National Research
FLAGSHIPS
Water for a Healthy Country

CSIRO

Preliminary quantitative risk assessment for the Salisbury stormwater ASTR Project

Declan Page, Karen Barry, Paul Pavelic, Peter Dillon and Antoine Chassagne

Water for a Healthy Country Flagship Report August 2008

www.csiro.au

Water for a Healthy Country Flagship Report Series

ISSN: 1835-095X

Enquiries should be addressed to:
Declan Page
Water for a Healthy Country | CSIRO Land and Water
Private Bag 2
Glen Osmond SA 5064

Australia is founding its future on science and innovation. Its national science agency, CSIRO, is a powerhouse of ideas, technologies and skills.

CSIRO initiated the National Research Flagships to address Australia's major research challenges and opportunities. They apply large scale, long term, multidisciplinary science and aim for widespread adoption of solutions. The Flagship Collaboration Fund supports the best and brightest researchers to address these complex challenges through partnerships between CSIRO, universities, research agencies and industry.

The Water for a Healthy Country Flagship aims to achieve a tenfold increase in the economic, social and environmental benefits from water by 2025. The work contained in this report is collaboration between CSIRO and City of Salisbury, United Water, SA Water Corporation and the SA Department of Water, Land and Biodiversity Conservation (DWLBC).

For more information about Water for a Healthy Country Flagship or the National Research Flagship Initiative visit www.csiro.au/org/HealthyCountry.html

Citation: Page, D, Barry, K, Pavelic, P, Dillon, P and Chassagne, A 2008. Preliminary quantitative risk assessment for the Salisbury stormwater ASTR project. CSIRO: Water for a Healthy Country National Research Flagship.

Cover Photograph:

Description: Exit of the cleansing reed bed at the Parafield stormwater harvesting system wetland.
Photographer: Declan Page

ACKNOWLEDGEMENTS

The Salisbury Aquifer Storage Transfer and Recovery (ASTR) Project is supported by the Australian Government Department of Innovation, Industry, Science and Research through its International Science Linkages Programme, enabling participation within the European Union Project 'Reclaim Water', which is supported in the 6th Framework Programme (Contract 018309). The ASTR project is also supported by the South Australian Premiers Science and Research Foundation and the Australian Government National Water Commission through the Water Smart Australia Programme commitment to Water Proofing Northern Adelaide Project. Project partners include CSIRO Land and Water, City of Salisbury, United Water, SA Water Corporation and the SA Department of Water, Land and Biodiversity Conservation (DWLBC).

Members of the ASTR HACCP committee provided valuable input into the production of this revised HACCP plan. Membership of the committee included:

- Declan Page (CSIRO, Chair of committee)
- Rudi Regel (United Water)
- Peter Dillon (CSIRO)
- Peter Newland, David Duncan (SA EPA)
- David Cunliffe (SA DHCS)
- Mark Purdie, Melissa Tscharke (City of Salisbury)
- Greg Ingleton, Jacqueline Frizenschaf (SA Water).

Members of the ASTR project team provided guidance and assistance throughout the preparation of this report, including Paul Pavelic (CSIRO Land and Water) and Stephanie Rinck-Pfeiffer (United Water International Pty Ltd).

Mark Purdie of the City of Salisbury, provided data on water quality and system operation for the Parafield wetlands stormwater harvesting system.

We are also grateful for the comments of Jochen Mueller, EnTox, Mark Labaz, Manager Pollution Avoidance, Adelaide and Mount Lofty Ranges Natural Resources Management Board, Peter Scott, SA EPA and Joanne Vanderzalm and Hartmut Holländer, CSIRO Land and Water.

EXECUTIVE SUMMARY

This report extends the preliminary qualitative risk assessment prepared by Swierc *et al.* (2005) for the Salisbury Aquifer Storage Transfer Recovery (ASTR) project by incorporating the results of water quality monitoring undertaken in 2006. Overall the quality of the stormwater entering the holding basin/cleansing reed bed pre-treatment system was similar to that reported in the literature, but below average rainfall during this period may not be representative of normal operating conditions. In addition to the traditional water quality monitoring, sediment analysis and passive samplers for assessment of micropollutants were trialled as tools for risk assessment where concentrations of hazards are below standard laboratory detection limits. While no micropollutants or enteric pathogens were detected using the conventional water or sediment monitoring programs, the passive samples detected trace quantities of micropollutants throughout the system of which diuron, simazine and chlorpyrifos appeared to be most significant.

The results of the monitoring program were used to construct a water and chloride balance for the holding storage and cleansing reed bed components of the ASTR system as a step to quantitatively assess their treatment performance for physical, chemical and microbial hazards. The performance of the system over August-September 2006 was found to have removal efficiencies of 0.66, 0.77 and 0.64 for total nitrogen, phosphorous and organic carbon respectively. The water produced after the cleansing reed bed was of near-potable quality, with the exceptions of colour caused by elevated iron concentrations and small numbers of faecal indicator bacteria caused by indigenous fauna in the cleansing reed bed.

In addition to the treatment performance, a quantitative risk assessment (QRA) for several index pathogens (rotavirus, *Campylobacter* and *Cryptosporidium*) and chemicals (diuron, simazine, and chlorpyrifos) was also performed. The QRA incorporated a Monte Carlo simulation to describe the behaviour of these hazards in the ASTR system - from stormwater harvesting through to the water user. For the micropollutants, final estimated concentrations were compared to the appropriate water quality guidelines. However, for the pathogens, the exposure estimate was then used as an input to a dose-response model to estimate the risk in terms of public health burden using the disability adjusted life years (DALYs) metric.

The QRA indicated that the risks posed by the index microbial and chemical hazards were currently unacceptable for use of the recovered water for drinking. These initial QRA results stem from the uncertainty of subsurface treatment component. If the recovered water is to be intended for drinking, the treatment provided by the aquifer must be quantified and validated to be incorporated into the multi-barrier approach of a risk management plan.

Recommendations for the risk assessment and management of the system focus on specific actions for partners with milestones that are detailed in Section 7. This includes monitoring and acquiring information to refine the risk assessment, involving the surface catchment, the stormwater capture and treatment system and aquifer processes. This leads to recommendations concerning communication about the risks and other steps required before recovered water could be introduced into mains supplies.

CONTENTS

LIST OF FIGURES

LIST OF TABLES

1. INTRODUCTION

This report continues to build upon the qualitative risk assessment by Swierc *et al.* (2005) of the Aquifer Storage Transfer Recovery (ASTR) project located at Salisbury in South Australia. Specifically, this report moves the risk assessment from the qualitative to the quantitative domain by assessing the results of the 2006 water quality monitoring program. Analysis of these results also gives a rationale for further risk-based water quality monitoring and research to support the implementation of the evolving Hazard Analysis and Critical Control Point (HACCP) plan for the ASTR project.

Although HACCP plans were first introduced over 40 years ago, their development for use with drinking water quality management has only recently been promoted by the World Health Organisation (WHO 2004) and Australian Drinking Water Guidelines (NHMRC 2004). This has resulted in several Australian water utilities incorporating HACCP into drinking water and wastewater quality management systems, some even gaining third party certification. Further to this, Dillon (2005) suggested that HACCP would be suitable to manage water quality in ASTR type applications to manage the variability and uncertainty surrounding perceived water quality risks both to human health and the environment. While previously HACCP has been applied systematically but in a largely qualitative manner, more recently, in order to have a conceptual model of the system, elements of quantitative risk analysis (QRA) have been incorporated (Medema *et al.* 1996; Notermans *et al.* 1996; Notermans and Teunis 1996).

It has been proposed that the application of QRA in the HACCP context should be considered as a next step in providing a quantitative framework for the ASTR system. This involves the stepwise analysis of the risk of occurrence of a hazard to understand its nature and facilitate appropriate control measures (Notermans *et al.* 1996). Simply put, a risk management framework consists of several components that can be grouped into: (i) risk assessment, (ii) risk management and (iii) risk communication (FAO/WHO 1995; Notermans *et al.* 1996). This report focuses on (i) the risk assessment component, which flows naturally into providing recommendations for (ii) risk management, and provides the content upon which subsequent risk communication at technical and public levels will rely.

In moving from a qualitative to quantitative approach, it is important to note that QRA models differ from the previous qualitative risk assessment (Swierc *et al.* 2005). The previous approach's aim was to reduce hazards without data to develop a numerical understanding. However, the QRA approach described in this report differs in that it incorporates stochastic mathematical models intended to describe the behaviour of hazards and is directly informed by the water quality monitoring program. The subsequent exposure estimates can then be used as an input to a dose-response model to estimate the public health burden in terms of disability adjusted life years (DALYs).

A similar approach can be used to determine risks to the environment, but is simplified for regulatory purposes to a comparison with water quality guideline values (NRMMC/EPHC 2005). Hence, by quantifying the risks associated with ASTR in the HACCP framework from 'catchment-to-tap', QRA is able to provide a conceptual framework to describe the process by which hazards impact on the endpoint of interest.

Finally, a sensitivity analysis of the QRA model allows for positioning of additional Critical and Quality Control Points and scenario testing for optimal risk reduction. Thereby when QRA is embedded within the HACCP framework it facilitates identification of appropriate future risk management strategies, including where to implement control measures to reduce risk or focus research to reduce consequential uncertainties.

This report discusses the results of the 2006 monitoring program and investigates the initial application of QRA to the HACCP system for the Salisbury ASTR project and covers stormwater harvesting, cleansing reedbed treatment, subsurface storage to drinking water supply, including drinking and other uses of potable water (Swierc *et al.* 2005). This is used to further evolve the implementation of HACCP system, with the aim to ensure product safety and proactively meet evolving regulator requirements.

2. REVIEW OF THE ASTR SYSTEM

Swierc *et al.* (2005) previously identified hazards in a qualitative risk assessment framework which could be controlled by means of critical control points (CCPs) and quality control points (QCPs) that exist at various stages throughout the water production system (Figure 1).

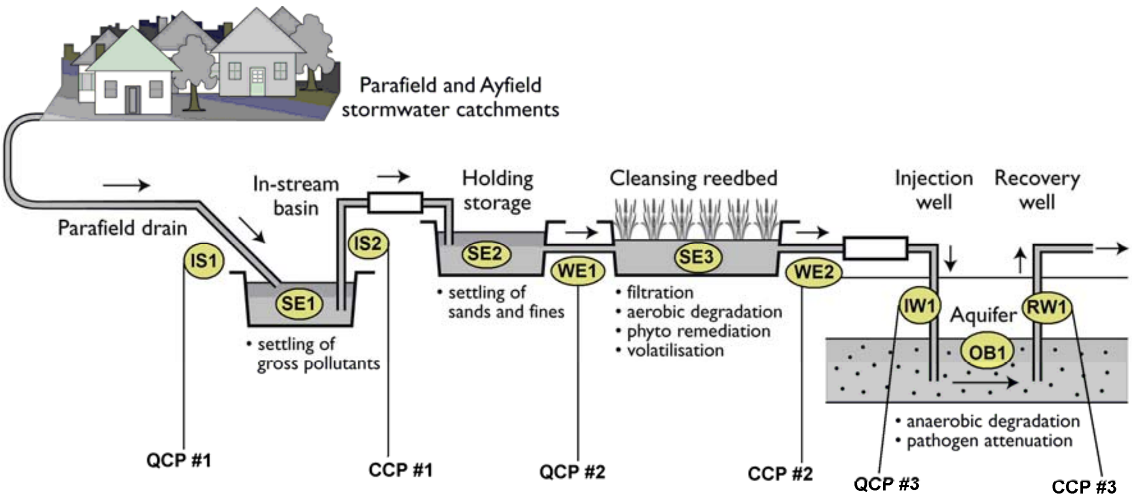

Figure 1 Conceptual of the ASTR system, showing CCPs, QCPs, water and sediment quality sampling points

Figure 1 shows the water and sediment quality monitoring points (yellow circles) which were adopted as part of the 2006 monitoring program in relation to the CCPs and QCPs. The CCPs discussed in this report correspond to the transfer point between the in-stream basin and the holding storage (sampling point IS2) and the outlet of the cleansing reed bed (sampling point WE2). These sampling points were located at the inflow and outflow of each basin with the aim to be able to quantify treatment performance of the in-stream basin, holding storage or cleansing reed bed in terms of hazard removal efficiencies. Similarly, sediment quality was monitored in each of the basins to assess if there was potential for accumulation of hazards. At the time of writing, the injection and recovery wells for the subsurface component were yet to be drilled and information on this, together with further source water quality and harvesting system treatment process evaluations, will be covered in the third report of this series.

The CCPs adopted are relatively easy to define for the operational parameters currently used to manage the system such as pH, conductivity and turbidity. QCPs were adopted to assess the treatment performance of the basins and reedbed. However, a key point in the application of HACCP to ASTR is that many of the potential hazards are difficult (if not impossible) to monitor in real-time and this provides the motivation to prevent the occurrences of hazards through use of supporting programs (SPs). Each of these preventative measures, CCPs, QCPs and SPs, are reviewed in the following sections.

2.1 Review of Critical Control Points

Of primary concern in evolving the HACCP plan is the continual refinement and review of CCPs used to manage the system. The 2006 year was remarkable in that very low volumes of stormwater were captured by the Parafield stormwater harvesting system due to the record drought/low rainfall received by Adelaide (and indeed most of Australia) during this period. 2006 was an especially dry year (260 mm) with the ASTR catchment receiving just over half of the rainfall of previous year's average (1889-2006: 456 mm) measured at the nearby Parafield airport weather station (ID number: 023013). Figure 2 shows the monthly volumes of stormwater captured by the Parafield stormwater harvesting system since its commissioning in February 2003 to December 2006.

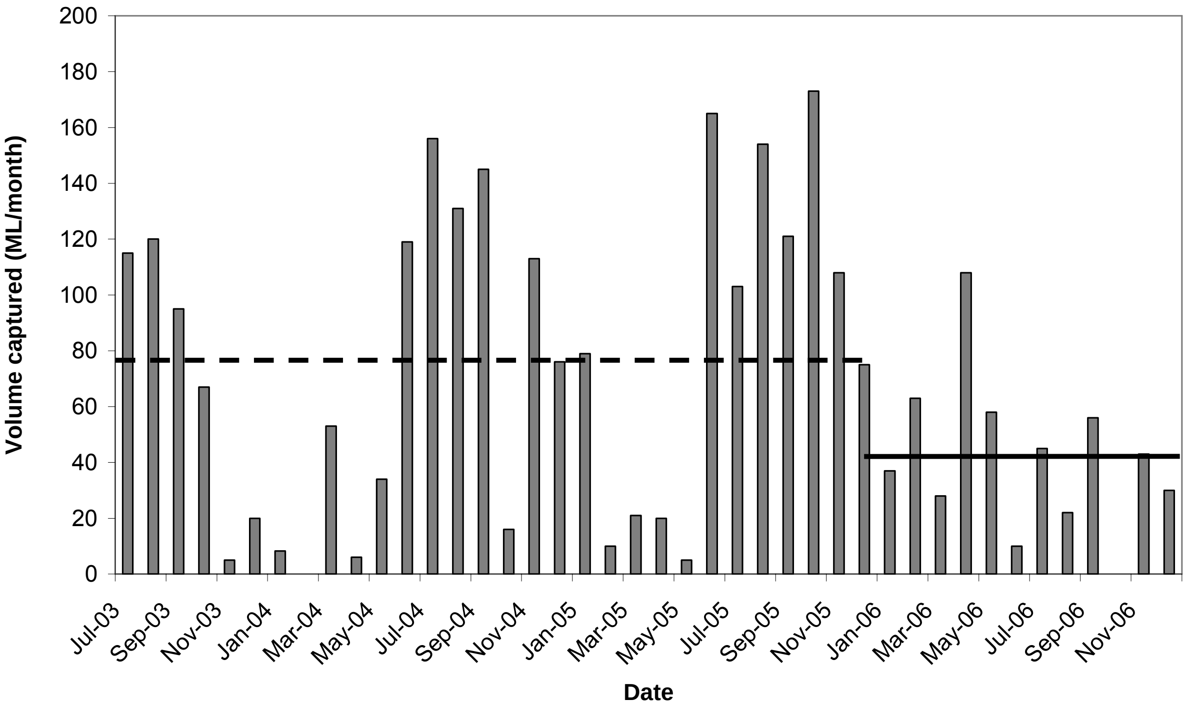

Figure 2 Monthly volumes of stormwater harvested by the Parafield stormwater harvesting facility (dashed line represents average flow (77ML/month) 2003-2005, solid line (42ML/month) 2006)

Figure 2 illustrates that that the poor rainfall affected the ability of the system to capture stormwater with an average of 77 ML/month (2003-2005: dashed line) dropping to 42 ML/month (2006: solid line). As a consequence the ASTR system was operated in a modified way with the objective of maximising stormwater captured to ensure sufficient water to flush the aquifer. These modified operations resulted in the in-stream basin being effectively bypassed as raw stormwater was pumped directly from the intake structure to the holding storage, thereby reducing treatment functions of the in-stream basin (settling of gross pollutants). The lack of this treatment caused blockages in the water quality monitoring station at CCP #1 (transfer point to the holding storage) thereby rendering it ineffective. Interestingly, there was no observed impact on downstream product water quality at the outlet of the reed bed (CCP #2). Nevertheless, for CCPs to be used to manage the system they must be reliable, which will lead to a recommendation that the monitoring station (CCP#1) automatically shuts down pumping if the turbidity concentration exceeds its critical limit of 100 NTU.

The conductivity and turbidity trace of CCP #2 at the exit of the cleansing reed bed, polled at 60 minute intervals, is shown in Figure 3. This trace represents unfiltered data during the winter period.

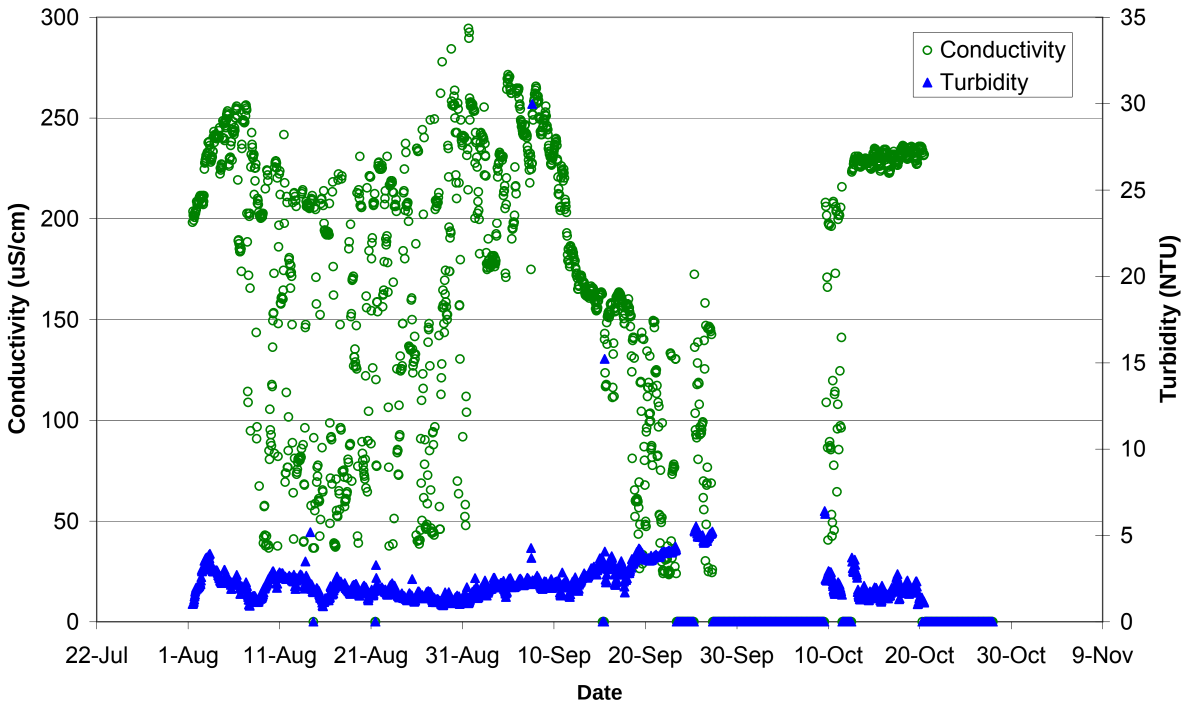

Figure 3 WE2, cleansing reedbed outlet (CCP #2) trace (August to October 2006)

The figure highlights the difficulty of using real-time systems to manage water quality; instability was observed in the conductivity data and both conductivity and turbidity data went off-line in September 2006. It is shown later in Section 4.2, during the treatment performance assessment of the cleansing reed bed, that the outflows were constant during the period shown in Figure 3 and as such, it is unlikely that the conductivity meter readings were reliable. To resolve this for the 2007 monitoring year a second redundant conductivity probe was temporarily deployed near the wetland outlet.

In 2006, CCPs #1 and #2 were largely ineffective in managing the quality of water produced by the stormwater harvesting system. CCP # 1 was blocked and inoperative for the majority of the season while CCP #2's efficacy was hampered by instability of the signal, probably due to either a faulty probe or insufficient maintenance. Hence it is recommended that in addition to a more frequent and auditable maintenance campaign, when the newer SCADA system is installed, CCPs #1 and #2 have their respective Critical Limits (CLs) reviewed. CCP# 1 is an important barrier for minimizing poor quality raw stormwater entering the stormwater harvesting system, while CCP #2 is critical in limiting poor quality water being injected into the subsurface. Additional installation and refinement of CCP #3 for the water recovered from the ASTR well-field is required when the subsurface component of the ASTR system becomes active.

2.2 Review of Quality Control Points

The quality control point (QCP #1) located at the Parafield drain, (upstream early warning water quality monitoring point) was not utilised as the inlet structure and pump become blocked during periods of high flow. QCP #1 provides little additional information to CCP #1 as there is little hydraulic residence time (< 5 mins) between them but is still important in assessing the treatment performance of the in-stream basin. Similarly, QCP #2 is valuable for assessing the treatment performance of the cleansing reed bed and QCP #3 for assessing the water quality being injected into the subsurface.

2.3 Review of Supporting Programs

There are seven identified supporting programs (SPs) operating at ASTR which are further discussed in this report:

1. Street sweeping in the urban catchment
2. Stormwater pollution prevention program: Be Stormwater Smart
3. SA EPA industry licensing program and codes of practice
4. Pesticide application programs in the Parafield stormwater catchment
5. ASTR site security
6. Sewer leak detection program
7. Equipment maintenance

Supporting programs are sometimes difficult to quantify in terms of their impact on water quality in a risk management sense. As a result of the ASTR HACCP committee meeting, each of the stakeholders was contacted to discuss SPs that operate within the Parafield catchment, with an aim to developing metrics and quantifying hazard reduction or risk mitigation with respect to source water quality. A summary of the supporting programs, their impacts in reducing hazards and hazardous events are given in Table 1.

Table 1 Summary of supporting programs impact on hazards and hazardous events

	Supporting program	Actions	Impact on hazards or hazardous events
1	Street sweeping in the urban catchment	Removal of gross pollutants from streets in catchment	None quantified, though may lead to improvements in source water quality
2	Stormwater pollution prevention program: Be Stormwater Smart	Employee awareness of stormwater quality in SMEs	None quantified, though improved industry awareness of stormwater may improve source water quality
3	SA EPA industry licensing program and codes of practice	Best management practices for specific industries and stormwater management	None quantified, though industry stormwater management may be improved
4	Pesticide application programs in the Parafield stormwater catchment	Best management practice, infrequent use of herbicides other than glyphosate	None quantified, though pesticide and herbicide management may be improved through adherence to EPA best practice codes.
5	ASTR site security	Improved fencing and signage	None quantified, though a reduction in unauthorised access may lead to reduced incidences of vandalism and associated hazardous events
6	Sewer leak detection program	Repair of sewer chokes	None quantified, though reduction in sewer chokes may lead to improved source water quality
7	Equipment maintenance	Calibration and servicing of on-line monitoring equipment	None quantified, though reduction in equipment malfunction will lead to increased confidence in CCPs.

More specific comments on each of the supporting programs are given in the following sub sections.

2.3.1 Street sweeping programs in the urban catchment

Street cleansing is a common (and expensive) practice undertaken by many urban municipalities. Street sweeping, essentially involving the operation of large trucks for cleaning street surfaces, is primarily performed for aesthetic purposes (Walker and Wong 1999). It is often perceived to lead to improvements in stormwater quality by preventing hazards deposited on street surfaces from reaching the stormwater system. There is, however, little available evidence to quantify the extent to which street sweeping can improve stormwater quality. A review by Walker and Wong (1999) investigated the effectiveness of street sweeping for stormwater quality improvement. The research literature on street cleaning indicated a lack of studies that address the issues of the effectiveness of street sweeping for sediment and associated hazard removal.

