilises proportions of different taxa in calculating and indices compared with MCI which uses presence/absence only). Yet pasture sites still had high taxonomic richness and comparable numbers of sensitive EPT taxa. Two possible explanations exist for this. Firstly, many pasture sites in the study were spring fed, characterised by high taxonomic diversity and water quality, but with habitat suitable for dominance by 'pollution tolerant' taxa. Another possibility is that slight levels of enrichment have enhanced diversity, encouraged more tolerant taxa without significantly effecting many highly sensitive taxa.

6.2.5 Stream order

Mid order sites were most well represented. High order sites had fewer replicates, but based on this and other data (refer 3.1.5), high order sites appeared to be stable, buffered by larger quantities of clean water sourced from undeveloped catchments. Mid order sites were also stable, changing little over time. Low order siites varied more; a product of their smaller size, and less replication for this class. Low order taxonomic diversity took a hit in 2003, and this was reciprocated to a lesser extent in % EPT and SQMCI results.

6.2.6 Spring-fed vs. non spring-fed

EPT and taxonomic diversity were more stable in spring fed streams over time, and did not display the 2002 dip in diversity observed for other sites. Whatever the event that effected community structure or macroinvertebrate sampling during this time, it had a less significance impact on the spring streams. There seemed to be a change in the ratio, and particularly the abundance, of pollution intolerant taxa to tolerant kinds, with more tolerant taxa present from 2001 onwards. This may have been related to changes in water and habitat quality.

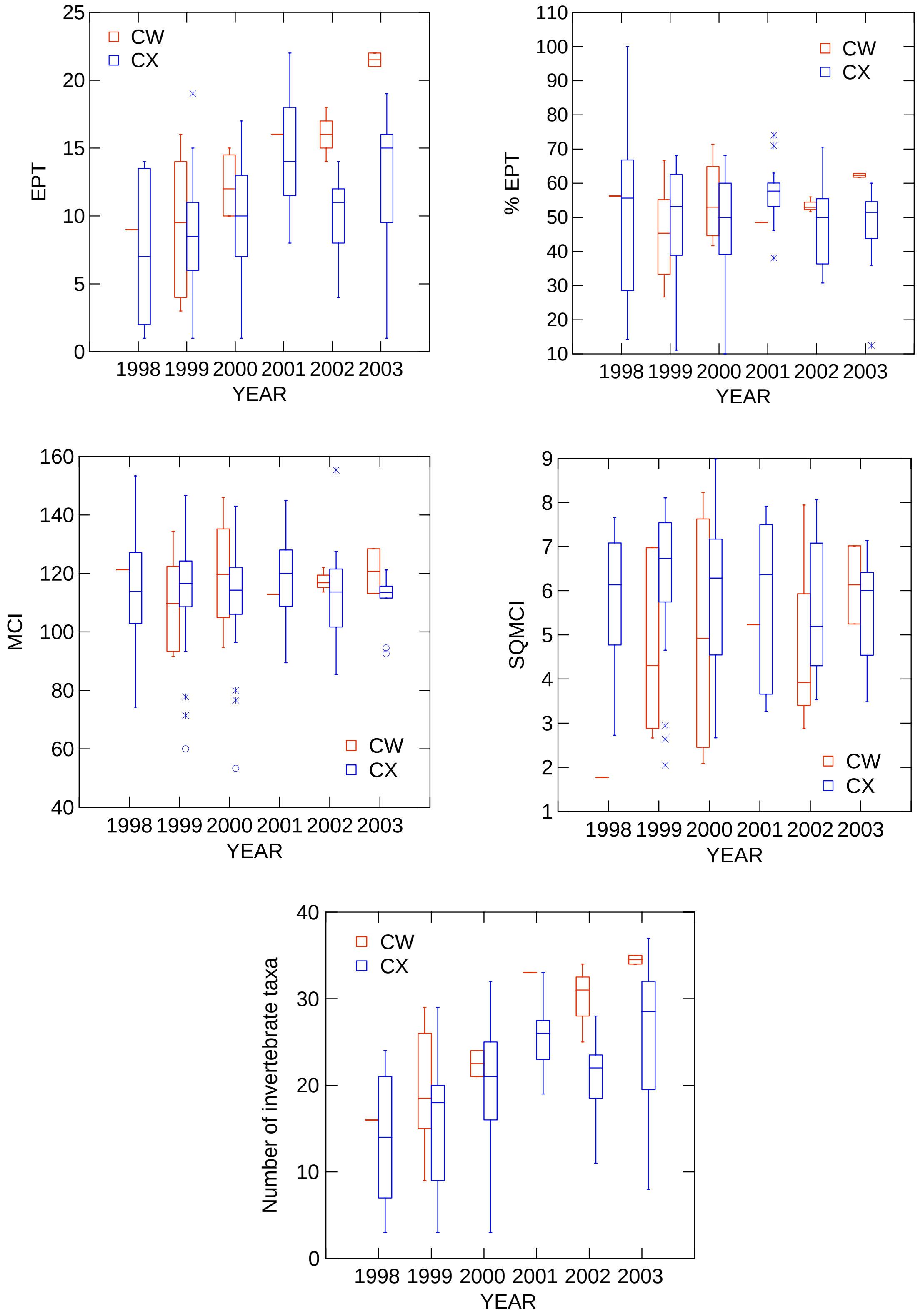


Figure 6.2.1 *EPT, % EPT, MCI, SQMCI, and number of invertebrate taxa plotted according to climate. Sites are combined into cool wet (CW) and cool extremely wet (CX) classes.*

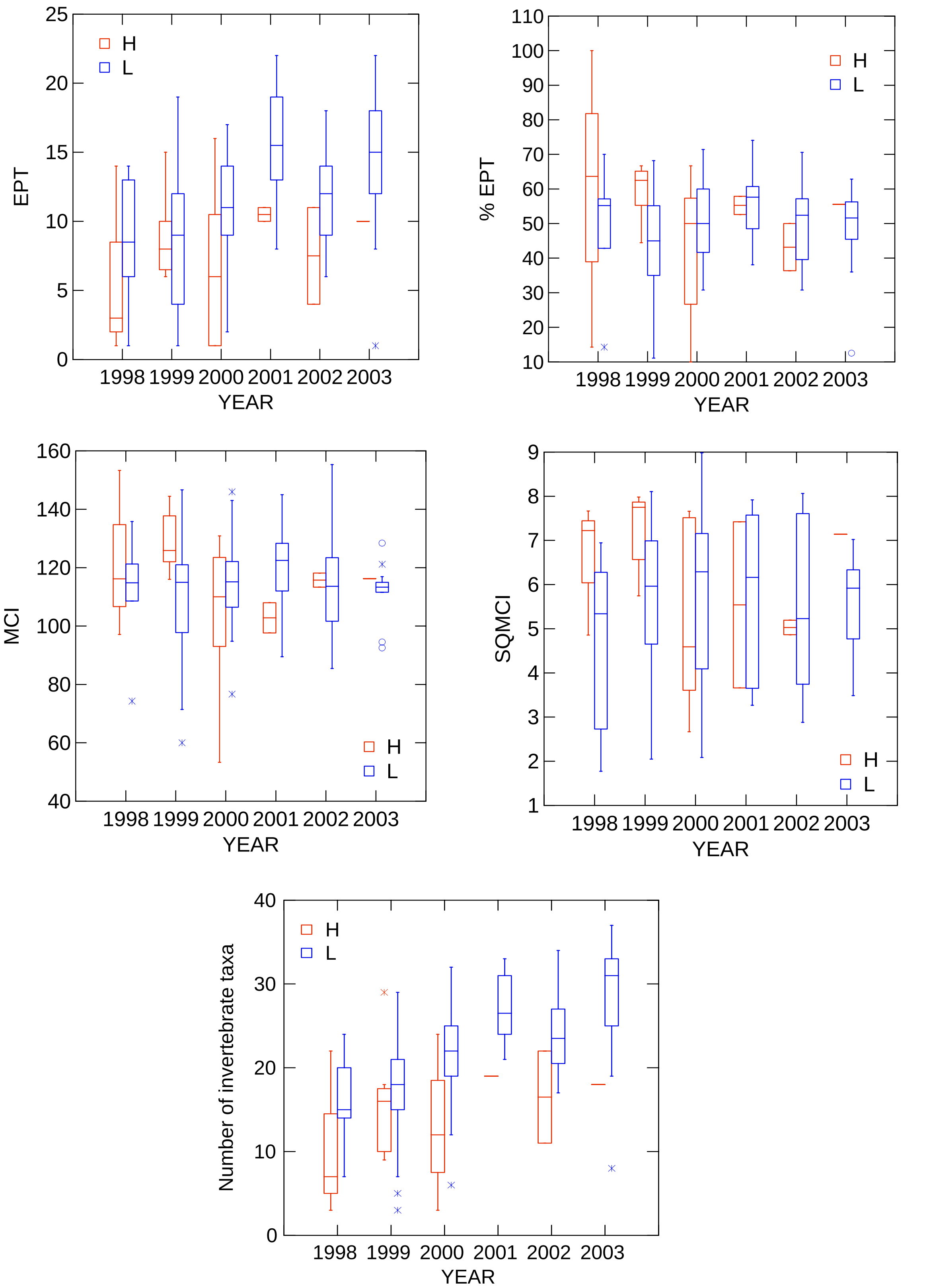


Figure 6.2.2 EPT, % EPT, MCI, SQMCI and number of invertebrate taxa plotted according to source of flow. Sites are combined into low elevation (L) and high elevation (H) classes.

