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Risk Assessment of Aquifer Storage Transfer and Recovery with Urban Stormwater for Producing Water of a Potable Quality

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The objective of the Parafield Aquifer Storage Transfer and Recovery research project in South Australia is to determine whether stormwater from an urban catchment that is treated in a constructed wetland and stored in an initially brackish aquifer before recovery can meet potable water standards. The water produced by the stormwater harvesting system, which included a constructed wetland, was found to be near potable quality. Parameters exceeding the drinking water guidelines before recharge included small numbers of fecal indicator bacteria and elevated iron concentrations and associated color. This is the first reported study of a managed aquifer recharge (MAR) scheme to be assessed following the Australian guidelines for MAR. A comprehensive staged approach to assess the risks to human health and the environment of this project has been undertaken, with 12 hazards being assessed. A quantitative microbial risk assessment undertaken on the water recovered from the aquifer indicated that the residual risks posed by the pathogenic hazards were acceptable if further supplementary treatment was included. Residual risks from organic chemicals were also assessed to be low based on an intensive monitoring program. Elevated iron concentrations in the recovered water exceeded the potable water guidelines. Iron concentrations increased after underground storage but would be acceptable after postrecovery aeration treatment. Arsenic concentrations in the recovered water continuously met the guideline concentrations acceptable for potable water supplies. However, the elevated concentration of arsenic in native groundwater and its presence in aquifer minerals suggest that the continuing acceptable residual risk from arsenic requires further evaluation.

Managed aquifer recharge (MAR) is a water resource management tool that is increasingly being used to facilitate water recycling in areas where water scarcity can be reduced by harvesting available stormwater and wastewater (Page et al., 2010). Previous MAR operations reported in the literature include recycling treated wastewater (Toze et al., 2010) and urban stormwater (Dillon et al., 1997; Marks et al., 2005; Dillon et al., 2008a) for horticultural and domestic irrigation to reduce the demand on groundwater resources or reticulated supply. During MAR, it has been reported that natural treatment can be achieved in the aquifer through removal of pathogens (Dillon and Toze, 2005; Page et al., 2010), nutrients (Vanderzalm et al., 2006), and micropollutants (Ying et al., 2003; Pavelic et al., 2005, 2006a); however, subsurface storage can add hazards to the stored water or create environmental risks. The evaluation of risk from MAR schemes includes studies of hydraulics (e.g., injection rates and clogging [Pavelic et al., 2007]) and salinity (recovery efficiency [Pavelic et al., 2006b]) and now incorporate a wider suite of hazards and hazardous events (NRMMC–EPHC–NHMRC, 2009a), including quantitative pathogen risk assessment (Toze et al., 2010; Page et al., 2010), the impact of geochemical reactions on inorganic chemicals and aquifer dissolution and the fate of nutrients (Vanderzalm et al., 2006; Vanderzalm et al., 2010), and micropollutants (Ying et al., 2003). Multidisciplinary projects, such as Water Reuse Foundation projects (Dillon and Toze, 2005; Vanderzalm et al., 2009), AISUWRS (Wolf et al., 2006), RECLAIM WATER in Europe (Kazner et al., 2009; Page et al., 2010), and MAR in Western Australia (Toze et al., 2010) have addressed many of these risks. However, other health and environmental risks, such as turbidity, radionuclides, contaminant migration in fractured rock aquifers, groundwater dependent ecosystems, and greenhouse gas considerations, have not been reported in the MAR literature or have not been considered in a unified risk assessment framework.

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Abbreviations: ASR, aquifer storage and recovery; ASTR, aquifer storage transfer and recovery; DALY, disability-adjusted life year; IW, injection well; MAR, managed aquifer recharge; QMRA, quantitative microbial risk assessment; RW, recovery well.

In 2006, the Australian Guidelines for Water Recycling (NRMMC–EPHC–AHMC, 2006) were released and then extended with the second phase guidelines: Phase 2A, Augmentation of Drinking Water Supplies applicable to advanced wastewater treatment for drinking water supply (NRMMC–EPHC–NHMRC, 2008); Phase 2B, Stormwater Harvesting and Reuse for nonpotable end use only (NRMMC–EPHC–NHMRC, 2009b); and Phase 2C, Managed Aquifer Recharge for recycling via the aquifer ("MAR guidelines") (NRMMC–EPHC–NHMRC, 2009a). This common holistic risk assessment framework applied to MAR, which provides a staged approach to assess the treatment capacity of the aquifer as part of the larger treatment train in water recycling with the same rigor as previously applied to engineered water treatment components, is to date unreported in the literature.

An aquifer storage transfer and recovery (ASTR) trial at Parafield in South Australia (Dillon et al., 2008b) provides a case study to apply the risk-based approach outlined in the Australian MAR Guidelines. The Parafield ASTR operation is investigating the viability of storing wetland-treated urban stormwater in a brackish aquifer for recovery at a water quality that meets potable standards and is the first scheme of this type constructed with the intention of investigating the potential of recycling stormwater for drinking water supply. The risk assessment is used to focus effort toward the highest-priority hazards commonly encountered in MAR operations and provides a rationale for further risk-based management plans. Given the aim to produce water of a potable quality, it was necessary to undertake a thorough assessment of the potential pathogen risks to human health using quantitative microbial risk assessment (QMRA). Details of the QMRA approach for assessing human health risks for water recycling via aquifers for the ASTR site are reported by Page et al. (2010).

This is the first reported application of the risk assessment framework outlined in the Australian MAR Guidelines to a case study site. The objectives of this study were (i) to document the application of the Australian MAR Guidelines to the ASTR case study site and (ii) to report the outcome of the risk assessment to human health and the environment when urban

Table 1. Summary of key hazards in source water, groundwater, and aquifer materials for managed aquifer recharge projects, with examples of specific hazards and acceptance criteria.