Experimental studies found street sweeping to be highly effective in the removal of large solids >2 mm under test conditions. However, field conditions are expected to significantly reduce the efficiencies because of limitations with sweeper access to source areas (mainly due to street design and car parking), sweeping mechanisms used and operator skills. Field studies undertaken by the Cooperative Research Centre for Catchment Hydrology in Australia found significant stormwater pollutant loads generated from source areas in spite of a daily street sweeping regime (Walker and Wong 1999). They concluded that quantities of gross pollutants entering the stormwater system depend more on the available energy in rainfall to re-mobilise and transport deposited hazards on street surfaces than the quantity of hazards deposited on street surfaces.

These studies indicated that street sweeping is relatively ineffective at reducing the street surface load of fine particles < 125 µm (Walker and Wong 1999). The particle size distribution of suspended solids conveyed in stormwater in Australian conditions typically range from 1 µm - 400 µm with approximately 70 % of the particles < 125 µm (Walker and Wong 1999). By comparison, at the ASTR site, 90% of particles in the water column at the in-stream basin sampling point (ISB1) were < 60 µm indicating a very fine sediment load. Therefore, street sweeping as it is currently practiced cannot be expected to be effective in the reduction of suspended solids and associated trace metals, pathogens and nutrient concentrations in stormwater.

Walker and Wong (1999) concluded that the performance of street sweeping for stormwater quality improvement is limited and must be accompanied by structural pollutant treatment measures to effectively reduce the discharge of gross and sediment associated pollutants in stormwater. The incremental benefits in increasing the frequency of street sweeping beyond what is required to meet street aesthetic criterion is expected to be small in relation to water quality improvements. As a result, there seems to be little benefit in conducting further in-depth field-based studies into the effectiveness of street sweeping for stormwater pollution control in the Parafield catchment.

2.3.2 Stormwater pollution prevention program: *Be Stormwater Smart*

The Adelaide and Mount Lofty Ranges Natural Resources Management Board fund a stormwater pollution prevention project (*Be Stormwater Smart*) that aims to reduce the level of stormwater pollution emanating from industry and businesses within a catchment area. It is important to note that no stormwater quality monitoring occurs as part of this program; instead environmental site reviews are conducted for small to medium sized enterprises (SMEs) to raise awareness, identify areas of concern and suggest potential stormwater management improvements.

In the Salisbury area, the program began in 1999 and initially identified ~2,000 businesses that should be targeted (note this number refers to all of the City of Salisbury, not just the Parafield urban stormwater catchment, numbers in the ASTR stormwater catchment remain undefined but are a subset of this total number).

During environmental site reviews, businesses were assigned an environmental performance category based upon their environmental site review report (% score). Of the businesses visited to date, 45% were automotive shops, 26% manufacturing industry, 3% transport related, 4% wholesalers and 22% other. A number of general metrics were used to assess the efficacy of the program in the absence of stormwater quality monitoring. These metrics are based on employee awareness, knowing what stormwater was, where it goes and knowing how to prevent stormwater pollution, e.g. use of bunding, spill kits. A summary of these metrics is given in Table 2.

Table 2 Summary of *Be Stormwater Smart* program metrics

Metric	2001	2002	2003	2004	2005	2006
Number of 1st visits	1	1	38	162	60	19
Number of businesses 100% compliant	0	0	0	8	2	2
% of businesses 100% compliant	0	0	0	5	3	11

Very few sites were found to score 100% compliance on their initial environmental site review. Encouragingly this fraction appears to be increasing with > 10% of all business; 100% compliant in 2006, up from 0% in 2003. This indicates an increasing understanding of the value of stormwater and awareness of how it can be managed at a local level. A second strategic review of the program is currently being conducted by the Adelaide and Mount Lofty Ranges Natural Resources Management Board in light of recent shifts in organisational structure and focus; as such the exact future of the program remains uncertain. However repeat visits to businesses could be helpful in evaluating the impact of the program. The proportion of businesses in the Parafield catchment that have been visited is unknown. If improvements are observed on repeat visits the program warrants extension to all potentially polluting industries.

2.3.3 SA EPA industry licensing program and codes of practice

The South Australian Environmental Protection Authority (SA EPA) licence specific larger businesses (e.g. light manufacturing) within the Parafield catchment. In the Parafield catchment none of the businesses were viewed as being of high risk (Scott 2007 pers. comm.). This is consistent with the landuse risk assessment performed by Swierc *et al.* (2005) and the stormwater pollution prevention program which found the majority of businesses were automotive workshops.

In addition to the industry licensing program, the SA EPA produces a number of codes of practice, including one for pesticide usage and stormwater management best practice (SA EPA 2005; 2006). For example, automotive shops are a dominant business type in the ASTR catchment; the stormwater management guidelines for automotive shops focuses on employee awareness, regular cleaning of the shop floor, use of bunding, use of biodegradable detergents and enhancing connections from wash areas to the sewerage network as opposed to the stormwater system.

SA EPA Guidelines for responsible pesticide use is similarly generic in nature and focuses on the legislative and general best practice such as following the instructions on the product label, applying the product only to the target area and considering the potential for contamination of the soil and nearby waters. They aim to minimise usage of pesticides/herbicides to target areas and reduce off target impacts from their application.

2.3.4 Pesticide application programs in the Parafield stormwater catchment

Although not technically an SP, a review was undertaken of the major pesticide and herbicide application programs located in the Parafield stormwater catchment to obtain an improved understanding of the types, frequencies and loads of herbicides and pesticides used. All stakeholders stated that they followed the instructions on the product labels and followed the SA EPA codes for responsible pesticide use. There were four major users of herbicides identified within the catchment:

1. **City of Salisbury:** The City of Salisbury uses a number of contractors in management of council parks and vegetation. Contractors primarily use herbicides containing Max™ or Duo™ with the active ingredient glyphosate, a systemic herbicide that translocates to the roots of plants. The manufacturer claims that the product is completely biodegradable, breaking down into natural products in both soil and water. Stated half-life is 60 days in the soil, a time frame described as rapid by manufacturer. The current Australian Drinking Water Guideline for glyphosate is 1 mg/L.

Other herbicides used include Oust and Dicamba. Oust™ contains the active constituent metsulfuron-methyl. Metsulfuron-methyl is a residual sulfonylurea compound, used as a selective pre- and post emergence herbicide for broadleaf weeds and some annual grasses. It is a systemic compound with foliar and soil activity. Metsulfuron methyl is practically non-toxic to fish and aquatic invertebrates. Metsulfuron methyl does not build up (bioaccumulate) in fish. Because metsulfuron-methyl is soluble in water, there is a potential for transport to the stormwater system. Manufacturer tests show that the half-life for metsulfuron-methyl in water, when exposed to sunlight, ranges from 1 to 8 days. The current Australian Drinking Water Guideline value is 0.03 mg/L.

Dicamba is a benzoic acid herbicide. It can be applied to the leaves or to the soil. Dicamba does not bind to soil particles and is highly soluble in water. Its leaching potential increases with precipitation and the volume applied. The current Australian Drinking Water Guideline is 0.1 mg/L.

Pesticides used include Striker™ and Biflex™. Striker™ contains the active ingredient deltamethrin, a pyrethroid insecticide that kills insects on contact and through digestion and Biflex™ contains the active ingredient bifenthrin, also a pyrethroid. The drinking water guidelines were calculated to be 3.5 and 0.35 μg/L for deltamethrin and bifenthrin respectively using the approach described in NRMMC / EPHC (2007).

Application of all herbicides and pesticides tends to be on an *ad-hoc* basis and is limited to road furniture such as poles and street signs. It is not used as a broad based spray. No estimate of quantities could be obtained.

2. **TransAdelaide:** TransAdelaide is a publicly owned corporation which provides suburban train services. TransAdelaide has a specialised team for control of vegetation within the railway verges. In addition to Roundup™ and similar formulations, Oust™ and Brushoff™ with the active ingredient metsulfuron-methyl are applied as pre-emergent herbicides utilising a specialised rail carriage. Applications are generally twice per year, applied at night time but no estimates of quantities could be obtained.
3. **The Department for Transport, Energy and Infrastructure:** The Department for Transport, Energy and Infrastructure manages vegetation along the major highway corridors. The herbicides used include Roundup™, Brushoff™ and Oust™ in various formulations. Application is predominantly along road corridors annually. No estimates of quantities could be obtained from the Department.
4. **Various smaller scale applications:** A myriad of smaller scale applications would occur within the catchment including industrial site management, construction site setup; concrete slab casting site preparation; construction site management as well as domestic garden use. No information could be obtained of the types, frequency or quantities of herbicide application.

2.3.5 ASTR site security

ASTR site security and access has improved due to changes initiated by the Parafield Airport Corporation. These include additional fencing and signage indicating that trespassing on the airport grounds is a federal offence which will result in prosecution. In 2006 there were no major incidences of vandalism at the site. At present these changes have not been quantified with respect to risk reduction and management.

2.3.6 Sewer leak detection program

WSSA / NWC (2007) lists 12.7 overflows per 100 km of sewer mains for SA Water. SA Water does not have a current proactive sewer leak detection program designed to protect stormwater quality - observed sewer overflows, chokes or cross connections to sewers are repaired as they arise. SA Water does not maintain records of sewer overflows, cross connections or chokes in the Parafield catchment (Hackney, P. pers. comm.).

2.3.7 Equipment maintenance

The on-line water quality probes which function as CCPs for management of water quality in the ASTR system are calibrated every three months. Given that CCP #1 at the in-stream basin was inoperable for the majority of the year and CCP #2 had an unstable signal a more frequent maintenance regime is recommended. CCPs are critical in the operational management of water quality and as such a sound maintenance schedule as well as call out when equipment fails is fundamental to operating the system to minimise water quality risks.

3. RISK-BASED WATER QUALITY MONITORING: SOURCE WATER QUALITY

For the purposes of the residual risk assessment, the production process is defined as all operations which begin with the runoff of rainfall as stormwater from the Parafield urban catchment through to the final product water derived from the ASTR recovery production wells. In future risk management plans this definition will be expanded to include blending in mains and delivery to customers. The following section describes the water quality sampling of stormwater in 2006 which aimed to characterise the variability of source water quality.

3.1 Methods used to characterise source water quality

A refrigerated ISCO water quality auto sampler was deployed to sample stormwater outflows at the inlet of the in-stream basin (IS2). Water samples (500 mL) were collected in glass bottles every three hours over a three day period when rainfall (>0.2 mm) had been recorded at the Parafield airport weather station. Samples were combined and submitted directly to the laboratory for analysis with the exception of microbiological analyses. For these a single grab sample was taken on the last day of the sampling period.

Samples were submitted to the Australian Water Quality Centre (a NATA certified laboratory) for chemical analyses and the CSIRO Land and Water laboratory (Perth) for analysis of environmental pathogens. Details of the analytes and associated methods are given in Appendix A.

In addition to the conventional water quality monitoring program, passive samplers for monitoring of micropollutants were trialled as an alternative method to assess average concentrations (Stephens *et al.* 2005). Conventional sampling/analytical techniques have limits of detections that are often orders of magnitude above the low concentrations that may pose a risk to the environments in which they occur (Huckins *et al.* 2002). For example, 17α-ethynylestradiol has been found to cause severe impact on fish populations in freshwater lakes at concentrations of 6 ng/L, a concentration below the detection limit of many chemicals in environmental waters. In view of the limitations of conventional sampling and analytical techniques, time integrated passive sampling techniques were deployed for the monitoring of organic micropollutants throughout the ASTR system (Stephens *et al.* 2005). Passive samplers were deployed at the in-stream basin inflows (IS1), in-stream basin outflows (IS2), cleansing reed bed inflows (WE1) and cleansing reed bed outflows (WE2). Further information on passive samplers is given in Section 3.3.

As a final assessment of the fates of micropollutants in the system, sediments were also collected in recognition that many micropollutants are hydrophobic and readily adhere to sediment surfaces. Sediment samples were analysed by Ecowise Environmental Pty. Ltd., a NATA certified laboratory for sediment analyses.

3.2 Composite sampling for routinely measured water quality parameters

A key limitation of existing water quality data in assessing risks identified by Swierc *et al.* (2005) was the absence of any hydrologically-linked water quality data (i.e. event-based monitoring). Stormwater can have highly variable quality. The ‘first flush’, the initial portion of the runoff, produces higher hazard concentrations early in the runoff event and a concentration peak preceding the peak flow (Deletic 1998). Recent work to evaluate water quality improvements by isolating first flush waters in roof runoff showed that more than half the runoff volume would need to be discarded in order to have appreciable improvements in quality retained in the rainwater tank (Deletic 1998). Hence the decision to utilise the first flush may have a significant trade-off between captured stormwater quality and quantity subject to the efficiency of settling within the storage ponds. The characterisation of this phenomenon was a principal objective in the sampling described here.

3.2.1 Composite sampling methods

To characterise the first flush, event-based sampling was performed during the winter of 2006 using a refrigerated ISCO sampler during August to October 2006 to capture the water quality character of stormwater through the system. Figure 4 illustrates the hydrologic flow through the ASTR system during the event-based sampling period.

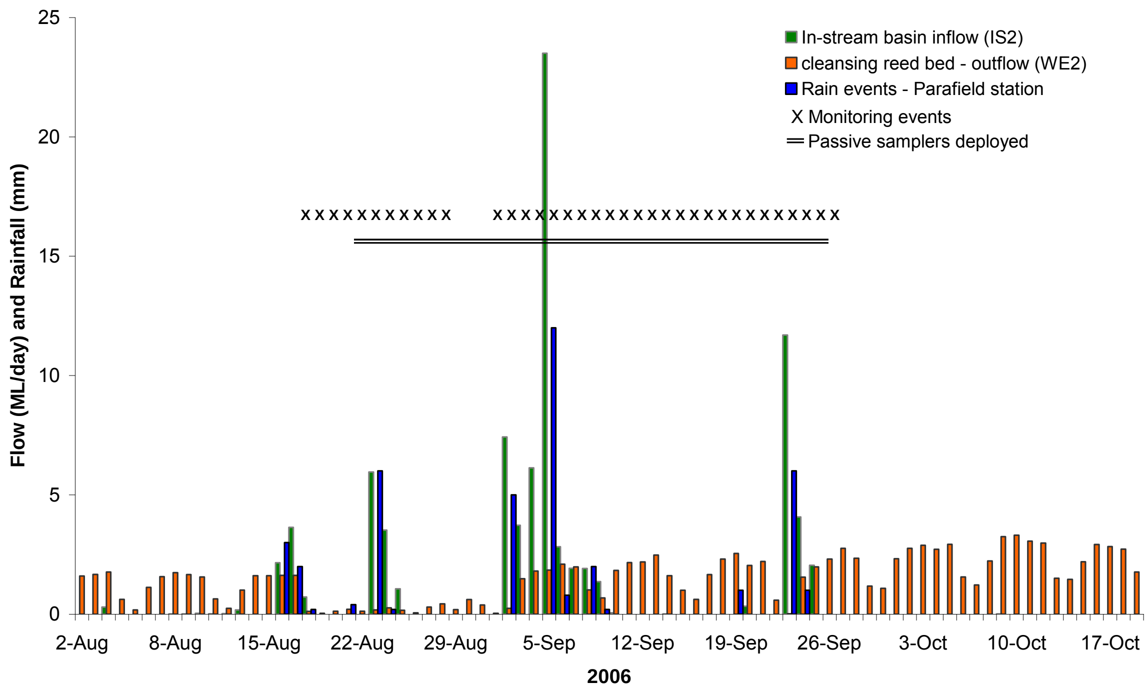

Figure 4 Daily flow through the ASTR system for the winter 2006, for monitoring points IS2 (in-stream basin outlet), WE2 (cleansing reed bed outlet) and rainfall measured at Parafield station

In deriving the flow data, the instantaneous flow rate readings were downloaded from the SCADA system at 60 minute intervals, multiplied by this time step and integrated over a 24 hour period to 9 am each day (to correspond with daily rainfall records) and recorded as daily flows for the day in which the period ends.

Figure 4 illustrates the event driven nature of the inflows into the system compared to the more steady outflows from the cleansing reed bed outflows. Figure 4 also illustrates the coupling between rainfall and inflows, and that the runoff events are clearly correlated to measured rainfall events. However, it is important to note that due to the lack of rain and hence inflows during 2006, the monitoring program was adapted to obtain samples from the in-stream basin outlet (IS2) and the cleansing reed bed outlet (WE2). No groundwater samples were collected due to on-ground infrastructure works at the time. Results of the groundwater sampling at Parafield ASR will be incorporated into groundwater assessments at the ASTR site in the next risk assessment report.

3.2.2 Composite sampling results

A summary of the 2006 stormwater quality monitoring results for the in-stream basin outlet (IS2) are given in Table 3. No turbidity or conductivity data was available from CCP #1 during this period due to the blockage of intake lines to the water quality monitoring station as discussed in Section 2.3.7.

Table 3 Volume averaged water quality monitoring results for IS2 (in-stream basin outflows)

	n*	Min	Median	Max	Mean	SD	Previous Mean**
Flow to Holding Storage (ML/day) for period 02/08/2006 – 27/10/2006	88	0	0.0008	23.5	0.96	3.1	
Physical characteristics (mg/L)^							
True Colour (HU)	7	17	48	86	51	33	
Conductivity (µS/cm)	7	164	250	321	247	81	483
pH (-)	7	7.2	7.6	8	7.6	0.3	6.9
Suspended Solids	7	3.0	4.5	12	3.5	3.5	32
TDS	7	90	139	180	137	46	265
Turbidity (NTU)	7	3.9	5.2	12	4.5	3.1	20
Major Ions (mg/L) ^							
Alkalinity	7	53	88	138	68	33	
Bicarbonate	7	65	122	168	125	35	85
Bromide	7	<0.05	0.12	0.71	0.38	0.29	
Sulphate	7	8.7	21	37	23	12	55
Chloride	7	17	25	89	24	27	78
Cyanide	7	< 0.05	< 0.05	< 0.05	< 0.05	0	
Fluoride	7	0.12	0.31	0.53	0.35	0.18	0.52
Calcium	7	15	34	51	38	14	29
Magnesium	7	3	8	29	17	11	6.8
Potassium	7	3.4	6.7	8.6	6.8	2.1	6.2
Sodium	7	11.6	18.9	98.8	22.3	35.4	59
Microbiological							
Coliforms (cfu/100mL)	7	2.8×10^{3}	2.1×10^{5}	4.8×10^{5}	3.5×10^{5}	2.1×10^{5}	
E. coli (cfu/100mL)	7	1.0×10^{0}	2.4×10^{2}	2.8×10^{3}	2.4×10^{2}	1.9×10^{3}	$1.8\text{x}10^{3}$
Enterococci (cfu/100mL)	7	0	5.2×10^{2}	2.7×10^{3}	7.0×10^{2}	1.1×10^{3}	
Faecal coliforms (cfu/100mL)	7	1.0×10^{0}	2.4×10^{2}	2.8×10^{3}	2.4×10^{2}	1.1×10^{3}	
Faecal Streptococci (cfu/100mL)	7	0	5.2×10	2.7×10^{3}	7.0×10^{2}	1.1×10^{3}	
f-RNA phages (pfu/100mL)	2	2	2	2			
somatic phages (pfu/100mL)	2	20	20	20			
Clostridium perfringens (n/20L)	2	< 1	< 1	< 1			
Cryptosporidium parvum (n/20L)	2	< 1	< 1	< 1			
Campylobacter (n/20L)	2	< 1	< 1	< 1			

	n*	Min	Median	Max	Mean	SD	Previous Mean**
Nutrients (mg/L)							
Ammonia	7	0.007	0.09	0.16	0.07	0.06	0.28
Nitrate + Nitrite	7	< 0.0005	0.003	0.086	0.006	0.03	0.14
Kjeldahl Nitrogen (t)	7	0.59	0.87	1.45	0.90	0.41	1.19
Nitrogen (t)	7	0.6	0.9	1.5	0.9	0.4	1.3
Organic Carbon (t)	7	9.9	12.4	24.8	16.8	5.9	
Organic Carbon (d)	7	8	11	23	16	5	
UV_{254} (cm^{-1})	7	0.31	0.39	0.70	0.51	0.14	
BOD_5 (d)	7	< 1.0	2.5	4.0	2.1	1.2	
BOD_5 (t)	7	2.0	4.5	6.0	3.7	1.6	
COD_5 (d)	7	37	54	68	60	11	
COD_5 (t)	7	46	68	94	82	17	
Phosphorus (t)	7	0.07	0.11	0.17	0.13	0.04	0.14
Phosphorus (d)	7	0.01	0.02	0.05	0.02	0.01	
Sulphur (t)	7	2.9	6.9	11.6	7.5	3.7	
Metals (mg/L)^							
Aluminium (t)	7	< 0.1	0.24	1.1	0.18	0.39	2.67
Antimony (t)	1	0.001	0.001	0.001			
Arsenic (d)	7	< 0.0005	0.0005	0.005	0.0006	0.002	0.0013
Arsenic (t)	7	< 0.0005	0.0005	0.006	0.0006	0.002	
Barium (t)	1	0.037	0.037	0.037			
Beryllium (t)	1	< 0.0005	< 0.0005	< 0.0005			
Boron (t)	1	0.093	0.093	0.093			
Cadmium (t)	7	< 0.0005	< 0.0005	< 0.0005	< 0.0005	0	0.0003
Chromium (t)	7	< 0.003	< 0.003	0.006	< 0.003		
Chromium (VI) (d)	7	< 0.01	< 0.01	< 0.01	< 0.01	0	
Cobalt (t)	1	0.01	0.01	0.01			
Copper (t)	1	0.015	0.015	0.015			0.008
Iron (d)	7	0.07	0.09	0.25	0.09	0.07	
Iron (t)	7	0.21	0.42	0.62	0.30	0.15	0.62
Lead (t)	7	< 0.0005	0.0025	0.0037	0.0027	0.0013	0.0078
Lithium (t)	1	0.005	0.005	0.005			
Manganese (d)	7	0.007	0.008	0.04	0.01	0.01	
Manganese (t)	7	0.035	0.043	0.058	0.043	0.010	0.034
Mercury (t)	1	< 0.0005	< 0.0005	< 0.0005			0.00025
Molybdenum (t)	1	0.0048	0.0048	0.0048			
Nickel (t)	7	0.0008	0.001	0.002	0.001	0.0005	
Selenium (t)	1	0.0015	0.0015	0.0015			
Silver (t)	1	< 0.001					
Thallium (t)	1	< 0.0005					
Vanadium (t)	1	0.003					
Zinc (t)	7	0.064	0.087	0.13	0.10	0.02	0.078

(t) = total; (d) = dissolved
^ unless otherwise indicated
* composite samples for (n=7) cover the period 16/08/06 – 26/08/07, including the first flush
** Reported averages by Marks *et al.* (2005), note, these are not volume weighted concentrations

Source water quality varied considerably among the samples. Overall the stormwater quality determined in the 2006 sampling had lower values of suspended solids, turbidity, conductivity, nutrients and *E. coli* than those previously reported for inflows by Marks *et al.* (2005) and Swierc *et al.* (2005) for baseline monitoring of the ASTR system. Given that the sampling occurred at the same location, differences may have been due to lower inflows to the system flushing less hazards out of the system or use of the autosampler in this study compared to previous grab sampling. For example, turbidity in the inflows is largely a function of the turbulence of the water entering the system and its ability to keep particles suspended in solution. Future monitoring of the source water quality will need to be performed to characterise a larger scale event (>25 mm) rainfall as is more typical during winter.

3.2.3 Composite sampling micropollutants results

In addition to the water quality results reported in Table 3, organic micropollutants were also sampled as they were previously identified as a key hazard in the preliminary HACCP plan (Swierc *et al.* 2005) largely due to the uncertainty surrounding their presence or absence. This uncertainty. when coupled to the lack of definitive information concerning uses of micropollutants such as herbicides within the catchment (Section 2.3.4, resulted in a screening of all possible micropollutants to obtain a qualitative list of those most likely to be present. Throughout the period August – October 2006, composite samples using the ISCO sampler were taken at the in-stream basin inlet and cleansing reed bed outlet in the system. During this period no micropollutants were detected in any locations. The list of micropollutants analysed and their detection limits are given in Table 4.

