Sydney WATER

2014 Sewage Treatment System Impact Monitoring Program

Interpretive Report

Volume 2 Trend analysis and case studies

Foreword

This report forms Volume 2 (of four) for the 2014 Sewage Treatment System Impact Monitoring Program (STSIMP). The 2014 interpretive report used long term trend analysis (greater than ten years) to identify if changes are occurring in wastewater system discharge quality and in downstream receiving waters. It also incorporates individual case studies aligned to three themes:

- treated wastewater discharges
- sewage overflows
- sensitivity of receiving environments.

Underlying each theme is the aim to better differentiate wastewater and recycled water discharge inputs from diffuse source inputs to receiving waters. Findings from the trend analysis and case studies will inform future case studies and monitoring.

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1 Introduction

Sydney Water monitors the environmental performance of its wastewater system through the Sewage Treatment System Impact Monitoring Program (STSIMP). It includes the monitoring of discharges from all Wastewater Treatment Plants (WWTP) and Water Recycling Plants (WRP), hereafter called 'plants' and environmental waters receiving discharges (Sydney Water 2010). Discharges include treated wastewater, recycled water or sewage overflows from wastewater networks.

The STSIMP is a requirement of Sydney Water's Environment Protection Licences (EPLs). As per the EPLs, a data report is produced annually with an interpretive report every three years. The previous interpretive report was published in 2011. The interpretive report is designed to identify changes in the environmental performance of the wastewater system and receiving waters through trend analysis.

The interpretive report also consists of case studies. The case studies for the 2014 interpretive report were based existing programs where Sydney Water had data available. The subjects for the papers were discussed between Sydney Water and the Environment Protection Authority (EPA) at the February 2014 Joint Officers Group. Future case studies will be jointly selected by the EPA and Sydney Water where adverse trends are flagged from the STSIMP or on matters of mutual interest.

The case studies for the 2014 interpretive report have been aligned to three themes with the aim to better differentiate wastewater and recycled water discharge inputs from diffuse source inputs to receiving waters. These include:

Treated wastewater discharge:

- Assessing the impact of the St Marys Water Recycling Initiative on the Hawkesbury Nepean River
- Hawkesbury Nepean River and South Creek model: a powerful tool to inform management decisions in the Hawkesbury Nepean catchment

Sewage overflows:

- Modelling wet weather overflows in the Upper Parramatta River
- Malabar Beach stormwater diversion: validation of the expected benefits

Sensitivity of the receiving environment:

- Assessing long term oceanographic fluctuations using deepwater ocean outfall dilution models
- Assessing ecological health and recreational amenity impacts of a large sewage overflow event at Glenfield on the Georges River in November 2013

Significant trends and findings from the interpretive report will be used to inform monitoring and case studies required for subsequent interpretive reports. Case studies may present validation of benefits or assessments of current/emerging environmental risks. These in turn will inform:

- key priorities for future action
- policies and strategies for improving Sydney Water's environmental performance
- development of an improved scientific evidence base

These outcomes will feed into business planning aimed at more efficiently meeting the environmental performance expectations of the community. The next interpretive report will be developed in 2016, then every four years to align with strategic business planning cycles.

The format of the 2014 interpretive report has changed in an effort to improve rigour, useability and reader friendliness. It is anticipated that the next interpretive report may vary slightly again. This is part of Sydney Water's strive for continual improvement.

2 Water quality trend analysis of receiving waters and discharges from Sydney Water's wastewater system

Abstract

Sydney Water's wastewater network spans 23 wastewater treatment and water recycling systems discharging to freshwater and ocean waters. These systems cover a range of treatment processes from primary sedimentation to ultrafiltration and reverse osmosis. They capture influent from an array of sources and discharge to waterways with varying sizes and characteristics. Many catchment changes have occurred over the past 20 to 30 years such as urban growth, the introduction of new chemical compounds, Water Sensitive Urban Design and nutrient reduction programs. To understand the contribution of the wastewater system to waterway health amongst these catchment changes requires a well-developed monitoring program.

The aim of this analysis is to provide a broad scale view of long term trends of Sydney Water's wastewater system discharge and their links to trends in key aspects of waterway health.

The parameters selected for analysis provide a measure of key environmental management concerns in Sydney's waterways. The two primary concerns include eutrophication and recreational amenity (focusing on swimmability at estuarine and beach locations). Therefore the key parameters analysed in plant discharges and the receiving waters were total nitrogen, total phosphorus and chlorophyll *a* as measures of eutrophication, and faecal coliforms and Enterococci for recreational amenity. Suspended solids and oil and grease were also assessed in plants with deepwater ocean outfalls. Toxicity was assessed at all treatment plants as a measure of the potential for plant discharge to directly affect aquatic biota. The method of assessment included a combination of temporal plots and regression analysis.

The key findings were:

1) total nitrogen concentrations in the discharge from most inland plants and in the Hawkesbury Nepean receiving waters are declining
2) watching briefs are recommended for total phosphorus at Winmalee and Hornsby Heights plants due to gradually increasingly concentrations
3) watching brief for North Head suspended solids due to current concentrations being close to EPL limits
4) oil and grease concentrations from the deepwater ocean outfall plants were gradually increasing until 2007, but have since remained stable in response to plant upgrades

There were no increases identified in the concentrations of the five key parameters, (total nitrogen, total phosphorus, chlorophyll *a*, faecal coliforms and Enterococci), from both the inland plant discharges and the corresponding downstream receiving waters.

Introduction

Sydney Water operates 23 wastewater systems, of which 15 discharge into the Hawkesbury Nepean River catchment and eight discharge to the ocean. These plants cover a range of treatment processes from primary sedimentation to ultrafiltration and reverse osmosis. They receive influent from a range of sources and discharge to a variety of waterways including inland freshwaters and ocean waters.

All inland discharges enter the Hawkesbury Nepean River via a number of tributaries, of which the largest are South, Eastern, Cattai and Berowra creeks. The catchment includes a diverse range of land uses such as urban, peri-urban and protected natural landscapes, agriculture and extractive industries. Many of these land uses provide a range of diffuse source pollutants and in combination with point sources, such as wastewater discharges, have the potential to put stress on the health of the Hawkesbury Nepean River and its tributaries.

In the Hawkesbury Nepean catchment, Sydney Water operates 15 plants that discharge between Maldon Weir in the upper Nepean catchment and Broken Bay. In addition, Hawkesbury Council operates two plants at McGraths Hill and South Windsor, which discharge to the Hawkesbury Nepean River. Total flow discharged into the Hawkesbury Nepean River from Sydney Water's plants has increased from 91 ML/day in 1980 to 127 ML/day in 2013-14. During this period, the catchment population increased from 335,000 to over 700,000 people.

Coastal plants discharge to ocean environments via a variety of outfalls. These include the deepwater ocean outfalls at Bondi, Malabar and North Head. These are Sydney Water's three largest plants discharging primary treated wastewater via a series of diffusers. The remaining five plants have near shore ocean outfalls, with secondary and tertiary treatment discharging along the Sydney and Illawarra coastlines.

In the early 1990s, Sydney Water investigations into the management of wastewater in the Hawkesbury Nepean River catchment identified the need to reduce the nutrient loads from treated wastewater discharges to the environment (Sydney Water 1997). In particular, phosphorus was identified as the key nutrient contributing to the development of potentially toxic algal blooms in the lower Hawkesbury Nepean River. A modelling study confirmed that phosphorus from plant discharges was the main contributor to eutrophication of the river during dry weather (Sydney Water 1996).

Since this time major treatment plant upgrades have occurred throughout Sydney Water's inland wastewater system. These upgrades include: the South Creek bubble licence upgrades (1998 to 2001) for the St Marys, Quakers Hill and Riverstone plants; the Berowra Nitrogen Reduction Program for Hornsby Heights and West Hornsby plants (2000 to 2002); the West Camden plant upgrade in 2009; and the St Marys Water Recycling Initiative commissioned in 2010 (Sydney Water 2014). The Wallacia plant (tertiary treatment) was opened in 2006 replacing the Warragamba plant (secondary treatment). The Blue Mountains Sewer Tunnel project to divert all sewage flows from the upper Blue Mountains towns to the Winmalee plant, located outside of the Blue Mountains World Heritage Area, was completed in June 2008 when the Blackheath and Mt Victoria plants were decommissioned. Between

1989 and 2005, sewage flows from all other Blue Mountains towns were diverted to the tertiary treatment plants at Winmalee or Penrith.

Various changes in catchment processes have occurred over the previous 20 to 30 years including urban growth, more extensive uptake of Water Sensitive Urban Design (WSUD), nutrient reduction programs such as the NSW Office of Water Nutrient Smart Management Program, and changes in chemical compounds used in industry and households. The changes in inputs to the Hawkesbury Nepean River, estuarine and ocean waters in Sydney, combined with changes to Sydney Water's wastewater systems, require monitoring and analysis. This is to better understand how receiving waters are responding to these changes and how Sydney Water's wastewater systems are contributing to these observed changes.

The aim of the trend analysis section of the 2014 STSIMP interpretive report is to provide a broad scale view of trends in discharge concentrations from Sydney Water's wastewater system plants and trends in the environmental condition of the receiving waters. The specific objectives are to:

1) identify long term trends, or other notable departures from typical conditions, for the selected parameters for key receiving water sites and plants.
2) provide screening level assessments of the significance of trends departing from zero trend and possible links to Sydney Water's wastewater systems.

Methods

Approach

This analysis examines long term trends in discharges from Sydney Water's wastewater systems and receiving water, acting as a screening level assessment. This analysis is primarily focused on the current operational configurations for each plant to provide the best indication possible of how current conditions may change in the future.

To ensure trends identified in the receiving waters could be linked to the wastewater system, where possible the trend analysis focused on water quality parameters monitored in both receiving waters and in treated wastewater. Treated wastewater discharge quality was also included to assess if treatment efficacy has changed over time.

Monitoring of ecological indicators was not included in this analysis as these indicators can be affected by a large range of *in situ* processes in their respective catchments which may confound results. Where a trend/change is identified through screening level analysis, additional monitored parameters, ecological monitoring and other studies can be included in future assessments.

The parameters selected for analysis provide a measure of key environmental management concerns of Sydney's waterways. The two primary concerns are:

1) eutrophication, being the enrichment of a waterbody with nutrients resulting in excessive growth of photosynthetic organisms and depletion of dissolved oxygen.
2) recreational amenity, including suitability for swimming at estuarine and beach locations.

Where treated wastewater is discharged from deepwater ocean outfalls, potential impacts from algal blooms and on recreational amenity are unlikely due to the high dilution of effluent and low use of the immediate receiving waters by the community. For these systems, the key parameters analysed (namely oil and grease and suspended solids) were targeted to measure the efficacy of treatment. Data was checked for completeness, relevance and length of record to determine the parameters were suitable for trend analysis. Receiving water analysis was carried out on available data from July 1994 to July 2014, while treatment plant discharge quality was assessed on data from July 1998 to July 2014. Historical receiving water and plant discharge data, (pre 1994), was not considered to ensure analysis is focused on identifying changes relevant to the current operating conditions of the wastewater system.

Monitoring programs

The Sewage Treatment System Impact Monitoring Program (STSIMP) is Sydney Water's core monitoring program to measure the impacts of its wastewater operations to the receiving water environment (Sydney Water 2010). It details monitoring activities and methods in all catchments of Sydney Water's area of operations. This includes the Hawkesbury Nepean River catchment where fifteen systems currently operate, and coastal waters where eight systems currently discharge to the ocean.

The STSIMP succeeded an earlier monitoring program, the Environmental Indicators Monitoring Program (EIMP,) which had similar broad objectives (Sydney Water 1995). This program ran consistently for a period of 14 years from July 1994 to June 2008 providing a long term dataset that is now being added to by the STSIMP. Combining the data enables long term analysis to identify trends in the quality of wastewater discharges and receiving waters.

Eutrophication and recreational amenity are the two key environmental management issues targeted in this study. These, combined with data availability from the STSIMP and former EIMP, guided the parameters selected for analysis. The parameters chosen to represent eutrophication were total nitrogen (TN), total phosphorus (TP) and chlorophyll *a* (Chl-a). Dissolved inorganic nitrogen (DIN) and filtered total phosphorus (FTP) data are presented for receiving waters to provide an indication of the bioavailable fractions of total nitrogen and total phosphorus. While filtered total phosphorus provides a guide to the bioavailable fraction of total phosphorus, it does not strictly represent it. Faecal coliforms or Enterococci (depending on data availability) were chosen to represent recreational amenity. For the plants that discharge to ocean waters, suspended solids and oil and grease were analysed to provide a measure of long term changes in the treated wastewater being discharged. Toxicity results from all treatment plants, either with inland discharges or ocean outfalls, were also analysed to provide an indication of the potential risk to aquatic biota from plant discharges. For inland plants the *Ceriodaphnia dubia* immobilisation EC_{50} (concentration that effects 50% of organisms) was reported as percent effluent, while for ocean plants the sea urchin sperm fertilisation EC_{50} was reported as percent effluent.

Site locations

Monitoring sites were chosen to represent key sections of the respective waterways to allow valid conclusions to be drawn. These sites covered a range of waterway characteristics, including downstream of plant discharge points, urban sites not directly affected by Sydney Water's discharges and sites upstream of all Sydney Water wastewater discharges. Receiving water sites were also selected to represent areas with potential for uncontrolled discharges (sewage overflows) to occur and impact on water quality. Broadly speaking receiving waters sites were chosen to represent:

- eutrophication in the Hawkesbury Nepean River system, estuaries (coastal lagoons) and urban rivers
- recreational amenity in estuaries (coastal lagoons) and Illawarra beaches

Sites for eutrophication monitoring in estuaries and urban rivers were located in coastal lagoons (estuaries) and in upper freshwater reaches of urban rivers ie Parramatta, Lane Cove and Georges rivers, where tidal flushing and dilution is reduced. This allows for clearer identification of changes in freshwater inputs to estuarine environments, where higher levels of tidal dilution decrease the ability to detect change.

Sites with long term records were selected to ensure trends more accurately identify potential future changes in water quality. A minimum of ten years of data was chosen for the long term datasets. Wallacia and Brooklyn plants and St Marys AWTP were not assessed in this study due to relatively short periods of operation. The Richmond plant was also not assessed as the old plant was decommissioned in 2006 and replaced with a new plant that provides the majority of discharge to reuse. The Picton plant was not analysed as water quality data is limited due to the majority of plant discharge being directed to irrigation reuse. It would be expected that in future STSIMP interpretive reports these plants will be assessed when the period of operation/data availability exceeds 10 years. For all plants with short term records, annual performance against EPL limits is presented in Volume 3 Data Report in Section 1.4.2 Results, while concentration and load data for the previous ten years are presented in Volume 3 Data Report Section 6.2 Appendix B.

Figure 2-1 and Figure 2-2 presents maps of the Illawarra Beachwatch and Sydney study sites respectively, including the locations of the plants. Currently operating plants are described in Table 2-1, while water quality sites analysed in this paper are described in Table 2-2.

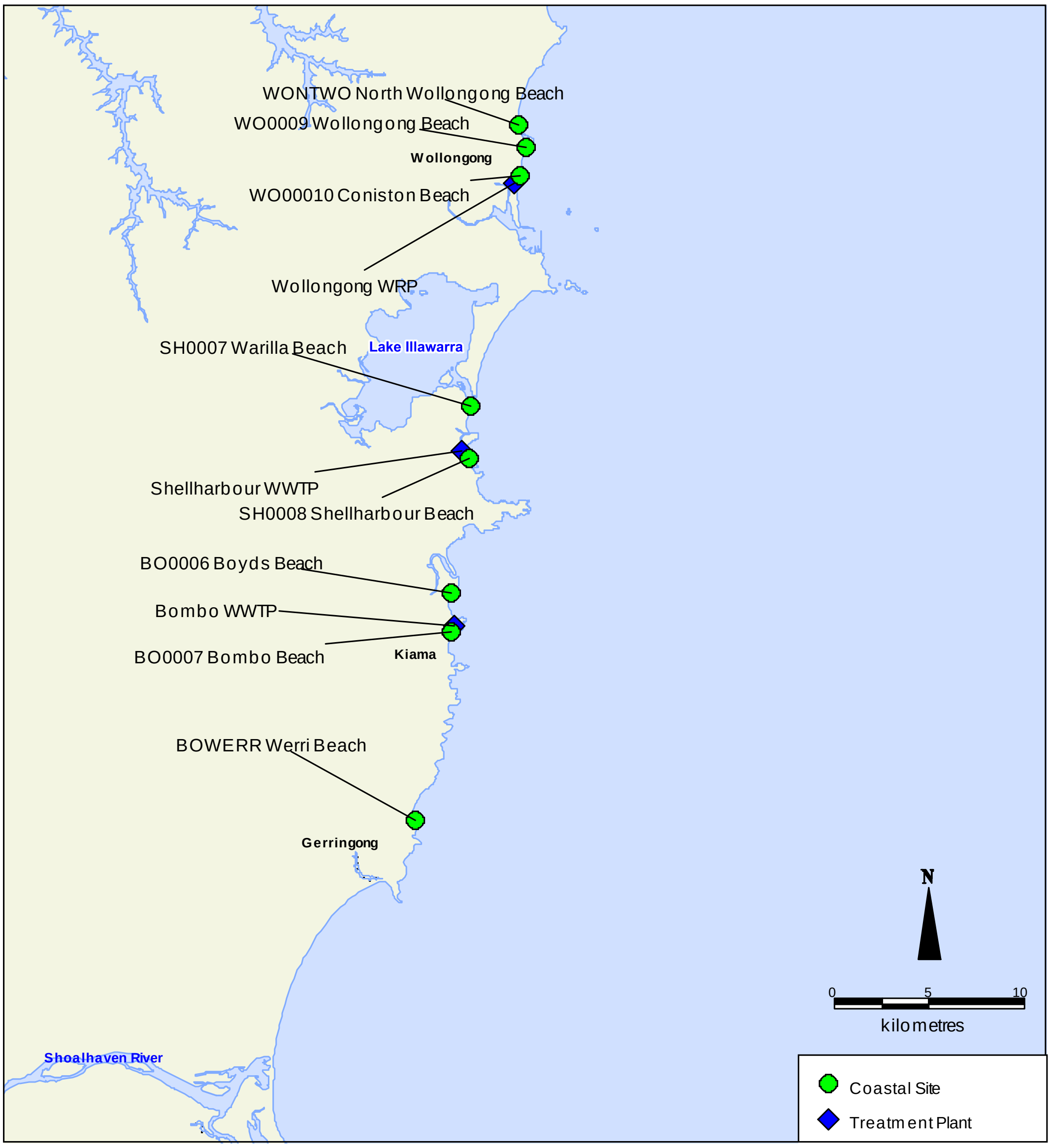


Figure 2-1 Location of Illawarra Beachwatch sites and treatment plants

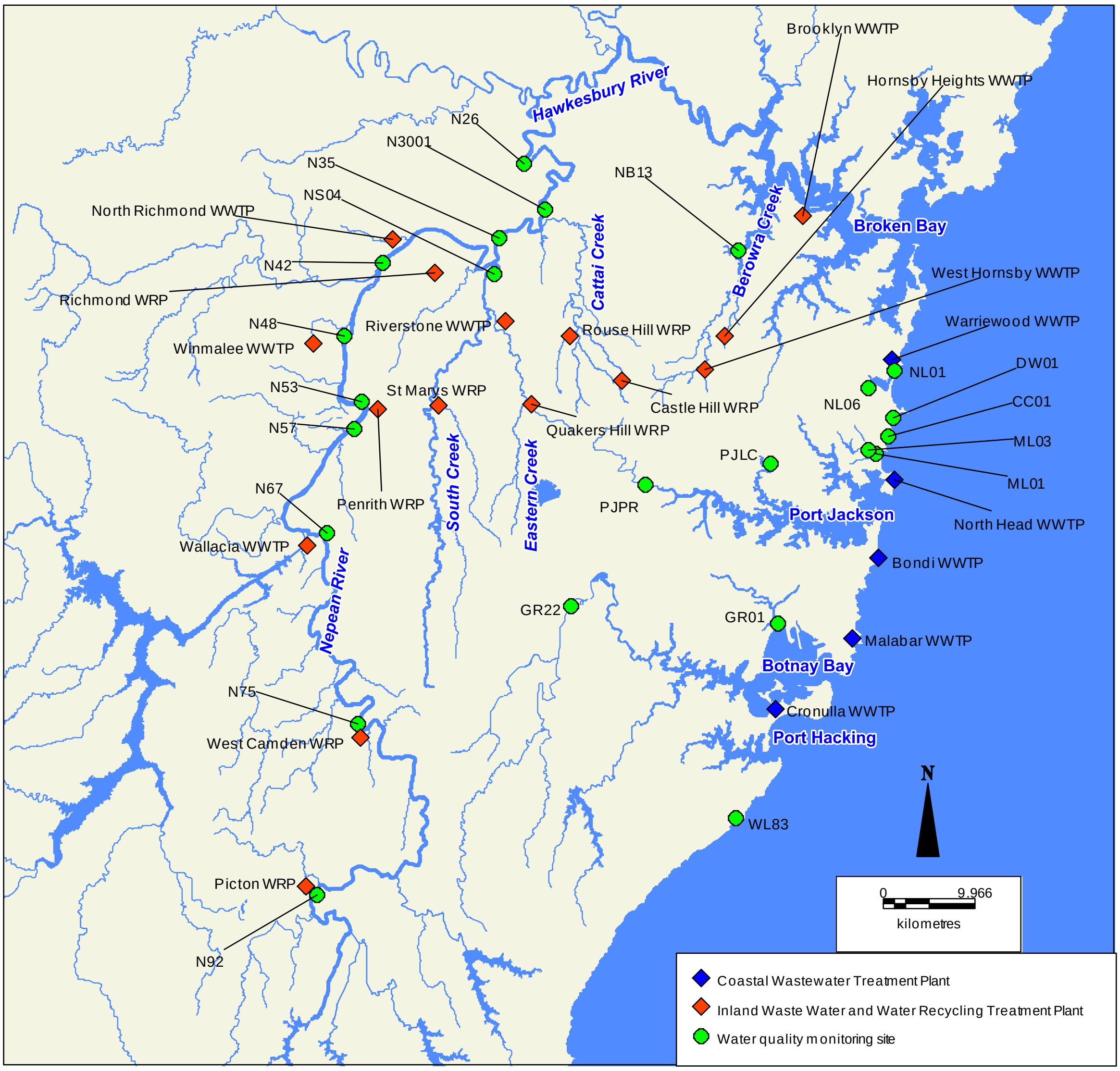


Figure 2-2 Location map of water quality sites and treatment plants assessed in this study

Table 2-1 List of plants currently operated by Sydney Water that discharge to coastal or inland waters

Catchment	Plants	Treated wastewater discharge location	Operation status	Upgrade history
Hawkesbury Nepean River	Picton WRP	Reused for onsite agricultural irrigation; wet weather overflows to Stone Quarry Creek to the Hawkesbury Nepean River	Operating since November 2000	NA
	West Camden WRP	Matahil Creek to the Hawkesbury Nepean River	Operating for the full period	Upgraded and amplified in 2009
	Wallacia WWTP	Warragamba River to the Hawkesbury Nepean River	Operating since September 2006	New plant replacing Warragamba WWTP in 2006
	Penrith WRP	Boundary Creek to the Hawkesbury Nepean River	Operating for the full period	Stage 8 upgrade 2003 and 2004
	Winmalee WWTP	Unnamed Creek to the Hawkesbury Nepean River	Operating for the full period	Upgrade to improve reliability 2007 to 2009 to receive flows from decommissioned Blue Mtns plants
	North Richmond WWTP	Redbank Creek to the Hawkesbury River	Operating for the full period	NA
	Richmond WRP	Reused for irrigation at the University of Western Sydney Richmond campus and Richmond Golf Club; excess overflows to Rickabys Creek to the Hawkesbury Nepean River	Operating since 2006	A new plant was commissioned in 2006 to replace the old plant
	St Marys WRP	Unnamed creek to South Creek	Operating for the full period	South Creek bubble licence upgrades between 1999 and 2001
	St Marys AWTP	Boundary Creek to the Nepean River – high quality recycled water discharge	Operating since September 2010	NA
	Quakers Hill WRP	Breakfast Creek to Eastern Creek	Operating for the full period	South Creek bubble licence upgrades between 1999 and 2001
	Riverstone WWTP	Eastern Creek to South Creek	Operating for the full period	South Creek bubble licence upgrades between 1999 and 2001
	Castle Hill WRP	Cattai Creek	Operating for the full period	NA
	Rouse Hill WRP	Second Ponds Creek to Cattai Creek; also reused for local recycling scheme	Operating for the full period	Various upgrades and amplifications since commissioning of the plant in

Catchment	Plants	Treated wastewater discharge location	Operation status	Upgrade history
				1994
	Hornsby Heights WWTP	Calna Creek to Berowra Creek	Operating for the full period	Berowra Creek Nitrogen Reduction Program 2000 to 2002
	West Hornsby WWTP	Waitara Creek to Berowra Creek	Operating for the full period	Berowra Creek Nitrogen Reduction Program 2000 to 2002
	Brooklyn WWTP	Hawkesbury Nepean River at 14 m depth on the second pylon of the old road bridge adjacent to Kangaroo Point	Operating since December 2007	NA
Deepwater ocean outfalls	North Head WWTP	Discharge from deepwater ocean outfall 3.7 km from shore at 65 m depth	Operating for the full period	Upgrades between 2005 to 2010 to improve plant reliability
	Bondi WWTP	Discharge from deepwater ocean outfall 2.2 km from shore at 63 m depth	Operating for the full period	Upgrades between 2005 to 2010 to improve plant reliability
	Malabar WWTP	Discharge from deepwater ocean outfall 3.6km from shore at 82 m depth	Operating for the full period	Upgrades between 2005 to 2010 to improve plant reliability
Near shore ocean outfalls	Cronulla WWTP	Discharge from Potter Point shoreline at a depth of ~ 5 m	Operating for the full period	Upgrade to provide tertiary treatment and UV disinfection 2001
	Warriewood WWTP	Discharge to Turrimetta Headland south of Warriewood Beach	Operating for the full period	UV disinfection in 2000 and improve reliability in 2009
Illawarra near shore ocean outfalls	Wollongong WRP	Discharge from offshore ocean outfall ~ 1 km from shore	Operating for the full period	Illawarra Wastewater Strategy upgrade in 2005 to include tertiary treatment and UV disinfection
	Shellharbour WWTP	Discharge from offshore ocean outfall 130 m from shore	Operating for the full period	Illawarra Wastewater Strategy upgrade completed in 2006
	Bombo WRP	Discharge from headland north of Bombo Beach at ~ 2 m deep	Operating for the full period	Illawarra Wastewater Strategy upgrade completed in 2005

NA Not applicable

Table 2-2 List and descriptions of water quality monitoring sites included in the trend analysis study

Site	Site type	Description	Parameters	Data range
N92	Hawkesbury Nepean River catchment	Hawkesbury Nepean River at Maldon Weir upstream of all treated wastewater discharges	TN, TP, Chl-a	1994-2014
N75	Hawkesbury Nepean River catchment	Hawkesbury Nepean River at Sharpes Weir, downstream of West Camden WRP inflow	TN, TP, Chl-a	1994-2014
N67	Hawkesbury Nepean River catchment	Hawkesbury Nepean River at Wallacia Bridge, upstream of Warragamba River inflow	TN, TP, Chl-a	1994-2014
N57	Hawkesbury Nepean River catchment	Hawkesbury Nepean River at Penrith Weir, upstream of Penrith WRP and St Marys AWTP inflow	TN, TP, Chl-a	1994-2014
N53	Hawkesbury Nepean River catchment	Hawkesbury Nepean River at BMG Causeway, downstream of Penrith WRP and St Marys AWTP inflow	TN, TP, Chl-a	1994-2014
N48	Hawkesbury Nepean River catchment	Hawkesbury Nepean River at Smith St, upstream of Winmalee WWTP discharge inflow	TN, TP, Chl-a	1994-2014
N42	Hawkesbury Nepean River catchment	Hawkesbury Nepean River at North Richmond, downstream of Winmalee WWTP discharge inflow	TN, TP, Chl-a	1994-2014
NS04	Hawkesbury Nepean River catchment	South Creek at Windsor before the inflow to the Hawkesbury River	TN, TP, Chl-a	1994-2014
N35	Hawkesbury Nepean River catchment	Hawkesbury Nepean River downstream of South Creek inflow	TN, TP, Chl-a	1994-2014
N26	Hawkesbury Nepean River catchment	Hawkesbury Nepean River at Sackville	TN, TP, Chl-a	1994-2014
NB13	Hawkesbury Nepean River catchment	Berowra Creek in tidal zone at Cunio Point	TN, TP, Chl-a	1994-2014
PJPR	Urban River	Parramatta River Weir	Chl-a	1994-2014
PJLC	Urban River	Lane Cove River Weir	Chl-a	1994-2014
GR22	Urban River	Georges River upstream of Liverpool Weir	Chl-a	1994-2014

Site	Site type	Description	Parameters	Data range
GR01	Urban River (estuarine)	Cooks River downstream of Muddy Creek near the Botany Bay entrance	Chl-a	1994-2014
NL01	Estuarine coastal lagoon	Narrabeen Lagoon canal entrance upstream of Ocean Bridge	Chl-a, Enterococci	1994-2014
NL06	Estuarine coastal lagoon	Narrabeen Lagoon 150m north of confluence with South Creek	Chl-a, Enterococci	1994-2014
DW01	Estuarine coastal lagoon	Dee Why Lagoon entrance at Long Reef	Chl-a, Enterococci	1994-2014
CC01	Estuarine coastal lagoon	Curl Curl Lagoon entrance at North Curl Curl	Chl-a, Enterococci	1994-2014
ML01	Estuarine coastal lagoon	Manly Lagoon upstream of Queenscliff Bridge	Chl-a, Enterococci	1994-2014
ML03	Estuarine coastal lagoon	Manly Lagoon at Footbridge in Nolan Reserve	Chl-a, Enterococci	1994-2014
WL83	Estuarine coastal lagoon	Wattamolla Lagoon reference site	Chl-a, Enterococci	1994-2014
BO0006	Illawarra Beach	Boyds Beach, ~ 2km north of Bombo WRP discharge point	Enterococci	1998-2014
BO0007	Illawarra Beach	Bombo Beach, ~ 0.5 km south of Bombo WRP discharge point	Enterococci	1998-2014
SH0007	Illawarra Beach	Warilla Beach, ~ 2km north of Shellharbour WWTP discharge point	Enterococci	1998-2014
SH0008	Illawarra Beach	Shellharbour Beach, ~1 km south of Shellharbour WWTP discharge point	Enterococci	1998-2014
WO0009	Illawarra Beach	Wollongong Beach, ~ 2km north of Wollongong WRP discharge point	Enterococci	1998-2014
WO00010	Illawarra Beach	Coniston Beach, ~ 1km north of Wollongong WRP discharge point	Enterococci	1998-2014
WONTWO	Illawarra Beach	North Wollongong Beach, ~ 3km north of Wollongong WRP discharge point	Enterococci	1998-2014
BOWERR	Illawarra Beach	Werri Beach, unaffected by discharge from plants	Enterococci	1998-2014

Chl-*a* Chlorophyll *a*

Data analysis

Two techniques were employed to identify long term trends in the data sets analysed:

1. long term data was plotted temporally and visually inspected for step changes, trends, outliers and other notable characteristics.
2. regression analysis was then performed where appropriate to assess the significance of any trends that may be present in the data by comparing the slope of the trend line to zero trend. The null hypothesis that was applied to all regression analysis was that the slope of the trend line would not be different from zero.

Identifying and characterising trends was carried out using both methods in combination. For receiving waters, temporal plots and regression analysis gave an indication of change in ambient conditions over a period greater than ten years. The outcomes of this analysis will potentially represent many different processes occurring in a catchment including changes to wastewater discharges, changes in urban runoff and agricultural run off, and land use change.

For plant discharges, temporal plots were used in the first instance only. This was due to plant upgrades leading to many step changes in concentrations of targeted parameters. In this case regression analysis is not an appropriate technique to measure change over the full analysis period. Regression analysis was considered for plants and parameters where a visible trend was evident on temporal plots or current concentrations are close to EPL limits. In this case regression analysis was carried out to determine if significant changes in discharge concentrations were occurring over the long term (greater than ten years) if no further upgrades have been carried out in that period. This gives a measure of change in discharge quality relevant to a plants current operational setup.

Data analysis was conducted using concentration data for each parameter. Analytical results that were below the method detection limit were deemed as half of the method detection limit. The only exception to this was for microbiological parameters where 0 values and <1 results were both taken as 1. Both represent the same result, however <1 has been used in more recent times to account for difficulties in reporting a 0 count of bacterial colony forming units.

Chlorophyll *a* was measured using the sonication method until December 1995 and then the improved method of grinding was introduced. The relationship between sonication and grinding methods has been determined (AWT 1997a), and based on this relationship, a correction factor of 1.18 times was applied to the pre-December 1995 chlorophyll *a* data to make this dataset compatible for long-term analysis.

For receiving waters, data points were separated according to wet and dry weather conditions as physical and chemical processes can vary greatly in response to flow. The days when average rainfall (previous three days moving average) exceeded 7 mm/day or actual rainfall on the day exceeded 10 mm were categorised as wet weather. The remaining data were categorised as dry weather data. Dry weather data were used for analysis as this represents the majority of data points and represents the conditions for which discharges are expected to have greatest influence on water quality. All data points were averaged monthly to remove bias associated with seasonal events where extra data may have been collected, for example when algal blooms necessitated increased monitoring for chlorophyll *a*. Only routine data has been used rather than event data to avoid over representing one-off events with a high number of data points.

Data were tested for normality using the Jacques-Bera test and by plotting histograms to test the underlying assumption of data being normally distributed for regression analysis. Where a normal distribution was not evident, data were $\log_{10}$ transformed and tested again for normality. The vast majority of data sets required $\log_{10}$ transformation to meet the requirement of normal distribution for regression analysis.

Using the long term datasets a large number of data points were included in each regression analysis resulting in the analysis being sensitive to small differences in trend lines when compared to zero. Careful interpretation is required to identify meaningful trends in the context of long term changes in treated wastewater discharge and in downstream receiving waters. For this reason a p value of <0.01 was used in this report to identify a significant trend. p values of less than <0.05 are also noted to indicate where there may be potential for a more significant trend.

Results

Receiving waters

Hawkesbury Nepean River

Results of regression analysis are provided in Table 2-3 for the Hawkesbury Nepean River catchment sites. Temporal plots including total nitrogen, dissolved inorganic nitrogen, total phosphorus, filtered total phosphorus and chlorophyll *a* to support regression analysis results are provided Appendix A (Figure 9-1). Sites are represented longitudinally from upstream to downstream along the Hawkesbury Nepean River with the location of each treatment plants discharge to the river indicated. Red and orange cells represent increasing trends at p<0.01 and p<0.05 respectively. Yellow cells represent no trend while green and light green cells represent decreasing trends at p<0.01 and p<0.05 respectively.

Trend analysis for the Hawkesbury Nepean River focused on eutrophication and concentrations of nitrogen and phosphorus. Results indicated all receiving water sites except for the Hawkesbury Nepean River at Penrith Weir (N57) had declining total nitrogen trends (p<0.01). Trends in total phosphorus concentrations in the Hawkesbury Nepean River from Penrith downstream were declining and significant at the p<0.01 level, with the exceptions of Berowra Creek (NB13). Total phosphorus in the Hawkesbury Nepean River at Wilberforce downstream of the South Creek inflow (N35) had a declining trend significant at the p<0.05 level. Upstream of Penrith, no trends in total phosphorus concentrations were evident except for an increasing trend at Penrith Weir at the p<0.05 level.

Concentrations of dissolved inorganic nitrogen and filtered total phosphorus (Appendix A - Figure 9-1) broadly followed changes in concentrations for total nitrogen and total phosphorus respectively at all sites. In the period from 2012 to 2014 increases in total phosphorus concentrations are evident in some sites between Penrith Weir and North Richmond that are not reflected in filtered total phosphorus concentrations. This is particularly so for North Richmond (N42). The reasons for this are not clear however this was a period of high rainfall after which large quantities of macrophytes were scoured from the river (Sydney Water 2014), providing a potential explanation for changes in the proportion of bioavailable phosphorus. The sites at which this observation is most notable are not directly downstream of Sydney Water wastewater system discharges ie increased chlorophyll *a* concentrations are not evident in the Nepean River immediately downstream of where the West Camden, Penrith and St Marys plants discharge to the Nepean River.

Penrith Weir (N57) and North Richmond (N42), sites not directly affected by discharges from plants, both had increasing trends for chlorophyll *a* ($p<0.01$). It is likely the increasing chlorophyll *a* trend at North Richmond has been influenced by the macrophyte washout caused by flooding in 2012 (Sydney Water 2014), with high chlorophyll *a* levels observed since this time, as indicated by Figure 9-1 in Appendix A. All other river sites showed no long term trend in chlorophyll *a* concentrations except for a declining trend at the control site upstream of all plant discharges, Maldon Weir (N92) at the $p<0.01$ level.

Table 2-3 Regression results presented longitudinally for the Hawkesbury Nepean River and the location of plants along the river

Site code	Site description	Treatment plant location and discharge waterway	Hawkesbury Nepean River and tributaries trends		
			TN	TP	Chl-*a*
N92	Hawkesbury Nepean River at Maldon Weir				
		West Camden WRP via Matahil Creek			
N75	Hawkesbury Nepean River at Sharpes Weir, d/s West Camden WRP		down p<0.01	no trend	no trend
N67	Hawkesbury Nepean River at Wallacia Bridge, u/s Warragamba River		down p<0.01	no trend	no trend
N57	Hawkesbury Nepean River at Penrith Weir		no trend	up p<0.05	up p<0.01
		Penrith WRP via Boundary Creek			
N53	Hawkesbury Nepean River at BMG Causeway, d/s Penrith WRP		down p<0.01	down p<0.01	no trend
N48	Hawkesbury Nepean River at Smith St, upstream of Winmalee plant		down p<0.01	down p<0.01	no trend
		Winmalee WWTP via Winmalee Lagoon			
N42	Hawkesbury River at North Richmond, d/s of Winmalee		down p<0.01	down p<0.01	up p<0.01
		North Richmond WWTP via Redbank Creek			
		St Marys WRP via South Creek			
		Quakers Hill WRP via South Creek			
		Riverstone WWTP via South Creek			
NS04	Lower South Creek, at Fitzroy Bridge		down p<0.01 [a]	down p<0.01	no trend
N35	Hawkesbury Nepean River at Wilberforce, d/s of South Creek		down p<0.01	down p<0.05	no trend
		Rouse Hill WRP via Cattai Creek			
		Castle Hill WRP via Cattai Creek			
N3001	Hawkesbury Nepean River off Cattai SRA, d/s of Cattai Creek		down p<0.01 [a]	down p<0.01	no trend
N26	Hawkesbury Nepean River at Sackville Ferry		down p<0.01	down p<0.01	no trend
		West Hornsby WWTP via Berowra Creek			
		Hornsby Heights WWTP via Berowra Creek			
NB13	Berowra Creek at Cunio Point		down p<0.01	no trend	no trend

[a] trends were analysed using a parametric t-test as available data was split into two periods; 1996 to 2001 and 2008 and 2014. TN=Total Nitrogen; TP=Total Phosphorus; Chl-*a*=chlorophyll *a*

Yellow	no trend
Green	down p<0.01
Light green	down p<0.05
Orange	up p<0.05
Red	up p<0.01

Estuaries and urban rivers

Regression analysis results are presented for estuarine coastal lagoons and urban rivers in Table 2-4 and are plotted temporally in Appendix A in Figure 9-2 and Figure 9-3. The majority of sites provided no long term trend for either chlorophyll *a* concentrations or Enterococci densities. The only trends detected ($p<0.05$) were increasing chlorophyll *a* concentrations in the Georges River at Liverpool Weir (GR22) and decreasing Enterococci concentrations ($p<0.05$) at Dee Why Lagoon. Chlorophyll *a* in the Georges River at Liverpool Weir is discussed in in detail in Chapter 8 which looks at the impact of a large sewage overflow at Glenfield.

Table 2-4 Regression results presented for estuaries and urban rivers

Site code	Site description	Estuaries and urban rivers	
		Chlorophyll *a*	Enterococci
NL01	Narrabeen Lagoon, canal entrance upstream of Ocean Bridge	(no trend)	(no trend)
NL06	Narrabeen Lagoon, 150m north of confluence with South Creek	(no trend)	(no trend)
DW01	Dee Why Lagoon, entrance at North Curl Curl	(no trend)	(down p<0.05)
CC01	Curl Curl Lagoon, entrance at North Curl Curl	(no trend)	(no trend)
ML01	Manly Lagoon, upstream of Queenscliff Beach Bridge	(no trend)	(no trend)
ML03	Manly Lagoon, at footbridge in Nolan Reserve	(no trend)	(no trend)
WL83	Wattamolla Lagoon	(no trend)	(no trend)
PJLC	Lane Cove Weir	(no trend)	nm
PJPR	Parramatta River Weir	(no trend)	nm
GR01	Cooks River, downstream of Muddy Creek	(no trend)	nm
GR22	Liverpool Weir	(up p<0.05)	nm

nm: not monitored

(yellow)	no trend	(green)	down p<0.05	(orange)	up p<0.05

Illawarra beaches

The majority of samples from the Illawarra beach sites had Enterococci densities below the method detection limit, heavily skewing each dataset. As such only temporal plots were used in the analysis to identify if any clear trends were evident (Figure 2-3). No clear trends were visible for each monitored beach. The vast majority of Enterococci densities at each site are below the threshold of 40 cfu/100mL for microbial assessment Category A, as outlined in NHMRC (2008). Beachwatch results from 2013-14 (OEH 2014) also indicate that all beaches in this analysis achieved NHMRC (2008) Beach Suitability Grades of good or very good, and have provided water quality of a high standard since monitoring began in the 1990s.

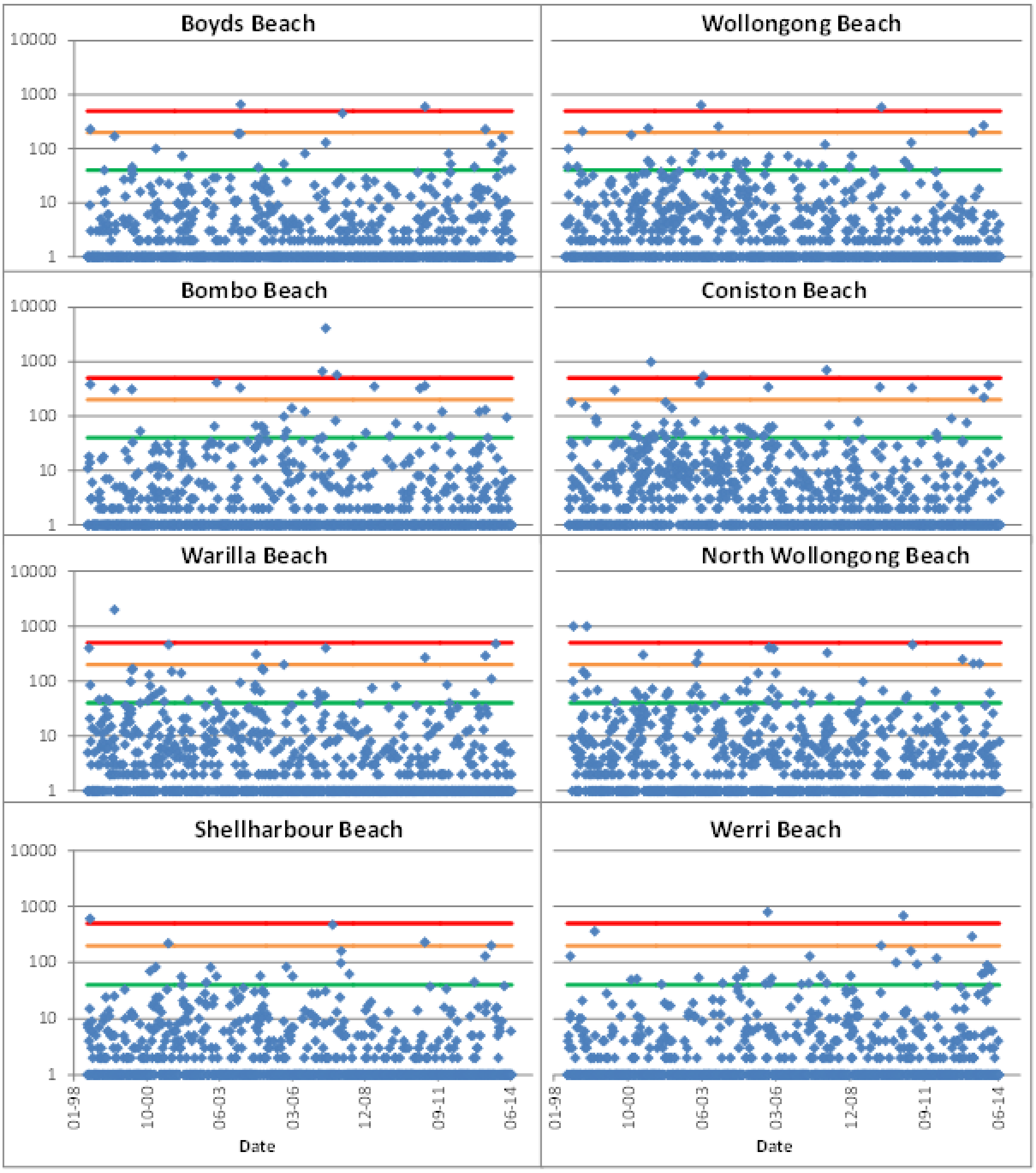


Figure 2-3 Temporal Enterococci plots (cfu/100mL) for selected Illawarra Beachwatch sites with NHMRC (2008) microbial assessment categories marked. Below 41 (green) is Category A, below 200 (orange) is Category B, below 500 (red) is Category C and above 500 (red) is category D

Treatment plants

Inland plants

Temporal plots of total nitrogen and total phosphorus over the period of analysis for plants with greater than ten years of data are presented in Figure 2-4. EPL limits are also plotted where applicable. Results are considered longitudinally from the furthest upstream sites first.

The West Camden plant shows a clear decrease in total nitrogen concentrations after a major upgrade and amplification of the plant in 2009. This decrease is also evident for total phosphorus concentrations, although smaller in size. Concentrations of both parameters at the West Camden plant remain below EPL limits, particularly for total nitrogen. At the Penrith plant, a decrease in total nitrogen and total phosphorus concentrations is evident after an upgrade in 2003-2004 to amplify and improve reliability of the plant. Concentrations of both parameters are currently within EPL limits. The Winmalee plant underwent an upgrade to improve reliability between 2007 and 2009, although was followed by a period of variable total nitrogen and total phosphorus concentrations (Figure 2-4). Concentrations of total nitrogen have since steadied and started to decrease to levels well below EPL limits. The temporal plot for total phosphorus discharged from the Winmalee plant indicates a very gradual increase in discharge concentrations, however total phosphorus concentrations currently remain well within EPL limits. Total phosphorus trends will be subject to a watching brief in subsequent STSIMP reports to determine if this gradual increase is a significant and ongoing. Sydney Water is currently developing an intensive monitoring plan for Winmalee to inform possible future upgrades (as part of Pollution Reduction Plan (PRP) 800). At the North Richmond plant, a decreasing trend in total phosphorus concentrations is apparent, while total nitrogen concentrations remain steady. Concentrations of both parameters are currently well below EPL limits.

In the South Creek catchment, the discharge concentrations of total phosphorus and total nitrogen from the St Marys and Riverstone plants showed a clear decrease between 1998 and 2002. This was in response to upgrades associated with the South Creek bubble licence conditions. The South Creek bubble licence applies to the St Marys, Quakers Hill and Riverstone plants. Since this time concentrations of both nutrient parameters have generally remained steady at the St Marys and Quakers Hill plants. A slight decrease in total nitrogen was observed in 2008 at Riverstone followed by a slight increase in 2012, however total nitrogen concentrations currently remain slightly below pre-2008 levels and are well below EPL limits. This variability will be monitored year by year to better understand if it represents an ongoing change that requires further investigation.

In the Cattai Creek catchment, total nitrogen and total phosphorus concentrations discharged from the Castle Hill plant have been relatively constant and remain below EPL limits. Slight decreases in total nitrogen concentrations occurred between 2007 and 2010, likely in response to minor works at this time designed to improve the plants reliability. Rouse Hill WRP has undergone multiple upgrades and amplifications since 1998 resulting in variable total nitrogen concentrations, however no net change is evident since 1998. Total phosphorus concentrations have declined in response to the works at Rouse Hill and both parameters remain below EPL limits.

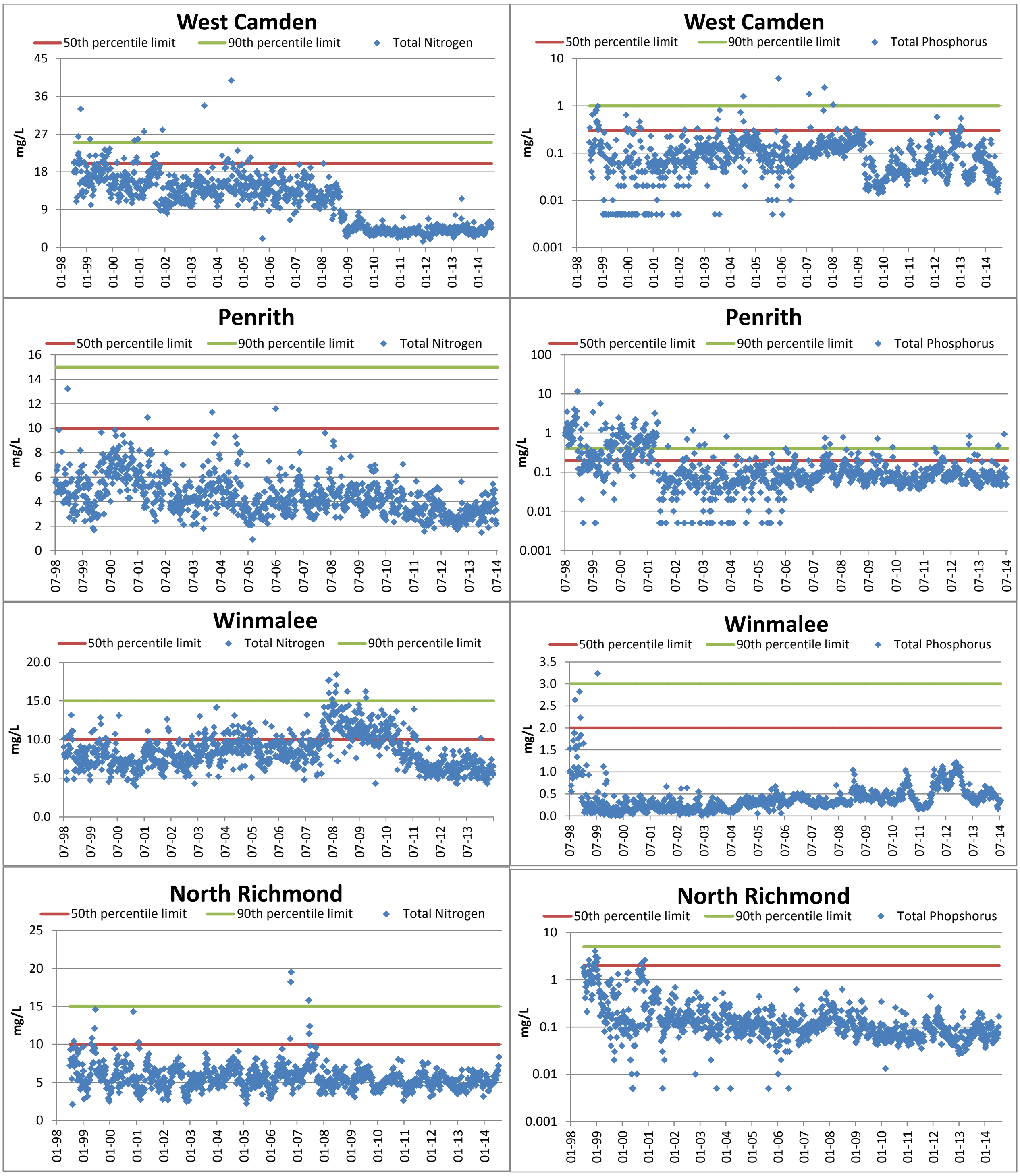
In the Berowra Creek catchment, clear decreases in total nitrogen concentrations occurred in response to the Berowra Creek nitrogen reduction program which saw upgrades to both the West Hornsby and Hornsby Heights plants. Total phosphorus was generally stable from the West Hornsby plant. There was a slight increase in total phosphorus concentrations discharged from the Hornsby Heights plant (Figure 2-4). Total phosphorus trends will be subject to a watching brief in subsequent STSIMP reports to determine if this gradual increase is significant and ongoing.

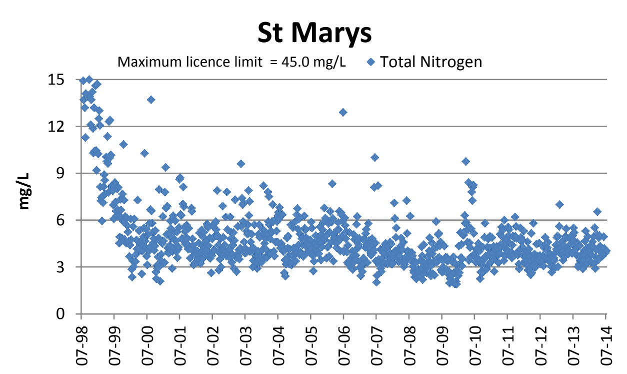
Further analysis of the two plants identified for watching briefs, (Winmalee and Hornsby Heights plants), was considered to ascertain if the trends visible on temporal plots were significant at the $p<0.01$ level. The criteria for this analysis ie greater than ten years of post upgrade data available to ensure a meaningful analysis of potential discharge changes in the plants current operational environment, was met for Hornsby Heights plant. The significant upgrade works conducted at the Winmalee plant between 2007 and 2009 mean that only a short post upgrade dataset was available that will be susceptible to temporary variations in plant discharge quality.

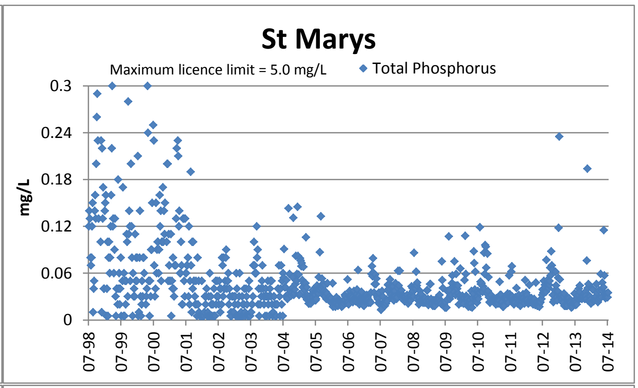
Hornsby Heights plant data was subjected to regression analysis according to the methods outlined in this study for data from July 2003 to July 2014. This represents the period since the

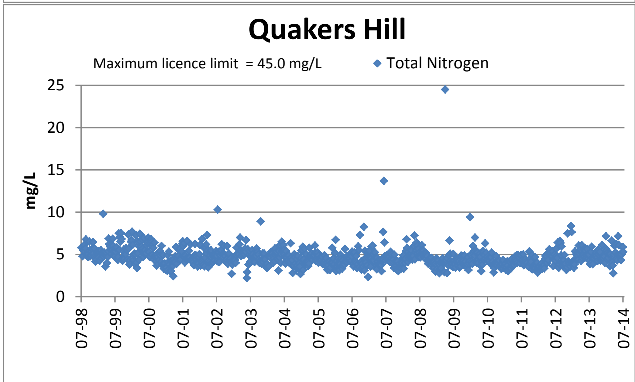
Berowra Creek nitrogen reduction program was implemented in 2002. The analysis found an increasing trend at the $p<0.01$ level, confirming the need for a watching brief of this plant. It should be noted that in the estuarine Berowra Creek receiving waters, phosphorus and chlorophyll *a* have not changed significantly over the full analysis period.

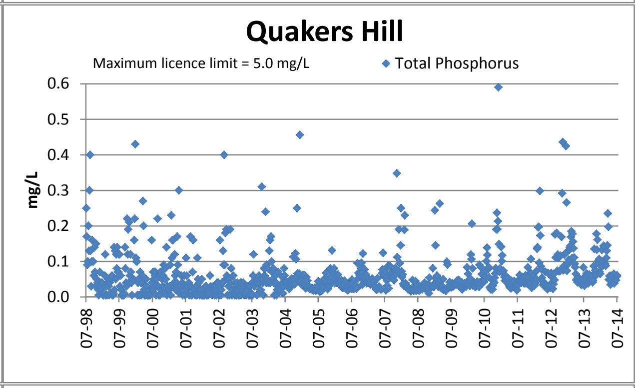
Across all inland plants, the vast majority of EC_{50} toxicity results for *Ceriodaphnia dubia* immobilisation test were 100% effluent (Figure 2-5), meaning that even at 100% effluent a toxic effect was not being observed on at least 50% of test organisms. There were also no clear trends evident overtime.

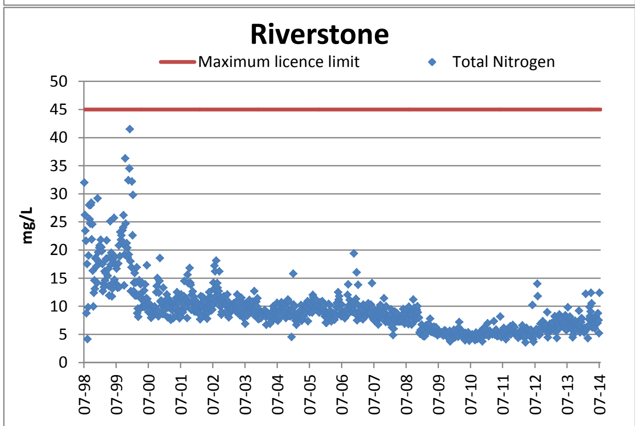


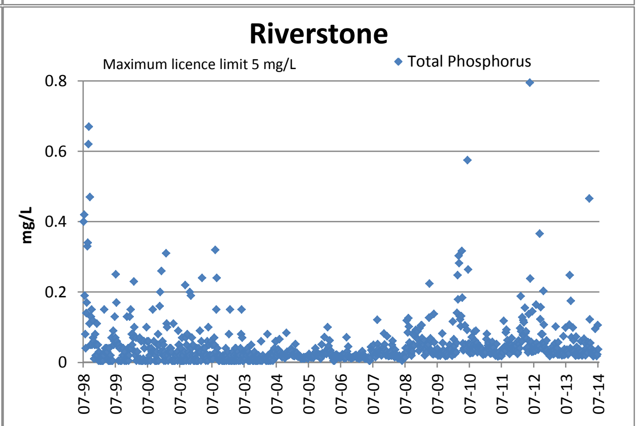

St Marys
Maximum licence limit = 45.0 mg/L
Total Nitrogen
mg/L

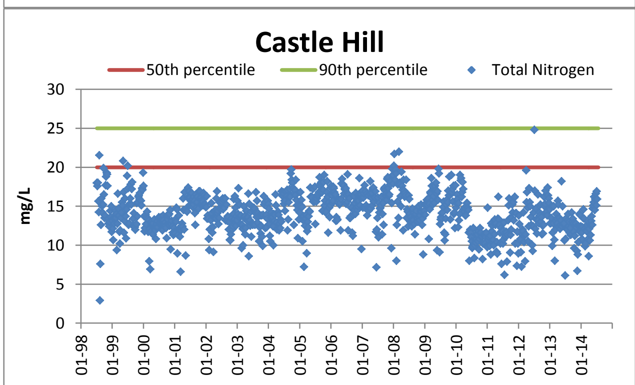

St Marys
Maximum licence limit = 5.0 mg/L
Total Phosphorus
mg/L

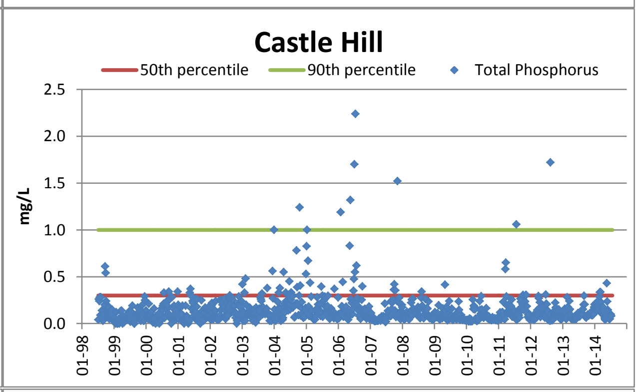

Quakers Hill
Maximum licence limit = 45.0 mg/L
Total Nitrogen
mg/L

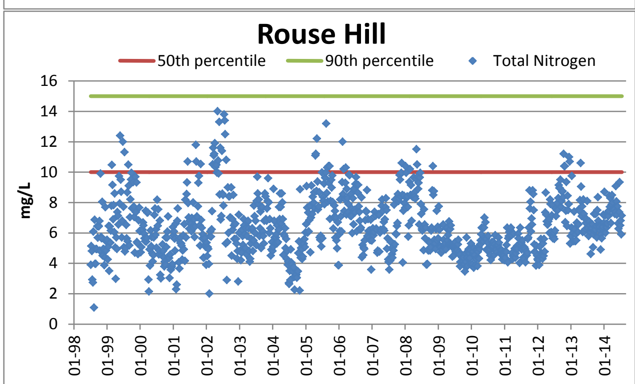

Quakers Hill
Maximum licence limit = 5.0 mg/L
Total Phosphorus
mg/L

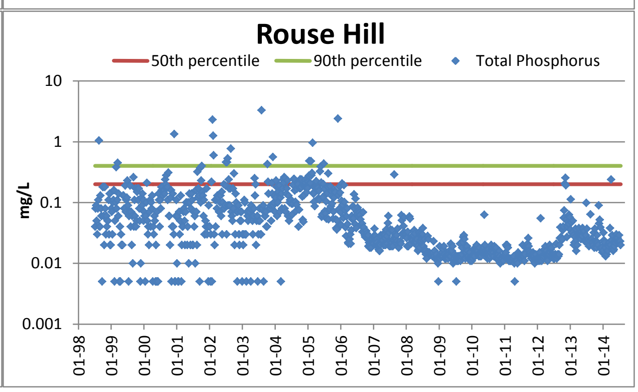

Riverstone
Maximum licence limit
Total Nitrogen
mg/L


Riverstone
Maximum licence limit 5 mg/L
Total Phosphorus
mg/L


Castle Hill
50th percentile
90th percentile
Total Nitrogen
mg/L


Castle Hill
50th percentile
90th percentile
Total Phosphorus
mg/L


Rouse Hill
50th percentile
90th percentile
Total Nitrogen
mg/L


Rouse Hill
50th percentile
90th percentile
Total Phosphorus
mg/L

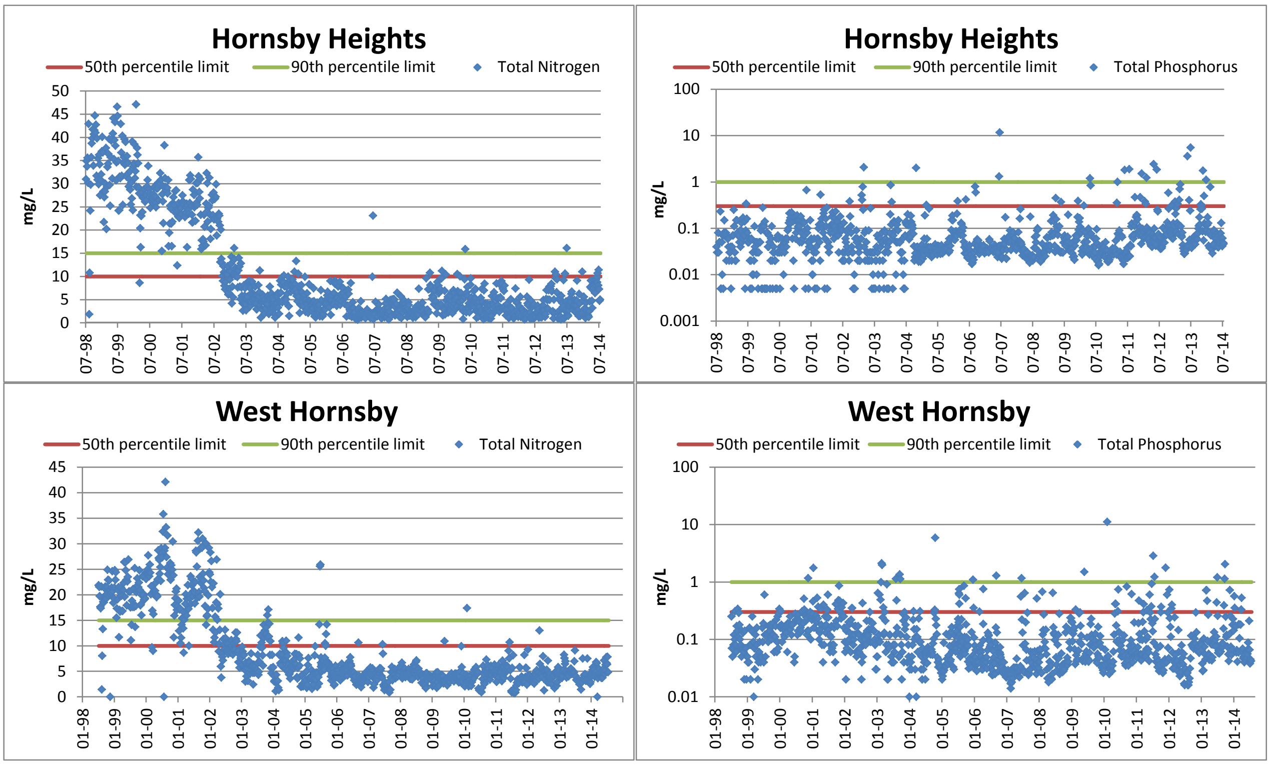


Figure 2-4 Temporal plots of discharged nitrogen (left) and phosphorus (right) concentrations from each inland plant analysed including EPL limits where applicable

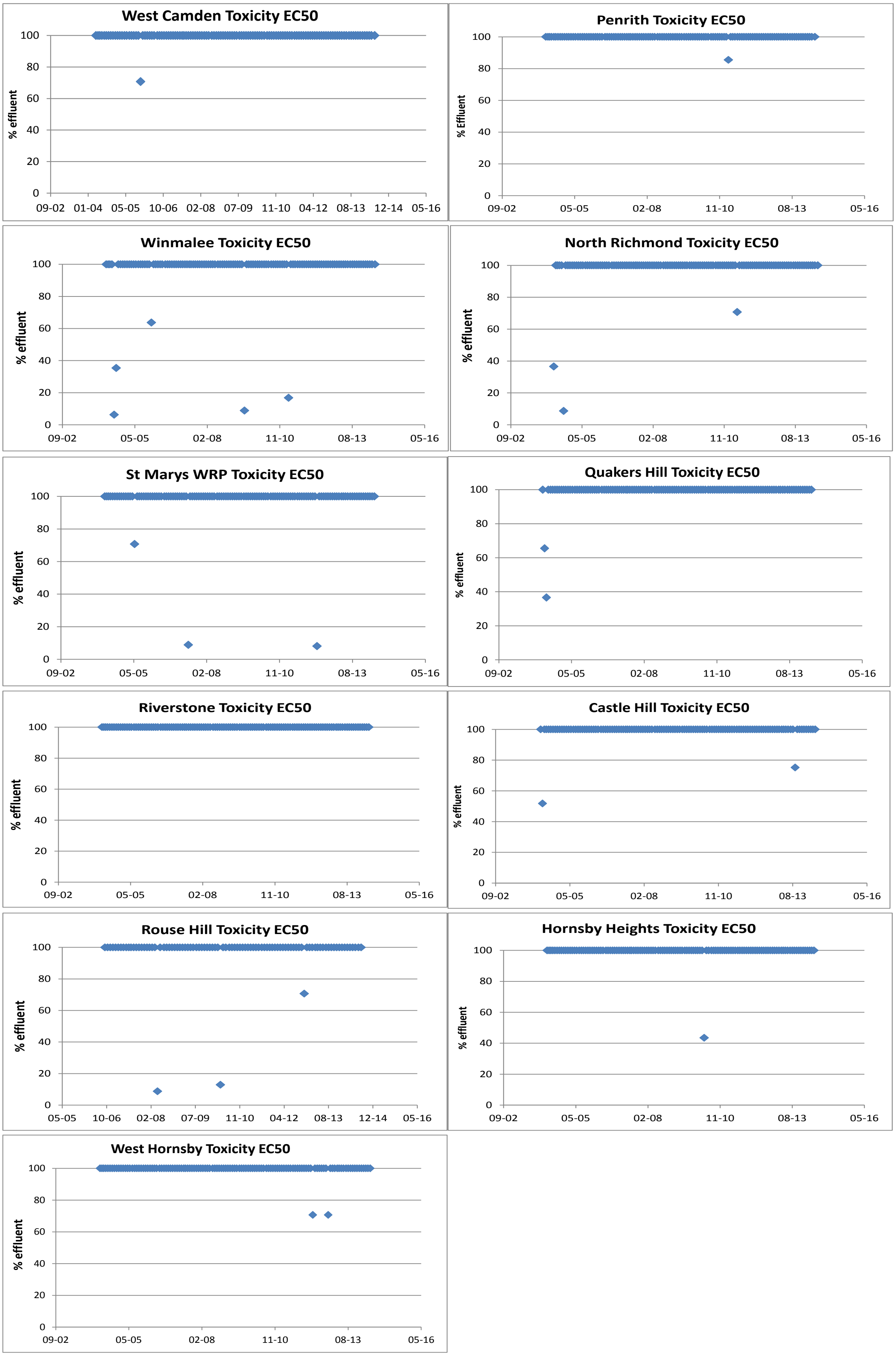


Figure 2-5 Temporal plots of *Ceriodaphnia dubia* immobilisation EC_{50} toxicity from each inland plant

Ocean plants

Figure 2-6 presents suspended solids and oil and grease temporal plots for the deepwater ocean outfall plants (Bondi, Malabar and North Head), while Figure 2-7 presents suspended solids and faecal coliform temporal plots for the nearshore ocean outfall plants (Bombo, Shellharbour, Wollongong, Cronulla and Warriewood).