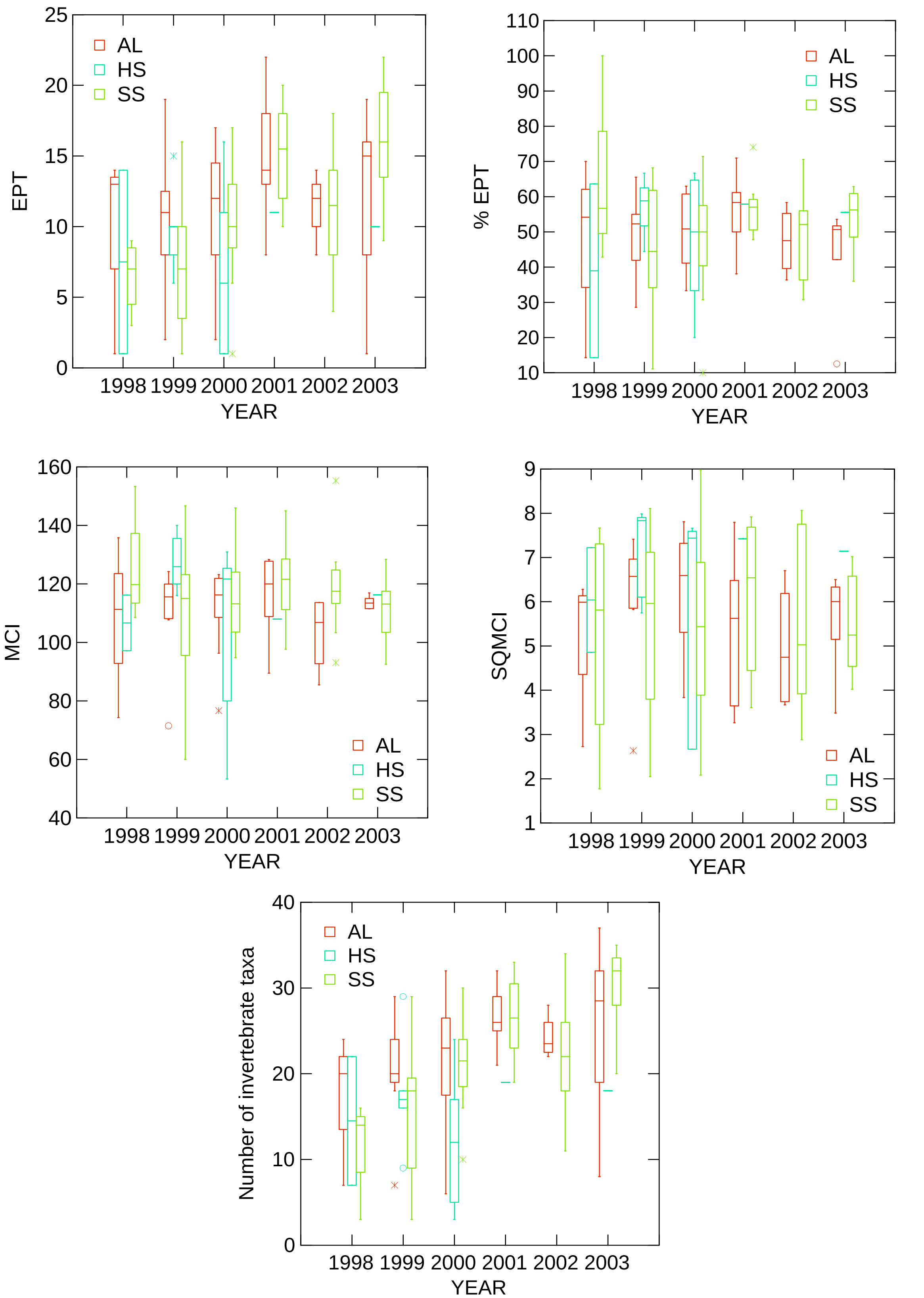


Figure 6.2.3 *EPT, % EPT, MCI, SQMCI and number of invertebrate taxa plotted according to geology. Sites are combined into alluvial and sand (Al), hard sedimentary (HS), and soft sedimentary (SS) classes.*

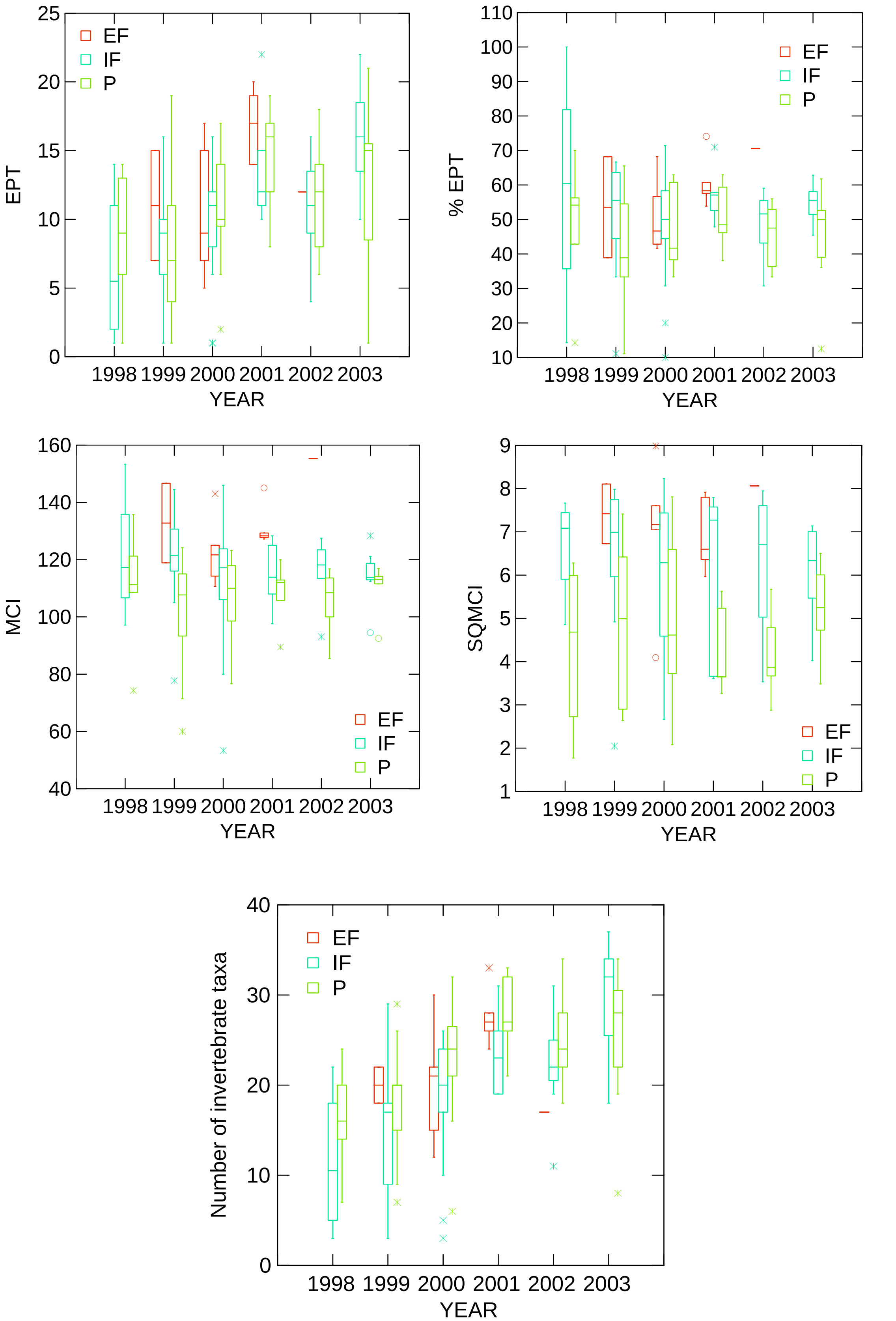


Figure 6.2.4 *EPT, % EPT, MCI, SQMCI and number of invertebrate taxa plotted according to landcover/landuse. Sites are combined into indigenous forest (IF), exotic forest (EF), and pasture (P) classes.*