Table 4 Micropollutants and detection limits monitored by composite sampling (n=7)

Pesticide	LOR (ng/L)
Aldrin, Chlordane-g, Dieldrin	10
Chlorothalonil, Chlorpyrifos, Chlorthal-Dimethyl, DDD, DDE, DDT, Endosulfan, Endosulfan Sulphate, Endrin, Heptachlor, Heptachlor Epoxide, Methoxychlor, Trifluralin, Vinclozolin	50
Parathion methyl	300
2 4 5-T, 2 4-D, Atrazine, Azinphos-methyl, Chlorsulfuron, Clopyralid, Diazinon, Fenitrothion, Hexazinone, MCPA, Metsulfuron-methyl, Parathion, Picloram, Prometryne, Silvex, Simazine, Sulfometuron, Triclopyr	500

LOR limit of reporting.

Of the herbicides analysed in Table 4, only Metsulfuron-methyl was identified as being utilised significantly in the management of vegetation on roadside verges (Section 2.3.4). It was anticipated that conventional sampling would have limitations in detection of spikes of micropollutants entering the system, which is due to the event-driven nature of flows. To address this, passive samplers were co-deployed across the ASTR system at the same points where micropollutants were monitored by the conventional program.

3.3 Passive samplers for monitoring of micropollutants

3.3.1 Passive sampling methods

Passive sampling techniques for micropollutants have the advantages of lower detection limits than the conventional sampling and analysis (e.g. Table 4). They overcome the problem of detecting intermittent contaminant spikes and they give time and flow integrated concentrations that better reflect the requirements for a risk assessment for stored water.

Passive sampling techniques provide a quantitative measure of the concentration of analytes that are detected in the samplers. Two types of passive samplers were deployed: Polydimethylsiloxanes (PDMS), which are used for the non-polar micropollutants (e.g. PAHs); and Empore™ Extraction Disks (ED) samplers, which are used for sampling polar micropollutants (e.g. simazine) (Figure 5).

Figure 5 PDMS passive sampler in wire cage with two ED samplers hanging in Teflon disk mountings beneath

Polydimethylsiloxane (PDMS) is a widely used silicon-based organic polymer. It is a hydrophobic polymeric sorption material that has been widely used as a non-polar coating for gas chromatographic columns. This polymer has been used in detecting a wide range of hazards: non-polar PAHs, organic insecticides and pesticides and other more polar chemicals (e.g. Komarova *et al.* 2006). In addition to the PDMS passive samplers, a series of more polar samplers (3M™ Empore™ Extraction Disks (EDs) contained in Teflon manifolds were deployed (Figure 5). Uptake for the EDs is limited to allow longer time integration through a membrane that allows rapid diffusion of polar chemicals. Four sites in the ASTR system were selected for passive sampler deployment (Table 5).

Table 5 Details of passive samplers deployed in 2006

Sampler location	Sample Types		Deployed	Retrieved	Days Deployed
	PDMS	ED			
In-stream basin inlet (IS1)	2	2	22/08/2006	25/09/2006	34
In-stream basin outlet (IS2)	2	2	22/08/2006	06/10/2006	45
Cleansing reed bed inlet (WE1)	2	2	22/08/2006	25/09/2006	34
Cleansing reed bed outlet (WE2)	2	2	22/08/2006	25/09/2006	34
Field Blank	1	1	22/08/2006	25/09/2006	34

Average water concentrations of micropollutants for the time of deployment are derived from the concentrations sequestered in the sampler using calibrations conducted in the laboratory. Polar sampler concentrations are converted into estimates of water concentrations (C_W) using a sampling rate (L d^{-1}) calculated in laboratory studies (Stephens *et al.* 2005):

$$C_w = \frac{C_p}{R_s t} \qquad (1)$$

Where C_w is the average concentration in water, C_p is the concentration measured in the passive sampler, R_s is the estimated sampling rate (= 0.08 L.day^{-1}) and t is the time in days the passive samplers were deployed. However, in order to achieve meaningful results with passive sampling techniques, it is necessary to consider site specific factors that influence the uptake of micropollutants in the sampler.

Full details of the methodology used for the analysis of the passive samplers are given in Kapernick *et al.* (2006) and repeated in Appendix B in addition to a complete list of the micropollutants analysed by GC-MS and their limits of detection.

3.3.2 Results of monitoring of micropollutants using polydimethylsiloxane (PDMS) samplers

A series of commonly occurring micropollutants were detected throughout the ASTR system at ng/L levels (Table 6). The numbers refer to sampling sites of the system (Figure 1).

Table 6. Estimated concentrations of micropollutants (ng L^{-1}) for the PDMS samplers

	LOD*	Field Blank	IS1	IS1 (repeat)	IS2	IS2 (repeat)	WE1	WE1 (repeat)	WE2	WE2 (repeat)
Diuron	n.d.	< 50	168	159	97	84	221	208	416	345
Trifluralin	< 50	< 0.5	< 0.5	< 0.5	< 0.5	1	< 0.5	< 0.5	< 0.5	< 0.5
Simazine	< 500	< 30	< 30	< 30	< 30	< 30	< 30	< 30	104	104
Atrazine	< 500	< 8	< 8	< 8	< 8	< 8	< 8	< 8	31	32
Diazinon	< 500	< 6	< 6	< 6	13	15	23	22	< 6	< 6
Terbutryn	n.d.	< 3	< 3	< 3	< 3	< 3	< 3	< 3	7	6
Methoxychlor	< 50	< 8	< 8	< 8	< 8	< 8	< 8	< 8	12	11
Chlorpyrifos	< 50	< 0.5	3	4	3	3	3	3	< 0.5	< 0.5
Chlordane	< 10	< 1	< 1	< 1	< 1	< 1	< 1	< 1	< 1	< 1
Nonachlor	n.d.	< 1	< 1	< 1	< 1	< 1	< 1	< 1	< 1	< 1
Oxadiazon	n.d.	< 0.5	2	2	2	2	3	3	1	1
Propiconazol	n.d.	< 2.5	< 2.5	< 2.5	7	8	13	14	< 2.5	< 2.5
Endosulfan	< 50	< 2	< 2	< 2	5	4	5	4	< 2	< 2
Piperonylbutoxide	n.d.	< 0.5	2	3	2	3	3	3	< 0.5	< 0.5
Oxyfluorfen	n.d.	< 0.5	1.8	1.7	2	2	3	3	< 0.5	< 0.5
Phenanthrene	n.d.	< 0.08	1	1.1	< 0.08	< 0.08	0.3	0.3	0.1	0.1
Fluoranthene	n.d.	< 0.08	1.2	1.4	0.4	0.5	0.6	0.6	< 10	< 10
Pyrene	n.d.	< 0.08	1.8	1.8	1.4	1.9	1.4	1.6	0.3	0.3

* Micropollutants in bold were also analysed by the conventional monitoring program (Table 4), detection limits (ng/L) are given in the first column for comparison

The 26 hazards analysed for, but not detected, in any sampler over the study period are shown in Appendix B.

Of all the micropollutants detected in the system, diuron was detected at consistently the highest concentration throughout the ASTR system, with sporadic detections of other micropollutants occurring throughout. The source of diuron in the catchment is unknown.

3.3.3 Results of monitoring of micropollutants using Empore™ Disks samplers

Table 7 shows the estimated concentration of micropollutants from the Empore™ Disk (ED) samplers. The numbers refer to sampling sites of the system (Figure 1). Ten chemical hazards analysed for, but not detected, are shown in Appendix B. It can be seen that only three herbicides and one breakdown product were detected, and each herbicide was detected at all sites.

Table 7. Estimated concentrations of micropollutants (ng L^{-1}) from ED samplers

	LOR*	Field Blank	IS1	IS1 (repeat)	IS2	IS2 (repeat)	WE1	WE1 (repeat)	WE2	WE2 (repeat)
Diuron	n.d.	<0.3	14	16	35	29	64	64	25	26
Simazine	< 500	<0.3	7.9	18	17	16	39	42	19	19
Atrazine	< 500	<0.3	3.2	3.5	10	11	15	16	21	18
desisopropyl Atrazine	n.d.	<0.3	<0.3	<0.3	1.5	<0.3	3.5	6.9	<0.3	4.4

* Micropollutants in bold were also analysed by the conventional monitoring program (Table 4), detection limits (ng/L) are given in the first column for comparison

Diuron was again found at the highest levels at all sites followed by simazine and atrazine. A breakdown product of atrazine - desisopropyl atrazine, was detected at low concentrations at most sites but not in all replicates. From Table 7 it appears that concentrations of micropollutants appear to increase with passage through the system. This may result from non-isokinetic uptake by the passive samplers at the locations deployed. This implies that the passive samplers underestimate the concentration of spikes that pass rapidly through the system. This may result from turbulence; for example, the ED sampler at IS1 is exposed to greater variations in hazard concentrations, spikes of hazards are the norm as stormwater is quickly pumped from the in-stream basin to the holding storage, while the sampler at WE2 is exposed to more steady-state like conditions. This shows the limitations of passive samplers for elevating mean concentrations when events have spiky inputs.

There are differences between the reported concentrations derived by the conventional water quality monitoring PDMS and ED passive samplers are summarised in Table 8.

Table 8 Comparison of maximum concentration (ng/L) micropollutants detected by three different methods

Micropollutant	Conventional	PDMS sampler	ED sampler
Trifluralin	< 50	1	n.d.
Simazine	< 500	104	42
Atrazine	< 500	32	21
Diazinon	< 500	23	n.d.
Methoxychlor	< 50	12	n.d.
Chlorpyrifos	< 50	4	n.d.
Chlordane	< 10	< 1	n.d.
Endosulfan	< 50	5	n.d.

n.d. not determined.

The conventional monitoring had a detection limit several times higher than the passive samplers, as such was unable to detect the micropollutants at the ng/L levels detected by the passive samplers. The passive samplers also gave differing results caused by their different affinities to the micropollutants based on polarity. Mueller (2007) suggested that the ED samplers were superior in the quantitative determination of simazine and atrazine concentrations and should be used in preference to the PDMS.

3.4 Sediment sampling and analysis

3.4.1 Sediment sampling methods

Sampling of sediments was performed to determine if organic micropollutants, metals and other hazards removed from the in-stream basin, holding storage and cleansing reed bed could be identified, and hence considered a sink for these hazards. Additionally, many of the hazards are hydrophobic and known to adhere to sediment particles and hence could pose a risk if remobilised into the system.

Ten random subsamples from the inlet and ten from the outlet of each basin (in-stream basin, holding storage, cleansing reed bed) were collected on 27/10/2006. At this time the in-stream basin and holding storage basins were mostly dry and samples were collected using a metal trowel for the micropollutant analysis and a plastic trowel for the metals analysis. Sediment samples were composed of the material which was above the basin clay liner to a depth ~5 cm. Samples from the cleansing reed bed were sampled using a coring tube to a depth of ~ 5 cm. All ten samples from each site were then stored in sediment sampling jars supplied by the laboratory and combined into a single composite sample prior to submission for analysis.

3.4.2 Sediment sampling results

The results are presented in Table 9. The numbers refer to sampling sites of the system (Figure 1).

Table 9 Sediment quality of the Parafield stormwater harvesting system

		In-stream basin		Holding storage		Cleansing reedbed	
	ANZECC sediment quality guideline	SE1 (inlet)	SE1 (outlet)	SE2 (inlet)	SE2 (outlet)	SE3 (inlet)	SE3 (outlet)
Organic carbon (%)		9	6.9	2.8	5.1	1.9	0.9
Arsenic (mg/kg)	70	< 5	< 5	< 5	< 5	6	< 5
Cadmium (mg/kg)	10	< 0.2	0.6	0.2	0.3	< 0.2	0.4
Lead (mg/kg)	220	53	100	49	69	13	10
Zinc (mg/kg)	410	450	460	320	260	93	22
Iron (mg/kg)		9,400	12,000	14,000	12,000	13,000	13,000
Manganese (mg/kg)		90	130	80	58	100	93
TPH (C_6-C_9, mg/kg)		< 100	< 20	< 20	< 20	< 40	< 20
TPH (C_{10}-C_{14}, mg/kg)		1,300	140	54	25	< 40	< 20
TPH (C_{15}-C_{28}, mg/kg)		2,800	1,100	610	880	< 100	< 50
TPH (C_{29}-C_{36}, mg/kg)		2,800	1,300	860	1,200	< 100	65
Benzene (mg/kg)		< 1	< 0.5	< 0.5	< 0.5	< 1	< 0.5
Toluene (mg/kg)		1.2	< 0.5	< 0.5	< 0.5	< 1	< 0.5
Ethylbenzene (mg/kg)		< 1	< 0.5	< 0.5	< 0.5	< 1	< 0.5
Xylene (mg/kg)		< 2	< 0.5	< 0.5	< 0.5	< 2	< 0.5

TPH: Total Petroleum Hydrocarbons

There was a generally observed decrease in sediment hazard concentration in moving from the in-stream basin to the holding storage to the cleansing reed bed. All metals except zinc in the in-stream basin met the ANZECC sediment quality guidelines. Total petroleum hydrocarbons (TPH) followed a similar trend, with the highest concentrations observed in the sediments in the in-stream basin, especially at the inlet. This contrasts to the aromatic hydrocarbons, where only toluene was only detected at the in-stream basin inlet.

In addition to the sediment quality data presented in Table 9, sediments were also analysed for 53 micropollutants. None were detected in any samples. A list of analytes and their corresponding detection limits is given in Appendix C. However, for many of these micropollutants there is a strong relationship between the adsorption on sediments and the organic carbon content (Table 9). Hazard partitioning with organic carbon can be described by:

$$K_{oc} = 100K_p / (\%OC) \qquad (2)$$

where K_{oc} is the adsorption coefficient adjusted for organic matter, K_p the adsorption coefficient and %OC the percentage organic carbon in the sediment, such that %OC is > 0.1% (Schwarzenbach and Westall 1981). Briggs (1981) showed that the general relationship

$$\log K_{om} = 0.52 \log K_{ow} + 0.62 \qquad (3)$$

where K_{om} is the adsorption coefficient for organic matter. Using equations 2 and 3, the published literature on K_{oc} and K_{ow} concentrations (Appendix B) and half the detection limit of the method for the micropollutants in Appendix C, the maximum theoretical concentration for these compounds can be determined for benzene, toluene, ethylbenzene, xylene (BTEX) and polyaromatic hydrocarbons (PAHs) (Table 10) and other micropollutants (Table 11).

For BTEX and PAHs (Table 10) and for the other organic micropollutants (Table 11), the aqueous concentrations calculated from the K_{oc} values were at least two orders of magnitude below the aqueous limit of reporting. This further corroborates the non detections of the micropollutants in Table 4. In all cases the highest calculated theoretical values occurred in the in-stream basin where the organic carbon concentration was the highest.

The results for the sediment quality were the same as previous studies (EES 2005a; 2005b) in that no micropollutants or PAHs were detected. There was a single detection of toluene in the in-stream basin inlet at slightly above the detection limit. Whilst micropollutants are known to bind to organic carbon in sediments, the quantities that have accumulated after two years since the cleaning of the basins in 2004 is still below the limit of reporting. Cleaning of the in-stream basin should be performed at a frequency that enables reuse of the sediment. At a frequency of two years, the desorption from sediments will have no measurable impact on water quality of harvested water. Similarly the cleaning frequency of the holding basin could be reduced.

In general, the methods for analysing sediments are too insensitive to provide suitable information for the quantitative risk assessments for any of the hazards of interest. As such, though sediment quality data is required for decisions regarding environmental disposal of sediments, the value of the information to inform the risk assessment with respect to water quality is limited. As such no further sampling of sediments is recommended unless improved detection limits can be obtained.

Table 10 Calculated equilibrium PAH concentrations (ng/L) based on sediment detection limits

				In-stream basin		Holding storage		Cleansing reedbed			
	Sediment Guideline** value (ng/kg)	K_{OC}	Sediment LOR (ng/kg)*	SE1 (inlet) (ng/kg)	SE1 (outlet) (ng/kg)	SE2 (inlet) (ng/kg)	SE2 (outlet) (ng/kg)	SE3 (inlet) (ng/kg)	SE3 (outlet) (ng/kg)	Aqueous LOR (ng/L)	Drinking Water Guideline Value (ng/L)
Total Organic Carbon (% w/w)				9	6.9	2.8	5.1	1.9	0.9		
Benzene		60	200	0.75	0.29	0.12	0.21	0.16	0.037	100	1000
Toluene		198	200	0.55	0.087	0.035	0.064	0.047	0.011	100	25000
Ethyl Benzene		418	200	0.11	0.041	0.017	0.030	0.022	0.0053	100	3000
Xylene		418	200	0.22	0.041	0.017	0.030	0.045	0.0053	100	20000
Naphthalene	210,000	260	200	0.35	0.039	0.022	0.049	0.0036	0.0017	100	70000
Acenaphthene	500,000	4198	200	0.021	0.0024	0.0013	0.0030	0.0002	0.00010	100	
Acenaphthylene	640,000	4009	200	0.022	0.0025	0.0013	0.0031	0.0002	0.00011	100	
Fluorene	540,000	7295	200	0.012	0.0014	0.00076	0.0017	0.0001	0.00006	100	
Phenanthrene	1,500,000	5200	200	0.017	0.0019	0.0011	0.0024	0.0001	0.00008	100	150000
Anthracene	1,100,000	13583	200	0.006	0.0007	0.00041	0.0009	0.00007	0.00003	100	150000
Fluoranthene	5,100,000	45000	200	0.002	0.0002	0.00012	0.0002	0.00002	0.00001	100	
Pyrene	2,600,000	45000	200	0.002	0.0002	0.00012	0.0002	0.00002	0.00001	100	150000
Benzo[a]anthracene	1,600,000	549541	200	0.0001	0.00001	0.00001	0.00002	0.000002	0.000001	100	
Chrysene	2,800,000	311172	200	0.0002	0.00003	0.00001	0.00004	0.000003	0.000001	100	
Benzo[b]fluoranthene	9,600,000	290402	200	0.0003	0.00003	0.00001	0.00004	0.000003	0.000002	100	
Benzo[k]fluoranthene	9,600,000	620869	400	0.0001	0.00001	0.000009	0.00002	0.000002	0.000001	200	
Benzo[a]pyrene	1,600,000	650130	200	0.0001	0.00001	0.000009	0.00002	0.000001	0.000001	100	10
Dibenzo[a,h]anthracene	260,000	1659587	200	0.00005	0.000006	0.000003	0.000008	0.000001	0.000001	100	
Benzo[ghi]perylene	9,600,000	2055891	200	0.00004	0.000005	0.000003	0.000006	0.000001	0.000001	100	
Indeno[1,2,3-cd]pyrene	9,600,000	2415461	200	0.00004	0.000004	0.000002	0.000005	0.000001	0.000001	100	

* half the limit of reporting was used for the calculation of the concentration in water; ** ANZECC (2000) for a highly impacted aquatic ecosystem

Table 11 Calculated micropollutant concentrations (ng/L) based on sediment detection limits

	K_{OC}	Sediment LOR (ng/kg)*	In-Stream Basin (inlet)	In-stream Basin (outlet)	Holding Storage (inlet)	Holding Storage (outlet)	Cleansing Reedbed (inlet)	Cleansing Reedbed (outlet)	Aqueous LOR (ng/L)	Australian Drinking Water Guideline Value (ng/L)
Total Organic Carbon (% w/w)			9	6.9	2.8	5.1	1.9	0.9		
Hexachlorobenzene	50000	2	0.0018	0.00020	0.00011	0.00026	0.000010	0.000005	100	20000
α-BHC	3800	2	0.024	0.0027	0.0015	0.0034	0.00013	0.000059	100	20000
β-BHC	1995	2	0.045	0.0052	0.0028	0.0064	0.00024	0.00011	100	20000
δ-BHC	275423	2	0.0003	0.000038	0.00002	0.000046	0.000002	0.000001	100	20000
Heptachlor	96000	2	0.0009	0.00011	0.000058	0.00013	0.000005	0.000002	50	50
Heptachlor-epoxide	3800	2	0.024	0.0027	0.0015	0.0034	0.00013	0.000059	50	50
Aldrin	6600	2	0.014	0.0016	0.00085	0.0019	0.000072	0.000034	10	10
Lindane	30200	2	0.0029	0.00034	0.00019	0.00042	0.000016	0.000007	100	50
DDE	251189	2	0.0003	0.00004	0.000022	0.000051	0.000002	0.000001	50	60
Dieldrin	16000	2	0.0056	0.00065	0.00035	0.00079	0.000030	0.000014	10	10
DDD	3800	2	0.024	0.0027	0.0015	0.0034	0.00013	0.000059	50	60
DDT	138038	2	0.0006	0.000075	0.000041	0.000092	0.000003	0.000002	50	60
Endrin	218776	2	0.0004	0.000047	0.000026	0.000058	0.000002	0.000001	50	60
Endrin aldehyde	79433	2	0.0011	0.00013	0.000070	0.00016	0.000006	0.000003	50	60
Methoxychlor	2089	2	0.043	0.0049	0.0027	0.0061	0.00023	0.00011	50	200
Chlordane	3236	2	0.028	0.0032	0.0017	0.0039	0.00015	0.000070	10	100
Endosulphan	3236	2	0.028	0.0032	0.0017	0.0039	0.00015	0.000070	50	50
Mevinphos	4571	2	0.020	0.0023	0.0012	0.0028	0.00010	0.000049	100	5000
Diazinon	26915	2	0.0033	0.00039	0.00021	0.00047	0.000018	0.000008	500	1000
Malthion	191	20	0.47	0.054	0.022	0.067	0.0049	0.0024	100	50000
Fenthion	132	20	0.68	0.079	0.032	0.097	0.0072	0.0034	500	500
Chlorpyriphos	245	20	0.37	0.042	0.017	0.052	0.0039	0.0018	50	100000
Ethion	1514	20	0.059	0.0068	0.0028	0.0084	0.00063	0.00029	500	3000
Dichlorvos	10233	20	0.0087	0.0010	0.00041	0.0012	0.000093	0.000044	100	1000
Stirfos	10965	20	0.0082	0.00094	0.00038	0.0012	0.000087	0.000041	100	
Ronnel	30	20	3.00	0.35	0.14	0.43	0.032	0.015	100	
Parathion	900	20	0.100	0.011	0.0047	0.014	0.0011	0.00050	300	10000

Monochlor	15	20	6.0	0.69	0.28	0.85	0.063	0.030	100	
2,4-D	3981	20	0.023	0.002	0.0011	0.0032	0.00024	0.00011	500	100
2,4,5-T	26	2000	3.4	0.66	0.43	0.48	0.36	0.10	500	50
Atrazine	26	2000	3.4	0.66	0.43	0.48	0.36	0.10	500	100
Propozine	49	2000	1.8	0.35	0.23	0.26	0.19	0.055	500	500
Prometryn	214	200	0.42	0.048	0.019	0.059	0.0044	0.0021	500	
Ametryn	363	200	0.25	0.028	0.012	0.035	0.013	0.0062	100	5000
Terbuthylazine	813	200	0.11	0.012	0.0051	0.016	0.0058	0.0028	100	1000
Prometon	389	200	0.23	0.026	0.011	0.033	0.012	0.0058	100	

* half the limit of reporting was used for the calculation of the concentration in water

3.5 Bio-indicators

The increased metal present in stormwater runoff can bioaccumulate in flora and fauna and thereby act as important bioindicators of water quality. However, for toxicity tests to be appropriate for detecting trace metal pollution in aquatic systems, careful consideration must be given to the selection of test species. Fish and daphnia bioassays are well accepted by scientists and regulatory communities. However, others such as the crustacea *Cherax destructor*, commonly known as the 'yabby', are widely distributed within the basins of the Parafield stormwater harvesting system. Metal concentrations of yabbies harvested from the inlet and outlet of the cleansing reed bed (Burges Per. Comm. 2007) are given in Table 12.

