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Marine and Freshwater Studies

Potential Impacts of Discharges from Wallerawang Power Station on the Upper Coxs Coxs River

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Cover Image: Site A at CR1 on Coxs River 22/04/2010. Photographer Matt Harper, Cardno Ecology Lab

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Executive Summary

Background and Aims

The water that is used in Wallerawang Power Station's (WPS) boiler and cooling systems is discharged into the Coxs River via seven licensed discharge points (LDPs). These discharges add a variety of contaminants to the river. The discharge at these LDP's is regulated by Environment Protection Licence (EPL) No. 766 issued under the Protection of the Environment Operations Act 1997. This licence specifies the concentration limits for sulfate, total suspended solids, turbidity, oil and grease and pH that must not be exceeded at each discharge point and stipulates that concentrations of a number of additional contaminants must be monitored at the discharge points and at two ambient water quality monitoring locations at certain intervals.

In March 2010, Cardno Ecology Lab was engaged by Delta Electricity to:

- review, compile and synthesise the monitoring records for their various aquatic discharge/monitoring locations;
- assess the potential impact of these contaminants on the aquatic environment and biota in the reach of the Coxs River between WPS and Lake Lyell; and
- assess the changes in periphyton, aquatic macroinvertebrates and fish that have occurred at a location (CR1) situated between Lakes Wallace and Lyell since the inception of the Upper Coxs River Health Monitoring Program.

Monitoring Records from Discharge/Monitoring Locations

Delta Electricity provided data from their licensed discharge points, ambient water quality monitoring locations and three additional water quality monitoring locations situated on the Coxs River downstream of Lake Wallace. The potential impact of the discharges was assessed by examining compliance with the concentration limits specified in the EPL and by comparing the concentrations of each contaminant at the various discharge/monitoring points with the appropriate ANZECC/ARMCANZ (2000) Water Quality Guidelines. The discharges generally complied with the percentile concentration limits for sulfate, total suspended solids, turbidity and pH specified in the EPL. A large proportion of the concentrations/levels of a number of other contaminants within the discharges, however, exceeded the default trigger values (DTV) or guideline trigger values (GTV) for protection of aquatic life in slightly-moderately disturbed ecosystems that are applied to receiving waters beyond the identified mixing zone.

A brief summary of the physico-chemical characteristics of each discharge and of the quality of water at each monitoring location is presented below arranged according to their position along the river.

The discharge from the Centennial-Springvale Water Transfer System bypass east of Kerosene Vale Ash Dam (LDP20) is released into Sawyers Swamp Creek which flows into the Coxs River upstream of the ambient water quality monitoring point LMP7. This intermittent discharge is characterised by high pH, turbidity and TSS levels, moderately high conductivities and low concentrations of sulfate. The conductivity and pH values exceeded the upper DTV on > 90% of the monitoring events and turbidity levels did so on >25% of the events.

The quality of the water at LMP7 provides an insight into the downstream extent of the inputs from the upstream tributaries, including that from LDP20. The conductivity, pH, aluminium and nutrient levels at LMP7 exceeded their respective upper DTVs. The boron, manganese and selenium concentrations, however, were low, indicating that trace metal inputs from the upstream tributaries are diluted rapidly by flows within the river and by inflows from Pipers Flat Creek. The high conductivity and nutrient levels are probably related to the discharge from the sewage treatment plant on Pipers Flat Creek.

The reach of the Upper Coxs River situated below the confluences with these tributaries but above Lake Wallace receives inputs from five licensed discharge points (LDP1, LDP3, LDP18, LDP5 and LDP21) associated with WPS.

The intermittent discharge from the Northern Retention Basin (Unit 7 Cooling Tower) released at LDP1 is characterised by high conductivities and sulfate concentrations. Most (95%) of the conductivity and a large proportion (45%) of the pH measurements taken at this point exceeded their respective upper DTVs.

The discharge from Sawyers Swamp Ash Dam that is released via LDP3 flows into Coxs River via Springvale Creek. This intermittent discharge is characterised by high conductivities, low pH, high concentrations of sulfate, fluoride, boron, manganese, and selenium and low iron concentrations. The input of all of these contaminants, except iron, is also highly variable. A large proportion of the conductivity, boron and selenium measurements taken in this discharge exceeded the upper DTV and GTVs for protection of 90% and 80% of aquatic species in slightly-moderately disturbed systems, respectively.

The discharge from the coal settling ponds that enters Coxs River at LDP18 is characterised by moderately high conductivities in excess of the upper default trigger value, neutral pH and relatively low TSS levels.

Further downstream, the river receives intermittent inputs from LDP21, the emergency discharge point for the Unit 8 Cooling Tower. This discharge is characterised by high pH levels, moderately high conductivities, low concentrations of sulfate, TSS, fluoride and boron and low turbidity. The conductivity and pH levels measured at LDP21 were also generally in excess of the upper default trigger value.

The discharge from the Southern Retention Basin enters the river via LDP5. This discharge is characterised by moderately high conductivity values, neutral pH and low concentrations of sulfate, fluoride, TSS, boron and selenium. The conductivity levels at this point exceeded the upper DTV on more than 75% of the monitoring events, while boron exceeded the 90% protection level for aquatic ecosystems during 23% of the events.

The quality of the water at LMP8, the downstream ambient monitoring location, provides an indication of the extent to which the inputs from the licensed discharge points and upstream tributaries are diluted. The water at this location was characterised by moderate conductivity, pH and sulfate levels and low concentrations of fluoride, boron, manganese, iron and selenium, which implies that the inputs from the licensed discharge points have undergone considerable dilution by the time they reach LMP8. The median conductivity, boron and manganese levels, however, were much greater than at LMP7, indicating that the discharges had elevated these parameters within the river. As a large proportion of the conductivity and pH measurements exceeded their respective upper DTVs and a fairly large proportion of boron measurements exceeded the trigger value for 95% protection, there is a possibility of adverse effects on aquatic biota.

The quality of the water at WX13, the monitoring point immediately downstream of Lake Wallace, provides an indication of any residual effects from the upstream discharge points, the effects of impoundment of the river and daily riparian releases. The water was characterised by high nitrogen levels, moderate conductivity, pH, sulfate and fluoride levels, low turbidity, highly variable manganese concentrations, low concentrations of TSS, iron, ammonium as nitrogen and nitrogen oxides. The lower median conductivity, pH, manganese and iron levels relative to LMP8 indicate that further dilution occurs within Lake Wallace. A large proportion of the conductivity levels, however, still exceed the upper DTV, as do the concentrations of most nutrients. The pH levels are also of concern, because 25% of the measurements exceeded their upper DTV. This suggests that the riparian releases from the dam may pose a potential threat to aquatic life.

Under normal operating conditions, the blowdown from Cooling Towers 7 and 8 is released into the Coxs River via licensed discharge point LDP4 which is situated downstream of Lake Wallace at the confluence of the tortuous watercourse and the river. This discharge was characterised by high conductivities, neutral pH, high sulfate and iron concentrations, moderate TSS, turbidity, fluoride boron and manganese levels, but low concentrations of selenium. The major concerns with this discharge are conductivity, which was always in excess of the upper DTV, and pH levels and boron concentrations which exceeded the upper DTV and 90% protection level for aquatic ecosystems on 22% of the events, respectively. Iron contamination may also be of concern, but is difficult to evaluate because of the small number of measurements and lack of guideline trigger values for Australian conditions.

The quality of the water at COX4, the monitoring point situated downstream of the confluence with LDP4 and the tortuous watercourse, reflects the combined effects of the discharge of the blowdown, the impoundment of the river and daily riparian releases from Lake Wallace. The water was characterised by high conductivity, pH and sulfate levels, moderate total suspended solids levels and relatively low concentrations of boron, manganese, iron and selenium. The conductivity levels and concentrations of trace metals were considerably lower than at LDP4, which suggests that the inputs from the discharge point had undergone some dilution by the time they reached COX4. Despite this, all the conductivity measurements and a large proportion of the pH levels exceeded their respective upper DTVs, while 49% of the boron concentrations exceeded the trigger value for

95% protection. The conductivity levels were twice as high as those recorded further upstream, suggesting that potential impacts of this parameter on aquatic life would be greater below than above Lake Wallace.

The water further downstream at COX5 was characterised by moderately high conductivity and pH levels, relatively low nutrient concentrations, low turbidity and aluminium concentrations. The conductivity and pH levels were considerably lower than at COX4, indicating that further dilution had occurred, most likely due to inflows from Marrangaroo Creek. Most of the conductivity and nutrient measurements and a large proportion of the pH levels, however, still exceeded their respective upper DTVs.

From the above, it is clear that the contaminants of particular concern are electrical conductivity, pH, boron, selenium and turbidity.

Assessment of Potential Impact of Contaminants Derived from WPS

The assessment of the effects of these discharges on the quality of the water in the Coxs River needs to take the following into consideration:

- natural background concentrations/values of the various water quality parameters;
- other sources of environmental disturbance, including naturally occurring events such as floods and droughts, that may have already had adverse effects on the aquatic ecosystem or be doing so now; and
- the possibility of adverse effects being ameliorated by the flows within the Coxs River and inflows from its tributaries;
- natural physical, chemical or biological processes that may result in changes in levels/concentrations of contaminants or their toxicity;
- changes in operational procedures at WPS.

Information about natural background concentrations/values of various water parameters is limited. In the late 1980s, the headwaters of Coxs River had very low conductivity and were not contaminated by trace elements of natural or anthropogenic origin. The quality of the water in the river upstream of its confluence with Neubecks Creek and Blue Lagoon was also of reasonable quality, despite some inputs.

The quality of water in the Upper Coxs River catchment has been adversely affected by a variety of activities, including clearing of land for agriculture and grazing, mining operations, urban and industrial development and the construction of impoundments. Two of the tributaries that flow into the Coxs River upstream of the power station have been impacted by mining and input elevated concentrations of trace metals into the river. A third tributary carries contaminants from a sewage treatment plant.

The extent to which flows within the river and its tributaries ameliorate the effects of the discharges depends on inflows from surface runoff and groundwater seepage, which, in turn, are determined by rainfall. The dilution of discharges would be much slower during periods of low rainfall, such as that which occurred in 2002/2003.

Changes to operational procedures, such as the conversion of WPS from a wet to dry ashing system and use of make-up water from different sources, may have affected the quantity and quality of the discharges.

The actual threat to aquatic biota posed by the discharges also depends on the rapidity with which their harmful components are diluted within the river and whether releases are continuous, periodic or intermittent. The intermittent discharges from LDP20 and LDP21, for example, are unlikely to have the same impact as those from the more regular discharge points.

Although the conductivity and pH levels at the monitoring locations exceeded the upper DTV, this does not necessarily mean that they will have had adverse effects on aquatic biota. The median conductivity values for the monitoring locations upstream and immediately downstream of Lake Wallace are well below the levels that the scientific literature indicates is harmful to some forms of aquatic biota. The median conductivity at COX4, however, was above this level, which suggests that the blowdown discharge may pose a threat to local aquatic biota. The median conductivity of the water at COX5 was below the level considered to be harmful. It should, however, be noted that the maximum conductivities recorded at LMP8, WX13, COX4 and COX5 exceeded the values considered to be harmful. The median pH levels at these locations fell within the 6.5-9.0 pH range that water quality guidelines around the world indicate is necessary for the protection of freshwater aquatic organisms, as did all the individual measurements. The effect of short-term fluctuations in these water quality parameters on aquatic organisms requires consideration, particularly as changes in pH can increase the toxicity of some contaminants.

Aquatic Biota

The aquatic biota that occurs at CR1 is influenced by natural features, the presence of the upstream and downstream impoundments, release of a constant riparian flow from Lake Wallace except when the dam overspills naturally, release of blowdown water from the Cooling Towers at Wallerawang Power Station and stocking of waters with introduced fish species. The biota comprises a diverse range of macroinvertebrate and periphyton taxa, but is limited to six species of fish, only two of which are native.

The macroinvertebrate fauna associated with the edge habitat was classed as equivalent to AUSRIVAS reference condition more often than significantly impaired. The riffle fauna, however, was classed as significantly impaired more frequently than equivalent to reference condition. The poorer condition of the riffle fauna is probably due to the reduction in flow. The Signal scores indicated the fauna was dominated by pollution-tolerant taxa and that the water was either severely or moderately polluted.

The quantitative samples collected from the riffle habitat have yielded 68 macroinvertebrate and 52 periphyton taxa since autumn 2002, whilst the number of taxa found per survey ranged from 21 to 42 for macroinvertebrates and 15-31 for periphyton. Chironomidae (non-biting midges) were the most abundant macroinvertebrate taxa, except in March 2006 and March 2009, when Hydropsychidae (net-spinning caddisflies) predominated. The composition of the periphyton assemblages was more varied, with either the diatoms *Fragilaria*, *Rhoicosphenia* and *Navicula* or the blue-green alga *Oscillatoria* and *Planktolyngbya* being the most abundant taxon. The multivariate structure and diversity of the macroinvertebrate and periphyton assemblages differed across both the spring and autumn surveys. Marked differences in the annual rainfall and in the number and duration of overspills from Lake Wallace over the period of investigation have probably led to the sampling of assemblages at different successional stages and structures.

Brown trout, mosquito fish and rainbow trout were the most frequently caught fish. The two native fish species (mountain galaxias and flathead gudgeon) were caught only occasionally. From 2002-2004, mosquito fish were the most abundant species, except in spring 2003, when brown trout predominated. The recent samples, however, have been dominated by either mountain galaxias or rainbow trout. The low diversity of fish is probably due to the impoundments restricting the movements of migratory species.

As CR1 is situated approximately 10 km downstream of LDP4, it is unlikely that the blowdown water from the cooling towers would have any detectable effects on aquatic biota, given the dilution effects of the daily 3 ML riparian release from Lake Wallace and natural inflows from the surrounding catchment during periods of rainfall. There is, of course, a possibility of the blowdown water having adverse effects on biota that occur close to its discharge point and of such effects being more widespread during periods of low flow resulting from drought conditions.

Conclusions

The water quality and aquatic ecology of the Upper Coxs River is influenced by a number of factors, including:

- • Presence of two impoundments;
- • Discharges associated with operation of WPS;
- • Inputs from the sewage treatment plant;
- • Inputs from previous and current coal mining; and
- • Stocking with highly predatory introduced fish species.

It is consequently difficult to attribute patterns in water quality and biota in this reach of the river solely to the effects of discharges from WPS.

Recommendations

1) The development of site-specific guidelines is considered inappropriate given the variability in quality and quantity of the discharges and their dilution due to the natural variation in flow within Coxs River.
2) Further site-specific investigations are needed to determine whether the contaminants within the discharges that exceeded the default guideline trigger values are actually having an adverse effect on the biota. As a precautionary approach, management and/or remedial actions that reduce the level of contamination and thereby the risk of adverse biological effects occurring could be implemented, instead of undertaking further investigations.
3) The presence of chemicals within the sediments of Lake Wallace derived from WPS and the sewage treatment plant and the contribution of chemical transformations within the lake to contaminant loads downstream in the river clearly require further investigation.
4) The water quality monitoring programmes undertaken by Delta should continue, but the parameters measured and concentration limits specified should be reviewed.
5) The levels/concentrations of contaminants from other sources that feed into the Coxs River in the vicinity of WPS need to be assessed, so that the contribution from the discharge points can be seen in context. The major additional inputs requiring assessment are those associated with the sewage treatment plant on Pipers Flat Creek, pastoral/agricultural activities in the surrounding catchment and acid deposition from the power stations air emission points.
6) The aquatic biota occurring in the vicinity of the water quality monitoring locations and licensed discharge points that feed directly into Coxs River need to be surveyed and compared with that in appropriate reference locations so that ecological impacts of the discharges from WPS can be assessed more fully.
7) The on-going implementation of the power stations drought management strategy should continue to reduce the salt and pollutant load with an ultimate goal of converting the station to a zero process water discharge site.
8) The composition and abundance of aquatic biota occurring at multiple locations on the Coxs River between Lake Wallace and Lake Lyell needs to be surveyed to gain a better understanding of the impact of variation in natural inflows, the presence of the impoundments, the consistent daily riparian release from Lake Wallace and the blowdown water from the cooling towers. Concurrent surveys also need to be conducted on reaches of similar length situated on nearby reference streams to gain an appreciation of the natural spatial variation in aquatic biota.
9) Consideration should also be given to the use of study methods, such as direct toxicity assessment and bioaccumulation studies, that are appropriate for the detection of changes in the biota associated with contaminants of particular concern.

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Glossary

Word or Acronym	Definition
Aquatic macroinvertebrates	Small animals lacking a backbone, such as insects, worms, crustaceans, molluscs and mites, that spend some or all of their life in freshwater.
Australian River Assessment System (AUSRIVAS)	A software package that provides an assessment of the health of Australian rivers based on samples of aquatic macroinvertebrates collected using a standardised protocol and predictive models that indicate the macroinvertebrates expected to occur at pristine sites with similar chemical and physical characteristics.
Cooling Tower Blowdown	Portion of water flow circulating through a cooling tower that is removed to ensure that the amount of dissolved solids and other impurities remain at an acceptable level.
Electrical conductivity	A measure of the total concentration of inorganic ions (salts) in the water.
Default Trigger Value (DTV)	Guideline trigger value derived from ecosystem data for undisturbed or slightly-modified ecosystems supplied by state agencies.
Guideline Trigger Value (GTV)	Concentration (or load) of a stressor below which there is a low risk that adverse ecological effects will occur
LDP	Licensed discharge point as specified within Environment Protection Licence No. 766 issued under Section 55 of the Protection of the Environment Operation Act 1997
MDS plot	A graphical representation of the relationship between assemblages based on their similarity with respect to overall taxonomic composition and abundance of individual taxa within and among places or times.
OE50 Taxa Score	An ratio calculated by the AUSRIVAS software package that represents the number of aquatic macroinvertebrate families with a greater than 50% predicted probability of occurrence that were actually observed at a site to the number of macroinvertebrate families with a greater than 50 % probability of occurrence predicted to occur at the site.
O0 Signal Score	A biological index that provides an indication of water quality in a river based on the sensitivity of all the different kinds of aquatic macroinvertebrates sampled to chemical pollution.
Percentile	Division of a frequency distribution into one hundredths
Periphyton	Complex mixture of benthic algae that grow on rocks, wood or other substrata in freshwater ecosystems
Permutational analysis of variance (PERMANOVA)	A multivariate statistical procedure that is used to test hypotheses about differences in the structure of assemblages among times or places

pH	A measure of the acidity or alkalinity of water which has a scale varying from 0 (extremely acidic) to 7 (neutral), through to 14 (extremely alkaline).
POEO Act	Protection of the Environment Operation Act 1997
Stressor	Physical or chemical factor that can have adverse effects on aquatic biota or freshwater ecosystems
Toxicant	Chemical contaminant that has the potential to exert toxic effects at concentrations that may occur in the environment
WPS	Wallerawang Power Station

1 Introduction

1.1 Background

Wallerawang Power Station (WPS) generates electricity using coal as the source of energy. The water that is used in the power station's system of cooling towers is primarily drawn from Lake Wallace, which is topped up during dry periods by water from Lake Lyell. Additional cooling water is obtained from mine dewatering projects, and the Fish River Water Supply Scheme provides water for both cooling and other station processes. Water used in the power plant boiler, cooling systems and for suppressing dust on coal stockpiles prior to combustion is discharged into the Coxs River. These discharges and the runoff produced when rain falls on the piles of coal in long-term storage input contaminants to the river. The type and location of the existing discharge points to water are specified in the Environment Protection Licence (EPL) No. 766 for WPS issued under Section 55 of the Protection of the Environment Operations Act 1997 (POEO Act). At present, water from the power station may be fed into the Coxs River at seven licensed discharge points (LDPs):

- LDP1 from Unit 7 Cooling Tower drain
- LDP3 from the Sawyers Swamp Creek Ash Dam via the caustic injection plant
- LDP4 from Unit 7 and 8 Cooling Tower blowdown
- LDP5 from the southern retention basin
- LDP18 the combined overflow drains from the coal stockpile settling basins
- LDP20 from the Centennial-Springvale Water Transfer System bypass point east of Kerosene Vale ash dam
- LDP21 from Unit 8 Cooling Tower drain.

The location of the licensed discharge points and ambient water quality monitoring locations associated with WPS's operations is shown in Figure 1.

All of these above discharge points, except LDP4, are situated upstream of Lake Wallace. The EPL also stipulates that the ambient water quality must be monitored at LMP 7 downstream of LDP20 and upstream of all power station discharge points, and at LMP 8 which is situated upstream of Lake Wallace, but downstream of LDP1, LDP3, LDP5, LDP18 and LDP21. Delta Electricity also monitors the quality of water in the Coxs River downstream of Lake Wallace; at WX13 which is situated upstream of LDP4, at COX4 which is 100 metres downstream of LDP4 and COX5 which is situated even further downstream. The quality of the water at the sites downstream of Lake Wallace is also likely to be influenced by the release of approximately 3 ML/day water from the dam into the Coxs River.

Electrical conductivity and pH have to be measured at all the discharge and monitoring points, but monitoring of the other specified contaminants (i.e. aluminium, arsenic, boron, copper, filterable iron, filterable manganese, fluoride, nickel, selenium, sulfate, zinc and turbidity) is only required at some discharge and monitoring points (see Table 1 for summary of monitoring requirements). The frequency of monitoring required also varies, with concentrations of contaminants at LDPs 4, 7 and 8 being determined at either weekly or monthly intervals, while those at LDPs 1, 3, 5 18, 20 and 21 are determined either weekly or monthly during any discharge. The monitoring records consequently differ across the discharge points and across the contaminants measured.

The Environment Protection Licence also indicates that the volume of liquids discharged to water must be monitored at LDPs 1, 3, 4, 5, 20 and 21 and limits the volume of liquids discharged to water at LDPs 1, 4, 20 and 21 to 210,000 KL/week, 105,000 KL/week, 30,000KL/day and 105,000 KL/week, respectively. The volume of the discharges at LDPs 1, 3, 4 and 5 is monitored continuously, whereas that at LDPs 20 and 21 is monitored daily.

Cardno Ecology Lab was engaged by Delta Electricity to:

- Review, compile and synthesise the monitoring records for the above-mentioned contaminants;
- Assess the potential impact of these contaminants on the aquatic environment and biota in the reach of the Coxs River between Wallerawang Power Station and Lake Lyell by comparing the concentrations of each contaminant with the appropriate ANZECC/ARMCANZ (2000) Water Quality Guidelines or, in the case of those for which there are no guidelines, appropriate scientific literature;

- Assess the changes in periphyton, aquatic macroinvertebrates and fish that have occurred at Location CR1, situated between Lakes Wallace and Lyell, since the inception of the Upper Coxs River Health Monitoring Program.

1.2 Existing Information

Three previous investigations on the Coxs River ecosystem contain information relevant to the current study area. The objectives of these studies, components studied and sites sampled are outlined below. The results of these studies are referred to in the discussion.

In the late 1980s, CSIRO's Division of Coal Technology undertook a three year study of the impact of ash disposal operations at coal-fired power stations in NSW, including that at Wallerawang, on trace metal concentrations in nearby lakes and streams (Jones *et al*. 1989). The study of the impacts of WPS involved measurement of:

- numerous water quality parameters at 17 sites spread across Sawyers Swamp Creek, Upper Coxs River, power station discharge points on two occasions (September 1988 and May 1989);
- pH, conductivity, fluoride, boron, iron, manganese, aluminium, selenium, arsenic and zinc at two sites near Lidsdale Cut on 7 occasions between July 1985 and May 1989;
- pH, conductivity, sodium, magnesium, calcium, sulfate, fluoride, boron, iron, aluminium, manganese, iron, selenium and arsenic in monthly water samples collected at five sites (on Coxs River upstream and downstream of its junction with Sawyers Swamp Creek, on Sawyers Swamp upstream of its junction with Coxs River, on Coxs River 100 m downstream of WPS, and on the drainage creek from oil-water separator 70 m upstream of Coxs River);
- concentrations of ultra-trace elements (cadmium, cobalt, chromium, copper, nickel, lead and zinc) in water at several sites on three occasions*;*
- concentration of mercury in water at 13 locations on 4 occasions; and
- trace metal contamination of sediment samples from key locations on two occasions.

In 1989, the potential threat of the blowdown from the two cooling towers at WPS on the Coxs River ecosystem was assessed (Harris and Hillman 1991). The assessment was based on surveys of aquatic macroinvertebrates and fish undertaken at nine sites representative of the major environmental disturbances that might affect the aquatic ecosystem in Coxs River at Wallerawang. Surveys were conducted at five sites on Coxs River located between Blackmans Flat and downstream of Lake Wallace, one site each on Neubecks Ck, Sawyers Swamp Ck and Piper Flat Ck and a site on the channel carrying the blowdown water from Unit 8 Cooling Tower. Macroinvertebrates were sampled using hand nets and artificial substratum traps. Fish were sampled using electrofishing, gill netting, poisoning and/or push netting techniques. Fish occurring in Lake Wallace were also sampled. Water quality was also measured at each site.

In March 1993, Australian Water Technologies Pty Ltd assessed sediment contamination and river health, as indicated by the aquatic macroinvertebrate fauna, at 46 sites on the Coxs River and its tributaries (O'Connor and Chessman 1994). Study sites in the upper catchment were located on the Coxs River above Lake Wallace (4 sites), on the river between Lake Wallace and Lake Lyell (1), Blackmans Flat Creek (1), Neubecks Creek (1), Sawyers Swamp Creek (1), Pipers Flat Creek (3), Marrangaroo Creek (2) and Farmers Creek (4).

The extent of the impact on the Coxs River catchment by sewage treatment plants, coal-based power stations, coal mining and urbanisation has been examined by measuring levels of trace metal contaminants in fluvial sediments (Birch *et al.* 2001). Samples of sediment were collected from sites on the Upper Coxs River and its tributaries (i.e. Neubecks Creek, Pipers Flat Creek, Sawyer Swamp Creek, Marrangaroo Creek and Farmers Creek). Background concentrations were determined by taking samples from the less impacted parts of the catchment. Samples were also collected from Lake Wallace and Lake Lyell to determine their role in reducing trace metal loading to the downstream reaches of the river and in the case of Lake Wallace the change in contamination with depth of sediment. The bioavailability of the metals and their chemical phases was also discussed.

2 Study Methods

2.1 Review of Water Quality Monitoring Data

2.1.1 Datasets

Delta Electricity provided data from their onsite, offsite and regional water quality monitoring programs in separate Excel spreadsheets. These data consisted of the concentrations/ levels of the various contaminants measured at each of the specified discharge and monitoring points on each sampling occasion. The datasets included contaminants that are directly toxic to biota (e.g. trace metals, ammonia, salinity, pH and DO) and those that are not toxic but which can have adverse effects on aquatic ecosystems and their biota (e.g. nutrients, total suspended solids and turbidity).

The variables recorded and the frequency with which they were measured differed across the three monitoring programs and across the discharge points/monitoring points included within each monitoring program in accordance with varying licence monitoring requirements and intermittent presence of discharge. Information on boron concentrations, for example, is available for eight locations, but only four of these have been monitored frequently. Iron concentrations have been monitored regularly at LMP7, LDP3, LMP8, WX13 and COX4, but only occasionally at LDP4 and once at LDP20. The periods for which each of the contaminants has been monitored at each location are summarised in Appendix 1. The most extensive dataset, that from the onsite water quality monitoring program, contains measurements of electrical conductivity, pH, sulfate and non-filterable residue taken at weekly intervals. It should also be noted that on occasions where there is no flow from the LDPs, a sample is still collected from the associated settling pond during the weekly sampling event to provide a result if flow is detected later in the week. Data from no flow events at LDP1, LDP3, LDP5 and LDP18 have been excluded from the subsequent analyses. The datasets examined therefore cover different time periods.

2.1.2 Summary of data

For the present review, the datasets were summarised in three ways. First, maximum, minimum, average (± 95% confidence interval) and median concentration/value of each of the water quality parameters measured at each discharge point were calculated. Second, composite box and whiskers plots were produced showing:

- median concentration/value (represented by the horizontal line across the box);
- confidence interval about the median (represented by the notch in the box);
- spread of data between the 25th and 75th percentiles (i.e. that within the box); and
- error bars (i.e. whiskers) from the edge of the box to the 5th and 95th percentiles

for each of the water quality parameters measured at each discharge/monitoring point. Third, time-series plots were produced for all the variables monitored on an extensive regular monthly or weekly basis. The linear curve fit function in the graphics package Slide Write Plus for Windows version 7.00 was used to fit the trend line.

2.1.3 Comparison with Concentrations Specified in the EPL

Compliance was assessed by determining how frequently the concentrations/values of sulfate, total suspended solids (TSS), turbidity and pH measured during the onsite water quality monitoring program exceeded the percentile concentration limits specified for each discharge point in the EPL (see Table 2). It is important to note that the 100 percentile concentration limit for pH (6.5-8.5 units) is the same at all discharge points, however, that for sulfate varies across the discharge points, being lower at LDP3 (1200 mg/L) than at LDP1, LDP4 and LDP21 (1600 mg/L) and non-existent for the other discharge points. The EPL also specifies 50th and 90th percentile concentration limits for sulfate at LDP1, LDP4 and LDP21. There are also 100 percentile concentration limits for TSS, turbidity, oil and grease, but these are only applicable to some of the discharge points.

2.1.4 Comparisons with Water Quality Guidelines

The potential impact of the discharges on aquatic biota was evaluated by determining the number of times over the reporting period that the concentrations/values of each of the water quality parameters measured exceeded the relevant ANZECC/ARMCANZ default trigger values (DTVs) for chemical and physical stressors and guideline trigger values (GTVs) for toxicants in slightly disturbed freshwater ecosystems in south-east Australia. DTV/GTVs are the concentrations or loads below which there is a low risk of adverse biological effects occurring. Values in excess of the DTV/GTV indicate that there is a potential risk to biota or ecosystems. It should be noted that these trigger values are normally used for receiving waters beyond the mixing zone rather than applied to discharges. Further site-specific investigations are needed to determine whether or not the stressor is actually having an adverse effect. As trigger values based on ecological effects data or data from appropriate reference systems are not available, this study uses DTVs derived from ecosystem data for modified or slightly-modified ecosystems in south-east Australia that have been provided by state agencies (ANZECC/ARMCANZ 2000). In the case of toxicants, the GTVs that afford 95% and 90% of species protection were considered the most appropriate. It should, however, be noted that the Upper Coxs River is considered a moderate to highly disturbed freshwater system and that the default trigger values applied may therefore be conservative. The median value of most stressors was also compared with the DTVs/GTVs, as recommended in ANZECC/ARMCANZ (2000). There are no water quality guidelines for fluoride, iron, nitrite or sulfate concentrations. In the case of total iron, the Canadian guideline value (CGV) of 0.3 mg/L was used as an interim indicative working level, as suggested by ANZECC/ARMCANZ (2000).

2.2 Aquatic Biota

The aquatic macroinvertebrates and periphyton occurring at three sites within a location known as CR1, situated about 11 km downstream of Lake Wallace, but upstream of Lake Lyell, have been sampled in autumn and spring each year as part of the Coxs River Health Monitoring Program (The Ecology Lab 2003, 2004, 2005, 2006, 2008a and b; Cardno Ecology Lab 2009). The fish occurring at these sites have also been sampled, but only in some years.

2.2.1 Macroinvertebrates

The aquatic macroinvertebrate fauna has been sampled in two ways: using the AUSRIVAS rapid assessment protocol and the more quantitative Surber sampler.

2.2.1.1 AUSRIVAS Methodology

Dip nets (250 µm mesh) were used to sample the aquatic macroinvertebrates occurring in association with riffle and pool edge habitats in accordance with the AUSRIVAS Rapid Assessment Method (RAM) described in Turak *et al.* (2004). Samples were sorted under a binocular microscope (at 40 X magnification), identified to family level and up to ten animals of each taxon counted, in accordance with the AUSRIVAS protocol (Turak *et al.* 2004). Data were analysed using the spring and autumn AUSRIVAS predictive models for edge and riffle habitats in New South Wales (Coysh *et al.* 2000). One-way analysis of variance (ANOVA) was used to assess the statistical significance of differences in mean OE50 Taxa and O0 SIGNAL scores among years.

2.2.1.2 Surber Methodology

At each site, two quantitative samples of fauna were collected from two randomly selected positions in the riffle habitat by placing a Surber sampler (0.29 m^2quadrat, 250 µm mesh) in an upstream direction and vigorously agitating the sediment within the sampler's quadrat for a period of two minutes. The animals occurring in these samples were extracted, identified to family level if feasible and counted in the laboratory. Two multivariate statistical techniques, multi-dimensional scaling (MDS) and permutational analysis of variance (PERMANOVA) were used to examine temporal patterns and differences in the structure of the riffle assemblages, respectively. PERMANOVA was also used to assess the statistical significance of differences in the total number of taxa collected each year. Data from the spring and autumn sampling events were analysed separately.

2.2.2 Periphyton

At each sampling site, two fist-sized cobblestones were collected from the riffle habitat. The periphyton occurring on each rock was removed by brushing the surface of the rock with a toothbrush and collecting the material dislodged in a bucket. The periphyton occurring within a sub-sample drawn from each bucket were identified to genus level and counted by staff at Enviro-Managers Pty Ltd and more recently Just Algae Pty Ltd. The surface area of each rock was also measured, so that periphyton counts could be converted into number of cells per cm^2 of rock surface area.

2.2.3 Fish

The fish and large mobile invertebrates, such as crayfish and shrimps, occurring at each site were sampled using a back-pack electrofisher (Model Smith-Root LR24). The operator of the electrofisher discharges an electric pulse into the water which stuns the fish, allowing them to be easily netted by a second person equipped with a dip net. Electrofishing was done in riffles, shallow pools and beneath overhanging banks, snags and vegetation. The fish captured were counted, identified and released.

3 Water Quality

The quality of the discharges complied with the percentile concentration limits for pH, sulfate and total suspended solids specified in the EPL. A large proportion of the electrical conductivity, pH, boron, selenium and turbidity measurements, however, exceeded their respective ANZECC/ARMCANZ water quality guidelines. The extent of the exceedances that occurred at each discharge point is indicated below.

3.1 Compliance with EPL Pollution Limits

3.1.1 pH

All the pH measurements taken at LDP18, LDP20 and LDP21 were within the 100 percentile limit of 6.5-8.5 pH units (Table 3). The measurements taken at LDP1, LDP4 and LDP5 exceeded the upper limit on a few occasions (3.8%, 1.5 % and 2.7% of monitoring events, respectively). Only the pH measurements taken at LDP 3 fell below 6.5, with 2% doing so.

3.1.2 Sulfate

None of the sulfate concentrations measured at LDP 1, LDP4 and LDP21 exceeded the 100 percentile concentration of 1600 mg/L stipulated in the EPL (Table 3). The concentrations measured at LDP4 exceeded the 90th percentile concentration (1200 mg/L) on 1 occasion, but none of the measurements taken at LDP1 and LDP21 did so. The concentrations measured at LDP1 and LDP 4 exceeded the 50th percentile concentration (1000 mg/L) on 11 (1.7%) and 36 (5.4%) occasions, respectively, but none of the measurements taken at LDP21 exceeded this limit. None of the sulfate concentrations measured at LDP3 exceeded the 100 percentile concentration limit for that location (1200 mg/L).

3.1.3 Total Suspended Solids (Non-Filterable Residue)

The non-filterable residue concentrations at LDP3 and LDP 18 exceeded the 30 mg/L limit on 11 and 10 occasions, respectively (i.e. 2.1% and 7.2 % of monitoring events) (Table 3).

3.2 Comparisons with Water Quality Guidelines for Freshwater Ecosystems

3.2.1 Electrical Conductivity

The median conductivity concentrations at all the discharge and monitoring points exceeded the upper DTV (350 µS/cm) (Table 4a). None of the individual EC measurements taken at the discharge/monitoring points fell below the lower DTV (Table 4a). All the EC measurements taken at LDP3, LDP4, LDP18 and LDP 21 and COX 4 exceeded the upper DTV. The EC levels at all the other monitoring/discharge points except one exceeded the upper DTV on over 90% of the monitoring events (Table 4a). The EC measurements at LDP5, however, only exceeded the upper DTV on 66% of the monitoring events.

3.2.2 pH

The median pH values for the two emergency discharge points (LDP20 and LDP21) and two of the monitoring locations (LMP7 and COX4) exceeded the upper DTV (8.0 units), but those for the other monitoring/discharge points were within the DTVs (Table 4b). None of the median pH measurements for the monitoring points or discharge points fell below the lower DTV (6.5 units) (Table 4b). The pH of the discharge at LDP3 (Sawyers Swamp Creek Ash Dam), however, fell below this limit on 1 occasion (2.1% of the time).

The pH of the water at the two emergency discharge points (LDP20 and LDP21) exceeded the upper DTV (8.0 units) on > 90% of the monitoring events. The pH measurements at LDP1, LDP3, LDP4 and LDP5 exceeded the upper DTV on 44.8%, 6.3% 22.1% and 5.8% of the monitoring events, respectively, but those for the other discharge point (LDP18) only did so on one occasion (0.2% of the total). The pH of the water at the monitoring points (LMP7, LMP8, COX4 and COX5) exceeded the upper DTV more frequently (40.7- 58.9% of the time) than that at the main discharge points. The measurements taken at the two most downstream monitoring points (COX4 and COX 5), in fact, exceeded the upper DTV twice as often as those taken at the monitoring point immediately downstream of Lake Wallace (WX13) and LDP4.

3.2.3 Trace Metals

3.2.3.1 Boron

Boron concentrations were measured at five licenced discharge points and three monitoring locations. The median boron concentration for all of these locations was greater than the 99% protection level (0.09 mg/L) (Table 5a). The median concentration for the monitoring locations (LMP7 and LMP8) and one of the discharge points upstream of the lake (LDP1) was below the 95% protection level (0.37 mg/L). The median value for LDP3 exceeded the 80% protection level (1.3 mg/L), but that for LDP5 was only greater than the 95% protection level. The median value for the discharge point below the lake (LDP4) fell between the 95% and 90% protection levels (0.68 mg/L), while that for the monitoring location further downstream (COX4) was equal to the 95% level.

Over 90% of the boron concentrations measured at LDP1, LDP3, LDP5 and LDP4 and between 50% and 75% of those from LMP7 and LMP8 exceeded the 99% protection level. The boron concentrations measured at LDP3 and LDP4 exceeded the 95% protection level on more than 50% of the monitoring events, while those measured at COX4, LDP5 and LMP8 did so on 20-50% of the monitoring events. Less than 10% of the boron measurements taken at LMP7 and LDP20 and none of those from LDP21 exceeded the 95% protection limit. A large proportion (> 93%) of the boron concentrations measured at LDP3 exceeded the 90%, but only 20-30% of the measurements from LDP5 and LDP4 did so. Only a small proportion (<10%) of the measurements taken at LMP7, LMP8 and COX4 exceeded the 90% protection level. A large proportion (88.6%) of the boron concentrations measured at LDP3 also exceeded the 80% protection level, but less than 10% of the measurements from LDP4, COX4 and LMP8 did so.

3.2.3.2 Manganese

Manganese concentrations were measured at two discharge points and four monitoring locations. The median concentrations at these locations were below the 99% protection level of 1.2 mg/L (Table 5b). The manganese concentrations were below the 99% protection level at three of the monitoring locations (LMP7, LMP8 and COX4) and below on 96.5% of the monitoring events at the fourth monitoring location (WX13). A few of the measurements taken at WX13 exceeded the 95%, 90% and 80% protection levels. The measurements taken at LDP3 exceeded the 99% protection level on 22 % of the monitoring events, but never the 95% level.

3.2.3.3 Selenium

Selenium concentrations were measured at four discharge points and three monitoring locations. The median concentrations at all of the locations except LDP3 were below the 99% protection level (0.005 mg/L) (Table 5c). The median concentration at LDP3 (0.02 mg/L) was in excess of the 90% protection level.

All of the individual selenium concentrations measured at LMP7 and COX4 and the majority (>80%) of the measurements made at LDP1, LDP5, LMP8 and LDP4 were below the 99% protection level. Approximately 84% of the measurements taken at LDP3, however, exceeded this limit. Sixty-eight percent of the measurements made at LDP3 also exceeded the 95% protection level (0.011 mg/L), but less than 10% of the concentrations measured at LDP1, LMP8 and LDP4 did so. Over 50% of the selenium measurements taken at LDP3 also exceeded the 90% level (0.018 mg/L) but less than 5% of the concentrations measured at LDP1, LMP8 and LDP4 did so. Approximately 36% of the selenium concentrations measured at LDP3 exceeded the 80% protection level (0.034 mg/L). A few (1.4%) of the measurements taken at LDP1 also exceeded the 80% protection level.

3.2.3.4 Iron

Iron concentrations were measured at two discharge points and four monitoring locations. As a reliable trigger value for iron has yet to be derived for Australian and New Zealand waters, the Canadian guideline value (CGV) of 0.3 mg/L has been used as an interim indicative working level, as suggested by ANZECC/ARMCANZ (2000). The median iron concentrations at the monitoring locations and one of the discharge points were well below the CGV (Table 5d). The median concentration for LDP4, however, was close to the CGV, but it should be noted that less than 10 measurements were taken at this point.

3.2.3.5 Aluminium

Aluminium concentrations were measured at LMP7 and WX13 (Table 5e). The median concentration at LMP7 exceeded the 90% protection level (0.08 mg/L), but not the 80% level (0.15 mg/L). Approximately 69% of the individual aluminium concentrations measured at LMP7 exceeded the 90% protection level and 40% the 80% protection level. The median concentration for WX13 was below the 99% protection level (0.027 mg/L), however, almost 50% of the measurements exceeded this level and approximately 13 % exceeded the 95% protection level. A small proportion of the measurements also exceeded the 90% and 80% protection levels.

3.2.3.6 Arsenic, Fluoride, Nickel and Zinc

It was also noted that the power station contributed low levels of copper, nickel, zinc and arsenic to the Coxs River, but the concentrations of these elements discharged at LDP4 were below their respective water quality limits.

3.2.4 Nutrients

Nutrient concentrations were measured at LMP7, WX13 and COX5 as part of the regional water quality monitoring program. At all three locations, the median concentrations of ammonium, total nitrogen, filtered ortho-phosphate and total phosphorus exceeded their respective default DTVs (Table 6). The median concentration of nitrogen oxides, however, only exceeded its default DTV at LMP7 and WX13. The median concentration of nitrate at LMP7 and WX13 exceeded the 99% protection level but not the 95% level. The median nitrate concentration at COX5 was below the 99% protection level.