For the deepwater ocean outfall sites, gradual increases in oil and grease concentrations were apparent between 1998 and 2006-07 at all three plants (Figure 2-6). After 2007, oil and grease concentrations remained steady with the majority of results below the 50^{th} percentile EPL limit for each plant. This was likely in response to works carried out at the plants in the 2005 to 2010 period with the objective to improve the reliability of operation and ability to meet EPL limits. Concentrations of suspended solids at all three deepwater ocean outfall plants were generally steady with the only change of note being slight increases between 1998 and 2006 at the Bondi and Malabar plants. Suspended solids concentrations at each plant remained within EPL limits in 2013-14, although by only a small margin at North Head plant. A watching brief will be applied to suspended solids discharged from the North Head plant.

At the nearshore ocean outfall plants, faecal coliforms concentrations were generally steady with the majority of results within the 50^{th} percentile EPL limits (Figure 2-7). Decreasing concentrations of suspended solids was apparent for the Shellharbour, Bombo and Wollongong plants. Wollongong, Shellharbour and Bombo plants were part of the Illawarra Wastewater Strategy which included a program of works to amplify and upgrade these plants, likely explaining decreases in suspended solids concentrations. These changes are detailed in Sydney Water (2008). For the Warriewood plant, suspended solids concentrations generally remained steady in the plants discharge through the analysis period.

Given the North Head plant suspended solids concentrations are currently below but close to EPL limits, further analysis was considered. Plant upgrades have been carried out at North Head between 2005 and 2010 to increase reliability and improve compliance with EPL limits. As such only a short term dataset representing the current operational environment was available, meaning regression analysis was not suitable. If suspended solids concentrations remain close to EPL limits in future reports when a longer dataset is available, then regression analysis may be appropriate when temporary fluctuations will have less influence on the outcome.

Sea urchin fertilisation EC_{50} toxicity results (Figure 2-8) were all within EPL limits at ocean discharge plants with no clear trends identified.

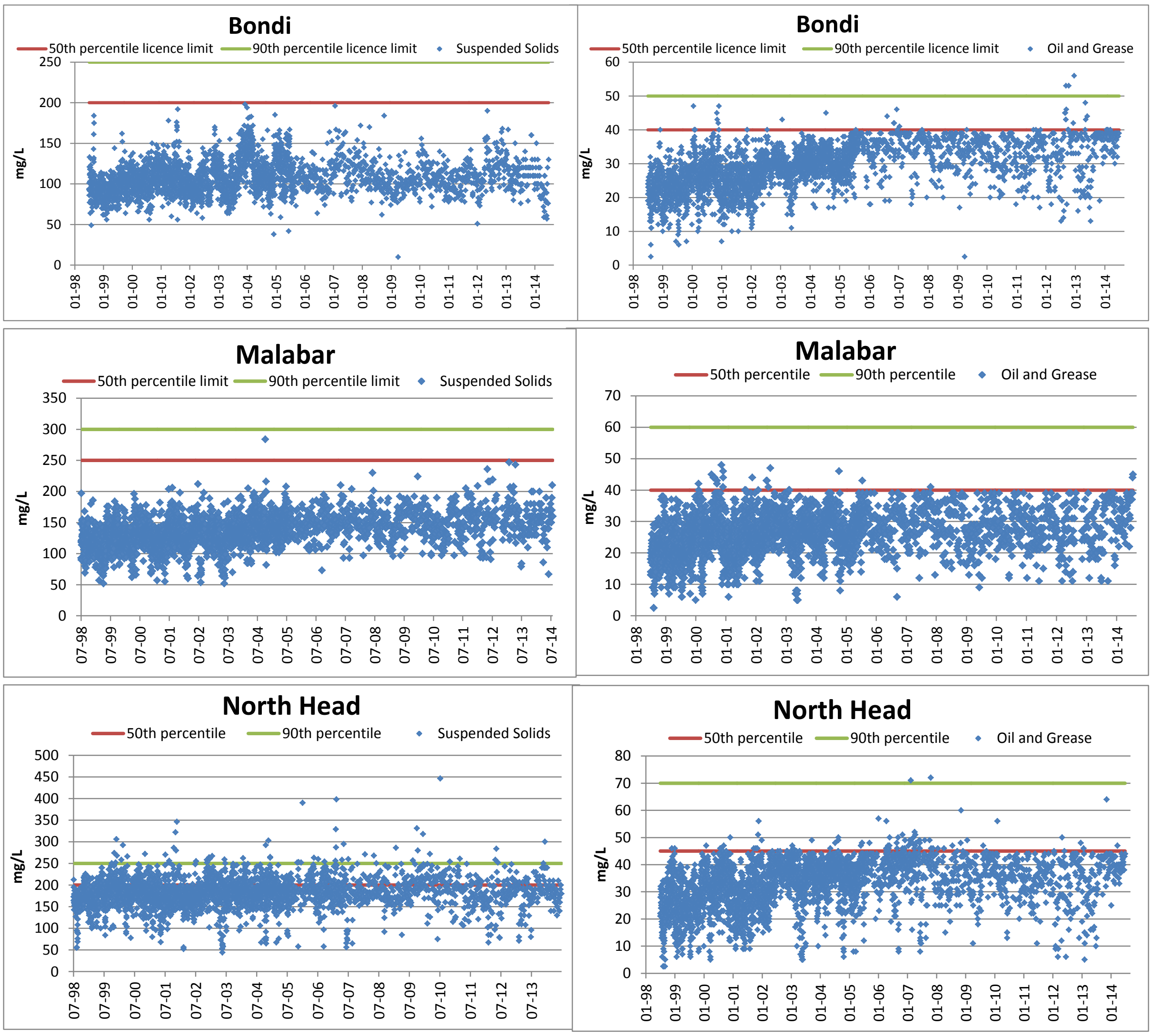


Figure 2-6 Temporal plots of discharged suspended solids (left) and oil & grease (right) concentrations from each deepwater ocean outfall plant analysed, including EPL limits were applicable

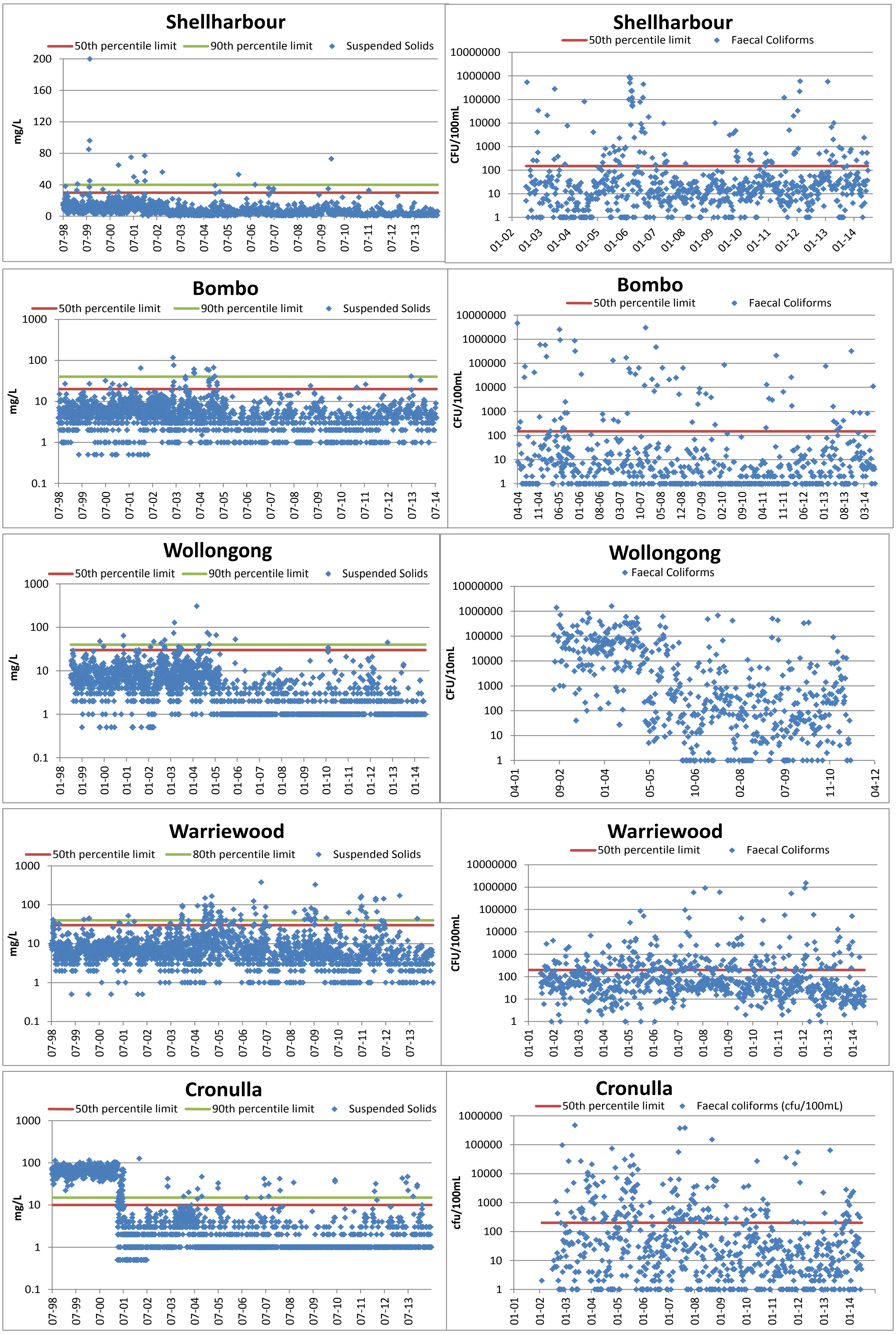


Figure 2-7 Temporal plots of discharged suspended solids (left) and faecal coliforms (right) concentrations from each near shore ocean outfall plant analysed, including EPL limits were applicable

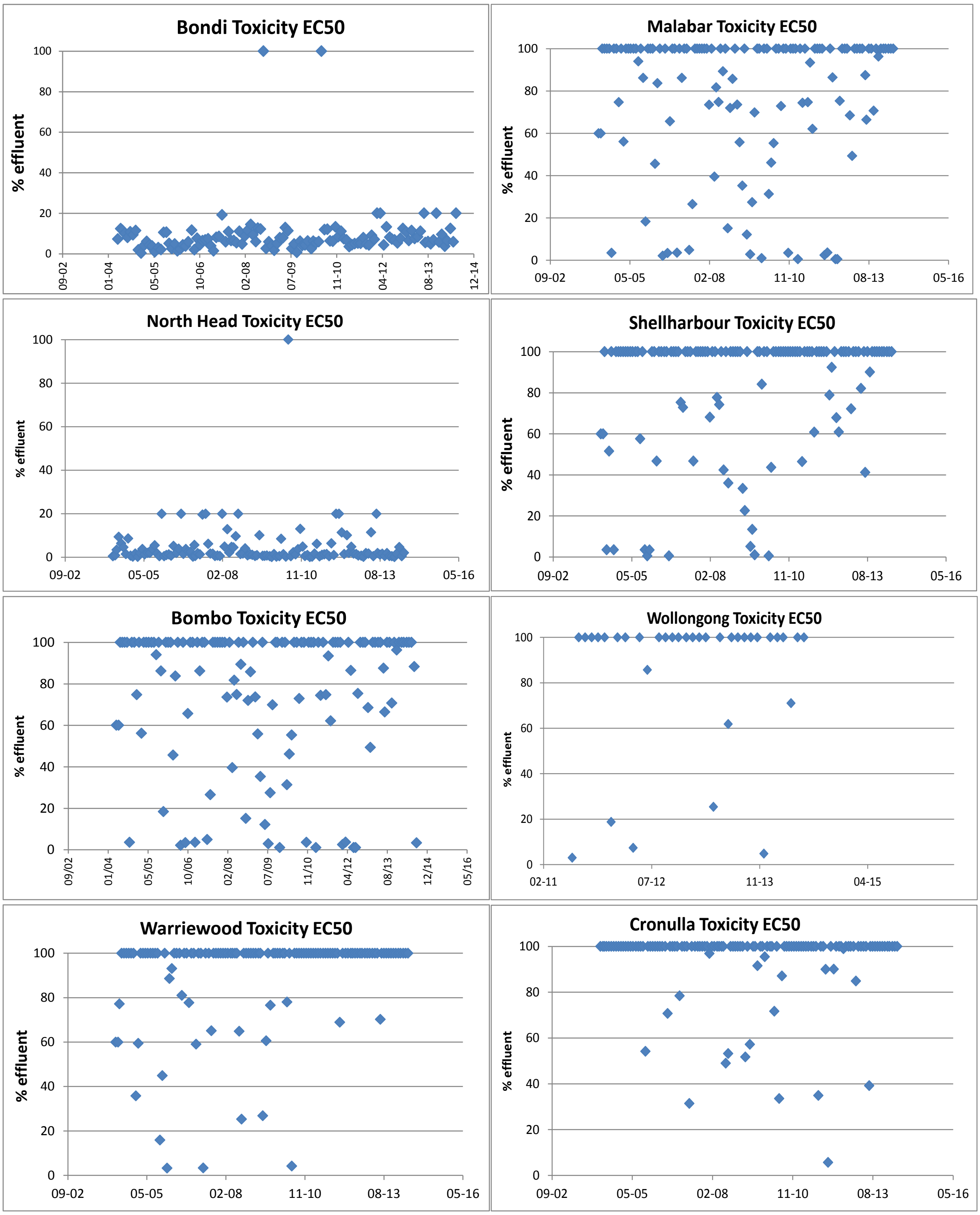


Figure 2-8 Temporal plots of sea urchin fertilisation EC_{50} toxicity from each ocean discharge plant

Conclusion

The most notable trends from the long term analysis were:

- the total phosphorus concentration in the discharge from the Winmalee and Hornsby Heights plants is increasing
- total nitrogen concentrations in the discharge from the Hawkesbury Nepean catchment plants is decreasing in response to plant upgrades. Total nitrogen concentrations are also decreasing in associated Hawkesbury Nepean River receiving waters
- suspended solids concentrations from the North Head plant are close to EPL limits
- oil and grease concentrations from the deepwater ocean outfall plants gradually increased until 2007, before steadying due to plant upgrades

Gradual increasing trends in total phosphorus concentrations at the Hornsby Heights and Winmalee plants contrasted with declining total phosphorus trends in the Hawkesbury Nepean River and steady total phosphorus trends in Berowra Creek. These trends will be monitored closely and reported in future STSIMP reports to better understand if changes are significant over the longer term or due to shorter term fluctuations.

There were no cases identified where increasing concentrations in plant discharges occurred in parallel with increasing concentrations for the same parameter in respective receiving waters. Decreasing concentrations in total nitrogen over the analysis period were observed for most Hawkesbury Nepean receiving water sites and most plants discharging to the Hawkesbury Nepean River.

A watching brief will be applied to suspended solids concentrations at North Head plant and reported on in future trend analysis studies in STSIMP reports. This is due to current concentrations being close to the EPL limit. Currently there is no statistically significant trend for suspended solids concentration at North Head, so no further action beyond closely monitoring concentrations in future is proposed.

The following action is proposed in response to increasing long term trends:

- Hornsby Heights plant – Sydney Water plans to build a new clarifier within the next 3 years to address the operational issues associated with the secondary clarifiers and tertiary filters that were not typical of the norm. The water quality will continue to be monitored to determine impacts.
- Winmalee plant – currently developing a Pollution Reduction Program to inform future upgrade planning.
- North Head – further action will be considered if the long term trend in suspended solids concentration continues to increase.

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Theme one: Treated wastewater discharges

The purpose of the 'treated wastewater discharges' theme is to better understand the effects of discharges on receiving water quality and their contribution to changes in waterway health. Where a deleterious impact is detected and a contribution to this impact from wastewater discharges can be inferred, this identifies priority areas for further assessment or management actions. Key to this theme is being able to differentiate sources of pollutants in a waterway to allow Sydney Water to better understand its contribution to the condition of waterway.

Two case studies were undertaken for this theme:

- an assessment of the effects of the St Marys Water Recycling Initiative (SMWRI) on the aquatic environment of the Hawkesbury Nepean River
- an overview of the Hawkesbury Nepean River and South Creek model and preliminary findings.

3 Assessing the impact of the St Marys Water Recycling Initiative on the Hawkesbury Nepean River

Abstract

A weight of evidence approach was taken to determine the impact of the St Marys Water Recycling Initiative (SMWRI) on the Hawkesbury Nepean River. The SMWRI takes tertiary treated wastewater from Penrith, St Marys and Quakers Hill Water Recycling Plants, and transfers it to the St Marys Advanced Water Recycling Plant to produce of high quality recycled water. This high quality recycled water is then transferred to Penrith and discharged to Boundary Creek which flows into the Hawkesbury Nepean River system. This scheme was designed to replace releases from Warragamba Dam to conserve drinking water supply, with a secondary objective of improving river health.

The potential water quality disturbance of this discharge on the river was assessed in a multidisciplinary project investigating changes in wastewater, water quality and stream health (macroinvertebrates), and included observational studies of macrophytes and fish assemblages. These five linked studies allowed for the impact from the discharge of high quality recycled water on the health of the Hawkesbury Nepean River to be assessed.

This paper presents the results from the quantitative impact assessment studies only, namely wastewater, water quality and macroinvertebrate studies.

Marked increases in discharge quantity and significant improvements in discharge quality have provided considerable environmental benefit to Boundary Creek, a small stream in poor health. The significance of these benefits in the much larger Hawkesbury Nepean River with higher levels of dilution, more complex land use patterns and more variable in-stream processes is difficult to establish. Water quality improvements through reduced nitrogen and phosphorus concentrations were evident for a short distance downstream of Boundary Creek after commissioning. However there was no measurable change in stream health according to the macroinvertebrate indicator.

Introduction

The Hawkesbury Nepean River drains a catchment of 22,000 square km and runs 470 km in length from Lake Bathurst to Broken Bay. The catchment contains a diverse range of land uses including urban, agriculture, natural landscapes, peri-urban and extractive industries and houses 800,000 people, mostly in Western Sydney. The catchment supplies water to over four million people in the Sydney basin and the Illawarra. The Hawkesbury Nepean River also provides raw water to the North Richmond Water Filtration Plant (WFP) for treatment.

The St Marys Water Recycling Initiative (SMWRI) was designed save drinking water being released from Warragamba Dam to maintain the health of the Hawkesbury Nepean River. The SMWRI provides this water by taking tertiary treated wastewater from Penrith, St Marys and Quakers Hill Water Recycling Plants (WRPs) for reverse osmosis treatment at St Marys Advanced Water Treatment Plant (AWTP). The highly treated recycled water is then piped to Penrith WRP for discharge to the Hawkesbury Nepean River via Boundary Creek. The net result is that ~20 ML/day of tertiary treated effluent previously discharged from Penrith WRP to Boundary Creek is now replaced by ~45 ML/day of highly treated recycled water. A secondary benefit expected from the scheme was an improvement in river health downstream of Boundary Creek through reduced concentrations of nutrients, particularly nitrogen (SKM 2006).

An improvement in river condition since the 1990s was documented by DECC (2009). This is primarily through reduced nitrogen and phosphorus concentrations while chlorophyll *a* levels were found to be stable or declining. Nutrient levels have remained elevated in some locations and frequently exceed guidelines for the protection of aquatic ecosystems (ANZECC 2000). It was also found that flows have decreased in the river in the last 100 years (DECC 2009).

The objective of this study was to identify and quantify changes in the water quality and stream health of the Hawkesbury Nepean River in response to increased dry weather flows brought about by the replacement of tertiary treated wastewater discharge with a greater volume of high quality recycled water.

Methods

Approach

This study used a weight of evidence approach to investigate changes, if any, in the Hawkesbury Nepean River and in Boundary Creek in response to the SMWRI. To achieve this, five separate studies were combined to form a multidisciplinary program. These studies investigated: the high quality recycled wastewater discharge; river water quality changes from altered wastewater discharge quality and stream health and macroinvertebrates. Observational macrophytes and fish surveys were also part of the larger SMWRI Aquatic Environmental Assessment Program (AEAP) but are not considered here.

A Multiple Before After Control Impact (MBACI) or BACI approach (except wastewater discharge as control sites are not feasible) was taken to assess impacts on the Hawkesbury Nepean River from the SMWRI. This approach was developed in response to difficulties in detecting impacts against the background of high variability in the observed environmental systems (Underwood 1993 and 1994). This case study focusses on the wastewater characterisation, water quality and macroinvertebrate studies as they were designed to directly and quantitatively assess impacts from the SMWRI. Details of the full Aquatic Environmental Assessment Program (AEAP) can be found in the Sydney Water Aquatic Environmental Assessment Program Baseline and Post Commissioning reports (Sydney Water 2010, 2012, 2013 and 2014).

Study area

The study area is centred above and below Boundary Creek and its junction with the Hawkesbury Nepean River. The upper limit is at Wallacia, approximately 18 km upstream of Penrith, while the lower limit is at North Richmond in the freshwater tidal reaches of the river. The area includes natural bushland, urban areas of Western Sydney, industrial, horticultural and peri-urban land uses. The SMWRI also affects South and Eastern creeks due to reduced plant discharge from St Marys and Quakers Hill plants. This case study focuses on the Hawkesbury Nepean River where the greatest change from the SMWRI was expected. Study sites are described and mapped in Table 3-1 and Figure 3-1.

Table 3-1 Water quality and macroinvertebrate sites on the Hawkesbury Nepean River

Site	Description	Site type	Easting MGA94	Northing MGA94
N67	Hawkesbury Nepean River at Wallacia Bridge	Upstream control site	281405	6250290
N57	Hawkesbury Nepean River at Penrith Weir	Upstream control site	281405	6263990
N53	Hawkesbury Nepean River at BMG Causeway	Impact site	284905	6264990
N48	Hawkesbury Nepean River at Smith St	Downstream site	283405	6271990
N42	Hawkesbury River at North Richmond	Downstream site	287705	6280790
N542	Boundary Creek upstream	Upstream control site	287129	6263811
N541	Boundary Creek downstream	Impact site	286305	6263990

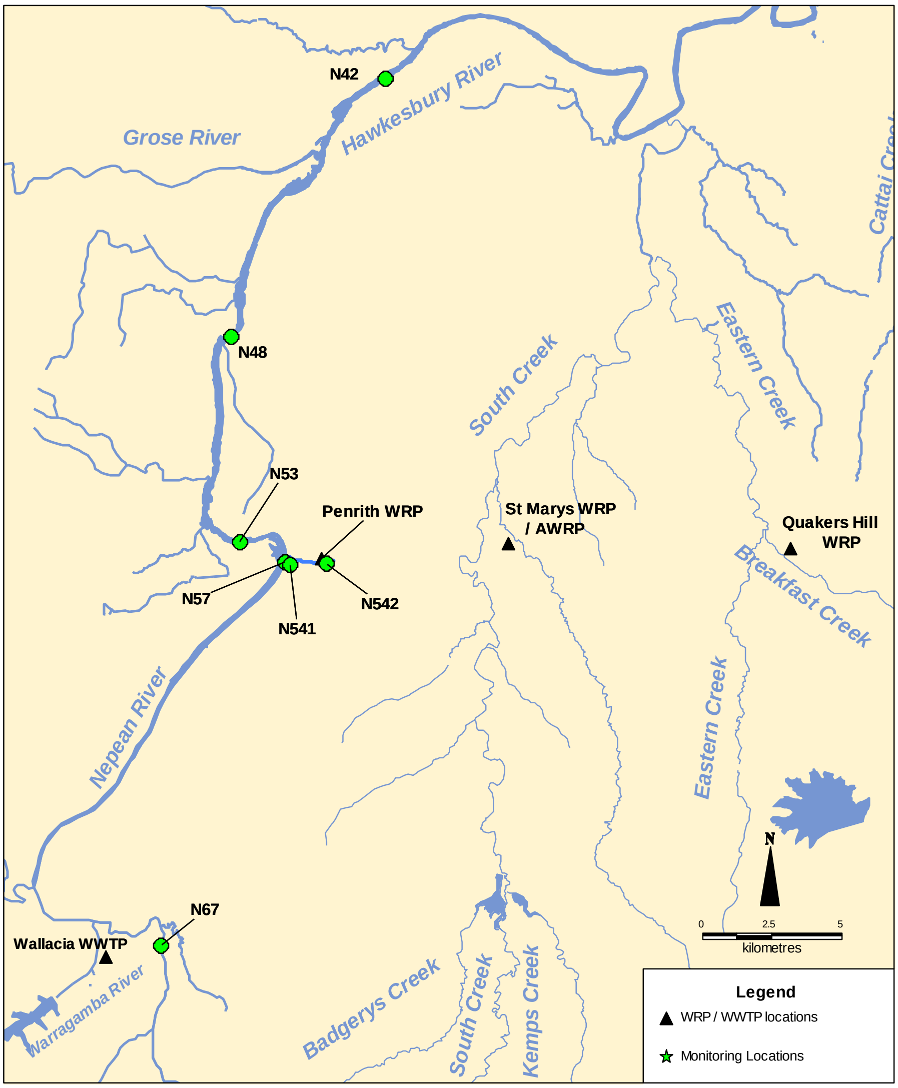

Figure 3-1 Water quality and macroinvertebrate monitoring sites

Wastewater quality

The expected outcome from the SMWRI was a significant increase in discharge volume and improvement in discharge quality to the Hawkesbury Nepean River. The wastewater quality study was set up to monitor changes in wastewater quality and quantity before and after commissioning of the SMWRI.

Monitoring program

Sydney Water monitors both quality and quantity of discharges to meet the requirements outlined in Environment Protection Licences (EPLs) issued by the NSW Environment Protection Authority (EPA). The data collected as part of the licence compliance monitoring formed the basis of this wastewater quality assessment.

Sampling frequency for wastewater discharge quality was in accordance with EPL requirements, which is either every six days or monthly. Variables monitored and their sampling frequencies are outlined in Table 3-2. For the purpose of establishing baseline conditions, data analysis was limited to the three years of data prior to commissioning of the SMWRI.

Whole sample toxicity tests were conducted on a monthly basis with the water flea, *Ceriodaphnia dubia*. The test organism was exposed to tertiary treated wastewater discharges from Penrith and advanced water treatment discharges from the St Marys AWTP. The toxicity results are reported as 48 hour median-effect concentrations (EC_{50}) (with the effect based on the immobilisation of 50% of the exposed *C. dubia*), lowest observable effect concentrations (LOEC) and no observable effect concentrations (NOEC). This is a unique case in that lower conductivity levels present increased risk of toxicity in aquatic environments. As such the LOECs reported in this paper are actually the highest conductivity levels at which a toxic effect was observed. Above this level toxic effects would not be expected. The term LOEC has still been applied in this paper to maintain consistency with the intent of the term LOEC in the wider literature.

Table 3-2 Wastewater quality parameters, sampling and analysis schedule

Parameters	Daily	Sampled once every six days	Monthly
Faecal coliforms		✓	
Enterococci		✓	
Total phosphorus		✓	
Total nitrogen		✓	
Ammoniacal nitrogen		✓	
Oxidised nitrogen		✓	
Toxicity bioassay 48 hour *Ceriodaphnia dubia*			✓
Discharge rate	✓		

Data analysis

The wastewater discharge null hypothesis (H_0) is that there is no difference between pre commissioning and post commissioning concentrations for each parameter; or, that there is no significant difference between the Penrith plant pre commissioning discharge and the post commissioning AWTP discharge.

The data analysis includes the pre commissioning period, 1 January 2007 to 31 December 2009, and the post commissioning period from 1 September 2010 to 31 August 2013. The treated wastewater quality from the Penrith plant in the pre commissioning period was compared to the highly treated recycled water from the St Marys AWTP in the post commissioning period. This represents the change in the quality of the discharge entering Boundary Creek. Nutrient and bacterial levels in the discharge were subject to statistical significance testing comparing pre to post-commissioning concentrations. Penrith WRP discharge rate and the AWTP discharge rate was plotted temporally to test if the expected change in flow occurred after commissioning. Post commissioning toxicity bioassay and conductivity results from the AWTP discharge were combined to calculate Lowest Observed Effect and No Observed Effect Concentrations (LOEC and NOEC) to test the potential toxicity of the highly treater recycled water discharge to Boundary Creek.

Where discharge quality measurements were less than the analytical limit of detection, a value equivalent to half the detection limit was used in analysis. Datasets were tested for normality and homogenous variances with data $\log_{10}$ transformed where these assumptions were not met. If these assumptions were still not met then non parametric analysis using the Mann-Whitney *U*-test for two-tailed significance (Fowler et al 2005 and Kanji 2001) was carried out. The independent samples *t*-Test was used for parametric analysis.

Water quality

Monitoring program

For water quality in Boundary Creek and the Hawkesbury Nepean River downstream of the discharge, the null hypothesis is that there was no change in concentrations of nutrients, bacterial indicators, chlorophyll *a* and conductivity due to the SMWRI, relative to changes in the upstream control sites.

The water quality monitoring program was aligned to the wastewater quality study to determine if changes in the discharge could be traced in the river, with the same parameters analysed, excluding toxicity. The water quality monitoring program involved three weekly sampling in the Hawkesbury Nepean River with replicate samples collected five minutes apart at each site on each sampling date. Further detail on the sampling and analytical methods can be found in Sydney Water (2007) 'Sydney Water Sewage Treatment Plant Compliance and Operational Monitoring Sampling Programme'.

A Multiple Before After Control Impact (MBACI) statistical design was used to compare results for each water quality variable. 'Impact' sites are located downstream of the AWTP discharge point in Boundary Creek (N541) and in the Hawkesbury Nepean River downstream of the Boundary Creek inflow (N53). Sites designated as 'downstream' sites are located further downstream than the impact site and provide an indication as to the longitudinal extent of an impact in a waterway (N48 and N42). 'Upstream' sites are located above the high quality recycled water discharge in Boundary Creek and the Hawkesbury Nepean River and are used as control sites (N57 and N67). The control sites are typically in areas affected by urban and agricultural run-off so cannot be considered as pristine reference sites.

Data analysis

Data was divided into wet weather and dry weather sampling events. The criterion for a wet weather event was daily rainfall of greater than 25 mm in the previous 72 hours. Dry weather conditions were considered to have resumed after 72 hours if a daily rainfall of no more than 2 mm had fallen in the 24 hours before sampling. Rainfall data was sourced from the Bureau of Meteorology Penrith Lakes Automatic Weather Station (BOM Climate Data Online: http://www.bom.gov.au/climate/data/?ref=ftr).

Physical and chemical processes can vary greatly between dry and wet weather flows, potentially biasing data if either the pre or post commissioning period is significantly wetter or drier than the other. Statistical analyses were performed on dry weather data, as the majority of sampling events occurred during dry weather and the SMWRI is expected to have greater effect in dry weather when discharges contribute a much larger portion of total flow in the Hawkesbury Nepean River. Due to the small number of wet weather sampling events and large variation in the number of wet weather events between periods and sites, wet weather statistical analyses were not possible.

A two-way ANOVA with replication (sites and periods) was employed for statistical analysis. Two post commissioning periods were used. These were year one and year three of the post commissioning period. This allows any post commissioning dry weather water quality trends not associated with the SMWRI to be identified. This also removes a period between January and July 2012 when flow conditions in the Hawkesbury Nepean River were not representative of the operation of the SMWRI. This was due to high flow events in early 2012 leading to an extended period of high flow in Hawkesbury Nepean River with reduced AWTP discharges and a period from April to July 2012 were AWTP discharge was reduced four fold as Warragamba Dam spilled.

The main ANOVA outcome of interest was the interaction between time periods and sites to determine when sites in a comparison had different pre to post commissioning trends. Summary statistics are presented as box plots with 5^{th}, 25^{th}, 50^{th}, 75^{th} and 95^{th} percentiles help to interpret statistically significantly ANOVA interactions. The analysis was done on $\log_{10}$ transformed data using a significance level (α) of 0.05. Replicate samples at each site on each sampling occasion were treated as independent samples in statistical analysis, with samples collected five minutes apart from different locations at a site. Small departures from normal distributions and homogenous variances were observed at most sites for most parameters. It has been shown that ANOVA is still robust in these circumstances (Sahai and Ageel 2000).

Ammoniacal nitrogen, oxidised nitrogen and field conductivity results from Boundary Creek had larger and more frequent departures from normal distributions and homogenous variances. For these the non-parametric Welch's *t-Test* was used to compare periods and sites. For this post commissioning year one and year three data were grouped together. This reduced the number of *t-Test*s required, lowering the probability of incorrect conclusions.

Macroinvertebrates

Monitoring program

The macroinvertebrate study design for the Hawkesbury Nepean River focused on two pairs of sites. The Boundary Creek control site (N542) upstream of the AWTP discharge point and the Boundary Creek impact site (N541) were paired to test the impact of the AWTP discharge on macroinvertebrate communities in Boundary Creek. In the Hawkesbury Nepean River the upstream control site at Penrith Weir (N57) and the impact site (N53), downstream of the Boundary Creek inflow were paired to test the impact of the AWTP discharge on macroinvertebrate communities in Hawkesbury Nepean River. These pairs were monitored before and after commissioning of the SMWRI allowing for a BACI analysis. Another upstream control site (N67) and downstream sites (N48 and N42) were also included for context.

Collection of macroinvertebrates was based on rapid assessment methods (eg Chessman 1995, Turak et al, 2004). Macroinvertebrates were collected in autumn and spring from available dominant habitats: pool edges; riffles (broken flowing water); macrophytes (aquatic plants); and pool rocks. Macroinvertebrate identification and enumeration was undertaken in the Sydney Water laboratory by trained laboratory analysts using the National Association of Testing Authorities (NATA) accredited in-house test method for macroinvertebrate identification and enumeration. Identification and counting was carried out up to genus taxonomic level where possible. This was done using published keys (Hawking, 2000), web links (www.mdfrc.org.au/bugguide/index.htm and www.taxonomy.org.au/), or using descriptions and reference specimens maintained by the Sydney Water Laboratory.

Data analysis

Data analysis was carried out by comparing pre commissioning data, collected between 1995 and 2009, to post commissioning data collected between spring 2010 and autumn 2013.

An analysis of stream health was carried out based on ANZECC (2000) guidelines using the Sydney region specific Stream Invertebrate Grade Number Average Level genus taxonomic version (SIGNAL-SG) biotic index (Chessman et al, 2007). The SIGNAL-SG score is simplistically an average of the sensitivity grades of the macroinvertebrate types present that also incorporates a measure of the animal counts (abundance) (Besley and Chessman 2008). The SIGNAL-SG score of each site and each period (pre and post commissioning) has been plotted with ± one standard deviation of the mean to assess if discharges from the SMWRI resulted in a new ecological equilibrium.

A two way BACI style ANOVA was used for each site pair to determine if stream health had been impacted by the SMWRI. The factor Site was comprised of two levels, upstream and downstream of the discharge point. The factor Period was also comprised of two levels, pre and post. ANOVA was used to determine if there was a difference in SIGNAL-SG scores comparing the post commissioning to the pre commissioning period.

Multivariate annotated ordination plots were used to group sample results to allow identification of similar sites and periods. BVSTEP was used to identify key macroinvertebrate data contributing to variations observed in data. SIGNAL-SG scores of these taxa were then annotated on the ordination plot to explore differences in site period groupings. Multivariate hypothesis testing was carried out with PERMANOVA as outlined in Anderson et al (2008).

The BIOENV and DISTLM routines were used to search for relationships between water quality parameters and similarities identified in biotic data. The water quality parameters identified for analysis acted as surrogates for other water quality parameters that were highly correlated with them. DISTLM output was displayed in a constrained ordination plot from the dbRDA routine.

Results

Treated wastewater quality

Changes in daily treated wastewater discharges from each of the plants affected by the SMWRI are presented in Figure 3-2. This demonstrates the reduced discharges from the St Marys, Penrith and Quakers Hill plants, and the subsequent increase in flow from the St Marys AWTP.

The increased discharge of high quality recycled water from the AWTP to the Hawkesbury Nepean River via Boundary Creek, compared to the previous tertiary treated discharges from the Penrith plant to Boundary Creek forms the basis of the hypothesis for this study.

Figure 3-2 shows the reduced discharge from the AWTP in mid 2012 in response to the spilling of Warragamba Dam. Several dips in AWTP discharge also occur in the lead up to this spill in early 2012 due to wet weather events. Discharge from the St Marys AWTP was reduced in these times to mitigate potential flood impacts in the Hawkesbury Nepean River.

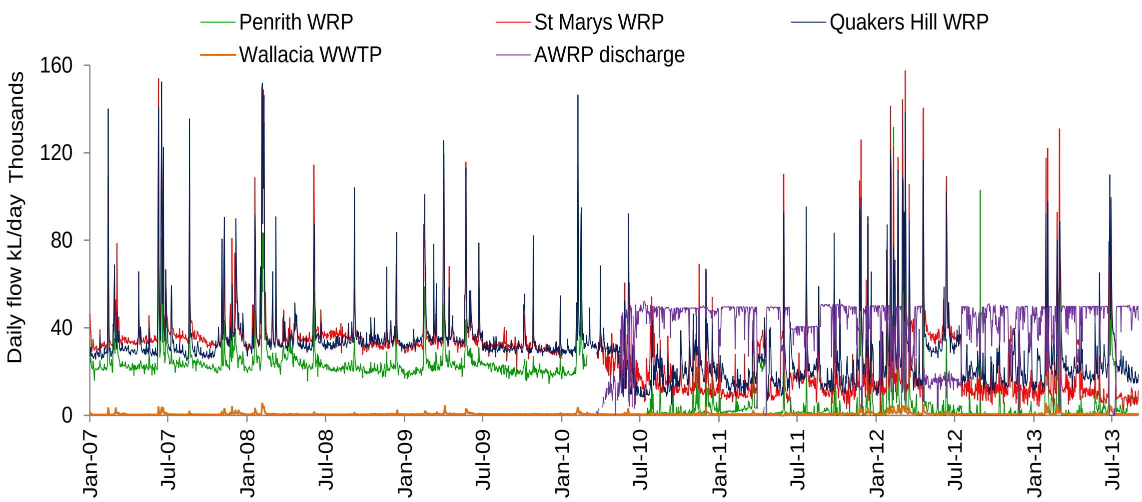

Figure 3-2 Discharges from Penrith, St Marys, Quakers Hill plants and St Marys AWTP

Univariate U and t test results comparing pre to post commissioning discharges to Boundary Creek are presented inTable 3-3. Clear reductions in total nitrogen, total phosphorus, ammoniacal nitrogen, oxidised nitrogen, Enterococci and faecal coliforms are evident, with all decreases being signficant ($p<0.01$).

Table 3-3 Results of statistical significant testing comparing the Penrith plants pre commissioning results to St Marys AWTP post commissioning results

Variable	Statistic*	Sample numbers: Pre / Post	Test statistic	p-value	Mean pre	Mean post
Enterococci	U	100 / 170	-15.223	<0.01	21	1
Faecal coliforms	U	183 / 174	-17.125	<0.01	58	1
Ammoniacal nitrogen	U	182 / 172	-6.746	<0.01	0.24	0.03
Oxidised nitrogen	t	112 / 172	46.780	<0.01	3.40	0.22
Total nitrogen	t	182 / 172	15.889	<0.01	4.47	0.28
Total phosphorus	U	182 / 172	-17.267	<0.01	0.12	0.01

The low conductivity of the high quality recycled water had the potential to induce deleterious effects on exposed aquatic organisms. Whole sample toxicity tests were conducted on a monthly basis with the water flea, *Ceriodaphnia dubia* to determine the toxicity of the AWTP discharge. The lowest observed effect concentration (LOEC) is the conductivity concentration where the high quality recycled water had a statistically significant effect on test organisms, relative to control organisms. In this unique case it is actually the highest observed effect concentration that is recorded as the LOEC, as the lower the conductivity the more likely it is to be toxic. No observed effect concentrations (NOEC) were also calculated.

A summary of LOEC and NOEC values are shown in Table 3-4. The mean LOEC value ± SD for the entire post commissioning period was 32.0±31.6 µS/cm. The standard deviation was unusually high in the third year, 49.1µS/cm, due to two elevated results on 16 October 2012 and 13 June 2013. Otherwise the LOEC ranges were between 14 and 34 µS/cm.

The overall trend for observable toxicity stress is shown in Figure 3-3 and indicates the threshold for toxicity stress is between 95 and 122 µS/cm, meaning toxicity stress is likely below this range. This indicates what conductivity levels may provide a toxic risk to aquatic biota in Boundary Creek and the Hawkesbury Nepean River due to the AWTP discharge. The ANZECC (2000) water quality guidelines for conductivity in slightly disturbed lowland rivers in South East Australia, recommends a range of 125-2200 µS/cm for the protection of aquatic ecosystems.

Table 3-4 Summary of LOEC and NOEC bioassay results for the post commissioning AWTP

	Year 1		Year 2		Year 3	
	LOEC (µS/cm)	NOEC (µS/cm)	LOEC (µS/cm)	NOEC (µS/cm)	LOEC (µS/cm)	NOEC (µS/cm)
Mean	18.5	73.9	24.8	89.9	49.4	86.0
Standard deviation	2.3	39	6.1	29.9	49.1	48.5
Mean ± Standard deviation	+20.8 -16.2	+112.9 -35.0	+31.9 -19.7	+119.8 -60.0	+98.5 -0.4	+134.4 -37.5
Overall post commissioning:			LOEC mean 32.0 ± 31.6 µS/cm		NOEC mean 83.8 ± 38.9 µS/cm	

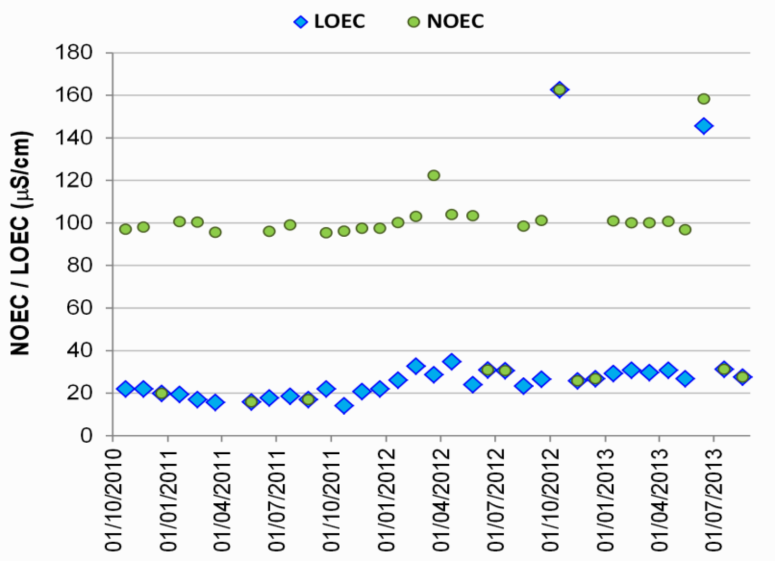

Figure 3-3 The NOEC and LOEC trend for the *Ceriodaphnia dubia* 48 hour toxicity bioassay from AWTP discharge in the post commissioning period

Receiving water quality

Nepean River and Boundary Creek summary statistics are presented in Figure 3-4, Figure 3-5, Figure 3-6 and Figure 3-7, and tabulated in Table 3-5.

In Boundary Creek, large decreases in previously elevated oxidised and total nitrogen concentrations occurred at the impact site post commissioning, bringing them in line with results from the upstream control site. This result was significant for total nitrogen (ANOVA interaction $p<0.01$), for which median concentrations decreased at the impact site from 4.73 mg/L to 0.34 mg/L. Median oxidised nitrogen concentrations at the impact site decreased pre to post commissioning from 4.0 mg/L to 0.26 mg/L, with a smaller magnitude decrease observed at the upstream control site. These trends were reflected in the Hawkesbury Nepean River immediately downstream of Boundary Creek. A significant total nitrogen ANOVA interaction ($p<0.01$) was due to decreased post commissioning total nitrogen concentrations at the impact site (N53). The downstream Hawkesbury Nepean River site at Smith St (N48) did not provide a similar trend, with a significant ANOVA interaction ($p=0.01$) for total nitrogen due to increased post commissioning concentrations at the Penrith Weir control site (N57).

Statistical analysis of the ammoniacal nitrogen concentrations was not considered appropriate due to a large number of below detection limit results and non-normally distributed data. Summary statistics indicated no clear trend in ammoniacal nitrogen concentrations in Boundary Creek pre to post commissioning. In the Hawkesbury Nepean River decreased ammoniacal nitrogen concentrations are indicated at the impact site (N53) compared to increased concentrations at the upstream control site (N67).

Pre commissioning total and filtered total phosphorus concentrations were lower at the Boundary Creek impact site (N541) compared to the upstream control site (N542). The differences increased in the post commissioning due to significant decreases in total and filtered phosphorus concentrations at the impact site, downstream of the recycled water inflow (ANOVA interaction $p<0.01$ for both total and filtered phosphorus). These changes were reflected at the impact site in the Hawkesbury Nepean River with significant ANOVA interactions ($p<0.01$) for total phosphorus and filtered total phosphorus. These ANOVA results were due to increased concentrations at the control sites contrasting with decreased concentrations at the Hawkesbury Nepean River impact site (N53). For filtered total phosphorus, the ANOVA interaction term was significant at $p<0.01$ at the next downstream site (N48) where a slight decrease in concentrations contrasted with increasing concentrations at the control sites (N57 and N67).

A significant ANOVA interaction ($p<0.01$) was observed for chlorophyll *a* in Boundary Creek. This was due to slightly reduced chlorophyll *a* concentrations at the impact site (N541) in the post commissioning period compared to increased concentrations at the upstream control site (N542) in the post commissioning. Overall the impact site had significantly lower chlorophyll *a* concentrations compared to the upstream site (ANOVA sites factor $p<0.01$). Chlorophyll *a* results in the Hawkesbury Nepean River provided no indication of an impact from the SMWRI. A significant ANOVA interaction ($p<0.01$) resulted from increased post commissioning chlorophyll *a* concentrations both upstream and downstream of the Boundary Creek inflow compared to steady concentrations at the upstream control site (N67).

Bacterial indicator densities in Boundary Creek at the impact site were significantly lower than the upstream site in the pre commissioning and post commissioning periods. The magnitude of these differences increased in the post commissioning period for Enterococci due to a significant decrease at the impact site (ANOVA sites and interaction $p<0.01$). This change in Enterococci concentrations in Boundary Creek was not reflected in the Hawkesbury Nepean River. This is despite a significant ANOVA interaction ($p<0.01$), likely due to increased concentrations at the Penrith Weir control site (N57) post commissioning compared to the impact site (N53) and upstream control site (N67).

Field measured conductivity levels were lower at the Boundary Creek impact site in both the pre commissioning and post commissioning periods. A highly significant post commissioning reduction in conductivity levels occurred at the impact site (median 779 µs/cm to 32 µs/cm, $p<0.01$ using Welch's t test). There was no significant change in conductivity levels at the upstream control site post commissioning, with highly variable conductivity levels observed. Conductivity levels decreased at all sites in the Hawkesbury Nepean River post commissioning indicating other processes in the river were impacting water quality in the study period. ANOVA returned a significant interaction ($p<0.01$), likely due to the Wallacia Bridge control site (N67) recording a large reduction from pre to post commissioning. All conductivity results in the Hawkesbury Nepean River were above the recommended minimum of 125 µS/cm trigger value for the protection of aquatic ecosystems. This indicates that reduced conductivity levels in Boundary Creek due to the SMWRI will not likely impact on the biota of the Hawkesbury Nepean River.

Table 3-5 Summary of results from the SMWRI Aquatic Environmental Assessment Program (AEAP) for the Hawkesbury Nepean River and Boundary Creek

Key indicators	Boundary Creek		Hawkesbury Nepean River		
	AWTP discharge (pure recycled water)	Impact site: Lower Boundary Creek (N5401)	Impact site: Hawkesbury Nepean River 1.2 km downstream (N53)	Downstream site: Hawkesbury Nepean River ~10 km downstream (N48)	Downstream site: Hawkesbury Nepean River ~20 km downstream (N42)
Nutrients	Clear positive impact	Clear positive impact	Clear positive impact	No impact detected	No impact detected
Chlorophyll *a*	No study required	Positive impact likely	No impact detected	No impact detected	No impact detected
Bacterial	Clear positive impact	Positive impact likely	No impact detected	No impact detected	No impact detected
Conductivity	No study required	Clear negative impact	No impact detected	No impact detected	No impact detected
Macroinvertebrates	No study required	Clear positive impact	No impact detected	No impact detected	No impact detected

Legend *

Clear positive impact[1]	Positive impact likely[2]	No impact detected[3]	Negative impact likely[4]	Clear negative impact[5]	No study required [6]

1 Clear positive Impact – positive statistically significant change in water quality or ecological parameter that is attributable to the Project

2 Likely positive impact – a positive change in water quality or ecological parameters is detected after commissioning but is statistically too small to be clearly attributable to the Project

3 Minimal change – no change evident (positive or negative)

4 Likely negative impact – a negative change in water quality or ecological parameters is detected after commissioning but is statistically too small to be clearly attributable to the Project

5 Clear negative impact – negative statistically significant change in water quality or ecological variable that is attributable to the project

6 No study possible or required by the AEAP

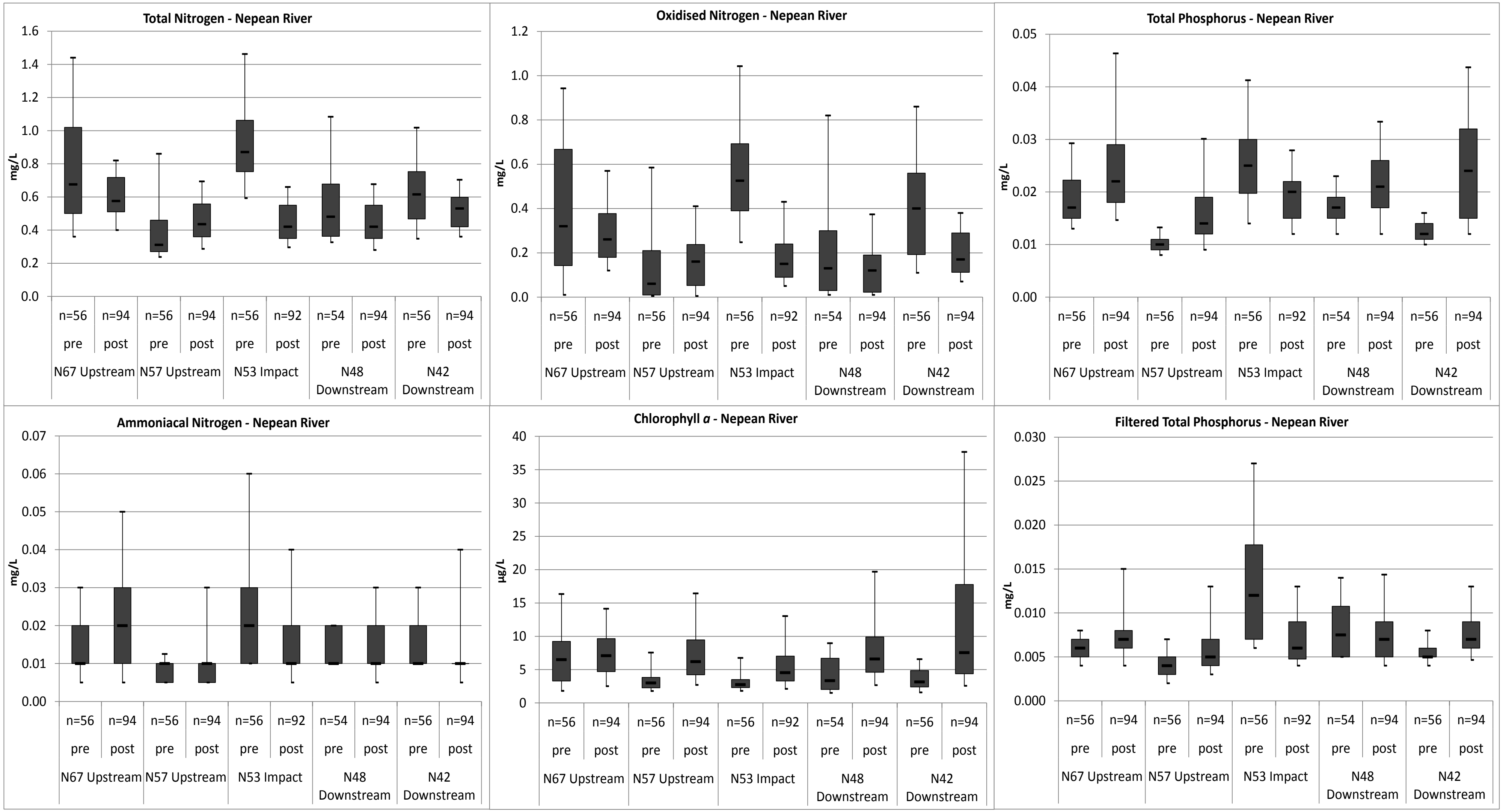

Figure 3-4 Hawkesbury Nepean River dry weather summary statistics for each site for nutrient parameters and chlorophyll *a* comparing pre commissioning to the post commissioning

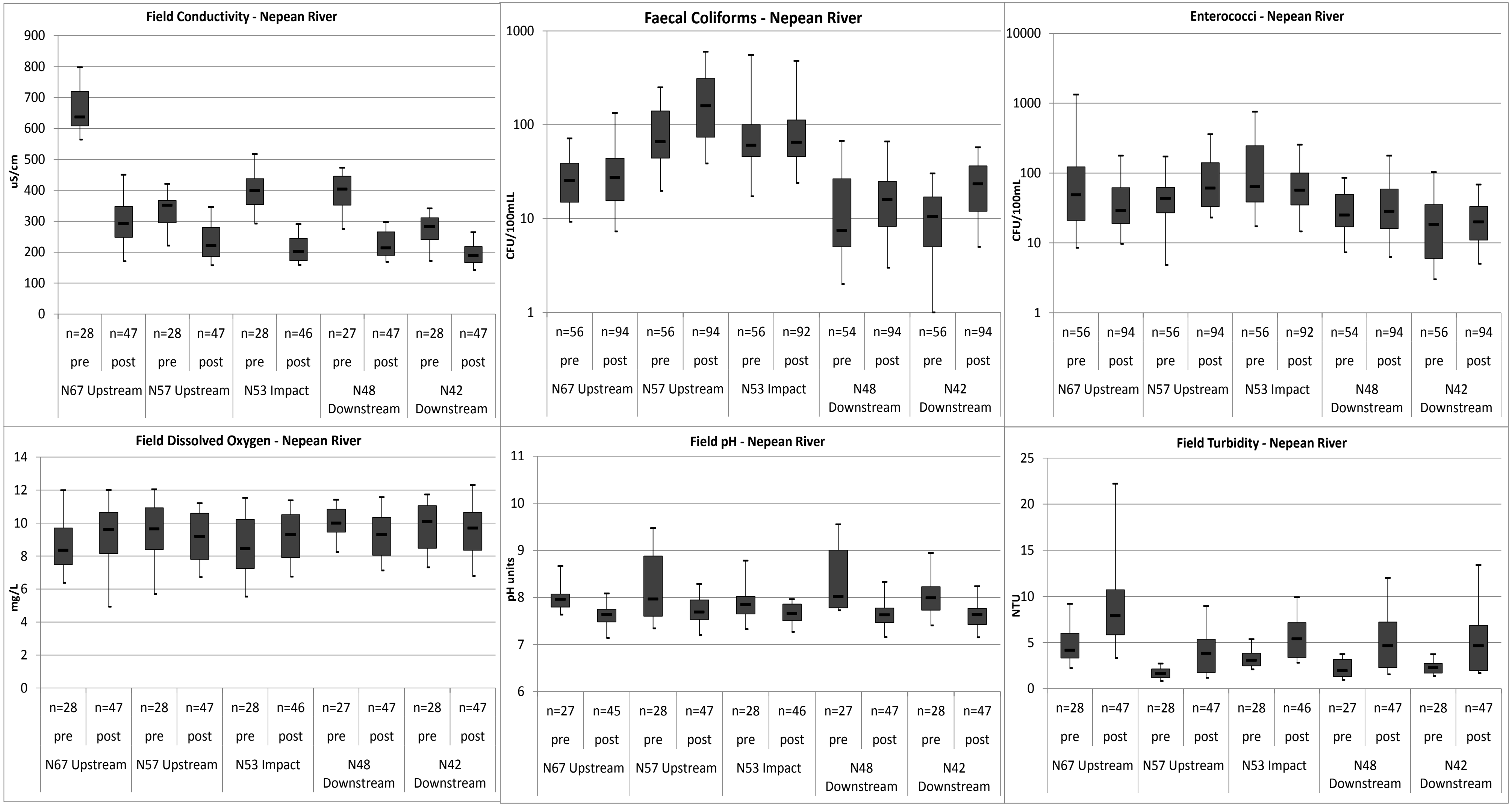

Figure 3-5 Hawkesbury Nepean River dry weather summary statistics for each site for bacterial indicators and field measured parameters comparing pre commissioning to the post commissioning

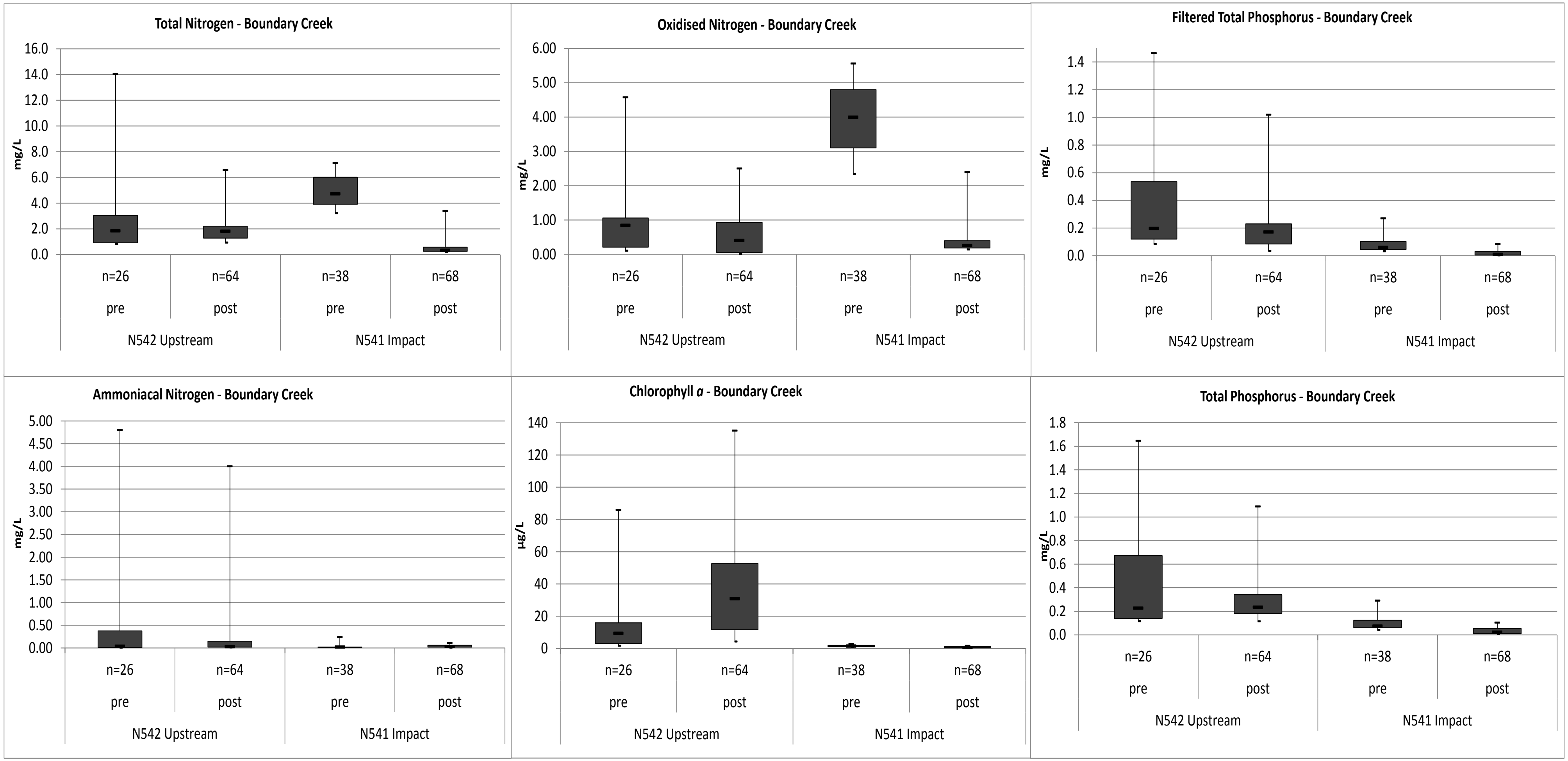

Figure 3-6 Boundary Creek dry weather summary statistics for each site for nutrient parameters and chlorophyll *a* comparing the pre and post commissioning periods

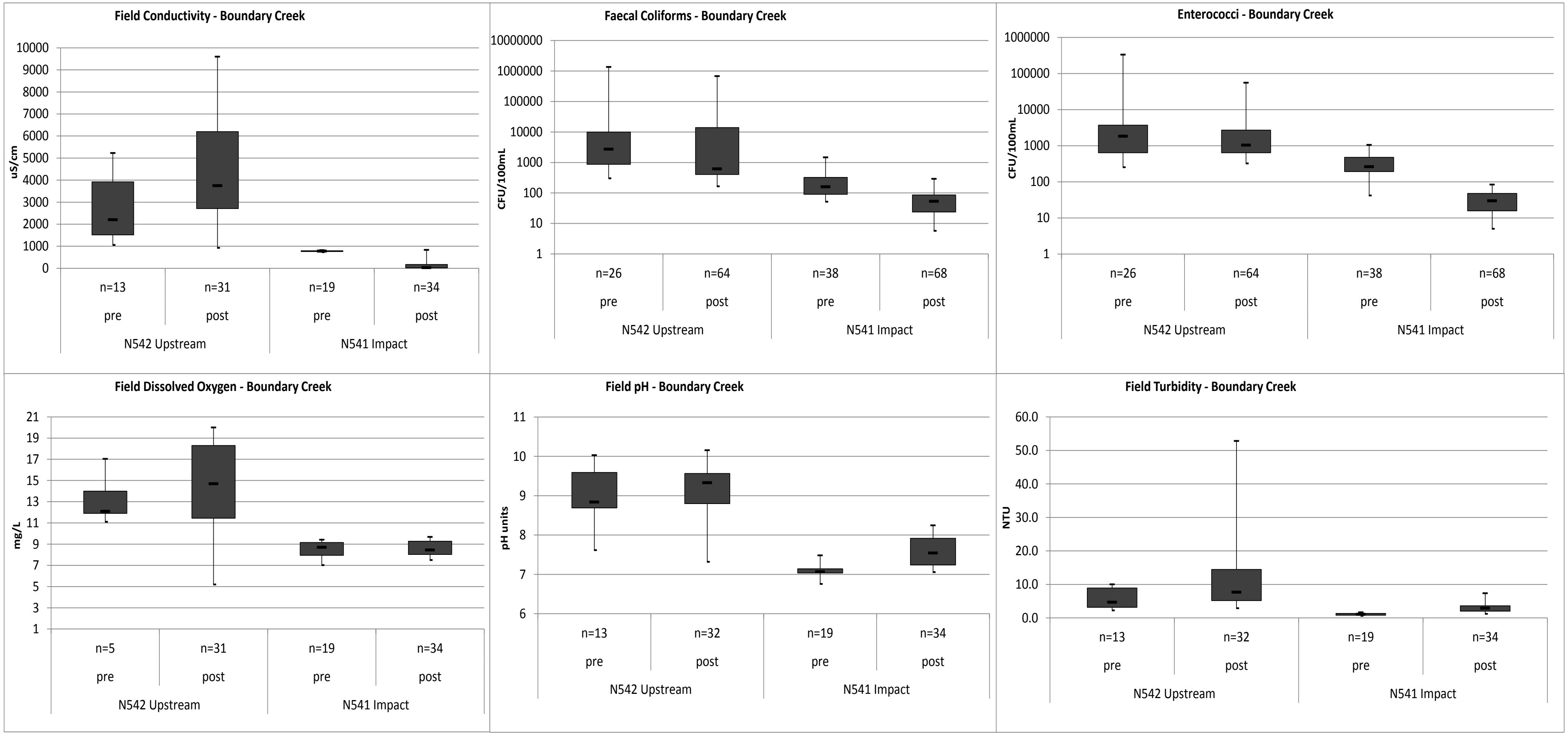

Figure 3-7 Boundary Creek dry weather summary statistics for each site for bacterial indicators and for field measured parameters comparing the pre and post commissioning periods

Macroinvertebrates

ANZECC 2000 assessment of SIGNAL_SG (used to assess stream health)

The direct connection between a stream and sources of surface runoff in urban and rural streams allow even small rainfall events to produce detectable impacts on stream health upstream of plants. As such, upper catchment stream health may limit downstream stream health in urban and rural streams. It is against this background that potential stream health changes from the SMWRI were assessed.

Pre commissioning stream health at the upstream Boundary Creek site was similar to post commissioning stream health. In contrast, stream health of the Boundary Creek impact site improved post commissioning. High variability was observed in year 3 (2012/13) of the post commissioning period (Figure 3-8). Field observations from autumn 2012, spring 2012 and autumn 2013 indicated aquatic plants (macrophytes) had been scoured out from the impact site, likely due to a period of high rainfall in early 2012. This loss of habitat may have influenced the increase in the underlying taxonomic variability seen in latter SIGNAL-SG scores from the downstream site. Stream health did not change from pre to post commissioning in the Hawkesbury Nepean River, indicating that the positive impact on stream health in Boundary Creek did not extend to the Hawkesbury Nepean River (Figure 3-8).

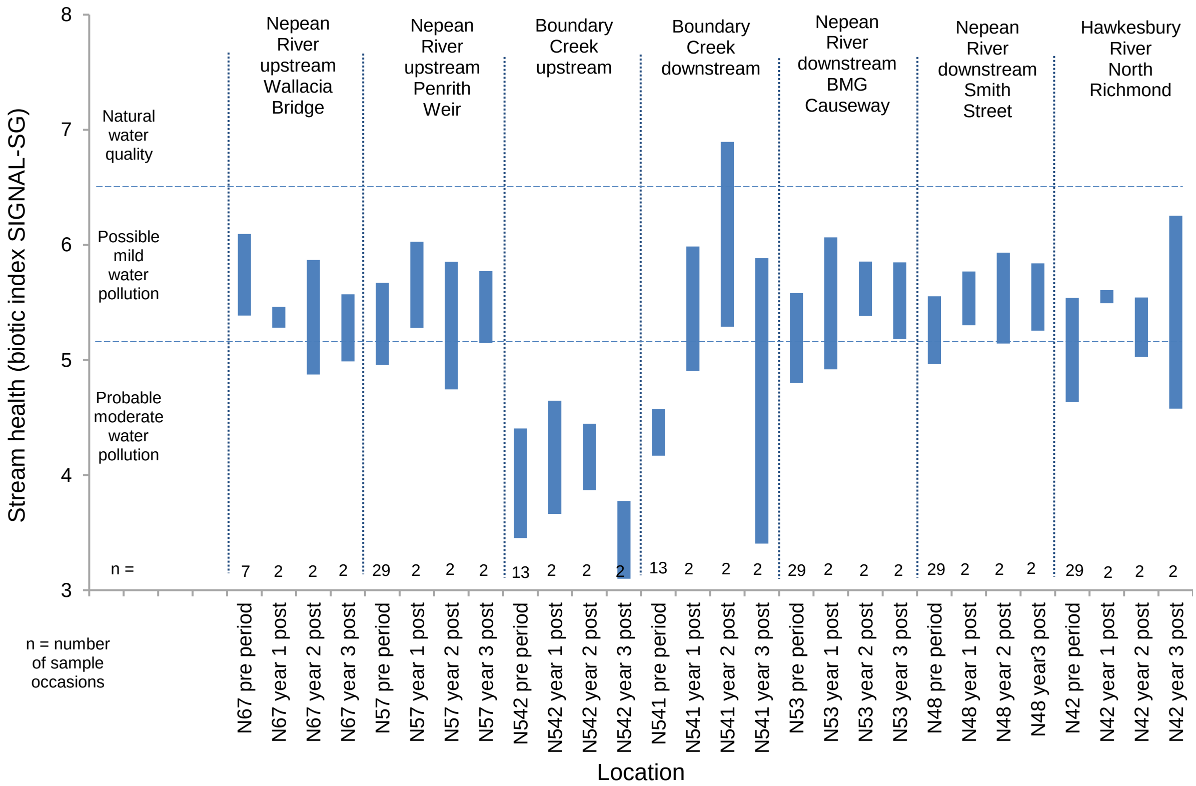

Figure 3-8 SIGNAL_SG scores at each site in the study area from pre commissioning and each year of post commissioning monitoring

Analysis of variance

'Period X Site' interaction test results from ANOVA indicated stream health was similar in the pre and post commissioning periods for the Hawkesbury Nepean River comparison. Changes observed in SIGNAL_SG scores in Boundary Creek stream health (Figure 3-8) were confirmed by the significant 'Site X Period' interaction (Table 3-6).

Table 3-6 'Period x Site' interaction result from ANOVA of all habitat SIGNAL-SG scores on each pair of sites

Plant and waterway	df	MS	F value	P value
Penrith WRP - Hawkesbury Nepean River	1	0.26047996	1.88	0.1714
Penrith WRP – Boundary Creek	1	3.76682931	11.04	**<0.0016**

Exploration of Boundary Creek with multivariate statistics

To further explore if the change detected in Boundary Creek between pre and post commissioning periods reflected a real change in taxonomic composition of the macroinvertebrate community, multivariate statistical analysis techniques were applied.

An MDS ordination plot of results is presented in Figure 3-9 to identify groups of samples. The pre commissioning samples from the impact site (N541) formed one distinct group of samples in the ordination plot. Another distinct group of samples was formed from pre and post commission period samples from the upstream site (N542).