Table 12 Metal concentrations in tissue of yabbies sampled in the cleansing reed bed

	Pb	Zn	Cu	Cd	Cr
ANFZA Food Standard (µg/g)	2.0	1,000	70	2.0	n/a
Cleansing Reed bed inlet (SE3)	5.9 (<0.2 – 7.6)	28 (20 – 34)	7.1 (6.1 – 10)	< 0.2	< 1.0
Cleansing reed bed outlet (SE3)	4.8 (<0.2 – 9.3)	26 (21 – 36)	6.7 (4.2 – 11)	< 0.2	< 1.0
Sampled commercial yabbies*	3.8	36	7.8	0.45	0.36
Corresponding sediment concentrations (µg/g)*	14	81	3.0	3.4	5.3

Brackets represent ranges
* from Bruno *et al.* (2006)

Of the five metals analysed only lead was present in concentrations greater than the ANFZA Food standard for molluscs. There is little information available on metal concentrations and toxicity for Australian fauna. Lead concentrations in the reed bed sediments (10-13 µg/g) are of similar range to that reported by Bruno *et al*. (2006) for a commercial yabby farm. However lead concentration in the reed bed yabbies (<0.2 to 9.3 µg/g) was on average higher than for commercial yabbies (3.8 µg/g). This is possibly due to wide variation in the age of reed bed yabbies and bioaccumulation. Bruno *et al.* (2006) concluded that the yabby biomagnified these metals in muscle tissue. Khan and Nugegoda (2007) studied *Cherax destructor* as a potential test species for toxicity testing of trace metals in Australian freshwater systems. The 96-h LC_{50} value for cadmium was 379 µg/L, 494 µg/L for copper, 50 mg/L for iron and 327 mg/L for nickel. Each of these values is higher than the comparative values for the metals in water in the untreated and treated stormwater at Parafield. Comparison of LC_{50} values by the same authors with those for other aquatic organisms reveals that *C. destructor* is less sensitive to trace metals than most other tested species and so is unlikely to be a suitable bioindicator of water quality for the ASTR system.

4. ASSESSMENT OF ASTR STORMWATER HARVESTING TREATMENT PERFORMANCE

Water harvested from the Parafield catchment that meets the target turbidity of 100 NTU was pumped from the in-stream basin to the holding storage and then gravity fed to the cleansing reed bed. During 2006, no water was rejected from the system.

CCP #1 at IS2 failed for the majority of 2006, as described in Section 2.1. The low rainfall and subsequent flows to the system caused physical blockages to the water quality station at this CCP. This resulted in the turbidity and conductivity meter being inoperative for the period. Confidence in CCPs is critical for the effective implementation of the HACCP plan if water is to be recovered for drinking water supplies.

This target CCP limit was originally defined by the South Australian Department of Health and Community Services for transfer of stormwater at CCP#1 (IS2). Water that does not meet the critical limit of the CCP is not permitted to enter further into the stormwater harvesting system. Although this value was subjective, it has been deemed sufficient to meet current regulatory objectives.

The efficacy of Parafield stormwater harvesting cleansing reed bed pre-treatment barrier for ASTR has not been previously quantitatively assessed. Reported removal efficiencies (Marks *et al.* 2005) were based on a comparison of average inflow and outflow concentration values. Due to the large temporal variations in runoff quality and relatively large residence time within the system (~10 days), this approach is prone to large errors in estimated removal efficiencies of the system. This revised quantitative assessment incorporates a more sophisticated approach to the calculation of removal efficiencies utilising a water and chloride balance before determining the actual removal efficiencies of reactive contaminants. This approach is discussed in the following section after an assessment of the cleansing reed bed outflow quality.

4.1 Water quality characterisation of the cleansing reed bed product water

Of primary interest is the quality of the product water leaving the cleansing reed bed of the Parafield stormwater harvesting system. This is because the outflows from the cleansing reed bed will be used as an injectant for the subsurface component of the ASTR system. Even though outflows have been largely steady (Figure 4), volume-averaged sampling was performed to obtain estimates of average hazard concentrations. The exception to this was pathogen and faecal indicator concentrations which were grab samples only. A summary of the 2006 volume averaged stormwater quality for WE2 is given in Table 13.

Table 13 Volume average water quality monitoring results for WE2 (Cleansing reed bed outlet)

	n	Min	Median	Max	Mean	SD	ADWG Guideline
Physical characteristics							
*** True Colour (HU)**	**7**	**17**	**66**	**83**	**52**	**27**	**15**
Conductivity (µS/cm)	7	182	296	361	293	76	
pH	7	7.2	7.3	7.5	7.3	0.12	6.5 – 8.5
Suspended Solids	7	< 1	1	3	1.3	0.9	
TDS	7	150	180	200	183	18	500
Turbidity (NTU)	7	1.3	3.1	4.8	2.9	1.4	5
Major Ions (mg/L)							
Alkalinity	7	53	83	116	87	29	
Bicarbonate	7	65	101	142	106	35	
Bromide	7	0.05	0.08	0.14	0.08	0.04	
Sulphate	7	8.4	9.6	11.4	9.9	1.1	250
Chloride	7	26	35	46	35	8	250
Cyanide	7	< 0.03	< 0.03	< 0.03	< 0.03	0	0.08
Fluoride	7	0.13	0.20	0.27	0.21	0.06	1.5
Calcium	7	16	26	35	27	9	
Magnesium	7	3.6	6.4	8.0	6.3	1.9	
Potassium	7	3.5	5.0	7.8	5.4	1.7	
Sodium	7	13	21	28	21	5	180
Microbiological							
*** Coliforms (cfu/100mL)**	**7**	**6.9×10^2**	**2.2×10^3**	**4.8×10^3**	**2.3×10^3**	**1.5×10^3**	**0**
*** *E. coli* (cfu/100mL)**	**7**	**< 1**	**1.0×10^1**	**1.1×10^2**	**2.3×10^1**	**4.2×10^1**	**0**
*** Enterococci (cfu/100mL)**	**7**	**< 1**	**7.0×10^0**	**1.1×10^1**	**6.0×10^0**	**4.2×10^0**	**0**
*** Faecal coliforms (cfu/100mL)**	**7**	**< 1**	**1.2×10^1**	**1.1×10^2**	**2.5×10^1**	**4.5×10^1**	**0**
*** Faecal Streptococci (cfu/100mL)**	**7**	**< 1**	**7.0×10^0**	**1.1×10^1**	**6.0×10^0**	**4.2×10^0**	**0**
f-RNA phages (pfu/100mL)	6	> 1		> 20			
somatic phages (pfu/100mL)	6	< 1		> 20			
Clostridium perfringens (n/20L)	4	< 1		< 1			
Cryptosporidium parvum (n/20L)	5	< 1		< 1			
Campylobacter (n/20L)	4	< 1		< 1			
Nutrients (mg/L)							
Ammonia	7	< 0.005	0.012	0.039	0.014	0.013	0.5
Nitrate + Nitrite	7	< 0.0005	< 0.0005	< 0.0005	< 0.0005	0	3
Kjeldahl Nitrogen (t)	7	0.26	0.50	0.74	0.48	0.19	
Nitrogen (t)	7	0.26	0.44	0.74	0.45	0.18	
Organic Carbon (t)	7	5.1	9.4	11.9	8.6	2.8	
Organic Carbon (d)	7	4.5	8.6	11.1	7.8	2.6	
UV_{254} (cm^{-1})	7	0.173	0.370	0.443	0.320	0.107	
BOD_5 (d)	7	1	1	2	1.1	0.4	
BOD_5 (t)	7	1	1	3	1.2	0.8	
COD_5 (d)	7	8	25	39	23	12	
COD_5 (t)	7	20	35	94	42	28	
Phosphorous (t)	7	0.016	0.042	0.075	0.044	0.026	
Phosphorous (d)	7	< 0.005	0.02	0.04	0.02	0.01	
Sulphur (t)	7	2.8	3.3	3.6	3.3	0.34	
Metals (mg/L)							
Aluminium (t)	7	0.02	0.07	0.71	0.17	0.27	
* Antimony (t)	2	0.0006		0.0131			0.003
Arsenic (d)	7	< 0.001	0.001	0.006	0.001	0.002	0.007
Arsenic (t)	7	< 0.001	0.001	0.006	0.001	0.002	0.007

	n	Min	Median	Max	Mean	SD	ADWG Guideline
Barium (t)	2	0.02		0.024			0.7
Beryllium (t)	2	< 0.0005		< 0.0005			
Boron (t)	2	0.048		0.06			4
Cadmium (t)	7	< 0.0005	< 0.0005	< 0.0005	< 0.0005	0	0.002
Chromium (t)	7	< 0.003	< 0.003	< 0.003	< 0.003	0	0.05
Chromium (VI) (d)	7	< 0.01	< 0.01	< 0.01	< 0.01	0	0.05
Cobalt (t)	2	< 0.01		< 0.01			
Copper (t)	2	< 0.03		< 0.03			1
*** Iron (d)**	**7**	**0.05**	**0.30**	**0.53**	**0.33**	**0.18**	**0.3**
*** Iron (t)**	**7**	**0.24**	**0.62**	**0.95**	**0.55**	**0.31**	**0.3**
Lead (t)	7	< 0.0005	0.0005	0.0008	0.0005	0.0003	0.01
Lithium (t)	2	< 0.01		< 0.01			
Manganese (d)	7	0.003	0.02	0.03	0.02	0.01	0.1
Manganese (t)	7	0.03	0.046	0.056	0.042	0.01	0.1
Mercury (t)	2	< 0.0005		< 0.0005			0.001
Molybdenum (t)	2	0.0022		0.0045			0.05
Nickel (t)	7	< 0.0005	< 0.0005	< 0.0005	< 0.0005	0	0.02
Selenium (t)	2	< 0.003		0.004			0.01
Silver (t)	2	< 0.001		< 0.001			0.1
Thallium (t)	2	< 0.0005		< 0.0005			
Vanadium (t)	2	< 0.003		0.006			
Zinc (t)	7	0.021	0.023	0.048	0.030	0.010	3

Bold indicates mean value exceeded the Australian Drinking Water Guidelines (2004)
* indicates maximum value exceeded the Australian Drinking Water Guidelines (2004)

An initial assessment of these results shows that the majority of these parameters already meet the Australian Drinking Water Guidelines (NHMRC 2004) prior to transfer to the subsurface storage component of the system. Exceptions to this include: colour, iron and the majority of the microbiological indicators. It is worth noting that colour, iron and the indicator organisms have likely sources within the system itself. Colour and iron are mostly derived from the release of iron from the sediments (Table 3 and Table 9) and faecal indicator organisms may originate from warm blooded animals that have been observed in the cleansing reed bed itself including hares, ducks, pigeons, sparrows and rodents. The concentrations reported in Table 13, when compared to the data presented in Table 3 do not necessarily reflect the actual removal efficiency of the system as a whole. To address this; a water and chloride balance was determined for the entire pre-treatment system for a three week period over the winter when the temperatures, stormwater inflows and hydraulic retention time of the cleansing reed bed were likely to be most representative of when stormwater injection would occur.

4.2 Water and chloride balances for determination of treatment process efficiency

Wetlands are a commonly used method for improving stormwater quality. They are passive systems and have been found to be robust in removing a variety of hazards over a range of environments. A summary of the reported removal efficiencies (REs) of wetlands for a variety of hazards in literature is in Appendix D.

However, wetland systems also have limitations such as the large land area needed for treatment in comparison with engineered treatment processes. Additionally, their REs have not been completely verified in a risk management sense, particularly the removal of dissolved nutrients, pathogens and micropollutants due to a combination of a lack of data and variability of inputs. The lack of process verification is complicated by the lack of a standard reporting method in the literature on details regarding wetland design, flow, loading and sampling strategies. Frequently, aspects such as hydraulic loading, short circuiting, vegetation types, ages and densities are not reported (Appendix D).

4.2.1 Water balance calculations

To address this, a water balance was determined (for the period 21/08/06 to 15/09/06) from the holding storage inlet to the cleansing reed bed outlet to determine the REs of the reactive contaminants in Table 3. This period was chosen to coincide with sampling, observed storm events and where there was no extraction from the nearby ASR wells into the cleansing reed bed. A summary of the flows, changes in storage volume and the times of sampling for this period are shown in Figure 6.

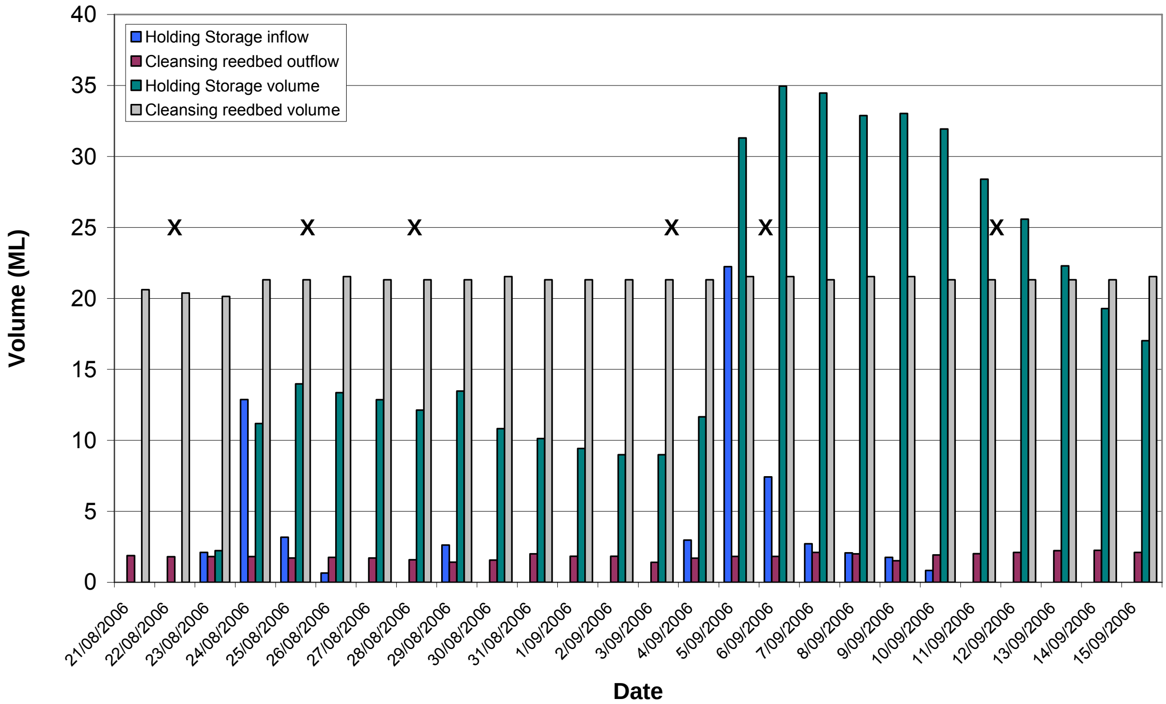

Figure 6 Flow, storage volumes and sampling times for determination of the ASTR system removal efficiencies, X indicates date of composite sampling for the in-stream basin inlet and cleansing reedbed outlet used for determination of treatment performance

This period covers three storm events which resulted in water being transferred from the in-stream basin from the 23-26/08, 29/08 and 4-9/09/06 to the holding storage, resulting in the observed rise in the volume of holding storage water. Water was then gravity fed to the cleansing reed bed resulting in a gradual decline in the holding storage volume, while flows from the reed bed remain fairly constant at ~1.8 ML/day. These constant flows are in contrast to the highly variable conductivity values displayed in Figure 3.

From 21/08/06 - 15/09/06 there was 26.6 mm of rainfall and 80 mm of class A panevaporation recorded at the nearby Parafield Airport meteorological station. To determine the relevance of rainfall and evaporation to the water budget it was assumed that the combined area of the holding storage and cleansing reed bed is 10^4 m^2, which equates to V_r of only ~0.3 ML. Similarly, applying a 0.7 multiplier to account for the scale effect in translating from pan to basin scale as well as the reduction in exchange with the atmosphere caused by covers over the basin, V_e is estimated at ~0.6 ML.

Furthermore, additional sources of uncertainty arise for the water balance stem from the nature of the polled SCADA data. The SCADA system used to operate the Parafield stormwater harvesting system was polled at hourly intervals for this period and the data used to develop an initial water balance across the system. However, SCADA data does not give cumulative water flows through the system, but rather an instantaneous reading at the given time of polling. As such the time of polling may not be representative compared to the period as a whole (currently hourly). The potential cumulative error associated with the method of calculating the volume pumped to the holding storage was calculated to be ~3ML consistent with the observed error.

The combined water balance for the holding storage and cleansing reed bed can be written for any given time period as:

$$V_{cr} = V_{hs} + V_{asr} + V_r - V_e + \left(S_{hsi} - S_{hsf}\right) + \left(S_{cri} - S_{crf}\right) \quad (4)$$

where: V_{cr} is the outflow volume from the cleansing reed bed over the period;

V_{hs} is the inflow volume pumped from the in-stream basin to the holding storage;

V_{asr} is the inflow volume from the Parafield ASR wells pumped directly to the cleansing reed bed;

V_r is the volume contributed by rainfall to the holding storage and the cleansing reed bed;

V_e is the volume of evaporation from the holding storage and cleansing reed bed;

S_{hs} is the volume held in the holding storage basin, , *i* initially and *f* finally with the difference representing the change in storage volume; and

S_{cr} is the volume held in the cleansing reed bed, *i* initially and *f* finally with the difference representing the change in storage volume;

A summary of the water balance calculations for a 24 day period in 2006 is presented in Figure 7.

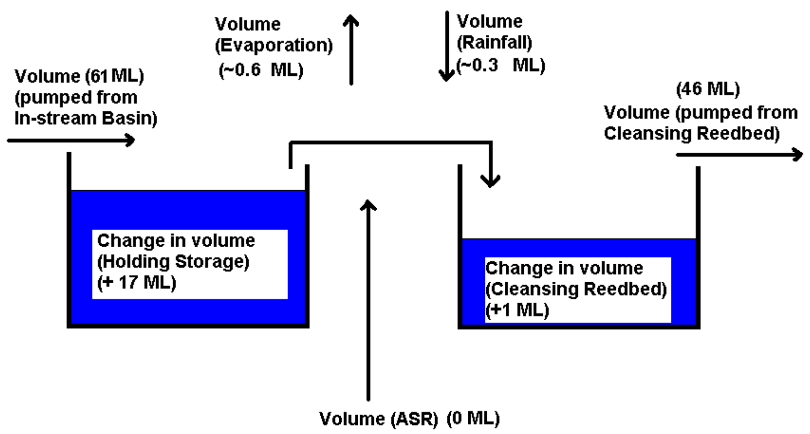

Figure 7 Conceptual diagram for the water balance of the Parafield system

During the period of 22/08 – 15/09, 61.4 ML of stormwater was captured by the system, with 45.9 ML exiting the system through the cleansing reed bed. The estimated error of the water balance was -2.7 ML for the same period, 4% of the inflows.

The water balance was corrected by adjusting the cleansing reed bed volume. Corrected water balance is given in Table 14

Table 14 Summary of the corrected daily water balance (ML), 22/06 – 15/09/2006

Date	Volume Inflow (V_{hs})	Volume rainfall (V_r)	Volume Evaporation (V_e)	Δ Holding Storage Volume (S_{hsf}-S_{hsi})	Corrected Δ Cleansing Reedbed Volume (S_{crf}-S_{cri})	Volume Outflow (V_{cr})
22/08/2006	0.0	0.0	0.0	0.0	-1.8	1.8
23/08/2006	2.1	0.0	0.0	2.2	-1.9	1.8
24/08/2006	12.9	0.1	0.0	9.0	2.2	1.8
25/08/2006	3.2	0.0	0.0	2.8	-1.3	1.7
26/08/2006	0.6	0.0	0.0	-0.6	-0.5	1.8
27/08/2006	0.0	0.0	0.0	-0.5	-1.2	1.7
28/08/2006	0.0	0.0	0.0	-0.7	-0.9	1.6
29/08/2006	2.6	0.0	0.0	1.3	-0.2	1.4
30/08/2006	0.0	0.0	0.0	-2.6	1.1	1.6
31/08/2006	0.0	0.0	0.0	-0.7	-1.3	2.0
1/09/2006	0.0	0.0	0.0	-0.7	-1.2	1.8
2/09/2006	0.0	0.0	0.0	-0.5	-1.4	1.8
3/09/2006	0.0	0.1	0.0	0.0	-1.4	1.4
4/09/2006	3.0	0.0	0.0	2.7	-1.4	1.7
5/09/2006	22.2	0.0	0.0	19.6	0.7	1.8
6/09/2006	7.4	0.1	0.0	3.6	2.1	1.8
7/09/2006	2.7	0.0	0.0	-0.5	1.1	2.1
8/09/2006	2.1	0.0	0.0	-1.6	1.6	2.0
9/09/2006	1.8	0.0	0.0	0.2	0.1	1.5
10/09/2006	0.8	0.0	0.0	-1.1	0.0	1.9
11/09/2006	0.0	0.0	0.0	-3.5	1.5	2.0
12/09/2006	0.0	0.0	0.0	-2.8	0.7	2.1
13/09/2006	0.0	0.0	0.0	-3.3	1.0	2.2
14/09/2006	0.0	0.0	0.0	-3.0	0.7	2.3
15/09/2006	0.0	0.0	0.0	-2.2	0.1	2.1
Sum	61.4	0.3	0.5	17.0	-1.7	45.9

In addition, using the water balance data, a retention time in the cleansing reed bed could also be calculated. Volumetrically, it was calculated during this period by comparing the reed bed volume to the reed bed outflows. Dividing the reed bed storage volume by the daily outflow gave an average hydraulic retention time for the cleansing reed bed of 11.6 days with a minimum of 9.3 and a maximum of 14.9. These results should be considered in light of the design retention time of 7 days.

4.2.2 Chloride balance calculations

From the water balance, a chloride mass balance was also constructed using the law of conservation of mass by including a chloride concentration term for each of the inputs and outputs. Assuming no other sources or sinks of chloride in the system this can be represented by:

$$V_{cr}Cl_{cr} = V_{hs}Cl_{hs} + V_{asr}Cl_{asr} + V_{r}Cl_{r} + \left(S_{hsi} - S_{hsf}\right)\left(Cl_{hsi} - Cl_{hsf}\right) + \left(S_{cri} - S_{crf}\right)\left(Cl_{cri} - Cl_{crf}\right) \quad (5)$$

where Cl stands for the mean chloride concentration during the interval for all flows, and to the starting and ending concentrations for all storages and all other subscripts as per the water balance equation.

This chloride balance allowed the calculation of the mass of chloride (i.e. load) entering and leaving the system and corrected for the error in the water balance. Rainfall was assumed to have an average chloride concentration of 5.7 mg/L (Keywood *et al.* 1997). A summary of the chloride balance for the system for the seven periods over which samples were taken is given in Table 15.

Table 15 Summary of the chloride balance (kg), 22/06 – 15/09/2006

Period	Holding Storage inflow mass ($V_{hs}Cl_{hs}$)	Rain (V_rCl_r)	Δ Holding Storage mass (S_{hsi}-S_{hsf}) (Cl_{hsi}-Cl_{hsf})	Δ Cleansing Reedbed mass (S_{cr}-S_{crf}) (Cl_{cri}-Cl_{crf})	Cleansing Reed bed outflow mass - calculated ($V_{cr}Cl_{cr}$)	Cleansing reed bed outflow mass – measured ($V_{cr}Cl_{cr}$)	Error (Measured – Calculated)
22/08-25/08	599	0.7	461	22	116	178	62
26/08-01/09	121	0.0	-168	25	264	345	81
02/09-04/09	98	0.5	73	16	9.2	99	90
05/09-09/09	797	1.5	470	29	299	213	-86
10/09-13/09	21	0.0	-268	26	263	190	-73
14/09-15/09	0	0.0	-137	22	115	92	-24
Sum	1,636	2.7	431	140	1,066	1,117	50

During this period, $V_{asr} = 0$

Using equation 5 the calculated error in the chloride balance relative to the inflows was 50.2 kg or approximately a 3% error.

4.3 Calculation of removal efficiencies for the cleansing reedbed

Having closed the chloride balance the removal efficiencies of the reactive contaminants across the system for the same period were calculated. The period was broken into the same distinct periods for which water quality averages of inputs and outputs were available (Table 3 and Table 13). These distinct periods were defined by the storm events that occurred and their associated monitoring results. For the remainder of the discussion on the cleansing reed bed performance, the removal efficiency (RE) is discussed. Previous efforts to calculate REs of the system (e.g. Marks *et al.* 2005) have calculated the relative reduction in concentration, C of any solute between inflows to the holding storage and outflows from the reed bed based on concurrent grab sample concentrations:

$$RE^{\dagger} = \frac{C_{hs} - C_{cr}}{C_{hs}} \quad (6)$$

where $RE^{\dagger}$ is an approximation for the removal efficiency. However this fails to take into account the variations in flow rates and concentrations of hazards in stormwater. Removal efficiency was determined in this report using a mass balance approach and using volume averaged concentrations:

$$RE* = \frac{C_{hs}V_{hs} - C_{cr}V_{cr}}{C_{hs}V_{hs}} \tag{7}$$

This equation applies equally to chloride (RE_{Cl}) yet no chloride removal is anticipated and therefore serves as a check on the accuracy of measurement of flows, volumes, concentrations and mass. $RE*_{Cl}$ was calculated to be -0.02, indicating a 2% error in the chloride balance. For other reactive contaminants, the calculated RE is corrected for Cl, where:

$$RE^{**} = \frac{RE^{*} - RE^{*}_{Cl}}{1 - RE^{*}_{Cl}} \tag{8}$$

Using this equation, RE** was calculated for the other reactive parameters (Table 16).