The individual concentrations of total nitrogen, ortho-phosphate and total phosphorus measured at LMP7, WX13 and COX5 exceeded their respective DTVs on 90% or more of the monitoring events (Table 6). The ammonium and nitrogen oxides concentrations measured at these locations were also generally in excess of their respective DTVs, with concentrations at LMP7 and WX13 being more frequently so than those at COX5. Approximately 13% of the individual nitrate concentrations measured at LMP7 exceeded the 95% protection level, but none of the measurements taken at WX13 or COX5 did so.

3.2.5 Turbidity

The median turbidity values for all of these locations except COX5 were within the range of default DTVs for upland rivers (Table 7). The median turbidity value for COX5 was marginally smaller than the lower DTV. Approximately 59% of the turbidity measurements taken at COX5 and 25% of those from WX13 were below the lower DTV. Some of the measurements made at LMP7 and LDP21 were also smaller than the lower DTV. Approximately 28% of the turbidity measurements taken at LDP21 exceeded the upper DTV, but less than 5% of the values for LMP7, WX13, LDP4 and COX5 did so. The latter exceedances are probably due to rainfall in the catchment.

3.3 Spatial Trends in Water Quality Parameters

3.3.1 Electrical Conductivity

The conductivity levels in the occasional discharge to the Sawyers Swamp Creek from the Centennial-Springvale Water Transfer Scheme (LDP20) were greater than those measured at the downstream ambient monitoring location on the Coxs River (LMP7). The overall range and spread of the conductivity measurements in this discharge were also very narrow compared with those for the other locations, indicating that they were less variable (Figure 2a).

The median value, spread and overall range of the conductivity measurements taken in the discharges from the Unit 7 Cooling Tower drain (LDP1) and Sawyers Swamp Creek Ash Dam via the Caustic Injection Plant (LDP3) were much greater than those for LMP7. The overlap in the position of the whiskers, however, indicates that some of the lower conductivity levels recorded at LDP1 were comparable with some of the measurements from LMP7.

The conductivity levels of the discharges from the Coal Settling Ponds (LDP18), Unit 8 Cooling Tower Drain (LDP21) and Southern Retention Basin (LDP5) were often considerably lower than those associated with LDP1 and LDP3. Many of the conductivity levels recorded at LDP18 and LDP21 were elevated relative to those recorded at LMP7. The overall range and spread of the conductivity levels in the discharge from the Southern Retention Basin (LDP5) was smaller than those for LDP18 and LDP21 and more comparable to those at LMP7.

The median conductivity at LMP8, the downstream ambient monitoring location, was considerably greater than that at LMP7 (863 µS/cm vs 668 µS/cm), which implies that the discharges to the river between these two locations were responsible for the increase in conductivity of the river. The larger overall range and spread of the conductivity levels at LMP8 compared with LMP7, indicates that the discharges also increased the variability in conductivity levels within the river.

Although the spread and median value of the conductivity levels recorded at WX13, the monitoring location immediately downstream of Lake Wallace, were smaller than for those for LMP8, their overall range was similar. This either suggests that relatively little dilution of the discharges occurs in Lake Wallace or that the riparian releases contribute to the maintenance of elevated conductivity levels.

The blowdown discharge from the power station cooling towers Units 7 and 8 (LDP4) was also characterised by high, but variable conductivity levels. Most of the conductivity measurements taken at the two monitoring stations further downstream in the river (COX4 and COX5) were lower than the levels in the blowdown discharge, but in excess of the levels recorded at the three monitoring locations further upstream.

The median values for the monitoring locations increased from 668 µS/cm at LMP7 to 863 µS/cm at LMP8, dropped down to 788 µS/cm at WX13, just downstream of Lake Wallace, but rose again to 1612 µS/cm at COX4, but declined further downstream to 1153 µS/cm at COX 5. The median EC concentrations at the discharge points upstream of Lake Wallace ranged from 470 µS/cm at LDP5 to 1754 µS/cm at LDP1 and increased to 2030 µS/cm at LDP4 further downstream.

3.3.2 pH

Figure 2b shows that the pH at the two emergency discharge points (LDP20 and LDP21) was generally greater (i.e. more alkaline) than that at the other locations. The overall range and spread of the pH measurements taken at LDP20 were much narrower than at the other locations and hence less variable. The pH of the discharge from LDP21, in contrast, was characterised by considerable variability, however, it is difficult to assess the significance of these records because of the small number of releases over the monitoring period.

The spread and overall range of the pH levels recorded in the discharge from the Unit 7 Cooling Tower drain (LDP1) was similar to that at the upstream monitoring location (LMP7). This implies that this particular discharge may have had little effect on the pH of the river. The discharge from the ash dam via the Caustic Injection Plant (LDP3), however, was characterised by considerable variation in the overall range and spread of pH values and it was more acidic than any of the other discharges and the water in the river. This acidity is normally corrected at the Caustic Injection Plant when the LDP is in service and flowing. The discharge from the Coal Settling Ponds (LDP18) was slightly more alkaline and consistent in terms of pH than that from LDP3. Most of the pH measurements taken in the discharges from LDP18 were also lower than those recorded at LMP7. The pH levels in the discharge from LDP5 were generally higher than those in the discharges from LDP3 and may help counteract its effects on the Coxs River. The pH at LMP8, the downstream ambient monitoring location, was comparable with that at LMP7. This implies that the discharges from the power station released further upstream had limited effects on the pH of the river.

The range of the pH values recorded at WX13 was comparable with that for LMP8, although the median value was considerably smaller. This implies that the water in Lake Wallace or the riparian release causes a slight reduction in pH. The pH levels in the blowdown discharge (LDP4) were slightly lower than those recorded at WX13. The pH levels recorded at COX4, the monitoring location situated immediately downstream of the confluence with the tortuous watercourse, were slightly higher than at the monitoring locations immediately upstream and downstream and the discharge from LDP4. This suggests that a local source was responsible for the elevation in pH at COX4. The pH levels at COX4 were comparable with those at the two monitoring locations upstream of Lake Wallace.

LDP20 and LDP21 had the largest median pH values. The median pH values for LDP3 (Sawyers Swamp Creek Ash Dam), LDP18 (Coal Settling Ponds) and LDP5 (Southern Retention Basin) were substantially lower than those for the other locations. The median pH values for the ambient monitoring locations upstream and downstream of the power station were similar at 8.05 and 8.00, respectively. The median pH levels below Lake Wallace increased from 7.85 at WX13 to 8.10 at COX4, but declined to 8.00 at COX5.

3.3.3 Sulfate

Sulfate levels in LDP20 and LDP21 were low compared with the other locations (Figure 3a). The levels in the discharge from the Centennial-Springvale Water Transfer Scheme (LDP20) were also less variable than at the other locations. Sulfate levels at the upstream monitoring location (LMP7) were also relatively low (~ 150 mg/L) and fairly stable. The overall range and spread of the sulfate levels in the discharges from the Unit 7 Cooling Tower drain (LDP1), Sawyers Swamp Creek Ash Dam via the Caustic Injection Plant (LDP3) and Southern Retention Basin (LDP5) were much greater than those for LMP7. The median sulfate concentration for LDP1 (524 mg/L) and LDP3 (1016 mg/L) was much greater than that for LMP7 (150 mg/L) and LDP5 (130 mg/L)(Table 8a). The sulfate levels at the downstream ambient monitoring location (LMP8) indicate that the inputs from LDP1 and LDP3 had undergone some dilution. The median concentration (230 mg/L), spread and overall range at LMP8, however, were elevated relative to those at LMP7.

The median sulfate concentration (223 mg/L) and overall range at the monitoring station immediately downstream of Lake Wallace (WX13) was similar to that at LMP8, although the spread of values was smaller. The discharge from the Units 7 and 8 Cooling Tower Blowdown downstream of Lake Wallace (LDP4) was characterised by high sulfate levels (median value of 760 mg/L) and considerable variability. Although the lower median sulfate level at COX4 (561 mg/L) suggests that the inputs from LDP4 had undergone some dilution, the levels were highly variable and considerably greater than at any of the other monitoring locations in the river.

3.3.4 Total Suspended Solids

The median value, range and spread of the TSS levels in the discharge from the Centennial-Springvale Water Transfer Scheme (LDP20) were larger than those for the other locations (Figure 3b). Considerable variation in TSS levels was also evident in the discharges from the Unit 7 Cooling Tower drain (LDP1), Ash Dam via the Caustic Injection Plant (LDP3), Coal Settling Ponds (LDP18) and Units 7 and 8 Cooling Tower Blowdown (LDP4). The median TSS level in the discharge from LDP1 was elevated relative to that for the upstream monitoring location (LMP7) but similar to that for the discharge points upstream of Lake Wallace. The median TSS level for LMP8, the downstream monitoring location, (8.4 mg/L) was greater than that for LMP7 (5.4 mg/L), suggesting that the discharges from the power station may be responsible for the increased levels. Most of the TSS levels recorded at WX13, the monitoring location downstream of Lake Wallace were smaller than those for LMP8. This implies that the TSS levels are reduced by some process within the lake or by the riparian releases. The TSS levels recorded in the blowdown discharge (LDP4) were generally greater than those at WX13 and presumably contribute to the elevated levels recorded at COX4. It should, however, be noted that the median TSS concentration is greater for COX4 (9.25 mg/L) than for LDP4 (6 mg/L). This suggests that the discharge may not be the only source of the TSS. The median TSS level at COX4 was also considerably greater than that for LMP7.

3.3.5 Turbidity

The range and spread of the turbidity measurements taken at the discharge point for the Centennial-Springvale Water Transfer Scheme (LDP20) were larger than those for any of the other locations (Figure 4a). The discharges from LDP21 and LDP4 also tended to be more turbid than the water in the river (i.e. at LMP7, WX13 and COX5). Figure 4a also shows that the water at LMP7 was generally more turbid than at the two monitoring locations downstream of Lake Wallace.

3.3.6 Fluoride

The discharges from the Unit 7 cooling tower drain (LDP1), the Unit 8 cooling tower drain (LDP21) and Units 7 and 8 Cooling Tower Blowdown (LDP4) were characterised by moderate, but relatively stable fluoride levels compared with the elevated and highly variable levels recorded in the discharge from Sawyers Swamp Creek Ash Dam via the Caustic Injection Plant (LDP3) (Figure 4b) The fluoride concentrations at the upstream monitoring location (LMP7) were relatively low (median = 0.38 mg/L) and stable in comparison to these discharges (medians varied from 1.04-3.16 mg/L), but comparable with those recorded in the discharge from the Southern Retention Basin (LDP5)(median = 0.50 mg/L) (Table 8b). The fluoride levels at the downstream monitoring location (LMP8) (median = 0.96 mg/L) were greater than those at LMP7, which suggests that the elevation was due to the discharges from the power station. The limited data available from WX13 (median = 2.42 mg/L) suggests that fluoride levels may be elevated further by inputs from Lake Wallace.

3.3.7 Trace Metals

3.3.7.1 Boron

The discharge from the Sawyer Swamp Creek Ash dam via the Caustic Injection Plant (LDP3) was characterised by large, but highly variable boron concentrations (Figure 5a). Some of the measurements from LDP1 and LDP5 were elevated relative to the concentrations at the upstream ambient monitoring location (LMP7). The inputs from these two discharge points, however, are difficult to assess, because only a few measurements were taken at each. The data from LMP8 indicated that the inputs of boron from the ash dam and the other discharge points had undergone considerable dilution before the Coxs River enters Lake Wallace.

The inputs from the power station raised the median boron concentration from 0.12 mg/L at LMP7 to 0.28 mg/L at LMP8. The median boron concentration at LDP4 (0.46 mg/L), indicated that there was some input from Units 7 and 8 Cooling Tower blowdown The lower boron concentrations measured at COX4 indicate that there has been some dilution of these inputs, however, the median value (0.37 mg/L) was still considerably greater than at LMP7.

3.3.7.2 Manganese

The discharge from the Sawyer Swamp Creek Ash Dam via the Caustic Injection Plant (LDP3) was also characterised by large, but highly variable manganese levels (Figure 5b). Most of the measurements taken at this discharge point exceeded the concentrations recorded at the upstream ambient monitoring location (LMP7). The manganese concentrations recorded at the downstream monitoring location (LMP8) indicated that the inputs from LDP3 were rapidly diluted; however, the levels were still elevated relative to LMP7. The manganese concentrations recorded at the monitoring location below Lake Wallace were generally similar to those at LMP8, but highly variable. This suggests that there may be some addition of manganese from Lake Wallace. Moderate concentrations of manganese were recorded at LDP4, however, the significance of these is also difficult to assess because of the small number of records. The manganese concentrations recorded at COX4 were generally much lower and more comparable to LMP7, suggesting that the inputs from Lake Wallace and LDP4 had undergone considerable dilution.

The inputs from the power station increased the median manganese concentration from 0.06 mg/L at LMP7 to 0.10 mg/L at LMP8. The median manganese concentration in the river downstream of Lake Wallace declined from 0.06 mg/L at WX13 to 0.01 mg/L at COX4.

3.3.7.3 Iron

The iron concentrations measured at the two ambient monitoring locations (LMP7 and LMP8) were generally low, but characterised by considerable variation (Figure 6a). The iron levels at LDP3, were much lower and more consistent. The concentrations recorded at the other two monitoring locations (WX13 and COX4) were also generally lower than at LMP7 and LMP8, suggesting that Lake Wallace contributed to their decline. The discharge at LDP4 was characterised by high, but variable iron concentrations on the few occasions that levels were measured. The lower iron concentrations recorded at COX4 indicate that the iron inputs from LDP4 were rapidly diluted.

The median iron concentration at LMP7 (0.060 mg/L) was twice that at LMP8 (0.030 mg/L), suggesting that the discharges from the power station may have reduced the levels in the river. Further decreases in median iron concentration were evident at WX13 (0.020 mg/L) and COX4 (0.010 mg/L).

3.3.7.4 Selenium

Low levels of selenium were recorded at the upstream ambient monitoring location (LMP7) (Figure 6b). The discharge from the ash dam (LDP3) was characterised by large, but highly variable selenium levels. The inputs from LDP1 and LDP5, in contrast, were elevated only slightly relative to LMP7. The selenium concentrations recorded at LMP8 indicate that the inputs from LDP3 had undergone considerable dilution, although some of the levels recorded still exceeded that recorded at LMP7. The selenium levels recorded in the discharge from the Units 7 and 8 Cooling Tower blowdown were slightly higher than at LDP1 and LDP5, but comparable to those recoded at LMP8. The selenium levels at COX4 were also low, indicating that the inputs from LDP4 were rapidly diluted. The median selenium concentration was 0.003 mg/L at all three monitoring locations (i.e. LMP7, LMP8 and COX4).

3.3.7.5 Nutrients

The range and spread of Total Nitrogen concentrations was greater at LMP7 than at WX13 and COX5. The spread of values at WX13 and COX5 were within that at LMP7 (Figure 7a). The median total Nitrogen concentrations, however, were the same for LMP7 and WX13 (0.700 mg/L) and only slightly greater than for COX5 (0.600 mg/L).

The spatial trend in Total Kjeldahl Nitrogen (TKN) did not match that in Total Nitrogen concentrations. The range and spread of TKN concentrations at LMP7 was also greater than those at WX13 and COX5. The spread of TKN concentrations at LMP7 overlapped some of the spread at COX5, but none of that at WX13 (Figure 7b). The elevation in TKN levels between LMP7 and WX13 may be due to discharges from the power station or the riparian releases from Lake Wallace or a combination thereof.

The range of total Ammonium as Nitrogen concentrations was much smaller at COX5 than at WX13 and LMP7 (Figure 7c). The spread of the Ammonium as Nitrogen concentrations at COX5, however, showed some overlap with those at the other two locations. The median concentrations increased from 0.030 mg/L at LMP7 to 0.040 mg/L at WX13, but had declined to 0.020 mg/L by COX5. The slight elevation in Ammonium as Nitrogen levels between LMP7 and WX13 may be associated with discharges from the power station or the riparian releases from Lake Wallace or both.

The spatial trends in Total Nitrogen Oxide and Nitrate concentrations were similar (Figure 7d and e). The range and spread of concentrations of these two nutrients was greater at LMP7 than at WX13 and COX5. The spread of the values at WX13 and COX5, however, was within that at LMP7. The median concentration of Total Nitrogen Oxide decreased from 0.165 mg/L at LMP7 to 0.050 mg/L at WX13 and to 0.010 mg/L at COX5. The corresponding change in median nitrate concentrations was from 0.160 mg/L to 0.040 mg/L and 0.010 mg/L. The range of Total Nitrite concentrations at LMP7 and WX13 was greater than at COX5, however, the spread of values at COX5 fell within that at the other two locations (Figure 7f).

The spatial trends in Total Ortho-Phosphate and Total Phosphorus were similar (Figure 8a and b). The range and spread of values for these two nutrients was greater at LMP7 than at WX13, which in turn was greater than that at COX5. The median orthophosphate concentration declined from 0.615 mg/L at LMP7 to 0.270 mg/L at WX13 and 0.120 mg/L at COX5. The corresponding change in median total phosphorus concentrations was from 0.255 mg/L to 0.140 mg/L and 0.070 mg/L. These data suggests that there may be a downstream gradient in orthophosphate and total phosphorus.

3.3.7.6 Other Water Quality Parameters

The dissolved oxygen concentrations at WX13 were more skewed towards low values than the measurements from LMP7 and COX5 (Figure 8c).

Measurements of aluminium concentrations were available for two monitoring locations, LMP7 and WX13. Limited data were also available from three discharge points, LDP1, LDP3 and LDP4. The range and spread of concentrations was much greater at LMP7 than at WX13 (Figure 8d). The median value declined from 0.115 mg/L at LMP7 to 0.020 mg/L at WX13. The aluminimum levels in the discharges from LDP1 and LDP4 were elevated relative to these monitoring locations. The range and spread of concentrations recorded at the other discharge point (LDP3) was much smaller and the median concentration at this point was similar to that at LMP7. The difference in concentration between LMP7 and WX13 implies that aluminium inputs between these points were diluted by inflows into the river and Lake Wallace.

3.4 Temporal Trends in Water Quality Parameters

3.4.1 Electrical Conductivity

The trend lines superimposed on the time series indicate that EC values have increased over time at both of the ambient monitoring locations, two discharge points (LDP1 and LDP4) and at COX4, the monitoring point downstream of the confluence with LDP4. There has been a marked decrease in variability of EC over time at WX13, but little change at the other locations (Figure 9). The increases at LDP1 and LDP4 were the most marked. At LDP3 and LDP5 a slight decrease and increase in EC values is evident over time. The gradual increase in background EC levels over time that occurred at LMP7 implies that some factor other than activities at WPS may be a contributor to this trend. The time series also showed that conductivity levels were highly variable at all locations and that values occasionally dropped below the upper DTV at all the locations except LDP18, LDP4 and COX4. The box and whisker plots also showed that the EC levels were less variable at LDP20, LMP7, LDP18 and LDP21 than at the other locations (Figure 5a). The low variability at LDP20 and LDP21 reflected the relatively small number of monitoring events at these locations.

3.4.2 pH

The trend lines showed the pH values have declined slightly over time at the upstream monitoring point (LMP7), there has been little change at LDP18 and COX4, but an increase at all the other monitoring and discharge points (Figure 10). The extent of the increase varied from less than 0.5 of a pH unit at LDP3, LDP5, LMP8, WX13 and COX5 to between 0.5 and 1.0 pH unit at LDP1 and LDP4. The time series (Figure 10) and box and whisker plots (Figure 5b) both show that the pH levels at LDP3 were more variable than at the other monitoring and discharge points.

3.4.3 Sulfate

There has been a marked increase in sulfate concentrations over time at LDP1, LDP4 and COX4, but only a slight increase at LMP8 (Figure 11). At LDP3, there has been a decrease in sulfate concentrations over time. Considerable temporal variability in sulfate levels was also evident at LDP1, LDP3, LMP8, LDP4 and COX4 throughout the monitoring period. At LDP5 and WX13 there has been a gradual decline in sulfate levels over time. At WX13, sulfate concentrations were less variable during the second half of the monitoring period.

3.4.4 Total Suspended Solids

Figure 12 shows that the TSS concentration has increased slightly over time at LDP1, LDP4 and LDP5, but remained fairly stable at the other discharge points. At three monitoring locations (LMP7, WX13 and COX4) there has been a slight decline in TSS concentrations over time; LMP8, however, shows no trend.

3.4.5 Fluoride

There appears to have been a marked decrease in fluoride concentrations over time at LDP3, particularly during the latter half of the monitoring period, and slight decreases over time at three of the other discharge points (LDP1, LDP5, LDP4) and one monitoring location (WX13) (Figure 13). The fluoride concentrations at the two ambient water monitoring locations (LMP7 and LMP8), however, have remained relatively stable over time.

3.4.6 Trace Metals

3.4.6.1 Boron

The temporal trends in boron concentrations differed among the monitoring locations, with a slight increase being evident at LMP7, a slight decline at LMP8, but no perceptible change at COX4 (Figure 14). The temporal trends in boron concentrations also differed at the two discharge points where routine monitoring was undertaken, with a marked decline being evident at LDP3, but only a slight decline at LDP4.

3.4.6.2 Manganese

The temporal trends in manganese concentrations also differed among the monitoring locations with a slight decline being evident at LMP7 and LMP8, a slight increase at LDP3 and COX4, but a marked decline at WX13 during the latter half of the monitoring period (Figure 15).

3.4.6.3 Iron

At LMP7, LDP3, WX13 and COX4 there has been a slight decline in iron concentration over time, however, at LMP8 there has been a slight increase (Figure 16).

3.4.6.4 Selenium

At LMP7 and LMP8 there have been slight increases and decreases in selenium concentration over time, respectively, but little change at COX4 (Figure 17). At two of the licensed discharge points there has been a slight decline in selenium concentrations over time (LDP1 and LDP4), but a marked decline at LDP3.

3.4.6.5 Aluminium

At LMP7, there has been a slight increase in aluminium concentration over time, but no obvious change at WX13 (Figure 18). Trends at the discharge points were not examined because of the paucity of data.

3.4.7 Nutrients

At LMP7, there is no clear trend in total nitrogen concentrations, however, at WX13 and COX5 a slight decrease has occurred over time (Figure 19). All three locations, however, showed a clear downward trend in total Kjeldahl Nitrogen (Figure 20). At LMP7, there is no clear trend in total ammonium as nitrogen concentrations, however, at WX13 and COX5 an increase has occurred over time (Figure 21).

The total nitrogen as oxides and nitrate concentrations increased over time at LMP7 and COX5, but remained relatively stable at WX13 (Figures 22 and 23, respectively). The total nitrite concentrations at all three locations, however, declined slightly over time (Figure 24).

The total ortho-phosphate and phosphorus concentrations decreased slightly over time at LMP7 and WX13, but increased slightly at COX5 (Figures 25 and 26, respectively).

3.4.8 Dissolved Oxygen

A downward trend in dissolved oxygen levels was evident at all three monitoring locations (Figure 27).

4 Aquatic Biota

4.1 Macroinvertebrates

4.1.1 AUSRIVAS Samples

AUSRIVAS indices for macroinvertebrate assemblages associated with the edge and riffle habitats at CR1 sampled from 2002 -2009 are presented in Table 9. CR1 is a location situated about 11 km downstream of Lake Wallace, but upstream of Lake Lyell, that has been sampled in autumn and spring each year as part of the Coxs River Health Monitoring Program.

4.1.1.1 Edge Habitat

In spring 2002 and 2008, the macroinvertebrate fauna associated with the edge habitat of all three study sites was classed as significantly impaired (AUSRIVAS Band B). In the intervening years, the fauna was generally equivalent to the AUSRIVAS reference condition (AUSRIVAS Band A). This implies that the overall condition of the fauna associated with the edge habitat was better in 2003-2007. Statistically significant differences in the mean OE50 taxa scores, the index from which the AUSRIVAS bands are derived, were restricted to two pairs of years: 2002 vs 2003 and 2002 vs 2005, with values being lower in 2002 (Figure 28a).

The mean O0Signal scores for the spring of 2002, 2003, 2004 and 2006 were < 4 which indicates that the majority of taxa found were pollution-tolerant and that the water was severely polluted (Figure 28a). The O0Signal scores for 2007-2009, however, were >4, indicating that the water was moderately polluted. It should, however, be noted that the differences in O0Signal scores among years were not statistically significant. This reflects the small range in values (3.82-4.09) over the study period.

In autumn 2002, 2004, 2007 and 2009, the macroinvertebrate fauna associated with the edge habitat were in a slightly better condition on average (all sites classed as AUSRIVAS Band A) than in 2003, 2006 and 2008 (some sites with B band others with A band) (Table 9). The differences in the mean OE50 taxa scores among years, however, were not statistically significant (Figure 29a). The mean O0Signal scores were similar between 2002 and 2008, with values for 2002, 2003 and 2004 being just below 4, whilst those for 2006, 2007 and 2008 were just greater than 4. The mean O0 Signal score for 2009 was significantly greater than that for three of the previous samples (2004, 2006 and 2008) (Figure 29a)

4.1.1.2 Riffle Habitat

The AUSRIVAS bands indicate that the riffle fauna sampled in the spring of 2002 and 2006-2009 was more impaired than that sampled in the spring of 2003 and 2004 (Table 9). The differences in OE50 taxa scores among years were not statistically significant. Figure 28b shows that the variation in OE50 taxa scores among sites was greater in 2002-2004 than in 2006-2008. The mean O0Signal scores were generally > 4, indicating that the water may have been moderately polluted. The O0Signal scores for 2005, however, was < 4. The mean O0Signal score for 2005 was significantly smaller than that for 2002, 2003, 2004 and 2009 (Figure 28b).

The AUSRIVAS bands indicate that the riffle fauna sampled in autumn 2002 and 2003 was in a poorer condition than that in the subsequent autumn surveys (Table 9b). The mean O0 Signal scores for all seven samples were between 4 and 5, indicating that there may have been exposure to moderate pollution (Figure 29b). The differences in mean OE50 taxa scores and O0 Signal scores among years were not statistically significant (Figure 29b).

4.1.2 Surber Samples

The riffle habitat at CR1 supported a diverse range of macroinvertebrates, with a total of 68 separate taxa having been recorded since autumn 2002. The number of taxa found per survey was considerably lower, varying from 24 to 42 in autumn and 21 to 36 in spring. Chironomidae (non-biting midges) were the most abundant taxa on all the sampling occasions except for March 2006 and March 2009, when Hydropsychidae (net-spinning caddisflies) predominated (Table 10). These two taxa accounted for between 34% and 81% of the macroinvertebrates found per survey. Caenidae (mayflies), the second most abundant taxon found between March 2002 and March 2004, accounted for between 10% and 30% of the total number of macroinvertebrates per survey. At other times, either

Chironomidae, Hydropsychidae or Oligochaeta (segmented worms) was the second most abundant taxon. On these occasions, their contribution to the overall numbers varied from 15-37%, 7-17% and 16-19%, respectively.

There were significant differences in the multivariate structure of riffle assemblages among years (Table 11). Significant differences in structure were detected between all pairs of autumn surveys and most pairs of spring surveys, except for 2002 vs 2003, 2002 vs 2004, 2006 vs 2009 and 2007 vs 2008. The MDS plot indicates the assemblages surveyed in the spring of 2002, 2003 and 2004 were generally distinct from those sampled during the other spring surveys (Figure 30a). It also shows that the assemblages surveyed in spring 2008 and 2002 were the least and most variable, respectively. The assemblages sampled in autumn 2002 and 2003 were also distinct from those sampled during the other autumn surveys (Figure 30b). Significant differences in the average numbers of macroinvertebrate taxa per sample were also detected, but only among some pairs of years (Table 12). Figure 31 shows that the samples collected in the spring of 2006 and 2007 and autumn of 2004 and 2008 had the least and most diverse macroinvertebrate fauna, respectively.

4.2 Periphyton

The riffle habitat supports a diverse range of periphyton with a total of 52 taxa having been recorded since autumn 2002. Considerably fewer taxa were found per survey, with numbers varying from 15 (October 2009) to 29 (November 2005) in spring and from 16 (March 2006) to 31 (March 2002) in autumn. The diatoms *Rhoicosphenia, Navicula, Amphora, Fragilaria* and *Nitzschia* were the most frequently occurring taxa. The numerically dominant taxa were not consistent across either survey period. In autumn, either the diatoms *Fragilaria* (March 2002, March 2003, March 2004 and April 2008), *Navicula* (April 2007) and *Rhoicosphenia* (March 2009) or the blue-green alga *Oscillatoria* (March 2006) were the most abundant taxon (Table 13). In spring, either the blue-green alga *Planktolyngbya* or the diatoms *Fragilaria* (November 2003), *Rhoicosphenia* (November 2005) and *Navicula* (October 2006, November 2007, November 2008 and October 2009) were the most abundant taxa. The dominant taxon accounted for between 21% and 73% of the total periphyton cells counted each survey.

Analyses based on the composition and relative abundance of periphyton indicated that there were significant differences in the multivariate structure of these assemblages among years (Table 14). This difference was evident during both survey periods and between all pairs of years. The MDS plots indicate the assemblages surveyed in 2002 and 2003 were distinct from those sampled subsequently and that this difference was evident in both spring and autumn (Figure 32a and b, respectively). The spring 2004 and 2009 samples appeared to be distinct from those collected in 2006, 2007 and 2008. The samples surveyed in autumn 2004 were also distinct from those sampled at other times. Significant differences in numbers of periphyton taxa were also detected, but only among some pairs of years (Table 15). Figure 33 shows that the samples collected in the spring of 2006 and 2007 and autumn of 2004 and 2008 had the least and most diverse periphyton, respectively.

4.3 Fish

The fish fauna appears to be limited to two native species, flathead gudgeon (*Philypnodon grandiceps*) and mountain galaxias (*Galaxias olidus)* and four alien species, brown trout (*Salmo trutta*), rainbow trout (*Oncorhynchus mykiss*), goldfish (*Carassius auratus*) and mosquito fish (*Gambusia holbrooki*) (Table 16). The native fish were caught less frequently than the other species, with mountain galaxias being found in spring 2006 only and flathead gudgeon in autumn 2002 and spring 2008. Brown trout, mosquito fish and rainbow trout were the most frequently caught fish, being represented on seven, six and five of the nine sampling events, respectively. From 2002-2004, mosquito fish were generally the most abundant species caught, except in spring 2003, when brown trout predominated. Mountain galaxias were the most abundant species caught in spring 2006, whereas rainbow trout were in spring 2008 and autumn 2009.

5 Discussion

5.1 Water Quality

5.1.1 Important Considerations

The assessment of the effects of the discharges from WPS on the quality of the water in Coxs River needs to take the following into consideration:

- natural background concentrations/values of the various water quality parameters;
- other sources of environmental disturbance, including naturally occurring events such as floods and droughts, that may have already had adverse effects on the aquatic ecosystem or be doing so now;
- the possibility of adverse effects being ameliorated by flows within Coxs River and inflows from its tributaries;
- natural physical, chemical or biological processes that may result in changes in levels/concentrations of contaminants or their toxicity; and
- changes in operational procedures at WPS.

Further information on natural background concentrations/values of water quality parameters is needed to determine whether the exceedances of guidelines values reported above are likely to pose a threat to aquatic biota. Studies undertaken prior to the construction of the power station or in undisturbed regions of the Upper Coxs River catchment would provide the most useful basis for comparison. In the late 1980s, CSIRO noted that the headwaters of Coxs River had very low conductivity and were not contaminated by trace elements of natural or anthropogenic origin (Jones *et al.* 1989). Limited data are also available from Blackmans Flat on the Coxs River upstream of its confluence with Neubecks Creek and Blue Lagoon. These data indicated the water was of reasonable quality, despite some inputs of major ions (Harris and Hillman 1991).

The Upper Coxs River catchment has been degraded by the clearing of land for agriculture and grazing, mining operations, urban and industrial development and construction of impoundments. Two of the tributaries that flow into the Coxs River upstream of LMP7 and the power station, Neubecks Creek and Sawyers Swamp Creek, have been impacted by mining. Neubecks Creek receives acid mine drainage from exposed coal deposits associated with a derelict mine and water pumped from a nearby underground mine site (Battaglia *et al.* 2005). The water in this creek has a lower pH, higher conductivity and higher concentrations of dissolved calcium, cadmium, magnesium, manganese, nickel, potassium, sulphur and zinc than nearby reference sites. Sawyers Swamp Creek receives inputs of iron, manganese, boron, fluoride, selenium and acid due to seepage from the old open cut mine “Lidsdale Cut” and inputs of iron, manganese and acid due to seepage from the base of the Sawyers Swamp Creek ash dam associated with WPS. There are also small inputs of selenium and antimony downstream of Lidsdale Cut, which have been attributed to the groundwater that percolates through the old colliery and ash fill area (CSIRO 1989). Jones *et al.* (1989) reported high concentrations of manganese in the Coxs River downstream of its confluence with Neubecks Creek and that further increases occurred as the river passed the disused Newcom Colliery and the junction with Sawyers Swamp Creek. They also noted that although boron, selenium and fluoride concentrations in Sawyers Swamp Creek exceeded their respective water quality guidelines, levels dropped below these limits in the Coxs River immediately downstream of its junction with this creek. Previous studies have shown that Pipers Flat Creek, a relatively large tributary, which flows into Coxs River just above the Wallerawang Power Station has high levels of phosphorus in the sediment (Jones *et al.* 1989), relatively high conductivity and chloride levels and a different major ion composition to that of nearby sites (Harris and Hillman 1991). These authors attributed the high conductivity and phosphorus levels to the discharges from the sewage treatment plant situated on this tributary. Agricultural practices in the catchment, acid deposition from the power stations air emission points and other anthropogenic activities may also result in changes in levels/concentrations of contaminants and could even affect toxicity by altering the pH of the water.

The amelioration of adverse effects resulting from the discharges depends on the rapidity of dilution by flows within the Coxs River and inflows from its tributaries. The flows within the river and its tributaries depend on inflows from surface runoff and groundwater seepage, which, in turn, are determined by rainfall. The dilution of discharges will consequently be slower during periods of low rainfall. The annual rainfall at Lithgow since 1997/1998 has varied from 446 mm in 2002/2003 to 1092 mm in 1999/2000 and been below the long-term average on 7 out of the past 12 years (Delta Electricity 2009). The rate of dilution of the blowdown discharge (LDP4) may be less variable than at the other LDPs, because of the daily riparian release from Lake Wallace.

Natural physical, chemical or biological processes may also result in changes in levels/concentrations of contaminants or their toxicity. The pH of the water, for example, may be influenced by the amount of plant growth and organic material present within Coxs River and the geology of the bedrock. Changes in pH may also increase the toxicity of contaminants such as ammonia and aluminium (Alabaster & Lloyd 1982; Collier & Winterbourn 1987; CCREM 1991). The toxicity of trace metal contaminants also depends on the hardness of the water.

Changes in operational procedures at Wallerawang Power Station could affect both the quality and quantity of the discharges. The conversion of Wallerawang Power Station to a dry ashing system has significantly reduced the need to discharge from the Sawyers Swamp Creek Ash Dam. The water level within this dam varies with rainfall and is currently managed by beneficially reusing this water for ash conditioning and dust suppression on the ash repository. Excess water may be discharged to the Coxs River through the caustic injection plant settling ponds and LDP3. The discharge is consequently quite variable with none having occurred in 2005/2006 (Parsons Brinckerhoff 2006), but a total of 415 ML, 786ML and 359.4 ML having been discharged in 2006/2007, 2007/2008 and 2008/2009, respectively (Delta Electricity 2007, 2008 and 2009). The average discharge volume, on the days discharge occurred, over this three year period has varied from 4.4-7.3 ML/day.

The evaporative cooling process used by the power station causes an increase in concentration of many of the contaminants. This concentration effect is proportional to the number of cycles of concentration and conductivity set point for the cooling towers. Typically, the towers are set to a maximum conductivity of 2,600µS/cm and a limit of between 2 and 3 cycles. This process is limited by the quantity and quality of make-up water available and the volumetric limitation in transferring and discharging water. Historically, pre 2000, the conductivity of Lake Wallace was in the order of 500 - 600µS/cm and the power station was able to access 100% of its allocation of Fish River water with a typical conductivity of <100µS/cm. The lowering of the cooling tower conductivity set point, while maintaining generation capacity can only be achieved by a lowering of the conductivity for the cooling tower make up supply.

With the implementation of Delta's Drought Management Strategy and Pollution Reduction Program, a measureable improvement in general water quality is expected. The reinstatement of the Kerosene Vale Ash Dam seepage collection and diversion system and the construction of the Sawyers Swamp Creek Ash Dam seepage collection and return system should result in decrease in trace metals in upstream water quality.

Condenser replacement with more corrosion resistant material at both unit 7 and 8, coupled with a conversion to less corrosive chlorine dioxide and use of corrosion inhibitors, should see a reduction in trace metal concentration in the cooling tower blowdown.

The construction and operation of the Reverse Osmosis Plant will see a significant reduction in the volume of cooling tower blowdown discharged which will result in a reduction in salt load discharged and a consequent reduction in salininty (electrical conductivity) in the Coxs River below Lake Wallace.

5.1.2 Effects of the Licensed Discharges on Water Quality

The discharge point from the Centennial-Springvale Water Transfer System bypass east of Kerosene Vale Ash Dam (LDP20) flows into the Sawyers Swamp Creek upstream of the ambient water quality monitoring point LMP7. This intermittent discharge is characterised by high pH, turbidity and TSS levels, moderately high conductivities and low concentrations of sulfate. The conductivity and pH values exceeded the upper DTV on > 90% of the monitoring events and turbidity levels did so on >25% of the events.

The water quality monitoring undertaken at the upstream ambient monitoring location (LMP7) provides an insight into the downstream extent of the various inputs from the upstream tributaries. The water at LMP7 was characterised by relatively high conductivities, moderately high pH, and low concentrations of sulfate, TSS, turbidity, fluoride, boron, manganese and selenium. This suggests that inputs from the two upstream tributaries are diluted rapidly by flows within the river and inflows from Pipers Flat Creek. Jones *et al.* (1991) also noted that trace metal inputs from the upstream tributaries were rapidly diluted in the Coxs River. Nevertheless, the conductivity, pH, aluminium, and nutrient levels generally exceeded their respective upper DTVs. The high conductivity and nutrient levels recorded at LMP7 may reflect inputs from the sewage treatment plant on Pipers Flat Creek.

The reach of the Upper Coxs River below LMP7 but above Lake Wallace receives inputs from five of the discharge points associated with Wallerawang Power Station (LDP1, LDP3, LDP18, LDP5 and LDP21). The intermittent discharge from the Unit 7 Cooling Tower drain released at LDP1 is characterised by high conductivities and sulfate concentrations. Most (95%) of the conductivity and a large proportion (45%) of the pH measurements taken at this point exceeded their respective upper DTVs. The discharge from Sawyers Swamp Ash Dam is released via LDP3 into Springvale Creek, which flows into Coxs River. This discharge is characterised by high conductivities, low pH, high concentrations of sulfate, fluoride, boron, manganese, and selenium and low iron concentrations. The input of all of these contaminants, except iron, is also highly variable. A large proportion of the conductivity, boron and selenium measurements taken in this discharge exceeded the upper DTV and GTVs for protection of 90% and 80% of aquatic species in slightly-moderately disturbed systems, respectively. The irregular discharge from the coal settling ponds that enters Coxs River at LDP18 is characterised by moderately high conductivities in excess of the upper default trigger value, neutral pH and relatively low TSS levels. Further downstream, the river receives intermittent inputs from LDP21, the discharge point for the Unit 8 Cooling Tower drain. This discharge is characterised by high pH levels, moderately high conductivities, low concentrations of sulfate, TSS, fluoride and boron and low turbidity. The conductivity and pH levels measured at LDP21 were also generally in excess of the upper default trigger value. Water discharged through LDP5 enters the Coxs River after settling in the Southern Retention Basin and is characterised by moderately high conductivity values, neutral pH and low concentrations of sulfate, fluoride, TSS, boron and selenium. The conductivity levels at this point exceeded the upper DTV on more than 75% of the monitoring events, while boron, which exceeded the 90% protection level for aquatic ecosystems during more than 25% of the events.

It is clear that aquatic organisms living in the section of the Coxs River receiving discharges from the power station are subject to variable inputs of a variety of contaminants. In the late 1980s, the irregular discharge from the fly ash neutralisation plant was considered to be the most significant input from the power station, because the relatively high volume input raised the selenium levels above its statutory limit and increased fluoride levels towards their limit (Jones *et al.* 1989). It was also noted that the power station contributed low levels of copper, nickel, zinc and arsenic to the Coxs River, but the concentrations of these elements were below their respective water quality limits. Conductivity, pH, boron and selenium are the contaminants of primary concern, because their levels frequently exceeded the water quality guidelines. Whether these contaminants actually pose a potential threat to aquatic life depends on the rapidity with which they are diluted by the receiving waters.

The water quality monitoring undertaken at LMP8, the downstream ambient monitoring location, provides an indication of the extent to which inputs into the Coxs River from the licenced discharge points and upstream tributaries are diluted. The water at this location was characterised by moderate conductivity, pH and sulfate levels and low concentrations of fluoride, boron, manganese, iron and selenium. This implies that the inputs from the licenced discharge points have undergone considerable dilution by the time they reach LMP8. The median conductivity, boron and manganese levels at LMP8 (863 µS/cm, 0.28 mg/L and 0.10 mg/L, respectively) were much greater than at LMP7 (668 µS/cm, 0.12 mg/L and 0.06 mg/L, respectively), which indicates that some of the contaminants had not been adequately diluted by the flows along the section of the river passing through the power station property. The conductivity and pH levels are of particular concern, because a large proportion of their measurements at LMP8 exceeded their respective upper DTVs. Boron is also of concern, because 22% of the concentrations measured at this location exceeded the GTV for 95% protection.