The remaining four groups of samples in the ordination plot (Figure 3-9) were from the impact site during post commissioning, indicating a different taxonomic composition to upstream site samples and pre commissioning impact site samples. These four groups indicated that samples from the impact site had more variable taxonomic composition in the post commissioning period. The annotation of seasons and years onto the plot reflects greater variability in taxonomic composition after aquatic plants had been scoured out from the impact site in early 2012. Despite the variability displayed in the ordination plot for the impact site, this plot did confirm a real change in taxonomic composition occurred in Boundary Creek in response to the SMWRI.

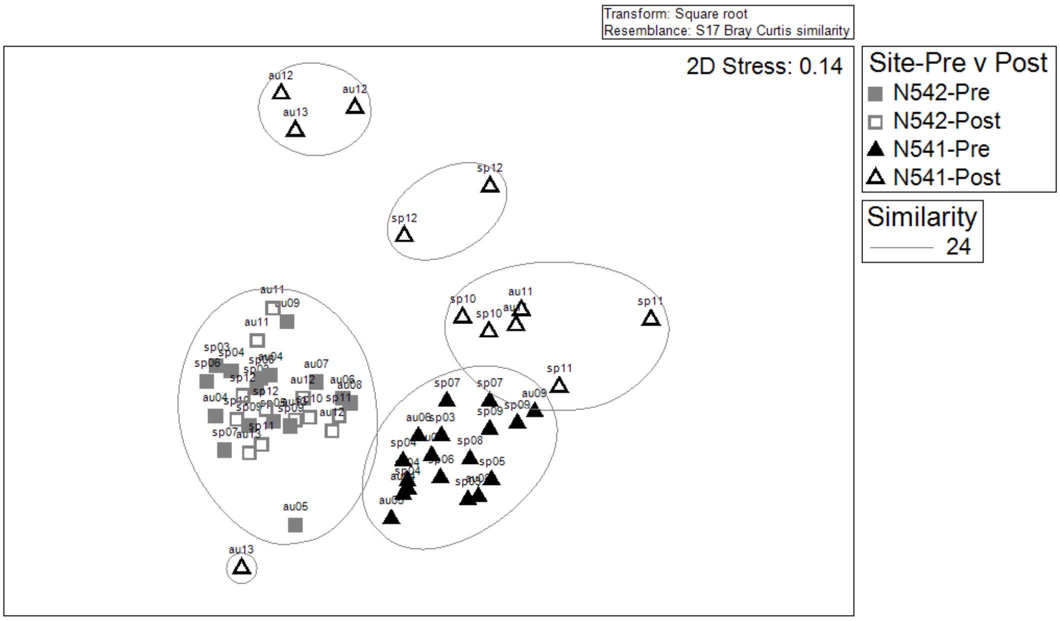

Figure 3-9 MDS ordination plot of macroinvertebrate samples with autumn 2010 samples omitted and first three classification groups overlaid. au = autumn; sp = spring

Macroinvertebrate biota with significant correlations (by the BVSTEP routine) were overlaid onto the ordination plot. Also annotated onto this plot were SIGNAL-SG sensitivity grades. The results in Figure 3-10 help explain the post commissioning increase in SIGNAL-SG scores from the impact site, as dominant taxa in these samples were macroinvertebrates with relatively higher SIGNAL-SG sensitivity grades.

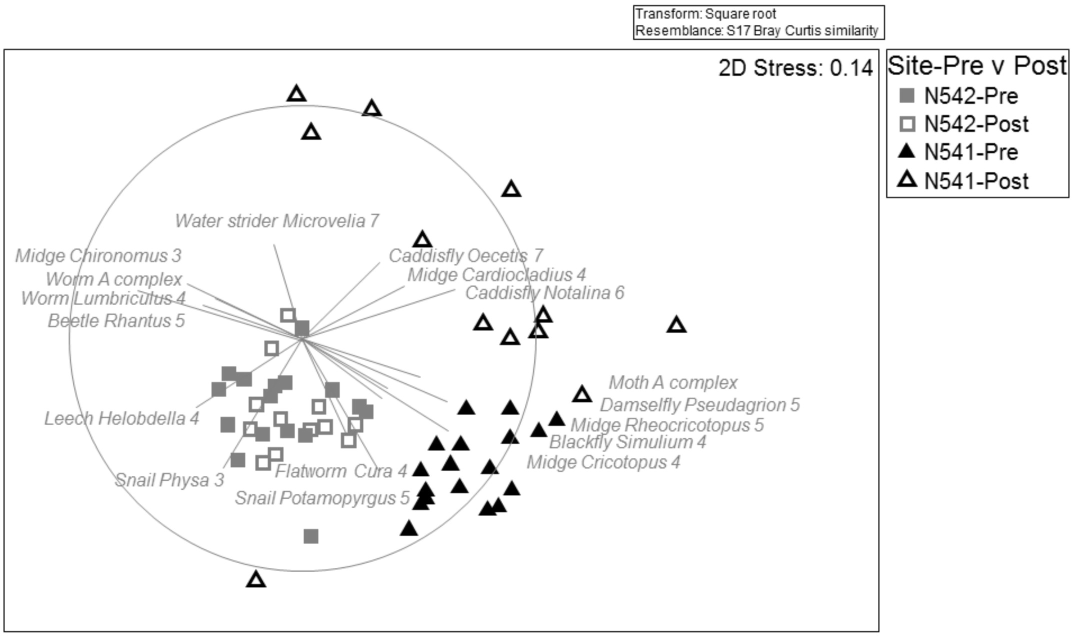

Figure 3-10 MDS ordination plot of macroinvertebrate samples with correlation vectors of taxa identified under BVSTEP overlaid and SIGNAL-SG sensitivity grades annotated

Hypothesis testing of macroinvertebrate community composition under PERMANOVA yielded results similar to ANOVA with a significant 'Site x Period' interaction (df = 1, MS = 10723, Pseudo F = 5.3123 P perm = 0.0001). Pairwise tests on the 'Period' factor within the 'Site' factor compared 'Pre' versus 'Post' groups of macroinvertebrate samples and were not significant for the upstream site (t = 1.3135, P perm = 0.0614) but were significant for the impact site (t = 2.7273, P perm = 0.0001).

Macroinvertebrate and water quality data compared in the BIOENV and DISTLM routines helped assess if water quality parameters were potentially responsible for structuring the observed biotic patterns.

To account for multi-collinearity within the water quality data, conductivity, Enterococci, faecal coliforms and total phosphorus (which were negatively correlated with flow at the macroinvertebrate habitat level) were omitted. Filterable phosphorus was omitted as it was well correlated with total phosphorus and faecal coliforms. Dissolved oxygen was omitted as it was well correlated with pH. Chorophyll *a* was omitted as it was correlated with turbidity.

Of the five parameters input into the BIOENV routine, flow at the macroinvertebrate habitat level best explained the macroinvertebrate sample pattern and by implication the correlated omitted parameters (listed above for flow). The next best combination of parameters was flow with total nitrogen. Correlations in both cases were only moderate (0.5 to 0.6).

DISTLM output is displayed in the constrained ordination plot in Figure 3-11. Flow was most correlated with axis 1 of the plot while total nitrogen was most correlated with axis 2. However, due to parameters being omitted to take account of multi-collinearity in this statistical test, the omitted parameters may potentially explain the biological community pattern. The total variation of the first two axes explained about a third (36%) of the inherent variation in the macroinvertebrate resemblance matrix.

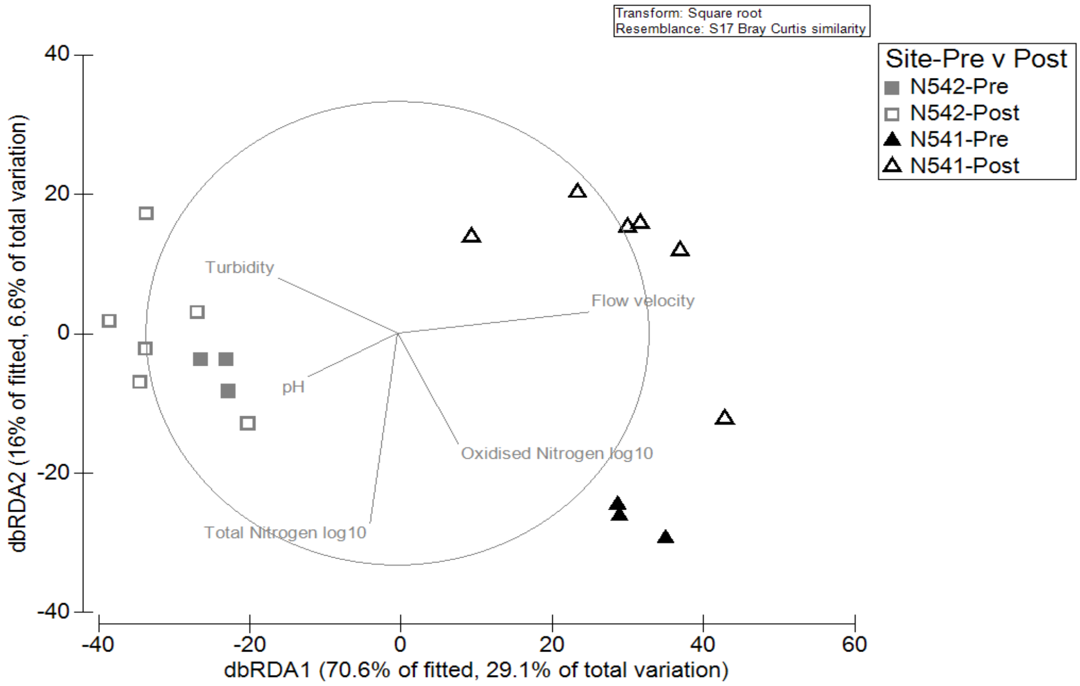

Figure 3-11 dbRDA ordination plot of macroinvertebrate samples based on DISTLM output with water quality correlation vectors overlaid

Discussion

This project was designed to provide a weight of evidence approach to identify change from the SMWRI on the water quality and aquatic ecology of the Hawkesbury Nepean River. It was also designed to allow individual studies to be linked to provide inferences about causes of change.

The wastewater quality study clearly identified the expected increase in volume of discharge to Boundary Creek and the significant improvement in discharge quality through reduced concentrations of nitrogen, phosphorus and bacterial indicators.

Boundary Creek macroinvertebrate data confirmed a positive change in the downstream community structure due to the SMWRI. The macroinvertebrate study indicated that flow (and by correlation total phosphorus, bacterial indicators and conductivity) and total nitrogen accounted for just over a third of the variation measured in the macroinvertebrate communities. This included lower total nitrogen concentrations post commissioning. These results, considered with the wastewater quality and water quality study, indicate a link between the SMWRI and the positive change in stream health observed in Boundary Creek post commissioning. Improvements in stream health in the Hawkesbury Nepean River downstream of the Boundary Creek inflow were not detected post commissioning.

The very low conductivity levels of the high quality recycled water discharged from the AWTP was potentially toxic to the receiving water. Results from this study indicate the low conductivity water observed in Boundary Creek post commissioning (median 32 µs/cm) did not have a significant impact on the downstream aquatic biota. This was demonstrated by the improvement in stream health shown by the macroinvertebrate indicator in Boundary Creek (Sydney Water 2014). In the Hawkesbury Nepean River, post commissioning conductivity levels below the Boundary Creek inflow were within the recommended ANZECC (2000) guideline for the protection of aquatic ecosystem. Conductivity levels in the Hawkesbury Nepean River also changed significantly in the study period in response to other processes occurring in the upper Hawkesbury Nepean River. In 2010 an environmental flows regime from the metropolitan water supply reservoirs (Cataract, Cordeaux, Nepean and Avon) was instigated. It is possible this project has altered conditions in the river since commencement.

This paper identifies spatially limited improvements in water quality and stream health due to the SMWRI. Other changes outside the area of influence of the SMWRI, such as increased chlorophyll *a* levels in Penrith Weir and decreased conductivity levels upstream of the Boundary Creek inflow, indicate other catchment factors are influencing the health of the Hawkesbury Nepean River.

Conclusion

Marked increases in discharge quantity and significant improvements in discharge quality have provided considerable environmental benefit to Boundary Creek, a small stream in poor health. The significance of these benefits in the much larger Hawkesbury Nepean River with higher levels of dilution, more complex land use patterns and more variable in stream processes is difficult to establish. The Hawkesbury Nepean River downstream of the recycled water inflow had significantly reduced total nitrogen and total phosphorus concentrations within a limited spatial extent (~1 km). There were no detectable impacts (positive or negative) on the aquatic ecology of the Hawkesbury Nepean River.

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4 Hawkesbury Nepean River and South Creek model: a powerful tool to inform management decisions in the Hawkesbury Nepean catchment

Abstract

Sydney Water operates 15 plants that discharge into the Hawkesbury Nepean River system, one of the largest river/estuary systems in NSW. With this comes a responsibility to minimise impact on the environment, while maintaining an affordable high quality service for customers. Considerable urban growth is planned for the Hawkesbury Nepean catchment over the next 30 years to accommodate Sydney's growing population. New water and wastewater services will be required. To plan for the most efficient and effective service for customers while protecting the environment requires a holistic understanding of the various impacts on the waterway and the interrelationships between them.

To address this knowledge gap, a water quality and hydrodynamic model of the Hawkesbury Nepean catchment has been developed. The model provides guidance on likely changes in water quality and quantity when testing different catchment, environmental flow, wastewater and landuse options over time. It provides the ability to differentiate between diffuse and point sources of pollution, and better understand the impact of wastewater treatment plant discharge in wet compared to dry weather conditions, and the complex interactions within such a large river system. This understanding will guide future expenditure to provide the maximum benefits to both the community and the environment.

The Hawkesbury Nepean River and South Creek model will provide scientific evidence to support future management and investment decisions for river managers, regulators and users.

Introduction

The Hawkesbury Nepean River and South Creek model (the model) was built to provide Sydney Water with the ability to compare and interpret different options for urban development and wastewater treatment plant discharges to the Hawkesbury Nepean River and South Creek waterways. The model simulates hydrology, hydrodynamics and biogeochemical processes to examine water quality benefits (or impacts) resulting from different scenarios across broad spatial and temporal scales. The model extends from Warragamba Dam on the Warragamba River, and Pheasants Nest and Broughton Pass weirs downstream of the Upper Nepean dams, to the ocean, covering an area of 12,000 km^2. A map showing the model domain is presented in Figure 4-1.

Sydney Water's main objectives in developing the model were to:

- provide science based evidence to inform our discussions about our environment protection licence requirements with the Environment Protection Authority; and
- inform the planning process for the North West and South West growth sectors (a future investment of a minimum of $2.5 billion).

Other NSW government agencies also have a vested interest in the model. Their drivers are to inform the Warragamba Dam environmental flow decision and the 2015 Metropolitan Water Plan review.

The Hawkesbury Nepean River is an iconic waterway of Sydney, with the catchment supporting a population of 800,000 people and providing nearly all of the drinking water to four million people living in Sydney, the Illawarra and the Blue Mountains. It has high economic value in terms of its recreational opportunities, agricultural and fisheries produce, as well as tourism and mining resources for the Sydney Metropolitan area (DECCW 2010). However these activities place considerable pressure on the Hawkesbury Nepean River system and need to be managed effectively if river health is to be protected and/or enhanced (HRC 1998).

In addition to these pressures, major urban growth has been planned for the Hawkesbury Nepean catchment over the next 30 years. These are expected to place further demand on the rivers' resources.

Figure 4-1 Hawkesbury Nepean River and South Creek model domain (shaded area) (SKM 2014b)

Background

The previous in-stream water quality model for the Hawkesbury Nepean River system (SALMON-Q) was developed for Sydney Water in the 1990s. This one-dimensional longitudinal model had basic water quality functionality, with some in-stream microbiological capability (related to primary productivity of benthic and planktonic algae). The key driver for the model at the time was high wastewater treatment plant nutrient discharges and prevalent algal blooms in the Hawkesbury Nepean River. Based in part from output from SALMON-Q an extensive upgrade program was implemented for all inland wastewater treatment plants. Following implementation of the major upgrades, the SALMON-Q platform was used infrequently, and the quality of model output became questionable as the calibration became outdated. SALMON-Q was also much restricted in its spatial extent and not applicable to the estuarine section of the river.

In 2008, the need for a water quality and quantity model of the Hawkesbury Nepean River resurfaced. This time the drivers were to assess the impacts of various activities planned for the Hawkesbury Nepean River catchment, such as:

- implementation of Metropolitan Water Plan initiatives, particularly environmental flow releases
- understanding the impacts of discharges from wastewater treatment plants and the benefits of treatment upgrades

- planning for growth and service delivery in the North West and South West sectors
- understanding the impacts of point source discharges and catchment runoff, as well as the effects of improvement activities to both
- ensuring the benefits of past investments are verified and recognised in the longer term.

Model build commenced in 2011, taking over three years to complete. This included extensive data collection (collating existing data and undertaking targeted campaign monitoring programs), as well as model calibration/validation. The Hawkesbury Nepean River and South Creek Model was installed on Sydney Water computers in early 2014.

The model was developed for Sydney Water by Sinclair Knight Merz Pty Ltd (SKM, now known as Jacobs) in partnership with BMT WBM, eWater, UWA and Yorb, and was reviewed by the Commonwealth Scientific and Industrial Research Organisation (CSIRO). Extensive data sets were provided by the NSW Office of Water, Office of Environment and Heritage, Sydney Catchment Authority, Manly Hydraulics Laboratory, Land and Property Information, Bureau of Meteorology, Hornsby Shire Council, Penrith City Council, The Hills Shire Council, Blacktown Council and Camden Council. These data were critical for building and calibrating the model.

Campaign monitoring programs

An assessment of existing data and their suitability for calibrating, validating and running the numerical models was undertaken prior to building the model. The existing data was collated from Sydney Water's extensive dataset as well as from other NSW state and local government agencies. Critical data gaps were identified. Targeted campaign monitoring programs were established to fill these critical data gaps.

They included:

- bathymetry surveys
- water current velocity profiles
- wet weather event water quality monitoring using autosamplers
- baseline dry weather water quality monitoring
- total and dissolved organic carbon measurements
- macrophyte surveys.

Bathymetry surveys of the river bed shape and depth were required to build the model mesh. Over 210 km of river was surveyed by Sydney Water between the Upper Nepean catchment and Spencer, and in South and Eastern creeks between January 2011 and March 2012. Bathymetry data was also obtained from other NSW government agencies. Historical bathymetry data was used for the estuary.

An Acoustic Doppler Current Profile (ADCP) study was undertaken to capture water velocity profiles at six sites between Wilberforce and the lower estuary. This study was critical to inform the advection and dispersion coefficients in the hydrodynamic model, and in turn, allow better replication of the physical processes which influence water quality. Two surveys were conducted - one on a spring tide (November 2011) and one on a neap tide (December 2011). Each survey was conducted continuously over a full ebb-flood tide cycle (~14 hours). Physico-chemical water quality profiles were measured from the thalweg during the surveys.

Autosamplers were setup at six sites in the Hawkesbury Nepean River catchment to measure stormwater runoff concentrations. The specific landuse types targeted were forested; rural/peri-urban; and urban. This study aimed to better understand the variability in water quality during a high flow event. Higher concentrations tend to occur early in an event (on the rising limb, often referred to as the “first flush”). This variability is vital for deriving accurate loads of water quality constituents from the model. Flow was recorded at each site to understand the relationship between water quality and flow related to each landuse. Dry weather data at the two forested catchment sites on the Colo and Grose rivers were limited and additional sampling was undertaken to supplement the data

Total and dissolved organic carbon data was required as a precursor to processes such as nitrification and denitrification in the water quality model. Due to the general lack of carbon data, an intensive monitoring program was established involving the collection of 220 samples from 28 sites over a six month period. This data was critical to inform the calibration of the detailed water quality model for the tidal and non-tidal reaches of the river system.

A macrophyte campaign monitoring program was implemented to assess spatial and temporal attributes of key macrophytes at four locations on the Hawkesbury Nepean River between Penrith Weir and North Richmond. The program also aimed to provide a better understanding of the relationship between macrophyte assemblages and hydraulic processes at these locations. One of the key macrophytes studied was *Egeria densa*, an introduced species that is rapidly spreading throughout the Hawkesbury Nepean River. Seven surveys were undertaken between November 2011 and January 2013. The data was used to inform the macrophyte ecological model.

Model concept and structure

The Hawkesbury Nepean River and South Creek model comprises four linked models: Source catchment model; TUFLOW FV (Three dimensional, Unsteady FLOW, Finite Volume) hydrodynamic model; Aquatic EcoDynamics (AED) water quality model; and EcoModeller macrophyte model. A conceptual diagram of how the models work together is shown in Figure 4-2.

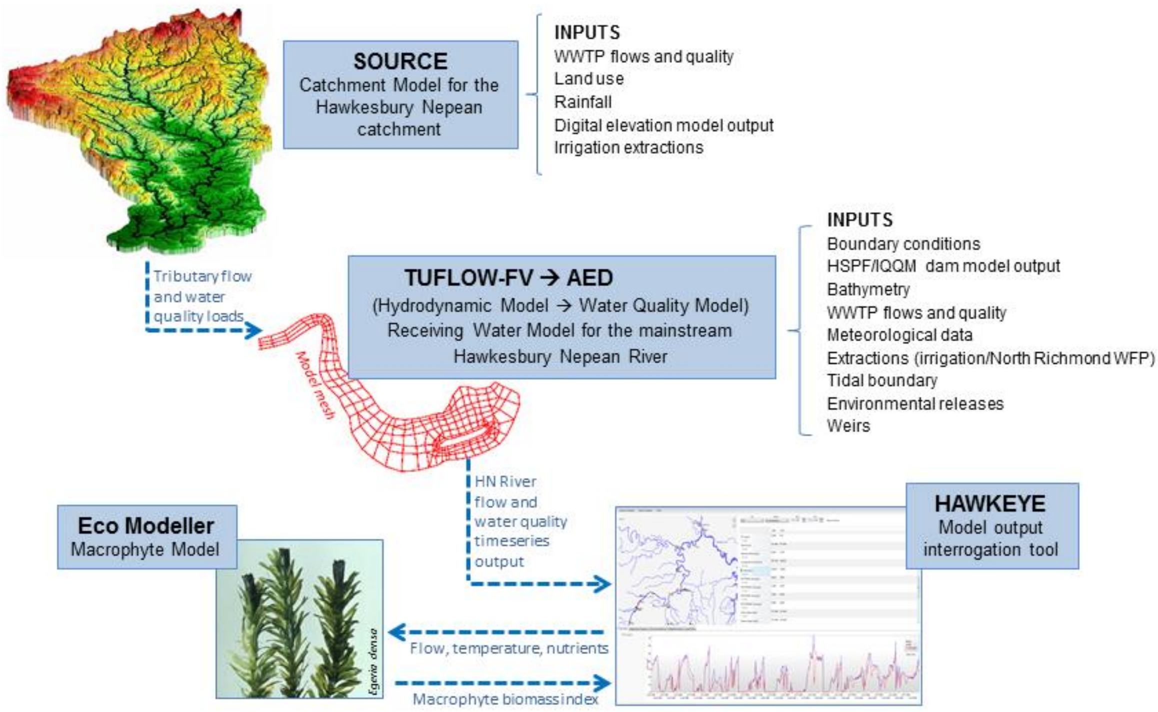

Figure 4-2 Hawkesbury Nepean River and South Creek model schematic (modified from BMT WBM 2014)

The Source catchment model is used to simulate the generation of flows and water quality constituent loads from the catchments that feed into the Hawkesbury Nepean River. It works by modelling the catchment as a series of nodes interconnected with links. This sequence of nodes and links forms a network in which water and materials are transported. The Source model also uses the concept of functional units. Functional units are areas within a subcatchment that have similar behaviour in terms of runoff and/or constituent generation. Functional units are based on combinations of landuse or cover (e.g. forest, crops, and urban areas), management activities, position in the landscape (flat, hillslope and ridge) and/or soil type. There are 555 subcatchments in the Source model, each of which comprise one or more functional units (SKM 2014a).

Other inputs into the Source model include tributary stream flow and water quality, rainfall data from 478 rainfall stations, plant discharge and irrigation extraction (SKM 2014a).

The daily time series of flows and water quality loads generated at the end of each of the tributary catchments in the Source model are inputs into the TUFLOW FV/AED hydrodynamic and water quality model.

The TUFLOW FV hydrodynamic model solves non-linear shallow water equations on a flexible mesh using a finite volume numerical scheme (BMT WBM 2014). TUFLOW FV uses the flows generated by the Source model and discharged from the dams (Warragamba Dam and the Upper Nepean dams, as represented by flow passing through Pheasants Nest and Broughton Pass weirs) to simulate the hydrodynamics of the river in three dimensions. Other key components of the TUFLOW FV model include tidal levels, flows, salinity and meteorological data to produce velocity, depth, salinity and temperature across the river system. Wastewater treatment plant discharge and irrigation extractions from the mainstream Hawkesbury Nepean River also occur directly from the TUFLOW FV model (SKM 2014a).

The results from TUFLOW FV are incorporated into the AED water quality model to simulate concentrations of sediment, nutrients, algae and bacteria in the river system over time (BMT WBM 2014). There are 30 variables modelled in the TUFLOW FV/AED linked model.

The macrophyte model is a plugin model for the eWater Eco Modeller platform. The macrophyte model generates a relative cover score for *Egeria densa*. The model structure is based on the growth of *Egeria densa* being potentially limited by temperature and nutrients, and through periodic removal/pruning through high velocity conditions (SKM 2014b). The *Egeria densa* model requires daily inputs of velocity, temperature, nitrogen and phosphorus. These are generated from TUFLOW FV/AED.

The calibration and validation of the full Hawkesbury Nepean River and South Creek model used a combination of data from historical sources and from targeted campaign monitoring programs. Thousands of measured data points were used to calibrate and validate the model. Detailed information on the calibration and validation of the model can be found in SKM 2014a.

The model has been independently peer reviewed by the CSIRO for design and technical quality. Improvement opportunities identified during the review were addressed by the model developers.

Scenarios

The model has been built to provide guidance on the likely quantitative differences in water quality and quantity when contrasting different catchment and environmental flow, wastewater and land use scenarios over time. Overall differences in flow and constituent concentrations between scenarios can be inferred by comparing scenarios. This includes differences between mean values, or differences between values that may be exceeded for a given proportion of time. It enables the assessment of the overall outcomes of a particular suite of management actions across a broad spatial and temporal domain, compared to an alternative suite of actions or a “do nothing” scenario (SKM 2014b). These management actions are incorporated in the model as scenarios.

The model has been set up as a scenario based model. That is, the same weather sequence is used for all model runs. The weather sequence chosen was the 1985-94 period as it includes a mixture of wet, dry and average years, and is the period frequently used for government modelling projects. A scenario model enables direct comparison of different outputs and hence the benefits of implementing different options. However, this approach precludes comparing scenario model outputs with observations because the timeframes of the model runs and data collection are different (SKM 2014c). The model has not been established to predict conditions at a particular time in the future. Predictive modelling requires input of accurate future conditions such as rainfall at specific locations in the model domain. The uncertainty associated with future climate models would create uncertainty in the Hawkesbury Nepean model output, such that it would not be possible to discriminate among scenarios.

An initial 100 scenarios were run to test different combinations of urban development, environmental flow, wastewater treatment and stormwater management measures over time. These combinations explore the system in its existing state (2011) and in 2020, 2030 and 2050 if the weather sequence between 1985 and 1994 was repeated. The scenarios investigated included:

- **Environmental flows:** Represented as changes in the input time series for flow from Warragamba Dam to the TUFLOW/AED model. Five different dam release regimes were used within the scenarios: measured and basecase releases, and 80/20, 95/20 and 90/10 transparent/translucent environmental flow releases from Warragamba Dam.
- **Wastewater treatment plant discharge (WWTP):** Changes to the discharge from 26 plants (existing and future proposed) were modelled within the Hawkesbury Nepean River catchment. The plants were altered to represent changes in discharge locations (local tributary, Hawkesbury Nepean River or out of the catchment), volume and quality, as well as commissioning and decommissioning.
- **Advanced Water Treatment Plant (AWTP):** Changes to the operation of the AWTP at St Marys. The AWTP is part of the St Marys Water Recycling Program and applies reverse osmosis to tertiary treated wastewater from Penrith, St Marys and Quakers Hill plants. The result of this process is the discharge of high quality recycled water into the Hawkesbury Nepean River, near Penrith. The options modelled were the operation of the AWTP at full capacity (50 ML/d recycled water return to catchment), partial capacity (25 ML/d recycled water return to catchment) and no capacity (0 ML/d recycled water return to catchment).
- **Population growth/landuse change:** Population growth was represented by changes in landuse in the catchment model and increased wastewater flows from the plants. There are three landuse options which have been modelled as part of the scenarios – 2011, 2030 and 2050. The years represent extensions of the growth boundaries and urban consolidation.
- **Water Sensitive Urban Design (WSUD):** Implementation of WSUD in “green field” or new urban areas to limit the loads of sediment and nutrients generated from these areas. WSUD effects were modelled as a reduction in concentration of suspended solids, nitrogen species (total nitrogen, oxidised nitrogen, total kjeldahl nitrogen and ammonium) and phosphorus species (total phosphorus and filterable reactive phosphorus) in the runoff from the new urban regions within the 2030 and 2050 landuse. The percentage reductions applied were 85% suspended solids, 65% phosphorus species and 45% nitrogen species.
- **Rehabilitation of sections of South Creek:** Assimilation of nutrients in South Creek was incorporated as a decay function within the catchment model. It represents an option to manage activities that reduce nutrient loads into streams. The management activities include revegetation of stream banks or installation of silt traps. The removal efficiency for each nutrient constituent is based on grass buffers at least 7 m wide and restricted stock access to protect the riparian vegetation and streambank.
- **Climate change:** Climate change scenarios were incorporated by using the NSW and ACT Regional Climate Modelling (NARCliM) downscaling project and changed rainfall-runoff parameterisation of the catchment model. The 2050 scenario was based on a subset of the NARCliM data, where the 1985 to 1994 results were adjusted to represent 2050 conditions. Climate change was only applied downstream of the dams. The model boundaries were adjusted to include sea level rise (0.7 m). Climate change scenarios are included in the model as proof of concept only due to the limited subset of NARCliM data available at the time.

Hawkeye

Hawkeye is an SQL Server database and associated interface that allows site based interrogation of the model results. Multiple scenarios can be simultaneously compared. There are 52 sites uploaded into Hawkeye, a small subset of the >40,000 sites in the model. To include all sites for the 100 scenarios is approximately 1.7 petabytes of modelled output, which is impractical to handle.

The model output is stored in Hawkeye at daily timesteps for each constituent for each site and scenario.

A screen shot of the Hawkeye interface is presented in Figure 4-3.

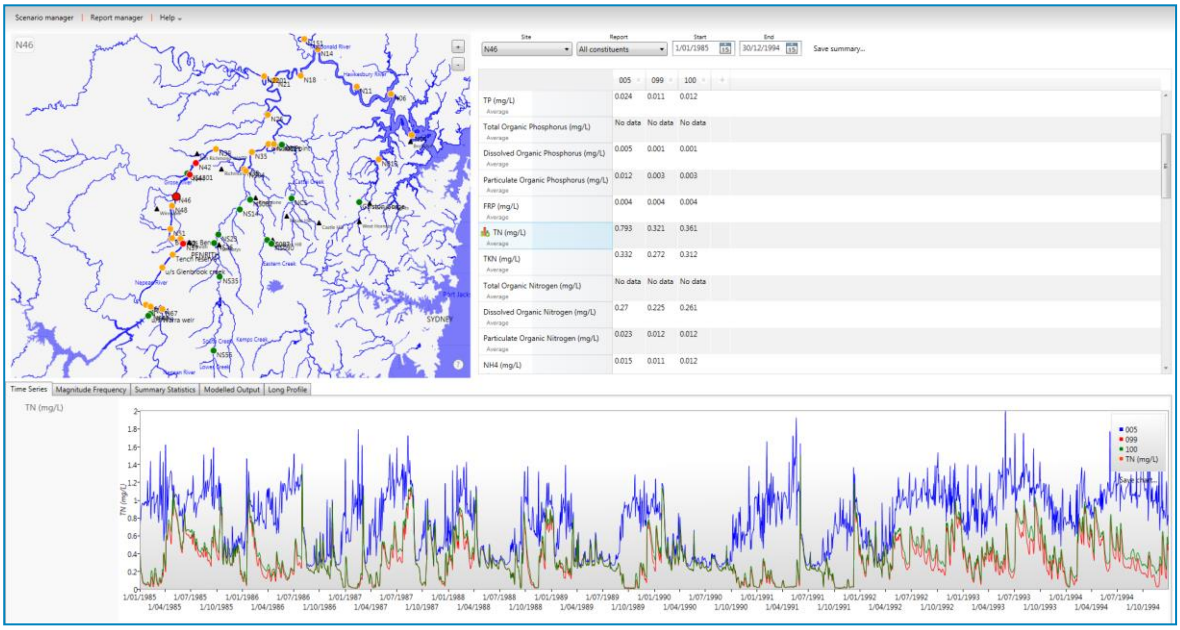

Figure 4-3 Screenshot of Hawkeye

Preliminary findings

Sydney Water has completed a preliminary analysis of the scenarios outputs. Due to the sheer volume of modelled output from the full suite of 100 scenarios, a subset of 19 scenarios and 10 sites was chosen for these initial analyses (Figure 4-4). These investigated a range of servicing options to manage future challenges including: future urban growth; plant discharge location; Sydney Water's contribution/influence on water quality; diffuse source management; treatment of discharge to recycled water quality standard; St Marys AWTP options under current and future conditions; and extreme options ie no discharge or all discharge of recycled water quality. The extreme options, while unrealistic in terms of cost, were chosen to better understand contributions from other sources (point and diffuse), and the extent to which Sydney Water could influence water quality with the discharge of very low nutrient water. Three parameters: total nitrogen, total phosphorus and chlorophyll *a* were analysed in 'all weather' and 'dry weather' conditions. This was the 'initial cut' of analysis that will prompt further analyses. Additional scenarios, parameters and sites will be analysed as new questions arise.

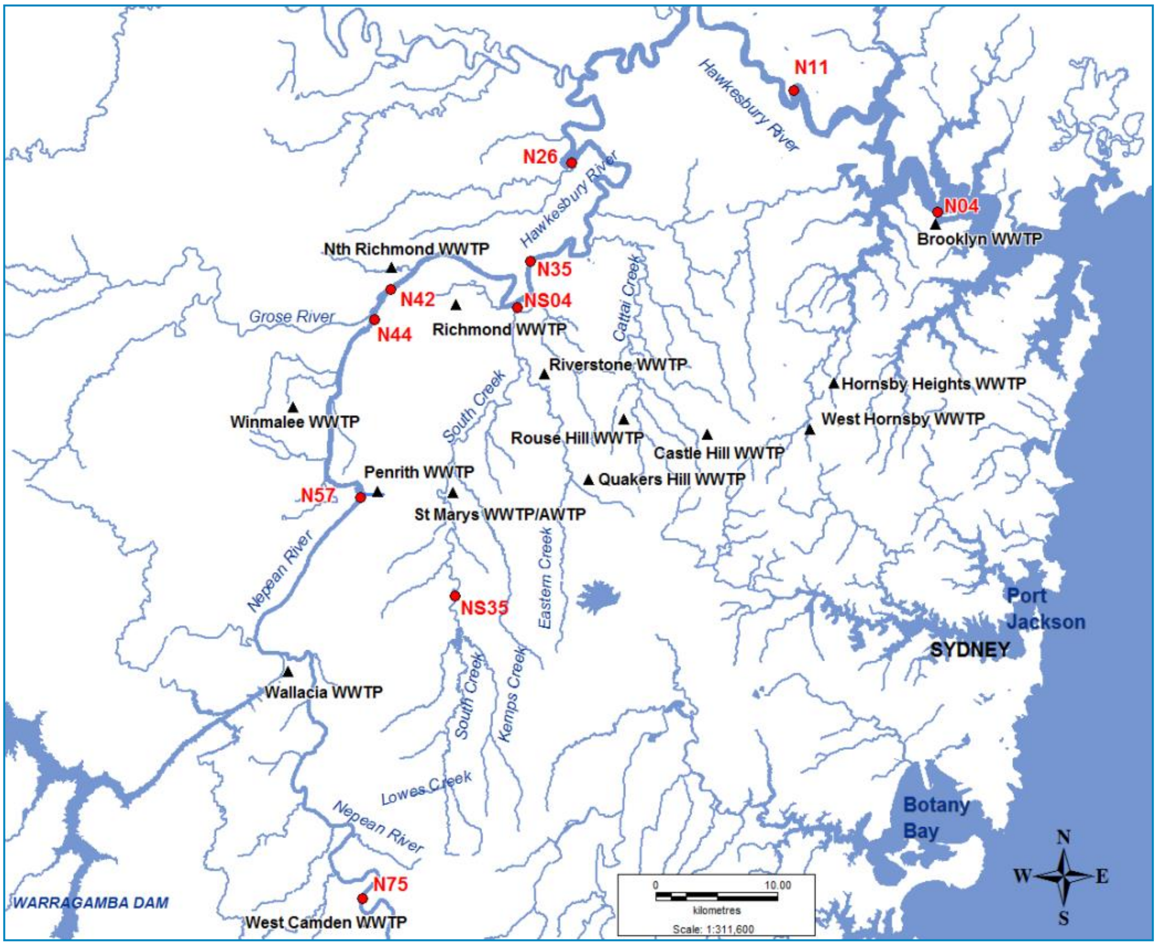

Figure 4-4 Ten sites (in red) chosen for initial scenario analysis

The statistical analysis of the modelled output incorporated two indicators to assess the relative performance of the selected scenarios:

- integration of the cumulative distribution function of each variable
- comparison of the model output with Healthy Rivers Commission objectives (HRC 1998).

The metrics are presented in a graphical form (examples are provided in Figure 4-5, Figure 4-6 and Figure 4-7). The graphs include the site locations on the x axis from the uppermost site, (N75, Hawkesbury Nepean River near Camden), to the furthermost downstream site, (N04, Hawkesbury Nepean River at Brooklyn). In the centre of the plot between the dashed lines, are two sites located in South Creek (NS35 and NS04). South Creek is an important tributary in the Hawkesbury Nepean River catchment as it will house much of Sydney's growth in the next 30 years. The graph has two y axes. The left hand side y axis is the integral metric as represented by the bar graph; the longer the bar, the poorer the performance. The right hand y axis is for the percentage of scenario variable records within the Healthy Rivers Commission objectives; the lower the line the poorer the performance.

Preliminary analysis indicates there are three key zones in the Hawkesbury Nepean system that show differing sensitivity to the management options tested:

- The upper Hawkesbury Nepean River responded to the majority of the management options for total nitrogen, had a variable response for total phosphorus, but minimal response for algae.
- South Creek and the region below the junction with the Hawkesbury Nepean River was sensitive to most scenarios tested. This zone is the main area where Sydney Water has the opportunity to improve water quality outcomes through wastewater infrastructure and treatment choices.
- The lower Hawkesbury Nepean River near Brooklyn (N04) showed little change among scenarios.

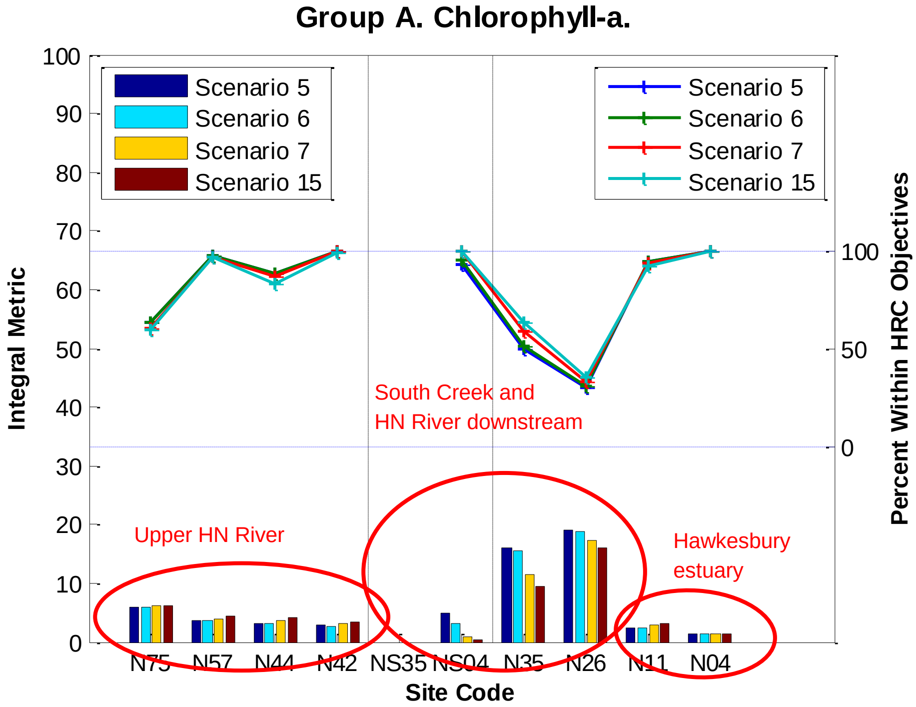

Figure 4-5 Three zones in the Hawkesbury Nepean system (Sc5=2011; Sc6=2020; Sc7=2030; Sc15=2050)

A second key finding is the importance of flow as a critical factor for managing river health. Sydney Water's discharge has been found overall to reduce, or have a neutral effect on, concentrations of total phosphorus and chlorophyll *a* in the waterways, while generally contributing to increased nitrogen levels

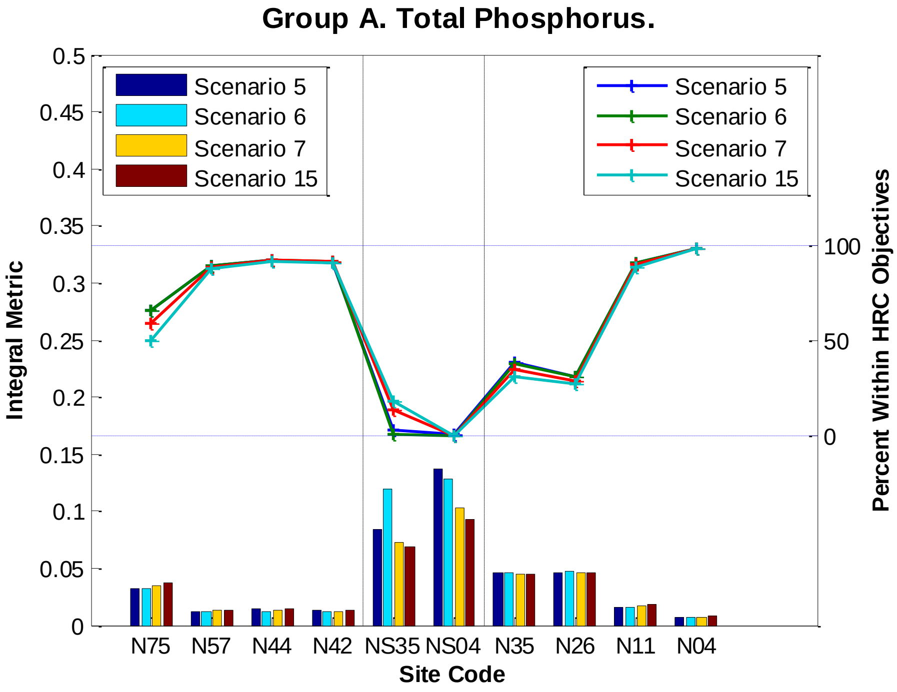

Figure 4-6 Influence of increased flow with population growth on total phosphorus in the Hawkesbury Nepean River system (Sc5=2011; Sc6=2020; Sc7=2030; Sc15=2050)

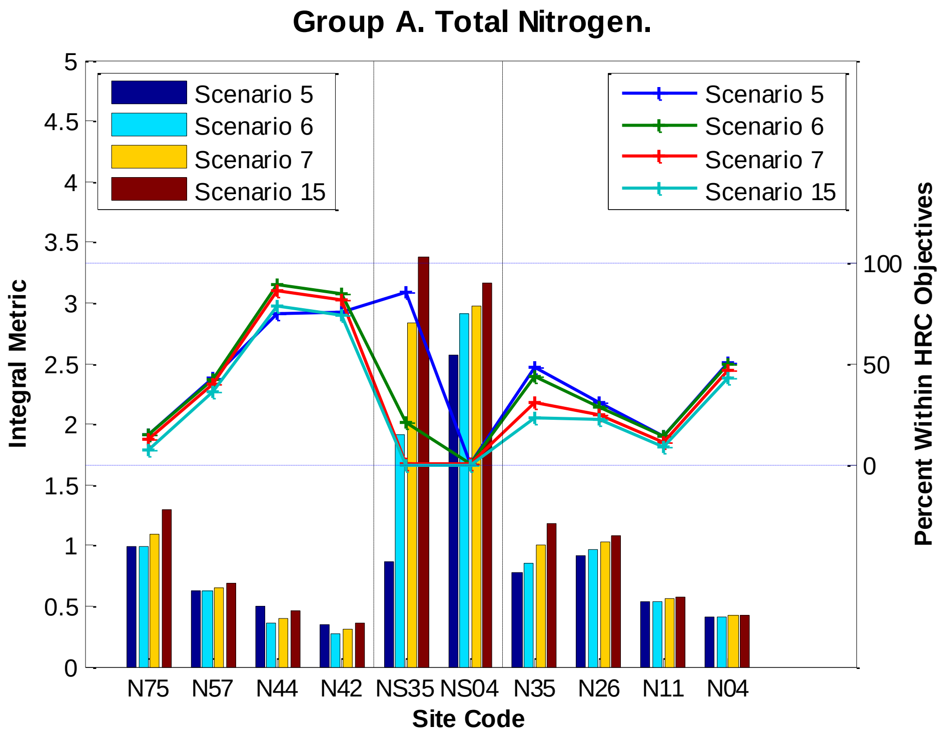

Figure 4-7 Influence of increased flow with population growth on total nitrogen in the Hawkesbury Nepean River system (Sc5=2011; Sc6=2020; Sc7=2030; Sc15=2050)

The examples provided are high level findings to show how the model can be used in the management of the Hawkesbury Nepean River. Scenarios, variables and sites are being further refined and interrogated to answer specific questions. This understanding will enable Sydney Water to plan for an improved environmental outcome when considering future management options in the Hawkesbury Nepean River catchment.

Model limitations

While the model is based on the best available scientific information specifically tailored for the Hawkesbury Nepean River catchment, as with all models, it is not without its limitations. It is important to be aware of these limitations when using the model and analysing the output.

Model limitations:

- The extent of the model mesh is limited to the main stem of the river and does not include the broader flood plain (except south of Penrith Lakes).
- The hydraulic performance of the weirs exceeded the weir rating curves during high flow events on eight occasions during the ten year simulation.
- Differences between spatial and temporal sampling of the field data and the predictions produced by the models may result in the modelled and measured concentrations for a constituent varying considerably during the calibration/validation period.
- The TUFLOW FV/AED model runs at sub-daily time step while the input time series from the Source model are daily. This may over-represent the scatter between observed and modelled concentrations during the calibration/validation period and may not necessarily reflect the performance of the model.
- Flow extractions for irrigation in South Creek during low flows had to be estimated as there were no measured extractions.
- Environmental flow releases from the Upper Nepean dams had to be estimated as there was no measured data to verify how much flow was actually released.
- Macrophyte beds influence both hydraulic and water quality behaviour upstream of South Creek. Macrophyte behaviour was not directly incorporated into TUFLOW FV.

Finally, it is important to note that the model has been based on the system as it is currently configured, such as landuse, weir location and the bathymetry of the river. In the future, as the catchment, river bathymetry and climate changes, it will be necessary to review the status of the model and update it with current data.

Conclusion

Water quality and quantity modelling is a key planning tool for understanding environmental impacts under different scenarios. It provides a means for guiding capital works programs, by allowing objective comparisons of likely water quality benefits against expenditure under different management options.

The Hawkesbury Nepean River and South Creek Model enables robust assessment of whole of system impacts of changes in the river system, such as those from wastewater treatment plant discharges, irrigation, catchment runoff and environmental flows. The model will enable Sydney Water and other river managers to develop affordable and cost effective management decisions that achieve environmental outcomes, consider different pollutant sources, are site specific, consider community goals and contribute to liveability.

Acknowledgements

Sydney Water would like to acknowledge Jacobs (previously known as Sinclair Knight Merz), BMT WBM, eWater, Yorb and UWA for the development of the model, the CSIRO for the review of the model, and Daniel Large for the analysis of the scenarios.

Extensive data sets were provided by the NSW Office of Water, Office of Environment and Heritage, Sydney Catchment Authority, Manly Hydraulics Laboratory, Land and Property Information, Bureau of Meteorology, Hornsby Shire Council, Penrith City Council, The Hills Shire Council, Blacktown Council and Camden Council. This data was critical for building and calibrating the model.

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Theme two: Sewage overflows

This theme assesses the potential for untreated discharges to impact on receiving water environments. The wastewater system contains a large network across Sydney Water's operational area for transporting sewage from private premises to wastewater and water recycling treatment plants. During periods of high rainfall, runoff can infiltrate into the network leading to sewer mains exceeding their capacity to hold flow. Sewage overflow points have been designed as part of this system to alleviate pressure during high flow events which ensure sewage flows in the network do not back up into private premises. During high flow events sewage may discharge from overflows to waterways. These discharges are episodic, typically of short duration and often diluted by large volumes of stormwater.

Consistent with the 'treated wastewater discharges' theme, a key aspect of this theme is to differentiate sources of pollutants in a waterway. In this case differentiating the effects of sewage overflows from stormwater will allow Sydney Water to better understand its contribution to the condition of a waterway. Two case studies were developed for this theme: modelling wet weather overflows in the upper Parramatta River to improve understanding of the contributions to the rivers water quality and a validation of the expected benefits of the Malabar stormwater diversions. Key to both case studies are techniques to improve our ability to differentiate stormwater from sewage overflow contribution to water quality.

5 Modelling wet weather overflows in the Upper Parramatta River

Abstract

The wet weather overflow abatement program describes actions that can be taken by Sydney Water to mitigate the impacts of sewer overflows during rainfall events. Currently, Sydney Water measures sewer overflow impact by overflow frequency, but this does not guarantee protection for the public or the environment. A new Effects Based Assessment approach is proposed to capture the impacts from an overflow on social use, public health and the environment.

One of the key pieces of information feeding into this assessment is output from hydrodynamic and water quality models. The Upper Parramatta River domain is used as a pilot area to test this new approach and it has its own dedicated models. The pilot models have been calibrated and validated against field data collected for that purpose.

The model was calibrated and validated using data from 2006, 2007 and 2013. Overall, the agreement between model output and observations during those periods was good (with a Nash-Sutcliffe of model efficiency generally exceeding 0.5, except for total suspended solids, e.g. Moriasi et. al., 2007) and the model is regarded as fit for its intended purpose.

Output from the model shows that water quality is quite poor but could be improved through water sensitive urban design. Sewer abatement would likely achieve little benefit for this area. However, concentrations are higher during rainfall events. This information will be important for the Effects Based Assessment to ensure abatement works achieve the best outcome for Sydney's waterways.

Introduction

Long term targets for improved performance in wet weather sewage overflows were set by the NSW Environment Protection Authority (EPA) in the late 1990s, based on the Environmental Impact Statements (EIS) (1998) submitted by Sydney Water. This approach was initiated by the enactment of the Protection of the Environment Operations (POEO) Act, which states that overflows are “scheduled activities” that form part of the sewerage treatment system and should be licenced. The target date in the EIS for meeting these long-term targets is 2021. The present target is frequency based (number of overflows per ten years) and applies to all overflows in a system, regardless of system size, environmental value, cost/benefit or demonstrated customer/community support.

It is not considered practicable to meet the targets by 2021 and the metric of overflow frequency alone will not guarantee protection for the public or the environment. Therefore, an Effects Based Assessment (EBA) approach has been proposed by Sydney Water. Broadly, this approach relates proposed changes in overflow discharges to social, public health and environmental benefits. Parameters that will be important for input to an EBA may include (but not be limited to):

- frequency, volume and duration of the overflow,
- types, concentrations and variability of contaminants in the overflow waters,
- sensitivity of the environment into which the overflow discharges, and
- public use of the waters into which the overflow discharges.

It is also noted that any EBA approach needs to assess the relevant water body as a whole. There is no benefit in solving an overflow problem at one location by creating a problem somewhere else in the system.

Overflows are intermittent and each overflow event is unique, hence monitoring alone will not be effective or practical in assessing environmental impacts. This is further complicated by discharges from stormwater systems, which likely occur simultaneously with overflow events. The management of the stormwater system is largely outside the jurisdiction of Sydney Water. Therefore, where stormwater is the major contributor to water contamination, mitigation measures applied to Sydney Water assets may have a negligible benefit.

The EBA approach uses a range of tools to help assess social, public health and environmental impacts and to separate the contributions from stormwater and overflow sources. Central to EBA are the results from numerical models. The Upper Parramatta River (UPR) model is a subset of the Sydney Harbour Model and is used as a trial of the EBA approach. Three scenarios (existing, “no overflows” and water sensitive urban design or WSUD) representing different potential mitigation options, have been modelled and are assessed in this report.

The model selected for use in the wet weather overflow abatement project is Resource Modelling Associates (RMA) model. RMA comprises a suite of models for simulating hydrodynamics and water quality in water bodies (King, 1993). The models can be operated in one, two or three dimensions using a finite element formulation.

The UPR model extends upstream from the Charles Street Weir at Parramatta. The weir prevents any exchange of estuarine waters with those from the UPR. Waters in the UPR are relatively shallow and stratification is minimal. Therefore, the two dimensional version of the hydrodynamic RMA model (RMA-2) is used. The 10-year period from 1985 to 1994 is used for all model scenarios. This enables the direct comparison of model results from different scenarios. This period contains a range of weather conditions considered to be representative of the long-term.

Calibration and validation of the Upper Parramatta River model

Results from the calibration and validation of the Upper Parramatta River (UPR) model are presented in SWC (2014). The calibration period for the hydrodynamic model was the period 2006-2007. A brief description of this process is provided below. It is recognised that numerical models are only approximations to the "real world". Uncertainty in the model output may arise for a number of reasons, some of which are outlined in SWC (2014).

Therefore, it is unrealistic to expect a perfect match between observations and model results over the whole model domain. Rather, the model calibration and validation process attempts to minimise differences between observations and model results, while simultaneously ensuring that the relevant processes are appropriately included in the model.

Cross-wavelet analysis is used to compare the observations with modelled output for both period and time e.g. Torrence and Compo (1998) and Grinsted, Moore and Jevrejeva (2004). Wavelet analysis allows us to incorporate both events and background information in a single analysis. The following points are used to assist in the interpretation of these figures.

- The heavy black lines represent the 95% confidence limits (i.e. good agreement in amplitude between the model results and observations).
- The curved line at the bottom of the plot indicates the "cone of influence". Outside this curve, the time series is affected by the ends of the time series (so-called edge effects) and results are not reliable, hence not shown.
- Arrows represent the phase difference between the two time series. Arrows pointing to the right indicate no phase difference between the model output and the data time series. Note: when the coherence is low phase has no meaning.
- The scale on the right hand side of each plot indicates the level of coherence between the observations and modelled output – time series that are highly correlated have coherence values close to unity.
- The scale on the left hand side of each plot is the Fourier period. In this analysis, the Fourier period should be multiplied by 17/24 to obtain "days". This enables both a radix2 Fast Fourier Transform algorithm to be used and the whole time series to be included in the analysis.

An example of the use of wavelet analysis to this problem is shown in Figure 5-1 for flow over Marsden Weir. In general there is high coherence (>95% confidence level) between the observations and the model results over the whole time series and for most periods. Some low period (i.e. high frequency) features have lower coherence, generally expected for analyses based on Fourier theory. Results from the cross-wavelet analysis indicate significant coherence between the observations and the model results at periods greater than a week or so over the whole time period (2006-2007). During periods of rainfall, this significant coherence extends to much shorter periods – as low 1.5 days. (Note: features in the time series with a period of less than about one and a half days cannot be resolved). Further, the arrows in the areas of significant coherence almost all point to the right, indicating zero lag between the observations and the model output (i.e.

the two time series are not temporally offset). Similar results were obtained for water level at other locations (SWC, 2014).

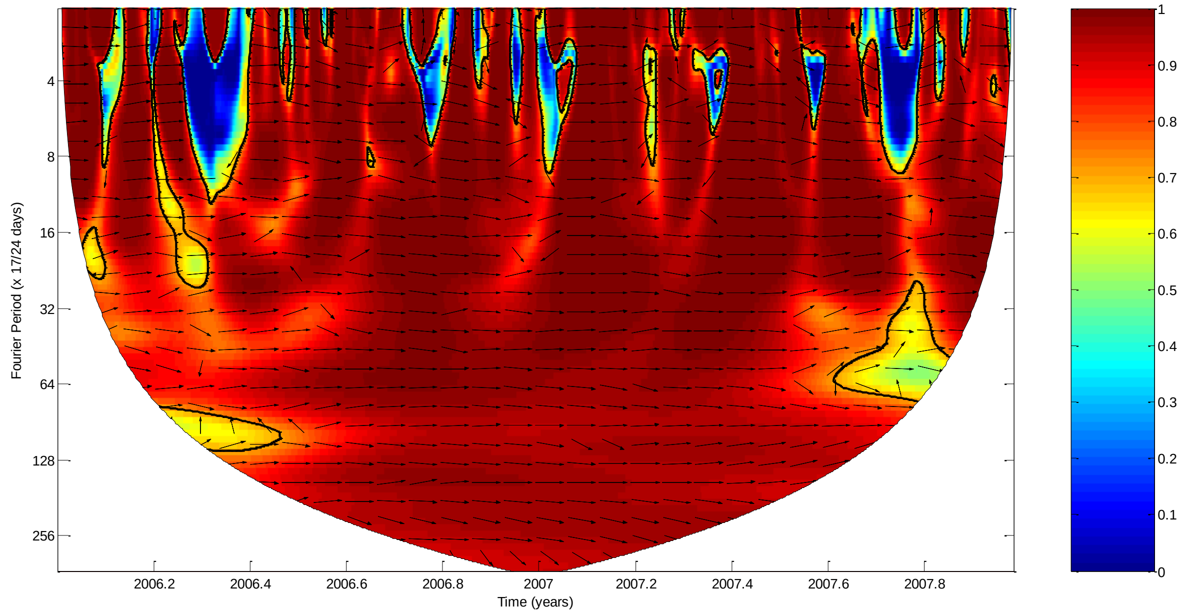

Figure 5-1 Wavelet coherence and phase between observed and modelled flow over Marsden Weir

The Nash-Sutcliffe coefficient (Nash and Sutcliffe, 1970) is a measure of the overall model performance against observed data. These are given in Table 5-1 for flow over Marsden Weir and water level at several locations in the model domain. Coefficients close to unity indicate a near perfect match between the observations and the model results. Coefficients close to zero indicate the model predictions are about as accurate as the median (for these analyses) of the observations, while coefficients less than zero indicate that the median is a better estimator of the observations than is the model. The Nash-Sutcliffe coefficient is always positive, always above 0.5, and, for most sites, is close to unity.

Table 5-1 The Nash-Sutcliffe coefficients assessing hydrodynamic model performance at several locations in the model domain

Gauging variable and location	Nash-Sutcliffe coefficient
FLOW over Marsden Weir	0.78
WATER LEVEL	
Marsden Weir	0.71
North Parramatta River Viaduct	0.78
Redbank Road	0.88
Johnstons Bridge	0.90
Sierra Place Basin	0.71
Loyalty Road Basin	0.58

In contrast to the volume of data available to calibrate and validate the hydrodynamic model, only limited data from limited events are available to calibrate and validate the water quality model. Therefore, focus is placed on the Nash-Sutcliffe coefficient of efficiency as the primary tool for judging the overall agreement between the observations and the water quality model results.

Water quality monitoring was undertaken at four sites, identified in this report as: Darling Mills Creek, Johnstons Bridge, Briens Road and Cumberland Hospital. The water quality model calibration period was between 1 January and 30 June 2013.

The Nash-Sutcliffe coefficients of efficiency for each variable modelled and at each of the four monitoring sites are presented in Table 5-2. All coefficients exceed zero (and most exceed 0.5), indicating that the model provides an acceptable fit to the observations. However, for total suspended solids the coefficients are small, indicating that the model does not perform quite as well, for this variable.

Table 5-2 The Nash-Sutcliffe coefficient of efficiency at the four water quality sites and for seven parameters

Variable	Darling Mills Creek	Johnstons Bridge	Briens Road	Cumberland Hospital
Enterococci	0.52	0.39	0.52	0.46
Total nitrogen	0.76	0.36	0.88	0.84
Oxidised nitrogen	0.84	0.68	0.71	0.75
Unionised ammonia	0.75	0.53	0.66	0.62
Total phosphorus	0.89	0.83	0.80	0.82
Orthophosphorus	0.57	0.89	0.80	0.75
Total suspended solids	0.23	0.08	0.34	0.00

Model uncertainty is quantified using two approaches. The first uses a Monte Carlo type approach, randomly selecting values for parameters (within their recommended range) and repeating the model run. This was done 5,000 times. Examples of the results for total phosphorus, organic nitrogen and Enterococci are provided in Figure 5-2, Figure 5-3 and Figure 5-4, respectively. For most variables there was virtually no difference in the output concentrations. Slight differences (but generally less than a few percent) were observed during dry weather periods. Example plots are provided below for total phosphorus, organic nitrogen and Enterococci.

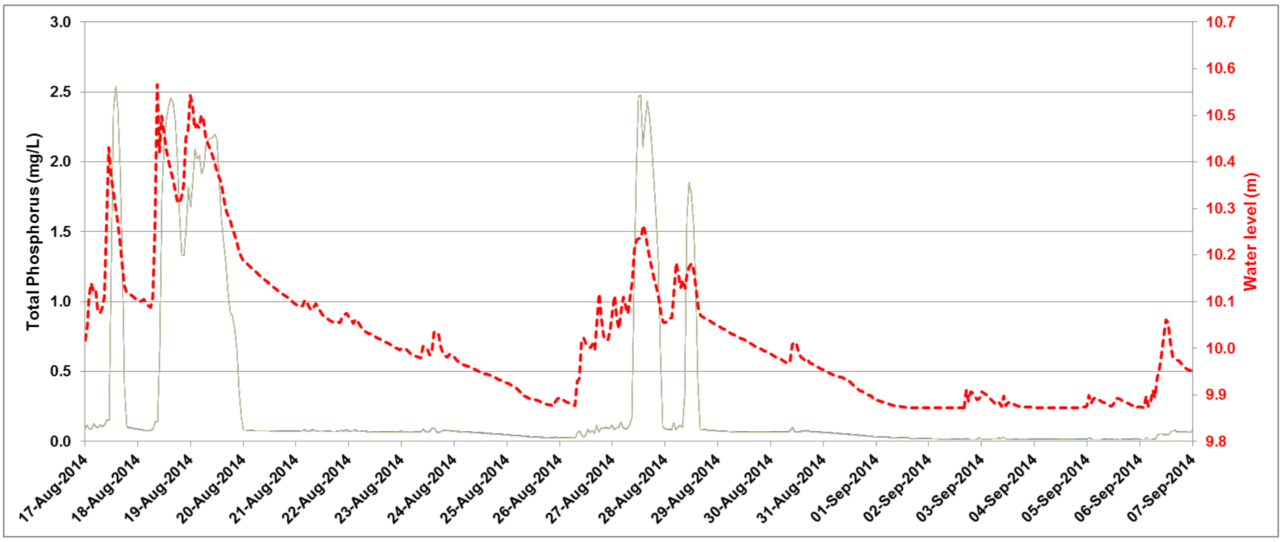

Figure 5-2 Results from the Monte Carlo simulations for total phosphorus

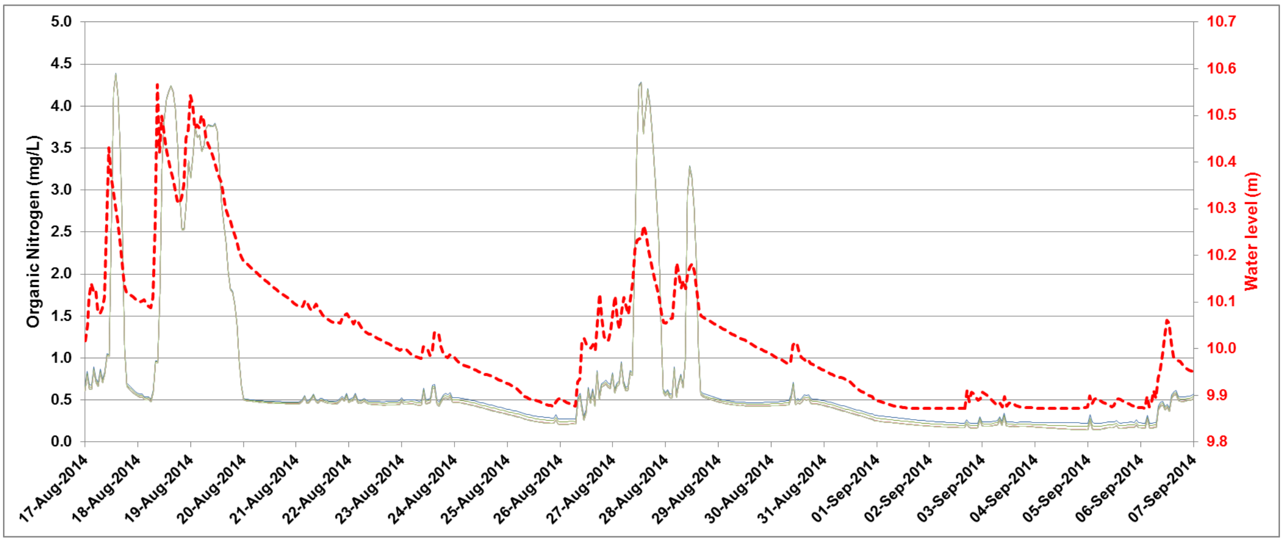

Figure 5-3 Results from the Monte Carlo simulations for organic nitrogen

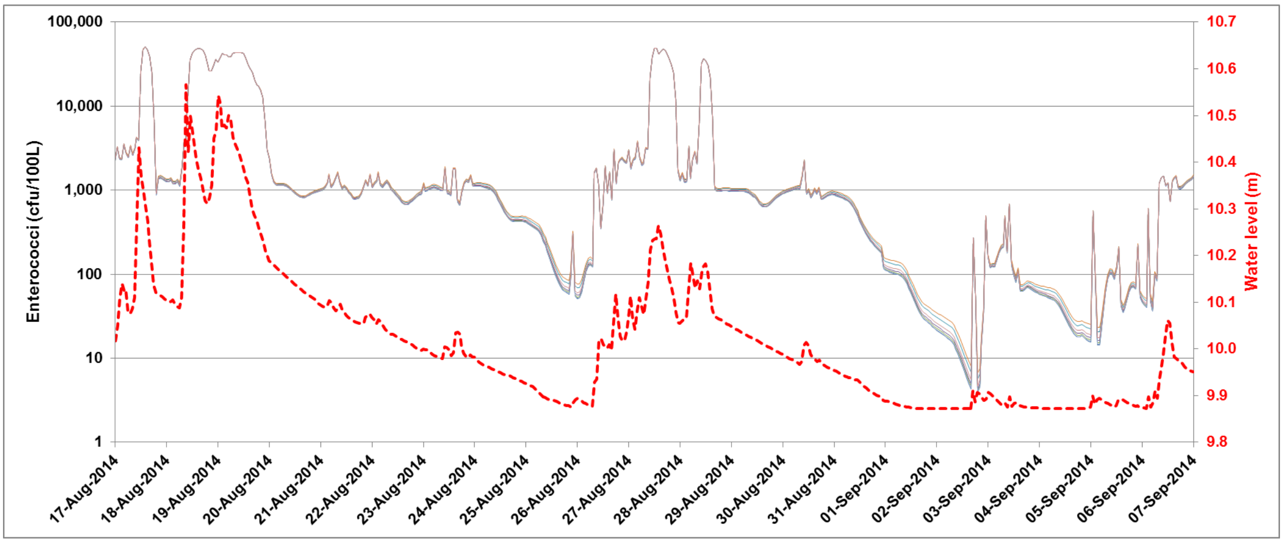

Figure 5-4 Results from the Monte Carlo simulations for Enterococci

The second approach perturbs the values of the input variables. In a general sense, a percentage change in the input concentrations results in the approximate same percentage change in the concentrations of the output variable. For example, a 10% change in the input concentration for nitrate results in a 10% change (approximately) in the output concentrations for nitrate.

From these results, it is concluded that the model reproduces the observations for flow over Marsden Weir (and water level at several locations – not shown here) over a range of time and space scales for flow and water level during 2006-2007. Overall, the model provides an acceptable representation of the observed water quality parameters at the monitored sites. The hydrodynamic and water quality models can be regarded as well-calibrated.

A comparison of modelled scenarios

The results presented below are based on the water quality model output. Using results from the calibrated UPR model, comparisons are made among the existing conditions, a "no overflow" scenario and a WSUD scenario. Existing conditions reflect the present waterway conditions, the "no overflows" scenario represents present conditions in the absence of overflows and WSUD scenario model the anticipated improvements under implementing WSUD. Five sites were selected for comparison. They represent conditions at the following sites: downstream in Finlaysons Creek (P14), upstream in Darling Mills Creek (P17), downstream in Darling Mills Creek (S3), Lake Parramatta (S6) and Charles Street Weir (S1). These sites are shown in Figure 5-5. Six key parameters were used including Enterococci as a measure of public health and the water quality parameters total nitrogen (TN), ammonia (NH_3), total phosphorus (TP), total suspended solids (TSS) and chlorophyll *a*.

Concentrations were extracted from the Upper Parramatta River model as a 10 year time series recorded hourly. The median concentrations of the water quality parameters and the 95%ile value of Enterococci were calculated for each scenario and site. Results were also compared with the ANZECC (2000) default trigger values and NHMRC (2008) guidelines for managing risks in recreational waters, these provide an indication of waterway condition.