Removal efficiencies could not be calculated for all parameters listed in Table 3 as many of these hazard concentrations were below the method detection limits in the input water quality. Where a concentration was below the detection limit, half the detection limit was substituted in the calculation of RE. Where the majority of the hazard concentrations were below the detection limit no RE was calculated.

Table 16 ASTR system pre-treatment performance (holding storage and cleansing reed bed combined 22/08 – 15/09/06)

Parameter	Holding storage inlet*	Cleansing reed bed outlet*	Uncorrected for chloride RE**	Corrected for chloride RE***	Marks *et al.* (2006) RE†	RE‡
Physical Characteristics						
True Colour (HU)	51	52	0.24	0.27		
Conductivity (µS/cm)	247	293	-0.07	-0.03		
Suspended Solids (mg/L)	3.5	1.3	0.73	0.74	0.77	0.77 ± 0.23
TDS (mg/L)	130	191	-0.06	-0.02	0.60	
Turbidity (NTU)	4.5	2.9	0.51	0.53	0.79	0.96
Major Ions (mg/L)						
Alkalinity	67.5	86.6	0.04	0.08		
Bicarbonate	125	105	0.37	0.39		
Sulphate	23.0	9.9	0.68	0.69		
Chloride	24.0	34.5	-0.04	0.00		
Fluoride	0.35	0.21	0.56	0.57		
Calcium	37.9	26.9	0.47	0.49		
Magnesium	17.1	6.3	0.73	0.74		
Potassium	6.8	5.4	0.40	0.43		
Sodium	22.3	21.0	0.05	0.08		
Microbiological (n/100mL)						
Coliforms	3.5×10^5	2.3×10^3	0.99	0.99		
E. coli	2.4×10^2	2.3×10^1	0.93	0.93	0.65	
Enterococci	7.0×10^2	6×10^0	0.99	0.99		
Faecal coliforms	2.4×10^2	2.6×10^1	0.93	0.93		0.98 ± 0.01
Faecal Streptococci	7.0×10^2	6×10^0	0.99	0.99		0.98
Nutrients (mg/L)						
Ammonia	0.07	0.02	0.83	0.84		0.46 ± 0.51
Nitrogen (t)	0.91	0.45	0.63	0.65	0.59	0.53 ± 0.23
Organic Nitrogen	0.90	0.48	0.62	0.64		0.46 ± 0.35
BOD (t)	3.7	1.2	0.76	0.77		0.66 ± 0.31
COD (t)	82.7	42.4	0.62	0.63		0.50 ± 0.26
Organic Carbon (d)	15.7	7.9	0.63	0.64		
Organic Carbon (t)	16.8	8.6	0.62	0.63	0.81	0.42 ± 0.23
UV_{254} absorbance	0.51	0.32	0.53	0.55		
Phosphorous (t)	0.13	0.04	0.75	0.76	0.64	0.49 ± 0.31
Sulphur (t)	7.5	3.3	0.67	0.68		
Metals (mg/L)						
Aluminium (t)	0.18	0.17	0.28	0.31		0.81
Iron (d)	0.09	0.33	-1.82	-1.71		
Iron (t)	0.3	0.55	-0.37	-0.32	0.35	0.99
Lead (t)	0.003	<0.0005	0.86	0.86	0.67	0.87 ± 0.12
Manganese (d)	0.01	0.02	-0.55	-0.49		
Manganese (t)	0.043	0.042	0.27	0.29	0.69	
Zinc (t)	0.10	0.03	0.80	0.81	0.48	0.71 ± 0.26

(d) dissolved, (t) total
* Volume averaged mean concentrations, n = 7
** uncorrected for chloride removals (Equation 5)
*** corrected for chloride (Equation 6)
† RE reported by Marks *et al.* (2005), n=12
‡Mean RE of surveyed wetlands from Appendix D (methods vary)

4.4 Review of cleansing reed bed treatment performance

A literature review of wetland treatment REs of the hazards commonly measured in international studies is presented in Appendix D. The average reported REs for wetland systems are compared to the calculated REs of the cleansing reed bed in Table 16. In attempting to understand the variability of the results reported for wetlands in Appendix D, Owens and Walling (2002) and Wahl *et al.* (1997) report that they were unable to derive firm conclusions between sites, nor identify cause effect relationships. Further work being performed by a University of Adelaide PhD student uses a sophisticated approach and will be reported in the next risk assessment report.

4.4.1 Cleansing reed bed performance and physical water quality

Treatment performance of urban wetlands has been mainly focussed on physical characteristics such as suspended solids and turbidity. In this respect the ASTR pre-treatment system has a similarly high removal efficiency of suspended solids (0.74) as reported in the general literature (0.77) (Appendix D) and in Table 16.

Hazards such as suspended solids are reasonably easy to manage as suitable CCPs such as turbidity can be implemented with appropriate critical limits at IS2 (in-stream basin outlet) and WE2 (cleansing reed bed outlet). The removal of turbidity and suspended solids in stormwater is a commonly adopted management measure and in the context of the ASTR project, is the function of all three treatment basins. However, it is important to recognise that the size range of sediments removed by the discrete sub-processes (in-stream basin, holding storage, vs. the cleansing reed bed) differs. For example, at IS1 (in-stream basin inlet) the d50 (median particle size) of the suspended solids in the water column was ~17 µm which dropped to ~8.7 µm at WE1 (cleansing reed bed inlet) and finally ~3.4 µm at WE2 (cleansing reed bed outlet).

The removal of suspended solids by sedimentation deserves attention as suspended solids may act as a mobile substrate for hazards such as BTEX and heavy metals (Table 9), viruses (Davies *et al.* 2003) and PAHs (Shinya *et al*. 2000). There is no doubt as to the importance of the removal of suspended solids from stormwater runoff and the use of turbidity as a CCP. However, at the same time it is important that the treatment barriers are capable of removing the critical size range of sediments which would be carrying a significant hazard load. Due to their physico-chemical characteristics, finer particulates are more efficient in the adsorption of pollutants and hence carry a relatively higher hazard concentration (Roger *et al.* 1998; Davies *et al.* 2003). Furthermore, the abundance of pathogens has also been correlated with finer particle sizes in reservoir studies (Brookes *et al*. 2005; Hipsey *et al*. 2006). Furthermore, even though some hazards are not directly monitored at a CCP, the hazards present in the stormwater may be reduced through supporting programs. Therefore, although activities such as catchment management are not classified as a CCP (as it cannot be monitored in real-time) the risks at this step could still be reduced by supporting programs for example by preventing fine sediment from entering the system.

4.4.2 Cleansing reed bed performance and microbiological indicator removal efficiency

The cleansing reed bed system is able to greatly reduce faecal indicator levels (0.93 – 0.99 removal efficiency), though measured outflow concentrations are sometimes higher than inflows (Table 16). Such observations have been previously noted in the literature (e.g. Perkins and Hunter 2000) typically where inflow levels are extremely low, there has been no account of flow weighted concentration or when there are extraneous input sources of animal faecal matter (wildlife) within the wetland.

To explain pathogen removal efficiencies, it is necessary to better understand what removal mechanisms are dominant, and how these mechanisms may be intensified through the manipulation of system operational parameters.

More recently, research efforts have begun to consider possible mechanisms for pathogen removal (Khatiwada and Polprasert 1999; Davies *et al.* 2003). Physical, chemical, and biological pathogen removal mechanisms have been identified (Khatiwada and Polprasert 1999; Davies *et al.* 2003) though it has not been determined which mechanisms are the most sensitive or amenable to process manipulation. Physical removal processes include various forms of filtration and settling. Chemical removal processes include oxidation and UV irradiation. Biological removal processes include biofilm adsorption (retention followed by natural die-off), predation (protozoan and/or viral) and the release of antibiotics by plants and other microbes. Controlled experimentation into pathogen removal is lacking in the research literature and is one issue that will be addressed with the use of pathogen diffusion chambers to assess die-off rates as has been previously done in aquifers (e.g. Toze and Hanna 2002; Gordon and Toze 2003). The pathogen die-off rates in the cleansing reed bed will be assessed both for abiotic and biotic factors that contribute to pathogen attenuation and the results used for further refinements of the quantitative risk assessment.

4.4.3 Nutrient removal efficiency in the cleansing reed bed

Nutrient removals in wetlands has been extensively studied (Appendix D). The cleansing reed bed had superior REs to the average REs reported (Table 16) for the wetlands studies collated in Appendix D. Factors affecting nutrient removals in wetlands cited included:

- Hydraulic residence time was the most important factor for nitrogen removal.
- Oxygen concentration in the sediments was the most commonly cited factor promoting phosphorous removal.
- Vegetative processes such as the flowering phases of the macrophytes can influence phosphorous removal.

However, the studies cited in Appendix D are difficult to directly compare as there is insufficient consistency of the reporting of wetland size and residence time and vegetation to examine these factors in more detail.

4.4.4 Metal removal efficiency in the cleansing reed bed

There have been many studies on the removal of heavy metals by urban wetlands. The REs for the cleansing reed bed: lead (0.86) and zinc (0.81), the two most commonly reported metals, are very similar to the average reported in the literature, Appendix D (0.87 and 0.71 respectively). While the RE differences may be explained in terms of hydraulic residence time and source water loadings, other metals such as iron and manganese had higher dissolved and total concentrations exiting the reed bed than in the raw stormwater samples (dissolved iron RE = -1.71).

It was observed that the sediments initially act as sinks for total iron brought into the in-stream basin (Table 9). During the sediment sampling, large quantities of iron were visible in the sediments as gross pollutants (e.g. nails, staples, rusty scrap metal pieces). Typically ~100 tonnes of sediment are removed from the in-stream basin each year (Purdie, M. Pers. Comm. 2007), which given the results from the sediment monitoring (Table 9) equates to ~1.4 tonnes of iron present in the sediments.

Geochemical modelling of equilibrium concentrations of iron and manganese using PHREEQC (Parkhurst and Appelo 1999), indicate that there would be releases of iron and manganese from the sediments in excess of concentrations expected from equilibrium alone. Microbial cycling which converts Fe (III) to Fe (II) under anoxic conditions in the sediments is then re-released into the water column. A similar release of total and dissolved iron from an artificial surface flow wetland was previously reported by Chagúe-Goff (2005). She reported that during summer, Fe accumulated in the sediments but was subsequently released in winter. The results of that study suggested that anoxic conditions promote the release of Fe from the sediments, and thus the transformation of particulate to dissolved iron.

The gross pollutants containing particulate Fe are a source of dissolved Fe in the holding storage and cleansing reed bed. Given the large quantities of Fe that wash into the system, there will always be a significant source of iron present in the sediments available for release. Additional treatment provided during subsurface storage will be evaluated to determine its adequacy and sustainability to achieve water quality of a potable standard.

4.4.5 Cleansing reed bed performance and micropollutant removal

No removal efficiencies for micropollutant hazards could not be calculated (due to no detections) for the ASTR system (Table 4). No attempt was made to use the passive sampling data due to the evidence of non-isokinetic uptake by the samplers which caused them to underestimate the concentration of micropollutants entering the system. To date there have been very few reported removal efficiencies for micropollutants in wetlands available in the literature. This stems from the difficulty of detecting micropollutants continuously when source concentrations vary enormously and hence in determining mass balances for highly variable systems. Of note is a study by Kadlec and Hey (1994) who observed a highly variable 0.26 - 0.95 mean removal efficiency for atrazine in four different artificial wetlands. Future research in this area is required if the design, operational and environmental features that affect removal efficiencies of micropollutants in wetlands are to be identified and reliance is to be made on wetlands as robust treatment systems.

4.5 Summary and recommendations arising from treatment performance investigations

In summary, the majority of the hazards passing through the cleansing reed bed system have a removal efficiency of a similar level to those previously reported in the international literature. The water quality of samples of the harvested stormwater meets the Australian drinking water guidelines for the majority of parameters with the exception of colour, iron and microbial indicators. Further validation monitoring will include:

- Use of pathogen attenuation chambers to assist in determining decay rates for the quantitative risk assessments described later and to help delineate if the observed microbial indicators are a result of short circuiting through the cleansing reed bed or because they were introduced by fauna present within the system.

- Assessment of iron and colour of water recovered from the subsurface. The colour of the water leaving the wetland is higher than the stormwater entering the system despite a reduction in DOC concentration. This is attributed to a similarly larger concentration of total and dissolved iron and manganese released from the sediments. This is confirmed by both geochemical modelling and a mass balance which determined that significantly more dissolved iron leaves than enters the system. Interventions required to prevent the release of iron should focus on either the more frequent cleaning of the basins to remove the particulate iron source or preventing the sediments from becoming anoxic and releasing iron. The elevated iron and colour in the treated water may still be effectively removed in the subsurface and as such additional assessments of the pre-treatment should be postponed until water is recovered via the ASTR extraction wells.
- Assessment of micropollutants and degradation rates. The work on micropollutants confirmed that many were present at nanogram per litre levels both in the raw and treated stormwater. Further assessment utilising passive samplers in a quantitative sense is required to obtain confidence in the robustness of the cleansing reed bed and the capability of the aquifer to attenuate these hazards.

5. QUANTITATIVE RISK ASSESSMENT METHODS

In the preliminary HACCP plan Swierc *et al.* (2005) performed an initial qualitative risk assessment of the Parafield stormwater harvesting scheme and identified chemical and microbial hazards. The following sections extend this approach by assessing risks in a quantitative framework.

5.1 Hazard identification

The preliminary HACCP plan for the Salisbury ASTR system identified physical, chemical and microbial risks (Swierc *et al.* 2005). The quantification of these risks is performed with respect to normal people, i.e. it specifically excludes immunocompromised or other groups who may have requirements different from the Australian Drinking Water Guidelines (NRMMC / EPHC 2007). These hazards are described in the three categories: chemical hazards to human health, microbial hazards to human health and hazards to the environment.

5.1.1 Chemical hazards to human health

In the HACCP plan, stakeholders had previously expressed concern about the detection of simazine and chlorpyrifos within the Parafield drain. While there were no detections in 2006, evidence of a variety of micropollutants was found throughout the Parafield stormwater harvesting system by use of passive sampling monitoring techniques (Table 6). In further assessing the value of the passive sampler micropollutant data, a screening level risk assessment was performed using assumed application rates and published half-lives for the micropollutants detected. A three-step approach was used in the screening risk assessment to determine the chemical hazards of highest risk:

1. Annual application loads for micropollutants were estimated from conservative assumed maximum usages within the catchment to determine maximum annual loads of micropollutants in the stormwater. This was calculated based on a single application to 1.6 Ha in the catchment (0.1 % of catchment land area) for each of the hazards. This assumption was based on there being no recorded usage of these micropollutants by any of the major stakeholders. Usage is confined to *ad hoc* commercial (e.g. preparation of building sites concrete slabs) or domestic (e.g. residential garden) applications.
2. Using data from the PDMS samplers and published environmental half-life data, concentrations were calculated for the water entering the in-stream basin and treated water exiting the cleansing reed bed.
3. Inlet concentrations were assumed to apply to runoff for the whole year and were used to calculate loads entering the system and these were compared to loads applied in the catchment to derive the % runoff coefficient, assuming no degradation of herbicides within the catchment.

The % runoff coefficient was calculated as a reality check for the concentrations derived from the passive samplers. Runoff coefficient values of the order of 1% were deemed to be plausible, whereas values significantly higher were deemed implausible. Using this simple decision criteria, the values obtained for diazinon, piperonyl butoxide and propiconazole appear unsupported whereas all other micropollutants had runoff % within an order of magnitude of 1%.

Assessment of the estimated final concentrations in the recovery well results in a rank list - diuron as the highest, followed by simazine and atrazine. Lastly, previous detections of simazine and chlorpyrifos by a conventional water quality monitoring program should also be considered (Swierc *et al.* 2005). Simazine has a longer half-life than atrazine, hence for the chemical hazards, three index chemicals were chosen to be further quantitatively assessed: diuron, chlorpyrifos and simazine.

A summary of the screening level risk assessment for micropollutant hazards detected at one or more locations by passive samplers from Table 6 is given in Table 17.

It is acknowledged that there are other chemical hazards that are used within the catchment and are present in the stormwater that also impact upon water quality. The list is extensive but definitive evidence in terms of hazard concentrations are still largely unknown or below conventional detection limits (Table 4). These additional hazards will be addressed on a case by case basis, with a QRA being triggered with any detection via the conventional monitoring program above the drinking water guideline values. The chemicals that are the focus of the QRA in this report were deemed the highest risk chemicals which were based on toxicity, detections in the stormwater system, availability of appropriate reference water quality guidelines and suspected uses within the catchment. Other chemicals are used in lesser amounts, are less toxic and have to date not been detected in the stormwater. The QRA forms the basis of an upper bound to the risk envelope, other unmonitored chemicals are assumed to have a lower associated risk.

Other chemical parameters of human health and environmental concern already effectively met the drinking water and ecosystem protection water quality guidelines and as such, did not warrant further a QRA. The exception to this is iron, colour and faecal indicator organisms. The risks posed by these hazards to drinking water quality are thought to be low but will still be determined during the assessment of the subsurface component to act as a final barrier in the treatment train in a future report.

Table 17 Micropollutant screening risk assessment

Chemical hazard	Chlorpyrifos	Diuron	Simazine	Atrazine	Diazinon	Oxadiazon	Endosulfan	Piperonyl butoxide	Propiconazole
Guideline value (ng/L)*	10,000	30,000	20,000	20,000	3,000	1,750	30,000	3,500	1,750
Predominant usage[a]	insecticide	herbicide	herbicide	herbicide	insecticide	herbicide	insecticide	insecticide synergist	fungicide
Area applied (Ha)[a, b]	1.6	1.6	1.6	1.6	1.6	1.6	1.6	1.6	1.6
Catchment %[a]	0.1	0.1	0.1	0.1	0.1	0.1	0.1		0.1
Application rate (g/Ha)[e]	500	50000	3000	3000	500	1000	1000		500
Total load applied (g)[c]	800	80000	4800	4800	800	1600	1600	64[f]	800
% runoff	**1.3**	**0.9**	**4.2**	**1.3**	**21**	**0.4**	**0.7**	**3,4226**	**61**
Load in runoff (g)[c]	10	745	203	62	170	6	12	21,904	492
Total annual flow (ML)[d]	1500	1500	1500	1500	1500	1500	1500	1,500	1500
Concentration at in-stream basin (ng/L)[c]	7	496	136	42	113	4	8	14,603	328
Previous detection at in-stream basin (ng/L)[d]	50		50						
Average retention time in cleansing reed bed (days)[c]	10	10	10	10	10	10	10	10	10
Log decay rate[a]	0.02	0.01	0.01	0.01	0.07	0.01	0.01	0.14	0.01
Total log removal in cleansing reed bed[c]	0.23	0.08	0.12	0.12	0.69	0.12	0.14	1.39	0.07
PDMS max concentration (ng/L)[d]	4	416	104	32	23	3	6	600	280
ED max concentration (ng/L)[d]		64	42	21					

[*] Derived guidelines using the NRMMC / EPHC (2007) approach
[a] Assumed parameter
[b] Usage data based on an assumed maximum 0.1% of the catchment (1.6 ha) for miscellaneous uses (Section 2.3.4), though detected none of these hazards have identified usages in the catchment
[c] Calculated
[d] Data from this report
[e] Application rates were based on recommendations by the Australian Pesticides and Veterinary Medical Authority (APVMA) once per year
[f] Piperonyl butoxide is an insecticide synergist, loads were estimated based on 2% of the total loads applied for chlorpyrifos, diazinon and endosulfan

5.1.2 Microbiological hazards to human health

For the microbial hazards, the hypothesis to be tested was that risk attributable to consuming water from the ASTR recovery well resulted in a disease burden of 1×10^{-6} disability adjusted life years (DALYs) per year or less. The scope of the assessment was for three 'index' pathogens, surrogates for the three microbiological groups: bacteria, protozoa and viruses as described by WHO (2004): *Campylobacter*; *Cryptosporidium parvum* and rotavirus respectively. These pathogens were selected as they contribute the greatest population health burden in terms of DALYs. Analogous to chemical hazards, these index pathogens were chosen due to their high infectivity, DALYs per case of infection and possible prevalence in stormwater. To date none of these pathogens have been detected anywhere in the Parafield stormwater harvesting system, but there remains continued detection of faecal indicators (Table 3) so their presence cannot be ruled out. These pathogens form a risk envelope and characterise the upper bound, if their risks can be adequately controlled, then other pathogens with similar characteristics but not monitored will also be under control.

5.1.3 Environmental hazards from product water

For the environmental endpoint (including soils, plants and water bodies), the NRMMC / EPHC (2006) recommends that the hazards: boron, cadmium, chlorine residuals, hydraulic loading, nitrogen, phosphorous, salinity, chloride and sodium be investigated for recycled water applications used for agricultural, municipal, residential irrigation and fire-control purposes (Table 18).

Table 18 Environmental risks for the untreated stormwater

Hazard	NRMMC / EPHC trigger value	Maximum concentration at reed bed outlet (from Table 3)
Boron	0.5 mg/L	0.093 mg/L
Cadmium	n.d.	< 0.0005 mg/L
Chlorine residual	< 0.003 mg/L	0 mg/L (chlorine not used)
Hydraulic loading	No symptoms of water logging in plants.	n/a for aquatic macrophytes
Total Nitrogen	No excessive vegetative growth, disease or delayed maturity of fruits.	1.5 mg/L
Total Phosphorous	No symptoms of iron deficiency.	0.17 mg/L
Salinity (TDS)	< 1000 mg/L	180 mg/L
Chloride	< 175 mg/L	89 mg/L
Sodium	< 115 mg/L	99 mg/L

In the ASTR project, stormwater is not intended to be utilised for irrigation and any reclaimed stormwater will be of an improved quality relative to when it was harvested from the urban catchment (Table 16). From the data available there is no indication that cadmium, nitrogen or phosphorous pose a risk either to the cleansing reed bed vegetation or if it was disposed of via the stormwater network, as concentrations are lower than the associated trigger values (Table 18). As such, as long as recovered water continues to meet the appropriate trigger values for use, risks associated with use of recycled water for agricultural, municipal, residential irrigation and fire-control purposes will remain negligible; as a result, further QRA calculations focus on human health risks posed by pathogens and chemicals.

5.2 Exposure Assessment

In order to assess the risk associated with the consumption of the index chemical and microbial hazards, the potential yearly exposure was estimated. The exposure was characterised by the probability that hazards were present in the water at the time of consumption and the distribution of the ingested dose. The dose in terms of concentration was an output of QRA model and it was further assumed that each person consumes 2 L of ASTR produced water per day (NRMMC / EPHC 2007). It is acknowledged that this exposure is high, but this value ensures that the risk assessment will be conservative. Final estimates of exposure can be evaluated by determining the mixing ratios of ASTR and mains supplied water and will be done once water has been recovered.

5.3 Dose-Response Assessment

For the chemical risk assessment, QRA model outputs were compared to appropriate drinking water quality guidelines (NRMMC / EPHC 2007). However for the microbial risk assessment, the dose-response model estimated the probability of illness resulting from a certain level of exposure. Dose response models for the infection endpoint have been developed for each of the index pathogens. The two models described by WHO (WHO 2004; NRMMC / EPHC 2007) were used: beta-Poisson and exponential. Different dose-response models are utilised depending on the specific pathogen under investigation, with constant parameters derived from observed data of published feeding trials and recommended by appropriate agencies. The beta-Poisson model was used for rotavirus and *Campylobacter,* and the exponential model was used for *Cryptosporidium* (WHO 2004; NRMMC / EPHC 2007). The dose response model for beta-Poisson is as follows:

$$\rho_{\text{inf}} = 1 - (1 + \delta/\beta)^{\alpha} \quad (9)$$

where ρ_{inf} is the probability of infection, δ is the dose and α and β are parameters fitted to the observed data. For rotavirus, α and β are 0.253 and 0.426 respectively whereas for *Campylobacter*, α and β are 0.145 and 7.58 respectively (NRMMC / EPHC 2007). The exponential dose response model is as follows:

$$\rho_{\inf} = 1 - e^{-\delta\kappa} \qquad (10)$$

where ρ_{inf} = 1 - $e^{-\delta\kappa}$ where κ is a parameter fitted to the observed data, 0.059 for *Cryptosporidium* (Teunis and Havelaar 2002; NRMMC / EPHC 2007).