The quality of the water at WX13, the monitoring point situated downstream of Lake Wallace provides an indication of any residual effects from the upstream discharge points, the effects of impoundment of the river and daily riparian releases. The water at this location was characterised by high nitrogen levels, moderate conductivity, pH, sulfate and fluoride levels, low turbidity, highly variable manganese concentrations, low concentrations of TSS, iron, ammonium as nitrogen and nitrogen oxides. The lower median conductivity, pH, manganese and iron levels relative to LMP8 indicate that further dilution occurs within Lake Wallace. Despite this, a large proportion of the conductivity levels and concentrations of nutrients, other than nitrate, exceeded their respective upper DTVs and could therefore pose a potential risk to aquatic life. The pH levels are also of concern, because 25% of the measurements exceeded their upper DTV. The reduction in pH and levels of metals is probably related to the fact that sediments within Lake Wallace act as a sink for trace metals and reduce metal loading to the downstream reaches of the Coxs River (Jones *et al* 1989; Birch *et al.* 2001). Previous workers have noted that the riparian releases contain greater concentrations of manganese, iron and aluminium than the surface water and have attributed the release of these elements from deposits within the lake to redox changes brought about by stratification of the lake (Jones *et al.* 1989).

The blowdown from Cooling Towers 7 and 8 is released into the Coxs River via licensed discharge point LDP4 which is situated downstream of Lake Wallace at the confluence of the tortuous watercourse and the river. This discharge was characterised by high conductivities, neutral pH, high sulfate and iron concentrations, moderate TSS, turbidity, fluoride, boron and manganese levels, but low concentrations of selenium. The major concerns with this discharge are conductivity, which was always in excess of the upper DTV, and pH levels and boron concentrations which exceeded the upper DTV and 90% protection level for aquatic ecosystems on 22% of the events, respectively. Iron contamination may also be of concern, but is difficult to evaluate because of the small number of measurements and lack of guideline trigger values for Australian conditions. The processes that occur within the cooling towers lead to concentration of major ions and elevated conductivity levels. Contaminants within the blowdown discharge may be derived from the makeup water source or leaching from the cooling system. Some of the trace metals in the blowdown discharge may be removed by adsorption onto sediment before the water is returned to the river (Harris and Hillman 1991).

The quality of the water at the monitoring points situated immediately downstream of the confluence with LDP4 and the tortuous watercourse, is likely to reflect the combined effects of the discharge of the blowdown, the impoundment of the river and daily riparian releases from Lake Wallace. The water at COX4 was characterised by high conductivity, pH and sulfate levels, moderate TSS measurements and relatively low concentrations of boron, manganese, iron and selenium. The conductivity levels and concentrations of these trace metals were considerably lower than at LDP4, which suggests that the inputs from the discharge point had undergone some dilution by the time they reached COX4. Although the manganese and selenium concentrations did not exceed the trigger values for 99% protection, a large proportion (49%) of the boron measurements were in excess of the trigger value for 95% protection. Despite the dilution, all the conductivity measurements were greater than their respective upper DTV, as were a large proportion of the pH measurements. The conductivity levels were also twice as high as recorded further upstream, suggesting that its potential risk to aquatic life was greater below than above Lake Wallace. Jones *et al.* (1989) noted there was a marked decrease in the concentration of manganese, copper, zinc and nickel between Lake Wallace and Lake Lyell and attributed this to physical and chemical removal mechanisms.

The water further downstream at the final monitoring location upstream of Lake Lyell, COX5, was characterised by moderately high conductivity and pH levels, relatively low nutrient and aluminium concentrations and low turbidity. The conductivity and pH levels were considerably lower than at COX4, indicating that further dilution had occurred, most likely due to flows within the river and inflows from Marrangaroo Creek. Approximately 98% of the conductivity and a large proportion of the pH measurements, however, still exceeded their respective DTVs, as did most of the nutrient levels.

5.1.3 Threats to Aquatic Biota

Although the conductivities recorded in the section of the Coxs River encompassing the monitoring locations were well in excess of the upper DTV (350 µS/cm), this does not necessarily mean that this contaminant poses a threat to aquatic life. The initial review of sensitivity of Australian freshwater biota to salinity undertaken by Hart *et al.* (1991) indicates that adverse effects on freshwater macroinvertebrates are likely to become apparent when salinity rises to around 1,000 mg/L (approximately 1,562 µS/cm). Aquatic macrophytes and riparian plants are slightly more tolerant, being sensitive to salinities within the 1000-2000 mg/L (1562-3134 µS/cm) and above 2,000 mg/L (>3,134 µS/cm). Adult fish, in contrast, are tolerant of salinities up to 10,000 mg/L (15,620 µS/cm). A subsequent review of the effects of increasing salinity on freshwater ecosystems in Australia undertaken by Nielsen *et al.* (2003) indicates the following:

- Majority of algae do not tolerate salinities > 10,000 mg/L (15620 µS/cm);
- Diatoms decrease in abundance and richness as salinity increases;
- Freshwater plants tolerate salinities up to 4,000 mg/L (6,250 µS/cm), but adverse effects on growth and development of roots and leaves become apparent above 1,000 mg/L (1,562 µS/cm);
- Macroinvertebrate fauna of rivers appear to be tolerant and fairly resilient to increasing salinity;
- Structurally simple macroinvertebrates such as soft-bodied hydra, insect larvae and molluscs are more sensitive to increased salinity;

- Salinity tolerance testing of 59 macroinvertebrate taxa indicated tolerance ranged from 5,000 to 50,000 mg/L (7,810-78,100 µS/cm), with baetid mayflies and macrocrustaceans being the least and most tolerant groups, respectively.
- Majority of native and introduced fish appear to be tolerant of salinities in excess of 3000 mg/L (4686 µS/cm).

The median conductivity levels at the monitoring locations situated upstream of Lake Wallace (668 µS/cm at LMP7 to 863 µS/cm at LMP8) and the first downstream monitoring location (788 µS/cm) were well below the salinity levels (1562 µS/cm) identified by Hart *et al* (*1991*) and Nielsen *et al* (2003) as likely to have adverse effects on aquatic biota. The larger median value at COX4 (1612 µS/cm) suggests that the elevation of the conductivity levels associated with the LDP4 discharge may be having a localised impact on aquatic plants and macroinvertebrates, however, this is dissipated by the time the river reaches COX5, where the median conductivity drops to 1153 µS/cm, as a result of inflows from Marrangaroo Creek. It should, however, be noted that the maximum conductivities recorded at LMP8, WX13, COX4 and COX5 exceeded the values considered to be harmful. The considerable variation in the conductivity levels noted at the monitoring locations probably reflects periods of low and high flow (Hart *et al.* 2003).

It should also be noted that recent studies on the effects of saline discharge from mines have indicated that chemical composition of the salts in the discharge is more important than conductivity *per se* (Cardno Ecology Lab 2010).

Although a fairly large proportion (25.6-58.9%) of the pH measurements taken at the monitoring locations exceeded the upper DTV, this does not necessarily mean this parameter is having adverse effects on aquatic biota in the river. The median pH levels at these locations fell within the 6.5-9.0 pH range that water quality guidelines around the world indicate is necessary for the protection of freshwater aquatic organisms, as did all the individual measurements (Alabaster & Lloyd 1982; USEPA 1986b; CCREM 1991; ANZECC 1992). The ANZECC (1992) guidelines also recommended that the maximum variation in pH should not be greater than 0.5 pH units from the natural seasonal maximum or minimum values. The difference in the minimum and maximum pH levels at the monitoring locations, however, varied between 1.3 units at COX4 to 2.3 units at COX5. The time series plots also show that rapid changes in pH in excess of 0.5 units occasionally occur at the monitoring sites. Information on the sensitivity of aquatic organisms to changes in pH of these magnitudes is needed to assess the implications of such variation.

5.2 Aquatic Flora and Fauna

The aquatic biota occurring at CR1 is influenced by the structure of the physical habitat, quality of water and sediment, inflows into Coxs River and Marrangaroo Creek plus impacts associated with the presence of the upstream and downstream impoundments, release of a constant riparian flow of approximately 3 ML/d from Lake Wallace, except when the dam overspills naturally, release of blowdown water from the Cooling Towers at Wallerawang Power Station and stocking with introduced fish species. The organisms that are sampled each survey period consequently reflect the cumulative influence of natural environmental features and various forms of disturbances that have occurred over the previous weeks and/or months.

The results show that CR1 supports a diverse range of macroinvertebrate and periphyton taxa, but the fish fauna is limited to six species, only two of which are native. The low diversity of fish at CR1 is not surprising, because only six species were recorded in the river between Blackmans Flat and a site downstream of Lake Wallace and the cooling tower blowdown discharge point in the late 1980s (Harris and Hillman 1991) and only seven species have been caught in the lower Coxs River (Young et al. 2000). The low diversity is probably a result of species that migrate between freshwater and the sea to breed having become locally extinct upstream of impoundments that prevent their movement (Gehrke *et al.* 2002). It should, however, be noted that a broad-scale survey of the Coxs River between Lakes Wallace and Lyell undertaken prior to the construction of the dams indicated that fish catches were very poor (Llewellyn and Tilzey 1974). It has subsequently been suggested that the poor catches may have been due to poor water quality (Harris and Hillman 1991). Three of the four species of alien fish (rainbow trout, goldfish and mosquito fish and one of the two species of native fish (flathead gudgeon) found at CR1 between 2002 and 2009 are known to have been present at a site downstream of Wallerawang Reservoir and the cooling tower discharge point in the late 1980s (Harris and Hillman 1991). The relative scarcity of native fish from 2002 to 2009 and absence of mosquito fish from the last three surveys may reflect predation by trout. The abundance of mosquito fish noted between 2002 and 2004 probably reflects the rapid maturation of this live-bearing species and its ability to produce several broods a year, characteristics which can cause rapid increases in the size of local populations

(McDowall 1980). The large abundances of trout species noted in the spring 2008 and autumn of 2009 may either be due to the impact of fish stocking and the subsequent upstream migration of fish or more successful reproduction and recruitment during wetter periods. Llewellyn and Tilzey (1974) hypothesised that trout may accumulate in the Coxs River below Lake Wallace, because further upstream migration is prevented by the dam wall. Lake Wallace is known to have been stocked on three occasions with Australian bass (*Macquaria novemaculeata*), a native fish species, and on numerous occasions with rainbow trout (*Oncorhynchus mykiss*) and brown trout (*Salmo trutta*), both of which are introduced species (NSW Fisheries 2003).

In addition to restricting the migration of fish, Lake Wallace reduces the downstream transport of sediment, nutrients, periphyton and macroinvertebrates. The daily riparian release from the dam also results in the periphyton and macroinvertebrate assemblages in that reach of the river being subject to lengthy periods of stable low flows, except when rainfall causes natural inflows from the surrounding catchment and the dam to overspill. The blowdown from the cooling towers that is released downstream of the dam is the only other source of variable flow.

There have been marked differences in the annual rainfall and differences in the number and duration of overspills from Lake Wallace over the period of investigation (Parsons Brinckerhoff 2006; Delta Electricity, 2007, 2008 and 2009). Periodic flood and overspill events would increase the rate of sloughing and erosion of periphyton, dislodge aquatic macroinvertebrates from their habitats and result in downstream drift and mortalities in both groups of organisms. The susceptibility of periphyton to damage or dislodgement during floods varies widely, with prostrate diatoms and taxa with specialised holdfast structures being the most resilient to such disturbances (Biggs 1995). Experimental studies indicate that an increase in mean water column velocity from <0.3 m/s to > 1m/s can cause 50% sloughing of mature filamentous diatom assemblages and that at velocities > 1.5 m/s > 70% of the assemblage is removed simply by water shear (Biggs and Thomsen 1994). Macroinvertebrates also differ in their susceptibility to disturbance by floods, with some species responding to rising discharges by seeking refuge (Brooks and Boulton 1991). The differences in the diversity and structure of the periphyton and macroinvertebrate assemblages detected among surveys may reflect differences in the amount of time for their accrual after each flood. This would result in surveys of assemblages at different successional stages. The rate of development of periphyton assemblages after flood events depends on the availability of nutrients and light, favourable temperatures, pollution and grazing pressure. Floods may also affect the availability of nutrients with frequent flood events leading to depletion of the nutrient pool and reducing the rate of recovery of the periphyton.

The fact that the periphyton and macroinvertebrate assemblages surveyed in 2002 and 2003, the two driest years of the study period, were distinct from those surveyed subsequently suggests that drought may also influence the structure of their assemblages. The low flows that prevailed during this period are likely to have enabled the periphyton assemblages to attain a more mature stage of succession than under greater flows. In New Zealand, the structure of periphyton assemblages changes appreciably during periods of low flows, with nutrient-poor and nutrient-rich catchments being dominated by diatoms and filamentous green algae, respectively (Biggs and Hickey 1994; Biggs *et al.* 1998).

As CR1 is situated approximately 10 km downstream of LDP4, it is unlikely that the blowdown water from the cooling towers would have any detectable effects on aquatic biota, given the dilution effects of the daily 3 ML riparian release from Lake Wallace and natural inflows from the surrounding catchment during periods of rainfall. There is, of course, a possibility of the blowdown water having adverse effects on biota that occur close to its discharge point and of such effects being more widespread during periods of low flow resulting from drought conditions. When spills from Lake Wallace have occurred the combined contribution of the riparian release and blowdown water to the overall flows recorded has varied from <10% in 2005/20-06 and 2006/2007 to > 20% in 2007/2008 and 2008/2009.

5.3 Conclusions

The water quality and aquatic ecology of the Upper Coxs River is influenced by a number of factors, including:

- Presence of two impoundments;
- Discharges associated with operation of WPS;
- Inputs from the sewage treatment plant upstream of WPS;
- Inputs from previous and current coal mining; and
- Stocking with highly predatory introduced fish species.

It is consequently difficult to attribute patterns in water quality and biota in this reach of the river solely to the effects of discharges from WPS.

6 Recommendations

1) The development of site-specific guidelines is considered inappropriate given the variability in quality and quantity of the discharges and their dilution due to the natural variation in flow within Coxs River.
2) Further site-specific investigations are needed to determine whether the contaminants that exceeded the water quality guidelines are actually having an adverse effect on the biota. Consideration needs to be given to the possibility that factors such as pH and hardness may modify impacts on freshwater ecosystems, the value of comparison with reference conditions and the need for direct assessments of toxicological effects (ANZECC/ARMCANZ 2000). As a precautionary approach, management and/or remedial actions that reduce the level of contamination and thereby the risk of adverse biological effects occurring could be implemented, instead of undertaking further investigations.
3) The presence of chemicals within the sediments of Lake Wallace derived from WPS and the sewage treatment plant and the contribution of chemical transformations, such as redox reactions, within the lake to contaminant loads downstream in the river clearly require further investigation.
4) The water quality monitoring programmes undertaken by Delta should continue, but the parameters measured and concentration limits specified should be reviewed.
5) The levels/concentrations of contaminants from other sources that feed into the Coxs River in the vicinity of WPS need to be assessed, so that the contribution from the discharge points can be seen in context. The major additional inputs requiring assessment are those associated with the sewage treatment plant on Pipers Flat Creek, pastoral/agricultural activities in the surrounding catchment and acid deposition from the power stations air emission points.
6) The on-going implementation of the power stations drought management strategy should continue to reduce the salt and pollutant load with an ultimate goal of converting the station to a zero process water discharge site.
7) The aquatic biota occurring in the vicinity of the water quality monitoring locations and licenced discharge points that feed directly into Coxs River need to be surveyed and compared with that at appropriate reference locations, so that ecological impacts of the discharges from Wallerawang Power Station can be assessed more fully.
8) The composition and abundance of aquatic biota occurring at multiple locations on the Coxs River between Lake Wallace and Lake Lyell needs to be surveyed to gain a better understanding of the impact of variation in natural inflows, the presence of the impoundments, the consistent daily riparian release from Lake Wallace and the blowdown water from the cooling towers. Concurrent surveys need to be conducted on appropriate reference streams to gain an appreciation of the natural spatial variation in aquatic biota along reaches of similar length to that between Lakes Wallace and Lake Lyell.
9) Consideration should also be given to the use of biological assessment methods that are appropriate for early detection of short or longer-term changes in the biota associated with contaminants of particular concern. These methods include direct toxicity assessment to determine the sensitivity of particular indicator organisms under laboratory conditions and field-based bioaccumulation studies and translocation experiments to measure toxicity *in situ.*

7 Acknowledgements

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9 Tables

Table 1: Contaminants that must be monitored at the various licenced discharge points and the frequency with which monitoring is required, as specified in Environmental Protection Licence No. 766.

Table 2: Concentration limits for contaminants at the various licenced discharge points at Wallerawang Power Station as specified in Environmental Protection Licence No. 766.

Table 3: Frequency with which key contaminants exceeded the concentration limits specified in Environment Protection Licence No. 766.

Table 4: Summary statistics for (a) electrical conductivity and (b) pH data from the monitoring and discharge locations on and adjacent to the Coxs River in the vicinity of Wallerawang Power Station.

Table 5: Summary statistics for (a) boron, (b) manganese, (c) selenium, (d) iron and (e) aluminium data from the monitoring and discharge locations on and adjacent to the Coxs River in the vicinity of Wallerawang Power Station.

Table 6: Summary statistics for (a) nitrate, (b) ammonium, (c) oxides of nitrogen, (d) total nitrogen, (e) filtered ortho-phosphate and (f) total phosphorus data from the three locations assessed in the regional monitoring program.

Table 7: Summary statistics for turbidity data from six locations in the vicinity of the Wallerawang Power Station.

Table 8: Summary statistics for (a) sulfate and (b) fluoride data from the monitoring and discharge locations on and adjacent to the Coxs River in the vicinity of Wallerawang Power Station.

Table 9: AUSRIVAS indices for macroinvertebrate assemblages associated with (a) edge and (b) riffle habitats at CR1 from 2002 -2009.

Table 10: Numerically dominant macroinvertebrate taxa recorded in Surber samples collected from riffle habitat at CR1 during each of the (a) spring and (b) autumn surveys and their percentage contribution to the total number of periphyton cells.

Table 11: Results of PERMANOVA tests comparing the structure of macroinvertebrate assemblages associated with riffle habitat.

Table 12: Results of PERMANOVA tests comparing the diversity of macroinvertebrate assemblages associated with riffle habitat.

Table 13: Numerically dominant periphyton taxa recorded on stones collected from riffle habitat at CR1 during each of the (a) spring and (b) autumn surveys and their percentage contribution to the total number of macroinvertebrates.

Table 14: Results of PERMANOVA tests comparing the structure of periphyton assemblages associated with stones in the riffle habitat at CR1 in (a) autumn and (b) spring of 2002-2009.

Table 15: Results of PERMANOVA tests comparing the diversity of periphyton assemblages associated with riffle habitat.

Table 16: Numbers of each species of fish and mobile invertebrate caught in the electrofishing undertaken at each site at CR1 since autumn 2002.

Table 1: Contaminants that must be monitored at the various licenced discharge points and the frequency with which monitoring is required, as specified in Environmental Protection Licence No. 766.

Pollutant	Location								
	LDP1	LDP3	LDP4	LDP5	LMP7	LMP8	LDP18	LDP20	LDP21
Conductivity	W	W*	W	M*	W	W	M*	W*	W*
Sulfate	W	W*	W		W	W			W*
Total suspended solids	W	W*	W	M*			M*	W*	W*
Turbidity			W					W*	
pH	W	W*	W	M*	M	M	M*	W*	W*
Fluoride		W*	M		M	M		M*	
Aluminium			M					M*	
Arsenic			M					M*	
Boron		M*	M		M	M		M*	
Copper			M					M*	
Nickel			M					M*	
Selenium	M	M*	M		M	M			
Zinc			M					M*	
Filterable iron		M*			M	M		M*	
Filterable Manganese		M*			M	M		M*	
Oil and Grease								W*	
Temperature								W*	

W, weekly; M, monthly and * during discharge

Table 2: Concentration limits for contaminants at the various licenced discharge points at Wallerawang Power Station as specified in Environmental Protection Licence No. 766.

Pollutant	Location	Percentile Concentration llimits		
		50th	90th	100th
pH	LDP1, LDP3, LDP4, LDP5, LDP18, LDP20, LDP21			6.5-8.5 units
Sulfate	LDP1 and LDP21	1000 mg/L	1200 mg/L	1600 mg/L
	LDP3			1200 mg/L
	LDP4			1600 mg/L
TSS	LDP3 and LDP18			30 mg/L
Turbidity	LDP4			25 ntu
Oil and grease	LDP5, LDP18 and LDP20			10 mg/L

Table 3: Frequency with which key contaminants exceeded the concentration limits specified in Environment Protection Licence No. 766.

Pollutant	Percentile Limit	Location						
		LDP1	LDP3	LDP4	LDP5	LDP18	LDP20	LDP21
pH	100% - < 6.5	0.00%	2.10%			0.00%	0.00%	0.00%
pH	100% - >8.5	5.00%	6.30%	1.50%	5.80%	0.00%	0.00%	0.00%
Total suspended solids	100% - 30 mg/L		2.10%			7.20%		
Sulfate	100% - 1600 mg/L	0.00%		0.00%				0.00%
	100% - 1200 mg/L		0.00%					
	90% - 1200 mg/L	0.00%		0.15%				0.00%
	50% - 1000 mg/L	1.70%		5.37%				0.00%
Oil and Grease	100% - 10 mg/L							

Table 4: Summary statistics for (a) electrical conductivity and (b) pH data from the monitoring and discharge locations on and adjacent to the Coxs River in the vicinity of Wallerawang Power Station (*n* = sample size, CI = 95% confidence interval, < LGTV = the number of data points below the lower guideline trigger value, > UGTV = number of data points above the upper guideline trigger value.

(a) Electrical conductivity

Location	n	Median	Mean	CI	Minimum	Maximum	< LGTV	> UGTV	% > UGTV
LDP20	72	1100	1088	27	129	1148	0	71	98.6
LMP7	656	667.5	682	14	177	1261	0	640	97.6
LDP1	178	1754	1635	90	205	2715	0	175	98.3
LDP3	43	2139	2088	56	1398	2301	0	43	100.0
LDP18	75	1318	1247	53	612	1536	0	75	100.0
LDP21	11	1053	1034	65	782	1194	0	11	100.0
LDP5	58	432	499	61	174	1121	0	38	65.5
LMP8	663	863	916	26	209	2268	0	650	98.0
WX13	653	788	873	22	173	2010	0	641	98.2
LDP4	522	2030	2005	33	586	2961	0	522	100.0
COX4	231	1612	1579	51	715	2315	0	231	100.0
COX5	86	1152.5	1128	81	268	1983	0	85	98.8

(b) pH

Location	n	Median	Mean	CI	Minimum	Maximum	< LGTV	> UGTV	% > UGTV
LDP20	72	8.38	8.37	0.03	7.56	8.53	0	70	97.2
LMP7	656	8.05	7.97	0.03	6.59	8.76	0	324	49.4
LDP1	261	8.00	8.00	0.04	6.9	9.1	0	117	44.8
LDP3	48	7.28	7.29	0.12	5.9	8.12	1	3	6.3
LDP18	482	7.52	7.52	0.02	6.55	8.14	0	1	0.2
LDP21	11	8.23	8.19	0.25	6.96	8.44	0	10	90.9
LDP5	138	7.58	7.57	0.04	6.84	8.41	0	8	5.8
LMP8	664	8.00	7.97	0.03	6.87	8.78	0	291	43.8
WX13	653	7.85	7.89	0.02	6.8	8.97	0	167	25.6
LDP4	673	7.71	7.77	0.02	7.04	8.7	0	149	22.1
COX4	231	8.10	8.10	0.03	7.52	8.85	0	136	58.9
COX5	86	8.00	7.94	0.07	6.7	9	0	35	40.7

Table 5: Summary statistics for (a) boron, (b) iron, (c) managanese and (d) selenium data from the monitoring and discharge locations on and adjacent to the Coxs River in the vicinity of Wallerawang Power Station(n = sample size, CI = 95% confidence interval, >WQG = the number of data points greater than the recreational water quality guideline.

(a) Boron (mg/L)

Location	n	Median	Mean	CI	Minimum	Maximum	% >GTV 99%	% >GTV 95%	% >GTV 90%	% >GTV 80%
LDP20	1		0.06							
LMP7	141	0.12	0.16	0.02	0.05	1.20	54.6	4.3	1.4	0
LDP1	11	0.32	0.30	0.04	0.19	0.38	100	9.1	0	0
LDP3	44	2.75	3.24	0.64	0.01	8.8	93.2	93.2	93.2	88.6
LDP21	6		0.10	0.07	0.05	0.28	16.7	0	0	0
LDP5	7	0.19	0.41	0.33	0.1	1.3	100.0	28.6	28.6	0.0
LMP8	141	0.28	0.36	0.05	0.01	1.6	63.8	22.7	9.2	1.4
WX13	655									
LDP4	71	0.46	0.60	0.09	0.15	2.1	100.0	69.0	22.5	7.0
COX4	37	0.37	0.46	0.16	0.17	3.2	100.0	48.6	5.4	2.7

(b) Manganese (mg/L)

Location	n	Median	Mean	CI	Minimum	Maximum	% >GTV 99%	% >GTV 95%	% >GTV 90%	% >GTV 80%
LDP20	1		0.03							
LMP7	141	0.06	0.05	0.01	0.01	0.41	0.0	0.0	0.0	0.0
LDP3	41	1.00	0.83	0.16	0.01	1.70	22.0	0.0	0.0	0.0
LMP8	141	0.10	0.10	0.02	0.005	0.9	0.0	0.0	0.0	0.0
WX13	86	0.06	0.22	0.13	0.001	5.1	3.5	1.2	1.2	1.2
LDP4	8	0.26	0.33	0.13	0.18	0.7	0.0	0.0	0	0
COX4	37	0.01	0.04	0.02	0.005	0.3	0.0	0.0	0.0	0

Table 5 continued.

(c) Selenium (mg/L)

Location	n	Median	Mean	CI	Minimum	Maximum	% >GTV 99%	% >GTV 95%	% >GTV 90%	% >GTV 80%
LMP7	142	0.003	0.002	0.000	0.000	0.003	0.0	0.0	0.0	0.0
LDP1	73	0.003	0.00	0.002	0.001	0.059	9.6	4.1	4.1	1.4
LDP3	44	0.020	0.090	0.060	0.003	0.950	84.1	68.2	52.3	36.4
LDP5	7	0.003	0.003	0.001	0.003	0.006	14.3	0.0	0.0	0.0
LMP8	141	0.003	0.004	0.000	0.001	0.018	10.6	1.4	0.7	0.0
WX13										
LDP4	84	0.003	0.004	0.001	0.001	0.025	17.9	6.0	1.2	0.0
COX4	37	0.003	0.003	0.001	0.001	0.003	0.0	0.0	0.0	0.0

(d) Iron (mg/L)

Location	n	Median	Mean	CI	Minimum	Maximum	% > CGL
LDP20	1		0.46				
LMP7	141	0.060	0.09	0.02	0.005	0.72	2.8
LDP3	41	0.010	0.01	0.01	0.01	0.16	0.0
LMP8	141	0.030	0.06	0.02	0.005	0.59	4.3
WX13	86	0.020	0.03	0.01	0.01	0.5	2.3
LDP4	8	0.290	0.33	0.03	0.22	0.50	37.5
COX4	37	0.010	0.03	0.02	0.005	0.35	2.7

(e) Aluminium (mg/L)

Location	n	Median	Mean	CI	Minimum	Maximum	% >GTV 99%	% >GTV 95%	% >GTV 90%	% >GTV 80%
LMP7	86	0.115	0.302	0.108	0.020	3.000	97.67	81.40	68.60	39.53
WX13	86	0.020	0.038	0.011	0.005	0.410	48.84	12.79	6.98	2.33
COX5										

Table 6: Summary statistics for (a) nitrate, (b) ammonium, (c) oxides of nitrogen, (d) total nitrogen, (e) filtered ortho-phosphate and (f) total phosphorus data from the three locations in the regional monitoring program (*n* = sample size, CI = 95% confidence interval, >WQG = the number of data points greater than the recreational water quality guideline.

(a) Nitrate (mg/L)

Location	n	Median	Mean	CI	Minimum	Maximum	% >GTV 99%	% >GTV 95%	% >GTV 90%	% >GTV 80%
LMP7	77	0.160	0.303	0.078	0.003	1.300	75.32	12.99	0.00	0.00
WX13	77	0.040	0.066	0.018	0.005	0.500	85.71	0.00	0.00	0.00
COX5	77	0.010	0.052	0.020	0.003	0.430	44.16	0.00	0.00	0.00

(b) Ammonium (mg/L) GTV = 0.013 mg/L

Location	n	Median	Mean	CI	Minimum	Maximum	% >GTV
LMP7	83	0.030	0.048	0.015	0.005	0.430	78.31
WX13	83	0.040	0.063	0.019	0.005	0.560	80.72
COX5	83	0.020	0.022	0.004	0.005	0.100	62.65

(c) Oxides of Nitrogen (mg/L) GTV = 0.015 mg/L

Location	n	Median	Mean	CI	Minimum	Maximum	% >GTV
LMP7	86	0.165	0.296	0.072	0.005	1.330	78.31
WX13	86	0.050	0.075	0.021	0.005	0.660	83.72
COX5	86	0.010	0.058	0.019	0.005	0.430	48.84

(d) Total Nitrogen (mg/L) GTV = 0.25 mg/L

Location	n	Median	Mean	CI	Minimum	Maximum	% >GTV
LMP7	86	0.700	0.783	0.101	0.030	2.100	89.53
WX13	86	0.700	0.757	0.057	0.050	1.600	95.35
COX5	86	0.600	0.651	0.049	0.200	1.400	97.67

(e) Filtered Ortho-phosphate (mg/L) GTV = 0.015 mg/L

Location	n	Median	Mean	CI	Minimum	Maximum	% >GTV
LMP7	86	0.615	0.883	0.190	0.080	6.400	100.00
WX13	86	0.270	0.378	0.068	0.020	1.400	100.00
COX5	86	0.120	0.160	0.030	0.005	0.650	97.67

(f) Total Phosphorus (mg/L) GTV = 0.020 mg/L

Location	n	Median	Mean	CI	Minimum	Maximum	% >GTV
LMP7	86	0.255	0.357	0.070	0.050	2.200	100.00
WX13	86	0.140	0.177	0.027	0.030	0.800	100.00
COX5	86	0.070	0.083	0.011	0.010	0.230	94.19

(f) Turbidity GTV ranges from 2-25 ntu

Location	n	Median	Mean	CI	Minimum	Maximum	% < LGTV	% > UGTV
LDP20	74	14	30.24	10.42	3.60	290.00	0.00	28.38
LMP7	86	3.7	8.60	3.31	1.10	110.00	3.49	4.65
LDP21	11	6.1	8.21	4.24	0.80	24.00	9.09	0.00
WX13	86	2.5	3.52	1.56	0.80	70.00	25.58	1.16
LDP4	149	8.9	10.05	0.76	3.30	27.25	0.00	1.34
COX5	86	1.5	4.46	3.63	0.65	160.00	59.30	1.16

Table 7: Summary statistics for turbidity data from six locations in the vicinity of the Wallerawang Power Station (n = sample size, CI = 95% confidence interval, >WQG = the number of data points greater than the recreational water quality guideline.

GTV ranges from 2-25 ntu

Location	n	Median	Mean	CI	Minimum	Maximum	% < LGTV	% > UGTV
LDP20	74	14	30.24	10.42	3.60	290.00	0.00	28.38
LMP7	86	3.7	8.60	3.31	1.10	110.00	3.49	4.65
LDP21	11	6.1	8.21	4.24	0.80	24.00	9.09	0.00
WX13	86	2.5	3.52	1.56	0.80	70.00	25.58	1.16
LDP4	149	8.9	10.05	0.76	3.30	27.25	0.00	1.34
COX5	86	1.5	4.46	3.63	0.65	160.00	59.30	1.16

Table 8: Summary statistics for (a) sulfate and (b) fluoride data from the monitoring and discharge locations on and adjacent to the Coxs River in the vicinity of Wallerawang Power Station (*n* = sample size, CI = 95% confidence interval.

(a) Sulfate (mg/L)

Location	n	Median	Mean	CI	Minimum	Maximum
LDP20	69	42	42.00	1.84	0.80	58.40
LMP7	659	150	157.16	3.36	34	374
LDP1	652	500	522.88	19.34	10	1200
LDP3	48	1016	996.86	41.43	354	1172
LDP18	ND					
LDP21	11	58	80.35	32.03	43.6	231.2
LDP5	78	130	208.03	43.03	11	770
LMP8	624	230	264.00	10.61	54	876
WX13	655	223	272.61	10.78	40	850
LDP4	671	760	739.30	16.09	62	1250
COX4	230	561	558.23	23.32	195	940

(b) Fluoride

Location	n	Median	Mean	CI	Minimum	Maximum
LDP20	10		1.03	0.09	0.67	1.19
LMP7	518	0.38	0.47	0.02	0.07	1.51
LDP1	345	1.71	1.75	0.08	0.2	5.08
LDP3	53	3.30	3.78	0.58	1.5	11.2
LDP18	ND					
LDP21	10	1.04	1.03	0.09	0.67	1.19
LDP5	78	0.50	0.67	0.10	0.1	2.2
LMP8	524	0.96	1.06	0.05	0.09	4
WX13	22	2.42	2.16	0.29	0.864	3.07
LDP4	137	2.12	2.04	0.07	1.03	3.45
COX4	ND					
COX5	ND					

Table 9: AUSRIVAS indices for macroinvertebrate assemblages associated with edge and riffle habitats at CR1 from (a) spring 2002-2008 and (b) autumn 2002 to 2009.

(a) Spring

Edge Habitat

Index	Site																				
	2002			2003			2004			2005			2006			2007			2008		
Site	A	B	C	A	B	C	A	B	C	A	B	C	A	B	C	A	B	C	A	B	C
NTC50	10	10	9	14	12	13	11	14	13	11	13	14	8	11	11	9	9	10	10	10	10
OE50 taxa	0.8	0.8	0.72	1.11	0.96	1.03	0.88	1.11	1.03	0.93	1.04	1.17	0.77	1.06	1.06	0.90	0.90	0.84	0.8	0.8	0.79
BAND Grade	B	B	B	A	A	A	A	A	A	A	A	X	B	A	A	A	A	A	B	B	B
O50SIGNAL	4.3	4.3	3.89	4.29	4.75	4.54	3.91	4.5	4.38	4.45	4.08	4.29	4.13	4	3.91	4.00	4.22	4.10	5.1	4.7	4.3
OE50SIGNAL	0.88	0.88	0.79	0.87	0.97	0.92	0.8	0.92	0.89	0.93	0.83	0.89	1.1	1.06	1.03	0.89	0.94	0.86	1.04	0.96	0.88
O0SIGNAL	3.95	3.84	3.67	3.5	4.24	3.88	3.52	4.14	4	4	3.77	3.74	3.94	4.09	4.23	4.05	4.32	3.89	4.2	4.21	3.63

Riffle Habitat

Index	Site																				
	2002			2003			2004			2005			2006			2007			2008		
Site	A	B	C	A	B	C	A	B	C	A	B	C	A	B	C	A	B	C	A	B	C
NTC50	10	8	5	11	6	8	6	12	12	6	8	8	9	10	Not sampled	8	8	10	6	10	10
OE50 taxa	0.77	0.62	0.39	0.85	0.46	0.62	0.46	0.93	0.93	0.46	0.62	0.62	0.7	0.77		0.62	0.62	0.77	0.46	0.77	0.77
BAND Grade	B	B	C	A	B	B	B	A	A	B	B	B	B	B		B	B	B	B	B	B
O50SIGNAL	5.4	5.38	5	5.82	5.67	5.38	5.33	5.83	6.08	5.33	5.25	5.13	5.56	5.80		5.00	5.75	5.50	6	5.7	5.5
OE50SIGNAL	0.9	0.89	0.83	0.97	0.94	0.89	0.89	0.97	1.01	0.89	0.87	0.85	0.92	0.96		0.83	0.96	0.91	1.00	0.95	0.91
O0SIGNAL	4.5	4.55	4.06	4.68	4.41	4.45	4.31	4.7	5.27	4	3.77	3.74	4.28	4.65		4.13	4.94	4.55	4.67	4.78	5

(b) Autumn

Edge Habitat

Index	Site																				
	2002			2003			2004			2006			2007			2008			2009		
Site	A	B	C	A	B	C	A	B	C	A	B	C	A	B	C	A	B	C	A	B	C
NTC50	10	11	9	8	10	8	11	11	9	8	10	10	11	12	11	11	6	11	9	9	10
OE50 taxa	0.99	1.08	0.89	0.79	0.99	0.79	1.08	1.09	0.89	0.7	0.88	0.9	1.14	1.05	1.1	1.08	0.61	0.98	0.9	0.91	0.95
BAND Grade	A	A	A	B	A	B	A	A	A	B	A	A	A	A	A	A	B	A	A	A	A
O50SIGNAL	4.1	4.27	4.44	4.5	4.6	3.63	3.82	3.82	4.11	3.88	4.1	4.1	4.36	3.92	3.82	3.82	3.67	4.27	4.3	4.1	4.2
OE50SIGNAL	0.92	1.05	1.1	1.01	1.14	0.89	0.86	0.86	0.93	1.0	1.1	1.1	1.05	1.05	1.01	0.86	0.95	1.13	1.1	1.1	1.1
O0SIGNAL	4	4.19	3.79	3.95	4.32	3.65	3.92	3.75	4.06	3.97	4.17	4.05	4.15	4.09	3.86	4.06	3.84	4.23	4.3	4.85	4.48

Riffle Habitat

Index	Site																				
	2002			2003			2004			2006			2007			2008			2009		
Site	A	B	C	A	B	C	A	B	C	A	B	C	A	B	C	A	B	C	A	B	C
NTC50	9	9	9	6	12	8	9	10	10	9	11	8	10	9	10	10	8	8	10	8	10
OE50 taxa	0.8	0.8	0.8	0.53	1.07	0.71	0.8	0.89	0.89	0.8	0.98	0.71	0.89	0.8	0.89	0.89	0.71	0.71	0.89	0.71	0.89
BAND Grade	B	B	B	C	A	B	B	A	A	B	A	B	A	B	A	A	B	B	A	B	A
O50SIGNAL	5.22	4.89	4.67	5	5.25	5.25	4.89	5.2	5.1	5	5	4.63	4.50	4.44	5.30	5.40	5.25	4.50	5.4	5	4.8
OE50SIGNAL	0.97	0.91	0.87	0.93	0.98	0.98	0.91	0.97	0.95	0.93	0.93	0.86	0.84	0.83	0.99	1.00	0.98	0.84	1	0.93	0.89
O0SIGNAL	4.63	4.33	3.93	4.67	5.05	4.82	4.73	4.82	4.8	4.55	4.68	4.4	4.47	4.47	4.65	5.06	5.00	4.56	5.07	4.57	4.31

Table 10: Numerically dominant macroinvertebrate taxa recorded in Surber samples collected from riffle habitat at CR1 during each of the (a) spring and (b) autumn surveys and their percentage contribution to the total number of animals collected. Note that only taxa accounting for >1% of the total animals are listed (*n* = 6 samples per survey).