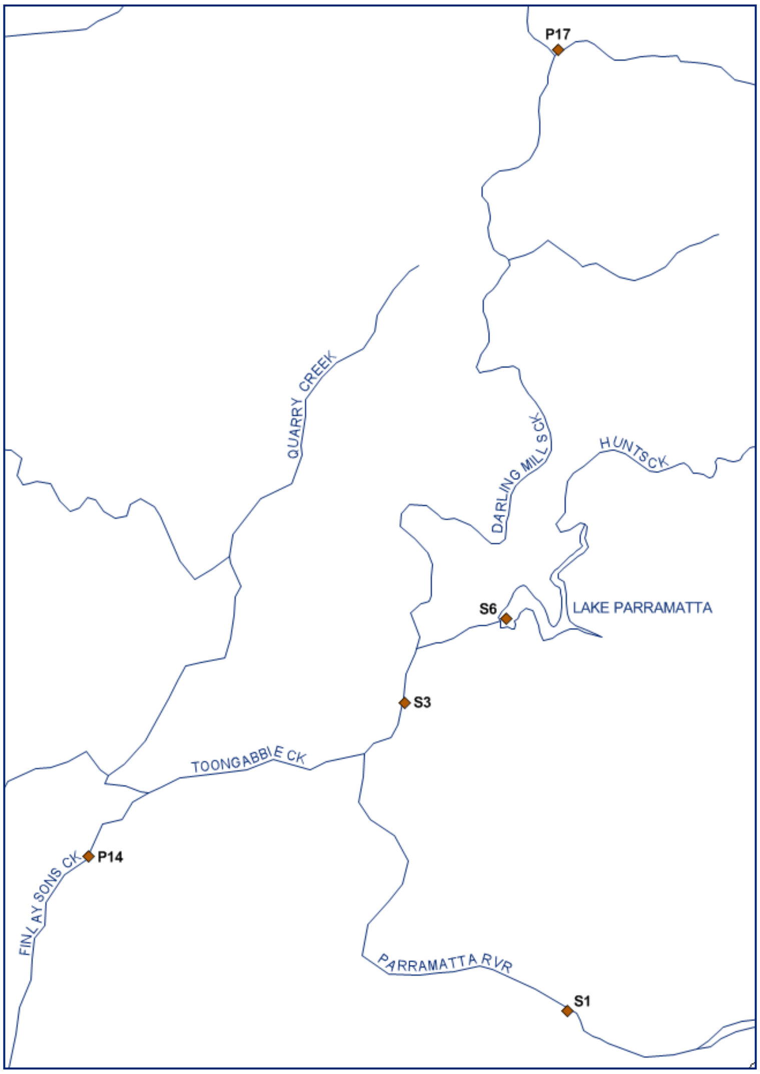

Figure 5-5 The sites from which water quality model results were extracted and analysed

Model uncertainty primarily arises through differences in parameter values and/or uncertainty in input values. Monte Carlo testing of the variable parameters resulted in virtually no difference in model output under wet weather conditions and less than about 1% difference in output under dry weather conditions. The uncertainty in the model output is less than 1%. However, results from sensitivity testing of the model input variables indicate that, for example, if there is a 10% change in the input variables, then there will be a similar percentage change (approximately) in the model output.

Based on the calibration data, the 95% confidence limits for the model output in Figure 5-6 through Figure 5-10 are +/- 0.17 for total nitrogen, +/- 0.011 for total phosphorus, +/- 0.006 for reactive phosphorus and +/- 11 for chlorophyll *a*. In general, this implies that, at most locations examined, there is no statistically significant difference between the “existing” and “no overflow” scenarios. However, concentrations of these substances for the WSUD scenario are significantly different from the “existing” and “no overflow” scenarios.

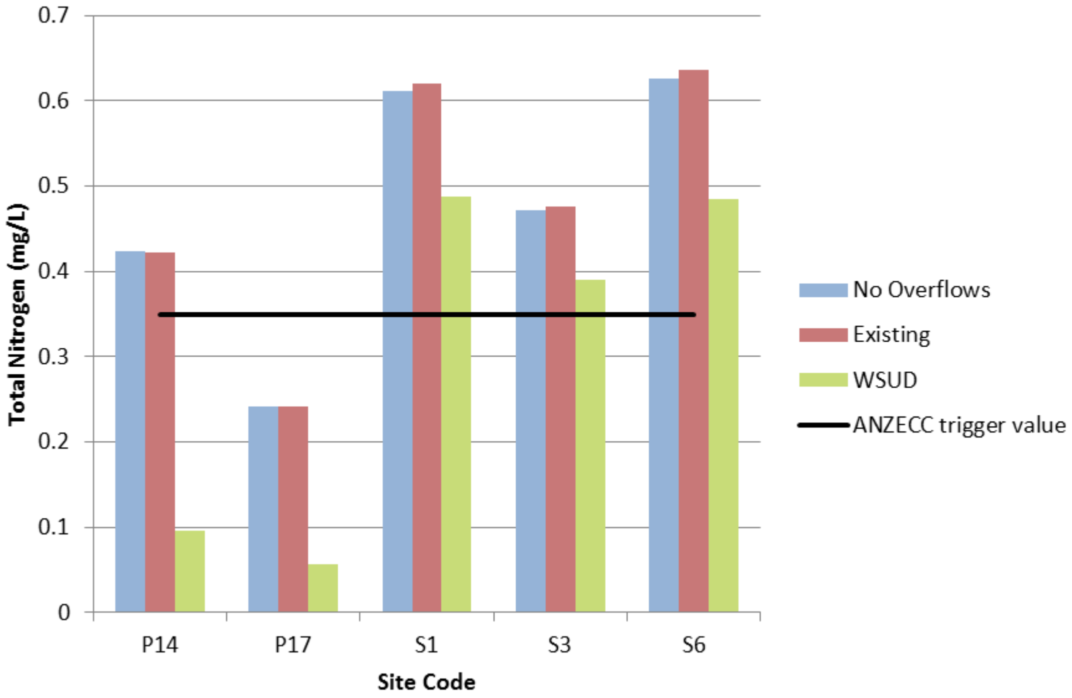

Figure 5-6 Median total nitrogen concentrations for sites and scenarios over the ten year period. The ANZECC (2000) default trigger value is superimposed

On average, total nitrogen currently exceeds the ANZECC trigger value at all sites except for upstream Darling Mills Creek, with levels being particularly high in the Parramatta River and in Lake Parramatta (Figure 5-6). This graph demonstrates that even if we could eliminate all the sewage overflows, we would not see any reductions in average total nitrogen. This suggests that sewage overflows are not a significant contributor to overall levels of TN. Alternatively, reducing the stormwater impact through WSUD results in decreases in total nitrogen concentrations. The greatest improvement is observed at Finlaysons Creek where implementation of WSUD is expected to lower the median concentration below the guideline to approximately 75% of the original value (0.42 mg/L to 0.10 mg/L). Concentrations of total nitrogen also decreased at the other sites but not by the same magnitude, and not below the trigger value.

Ammonia concentrations follow a similar trend to total nitrogen. Concentrations are above the ANZECC trigger value for a 99% level of protection of species for all sites except upstream Darling Mills Creek and on Finlaysons Creek (Figure 5-7). Ammonia concentrations also would not see an overall improvement if all sewer overflows were removed. However, implementing WSUD would make a small improvement to overall ammonia levels and could improve guideline compliance.

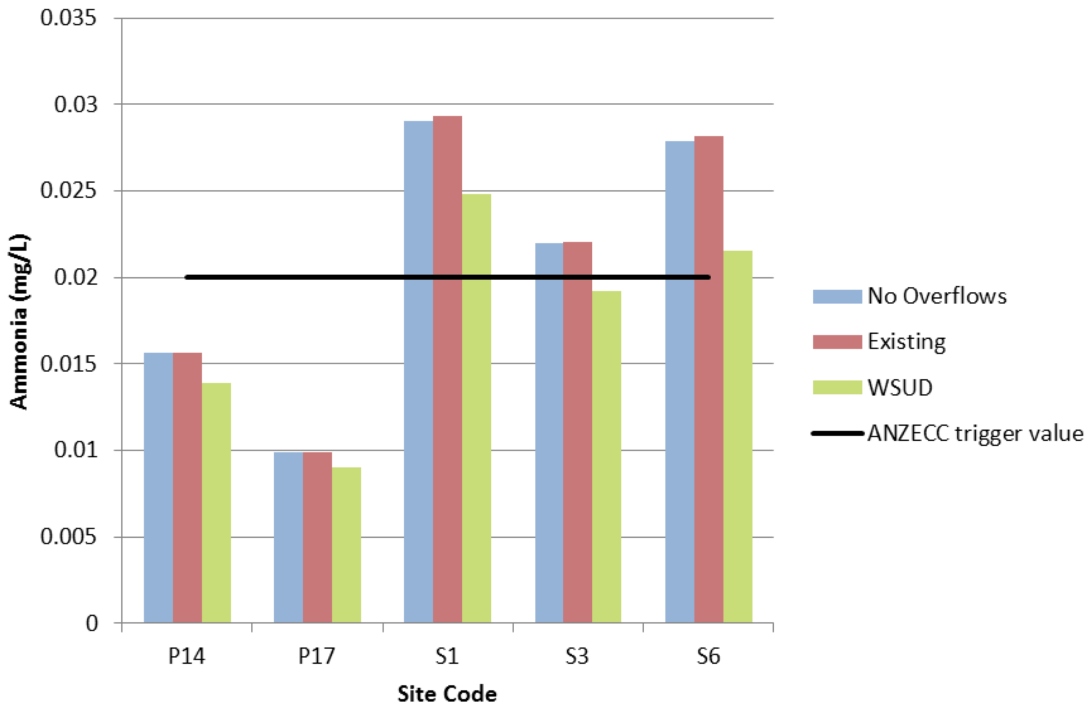

Figure 5-7 Median ammonia concentrations for sites and scenarios over the ten year period. The ANZECC (2000) default trigger value is superimposed

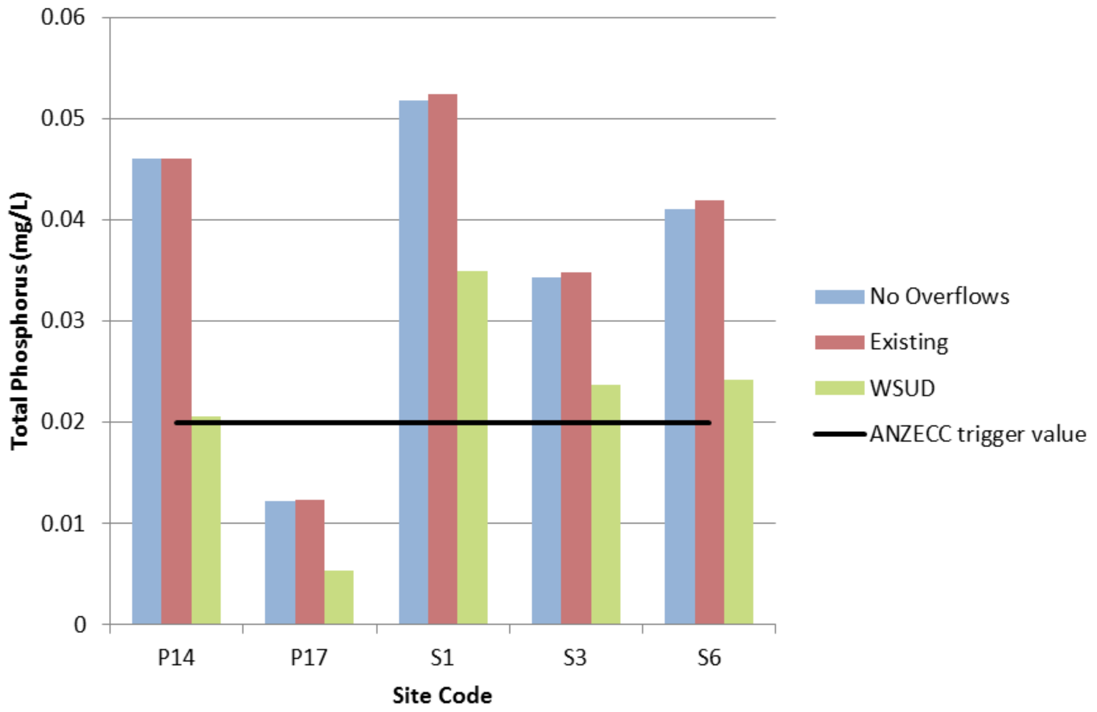

Figure 5-8 Median total phosphorus concentrations for sites and scenarios over the ten year period. The ANZECC (2000) default trigger value is superimposed

Total phosphorus levels under existing conditions exceed the ANZECC trigger values on average at all sites except upstream on Darling Mills Creek (Figure 5-8). Eliminating all sewage overflows does not improve total phosphorus concentrations. WSUD implementation improved total phosphorus concentrations by over half along Finlaysons Creek to the ANZECC trigger value. Concentrations were also improved by approximately a third in response to WSUD for Parramatta River, downstream Darling Mills Creek and in Lake Parramatta, and result in an incremental improvement in guideline compliance.

Total suspended solids were well below the ANZECC aquatic ecosystem trigger value at all sites (Figure 5-9). Thus, little improvements in this variable can be achieved through sewer or stormwater abatement.

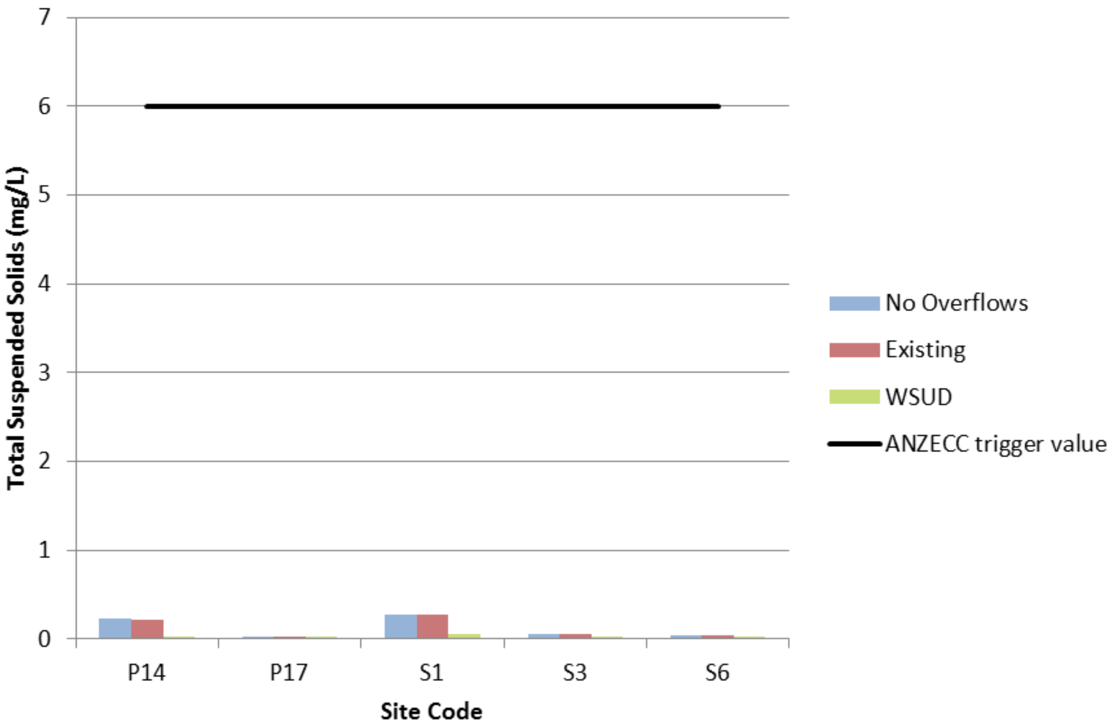

Figure 5-9 Median suspended solids concentrations for sites and scenarios over the ten year period. The ANZECC (2000) aquatic ecosystem default trigger value for lowland coastal rivers in NSW is superimposed

Chlorophyll *a* levels are almost 20 times the default trigger value in Lake Parramatta and almost 10 times the default trigger value at sites downstream of the lake (Figure 5-10). This suggests the high chlorophyll *a* levels may be a result of the lake which is a still water body that encourages the plant growth. Figure 5-10 shows that even by eliminating all sewage overflows, we would not see an improvement in overall chlorophyll *a* levels. However, implementing WSUD would be beneficial within and downstream of Lake Parramatta for increasing overall compliance with trigger values.

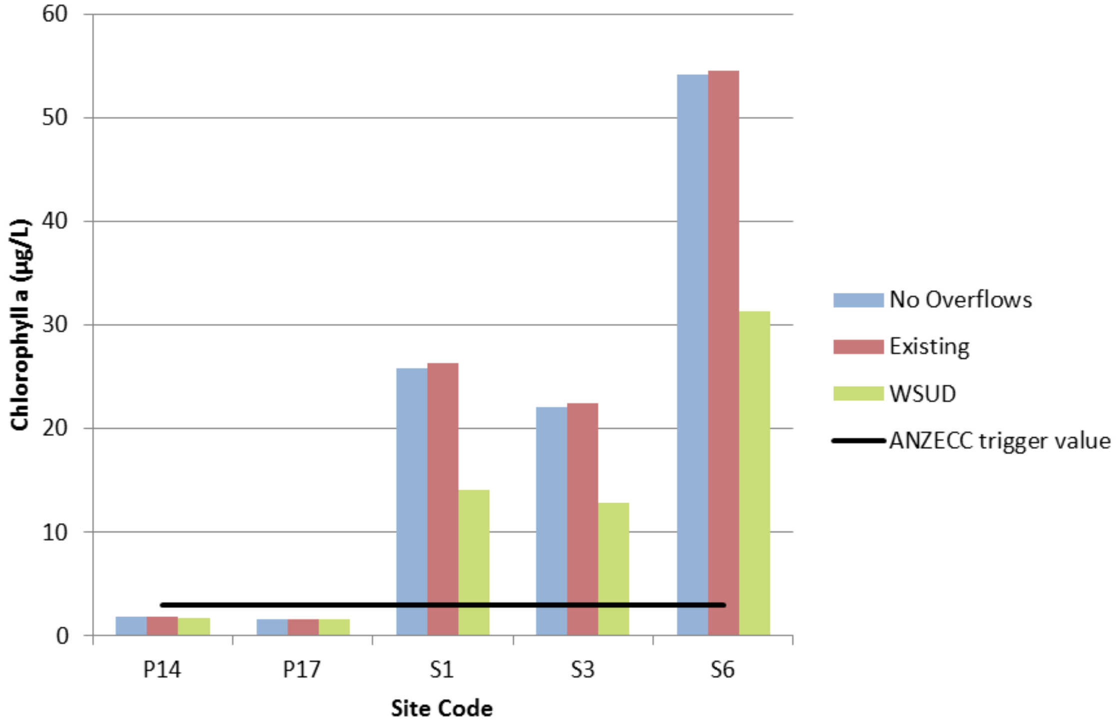

Figure 5-10 Median chlorophyll *a* concentrations for sites and scenarios over the ten year period. The ANZECC (2000) default trigger value is superimposed

Enterococcus is used as an indicator for the level of risk to public health and is based on exposure conditions. This indicator is relevant for sites where people are exposed to water through primary recreation (direct contact) or secondary recreation (intermediate contact). Secondary recreation takes place at Lake Parramatta in the form of boating. Parramatta City Council has aspirations to make a site on Parramatta River at Charles Street Weir open for recreation. Therefore, Enterococci levels were investigated at these sites. In line with the NHMRC Guidelines (2008), the 95%ile is placed in one of four categories (A = smallest, D = largest) to determine risk of exposure at a site. Levels in Lake Parramatta fall within Category B (Figure 5-11) meaning the risk of gastroenteritis (GI) is 1-5% and the risk of Acute Febrile Respiratory Illness (AFRI) is 0.3-1.9%. Removing sewage overflows reduces the 95%ile marginally, while implementing WSUD increases the percentile. At Charles Street Weir, the 95%ile is well into Category D, and is more than four times the maximum value in Category C (Figure 5-11). In this category the risk of GI is greater than 10% and the risk of AFRI is greater than 3.9%. As with Lake Parramatta, eliminating sewage overflows reduces Enterococci levels, and implementing WSUD increases levels. However, these changes are inconsequential for changing guideline classification. High levels at Charles Street Weir are likely the result of accumulation of all Enterococci from the Upper Parramatta River catchment which drains to this point. It should be noted that Enterococci can reach very high levels during storm events making recreational activities unsafe.

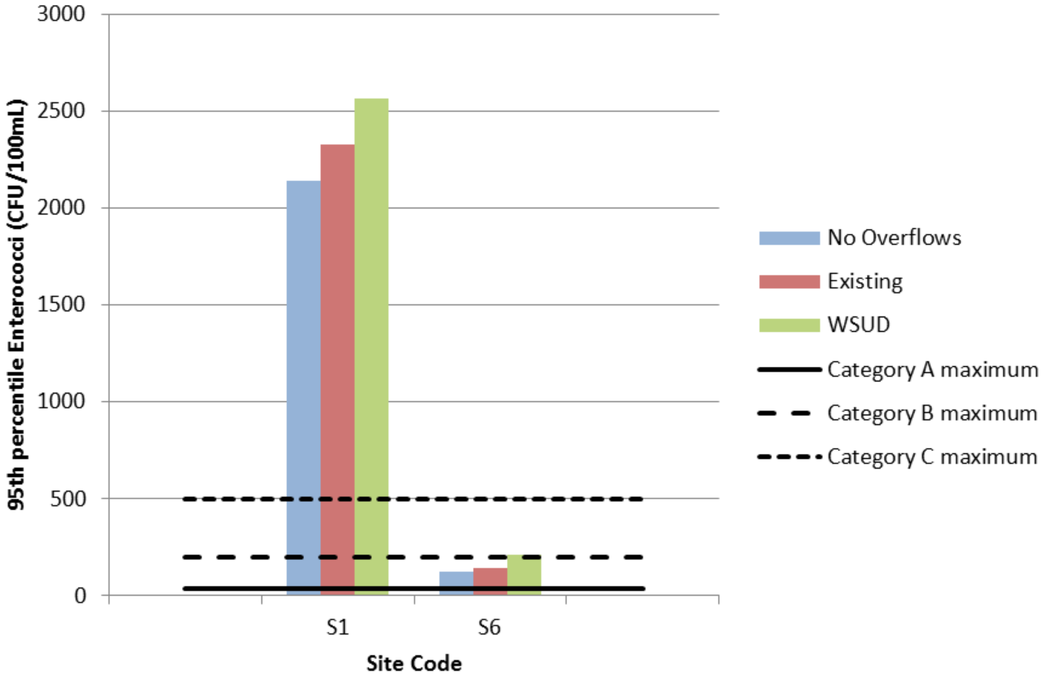

Figure 5-11 95th percentile values for Enterococci for sites and scenarios over the ten year period. The maximum NHMRC values for three categories are superimposed

Using medians to summarise the ten year data set does not allow us to see the effects during wet weather events where water quality parameters and Enterococci can exhibit much higher levels. Even though these effects may be short lived, they have the potential to significantly impact waterways. To identify extreme events, results are presented as probability of exceedance plots (for total nitrogen, ammonia, total phosphorus, total suspended solids, chlorophyll *a* and Enterococci, respectively). A description of the main findings from this analysis is provided below.

Emphasis is placed on the Charles Street Weir site, as it lies at the bottom of the catchment and will represent effects from the whole catchment. However, it is noted that results from other sites may differ substantially from those presented below. In general, the impact associated with rainfall events is near the low probability of exceedance end of the plots.

Total nitrogen (Figure 5-12), ammonia (Figure 5-13) and total phosphorus (Figure 5-14) show similar patterns as identified in median graphs. There are no meaningful differences between the existing and “no overflow” scenarios for about 95% of the time. The remaining 5% comprises primarily rainfall events (overflows) implying that overflows are a substantial contributor to the concentrations of these nutrients at the Charles Street Weir site. The plots for both the existing and “no overflow” scenarios are consistently greater than those for the WSUD scenario. This highlights the benefits (particularly for total nitrogen and total phosphorus) that could be potentially achieved by implementing WSUD.

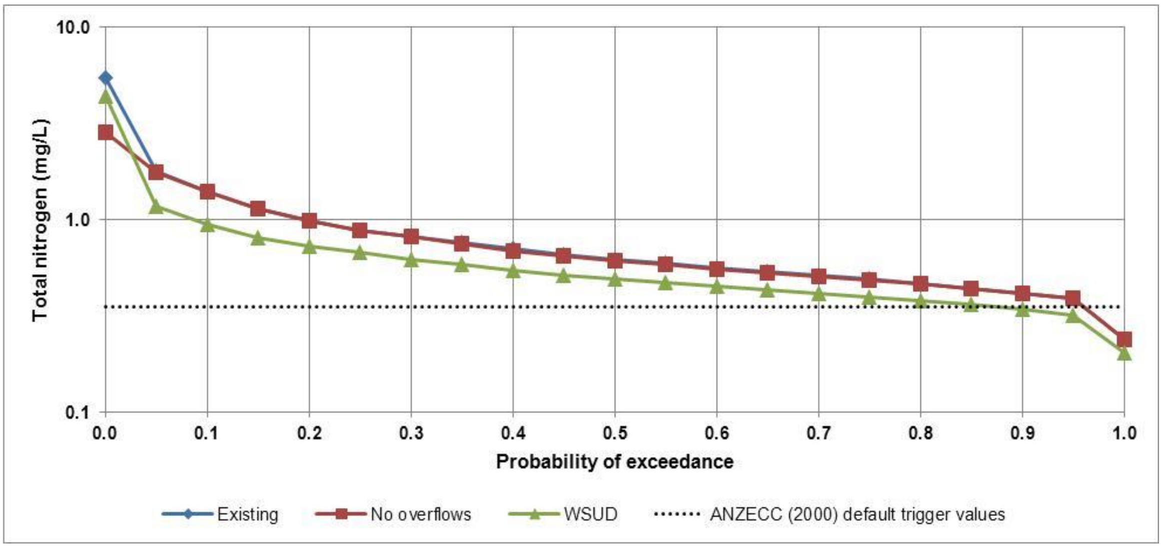

Figure 5-12 Probability of exceedance for total nitrogen at site S1 (Charles Street Weir) for three scenarios over the ten year period

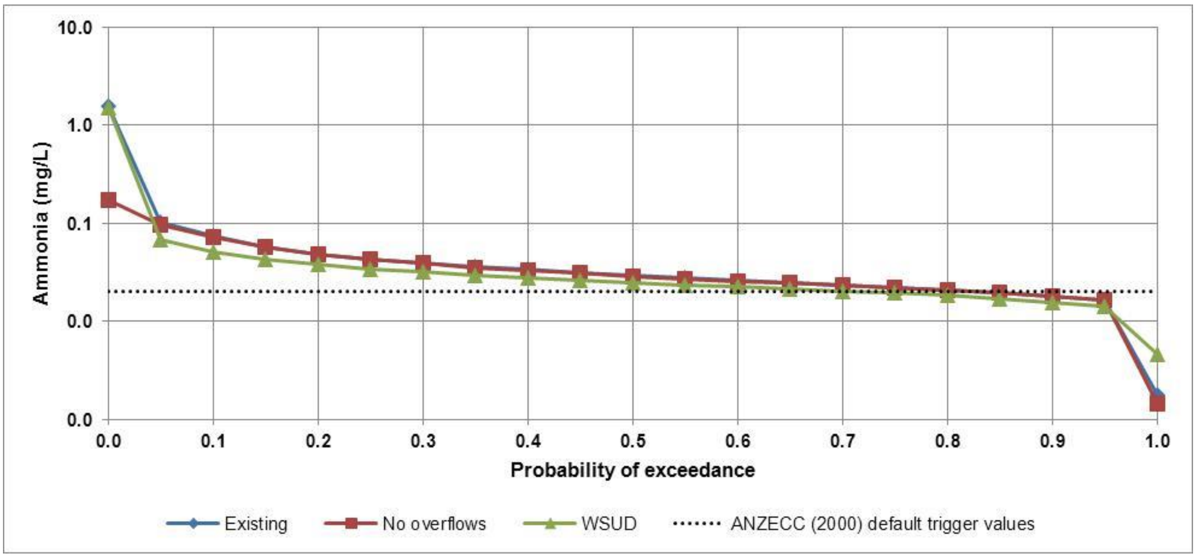

Figure 5-13 Probability of exceedance for ammonia at site S1 (Charles Street Weir) for three scenarios over the ten year period

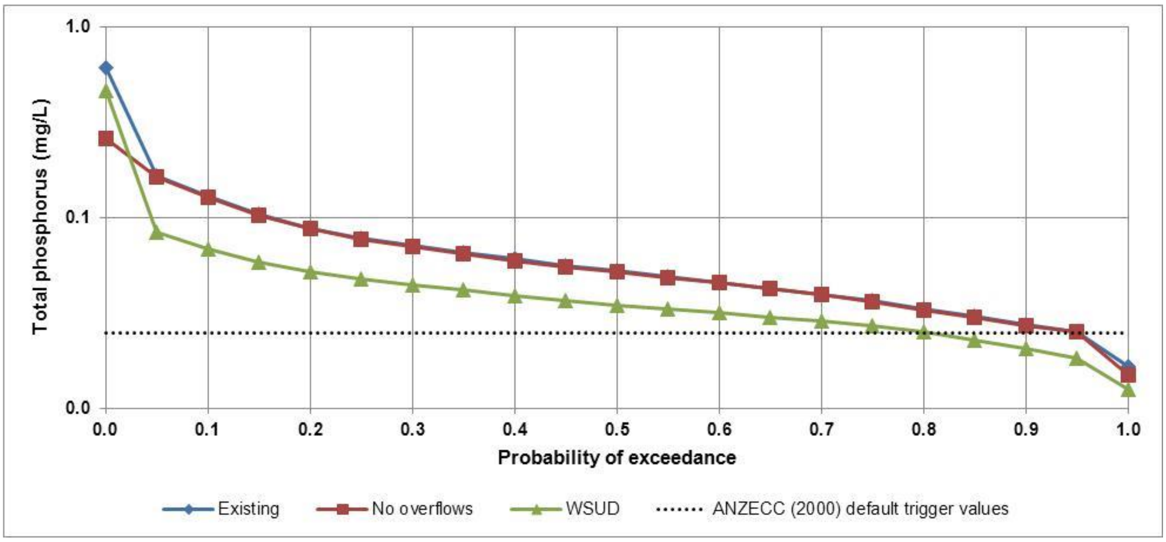

Figure 5-14 Probability of exceedance for total phosphorus at site S1 (Charles Street Weir) for three scenarios over the ten year period

Suspended solids (Figure 5-15) and chlorophyll *a* (Figure 5-16) show similar patterns. There is no meaningful difference in the respective concentrations between the “existing” and “no overflows” scenarios, implying that the overflows have little effect on chlorophyll *a* and total suspended solids. Both the existing and “no overflow” scenarios lie markedly above the WSUD scenario. Again, this highlights the potential benefit on implementing WSUD. For suspended solids, implementing WSUD results in concentrations meeting the default trigger values about 85% of the time, an increase of about 10%. While WSUD reduces the concentrations of chlorophyll *a*, concentrations still lie above the relevant default trigger values for about 95% of the time.

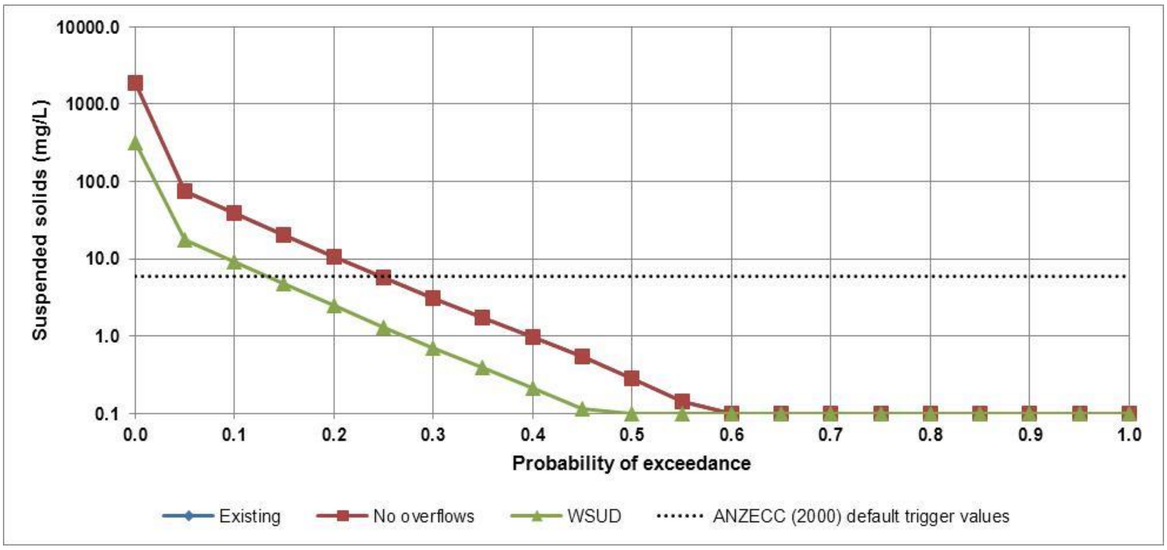

Figure 5-15 Probability of exceedance for suspended solids at site S1 (Charles Street Weir) for three scenarios over the ten year period

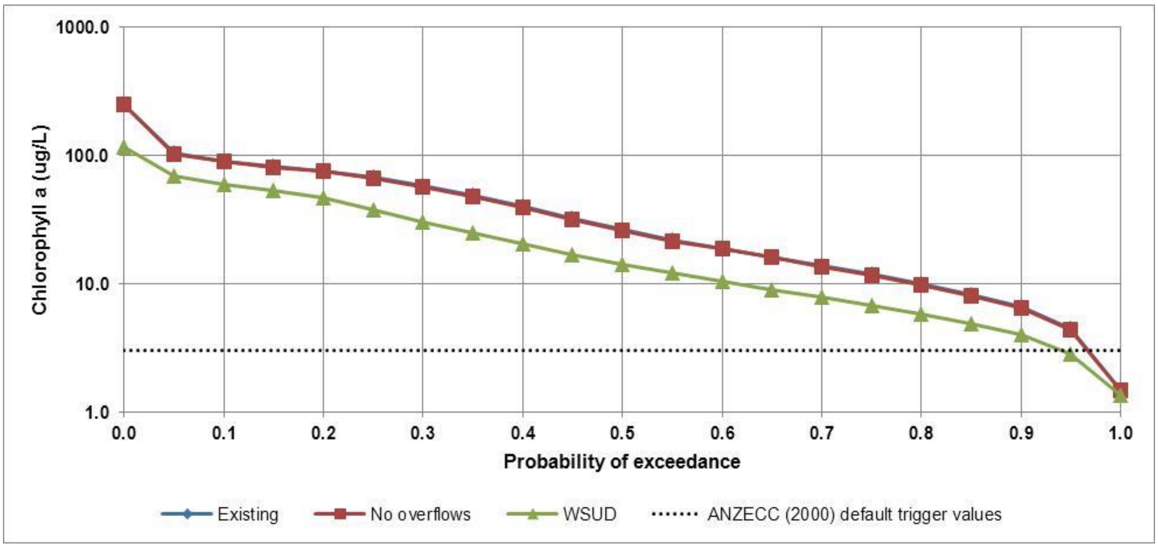

Figure 5-16 Probability of exceedance for chlorophyll *a* at site S1 (Charles Street Weir) for three scenarios over the ten year period

For Enterococci (Figure 5-17), there is little difference between the existing and WSUD scenarios. About 5% of the time, the “no overflow” scenario is less than that for the other two scenarios. This implies that rainfall events (overflows) are a substantial contributor to the concentrations of Enterococci at the Charles Street Weir site. All scenarios show that the 95% value (i.e. exceeded 5% of the time) lie above all of the three NHMRC categories for the protection of “healthy adult bathers”.

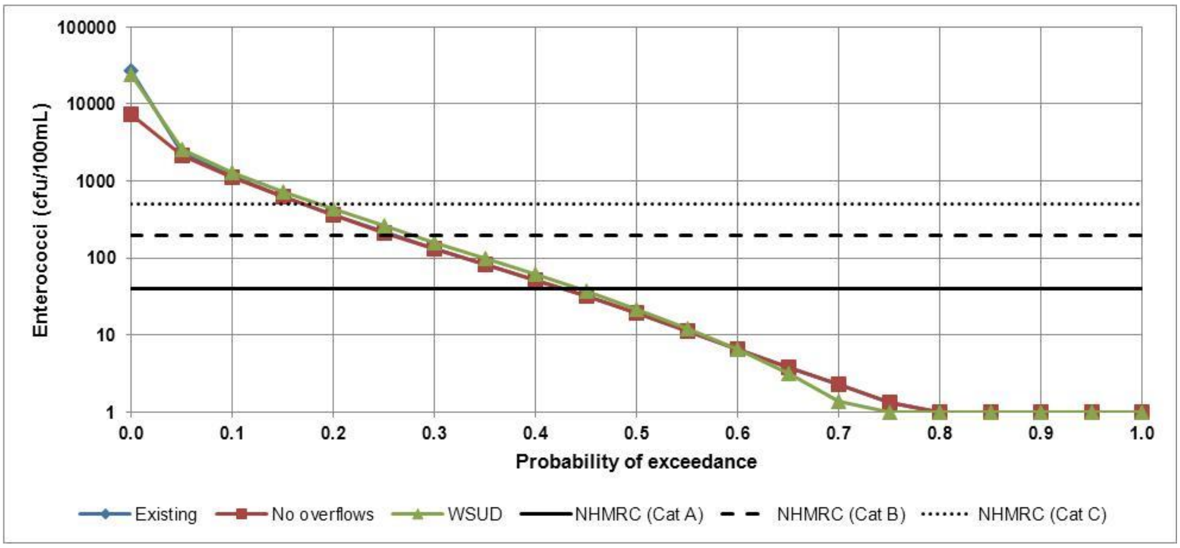

Figure 5-17 Probability of exceedance for Enterococci at site S1 (Charles Street Weir) for three scenarios over the ten year period

Summary and conclusions

The focus of this study is the potential improvements that can be made to water quality when overflows occur as a result of large wet weather events. The study was not designed to examine impacts under dry weather or small rainfall events in which sewage overflows do not occur.

Water quality in the Upper Parramatta River catchment is generally poor. This is true for many highly urbanised catchments. Overall, water quality is better upstream in the natural Darling Mills Creek but poorer downstream on Parramatta River which receives flows from the entire catchment. Nutrient levels including total nitrogen, ammonia and total phosphorus can be improved by implementing WSUD, with some sites benefiting more than others. Chlorophyll *a* levels can also be reduced through WSUD, particularly in Lake Parramatta. Suspended solids concentrations are generally low and benefits from abatement will be small. Enterococci levels fall in the same NHMRC guideline category regardless of abatement options.

Exceedance plots demonstrate that nutrients and Enterococci concentrations can be much higher under wet weather events when sewer overflows discharge. However, neither sewer abatement nor WSUD implementation can bring these wet weather values below the guidelines. The exceedance plots do support the idea that WSUD implementation will provide the greatest benefits to the catchment.

From the modelling of the Upper Parramatta River catchment, it is concluded that sewage overflows do not appear to be the major contributing factor to poor water quality. Better outcomes are likely to be achieved by investing in stormwater abatement strategies.

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6 Malabar Beach stormwater diversion: validation of the expected benefits

Abstract

Historically Malabar Beach water quality has been poor in comparison to many other Sydney beaches. The main Council stormwater drain and the drain from the Malabar plant deliver about 75% of the stormwater to Malabar Beach with ~65% from the main council stormwater drain. The intertidal zone ecological community had also been affected by the stormwater drain, with communities of green algae dominating the discharge area.

Source detection monitoring and modelling of Malabar Beach was undertaken to determine which sources, stormwater or sewage overflows, were providing the greatest contribution to poor water quality. This study established that the main council drain was the dominant source of faecal bacteria pollution at Malabar Beach. Sydney Water and Randwick Council entered into an agreement to divert stormwater from the two drains away from the beach to the South Western Ocean Outfall Sewer 2 that leads to the Malabar plant cliff face outfall. The aim of this paper was to validate the results of the pollution source detection study and the water quality and ecological health benefits of the stormwater diversion.

Beach bathing water quality monitoring and rocky platform intertidal zone monitoring was carried out before and after the diversion of stormwater to the Malabar plant cliff face outfall. The viability of the methods used for source detection and the benefits of the stormwater diversion were confirmed. Microbial monitoring of the beach found a notable improvement in the Beach Suitability grade for recreation after completion of the diversion. After the diversion a localised intertidal ecological change was also detected in the stormwater discharge zone, with the ecological community structure now more typical of other coastal locations unaffected by stormwater discharges.

Introduction

Malabar Beach water quality has been poor in comparison to other Sydney coastal beaches due to stormwater discharges from the main Council stormwater drain and the drain from the Malabar plant. Together these drains deliver about 75% of the stormwater to Malabar Beach. The main council stormwater drain contributes about 65% itself. It has also been observed that the ecological intertidal zone community had been affected by the stormwater drain discharge, with dense communities of green algae dominating the discharge zone.

Source detection monitoring and modelling of Malabar beach was undertaken to determine which sources were providing the greatest contribution to the poor water quality previously observed at this beach (OEH 2012). This monitoring established that the main council drain was the dominant source of faecal bacteria pollution at Malabar Beach. As such, Sydney Water and Randwick Council entered into an agreement to divert stormwater from the two drains away from the beach to the South Western Ocean Outfall Sewer 2 that leads to the Malabar plant cliff face outfall. It was expected the diversion would lead to improvements in beach bathing water quality and improvements in the ecological health of the intertidal rock platform communities. The diversion was completed on 30 November 2012.

Beach bathing water quality monitoring and rocky platform intertidal zone monitoring was carried out before and after the diversion of stormwater to the Malabar plant cliff face outfall. The aim of this paper was to validate the results of the pollution source detection study by assessing Malabar Beach water quality and intertidal zone community structure before and after the stormwater diversion.

Microbial pollution source detection

Malabar Beach is 150 m long and situated at the end of a long, narrow bay. It is backed by a small park and picnic area. The beach is not patrolled. Beach water quality is affected by its location, being within the Long Bay embayment. This reduces flushing from tidal movements, allowing discharges from land to have a greater influence on water quality. As a result it was expected that drains discharging to the beach would have a significant impact on beach water quality

To determine what the primary sources of poor microbial water quality were at Malabar Beach, a technique for detecting the source of pollution was required. This led to the development of a model to investigate the sources of water in various locations in the embayment. The model was then used to attribute that water sources contribution to overall microbial pollution.

Monitoring to inform the model was carried out through the collection of high spatial resolution conductivity data from within the embayment and adjacent to stormwater drains to represent salinity. The use of conductivity data allows a determination to be made of the contribution from freshwater discharged by the stormwater drain to the overall salinity levels observed at each sample point. In turn Enterococci data from samples collected at the stormwater drain could be combined with conductivity data to estimate the levels of microbial contamination at various points on the beach. Enterococci results can also be used to indicate if sewage overflows contribute to poor water quality. Through this, the impact of the stormwater drains on microbial pollution at Malabar Beach could be determined.

Figure 6-1 and Figure 6-2 present results of the source detection and provide a diagrammatic representation of the beach, including the dark blue arrow at centre top indicating the path of the main council stormwater drain. Results indicated that levels of microbial pollution at Malabar Beach were sufficient to create poor beach bathing water quality, as indicated by the Beachwatch monitoring (OEH 2012). The primary source of this pollution was the main council stormwater drain. Generally levels of microbial pollution observed were not high enough to suggest inputs from sewerage related sources.

Figure 6-1 Enterococci estimates based on conductivity at Malabar Beach in wet weather

Figure 6-2 Enterococci estimates based on conductivity at Malabar Beach in dry weather

It was concluded that the likely outcome of diverting the stormwater to the otherwise now unused cliff face outfall would provide the maximum benefit for Malabar Beach water quality and ecological health. As such beach bathing water quality programs and a new monitoring program to assess ecological health at Malabar Beach was instigated to validate the expected benefits.

Methods

Beach bathing assessment

The objective of the beach bathing assessment was to ascertain if microbial water quality had improved after the stormwater diversion and if this resulted in improved recreational amenity for bathers at Malabar Beach.

The beach bathing assessment was based on the NHMRC (2008) guidelines for managing risks in recreational waters. This involves determining a Beach Suitability Grade for the use of each beach for swimming. In this case a microbial assessment category is assigned according to the 95th percentile of Enterococci densities from monitoring data. Table 6-1 presents the microbial assessment category thresholds. Sanitary Inspection categories for each beach are also assigned according to the presence of stormwater drains, sewage overflows, sewage bypasses, animals, toilets and bather densities. This leads to the assignment of categories of very high, high, moderate, low and very low. The results of these two assessments are placed in a matrix to find the beach suitability grade which can range from very good to very poor and include an option for follow up where further monitoring and assessment may be required.

Table 6-1 NHMRC (2008) recommended microbial assessment categories

Microbial Assessment Category	Standardised 95%ile Enterococci density (cfu/100mL)
A	<41
B	41-200
C	201-500
D	>500

Since the uptake of the NHMRC approach to assessing beach suitability for recreation and bathing, sanitary inspections have been conducted under the Beachwatch program. Enterococci densities have been routinely monitored at Malabar Beach, also as part of the Beachwatch program (OEH 2014) since the 1990s. This information was used to plot trends over the long term and to determine the Beach Suitability Grade for Malabar Beach before and after the stormwater diversion.

Intertidal community recovery assessment

Monitoring of rocky-intertidal communities assessed the potential ecological recovery from the cessation of stormwater discharges at the northern end of Malabar Beach. The structures of natural communities (without anthropogenic impacts) from two reference (control) sites were used in assessment of the drain (impact) site adjacent to the stormwater pipe.

Rocky-intertidal communities are comprised of macro algae and macroinvertebrates. These organisms will also colonise a variety of man-made structures such as breakwaters, jetties, docks, groynes, dykes and seawalls (Crowe et al., 2000). Rocky-intertidal community structure was recorded from wave-exposed ocean headland locations on naturally occurring rock platforms that could be safely accessed at low tide. Sites were selected to have similar levels of wave exposure in an attempt to minimise this natural influence. Wave exposure is known to influence distribution and abundance of rocky-intertidal communities between exposed headlands and sheltered bays or inlets (Crowe et al., 2000).

Photo quadrat measurements were taken 19 months apart. The initial images, formed the before period, were captured during the construction period of the stormwater diversion. The second set of images formed the after period. At each period eight replicate photo quadrats were taken at each site.

In the period before stormwater was diverted away from Malabar Beach, an extensive cover of green macro algae occurred on the intertidal rock platform adjacent to the stormwater outfall (Figure 6-3). The study of cessation of shoreline effluent discharge at North Head and Malabar recorded a decrease in the percentage cover of green macro algae together with an increase in other species present to levels comparable with reference locations (Archambault et al. 2001). Hence the statistical analysis of Malabar Beach intertidal rock platform data should focus on changes in community structure.

Prior to multivariate analysis of community data, data were transformed with a fourth root transformation and an association matrix was constructed based upon the Bray-Curtis resemblance measure. The Bray-Curtis resemblance measure is focused on compositional changes in taxa identities (Anderson and Walsh 2013).

Data patterns were visually displayed in an ordination plot of the rocky-intertidal community photo quadrat data. The nonmetric multidimensional scaling (nMDS) ordination routine of PRIMER was employed to produce a two–dimensional ordination plot. In this plot the relative distance between samples is proportional to the relative similarity in taxonomic composition and abundance – the closer the points on the graph the more similar the community (Clarke 1993).

The group average classification technique was then used to place the photo quadrat samples into groups, each of which had a characteristic community structure based on relative similarity of their attributes. This classification technique initially forms pairs of samples from the most similar taxa and gradually fuses the pairs into larger and larger groups (clusters) with increasing internal variability.

The similarity percentages (SIMPER) routine was used to explore which taxa were principally responsible for differences between sets of samples defined a-priori. These groups were from control and the stormwater drain site. This routine employed Bray Curtis similarities to examine the contribution of individual taxa to the average similarity between groups and within groups.

Data were then further explored with hypothesis testing. An asymmetrical permutational multivariate analysis of variance (PERMANOVA) model was constructed. This was comprised of fixed factors ‘Control / Impact’ and ‘Before / After’, together with a random factor ‘Site (Control / Impact)’ where Sites were nested within ‘Control / Impact’. The outfall site was the only site under the ‘Impact’ location and the two reference sites formed the ‘Control’ location.

Figure 6-3 Pre stormwater diversion image (2012) looking toward the water at the northern end of Malabar Beach

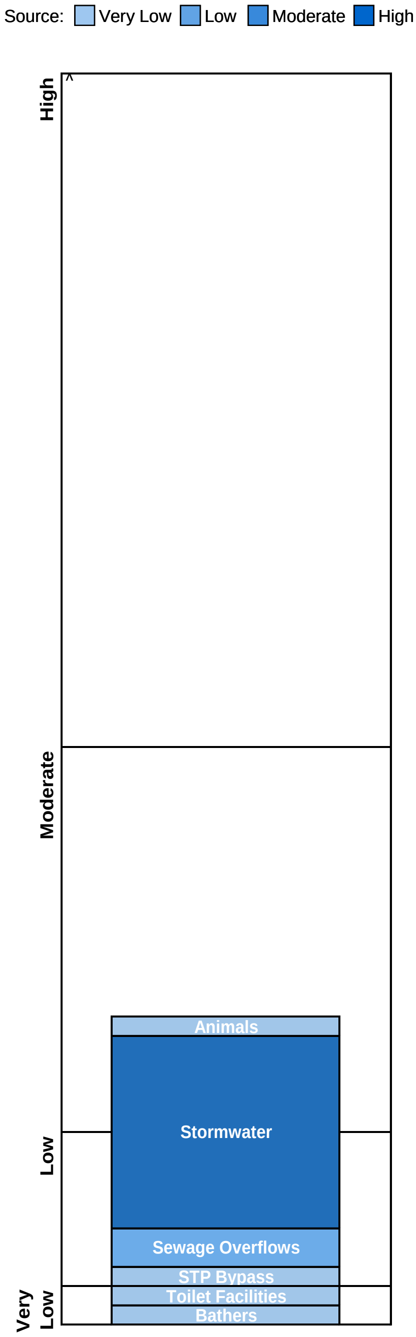
Figure 6-4 Post stormwater diversion image (2014) looking toward the water at the northern end of Malabar Beach

Results

Beach bathing

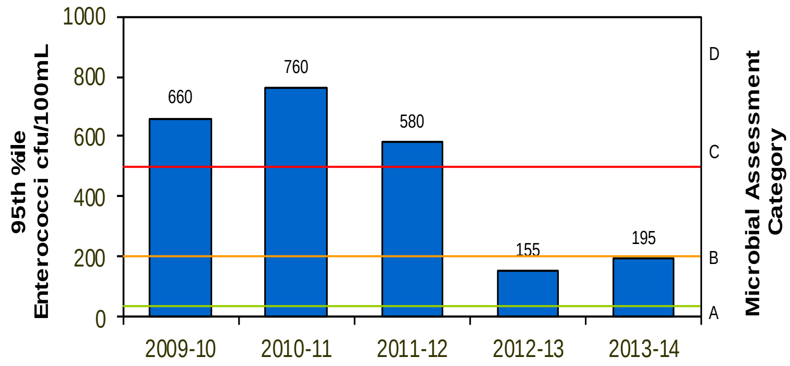
Results of the beach bathing water quality assessment, based on Beachwatch monitoring by OEH (2014) indicate a definite improvement in the recreational amenity of Malabar Beach after the stormwater diversion (Figure 6-5). The sanitary inspection and microbial assessment gave a beach suitability grade after the diversion of Good. Trends in Enterococci also reflect this, with the proportion of results accounted for by microbial assessment categories A and B being the highest in 2012-2013 and 2013-14 over 20 years of data. These results align with the expectation of improved beach water quality due to the stormwater diversion. This also supports the conclusion of the source detection study that the main stormwater drain was responsible for the majority of microbial pollution on Malabar Beach.

Sanitary Inspection: Moderate



Microbial Assessment: B

Monitoring period for 2013–14 result is June 2012 to April 2014.



Beach Suitability Grade assessment

Year	Enterococci 95%ile (cfu/100mL)	Microbial Assessment Category	Sanitary Inspection Category	Beach Suitability Grade
2011-2012	580	D	Moderate	Poor
2012-2013	155	B	Moderate	Good

Trends in Enterococci data through time

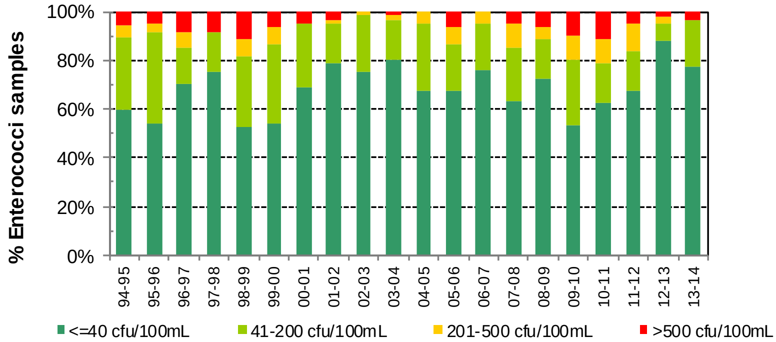


Figure 6-5 Beach Suitability Grade assessment and long term trends for Malabar Beach

Intertidal community recovery

The MDS ordination of photo quadrat data displayed after period samples from the stormwater impact site to be similar to the southern reference site. While before period samples from the stormwater impact site were clearly separated from the two reference sites, with one exception (Figure 6-6). This indicates an obvious change in the community structure at the stormwater impact location between before and after periods.

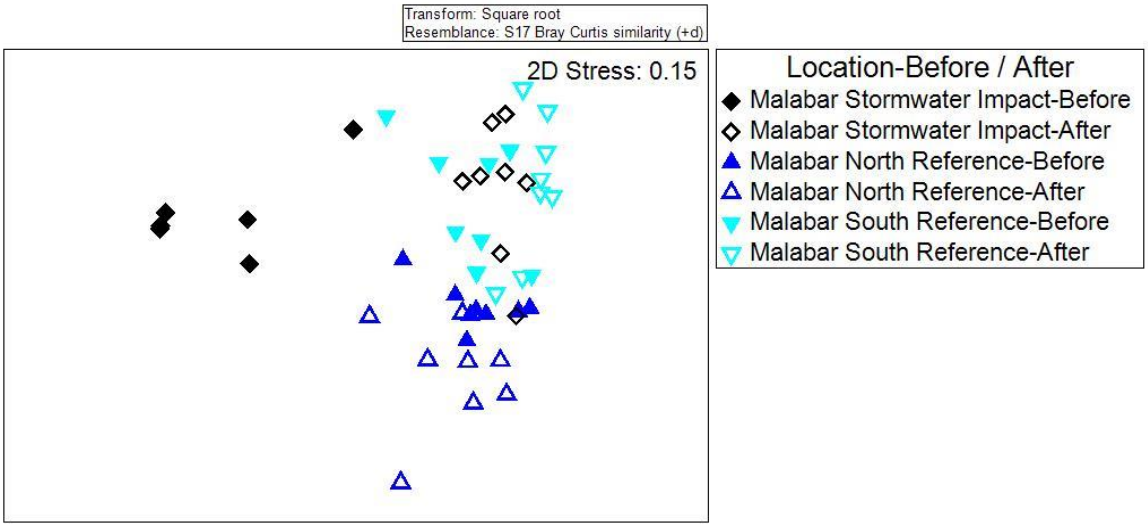


Figure 6-6 Two dimensional non-metric MDS ordination plot of rocky intertidal community structure

The tree diagram output from the group average classification analysis was checked to see if control (north and south reference sites and stormwater drain (a-priori) groups of samples were separated high up in the tree diagram. This was the case, with the first split separating seven of the eight 'stormwater impact before period' samples from all other samples (Figure 6-7). This plot confirmed a change occurred in the community structure of the rocky intertidal platform at the stormwater impact location in the period after diversion of the stormwater discharge.

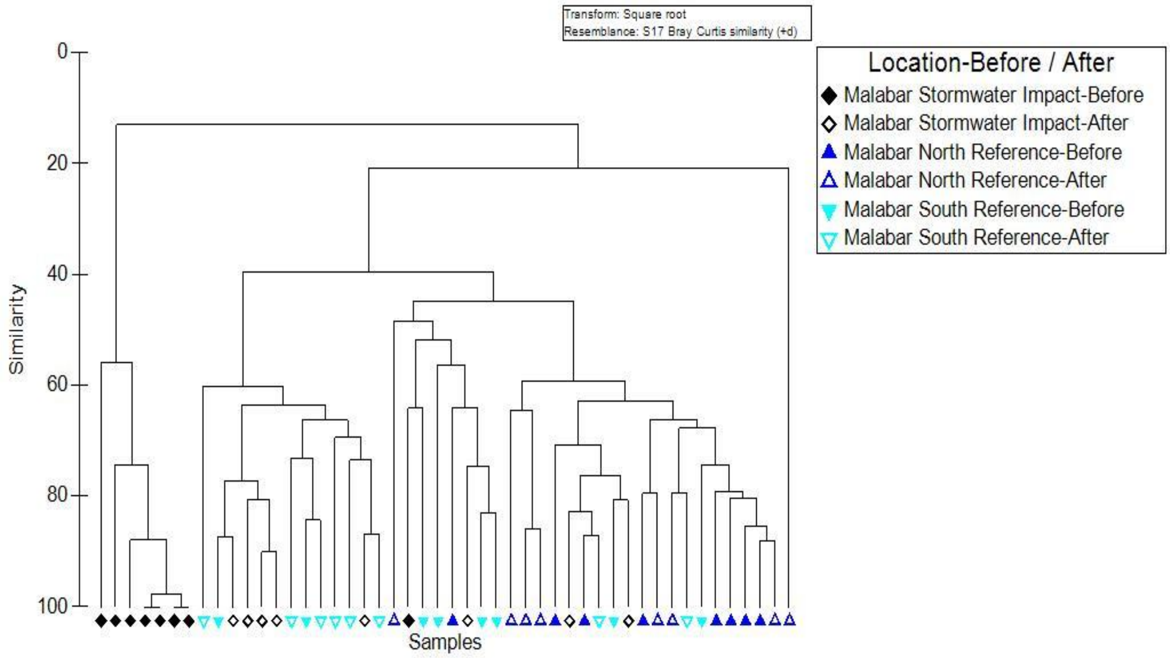


Figure 6-7 Tree diagram of rocky intertidal community structure at stormwater impact site and two reference sites before and after the stormwater diversion

Simper analysis of before after period data indicated the percentage contribution of each taxon to the community structure (Figure 6-2). A clear change in taxonomic composition of the community at the stormwater impact site occurred between before and after periods. In contrast at the reference sites most taxa persisted between before and after periods although abundances varied. Variability in community structure at the two control sites between the before and after periods was not surprising, as Underwood and Chapman's (1998) study of sheltered rocky-intertidal communities generally supported the view that communities on rocky-intertidal shores are haphazardly constructed and temporally dynamic in composition.

Table 6-2 SIMPER analysis results with percentage contribution of each taxon to community structure for each location before and after the stormwater diversion

	Malabar impact		North reference		South reference	
	Before	After	Before	After	Before	After
Green algae (*Chlorophyta*)	99.29					
Red Algae (*Rhodophyta*)	0.71				7.04	
Nerite (*Nerita Nertidae*)		10.11	41.07	32.63	10.95	10.09
Conniwinks (*Lottorinidae Bembiciume*)		4.43	30.84	44.48	11.12	9.92
Brown algae (*Phaeophyta*)		26.5	14.44	6.25	24.80	
False limpets and rock limpets (*Patellogastropoda)*		57.15	9.46	1.79	37.61	78.11
Zebra top shell (*Trochidae Austrocochlea*)		1.69	2.55	3.25	5.60	1.88
Barnacles (*Cirripedia*)			1.31	9.90		
Oyster borer (*Muricidae Morula marginalba*)		0.42	0.33	0.23	2.54	
Chitons (*Neoloricata*)					0.34	
Periwinkles (*Nodilittorina*)				1.20		
Star Fish				0.28		

A significant PERMANOVA result was returned for the interaction term 'Before / After' x 'Control / Impact' which indicated an ecological change had occurred after the stormwater diversion (Figure 6-3). This test result was further explored with a pairwise test of the 'Before / After' x 'Control / Impact' factor. Under the before period a P(MC) value of 0.0627 was returned, which indicated a weak non-significant difference between the community structures at the stormwater impact site and control sites. In contrast under the after period a much stronger non-significant difference was indicated by a P(MC) value of 0.7709. This suggested the community structures were similar at the stormwater impact site and control sites. While both results were non-significant the degree of difference in probability values reflects a change in community structure occurred between time periods.

Table 6-3 PERMANOVA results comparing fixed factors control/impact and before/after

Source	df	SS	MS	Pseudo-F	P(MC)
Before / After	1	14420	14420	5.2178	0.0591
Control / Impact	1	15656	15656	1.1156	0.4593
Location (Control/Impact)	1	14034	14034	15.103	0.0001
Before / After x Control/Impact	1	17974	17974	6.5037	**0.0394**
Before / After x Location(Control/Impact)	1	2763.6	2763.6	2.9741	**0.0174**
Residuals	42	39027	929.22		
Total	47	97582			

Summary and conclusions

The source detection study, using intensive spatial monitoring and modelling of the Malabar Beach embayment, indicated that the majority of microbial pollution was likely caused by the main stormwater council drain discharging to the beach. The viability of the methods used for source detection and the benefits of the stormwater diversion were confirmed. Microbial monitoring of the beach found a notable improvement in the Beach Suitability Grade for recreation after completion of the diversion.

A localised ecological change was also expected in response to the stormwater diversion and was detected at the northern end of Malabar Beach where the stormwater discharged to. Community structure of this rocky intertidal platform is now more typical of that observed at the control sites that are unaffected by stormwater and sewage.

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Theme three: Sensitivity of the receiving environment

The purpose of the 'sensitivity of the receiving environment' theme is to improve understanding of the current condition and resilience of the receiving waters. A water quality disturbance resulting from the input of wastewater discharges or stormwater does not necessarily reflect a deleterious impact in a waterway. Improved knowledge of the natural variations in a waterway and how they respond to inputs of wastewater or stormwater improves the ability to detect or predict an impact in that waterway. A highly sensitive receiving environment will be more susceptible to water quality and ecological conditions being pushed beyond the bounds of existing variability when a discharge containing pollutants occurs. This is when the potential is highest for a deleterious impact on the receiving environment.

Two case studies are presented in this theme. The first case study presents an analysis of the effect of longer term fluctuations in climate on the oceanography of ocean environments off Sydney's coast and of the performance of the deepwater ocean outfalls with respect to design criteria. The second is an assessment of the effects of a large sewer overflow incident at Glenfield on the Georges River in November 2013, including recovery rates for the river.

7 Assessing long term oceanographic fluctuations using deep water ocean outfall plume models

Abstract

Sydney's deepwater ocean outfalls have been operating for almost 25 years, discharging treated wastewater to the waters offshore from Sydney. Numerical modelling is undertaken to help assess the environmental performance of these outfalls. Model output showed plume characteristics were within the original design criteria, and that differences over time were not statistically significant. Long period fluctuations in the ocean currents were examined again showing no meaningful temporal trends in the data. From these results, it is concluded that the deepwater ocean outfalls continue to operate well within the original design criteria and that there has been little or no change in performance over time.

Introduction

Approximately 80% of Sydney's wastewater is treated and discharged through three deepwater ocean outfalls. The outfalls were commissioned in the early 1990s in response to "pollution of ocean bathing beaches and accumulation of contaminants in marine biota near the (old) shoreline outfalls" (Philip and Pritchard, 1996). The outfalls service the plants located at North Head, Bondi and Malabar and discharge primary treated wastewater into waters between 60 m and 80 m deep, between 2 km and 4 km offshore (Figure 7-1). The configuration of each outfall is broadly similar, differing mainly as a result of the different populations served by the wastewater systems.

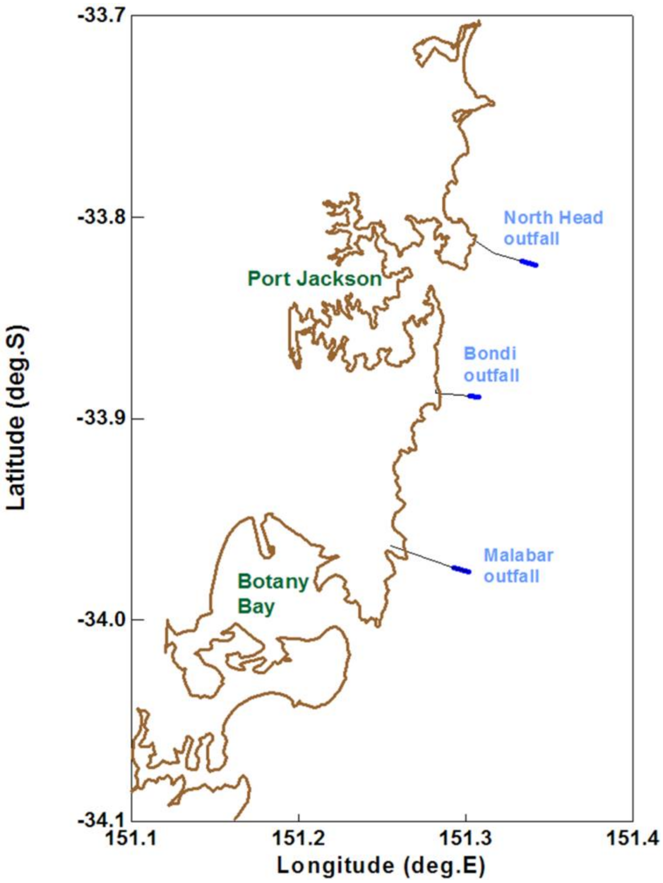

Figure 7-1 Schematic showing the locations of the three deepwater ocean outfalls

The Environment Protection Authority (EPA) managed an extensive, five-year environmental monitoring program (EMP) to examine the environmental performance of the deepwater ocean outfalls during the first two years of their operation. Results from these studies were published in a special edition of Marine Pollution Bulletin (MPB, 1996). Results from the EMP found that "the deepwater outfalls performed well during the first two years of their operation: they mitigated most of the environmental problems previously experienced when shoreline outfalls were operating without creating any major new problems in the ocean waters in the short term" (Philip and Pritchard, 1996).

However, Philip and Pritchard (1996) do note that the duration of the EMP was relatively short (compared with the design life of the outfalls) and ongoing monitoring is required to identify and quantify whether accumulative impacts are occurring. A monitoring program to help assess potential environmental impacts forms part of Sydney Waters current environment protection licence.

Tacit to the ongoing environmental performance of the deepwater outfalls is that they are operating, as designed, into the long term. A major element in assessing this performance is provided by near-field numerical modelling. The work presented here examines the results from the near-field modelling, compares them with the original design criteria and examines potential long-term trends.

How the deepwater ocean outfalls work

Wastewater from the deepwater ocean outfalls is discharged as a buoyant jet – that is, at velocities much higher than the surrounding ocean currents and at densities much less than that of the ocean waters. These two factors cause ocean waters to be entrained into the wastewater plume. The velocity difference between the wastewater and the ocean waters causes shear between these two fluids (Figure 7-2). The shear causes instabilities at the interface between the two fluids. As these instabilities grow they engulf (or entrain) ocean water into the wastewater plume which then becomes more dilute.

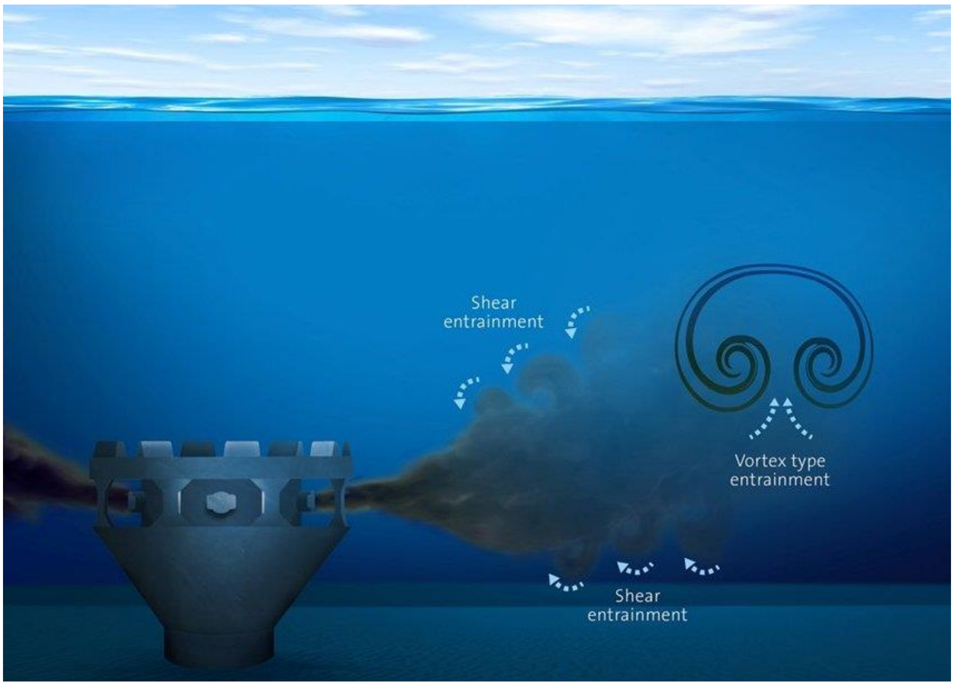

Figure 7-2 Schematic of an outfall diffuser and the entrainment process (not to scale)

The lower density of the wastewater (compared with the surrounding ocean water) causes the wastewater plume to rise through the water column. As the plume rises, it pushes through the ocean water above. This vertical movement of the plume causes a wake on the underside of the plume, resulting in vortex-type flow. Ocean waters are sucked into the plume (or entrained) via this vortex flow and the wastewater plume dilutes (Figure 7-2).

These two entrainment processes cause the wastewater plume to eventually have velocity and density properties close to that of the ocean waters. When that occurs, the vertical movement of the plume ceases. For Sydney's deepwater ocean outfalls, this process takes about 10 minutes and occurs within about a hundred metres of the discharge point (although these values can vary considerably, depending on oceanic conditions). Further dispersion of the wastewater occurs via oceanic turbulence. In general, about 90% of the wastewater dilution occurs via the entrainment process.