5.4 Risk Calculations

For micropollutants the risk assessment used the current guideline values as endpoints. Though simplistic, they are easy to assess by regulatory bodies, and hence establishment of critical limits for CCPs is simplified. However, for the pathogens, a more sophisticated QRA model was adopted where the yearly disease burden in terms of DALYs was calculated and compared to current Australian acceptable thresholds (NRMMC / EPHC 2007).

Information and data for the development of the QRA model were obtained from water quality monitoring and literature and are summarised in Table 19 and Table 20.

Table 19 Chemical QRA model input parameters

Parameter (units)	Diuron	Simazine	Chlorpyrifos	Reference
Initial concentration in stormwater (µg/L)	(0.003, 0.03, 0.3)*	(0.05 , 0.5, 5)*	(0.005, 0.05, 0.5)*	This report, Section 3.4
Residence time in wetland (days)		(9, 12, 15)*		This report, Section 4.2
Aerobic half life (days)	(20, 70, 120)*	(10, 60, 300)*	(50, 100, 150)*	AGWR (2C)
Residence time in the aquifer (days)		(100, 200, 400)*		Pavelic *et al.* (2004)
Anaerobic half life (days)	(995, 995, ∞**)*	(∞)**	(∞)**	AGWR (2C)

Table 20 Microbial QRA input parameters

Parameter (units)	Rotavirus	*C. pavum*	*Campylobacter*	Reference
Initial concentration in stormwater (μ, σ)	(254, 303)[a]	(17, 21)[a]	(33, 19)[a]	AGWR (2B)
Residence time in wetland (days)		(9, 12, 15)*		This report, Section 4.2
Aerobic decay rate (T_{90} :days/log_{10})	(∞)**	(∞)**	(∞)**	AGWR (2C)
Residence time in the Aquifer (days)		(100, 200, 400)*		Pavelic *et al.* (2004)
Anaerobic decay rate (T_{90} :days/log_{10})	(∞)**	(∞)**	(∞)**[b]	AGWR (2C)
Dose response model	Beta-Poisson	Exponential	Beta-Poisson	NRMMC / EPHC 2007
Disease Burden (DALYs per case)	1.4×10^{-2}	1.5×10^{-3}	4.6×10^{-3}	AGWR (2A) (2007)
Population susceptibility (percentage)	6%[c]	100%	100%	NRMMC / EPHC 2007

AGWR (2C) Australian Guideline for Water Recycling: Managing Health and Environmental Risks (Phase 2C Managed Aquifer Recharge
* Triangular distribution used
** No reported degradation rate
[a] Australian Guidelines for Water Recycling Managing Health and Environmental Risks (2007) Volume 2B – Stormwater Reuse – based on reported log normal distributions
[c] As recommended for Australia in the Australian Drinking Water Guidelines (2004)

All distributions were initially assumed to have triangular functions. Triangular functions serve as an initial estimate of distributions which can be updated in subsequent revised risk assessments when additional specific information becomes available. For the chemical hazards in Table 19 initial mean concentrations of each hazard was assumed to be at the detection limit and current guideline value, with maximum values at ten times this value. For the initial pathogen numbers in stormwater, concentrations were derived from stormwater quality literature and are a log normal distribution.

The QRA model was further developed to facilitate Monte Carlo simulation, which entails generating hypothetical scenarios in terms of the values attributed to the identified factors in the exposure and dose-response assessments. The simulation represents the inherent variability in the process of stormwater harvesting, wetland treatment and subsurface storage and influence on expectant risk as well as the uncertainty in the mathematical model of the process. Ten thousand iterations were performed for each simulation, using Latin Hypercube sampling, with @RISK Industrial v4.5 [Palisade, Newfield, NY] and Microsoft Excel [Microsoft Corp., CA] software. The outcome is a statistical distribution of risk experienced by the diverse members of the population. After the QRA model was developed and results obtained, analysis and experimentation with the model were performed.

5.5 Chemical hazard risk assessment

Monte Carlo type applications and the QRA calculation is actually a relatively simple model simulated 10,000 times. An example of a QRA calculation for the chemical hazard simazine is given in Table 21. This simulation represents one possible combination of variables used in Table 19 and yields a result above current guideline value. In the example below, 1 µg/L was extracted from the triangular distribution (0.05, 0.5, 5).

Table 21 Example QRA calculation for simazine

Initial concentration in stormwater	1.0 µg/L	Concentration of simazine measured in the stormwater network, from Table 19*
Residence time in the cleansing storage	12 days	From Table 19*
Half life in the wetland (aerobic)	60 days	From Table 19*
Calculated concentration exiting the wetland	0.64 µg/L	Concentration after aerobic degradation in the wetland.
Residence time in the aquifer	200 days	From Table 19*
Half life in the aquifer (anaerobic)	n.d. days	From Table 19
Calculated concentration at extraction well	0.64 µg/L	Concentration after anaerobic degradation in the aquifer.
Compare with guideline value / detection limit	0.5 µg/L	

* Triangular distribution

For each hazard, 10,000 simulations with calculations such as in Table 21 were performed with differing parameters chosen randomly from the distributions given in Table 19. The results for the chemical risk assessment of each of the three principal hazards are given in Table 22.

The chemical risk assessment provides the expected value of risk, in this case the mean concentration of the chemical hazard and comparison to the guideline value. This is based on the guidelines being point estimates of the probability of health effects for a random individual and is consistent with the reporting of mean concentrations as reference to applicable water quality guidelines. Although this practice is useful from a regulator perspective, information regarding the range of risk experienced by the population can sometimes be lost. For this reason it is informative to report the extremes of the distribution of risk (Table 22), the 95th percentile, especially if those risks are focused on an identifiable high-risk segment of the population and are captured to some degree by the distribution of concentrations of the hazards.

Table 22 QRA results for chemical risk assessments

Hazard	Diuron	Chlorpyrifos	Simazine
Guideline value (µg/L)	0.03	0.05	0.5
Mean	0.042	0.073	0.72
5%	0.074	0.0088	0.089
Median (50%)	0.035	0.052	0.53
95%	0.98	0.19	2.0
Mode	0.036	0.018	0.74

For the three chemical hazards studied, all had predicted concentrations greater than the guideline values (NRMMC / EPHC 2007). This is because the initial assumptions used in the risk assessment were extremely conservative. For example, even though these chemicals have not been detected in the cleansing reed bed outlet, an initial concentration distribution range was chosen that exceeded the detection limit by an order of magnitude. Hence, given the inputs used for the QRA model, these chemicals pose a significant risk. It is important to note that the estimated concentrations in Table 22 do not incorporate any treatment in the subsurface. While some degradation is likely, the aquifer cannot yet be relied upon as an effective barrier until its treatment capacity has been validated though further research. Further sampling and assessment of the system will help to constrain this risk assessment further to give greater confidence in the results. Recommendations of validation monitoring, such as batch studies using aquifer core material and micropollutants, are given at the end of the report.

5.6 Microbial hazard risk assessment

The results for the microbial risk assessment are given in Table 23.

Table 23 QRA results for microbial risk assessments

Parameter	*C. pavum*	*Campylobacter*	Rotavirus
Guideline value (DALYs)		1.0×10^{-6}	
Mean	1.5×10^{-3}	4.6×10^{-3}	8.4×10^{-3}
5%	1.5×10^{-3}	4.6×10^{-3}	8.4×10^{-3}
Median (50%)	1.5×10^{-3}	4.6×10^{-3}	8.4×10^{-3}
95%	1.5×10^{-3}	4.6×10^{-3}	8.4×10^{-3}
Mode	1.5×10^{-3}	4.6×10^{-3}	8.4×10^{-3}

As with the chemical risk assessments the mean is used for the best estimate of average risk. When more data are available, the probability frequency distributions can be used. When considering the results, it is important to note that there is no one probability for illness resulting from the consumption of water, but rather there is a range of risk. As shown in Table 23, the risk for most simulations (as exemplified by the mode) is of the order of 10^{-3} DALYs. Similar to the chemical hazards, the treatment capacity of the aquifer and the cleansing reed bed was assumed to be zero until suitable pathogen decay studies have been completed as part of validation monitoring.

5.7 Sensitivity analysis

QRA coupled with stochastic Monte Carlo simulations can provide a sensitivity analysis of the factors that most significantly influence risk, i.e. those factors which are highly correlated with increased or decreased risk. This analysis helps prioritize risk management efforts and for subsequent improvement of the QRA model by focusing research priorities.

5.7.1 Chemical hazard - sensitivity analysis

A sensitivity analysis was performed by repeating the risk assessment varying only one factor (e.g. initial hazard concentration) at a time whilst fixing all others at their most likely value. The results were plotted of the mean hazard concentration as a function of changing percentile values of the distributions of each of the factors in Table 19. These results were then fitted with either a linear or exponential function using Microsoft Excel. The results are shown in Table 24.

Table 24 Sensitivity analysis of chemical hazards

Factor	Diuron	Chlorpyrifos	Simazine
Initial hazard concentration	$y = 0.04x + 0.04$	$y = 0.07x + 0.07$	$y = 0.7x + 0.7$
Wetland decay rate (half life days)	$y = 0.04e^{-0.7x}$	$y = 0.06e^{-1.0x}$	$y = 0.6e^{-1.0x}$
Wetland residence time (days)	$y = 0.04e^{-0.6x}$	$y = 0.07e^{-0.9x}$	$y = 0.8e^{-0.8x}$
Aquifer decay rate (half life days)	$y = 0.04e^{-0.4x}$	n.d.	n.d.
Aquifer residence time (days)	$y = 0.04e^{-0.4x}$	n.d.	n.d.
Example, substituting a 10% increase for the factor (from Table 19).			
Initial hazard concentration	0.045	0.077	0.77
Wetland decay rate (half life days)	0.037	0.054	0.54
Wetland residence time (days)	0.038	0.064	0.72
Aquifer decay rate (half life)	0.038	n.d.	n.d.
Aquifer residence time (days)	0.038	n.d.	n.d.

y = mean hazard concentration, x = percentile change from most likely value; n.d. not determined – no current degradation rate published

Table 24 illustrates that the final hazard concentration is linearly related to the percentile changes across the initial hazard concentration, whereas changes in each of the other factor percentiles have an exponential relationship. From the slope, the influence of each factor can be determined. Those factors with the greatest slope have the greatest influence on expectant risk. From the sensitivity analysis, it can be inferred where resources should be deployed to maximise the reduction of risk. In the current QRA model, the initial concentration of each chemical is the most sensitive factor affecting risk. The residence time with respect to the half-lives of the chemicals for both the wetland and the aquifer is very small and as such is less sensitive. Note, there is no reported anaerobic half-life for chlorpyrifos or simazine (NRMMC / EPHC 2008 2C) and as such, no sensitivity analysis of it to resultant risk could be determined. However, the half-lives of the chemicals and the residence time in the aquifer could be potentially the most sensitive again, highlighting the need for validation monitoring of the aquifer treatment barrier. This demonstrates the importance of the role that observation bores will play in managing the ASTR system in measurement of residence times for hazards and confirmation of the chemical half-lives both by monitoring and through batch experiments.

5.7.2 Microbial hazard – sensitivity analysis

The mean risks from microbial hazards were estimated to be above the guideline value for the majority of values for all three index microbial hazards (Table 23). This precludes a detailed sensitivity analysis as all calculated risks in the Monte Carlo analysis were high and contained a single value of DALYs. This is again due to the aquifer barrier not being given any treatment credit within the QRA model as there were no reliable published pathogen decay rates. Recommendations for validation monitoring of pathogen decay rates are given at the end of this report.

5.8 Worst case scenario analysis

Frequently, the worst case scenario is used as a planning tool in the regulation and formation of public policy. The QRA models can be used to assess worst case scenarios and aid in their management. This may be accomplished by the use of QRA models which allows the effects of changing distributions to be simulated with the QRA model. Changing a distribution can be used to restrict samples drawn from the distribution to values between a specified pair of percentiles. Alternatively, re-sampling can be performed by specifying a new distribution that will be sampled instead of the original distribution in the QRA model. Such analyses are performed on the previous QRA models for the chemical and microbial hazards in the following sections to generate an understanding of the possible worst case scenarios.

5.8.1 Chemical hazards – worst case scenario analysis

For each of the chemical hazards, an analysis was performed utilising the original QRA model. Bearing in mind the need to move away from intuited probabilities, selection of extreme values posed a challenge. To simulate a worst case scenario, distributional sampling for the Monte Carlo simulation was performed from the upper 95th percentile for input ranges of the initial chemical hazard concentrations in stormwater, and the lower 5th percentiles for the residence times in the aquifer, wetland, anaerobic and aerobic degradation rates (Table 19). The results of these simulations are shown in Table 25.

Table 25 QRA results for "worst case scenario" chemical risk assessments (µg/L)

Hazard	Diuron	Chlorpyrifos	Simazine
Guideline value (µg / L)	0.03	0.05	0.5
Mean	0.07	0.33	3.5
5%	0.06	0.30	3.2
Median (50%)	0.07	0.32	3.5
95%	0.08	0.36	4.0
Mode	0.07	0.30	3.2

It is informative to note that under the modelled worst case scenario, the QRA models predict that diuron, simazine, and chlorpyrifos are all above their respective guideline values for all values in the simulation. This is primarily due to their assumed lack of removal in the aquifer and low residence time relative to half-life in the cleansing reed bed.

5.8.2 Microbial hazards – worst case scenario analysis

For each of the microbial hazards a similar analysis was performed to estimate likely impacts of the worst case scenario. The scenario inputs involved distributional sampling from the upper 95th percentile for input ranges of the initial pathogen concentrations in stormwater, and the lower 5th percentiles for the residence times in the aquifer and wetland and associated aerobic and anaerobic log removal rates (Table 20). The results of the worst case scenario simulations were identical to those from the baseline simulation (Table 23), i.e. an infection resulting from every consumption of water. This again alludes to the importance of using tracer studies to more accurately determine the distribution of residence times as well as using pathogen decay chambers to determine decay rates and the utility of piezometers located in the most permeable strata within the aquifer.

6. REVISED WATER QUALITY SAMPLING TO SUPPORT THE HACCP PLAN AND IMPROVED QRA

Water quality sampling should be continued to be performed within the framework provided by the preliminary HACCP plan (Swierc *et al.* 2005) and will include the four broad objectives as described by NHMRC /EPHC (2005):

1. **Baseline monitoring.** This includes the gathering of further information that will underpin and update this risk assessment.
2. **Validation monitoring**. This includes obtaining evidence that the elements of the HACCP plan will achieve the specified performance requirements, and includes scientific investigations to improve process understandings and knowledge gaps.
3. **Operational monitoring.** This includes investigations on CCPs and control parameters to assess whether a preventive measure is operating within specified design specifications and is under control.
4. **Verification monitoring**. This includes the application of procedures, tests and other evaluations in addition to those used in operational monitoring, in order to determine compliance with the HACCP plan.

Usually, baseline monitoring is undertaken prior to the establishment of a HACCP plan, whereas validation, operational and verification monitoring are undertaken in implementing the plan. To date, monitoring has focussed on the stormwater harvesting system; future monitoring will also include the groundwater. With respect to these four water quality sampling program objectives, specific recommendations are made in the following sections. Note: this monitoring program only covers those components of the ASTR project that are currently within the scope of the HACCP plan. A list of the analytes of the revised monitoring program is given in Appendix E.

6.1 Baseline monitoring

The baseline monitoring program provides evidence that there are no detrimental effects on human health or the environment from the ASTR project. Such effects can be measured as changes in the environment or human population that have a demonstrative detrimental effect on human health or the environment or exceedances of relevant CCP limits for human health or environmental protection.

The baseline water quality monitoring can largely be continued to be addressed by monthly grab samples using the modified list of analytes presented in Appendix E.

The focus of the analytes has been changed to reflect the updated knowledge of the risks inherent in the system. There are also several recommended changes which include:

- discontinuing sediment sampling as standard analytical methods are too insensitive to detect the analytes of interest

- providing a complete revision of the micropollutant/pesticide suite in light of the insensitivity of the conventional methods.

6.2 Validation monitoring

Validation monitoring is used to determine whether the current ASTR project and HACCP plan meet health and environmental targets for drinking water supplies. Research and investigative studies and monitoring are included within the validation monitoring program and include strategic programs designed to increase understanding of the system, to identify and characterise potential hazards and to fill knowledge gaps. Recommended validation monitoring includes:

- Investigation of the subsurface system, especially with regard to the extent of mixing with ambient groundwater and residence times in subsurface storage. This is likely to be one of the dominant factors in controlling risk. This requires fully penetrating wells and piezometers in the high permeability zones.
- Measurement of micropollutant decay rates by batch studies using aquifer sediment material. This again is one of the main factors affecting risk from micropollutants. While not controllable in an operational sense it may have a direct impact to future ASTR operations, especially in determining minimum residence times for the water in the subsurface.
- Measurement of micropollutants in piezometers and recovered water using passive samplers.
- Use of pathogen diffusion chambers to determine likely decay rates of the pathogens in the cleansing reed bed and in the aquifer. Again the decay rates of pathogens have a large effect on risk.
- Continue source water monitoring to understand the temporal and spatial variability of water quality parameters to assess if the poor rainfall and low flows experienced in 2006 has resulted in atypical source water quality for the system.
- Conduct a second series of event-based sampling that characterises typical inflows and storm events in order to refine statistics of source water concentrations. These will include samplers deployed for differing periods of time to assess removal efficiency of the wetland and loads entering the subsurface.
- Use of faecal sterols to determine if the stormwater is impacted by human faecal material as an adjunct to pathogen sampling which is significantly more costly, and has had no detections to-date.
- Future recovery of the ASTR water to mains may require additional treatment such as chlorination. Treatment requirements need to be determined based on the recovered water quality as well as any potential effects that may arise from blending with mains water.

One of the objectives of validation monitoring is to prove that the system delivers the expected water quality. Therefore, operational monitoring, discussed below, is generally performed at the same time as validation monitoring especially when considering appropriate critical limits.

6.3 Operational monitoring

Operational monitoring is the routine monitoring of control parameters and CCPs identified in the ASTR project, which is used to confirm that the system is being appropriately managed. It provides advance warning that systems may be deviating to a point where control will be lost. Future operational monitoring should assess and confirm the performance of preventive measures. Key elements include:

- Addition of conductivity measurements for each of the Parafield ASR wells if they continue to supply water to the ASTR system. This will help improve the water and chloride balance for future assessment of system treatment performance and residence time in the cleansing reedbed when switching between stormwater and ASR groundwater.
- Identification and revision of the parameters and criteria at CCPs used to measure operational effectiveness and, where necessary, trigger corrective actions. Specifically a calibration review of CCPs and associated equipment should be included. This includes conductivity and turbidity probes at CCP#1, CCP#2 and future CCPs that will be utilised in the subsurface to manage water quality.
- Ongoing review and interpretation of results to confirm operational performance. Current CCPs are never breached and so tighter constraints to water quality could be considered to reduce the overall risk. This includes determining response times and suitable actions documented in a plan if the CLs are exceeded.
- Correlation of CCPs to other hazards to determine the utility of CCPS to be surrogate measures for other hazards, e.g. pathogens, micropollutants.
- Regular inspection of the ASTR facilities and maintenance of plant and equipment, including site inspections and intake structures of the water quality stations. Development of a regular inspection schedule that is auditable.

6.4 Verification monitoring

The purpose of verification monitoring is to confirm compliance with the risk management plan. Verification of water quality assesses the overall performance of the ASTR system, the ultimate quality of water being supplied or discharged.

Verification monitoring should be combined with validation during the initial operation of the ASTR project; at which stage, verification assesses whether the ASTR project is performing and validation assesses whether it will continue to perform in future. In terms of water quality monitoring, this will be performed at the recovery wells on the extracted water to confirm that it meets the Australian Drinking Water Guidelines - once full breakthrough has been established and the system is operating to produce recovered water intended for but prior to distribution.

Finally, in completing the monitoring plan, auditing is an essential part of a HACCP plan. The aim of auditing as part of verification monitoring is to verify compliance in the activities of the ASTR project; for example, to verify that operators are following the appropriate practices and that calibration schedules are being adhered to.

7. CONCLUSIONS AND RECOMMENDATIONS

A summary of the recommendations of this report with respect to fulfilment of the 12 elements of the HACCP framework described in the Australian National Guidelines for Water Recycling: Managing Health and Environmental Risks (2005) is given in Table 26. These recommendations should be adopted prior to the recovery of water to the mains system.

Table 26 Summary of recommendations

Time frame	Task	Action	Responsibility
By end winter injection (October) 2008	Continued assessment and documentation of key characteristics of the ASTR system	1. Assessment of 2007-08 performance of stormwater harvesting system by August 08. 2. Deployment of pathogen decay chambers in the Parafield wetland by August 08. 3. Review risks of new Cobbler Creek catchment area to be connected to ASTR system by August 08. 4. Sample for faecal sterols to assess source of faecal indicators by August 08.	CSIRO
	Review of flow diagram and new SCADA system	5. Produce updated flow diagram based on SCADA system for inclusion in the risk management plan by August 08.	CoS
	Establish mechanisms for operational control, determine critical control points and responses	6. Draft CLs, CCPs and emergency response plans for consideration by HACCP committee by August 08. 7. Determine CLs for all CCPs for inclusion in the risk management plan by August 08	CSIRO, CoS, SA EPA, SA DHCS, DWLBC plus others as required.
	Identify procedures required for all processes and activities applied within the ASTR system	8. Determine actions for exceedence of CCPs for input into operational plan and communication strategy by August 08.	CoS
	Continuation and strengthening of risk management (HACCP) committee	9. Attendance of HACCP meetings in July, September to agree on content and implementation of risk management plan.	CSIRO, CoS, SA EPA, SA DHCS, DWLBC plus others as required.

Time frame	Task	Action	Responsibility
	Define communication protocols with the involvement of relevant agencies and prepare a contact list of key people, agencies and stakeholders	10. Produce a draft contact list for input into the communication strategy.	CoS, SA EPA, SA DHCS, DWLBC plus others as required.
	Define potential incidents and emergencies and document procedures and response plans with the involvement of relevant agencies for input into risk management plan.	11. Determine which agencies would be contacted and associated protocol.	CoS, SA EPA, SA DHCS plus others as required.
	Document the preventive measures and strategies addressing each significant risks	12. Report on effectiveness of preventative measures for protecting water quality.	SA Water – sewer choke reporting and risks CoS – contractor pesticide usage.
By March 2009	Development of ASTR water quality policy	13. Produce first draft of a water quality policy for sign off by senior management.	CoS
	Development of ASTR risk communication strategy	14. Produce first draft of communication strategy including: a. List of key contacts b. Contacts for incidences and emergencies c. Annual reporting d. Auditing e. Community involvement	CoS
	Develop ASTR risk management plan	15. Produce first draft of an ASTR risk management plan including: a. Stakeholder contact list b. Process diagram c. CCPs and CLs d. Incident and emergency procedure e. List of required SOPs	CoS, SA Water, UWI, CSIRO
	Development of Standard Operating	16. Produce the required Standard	UWI, CoS, SA

Time frame	Task	Action	Responsibility
	Procedures (SOPs) to support the ASTR risk management plan	Operating Procedures (SOPs)	Water
	Characterise recovered water quality and residence time in the aquifer	17. Water quality sampling of recovered water quality 18. Deploy passive samplers in groundwater monitoring bores by March 09. 19. Determine residence times of water in the aquifer	CSIRO
	Establish a program for regular inspection and maintenance of all equipment, including on-line monitoring equipment	20. Determine adequate maintenance schedule for all CCPs and implement	CoS
	Establish procedures for the short-term annual review of monitoring data.	21. Determine who should be involved in short term review	CoS
Prior to recovery of to mains supplies	Determine post treatment requirements and effects of mixing recovered water with mains water. Note: This is currently not funded by any partner but was part of an NWC proposal as part of Water Proofing Northern Adelaide that requires SA Government matching support.	22. Determine recovered water quality and requirements for mixing with mains water	CSIRO
	Produce a definitive list of risk assessment / management and operational requirements to be addressed such that ASTR recovered water can be introduced into the Adelaide mains supply.	23. Production of an ultimate list of requirements that must be met prior to ASTR recovered water being introduced into the mains.	SA Water, UWI
	Establish a records management system and ensure that employees are trained to fill out records	24. Implement a document / records management system	CoS
	Review sensor calibration and maintenance program	25. Review SCADA data for all sensors and determine if sufficiently reliable/additional regular maintenance is required.	CoS
	Senior managers review the effectiveness of	26. Implement an auditing system for the	CoS

Time frame	Task	Action	Responsibility
	the management system and evaluate the need for change	risk management plan	
	Train employees and regularly test emergency response plans, investigate any incidents or emergencies and revise protocols as necessary	27. Test communication protocol and operational risk management plan	CoS
	Develop a water quality management improvement plan and ensure that the plan is communicated and implemented, and that improvements are monitored for effectiveness	28. Determine how results of the audit will lead to improvement of the risk management plan	CoS

CoS City of Salisbury; SA EPA South Australian Environmental Protection Authority; DWLBC Department of Water Land and Biodiversity; CSIRO Commonwealth Scientific and Industrial Research Organisation; SA DHCS South Australian Department of Health and Community Services.