(a) Autumn surveys

March 2002		March 2003		March 2004		March 2006		April 2007		April 2008		March 2009	
Taxon	**%**	**Taxon**	**%**	**Taxon**	**%**	**Taxon**	**%**	**Taxon**	**%**	**Taxon**	**%**	**Taxon**	**%**
Chironomidae	38.46	Chironomidae	35.07	Chironomidae	34.79	Hydropsychidae	61.52	Chironomidae	59.35	Chironomidae	42.29	Hydropsychidae	41.45
Caenidae	17.24	Caenidae	29.57	Caenidae	16.24	Chironomidae	15.29	Hydropsychidae	7.35	Hydropsychidae	17.11	Chironomidae	37.35
Hydropsychidae	15.10	Oligochaeta	7.54	Gomphidae	15.21	Baetidae	5.86	Oligochaeta	7.22	Baetidae	16.73	Baetidae	4.12
Gomphidae	7.41	Gomphidae	5.80	Hydropsychidae	13.66	Caenidae	3.74	Caenidae	6.75	Oligochaeta	6.49	Simuliidae	3.26
Hydroptilidae	3.99	Hydropsychidae	3.19	Oligochaeta	3.61	Oligochaeta	2.03	Baetidae	4.44	Ecnomidae	3.29	Ecnomidae	2.55
Nematoda	3.28	Hydroptilidae	2.75	Baetidae	3.35	Corbiculidae	1.62	Corbiculidae	3.62	Caenidae	2.96	Oligochaeta	1.95
Pyralidae	2.42	Acarina	2.61	Pyralidae	3.09	Hydroptilidae	1.53	Ecnomidae	2.70	Corbiculidae	2.54	Caenidae	1.74
Baetidae	1.85	Pyralidae	1.81	Ecnomidae	2.32	Gomphidae	1.44	Corixidae	1.66	Tipulidae	1.17	Hydroptilidae	1.46
Physidae	1.71	Corixidae	1.38	Hydroptilidae	2.06	Simuliidae	1.44	Simuliidae	1.28	Gomphidae	1.05	Corbiculidae	1.29
Oligochaeta	1.28	Nematoda	1.23	Corbiculidae	1.03	Tipulidae	1.22					Corydalidae	1.16
		Nemertea	1.23			Pyralidae	1.08						
		Elmidae	1.16										
		Diphlebiidae	1.01										

(b) Spring surveys

November 2002		November 2003		November 2005		October 2006		November 2007		November 2008		October 2009	
Taxon	**%**	**Taxon**	**%**	**Taxon**	**%**	**Taxon**	**%**	**Taxon**	**%**	**Taxon**	**%**	**Taxon**	**%**
Chironomidae	66.47	Chironomidae	44.13	Chironomidae	62.26	Chironomidae	43.94	Chironomidae	63.52	Chironomidae	33.49	Chironomidae	80.64
Caenidae	10.07	Caenidae	17.08	Hydropsychidae	13.14	Hydropsychidae	12.42	Hydropsychidae	6.68	Oligochaeta	18.64	Oligochaeta	6.12
Hydroptilidae	8.43	Gomphidae	7.92	Tipulidae	5.31	Gripopterygiidae	11.77	Oligochaeta	4.24	Gripopterygiidae	14.48	Gripopterygiidae	5.37
Acarina	2.93	Ostracoda	7.65	Oligochaeta	4.61	Corbiculidae	8.85	Tipulidae	4.00	Hydropsychidae	11.44	Hydropsychidae	2.16
Oligochaeta	2.46	Gripopterygiidae	6.15	Hydroptilidae	3.35	Tipulidae	6.48	Baetidae	3.71	Tipulidae	5.70	Corbiculidae	1.47
Gomphidae	2.22	Oligochaeta	5.87	Baetidae	2.45	Caenidae	5.18	Caenidae	3.58	Corbiculidae	4.45		
Physidae	1.29	Tipulidae	1.78	Simuliidae	1.54	Oligochaeta	1.84	Corbiculidae	3.31	Caenidae	1.96		
Copepoda	1.09	Lymnaeidae	1.50	Caenidae	1.47	Elmidae	1.51	Simuliidae	1.63	Nematoda	1.50		
		Hydroptilidae	1.09	Gripopterygiidae	1.40	Gomphidae	1.51	Ostracoda	1.56	Ecnomidae	1.37		
		Nematoda	1.09			Corydalidae	1.30	Hydroptilidae	1.29	Hydroptilidae	1.12		
		Physidae	1.09					Physidae	1.10	Baetidae	1.00		

Table 11: Results of PERMANOVA tests comparing the structure of macroinvertebrate assemblages associated with riffle habitat at CR1 in (a) autumn and (b) spring of 2002-2009. Bold indicates significant at $P < 0.05$

(a) Autumn

Main test (based on 4967 unique permutations)

Source of variation	df	SS	MS	Pseudo-F	P(MC)
Year	6	33583.00	5597.10	7.16	**<0.001**
Residual	33	25805.00	781.96		
Total	39	59387.00			

462 permutations)

Years	t	P(MC)
02, 03	2.159	**0.011**
02, 04	2.675	**0.001**
02, 06*	2.539	**0.004**
02, 07	2.910	**0.001**
02, 08	3.193	**0.001**
02, 09	2.614	**0.001**
03, 04	3.549	**<0.001**
03, 06*	2.550	**0.003**
03, 07	4.382	**<0.001**
03, 08	4.122	**<0.001**
03, 09	3.594	**<0.001**
04, 06*	2.037	**0.013**
04, 07	1.979	**0.009**
04, 08	2.302	**0.001**
04, 09	2.264	**0.005**
06, 07*	2.528	**0.004**
06, 08*	2.050	**0.013**
06, 09*	1.967	**0.019**
07, 08	2.370	**0.002**
07, 09	2.081	**0.007**
08, 09	1.771	**0.026**

(b) Spring

Main test (based on 4968 unique permutations)

Source of variation	df	SS	MS	Pseudo-F	P(MC)
Year	6	30154.00	5025.70	5.45	**<0.001**
Residual	35	32253.00	921.51		
Total	41	62407.00			

Pairwise tests (based on 462 unique permutations)

Years	t	P(MC)
02, 03	1.661	0.053
02, 04	1.277	0.168
02, 06	2.149	**0.007**
02, 07	2.281	**0.009**
02, 08	2.813	**0.001**
02, 09	2.108	**0.011**
03, 04	2.288	**0.005**
03, 06	3.065	**0.001**
03, 07	2.395	**0.003**
03, 08	3.283	**<0.001**
03, 09	2.740	**<0.001**
04, 06	2.899	**0.001**
04, 07	2.778	**0.001**
04, 08	3.764	**<0.001**
04, 09	2.516	**0.001**
06, 07	2.003	**0.012**
06, 08	2.065	**0.006**
06, 09	1.500	0.064
07, 08	1.408	0.113
07, 09	1.660	**0.042**
08, 09	1.903	**0.019**

Table 12: Results of PERMANOVA tests comparing the diversity of macroinvertebrate assemblages associated with the riffle habitat at CR1 in (a) autumn and (b) spring of 2002-2009. Bold indicates significant at *P* < 0.05

(a) Spring

Main test (based on 1467 unique permutations)

Source of variation	df	SS	MS	Pseudo-F	P(MC)
Year	6	441.63	73.60	7.15	**<0.001**
Residual	33	339.75	10.30		
Total	39	781.38			

Pairwise tests (based on 462 unique permutations)

Years	t	P(MC)	# perms
02, 03	1.182	0.262	19
02, 04	0.901	0.378	16
02, 06	2.429	**0.043**	30
02, 07	3.528	**0.004**	21
02, 08	1.103	0.291	19
02, 09	0.455	0.660	19
03, 04	0.492	0.631	14
03, 06	1.377	0.204	23
03, 07	5.404	**<0.001**	26
03, 08	2.334	**0.038**	22
03, 09	0.726	0.496	16
04, 06	2.492	**0.033**	20
04, 07	6.281	**<0.001**	24
04, 08	2.215	**0.051**	19
04, 09	0.371	0.719	14
06, 07	9.121	**<0.001**	43
06, 08	3.551	**0.008**	38
06, 09	2.025	0.072	24
07, 08	2.098	0.064	16
07, 09	4.218	**0.003**	25
08, 09	1.575	0.150	20

(b) Autumn

Main test (based on 606 unique permutations)

Source of variation	df	SS	MS	Pseudo-F	P(MC)
Year	6	573.14	95.52	6.49	**<0.001**
Residual	35	515.33	14.72		
Total	41	1088.50			

Pairwise tests (based on 462 unique permutations)

Years	t	P(MC)	# perms
02, 03	1.418	0.183	26
02, 04	0.535	0.611	20
02, 06	1.008	0.339	23
02, 07	2.029	0.072	31
02, 08	3.393	**0.008**	36
02, 09	1.947	0.084	26
03, 04	3.247	**0.008**	19
03, 06	0.829	0.426	13
03, 07	0.884	0.407	20
03, 08	2.781	**0.020**	23
03, 09	0.636	0.544	14
04, 06	3.250	**0.008**	17
04, 07	4.113	**0.003**	25
04, 08	6.656	**<0.001**	33
04, 09	5.124	**0.001**	21
06, 07	1.826	0.090	17
06, 08	4.189	**0.002**	23
06, 09	1.929	0.074	13
07, 08	1.779	0.101	20
07, 09	0.430	0.674	15
08, 09	2.663	**0.024**	19

Table 13: Numerically dominant periphyton taxa recorded on stones collected from the riffle habitat at CR1 during each of the (a) spring and (b) autumn surveys and their percentage contribution to the total number of periphyton cells. Note that only taxa accounting for >1% of the total cells are listed.

(a) Autumn surveys

March 2002		March 2003		March 2004		March 2006		April 2007		April 2008		March 2009	
Taxon	**%**	**Taxon**	**%**	**Taxon**	**%**	**Taxon**	**%**	**Taxon**	**%**	**Taxon**	**%**	**Taxon**	**%**
Fragilaria	35.80	*Fragilaria*	39.27	*Fragilaria*	62.09	*Oscillatoria*	21.05	*Navicula*	49.15	*Fragilaria*	26.66	*Rhoicosphenia*	55.44
Oscillatoria	22.95	*Planktolyngbya*	16.10	*Leptolyngbya*	7.60	*Rhoicosphenia*	15.88	*Nitzschia*	15.72	*Navicula*	20.14	*Navicula*	8.53
Planktolyngbya	8.11	*Amphora*	14.89	*Amphora*	6.42	*Fragilaria*	13.51	*Leptolyngbya*	13.15	*Leptolyngbya*	14.67	*Amphora*	8.21
Scenedesmus	5.42	*Oscillatoria*	12.60	*Oscillatoria*	6.16	*Navicula*	12.65	*Oscillatoria*	11.45	*Synedra*	14.57	*Melosira*	5.75
Rhoicosphenia	4.36	*Rhoicosphenia*	6.85	*Melosira*	4.02	*Bacillaria*	11.97	*Rhoicosphenia*	5.56	*Melosira*	10.56	*Cocconeis*	4.71
Navicula	3.50	*Scenedesmus*	2.13	*Navicula*	2.54	*Nitzschia*	9.88	*Melosira*	1.34	*Rhoicosphenia*	4.22	*Synedra*	3.99
Synedra	3.00	*Melosira*	1.89	*Synedra*	2.07	*Amphora*	5.28			*Nitzschia*	1.67	*Fragilaria*	2.88
Anabaena	2.00	*Nitzschia*	1.63	*Oedogonium*	1.84	*Melosira*	3.74			*Surirella*	1.43	*Epithemia*	1.98
Cymbella	1.74	*Gomphonema*	1.28	*Rhoicosphenia*	1.84	*Scenedesmus*	3.25			*Gyrosigma*	1.27	*Tabellaria*	1.89
Oocystis	1.40	*Navicula*	1.26	*Nitzschia*	1.71					*Amphora*	1.05	*Oedogonium*	1.68
Ankistrodesmus	1.33	*Phormidium*	1.09							*Oedogonium*	1.01	*Nitzschia*	1.49
Raphidiopsis	1.22											*Bacillaria*	1.01
Oedogonium	1.17												
Melosira	1.11												

(b) Spring surveys

November 2002		November 2003		November 2005		October 2006		November 2007		November 2008		October 2009	
Taxon	**%**	**Taxon**	**%**	**Taxon**	**%**	**Taxon**	**%**	**Taxon**	**%**	**Taxon**	**%**	**Taxon**	**%**
Planktolyngbya	38.34	*Fragilaria*	66.61	*Rhoicosphenia*	42.53	*Navicula*	44.65	*Navicula*	48.34	*Navicula*	72.94	*Navicula*	55.62
Oscillatoria	30.59	*Oscillatoria*	8.20	*Leptolyngbya*	15.56	*Fragilaria*	21.85	*Melosira*	38.41	*Fragilaria*	6.28	*Rhoicosphenia*	15.67
Fragilaria	8.25	*Nitzschia*	4.34	*Navicula*	15.35	*Rhoicosphenia*	11.52	*Rhoicosphenia*	3.89	*Melosira*	6.22	*Surirella*	9.97
Scenedesmus	3.98	*Amphora*	3.70	*Oscillatoria*	6.98	*Audouinella*	7.35	*Synedra*	3.19	*Surirella*	5.22	*Melosira*	5.84
Amphora	3.76	*Raphidiopsis*	3.48	*Mougeotia*	3.52	*Synedra*	5.11	*Fragilaria*	2.77	*Rhoicosphenia*	3.36	*Leptolyngbya*	4.15
Rhoicosphenia	3.66	*Stigeoclonium*	2.30	*Fragilaria*	3.49	*Leptolyngbya*	2.42	*Leptolyngbya*	1.95	*Leptolyngbya*	2.05	*Fragilaria*	2.50
Nitzschia	2.74	*Navicula*	2.16	*Cladophora*	2.72	*Oscillatoria*	1.33			*Oedogonium*	1.26	*Oscillatoria*	2.28
Phormidium	1.71	*Rhoicosphenia*	2.14	*Melosira*	1.81	*Amphora*	1.31					*Synedra*	1.73
Oedogonium	1.59	*Anabaena*	1.66	*Stigeoclonium*	1.42	*Nitzschia*	1.28						
Navicula	1.58	*Scenedesmus*	1.15	*Nitzschia*	1.06	*Ankistrodesmus*	1.04						
				Tabellaria	1.05	*Cocconeis*	1.03						

Table 14: Results of PERMANOVA tests comparing the structure of periphyton assemblages associated with stones in the riffle habitat at CR1 in (a) autumn and (b) spring of 2002-2009. Bold indicates significant at *P* < 0.05.

(a) Autumn

Main test (based on 4966 unique permutations)

Source of variation	df	SS	MS	Pseudo-F	P(MC)
Year	6	35297.00	5882.90	11.80	**<0.001**
Residual	35	17446.00	498.47		
Total	41	52744.00			

Pairwise tests (based on 462 unique permutations)

Years	t	P(MC)
02, 03	2.054	**0.005**
02, 04	2.951	**0.001**
02, 06*	3.810	**<0.001**
02, 07	3.834	**<0.001**
02, 08	3.981	**<0.001**
02, 09	3.539	**<0.001**
03, 04	4.028	**<0.001**
03, 06*	4.731	**<0.001**
03, 07	4.967	**<0.001**
03, 08	5.281	**<0.001**
03, 09	4.372	**<0.001**
04, 06*	3.417	**<0.001**
04, 07	2.893	**<0.001**
04, 08	2.668	**0.001**
04, 09	3.322	**<0.001**
06, 07*	2.331	**0.003**
06, 08*	2.982	**0.001**
06, 09*	2.866	**0.000**
07, 08	2.215	**0.003**
07, 09	2.685	**0.002**
08, 09	2.472	**0.002**

(b) Spring

Main test (based on 4968 unique permutations)

Source of variation	df	SS	MS	Pseudo-F	P(MC)
Year	6	37121.00	6186.80	11.52	**<0.001**
Residual	33	17731.00	537.29		
Total	39	54852.00			

Pairwise tests (based on 462 unique permutations)

Years	t	P(MC)
02, 03	2.166	**0.006**
02, 04	3.836	**<0.001**
02, 06*	3.736	**0.001**
02, 07	5.230	**<0.001**
02, 08	5.591	**<0.001**
02, 09	5.990	**<0.001**
03, 04	2.817	**0.001**
03, 06*	2.423	**0.002**
03, 07	3.564	**0.001**
03, 08	4.047	**<0.001**
03, 09	4.185	**<0.001**
04, 06*	1.944	**0.014**
04, 07	2.851	**0.001**
04, 08	3.098	**<0.001**
04, 09	2.890	**<0.001**
06, 07*	2.448	**0.002**
06, 08*	2.773	**0.001**
06, 09*	2.751	**0.001**
07, 08	2.317	**0.001**
07, 09	2.962	**0.001**
08, 09	1.971	**0.006**

Table 15: Results of PERMANOVA tests comparing the diversity of periphyton assemblages associated with stones in the riffle habitat at CR1 in (a) autumn and (b) spring of 2002-2009. Bold indicates significant at $P < 0.05$

(a) Spring

Main test (based on 617 unique permutations)

Source of variation	df	SS	MS	Pseudo-F	P(MC)
Year	6	238.33	39.72	3.96	**0.005**
Residual	35	351.50	10.04		
Total	41	589.83			

Pairwise tests (based on 462 unique permutations)

Years	t	P(MC)	# perms
02, 03	0.515	0.515	12
02, 04	0.092	0.092	9
02, 06*	0.094	0.094	19
02, 07	0.190	0.190	8
02, 08	0.006	**0.006**	9
02, 09	0.000	**<0.001**	12
03, 04	0.518	0.518	14
03, 06*	0.073	0.073	25
03, 07	0.183	0.183	15
03, 08	0.045	**0.045**	15
03, 09	0.012	**0.012**	17
04, 06*	0.039	**0.039**	24
04, 07	0.029	**0.029**	13
04, 08	0.004	**0.004**	16
04, 09	0.001	**0.001**	18
06, 07*	0.215	0.215	21
06, 08*	0.384	0.384	18
06, 09*	0.637	0.637	18
07, 08	0.291	0.291	9
07, 09	0.039	**0.039**	11
08, 09	0.129	0.129	6

(b) Autumn

Main test (based on 606 unique permutations)

Source of variation	df	SS	MS	Pseudo-F	P(MC)
Year	6	231.90	38.65	6.19	**<0.001**
Residual	35	218.67	6.25		
Total	41	450.57			

Pairwise tests (based on 462 unique permutations)

Years	t	P(MC)	# perms
02, 03	2.834	**0.017**	15
02, 04	0.686	0.493	12
02, 06*	3.689	**0.004**	17
02, 07	0.560	0.584	13
02, 08	2.217	**0.050**	10
02, 09	3.233	**0.007**	20
03, 04	3.822	**0.002**	17
03, 06*	0.775	0.463	11
03, 07	1.861	0.092	17
03, 08	1.686	0.123	10
03, 09	0.749	0.477	13
04, 06*	4.833	**0.001**	18
04, 07	1.184	0.251	13
04, 08	3.612	**0.004**	11
04, 09	4.098	**0.002**	22
06, 07*	2.538	**0.029**	17
06, 08*	3.024	**0.013**	9
06, 09*	0.112	0.915	13
07, 08	1.059	0.311	10
07, 09	2.331	**0.040**	20
08, 09	2.284	**0.046**	12

Table 16: Numbers of each species of fish and mobile invertebrate caught in the electrofishing undertaken at each site at CR1 since autumn 2002

	Autumn 2002			Spring 2002			Autumn 2003		
	A	B	C	A	B	C	A	B	C
Fish									
Mountain galaxias	0	0	0	0	0	0	0	0	0
Flathead gudgeon	0	4	0	0	0	0	0	0	0
Rainbow trout*	2	2	0	0	0	0	0	0	0
Brown trout*	3	0	0	1	0	0	1	1	0
Goldfish*	0	0	0	0	0	0	0	0	0
Mosquito fish*	4	0	1	0	0	30	24	45	42
Invertebrates									
Yabbies	0	0	0	0	0	0	0	0	0
Shrimp	0	0	0	0	0	0	0	0	0

	Spring 2003			Autumn 2004			Spring 2004		
	A	B	C	A	B	C	A	B	C
Fish									
Mountain galaxias	0	0	0	0	0	0	0	0	0
Flathead gudgeon	0	0	0	0	0	0	0	0	0
Rainbow trout*	0	0	0	4	1	3	1	0	0
Brown trout*	21	1	6	0	0	0	0	0	0
Goldfish*	0	0	2	0	0	0	0	0	0
Mosquito fish*	0	17	0	83	164	24	0	1	45
Invertebrates									
Yabbies	0	0	0	10	6	13	0	0	0
Shrimp	0	0	0	0	0	0	0	0	0

	Spring 2006			Spring 2008			Autumn 2009		
	A	B	C	A	B	C	A	B	C
Fish									
Mountain galaxias	4	7	3	0	0	0	0	0	0
Flathead gudgeon	0	0	0	1	0	0	0	0	0
Rainbow trout*	0	0	0	14	3	2	71	15	47
Brown trout*	0	1	0	1	2	2	2	8	2
Goldfish*	0	0	0	0	0	7	0	0	11
Mosquito fish*	0	0	0	0	0	0	0	0	0
Invertebrates									
Yabbies	10	6	13	2	2	1	0	1	0
Shrimp	0	0	0	4	3	12	0	0	0

10 Figures

Figure 1: Aerial photographs showing the location of the licenced discharge points and ambient water quality monitoring locations associated with Wallerawang Power Station operations.

Figure 2: Box and whisker plots for (a) electrical conductivity and (b) pH data from the various monitoring and discharge locations on and adjacent to the Coxs River in the vicinity of Wallerawang Power Station.

Figure 3: Box and whisker plots for (a) sulfate and (b) non-filterable residue data from the various monitoring and discharge locations on and adjacent to the Coxs River in the vicinity of Wallerawang Power Station.

Figure 4: Box and whisker plots for (a) turbidity and (b) fluoride data from the various monitoring and discharge locations on and adjacent to the Coxs River in the vicinity of Wallerawang Power Station.

Figure 5: Box and whisker plots for (a) boron and (b) manganese data from the various monitoring and discharge locations on and adjacent to the Coxs River in the vicinity of Wallerawang Power Station.

Figure 6: Box and whisker plots for (a) iron and (b) selenium data from the various monitoring and discharge locations on and adjacent to the Coxs River in the vicinity of Wallerawang Power Station.

Figure 7: Box and whisker plots for (a) total nitrogen), (b) total Kjeldahl nitrogen, (c) total ammonia as nitrogen, (d) total nitrogen oxides, (e) nitrate and (f) nitrite data from three monitoring locations on the Coxs River in the vicinity of Wallerawang Power Station.

Figure 8: Box and whisker plots for (a) total phosphate, (b) total phosphorus, (c) dissolved oxygen and (d) total aluminium data from monitoring locations on the Coxs River in the vicinity of Wallerawang Power Station.

Figure 9: Time series of electrical conductivity data from the various monitoring and discharge points on and adjacent to the Coxs River in the vicinity of Wallerawang Power Station.

Figure 10: Time series of pH data from the various monitoring and discharge points on and adjacent to the Coxs River in the vicinity of Wallerawang Power Station.

Figure 11: Time series of sulfate data from the various monitoring and discharge points on and adjacent to the Coxs River in the vicinity of Wallerawang Power Station.

Figure 12: Time series of total suspended solids (i.e. non-filterable residue) data from the various monitoring and discharge points on and adjacent to the Coxs River in the vicinity of Wallerawang Power Station. Blue line = trend

Figure 13: Time series of fluoride data from the various monitoring and discharge points on and adjacent to the Coxs River in the vicinity of Wallerawang Power Station.

Figure 14: Time series of boron data from the various monitoring and discharge points on and adjacent to the Coxs River in the vicinity of Wallerawang Power Station.

Figure 15: Time series of manganese data from the various monitoring and discharge points on and adjacent to the Coxs River in the vicinity of Wallerawang Power Station.

Figure 16: Time series of total iron data from the various monitoring and discharge points on and adjacent to the Coxs River in the vicinity of Wallerawang Power Station.

Figure 17: Time series of selenium data from the various monitoring and discharge points on and adjacent to the Coxs River in the vicinity of Wallerawang Power Station.

Figure 18: Time series of aluminium data from two monitoring locations on the Coxs River in the vicinity of Wallerawang Power Station.

Figure 19: Time series of total nitrogen data from three monitoring points on the Coxs River in the vicinity of Wallerawang Power Station.

Figure 20: Time series of total Kjeldahl nitrogen data from three monitoring points on the Coxs River in the vicinity of Wallerawang Power Station.

Figure 21: Time series of ammonia as nitrogen data from three monitoring points on the Coxs River in the vicinity of Wallerawang Power Station.

Figure 22: Time series of total nitrogen oxides data from three monitoring points on the Coxs River in the vicinity of Wallerawang Power Station.

Figure 23: Time series of total nitrate data from three monitoring points on the Coxs River in the vicinity of Wallerawang Power Station.

Figure 24: Time series of total nitrite data from three monitoring points on the Coxs River in the vicinity of Wallerawang Power Station.

Figure 25: Time series of total filtered orthophosphate data from three monitoring points on the Coxs River in the vicinity of Wallerawang Power Station.

Figure 26: Time series of total phosphorus data from three monitoring points on the Coxs River in the vicinity of Wallerawang Power Station.

Figure 27: Time series of dissolved oxygen concentration data from three monitoring points on the Coxs River in the vicinity of Wallerawang Power Station.

Figure 28: Mean (± S.E.) OE50 taxa scores and O0 Signal scores for macroinvertebrate assemblages associated with (a) edge and (b) riffle habitat at CR1 in spring 2002 - 2009.

Figure 29: Mean (± S.E.) OE50 taxa scores and O0 Signal scores for macroinvertebrate assemblages associated with (a) edge and (b) riffle habitat at CR1 in autumn 2002 - 2009.

Figure 30: MDS plots comparing the structure of aquatic macroinvertebrate assemblages at CR1 in (a) spring and (b) autumn 2002 -2009.

Figure 31: Mean (± S.E.) number of macroinvertebrate taxa per sample collected at CR1 in (a) spring and (b) autumn 2002 -2009.

Figure 32: MDS plots comparing the structure of periphyton assemblages at CR1 in (a) spring and (b) autumn 2002 -2009.

Figure 33: Mean (± S.E.) number of periphyton taxa per sample collected at CR1 in (a) spring and (b) autumn 2002 -2009.

Figure 1: Aerial photographs showing the location of the licensed discharge points and ambient water quality monitoring locations associated with Wallerawang Power Station operations.

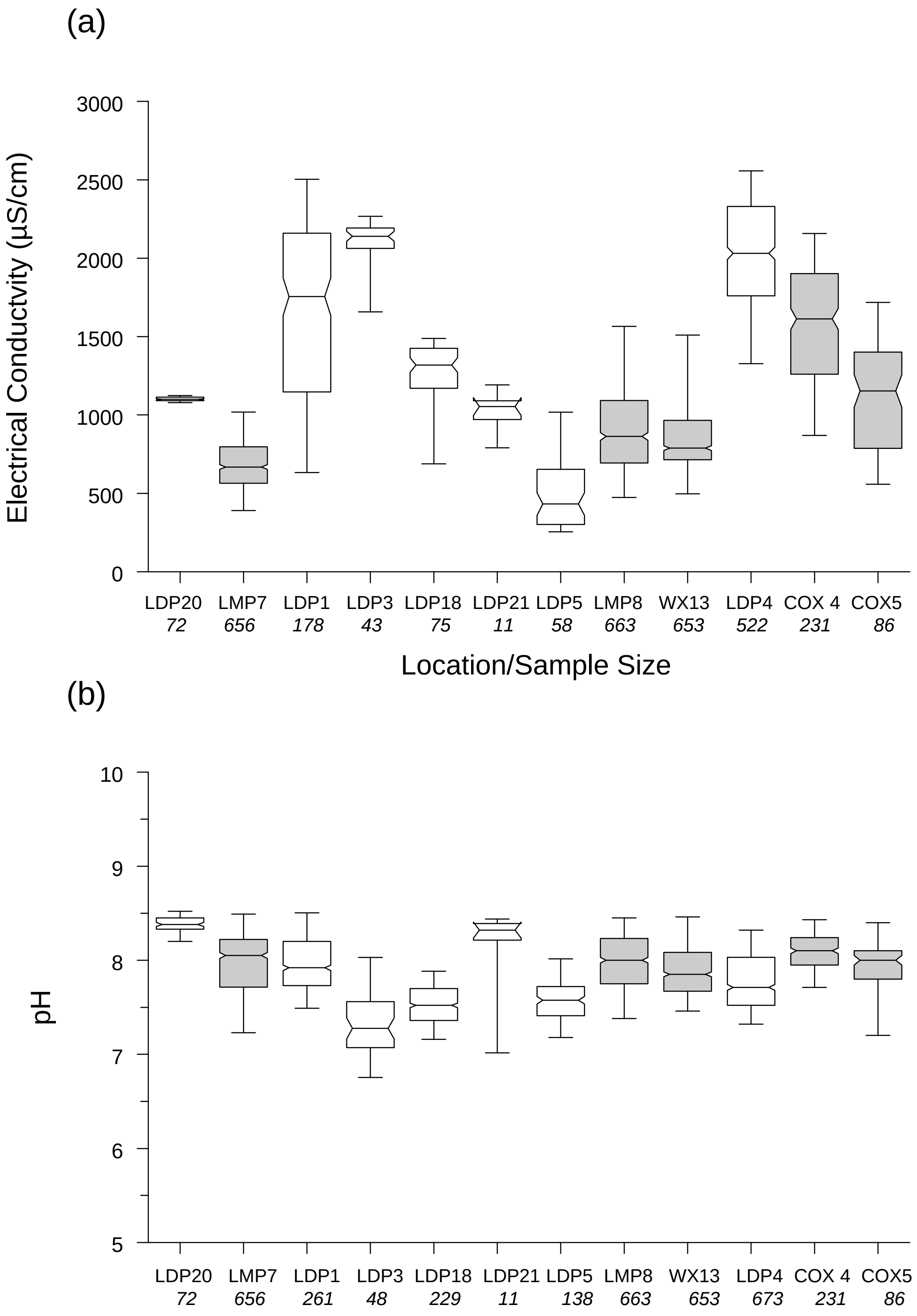
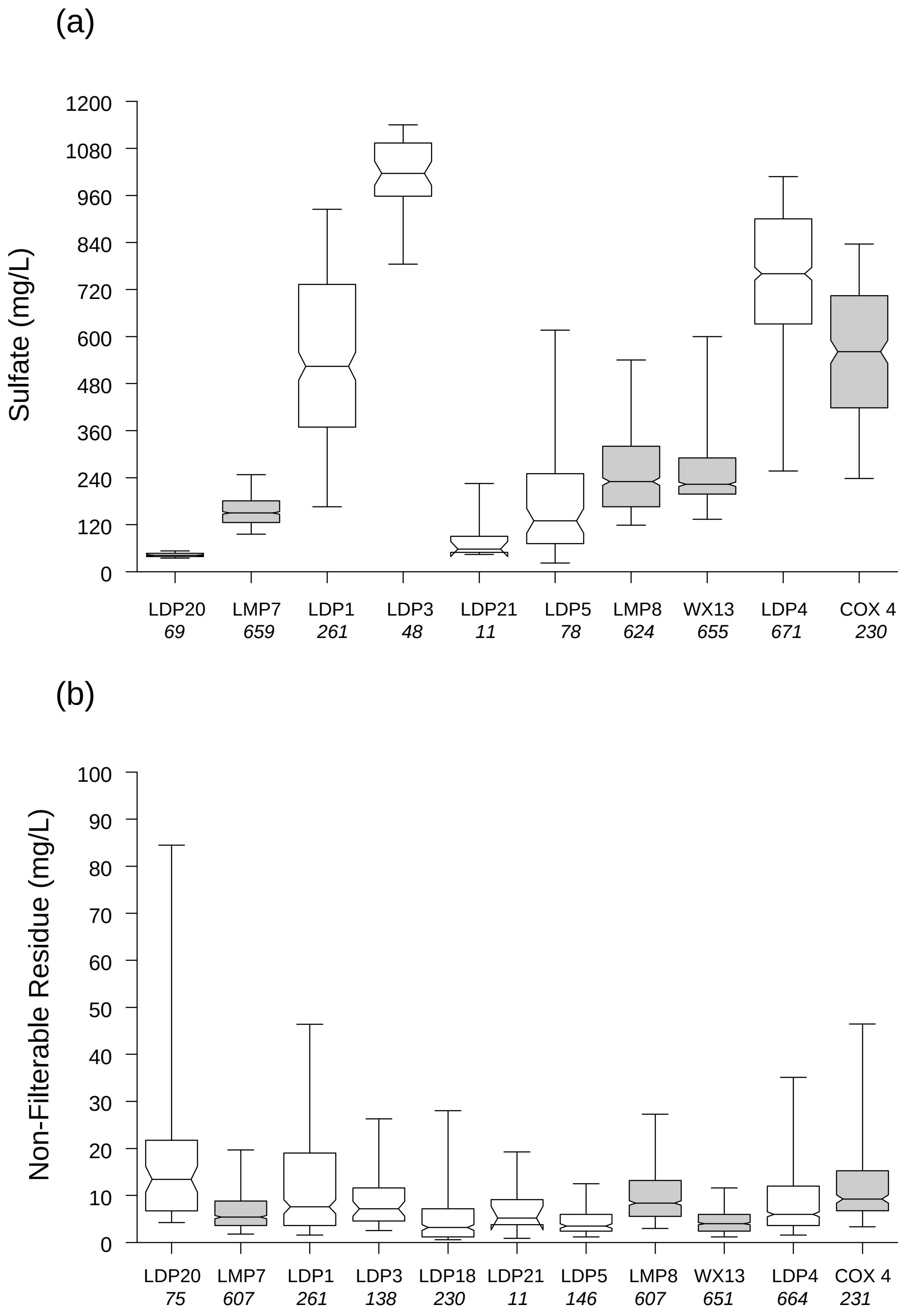


Figure 2: Box and whisker plots for (a) electrical conductivity and (b) pH data from the various monitoring (shaded) and discharge (unfilled) locations on and adjacent to the Coxs River in the vicinity of Wallerawang Power Station. Note that sites are arranged from upstream to downstream , the box represents data between the 25th and 75th percentile, whiskers extend out from the box to the 5th and 95th percentiles, the horizontal line represents the median and the notches show the confidence interval about the median.



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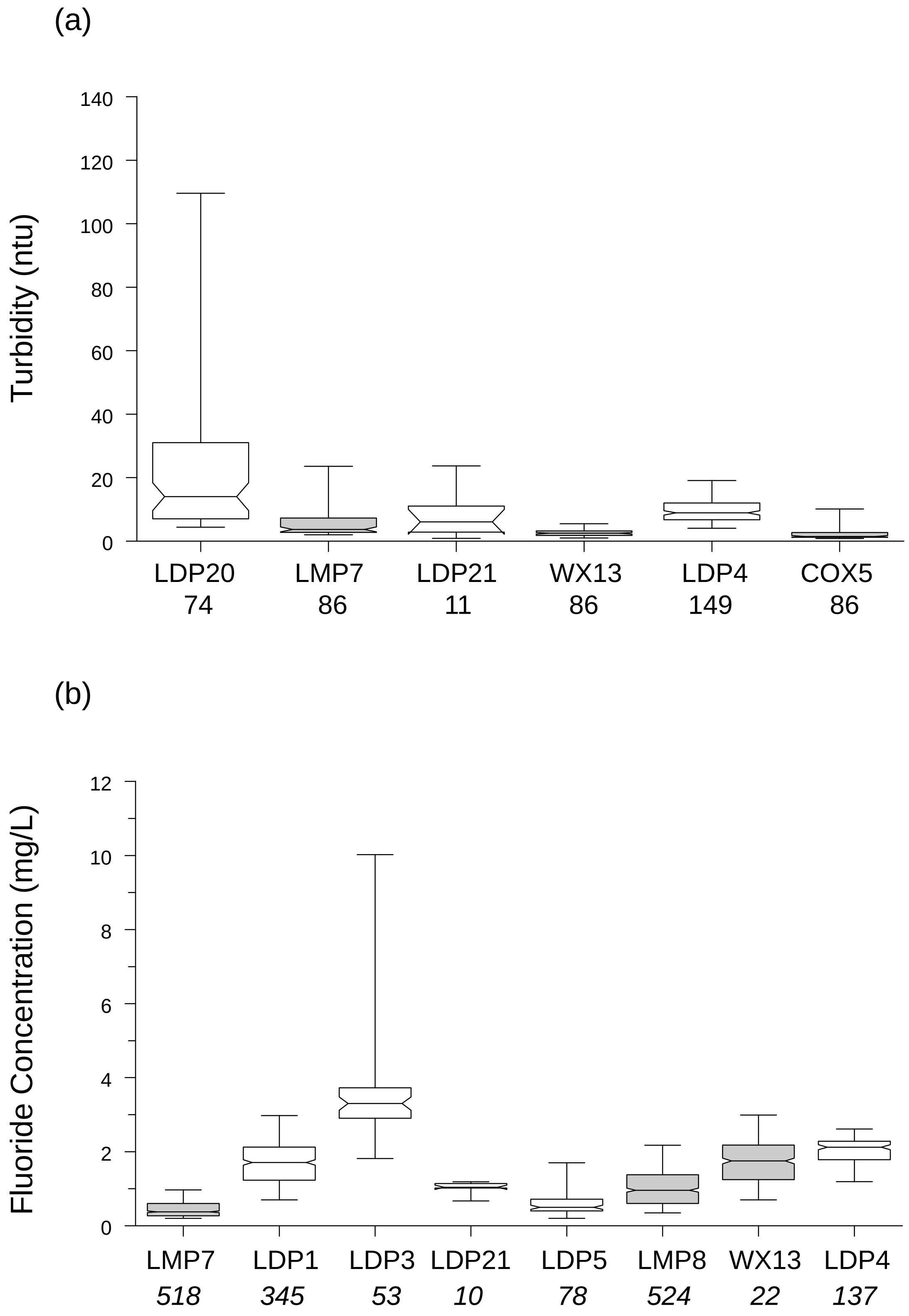


Figure 4: Box and whisker plots for (a) fluoride and (b) turbidity data from the various monitoring (shaded) and discharge (unfilled) locations on and adjacent to the Coxs River in the vicinity of Wallerawang Power Station.

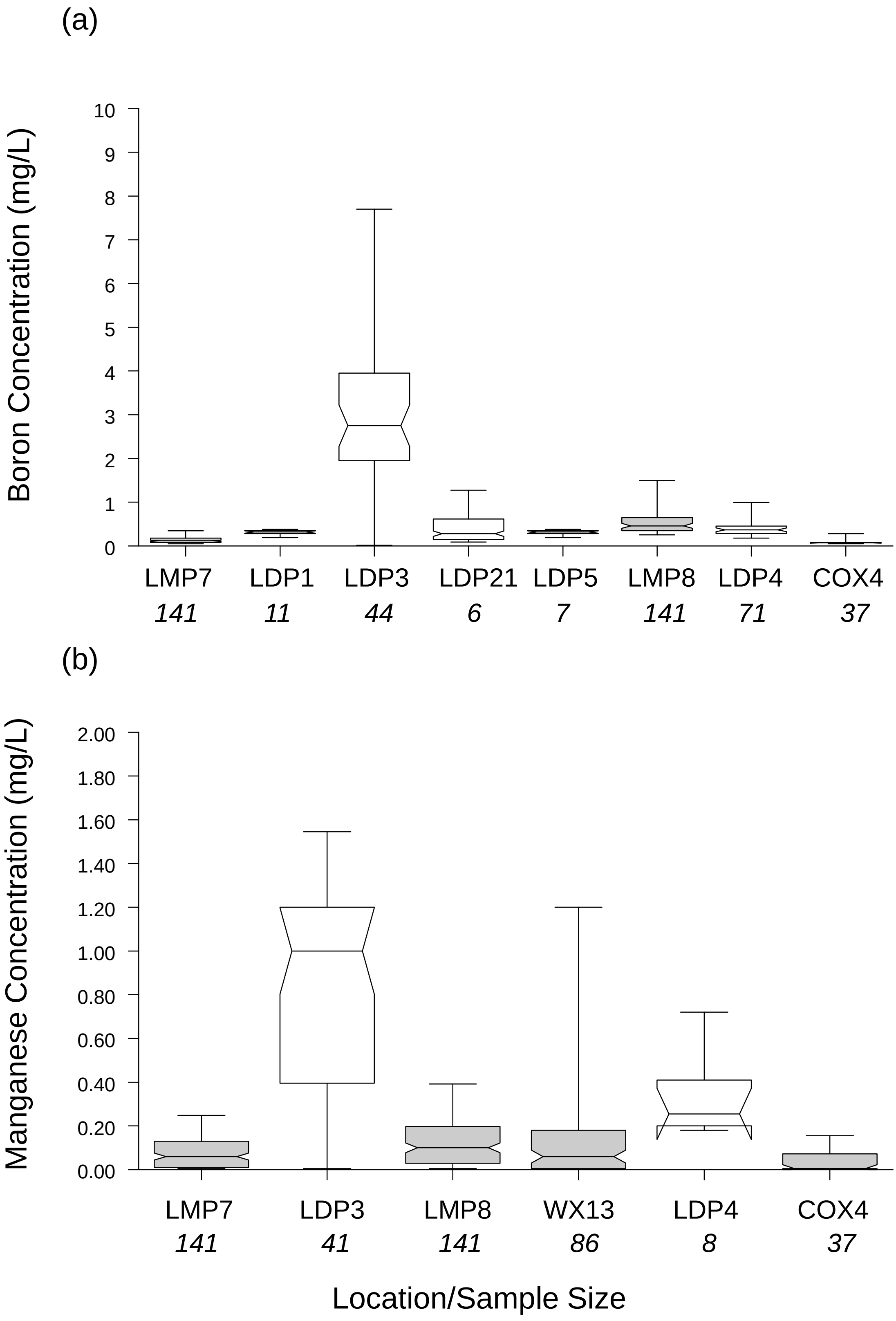


Figure 5: Box and whisker plots for (a) boron and (b) manganese data from the various monitoring (shaded) and discharge (unfilled) locations on and adjacent to the Coxs River in the vicinity of Wallerawang Power Station.

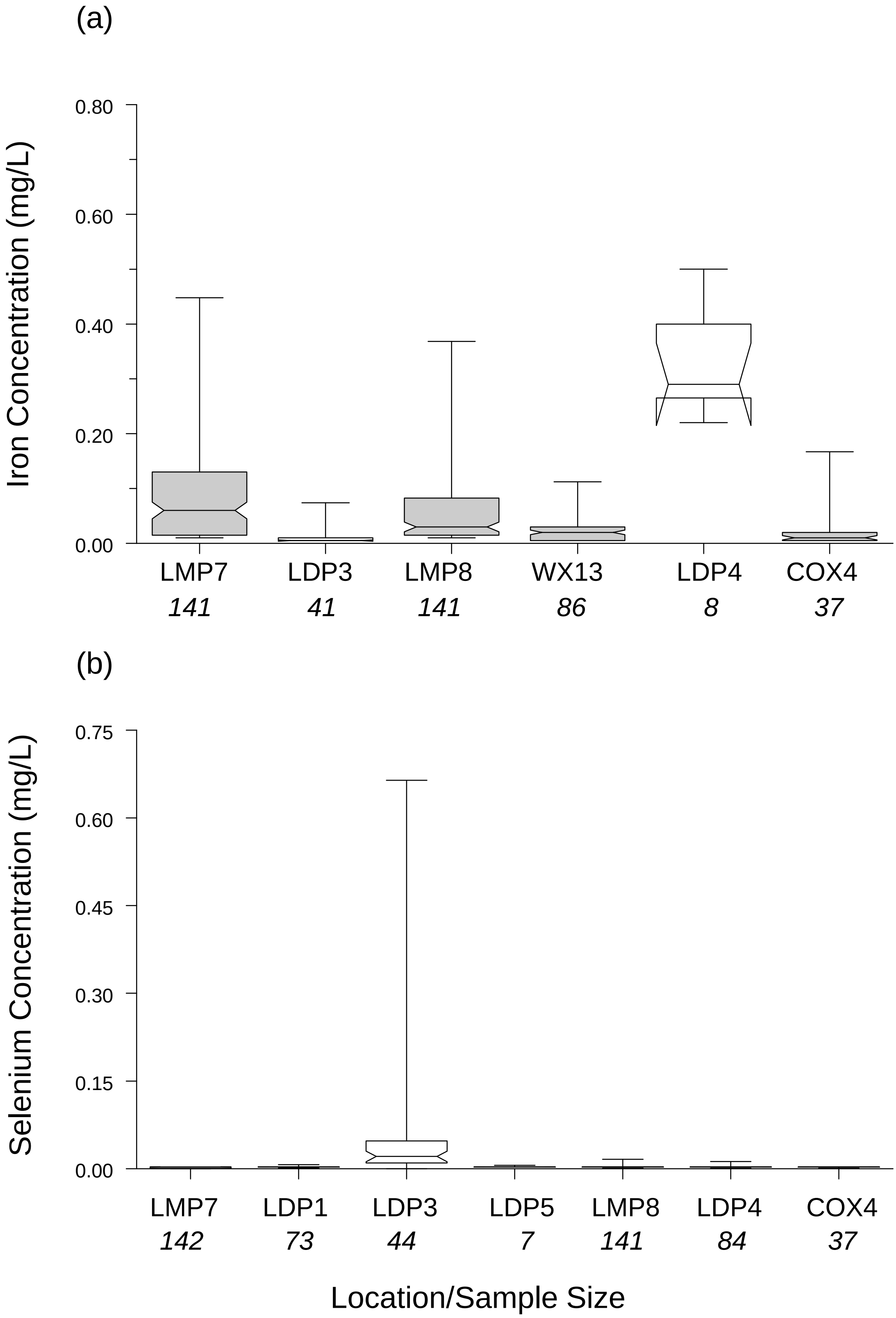
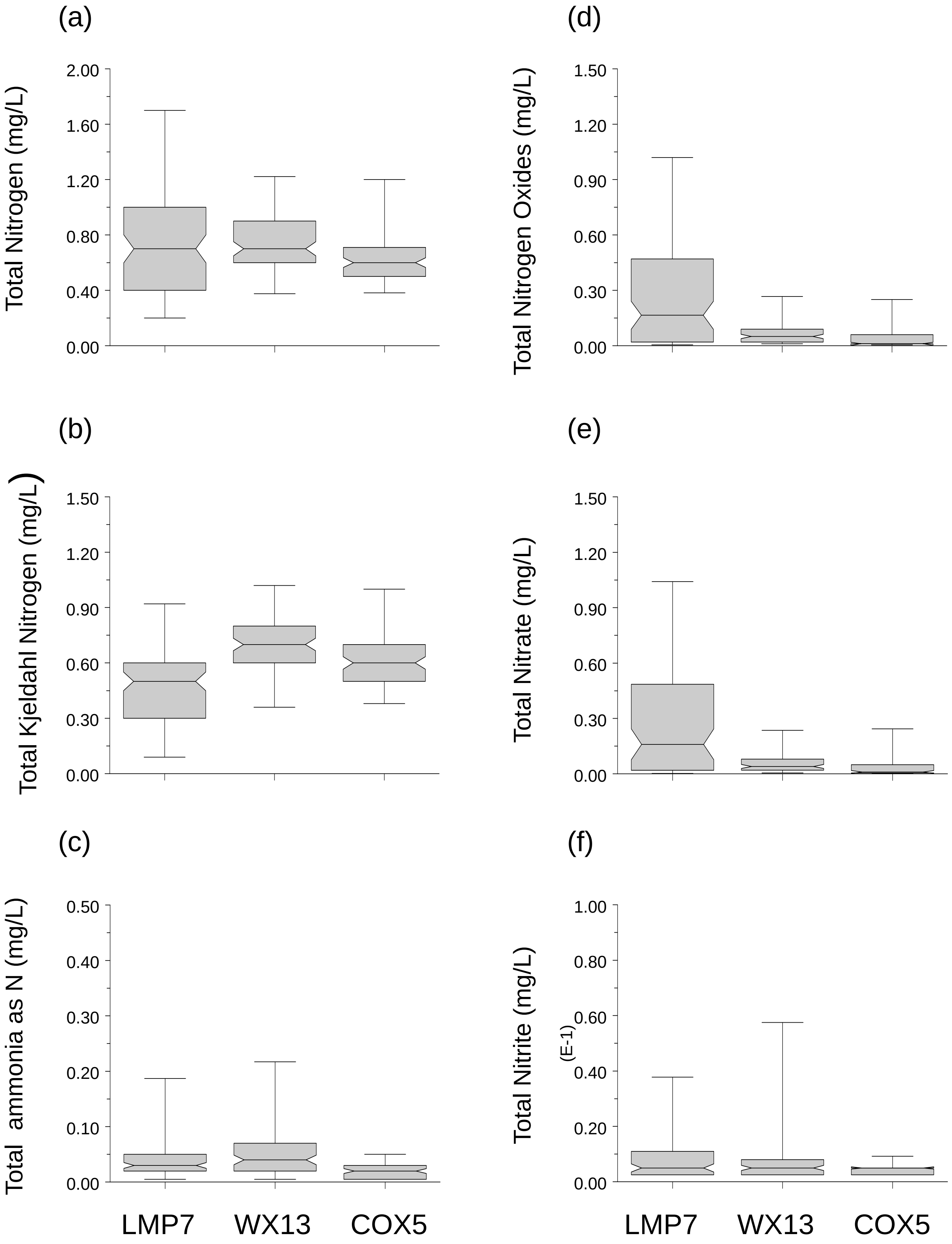


Figure 6: Box and whisker plots for (a) iron and (b) selenium data from the various monitoring (shaded) and discharge (unfilled) locations on and adjacent to the Coxs River in the vicinity of Wallerawang Power Station.