Ocean currents enhance dilution, enabling the organic material in the wastewater to be easily broken down by natural processes. The dominant current system off Sydney is the pole-ward flowing East Australian Current (EAC). This is an example of a western boundary current (WBC) and similar such currents occur along the western boundaries of all oceans (eg the Gulf Stream in the North Atlantic Ocean). WBC's are the intensification of ocean waters along the western boundaries of the ocean due to the rotation of the Earth. The mainstream of the EAC lies about 100 km offshore from Sydney and it is only the western edge of this current into which wastewaters from the deepwater ocean outfalls are discharged. The EAC is complex and highly variable. While currents in the mainstream EAC often reach speeds exceeding 2 m/s, the currents near the deepwater are typically in the range 0.2-0.5 m/s (but can exceed speeds of 1 m/s).

Stratification of the water column (ie the variation in water density) governs the height to which the wastewater plumes rise in the water column. If the wastewater plumes reach the surface of the ocean they may become visible and they may move towards the ocean bathing beaches, under appropriate wind conditions. Variations in density are dominated by variations in temperature and salinity. In summer, solar heating of the surface waters lowers their density compared with deeper waters and the water column becomes stratified. Results from the modelling indicate that a temperature difference of 1^{o}C over 50 m is sufficient to produce a submerged wastewater plume. This occurs more than 96% of the time. It is only during the coldest time of the year or when large storms break down the stratification, that the wastewater plumes reach the surface.

The near-field model

Modelling of ocean outfalls is usually carried out in two phases – the near-field and the far-field. The time and space scales for each phase are considerably different and it is not practicable to incorporate both phases into a single model. In the near-field the dominant processes responsible for the dilution of the wastewater are the momentum and buoyancy of the wastewater. Far-field wastewater dispersion is dominated by oceanic turbulence. Work here focusses on the results from the near-field modelling. Results presented here are provided at the “boundary of the initial dilution zone”. The distance from the discharge point to this boundary varies considerably, depending on ocean and discharge conditions. It is defined to occur when the vertical momentum and buoyancy of the wastewater are the same as that of the surrounding water. The near-field model automatically outputs this distance. The initial dilution zone is also referred to as the initial mixing zone or the end of the near-field.

A near-field model (PLOOM – Primary Lagrangian Ocean Outfall Model) was developed specifically for Sydney's deepwater ocean outfalls. PLOOM overcomes problems experienced by some other near-field models including:

- no restriction to the number of layers in the water column
- discharge can be either positively or negatively buoyant
- discharge can be in any direction to the ocean waters

- merging of discharges from individual outlet ports is automatically incorporated into the model
- entrainment allows for both shear and vortex type processes

The near-field model has been formally reviewed in international journals (Tate and Middleton, 2000; Tate and Middleton, 2004). Model calibration and validation is described in Tate (2002), based on laboratory modelling undertaken by Couriel and Wilkinson (1993) and field data results presented in Cox and Wilson (1993). Further, PLOOM has been compared with other near-field models including: CORJET (Jirka, and Akar, 1991; Jirka and Doneker, 1991), IMPULSE (Chu, 1976), JETLAG (Lee and Cheung, 1990) and OSPLM (Davidson, Knudsen and Wood, 1991). All models produce comparable results using a range of outfalls and environmental configurations.

Input data

Data required to operate the models include the configuration of outfalls, wastewater flow and oceanic conditions (profiles of currents and water density).

The configuration of the outfalls is essentially fixed, although it is possible to alter the number of outlet ports that are operating (which have remained fixed since commissioning of the outfalls). Wastewater flow data for each of the three deepwater ocean outfalls are provided by Sydney Water's HYDSTRA system. Ocean data are provided by a moored instrument system, designated the Ocean Reference Station (ORS). A schematic of the ORS showing its major components is presented in Figure 7-3.

The ORS is located approximately 3 km east of Bondi Beach in waters approximately 67 m deep. Commencing operation in November 1990, the ORS underwent a major re-configuration in May 2006. Since that time, the ORS instrumentation includes:

- a bottom mounted Acoustic Doppler Current Profiler (ADCP) returning current speed and direction data from every 2 m in the water column
- 14 temperature sensors located every 4 m in the water column to estimate density
- two conductivity, temperature, depth (CTD) sensors located about 10 m above the sea floor and about 10 m below the sea surface

All data are recorded internally at 5 min intervals. The ORS is serviced (nominally monthly) to upload data from the instruments. The data are examined under a Third Party Certified Quality Management System prior to dissemination to relevant authorities and as input to the near-field models. The models are run annually, outputting the estimated location, height of rise and dilution of the wastewater plumes.

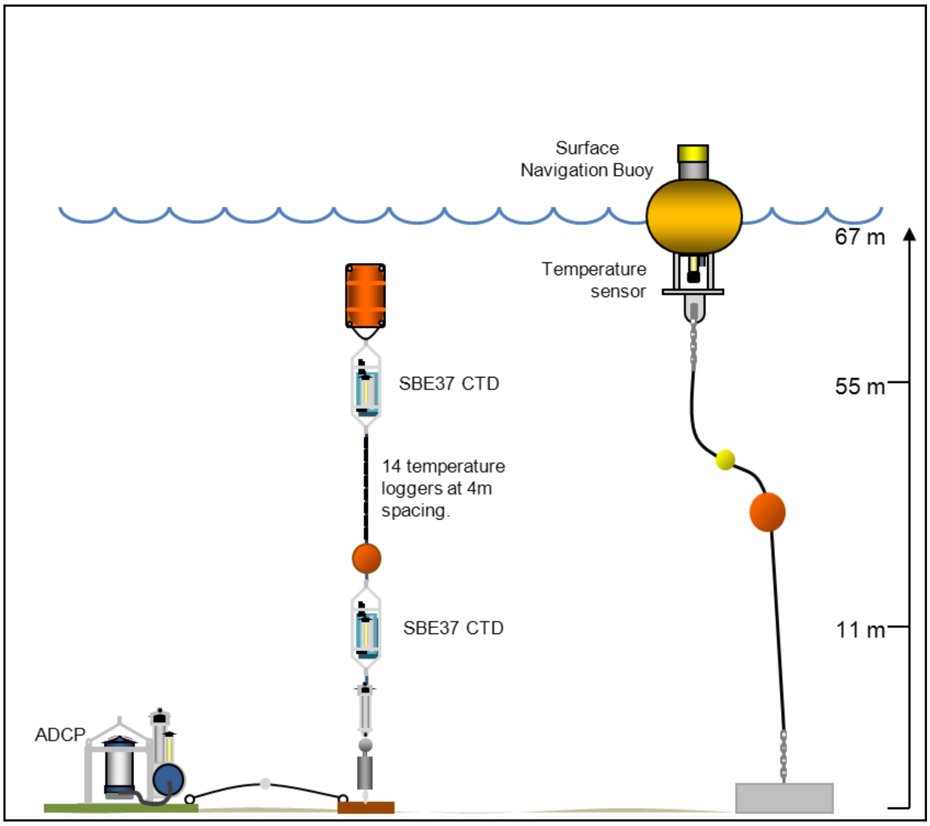

Figure 7-3 Schematic of the Ocean Reference Station

Results

Comparison with design criteria

CCE (1976) reported on the likely impacts that a proposed deepwater ocean outfall would have on the marine environment. Based on their observations (and other information) they provided preliminary deepwater outfall design criteria. In summary, these criteria include dilution at the boundary of the initial dilution zone should exceed 40:1 at least 98% of the time and that, between 1 November and 1 May, plumes should remain submerged at least 90% of the time.

Modelled plume dilutions at the boundary of the initial dilution zone are presented in Figure 7-4. Plume dilutions are highly variable, but generally lie between about 100:1 and 1,000:1. They are lowest in the warmer months when stratification of the water column is at its greatest. At these times, the wastewater plumes are trapped below the thermocline and have less receiving water with which to mix, resulting in lower dilutions.

Modelled dilutions that are exceeded 98% of the time from 2007 to 2013 are presented annually in Table 7-1 for each of the three deepwater ocean outfalls. These values always exceed the design criteria of 40:1. Further, the annual values vary little from year-to-year. Highest dilutions are observed for the Bondi discharge (corresponding to the lowest population served) and the lowest dilutions are obtained for the Malabar discharge (corresponding to the highest population served).

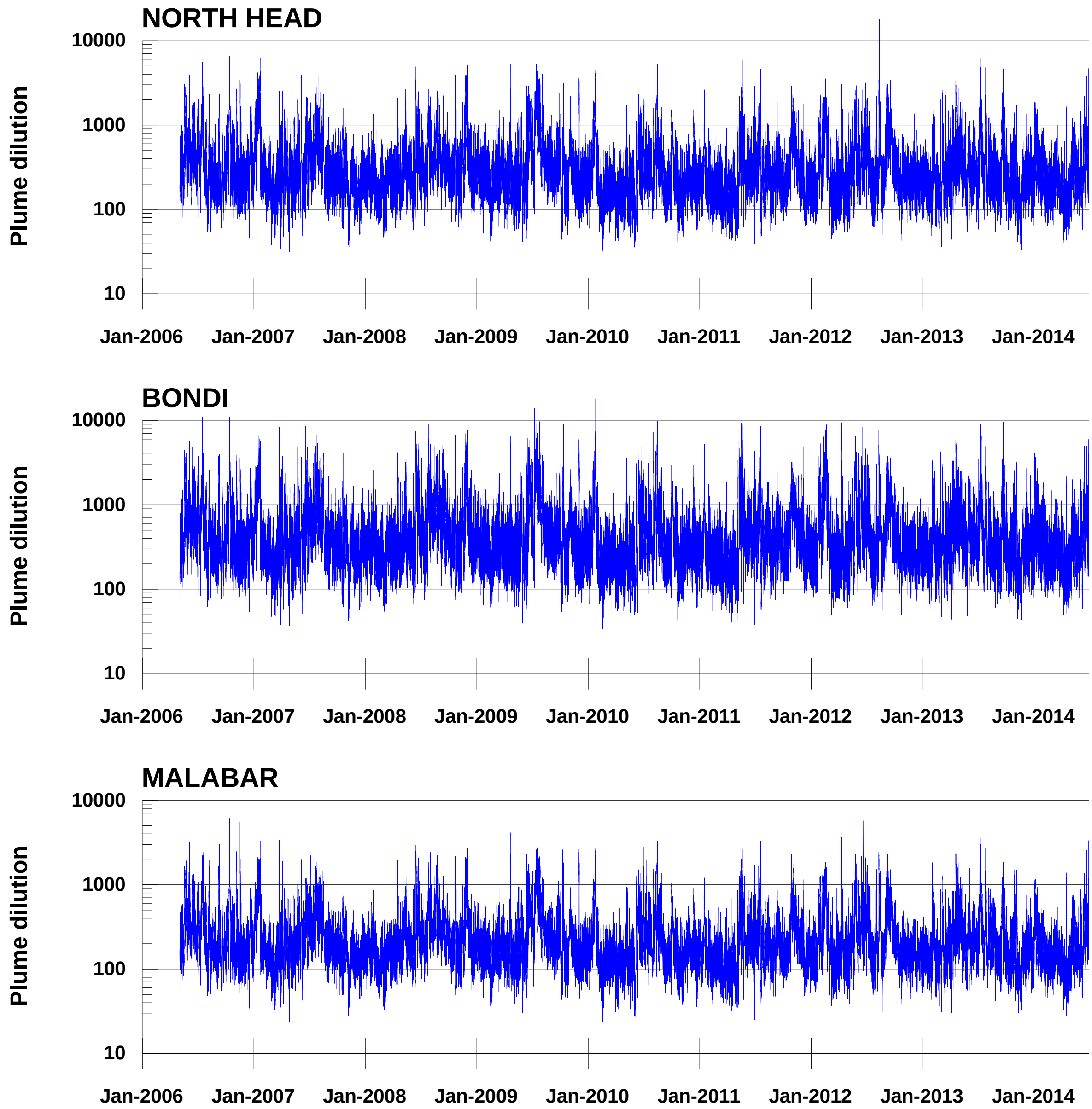

Figure 7-4 Modelled estimates of the plume dilutions for each of the three deepwater ocean outfalls between May 2006 and June 2014

Table 7-1 Dilution exceeded 98% of the time. Design criteria >40

Year	North Head	Bondi	Malabar
2007	74.3	96.2	57.5
2008	86.0	109.0	67.7
2009	78.1	103.0	63.7
2010	66.7	85.0	52.7
2012	75.2	94.9	56.0
2013	81.3	99.5	65.5
2014	74.3	85.1	59.3

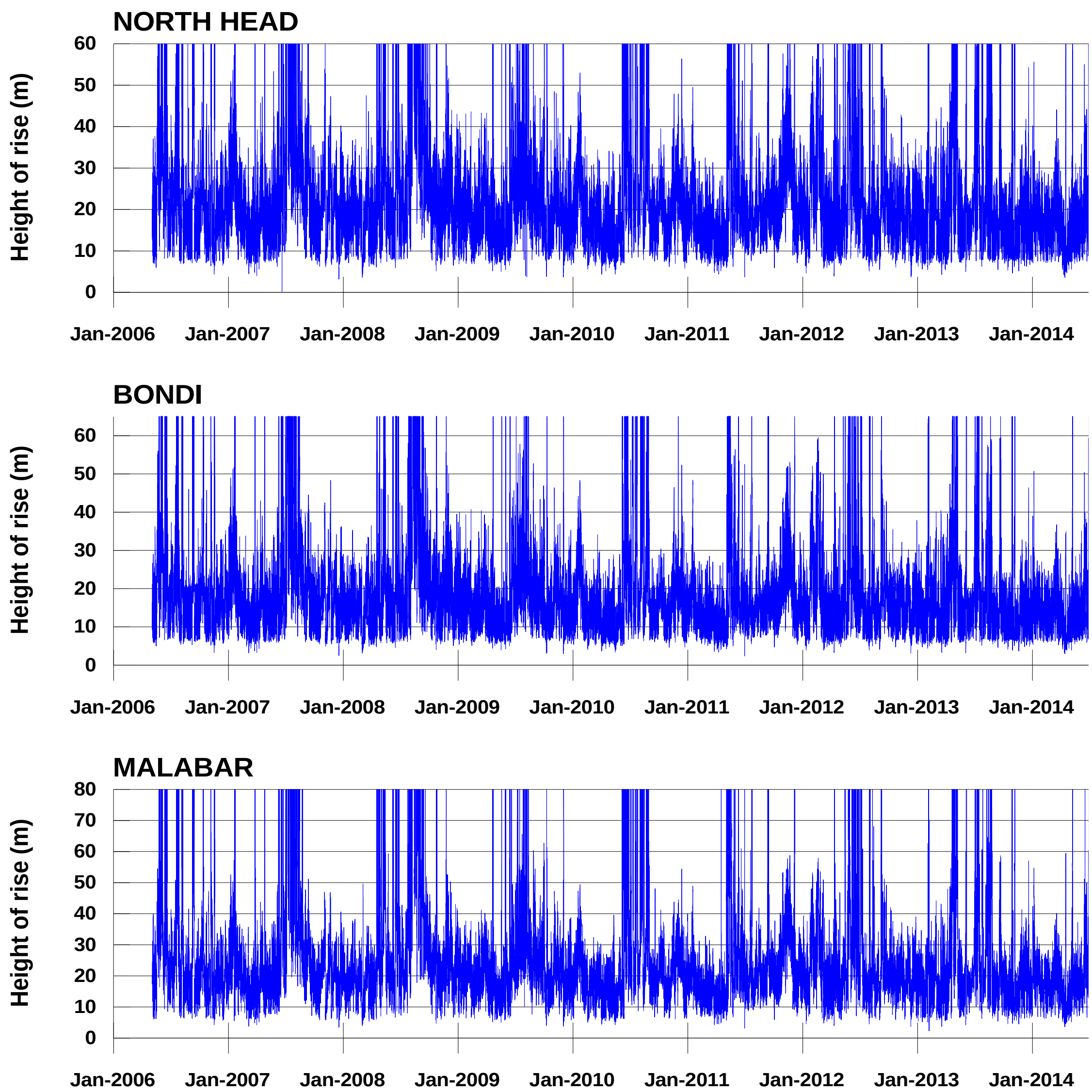

Figure 7-5 Modelled estimates of the height of plume rise for each of the three deepwater ocean outfalls between May 2006 and June 2014

Modelled maximum heights of plume rise are provided in Figure 7-5. The plots show a clear annual pattern with the greater heights of plume rise occurring in the colder months (when the thermal stratification is least) and low heights of plume rise in the warmer months. Generally, heights of plume rise are less than 40 m, although this value is highly variable.

The percentage of time the plumes are submerged (between 1 November and 1 May each summer) is detailed in Table 7-2. For all years and all outfalls, this value is above 96%. The design criteria specified at least 90%, although CCE (1976) modelling predicted this to be at least 96%.

Table 7-2 Percent of time during summer (between 1 November and 1 May) that plumes remain submerged. Design criteria > 90%.

Year	North Head	Bondi	Malabar
2006/07	98.3%	98.9%	98.3%
2007/08	99.2%	99.6%	98.4%
2008/09	99.9%	99.9%	99.7%
2009/10	99.8%	99.9%	99.9%
2010/11	100.0%	100.0%	100.0%
2011/12	99.1%	99.9%	99.7%
2012/13	96.9%	98.5%	98.4%
2013/14	99.5%	100.0%	100.0%

Based on these design criteria, the deepwater outfalls are performing better than expected. (Although it is noted that engineering structures are inherently conservative, so these results are not unexpected). Philip and Pritchard (1996) also state that the "outfalls have performed as well or better than was predicted at the time the EIS's were prepared". There is no apparent trend in the modelled plume characteristics through time. This evidence suggests that the deepwater ocean outfalls are continuing to operate as (or better than) designed.

Long-period fluctuations

From a climate perspective, the deepwater ocean outfalls have only been operating for a relatively short period of time. Here, we place the last 25 years into the context of long-term climate conditions. The Southern Oscillation Index (SOI) is used as a surrogate for the long-term variations and this index is compared with ocean conditions. By inferring likely conditions into the future, it may be possible to identify whether the deepwater ocean outfalls will continue to perform as designed. If not, then planning can be made to mitigate potential future environmental impacts.

With such a long data set, the time series will be non-stationary and more commonly used time series analysis techniques, such as Fourier transforms, are not applicable. Wavelet analysis is applicable to non-stationary time series and is used here. The Morlet wavelet is used, primarily because it can accommodate complex time series (hence coherence can be determined) and its scale is directly related to the Fourier period. The cross-wavelet and wavelet coherence software was obtained from Grinsted et al (2004).

Current data from the ORS from the near-surface and the near-sea floor are compared with the SOI (obtained from the Bureau of Meteorology). ORS data were averaged into hourly bins, then rotated according to their principal axes (by about 15^{o} clockwise) to generate along-shore and across-shelf current components. Only the along-shore currents are used in this analysis. Current data were then filtered to remove tidal and inertial periods using the 51G113 filter (Thompson, 1983) and averaged into monthly bins.

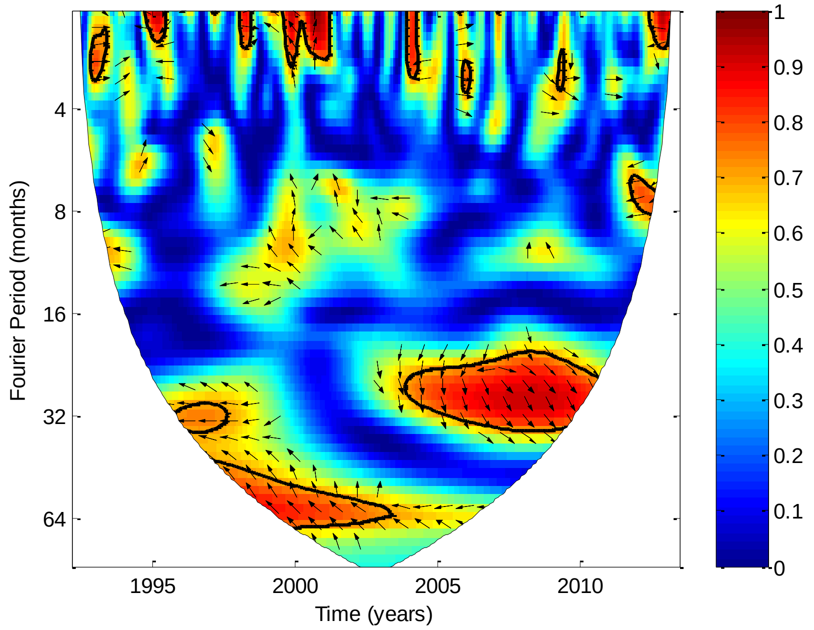

Figure 7-6 Wavelet coherence between the currents in the upper layer and the Southern Oscillation Index

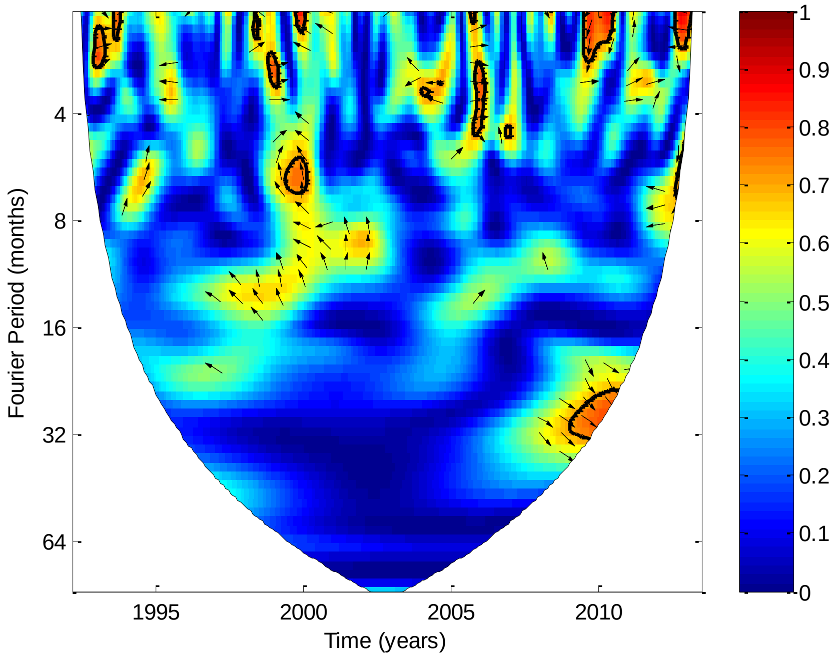

Figure 7-7 Wavelet coherence between the currents in the bottom layer and the Southern Oscillation Index

Wavelet coherence plots are presented in Figure 7-6 (upper layer currents and SOI) and Figure 7-7 (lower layer currents and SOI). In these plots, the 95% confidence limits are indicated by the solid black lines. The phase difference between the two time series is indicated by the arrows. Arrows pointing to the right indicate the two time series are in-phase, pointing to the left indicated they are 180^o out of phase.

For the currents in the upper layer, there is a significant coherence (above the 95% confidence limits) at a period between about 2 and 5 years. This region of high coherence is split into two, separated by the drought in eastern Australia in the early 2000s. Prior to the drought, the phase difference is about 90^o, with the SOI leading the currents. However, after the drought, the phase difference is about -90^o, with the SOI lagging the currents. This may suggest a different surface ocean current structure associated with an approaching El Nino event, compared with a receding event. There is little coherence between the near bottom currents and the SOI (Figure 7-7) across all times and Fourier periods.

An unsuccessful attempt was made to use the SOI (and other information including the Tasman Sea surface temperature) to estimate the along-shore currents and dilutions. Some 15,000 different sets of curves were trialed. The agreement between observations and predictions lay well below acceptable levels for all curves tested. The largest correlation coefficient, R^2 = 0.09, is not significant at the 5% level. Similarly, the largest Nash-Sutcliffe coefficient of efficiency was <0.09 indicating that the curves fitted were only slightly better than using the median value. Predicting the currents and plume dilutions using the SOI and Tasman Sea surface temperature do not appear possible. Attempts at these predictions using other parameters are presently underway.

Summary and conclusions

This report outlines the operation of the deepwater ocean outfalls and the results from the near-field modelling carried out between 1 May 2006 and 30 June 2014. Compared with the original design assumptions, the plumes from the deepwater ocean outfalls continue to operate better than anticipated in terms of dilutions achieved and frequency of surfacing plumes. Monitoring will continue into the future to ensure that this is maintained.

Using the SOI, ocean conditions between 1992 and 2014 are placed into a long-term context to assess potential conditions into the future and possible effects on the movement of wastewater from the deepwater ocean outfalls. The results from these analyses suggest that (a) there is a coupling of the SOI and the surface currents off Sydney during non-drought years and a decoupling during drought years, and (b) the lead up to an El Nino event is characterised by a changing phase difference between the SOI and the surface currents.

Unsuccessful attempts were made to predict the currents and wastewater plume dilutions using SOI and the Tasman Sea surface temperature. However, trials using other parameters continue to be examined. Such predictions will allow Sydney Water to plan for likely changes in plume dilution and movement.

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8 Assessing the ecological and recreational amenity impacts of a large sewage overflow at Glenfield on the Georges River in November 2013

Abstract

An electrical storm followed by intense rainfall on 22 November 2013 resulted in flooding in the inflow pumping station (SPS 353) at the Glenfield plant after a valve failed. This lead to the submergence of the station's pumps and caused the release of approximately 153 ML of untreated wastewater into Bunbury Curran Creek which flows into the Georges River. Sydney Water immediately moved to repair the pumping station and monitor and remediate any impacts in the Georges River.

A monitoring and assessment program was instigated in response to the event. The purpose of the program was to identify and quantify water quality, recreational amenity and ecological health impacts to the creek and the Georges River, and to chart the rate of recovery after the overflow event.

The monitoring program measured a number of water quality and environmental health indicators along the Georges River after the overflow event. Data included: macroinvertebrates and the Stream Invertebrate Grade Number Average Level (SIGNAL) scores as an indication of ecological health; microbiological parameters as potential factors for public health and recreational amenity; and key water quality parameters such as dissolved oxygen, nutrients, chlorophyll *a* and algal populations (focusing on cyanobacteria). The data were compared with all available 'pre overflow' data.

Results indicated that there was a change in some water quality parameters in the Georges River, however the *main effects* were limited to the reach from the confluence with Bunbury Curran Creek to Liverpool Weir on the Georges River. Chipping Norton Lake, downstream from Liverpool Weir, showed a pulse effect where some parameters immediately returned to background ranges, while others were slower.

Microbiological indicators were impacted by the overflow event with *median* values for faecal coliforms exceeding primary contact guidelines (ANZECC 2000) at the Cambridge Avenue site (site 2) and Liverpool Weir (site 8) immediately after the event. Dissolved oxygen concentrations declined to a level harmful to aquatic biota for approximately eight days at sites located immediately adjacent to the overflow ie the lower reach of Bunbury Curran Creek, and the confluence with the Georges River to Liverpool Weir.

Changes in cyanobacteria populations and algal community structures were not statistically linked to the overflow event. Before-after-control-impact analyses were not conclusive. However, total nitrogen, total phosphorus and chlorophyll *a* (an indicator of algal population growth) were all elevated during the 'event' period at the Cambridge Avenue site (site 2), Liverpool Weir (site 8) and one site in Chipping Norton Lake (site 12). These sites returned to almost background nutrient and chlorophyll *a* concentrations in the 'after' period (2 weeks later). Cyanobacteria populations were generally low with negligible potentially toxic cyanobacteria present, except for one site in Chipping Norton Lake in April. This occurrence is presumed to be unconnected to the overflow.

An assessment of stream health using analysis of macroinvertebrate populations, showed a decline in stream health that was not statistically significant. SIGNAL scores returned to the normal ranges experienced before the overflow incident, while lower estuary sites were unaffected.

These 'impacts' appeared to be indicative of a 'pulse' of contaminants. The lower estuary sites showed only mild perturbation, and then only at the site just downstream of the confluence with Harris Creek. These sites were only sampled for 8 days, up until 10 December 2013.

Introduction

A large sewage overflow occurred at the pumping station at Glenfield Water Recycling Plant (WRP) on 22 November 2013 after an electrical storm followed by intense rainfall. This resulted in flooding in the inflow pumping station (SPS 353) after a valve failed – a rare event for these pumping stations. The pumping station feeds floodwater into the plant. The pumps were submerged and rendered inoperable, allowing approximately 150 ML of diluted, but untreated wastewater and stormwater to discharge into Bunbury Curran Creek. The overflow may have caused a fish kill in the local creek. The station was back online within two days, and the damaged pumps were progressively repaired and replaced. From Thursday 28 November 2013, full pumping capacity of SPS 353 was restored.

The Department of Health requested a monitoring program to identify and quantify water quality and ecological health impacts to the creek and the Georges River and to chart the rate of recovery.

This case study assesses the sensitivity of the Georges River receiving waters to a large unexpected sewage overflow event from the Glenfield plant, and the consequent impact on the river's environmental values.

Two categories describe the environmental values of the river:

- Ecosystem health – protection of aquatic ecosystems
- Recreational amenity – primary and secondary contact recreation such as swimming, boating, water skiing and fishing (particularly in the Chipping Norton Lakes area)

This paper presents outcomes from the monitoring to address the following questions:

- Were there any observable changes in environmental indicators that can be linked to the overflow incident?
- If there was an observable change, what was the spatial and temporal extent of that change?
- Were the observed changes likely to have impacted the environmental values of the river?

Outcomes from this case study will inform future incident responses to minimise disruption to the community and potentially inform future decision making around wastewater infrastructure investment.

Methods

Approach

The environmental and recreational indicators presented in Table 8-1 were examined to see if there was any putative impact from the overflow event.

Monitoring data were compared with the ANZECC (2000) water quality guidelines and the NHMRC (2008) recreational water quality guidelines to assess water quality changes in the Georges River following the overflow. The NHMRC guidelines were used for the indicators*:* Enterococci, cyanobacteria and pH. The ANZECC guidelines were used for faecal coliforms, nutrients, chlorophyll *a* and dissolved oxygen.

Table 8-1 Environmental and recreational indicators

Indicator type	Indicator	Comment
Potential effects on ecological and environmental health of the river	Macroinvertebrates – SIGNAL SG	Macroinvertebrates provide a snapshot of the ecological health of the river.
	Water quality parameters: Chlorophyll *a* Total nitrogen Total phosphorus Dissolved oxygen pH	Chlorophyll *a* and nutrient species can be used to predict algal status and the eutrophic state of the river. Dissolved oxygen is necessary to aquatic fauna and low concentrations often result in fish kills. pH when high can indicate high vegetation productivity and pollutants, and when low can also cause injury to fish.
Potential effects on public health and recreational amenity	Algal status (Cyanobacteria) Enterococci Faecal coliforms	Potential harm to biota including humans from exposure to microcystin. Indicator of potential harm from pathogens

Study area

The Georges River is bedrock confined in its upper reaches down to Macquarie Fields, meanders through Chipping Norton Lake and downstream to Pleasure Point, where it is once again bedrock confined. It flows through natural bushland and mixed-rural land use upstream of the Glenfield plant, as well as part of the Campbelltown urban area. Much of the upper catchment is a protected water catchment (on the right bank) for the Woronora Reservoir. Downstream of the plant includes urban areas, light industry, recreational activities and an airport. Runoff from urban and agricultural areas transports nutrients, heavy metals and other contaminants to the Georges River freshwater reaches and Bunburry Curran Creek which joins the river at Glenfield.

The Georges River is freshwater upstream of Liverpool Weir, while saline and tidal downstream. The downstream estuarine reach extends to Botany Bay.

Table 8-2 and Figure 8-1 present locations and descriptions of study sites.

Table 8-2 Study sites on the Georges River (historical site identification codes are in brackets)

Site name	Description	Site Type	Location	Lat. MGA94	Long. MGA94
Site 1	8 m downstream of the SPS 353 overflow outlet	Downstream, impact	Bunbury Curran Creek	-33.9828	150.9037
Site 2	upstream side of the Cambridge Avenue bridge (GR23)	Downstream, impact	Georges River	-33.9701	150.912
Site 3	~140 m upstream of the confluence with the Georges River	Downstream, impact	Bunbury Curran Creek	-33.9792	150.9097
Site 4	~10 m upstream of the overflow point, SPS 353	Upstream control	Bunbury Curran Creek	-33.9831	150.9033
Site 5	End of Belmont Road	Downstream, impact	Georges River	-33.9762	150.9114
Site 6	Pavillion in Helles Park	Downstream, impact	Georges River	-33.9369	150.921
Site 7	End of Victoria Road, Glenfield	Upstream control	Georges River	-33.9869	150.9089
Site 8	Upstream edge of Liverpool Weir (GR22)	Downstream, impact	Georges River	-33.9254	150.9286
Site 9	North east point of Haigh Park	Downstream, impact	Georges River	-33.9232	150.9381
Site 10	Sewer pipe across the Georges River at Epsom Road	Downstream, impact	Georges River	-33.9187	150.9527
Site 11	Upstream site of Ingleburn Weir (GR24)	Upstream control	Georges River	-34.0067	150.8881
Site 12	Angle Park boat ramp (GRe12)	Chipping Norton Lake	Georges River	-33.9063	150.9528
Site 13	Grand Flaneur Beach	Chipping Norton Lake	Georges River	-33.9057	150.9574
Site 14	End of Georges River Road, north shore of the lake	Chipping Norton Lake	Georges River	-33.8989	150.9593
Site 15	Davy Robinson boat ramp, ~900 m downstream of Newbridge Road bridge	Downstream, estuarine	Georges River	-33.9307	150.9692
Site 16	150 m downstream of the confluence with Harris Creek (GR19), mid-channel	Downstream, estuarine	Georges River	-33.9728	150.9955
Site 17	Confluence with Salt Pan Creek (GR18), mid-channel	Downstream, estuarine	Georges River	-33.9773	151.0365
Site 18	Confluence with Woronora River, mid-channel	Downstream, estuarine	Georges River	-33.9938	151.0693
Site 19	Georges River at the entrance to Kogarah Bay	Downstream, estuarine	Georges River	-33.9997	151.1193
Site 20	Woolooware Bay (GR09), mid-bay	Downstream, estuarine	Georges River	-34.0224	151.1403
Site 21	Georges River entrance to Botany Bay	Downstream, estuarine	Georges River	-33.9993	151.1502
Site 22	Pool on Bunbury Curran Creek, ~100 m from the Georges R.	Downstream, impact	Bunbury Curran Creek	-33.9790	150.909

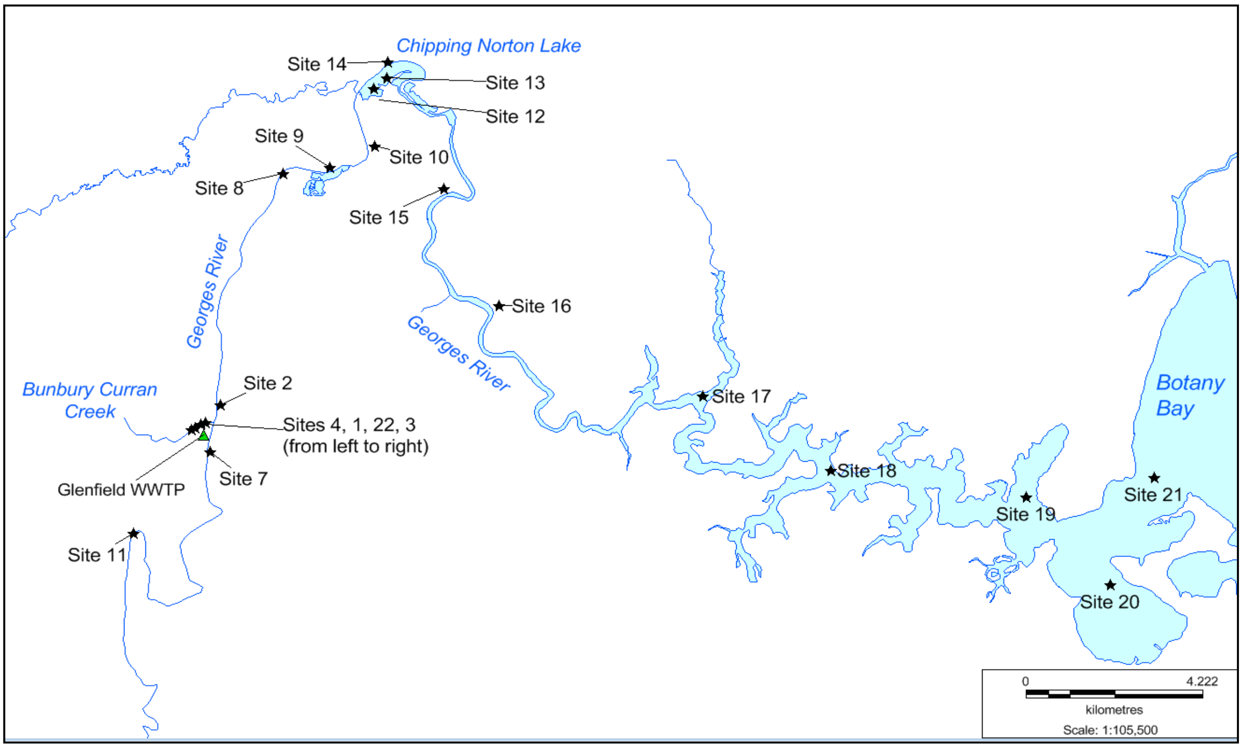


Figure 8-1 Study area with sampling sites ★, and the Glenfield plant ▲

Data analysis

Different parameters and sites had different sampling regimes, available historical data and suitable analysis techniques. As such, not all data could be compared long term.

At the start of the incident, Sydney Water followed the normal Environmental Response protocols and a specific number of variables were examined. As the incident unfolded additional variables were added, consistent with those used in previous programs (EIMP 1995 to 2008). Macroinvertebrate data was available under both EIMP and STSIMP (2008 to 2014) programs.

Data analysis is divided into three groups where the indicators are treated differently: water quality; free-floating algal communities; and stream health.

Software and statistical analyses

The software used included SAS Version 9.4 software package (SAS Institute Inc., Cary, NC, USA) for ANOVA tests and PRIMER Version 6.1.16 software package (Clarke and Warwick 2001) and the PERMANOVA+ Version 1.0.6 (Anderson *et al*, 2008) add-on module to PRIMER for multivariate analyses. In general:

- Before processing in PRIMER, the dataset was checked for homogeneity, normality and where necessary, log 10 transformed prior to hypothesis testing. The basis for hypothesis decisions was Type III Sums of Squares.
- Principal Component Analysis (PCA) was used to discern the trend of impact and recovery
- Summary statistics, box and whisker plots and time series plots were generated for all water quality data (using Palisade *StatTools* for univariate statistics and Excel for plotting)

Table 8-3 describes the sites and parameters used in the water quality analysis. The parameters analysed were dissolved oxygen, pH, ammonia, total nitrogen, total phosphorus, chlorophyll *a* (where available), faecal coliforms and Enterococci.

Analysis of chlorophyll *a* used a BACI (Underwood and Chapman 1995) design with formal hypothesis testing. To further explore trends in chlorophyll *a,* formal hypothesis testing of Georges River sites 7 and 12 was conducted based on data from event and after periods. Site 7 is upstream of the Bunbury Curran Creek inflow, while Site 12 is in the upper estuary. The ANOVA model was comprised of two fixed factors 'Site' and 'Period'. An interaction term was also possible to test 'Site by Period'. To better meet the assumptions of ANOVA data were 'log 10' transformed.

Period of assessment

Water quality data periods for the purposes of statistical analyses included:

- 'before' – July 1996 to June 2008 depending on data coverage (filtered to remove wet-weather events)
- 'event' – 22 November to 12 December 2013 (average daily data)
- 'after' – 18 December 2013 to 23 June 2014, using unfiltered data

For specific PRIMER analyses:

- 'event1' – 23 to 30 November 2013
- 'event2' – 1 to 5 December 2013
- 'event3' – 7 to 9 December 2013
- 'after' – 18 December 2013 to 23 June 2014 with weekly sampling, averaged by month

Data filtering for the 'before' period was based on dry weather conditions. Data was omitted from the analyses where the rainfall was greater than 10 mm over the previous 72 hours (unless otherwise stated in the text). Rainfall data from three stations were combined to calculate the dry weather dates. The rainfall stations used were Fairfield (station 567077), Glenfield (station 567078) and Liverpool (station 566049).

Replicate data was collected during the 'event' period at some sites, such as in Bunbury Curran Creek. Replicate data was averaged for analysis, unless otherwise stated, such as for dissolved oxygen analysis.

Water quality comparisons included:

- Georges River sites 2, 8, and 12 comparing the before, event and after periods
- Georges River sites 17 and 20 comparing the before and event periods as representative of the lower estuarine reaches
- Georges River sites 7, 2, 8, 9, 10, 12, 13, 14 and 15 comparing the event and after periods to assess the downstream affects to Chipping Norton Lakes
- Bunbury Curran Creek sites 4, 1 and 3 comparing the event and after periods
- mid and outer estuary Georges River sites 16, 18, 19 and 21 in the event period to see if there were immediate downstream affects

The sites examined in the post-incident monitoring program covered various periods depending on whether there was historical data (Table 8-3). The 'before' data were available for five sites, while the 'after' period covered 13 of the 20 sites. Inconsistencies in the number of sites covered in each period limited the statistical analyses performed.

Table 8-3 Sites and parameters for water quality characterisation

Sites	Available data	Parameters	Rationale
2, 8, 12	Before, event, after. Historical data from 1995.	DO, pH, AMM, TN, TP, FC, Ent	
17, 20	Before, event (up to 2/12/2013).	DO, AMM, TN, TP	Lower estuary downstream effects
9, 10, 13, 14, 15	Event, after	DO, AMM, TN, TP, FC, Ent	Local downstream effects
7, 4	Event, after	DO, AMM, TN, TP, FC, Ent	Two upstream control sites, one in BCC* **
1, 22, 3	Event, after	DO, AMM, TN, TP, FC, Ent	Local effects in BCC
16, 18, 19, 20, 21	Event only	DO, AMM, TN, TP, FC, Ent	Recovery times in the lower estuary
7, 12	Event, after	Chlorophyll *a*	Indicator of algal activity

* Bunbury Curran Creek
** Both of these sites are downstream of urbanised areas and as such had impaired water quality
DO = dissolved oxygen, pH = pH units, AMM = ammoniacal nitrogen, TN = total nitrogen, TP = total phosphorus, FC = faecal coliforms, Ent = Enterococci

Free-floating algal communities

Algal status was determined by using multivariate analysis of community structure based on phylum taxonomic groups of free-floating algae. Periods covered 'before' and 'after' for sites 2, 8 and 12.

'After' period data includes weekly samples collected between 18 December 2013 to 29 January 2014, 27 May to 23 to June 2014, and 22 April 2014.

Prior to analysis, the data was square root transformed. Rare taxa were removed when observed in only one sample. For each site sample, data were averaged by phylum group. Methods included:

- Permutational multivariate analysis of variance (PERMANOVA) with Type III sums of squares
- The Bray-Curtis measure of similarity between the fauna of each pair of samples
- Cluster analysis, to group sites by characteristic taxa using the relative similarity of their attributes
- Nonmetric multidimensional scaling (nMDS) ordination plots used to check the validity of the output from classification techniques

Stream health

The Stream Invertebrate Grade Number Average Level or SIGNAL is a biotic index used to assess stream health. In this case the Sydney region version of SIGNAL-SG (Chessman et al 2007) was used. This index follows that outlined in Besley and Chessman (2008) that used habitat data from autumn and spring seasons. That work demonstrated impacts and recovery from wastewater discharge using multi-season and year data. The Sydney region specific version is considered to provide a more sensitive assessment than afforded by the state-wide derived SIGNAL2 grades as SIGNAL2 grades were used in an initial step in the calculation of the Sydney version. The Sydney region specific version employed here is also based on finer genus level taxonomy compared with SIGNAL2 that is based upon coarser family level taxonomy. The Sydney region version has 367 grades versus 174 grades for SIGNAL2.

Sydney Water sampled three sites in the Georges River between 1995 and 2013 twice per year (spring and autumn) as part of the STSIMP and preceding EIMP. These data represent the 'before' period. The sites include:

- Site 11 - Georges River at Ingleburn Reserve (upstream reference site)
- Site 2 - Georges River at Cambridge Causeway (downstream impact site)
- Site 8 - Georges River at Liverpool Weir (spatial distant downstream impact site). Liverpool Weir is a barrier to saline estuarine water which limits the extent of freshwater macroinvertebrates.

Sydney Water conducted two other surveys after the initial post incident survey, about eight weeks apart, adding three 'post' event data points:

- Post incident 2013
- Post incident 2013 - Summer 2014
- Post incident 2013 - Autumn 2014

The Stream Invertebrate Grade Number Average Level biotic index, Sydney genus taxonomic level version (SIGNAL-SG, Chessman et al 2007), allows the calculation of stream health scores for each data point.

ANOVA hypothesis testing compared the fixed factor 'Period' with two levels 'before' and 'after' and the fixed factor 'Site' with 'upstream' and 'downstream' of the overflow point (the confluence with Bunbury Curran Creek). The model used Site 11 as the upstream site and initially used Site 2 as the downstream site, followed by Site 8 in a second run.

Results and discussion

Water quality

Principal components analysis (PCA)

PCA combines multiple variables into a series of ‘components’ that describe the variation present in a dataset. PC1 is usually the component that describes most of the variability.

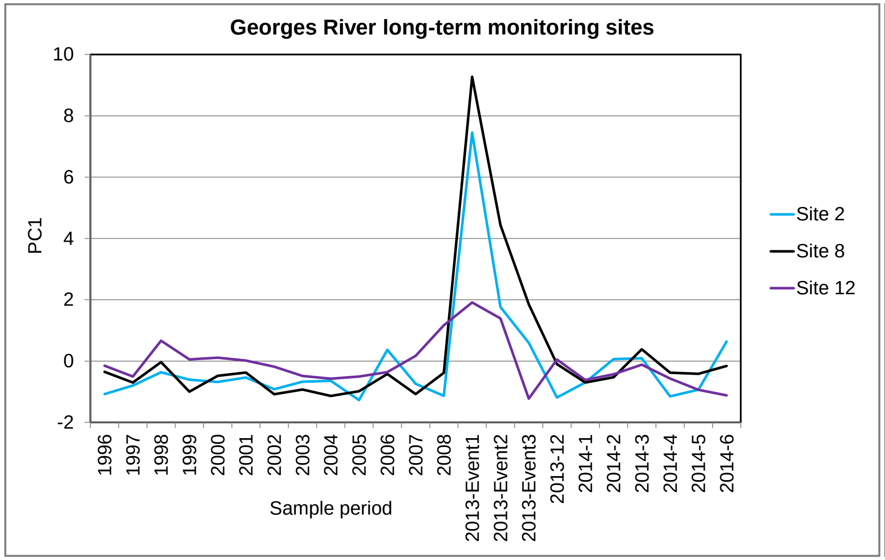
Figure 8-2 and Figure 8-3 present four PC1 plots against time. This analysis illustrates that the incident at Glenfield created a transient pulse disturbance on the water quality of the Georges River. While one of the four plots accounted for 43% of variation in water quality data, the other three plots accounted for over two thirds of the original variation (65%, 65% and 68%) and describe the overall structure reasonably well.

In the Georges River (Figure 8-2 – left) this analysis indicates that water quality parameters were adversely affected during the ‘event’ periods, and recovered toward dry weather levels in each subsequent period. Figure 8-2 (left) shows the similarity of the ‘after’ event water quality (December 2013 to June 2014) to the extensive ‘before’ period (1996 to 2008), suggesting that the disturbance had returned to baseline conditions for the three sites where historical data were available. PC1 accounted for 43% of the total sample variability. The overflow clearly impacted sites 2 and 8, Cambridge Avenue and Liverpool Weir respectively, in the first week of the ‘event’.

In Bunbury Curran Creek (Figure 8-2 – right) the initial water quality disturbance was evident at the two downstream sites (site 1 and site 3) below the overflow point of the pumping station. Recovery in water quality started in early December and returned to levels typical of the upstream site (site 4) through mid to late December (Figure 8-2, right). Over the next six months water quality of the two downstream sites in Bunbury Curran Creek was very similar to the upstream site. PC1 accounted for 68% of the total sample variability.

The water quality disturbance decreased along the length of the Georges River (Figure 8-3, left) with distance toward the ocean. During the period monitored, there was no indication of a water quality disturbance at the outer estuary sites (sites 19 to 21) compared to the upstream freshwater section (Liverpool Weir, site 8) and Chipping Norton Lake (site 12). PC1 accounted for 65% of the total sample variability. The evidence is obscured by other factors at these locations: local diffuse pollutions sources, as well as tributary and tidal inflows.

PCA of nutrient recovery rates ((Figure 8-3, right) found the most pronounced disturbance in nutrient parameters was at the sites 8 and 9, next to Liverpool Weir and Haigh Park, in the ‘event’ period. Disturbance at other downstream sites was evident at relatively lower levels. This may reflect increased tidal flushing or tributary inflows (increasing dilution) with distance toward the estuary mouth. Site 2, Cambridge Avenue, situated upstream of Liverpool Weir and just downstream of the confluence with Bunbury Curran Creek, had a lower level of disturbance in nutrient parameters when compared with the sites near Liverpool Weir. This could be an artefact of when sampling commenced for nutrient parameters, or it may reflect this was a free flowing section of the Georges River where the outflow from Bunbury Curran Creek meets the Georges River. Site 7, situated on the Georges River, upstream of the confluence with Bunbury Curran Creek had similar water quality for nutrient parameters throughout and after the event. This site acted as an upstream positive control location. PC1 accounted for 65% of the total sample variability. While PCA indicated a short-lived impact downstream of the Bunburry Curran Creek confluence, and in Chipping Norton Lake, this analysis does not isolate the parameters affected.



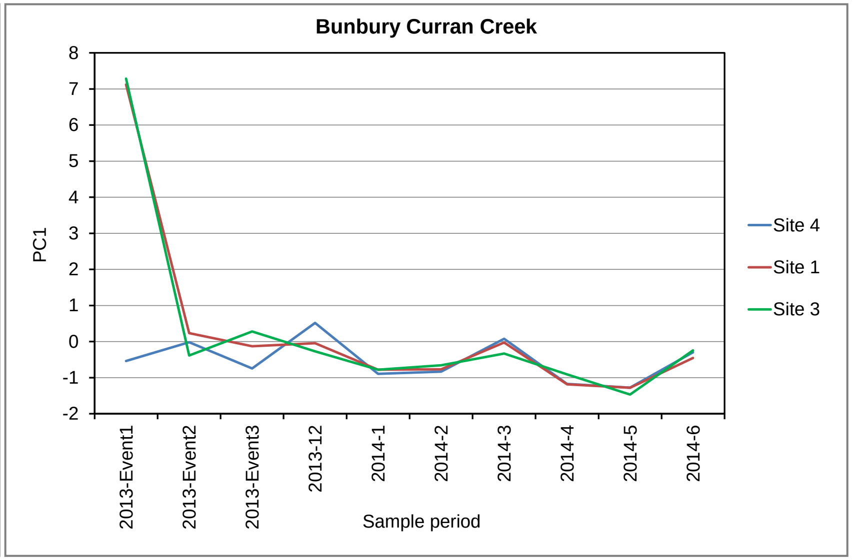
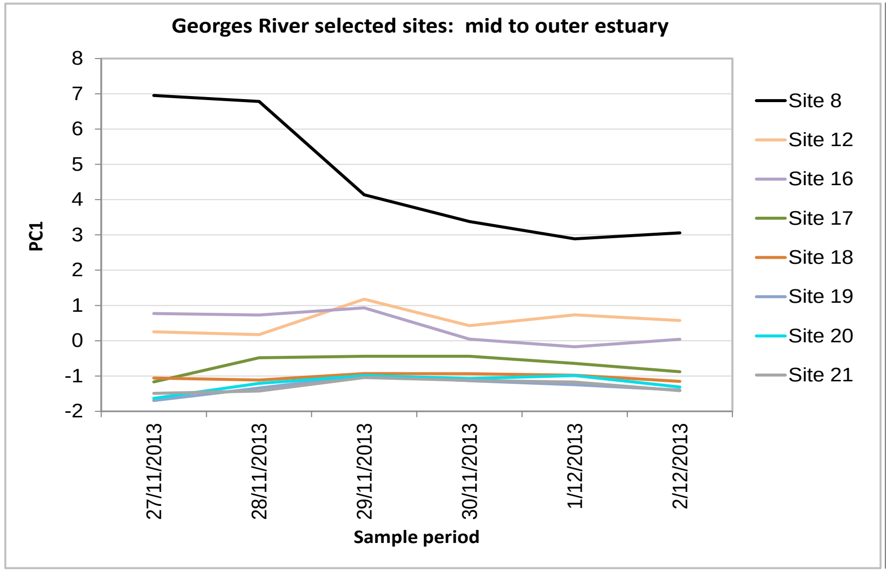


Figure 8-2 PC1 temporal plot for the long term Georges River monitoring sites (left) and Bunbury Curran Creek sites (right)



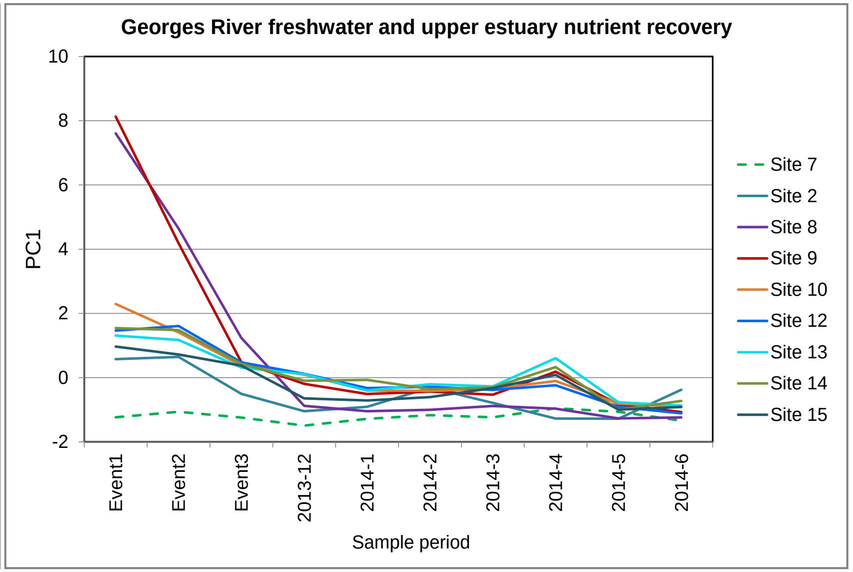


Figure 8-3 PC1 Georges River temporal plots for mid and outer estuary (left) and for freshwater and upper estuary nutrients (right)

Water quality: recreational amenity and microbiological indicators

The NHMRC guidelines only cover Enterococci in marine waters as there is insufficient data for a freshwater guideline (NHMRC 2008, Chapter 5). The ANZECC 2000 guidelines for median values for faecal coliforms suggests: 150 cfu/100mL for primary contact[1] and 1,000 cfu/100mL for secondary[2] contact users of the river; and for Enterococci, 35 cfu/100mL and 230 cfu/100mL, for primary and secondary contact respectively.

Faecal coliforms exceeded the primary contact guidelines for median values at the main upstream impact sites during the ‘event’ period at sites 2 and 8 (Cambridge Avenue and Liverpool Weir). These sites recovered to pre-event conditions by 7-9 December (Figure 8-4). The downstream sites, 17 and 20, had lower overall results during the event.

In Chipping Norton Lake, the three sites sampled (sites 12, 13 and 14) showed initial increases in faecal coliforms and Enterococci in the first week of December 2013. At all three sites, this impact was smaller than peaks occurring later in the monitoring period – 17 February and 18 March 2014 (Figure 8-5 for Enterococci), except for site 12 early in December. On 24 March site 14 had the highest densities of both faecal coliforms and Enterococci (60,000 and 9,900 cfu/100mL, maximum values respectively).

Site 12 at the Angle Park boat ramp (near the inflow to the lake) showed an increase in faecal coliforms in the event period, and then returned to pre-overflow conditions. The increase did not exceed the primary contact guideline (for median values). Similarly for Enterococci, with a wider spread of 50% of the data, levels also returned to pre-overflow conditions after the event (Figure 8-4 and Figure 8-7).

At site 15, just downstream of the lake, microbiological indicators were generally low except for elevated Enterococci in April and June (390 cfu/100mL and 1,500 cfu/100mL, respectively). Site 15 had overall lower microbiological results, (Figure 8-6), than in the lake, suggesting (but not proving) a local factor affecting water quality in the lake which is surrounded by suburbs. This could also be from dilution from Prospect Creek, downstream of the lake and upstream of site 15.

Enterococci results were elevated during the ‘event’ period for site 2 (Cambridge Avenue), but decreased to below the ‘before’ period results. The 95th percentile results were in the ‘poor’ water quality category C, in all the ‘before’ and ‘after’ and then in category D (>500 cfu/100mL) in the ‘event’ period (24,600 cfu/100mL). The 95th percentile results for site 8 (Liverpool Weir) were initially in category C (436 cfu/100mL), then improved in the ‘event’ and ‘after’ period to category B levels (as shown in Figure 8-7).

Limited data was available for the upstream control sites, (site 7 at the upstream edge of the Glenfield plant and site 11 at Ingleburn Weir). Site 7 only had ‘after’ event data and site 11 had five samples from 29 November to the 3 December. Enterococci, 95th percentile results for site 11 were 229 and 278 cfu/100mL, for ‘before’ and ‘after’, respectively (category C ‘poor’). These results were similar to sites 2, 8 and 12 in the ‘before’ period, suggesting that the microbiological indicators, especially Enterococci populations, are sometimes the result of other inflows. More

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[1] Primary contact = swimming, surfing and activities where people come into frequent physical contact with water.

[2] Secondary contact = sports and other water activities with much less contact with water, such as boating and fishing.

detailed studies are required to verify the source of microbiological contamination within the lake as well as transport dynamics around the lake.

The downstream estuarine sites, 16 to 21, were mildly affected by the upstream overflow from Bunbury Curran Creek in terms of biological indicators. These sites were only sampled between 27 November and 10 December 2013. Median faecal coliforms densities were within the primary contact guidelines (ANZECC 2000) for all sites. Enterococci results increased slightly for site 16 (Harris Creek confluence) on 1 December, and markedly for site 20 (Woolooware Bay) on 6 December. Sites in the middle of this transect, 17 to 19, had low bacterial densities of around 9 and 10 cfu/100mL. The results are shown as time series plots (Figure 8-9), where the sampling dates are the same in all six plots.

Formal hypothesis testing of Georges River sites 17 and 20, (long-term sites in the lower estuary), sampled in the 'before' (1995 to 2005) and 'event' period was conducted with an ANOVA model comprised of one factor 'Period'. The total faecal coliforms and Enterococci parameters were analysed untransformed as the homogeneity of variance was non-significant using Brown and Forsythe's test (Appendix 9.4).

ANOVA testing indicated no significant difference between the water quality samples collected from the event period and those samples collected before the incident under dry weather conditions (site 17, faecal coliforms: p=0.602 and Enterococci: p=0.655, and site 20, faecal coliforms: p=0.822 and Enterococci: p=0.307). The results suggested the effects did not extend to in the mid to lower estuary during the event monitoring. An earlier version of these testing outcomes were communicated to the EPA & NSW Health as part of the incident response and formed part of reasoning to lift the closure of the mid to lower estuary.

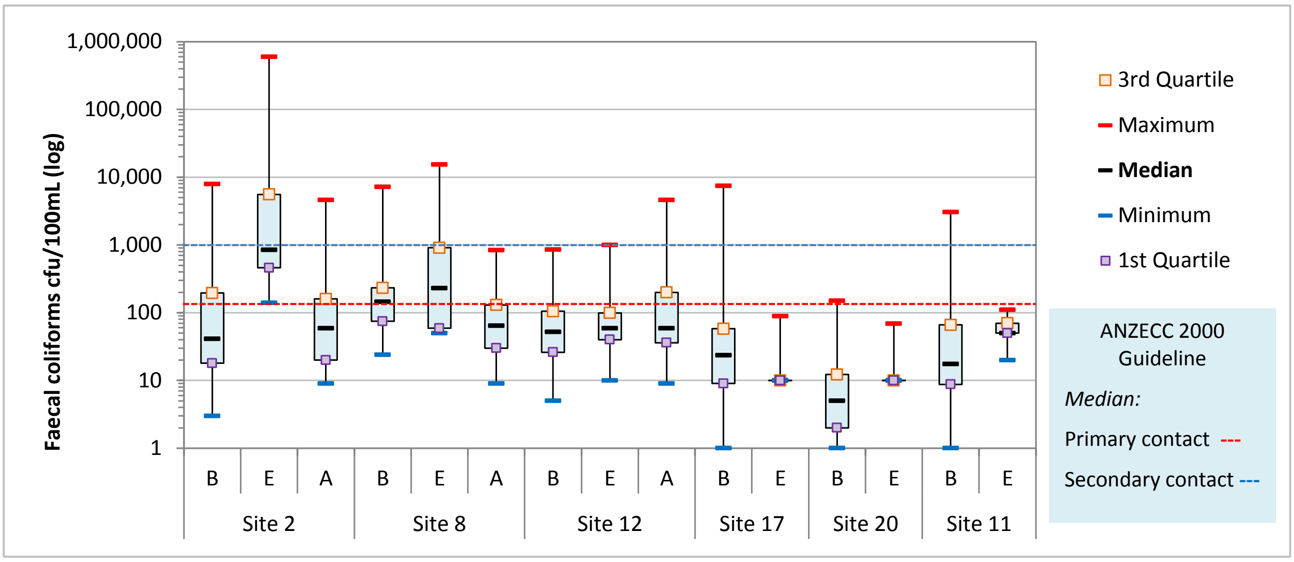


Figure 8-4 Faecal coliforms statistics, 'before', 'event' and 'after' for upstream long-term sites in the Georges River

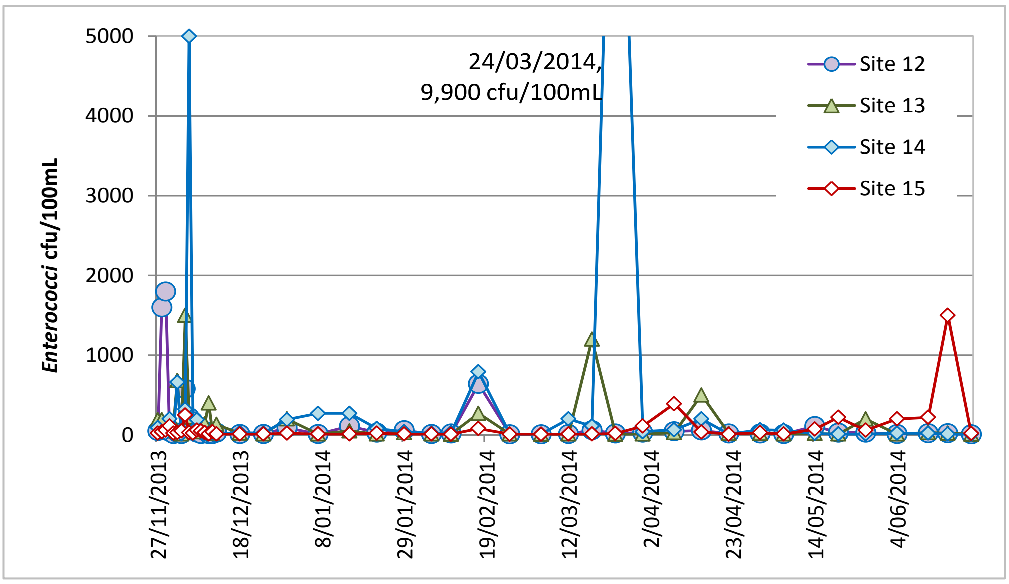


Figure 8-5 Enterococci time series plot for the Chipping Norton Lake sites

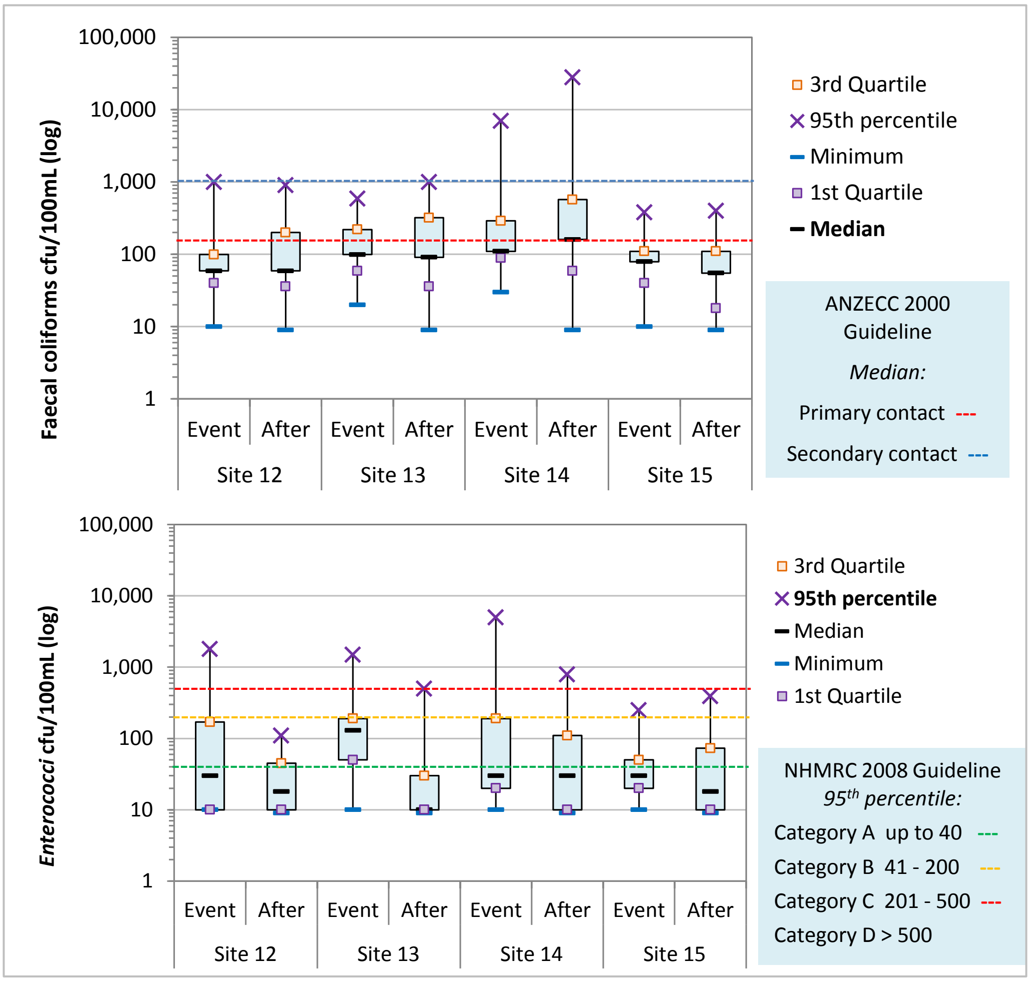


Figure 8-6 Faecal coliforms and Enterococci statistics for the Chipping Norton Lake sites plus site 15 downstream of the lake for the 'event' and 'after' data

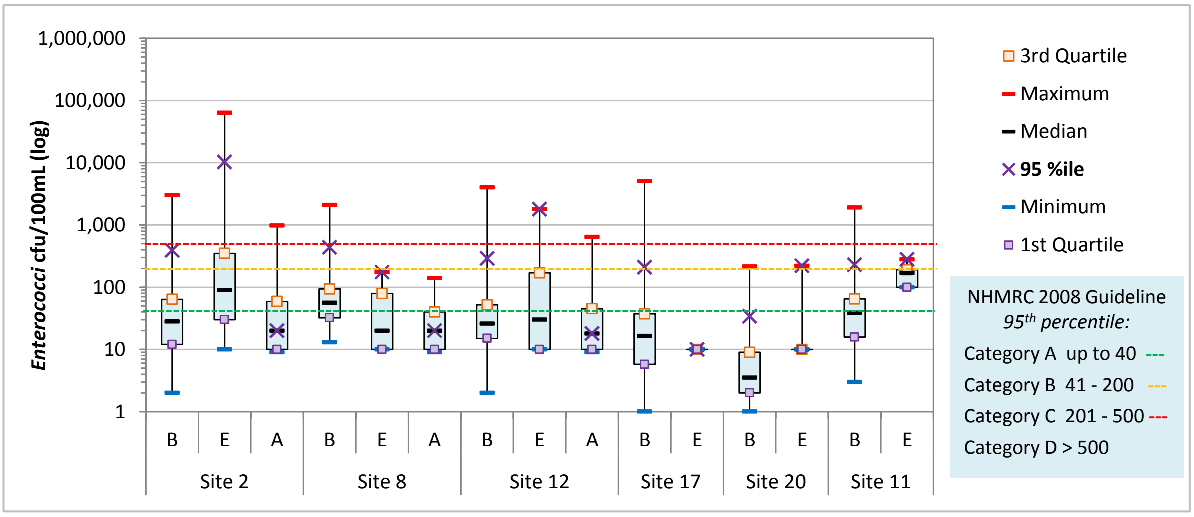
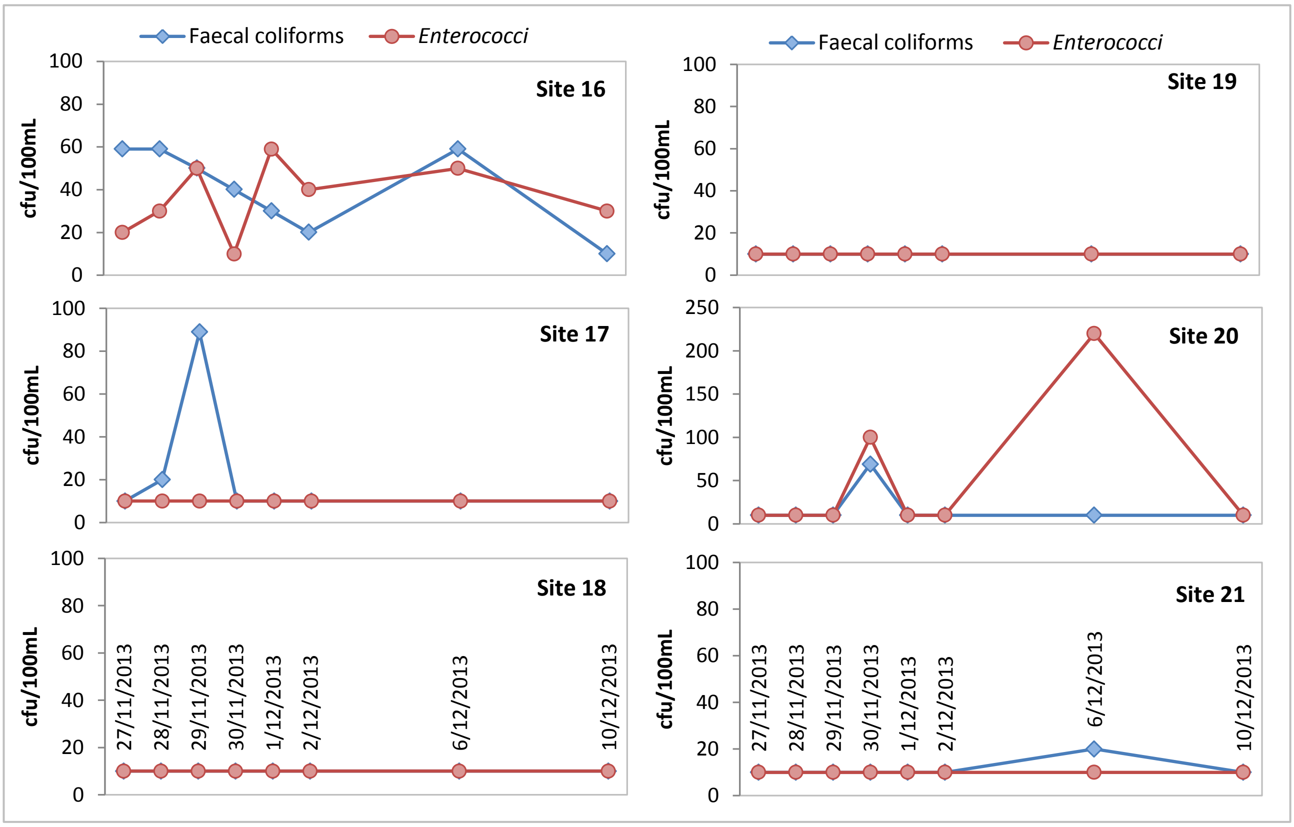


Figure 8-7 Enterococci statistics, 'before', 'event' and 'after' for long-term sites in the Georges River



Note: different scale for site 20

Figure 8-8 Enterococci and faecal coliforms statistics for 'event' data for downstream (estuarine) sites on the Georges River

Water quality: recreational amenity and blue green algal indicators

The NHMRC Recreational Water Guidelines 2008 were used to assess the impact of the overflow on the Georges River upper estuarine reaches (Table 8-4).