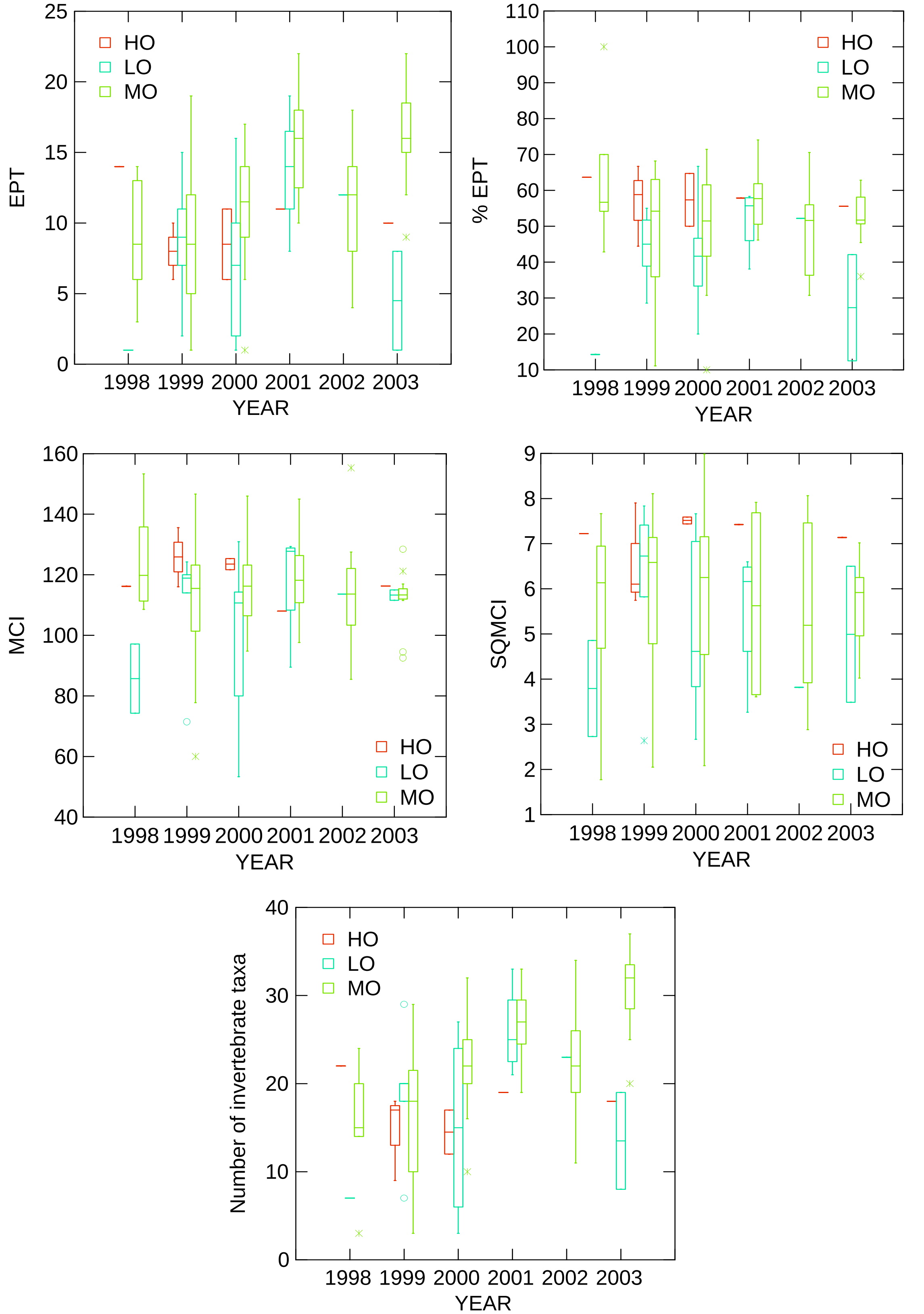


Figure 6.2.5 EPT, % EPT, MCI, SQMCI and number of invertebrate taxa plotted according to stream order. Sites are combined into high order (HO), mid order (MO), and low order (LO) classes.

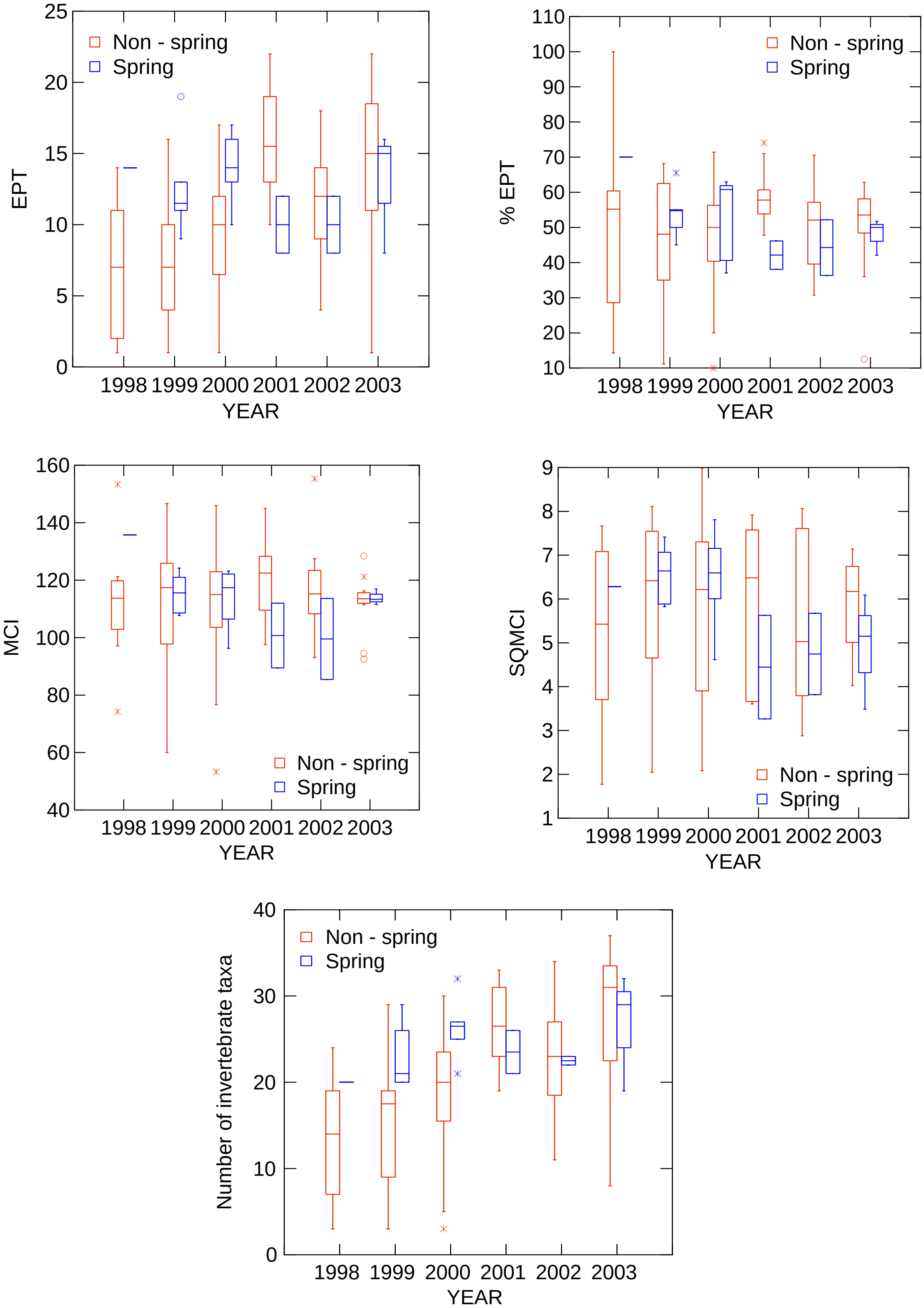


Figure 6.2.6 EPT, % EPT, MCI, SQMCI and number of invertebrate taxa plotted according to origin. Sites are combined into spring fed and non-spring fed classes.

7 Water quality trends in contact recreation areas

Water quality for contact recreation measured at the WCRC contact recreation monitoring sites in the Buller district was generally poor. Orowaiti Lagoon was particularly poor and showed evidence that it had deteriorated. Two sites at the mouth of the Buller River were to a lesser extent also poor, with the possibility that water quality had also deteriorated slightly.

The Grey District had far more sites. The brackish Blaketown and Rapahoe Lagoons had the lowest water quality, both poor, although Rapahoe had many individual values within guidelines. Beaches on either side of the Grey River mouth were generally good, with only occasional exceedences. River sites were also predominantly good, but did have occasionally high exceedences. Three sites were located at Lake Brunner. All were normally safe, but again, occasional large spikes were observed at Moana. Because these poor results are only periodic, chance events cannot be ruled out as the cause, and there was not any sufficiently stable evidence to cause concern.

Lake Kaniere had the highest water quality for contact recreation with no issues evident.

7.1.1 Areas in the Buller District

The two main areas in the Buller district where water quality was monitored for contact recreation purposes were located in the lower Orowaiti and Buller Rivers. The term water quality is used here to describe bacteriological water quality relating to its suitability for human contact, and does not necessarily relate to other forms such as biological or habitat quality.

The Orowaiti Lagoon (Figure 7.1.1) never recorded a contact recreation seasonal median within the Department of Health guidelines for any bacteriological indicator. There were no water quality samples since summer 2001 - 2002 when *E. coli* have been at an acceptable level. This, combined with individual and median *E. coli* and Enterococci levels, suggests water quality has deteriorated in the Lagoon for the last three years. While *E. coli* medians for Orowaiti at Excelsior Road have also consistently stayed above guidelines (Figure 7.1.2), they are lower than those recorded at the Lagoon. Unlike Lagoon, Excelsior water samples have frequently proved ‘acceptable’ and there doesn’t seem to have been any noticeable trend since monitoring began in 1999.

Like the Orowaiti Lagoon, the Buller River @ Shingle Beach has never had an *E. coli* bathing seasonal median within guidelines, although they often did not exceed by much - something reinforced by Enterococci medians (Figure 7.1.3). On the true left side of the Buller River, bacteriological contamination at Marrs Beach appeared to have been slightly lower (Figure 7.1.4). Deterioration in 2003 – 2004 was also apparent, but to a lesser extent.

While no clear pattern was observable for the district, all data suggests that bacteriological contamination was higher in the most recent 2003 – 2004 season, and that *average* water quality at most contact recreation sites was at best of dubious quality.

7.1.2 Areas in the Grey District

There were ten contact recreation monitoring sites in the Grey district, which covered a range of habitats from brackish/marine, to freshwaters encompassing both river and lake environments.