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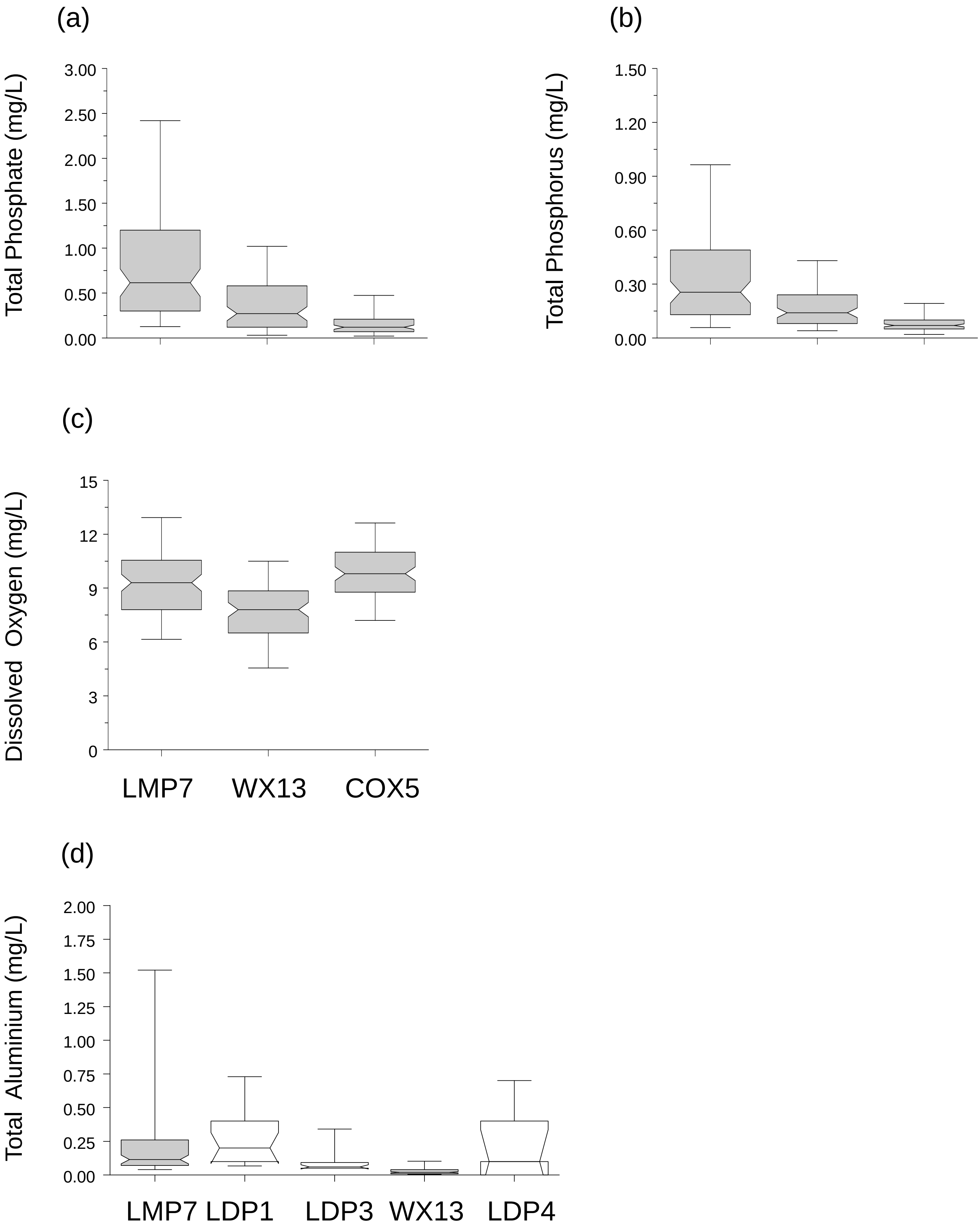


Figure 8: Box and whisker plots for (a) total phosphate, (b) total phosphorus, (c) dissolved oxygen from three monitoring locations on the Coxs River in the vicinity of Wallerawang Power Station and (d) total aluminium from five locations.

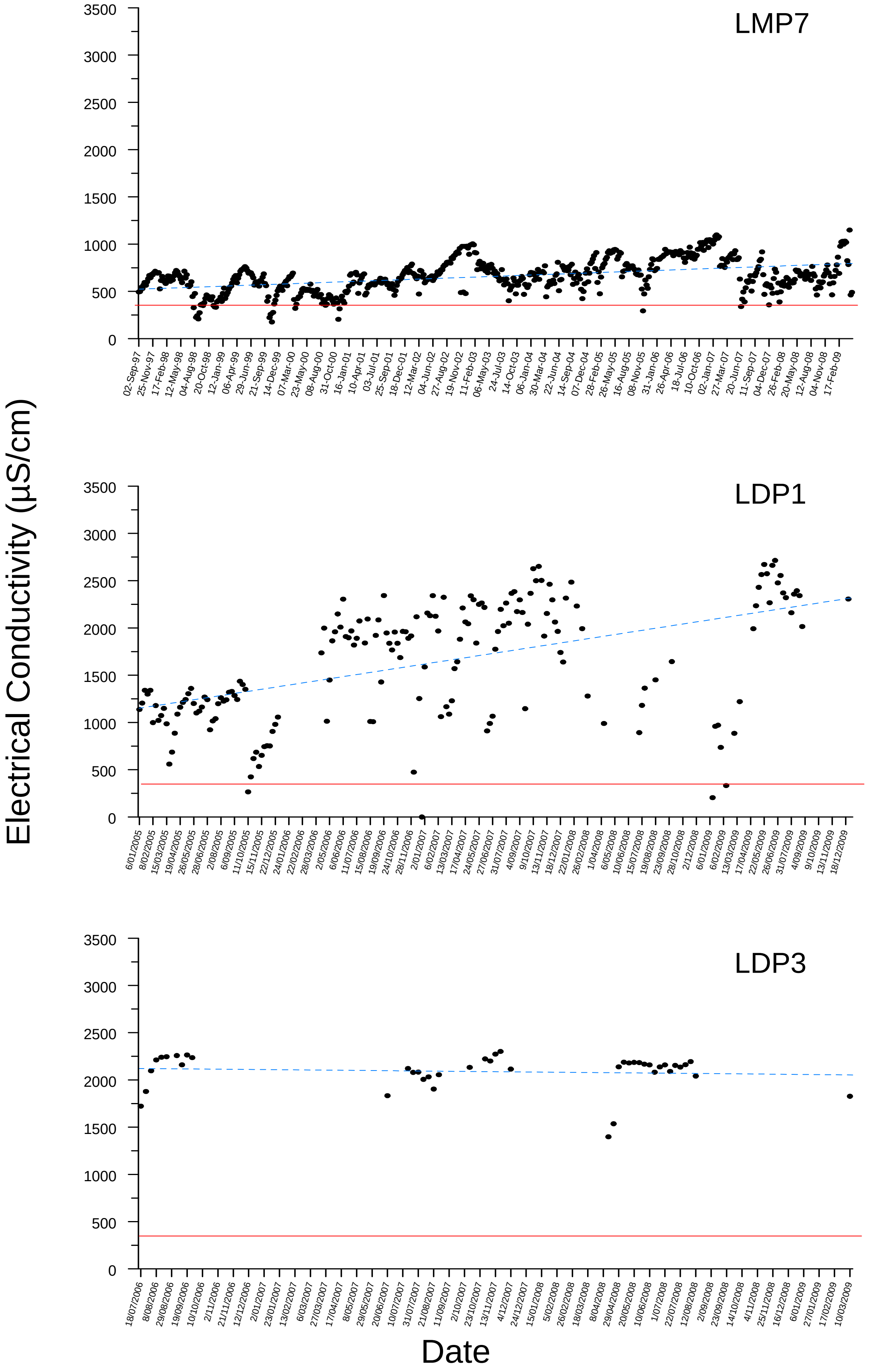


Figure 9: Time series of electrical conductivity data from the various monitoring and discharge points on and adjacent to the Coxs River in the vicinity of Wallerawang Power Station. Blue line = trend, Red line = upper guideline trigger value.

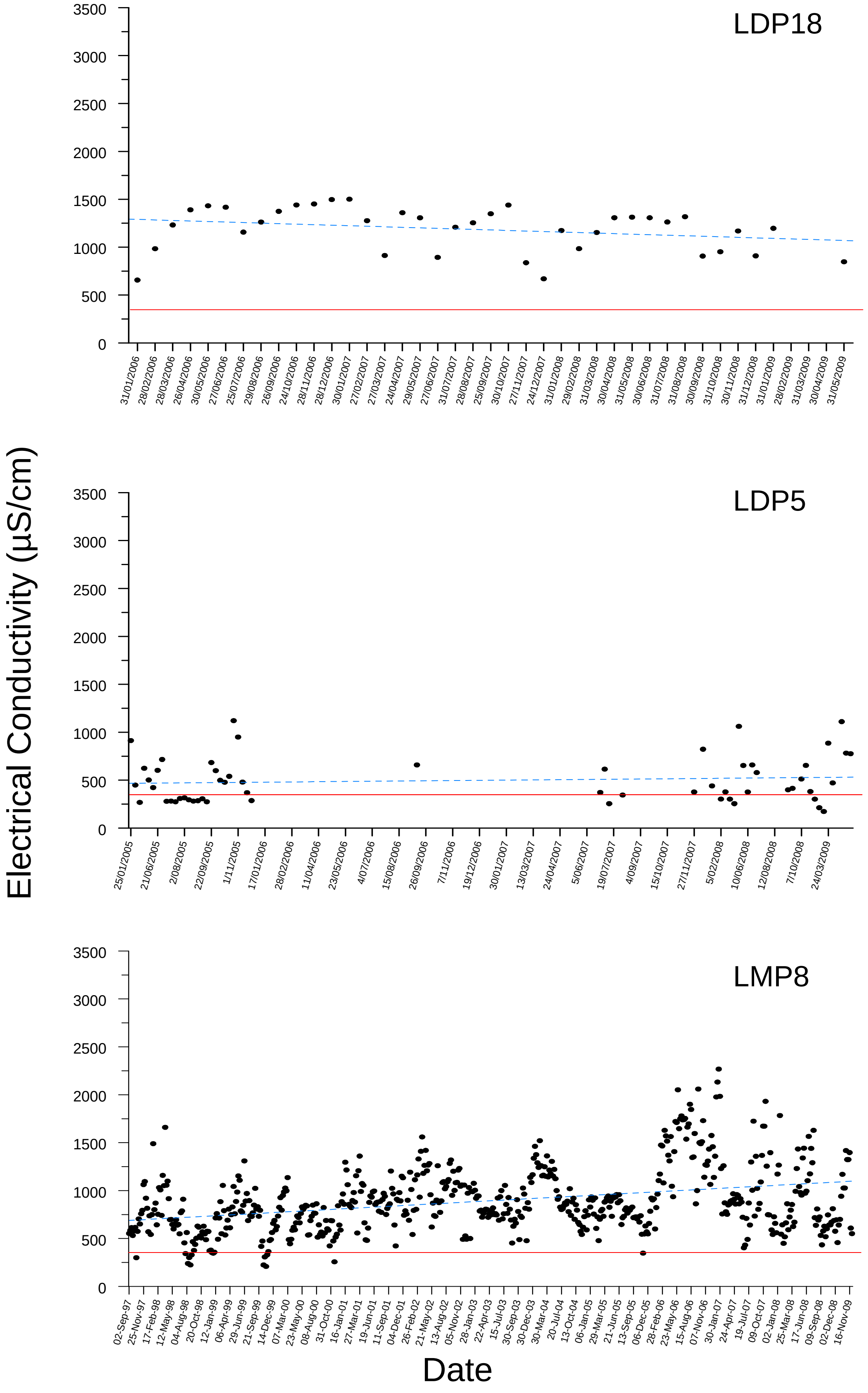


Figure 9 continued: Time series of electrical conductivity data from the various monitoring and discharge points on and adjacent to the Coxs River in the vicinity of Wallerawang Power Station. Blue line = trend, Red line = upper guideline trigger value.

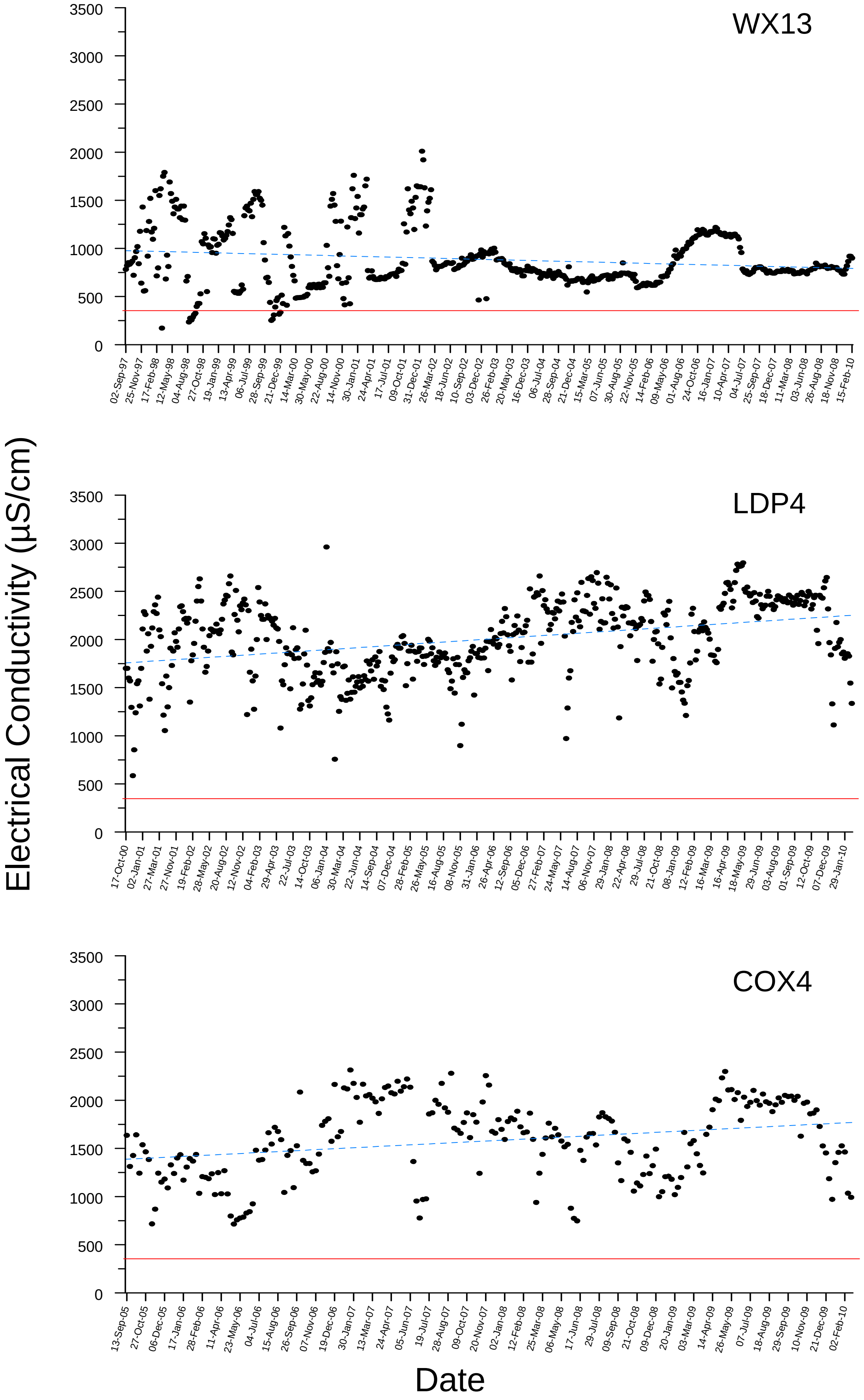
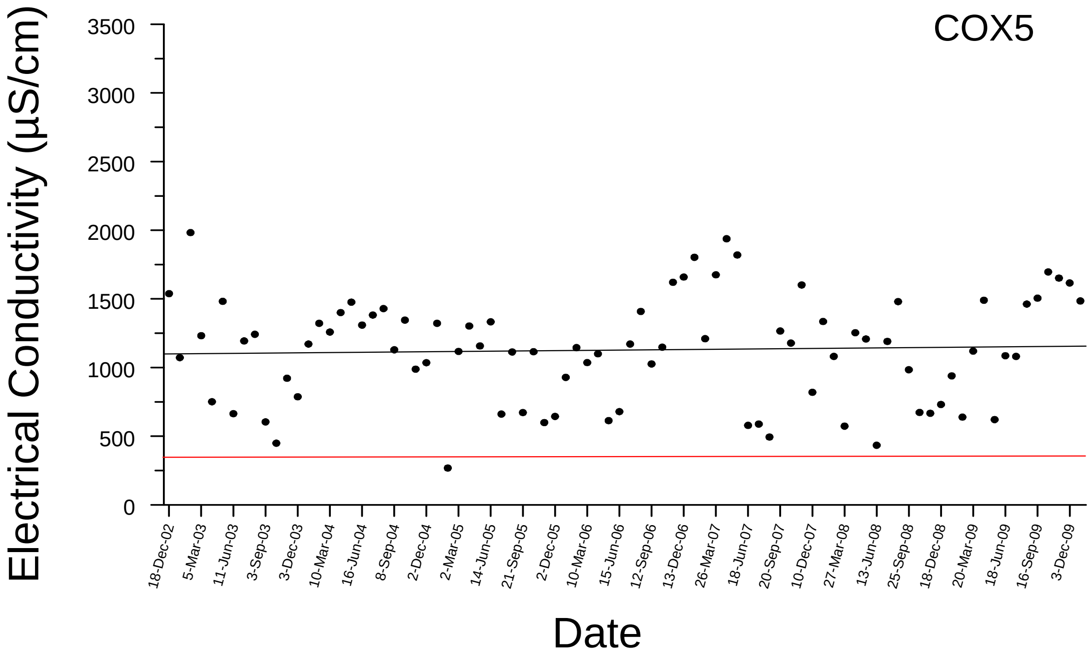


Figure 9 continued: Time series of electrical conductivity data from the various monitoring and discharge points on and adjacent to the Coxs River in the vicinity of Wallerawang Power Station. Blue line = trend, Red line = upper guideline trigger value.



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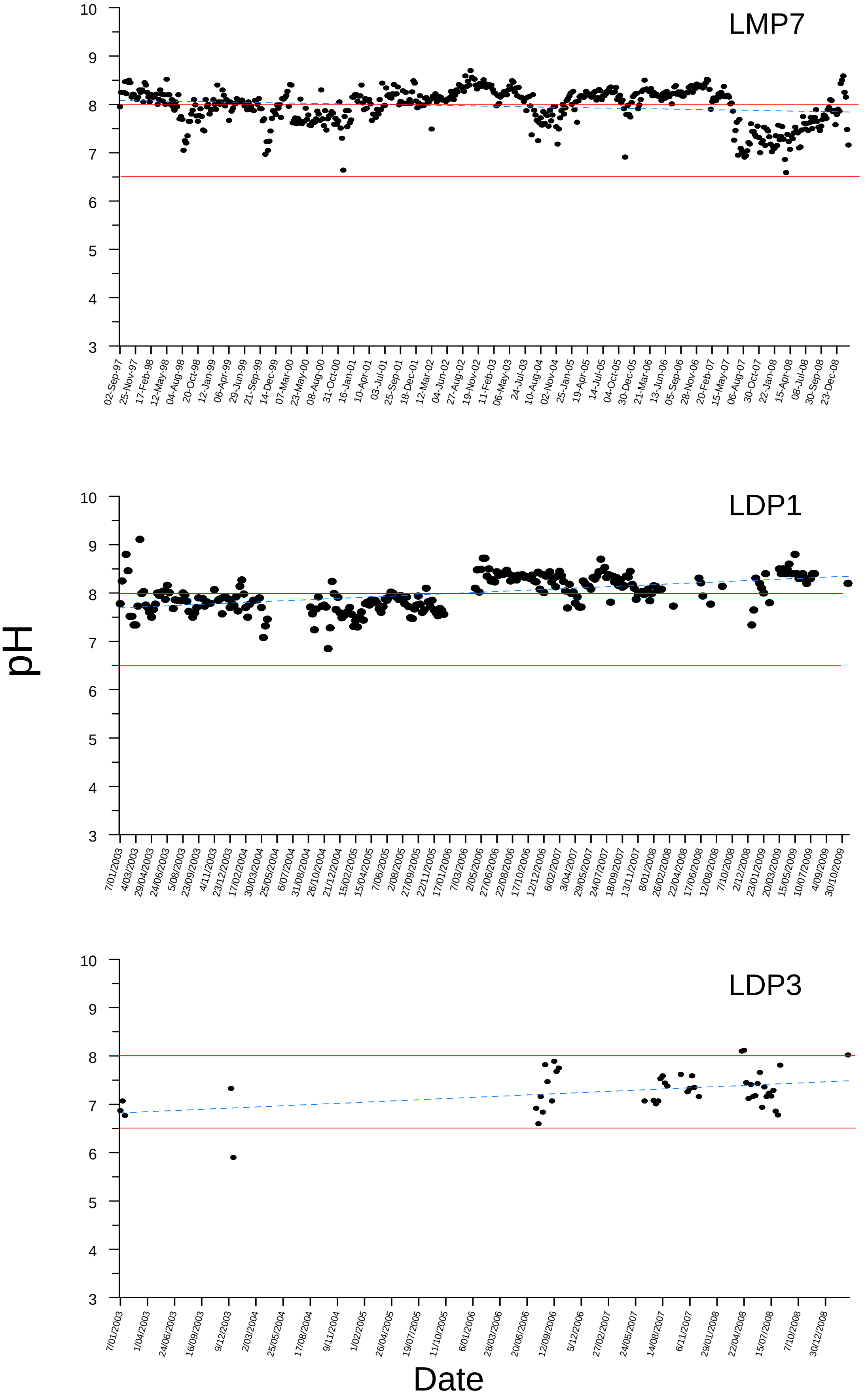


Figure 10: Time series of pH data from the various monitoring and discharge points on and adjacent to the Coxs River in the vicinity of Wallerawang Power Station. Blue line = trend, Red lines = upper and lower guideline trigger values

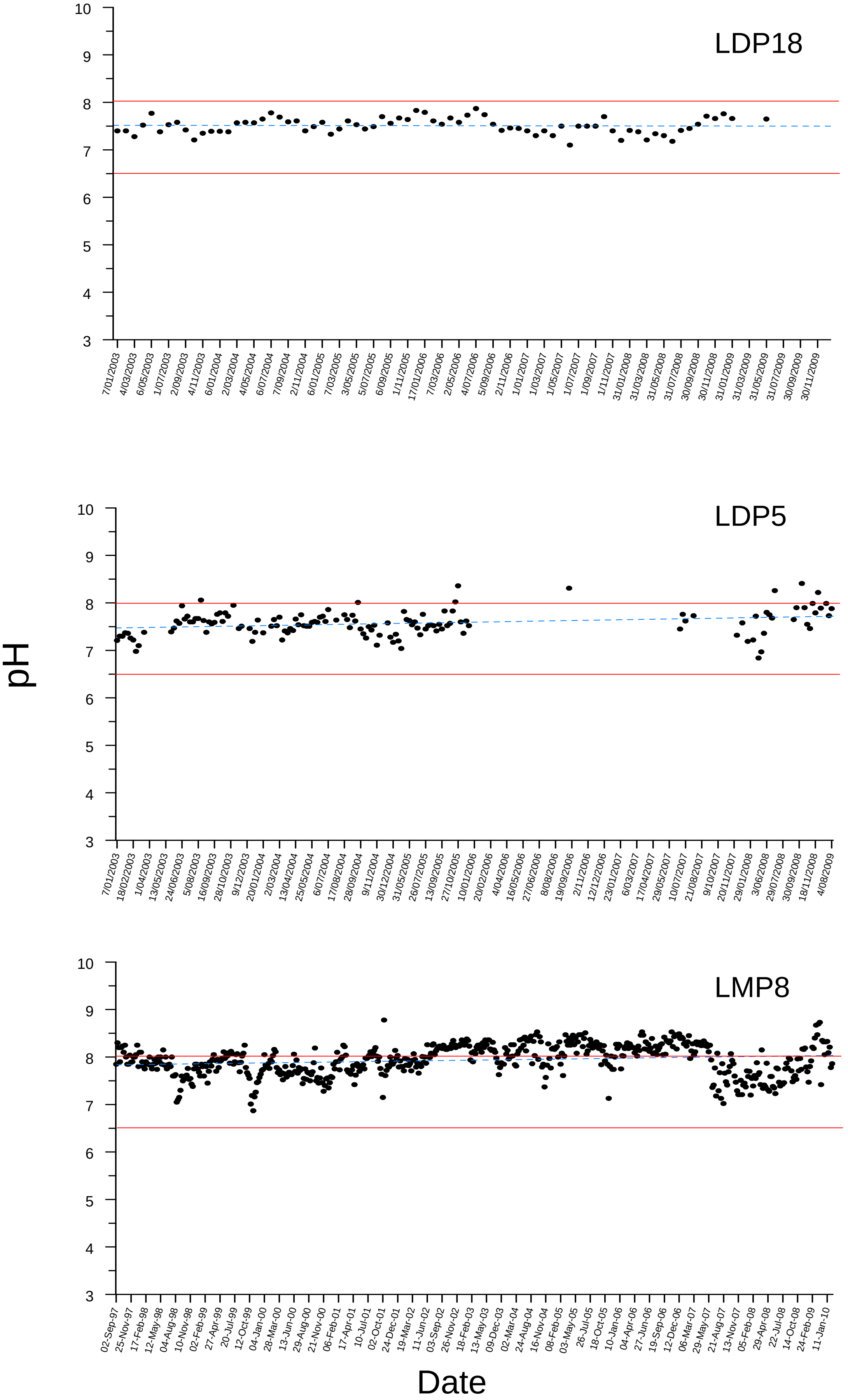


Figure 10 continued: Time series of pH data from the various monitoring and discharge points on and adjacent to the Coxs River in the vicinity of Wallerawang Power Station. Blue line = trend, Red lines = upper and lower guideline trigger values

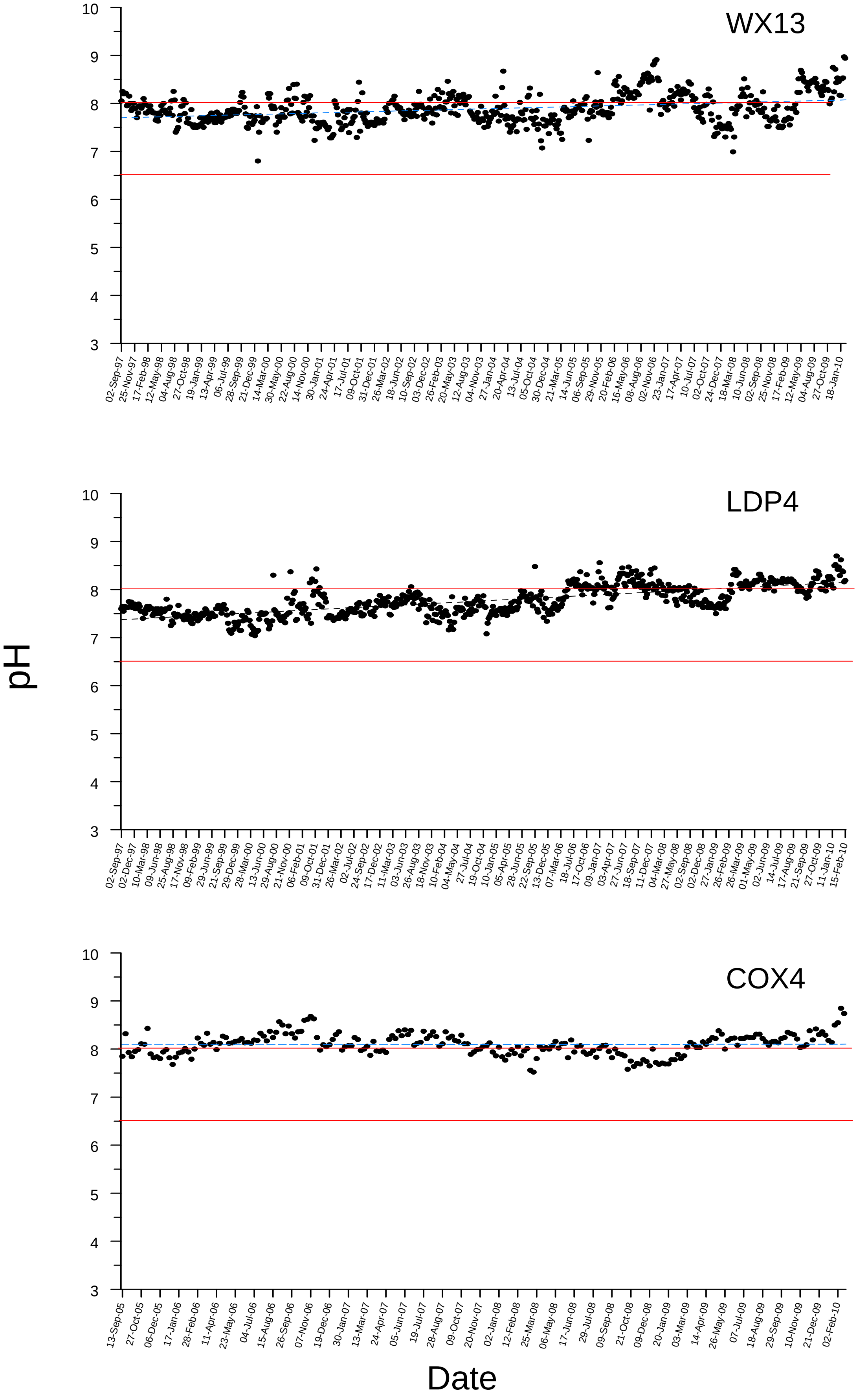


Figure 10 continued: Time series of pH data from the various monitoring and discharge points on and adjacent to the Coxs River in the vicinity of Wallerawang Power Station. Blue line = trend, Red line = upper guideline trigger value.

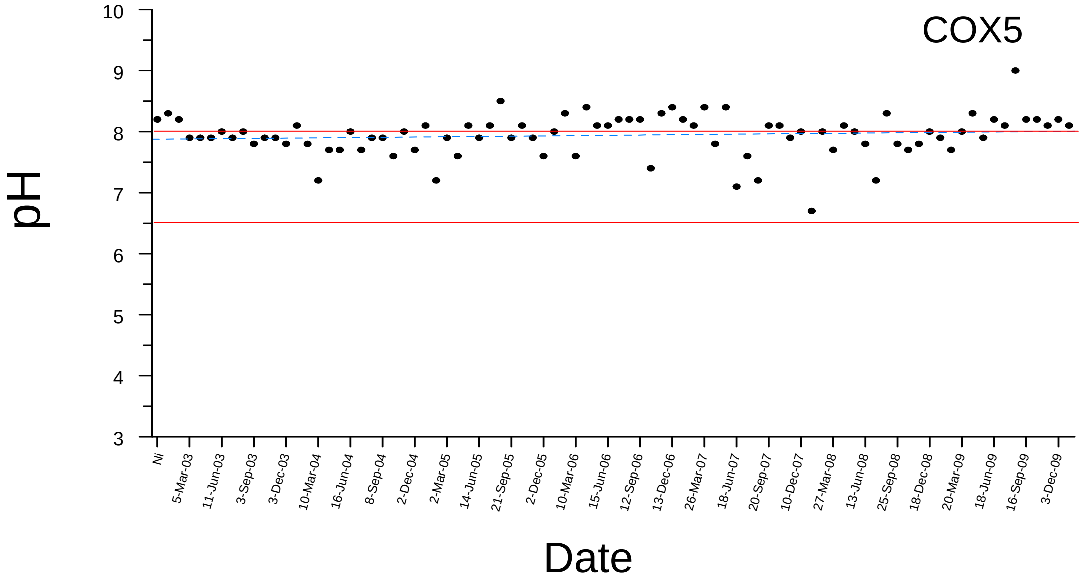


Figure 10 continued: Time series of pH data from the various monitoring and discharge points on and adjacent to the Coxs River in the vicinity of Wallerawang Power Station. Blue line = trend, Red line = upper guideline trigger value.

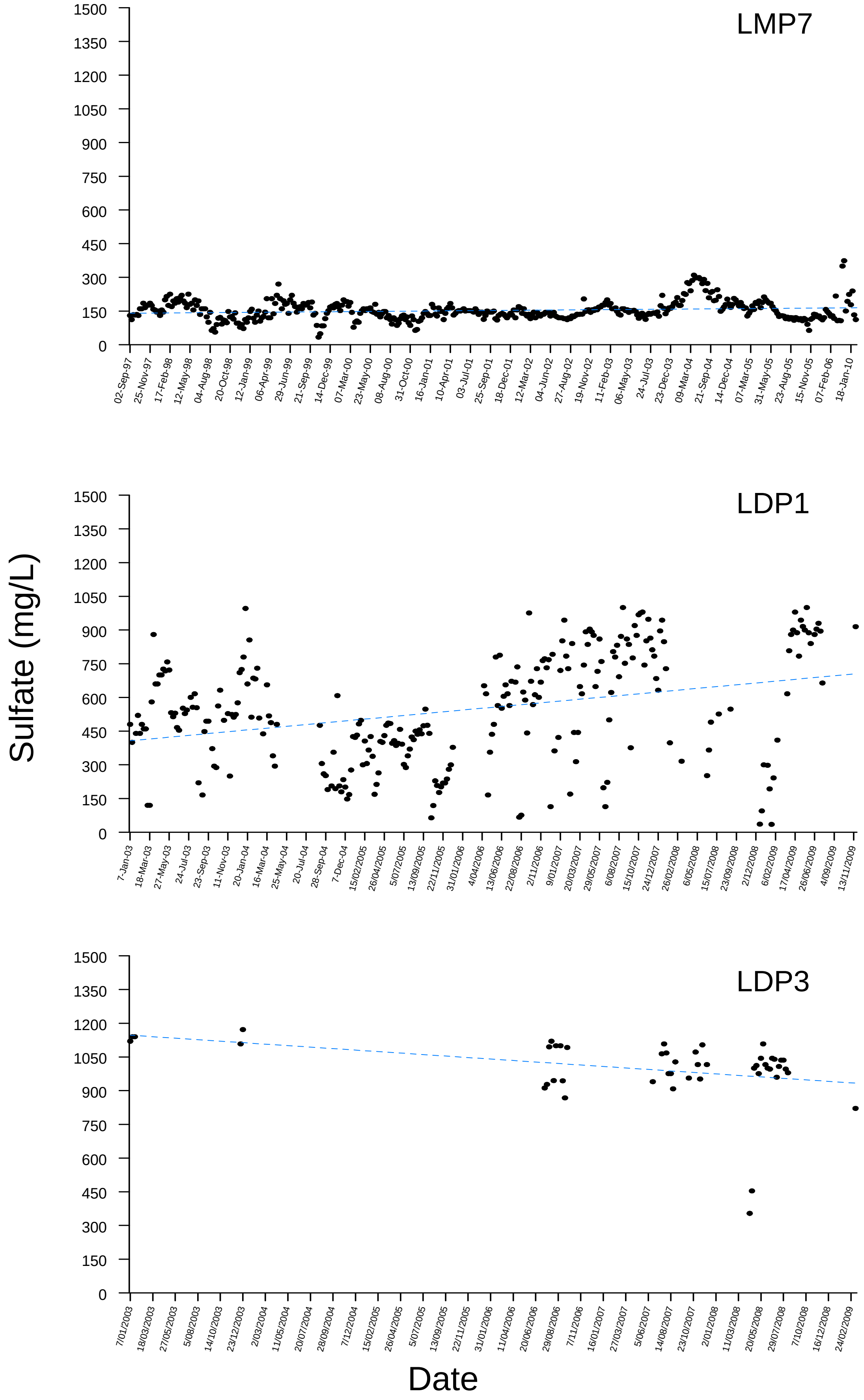


Figure 11: Time series of sulfate data from the various monitoring and discharge points on and adjacent to the Coxs River in the vicinity of Wallerawang Power Station. Blue line = trend.

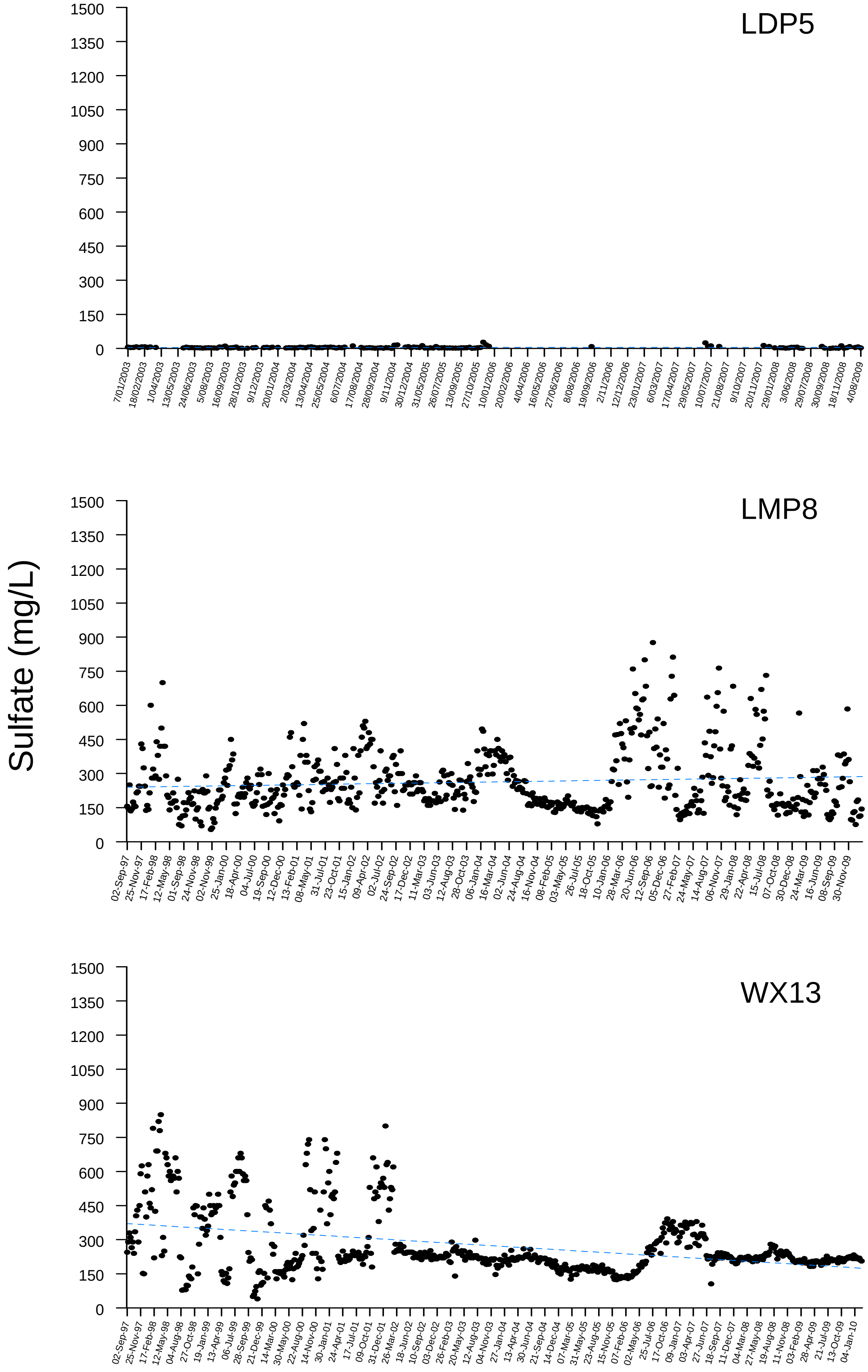


Figure 11 continued: Time series of sulfate data from the various monitoring and discharge points on and adjacent to the Coxs River in the vicinity of Wallerawang Power Station. Blue line = trend.