Table 8-4 NHMRC Recreational Guidelines for monitoring freshwater rivers for blue green algae

Source: NHMRC Recreational Water Guidelines 2008				
Alert level		Potentially toxic Cyanobacterial biovolume mm^3/L	Cyanobacterial biovolume mm^3/L	
Surveillance mode		-	>0.04 to <0.4	Regular monitoring
Alert mode		≥0.4 to <4	>0.4 to <10	Notify agencies, increase monitoring
Action mode		≥4	>10	Continue monitoring, assess toxicity and notify health authorities

Figure 8-9 shows the alert level samples for cyanobacteria, and their respective dates, on the Georges River. The red line along the Georges River is where the main impact of the overflow was expected. Beyond this (downstream of site 8, Liverpool Weir) the river is tidal and less conducive to algal population growth, although green, amber and red level alerts were also recorded at sites 8 to 14. This is not necessarily connected to the November overflow event since the lag time between an influx of nutrients to the river and the algal population response at these sites is unknown. The influx of nutrients from other sources in the area is also unknown. Figure 8-10 and Figure 8-11 for sites 2 and 8 showed much lower cyanobacteria cell counts in samples collected after the event compared with the pre-event period, 1995 to 2008 (available data). The cyanobacteria cell counts for site 12 in the lake are not presented due to the very low cell counts. Of the 38 algal samples for site 12, 15 from the 'before' period had no cyanobacteria, with a maximum count of 100 cells/100mL from the remaining 43 samples. In the 'after' period for site 12, the maximum was 18 cells/100mL.

Table 8-5 shows all of the post-overflow algal results for cyanobacteria. Most of the samples with green and amber alert levels for cyanobacteria occurred in summer and autumn (mid-December, April and May). Notably a red alert level for cyanobacteria and potentially toxic cyanobacteria occurred on 22 April 2014 in Chipping Norton Lake where the sample from site 13 had 820 cells/mL (7.272 mm^3/L BioVolume) of the potentially toxic genera *Oscillatoria* sp.

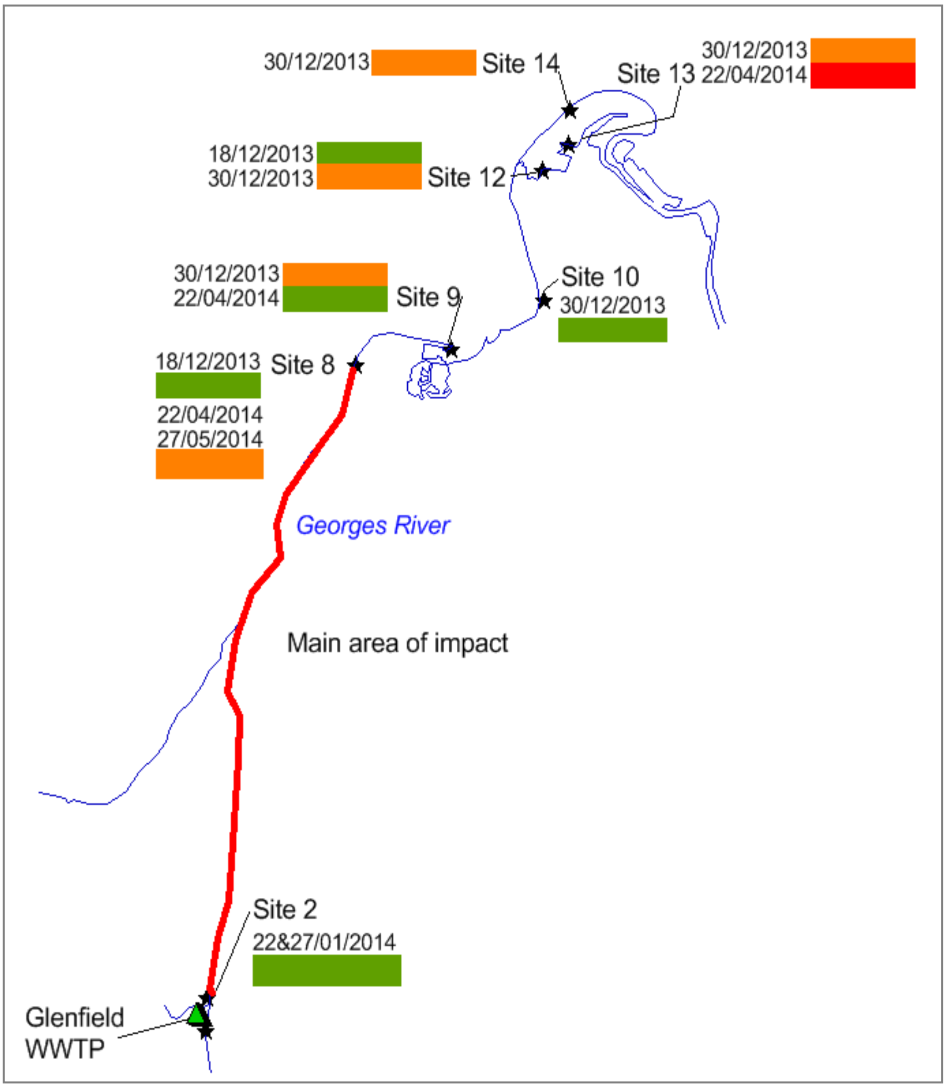


Figure 8-9 Cyanobacteria results after the overflow event on the Georges River. Dates are associated with the alert level indicated. For more information see Table 8-5.

Table 8-5 Summary of blue green algal results for the Georges River monitoring

Site location	Site ID	Number of samples assessed	Date of alert level result	Alert level	Max. cyanobacteria mm^3/L	Max. potentially toxic cyanobacteria mm^3/L
Upstream site	**7**	*4*	4/06/2014		0.042	0.037
Bunburry Curran Creek	**1**	*7*	na[1]	na	0.014	0
	3	*8*	na	na	0.023	0.003
	22	*8*	07/01/2014		0.055	0
Confluence of BCC and Georges River	**2**	*12*	22/01/2014		0.095	0
			29/01/2014		0.086	0
			17/06/2014		0.345	0.345
Georges River upstream of the Chipping Norton Lakes	**8**	*13*	18/12/2013		0.072	0
			22/04/2014		0.709	0.709
			27/05/2014		0.472	0.470
			4/06/2014		0.401	0.401
			23/06/2014		0.291	0.131
	9	*14*	30/12/2013		1.299	0
			22/04/2014		0.236	0
	10	*13*	30/12/2013		0.349	0
Chipping Norton Lakes	**12**	*13*	18/12/2013		0.265	0
			30/12/2013		1.312	0
	13	*14*	30/12/2013		2.604	0
			22/04/2014[2]		7.284	7.272
	14	*14*	30/12/2013		2.011	0
Downstream of the lakes	**15**	*5*	na	na	0.008	0

[1] na = no alerts across all the samples for that site

[2] potentially toxic cyanobacteria had a red alert status while the total cyanobacteria had an amber alert status

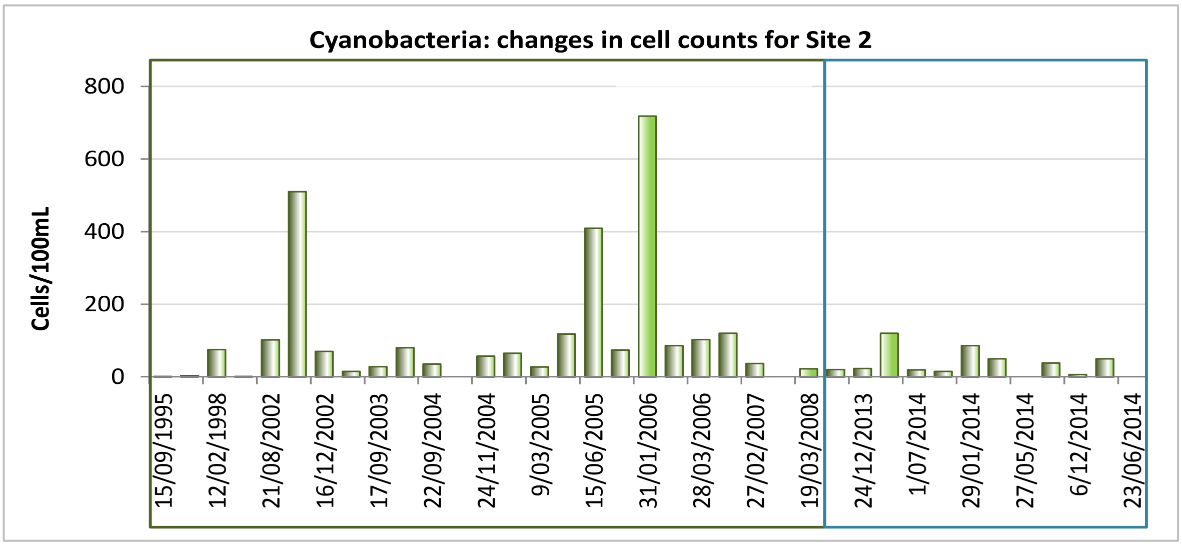


Figure 8-10 Cyanobacteria cell counts for site 2, Cambridge Avenue on the Georges River

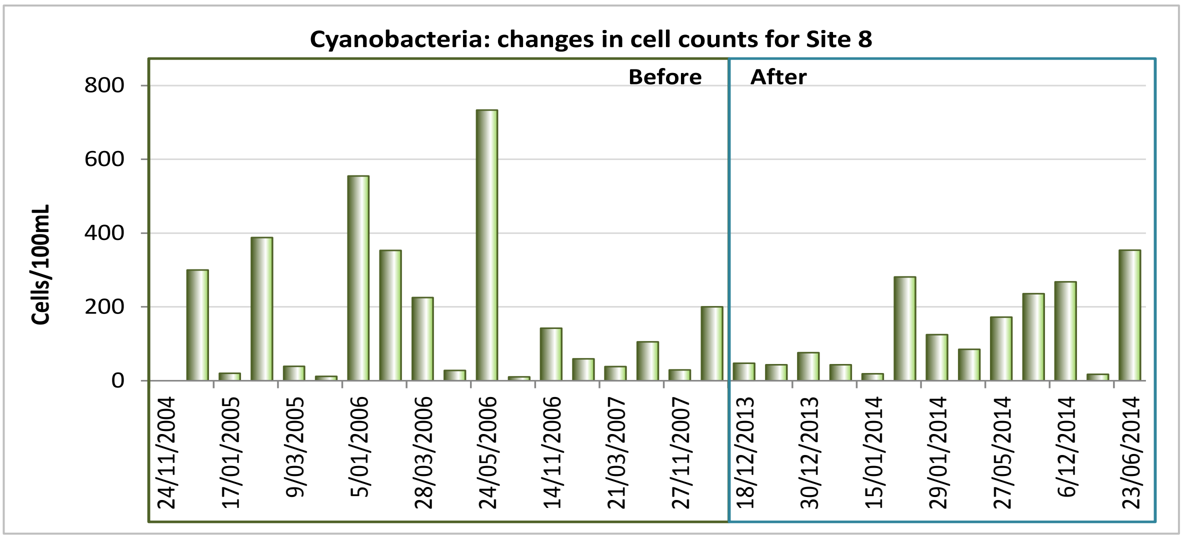


Figure 8-11 Cyanobacteria cell counts for Site 8, Liverpool Weir on the Georges River

Water quality: indicators of ecological health, dissolved oxygen

The monitoring program was instigated as an 'environmental response', not a regular study of dissolved oxygen processes, and this guided the type and frequency of measurements made. Dissolved oxygen was mostly measured in the afternoon during the 'event' period (with some samples near the overflow point also in replicate), but was measured in the morning during the 'after' period.

Dissolved oxygen measurements were made according to the standard Sydney Water operating methods. Results for replicate measurements have been included where available.

The 'event' period dissolved oxygen concentrations declined sharply at the two downstream sites (sites 2 and 8). At site 2, Cambridge Avenue Bridge, the dissolved oxygen declined to 1.4 mg/L (10:30 am on 24 November). This concentration is low enough to cause fish kills, although none were found in the Georges River. The next site down, site 8 at Liverpool Weir, had similarly low dissolved oxygen concentrations for four days between 25 and 29 November, before increasing on the 29th (afternoon measurement) to 7.4 mg/L. The red circles in Figure 8-12 show the period of very low oxygen concentration. By 18 December, both sites had a dissolved oxygen concentration above 7 mg/L. Dissolved oxygen at site 7, upstream of the Glenfield plant, remained high during the 'event' period.

An examination of available historical dissolved oxygen data (2002 to 2008), found that the concentration at sites 2 and 8 ranged between 2 and 12 mg/L (Figure 8-13). The upstream site, at Ingleburn Weir (site 11) also displayed a wide range of concentrations, but did not fall below 6.5 mg/L.

During the post overflow period, the dissolved oxygen concentration at site 2 increased to 12.3 mg/L on 30 December, before settling to a range between 5.8 to 10 mg/L (Figure 8-14). Site 7, the upstream reference site, had concentrations above 5.4 mg/L (25 February 2014) in the 'after' period. Sites 2, 9 and 10 maintained moderate to high concentrations, being all above 5.5 mg/L in the 'after' period. Site 9 at Haigh Park is at the entrance to the Lake Moore wetland where abundant macrophyte growth may maintain high dissolved oxygen levels during the day.

Bunbury Curran Creek does not have any pre-event dissolved oxygen data. Immediately after the event there was considerable variation in dissolved oxygen concentrations for the three downstream sites. Site 4, upstream of the overflow, maintained a high dissolved oxygen concentration until 3 December when it decreased to 3.2 mg/L (dashed green line in Figure 8-15). Throughout the monitoring period, dissolved oxygen concentrations ranged widely between 9.2 mg/L (site 3, 20 May 2014) and 1 mg/L (site 22, 4 June 2014).

Only a few replicate measurements were made during the 'event' period in Bunbury Curran Creek. These were for site 4 and site 1, upstream and downstream of the overflow point. The latter site, showed some diurnal variation as shown in Figure 8-16. Site 4, upstream of the overflow in Bunbury Curran Creek, was sampled daily from the 23 November to 2 December and had a downward trend from 7.3 mg/L to 3.9 mg/L – this site was sampled twice a day, with three additional afternoon samples. The morning and afternoon samples were similar to each other on each day.

In Bunbury Curran Creek, site 22 is in a pool just below a bedrock constriction in the channel. The nature of the substrate material, depth, volume and general behaviour of the pool under high flow conditions is unknown. Therefore it is unknown if the low dissolved oxygen at this site is a result of local conditions or the overflow. Dead fish were collected from Bunbury Curran Creek but there is insufficient data to confirm cause of mortality.

The ANZECC 2000 guidelines for dissolved oxygen in lowland rivers recommend a range of 90% to 110%. This is a narrow range for an essentially disturbed urban river. The maximum and minimum concentrations for site 2 (Cambridge Avenue) and site 12 (Angle Park boat ramp in Chipping Norton Lake) were outside the recommended range for the 'before', 'event' and 'after' periods. The maximum concentration for site 8, (Liverpool Weir), fell within the range in the 'after' period. Similarly the maximum concentrations for sites 17 and 20, were within range in the 'event' period. This is shown in Figure 8-17 and indicates the river at various times has a wide range of dissolved oxygen concentrations. Sites 2 and 8 showed a decline in dissolved oxygen during the 'event' period which may be a result of increased oxygen consumption. However dissolved oxygen increased in the lake, potentially due to surface mixing introducing more oxygen, and horizontal and vertical mixing processes.

The downstream sites, 16 to 21, were only sampled from 27 November to 10 December 2013. All minimum concentrations were below the ANZECC guideline. The lowest concentration was at site 16 downstream of Chipping Norton Lake where the minimum was 50.7% saturation (Figure 8-18). Dissolved oxygen concentrations declined at these sites, initially to around 5.5 mg/L, increasing slightly for the next two December measurements (6 and 10 December). There are insufficient data to conclude that the overflow had an impact at these sites as they would also be affected by tides and other inflows.

The dissolved oxygen results suggest that the overflow event impacted water quality near the confluence with Bunbury Curran Creek, but cannot be confirmed downstream of Chipping Norton Lake. From the data collected, it is unclear if the Georges River from the Glenfield plant to Liverpool Weir typically experiences low levels of dissolved oxygen as the 'before' results for dissolved oxygen saturation were mostly below the guideline range for mean and median measurements, less than 90% saturation (Figure 8-17).

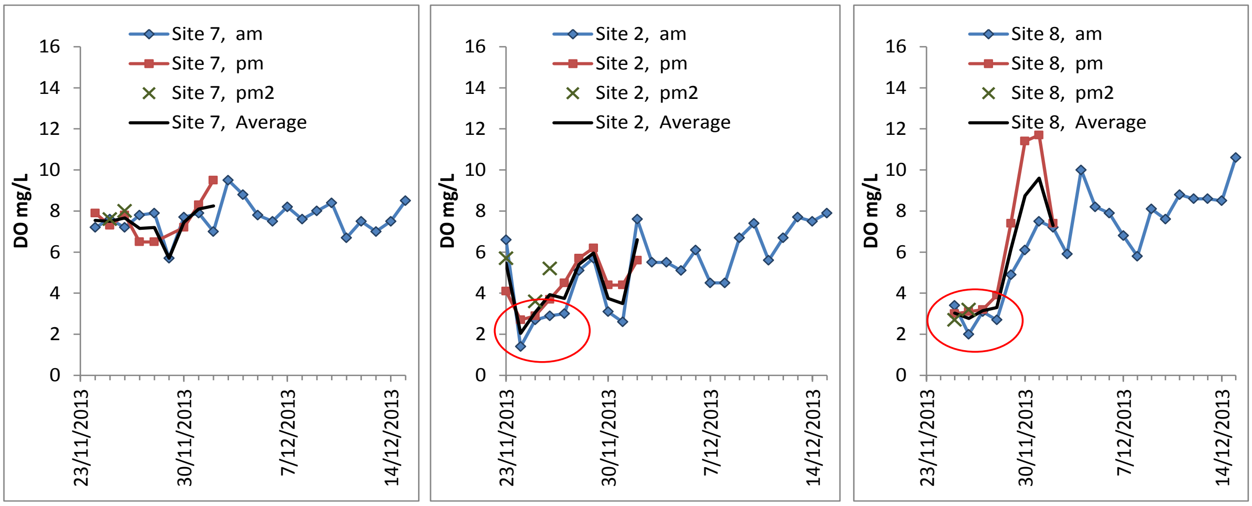


Figure 8-12 Dissolved oxygen concentration in the Georges River: upstream reference site 7 and impact sites 2 and 8

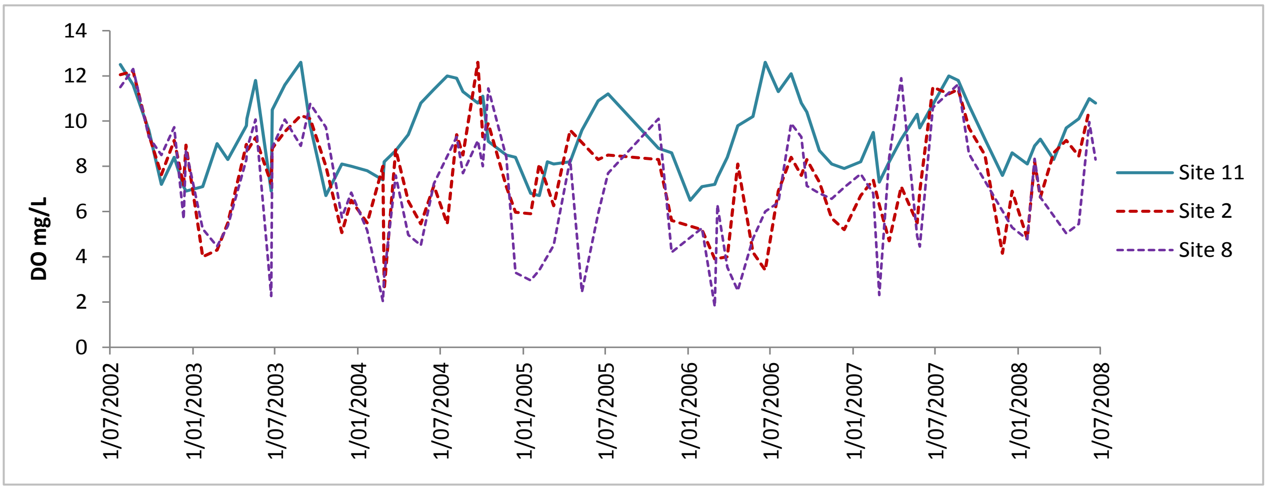


Figure 8-13 Historical data for dissolved oxygen concentration at sites 2 and 8 (impact sites) and 11 (upstream control site) in the Georges River

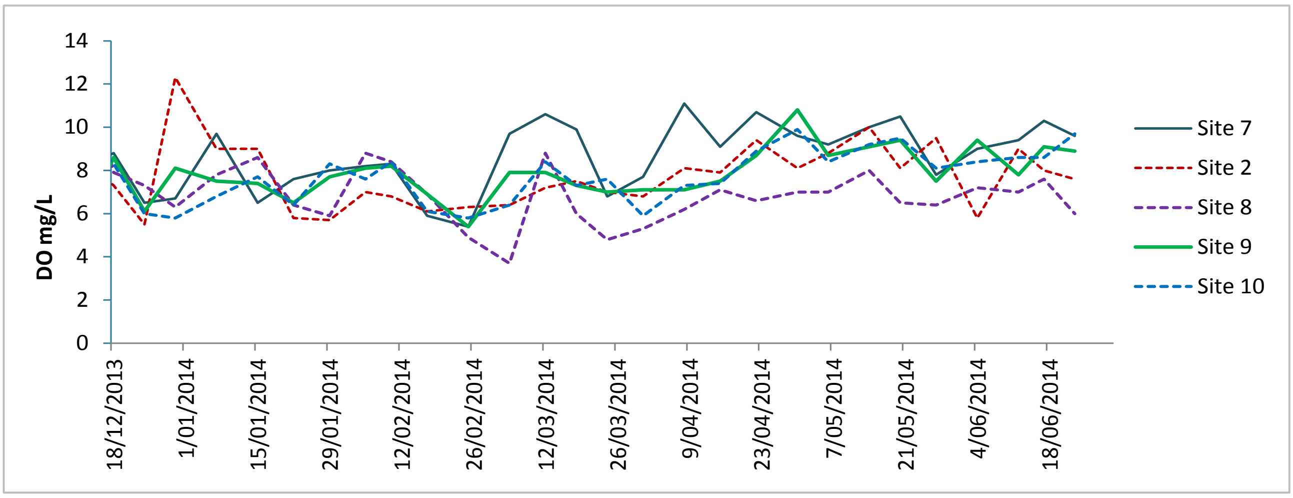


Figure 8-14 Post event data for dissolved oxygen concentration at sites 7 (upstream control site), 2, 8, 9 and 10 (impact sites) in the Georges River: period 18/12/2013 to 23/06/2014

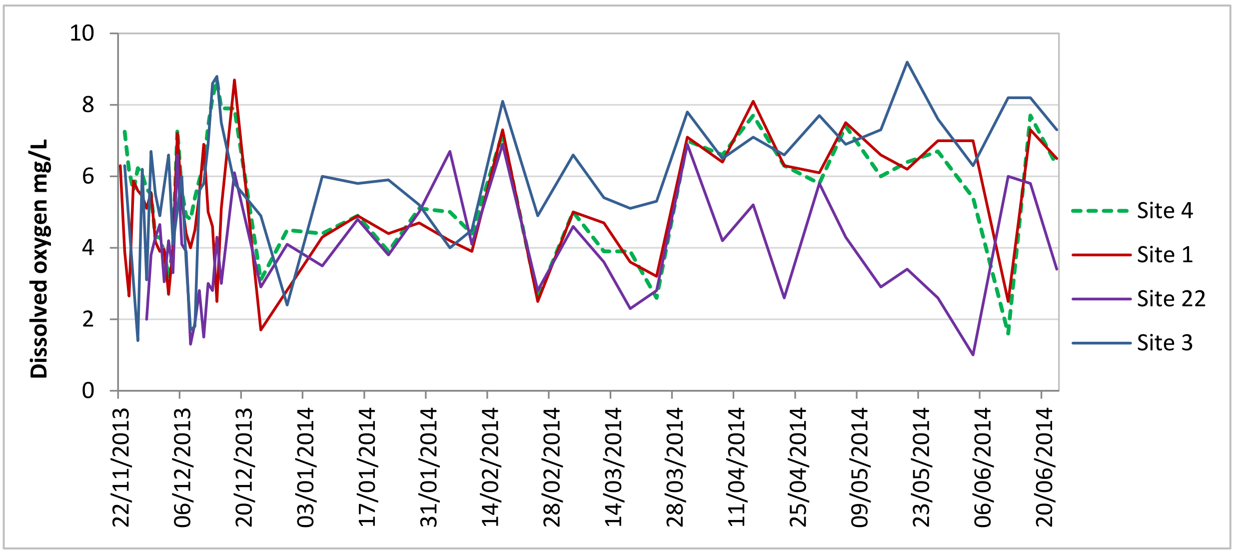


Figure 8-15 Post event data for dissolved oxygen concentration Bunbury Curran Creek sites

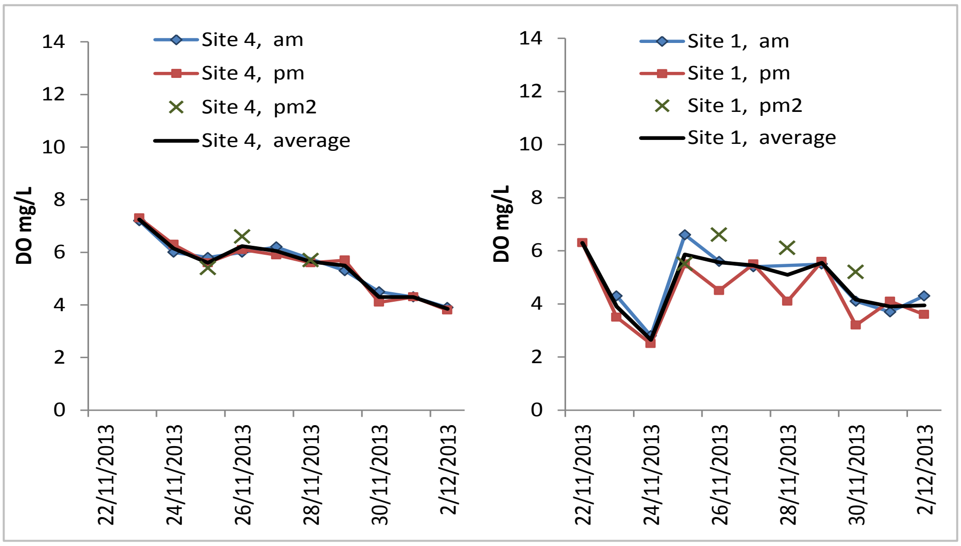


Figure 8-16 Diurnal dissolved oxygen concentrations at two sites in Bunbury Curran Creek

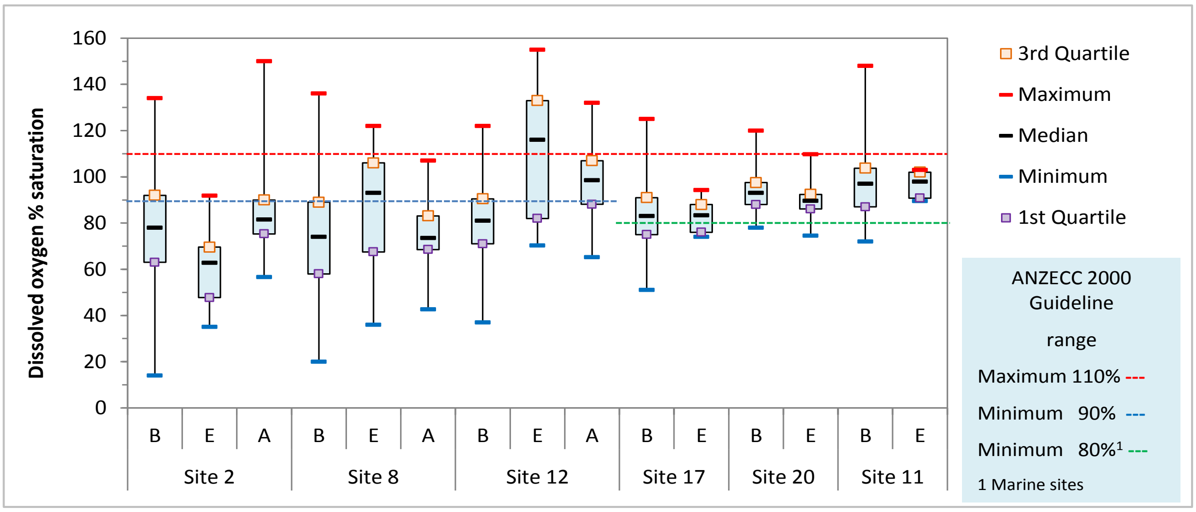


Figure 8-17 Dissolved oxygen statistics, ‘before’, ‘event’ and ‘after’ for long-term sites on the Georges River

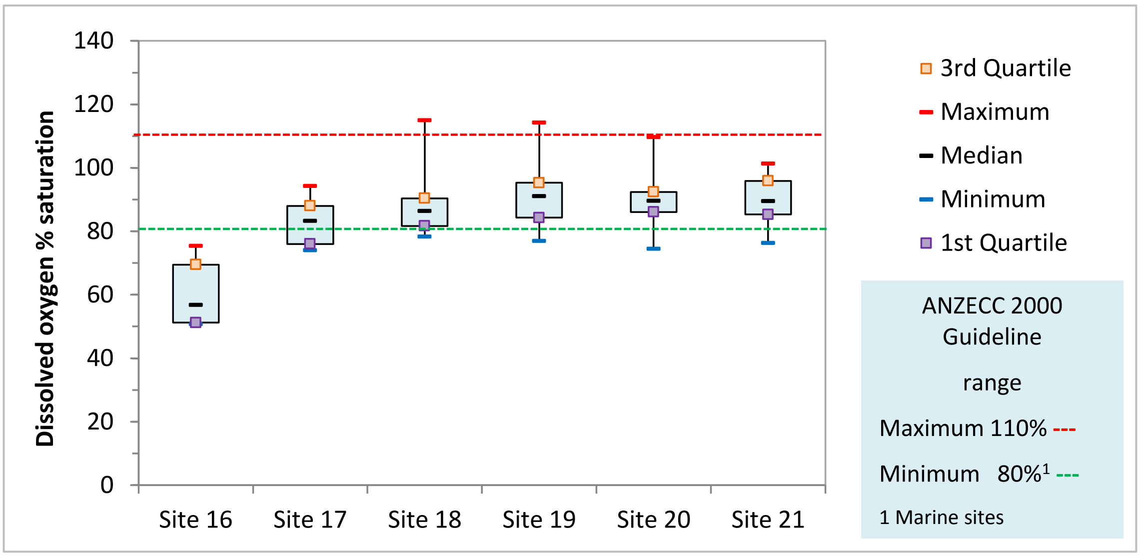


Figure 8-18 Dissolved oxygen, percent saturation statistics, 'event' only for the downstream (estuarine) sites on the Georges River

Water quality: indicators of ecological health, pH and nutrients

The ANZECC 2000 guideline values for marine and freshwater sites are presented in Table 8-6.

Table 8-6 ANZECC 2000 guidelines thresholds

Parameter	Statistic	Marine sites	Freshwater sites
Total nitrogen	maximum	0.3 mg/L	0.35 mg/L
Ammonium	maximum	0.015 mg/L	0.02 mg/L
Total phosphorus	maximum	0.03 mg/L	0.025 mg/L
pH	range	7 – 8.5 pH units	6.5 – 8.5 pH units

Values for pH were within the guideline range for all the downstream sites (Figure 8-19). Site 16, Harris Creek confluence, had a low pH on the 27 November 2013 (6.8 pH units). In Bunbury Curran Creek the pH was within range for all samples except the 'after' data, where the maximum was 9.3 pH units for site 4, which is upstream of the overflow point, suggesting high productivity in this area.

Sites 2, 8, 12 and 17 exceeded the ANZECC 2000 guidelines for total nitrogen regardless of the period, with elevated levels during the 'event' (Figure 8-20). The upstream sites 2, 8 and 12 recovered to near pre-event conditions by mid-December 2013. This showed the 'pulse' effect of the overflow.

Total phosphorus increased in the 'event' period for sites 2, 8, 12 and 20 (Figure 8-21). The magnitude of the increase was high at the first three sites. The concentration decreased to near 'before' period levels after the overflow event as suggested by the principal components analysis. All sites indicate an impact on the river during the 'event' period for nutrients – even site 17 and 20 in the lowest reaches. The greatest impact in nitrogen and phosphorus was at site 8 (Liverpool Weir) suggesting tidal trapping behind this obstruction is a factor (sampling is on the upstream side of the weir).

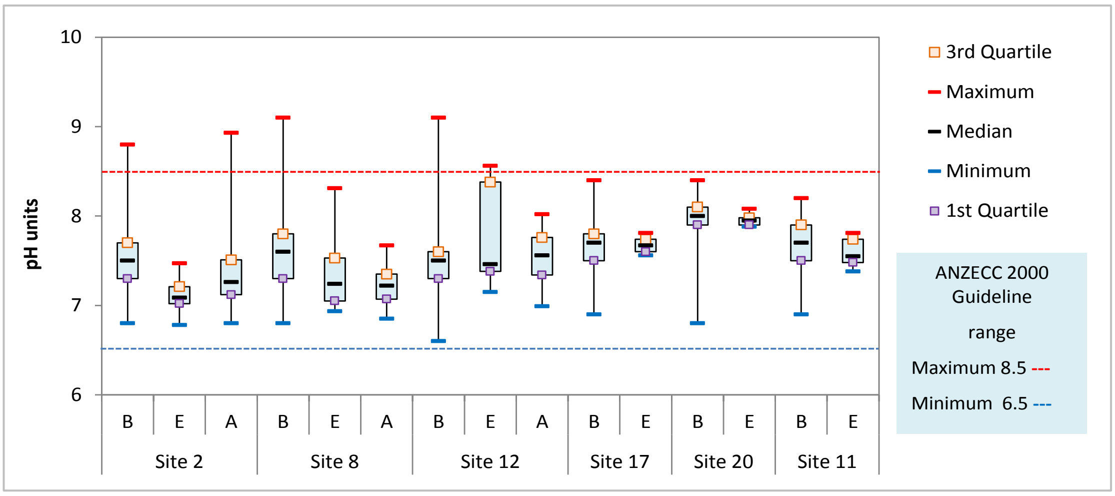


Figure 8-19 pH values statistics, 'before', 'event' and 'after' for the long-term sites on the Georges River

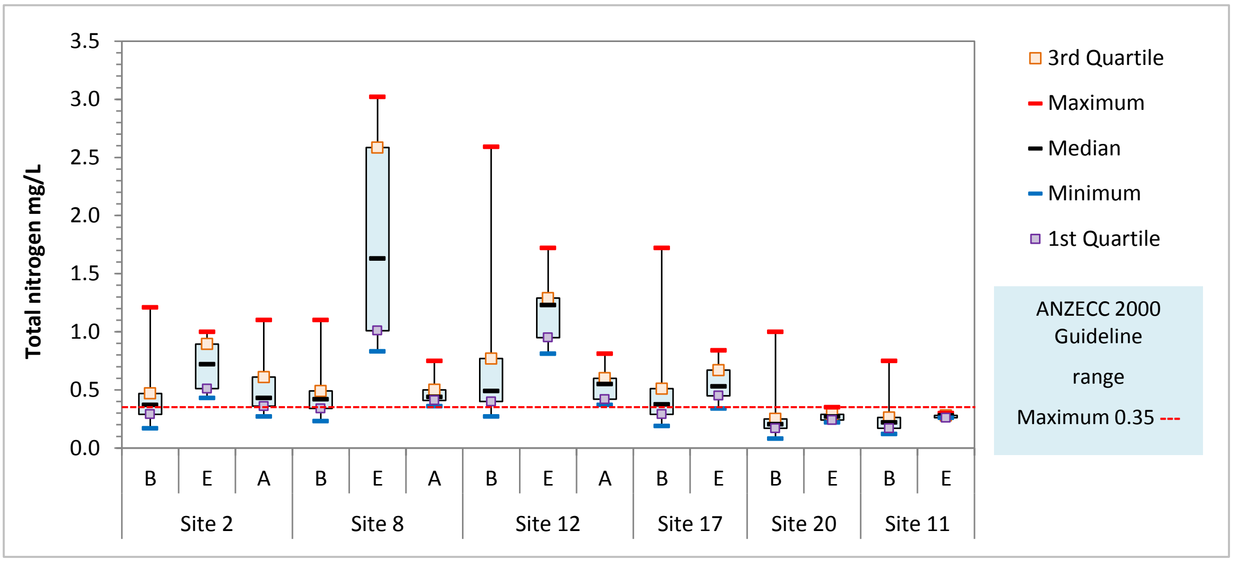


Figure 8-20 Total nitrogen statistics, 'before', 'event' and 'after' for long-term sites on the Georges River

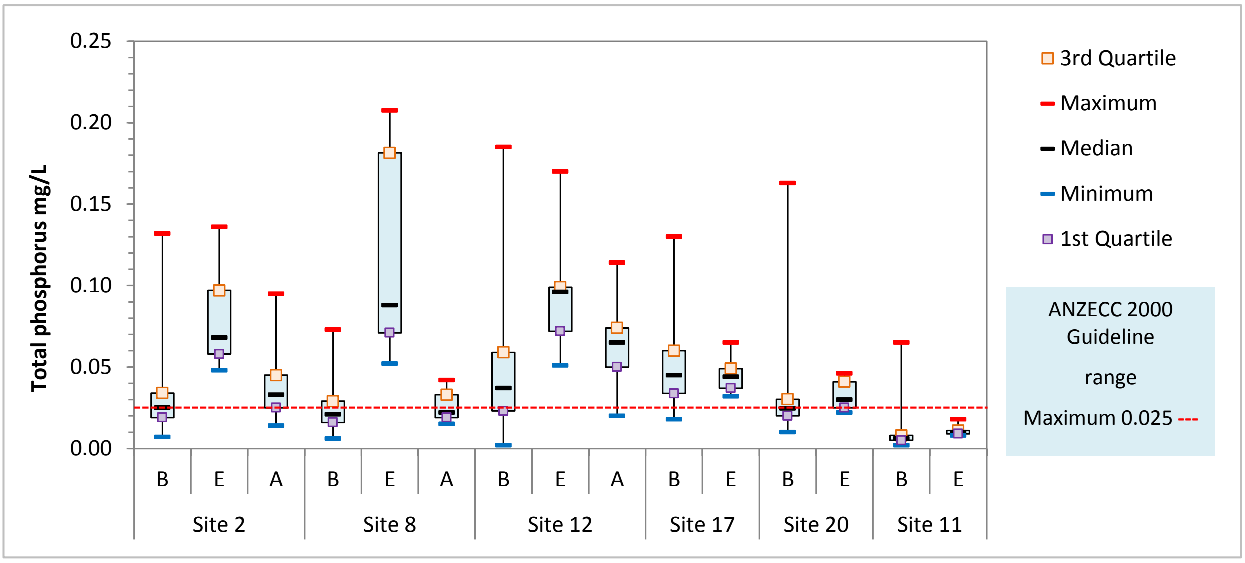


Figure 8-21 Total phosphorus statistics, 'before', 'event' and 'after' for long-term sites on the Georges River

Formal hypothesis testing of Georges River sites 17 and 20, long-term sites in the lower estuary, sampled in the before (1995 to 2005) and event period was conducted with an ANOVA model comprised of one factor 'Period'. The total nitrogen and total phosphorus parameters were analysed untransformed as Brown and Forsythe's for homogeneity of variance were non-significant (Appendix 9.4).

ANOVA testing indicated no significant difference between water quality samples collected from the event period and those samples collected under dry weather conditions before the incident:

- site 17: *p*-values of 0.141 and 0.507 for total nitrogen and phosphorus respectively
- site 20: *p*-values of 0.168 and 0.424 for total nitrogen and phosphorus respectively

These results suggested effects did not extend to the mid to lower estuary during the event monitoring. This was communicated to EPA & NSW Health as part of the incident response and formed part of reasoning to lift closure of mid to lower estuary.

Chlorophyll a (an indicator of algal growth) was significantly different in the 'event' period between site 7 (above the WWTP) and site 12 (a downstream impact site in Chipping Norton Lake). Site 7 also had significantly different chlorophyll a results between the 'event' and 'after' periods. While site 12 had higher chlorophyll a concentrations in the 'event' period and reduced concentrations in the 'after' period, site 7 had the opposite trend ie higher chlorophyll a concentrations in the 'before' period, than in the 'after' period. Significant changes in chlorophyll a concentrations occurred:

- for both sites between periods: *p*-values of 0.003 and 0.023 for sites 7 and 12 respectively
- for both sites, in the 'event' period (*p*-value = 0.001) (but not in the 'after' period (*p*-value = 0.118))

This suggests that the overflow event is not a contributing factor at site 7. Site 7 is in a part of the Georges River that receives urban inflows including nutrients.

The chlorophyll a concentration at site 12 took longer to return to 'before' levels than for nutrient and bacterial indicators. This may be due to local factors, such as aquatic vegetation (providing microhabitats for algal growth), local nutrient inflows and tidal influences. This study did not explore the tidal component. During the event, 27 November to 12 December 2013, site 15, just downstream of the lake, had stable, low chlorophyll a concentrations (Figure 8-22). At this time, site 12, nearest the Georges River inflow, had a peak in chlorophyll a concentration of 129 µg/L on the 5 December. All three sites were at background levels on 11 December 2013. Figure 8-23 shows that in the 'after' period, the four sites (12, 13, 14 and 15), returned to their usual variable behaviour, with elevated chlorophyll a in late April 2014.

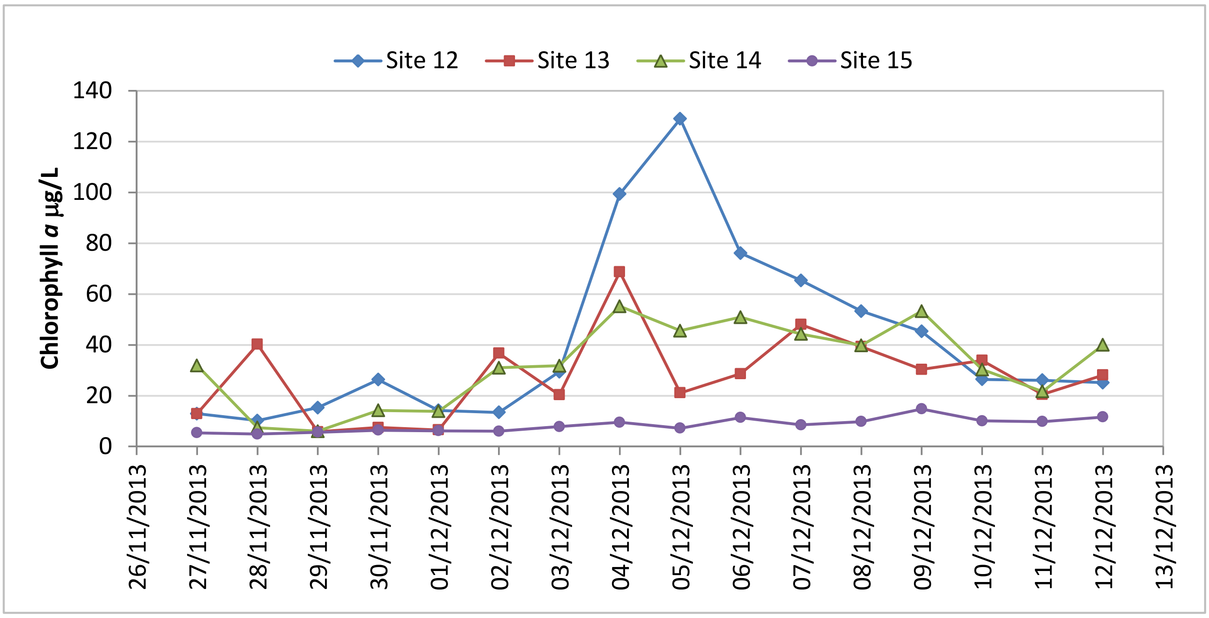


Figure 8-22 Chlorophyll *a* in the 'event' period for the lake sites and site 15, just downstream of the lake

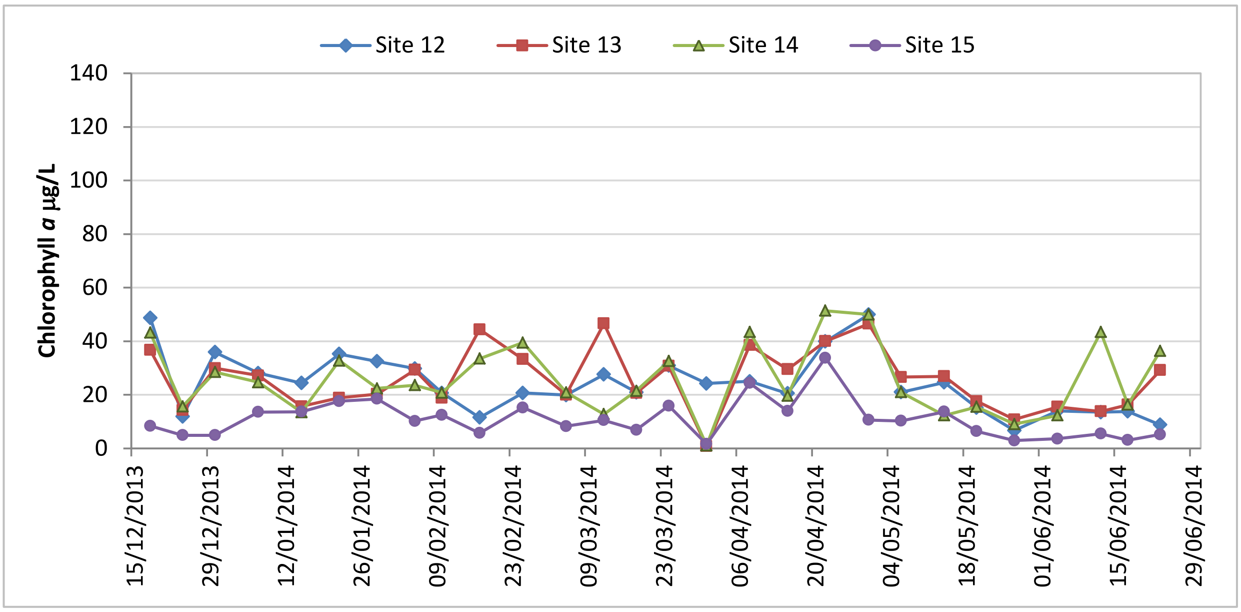


Figure 8-23 Chlorophyll *a* in the 'after' period for the lake sites and site 15, just downstream of the lake

Sites 12, 13 and 14 are in Chipping Norton Lake, which at times (during inflowing tides along the Georges River) holds nutrients, algae and dissolved organic matter longer than at other locations on the river. Given chlorophyll *a* concentrations were elevated through late summer and well into autumn at site 7, this could suggest that the recovery period for chlorophyll *a* at upper estuarine sites 9 to 14, is due to other natural weather influences. Figure 8-24 shows the chlorophyll *a* statistics for the main long-term sites, 2, 8, 12, 17 and 20.

The nonmetric multidimensional scaling (nMDS) ordination plots (Figure 8-25, Figure 8-26 and Figure 8-27) from multivariate analysis of community structure did not reveal a distinct temporal pattern for samples grouped by 'period' for site 2. Rather samples from the various sub-periods were interspersed. The nMDS pattern suggested the disturbance in water quality did not influence algal community structure in either 'after' sub-periods. This assessment was supported by non-significant PERMANOVA results for both period and sub-period factors of PERMANOVA model (Table 8-7). The other two long-term sites (sites 8 and 12) had similar non-significant PERMANOVA test results and interspersed samples from the various sub-periods (Figures 8-25, 8-26 and 8-27).

Table 8-7 PERMANOVA results for algal communities for sites 2, 8 and 12, extract of output

Source	df	SS	MS	Pseudo-F	P(perm)	Unique perms	P(MC)
Site 2							
Period	1	2378.8	2378.8	2.6473	0.1733	6	0.1095
sub period (Period)	2	1880.1	940.06	1.5603	0.1404	9927	0.1546
Res	33	19882	602.48				
Total	36	24145					
Site 8							
Period	1	1091.6	1091.6	1.3391	0.3304	6	0.3332
sub period (Period)	2	1643.2	821.61	1.6782	0.1373	9949	0.1387
Res	27	13208	489.57				
Total	30	15971					
Site 12							
Period	1	1753.5	1753.5	3.8466	0.168	6	0.0585
sub period (Period)	2	867.58	433.79	0.65677	0.7157	9930	0.6883
Res	34	22457	660.49				
Total	37	25109					

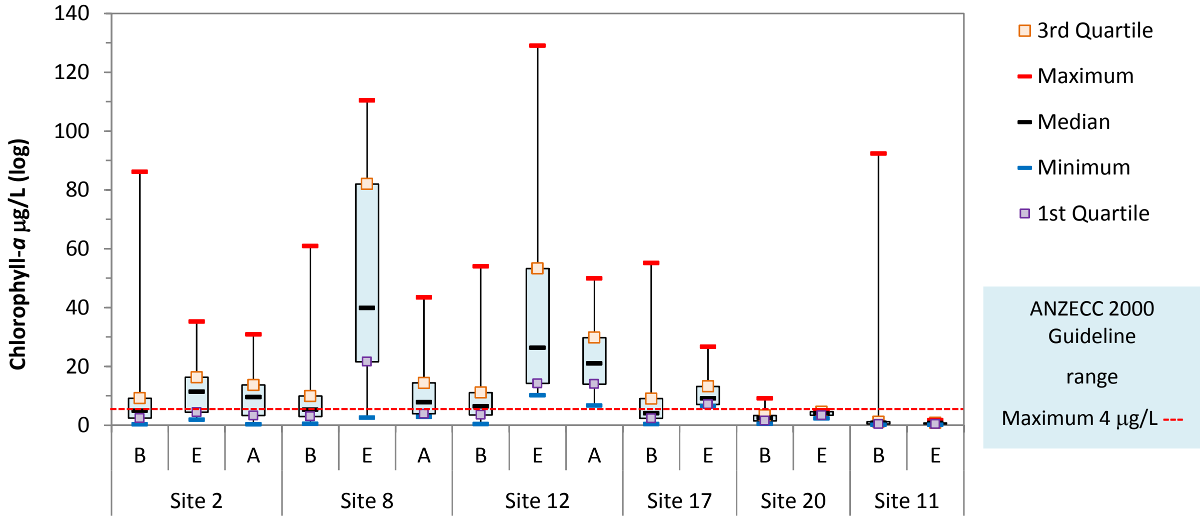


Figure 8-24 Chlorophyll *a* statistics, 'before', 'event' and 'after' for long-term sites on the Georges River

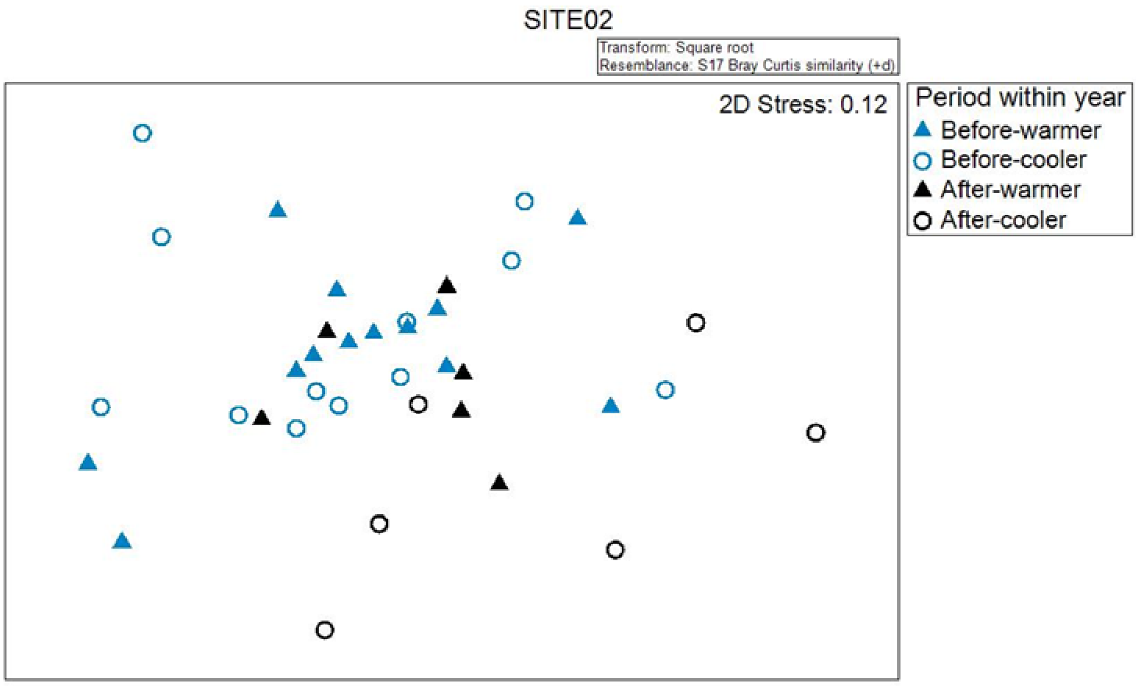


Figure 8-25 MDS ordination plot of algal community structure samples from site 2 coded by 'before' and 'after' periods

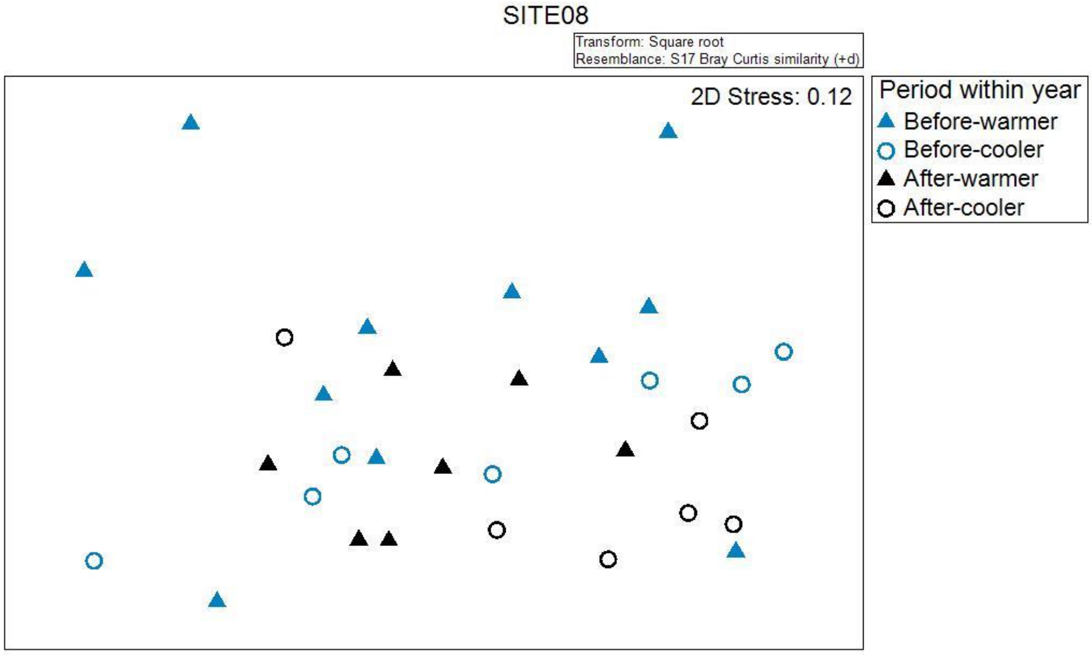


Figure 8-26 MDS ordination plot of algal community structure samples from site 8 coded by 'before' and 'after' periods

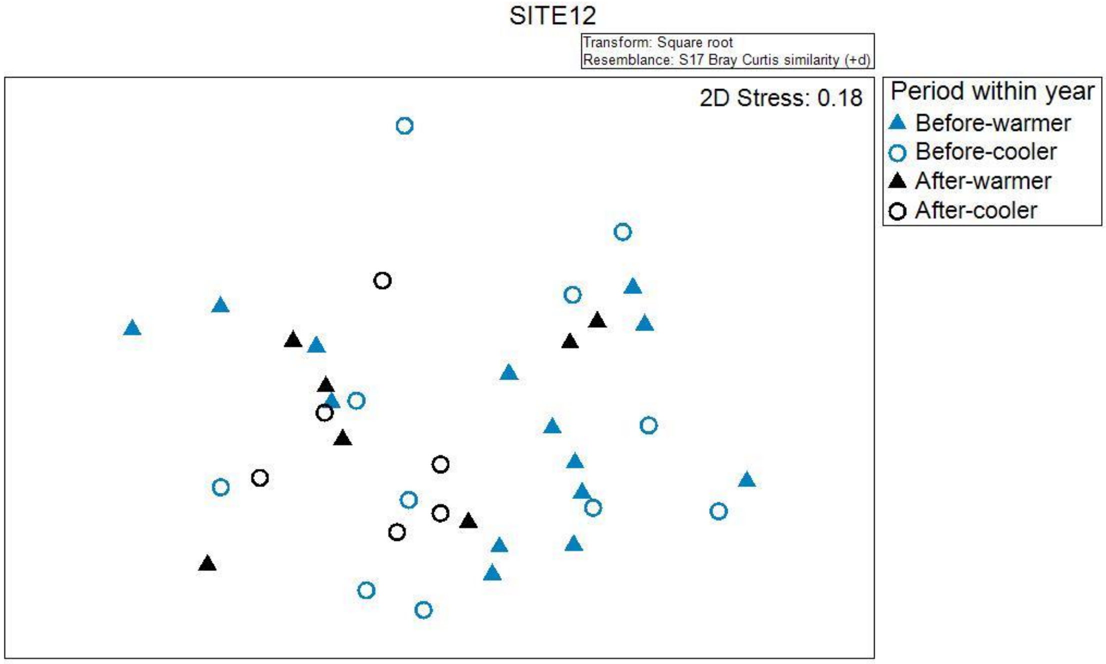


Figure 8-27 MDS ordination plot of algal community structure samples from site 12 coded by 'before' and 'after' periods

Macroinvertebrates

Sensitive freshwater macroinvertebrate taxa are typically absent from stream sites with a greater than 20% connection to impervious surfaces (Walsh, 2004). Surfaces like roofs, gutters, roads, paths and car parks, result in the frequent delivery of pollutants from smaller rainfall events (Walsh *et al*. 2005). Some water quality impairment is expected in streams located in urbanised areas such as site 2 at Cambridge Causeway and Site 8 at Liverpool Weir on the Georges River.

SIGNAL-SG scores from sites 2 and 8 indicate the stream health before the event was similar to the better scores recorded from 1995 to 2013 (Figure 8-28 and Figure 8-29). Given the preceding dry weather and subsequent reduced delivery of pollutants from the catchment, these results are expected.

SIGNAL-SG scores recorded immediately post the incident showed a decline in stream health. The second and third surveys post-incident have confirmed this decline in stream health levels at sites 2 and 8. Both of these post-incident data points, from both sites, were within the range recorded at these two sites pre-incident (Figure 8-28 and Figure 8-29).

Statistical testing returned non-significant interaction terms for sites 11 and 2, and then for sites 11 and 8. Site 11 was the control site while sites 2 and 8 were the impact sites. These results indicate an impact did not occur in stream health from the wastewater overflow based on the macroinvertebrate indicator (site 11 and site 2 Mean Square = 0.00001, df = 1, p = 0.9946; site 11 and site 8 Mean Square = 0.03785, df = 1, p = 0.6623).

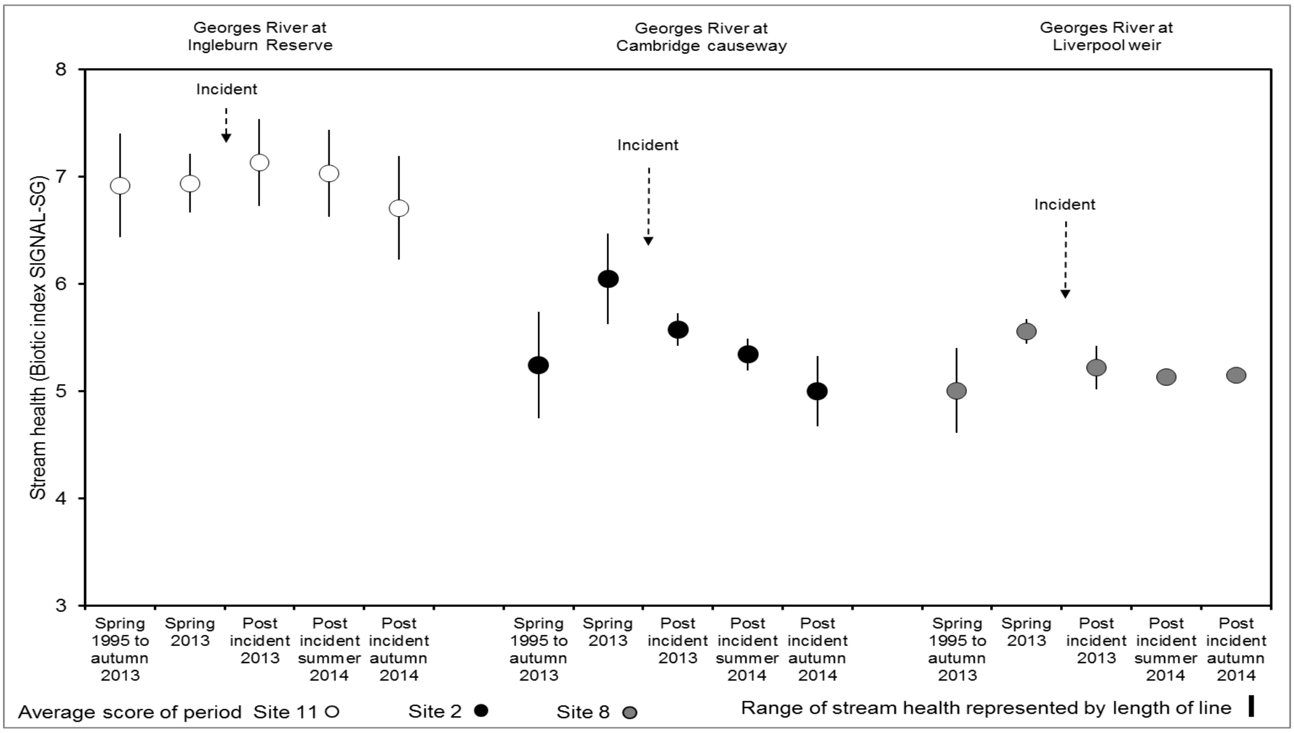


Figure 8-28 Stream health summary pre and post wastewater overflow

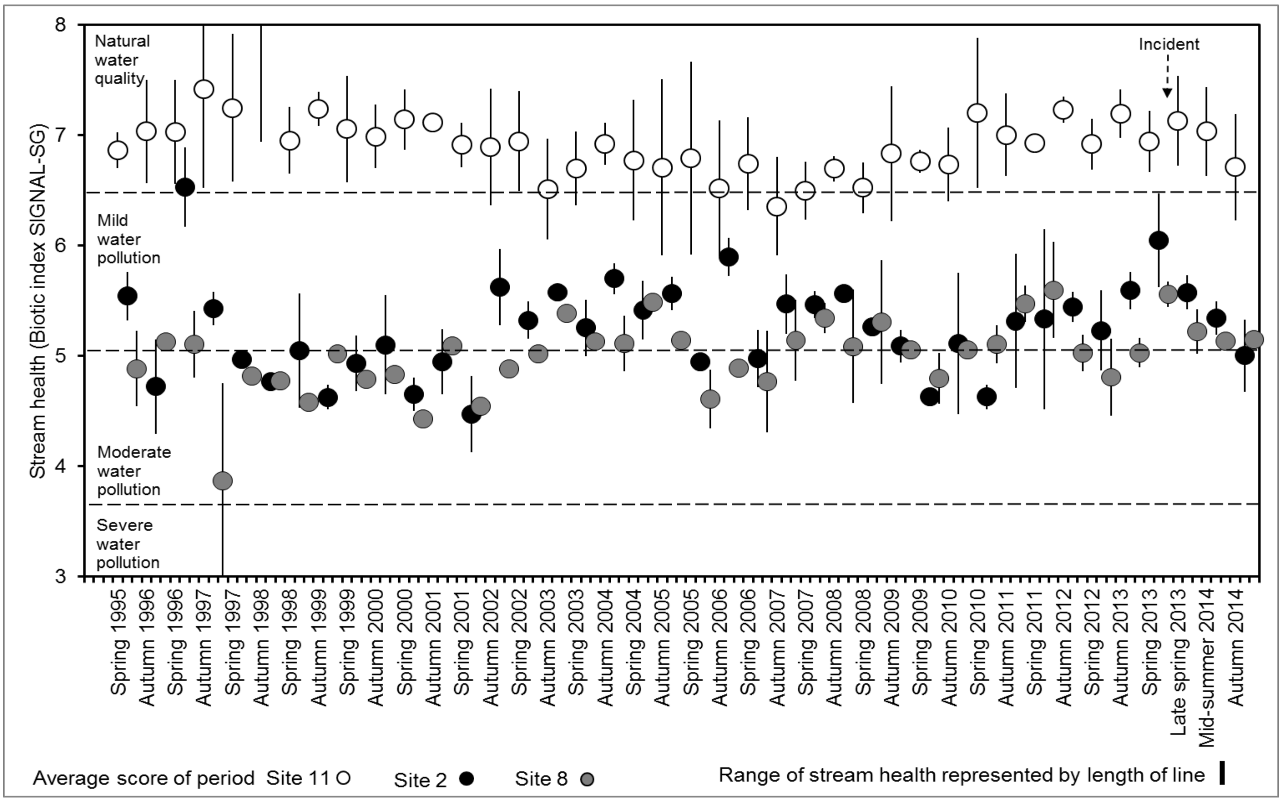


Figure 8-29 Stream health of each sample period

Conclusion

There was a change in water quality in part of the Georges River after the overflow event in November 2013. The main impact effects were limited to the area from the confluence of Bunbury Curran Creek and the Georges River, to Liverpool Weir on the Georges River and to a smaller degree in Chipping Norton Lake. The impact only lasted two weeks for most parameters. Smaller effects were noted in the downstream reaches towards Botany Bay.

Water quality parameters returned to background levels by mid- to late-December 2013 for all sites affected. The overflow event did not affect cyanobacteria populations and community structures as shown in the before-after data analyses. Local conditions and warmer temperatures appeared to have effected algal populations since isolated elevated algal populations were noted later in the monitoring period.

The most significant effect of the overflow was low dissolved oxygen from Bunbury Curran Creek to Liverpool Weir on the Georges River. The dissolved oxygen concentration declined to 1.4 mg/L in the creek and at Cambridge Avenue.

Stream health, indicated by macroinvertebrate populations, showed a decline of stream health which was not statistically significant and fell within the normal variation experienced before the overflow incident.

Lower estuary sites and Chipping Norton Lake were only slightly affected by the overflow event.

The impacts on the Georges River may be termed a ‘pulse disturbance’ as defined by Morris and Therivel (2009). This is where the disturbance is high intensity, but short-lived and does not induce a permanent ecological change. The impacts on the Georges River were short-lived both in real-time and statistically, since conditions appeared too returned to the pre-event variability (from the evidence available).

While a precautionary approach to limiting public access to places where these events happen is required, accounting for the local physical and landuse characteristics of the river is worth considering. For example, in this event Liverpool Weir formed a barrier to the transfer of pollutants downstream and Chipping Norton Lake acts, at times, as a tidally trapped basin for nutrients and algal populations, making sampling times important.