Monitoring results for Seven Mile Creek @ Rapahoe showed water quality to be consistent over time. While water quality as measured by *E. coli* failed to comply with the seasonal median guideline, it only just did so with many individual samples proving 'acceptable' (Figure 7.2.1).

Three monitoring sites are located near the mouth of the Grey River. Water quality at the west end of Cobden Beach (Figure 7.2.2) was normally within guidelines, with only the occasional exceedence. Enterococci were low in all years except 2003- 2004. A similar result was observed at Blaketown Beach (Figure 7.2.4) on the other side of the of the Grey River mouth. Neither site showed any significant pattern of change over the monitoring period. Water quality in Blaketown Lagoon (Figure 7.2.3) was poorer than that observed at either beach site. Season medians for *E. coli* and Enterococci mostly exceeded their respective guidelines, evident from the number of 'alert' and 'action' *E. coli* samples collected.

Water quality at river monitoring sites was normally adequate with occasionally elevated levels and spikes. While regression on medians for the Taylorville swimming hole (Grey River) indicated deterioration, the combination of high variability and few data points would indicate this to be a weak conclusion (Figure 7.2.5). The same consideration should be applied to the regression of medians for Nelson Creek (Figure 7.2.7). Water quality at both sites was frequently acceptable for swimming; with some spikes of poorer water quality occurring. Water quality in the Arnold River, as far down as Blair's Rd No.2 Bridge (Figure 7.2.6), had spikes of poor water quality, but remained acceptable since monitoring commenced in 2001.

At Lake Brunner, no seasonal medians exceeded the guideline. Cashmere Bay data was the shortest (Figure 7.2.9). Other than one major exceedence ('action'), water quality here was acceptable. Iveagh Bay (Figure 7.2.10) is near Cashmere Bay, but further from urban settlement. 'Action' levels of *E. coli* were not recorded here, with two 'alert' samples in 2001 and 2003. Moana had one 'alarm' sample per season since 2001, with all others 'acceptable' (Figure 7.2.8). There were no trends in water quality at Lake Brunner.

7.1.3 Areas in the Westland District

Water quality for contact recreation in Lake Kaniere, as indicated by results to date, was adequate (Figures 7.3.1 to 7.3.3). No values other than those deemed 'acceptable' by guidelines were recorded for this lake. There was a possibility of minor deterioration in water quality at Hans Bay Jetty (Figure 7.3.2), but levels were normally well below guidelines.

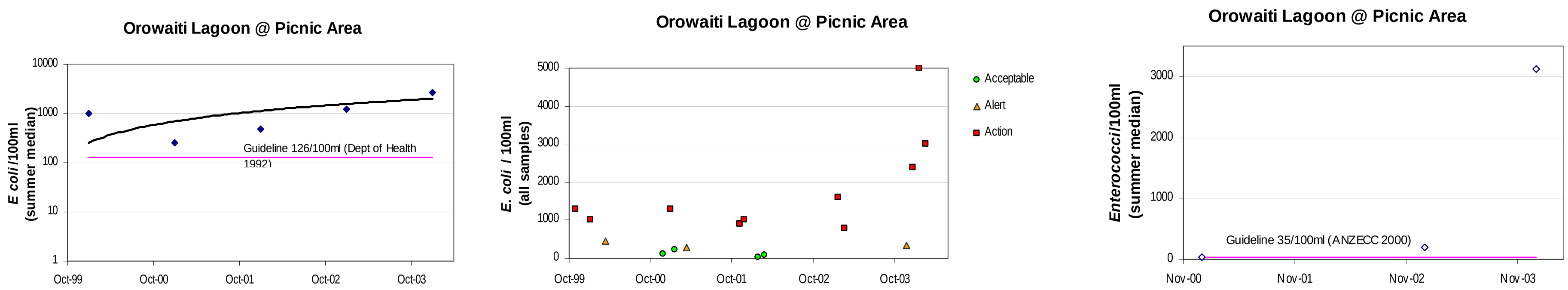


Figure 7.1.1 *Orowaiti Lagoon @ Picnic Area. From left to right are season medians for E. coli (trend shown by blue line), individual E. coli values, and season Enterococci medians.*

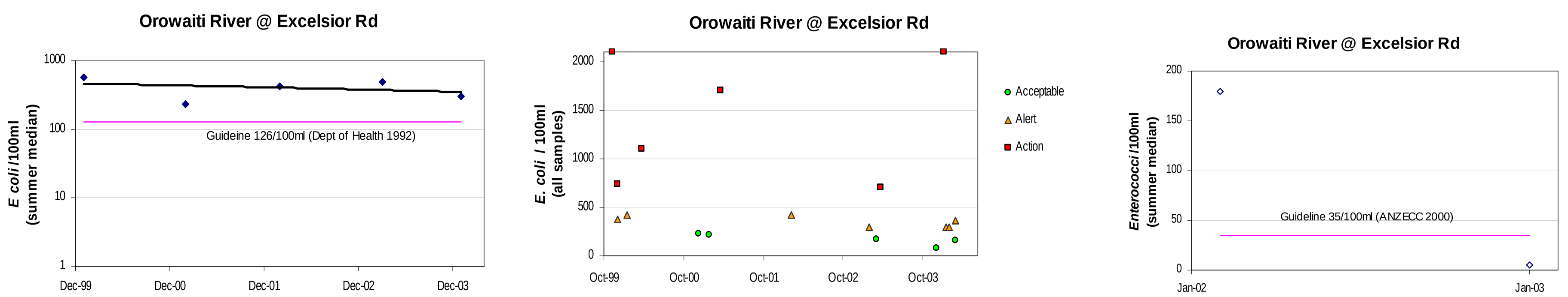


Figure 7.1.2 *Orowaiti River @ Excelsior Rd. From left to right are season medians for E. coli (trend shown by blue line), individual E. coli values, and season Enterococci medians.*

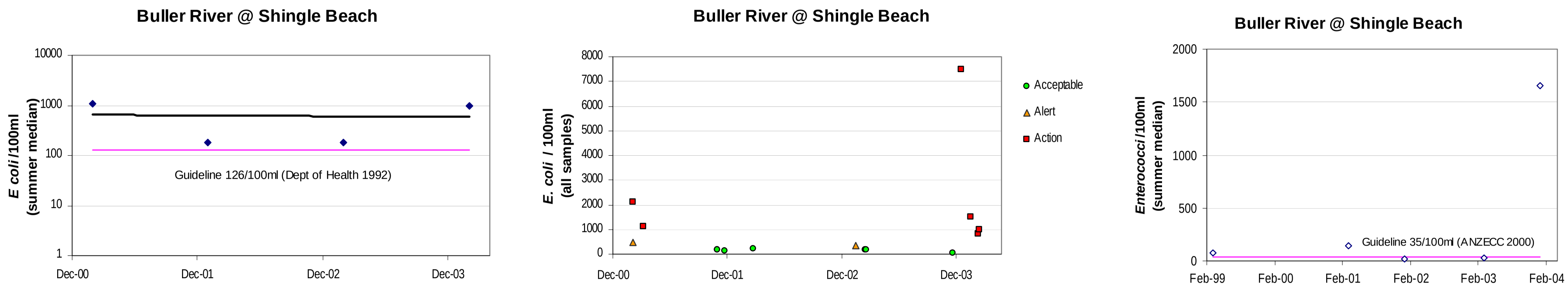


Figure 7.1.3 Buller River @ Shingle Beach. From left to right are season medians for E. coli (trend shown by blue line), individual E. coli values, and season Enterococci medians.

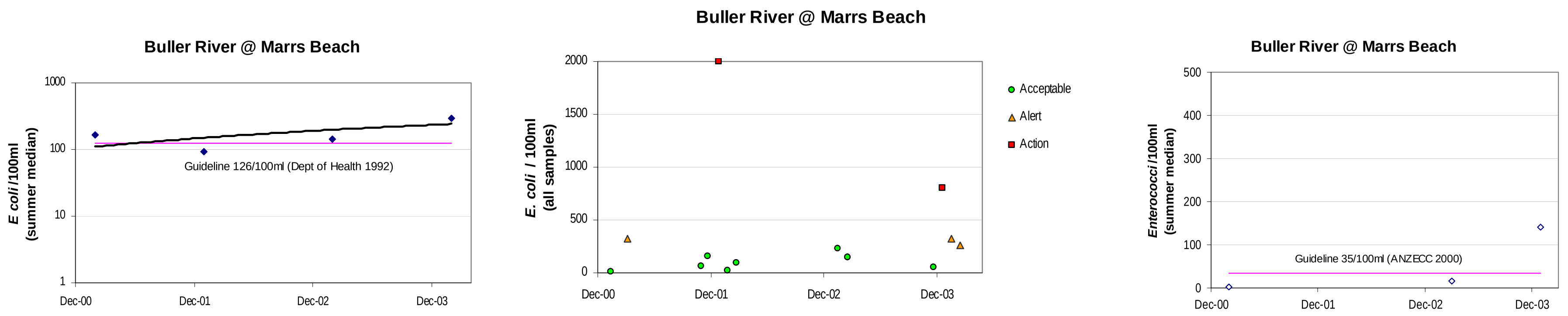


Figure 7.1.4 Buller River @ Marrs Beach. From left to right are season medians for E. coli (trend shown by blue line), individual E. coli values, and season Enterococci medians.