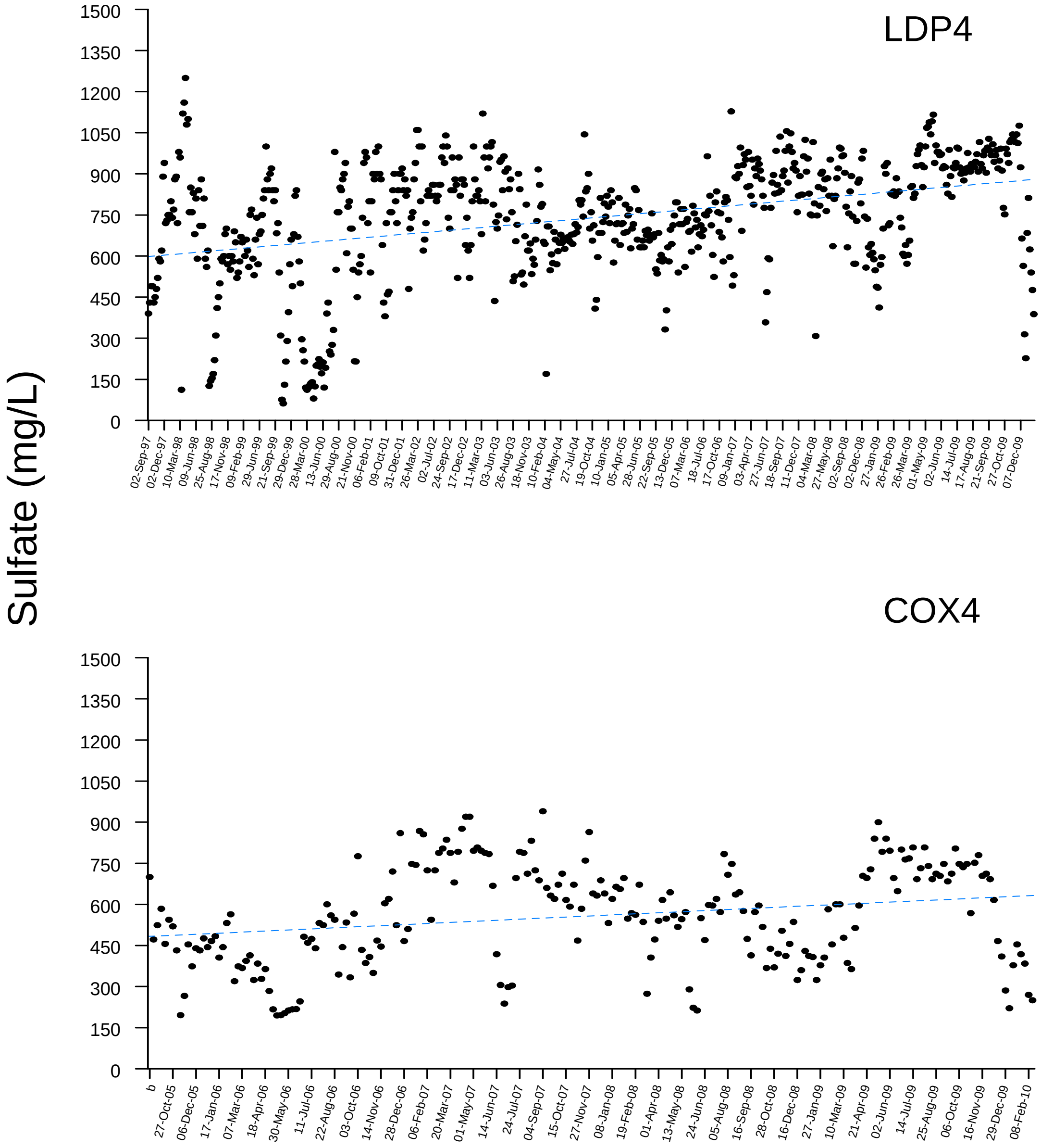


Figure 11 continued: Time series of sulfate data from the various monitoring and discharge points on and adjacent to the Coxs River in the vicinity of Wallerawang Power Station. Blue line = trend.

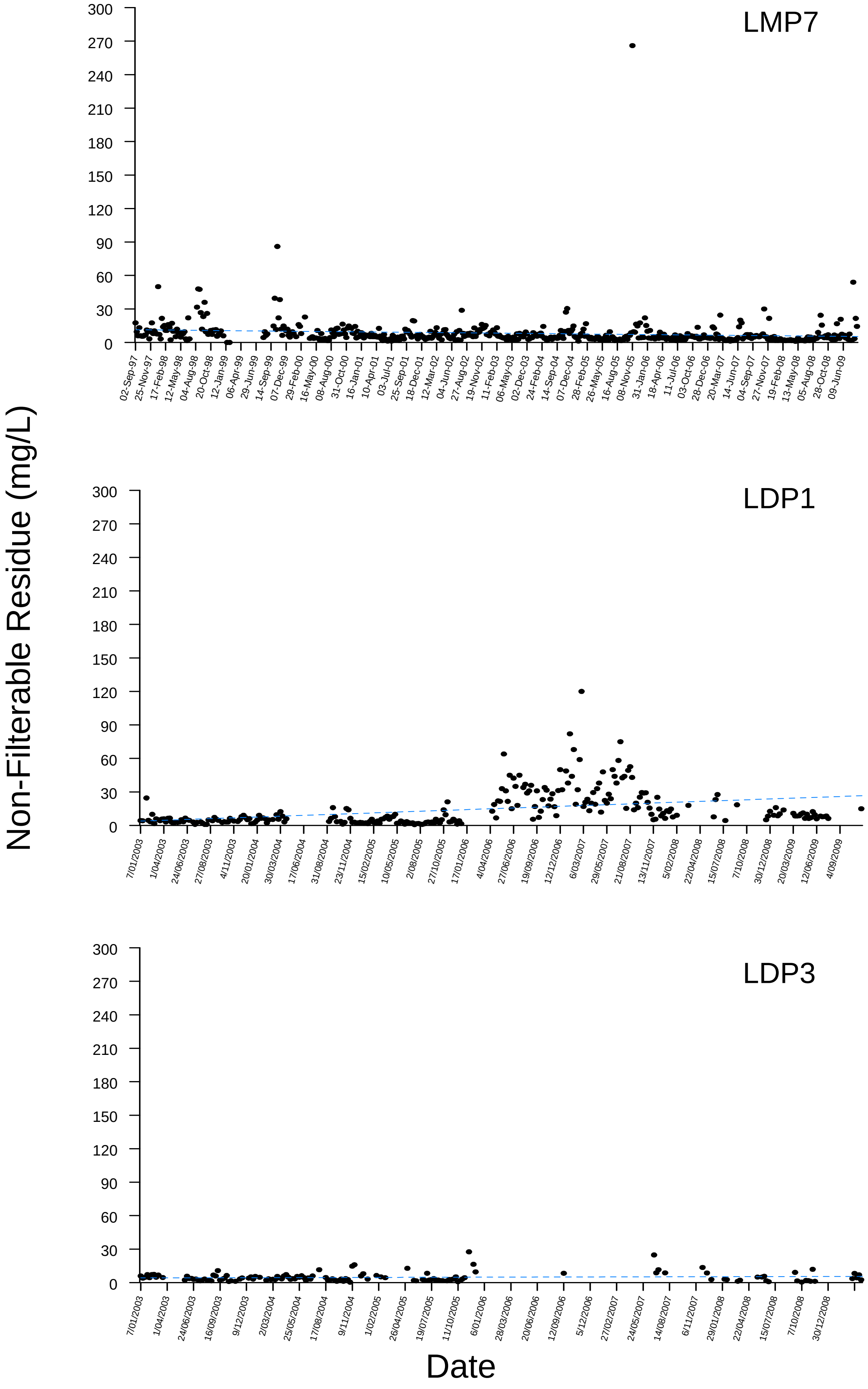


Figure 12: Time series of non-filterable residue data from the various monitoring and discharge points on and adjacent to the Coxs River in the vicinity of Wallerawang Power Station. Blue line = trend.

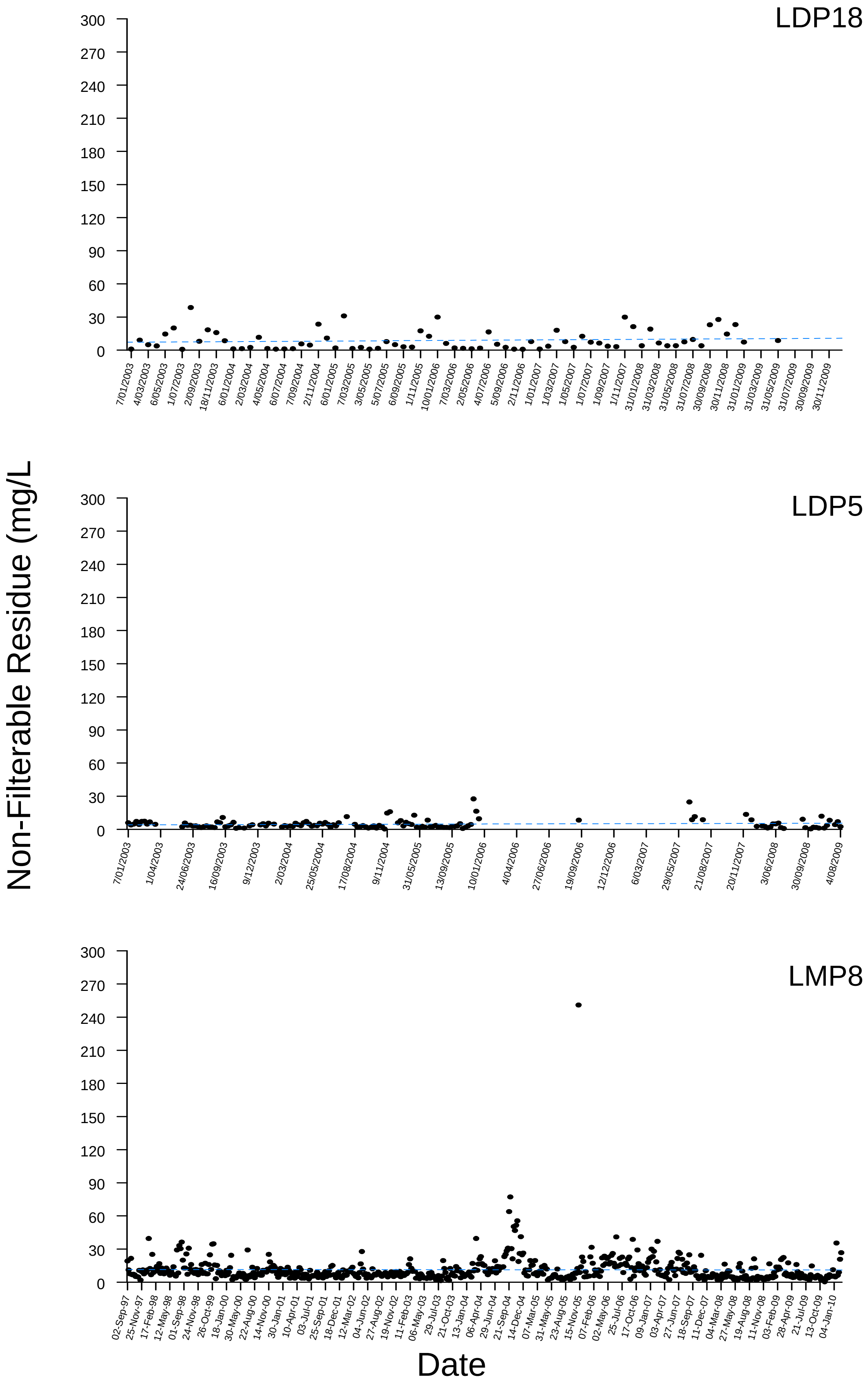


Figure 12 continued: Time series of non-filterable residue data from the various monitoring and discharge points on and adjacent to the Coxs River in the vicinity of Wallerawang Power Station. Blue line = trend, Red line = upper guideline trigger value.

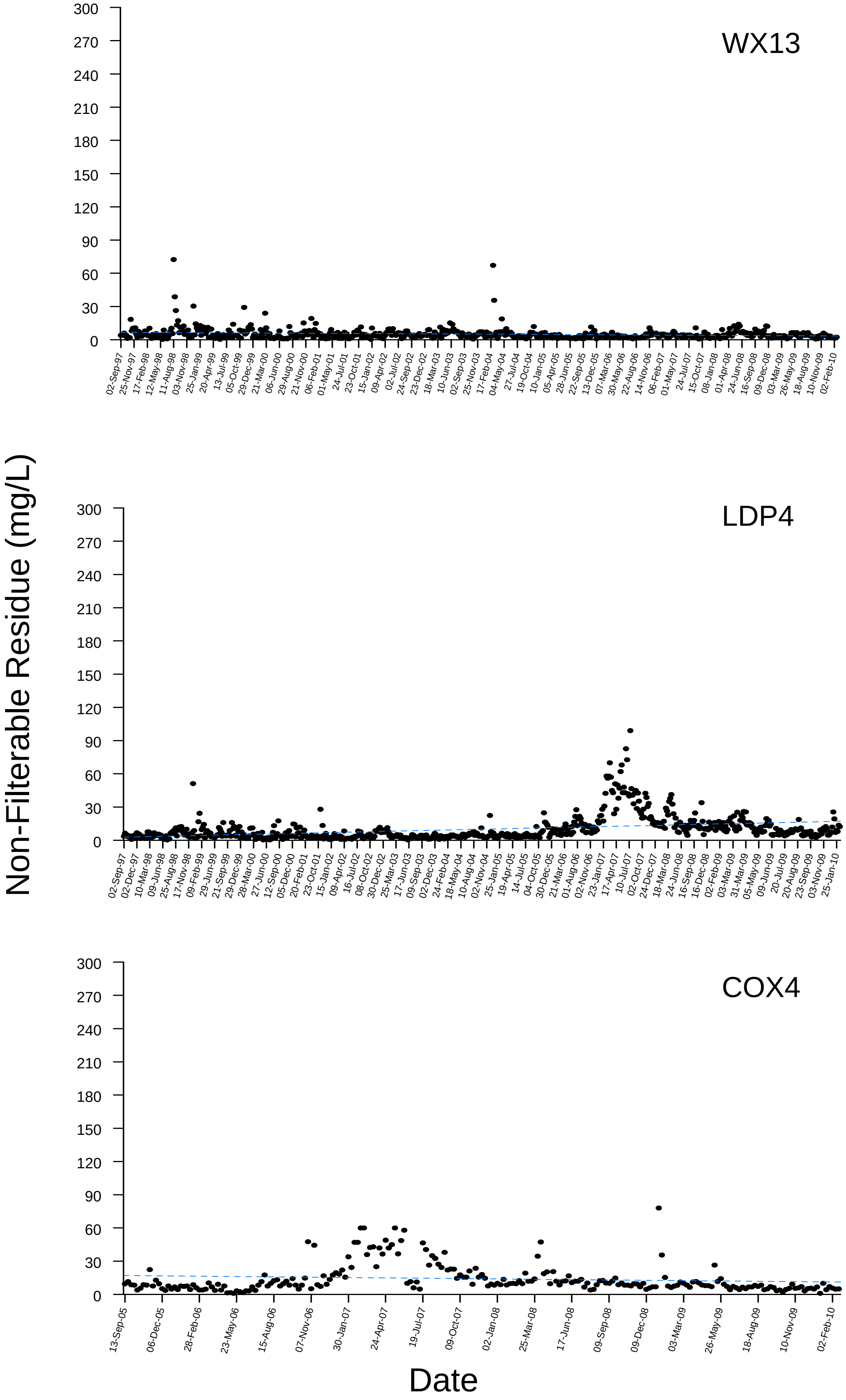


Figure 12 continued: Time series of non-filterable residue data from the various monitoring and discharge points on and adjacent to the Coxs River in the vicinity of Wallerawang Power Station. Blue line = trend, Red line = upper guideline trigger value.

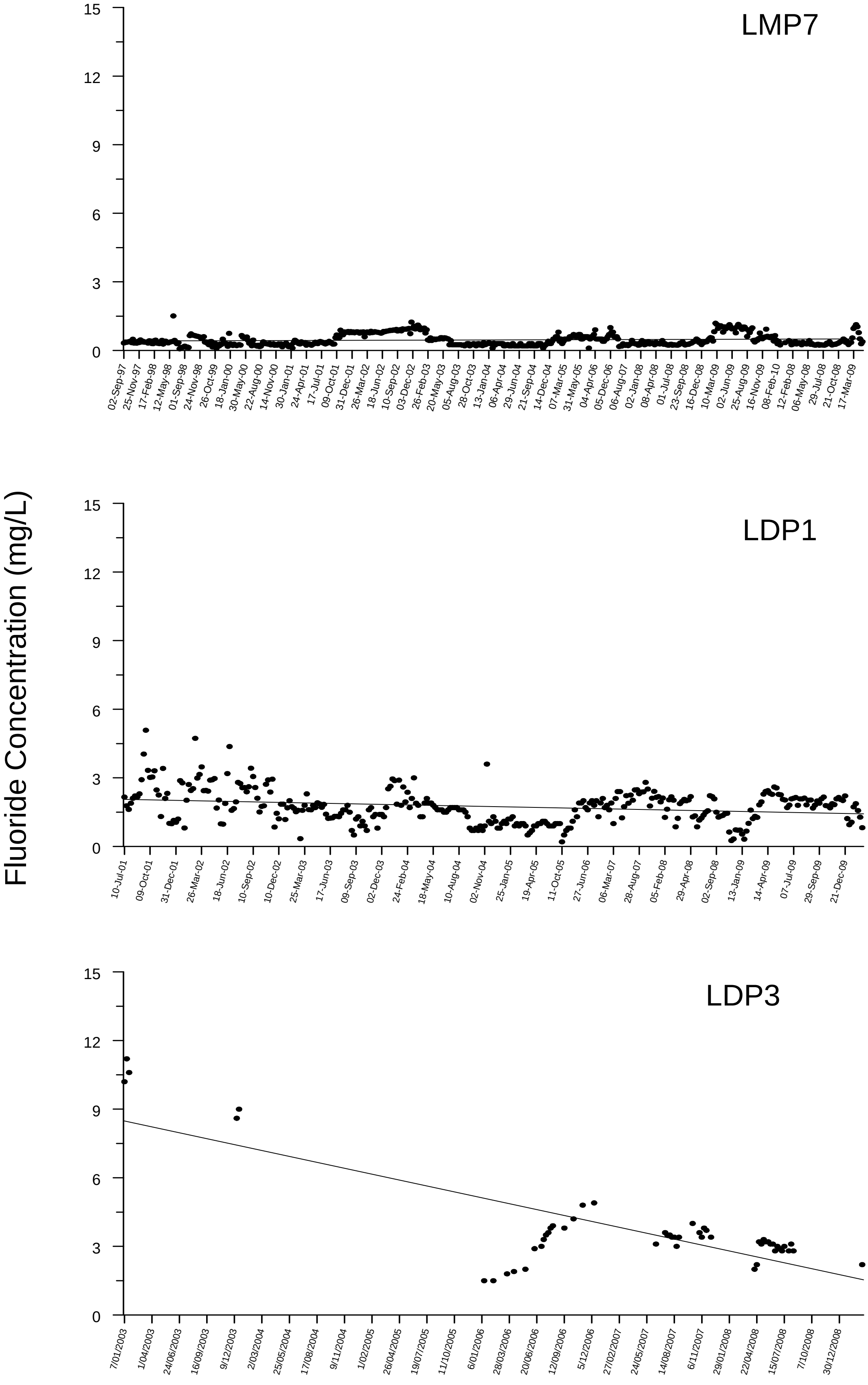


Figure 13: Time series of fluoride data from the various monitoring and discharge points on and adjacent to the Coxs River in the vicinity of Wallerawang Power Station. Blue line = trend, Red line = upper guideline trigger value.

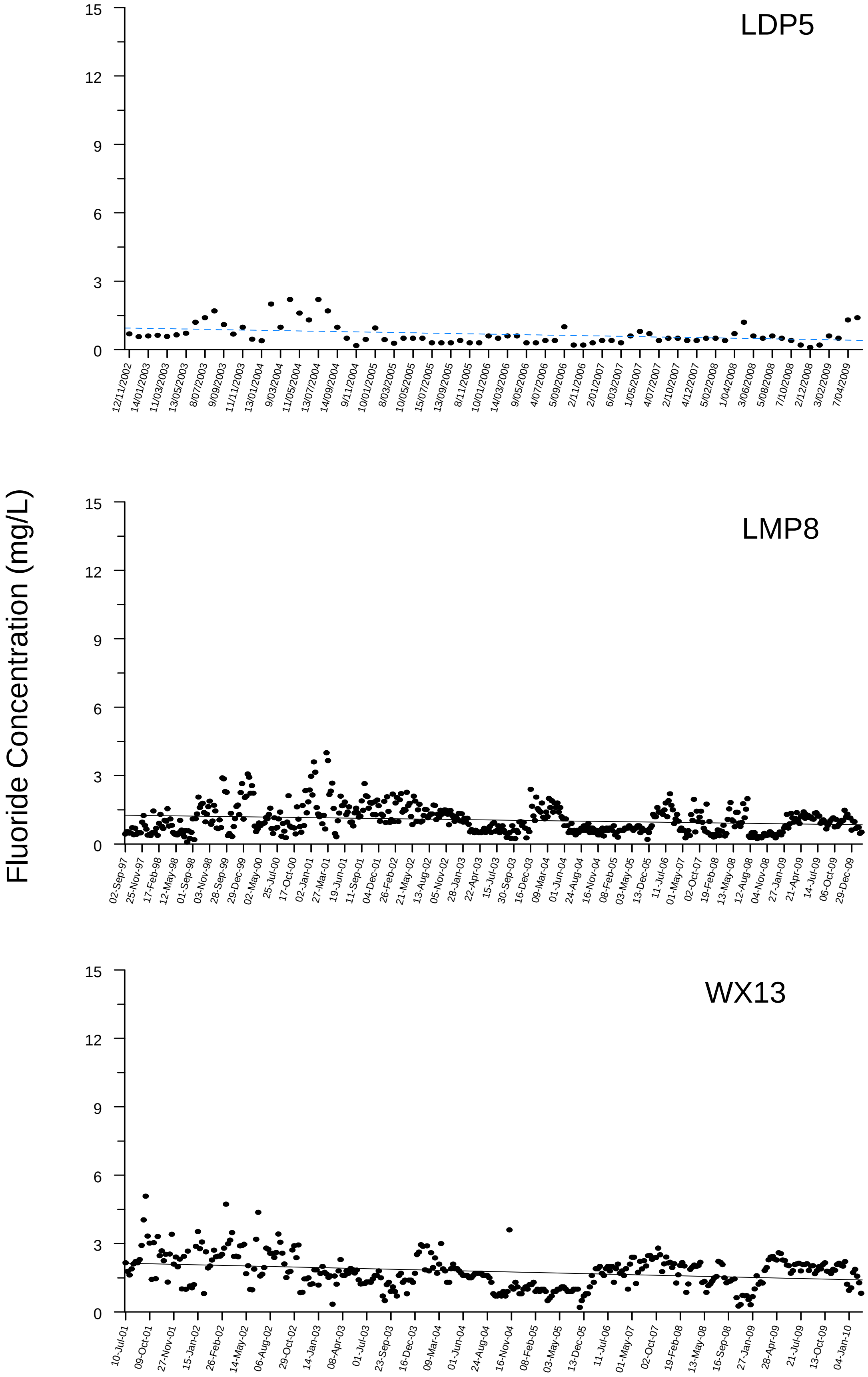


Figure 13 continued: Time series of fluoride data from the various monitoring and discharge points on and adjacent to the Coxs River in the vicinity of Wallerawang Power Station. Blue line = trend, Red line = upper guideline trigger value.

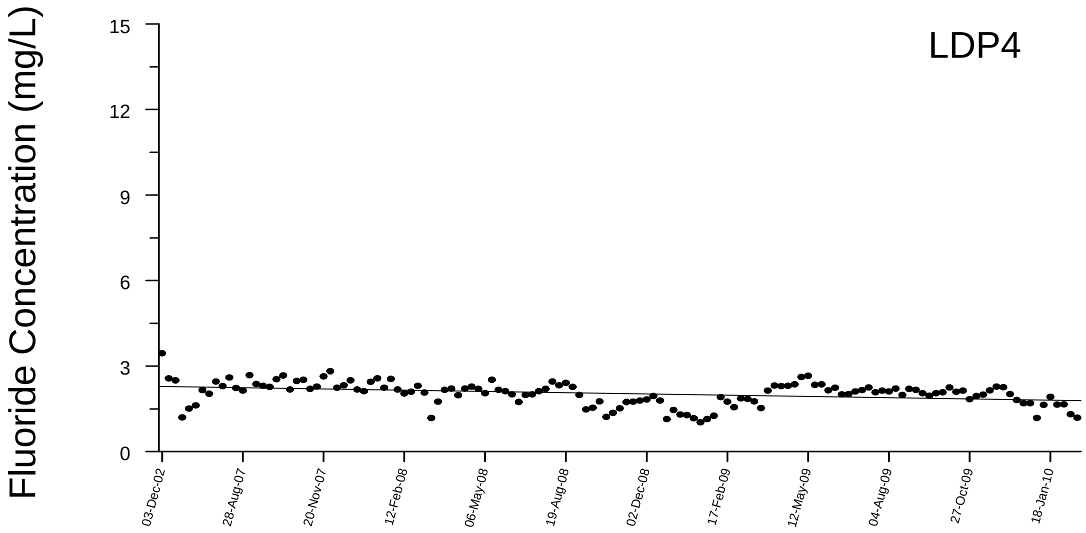


Figure 13 continued: Time series of fluoride data from the various monitoring and discharge points on and adjacent to the Coxs River in the vicinity of Wallerawang Power Station. Blue line = trend, Red line = upper guideline trigger value.

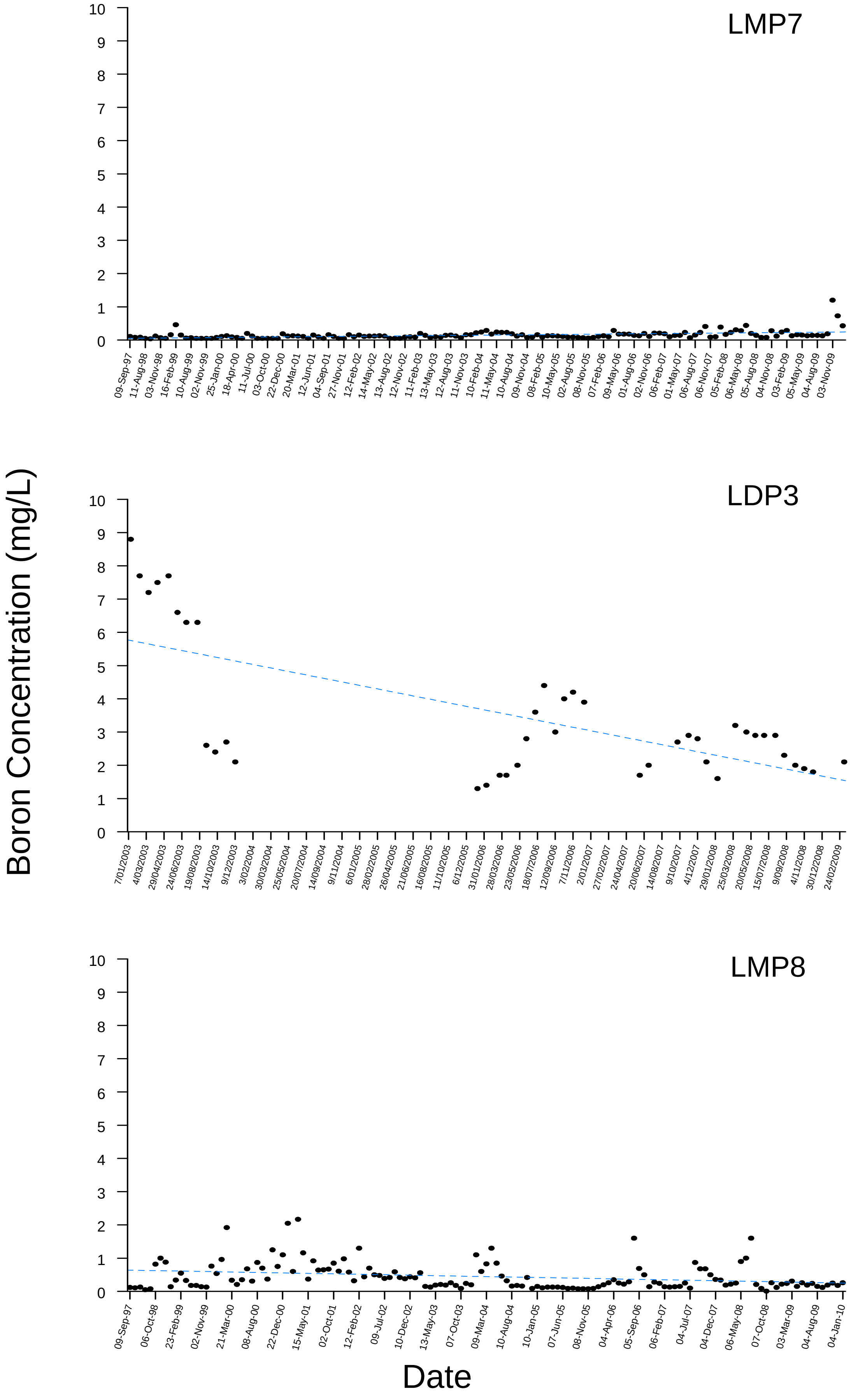


Figure 14: Time series of boron data from the various monitoring and discharge points on and adjacent to the Coxs River in the vicinity of Wallerawang Power Station. Blue line = trend line.

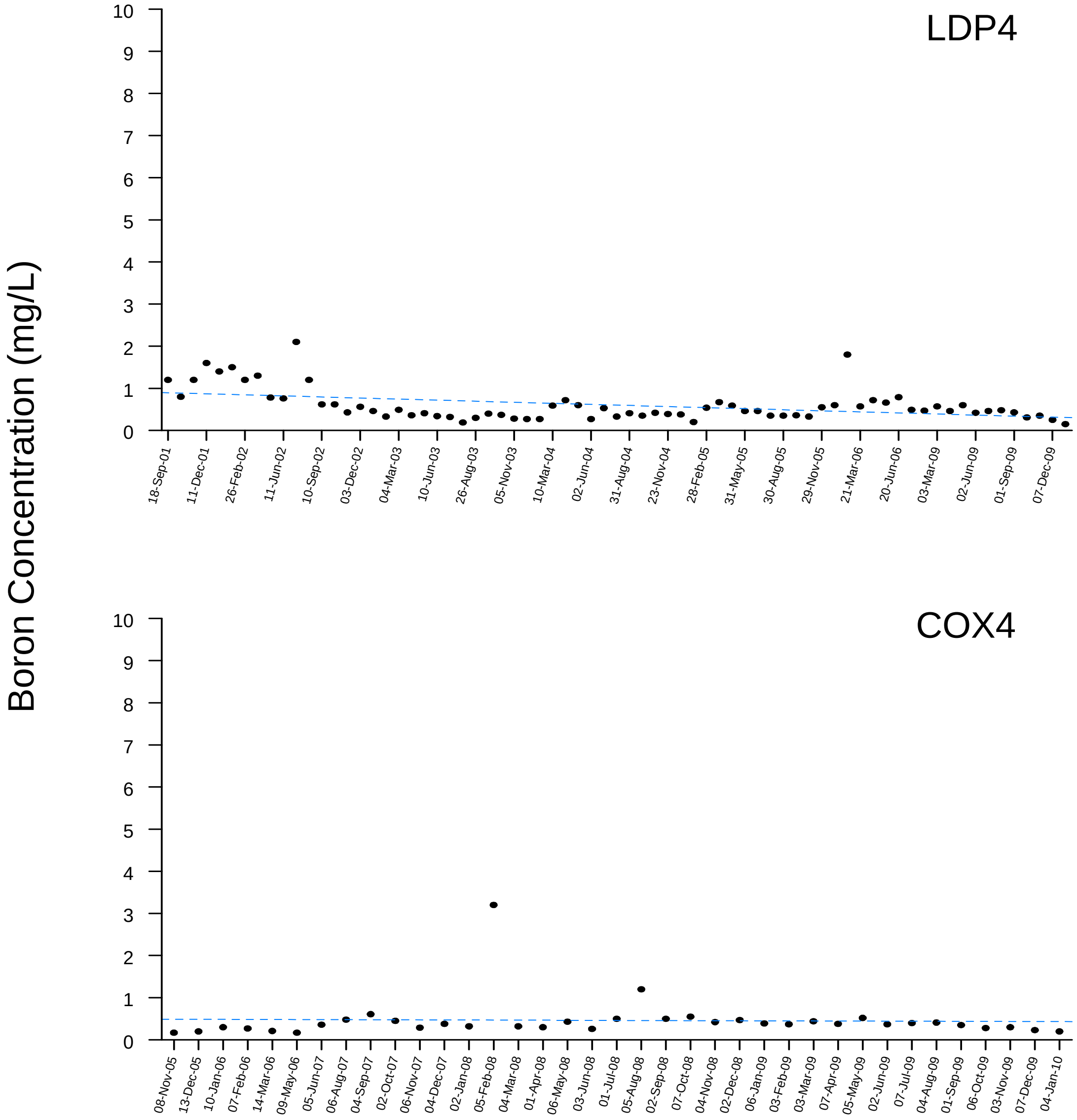


Figure 14 continued: Time series of boron data from the various monitoring and discharge points on and adjacent to the Coxs River in the vicinity of Wallerawang Power Station. Blue line = trend line.

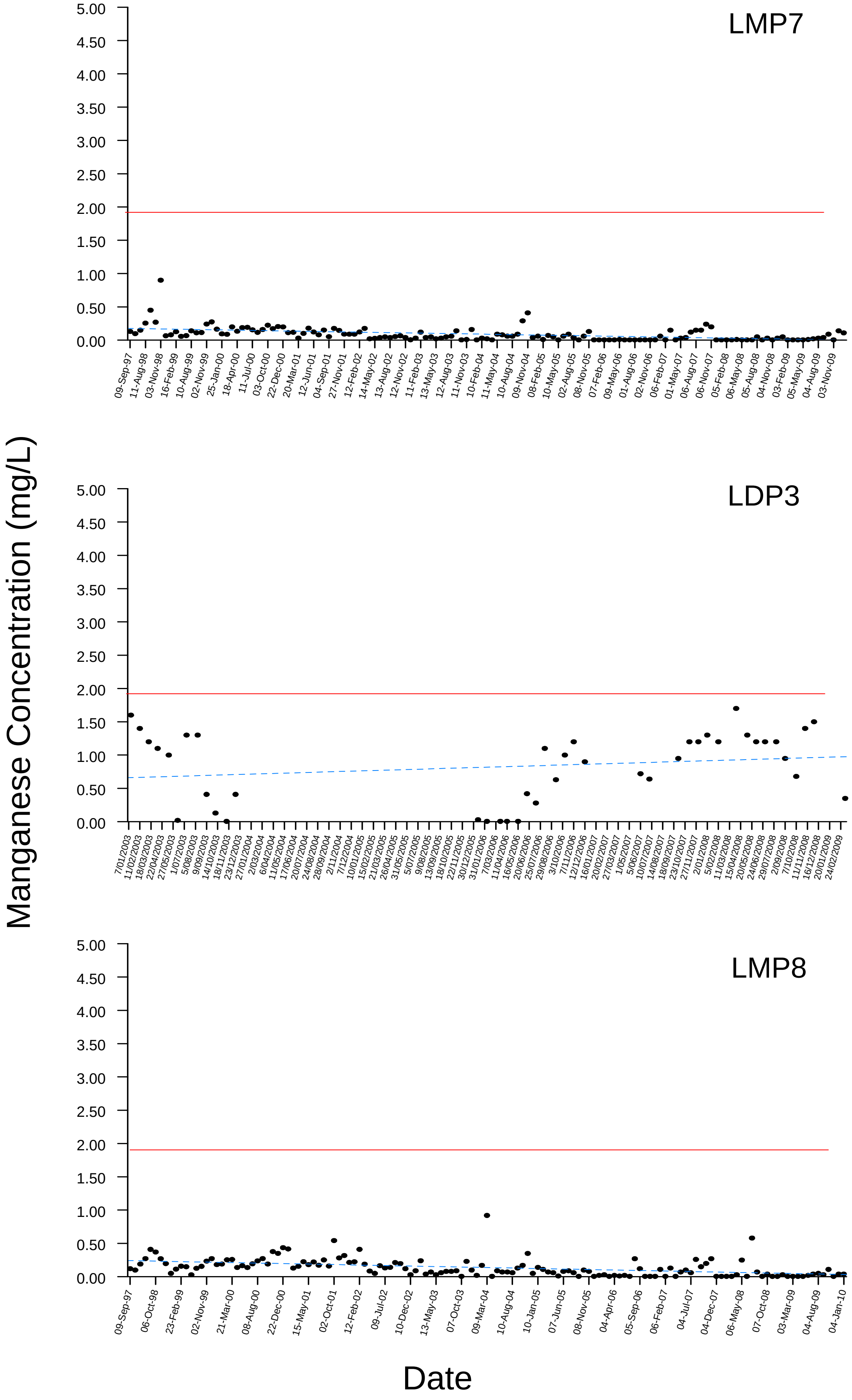


Figure 15: Time series of manganese data from the various monitoring and discharge points on and adjacent to the Coxs River in the vicinity of Wallerawang Power Station. Blue line = trend, Red line = 95% protection guideline trigger value

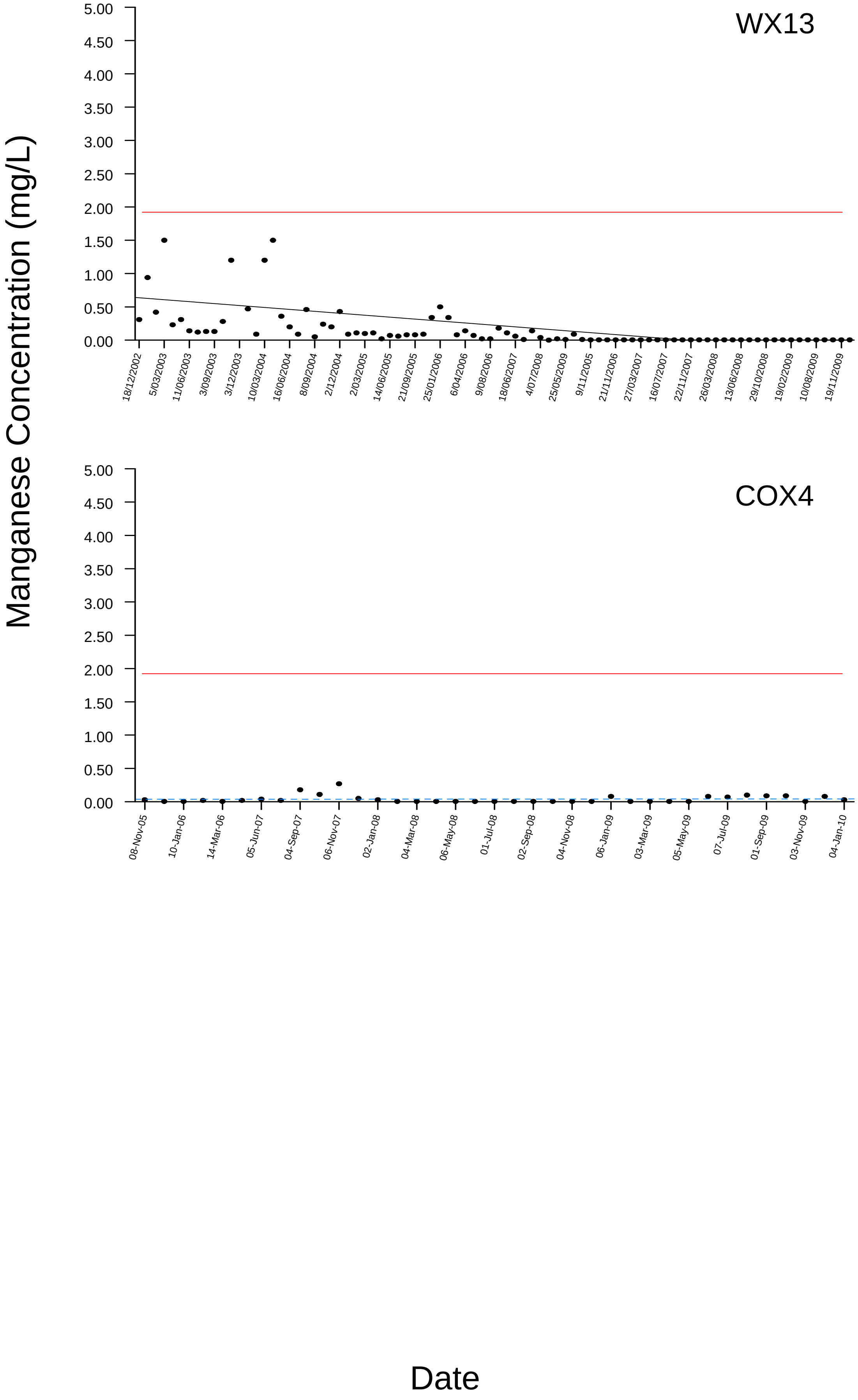


Figure 15: Time series of manganese data from the various monitoring and discharge points on and adjacent to the Coxs River in the vicinity of Wallerawang Power Station. Blue line = trend, Red line = 95% protection guideline trigger value

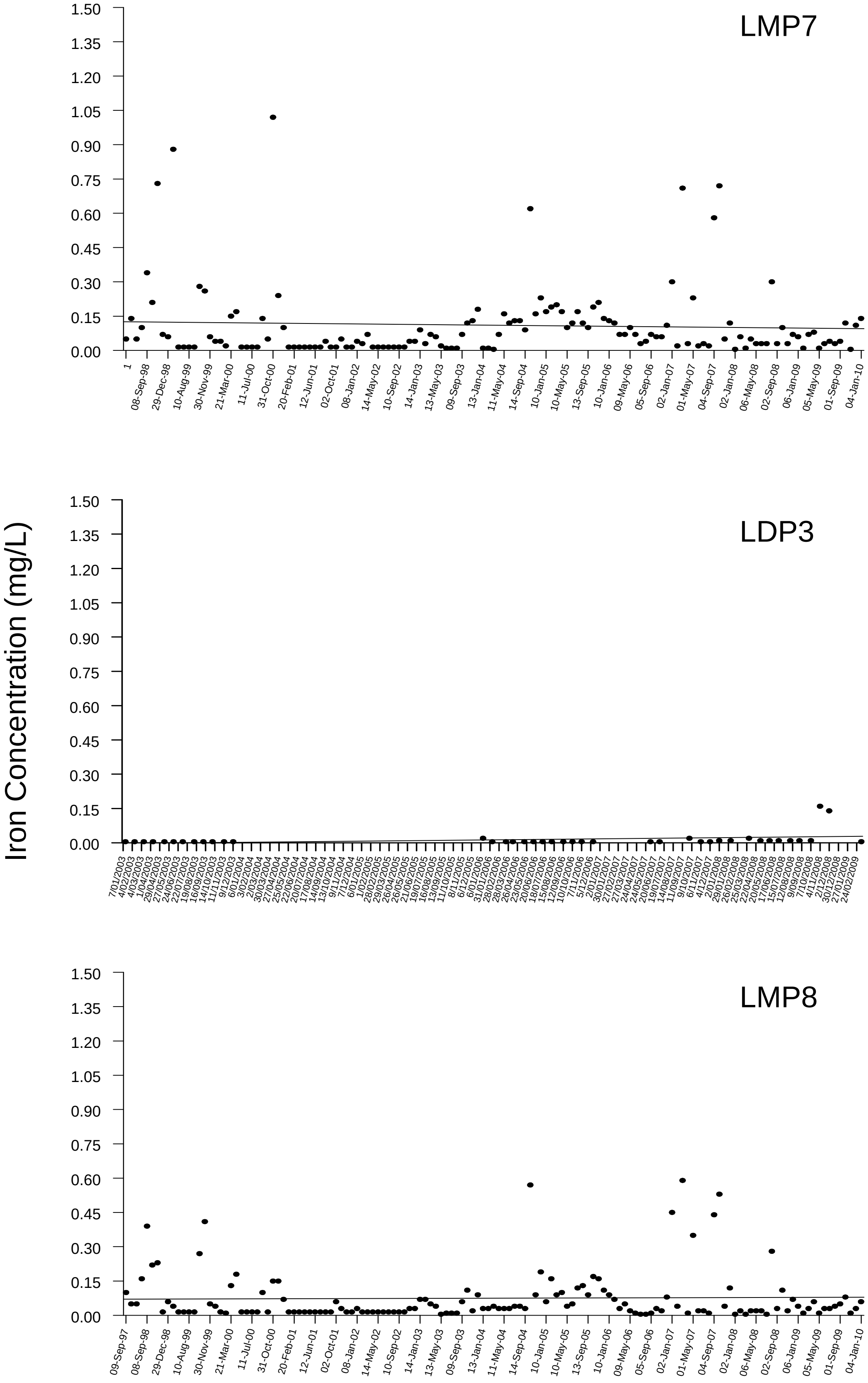


Figure 16: Time series of total iron data from the various monitoring and discharge points on and adjacent to the Coxs River in the vicinity of Wallerawang Power Station. Blue line = trend, Red line = upper guideline trigger value.