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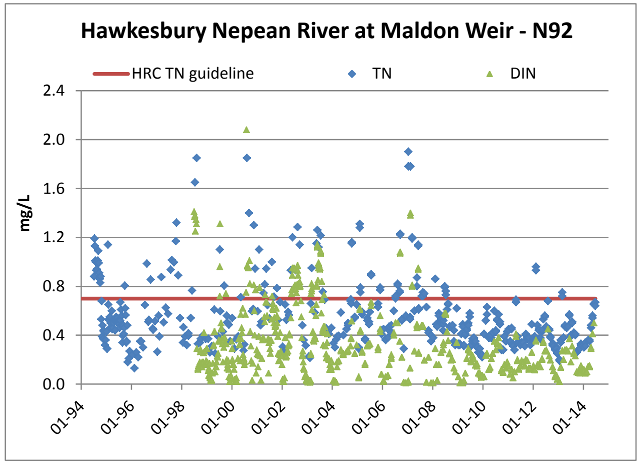
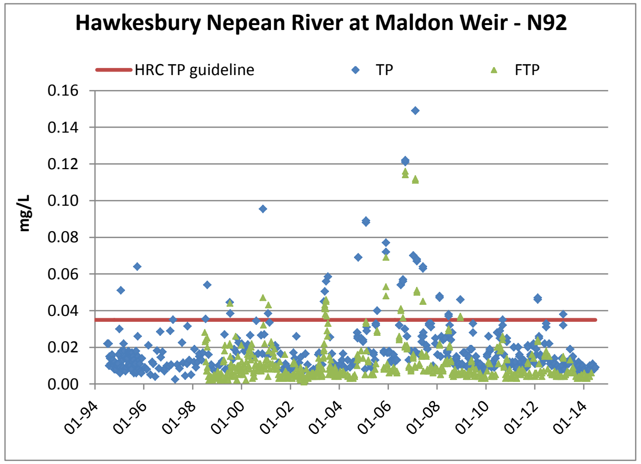
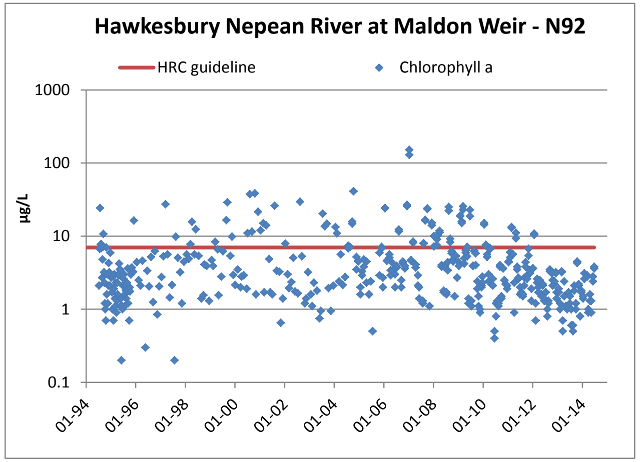
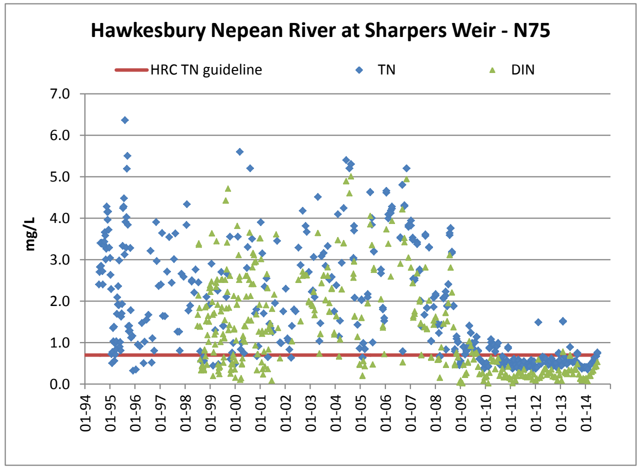
9 Appendices

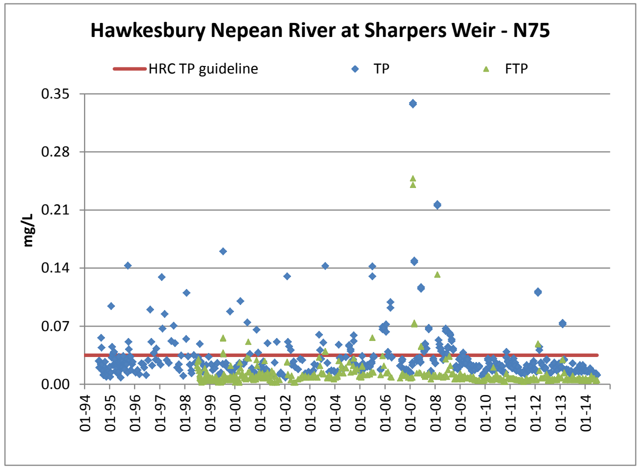
9.1 Appendix A Trend Analysis receiving waters temporal plots

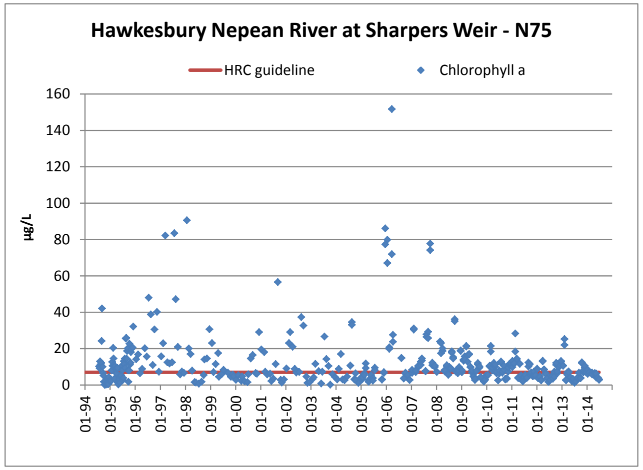


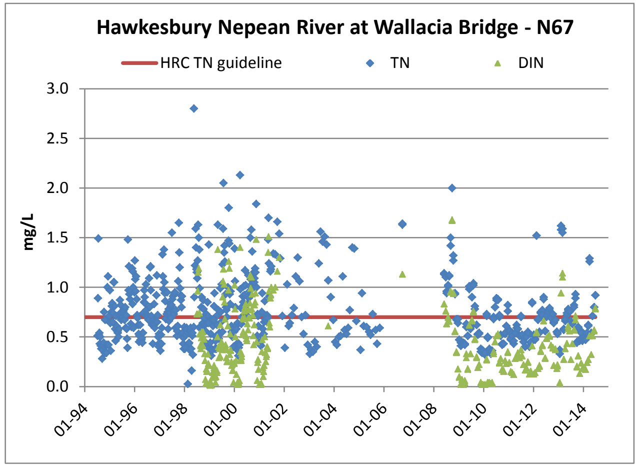








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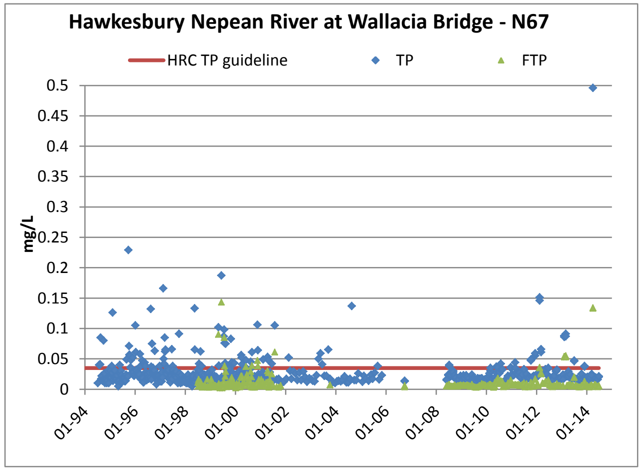
TN
DIN
mg/L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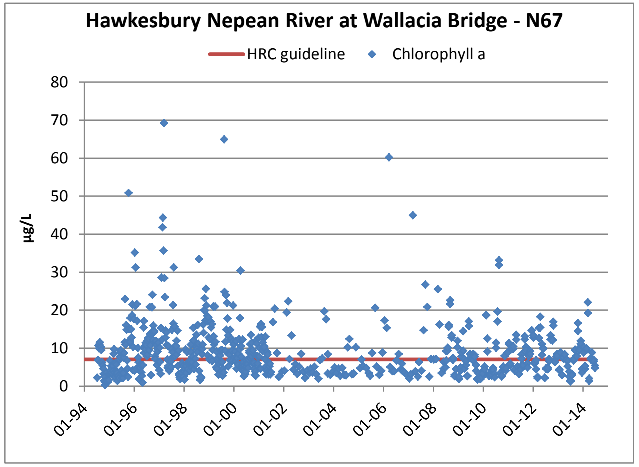
HRC TP guideline
TP
FTP
mg/L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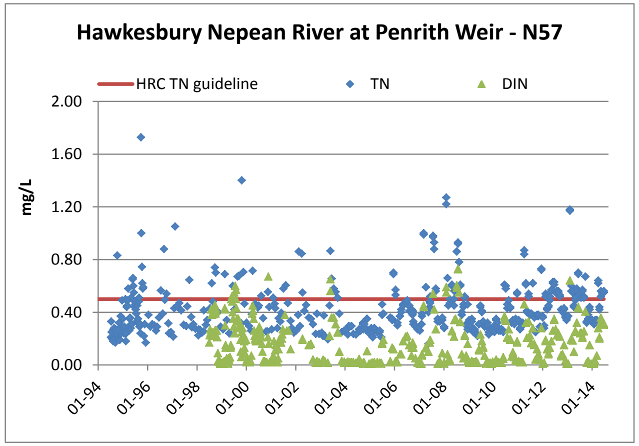
HRC guideline
Chlorophyll a
µg/L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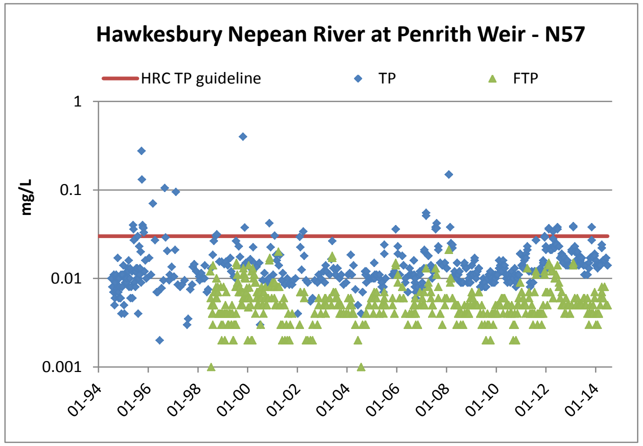
TN
DIN
mg/L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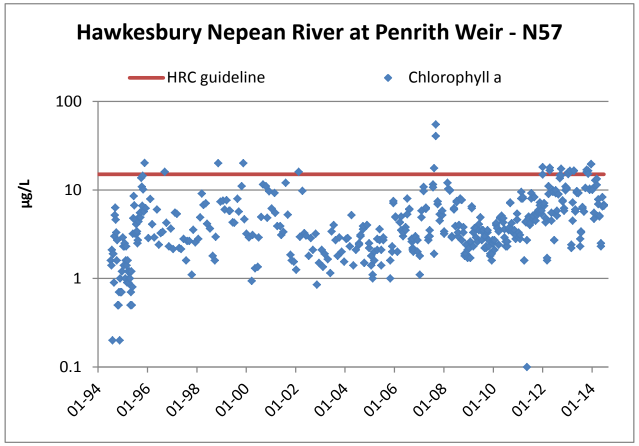
HRC TP guideline
TP
FTP
mg/L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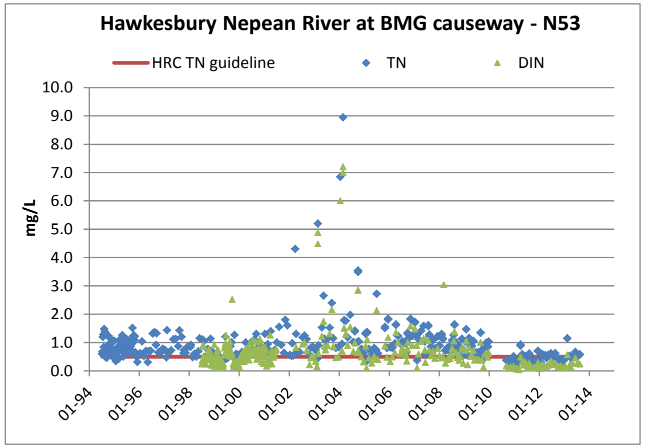
HRC guideline
Chlorophyll a
µg/L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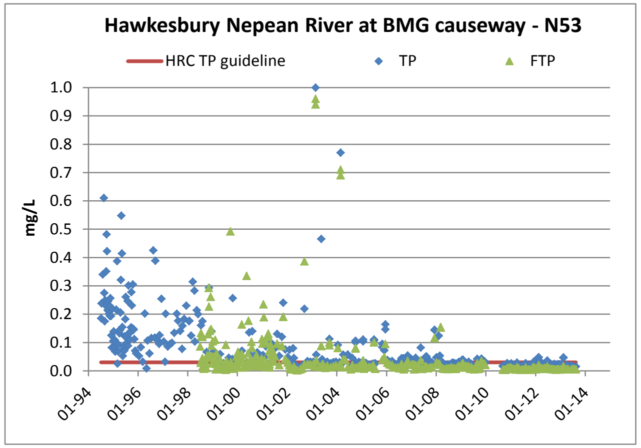
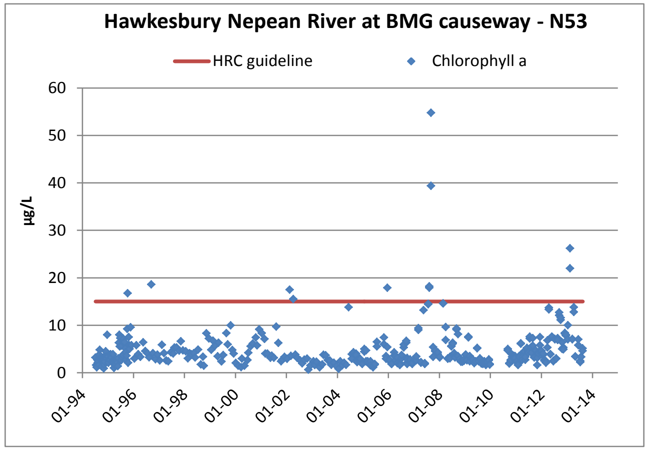
TN
DIN
mg/L


Hawkesbury Nepean River at BMG causeway - N53
HRC TP guideline
TP
FTP
mg/L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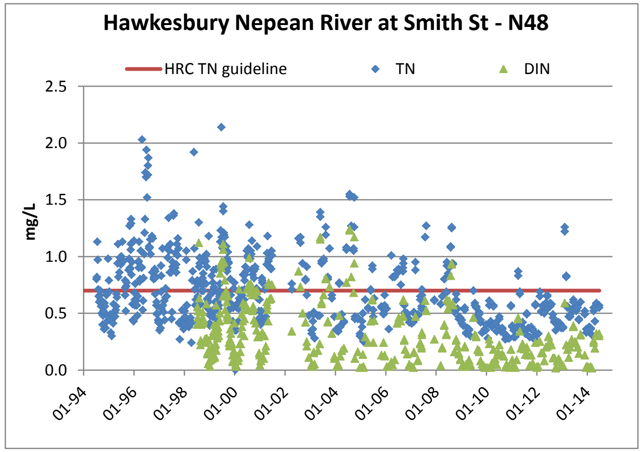
HRC guideline
Chlorophyll a
µg/L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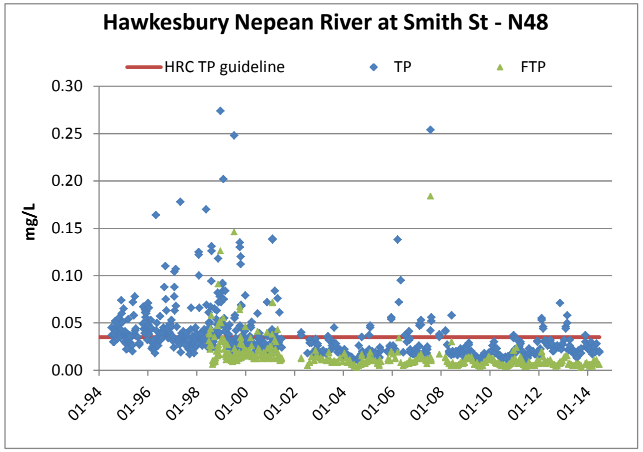
TN
DIN
mg/L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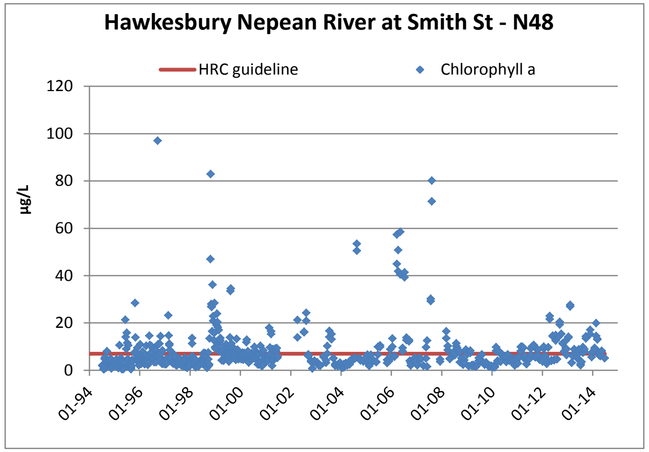
HRC TP guideline
TP
FTP
mg/L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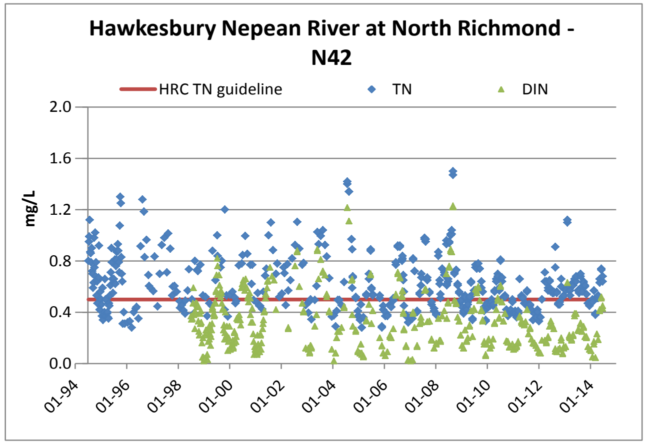
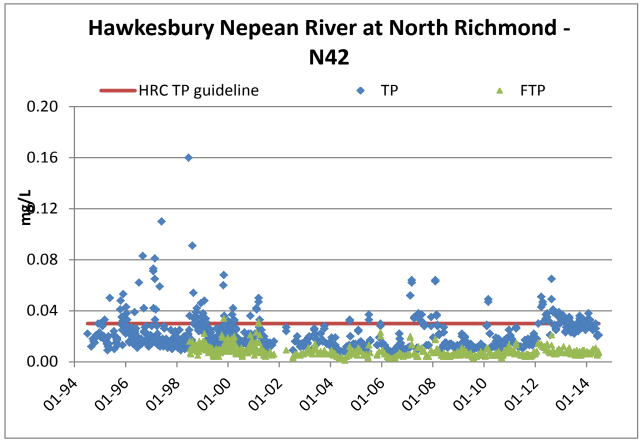
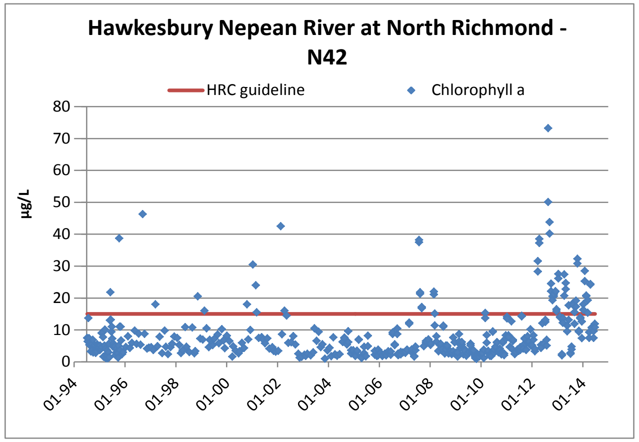
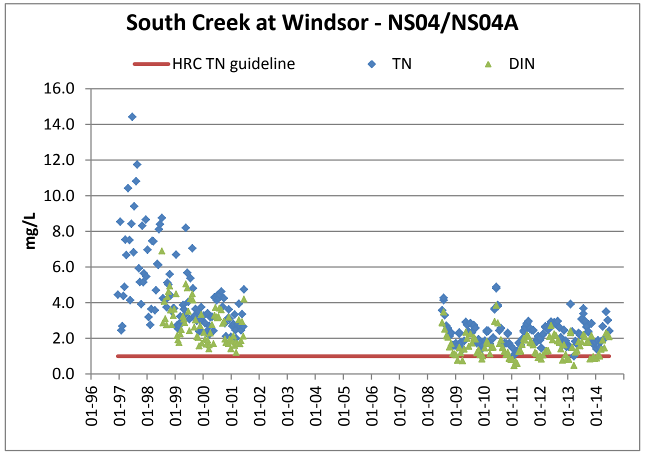
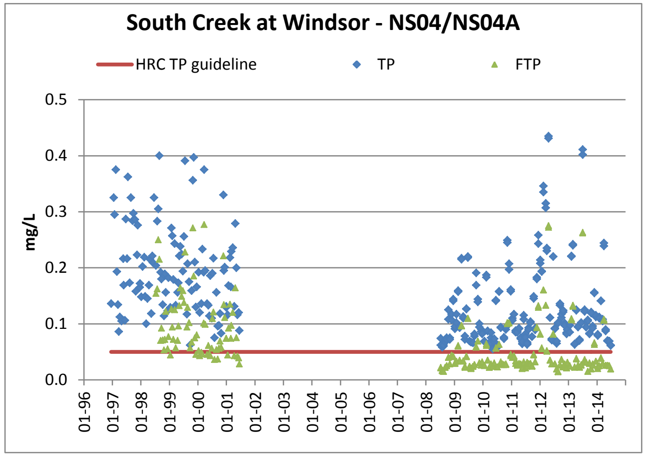
HRC guideline
Chlorophyll a
µg/L

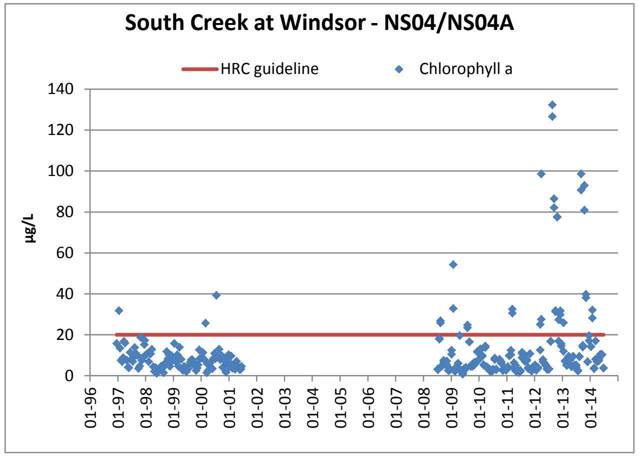

Hawkesbury Nepean River at North Richmond - N42
HRC TN guideline
TN
DIN
mg/L


Hawkesbury Nepean River at North Richmond - N42
HRC TP guideline
TP
FTP
mg/L


Hawkesbury Nepean River at North Richmond - N42
HRC guideline
Chlorophyll a
µg/L


South Creek at Windsor - NS04/NS04A
HRC TN guideline
TN
DIN
mg/L


South Creek at Windsor - NS04/NS04A
HRC TP guideline
TP
FTP
mg/L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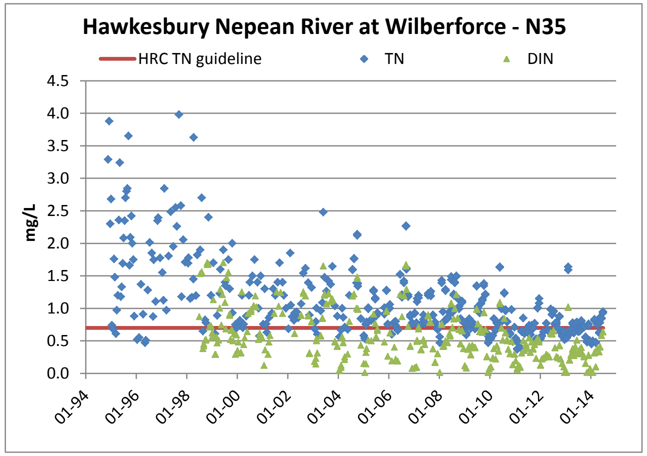
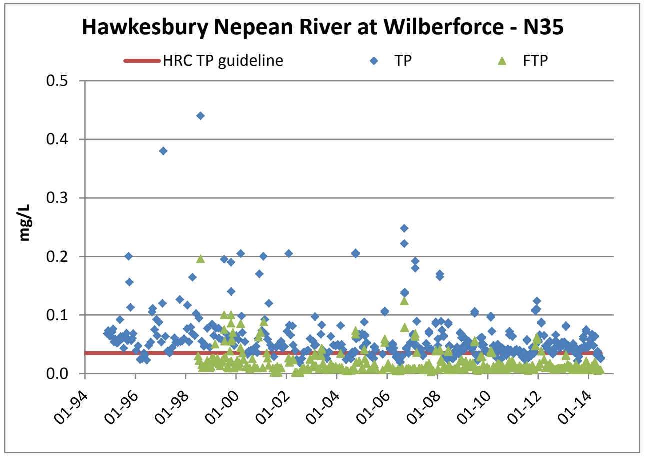
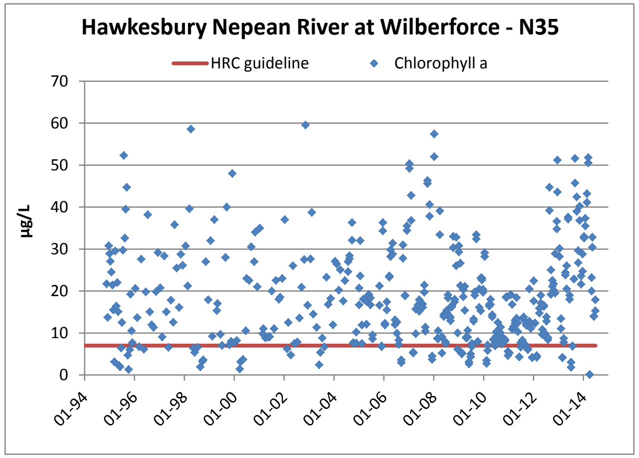
HRC guideline
Chlorophyll a
µg/L


Hawkesbury Nepean River at Wilberforce - N35
HRC TN guideline
TN
DIN
mg/L
4.5
4.0
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
01-94
01-96
01-98
01-00
01-02
01-04
01-06
01-08
01-10
01-12
01-14


Hawkesbury Nepean River at Wilberforce - N35
HRC TP guideline
TP
FTP
mg/L
0.5
0.4
0.3
0.2
0.1
0.0
01-94
01-96
01-98
01-00
01-02
01-04
01-06
01-08
01-10
01-12
01-14


Hawkesbury Nepean River at Wilberforce - N35
HRC guideline
Chlorophyll a
µg/L
70
60
50
40
30
20
10
0
01-94
01-96
01-98
01-00
01-02
01-04
01-06
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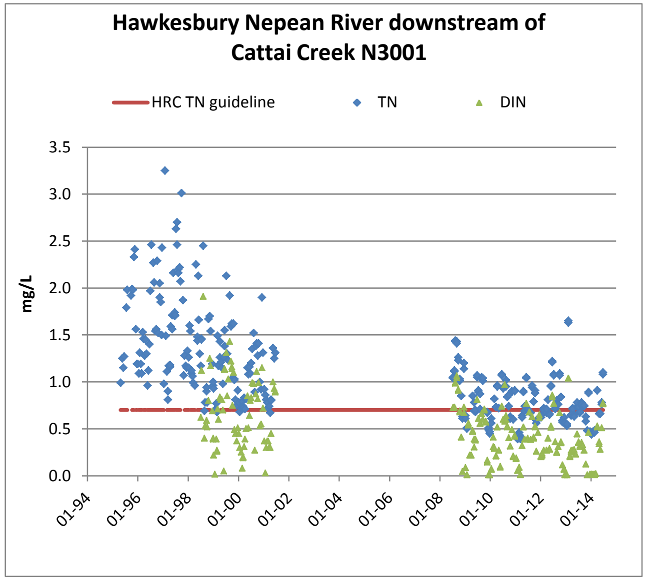
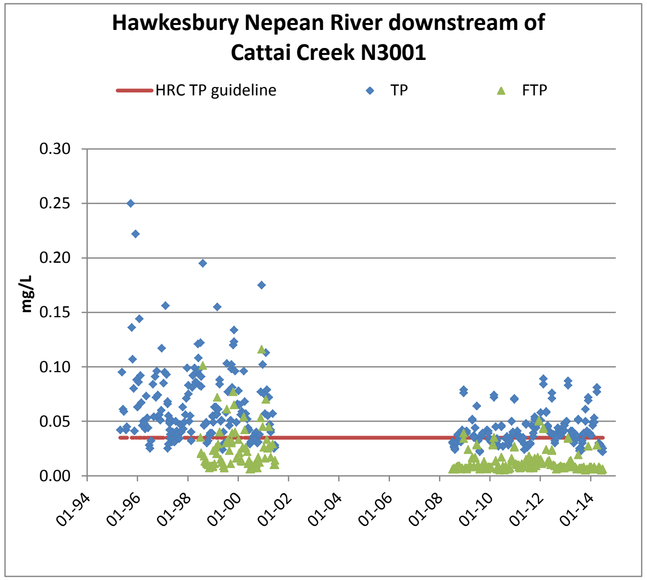
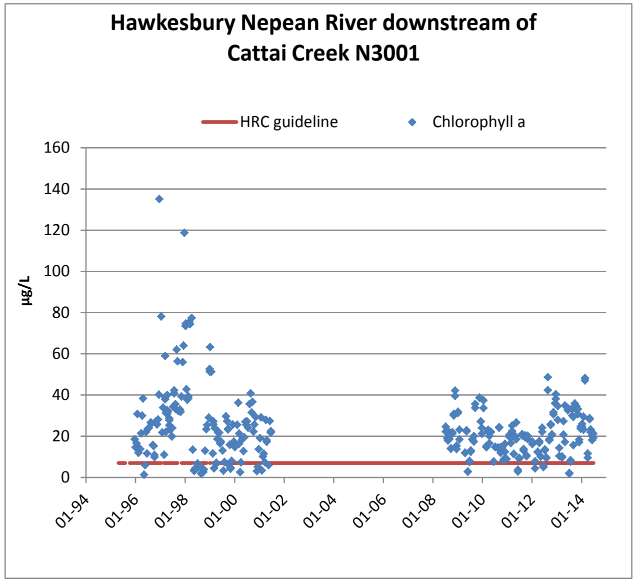
01-10
01-12
01-14


Hawkesbury Nepean River downstream of Cattai Creek N3001
HRC TN guideline
TN
DIN
mg/L
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
01-94
01-96
01-98
01-00
01-02
01-04
01-06
01-08
01-10
01-12
01-14


Hawkesbury Nepean River downstream of Cattai Creek N3001
HRC TP guideline
TP
FTP
mg/L
0.30
0.25
0.20
0.15
0.10
0.05
0.00
01-94
01-96
01-98
01-00
01-02
01-04
01-06
01-08
01-10
01-12
01-14


Hawkesbury Nepean River downstream of Cattai Creek N3001
HRC guideline
Chlorophyll a
µg/L
160
140
120
100
80
60
40
20
0
01-94
01-96
01-98
01-00
01-02
01-04
01-06
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01-10
01-12
01-14

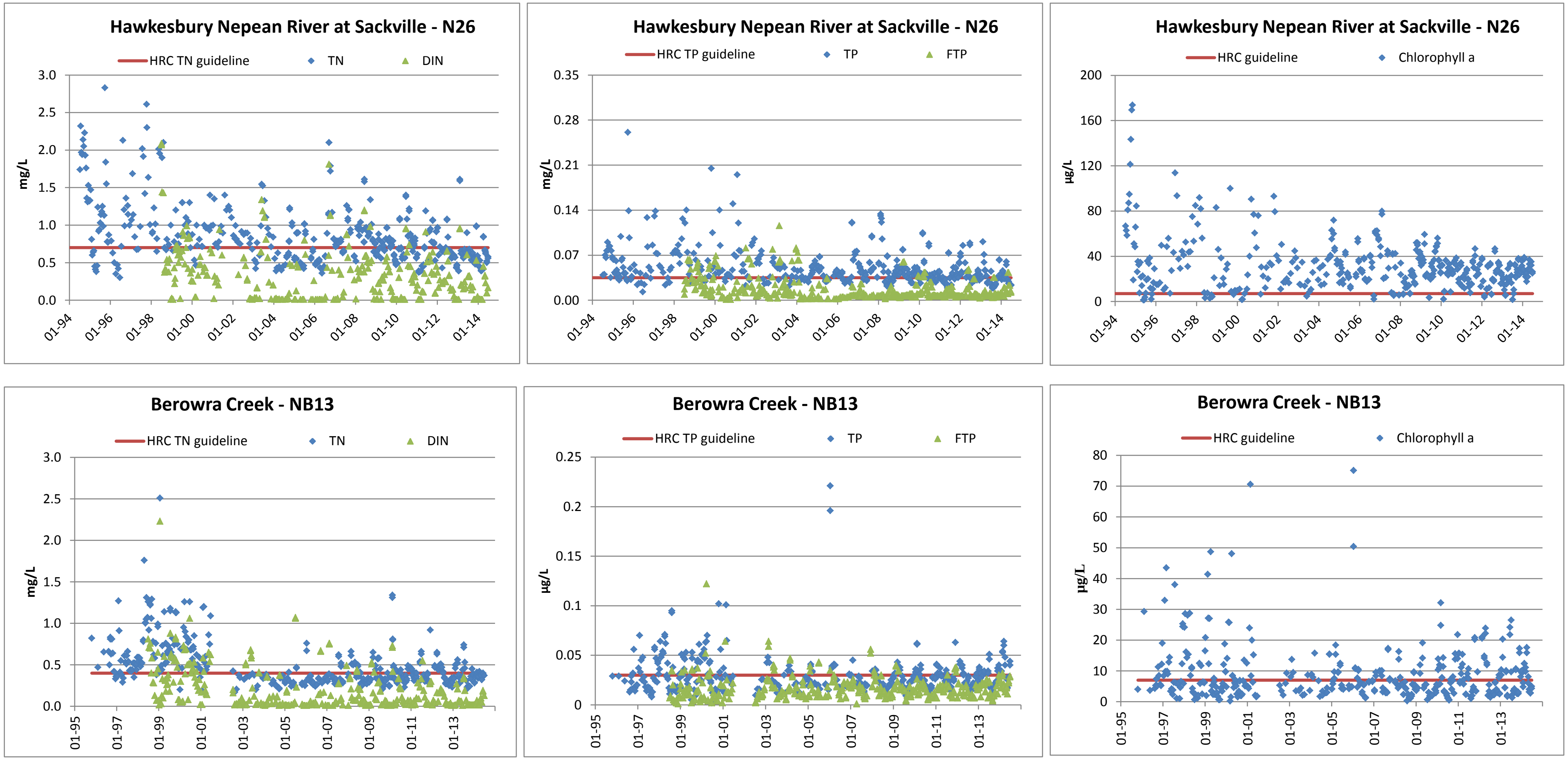
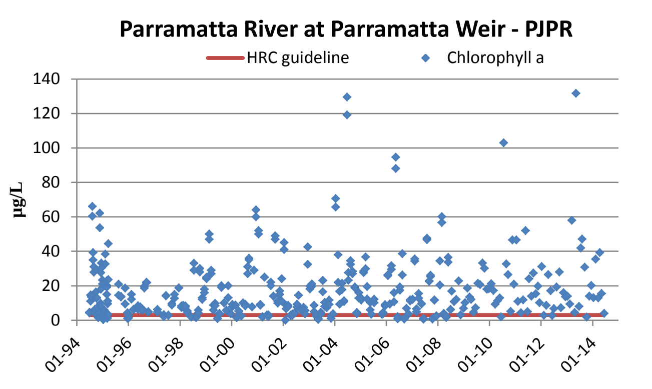


Figure 9-1 Trend analysis temporal plots for Hawkesbury Nepean River sites for total nitrogen (TN) and dissolved inorganic nitrogen (DIN) on the left, total phosphorus (TP) and filtered total phosphorus (FTP) in the middle and chlorophyll *a* on the right.


Parramatta River at Parramatta Weir - PJPR
HRC guideline
Chlorophyll a
µg/L
0
20
40
60
80
100
120
140
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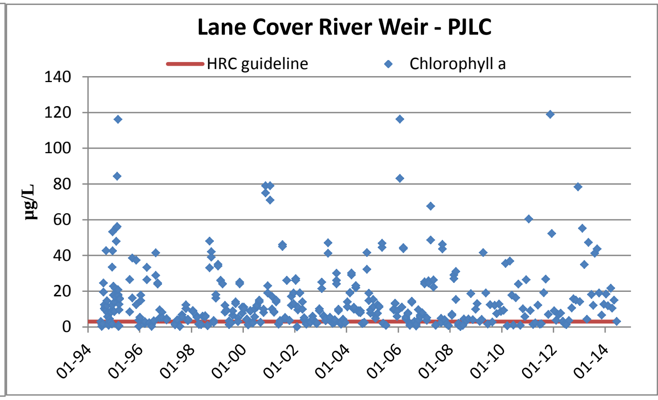
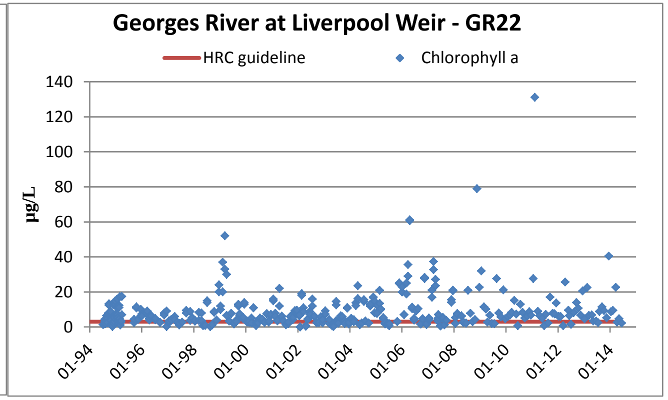
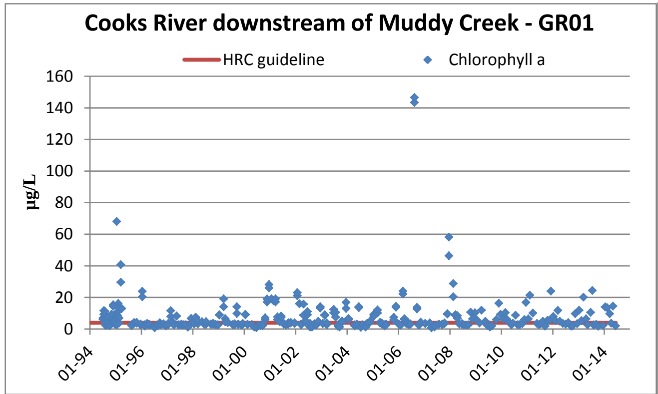
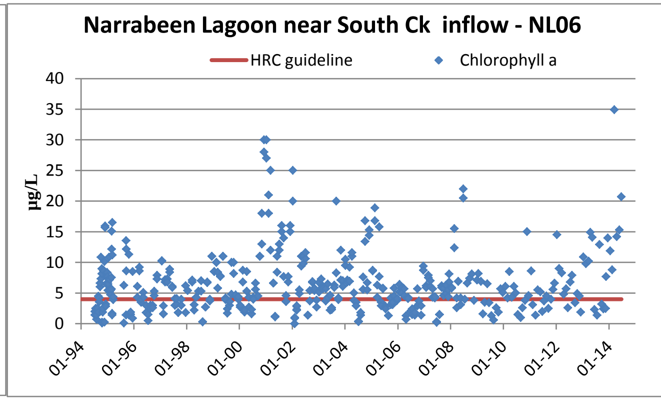
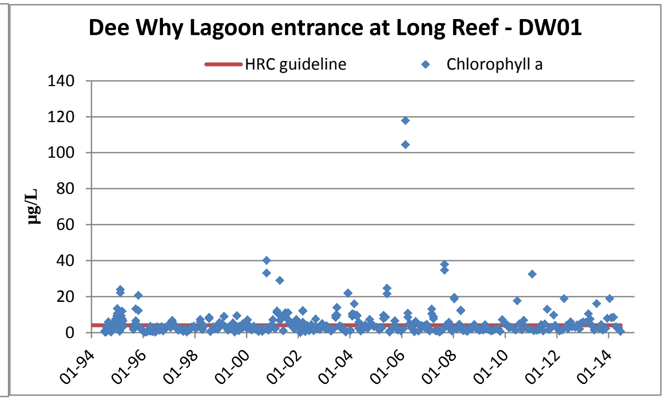
01-10
01-12
01-14

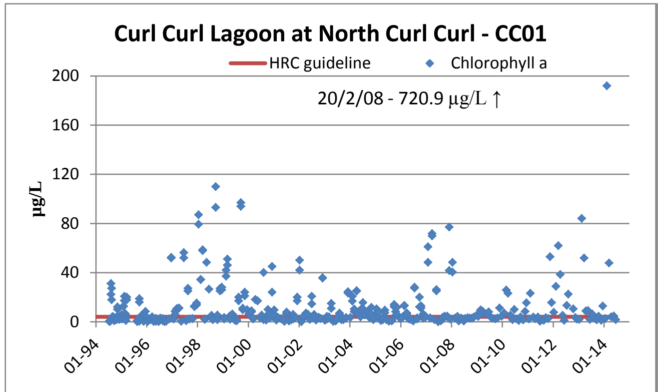

Lane Cover River Weir - PJLC
HRC guideline
Chlorophyll a
µg/L
0
20
40
60
80
100
120
140
01-94
01-96
01-98
01-00
01-02
01-04
01-06
01-08
01-10
01-12
01-14

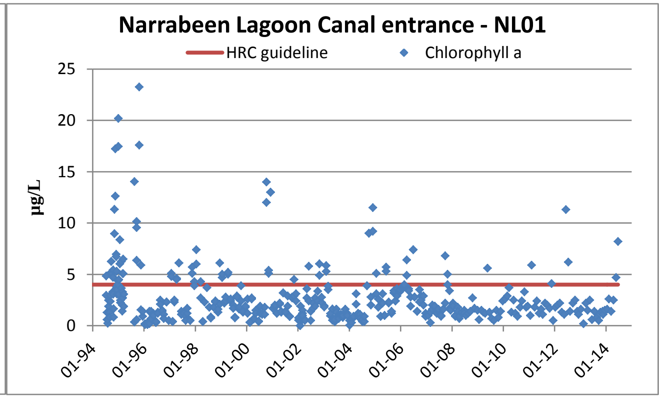

Georges River at Liverpool Weir - GR22
HRC guideline
Chlorophyll a
µg/L
0
20
40
60
80
100
120
140
01-94
01-96
01-98
01-00
01-02
01-04
01-06
01-08
01-10
01-12
01-14

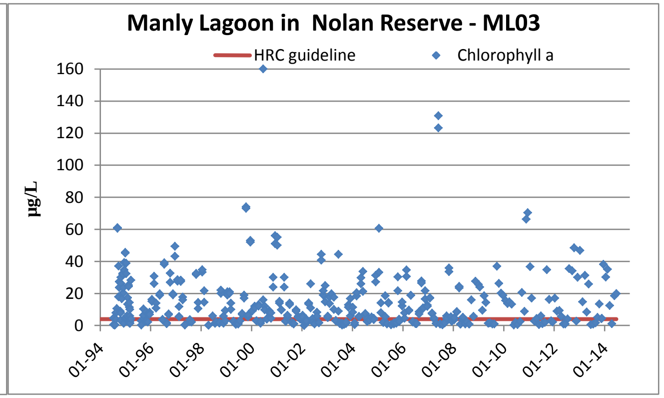

Cooks River downstream of Muddy Creek - GR01
HRC guideline
Chlorophyll a
µg/L
0
20
40
60
80
100
120
140
160
01-94
01-96
01-98
01-00
01-02
01-04
01-06
01-08
01-10
01-12
01-14


Narrabeen Lagoon near South Ck inflow - NL06
HRC guideline
Chlorophyll a
µg/L
0
5
10
15
20
25
30
35
40
01-94
01-96
01-98
01-00
01-02
01-04
01-06
01-08
01-10
01-12
01-14


Dee Why Lagoon entrance at Long Reef - DW01
HRC guideline
Chlorophyll a
µg/L
0
20
40
60
80
100
120
140
01-94
01-96
01-98
01-00
01-02
01-04
01-06
01-08
01-10
01-12
01-14


Curl Curl Lagoon at North Curl Curl - CC01
HRC guideline
Chlorophyll a
20/2/08 - 720.9 µg/L ↑
µg/L
0
40
80
120
160
200
01-94
01-96
01-98
01-00
01-02
01-04
01-06
01-08
01-10
01-12
01-14


Narrabeen Lagoon Canal entrance - NL01
HRC guideline
Chlorophyll a
µg/L
0
5
10
15
20
25
01-94
01-96
01-98
01-00
01-02
01-04
01-06
01-08
01-10
01-12
01-14


Manly Lagoon in Nolan Reserve - ML03
HRC guideline
Chlorophyll a
µg/L
0
20
40
60
80
100
120
140
160
01-94
01-96
01-98
01-00
01-02
01-04
01-06
01-08
01-10
01-12
01-14

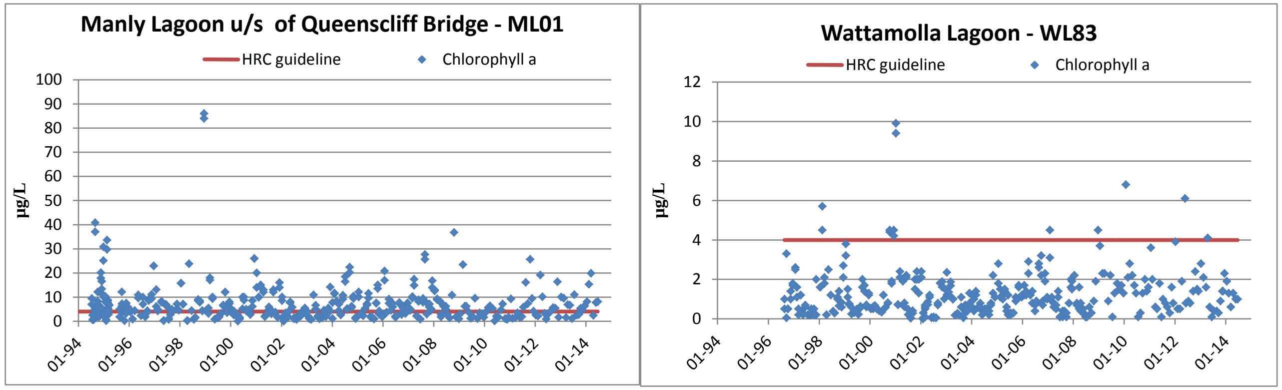


Figure 9-2 Trend analysis temporal plots for estuaries and urban river sites for chlorophyll *a* with NHMRC (2008) microbial assessment categories marked. Below 41 (green) is Category A, below 200 (orange) is Category B, below 500 (red) is Category C and above 500 (red) is category D

Table 9-1 Healthy Rivers Commission guidelines (HRC 1998)

Catchment	Healthy Rivers Commission Guideline	Sites	Water Quality Parameter		
			Total Nitrogen mg/L	Total Phosphorus mg/L	Chlorophyll *a* µg/L
Hawkesbury Nepean River catchment	Mixed use rural area and sandstone plateau	N92, N75, N67, N48, N35, N3001, N26	<0.70	<0.035	<7
	Predominantly urban	N57, N53, N42	<0.50	<0.030	<10-15
	Estuarine and brackish	NB13	<0.40	<0.030	<7
	Urban tributary	NS04/NS04A	<1.0	<0.05	<20
Non Hawkesbury Nepean River catchment	Freshwater	PJPR, PJLC, GR22			<3
	Estuarine and coastal lagoons	GR01, NL06, DW01, CC01, NL01, ML03, ML01, WL83			<4

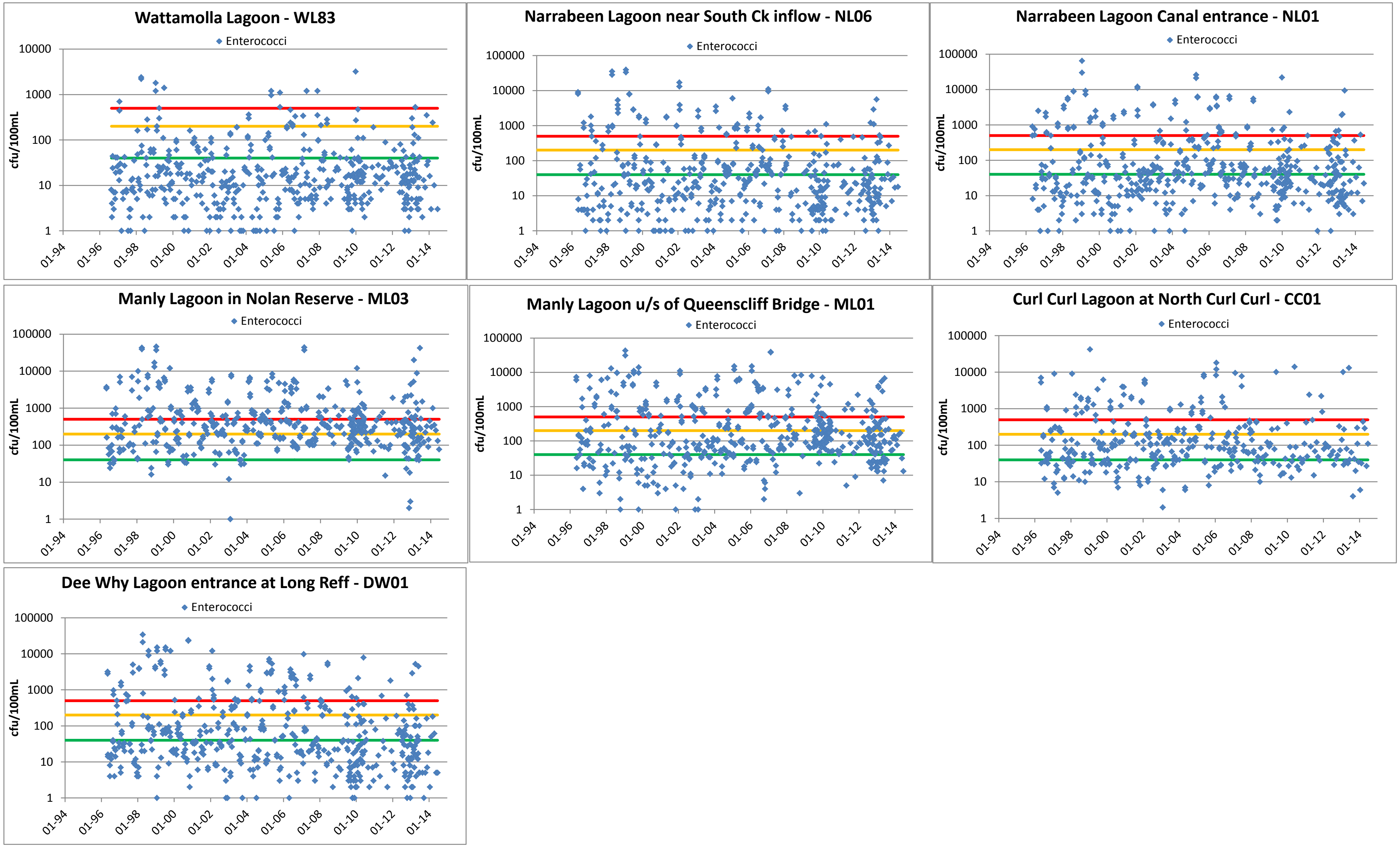


Figure 9-3 Trend analysis temporal plots for estuarine lagoon sites for Enterococci with NHMRC (2008) microbial assessment categories marked. Below 41 (green) is Category A, below 200 (orange) is Category B, below 500 (red) is Category C and above 500 (red) is category D

9.2 Appendix B PERMANOVA analyses for the Glenfield plant overflow in November 2013

PERMANOVA SITE02

Sums of squares type: Type III (partial)
Fixed effects sum to zero for mixed terms
Permutation method: Permutation of residuals under a reduced model
Number of permutations: 9999

Factors

Name	Type	Levels
Period	Fixed	2
sub period	Random	2

PERMANOVA table of results

Source	df	SS	MS	Pseudo-F	P(perm)	Unique perms	P(MC)
Period	1	2378.8	2378.8	2.6473	0.1733	6	0.1095
sub period(Period)	2	1880.1	940.06	1.5603	0.1404	9927	0.1546
Res	33	19882	602.48				
Total	36	24145					

Details of the expected mean squares (EMS) for the model

Source	EMS
Period	1*V(Res) + 8.1039*V(sub period(Period)) + 16.208*S(Period)
sub period(Period)	1*V(Res) + 9.24*V(sub period(Period))
Res	1*V(Res)

Construction of Pseudo-F ratio(s) from mean squares

Source	Numerator	Denominator	Num.df	Den.df
Period	1*Period	0.87705*sub period(Period) + 0.12295*Res	1	2.37
sub period(Period)	1*sub period(Period)	1*Res	2	33

Estimates of components of variation

Source	Estimate	Sq.root
S(Period)	91.327	9.5565
V(sub period(Period))	36.534	6.0443
V(Res)	602.48	24.546

PERMANOVA SITE08

Transform: Square root
Resemblance: S17 Bray Curtis similarity (+d)
Sums of squares type: Type III (partial)
Fixed effects sum to zero for mixed terms
Permutation method: Permutation of residuals under a reduced model
Number of permutations: 9999

Factors

Name	Type	Levels
Period	Fixed	2
sub period	Random	2

PERMANOVA table of results

Source	df	SS	MS	Pseudo-F	P(perm)	Unique perms	P(MC)
Period	1	1091.6	1091.6	1.3391	0.3304	6	0.3332
sub period(Period)	2	1643.2	821.61	1.6782	0.1373	9949	0.1387
Res	27	13218	489.57				
Total	30	15971					

Details of the expected mean squares (EMS) for the model

Source	EMS
Period	1*V(Res) + 7.3625*V(sub period(Period)) + 14.725*S(Period)
sub period(Period)	1*V(Res) + 7.5085*V(sub period(Period))
Res	1*V(Res)

Construction of Pseudo-F ratio(s) from mean squares

Source	Numerator	Denominator	Num.df	Den.df
Period	1*Period	0.98056*sub period(Period) + 1.9444E-2*Res	1	2.05
sub period(Period)	1*sub period(Period)	1*Res	2	27

Estimates of components of variation

Source	Estimate	Sq.root
S(Period)	18.772	4.3326
V(sub period(Period))	44.222	6.65
V(Res)	489.57	22.126

PERMANOVA SITE12

Sums of squares type: Type III (partial)

Fixed effects sum to zero for mixed terms
Permutation method: Permutation of residuals under a reduced model
Number of permutations: 9999

Factors

Name	Type	Levels
Period	Fixed	2
sub period	Random	2

PERMANOVA table of results

Source	df	SS	MS	Pseudo-F	P(perm)	Unique perms	P(MC)
Period	1	1753.5	1753.5	3.8466	0.168	6	0.0585
sub period(Period)	2	867.58	433.79	0.65677	0.7157	9930	0.6883
Res	34	22457	660.49				
Total	37	25109					

Details of the expected mean squares (EMS) for the model

Source	EMS
Period	1*V(Res) + 8.4771*V(sub period(Period)) + 16.954*S(Period)
sub period(Period)	1*V(Res) + 9.3908*V(sub period(Period))
Res	1*V(Res)

Construction of Pseudo-F ratio(s) from mean squares

Source	Numerator	Denominator	Num.df	Den.df
Period	1*Period	0.9027*sub period(Period) + 9.7298E-2*Res	1	2.71
sub period(Period)	1*sub period(Period)	1*Res	2	34

Estimates of components of variation

Source	Estimate	Sq.root
S(Period)	76.537	8.7485
V(sub period(Period))	-24.14	-4.9133
V(Res)	660.49	25.7

9.3 Appendix C Summary Statistics for the sewage overflow at Glenfield in November 2013

Summary statistics notes:

Periods of cover for statistical analysis

'Before' data covers the period 1995 to 2008 and have not been averaged or filtered to remove wet weather.

'Event' data are from 22 November to the 12 December 2013. Data are daily-averaged where there is replicate data, for each day of 26 November to 2 December.

'After' data covers the period 18 December 2013 to the 23 June 2014. Data have not been averaged or filtered to remove wet weather.

Sites

Site	Location
Site 1	BCC, 8 m downstream of the SPS 353 overflow outlet
Site 3	BCC, ~140 m upstream of the confluence with the Georges River
Site 4	BCC, ~10 m upstream of the overflow point, SPS 353
Site 22	BCC, Pool on Bunbury Curran Creek, ~100 m from the Georges River
Site 7	Georges River, end of Victoria Road, Glenfield, upstream site
Site 11	Georges River at Ingleburn Weir (GR24), upstream site
Site 2	Georges River, upstream at Cambridge Avenue bridge
Site 8	Georges River at Liverpool Weir upstream side
Site 9	Georges River at north east point of Haigh Park
Site 10	Georges River at Epson Road
Site 12	Chipping Norton Lake, Angle Park boat ramp
Site 13	Chipping Norton Lake, at Grand Flaneur Beach
Site 14	Chipping Norton Lake, at the end of Georges River Road
Site 15	Georges River, at Davy Robinson boat ramp
Site 16	Georges River, downstream of the confluence with Harris Creek
Site 17	Georges River, at the confluence with Salt Pan Creek
Site 18	Georges River, at the confluence with Woronora River
Site 19	Georges River, at the entrance to Kogarah Bay
Site 20	Georges River, at Woolooware Bay
Site 21	Georges River, at the entrance to Botany Bay

Analyte abbreviations

Analyte	Analyte name	Units
DOS	Dissolved oxygen	mg/L
pH	pH	pH units
AMM	Ammoniacal nitrogen	mg/L
TN	Total nitrogen	mg/L
TP	Tota phosphorus	mg/L
FC	Faecal coliforms	cfu/100mL
ENT	Enterococci	cfu/100mL
CHLA	Chlorophyll *a*	mg/m^3 (µg/L)

Column headings in the tables (examples)

DOS-1	Dissolved oxygen saturation for site 1
pH-1	pH for site 1
AMM-2	Ammoniacal nitrogen for site 2
FC-3	Faecal coliforms for site 3

Sites are presented in number order, as sets of ‘Before’, ‘Event’ and ‘After’.

‘Before’ results

Std.Dev. = standard deviation, 95 %ile = 95th percentile

‘Before’	DOS-2	pH-2	AMM-2	TN-2	TP-2	FC-2	ENT-2	CHLA-2
Minimum	14	6.8	0.002	0.170	0.007	3	2	0.3
1st Quartile	63	7.3	0.010	0.290	0.019	18	12	2.4
Mean	77	7.5	0.031	0.406	0.031	257	102	7.7
Median	78	7.5	0.020	0.370	0.025	41	28	4.9
3rd Quartile	92	7.7	0.032	0.470	0.034	195	64	9.2
Maximum	134	8.8	0.460	1.210	0.132	7900	3000	86.1
Count	117	118	121	121	121	121	121	119
Std. Dev.	23	0.4	0.050	0.164	0.021	814	305	10.4
SE Mean	2	0.032	0.005	0.015	0.002	74	28	0.956
95 %ile	110	8.2	0.084	0.720	0.076	1000	390	22.0

‘Before’	DOS-8	pH-8	AMM-8	TN-8	TP-8	FC-8	ENT-8	CHLA-8
Minimum	20	6.8	0.002	0.230	0.006	24	13	0.5
1st Quartile	58	7.3	0.010	0.340	0.016	74.75	32.25	3.0
Mean	73	7.6	0.034	0.451	0.024	247	135	8.6
Median	74	7.6	0.025	0.420	0.021	145	56	5.3
3rd Quartile	89	7.8	0.042	0.490	0.029	233	94	9.9
Maximum	136	9.1	0.245	1.100	0.073	7200	2100	60.9
Count	117	117	118	118	118	118	118	116
Std. Dev.	24	0.4	0.035	0.146	0.012	668	290	9.9
SE Mean	2	0.038	0.003	0.013	0.001	61	27	0.919
95 %ile	112	8.3	0.091	0.685	0.052	537	436	29.1

‘Before’	DOS-11	pH-11	AMM-11	TN-11	TP-11	FC-11	ENT-11	CHLA-11
Minimum	72	6.9	0.002	0.120	0.002	1	3	0.0
1st Quartile	87	7.5	0.010	0.170	0.005	8.75	15.75	0.4
Mean	96	7.7	0.010	0.229	0.008	110	90	1.8
Median	97	7.7	0.010	0.220	0.006	18	39	0.8
3rd Quartile	104	7.9	0.010	0.263	0.008	66	65	1.2
Maximum	148	8.2	0.030	0.750	0.065	3050	1900	92.3
Count	118	118	120	120	120	120	120	118
Std. Dev.	13	0.3	0.004	0.081	0.006	324	232	8.5
SE Mean	1	0.027	0.000	0.007	0.001	30	21	0.779
95 %ile	115	8.1	0.020	0.330	0.014	639	229	3.1

‘Before’	DOS-12	pH-12	AMM-12	TN-12	TP-12	FC-12	ENT-12	CHLA-12
Minimum	37	6.6	0.006	0.270	0.002	5	2	0.4
1st Quartile	71	7.3	0.011	0.400	0.023	26	15	3.5
Mean	80	7.5	0.049	0.612	0.048	109	95	9.7
Median	81	7.5	0.028	0.490	0.037	52	26	6.4
3rd Quartile	91	7.6	0.058	0.770	0.059	105	52	11.1
Maximum	122	9.1	0.320	2.590	0.185	850	4014	54.0
Count	119	119	121	121	121	121	121	119
Std. Dev.	15	0.3	0.057	0.321	0.036	161	382	9.8
SE Mean	1	0.027	0.005	0.029	0.003	15	35	0.894
95 %ile	103	7.9	0.185	1.160	0.135	420	290	29.3

‘Before’	DOS-17	pH-17	AMM-17	TN-17	TP-17	FC-17	ENT-17	CHLA-17
Minimum	51	6.9	0.010	0.190	0.018	1	1	0.4
1st Quartile	75	7.5	0.020	0.290	0.034	9	5.75	2.3
Mean	84	7.7	0.050	0.456	0.050	191	95	6.9
Median	83	7.7	0.025	0.375	0.045	24	17	4.1
3rd Quartile	91	7.8	0.061	0.510	0.060	58	37	9.1
Maximum	125	8.4	0.335	1.720	0.130	7450	5050	55.1
Count	119	119	120	120	120	120	120	118
Std. Dev.	13	0.2	0.061	0.265	0.021	780	486	7.6
SE Mean	1	0.022	0.006	0.024	0.002	71	44	0.701
95 %ile	106	8.0	0.221	0.992	0.083	655	210	21.4

‘Before’	DOS-20	pH-20	AMM-20	TN-20	TP-20	FC-20	ENT-20	CHLA-20
Minimum	78	6.8	0.010	0.080	0.010	1	1	0.5
1st Quartile	88	7.9	0.010	0.170	0.020	2	2	1.5
Mean	94	8.0	0.019	0.223	0.027	13	10	2.8
Median	93	8.0	0.010	0.205	0.025	5	4	2.5
3rd Quartile	98	8.1	0.020	0.250	0.030	12	9	3.4
Maximum	120	8.4	0.190	1.000	0.163	150	215	9.1
Count	119	118	120	120	120	120	120	119
Std. Dev.	7	0.2	0.020	0.103	0.015	24	24	1.6
SE Mean	1	0.020	0.002	0.009	0.001	2	2	0.150
95 %ile	107	8.2	0.040	0.330	0.045	68	34	5.9

‘Event’ results

‘Event’	DOS-1	pH-1	AMM-1	TN-1	TP-1	FC-1	ENT-1	CHLA-1
Minimum	32.5	6.8	0.005	0.670	0.051	250	160	ns
1st Quartile	44.95	7.3	0.045	0.800	0.073	690	405	ns
Mean	56	7.4	0.119	2.407	0.274	194946	45729	ns
Median	53	7.4	0.100	0.910	0.082	2500	770	ns
3rd Quartile	63	7.5	0.190	1.130	0.101	18000	1850	ns
Maximum	83	7.6	0.390	26.000	3.390	2013333	443333	ns
Count	15	19	17	20	20	21	21	ns
Std. Dev.	15	0.2	0.102	5.647	0.740	556877	128254	ns
SE Mean	4	0.046	0.025	1.263	0.165	121521	27987	ns
95 %ile	83	7.6	0.390	5.450	0.515	1700000	415000	ns
‘Event’	DOS-2	pH-2	AMM-2	TN-2	TP-2	FC-2	ENT-2	CHLA-2
Minimum	35.1	6.8	0.010	0.430	0.048	140	10	1.9
1st Quartile	47.7	7.0	0.060	0.510	0.058	460	30	4.5
Mean	61	7.1	0.224	0.708	0.077	46311	3519	13.0
Median	63	7.1	0.230	0.720	0.068	845	90	11.4
3rd Quartile	70	7.2	0.355	0.895	0.097	5567	350	16.3
Maximum	92	7.5	0.580	1.000	0.136	596333	63533	35.2
Count	15	20	19	19	19	22	22	19
Std. Dev.	15	0.2	0.172	0.192	0.026	143049	13578	8.8
SE Mean	4	0.044	0.039	0.044	0.006	30498	2895	2.027
95 %ile	92	7.5	0.580	1.000	0.136	345000	10300	35.2

'Event'	DOS-3	pH-3	AMM-3	TN-3	TP-3	FC-3	ENT-3	CHLA-3
Minimum	19	7.0	0.005	0.380	0.041	180	20	2.5
1st Quartile	38.2	7.1	0.010	0.850	0.114	480	110	5.2
Mean	54	7.3	0.487	1.676	0.213	119158	10926	24.9
Median	59	7.2	0.010	1.040	0.140	5000	380	9.2
3rd Quartile	72	7.4	0.370	1.410	0.173	32000	1200	25.4
Maximum	84	7.8	4.800	6.200	0.975	1200000	100000	128.6
Count	14	16	16	16	16	17	17	13
Std. Dev.	22	0.2	1.189	1.672	0.221	318861	28879	35.6
SE Mean	6	0.057	0.297	0.418	0.055	77335	7004	9.886
95 %ile	84	7.8	4.800	6.200	0.975	1200000	100000	128.6
'Event'	DOS-4	pH-4	AMM-4	TN-4	TP-4	FC-4	ENT-4	CHLA-4
Minimum	36.4	7.3	0.005	0.600	0.047	200	25	ns
1st Quartile	47.7	7.3	0.060	0.710	0.052	440	230	ns
Mean	61	7.4	0.106	0.867	0.069	4770	2354	ns
Median	63	7.4	0.090	0.785	0.058	850	470	ns
3rd Quartile	71	7.5	0.150	1.060	0.073	1600	660	ns
Maximum	87	7.6	0.310	1.375	0.164	37000	20000	ns
Count	15	18	17	19	19	20	20	ns
Std. Dev.	15	0.1	0.082	0.235	0.029	10164	5143	ns
SE Mean	4	0.027	0.020	0.054	0.007	2273	1150	ns
95 %ile	87	7.6	0.310	1.375	0.164	30500	12000	ns
'Event'	DOS-7	pH-7	AMM-7	TN-7	TP-7	FC-7	ENT-7	CHLA-7
Minimum	73.95	6.7	0.005	0.290	0.017	10	10	3.7
1st Quartile	86.1	7.1	0.010	0.310	0.018	54.5	25	6.2
Mean	92	7.4	0.011	0.361	0.027	984	73	13.9
Median	91	7.3	0.010	0.340	0.024	69	40	10.6
3rd Quartile	97	7.5	0.013	0.390	0.033	127	120	14.1
Maximum	116	9.1	0.025	0.560	0.044	17000	220	52.4
Count	15	17	17	17	17	19	19	17
Std. Dev.	10	0.5	0.006	0.069	0.009	3879	61	12.3
SE Mean	3	0.128	0.001	0.017	0.002	890	14	2.987
95 %ile	116	9.1	0.025	0.560	0.044	17000	220	52.4
'Event'	DOS-8	pH-8	AMM-8	TN-8	TP-8	FC-8	ENT-8	CHLA-8
Minimum	36	6.9	0.040	0.830	0.052	50	10	2.6
1st Quartile	67.5	7.1	0.110	1.010	0.071	59	10	21.6
Mean	85	7.4	0.918	1.888	0.120	2447	48	49.7
Median	93	7.2	0.460	1.630	0.088	230	20	39.9
3rd Quartile	106	7.5	1.540	2.585	0.182	910	79	82.0
Maximum	122	8.3	2.200	3.020	0.208	15437	175	110.4
Count	15	16	16	16	16	18	18	16
Std. Dev.	26	0.4	0.794	0.748	0.058	4474	51	35.7
SE Mean	7	0.101	0.199	0.187	0.015	1055	12	8.918
95 %ile	122	8.3	2.200	3.020	0.208	15437	175	110.4

‘Event’	DOS-9	pH-9	AMM-9	TN-9	TP-9	FC-9	ENT-9	CHLA-9
Minimum	92.2	7.1	0.010	0.840	0.050	50	10	21.5
1st Quartile	102	7.2	0.040	1.020	0.058	79	30	25.6
Mean	115	7.7	0.314	1.669	0.119	526	66	84.0
Median	119	7.5	0.080	1.220	0.068	210	50	41.1
3rd Quartile	125	8.1	0.365	2.535	0.197	450	95	148.7
Maximum	134	8.6	1.320	3.390	0.319	2463	170	224.1
Count	15	16	14	14	14	18	18	13
Std. Dev.	14	0.5	0.431	0.832	0.086	730	50	78.3
SE Mean	4	0.133	0.115	0.222	0.023	172	12	21.706
95 %ile	134	8.6	1.320	3.390	0.319	2463	170	224.1