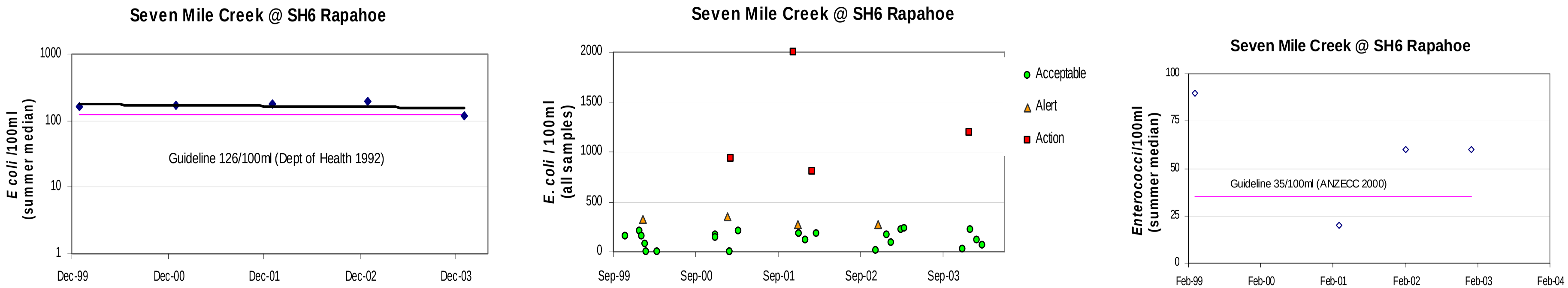


Figure 7.2.1 Seven Mile Creek @ SH6 Rapahoe. From left to right are season medians for E. coli (trend shown by blue line), individual E. coli values, and season Enterococci medians.

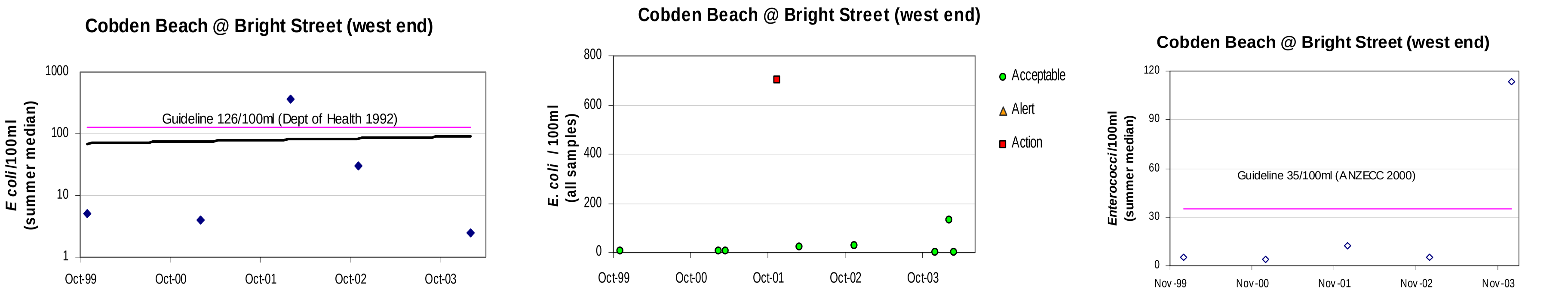


Figure 7.2.2 Cobden Beach @ Bright street. From left to right are season medians for E. coli (trend shown by blue line), individual E. coli values, and season Enterococci medians.

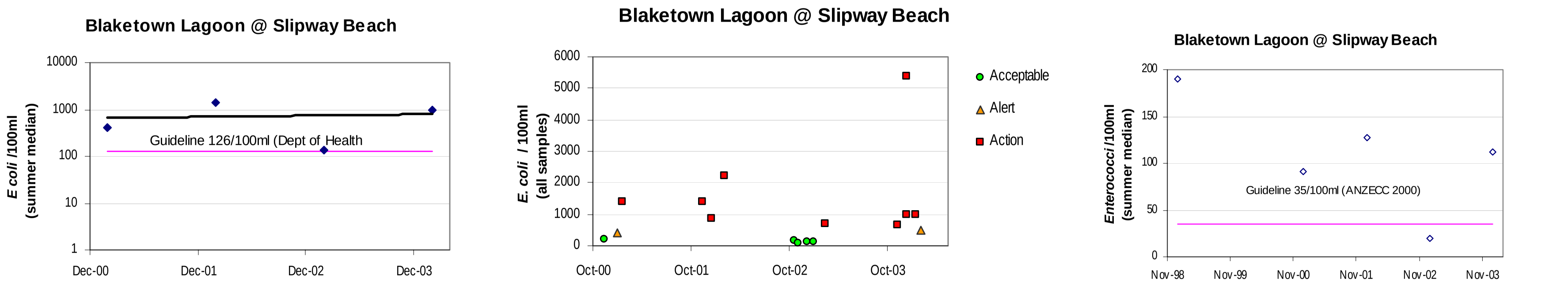


Figure 7.2.3 Blaketown Lagoon @ Slipway Beach. From left to right are season medians for E. coli (trend shown by blue line), individual E. coli values, and season Enterococci medians.

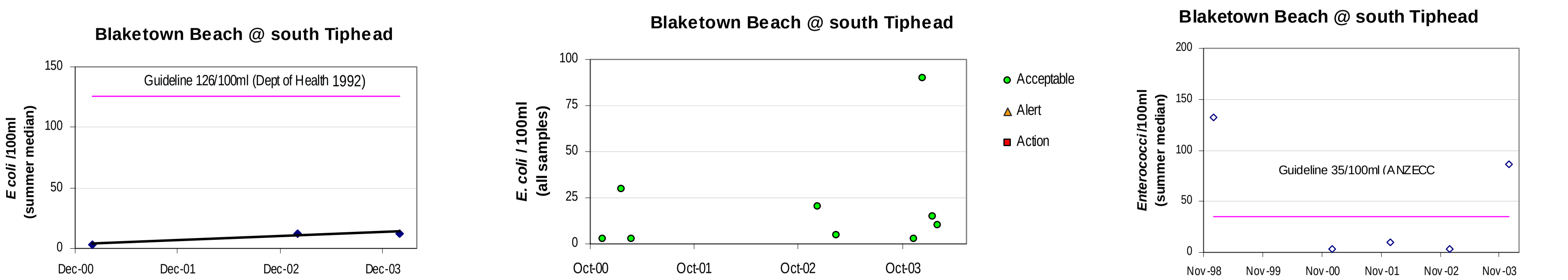


Figure 7.2.4 Blaketown Beach @ South Tiphead. From left to right are season medians for E. coli (trend shown by blue line), individual E. coli values, and season Enterococci medians.

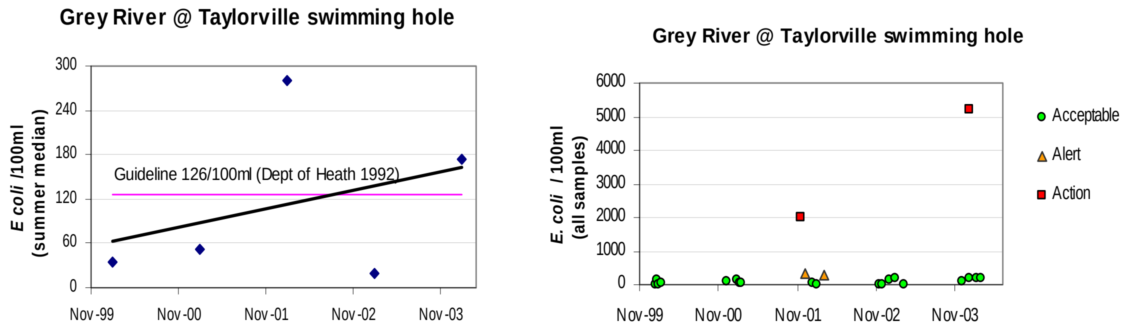


Figure 7.2.5 Grey River @ Taylorville swimming hole. From left to right are season medians for E. coli (trend shown by blue line), and individual E. coli values.

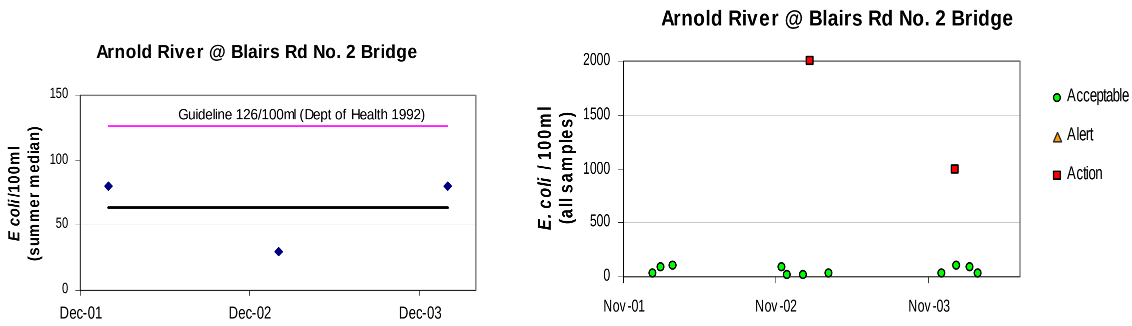


Figure 7.2.6 Arnold River @ Blairs Rd No.2 Bridge. From left to right are season medians for E. coli (trend shown by blue line), and individual E. coli values.