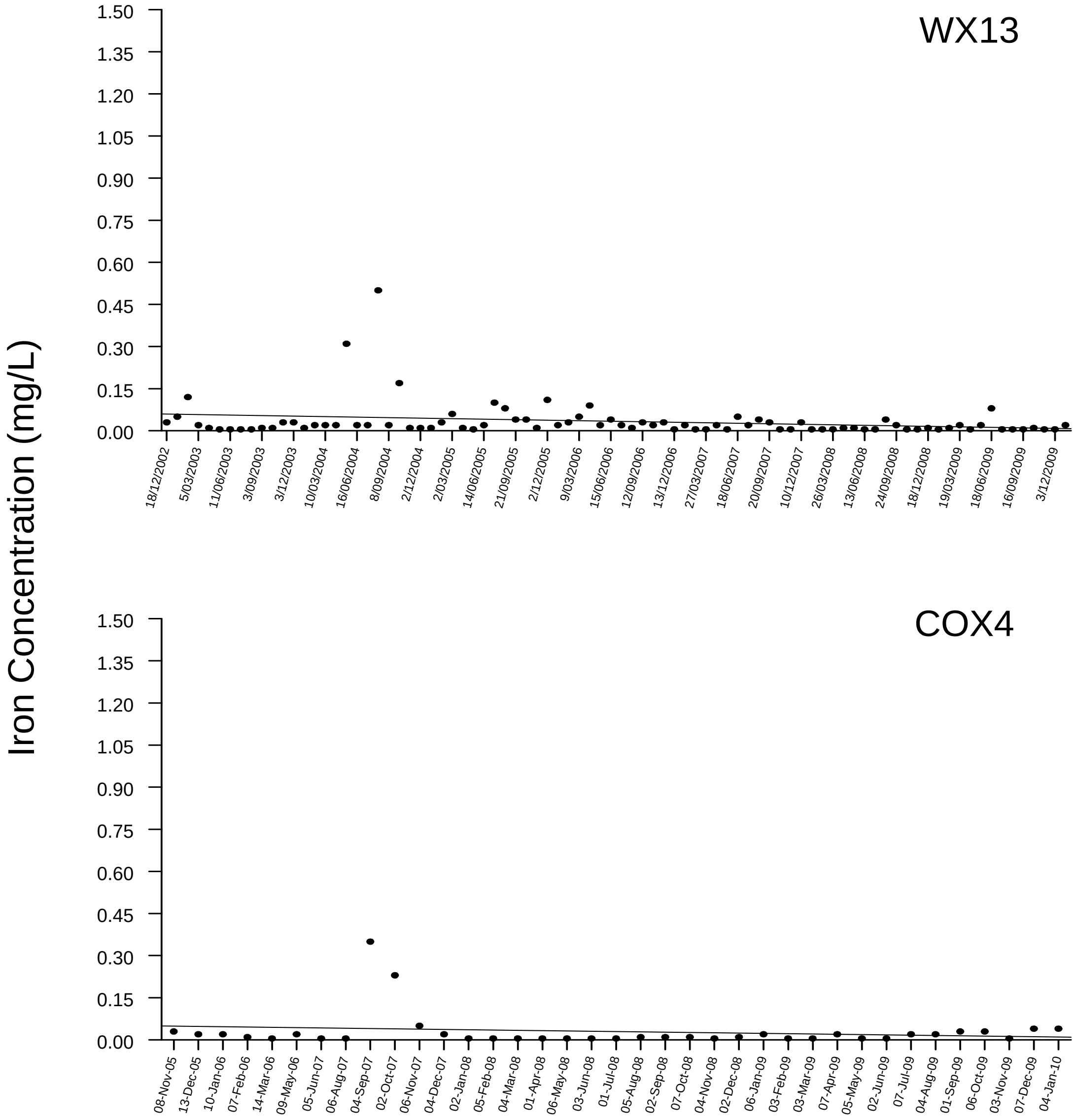


Figure 16 continued: Time series of total iron data from the various monitoring and discharge points on and adjacent to the Coxs River in the vicinity of Wallerawang Power Station. Blue line = trend, Red line = upper guideline trigger value.

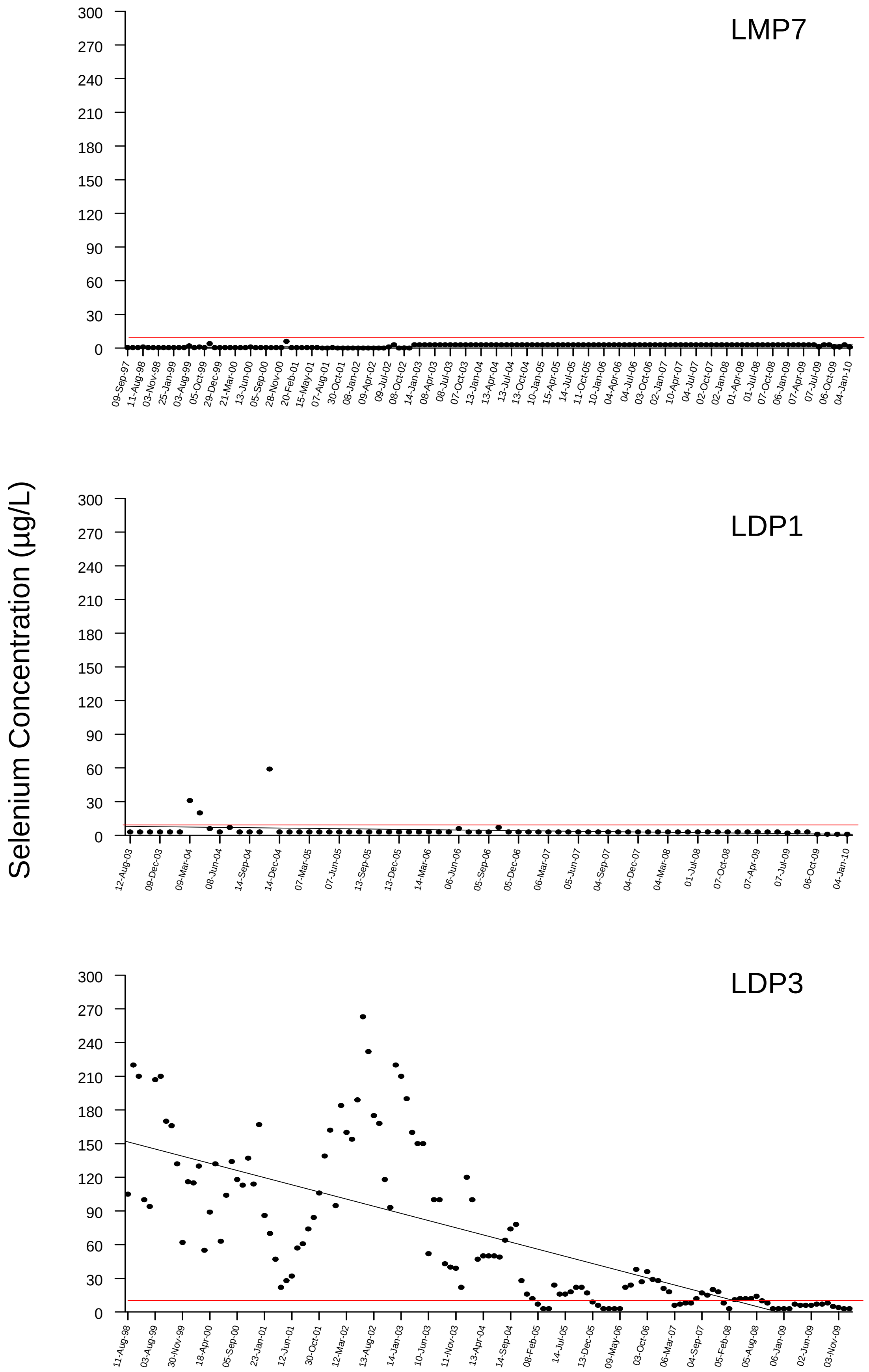


Figure 17: Time series of selenium data from the various monitoring and discharge points on and adjacent to the Coxs River in the vicinity of Wallerawang Power Station. Blue line = trend, Red line = upper guideline trigger value.

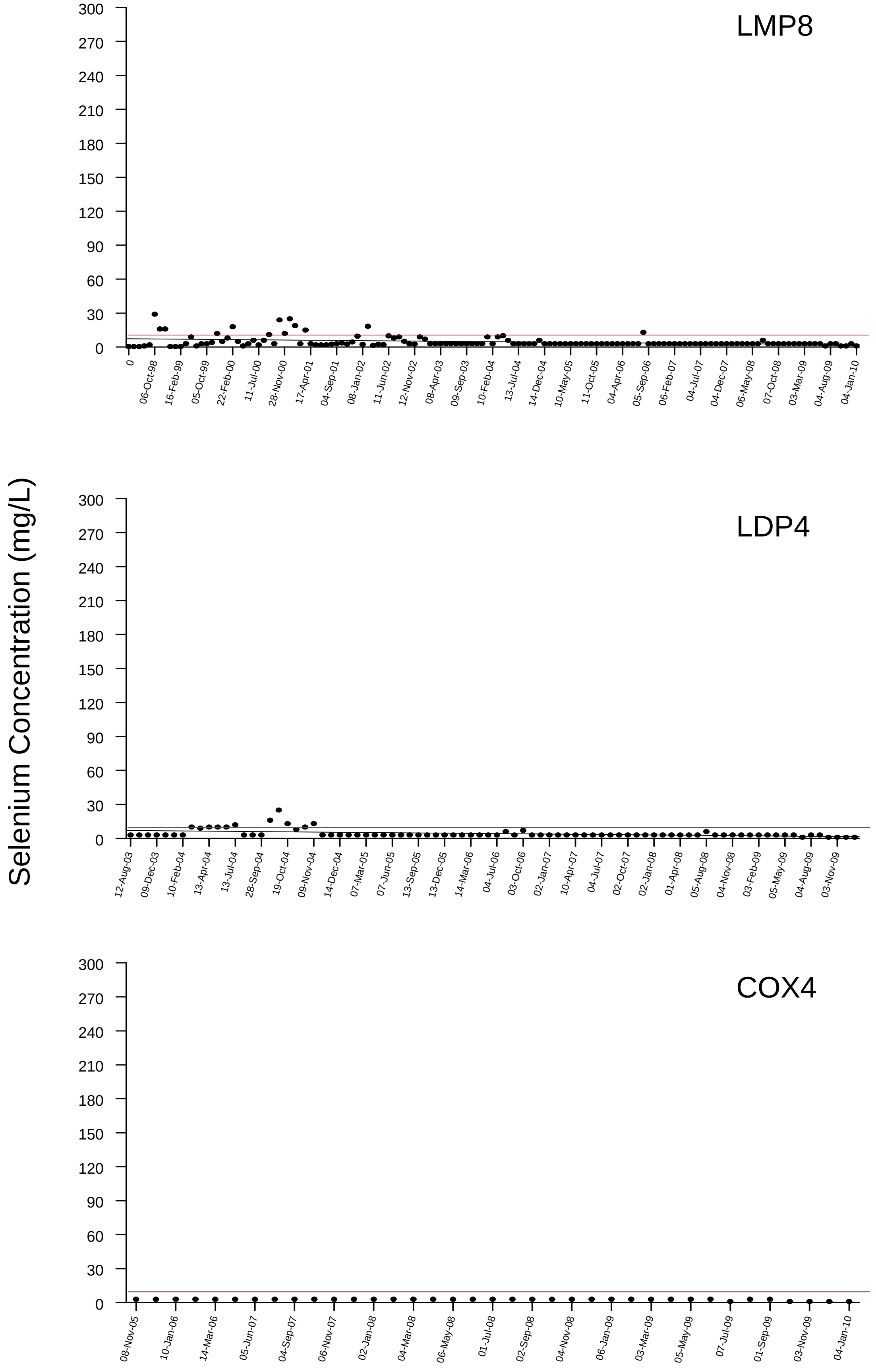


Figure 17 continued: Time series of selenium data from the various monitoring and discharge points on and adjacent to the Coxs River in the vicinity of Wallerawang Power Station. Blue line = trend, Red line = upper guideline trigger value.

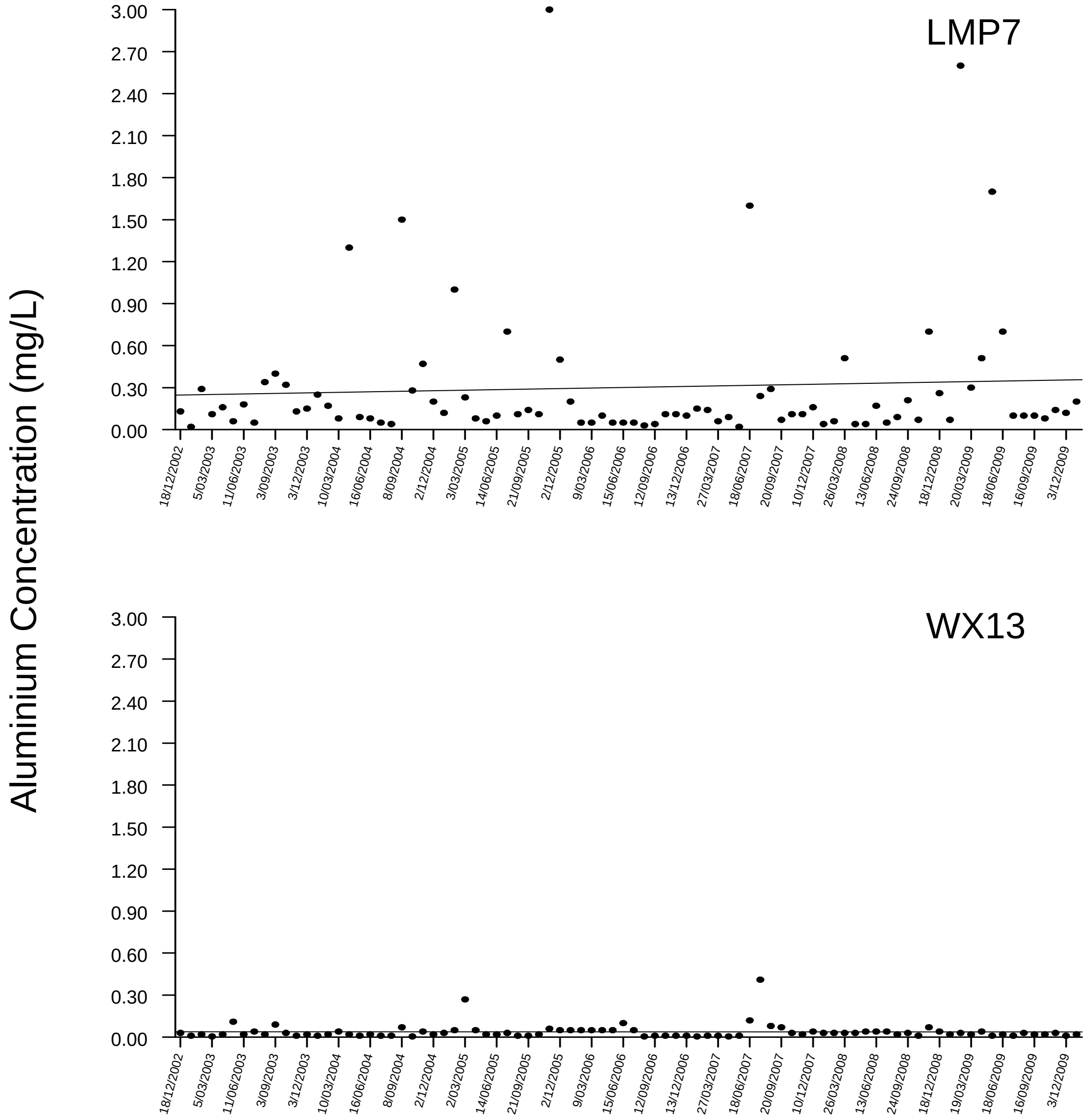


Figure 18: Time series of aluminium data from two monitoring locations on the Coxs River in the vicinity of Wallerawang Power Station. Blue line = trend, Red line = upper guideline trigger value.

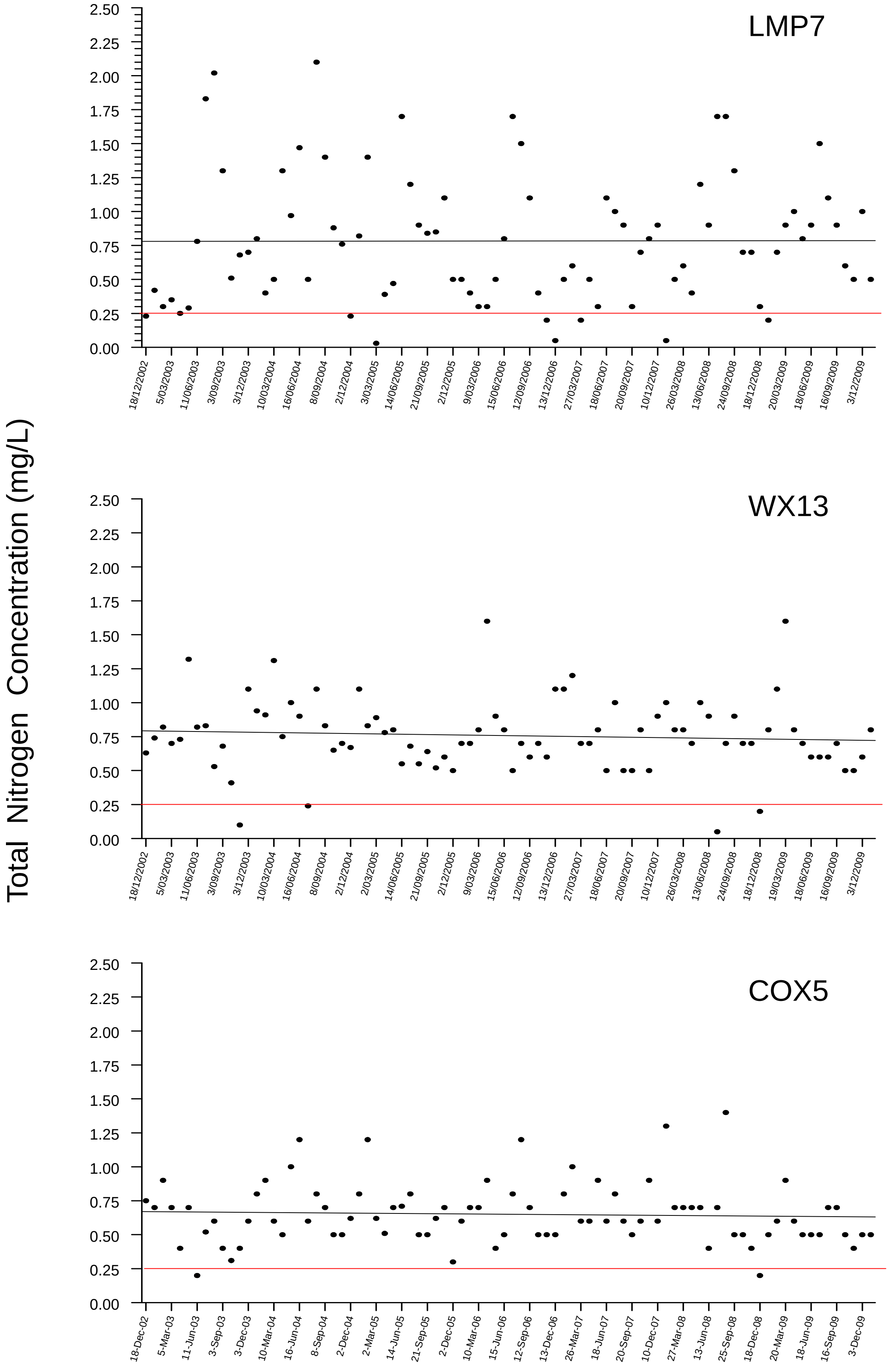


Figure 19: Time series of total nitrogen data from three monitoring points on the Coxs River in the vicinity of Wallerawang Power Station. Blue line = trend, Red line = default trigger value.

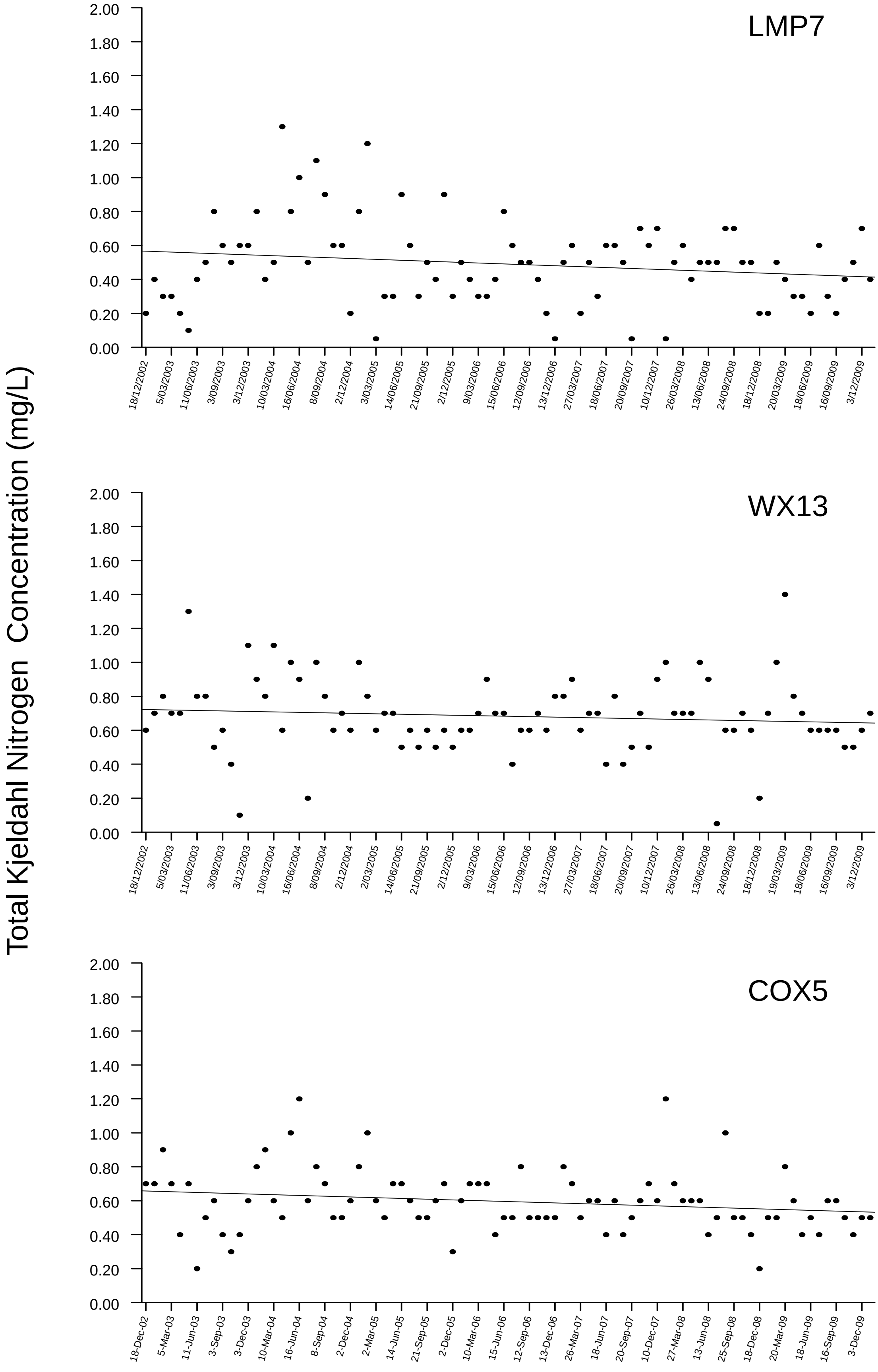


Figure 20: Time series of total Kjeldahl nitrogen data from three monitoring points on the Coxs River in the vicinity of Wallerawang Power Station. Blue line = trend, Red line = upper guideline trigger value.

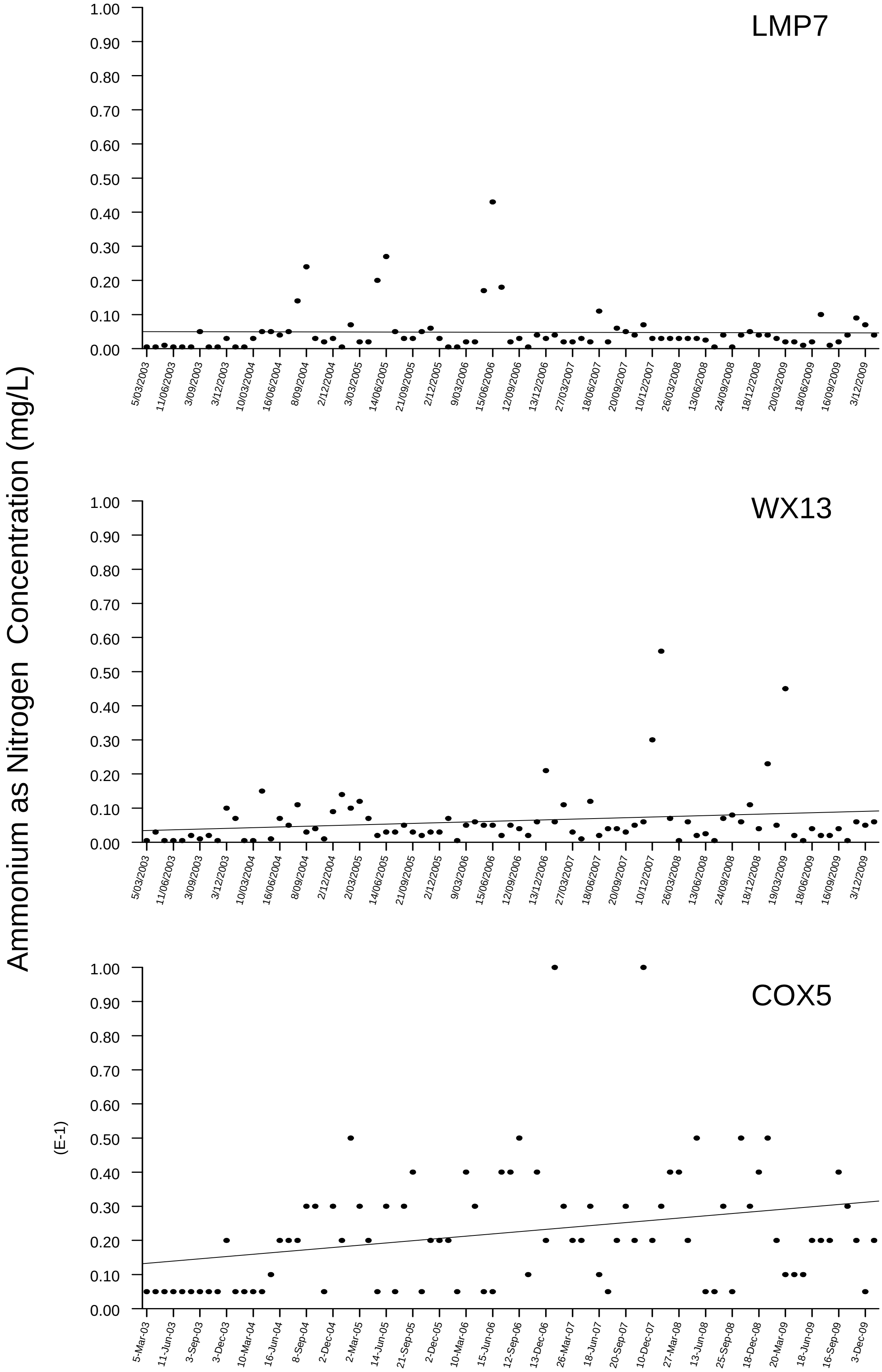


Figure 21: Time series of ammonia as nitrogen data from three monitoring points on the Coxs River in the vicinity of Wallerawang Power Station. Blue line = trend, Red line = upper guideline trigger value.

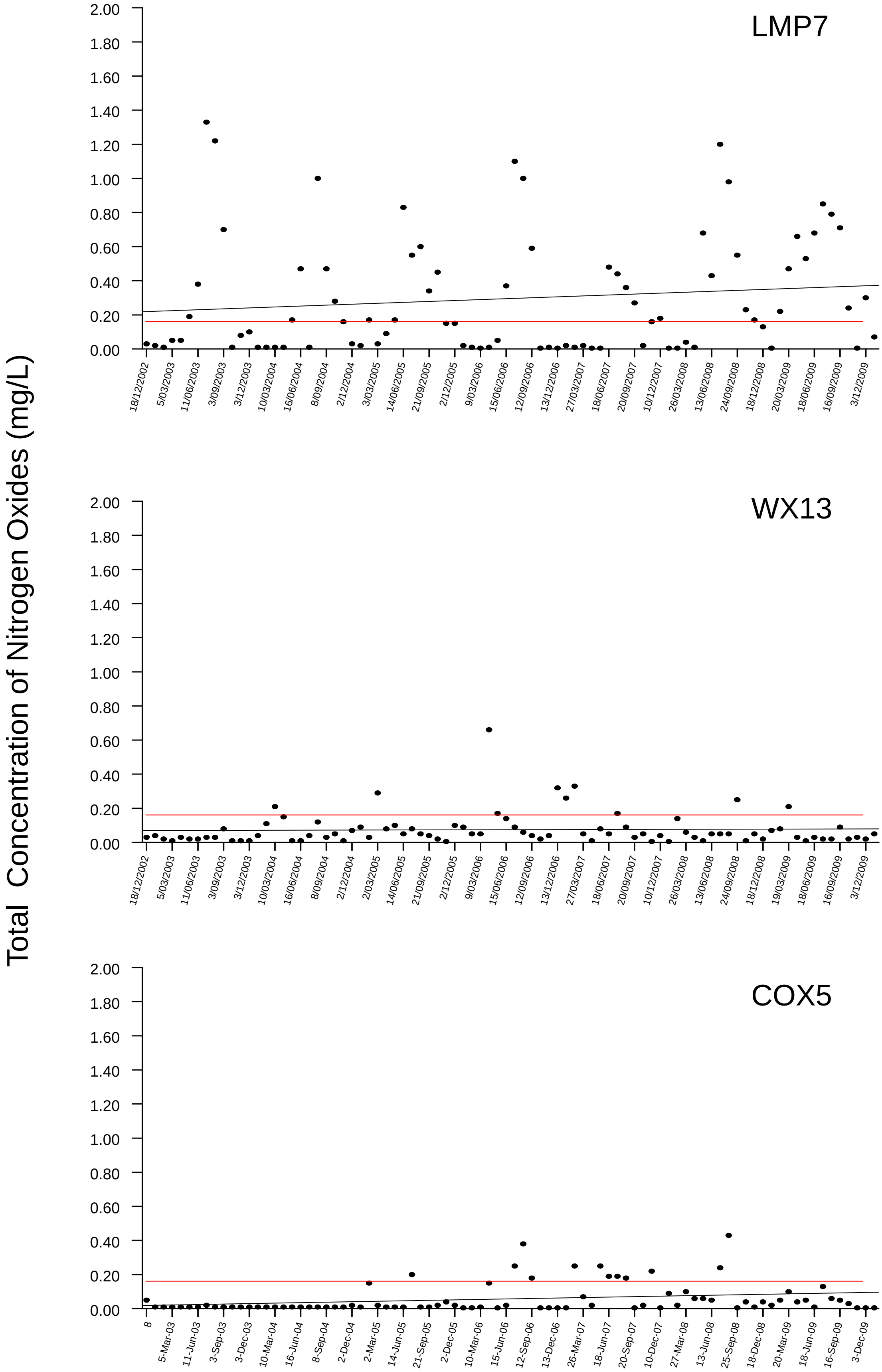


Figure 22: Time series of total nitrogen oxides data from three monitoring points on the Coxs River in the vicinity of Wallerawang Power Station. Blue line = trend, Red line = default trigger value.

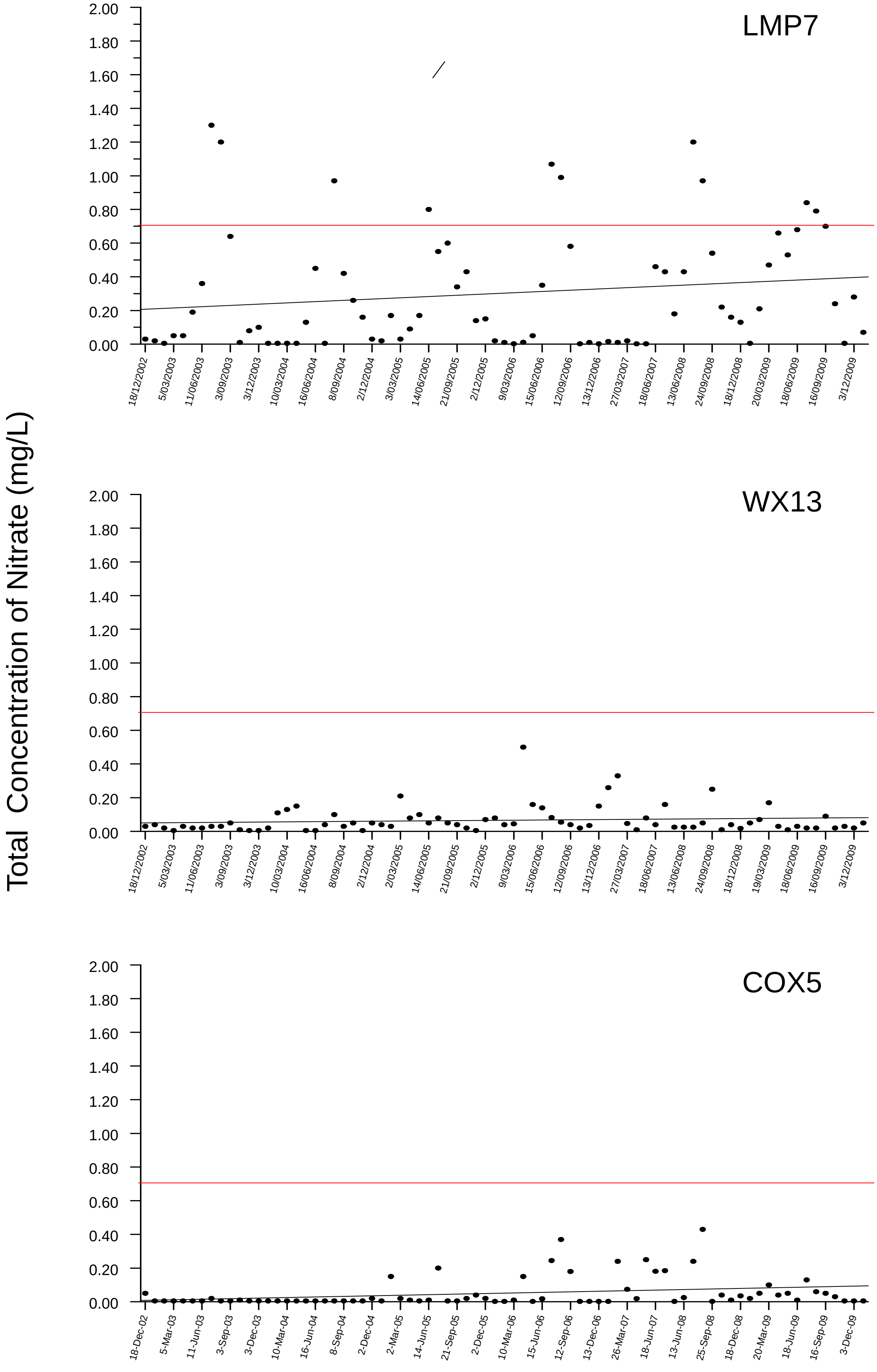


Figure 23: Time series of total nitrate data from three monitoring points on the Coxs River in the vicinity of Wallerawang Power Station. Blue line = trend, Red line = default trigger value.

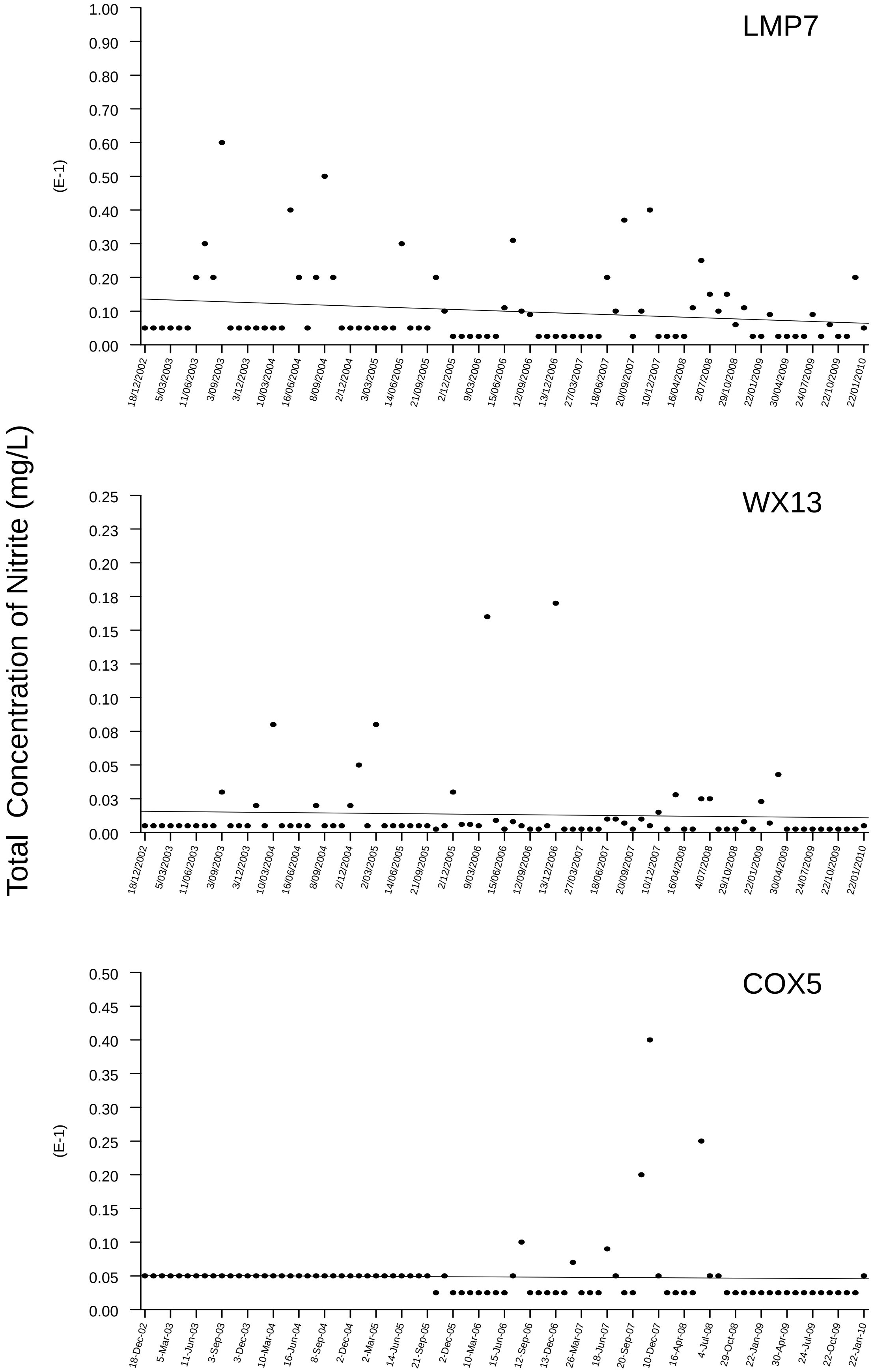


Figure 24: Time series of total nitrite data from three monitoring points on the Coxs River in the vicinity of Wallerawang Power Station. Blue line = trend, Red line = default trigger value.

