Available at www.sciencedirect.com

ScienceDirect

journal homepage: www.elsevier.com/locate/watres

Valuing the subsurface pathogen treatment barrier in water recycling via aquifers for drinking supplies

Declan Page [a,b,*], *Peter Dillon* [a,b], *Simon Toze* [b,c], *Davide Bixio* [d,1], *Bettina Genthe* [e], *Blanca Elena Jiménez Cisneros* [f], *Thomas Wintgens* [g]

[a] *CSIRO Water for a Healthy Country, Private Bag No. 2, Glen Osmond, SA 5064, Australia*
[b] *CSIRO Water for a Healthy Country, Queensland Bioscience Precinct, 306 Carmody Road, St Lucia, Brisbane, QLD 4067, Australia*
[c] *Water for a Healthy Country National Research Flagship, Canberra, Australia*
[d] *Aquafin NV, Dijkstraat 8, B-2630 Aartselaar, Belgium*
[e] *CSIR – Council for Scientific and Industrial Research, P.O. Box 320, Stellenbosch 7599, South Africa*
[f] *UNAM – Universidad Nacional Autónoma de México, Instituto de Ingeniería, Circuito Escolar, Ciudad Universitaria, 04510 México, D.F., México*
[g] *Fachhochschule Nordwestschweiz, Hochschule für Life Sciences, Institut für Ecopreneurship, Gründenstrasse 40, CH-4132 Muttenz, Switzerland*

ARTICLE INFO

Article history:
Received 8 October 2009
Received in revised form
30 November 2009
Accepted 6 December 2009
Available online 16 December 2009

Keywords:
Managed aquifer recharge
Aquifer treatment
QMRA
Water recycling

ABSTRACT

A quantitative microbial risk assessment (QMRA) was performed at four managed aquifer recharge (MAR) sites (Australia, South Africa, Belgium, Mexico) where reclaimed wastewater and stormwater is recycled via aquifers for drinking water supplies, using the same risk-based approach that is used for public water supplies. For each of the sites, the aquifer treatment barrier was assessed for its $\log_{10}$ removal capacity much like for other water treatment technologies. This information was then integrated into a broader risk assessment to determine the human health burden from the four MAR sites. For the Australian and South African cases, managing the aquifer treatment barrier was found to be critical for the schemes to have low risk. For the Belgian case study, the large treatment trains both in terms of pre- and post-aquifer recharge ensures that the risk is always low. In the Mexico case study, the risk was high due to the lack of pre-treatment and the low residence times of the recharge water in the aquifer. A further sensitivity analysis demonstrated that human health risk can be managed if aquifers are integrated into a treatment train to attenuate pathogens. However, reduction in human health disease burden (as measured in disability adjusted life years, DALYs) varied depending upon the number of pathogens in the recharge source water. The beta-Poisson dose response curve used for translating rotavirus and *Cryptosporidium* numbers into DALYs coupled with their slow environmental decay rates means poor quality injectant leads to aquifers having reduced value to reduce DALYs. For these systems, like the Mexican case study, longer residence times are required to meet their DALYs guideline for drinking water. Nevertheless the results showed that the risks from pathogens can still be reduced and recharging via an aquifer is safer than discharging directly into surface water bodies.

* *Corresponding author*. CSIRO Water for a Healthy Country, Private Bag No. 2, Glen Osmond, SA 5064, Australia.
E-mail address: Declan.Page@csiro.au (D. Page).
[1] Current address: European Commission, 170 Rue de la Loi/Wetstraat, 1040 Brussels, Belgium.

0043-1354/$ – see front matter © 2009 Elsevier Ltd. All rights reserved.
doi:10.1016/j.watres.2009.12.008

1. Introduction

Water reuse is increasingly regarded as an appropriate and cost effective option for augmentation of urban water supply needs (NRMMC-EPHC-AHMC, 2008). Drivers for the increased reuse of water include severe water shortages in dry periods, climate change, stricter regulations on waste discharge to the receiving environment and growing urban populations. Furthermore, unintentional water reuse may also exist as a result of lack of sanitation (Jimenez and Asano, 2008), and limited wastewater treatment facilities.