Hazard	Origin†	Examples	Acceptance criteria‡
Pathogens	S, (G)	viruses	$<10^{-6}$ Disability-adjusted life year per person per year for water recovered, based on additional data collection if dependent on treatment in the aquifer.
Inorganic chemicals	G, A, S	arsenic	Inorganic chemical concentrations in source water meet target for beneficial use. Geochemical modeling shows release from aquifer is unlikely to produce concentrations > target values.
Salinity and sodicity	G, (S)	salinity	Low anticipated mixing leading to acceptable recovery efficiencies.
Nutrients	S, (G)	nitrogen	Nutrient concentrations meet environmental values for aquifer beyond attenuation zone and beneficial use of recovered water.
Organic chemicals	S, (G)	pesticides	Any organic chemicals present in source water or formed in the subsurface (e.g., disinfection byproducts) are at, or attenuate to, concentrations that meet environmental values for aquifer beyond the attenuation zone and in recovered water.
Turbidity and particulates	S, (G)	suspended solids	Low risk of purge water waste stream affecting receiving environment; mobilizing clays from aquifer. Low risk of recovered water and aquifer turbidity beyond the attenuation zone not meeting required environmental values.
Radionuclides	G, A, (S)	alpha-radiation	Radioactivity of native groundwater and recharge water meets targets for beneficial use of recovered water. Low risk of release through geochemical reactions.
Pressure, flow rates, volumes and levels	S	waterlogging	Groundwater models predict defined maximum and minimum heads in injection well and other wells to be achieved by pump selection and placement. Recharged water is confined to target storage zone, and predicted upward and downward leakage is negligible. Predicted land subsidence is negligible.
Contaminant migration in fractured rock and karstic aquifers	S, (G)	PAHs§	Low potential for contamination of recharged water from other sources in the area. Attenuation zone enlarged to account for hydraulics and biodegradation and excludes groundwater-dependent ecosystems and other wells.
			Tangible evidence from nearby managed aquifer recharge operations in the same aquifer that human and environmental health is protected.
Aquifer dissolution and aquitard and well stability	S, A	excess sand recovery	Geochemical modeling shows that dissolution will not occur or is so slow that aquitard or well instability will not occur within the working life of the well (50-yr minimum).
			Clay cation exchange calculations show that dispersion and slumping of clays in the aquifer and aquitard will not occur within the working life of the well (50-yr minimum).
Impacts on groundwater-dependent ecosystems	S, A	levels outside historical range	Heads variation in groundwater dependent ecosystems are within historical range or closer to historical range than without managed aquifer recharge project. Heads do not fall below minimum levels for ecosystem maintenance, and mass/concentrations of nutrients and contaminants discharged to ecosystems are within acceptable range for indicator species present.
			Aquifer unlikely to contain stygofauna (i.e., aquifer is anaerobic or has no macropores).
Greenhouse gases	S	excessive energy use	Energy efficiency analysis has been performed, and energy use per m^3 water recovered (in lifecycle assessment) is low compared with options that meet all other criteria.

† A, aquifer minerals; G, groundwater; S, source water for recharge. Brackets show possible secondary source.

‡ Acceptance criteria are in some cases abbreviated from those in NRMMC–EPHC–NHMRC (2009a).

§ PAH, polycyclic aromatic hydrocarbon.

stormwater is recycled via ASTR with the intention of recovering it at a potable quality.

Staged Risk Assessment Methodology

The MAR risk assessment framework set out in the Australian Guidelines for Water Recycling: 2A Augmentation of Drinking Water Supplies (NRMMC–EPHC–NHMRC, 2008) and 2C Managed Aquifer Recharge (NRMMC–EPHC–NHMRC, 2009a) has been applied to the ASTR project. The risk assessment framework assessed 12 hazards common to MAR projects (Table 1) in a staged approach.

Risk Assessment Stages

The Australian MAR Guidelines suggest four steps in the risk assessment process with increasing costs of acquiring information as confidence in the viability of the project increases (Fig. 1). Managed aquifer recharge projects should be assessed as a series of stages depending on the complexity and perceived level of risk.

Stage 1 is a simple desktop study whereby all available information is used to undertake an entry-level assessment to determine if the MAR project is likely to be viable. This stage reveals the likely degree of difficulty of the project and identifies additional data that will be needed in a Stage 2 assessment. Stage 1 addresses the type and scale of the MAR project, the existence of a suitable aquifer, the availability of source water, and the intended uses of recovered water. Environmental values, management capability, and compatibility with catchment and groundwater management plans are also assessed. If the project is determined to be viable and if the investigations are to provide the information necessary for a Stage 2 risk assessment are not cost prohibitive, then the project may proceed to Stage 2.

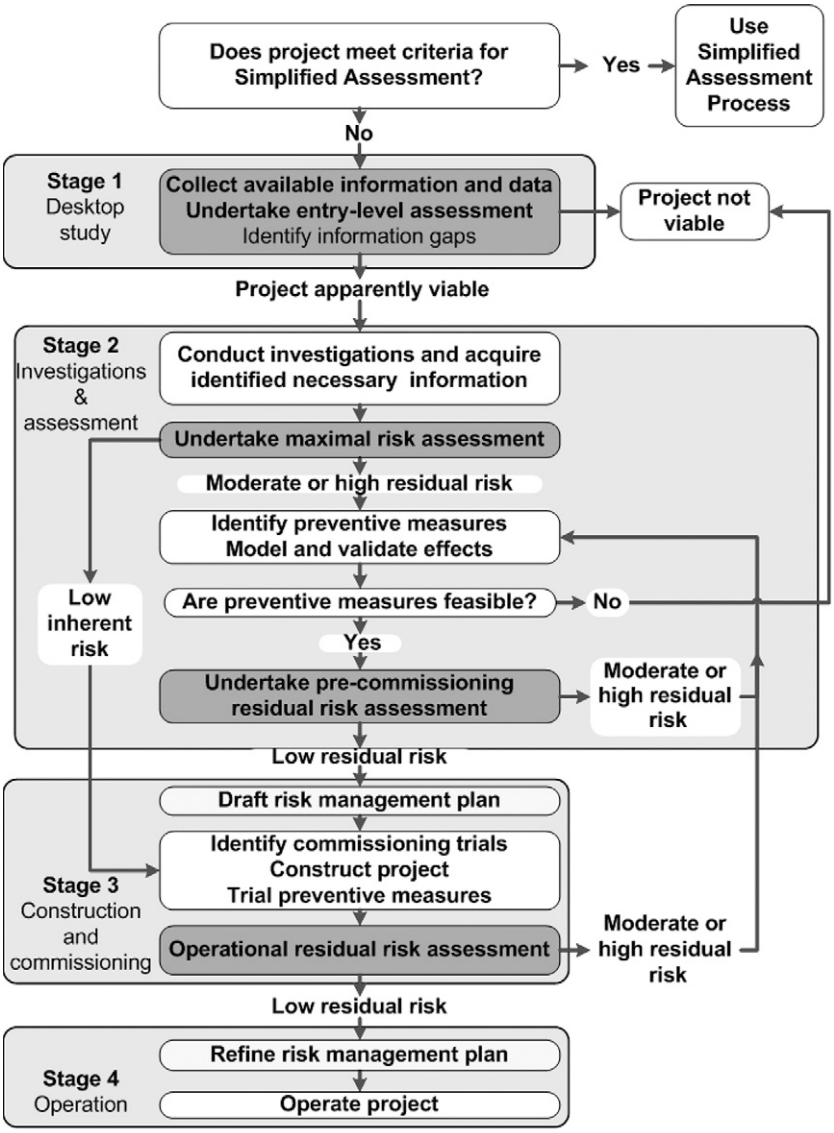

Fig. 1. Risk assessment stages in managed aquifer recharge project development (NRMMC–EPHC–NHMRC, 2009a).