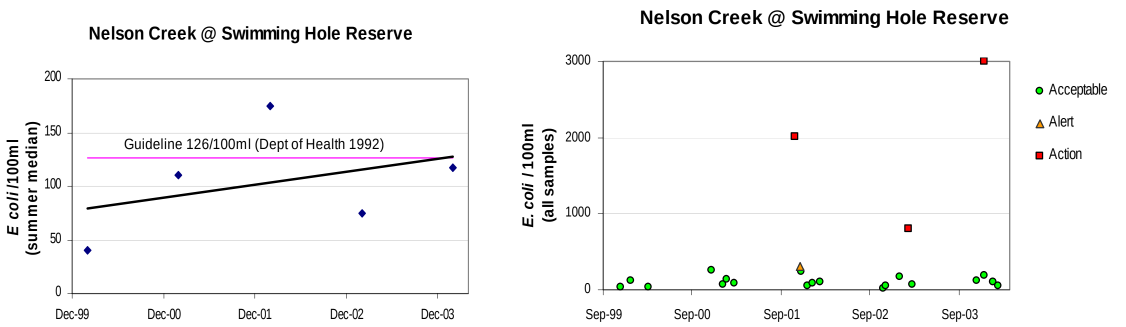


Figure 7.2.7 Nelson Creek @ Swimming Hole Reserve. From left to right are season medians for E. coli (trend shown by blue line), and individual E. coli values.

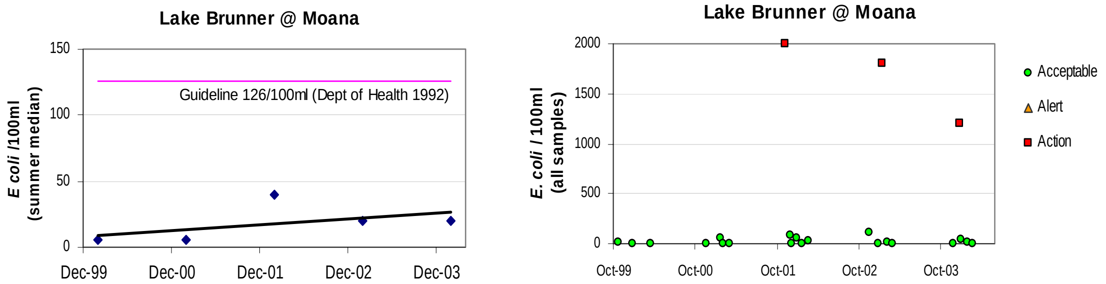


Figure 7.2.8 Lake Brunner @ Moana. From left to right are season medians for E. coli (trend shown by blue line), and individual E. coli values.

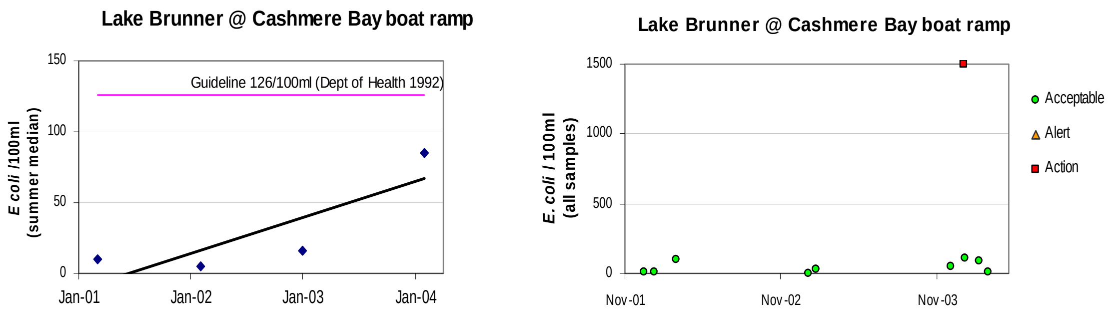


Figure 7.2.9 Lake Brunner @ Cashmere Bay boat ramp. From left to right are season medians for E. coli (trend shown by blue line), and individual E. coli values.

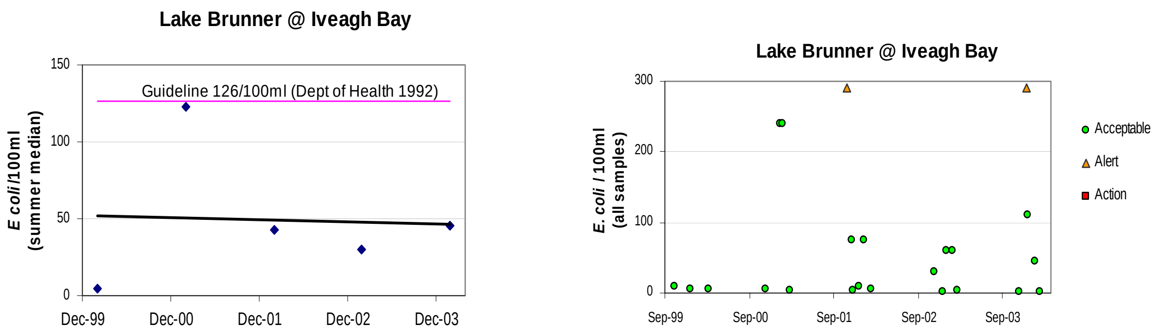


Figure 7.2.10 Lake Bruner @ Iveagh Bay. From left to right are season medians for E. coli (trend shown by blue line), and individual E. coli values.

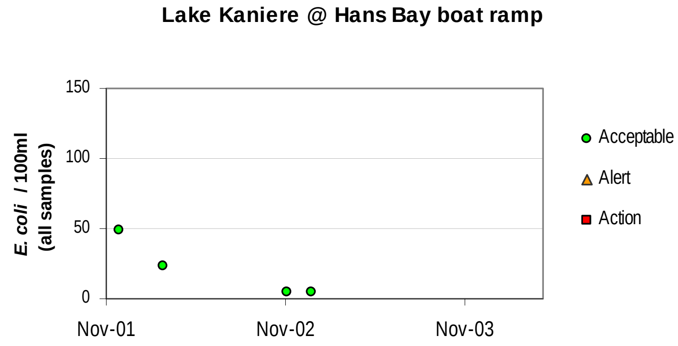


Figure 7.3.1 Lake Kaniere @ Hans Bay boat ramp individual E. coli values.

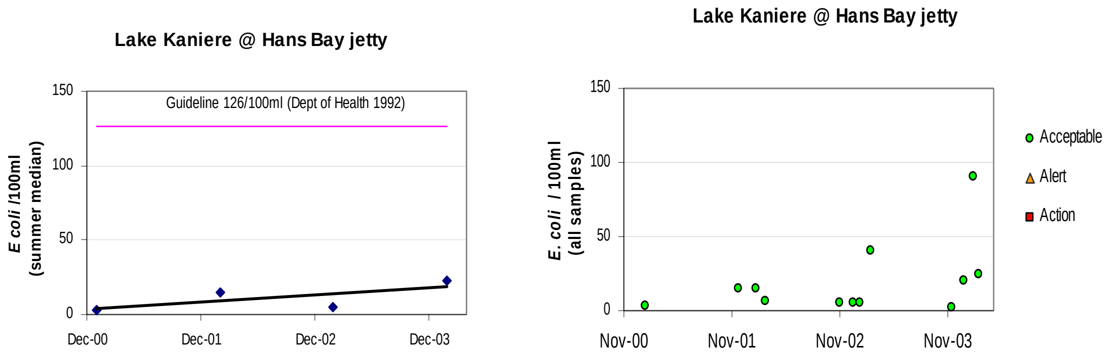


Figure 7.3.2 Lake *Kaniere* @ Hans Bay jetty. From left to right are season medians for *E. coli* (trend shown by blue line), and individual *E. coli* values.

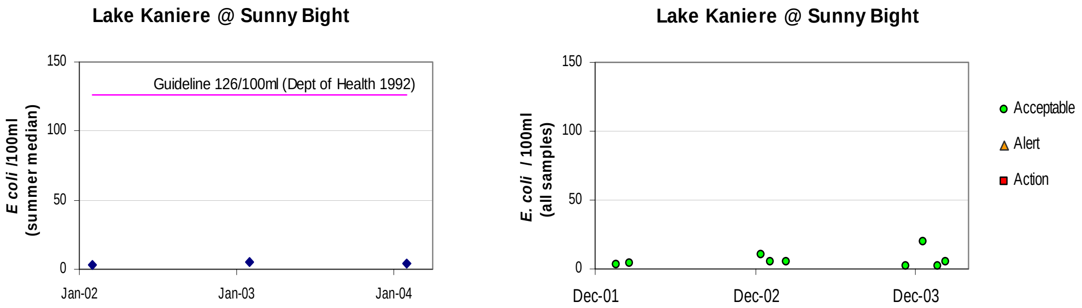


Figure 7.3.3 Lake Kaniere @ Sunny Bight. From left to right are season medians for E. coli (trend shown by blue line), and individual E. coli values.

8 Water quality trends in Lake Brunner

Lake Brunner is the largest Lake on the West Coast, at 36.1 km^2. It has been categorized by the WCRC as a special management area, owing to its high natural and human use values. It is also considered the most vulnerable lake on the West Coast, due to development pressure, recreational use, farming and forestry. Farming accounts for around 25 % of land use in the catchment (WCRC, 2004).

Epilimnetic nitrogen stabilized in 2003-04. However, trends in phosphrus and phytoplankton biomass were up, and clarity down. In Cashmere Bay anoxic conditions near the lake bed sediments caused nutrient releases from the sediment. Nutrients in the tributaries (Crooked, Orangipuku, and Hohonu) have stabilised and this may lead to more stable nutrient levels in the lake. While faecal coliforms were higher in the rivers with higher agricultural activity (Crooked and Orangipuku), they were within contact recreation guidelines.