Climate change and increasing urbanisation has had a detrimental effect on groundwater resources which has resulted in an increasing worldwide interest in the recharge of aquifers for augmenting urban drinking water supplies (Dillon, 2005). Aquifer recharge can utilise a variety of non-traditional source waters including urban stormwater and reclaimed water from sewage effluent which may contain hazards to human health. The role of the aquifer in the treatment train has not been considered with the same rigor as engineered components such as filtration or disinfection, even though it may lead to large improvements in water quality (Dillon and Toze, 2005). It has been documented that pathogens are removed during passage through aquifers (Yates et al., 1990; Nasser and Oman, 1999; Gordon and Toze, 2003; Toze et al., 2004) yet this information is often still to be incorporated into risk management plans that acknowledge the role of aquifers as active treatment systems. Consequently, many jurisdictions do not integrate the subsurface treatment into the entire risk management strategy for potable water supplies.

With new approaches such as water recycling via aquifers, sound risk management becomes even more important. Australia has been active in developing new approaches to managing risks associated with recycled water quality. In 2006, the National Water Quality Management Strategy added the Australian Guidelines for Water Recycling: Managing Health and Environmental Risks (Phase 1) (NRMMC-EPHC-AHMC, 2006) and subsequently in 2008/09 three supplementary guidelines, the Australian Guidelines for Water Recycling: Managing Health and Environmental Risks (Phase 2A: Augmentation of Drinking Water Supplies. Phase 2B: Stormwater Harvesting and Reuse and Phase 2C: Managed Aquifer Recharge, 2009) (NRMMC-EPHC–NHMRC, 2008, 2009a, 2009b). These guidelines form the basis of an integrated methodology for managing human health risks using quantitative microbial risk assessment (QMRA) and provide guidance and acceptability criteria for a range of risks common across many managed aquifer recharge (MAR) configurations. These efforts parallel international developments in the World Health Organisation Water Safety Plans (WHO, 2004, 2005).

In other countries such as Mexico, there is extensive use of wastewater for irrigation, some of which infiltrates into the underlying aquifers that are used as drinking sources (Jimenez and Chávez, 2004). It is, therefore, important to assess the risks of these practices to human health and to move from unintentional reuse to managed systems. In this regard, local standards to promote and control aquifer recharge have been proposed (e.g. NOM-014-CNA, 2003). Similarly, the European Commission 'RECLAIM WATER' project was developed to share knowledge on current practices at selected aquifer recharge sites (Kopac et al., 2007; Le Corre et al., 2007), with the aim to develop sound risk-based management approaches to aquifer recharge.

Hence the objectives of this paper are:

- to determine the value of the aquifer treatment barrier for pathogens at four drinking water case study sites, reusing both effluent (treated and untreated) and stormwater via the subsurface;
- to perform a quantitative microbial risk assessment on the case study sites which use water reclamation via aquifers to augment a potable supply;
- to standardise the valuing of the aquifer in relation to the other engineered treatment barriers;
- to develop an approach for integrating aquifer treatment with other treatment systems in assessment of drinking water supplies.

2. Case study sites

This study considers four case studies that form part of the larger RECLAIM WATER project. Each site utilises a non-traditional water source and an engineered water treatment train coupled to an aquifer recharge system for augmenting urban drinking water supplies. The four case studies: Tula Valley (Mexico); Parafield (Australia); Atlantis (South Africa) and Torreele/St-André (Belgium) treatment train diagrams are shown in Fig. 1. Each treatment train was assessed using a quantitative microbial risk assessment approach and the aquifer treatment contribution compared across the four case study sites. Special attention has been given to the contribution of the aquifer barrier within the broader treatment train and its importance in managing human health risks.

The treatment trains and important attributes of the four case studies are summarised in Table 1. These range from primary treatment with almost total reliance on the subsurface passage and residence time for water quality improvement at Tula Valley to advanced tertiary treatment where there is no reliance on the aquifer (Torreele/St-André) for water quality improvement. At Parafield and Atlantis, the aquifer plays an important complementary role to the engineered treatment systems. Even though the case study sites have very different treatment trains, seven key system components of MAR can be identified (Table 1). Each site is further described briefly below.