Stage 2 involves the collection of additional information, such as aquifer residence time, analysis of source and native groundwater quality, and characterization of the reactive aquifer minerals. A maximal risk assessment is undertaken to estimate the risks in the absence of any controls or preventive measures. The outcomes of the maximal risk assessment are used to identify necessary preventive measures to reduce risk, such as water treatment. Stage 2 is an iterative process that can be repeated until the precommissioning residual risk is acceptable (by the addition of extra preventative measures, such as pretreatment) before moving to MAR project commissioning in Stage 3.

Stage 3 involves commissioning trials and is the main subject of this paper. The MAR scheme is trialed to validate the effectiveness of preventive measures and operational controls and to assess the suitability of recovered water for the intended use(s). The aim of Stage 3 is to identify unforeseen residual risks and the required preventative measures. If residual risks are deemed to be low, the project can move into Stage 4 operation, with a management plan and regular operational monitoring. Stage 4 includes verification monitoring performed to assess whether the quality of the recovered water is acceptable and to verify that environmental values of the aquifer are protected.

This risk assessment approach (Fig. 1) identifies 12 hazards or hazardous events to human health or the environment that should be assessed for each MAR project (Table 1). Water recycling schemes typically consider the hazards that originate in the recycled water source only, but when assessing the risks associated with MAR it is necessary to assess risks related to aquifer storage, such as increased concentrations of arsenic after dissolution of aquifer minerals (Vanderzalm et al., 2006) or impacts on the aquifer, aquitards, and other groundwater users resulting from injection or recovery pressures. Central to this approach is the concept of an attenuation zone (Fig. 2), which surrounds the zone of recharge and is the area where natural attenuation takes place so that all relevant environmental values of the aquifer will be continually met beyond the attenuation zone.

Study Site

Urban stormwater from a mixed residential and industrial catchment is harvested from the stormwater network before passing through two settling ponds and a constructed wetland (Marks et al., 2005) located at the Parafield Airport within the City of Salisbury in the Greater Adelaide Metropolitan Area in South Australia. The wetland-treated stormwater is injected into the target aquifer, a confined limestone tertiary aquifer approximately 60 m thick (from 160 to 220 m below ground) within the Port Willunga formation known as the T2 aquifer. The T2 aquifer is overlain by a 7-m-thick clay aquitard of Munno Para Clay that prevents migration of injected water to the overlying aquifers. The ASTR system is a six-well system (Fig. 3, after Kremer et al., 2008) that was progressively drilled between May 2006 and January 2007, consisting of two inner recovery wells (RW1 and RW2) and four outer injection wells (IW1–IW4), with interwell spacing of 50 m between each injection well and its nearest recovery well, providing a mean aquifer residence time of 240 d between injection and recovery (Pavelic et al., 2004; Kremer et al., 2008, 2010). The six ASTR wells are completed over an open interval of about 17 m from 165 to 182 m below ground to preclude a zone of higher hydraulic conductivity in the lower part of the aquifer, which could compromise recovery efficiencies (Kremer et al., 2008, 2010). The mineralogy in the storage zone is dominated by calcite (65 ± 23%) and quartz (30 ± 22%) with tracers of ankerite, goethite, hematite, pyrite, albite, and microcline and a low organic carbon content (<0.5%).

Between September 2006 and August 2008, 377,000 m^3 of wetland-treated stormwater was injected via the two central (RW) wells to flush the brackish aquifer and create a bubble of lower-salinity water (Kremer et al., 2008, 2010) (Fig. 4). This was followed by 32,000 m^3 of injection via the outer (IW) wells between September 2008 and April 2009 and recovery of 105,000 m^3 from the RW wells between February and April 2009.

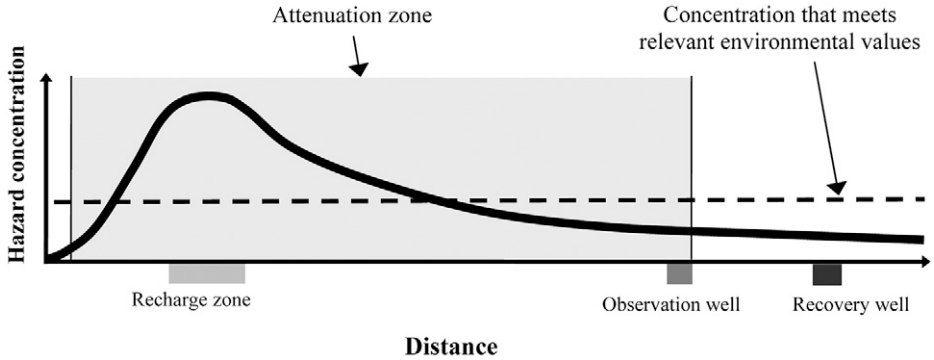

Fig. 2. Attenuation zone in an aquifer. This plot of hazard concentration on a transect through the aquifer from recharge zone to recovery well shows that an observation well on the perimeter of the predetermined attenuation zone would verify that the required attenuation is achieved within the zone.

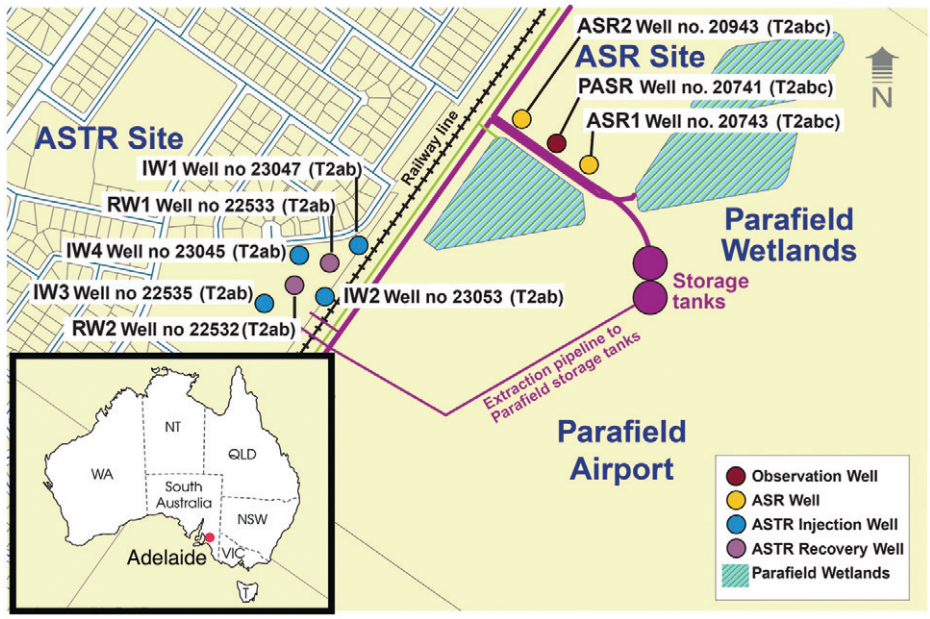

Fig. 3. City of Salisbury water harvesting facilities in the Parafield area, identifying the location of wells at the aquifer storage transfer and recovery (ASTR) and Parafield aquifer storage and recovery (ASR) sites (after Kremer et al., 2008).