The progress in the implementation of the risk management plan following the initial work by Swierc *et al.* (2005), but has been modified to align with the 12 elements described in the Australian National Guidelines for Water Recycling: Managing Health and Environmental Risks (2005), is shown in Figure 8.

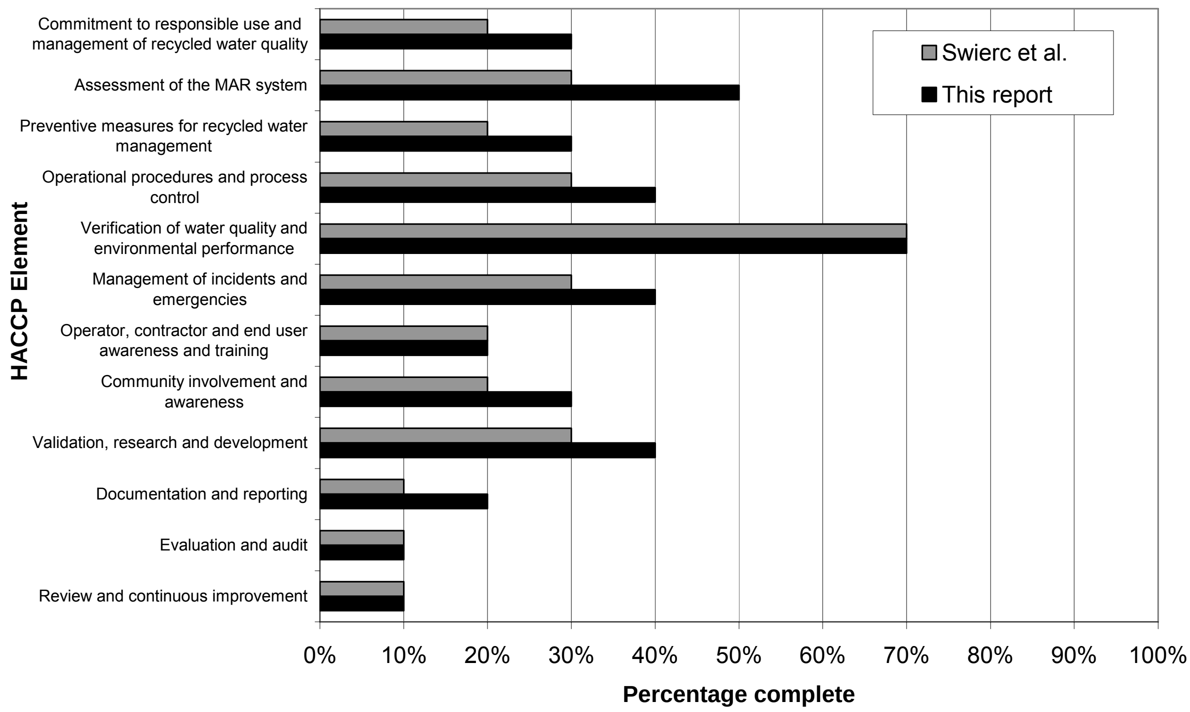

Figure 8 Progress of the ASTR plan against the 12 HACCP elements from Table 26

Figure 8 illustrates the progression of the ASTR project against each of the 12 elements. Most of the understanding of the system was gained in the quantitative risk assessment of the ASTR system (element 2), but additional work is required in establishing and documenting the operation of the system and providing timely responses that are auditable.

The next refinement of this risk assessment will include a risk management plan and will include a first evaluation of treatment effectiveness of the subsurface system. Continuation to forecast water quality changes and impacts on infrastructure by the introduction of recovered water into the mains system is recommended as an extension to the current project. Such an extension would also facilitate implementation of a risk management plan in the operational system to the ASTR project.

References

Bertrand-Krajewski, J.-L., Chebbo, G., Saget, A. (1998). Distribution of pollutant mass vs volume in stormwater discharges and the first flush phenomenon, *Water Research*, 32, pp. 2341–2356.

Bolton, K.G.E. and Greenway, M. (1999). Pollutant removal capability of a constructed *Melaluca* wetland receiving primary settled sewage, *Water, Science and Technology*, 39, pp. 199-206.

Briggs, G.G. (1981) Adsorption of pesticides on some Australian soils, *Australian Journal of Soil Research*, 19, pp. 61-68.

Brookes, J.D., Hipsey, M.R., Burch, M.D., Regel, R.H., Linden, L.G., Ferguson, C.M. and Antenucci J. (2005). Relative value of surrogate indicators for detecting pathogens in lakes and reservoirs, *Environmental Science Technology*, 39, pp. 8614-8621.

Bruno, G., Volpe, M.G., de Luise, G.E., Paolucci, M. (2006) Detection of heavy metals in farmed *Cherax Destructor*, *Bulletin Francais de la Peche et de la Pisciculture*, 380-381, pp 1341-1349.

Burges, M. (2008) Personal Communication.

Chagúe-Goff, C. (2005) Assessing the removal efficiency of Zn, Cu, Fe and Pb in a treatment wetland using selective sequential extraction: A case study, *Water, Air and soil Pollution*, 160, pp. 161-179.

Davies, C.M., Yousefi, Z. and Bavor, H.J. (2003). Occurrence of coliphages in urban stormwater ad their fate in stormwater management systems, *Letters in Applied Microbiology*, 37, pp. 299–303.

de Jong, J. (1976). The purification of wastewater with the aid of rush or reed ponds. In: Tourbier, J. and Pierson, R.W. [Eds.] *Biological Control of Water Pollution*, University of Pennsylvania Press, pp. 133-140.

Deletic, A. (1998). The first flush load of urban surface runoff, *Water Research*, 32, pp. 2462–2470.

Dillon, P. (2005). Future management of aquifer recharge, *Hydrogeology Journal*, 13, pp. 313-316.

EES, Economic Environmental Solutions (2005a). Interpretation of sediment analysis results for City of Salisbury Parafield Urban Stormwater Initiative, *Report prepared by Economic Environmental Solutions Pty. Ltd. for the City of Salisbury*, 17th February 2005.

EES, Economic Environmental Solutions (2005b). Interpretation of sediment analysis results for City of Salisbury Parafield Urban Stormwater Initiative, *Report prepared by Economic Environmental Solutions Pty. Ltd. for the City of Salisbury*, 18th April 2005.

FAO / WHO, Food Agriculture Organisation / World Health Organisation (1995). Application of Risk Analysis to Food Standards Issues, Geneva, Switzerland.

Ferrara, R.A. and Witkowski, P. (1983). Stormwater quality characteristics in detention basins, *Journal of the Environmental Engineering Division*, 109, pp. 428-447.

Finlayson, C.M. and Chick, A.J. (1983). Testing the potential of aquatic plants to treat abattoir effluent, *Water Research*, 17, pp. 415-422.

Gersberg, R.M., Elkins, B.V., Lyon, S.R. and Goldman, C.R. (1986). Role of aquatic plants in wastewater treatment by artificial wetlands, *Water Research,* 20, pp. 363-368.

Gordon, C., and Toze, S. (2003). Influence of groundwater characteristics on the survival of enteric viruses. *Journal of Applied Microbiology* 95, pp. 536-544.

Graham, R.A. (1989). Urban stormwater quality in a detention pond and wetland system. *University of Melbourne Masters Thesis*.

Hackney, P. (2007) SA Water – Personal Communication.

Hammer, D.A., Pullin, B.P. McCaskey, T.A., Eason, J. and Payne, V.W.E. (1993). Treating livestock wastewaters with constructed wetlands. In: Moshiri, G.A. [Ed.] *Constructed wetlands for water quality improvement*, Lewis Publishers, pp. 343-347.

Hey, D.L. (1982). Lake Ellyn and urban stormwater treatment. In DeGroot, W. [Ed.] Proc. *Conference on Stormwater Detention Facilities*, New England College, Henniker, New Hampshire, August, 1982, pp. 220-235.

Hipsey, M.R., Brookes, J.D., Regel, R.H., Antenucci, J., Burch, M.D. (2006). *In situ* evidence for the association of total coliforms and Escherichia coli with suspended inorganic particles in an Australian reservoir. *Water, Air and Soil Pollution*, 170, pp. 191-209

Huckins, J.N., Petty, J.D., Prest, H.F., Clark, R.C., Alvarez, D.A., Orazio, C.E., Lebo, J.A., Cranor, W.L., Johnson, B.T. (2002). A guide for the use of semi permeable membrane devices (SPMDS) as sampler of waterborne hydrophobic organic contaminants. *API Publication 4690*.

Kadlec, R.H. and Hey, D.L. (1994). Constructed wetlands for river water quality improvement. In: Bavor, H.J. and Mitchell, D.S. [Eds.] Wetland systems in water pollution control, *Water Science and Technology*, 29, pp. 159-168.

Kamedulski, G.E. and McCuen, R.H. (1979). Evaluation of alternative stormwater detention policies, *Journal of Water Resources Planning and Management Division*, 105, pp. 1146-1152.

Kapernick, A., Komarova, T., Dunn, A., Tan, B. and Mueller, J. (2006). Using passive samplers to estimate levels of herbicides, pesticides and PAHs in urban storm water drains in Adelaide, *EnTox Consultancy report to CSIRO Land and Water*, December 2006.

Kerr, R.I. and Scholes, J.D. (1979). Kananook Creek: rehabilitation strategy. *Dandenong Valley Authority Tech. Report No. 6*.

Keywood, M.D., Chivas, A.R., Fifield, L.K., Cresswell, R.G. and Ayers, G.P. (1997) The accession of chloride to the western half of the Australian continent, *Australian Journal of Soil Research*, 35, pp. 1177-89.

Khan. S. and Nugegoda, D. (2007) Sensitivity of juvenile freshwater crayfish *Cherax destructor* (Decapoda: Parastacidae) to trace metals*, Ecotoxicology and Environmental Safety*, 68, pp 463-469.

Khatiwada, N.R. and Polprasert, C. (1999). Kinetics of faecal coliform removal in constructed wetlands, *Water Science and Technology*, 3, pp. 109–116.

Komarova, T., Bartkow M. E., Müller J. F., Carter, S. and Vanderzalm J. (2006). Field evaluation of passive samplers: monitoring polyaromatic hydrocarbons (PAHs) in stormwater, *Polycyclic Aromatic Compounds,* 26, pp. 221-236.

Lager, J.A., Smith, W.G., Lynard, W.G., Finn, R.M. and Finnemore, E.J. (1977). Urban stormwater management and technology: update and users' guide. *USEPA Report EPA-600-8-77-014*.

Marks, R., Chapman, F., Lane, S. and Purdie, M. (2005). Parafield urban stormwater harvesting facility, *Water*, 32, pp. 42-45.

Medema, G.J., Tenuis, P., Havelaar, A.H., Haas, C.N. (1996). Assessment of the dose-response relationship of *Campylobacter jejuni*, *International Journal of Food Microbiology*, 30, pp. 101–111.

Meiorin, E.C. (1989). Urban runoff treatment in a fresh/brackish water marsh in Fremont, California. In: D.A. Hammer [Ed.], *Constructed Wetlands for Wastewater Treatment*, Lewis Publishers, pp. 677-685.

Mitchell, C.A. (1995). Artificial wetlands and septic tank effluent: a single household case study. *Proceedings of the AWWA 16th Convention*, Sydney, 2-6 Apr., 1995, pp. 899-906.

MMBW (1985). A preliminary investigation of the reed bed on the Bonview Main Road Drain. *Unpublished report of the Melbourne and Metropolitan Board of Works*, April, 1985.

Mueller, J. (2007) EnTox - Personal Communication

NHMRC, National Health Medical Research Council (2004). *Australian Drinking Water Guidelines*, Canberra, Australia.

NRMMC / EPHC, Natural Resource Management Ministerial Council / Environmental Protection Heritage Council (2005). *National Guidelines for Water Recycling Managing Health and Environmental Risks – Draft for Public Consultation*, Canberra, Australia.

NRMMC / EPHC, Natural Resource Management Ministerial Council / Environmental Protection Heritage Council (2007). *National Guidelines for Water Recycling Managing Health and Environmental Risks: Augmentation of Drinking Water Supplies – Draft for Public Consultation*, Canberra, Australia.

Notermans, S. and Mead, G.C. (1996). Incorporation of elements of quantitative risk analysis the HACCP system, *International Journal of Food Microbiology,* 30, pp. 157–173.

Notermans, S., Tenuis, P. (1996). Quantitative risk analysis and the production of microbiologically safe food an introduction, *International Journal of Food Microbiology*, 30, pp. 3–7.

Notermans, S., Mead, G.C., Louve, J.L. (1996). Food products and consumer protection: a conceptual approach and a glossary of terms, *International Journal of Food Microbiology*, 30, pp. 175–185.

Owens, P.N. and Walling, D.E. (2002). The phosphorus content of fluvial sediment in rural a nd industrialized river basins, *Water Research*, 36, pp. 685–701

Parkhurst, D. L. and Appelo, C. A. J. (1999). User's guide to PHREEQC-2 – A computer program for speciation, batch-reaction, one-dimensional transport and inverse geochemical calculations. *USGS Water Resources Investigations Report* 99-4259.

Patruno, J. and Russell, J. (1994). Natural wetland polishing effluent discharging to Wooloweyah Lagoon. In: Bavor, H.J. and Mitchell, D.S. [Eds.] Wetland systems in water pollution control, *Water Science and Technology*, 29, pp. 185-192.

Pavelic, P., Dillon, P. and Robinson, N. (2004). Groundwater modelling to assist well-field design and operation for the ASTR trial at Salisbury, South Australia, *CSIRO Land and Water Technical Report No. 27/04*.

Perkins, J. and Hunter, C. (2000). Removal of enteric bacteria in a surface flow constructed wetland in Yorkshire, England, *Water Research*, 34, pp. 1941–1947.

Pitman, C. and Ormsby, B. (1993). Performance of wetlands in a saline environment. *Background Papers for MFP Australia Catchment Management-Wetland Workshop*, 13-14 May, 1993.

Purdie, M. (2007) City of Salisbury – Personal Communication.

Randall, C.W., Ellis, K., Grizzard, T.J. and Knocke, W.R. (1982). Urban runoff pollutant removal by sedimentation. In: DeGroot, W. [Ed.] *Proceedings of Conference on Stormwater Detention Facilities*. New England College, Henniker, New Hampshire, August, 1982, pp. 205-219.

Reed, S.C. and Brown, D.S., (1995). Subsurface flow wetlands – a performance evaluation. Water *and Environmental Research*, 67, pp. 244–248.

Roger, S., Montrejaud-Vignoles, M., Andral, M.C., Herremans, L., Fortune, J.P. (1998). Mineral, physical and chemical analysis of the solid matter carried by motorway runoff water, *Water Research*, 32, pp. 1119–1125.

Roser, D.J., McKersie, S.A., Fisher, P.J., Breen, P.F. and Bavor, H.J. (1987). Sewage treatment using aquatic plants and artificial wetlands, *Water*, 14, pp. 20-24.

Rosich, R.S. and Cullen, P. (1979). Design and management of multi-purpose lakes. *IEA Hydrology and Water Resources Symposium*, Perth, pp. 215-219.

SA EPA, South Australian Environmental Protection Authority (2003). *Environment Protection (Water Quality) Policy and Explanatory Report 2003*, Adelaide, Australia.

SA EPA, South Australian Environmental Protection Authority (2005) EPA Guidelines for Responsible Pesticide Use, Adelaide, Australia.

SA EPA, South Australian Environmental Protection Authority (2006) Draft Code of Practice for Industrial, Retail and Commercial Sites, Adelaide Australia.

Scott, P. (2007) SA EPA - Personal communication.

Shinya, M., Tsuchinaga, T., Kitano, M., Yamada, Y., Ishikawa, M. (2000). Characterization of heavy metals and polycyclic aromatic hydrocarbons in urban highway runoff, *Water Science and Technology*, 42, pp. 201–208.

Soukup, A., Williams, R.J., Cattell, F.C.R. and Krogh, M.H. (1994). The function of a coastal wetland as an efficient remover of nutrients from sewage effluent: a case study. In: Bavor, H.J. and Mitchell, D.S. [Eds.] Wetland systems in water pollution control, *Water Science and Technology*, 29, pp. 295-304.

Stephens, B.S., Kapernick, A.P., Eaglesham, G. and Mueller, J. (2005). Aquatic passive sampling of herbicides on naked particle loaded membranes: Accelerated measurement and empirical estimation of kinetic parameters, *Environmental Science and Technology*, 39, pp. 8891-8897.

Swierc, J., Page, D.W., van Leeuwen, J.A., and Dillon, P. (2005). Preliminary Hazard Analysis and Critical Control Point (HACCP) – Salisbury stormwater to drinking water aquifer storage transfer and recovery (ASTR) project, CSIRO Land and Water Technical Report No. 20/05.

Teunis, P. F. M. and Havelaar, A. H. (2002). Risk assessment for protozoan parasites, International Biodeteriation and Biodegradation, 50, pp. 183-193.

Thompson, J.L. (1979). River Tame purification scheme: developments and design. Paper to Inst. Water Engin. and Sci., Midlands Section, UK, Jan. 1979.

Tomlinson, G.W., Fisher, A.G. and Clark, R.D.S. (1993). Stormwater - A Community Resource: "The Paddocks". EWS Report No. 93/11.

Toze, S. and Hanna J. (2002). The survival potential of enteric microbial pathogens in a treated effluent ASR project. In: *Management of Aquifer Recharge for Sustainability*. (ed) P. Dillon. Balkema Publishers Australia, pp. 139-142.

USEPA, (1983). Results of the nationwide urban runoff program. Vol. 1, Final Report. Water Planning Div. WH-554.

Wahl, M.H., McKellar, H.N., Williams, T.M. (1997). Patterns of nutrient loading in forested and urbanized coastal streams, *Journal of Experimental Marine Biology and Ecology*, 213, pp. 111–131.

Walker, T.A. and Wong, T.H.F. (1999). Effectiveness of street sweeping for stormwater pollution control, *Cooperative Research Centre for Catchment Hydrology*, December, 1999.

WHO, World Health Organisation (2004). Drinking Water Guidelines, 3rd Ed. Volume 1 Recommendations, Geneva, Switzerland.

WSSA / NWC Water Services Association of Australia / National Water Commission (2007) National performance report 2005-2006: Major urban utilities, (https://www.wsaa.asn.au/download/2007/npr2006major.pdf) accessed 23/01/08.

Yousef, Y.A., Wanielista, M.P. and Harper, H.H. (1986). Design and effectiveness of urban retention basins. In: Urbonas, B. and Roesner, L.A. [Eds.] *Urban runoff quality - impact and quality enhancement technology*, Proceedings of Conference at New England College, Henniker, New Hampshire, June 1986, pp. 338-350.

APPENDIX A –WATER QUALITY ANALYSIS METHODS

Table 27 Analytes measured as part of the 2006 monitoring program

Analyte	Method reference	Limit of reporting
General		
pH	APHA 4500-H B	0.1 (-)
Electrical conductivity	APHA 2520 B	1.0 (µS/cm and mg/L)
Total dissolved solids	APHA 2520 B	1 (mg/L)
Alkalinity	APHA 2320 B.	0 (mg/L)
Turbidity	APHA 2130 B	0.1 (NTU)
Suspended Solids	APHA 2540 D	1 (mg/L)
UV vis abs @ 254 nm	T0120-01	0.001
True Colour	T0029-01 W09-023 [a]	1 (HU)
Major Ions		
Calcium	USEPA 200.8	0.1 (mg/L)
Magnesium	USEPA 200.8	0.3 (mg/L)
Potassium	USEPA 200.8	1 (mg/L)
Sodium	USEPA 200.8	0.5 (mg/L)
Chloride	APHA 4500-Cl E.	4 (mg/L)
Bicarbonate	APHA 2320 B.	1 (mg/L)
Sulphate	APHA Inductively coupled plasma method APHA 3120 B	0.5 (mg/L)
Fluoride	APHA 4500-F- C	0.1 mg/L
Bromide	USEPA 300.0 (1993)	0.1 (mg/L)
Cyanide	APHA 4500-CN C/E	0.05 (mg/L)
Nutrients		
Total Nitrogen	APHA 4500-NO_3^- F .	0.05 (mg/L)
Total Kjeldahl Nitrogen	APHA 4500-NH3 G and 4500-Norg B	0.05 (mg/L)
Ammonia	APHA 4500-NH_3 H	0.005 (mg/L)
Nitrate plus nitrite	APHA 4500-NO_3^- F.	0.005 (mg/L)
Total organic carbon	W09-023 [a]	0.3 (mg/L)
Dissolved organic carbon	T0158-09 W09-023 [a]	0.3 (mg/L)
Biological oxygen demand (total and soluble)	APHA 5210 B	1 (mg/L)
Chemical oxygen demand (total and soluble)	APHA 5220 B	50 (mg/L)
Total phosphorous	APHA 4500-P E.	0.005 (mg/L)
Dissolved phosphate	APHA 4500-P F.	0.005 (mg/L)
Metals		
Arsenic, Antimony, Barium, Beryllium, Cadmium, Chromium, Cobalt, Copper, Lead, Lithium, Mercury, Molybdenum, Nickel, Thallium, Selenium, Vanadium	USEPA 200.8	As Cu Li 0.001 (mg/L), Sb Ba Be Cd Co Pb Mo Ni Tl Hg 0.0005 (mg/L) Cr Se V 0.003 (mg/L)
Silver	APHA Electro thermal atomic absorption method APHA 3113 B.	Ag 0.0002 (mg/L)
Chromium (VI)	T0530-51 [a]	0.01 (mg/L)
Aluminium, Boron, Iron, Manganese and Zinc	APHA Inductively coupled plasma method APHA 3120 B	Al 0.02, B 0.04, Fe 0.03, Mn 0.004 Zn 0.01 (mg/L)

Iron (soluble), Lead (soluble), Manganese (soluble)	APHA Inductively coupled plasma method APHA 3120 B	Fe 0.005, Pb 0.01, Mn 0.001 (mg/L)
Pathogen Indicators		
Colony Count at 20 C	T0083-01 WMZ-500 [a]	0 / 100mL (cfu)
Colony Count at 35 C	T0084-11 WMZ-500 [a]	0 / 100mL (cfu)
Total coliforms	T0080-07 WMZ-500 [a]	0 / 100mL (cfu)
Thermotolerant coliforms	T0076-01 WMZ-500 [a]	0 / 100mL (cfu)
Enterococci bacteria	T0482-10 WMZ-500 [a]	0 / 100mL (cfu)
f RNA bacteriophages	Enrichment - 500, 50 & 5 ml [b]	+ve or -ve
Somatic coliphages	Enrichment - 500, 50 & 5 ml [b]	+ve or -ve
Clostridium perfringens	5x100ml filtered [b]	+ve or -ve
Campylobacter Spp.	5x100ml filtered [b]	+ve or -ve
Cryptosporidium pavum	USEPA 1623	1/10 L
Pesticides		
Acidic Herbicides	In-house	0.2 (µg/L)
Organophosphate and Triazine Pesticides	USEPA, 507	parathion methyl 0.3, others 0.5 (µg/L)
Organochlorine Pesticides	USEPA, 508	Aldrin, Chlordane-a 0.01, others 0.05 (µg/L)
SEDIMENTS		
Total Petroleum Hydrocarbons	WSL 030	20 (mg/kg)
Polycyclic Aromatic Hydrocarbons	WSL 8100	0.1 (mg/kg)
Acidic Herbicides	WSL 070	0.5 (mg/kg)
Organochlorine Pesticides/PCB	WSL 8080	0.05 (mg/kg)
Organophosphate Pesticides	WSL 8140	0.5 (mg/kg)
Volatiles Scan (inc. BTEX/Halogenated Organics/Solvents)	WSL 3810 A & B	0.5 (mg/kg)
Metals (Arsenic, Cadmium, Lead, Iron, Manganese & Zinc)	WSL 023/032	Cd 0.2, Mn 0.05, As, Pb, Fe Zn 5 (mg/kg)

[a] internal reference method of the Australian Water Quality Centre, [b] in house laboratory developed method

APPENDIX B – MICROPOLLUTANTS ANALYSED BY PASSIVE SAMPLERS AND DETECTION LIMITS

Quality Control and Assurance

All EnTox laboratory procedures are performed by fully trained staff according to internally developed Standard Operation Procedures (SOPs), except in cases where procedures are still under development for research purposes. For the current project, EnTox used the following internal SOPs for the preparation and extraction of the samplers:

- ED 01 Preparation of EDs for herbicide passive sampling
- ED 04 - Extraction clean-up and analysis of EDs for herbicides
- PDMS 01 Precleaning
- PDMS 03 Extraction
- PDMS 05 Calculation of PAH
- ED 07 - Extraction and analysis of EDs for EDCs

These procedures include the use of procedural, fabrication and or field blanks that are analysed with the field samples to determine background levels of contamination associated with preparation, storage and transport of the samplers to and from the field. Additionally, the use of deuterated standards added to the samplers prior to deployment and during their extraction provides information regarding sample recoveries. Detailed guidelines on handling, storage and use of passive samplers were provided to CSIRO to maximize the quality and consistency of sampler treatment.