8.1 Lake monitoring

Epilimnetic nitrogen (both nitrate and TN) had similar ranges as observed in the 2003-04 monitoring season, and slightly lower than concentrations in 2001-2002. Alternatively, phosphorus, the limiting nutrient in the lake (based on N:P ratios > 20:1), remains higher than recent years (2001-2002), and well above concentrations recorded during the mid 1990s. Phytoplankton biomass appeared to trend upwards, which negatively affected visual clarity (secchi depth). Overall, secchi depth appeared to have decreased by approximately 1 m, from around 5.5 m in the 1990’s, to around 4.5 on average in 2004-2005 (Figure 7.1.1).

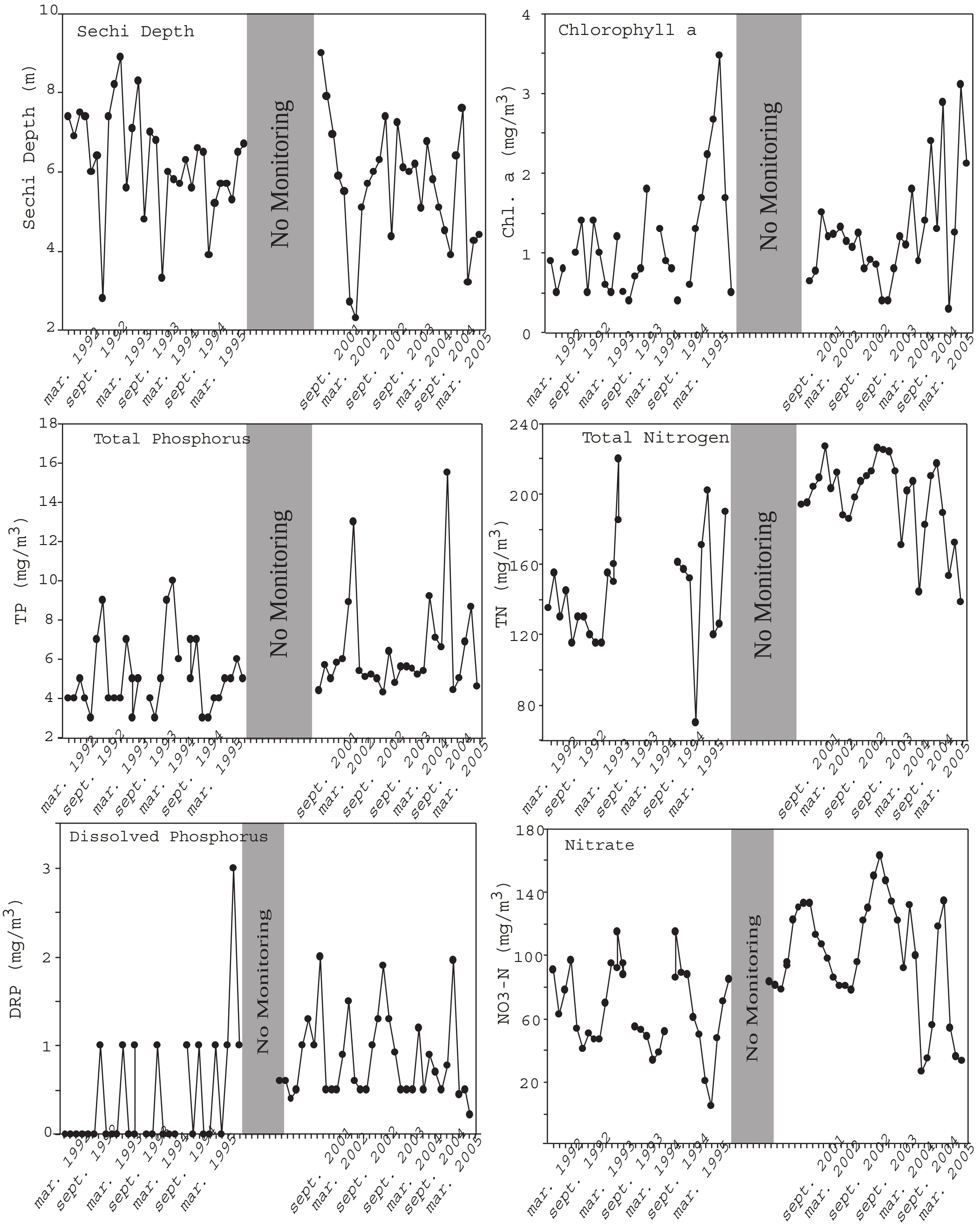


Figure 8.1.1. Lake Brunner trends in clarity, chlorophyll a, total phosphorus, total nitrogen, dissolved inorganic phosphorus, and nitrate for the mid lake station. sampled using a 20m integrated tube sampler.

Mid summer (March 05) vertical profiles for the three stations on the lake indicate that hypolimnetic oxygen depletion is still a concern for Cashmere Bay, where bottom waters are near-anoxic between January and May. At Cashmere Bay, there was also evidence that anoxic conditions at the sediment water interface are driving internal loading of nutrients from the sediments, as evidenced by high TDN levels in bottom waters, particularly ammonia. Instrument profiling also revealed increased conductivity in near bottom waters. Neighbouring Iveagh Bay had normal temperature and dissolved oxygen profiles over the summer period.

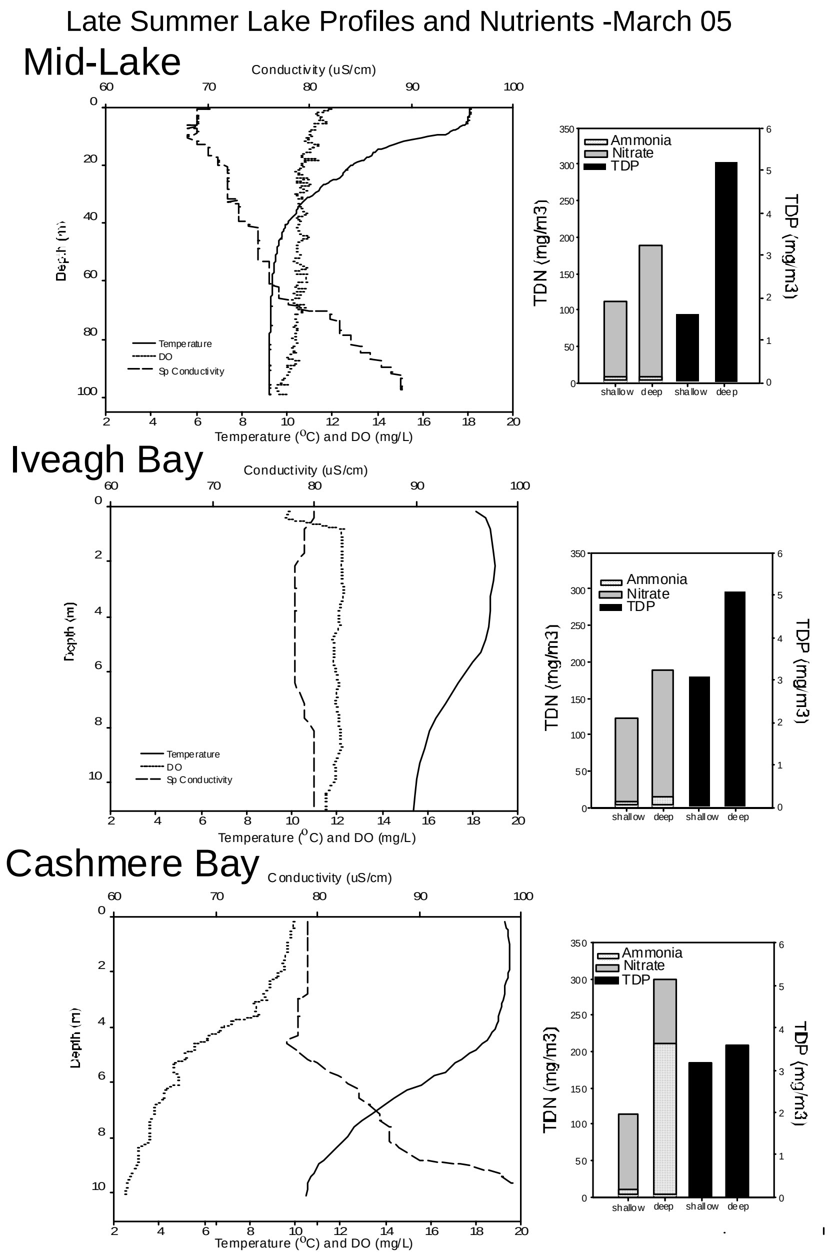
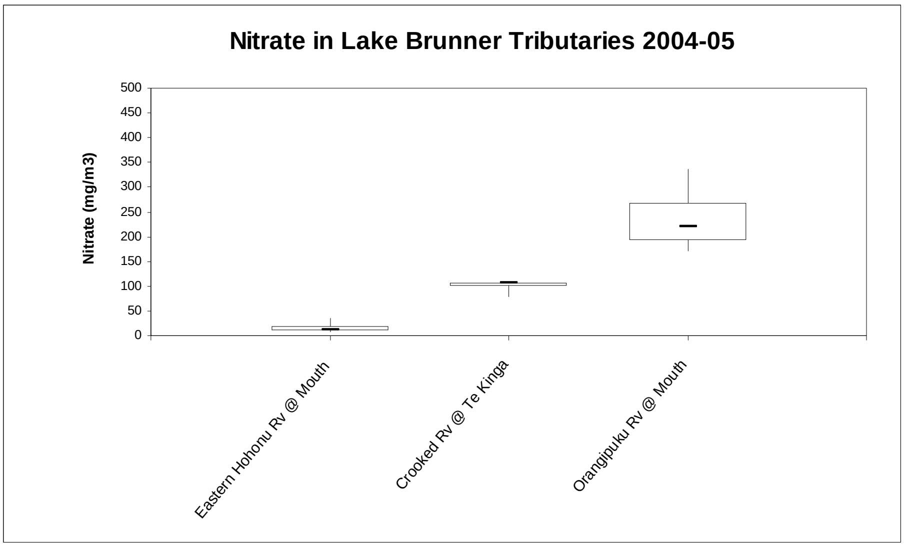


Figure 8.1.2 Dissolved oxygen, temperature, and conductivity profiles for Cashmere Bay, Iveagh Bay, and Mid-Lake for March 2005.