Harvested Volume

During the aquifer flushing phase from September 2006 to August 2008, there was a total of 688 mm of rainfall at the study site (compared with an annual average of 453 mm), which resulted in only a total of 1,610,000 m^3 of stormwater over the 2-yr period being harvested by the Parafield system (compared with a designed maximum of approximately 1,100,000 m^3 yr^{-1}). Monthly rainfall was low and variable, with a minimum of 0 mm, a mean of 28.7 mm, and a maximum of 83.2 mm in any given month but falling predominantly within the winter months of June to August. The below-average rainfalls (76% of average) limited the quantities of stormwater that were captured by the system and resulted in the extension of the aquifer flushing period over three winters.

Water Quality Monitoring

Water quality monitoring was undertaken from 2006 to 2009 at the inlet and outlet of the wetland of the Parafield stormwater harvesting system. In June 2006, before injection at the ASTR site, ambient groundwater samples were collected from the constructed RW1, RW2, and IW3 wells for an assessment of baseline groundwater quality. Periodic groundwater sampling and down-hole water quality profiling was also performed in the ASTR wells during the flushing operations to assess the quality of water and to track the breakthrough of the freshwater plume.

Wells were sampled using a conventional monitoring pump, with three bore volumes displaced before sample collection. Temperature, electrical conductivity, dissolved oxygen, pH, and redox potential (Eh) were measured in the field using a field lab analyzer (90FLMV; TPS Pty. Ltd., Queensland, Australia), and samples were collected once these values had stabilized. The recovered water quality was based on five samples from RW1 and RW2 from February to April 2009.

All samples were collected and maintained below 4°C before analysis. Major ions, micropollutants, metals, nutrients, and microbial fecal indicators (*Escherichia coli*, thermotolerant coliforms, fecal Streptococci, Enterococci, and *Clostridium perfringens*) were analyzed within 24 h according to standard methods based on APHA–AWWA–WEF (2005). Analysis for *Campylobacter*, rotavirus, and *Cryptosporidium* were performed within 48 h according to the methods reported by Toze et al. (2010).

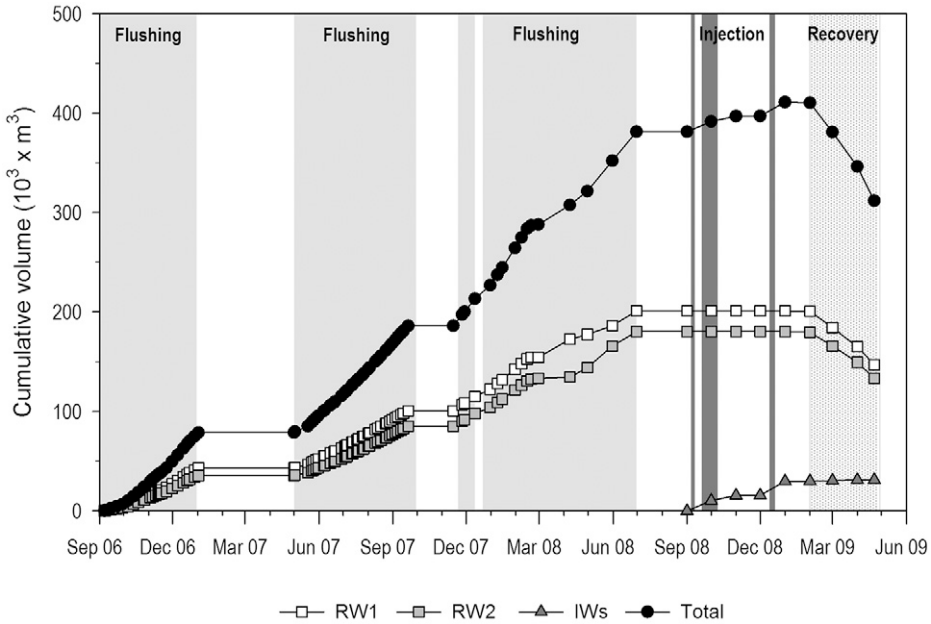

Fig. 4. Cumulative volume injected and recovered from the start of aquifer storage transfer and recovery operation in September 2006 up to April 2009. This includes flushing phase (injection into recovery wells [RWs]) from September 2006 to June 2008, commencement of injection into injection wells (IWs) in September 2008, and commencement of recovery in February 2009.

Quantitative Microbial Risk Assessment

The pathogen risk assessment of this case study site follows the approach outlined in Page et al. (2010) by estimating the pathogenic burden of disease in disability-adjusted life years (DALYs). Three representative pathogens (rotavirus, *Cryptosporidium*, and *Campylobacter*) were used to assess the risk of viruses, protozoa, and bacteria based on a tolerable risk of 10^{-6} DALYs per person per year as described in NRMMC–EPHC–NHMRC (2008, 2009a, 2009b). The Page et al. (2009) report on a QMRA for the three reference pathogens, developed based on the site-specific pathogen decay information obtained from in situ pathogen decay chamber studies (Page et al., 2010), is used to calculate the time for a 1-$\log_{10}$ reduction in pathogen numbers.

Results and Discussion

Risk assessment was pursued as the ASTR scheme progressed through the stages of project development (Fig. 1). The Stage 1 assessment completed in 2005 showed the Parafield ASTR scheme to be viable because there was supply and demand for the recycled stormwater, an existing capture and treatment system, a suitable storage aquifer, and the capability to contract and operate a project of this nature (Page et al. 2009). More detailed investigations were undertaken in Stage 2, including geochemical investigations to assess the effect of storing stormwater in the aquifer on the dissolution of minerals in the aquifer storage zone and on inorganic chemical concentrations (Vanderzalm et al., 2010), control of mixing between the injected stormwater with the brackish native groundwater to meet the salinity constraint for drinking water quality of 500 mg L^{-1} total dissolved solids (NHMRC–NRMMC, 2004), and provision of suitable aquifer residence times (>200 d) for pathogen inactivation (details of the Stage 1 assessment are reported in the Appendix of Page et al. [2009]).

During the Stage 2 assessment, the current injection well configuration was drilled (Kremer et al., 2008), groundwater and stormwater quality were investigated, and a maximal risk assessment of the untreated stormwater entering the wetland was performed (Page et al., 2008). The results of the maximal risk assessment indicated that the risks from turbidity, color, iron, and presence of fecal indicator organisms (*E. coli*, thermotolerant coliforms, Streptococci, and Enterococci) were high if the water was to be used for drinking. Additional engineered treatment may be required for recovered water if pathogen attenuation in the wetland and the aquifer is not sufficient to improve the water quality. This was investigated as the highest priority in Stage 3, the operational residual risk assessment (Page et al., 2009), when recovered water was used only for nonpotable purposes. The results of the water quality monitoring data are presented in Table 2 with respect to the Stage 3 assessment and the hazards defined by the Australian MAR Guidelines (NRMMC–EPHC–NHMRC, 2009b). Assessment of each of the risks from the 12 hazards (Table 1) is discussed in the following sections.