Blanks

CSIRO commissioned the use of one Field Blank of each sample type for this project. These blanks were also treated as a Fabrication and Procedural blanks for the purposes of standard recoveries. The field blanks traveled with the samplers to and from the sites and the PDMS blanks were exposed to air and light for the same time the deployed PDMS were.

Recovery’s & Performance Reference Compounds

EnTox used deuterated Performance Reference Compounds (PRCs) that are loaded into the ED membranes prior to exposure. These compounds are gradually removed from the samplers during deployment and provide an indication of the loss and uptake rates of the samplers. Note that there are currently no PRCs routinely loaded into PDMS before deployment. A variety of compounds and techniques are currently being trialed to determine a suitable methodology to load PDMS samplers with the standards. For ED samplers EnTox typically includes the use of deuterated diuron as a PRC. However recent work, as well as communications with other researchers in this field, indicates that the loss of chemicals from the EDs is not isokinetic and hence the use of PRCs in ED samplers is questionable. Therefore, at present, EnTox does not use a PRC based correction of the kinetics. Work is underway to introduce a novel technique for estimating the effect of flow and turbulence on the kinetics (ie the sampling rate). Surrogate standards were added to the sampler prior to extractions to monitor any analyte loss during extraction procedures and a Recovery Standard was added to the sample extracts immediately prior to analysis.

The surrogate and recovery standards allow calibration of the analyte mass measured in the sample; and allow for adjustments to be made due to sample loss and small volume variations during the extraction and analysis processes.

Table 28 Micropollutants analysed for on LC-MS that accumulate in ED samplers

Analyte	log Kow	Accumulates in Polar samplers	LOD# (ng/ED)	Equivalent Water LOD (ng/L)*
Ametryn	2.98	yes	1	0.3
Atrazine	2.61	yes	1	0.3
Diuron	2.68	yes	1	0.3
Flumeturon	2.42	yes	1	0.3
Hexazinone	1.85	yes	1	0.3
Prometryn	3.51	yes	1	0.3
Simazine	2.18	yes	1	0.3
Tebuthiuron	1.79	yes	1	0.3
Desethylatrazine	-	yes	1	0.3
desisopropylatrazine	-	yes	1	0.3

LOD- limit of detection. *Estimate for an Empore™ sampler deployed with a membrane for 30 days in standard conditions of flow and temperature.

Table 29 PAHs analysed for using GC-MS in PDMS

Analyte	log Kow	Accumulates in PDMS	LOD# (ng/PDMS)	Equivalent Water LOD (ng/L)*
Acenaphthene	4.08	Yes		0.3
Acenaphthylene	4.22	Unknown		
Anthracene	4.67	Unknown		
benz[a]anthracene	5.32	Yes		0.06
benz[a]pyrene	5.11	Yes		0.06
benz[e]pyrene	-	Yes		0.06
benzo[b+k]fluoranthene	5.55	Yes		0.06
benzo[ghi]perylene	4.04	Yes		0.3
Chrysene	5.32	Yes		0.08
dibenz[ah]anthracene	4.83	Unknown		
Fluoranthene	4.68	Yes		0.08
Fluorine	4.38	Unknown		
indeno[123cd]pyrene	4.51	Yes		0.3
Naphthalene	3.45	Yes		0.3

Analyte	log Kow	Accumulates in PDMS	LOD# (ng/PDMS)	Equivalent Water LOD (ng/L)*
Perylene	-	Yes		0.08
Phenanthrene	4.47	Yes		0.08
Pyrene	4.47	Yes		0.08

LOD- limit of detection. * Estimate for a standard PDMS deployed for 30 days in standard conditions of flow and temperature.

Table 30 Micropollutants analysed for using GC-MS and known to accumulate in PDMS strips

Analyte	log Kow	Accumulates in PDMS	LOD# (ng/PDMS)^	Equivalent Water LOD (ng/L)*
Ametrin	2.98	yes	100	22
Atrazine	2.61	yes	100	16
Bifenthrin	6.00	yes	100	2
Chlordane cis	6.00	yes	100	2
Chlordane trans	6.00	yes	100	2
Chlorfenvinphos	3.85	yes	100	<3
Chlorpyrifos	4.70	yes	100	1
Chlorpyrifos methyl	4.24	yes	100	1
Cypermethrin	6.60	yes	100	1
DDD op	6.02	yes	100	<2
DDD p,p	6.02	yes	100	2
DDDo,p	6.02	yes	100	2
DDE pp	5.69	yes	100	1
DEET	2.18	yes	100	61
Diazinon	3.30	yes	100	11
Dicofol p,p	4.28	yes	100	1
Dieldrin	4.54	yes	100	1
Diuron	2.68	yes	100	50
Endosulfan sulphate	3.66	yes	100	6
Fenamiphos	3.23	yes	100	<12
Fipronil	4.00	yes	100	3
HCB	5.50	yes	100	1
Heptachlor Epoxide	5.40	yes	100	2
Hexazinone	1.85	yes	100	<50
Metolachlor	3.13	yes	100	15

Analyte	log Kow	Accumulates in PDMS	LOD# (ng/PDMS)^	Equivalent Water LOD (ng/L)*
Nonachlor cis		yes	100	2
Nonachlor trans	6.20	yes	100	2
Oxadiazon	4.80	yes	100	1
Oxychlor	5.48	yes	100	1
Pendimethalin	5.18	yes	100	2
Permethrin	6.10	yes	100	2
Phorate	3.92	yes	100	3
Phosphate tri-n-butyl	4.00	yes	100	3
Piperonil Butoxide	4.75	yes	100	1
Pirimiphos methyl	4.20	yes	100	1
Procymidone	3.08	yes	100	16
Procymidone	3.08	yes	100	18
Prometrin	3.51	yes	100	6
Propiconazol isomers	3.72	yes	100	5
Simazine	2.18	yes	100	61
Tebuconazole	3.70	yes	100	5
Terbutrin	3.65	yes	100	6
Trifluralin	4.27	yes	100	1

LOD- limit of detection. ^ Based on analysis of a 1 mL sample extract * Estimate for a standard PDMS deployed for 30 days in standard conditions of flow and temperature.

APPENDIX C - MICROPOLLUTANTS ANALYSED IN SEDIMENTS AND DETECTION LIMITS

Table 31 Micropollutants analysed and associated detection limits in sediments

	In-stream basin (inlet)	In-stream basin (outlet)	Holding storage (inlet)	Holding storage (outlet)	Cleansing reedbed (outlet)	Cleansing reedbed (outlet)
Naphthalene; Acenaphthylene; Acenaphthene; Fluorene; Phenanthrene; Anthracene; Fluoranthene; Pyrene; Benzo(a)anthracene; Chrysene; Benzo(b)fluoranthene; Benzo(k)fluoranthene; Benzo(a)pyrene; Dibenzo(ah)anthracene; Benzo(ghi)perylene; Indeno(123-cd)pyrene	<2.0	<0.3	<0.4	<0.5	<0.1	<0.1
a-BHC; b-BHC; d-BHC; HCB; Aldrin; Chlordane; Dieldrin; Lindane; Endrin; Endrin-aldehyde; Heptachlor; Heptachlor-epoxide; DDT; DDE; DDD; Methoxychlor; Endosulphan 1; Endosulphan sulphate	<2.0	<0.3	<0.4	<0.5	<0.05	<0.05
Mevinphos; Diazinon; Malthion; Fenthion; Chlorpyrifos; Ethion; Dichlorvos; Stirofos; Ronnel; Parathion	<2.0	<0.3	<0.3	<0.5	<0.1	<0.1
Monochlorophenoxyacetic acid; 2,4-dichlorophenoxyacetic acid; 2,4,5-trichlorophenoxyacetic acid	<2.0	<0.5	<0.8	<0.5	<1.0	<0.6
Atrazine	<2.0	<0.3	<0.3	<0.5	<0.1	<0.1
Propazine; Prometryn; Ametryn; Terbutylazine; Prometon	<2.0	<0.3	<0.3	<0.5	<0.5	<0.5

NB: LOD differs across the sites due to differences in sediment matrix properties.

APPENDIX D - SUMMARY OF CASE STUDIES OF WETLANDS AND BASINS FOR TREATING STORMWATER AND WASTEWATER

	RE (1)	RE (2)	Reference
Suspended Solids	0.20 - 0.6		Lager *et al.* (1977)
[mean RE = 0.77 ± 0.23]	0.91	0.89	Kamedulski and McCuen (1980)
	0.90	0.87	Hey (1982)
	0.90	0.80 - 0.98	Randall *et al.* (1982)
	0.15	0.36	Ferrara and Witkowski (1983)
	0.91		USEPA (1983)
	0.70		Thompson (1979)
	0.64		Meiorin (1989)
	0.88	0.86 – 0.90	Kadlec and Hey (1994)
	0.64		MMBW (1985)
	0.94	0.82	Graham (1989)
	0.83		Tomlinson *et al*. (1993)
	0.98		Pitman and Ormsby (1993)
	0.94	0.98	Gersberg *et al*. (1986)
	0.91		Hammer *et al*. (1993)
	0.33 – 0.86		Kerr and Scholes (1979)
	0.83	0.84	Finlayson and Chick (1983)
	0.88	0.74 – 0.94	Roser *et al*. (1987)
	0.35	0.50	Mitchell (1995)
	0.98	0.97	Bolton and Greenway (1999)
Turbidity	0.96	0.97	Bolton and Greenway (1999)
Total Dissolved Solids	-0.49		Meiorin (1989)
[mean RE = -0.62]	-0.75		Tomlinson *et al*. (1993)
Total Organic Carbon	0.80	0.68	Kamedulski and McCuen (1980)
[mean RE = 0.42 ± 0.23]	0.32	0.14	Hey (1982)
	0.36	0.11 – 0.49	Randall *et al.* (1982)
	0.33		Tomlinson *et al*. (1993)
Biochemical Oxygen Demand (t)	0.30		Lager *et al.* (1977)
[mean RE = 0.66 ± 0.31]	0.88	0.81	Kamedulski and McCuen (1980)
	0.58	0.25	Hey (1982)
	0.64	0.60 – 0.69	Randall *et al.* (1982)
	0.69		USEPA (1983)
	0.40		Thompson (1979)
	-0.35		Meiorin (1989)
	0.57		Tomlinson *et al*. (1993)
	0.95	0.98	de Jong (1976)
	0.96	0.74	Gersberg *et al*. (1986)
	0.90		Hammer *et al*. (1993)
	0.48 – 0.62		Kerr and Scholes (1979)
	0.86	0.57 – 0.95	Roser *et al*. (1987)
	0.67	0.58	Mitchell (1995)
	0.93	0.92	Bolton and Greenway (1999)
Chemical Oxygen Demand (t)	0.34		Lager *et al.* (1977)
[mean RE = 0.50 ± 0.26]	0.67	0.45	Kamedulski and McCuen (1980)
	0.50	0.45	Hey (1982)
	0.46	0.18 – 0.67	Randall *et al.* (1982)
	0.10	0.11	Ferrara and Witkowski (1983)
	0.69		USEPA (1983)

	RE (1)	RE (2)	Reference
	0.85	0.98	de Jong (1976)
Phosphorous (t)	0.20		Lager *et al.* (1977)
[mean RE = 0.49 ± 0.31]	0.17	0.13	Kamedulski and McCuen (1980)
	0.51	0.60	Hey (1982)
	0.56	0.42 – 0.85	Randall *et al.* (1982)
	0.31	-0.11	Ferrara and Witkowski (1983)
	0.79		USEPA (1983)
	0.55		Yousef *et al.* (1986)
	0.48		Meiorin (1989)
	0.74	0.69 – 0.72	Kadlec and Hey (1994)
	0.65		Rosich and Cullen (1979)
	0.84		MMBW (1985)
	0.85	0.40	Graham (1989)
	0.67		Tomlinson *et al*. (1993)
	0.97		Pitman and Ormsby (1993)
	0.40	0.98	de Jong (1976)
	0.76		Hammer *et al*. (1993)
	0.78 – 0.85		Kerr and Scholes (1979)
	0.53	0.37	Finlayson and Chick (1983)
	0.12	-0.05 – 0.23	Roser *et al*. (1987)
	0.94		Patruno and Russell (1994)
	0.95		Soukup *et al*. (1994)
	0.04	0.13	Bolton and Greenway (1999)
Phosphorous (d)	0.65	0.50	Kamedulski and McCuen (1980)
[mean RE = 0.48 ± 0.59]	0.56	0.72	Hey (1982)
	0.71		USEPA (1983)
	0.90		Yousef *et al.* (1986)
	0.56		Meiorin (1989)
	0.91		MMBW (1985)
	0.95	0.19	Graham (1989)
	0.99		Pitman and Ormsby (1993)
	0.92 – 0.93		Kerr and Scholes (1979)
	0.46	0.28	Finlayson and Chick (1983)
	0.89		Soukup *et al*. (1994)
	0.57	-0.80	Mitchell (1995)
	-0.61	-0.39	Bolton and Greenway (1999)
Total Nitrogen	0.33	0.09 – 0.77	Randall *et al.* (1982)
[mean RE = 0.53 ± 0.23]	0.36		Yousef *et al.* (1986)
	0.78	0.52	Graham (1989)
	0.42		Tomlinson *et al*. (1993)
	0.95		Pitman and Ormsby (1993)
	0.14	0.25	Finlayson and Chick (1983)
	0.53	0.36 – 0.67	Roser *et al*. (1987)
	0.77		Patruno and Russell (1994)
	0.84	0.58	Bolton and Greenway (1999)
Ammonia	-0.69	-0.56	Hey (1982)
[mean RE = 0.46 ± 0.51]	0.82		Yousef *et al.* (1986)
	0.10		Meiorin (1989)
	0.96		MMBW (1985)
	0.98	0.94	Graham (1989)
	0.03		Tomlinson *et al*. (1993)
	0.94	0.28	Gersberg *et al*. (1986)
	0.94		Hammer *et al*. (1993)

	RE (1)	RE (2)	Reference
	0.61 – 0.84		Kerr and Scholes (1979)
	0.03	0.12	Finlayson and Chick (1983)
	0.69		Patruno and Russell (1994)
	0.92		Soukup *et al*. (1994)
	0.28		Mitchell (1995)
	0.81	0.46	Bolton and Greenway (1999)
Total Kjeldahl Nitrogen	0.38		Lager *et al.* (1977)
[mean RE = 0.46 ± 0.35]	0.20	-0.20	Ferrara and Witkowski (1983)
	0.60		USEPA (1983)
	0.11		Yousef *et al.* (1986)
	-0.28		Meiorin (1989)
	0.66		MMBW (1985)
	0.70	0.29	Graham (1989)
	0.67	0.95	de Jong (1976)
	0.91		Hammer *et al*. (1993)
	0.58 – 0.60		Kerr and Scholes (1979)
	0.56	0.37 – 0.71	Roser *et al*. (1987)
	0.63		Soukup *et al*. (1994)
Nitrate + Nitrite	0.84	0.90	Hey (1982)
[mean RE = 0.64 ± 0.32]	0.80		USEPA (1983)
	0.87		Yousef *et al.* (1986)
	0.15		Meiorin (1989)
	0.78	0.66 – 0.91	Kadlec and Hey (1994)
	0.95	0.85	Graham (1989)
	0.30		Tomlinson *et al*. (1993)
	0.92		Pitman and Ormsby (1993)
	-5.38	-0.60	Finlayson and Chick (1983)
	0.96		Patruno and Russell (1994)
	0.86		Soukup *et al*. (1994)
	0.76	0.64	Mitchell (1995)
Aluminium (t)	0.81		Tomlinson *et al*. (1993)
Cadmium (t)	0.77	0.73	Kamedulski and McCuen (1980)
[mean RE = 0.77 ± 0.05]	0.84	0.81	Hey (1982)
	0.71		USEPA (1983)
Chromium (t)	0.55		Thompson (1979)
[mean RE = 0.52 ± 0.20]	0.68		Meiorin (1989)
	0.63	0.23	Graham (1989)
Copper (t)	0.55		Thompson (1979)
[mean RE = 0.54 ± 0.25]	0.56		Yousef *et al.* (1986)
	0.33		Meiorin (1989)
	0.74	0.10	Graham (1989)
	0.85		Tomlinson *et al*. (1993)
	0.62		Pitman and Ormsby (1993)
Iron (t)	0.99		Pitman and Ormsby (1993)
Lead (t)	0.97	0.96	Kamedulski and McCuen (1980)
[mean RE = 0.87 ± 0.12]	0.89	0.87	Hey (1982)
	0.86	0.78 – 0.94	Randall *et al.* (1982)
	0.95		USEPA (1983)
	0.88		Yousef *et al.* (1986)
	0.88		Meiorin (1989)
	0.88	0.50	Graham (1989)
	0.89		Tomlinson *et al*. (1993)
	0.99		Pitman and Ormsby (1993)

	RE (1)	RE (2)	Reference
Manganese (t)	-1.11		Meiorin (1989)
	0.97		Pitman and Ormsby (1993)
Nickel (t)	0.19		Thompson (1979)
	0.20		Meiorin (1989)
Zinc (t)	0.97	0.95	Kamedulski and McCuen (1980)
[mean RE = 0.71 ± 0.26]	0.84	0.78	Hey (1982)
	0.44	0.12 – 0.73	Randall *et al.* (1982)
	0.71		USEPA (1983)
	0.53		Thompson (1979)
	0.92		Yousef *et al.* (1986)
	0.33		Meiorin (1989)
	0.93	0.28	Graham (1989)
	0.90		Tomlinson *et al*. (1993)
	0.98		Pitman and Ormsby (1993)
Thermotolerant Coliforms	0.98		Pitman and Ormsby (1993)
[mean RE = 0.98 ± 0.01]	0.98		de Jong (1976)
	0.99		Hammer *et al*. (1993)
	0.98	0.92 – 0.99	Roser *et al*. (1987)
	0.98		Patruno and Russell (1994)
	0.99	0.99	Bolton and Greenway (1999)
Faecal Streptococci	0.98		Hammer *et al*. (1993)
Atrazine	0.25 – 0.95		Kadlec and Hey (1994)

de Jong (1976) – 1.0 ha artificial pond, diluted wastewater, RE (1) hydraulic retention time 6 days, RE (2) hydraulic retention time 10 days.
Lager *et al.* (1977) - Various sedimentation basins, urban stormwater runoff.
Kerr and Scholes (1979) – 98 ha natural wetland, secondary treated wastewater.
Thompson (1979) – Detention basin, urban stormwater runoff.
Rosich and Cullen (1979) – 704 ha artificial lake, urban stormwater runoff.
Kamedulski and McCuen (1980) - 2.5 Ha lake, urban stormwater runoff from 4% of the catchment, RE (1) is a 1 in 2 year storm, RE (2) is a 1 in 10 year storm.
Randall *et al.* (1982) - Laboratory settling experiments, urban stormwater runoff, RE (1) is the mean removal efficiency for a series of seven experiments, RE (2) reports the range.
Hey (1982) - 4.1 ha detention basin, urban stormwater runoff from 1.9% of the catchment, RE (1) based on data from 91 storm events, RE (2) based on data for the complete year including baseflow.
Ferrara and Witkowski (1983) – Detention basin, urban stormwater runoff from a 258 ha catchment, 64% urban, 36% rural, RE (1) 2.5 mm storm event, RE (2) 10 mm storm event.
Finlayson and Chick (1983) – Artificial wetlands, livestock wastewater, RE (1) Typha vegetation, hydraulic retention time 2.7 days, RE (2) Phragmites vegetation, hydraulic retention time 3.6 days.
USEPA (1983) – Various detention basins, urban stormwater runoff, REs are maximums achieved on a contaminant load basis.
MMBW (1985) – Natural wetland with Typha and Phragmites vegetation, urban stormwater runoff, RE based on summer baseflows.
Gersberg *et al*. (1986) – artificial wetlands, primary treated wastewater hydraulic retention time 6 days, (RE (1) Scirpus vegetation, RE (2) Typha vegetation.
Yousef *et al.* (1986) – 1.2 ha retention basin, 6% of catchment, receiving urban stormwater runoff from a highway, REs based on average concentrations.
Roser *et al*. (1987) – various artificial wetlands, secondary treated wastewater, hydraulic retention time 3 – 11 days, RE (1) average , RE (2) range.
Graham (1989) – Various modified lakes and wetlands 0.42% of catchment, urban stormwater runoff, RE (1) hydraulic retention time 33 days, RE (2) hydraulic retention time 0.5 – 2 hours.
Meiorin (1989) – Artificial wetland, 22 ha, 1.6% of catchment, receiving urban stormwater runoff, RE based on 11 storms.
Hammer *et al*. (1993) – 0.54 ha artificial wetland, livestock wastewater.
Pitman and Ormsby (1993) – 25 ha artificial wetland, urban stormwater runoff, RER based on average concentrations.
Tomlinson *et al*. (1993) – 0.93 ha artificial wetland, urban stormwater runoff, RE based on average concentrations.
Patruno and Russell (1994) – 60 ha natural wetland, treated wastewater
Kadlec and Hey (1994) – Various artificial wetlands, 1.6 – 2.9 ha, river water from mixed urban and agricultural catchment, RE (1) is mean removal for all wetlands, RE (2) is the range of all four wetlands.
Soukup *et al*. (1994) – 90 ha natural wetland, treated wastewater.
Mitchell (1995) – artificial subsurface flow wetland, treated wastewater, RE (1) greywater, RE (2) blackwater
Bolton and Greenway (1999) – Artificial wetland, primary treated wastewater, RE (1) hydraulic loading rate 300 L. hr^{-1}, RE (2) hydraulic loading rate 600 L. hr^{-1}

APPENDIX E - SUMMARY OF REVISED MONITORING SUITE

Suite A	Suite B	Suite C
General	General	Dissolved Oxygen
Major Ions	Major Ions	Conductivity
Nutrients	Nutrients	Turbidity
Metals	Metals	Temperature
Microbiological	Microbiological	
Advanced Microbiological	Field parameters	
Micropollutants		
Field parameters		

General	Metals (total)	Major Ions	Micropollutants
conductivity	aluminium (plus dissolved)	calcium	Detergents
pH	antimony	magnesium	BTEX
		potassium	Poly aromatic
suspended solids	arsenic (plus dissolved)		hydrocarbons
alkalinity	barium	bromide	Diuron
colour	beryllium	sodium	Trifluralin
turbidity	boron	bicarbonate	Simazine
	cadmium	chloride	Atrazine
Microbiological	chromium	fluoride	Diazinon
E. Coli	cobalt	sulphate	Terbutryn
Faecal Coliforms	copper		Metolachlor
Faecal streptococci	iron	Nutrients	Chlorpyrifos
Faecal enterococci	lead	ammonia as N	Oxadiazon
	lithium	nitrate + nitrite as N	Propiconazol isomers
Advanced Microbiological	manganese	total nitrogen as N	Endosulfan sulphate
Cryptosporidium parvum	mercury	phosphorus (total)	Piperonyl butoxide
C. perfringens	molybdenum	phosphorous (dissolved)	Oxyfluorfen
Campylobacter jejuni	nickel	total organic carbon	
enteric viruses	selenium	dissolved organic carbon	
F-RNA phages	silver	UV vis abs @ 254nm	
somatic phages	thallium		
	vanadium		
Faecal sterols	zinc		

Contact Us
Phone: 1300 363 400
+61 3 9545 2176
Email: enquiries@csiro.au
Web: www.csiro.au

Your CSIRO