8.2 Tributary Monitoring

2004-05 trends in nitrate of the three major tributaries flowing into Lake Brunner were similar to those measured in 2001-03. Concentrations were highest in the Orangipuku River (median 245 mg/m^3), intermediate in the Crooked River (median 141 mg/m^3), and lowest in the forested Hohonu River (median 22 mg/m^3). Similar concentration ranges were measured in 2004-05, compared with 2001-2003, suggested that although quantities of nitrogen are higher for the two agricultural rivers, they appear to be stable over this time period, and may indicate that nutrient concentrations in the lake will stabilize.



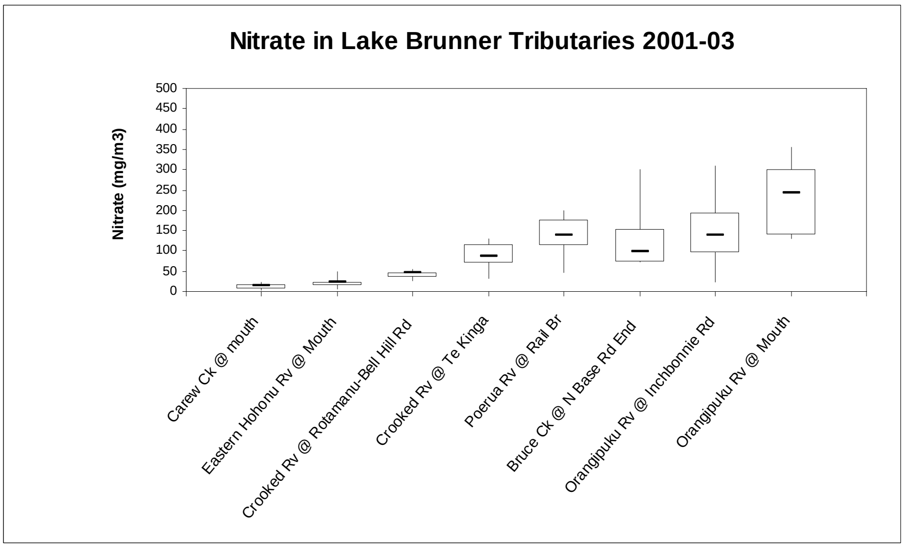
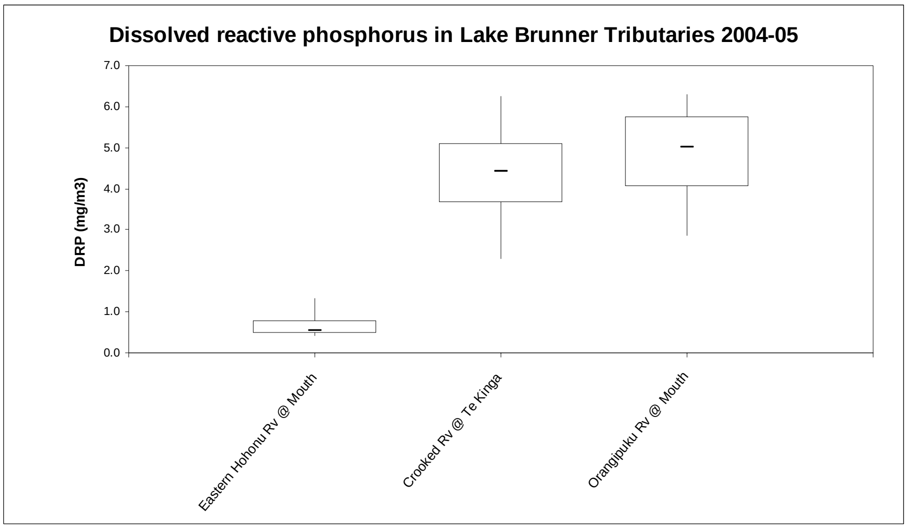


Figure 8.1.3 Nitrate levels in the Lake Brunner tributaries for the years of 2001-03 and 2004-05.

Similar to trends for nitrogen, concentrations of dissolved reactive phosphorus (DRP) during 2004-05 in the three major inflowing tributaries were similar to those measured in previous years (2001-03). Median concentrations in the Orangipuku River were 5.4 mg/m^3 over 2004-05, compared with 6.0 mg/m^3 in 2001-03. Again this indicates that phosphorous concentrations in agricultural catchments draining into the lake were approximately 10x greater than that of forested catchments, but were stable over this monitoring period (2001-2005), maybe being an indicator that nutrient concentrations in the lake could stabilize in the future.



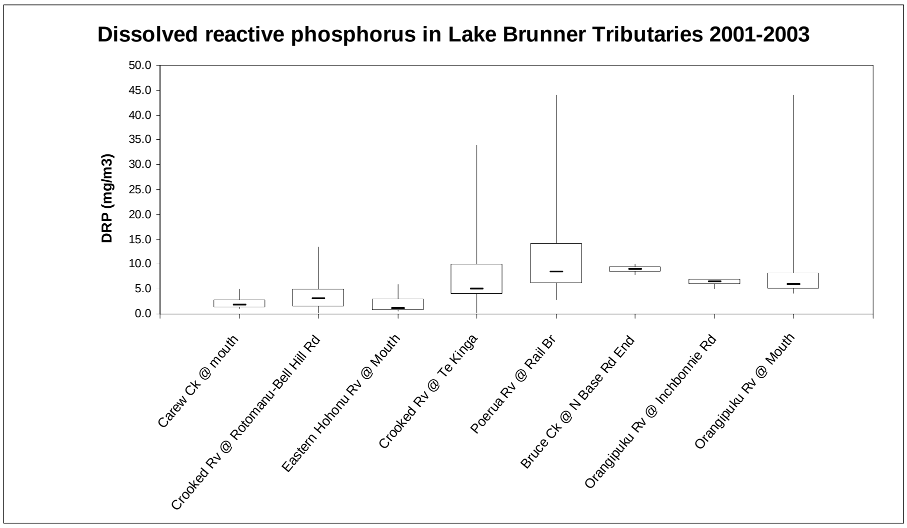


Figure 8.1.4 Dissolved reactive phosphorus levels in the Lake Brunner tributaries for the years of 2001-03 and 2004-05.

Faecal coliform concentrations in 2004-05 were similar among the three inflowing tributaries, despite large differences in land-use between the catchments. All were in the range of the recreational bathing standard of 100 cfu/100ml.

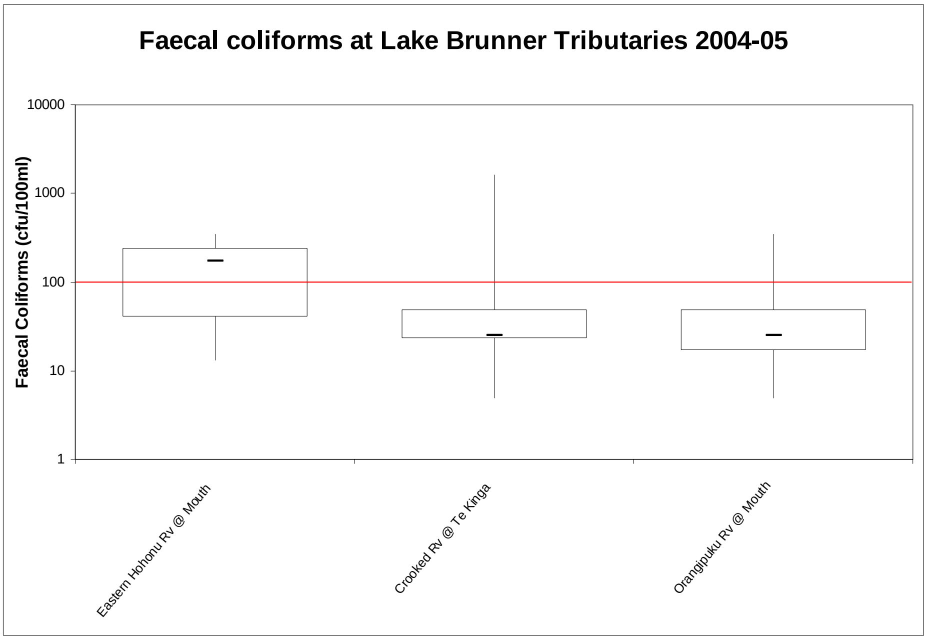


Figure 8.1.5 Faecal coliforms for the Eastern Hohonu River, Crooked River and Orongopuku River. The ANZECC (2000) guideline for contact recreation is shown as a red line.

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Figures and tables

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