Pathogens

The maximal risk assessment indicated that fecal indicator organisms (thermotolerant coliforms, *E. coli*, enterococci, streptococci) were frequently detected in low numbers in the source water (Table 2). In the absence of a defined site-specific probability distribution function for pathogen numbers in the source water, the 95th percentile of pathogen numbers of 1, 1.8, and 15 n L^{-1} for rotavirus, *Cryptosporidium*, and *Campylobacter*, respectively, were taken from the stormwa-

Table 2. Water quality data for the ambient groundwater in the T2 aquifer, stormwater before and after wetland treatment, and recovered water from the Salisbury ASTR trial in comparison to Australian drinking water guideline values (mean ± SD).

	Guideline value†	Ambient groundwater in T2 aquifer (n = 3)	Wetland-treated stormwater		Recovered water (RW1 after RW2)	
		mean ± SD	Samples (n)	mean ± SD	Samples (n)	mean ± SD
Physical characteristics						
Temperature, °C		25.5 ± 0.4	10	12.3 ± 5.0	5	17.9 ± 1.0
pH	6.5–8.5	6.9 ± 0.1	10	7.1 ± 0.5	5	7.5 ± 0.6
True color, HU	15	14 ± 24	19	47 ± 26	5	23 ± 5
Pathogens						
Campylobacter, per 100 mL		na‡		na	1	>1100
Clostridium perfringens, per 100 mL		na	1	<10	1	2
Cryptosporidium, per 10 L		na		na	1	<4
Escherichia coli, *per* 100 mL)	0	0	34	34 ± 47	4	<1
Enterococci, per 100 mL	0	0	34	24 ± 28	3	<1
Fecal Streptococci, per 100 mL	0	0	34	24 ± 28	3	<1
Giardia, per 10 L		na		na	1	<5
Rotavirus		na		na	1	absent
Thermotolerant coliforms, per 100 mL	0	0	34	36 ± 46	4	<1
Inorganic chemicals						
Aluminum (soluble)	0.2	<0.02	3	0.04 ± 0.03	2	<0.01
Arsenic (total)	0.007	0.011 ± 0.001	34	0.001 ± 0.002	5	0.002 ± 0.001
Barium (total)	0.7	na	17	0.017 ± 0.003	1	0.016
Bicarbonate		317 ± 11	34	84 ± 34	5	206 ± 24
Boron (soluble)	4	na	17	0.04 ± 0.02	1	0.08
Cadmium (total)	0.002	<0.0005	34	0.0003 ± 0.0003	1	<0.0005
Calcium		135 ± 5	34	23 ± 8	5	54 ± 4
Chloride	250	920 ± 7	34	27 ± 8	5	63 ± 19
Chromium (total)	0.05 as Cr(VI)	<0.003	34	0.002 ± 0.002	1	<0.003
Copper (total)	2,1	na	13	0.004 ± 0.005	1	<0.001
Iron (total)	0.3	1.54 ± 0.06	34	0.54 ± 0.27	5	0.36 ± 0.07
Lead (total)	0.01	<0.005	34	0.0006 ± 0.0006	1	<0.0005
Magnesium		83 ± 0.3	34	4.8 ± 1.9	5	9.5 ± 1.8
Manganese (total)	0.5, 0.1	0.007 ± 0.001	34	0.04 ± 0.04	5	0.059 ± 0.002
Mercury (total)	0.001	na	15	<0.0003	1	<0.0003
Molybdenum (total)	0.05	na	16	0.001 ± 0.001	1	0.0007
Nickel (total)	0.02	<0.005	34	0.0009 ± 0.001	1	<0.0005
Potassium		13.3 ± 0.2	34	3.8 ± 1.3	5	4.7 ± 0.6
Selenium (total)	0.01	na	7	0.004 ± 0.001	1	<0.003
Silver (total)	0.1	na	7	0.0004 ± 0.0004	1	<0.0002
Sulfate	500, 250	275 ± 5	34	10.4 ± 2.6	5	21.1 ± 5.8
Zinc (total)	3	0.038 ± 0.007	33	0.02 ± 0.02	1	<0.003
Salinity/sodicity						
Electrical conductivity, μS cm^{-1}		3670 ± 15	10	230 ± 70	5	526 ± 97
Sodium	180	501 ± 5	34	18 ± 4	5	47 ± 14
Total dissolved solids	500	2020 ± 10	34	133 ± 33	3	317 ± 21
Nutrients						
Ammonia as N	0.5	0.035 ± 0.03	34	0.024 ± 0.040	4	0.14 ± 0.01
Dissolved organic C		1.4 ± 0.2	34	6.1 ± 2.3	4	4.7 ± 0.8
Nitrate + nitrite as N	11.3 (NO_3–N)	<0.005	34	0.008 ± 0.026	4	0.006 ± 0.004
Total Kjeldahl N		0.04 ± 0.03	34	0.41 ± 0.21	4	0.33 ± 0.08
Total organic C		1.4 ± 0.2	34	7.0 ± 2.5	4	4.8 ± 0.7
Total P		0.017 ± 0.005	34	0.050 ± 0.024	4	0.030 ± 0.003
Organic chemicals						
Simazine, μg L^{-1}	5	na	12	0.2 ± 0.3	4	<0.5
Turbidity/particulates						
Suspended solids		3.0 ± 0.6	34	3.5 ± 3.1	3	<1
Turbidity, NTU§	5	25 ± 4	34	4.0 ± 2.6	3	0.3 ± 0.2

† Australian Drinking Water Guidelines (NHMRC–NRMMC, 2004).

‡ na, not analyzed.

§ NTU, nephelometric turbidity units.

ter Guidelines (NRMMC–EPHC–NHMRC, 2009b) as inputs to the QMRA model. Results for the QMRA in DALYs as a function of treatment steps are shown in Fig. 5. Residual bacterial risks for the recovered water were assessed to be very low after wetland treatment and then reduced to $<<1 \times 10^{-10}$ DALYs in the recovered water due to the high decay rate reported for *Campylobacter* (Toze et al., 2010). The average $\log_{10}$ removals calculated for the aquifer were low (1.4 and 2.8 for rotavirus and *Cryptosporidium*, respectively) (Page et al., 2010); therefore, additional postrecovery treatments of chlorination and UV disinfection were included until the residual risks from protozoan and viral hazards were acceptable, estimated at 2.8×10^{-8} and 3.0×10^{-7} DALYs per person per year, respectively. Other studies (e.g., Toze et al., 2010) have calculated similar aquifer removal rates for pathogens (e.g., 2.1 $\log_{10}$ for rotavirus with a 70-d residence time) in MAR systems and discuss the limitations of this approach.