‘Event’	DOS-10	pH-10	AMM-10	TN-10	TP-10	FC-10	ENT-10	CHLA-10
Minimum	69.95	6.9	0.010	0.870	0.064	20	10	16.3
1st Quartile	80.15	7.2	0.030	1.000	0.076	69	20	29.1
Mean	91	7.4	0.141	1.188	0.088	549	77	36.8
Median	93	7.4	0.060	1.130	0.083	99	40	35.7
3rd Quartile	102	7.6	0.225	1.310	0.096	200	110	45.7
Maximum	115	7.8	0.430	1.665	0.124	6967	280	64.2
Count	15	16	14	14	14	18	18	13
Std. Dev.	14	0.3	0.149	0.242	0.018	1616	84	13.1
SE Mean	4	0.068	0.040	0.065	0.005	381	20	3.632
95 %ile	115	7.8	0.430	1.665	0.124	6967	280	64.2

‘Event’	DOS-11	pH-11	AMM-11	TN-11	TP-11	FC-11	ENT-11	CHLA-11
Minimum	89.5	7.4	0.010	0.260	0.008	20	99	0.1
1st Quartile	90.8	7.5	0.010	0.260	0.009	50	99.5	0.4
Mean	97	7.6	0.010	0.279	0.011	65	181	0.7
Median	98	7.6	0.010	0.280	0.010	50	170	0.7
3rd Quartile	102	7.7	0.010	0.280	0.011	70	190	0.8
Maximum	103	7.8	0.010	0.300	0.018	110	280	1.7
Count	7	8	8	8	8	8	8	8
Std. Dev.	5	0.2	0.000	0.016	0.003	31	67	0.5
SE Mean	2	0.057	0.000	0.005	0.001	11	24	0.166
95 %ile	103	7.8	0.010	0.300	0.018	110	280	1.7

‘Event’	DOS-12	pH-12	AMM-12	TN-12	TP-12	FC-12	ENT-12	CHLA-12
Minimum	70.3	7.2	0.005	0.810	0.051	10	10	10.2
1st Quartile	82	7.4	0.010	0.950	0.072	40	10	14.2
Mean	110	7.8	0.111	1.194	0.095	127	291	41.7
Median	116	7.5	0.010	1.230	0.096	59	30	26.3
3rd Quartile	133	8.4	0.230	1.290	0.099	99	170	53.3
Maximum	155	8.6	0.330	1.720	0.170	1000	1800	129.0
Count	15	14	16	16	16	16	16	16
Std. Dev.	28	0.6	0.125	0.241	0.028	236	570	34.7
SE Mean	7	0.148	0.031	0.060	0.007	59	142	8.675
95 %ile	155	8.6	0.330	1.720	0.170	1000	1800	129.0

‘Event’	DOS-13	pH-13	AMM-13	TN-13	TP-13	FC-13	ENT-13	CHLA-13
Minimum	69.8	7.2	0.005	0.830	0.047	20	10	5.8
1st Quartile	91.9	7.5	0.010	0.900	0.073	59	50	12.9
Mean	110	8.0	0.103	1.142	0.091	193	250	28.0
Median	120	8.2	0.030	1.140	0.092	99	130	28.1
3rd Quartile	128	8.6	0.180	1.240	0.105	220	190	36.7
Maximum	150	8.7	0.310	1.480	0.128	590	1500	68.7
Count	15	14	16	16	16	16	16	16
Std. Dev.	25	0.5	0.115	0.201	0.023	169	373	16.8
SE Mean	6	0.145	0.029	0.050	0.006	42	93	4.191
95 %ile	150	8.7	0.310	1.480	0.128	590	1500	68.7
‘Event’	DOS-14	pH-14	AMM-14	TN-14	TP-14	FC-14	ENT-14	CHLA-14
Minimum	73.2	7.1	0.005	0.820	0.057	30	10	6.0
1st Quartile	92	7.5	0.005	0.960	0.082	89	20	14.2
Mean	110	8.0	0.095	1.161	0.098	618	420	32.3
Median	118	8.1	0.010	1.220	0.097	110	30	31.8
3rd Quartile	128	8.4	0.170	1.280	0.107	290	190	44.3
Maximum	143	8.6	0.270	1.660	0.148	7000	5000	55.2
Count	15	14	16	16	16	16	16	16
Std. Dev.	22	0.5	0.111	0.217	0.021	1710	1233	16.0
SE Mean	6	0.138	0.028	0.054	0.005	427	308	4.002
95 %ile	143	8.6	0.270	1.660	0.148	7000	5000	55.2
‘Event’	DOS-15	pH-15	AMM-15	TN-15	TP-15	FC-15	ENT-15	CHLA-15
Minimum	44.4	6.8	0.010	0.810	0.043	10	10	4.9
1st Quartile	55.6	7.1	0.030	0.890	0.066	40	20	6.0
Mean	69	7.3	0.086	1.013	0.084	98	52	8.4
Median	69	7.3	0.050	1.010	0.085	79	30	7.8
3rd Quartile	79	7.5	0.110	1.090	0.099	110	50	9.8
Maximum	95	7.7	0.260	1.130	0.110	380	250	14.8
Count	16	15	16	16	16	16	16	16
Std. Dev.	16	0.3	0.069	0.108	0.020	88	58	2.8
SE Mean	4	0.065	0.017	0.027	0.005	22	15	0.688
95 %ile	95	7.7	0.260	1.130	0.110	380	250	14.8
‘Event’	DOS-16	pH-16	AMM-16	TN-16	TP-16	FC-16	ENT-16	CHLA-16
Minimum	50.7	6.8	0.030	0.790	0.061	10	10	4.7
1st Quartile	51.2	7.1	0.060	0.800	0.063	20	20	5.0
Mean	61	7.2	0.096	0.959	0.077	41	36	7.3
Median	57	7.2	0.070	0.890	0.069	40	30	7.4
3rd Quartile	70	7.3	0.120	1.090	0.087	59	50	8.1
Maximum	75	7.5	0.190	1.130	0.101	59	59	10.6
Count	8	8	8	8	8	8	8	8
Std. Dev.	9	0.2	0.053	0.143	0.016	19	17	2.2
SE Mean	3	0.071	0.019	0.051	0.006	7	6	0.763
95 %ile	75	7.5	0.190	1.130	0.101	59	59	10.6

'Event'	DOS-17	pH-17	AMM-17	TN-17	TP-17	FC-17	ENT-17	CHLA-17
Minimum	74	7.6	0.010	0.340	0.032	10	10	6.5
1st Quartile	76	7.6	0.010	0.450	0.037	10	10	7.1
Mean	84	7.7	0.028	0.580	0.047	21	10	12.2
Median	83	7.7	0.020	0.530	0.044	10	10	9.1
3rd Quartile	88	7.7	0.030	0.670	0.049	10	10	13.2
Maximum	94	7.8	0.070	0.840	0.065	89	10	26.7
Count	8	8	8	8	8	8	8	8
Std. Dev.	7	0.1	0.021	0.169	0.012	28	0	6.6
SE Mean	2	0.032	0.007	0.060	0.004	10	0	2.339
95 %ile	94	7.8	0.070	0.840	0.065	89	10	26.7

'Event'	DOS-18	pH-18	AMM-18	TN-18	TP-18	FC-18	ENT-18	CHLA-18
Minimum	78.3	7.8	0.010	0.280	0.026	10	10	4.0
1st Quartile	81.7	7.9	0.010	0.300	0.028	10	10	4.9
Mean	90	7.9	0.021	0.394	0.039	10	10	8.1
Median	86	7.9	0.010	0.310	0.034	10	10	5.2
3rd Quartile	90	8.0	0.010	0.370	0.038	10	10	10.2
Maximum	115	8.0	0.090	0.840	0.062	10	10	13.4
Count	8	8	8	8	8	8	8	8
Std. Dev.	11	0.1	0.028	0.186	0.013	0	0	3.8
SE Mean	4	0.023	0.010	0.066	0.005	0	0	1.328
95 %ile	115	8.0	0.090	0.840	0.062	10	10	13.4

'Event'	DOS-19	pH-19	AMM-19	TN-19	TP-19	FC-19	ENT-19	CHLA-19
Minimum	76.9	7.9	0.005	0.220	0.020	10	10	2.9
1st Quartile	84.3	8.0	0.010	0.260	0.021	10	10	3.4
Mean	93	8.0	0.009	0.276	0.028	10	10	4.5
Median	91	8.0	0.010	0.270	0.027	10	10	4.2
3rd Quartile	95	8.0	0.010	0.290	0.031	10	10	4.7
Maximum	114	8.1	0.010	0.320	0.038	10	10	6.3
Count	8	8	8	8	8	8	8	8
Std. Dev.	11	0.1	0.002	0.032	0.006	0	0	1.2
SE Mean	4	0.020	0.001	0.011	0.002	0	0	0.416
95 %ile	114	8.1	0.010	0.320	0.038	10	10	6.3

'Event'	DOS-20	pH-20	AMM-20	TN-20	TP-20	FC-20	ENT-20	CHLA-20
Minimum	74.5	7.9	0.010	0.220	0.022	10	10	2.3
1st Quartile	86.1	7.9	0.010	0.240	0.025	10	10	3.4
Mean	91	8.0	0.015	0.281	0.033	17	48	4.0
Median	90	8.0	0.010	0.270	0.030	10	10	4.1
3rd Quartile	92	8.0	0.020	0.290	0.041	10	10	4.6
Maximum	110	8.1	0.030	0.350	0.046	69	220	5.0
Count	8	8	8	8	8	8	8	8
Std. Dev.	10	0.1	0.008	0.047	0.009	21	76	0.8
SE Mean	3	0.023	0.003	0.017	0.003	7	27	0.299
95 %ile	110	8.1	0.030	0.350	0.046	69	220	5.0

'Event'	DOS-21	pH-21	AMM-21	TN-21	TP-21	FC-21	ENT-21	CHLA-21
Minimum	76.3	8.0	0.005	0.200	0.017	10	10	1.9
1st Quartile	85.3	8.0	0.005	0.210	0.020	10	10	2.7
Mean	91	8.0	0.008	0.241	0.024	11	10	3.0
Median	90	8.0	0.010	0.220	0.023	10	10	2.9
3rd Quartile	96	8.1	0.010	0.260	0.026	10	10	3.3
Maximum	101	8.1	0.010	0.310	0.030	20	10	4.2
Count	8	8	8	8	8	8	8	8
Std. Dev.	8	0.1	0.003	0.039	0.005	4	0	0.7
SE Mean	3	0.018	0.001	0.014	0.002	1	0	0.233
95 %ile	101	8.1	0.010	0.310	0.030	20	10	4.2
'Event'	DOS-22	pH-22	AMM-22	TN-22	TP-22	FC-22	ENT-22	CHLA-22
Minimum	13.9	6.8	0.005	0.750	0.106	170	40	3.8
1st Quartile	24.2	7.1	0.005	0.880	0.124	240	160	4.6
Mean	36	7.1	0.132	1.443	0.197	19930	1314	12.2
Median	35	7.1	0.010	1.120	0.132	18500	635	9.2
3rd Quartile	43	7.2	0.090	1.605	0.197	37000	875	11.0
Maximum	71	7.4	1.110	3.565	0.610	49500	10000	37.4
Count	14	13	15	15	15	15	15	14
Std. Dev.	15	0.2	0.302	0.842	0.132	19265	2591	10.7
SE Mean	4	0.043	0.078	0.217	0.034	4974	669	2.849
95 %ile								

'After' results

'After'	DOS-1	pH-1	AMM-1	TN-1	TP-1	FC-1	ENT-1	CHLA-1
Minimum	20.2	6.7	0.010	0.480	0.033	30	2	0.2
1st Quartile	47.7	7.0	0.040	0.580	0.047	240	50	5.8
Mean	58.1	7.3	0.105	0.782	0.070	1806	414	10.3
Median	57.5	7.2	0.060	0.720	0.060	450	110	7.3
3rd Quartile	68.3	7.4	0.100	0.970	0.084	740	230	14.2
Maximum	109.0	9.1	0.560	1.450	0.147	21000	6000	55.4
Count	27	27	27	27	27	27	27	27
Std. Dev.	19.2	0.5	0.139	0.283	0.031	4556	1154	10.2
SE Mean	3.687	0.088	0.027	0.055	0.006	877	222	1.958
95 %ile	83.7	7.8	0.550	1.410	0.139	13000	1400	19.5

'After'	DOS-2	pH-2	AMM-2	TN-2	TP-2	FC-2	ENT-2	CHLA-2
Minimum	56.6	6.8	0.010	0.270	0.014	9	9	0.3
1st Quartile	75.3	7.1	0.010	0.360	0.025	20	10	3.3
Mean	85.9	7.4	0.029	0.511	0.038	336	98	10.6
Median	81.5	7.3	0.010	0.430	0.033	59	20	9.6
3rd Quartile	90.0	7.5	0.040	0.610	0.045	160	59	13.7
Maximum	150.0	8.9	0.110	1.100	0.095	4600	980	30.9
Count	28	28	28	28	28	28	28	28
Std. Dev.	18.1	0.5	0.028	0.235	0.019	884	197	8.7
SE Mean	3.424	0.086	0.005	0.044	0.004	167	37	1.640
95 %ile	112.0	8.2	0.100	1.060	0.074	1200	330	28.9
'After'	**DOS-3**	**pH-3**	**AMM-3**	**TN-3**	**TP-3**	**FC-3**	**ENT-3**	**CHLA-3**
Minimum	29.5	6.5	0.010	0.410	0.024	18	9	1.7
1st Quartile	59.4	7.0	0.050	0.660	0.051	40	36	3.4
Mean	66.9	7.2	0.118	0.811	0.076	2954	1291	9.6
Median	68.9	7.2	0.120	0.790	0.072	180	59	6.3
3rd Quartile	74.3	7.4	0.160	0.860	0.087	430	160	13.4
Maximum	91.4	7.6	0.340	1.420	0.199	48000	25000	37.0
Count	28	28	28	28	28	28	28	28
Std. Dev.	12.9	0.2	0.076	0.224	0.036	9502	4829	9.0
SE Mean	2.439	0.047	0.014	0.042	0.007	1796	913	1.698
95 %ile	86.9	7.5	0.260	1.300	0.140	18000	7000	28.1
'After'	**DOS-4**	**pH-4**	**AMM-4**	**TN-4**	**TP-4**	**FC-4**	**ENT-4**	**CHLA-4**
Minimum	14.6	6.8	0.010	0.470	0.031	45	9	2.1
1st Quartile	46.5	7.0	0.030	0.570	0.048	220	50	4.1
Mean	57.0	7.2	0.114	0.805	0.072	2034	472	9.8
Median	57.0	7.1	0.050	0.680	0.059	550	110	6.3
3rd Quartile	64.6	7.3	0.090	0.840	0.076	800	170	10.5
Maximum	98.4	9.3	0.810	1.990	0.179	29000	8000	38.8
Count	28	28	28	28	28	28	28	26
Std. Dev.	17.6	0.5	0.180	0.349	0.037	5478	1496	9.4
SE Mean	3.329	0.089	0.034	0.066	0.007	1035	283	1.835
95 %ile	80.7	7.6	0.630	1.450	0.154	6000	990	36.8
'After'	**DOS-7**	**pH-7**	**AMM-7**	**TN-7**	**TP-7**	**FC-7**	**ENT-7**	**CHLA-7**
Minimum	67.0	6.4	0.010	0.240	0.010	9	9	3.9
1st Quartile	84.5	7.3	0.010	0.330	0.016	20	10	16.6
Mean	95.8	7.6	0.010	0.398	0.024	78	120	39.2
Median	93.7	7.4	0.010	0.390	0.024	36	36	29.3
3rd Quartile	106.0	8.0	0.010	0.460	0.029	91	130	49.5
Maximum	126.0	9.1	0.010	0.560	0.045	500	670	123.6
Count	28	28	28	28	28	28	28	26
Std. Dev.	16.4	0.6	0.000	0.089	0.009	102	172	33.5
SE Mean	3.093	0.118	0.000	0.017	0.002	19	33	6.577
95 %ile	126.0	8.8	0.010	0.550	0.042	210	500	106.7

'After'	DOS-8	pH-8	AMM-8	TN-8	TP-8	FC-8	ENT-8	CHLA-8
Minimum	42.6	6.9	0.010	0.360	0.015	9	9	2.8
1st Quartile	68.5	7.1	0.010	0.410	0.019	30	10	3.9
Mean	76.9	7.2	0.042	0.475	0.026	147	39	10.4
Median	73.5	7.2	0.040	0.440	0.022	64	20	7.8
3rd Quartile	83.1	7.4	0.050	0.500	0.033	130	40	14.4
Maximum	107.0	7.7	0.180	0.750	0.042	840	140	43.4
Count	28	28	28	28	28	28	28	27
Std. Dev.	15.8	0.2	0.034	0.088	0.008	206	39	8.9
SE Mean	2.991	0.040	0.006	0.017	0.002	39	7	1.718
95 %ile	106.0	7.6	0.090	0.620	0.040	630	140	25.2
'After'	DOS-9	pH-9	AMM-9	TN-9	TP-9	FC-9	ENT-9	CHLA-9
Minimum	66.5	6.9	0.010	0.380	0.014	9	10	4.4
1st Quartile	82.1	7.1	0.010	0.470	0.042	50	30	16.2
Mean	89.9	7.3	0.029	0.598	0.054	255	98	29.3
Median	91.0	7.3	0.010	0.560	0.051	110	69	22.3
3rd Quartile	95.1	7.5	0.030	0.640	0.063	400	130	30.3
Maximum	120.0	7.8	0.120	1.440	0.141	980	320	188.7
Count	28	28	28	28	28	28	28	28
Std. Dev.	10.9	0.2	0.030	0.197	0.023	276	86	33.2
SE Mean	2.069	0.044	0.006	0.037	0.004	52	16	6.273
95 %ile	104.0	7.6	0.090	0.830	0.078	920	280	54.9
'After'	DOS-10	pH-10	AMM-10	TN-10	TP-10	FC-10	ENT-10	CHLA-10
Minimum	69.2	6.9	0.010	0.380	0.019	18	9	8.1
1st Quartile	78.0	7.2	0.010	0.470	0.054	40	10	19.6
Mean	88.0	7.4	0.026	0.562	0.063	164	70	25.9
Median	86.4	7.4	0.010	0.560	0.063	73	59	23.6
3rd Quartile	96.8	7.6	0.020	0.610	0.078	170	91	27.4
Maximum	111.0	8.1	0.140	0.830	0.119	880	310	56.7
Count	28	28	28	28	28	28	28	28
Std. Dev.	11.2	0.3	0.034	0.103	0.021	201	68	11.2
SE Mean	2.125	0.049	0.006	0.019	0.004	38	13	2.119
95 %ile	105.0	7.8	0.130	0.690	0.089	550	160	53.4
'After'	DOS-12	pH-12	AMM-12	TN-12	TP-12	FC-12	ENT-12	CHLA-12
Minimum	65.2	7.0	0.010	0.370	0.020	9	9	6.7
1st Quartile	88.1	7.3	0.010	0.420	0.050	36	10	14.0
Mean	98.1	7.6	0.019	0.540	0.062	300	53	24.1
Median	98.5	7.6	0.010	0.550	0.065	59	18	21.0
3rd Quartile	107.0	7.8	0.010	0.600	0.074	200	45	29.8
Maximum	132.0	8.0	0.110	0.810	0.114	4600	640	49.9
Count	28	28	28	28	28	28	28	28
Std. Dev.	14.9	0.3	0.025	0.117	0.021	866	119	11.0
SE Mean	2.822	0.053	0.005	0.022	0.004	164	22	2.083
95 %ile	121.0	8.0	0.090	0.740	0.091	900	110	48.7

'After'	DOS-13	pH-13	AMM-13	TN-13	TP-13	FC-13	ENT-13	CHLA-13
Minimum	83.3	7.2	0.010	0.360	0.029	9	9	0.8
1st Quartile	91.4	7.5	0.010	0.480	0.047	36	10	16.3
Mean	102.7	7.7	0.016	0.580	0.073	269	98	25.7
Median	100.0	7.7	0.010	0.520	0.066	91	10	26.6
3rd Quartile	111.0	7.9	0.010	0.620	0.077	320	30	30.8
Maximum	135.0	8.5	0.100	1.370	0.233	1900	1200	46.6
Count	28	28	28	28	28	28	28	28
Std. Dev.	12.0	0.3	0.021	0.182	0.040	414	242	11.4
SE Mean	2.277	0.057	0.004	0.034	0.008	78	46	2.154
95 %ile	120.0	8.2	0.080	0.810	0.149	1000	500	46.5
'After'	DOS-14	pH-14	AMM-14	TN-14	TP-14	FC-14	ENT-14	CHLA-14
Minimum	78.1	7.2	0.010	0.350	0.025	9	9	1.1
1st Quartile	93.9	7.4	0.010	0.450	0.053	59	10	15.5
Mean	105.9	7.7	0.016	0.565	0.070	3451	446	25.6
Median	105.0	7.7	0.010	0.570	0.067	160	30	21.5
3rd Quartile	118.0	7.9	0.010	0.610	0.078	570	110	33.5
Maximum	134.0	8.6	0.090	0.960	0.161	60000	9900	51.4
Count	28	28	28	28	28	28	28	28
Std. Dev.	14.8	0.3	0.020	0.134	0.030	12259	1859	13.1
SE Mean	2.796	0.064	0.004	0.025	0.006	2317	351	2.478
95 %ile	130.0	8.3	0.080	0.780	0.121	28000	790	50.0
'After'	DOS-15	pH-15	AMM-15	TN-15	TP-15	FC-15	ENT-15	CHLA-15
Minimum	51.2	7.0	0.010	0.380	0.027	9	9	1.6
1st Quartile	72.2	7.2	0.010	0.420	0.036	18	10	5.2
Mean	77.3	7.4	0.041	0.523	0.054	102	111	10.8
Median	80.2	7.4	0.030	0.490	0.047	55	18	10.2
3rd Quartile	82.9	7.5	0.040	0.520	0.060	110	73	13.7
Maximum	89.1	7.8	0.180	0.980	0.124	400	1500	33.8
Count	28	28	28	28	28	28	28	28
Std. Dev.	9.3	0.2	0.041	0.150	0.024	122	287	7.1
SE Mean	1.758	0.035	0.008	0.028	0.004	23	54	1.343
95 %ile	88.3	7.6	0.130	0.970	0.102	400	390	24.4
'After'	DOS-22	pH-22	AMM-22	TN-22	TP-22	FC-22	ENT-22	CHLA-22
Minimum	9.8	6.5	0.010	0.600	0.039	27	20	0.4
1st Quartile	30.8	6.9	0.070	0.690	0.054	160	59	4.3
Mean	45.2	7.1	0.150	0.910	0.089	3840	2716	11.9
Median	43.8	7.1	0.150	0.780	0.084	280	120	6.3
3rd Quartile	56.8	7.2	0.200	0.930	0.093	1200	250	18.2
Maximum	78.1	7.4	0.350	1.690	0.221	56000	62000	43.4
Count	28	28	28	28	28	28	28	28
Std. Dev.	18.0	0.2	0.078	0.291	0.045	11248	11715	10.7
SE Mean	3.401	0.043	0.015	0.055	0.008	2126	2214	2.030
95 %ile	76.6	7.4	0.300	1.570	0.184	21000	8000	34.4

9.4 Appendix D Hypothesis testing for the sewage overflow at Glenfield in November 2013

The GLM Procedure, Brown and Forsythe's Test for Homogeneity of Variance, Least Squares Means, Adjustment for Multiple comparisons: Tukey-Kramer

Charts with fit diagnostics and distribution (box and whisker plots) are available on request

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Class Level Information		
Class	Levels	Values
Period	3	1_Before 2_Event 3_After

Dependent Variable: TN

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	2	1.27965158	0.63982579	21.04	<.0001
Error	155	4.71429272	0.03041479		
Corrected Total	157	5.99394430			

R-Square	Coeff Var	Root MSE	TN Mean
0.213491	39.20737	0.174398	0.444810

Source	DF	Type I SS	Mean Square	F Value	Pr > F
Period	2	1.27965158	0.63982579	21.04	<.0001
Source	DF	Type III SS	Mean Square	F Value	Pr > F
Period	2	1.27965158	0.63982579	21.04	<.0001

Dependent Variable: TP

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	2	0.02482382	0.01241191	28.92	<.0001
Error	155	0.06651765	0.00042915		
Corrected Total	157	0.09134147			

R-Square	Coeff Var	Root MSE	TP Mean
0.271769	58.33369	0.020716	0.035513

Source	DF	Type I SS	Mean Square	F Value	Pr > F
Period	2	0.02482382	0.01241191	28.92	<.0001
Source	DF	Type III SS	Mean Square	F Value	Pr > F
Period	2	0.02482382	0.01241191	28.92	<.0001

Dependent Variable: LTN

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	2	0.91445771	0.45722885	18.79	<.0001
Error	155	3.77151070	0.02433233		
Corrected Total	157	4.68596841			

R-Square	Coeff Var	Root MSE	LTN Mean
0.195148	-40.23558	0.155988	-0.387687

Source	DF	Type I SS	Mean Square	F Value	Pr > F
Period	2	0.91445771	0.45722885	18.79	<.0001
Source	DF	Type III SS	Mean Square	F Value	Pr > F
Period	2	0.91445771	0.45722885	18.79	<.0001

Dependent Variable: LTP

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	2	2.30031942	1.15015971	23.14	<.0001
Error	155	7.70409113	0.04970381		
Corrected Total	157	10.00441056			

R-Square	Coeff Var	Root MSE	LTP Mean
0.229931	-14.59664	0.222944	-1.527362

Source	DF	Type I SS	Mean Square	F Value	Pr > F
Period	2	2.30031942	1.15015971	23.14	<.0001
Source	DF	Type III SS	Mean Square	F Value	Pr > F
Period	2	2.30031942	1.15015971	23.14	<.0001

Brown and Forsythe's Test for Homogeneity of TN Variance ANOVA of Absolute Deviations from Group Medians					
Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Period	2	0.0309	0.0154	0.94	0.3925
Error	155	2.5438	0.0164		

Brown and Forsythe's Test for Homogeneity of TP Variance ANOVA of Absolute Deviations from Group Medians					
Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Period	2	0.000662	0.000331	1.15	0.3189
Error	155	0.0446	0.000288		

Brown and Forsythe's Test for Homogeneity of LTN Variance ANOVA of Absolute Deviations from Group Medians					
Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Period	2	0.0200	0.0100	1.06	0.3487
Error	155	1.4619	0.00943		

Brown and Forsythe's Test for Homogeneity of LTP Variance ANOVA of Absolute Deviations from Group Medians					
Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Period	2	0.0538	0.0269	1.22	0.2973
Error	155	3.4134	0.0220		

Period	TN LSMEAN	LSMEAN Number
1_Before	0.40578512	1
2_Event	0.72923077	2
3_After	0.48750000	3

Least Squares Means for effect Period Pr > |t| for H0: LSMean(i)=LSMean(j) Dependent Variable: TN			
i/j	1	2	3
1		<.0001	0.0938
2	<.0001		0.0003
3	0.0938	0.0003	

Least Squares Means for Effect Period				
i	j	Difference Between Means	Simultaneous 95% Confidence Limits for LSMean(i)-LSMean(j)	
1	2	-0.323446	-0.443901	-0.202990
1	3	-0.081715	-0.173935	0.010505
2	3	0.241731	0.099608	0.383853

Period	TP LSMEAN	LSMEAN Number
1_Before	0.03118182	1
2_Event	0.07715385	2
3_After	0.03479167	3

Least Squares Means for effect Period Pr > |t| for H0: LSMean(i)=LSMean(j) Dependent Variable: TP			
i/j	1	2	3
1		<.0001	0.7160
2	<.0001		<.0001
3	0.7160	<.0001	

Least Squares Means for Effect Period				
i	j	Difference Between Means	Simultaneous 95% Confidence Limits for LSMean(i)-LSMean(j)	
1	2	-0.045972	-0.060280	-0.031664
1	3	-0.003610	-0.014564	0.007344
2	3	0.042362	0.025480	0.059244

Period	LTN LSMEAN	LSMEAN Number
1_Before	-0.42121079	1
2_Event	-0.14900604	2
3_After	-0.34795838	3

Least Squares Means for effect Period Pr > |t| for H0: LSMean(i)=LSMean(j) Dependent Variable: LTN			
i/j	1	2	3
1		<.0001	0.0928
2	<.0001		0.0009
3	0.0928	0.0009	

Least Squares Means for Effect Period				
i	j	Difference Between Means	Simultaneous 95% Confidence Limits for LSMean(i)-LSMean(j)	
1	2	-0.272205	-0.379945	-0.164465
1	3	-0.073252	-0.155737	0.009232
2	3	0.198952	0.071833	0.326072

Period	LTP LSMEAN	LSMEAN Number
1_Before	-1.57514662	1
2_Event	-1.13464377	2
3_After	-1.49917158	3

Least Squares Means for effect Period Pr > |t| for H0: LSMean(i)=LSMean(j) Dependent Variable: LTP			
i/j	1	2	3
1		<.0001	0.2820
2	<.0001		<.0001
3	0.2820	<.0001	

Least Squares Means for Effect Period				
i	j	Difference Between Means	Simultaneous 95% Confidence Limits for LSMean(i)-LSMean(j)	
1	2	-0.440503	-0.594488	-0.286518
1	3	-0.075975	-0.193865	0.041915
2	3	0.364528	0.182845	0.546211

Dependent Variable: FC

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	2	227381494334	113690747167	7.32	0.0009
Error	159	2.4684106E12	15524595046		
Corrected Total	161	2.6957921E12			

R-Square	Coeff Var	Root MSE	FC Mean
0.084347	953.4647	124597.7	13067.89

Source	DF	Type I SS	Mean Square	F Value	Pr > F
Period	2	227381494334	113690747167	7.32	0.0009
Source	DF	Type III SS	Mean Square	F Value	Pr > F
Period	2	227381494334	113690747167	7.32	0.0009

Dependent Variable: LFC

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	2	42.0287903	21.0143952	40.69	<.0001
Error	159	82.1234621	0.5164998		
Corrected Total	161	124.1522524			

R-Square	Coeff Var	Root MSE	LFC Mean
0.338526	36.83475	0.718679	1.951090

Source	DF	Type I SS	Mean Square	F Value	Pr > F
Period	2	42.02879032	21.01439516	40.69	<.0001
Source	DF	Type III SS	Mean Square	F Value	Pr > F
Period	2	42.02879032	21.01439516	40.69	<.0001

Period	FC LSMEAN	LSMEAN Number
1_Before	256.628	1
2_Event	122483.529	2
3_After	155.271	3

Least Squares Means for effect Period
Pr > |t| for H0: LSMean(i)=LSMean(j)

Dependent Variable: FC

i/j	1	2	3
1		0.0006	1.0000
2	0.0006		0.0065
3	1.0000	0.0065	

Least Squares Means for Effect Period

i	j	Difference Between Means	Simultaneous 95% Confidence Limits for LSMean(i)-LSMean(j)	
1	2	-122227	-198579	-45874
1	3	101.357266	-65768	65971
2	3	122328	28882	215775

Period	LFC LSMEAN	LSMEAN Number
1_Before	1.78210391	1
2_Event	3.43827958	2
3_After	1.74963494	3

Least Squares Means for effect Period
Pr > |t| for H0: LSMean(i)=LSMean(j)

Dependent Variable: LFC

i/j	1	2	3
1		<.0001	0.9777
2	<.0001		<.0001
3	0.9777	<.0001	

Least Squares Means for Effect Period				
i	j	Difference Between Means	Simultaneous 95% Confidence Limits for LSMean(i)-LSMean(j)	
1	2	-1.656176	-2.096576	-1.215775
1	3	0.032469	-0.347468	0.412406
2	3	1.688645	1.149646	2.227643

Dependent Variable: ENT

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	2	1254041657	627020828	5.40	0.0054
Error	158	18349598624	116136700		
Corrected Total	160	19603640281			

R-Square	Coeff Var	Root MSE	ENT Mean
0.063970	1017.660	10776.67	1058.966

Source	DF	Type I SS	Mean Square	F Value	Pr > F
Period	2	1254041657	627020828	5.40	0.0054
Source	DF	Type III SS	Mean Square	F Value	Pr > F
Period	2	1254041657	627020828	5.40	0.0054

Dependent Variable: LENT

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	2	9.11594129	4.55797064	10.88	<.0001
Error	158	66.21072674	0.41905523		
Corrected Total	160	75.32666803			

R-Square	Coeff Var	Root MSE	LENT Mean
0.121019	40.83849	0.647345	1.585134

Source	DF	Type I SS	Mean Square	F Value	Pr > F
Period	2	9.11594129	4.55797064	10.88	<.0001
Source	DF	Type III SS	Mean Square	F Value	Pr > F
Period	2	9.11594129	4.55797064	10.88	<.0001

Period	ENT LSMEAN	LSMEAN Number
1_Before	101.82645	1
2_Event	9181.64706	2
3_After	90.63043	3

Least Squares Means for effect Period Pr > |t| for H0: LSMean(i)=LSMean(j) Dependent Variable: ENT			
i/j	1	2	3
1		0.0040	1.0000
2	0.0040		0.0248
3	1.0000	0.0248	

Least Squares Means for Effect Period				
i	j	Difference Between Means	Simultaneous 95% Confidence Limits for LSMean(i)-LSMean(j)	
1	2	-9079.820613	-15684	-2475.567575
1	3	11.196011	-5788.776699	5811.168722
2	3	9091.016624	935.660615	17246

Period	LENT LSMEAN	LSMEAN Number
1_Before	1.52117328	1
2_Event	2.26947473	2
3_After	1.41580598	3

Least Squares Means for effect Period Pr > |t| for H0: LSMean(i)=LSMean(j) Dependent Variable: LENT			
i/j	1	2	3
1		<.0001	0.7547
2	<.0001		0.0002
3	0.7547	0.0002	

Least Squares Means for Effect Period				
i	j	Difference Between Means	Simultaneous 95% Confidence Limits for LSMean(i)-LSMean(j)	
1	2	-0.748301	-1.145013	-0.351590
1	3	0.105367	-0.243032	0.453766
2	3	0.853669	0.363784	1.343553

Brown and Forsythe's Test for Homogeneity of FC Variance ANOVA of Absolute Deviations from Group Medians					
Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Period	2	2.255E11	1.127E11	7.27	0.0010
Error	159	2.467E12	1.552E10		

Brown and Forsythe's Test for Homogeneity of LFC Variance ANOVA of Absolute Deviations from Group Medians					
Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Period	2	0.9173	0.4587	1.77	0.1736
Error	159	41.1983	0.2591		

Brown and Forsythe's Test for Homogeneity of ENT Variance ANOVA of Absolute Deviations from Group Medians					
Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Period	2	1.2434E9	6.2171E8	5.36	0.0056
Error	158	1.832E10	1.1598E8		

Brown and Forsythe's Test for Homogeneity of LENT Variance ANOVA of Absolute Deviations from Group Medians					
Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Period	2	1.4257	0.7128	3.76	0.0253
Error	158	29.9303	0.1894		

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Class Level Information		
Class	Levels	Values
Period	3	1_Before 2_Event 3_After

Dependent Variable: TN

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	2	37.49064261	18.74532130	347.15	<.0001
Error	153	8.26159329	0.05399734		
Corrected Total	155	45.75223590			

R-Square	Coeff Var	Root MSE	TN Mean
0.819428	38.15814	0.232373	0.608974

Source	DF	Type I SS	Mean Square	F Value	Pr > F
Period	2	37.49064261	18.74532130	347.15	<.0001
Source	DF	Type III SS	Mean Square	F Value	Pr > F
Period	2	37.49064261	18.74532130	347.15	<.0001

Dependent Variable: TP

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	2	0.16632328	0.08316164	192.00	<.0001
Error	153	0.06627031	0.00043314		
Corrected Total	155	0.23259359			

R-Square	Coeff Var	Root MSE	TP Mean
0.715081	59.55011	0.020812	0.034949

Source	DF	Type I SS	Mean Square	F Value	Pr > F
Period	2	0.16632328	0.08316164	192.00	<.0001
Source	DF	Type III SS	Mean Square	F Value	Pr > F
Period	2	0.16632328	0.08316164	192.00	<.0001

Dependent Variable: LTN

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	2	5.81216083	2.90608042	202.16	<.0001
Error	153	2.19938755	0.01437508		
Corrected Total	155	8.01154838			

R-Square	Coeff Var	Root MSE	LTN Mean
0.725473	-40.20566	0.119896	-0.298207

Source	DF	Type I SS	Mean Square	F Value	Pr > F
Period	2	5.81216083	2.90608042	202.16	<.0001
Source	DF	Type III SS	Mean Square	F Value	Pr > F
Period	2	5.81216083	2.90608042	202.16	<.0001

Dependent Variable: LTP

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	2	7.22747139	3.61373570	94.72	<.0001
Error	153	5.83691925	0.03814980		
Corrected Total	155	13.06439064			

R-Square	Coeff Var	Root MSE	LTP Mean
0.553219	-12.33442	0.195320	-1.583534

Source	DF	Type I SS	Mean Square	F Value	Pr > F
Period	2	7.22747139	3.61373570	94.72	<.0001
Source	DF	Type III SS	Mean Square	F Value	Pr > F
Period	2	7.22747139	3.61373570	94.72	<.0001

Brown and Forsythe's Test for Homogeneity of TN Variance ANOVA of Absolute Deviations from Group Medians					
Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Period	2	2.8326	1.4163	76.72	<.0001
Error	153	2.8246	0.0185		

Brown and Forsythe's Test for Homogeneity of TP Variance ANOVA of Absolute Deviations from Group Medians					
Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Period	2	0.0274	0.0137	135.24	<.0001
Error	153	0.0155	0.000101		

Brown and Forsythe's Test for Homogeneity of LTN Variance ANOVA of Absolute Deviations from Group Medians					
Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Period	2	0.0434	0.0217	3.73	0.0262
Error	153	0.8898	0.00582		

Brown and Forsythe's Test for Homogeneity of LTP Variance ANOVA of Absolute Deviations from Group Medians					
Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Period	2	0.0538	0.0269	2.00	0.1388
Error	153	2.0573	0.0134		

Period	TN LSMEAN	LSMEAN Number
1_Before	0.45093220	1
2_Event	2.17000000	2
3_After	0.47541667	3

Least Squares Means for effect Period Pr > |t| for H0: LSMean(i)=LSMean(j) Dependent Variable: TN			
i/j	1	2	3
1		<.0001	0.8852
2	<.0001		<.0001
3	0.8852	<.0001	

Least Squares Means for Effect Period				
i	j	Difference Between Means	Simultaneous 95% Confidence Limits for LSMean(i)-LSMean(j)	
1	2	-1.719068	-1.874528	-1.563607
1	3	-0.024484	-0.147635	0.098666
2	3	1.694583	1.509631	1.879536

Period	TP LSMEAN	LSMEAN Number
1_Before	0.02447458	1
2_Event	0.13892857	2
3_After	0.02579167	3

Least Squares Means for effect Period Pr > |t| for H0: LSMean(i)=LSMean(j) Dependent Variable: TP			
i/j	1	2	3
1		<.0001	0.9569
2	<.0001		<.0001
3	0.9569	<.0001	

Least Squares Means for Effect Period				
i	j	Difference Between Means	Simultaneous 95% Confidence Limits for LSMean(i)-LSMean(j)	
1	2	-0.114454	-0.128377	-0.100531
1	3	-0.001317	-0.012347	0.009713
2	3	0.113137	0.096572	0.129702

Period	LTN LSMEAN	LSMEAN Number
1_Before	-0.36455417	1
2_Event	0.31525477	2
3_After	-0.32985364	3

Least Squares Means for effect Period Pr > |t| for H0: LSMean(i)=LSMean(j) Dependent Variable: LTN			
i/j	1	2	3
1		<.0001	0.4016
2	<.0001		<.0001
3	0.4016	<.0001	

Least Squares Means for Effect Period				
i	j	Difference Between Means	Simultaneous 95% Confidence Limits for LSMean(i)-LSMean(j)	
1	2	-0.679809	-0.760021	-0.599597
1	3	-0.034701	-0.098242	0.028840
2	3	0.645108	0.549680	0.740537

Period	LTP LSMEAN	LSMEAN Number
1_Before	-1.65941780	1
2_Event	-0.90044576	2
3_After	-1.60890818	3

Least Squares Means for effect Period Pr > |t| for H0: LSMean(i)=LSMean(j) Dependent Variable: LTP			
i/j	1	2	3
1		<.0001	0.4820
2	<.0001		<.0001
3	0.4820	<.0001	

Least Squares Means for Effect Period				
i	j	Difference Between Means	Simultaneous 95% Confidence Limits for LSMean(i)-LSMean(j)	
1	2	-0.758972	-0.889643	-0.628301
1	3	-0.050510	-0.154023	0.053003
2	3	0.708462	0.553002	0.863923

Dependent Variable: FC

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	2	241284093	120642046	20.42	<.0001
Error	154	909947757	5908752		
Corrected Total	156	1151231850			

R-Square	Coeff Var	Root MSE	FC Mean
0.209588	386.3635	2430.792	629.1465

Source	DF	Type I SS	Mean Square	F Value	Pr > F
Period	2	241284092.6	120642046.3	20.42	<.0001
Source	DF	Type III SS	Mean Square	F Value	Pr > F
Period	2	241284092.6	120642046.3	20.42	<.0001

Dependent Variable: ENT

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	2	254659.18	127329.59	1.97	0.1431
Error	154	9957602.91	64659.76		
Corrected Total	156	10212262.09			

R-Square	Coeff Var	Root MSE	ENT Mean
0.024937	226.2021	254.2828	112.4140

Source	DF	Type I SS	Mean Square	F Value	Pr > F
Period	2	254659.1835	127329.5918	1.97	0.1431
Source	DF	Type III SS	Mean Square	F Value	Pr > F
Period	2	254659.1835	127329.5918	1.97	0.1431

Dependent Variable: LFC

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	2	9.08614721	4.54307361	20.17	<.0001
Error	154	34.68293443	0.22521386		
Corrected Total	156	43.76908164			

R-Square	Coeff Var	Root MSE	LFC Mean
0.207593	21.89042	0.474567	2.167921

Source	DF	Type I SS	Mean Square	F Value	Pr > F
Period	2	9.08614721	4.54307361	20.17	<.0001
Source	DF	Type III SS	Mean Square	F Value	Pr > F
Period	2	9.08614721	4.54307361	20.17	<.0001

Dependent Variable: LENT

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	2	4.59316378	2.29658189	12.36	<.0001
Error	154	28.62259250	0.18586099		
Corrected Total	156	33.21575629			

R-Square	Coeff Var	Root MSE	LENT Mean
0.138283	24.96885	0.431116	1.726616

Source	DF	Type I SS	Mean Square	F Value	Pr > F
Period	2	4.59316378	2.29658189	12.36	<.0001
Source	DF	Type III SS	Mean Square	F Value	Pr > F
Period	2	4.59316378	2.29658189	12.36	<.0001

Brown and Forsythe's Test for Homogeneity of FC Variance ANOVA of Absolute Deviations from Group Medians					
Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Period	2	2.306E8	1.153E8	20.72	<.0001
Error	154	8.5689E8	5564240		

Brown and Forsythe's Test for Homogeneity of ENT Variance ANOVA of Absolute Deviations from Group Medians					
Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Period	2	156069	78034.4	1.27	0.2837
Error	154	9459836	61427.5		

Brown and Forsythe's Test for Homogeneity of LFC Variance ANOVA of Absolute Deviations from Group Medians					
Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Period	2	2.5261	1.2631	15.00	<.0001
Error	154	12.9713	0.0842		

Brown and Forsythe's Test for Homogeneity of LENT Variance ANOVA of Absolute Deviations from Group Medians					
Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Period	2	0.1166	0.0583	0.68	0.5097
Error	154	13.2655	0.0861		

Period	FC LSMEAN	LSMEAN Number
1_Before	246.56780	1
2_Event	4441.20000	2
3_After	127.62500	3

Least Squares Means for effect Period Pr > |t| for H0: LSMean(i)=LSMean(j) Dependent Variable: FC			
i/j	1	2	3
1		<.0001	0.9740
2	<.0001		<.0001
3	0.9740	<.0001	

Least Squares Means for Effect Period				
i	j	Difference Between Means	Simultaneous 95% Confidence Limits for LSMean(i)-LSMean(j)	
1	2	-4194.632203	-5771.556473	-2617.707934
1	3	118.942797	-1169.215389	1407.100982
2	3	4313.575000	2420.129962	6207.020038

Period	ENT LSMEAN	LSMEAN Number
1_Before	135.381356	1
2_Event	55.866667	2
3_After	34.833333	3

Least Squares Means for effect Period Pr > |t| for H0: LSMean(i)=LSMean(j) Dependent Variable: ENT			
i/j	1	2	3
1		0.4906	0.1846
2	0.4906		0.9658
3	0.1846	0.9658	

Least Squares Means for Effect Period				
i	j	Difference Between Means	Simultaneous 95% Confidence Limits for LSMean(i)-LSMean(j)	
1	2	79.514689	-85.445828	244.475206
1	3	100.548023	-34.204955	235.301000
2	3	21.033333	-177.038114	219.104781

Period	LFC LSMEAN	LSMEAN Number
1_Before	2.15425802	1
2_Event	2.81681799	2
3_After	1.82953575	3

Least Squares Means for effect Period Pr > |t| for H0: LSMean(i)=LSMean(j) Dependent Variable: LFC			
i/j	1	2	3
1		<.0001	0.0074
2	<.0001		<.0001
3	0.0074	<.0001	

Least Squares Means for Effect Period				
i	j	Difference Between Means	Simultaneous 95% Confidence Limits for LSMean(i)-LSMean(j)	
1	2	-0.662560	-0.970425	-0.354695
1	3	0.324722	0.073233	0.576211
2	3	0.987282	0.617622	1.356942

Period	LENT LSMEAN	LSMEAN Number
1_Before	1.82340728	1
2_Event	1.51030358	2
3_After	1.38591765	3

Least Squares Means for effect Period Pr > |t| for H0: LSMean(i)=LSMean(j) Dependent Variable: LENT			
i/j	1	2	3
1		0.0240	<.0001
2	0.0240		0.6559
3	<.0001	0.6559	

Least Squares Means for Effect Period				
i	j	Difference Between Means	Simultaneous 95% Confidence Limits for LSMean(i)-LSMean(j)	
1	2	0.313104	0.033426	0.592781
1	3	0.437490	0.209027	0.665952
2	3	0.124386	-0.211428	0.460200

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Class Level Information		
Class	Levels	Values
Period	3	1_Before 2_Event 3_After

Dependent Variable: TN

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	2	5.52924779	2.76462389	32.78	<.0001
Error	155	13.07190158	0.08433485		
Corrected Total	157	18.60114937			

R-Square	Coeff Var	Root MSE	TN Mean
0.297253	44.34087	0.290405	0.654937

Source	DF	Type I SS	Mean Square	F Value	Pr > F
Period	2	5.52924779	2.76462389	32.78	<.0001
Source	DF	Type III SS	Mean Square	F Value	Pr > F
Period	2	5.52924779	2.76462389	32.78	<.0001

Dependent Variable: TP

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	2	0.03690536	0.01845268	16.32	<.0001
Error	155	0.17526216	0.00113072		
Corrected Total	157	0.21216752			

R-Square	Coeff Var	Root MSE	TP Mean
0.173944	60.91430	0.033626	0.055203

Source	DF	Type I SS	Mean Square	F Value	Pr > F
Period	2	0.03690536	0.01845268	16.32	<.0001
Source	DF	Type III SS	Mean Square	F Value	Pr > F
Period	2	0.03690536	0.01845268	16.32	<.0001

Dependent Variable: LTN

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	2	1.56758344	0.78379172	27.68	<.0001
Error	155	4.38878583	0.02831475		
Corrected Total	157	5.95636927			

R-Square	Coeff Var	Root MSE	LTN Mean
0.263178	-72.85341	0.168270	-0.230970

Source	DF	Type I SS	Mean Square	F Value	Pr > F
Period	2	1.56758344	0.78379172	27.68	<.0001
Source	DF	Type III SS	Mean Square	F Value	Pr > F
Period	2	1.56758344	0.78379172	27.68	<.0001

Dependent Variable: LTP

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	2	2.50616922	1.25308461	16.10	<.0001
Error	155	12.06315204	0.07782679		
Corrected Total	157	14.56932126			

R-Square	Coeff Var	Root MSE	LTP Mean
0.172017	-20.59750	0.278975	-1.354409

Source	DF	Type I SS	Mean Square	F Value	Pr > F
Period	2	2.50616922	1.25308461	16.10	<.0001
Source	DF	Type III SS	Mean Square	F Value	Pr > F
Period	2	2.50616922	1.25308461	16.10	<.0001

Dependent Variable: FC

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	2	20716.087	10358.043	0.37	0.6927
Error	155	4362346.097	28144.168		
Corrected Total	157	4383062.184			

R-Square	Coeff Var	Root MSE	FC Mean
0.004726	148.0889	167.7622	113.2848

Source	DF	Type I SS	Mean Square	F Value	Pr > F
Period	2	20716.08691	10358.04346	0.37	0.6927
Source	DF	Type III SS	Mean Square	F Value	Pr > F
Period	2	20716.08691	10358.04346	0.37	0.6927

Dependent Variable: ENT

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	2	950688.51	475344.25	3.33	0.0382
Error	155	22096112.09	142555.56		
Corrected Total	157	23046800.59			

R-Square	Coeff Var	Root MSE	ENT Mean
0.041250	354.6900	377.5653	106.4494

Source	DF	Type I SS	Mean Square	F Value	Pr > F
Period	2	950688.5088	475344.2544	3.33	0.0382
Source	DF	Type III SS	Mean Square	F Value	Pr > F
Period	2	950688.5088	475344.2544	3.33	0.0382

Dependent Variable: LFC

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	2	0.38301878	0.19150939	0.88	0.4163
Error	155	33.68195061	0.21730291		
Corrected Total	157	34.06496940			

R-Square	Coeff Var	Root MSE	LFC Mean
0.011244	26.16783	0.466158	1.781415

Source	DF	Type I SS	Mean Square	F Value	Pr > F
Period	2	0.38301878	0.19150939	0.88	0.4163
Source	DF	Type III SS	Mean Square	F Value	Pr > F
Period	2	0.38301878	0.19150939	0.88	0.4163

Dependent Variable: LENT

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	2	3.11900242	1.55950121	5.81	0.0037
Error	155	41.63977061	0.26864368		
Corrected Total	157	44.75877303			

R-Square	Coeff Var	Root MSE	LENT Mean
0.069685	34.41756	0.518308	1.505942

Source	DF	Type I SS	Mean Square	F Value	Pr > F
Period	2	3.11900242	1.55950121	5.81	0.0037
Source	DF	Type III SS	Mean Square	F Value	Pr > F
Period	2	3.11900242	1.55950121	5.81	0.0037

Brown and Forsythe's Test for Homogeneity of TN Variance ANOVA of Absolute Deviations from Group Medians					
Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Period	2	0.3628	0.1814	3.15	0.0456
Error	155	8.9243	0.0576		

Brown and Forsythe's Test for Homogeneity of TP Variance ANOVA of Absolute Deviations from Group Medians					
Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Period	2	0.00223	0.00112	1.56	0.2137
Error	155	0.1110	0.000716		

Brown and Forsythe's Test for Homogeneity of LTN Variance ANOVA of Absolute Deviations from Group Medians					
Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Period	2	0.2139	0.1069	8.67	0.0003
Error	155	1.9116	0.0123		

Brown and Forsythe's Test for Homogeneity of LTP Variance ANOVA of Absolute Deviations from Group Medians					
Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Period	2	0.5218	0.2609	7.88	0.0005
Error	155	5.1305	0.0331		

Brown and Forsythe's Test for Homogeneity of FC Variance ANOVA of Absolute Deviations from Group Medians					
Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Period	2	3160.1	1580.0	0.06	0.9374
Error	155	3786447	24428.7		

Brown and Forsythe's Test for Homogeneity of ENT Variance ANOVA of Absolute Deviations from Group Medians					
Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Period	2	876881	438440	3.14	0.0461
Error	155	21656259	139718		

Brown and Forsythe's Test for Homogeneity of LFC Variance ANOVA of Absolute Deviations from Group Medians					
Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Period	2	0.1898	0.0949	1.07	0.3445
Error	155	13.7095	0.0884		

Brown and Forsythe's Test for Homogeneity of LENT Variance ANOVA of Absolute Deviations from Group Medians					
Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Period	2	0.8248	0.4124	3.40	0.0360
Error	155	18.8140	0.1214		

Period	TN LSMEAN	LSMEAN Number
1_Before	0.61214876	1
2_Event	1.27307692	2
3_After	0.53583333	3

Least Squares Means for effect Period Pr > |t| for H0: LSMean(i)=LSMean(j) Dependent Variable: TN			
i/j	1	2	3
1		<.0001	0.4692
2	<.0001		<.0001
3	0.4692	<.0001	

Least Squares Means for Effect Period				
i	j	Difference Between Means	Simultaneous 95% Confidence Limits for LSMean(i)-LSMean(j)	
1	2	-0.660928	-0.861508	-0.460348
1	3	0.076315	-0.077247	0.229878
2	3	0.737244	0.500584	0.973903

Period	TP LSMEAN	LSMEAN Number
1_Before	0.04844628	1
2_Event	0.10307692	2
3_After	0.06333333	3

Least Squares Means for effect Period Pr > |t| for H0: LSMean(i)=LSMean(j) Dependent Variable: TP			
i/j	1	2	3
1		<.0001	0.1202
2	<.0001		0.0022
3	0.1202	0.0022	

Least Squares Means for Effect Period				
i	j	Difference Between Means	Simultaneous 95% Confidence Limits for LSMean(i)-LSMean(j)	
1	2	-0.054631	-0.077856	-0.031405
1	3	-0.014887	-0.032668	0.002894
2	3	0.039744	0.012341	0.067147

Period	LTN LSMEAN	LSMEAN Number
1_Before	-0.25676071	1
2_Event	0.10049009	2
3_After	-0.28048586	3

Least Squares Means for effect Period Pr > |t| for H0: LSMean(i)=LSMean(j) Dependent Variable: LTN			
i/j	1	2	3
1		<.0001	0.8033
2	<.0001		<.0001
3	0.8033	<.0001	

Least Squares Means for Effect Period				
i	j	Difference Between Means	Simultaneous 95% Confidence Limits for LSMean(i)-LSMean(j)	
1	2	-0.357251	-0.473473	-0.241028
1	3	0.023725	-0.065254	0.112704
2	3	0.380976	0.243848	0.518104

Period	LTP LSMEAN	LSMEAN Number
1_Before	-1.41720804	1
2_Event	-0.99615335	2
3_After	-1.23185513	3

Least Squares Means for effect Period Pr > |t| for H0: LSMean(i)=LSMean(j) Dependent Variable: LTP			
i/j	1	2	3
1		<.0001	0.0095
2	<.0001		0.0402
3	0.0095	0.0402	

Least Squares Means for Effect Period				
i	j	Difference Between Means	Simultaneous 95% Confidence Limits for LSMean(i)-LSMean(j)	
1	2	-0.421055	-0.613740	-0.228369
1	3	-0.185353	-0.332871	-0.037834
2	3	0.235702	0.008357	0.463046

Period	FC LSMEAN	LSMEAN Number
1_Before	109.082645	1
2_Event	151.076923	2
3_After	114.000000	3

Least Squares Means for effect Period Pr > |t| for H0: LSMean(i)=LSMean(j) Dependent Variable: FC			
i/j	1	2	3
1		0.6677	0.9906
2	0.6677		0.7973
3	0.9906	0.7973	

Least Squares Means for Effect Period				
i	j	Difference Between Means	Simultaneous 95% Confidence Limits for LSMean(i)-LSMean(j)	
1	2	-41.994278	-157.866332	73.877775
1	3	-4.917355	-93.628128	83.793417
2	3	37.076923	-99.637468	173.791314

Period	ENT LSMEAN	LSMEAN Number
1_Before	95.173554	1
2_Event	353.461538	2
3_After	29.500000	3

Least Squares Means for effect Period Pr > |t| for H0: LSMean(i)=LSMean(j) Dependent Variable: ENT			
i/j	1	2	3
1		0.0528	0.7168
2	0.0528		0.0365
3	0.7168	0.0365	

Least Squares Means for Effect Period				
i	j	Difference Between Means	Simultaneous 95% Confidence Limits for LSMean(i)-LSMean(j)	
1	2	-258.287985	-519.069364	2.493394
1	3	65.673554	-133.978708	265.325816
2	3	323.961538	16.272443	631.650634

Period	LFC LSMEAN	LSMEAN Number
1_Before	1.76468445	1
2_Event	1.94519070	2
3_After	1.77705009	3

Least Squares Means for effect Period Pr > |t| for H0: LSMean(i)=LSMean(j) Dependent Variable: LFC			
i/j	1	2	3
1		0.3826	0.9923
2	0.3826		0.5482
3	0.9923	0.5482	

Least Squares Means for Effect Period				
i	j	Difference Between Means	Simultaneous 95% Confidence Limits for LSMean(i)-LSMean(j)	
1	2	-0.180506	-0.502478	0.141465
1	3	-0.012366	-0.258865	0.234133
2	3	0.168141	-0.211745	0.548026

Period	LENT LSMEAN	LSMEAN Number
1_Before	1.50670320	1
2_Event	1.89792721	2
3_After	1.28977742	3

Least Squares Means for effect Period Pr > |t| for H0: LSMean(i)=LSMean(j) Dependent Variable: LENT			
i/j	1	2	3
1		0.0285	0.1500
2	0.0285		0.0024
3	0.1500	0.0024	

Least Squares Means for Effect Period				
i	j	Difference Between Means	Simultaneous 95% Confidence Limits for LSMean(i)-LSMean(j)	
1	2	-0.391224	-0.749216	-0.033232
1	3	0.216926	-0.057150	0.491001
2	3	0.608150	0.185765	1.030535

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Class Level Information		
Class	Levels	Values
Period	2	1_Before 2_Event

Dependent Variable: TN

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	1	0.15135750	0.15135750	2.20	0.1410
Error	124	8.54964250	0.06894873		
Corrected Total	125	8.70100000			

R-Square	Coeff Var	Root MSE	TN Mean
0.017395	56.67214	0.262581	0.463333

Source	DF	Type I SS	Mean Square	F Value	Pr > F
Period	1	0.15135750	0.15135750	2.20	0.1410
Source	DF	Type III SS	Mean Square	F Value	Pr > F
Period	1	0.15135750	0.15135750	2.20	0.1410

Dependent Variable: TP

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	1	0.00018566	0.00018566	0.44	0.5070
Error	124	0.05197870	0.00041918		
Corrected Total	125	0.05216436			

R-Square	Coeff Var	Root MSE	TP Mean
0.003559	41.56145	0.020474	0.049262

Source	DF	Type I SS	Mean Square	F Value	Pr > F
Period	1	0.00018566	0.00018566	0.44	0.5070
Source	DF	Type III SS	Mean Square	F Value	Pr > F
Period	1	0.00018566	0.00018566	0.44	0.5070

Dependent Variable: LTN

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	1	0.15851354	0.15851354	4.08	0.0457
Error	124	4.82313654	0.03889626		
Corrected Total	125	4.98165008			

R-Square	Coeff Var	Root MSE	LTN Mean
0.031819	-51.20542	0.197221	-0.385157

Source	DF	Type I SS	Mean Square	F Value	Pr > F
Period	1	0.15851354	0.15851354	4.08	0.0457
Source	DF	Type III SS	Mean Square	F Value	Pr > F
Period	1	0.15851354	0.15851354	4.08	0.0457

Dependent Variable: LTP

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	1	0.00495350	0.00495350	0.17	0.6814
Error	124	3.62748594	0.02925392		
Corrected Total	125	3.63243944			

R-Square	Coeff Var	Root MSE	LTP Mean
0.001364	-12.75328	0.171038	-1.341128

Source	DF	Type I SS	Mean Square	F Value	Pr > F
Period	1	0.00495350	0.00495350	0.17	0.6814
Source	DF	Type III SS	Mean Square	F Value	Pr > F
Period	1	0.00495350	0.00495350	0.17	0.6814

Dependent Variable: FC

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	1	159222.60	159222.60	0.27	0.6023
Error	124	72345242.33	583429.37		
Corrected Total	125	72504464.93			

R-Square	Coeff Var	Root MSE	FC Mean
0.002196	417.4453	763.8255	182.9762

Source	DF	Type I SS	Mean Square	F Value	Pr > F
Period	1	159222.6036	159222.6036	0.27	0.6023
Source	DF	Type III SS	Mean Square	F Value	Pr > F
Period	1	159222.6036	159222.6036	0.27	0.6023

Dependent Variable: ENT

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	1	45364.63	45364.63	0.20	0.6552
Error	124	28076024.30	226419.55		
Corrected Total	125	28121388.93			

R-Square	Coeff Var	Root MSE	ENT Mean
0.001613	524.6809	475.8356	90.69048

Source	DF	Type I SS	Mean Square	F Value	Pr > F
Period	1	45364.62857	45364.62857	0.20	0.6552
Source	DF	Type III SS	Mean Square	F Value	Pr > F
Period	1	45364.62857	45364.62857	0.20	0.6552

Dependent Variable: LFC

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	1	0.38223783	0.38223783	0.75	0.3867
Error	124	62.80363809	0.50648095		
Corrected Total	125	63.18587592			

R-Square	Coeff Var	Root MSE	LFC Mean
0.006049	49.32651	0.711675	1.442783

Source	DF	Type I SS	Mean Square	F Value	Pr > F
Period	1	0.38223783	0.38223783	0.75	0.3867
Source	DF	Type III SS	Mean Square	F Value	Pr > F
Period	1	0.38223783	0.38223783	0.75	0.3867

Dependent Variable: LENT

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	1	1.08193169	1.08193169	2.93	0.0892
Error	124	45.71844428	0.36869713		
Corrected Total	125	46.80037596			

R-Square	Coeff Var	Root MSE	LENT Mean
0.023118	49.10931	0.607204	1.236434

Source	DF	Type I SS	Mean Square	F Value	Pr > F
Period	1	1.08193169	1.08193169	2.93	0.0892
Source	DF	Type III SS	Mean Square	F Value	Pr > F
Period	1	1.08193169	1.08193169	2.93	0.0892

Brown and Forsythe's Test for Homogeneity of TN Variance ANOVA of Absolute Deviations from Group Medians					
Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Period	1	0.00360	0.00360	0.08	0.7834
Error	124	5.8719	0.0474		

Brown and Forsythe's Test for Homogeneity of TP Variance ANOVA of Absolute Deviations from Group Medians					
Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Period	1	0.000372	0.000372	1.84	0.1773
Error	124	0.0250	0.000202		

Brown and Forsythe's Test for Homogeneity of LTN Variance ANOVA of Absolute Deviations from Group Medians					
Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Period	1	0.0129	0.0129	0.77	0.3819
Error	124	2.0782	0.0168		

Brown and Forsythe's Test for Homogeneity of LTP Variance ANOVA of Absolute Deviations from Group Medians					
Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Period	1	0.0257	0.0257	2.51	0.1154
Error	124	1.2687	0.0102		

Brown and Forsythe's Test for Homogeneity of FC Variance ANOVA of Absolute Deviations from Group Medians					
Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Period	1	157795	157795	0.27	0.6024
Error	124	71740714	578554		

Brown and Forsythe's Test for Homogeneity of ENT Variance ANOVA of Absolute Deviations from Group Medians					
Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Period	1	43750.0	43750.0	0.19	0.6599
Error	124	27877906	224822		

Brown and Forsythe's Test for Homogeneity of LFC Variance ANOVA of Absolute Deviations from Group Medians					
Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Period	1	0.5335	0.5335	2.45	0.1203
Error	124	27.0309	0.2180		

Brown and Forsythe's Test for Homogeneity of LENT Variance ANOVA of Absolute Deviations from Group Medians					
Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Period	1	1.0289	1.0289	6.56	0.0116
Error	124	19.4360	0.1567		

Site20 BA testing dry weather

Class Level Information		
Class	Levels	Values
Period	2	1_Before 2_Event

Dependent Variable: TN

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	1	0.01966730	0.01966730	1.92	0.1683
Error	124	1.27000333	0.01024196		
Corrected Total	125	1.28967063			

R-Square	Coeff Var	Root MSE	TN Mean
0.015250	44.82083	0.101203	0.225794

Source	DF	Type I SS	Mean Square	F Value	Pr > F
Period	1	0.01966730	0.01966730	1.92	0.1683
Source	DF	Type III SS	Mean Square	F Value	Pr > F
Period	1	0.01966730	0.01966730	1.92	0.1683

Dependent Variable: TP

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	1	0.00014766	0.00014766	0.64	0.4236
Error	124	0.02841200	0.00022913		
Corrected Total	125	0.02855966			

R-Square	Coeff Var	Root MSE	TP Mean
0.005170	55.39538	0.015137	0.027325

Source	DF	Type I SS	Mean Square	F Value	Pr > F
Period	1	0.00014766	0.00014766	0.64	0.4236
Source	DF	Type III SS	Mean Square	F Value	Pr > F
Period	1	0.00014766	0.00014766	0.64	0.4236

Dependent Variable: LTN

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	1	0.08611330	0.08611330	4.11	0.0447
Error	124	2.59704694	0.02094393		
Corrected Total	125	2.68316025			

R-Square	Coeff Var	Root MSE	LTN Mean
0.032094	-21.46985	0.144720	-0.674062

Source	DF	Type I SS	Mean Square	F Value	Pr > F
Period	1	0.08611330	0.08611330	4.11	0.0447
Source	DF	Type III SS	Mean Square	F Value	Pr > F
Period	1	0.08611330	0.08611330	4.11	0.0447

Dependent Variable: LTP

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	1	0.04905782	0.04905782	1.92	0.1682
Error	124	3.16592544	0.02553166		
Corrected Total	125	3.21498326			

R-Square	Coeff Var	Root MSE	LTP Mean
0.015259	-9.997282	0.159786	-1.598297

Source	DF	Type I SS	Mean Square	F Value	Pr > F
Period	1	0.04905782	0.04905782	1.92	0.1682
Source	DF	Type III SS	Mean Square	F Value	Pr > F
Period	1	0.04905782	0.04905782	1.92	0.1682

Dependent Variable: FC

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	1	29.14325	29.14325	0.05	0.8218
Error	124	70954.32500	572.21230		
Corrected Total	125	70983.46825			

R-Square	Coeff Var	Root MSE	FC Mean
0.000411	176.9842	23.92096	13.51587

Source	DF	Type I SS	Mean Square	F Value	Pr > F
Period	1	29.14325397	29.14325397	0.05	0.8218
Source	DF	Type III SS	Mean Square	F Value	Pr > F
Period	1	29.14325397	29.14325397	0.05	0.8218

Dependent Variable: ENT

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	1	656.26825	656.26825	1.05	0.3065
Error	124	77173.20000	622.36452		
Corrected Total	125	77829.46825			

R-Square	Coeff Var	Root MSE	ENT Mean
0.008432	234.7537	24.94723	10.62698

Source	DF	Type I SS	Mean Square	F Value	Pr > F
Period	1	656.2682540	656.2682540	1.05	0.3065
Source	DF	Type III SS	Mean Square	F Value	Pr > F
Period	1	656.2682540	656.2682540	1.05	0.3065

Dependent Variable: LFC

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	1	0.08123920	0.08123920	0.34	0.5589
Error	124	29.32795059	0.23651573		
Corrected Total	125	29.40918979			

R-Square	Coeff Var	Root MSE	LFC Mean
0.002762	57.73004	0.486329	0.842419

Source	DF	Type I SS	Mean Square	F Value	Pr > F
Period	1	0.08123920	0.08123920	0.34	0.5589
Source	DF	Type III SS	Mean Square	F Value	Pr > F
Period	1	0.08123920	0.08123920	0.34	0.5589

Dependent Variable: LENT

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	1	0.35941719	0.35941719	1.71	0.1932
Error	124	26.04067901	0.21000548		
Corrected Total	125	26.40009620			

R-Square	Coeff Var	Root MSE	LENT Mean
0.013614	61.62266	0.458264	0.743661

Source	DF	Type I SS	Mean Square	F Value	Pr > F
Period	1	0.35941719	0.35941719	1.71	0.1932
Source	DF	Type III SS	Mean Square	F Value	Pr > F
Period	1	0.35941719	0.35941719	1.71	0.1932

Brown and Forsythe's Test for Homogeneity of TN Variance ANOVA of Absolute Deviations from Group Medians					
Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Period	1	0.00137	0.00137	0.19	0.6658
Error	124	0.9080	0.00732		

Brown and Forsythe's Test for Homogeneity of TP Variance ANOVA of Absolute Deviations from Group Medians					
Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Period	1	1.016E-7	1.016E-7	0.00	0.9806
Error	124	0.0211	0.000171		

Brown and Forsythe's Test for Homogeneity of LTN Variance ANOVA of Absolute Deviations from Group Medians					
Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Period	1	0.0109	0.0109	1.12	0.2912
Error	124	1.2020	0.00969		

Brown and Forsythe's Test for Homogeneity of LTP Variance ANOVA of Absolute Deviations from Group Medians					
Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Period	1	0.000969	0.000969	0.09	0.7597
Error	124	1.2784	0.0103		

Brown and Forsythe's Test for Homogeneity of FC Variance ANOVA of Absolute Deviations from Group Medians					
Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Period	1	1.7813	1.7813	0.00	0.9534
Error	124	64318.3	518.7		

Brown and Forsythe's Test for Homogeneity of ENT Variance ANOVA of Absolute Deviations from Group Medians					
Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Period	1	307.3	307.3	0.52	0.4736
Error	124	73756.8	594.8		

Brown and Forsythe's Test for Homogeneity of LFC Variance ANOVA of Absolute Deviations from Group Medians					
Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Period	1	0.2306	0.2306	2.28	0.1334
Error	124	12.5285	0.1010		

Brown and Forsythe's Test for Homogeneity of LENT Variance ANOVA of Absolute Deviations from Group Medians					
Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Period	1	0.1259	0.1259	1.31	0.2543
Error	124	11.8979	0.0960		