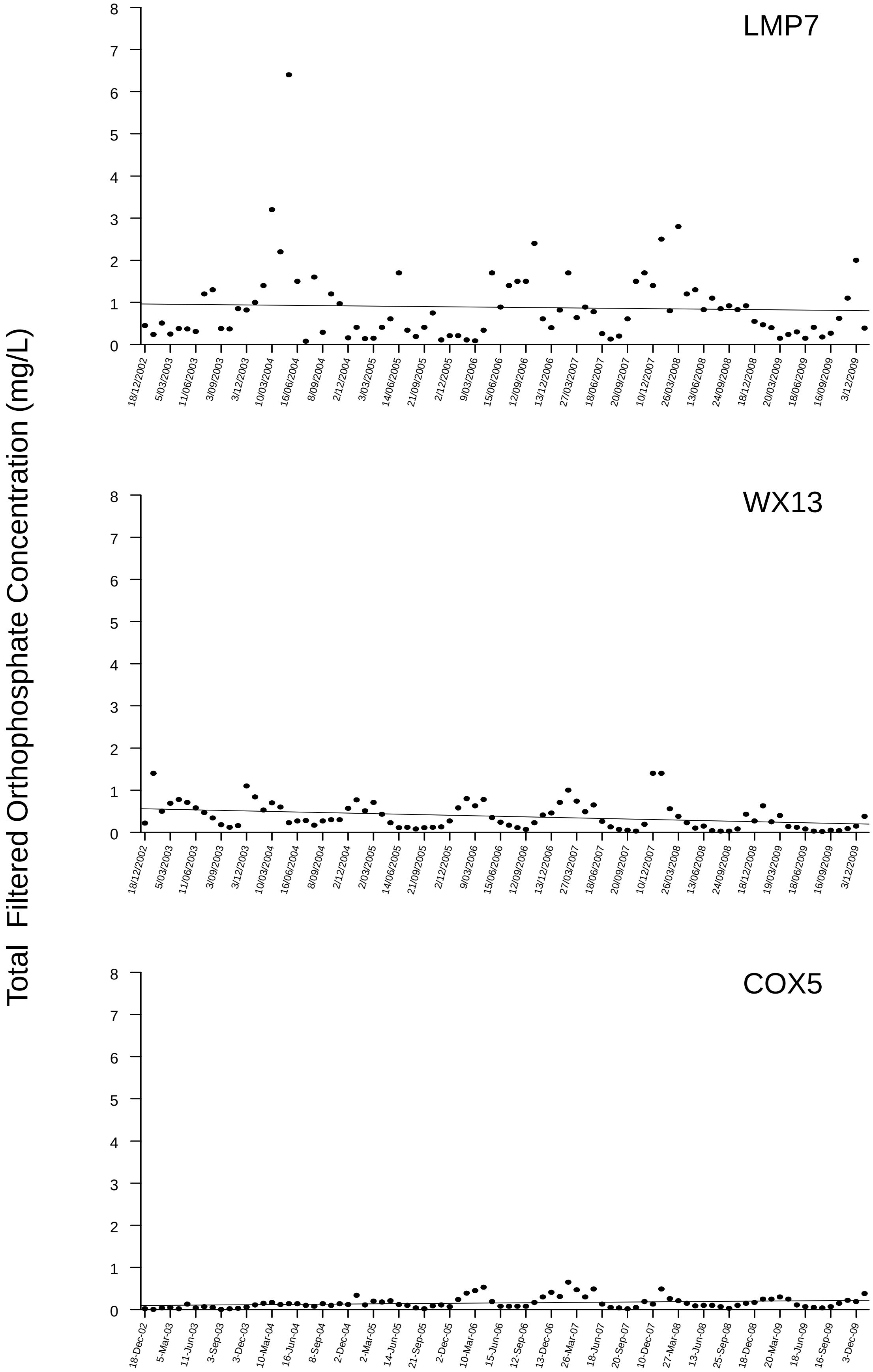


Figure 25: Time series of total filtered orthophosphate data from three monitoring points on the Coxs River in the vicinity of Wallerawang Power Station. Blue line = trend, Red line = default trigger value.

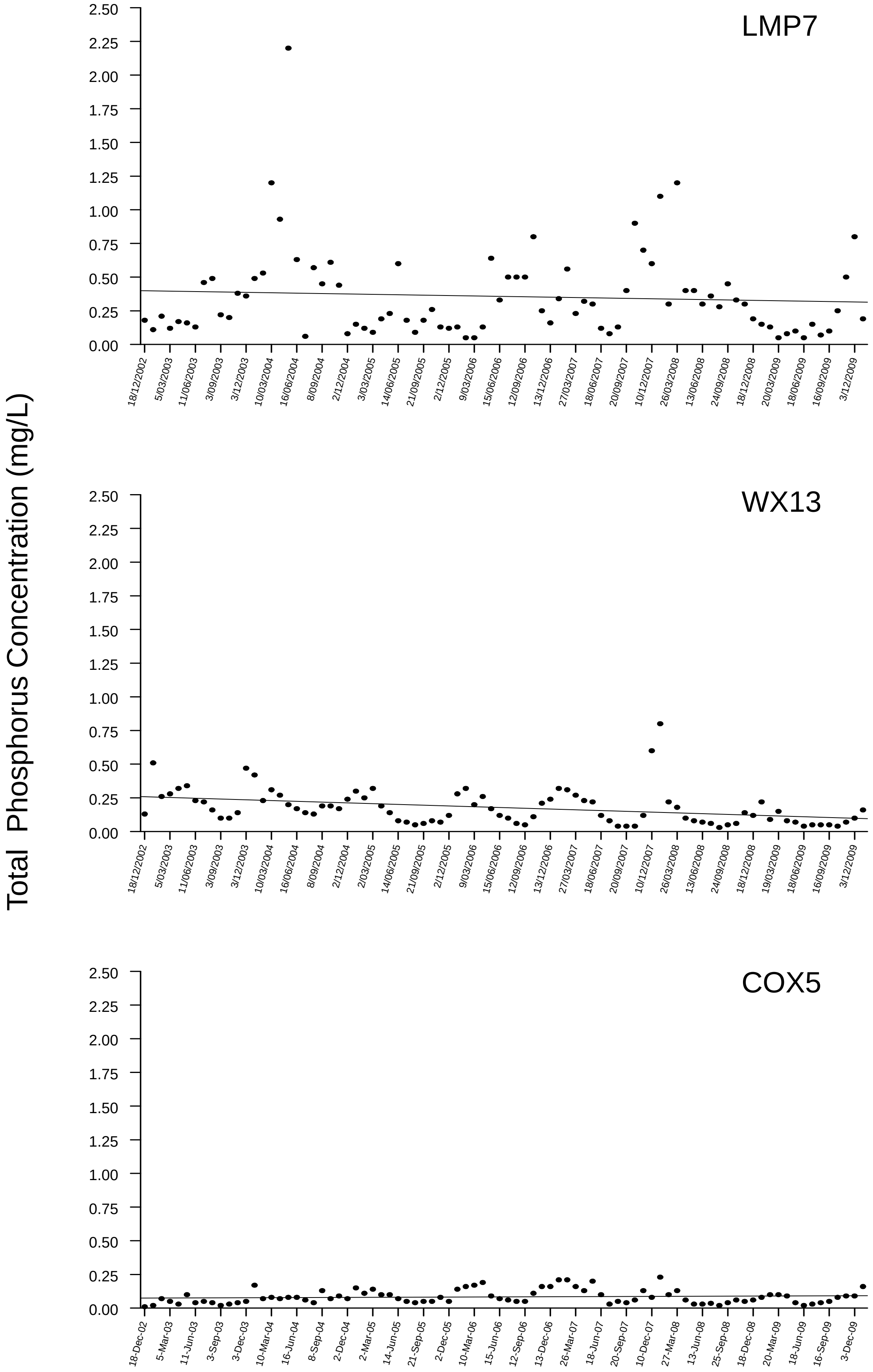


Figure 26: Time series of total phosphorus data from three monitoring points on the Coxs River in the vicinity of Wallerawang Power Station. Blue line = trend, Red line = default trigger value.

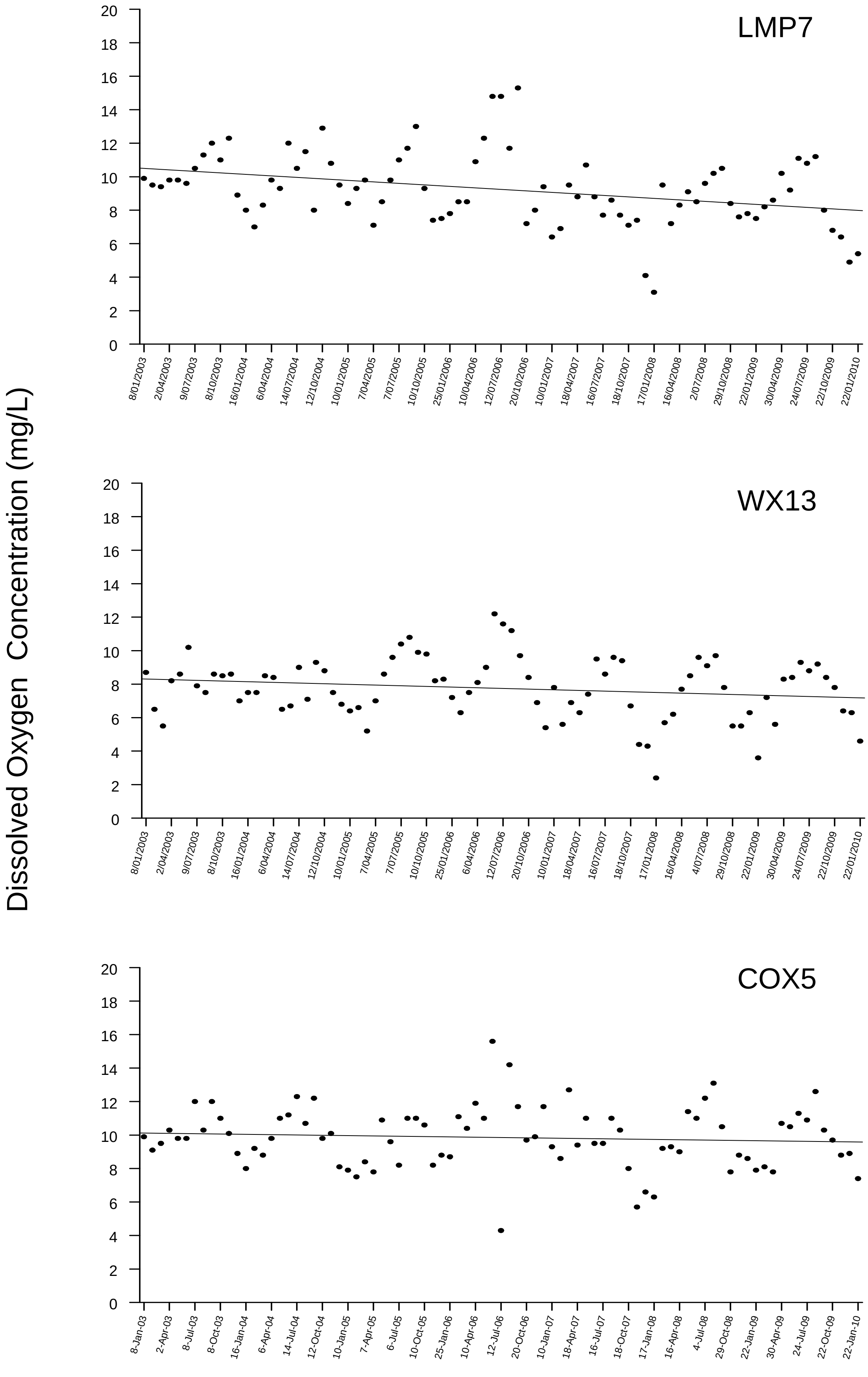


Figure 27: Time series of dissolved oxygen concentration data from three monitoring points on the Coxs River in the vicinity of Wallerawang Power Station. Blue line = trend, Red line = upper guideline trigger value.

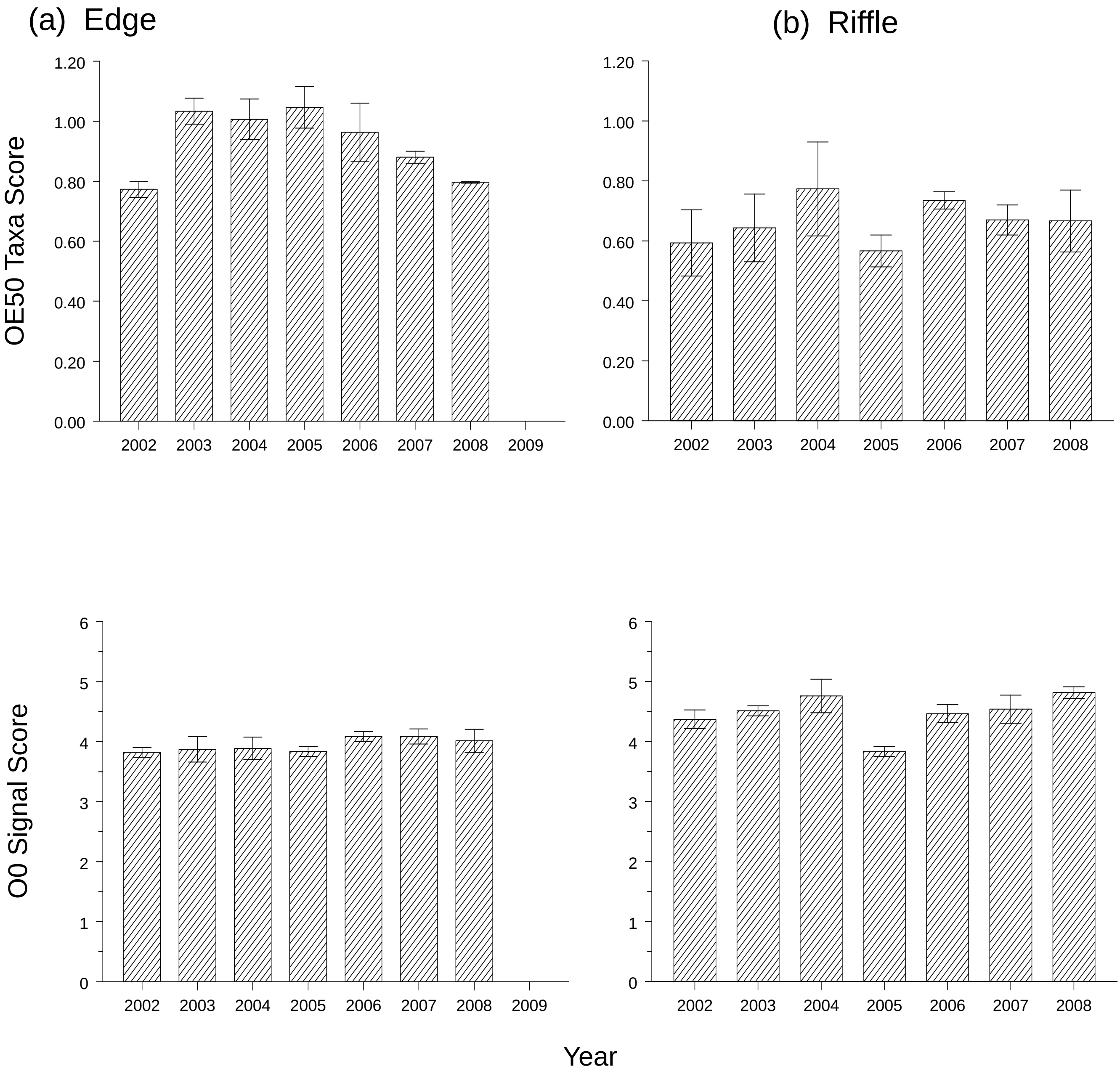


Figure 28: Mean (± S.E.) OE50 taxa scores and O0 Signal scores for macroinvertebrate assemblages associated with (a) edge and (b) riffle habitat at CR1 in spring 2002-2008 (*n* = 3). No data are available for 2005.

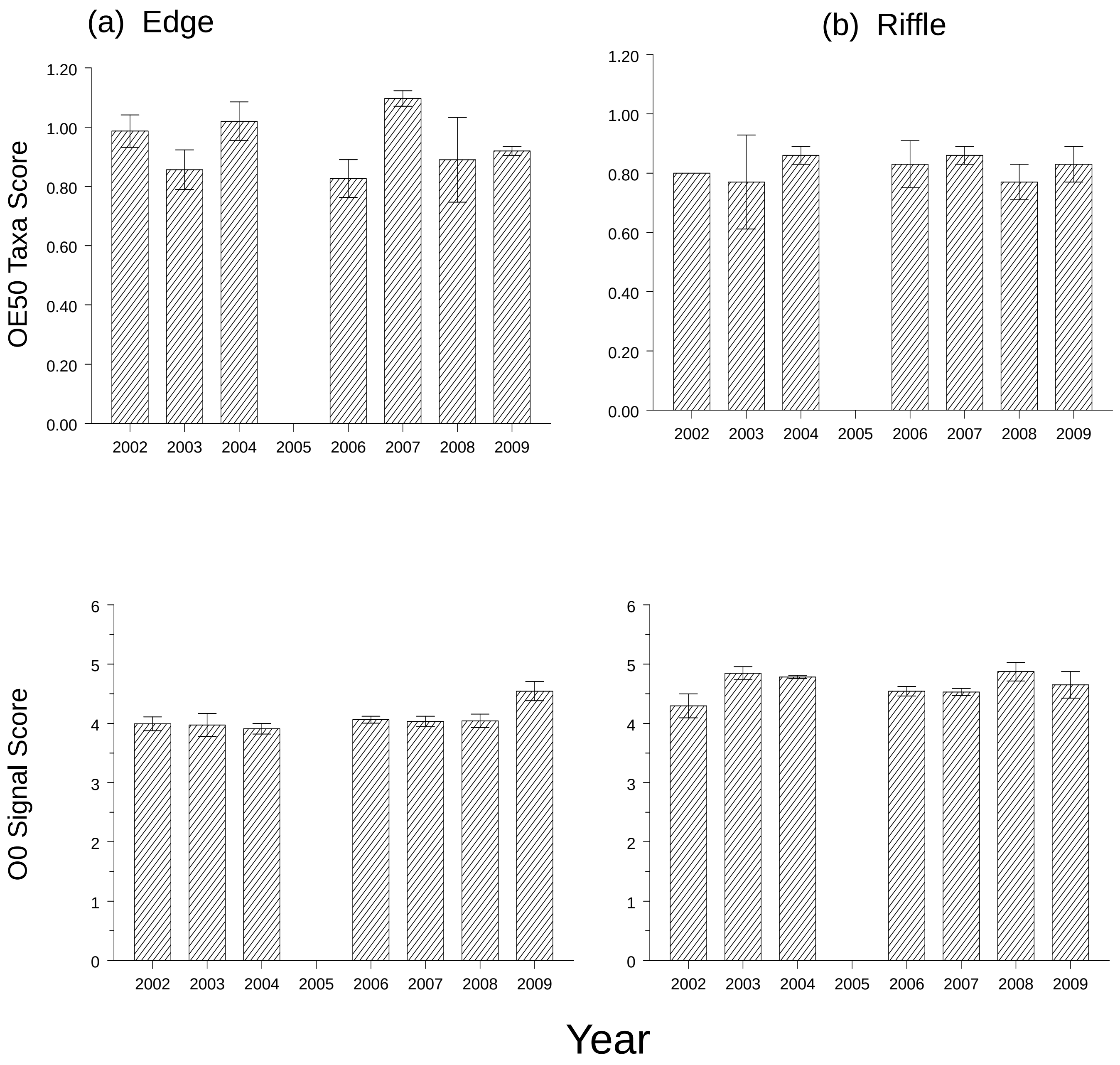
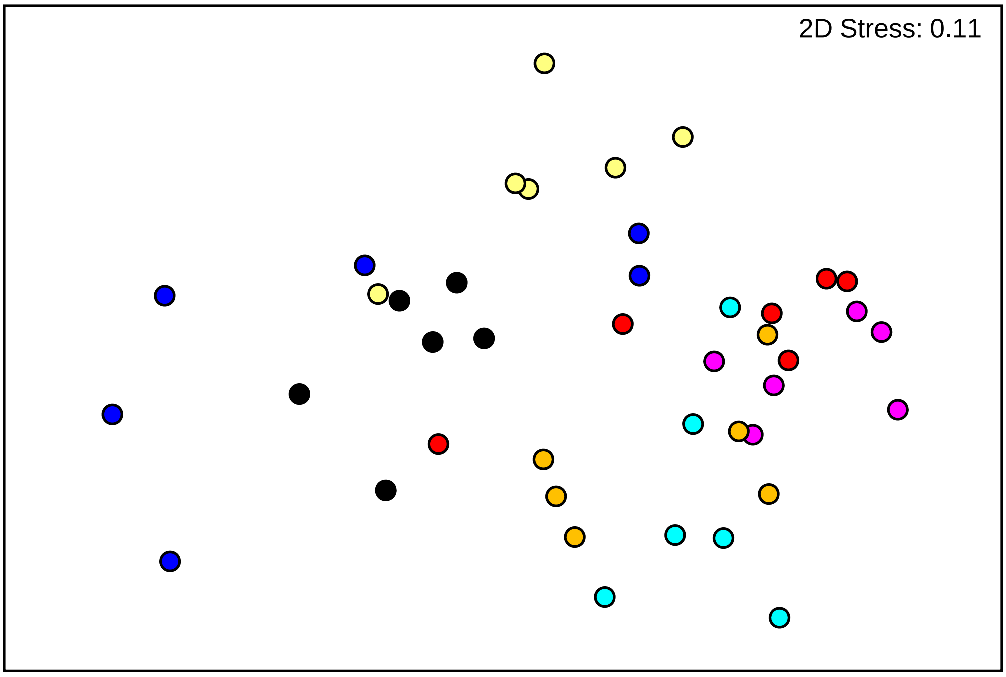


Figure 29: Mean (± S.E.) values of OE50 taxa scores and O0 Signal scores for macroinvertebrate assemblages associated with (a) edge and (b) riffle habitat at CR1 in autumn 2002- 2009 (*n* = 3). No data are available for 2005. (Different letters above bars indicate a statistically significant difference between years)

(a) Spring



(b) Autumn

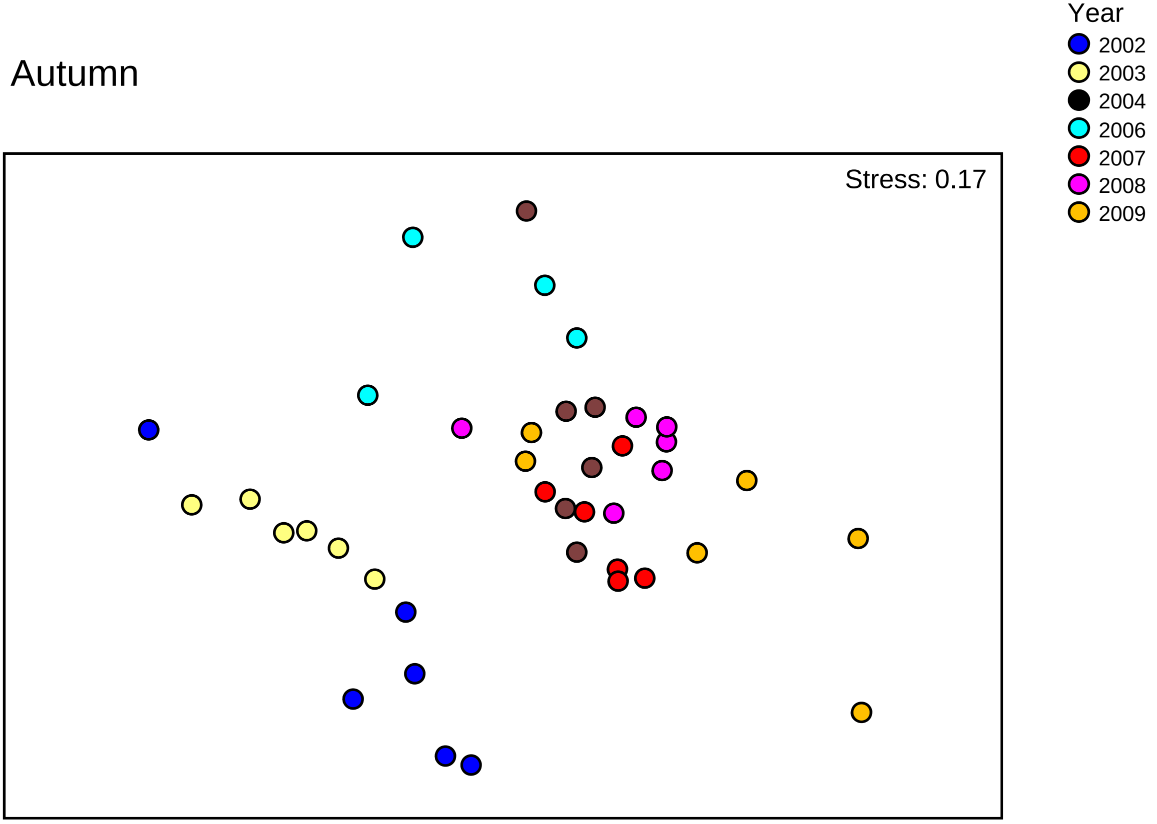
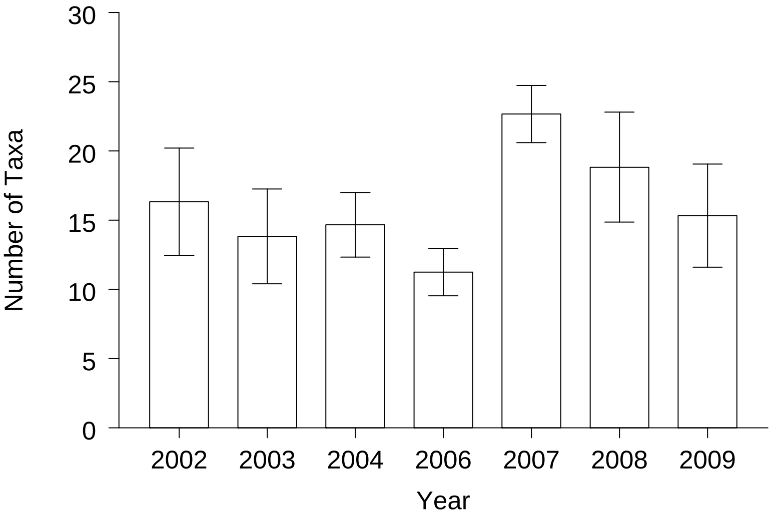


Figure 30: MDS plots comparing the structure of aquatic macroinvertebrate assemblages in the riffle habitat at CR1 as indicated by the Surber samples collected in (a) spring and (b) autumn 2002 -2009.

(a) Spring



(b) Autumn

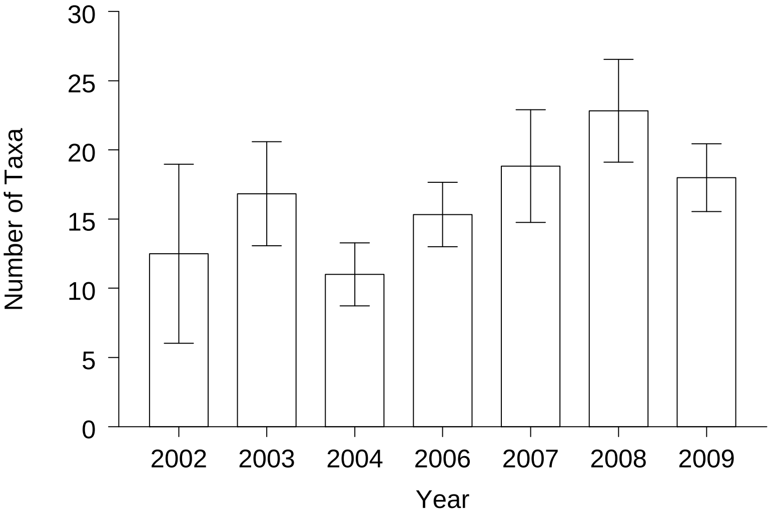
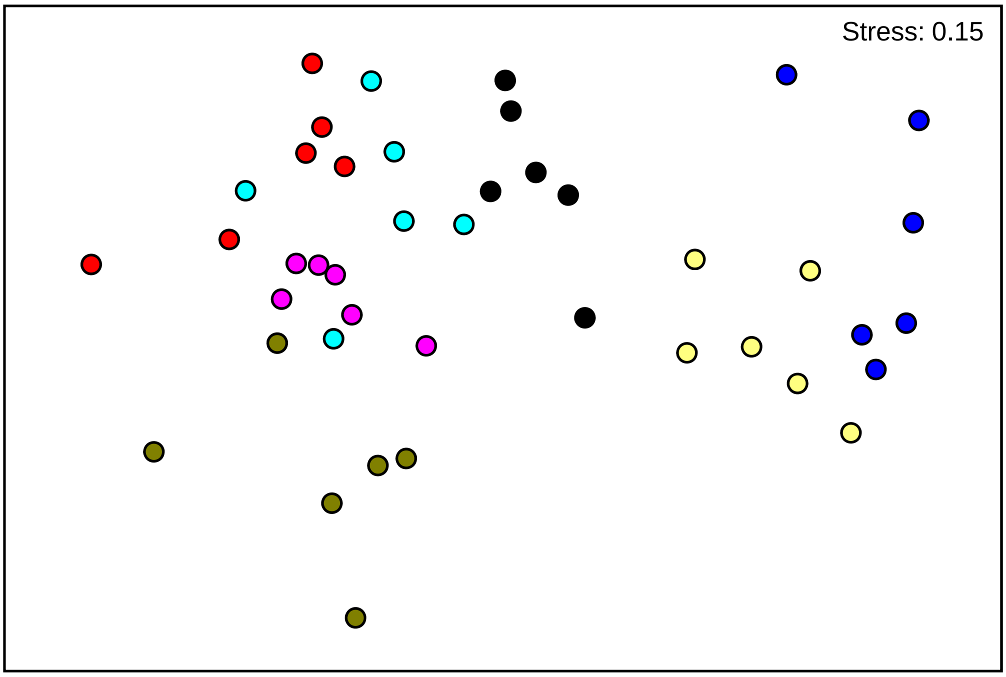


Figure 31: Mean (± S.E.) number of macroinvertebrate taxa per sample collected at CR1 in (a) spring and (b) autumn 2002 -2009 (*n* = 6).

(a) Spring



(b) Autumn

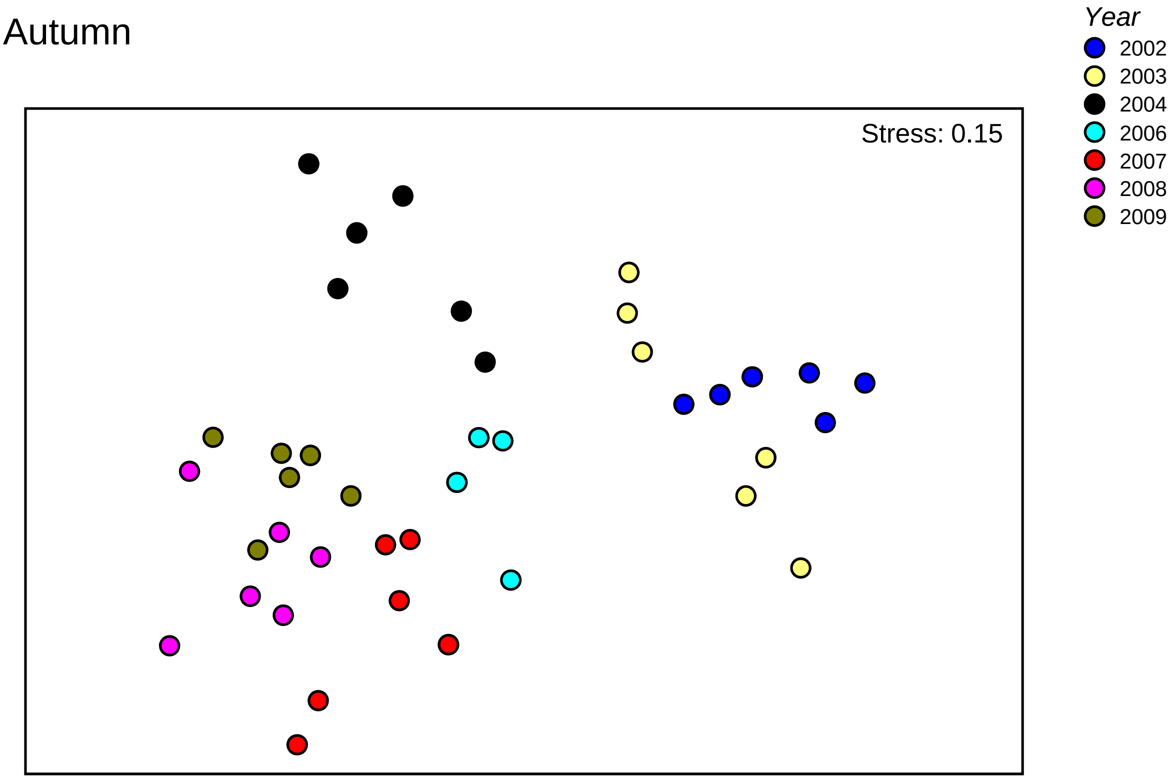
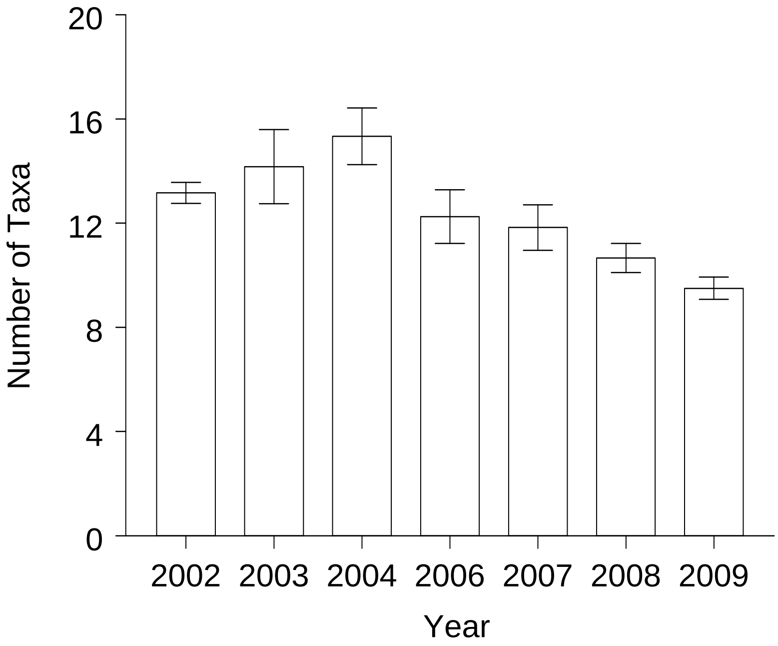


Figure 32: MDS plots comparing the structure of periphyton assemblages in the riffle habitats at CR1 in (a) spring and (b) autumn 2002 -2009.

(a) Spring



(b) Autumn

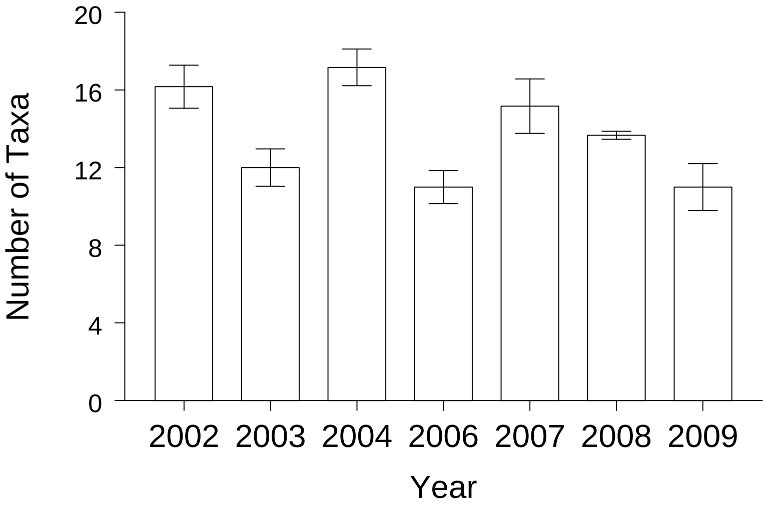


Figure 33: Mean (± S.E.) number of periphyton taxa per sample collected at CR1 in (a) spring and (b) autumn 2002 -2009 (*n* =6).

11 Appendices

Appendix 1: Duration of monitoring for each of the water quality parameters measured in (a) the onsite and (b) offsite water quality monitoring programs.

Appendix 2: Duration of the monitoring period for each of the water quality parameters measured in the regional water quality monitoring program.

Appendix 1: Duration of monitoring for each of the water quality parameters measured in (a) the onsite and (b) offsite water quality monitoring programs.

(a) Onsite Monitoring Program

Location	EC	pH	Turbidity	Sulfate	NFR	Fluoride	Iron	Manganese	Boron	Selenium
LDP20	02/09 - 09/09	02/09 - 09/09	02/09 - 09/09	02/09 - 09/09	02/09 - 09/09					
LMP7	09/97 - 02/10	09/97 - 02/10	09/08 - 02/10	09/97 - 02/10	09/97 - 02/10	09/97 - 02/10	09/97 - 01/10	09/97 - 01/10	09/97 - 01/10	09/97 - 01/10
LDP1	09/97 - 02/10	09/97 - 02/10		09/97 - 02/10	09/97 - 02/10	07/01 - 02/10			some	08/03 - 01/10
LDP3	10/00 - 02/10	09/98 - 02/10		08/98 - 02/10	08/98 - 02/10	08/98 - 02/10	08/98 - 01/10	08/98 - 01/10	08/98 - 01/10	08/98 - 01/10
LDP18	10/00 - 02/10	10/00 - 02/10			10/00 - 02/10					
LDP21	12/09 - 02/10	12/09 - 02/10	12/09 - 02/10	12/09 - 02/10	12/09 - 02/10	12/09 - 02/10			12/09 - 02/10	
LDP5	09/97 - 02/10	09/97 - 02/10			09/97 - 02/10					
LMP8	09/97 - 02/10	09/97 - 02/10		09/97 - 02/10	09/97 - 02/10	09/97 - 02/10	09/97 - 01/10	09/97 - 01/10	09/97 - 01/10	09/97 - 01/10
WX13	09/97 - 02/10	09/97 - 02/10		09/97 - 02/10	09/97 - 02/10	07/01 - 02/10			some	08/03 - 02/10
LDP4	10/00 - 02/10	09/97 - 02/10	06/08 - 02/10	09/97 - 02/10	09/97 - 02/10	12/02 - 02/10			09/01 - 01/10	08/03 - 01/10
COX4	09/05 - 02/10	09/05 - 02/10					11/05 - 01/10	11/05 - 01/10	11/05 - 01/10	11/05 - 01/10

(b) Offsite Monitoring Program

Location	TSS	Sulfate	Fluoride	Iron	Selenium
LMP7	11/02 - 05/09				
LDP1	11/02 - 05/09				
LDP3	11/02 - 05/09				
LDP5	11/02 - 05/09	11/02 - 05/09	11/02 - 05/09		08/03-08/09
LMP8	12/02 - 05/09				
WX13	11/02 - 05/09				
LDP4	11/02 - 01/10			10/08-12/08	
COX4	01/07 - 05/09				

Appendix 2: Duration of monitoring for each of the water quality parameters measured in the regional offsite water quality monitoring program.

Location	Temperature	pH	Conductivity	DO	TSS	Turbidity	NH3 _N	TKN	Nox	Nitrate
LMP7	01/03 - 01/10	12/02 - 01/10	12/02 - 01/10	01/03 - 01/10	12/02 - 01/10	12/02 - 01/10	03/03 - 01/10	12/02 - 01/10	12/02 - 01/10	12/02 - 01/10
WX13	01/03 - 01/10	12/02 - 01/10	12/02 - 01/10	01/03 - 01/10	12/02 - 01/10	12/02 - 01/10	03/03 - 01/10	12/02 - 01/10	12/02 - 01/10	12/02 - 01/10
COX5	01/03 - 01/10	12/02 - 01/10	12/02 - 01/10	01/03 - 01/10	07/03 - 01/10	12/02 - 01/10	03/03 - 01/10	12/02 - 01/10	12/02 - 01/10	12/02 - 01/10

Location	Nitrite	Total N	PO4	Total P	Aluminium	Iron	Manganese
LMP7	12/02 - 01/10	12/02 - 01/10	12/02 - 01/10	12/02 - 01/10	12/02 - 01/10	12/02 - 01/10	12/02 - 01/10
WX13	12/02 - 01/10	12/02 - 01/10	12/02 - 01/10	12/02 - 01/10	12/02 - 01/10	12/02 - 01/10	12/02 - 12/09
COX5	12/02 - 01/10	12/02 - 01/10	12/02 - 01/10	12/02 - 01/10			