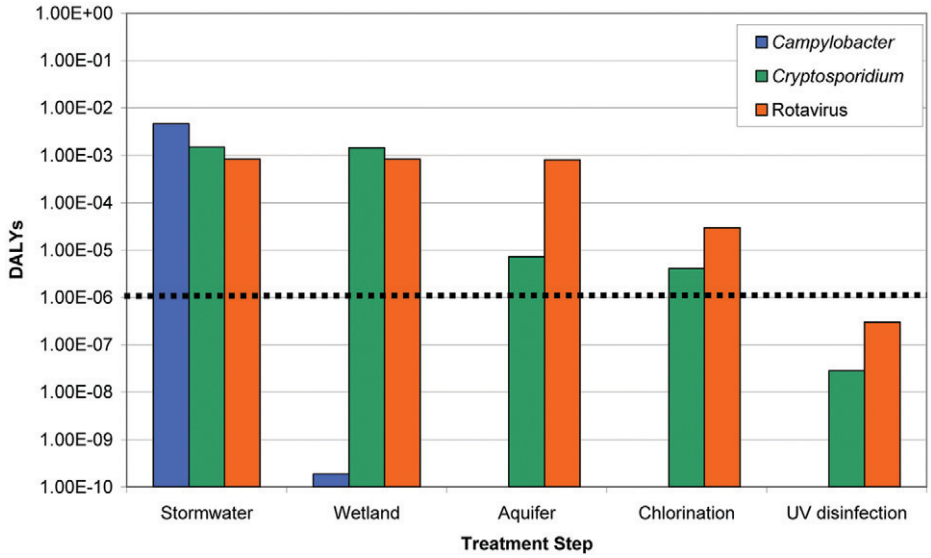

Fig. 5. Disability-adjusted life years (DALYs) as a function of the multiple barriers of the aquifer storage transfer and recovery scheme (dotted line indicates "tolerable risk" set at 1.0 × 10^{-6} DALYs per person per year).

Inorganic Chemicals

The key risks associated with inorganic chemical hazards during MAR arise from subsurface reactions, which can lead to increased concentrations of arsenic, iron, manganese, and trace species (e.g., cadmium, chromium, lead) or hydrogen sulfide in the recovered water. Stage 2 and 3 risk assessment investigations indicate that iron concentrations in the ambient groundwater (1.54 mg L^{-1}), in the wetland-treated stormwater (0.54 mg L^{-1}), and in the recovered water (0.36 mg L^{-1}) are all above the 0.3 mg L^{-1} aesthetic guideline value for iron in drinking water (Table 2). Iron oxide as hematite and goethite are significant components of the iron-bearing aquifer minerals at the study site quantified by X-ray diffraction (2.1% as Fe_2O_3), in addition to traces of pyrite and ankerite (Page et al., 2009). A lower iron concentration in the recovered water than in the wetland-treated stormwater illustrates removal of particulate iron by filtration during injection. However, the soluble iron concentration is affected by interaction between the injected water and the aquifer minerals within the storage zone (Vanderzalm et al., 2010). In this case, injection of a source water containing oxygen and organic matter can lead to increases in soluble iron due to oxidative dissolution of pyrite (under oxic conditions, immediately after injection) or reductive dissolution of Fe(III)oxides (under anoxic conditions). Both mechanisms for iron release have been reported within the T2 aquifer at nearby aquifer storage and recovery (ASR) operations using stormwater (Herczeg et al., 2004) and treated wastewater (Vanderzalm et al., 2006). The prevalence of this iron in the storage zone indicates that additional post-treatment measures, such as aeration to reduce iron concentrations, should be considered.

The arsenic concentration in the ambient groundwater (9–11 μg L^{-1}) and within the aquifer core samples (6–144 ppm) is sufficient to indicate a source of arsenic within the aquifer sediments that can lead to increased concentrations in the recovered water, as previously observed within the T2 aquifer at a nearby treated wastewater ASR site (Vanderzalm et al., 2006). Arsenic can be released under oxic and anoxic conditions. The oxidation of arsenian pyrite is expected to occur soon after injection of stormwater in the T2 aquifer (Herczeg et al., 2004); the arsenic that is released can then be controlled through sorption to iron oxides before recovery during transfer between the separate injection and recovery wells in ASTR (as opposed to ASR). However, the aquifer's capacity to remove arsenic by sorption may be affected by organic matter and phosphate in stormwater, which can result in loss or competition for sorption sites. Although the initial recovered water quality did not show any increase in arsenic and concentrations remained below the drinking water guideline value of 7 μg L^{-1} (NHMRC–NRMMC, 2004), additional groundwater quality monitoring is recommended as part of the Stage 4 assessment to confirm that the risk from arsenic remains low if the system is to be used for long-term operation as a drinking water supply.

Salinity and Sodicity

During the Stage 2 risk assessment, monitoring of the ASTR wells during the flushing phase showed that the aquifer was effectively flushed (Fig. 6), with the IW wells reaching an 85% fraction of source water (based on electrical conductivity data) in August 2008 after 377,000 m^3 of water had been recharged through the inner wells (RW1 and RW2). Predictive groundwater modeling during the Stage 3 assessment suggested that sufficient water had been injected to ensure the salinity of the recovered water (Table 2) remains below the acceptability criteria of 500 mg L^{-1} total dissolved solids for the operational scenarios evaluated (Kremer et al., 2010).

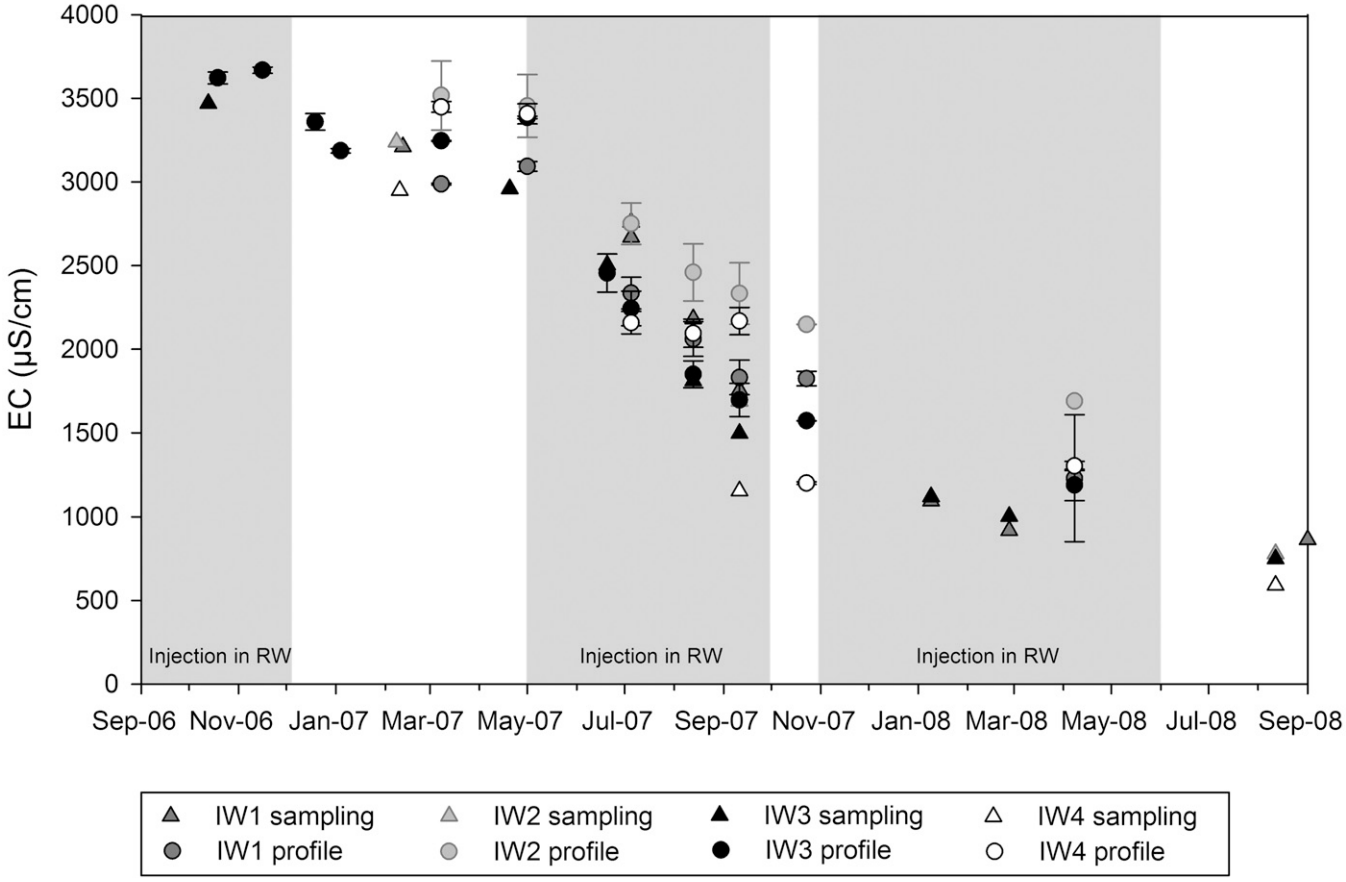

Fig. 6. Time versus depth average electrical conductivity (EC) data obtained from down-hole profiles over the opened intervals, and EC data collected during sampling, at the injection wells IW1, IW2, IW3, and IW4 during the flushing phase from September 2006 to August 2008, showing the breakthrough of source water in the aquifer. Injection periods at the recovery wells (RW) are shown with gray shading.

Nutrients

The maximal risk assessment completed as part of the Stage 2 assessment indicated that all of the nutrients in the untreated stormwater were low but could induce environmental concerns through aquifer clogging because the source water is higher in nutrient content than the receiving groundwater. After wetland treatment, the nutrient levels in the source water were considered a low risk for injection into the T2 aquifer based on previous experience in the carbonate T2 aquifer (Table 2). Further monitoring of water quality during the Stage 3 assessment confirmed that the risks from nutrients to human health and the environment were low based on the acceptance criteria in Table 1. Some removal of injected organic carbon was evident during aquifer storage and treatment, with approximately 35% dissolved organic carbon removed by microbial oxidation and sorption processes (Vanderzalm et al., 2010). Additional monitoring during Stage 4 will be used to quantify the long-term nutrient removal capacity of the subsurface treatment step for the ASTR scheme.

Organic Chemicals

Organic chemicals originating from the residential, commercial, and industrial land uses in the stormwater catchment were originally identified as part of the Stage 2 risk assessment by Swierc et al. (2005) and included light hydrocarbon fractions; benzene, toluene, ethylbenzene, and xylenes; and herbicides. During the Stage 3 risk assessment, over 300 organic chemicals were investigated as part of the monitoring program, but the majority of them were not detected (Page et al., 2008, 2009). The comprehensive monitoring suite was designed to address the potential organic chemical hazards in the source water as identified by Swierc et al. (2005) and their fate during aquifer storage and transfer, in addition to the potential for generation of disinfection byproducts in the aquifer. Simazine (6-chloro-N2,N4-diethyl-1,3,5-triazine-2,4-diamine) was the most frequently detected organic chemical at the outlet of the constructed wetland over a 3-yr period (Table 2). In 2006 simazine was not detected in any of the five grab samples (detection limit, 0.5 $\mu g\ L^{-1}$), in 2007 there were two detections of 0.46 and 0.57 $\mu g\ L^{-1}$ out of six weekly composite samples, and in 2008 there was a single detection of 0.76 $\mu g\ L^{-1}$ out of four weekly composite samples. Based on the 2007–2008 data, where a lower detection limit of 0.1 $\mu g\ L^{-1}$ was available, simazine was detected in 30% of samples. The simazine concentrations reported for the outlet of the constructed wetland are considerably lower than the Australian human health guideline value of 20 $\mu g\ L^{-1}$. Based on all of the existing data, the residual risks from organic chemicals were assessed to be low. Nonetheless, it is recommended that Stage 4 monitoring be undertaken for a reduced suite of organic chemical hazards representing a range of chemicals likely to persist in the aquifer to ensure that the risk from organic chemicals remains low.

Turbidity

Turbidity itself does not pose a human health risk, but if it is in excess of guideline values (5 nephelometric turbidity units [NTU], where drinking is an intended end use) it may interfere with disinfection performance. Although the turbidity values reported at the wetland outlet exceed this value (Table 2), turbidity levels in the recovered water after aquifer passage (0.3 NTU) were below the drinking water aesthetic guideline of 5 NTU (NHMRC–NRMMC, 2004), so the residual risk was deemed to be acceptable.

Radionuclides

The Stage 2 risk assessment found no hazardous land uses within the stormwater catchment that would lead to radionuclides entering the source water (Swierc et al. 2005). Furthermore, the T2 aquifer is considered a low-risk lithology, without granitic or coal deposits, and is low in organic carbon content (<0.5%). The residual risk assessment indicates that the potential exists for the release of radionuclides through geochemical reactions when organic matter in the source water leads to reductive dissolution of iron oxides in the aquifer sediments. During the Stage 3 assessment, gross α and β activity was measured in the recovered water in February 2009. Both measures of radioactivity remained within the Australian Drinking Water Guideline value of <0.5 Bq^{-1} (NHMRC–NRMMC, 2004). The samples taken represented wetland-treated stormwater that has been stored in the aquifer for at least 8 mo, which is a sufficient time interval to observe any potential increases in radioactivity due to hydrogeochemical reactions. As the gross α and β activity remains low, it confirms that the risk from radionuclides is low.

Pressure and Flow Rates

During the Stage 2 risk assessment, observations of drawdown in nearby wells during pump testing at the ASTR site indicated that no leakage from or to the overlying aquifer occurred. Between 2006 and 2008, the average injection flow rate observed at the RW1 and RW2 wells of 4.9 ± 1.7 L s^{-1} was stable or increased over time. Injection pressures measured at RW1 and RW2 wells did not exceed 500 kPa, the acceptable limit based on the maximum allowable injection pressure calculated using the MAR guidelines (NRMMC–EPHC–NHMRC, 2009b), which cannot induce over pressurization of the aquifer or rupture the aquitard. Similarly, drawdown is not capable of dewatering the aquitard (as determined by Kremer et al., 2008), so consolidation of compressible media and subsidence is unlikely to occur. Conceptual models defined in Kremer et al. (2008) showed that an area of 800 m radius is likely to be affected by drawdown during operations at the ASTR site. Based on groundwater monitoring, modeling, and observations from MAR operations in the T2 aquifer, the residual risks for pressure, flow rates, volume, and water levels are low.

Hazard Migration through Preferential Flow Paths

Preferential flow paths induced by high conductivity layers in the aquifer allow recharged water to travel faster than the average flow rate, resulting in residence time in the aquifer less than the mean calculated by Kremer et al. (2008) of 240 d, potentially affecting the treatment capacity of the aquifer with respect to pathogens and organic chemicals. For the Stage 2 risk assessment, the target T2 aquifer was characterized as a sandy-limestone aquifer, heterogeneous with respect to depth (Page et al., 2009), and pumping tests suggested that the flow was more likely to be through porous media than through fissures or karstic features by selecting a storage zone so as to avoid a high-permeability layer deeper in the aquifer (Pavelic et al., 2006b). Field observations and three-dimensional flow and solute transport modeling (Kremer et al., 2010) based on field data and accounting for the transmissive layer at depth suggest a low likelihood of contaminant migration in preferential flow paths along layers of higher hydraulic conductivity in the heterogeneous aquifer.

Aquifer Dissolution

The Stage 2 risk assessment indicated that recharge water (Table 2) may react with the aquifer matrix material, resulting in dissolution of carbonate minerals. Aquifer dissolution may increase the effective diameter of a well, thereby increasing yield, and may inhibit chronic clogging problems. However, aquifer dissolution can have negative effects, including collapse of uncased wells, undermining and collapse of overlying aquitards, the production of turbid water, and the development of preferential flow paths that alter aquifer residence time. The impact of aquifer dissolution on the stability of the overlying clay aquitard was considered in the Stage 3 assessment by assuming that dissolution of a 2-m radius around the injection well would result in stability concern. Page et al. (2009) estimated that the time required for dissolution of the calcite in a 2-m radius around the open interval of an ASTR injection well ranged from 120 to 200 yr, based on dissolution rates of 0.3 and 0.5 mmol L^{-1} and a total annual injection volume of 172,000 m^3 yr^{-1} expected under average rainfall conditions (Kremer et al., 2010), with 43,000 m^3 yr^{-1} injected into each IW well. These calculations indicate that aquifer dissolution is not a risk to the lifetime of the injection wells and hence the risk for aquifer dissolution and stability is low. However, whenever pumps are replaced, or at 10-yr intervals, it is recommended that caliper logs are run to verify the rates of dissolution and that monitoring enables mass balance calculations to estimate the mass of calcite dissolution occurring.

Aquifer and Groundwater Dependent Ecosystems

Managed aquifer recharge can affect groundwater-dependent ecosystems (e.g., stygofaunal assemblages) and connected rivers and wetlands by raising or lowering the water table, changing nutrient cycles, and introducing hazards to the system. The Stage 2 assessment revealed that there are no surface water ecosystems connected to the T2 aquifer within 10 km of the ASTR site. Furthermore, there are unlikely to be populations of stygofauna in the storage zone due to the depth, salinity, anoxic conditions, and lack of karst features. Remote connections to recharge sources leads to low nutrient availability and hence to low stygofauna populations (Tomlinson and Boulton, 2008). As part of the Stage 3 assessment, sampling of stygofaunal communities was performed, and none was detected. Therefore, the residual risk to groundwater-dependent ecosystems is deemed to be acceptable.

Energy and Greenhouse Gases

Energy consumption in the provision of water supplies comes from the treatment of water and pumping from source to treatment site to end user (Kenway et al. 2008). The risk assessment involved comparing the energy requirements and greenhouse gas emissions for the ASTR site on a per volume basis with those from alternative sources of water. The Stage 2 assessment simply considered that the sourcing of stormwater close to the ASTR site and end users would consume less energy than pumping from the River Murray and through the Mt Lofty Ranges. The Stage 3 assessment considered the 2008 volume and energy consumption data at the Parafield stormwater harvesting system and ASTR well field. Based on a recovery efficiency of 90%, the ASTR scheme consumes approximately 2.7 MJ m^{-3} of water produced (including distribution to end users). This compares with the energy cost of water supply from the River Murray and Mt Lofty Ranges catchments with conventional treatment (coagulation, filtration, and disinfection) and distribution, which varies from 3.5 MJ m^{-3} (50% River Murray water) to 6.9 MJ m^{-3} (90% River Murray water) (Kenway et al., 2008; SA Water Corporation, 2007). Seawater desalination with recovery and distribution typically consumes more than 14.4 MJ m^{-3} (Kenway et al., 2008). Because energy consumed per unit of water produced is less for the ASTR project than for current urban water supplies and desalination, the risks of excess energy consumption and greenhouse gas emissions are low.

Conclusions

The Australian MAR risk assessment framework presented in this paper is the first reported internationally to provide a staged approach to managing the risks typically associated with water recycling systems, which include an aquifer component. The ASTR project evaluated whether recycled urban stormwater could meet standards for potable quality. The QMRA indicated that the risks to human health from viral and protozoan pathogens were potentially high and that further postrecovery treatment would be required. This QMRA was limited by inadequate characterization of pathogen numbers in source water and thus uses a conservative approach of taking 95th percentile numbers from stormwater harvesting and reuse guidelines. The MAR staged risk assessment demonstrated that, although the risks from organic chemicals, turbidity, and inorganic chemicals were acceptable in the early stages of operation of the ASTR scheme, the assessment is also uncertain. In a longer-term assessment, these risks need be to better characterized to reduce uncertainty and to better define the residual risks from these hazards. Further monitoring and assessment by undertaking Stage 4 of the MAR risk assessment will confirm this through the adoption and verification of a risk management plan.

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