The Tula valley (also known as Mezquital valley) site is located 100 km north of Mexico City and has received untreated wastewater from Mexico City since 1986. The Tula valley is a semiarid area with an expanded economy due to the availability of wastewater used for irrigation (Jimenez, 2004). It has been estimated that $\sim$60 m^3/s of wastewater are used for irrigation in the area and as a result the local aquifer is being recharged with at least $\sim$25 m^3/s due to the infiltration of untreated wastewater from unlined irrigation channels, storage dams and excess water used for irrigation (Jimenez and Chávez, 2004). This infiltrated wastewater is hydraulically connected to local springs (aquifer residence time 20–40 days)

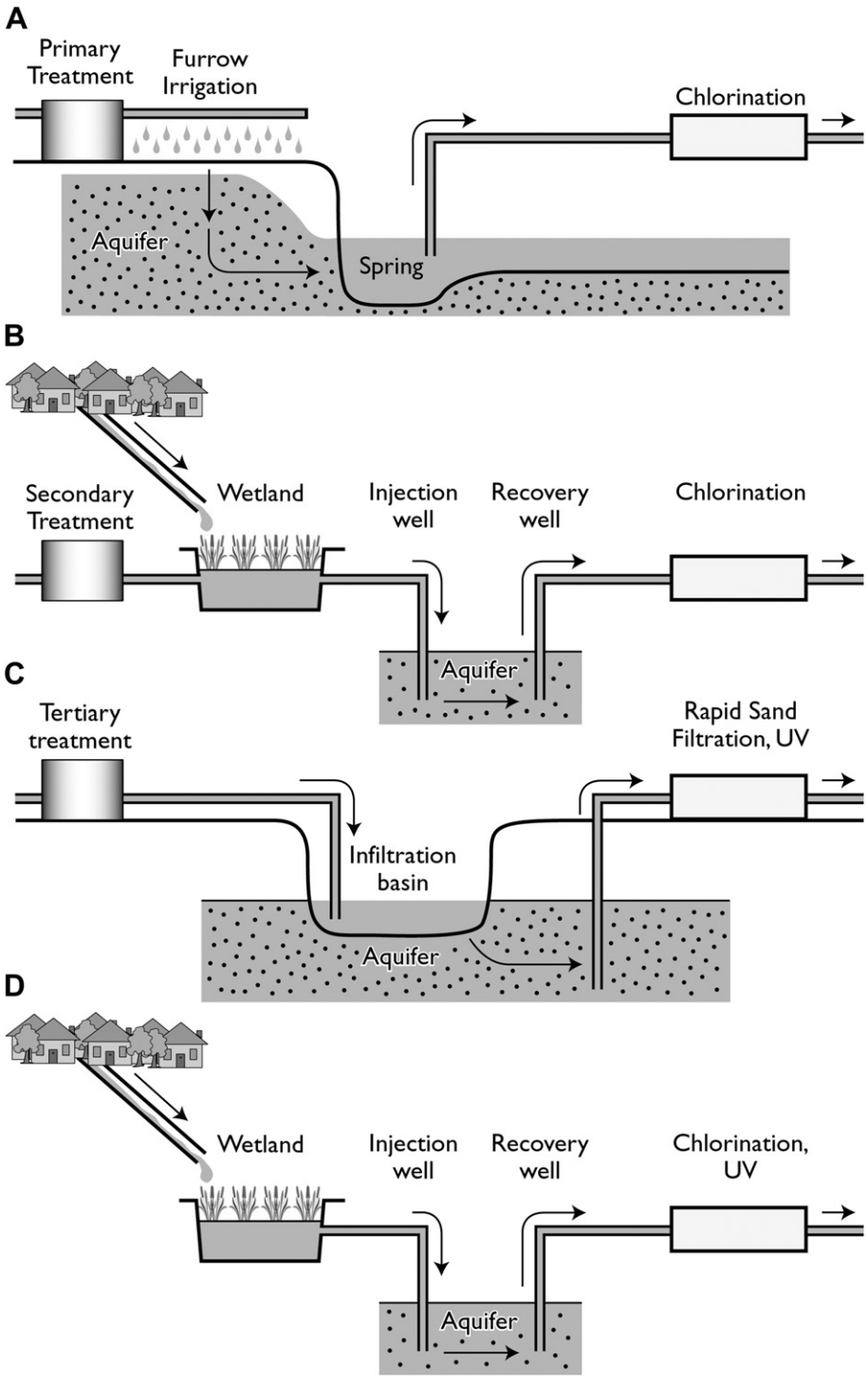

Fig. 1 – Case study site system diagrams: (A) Tula Valley; (B) Atlantis; (C) Torreele/St-André; (D) Parafield.

that are used as drinking water supplies (Jimenez and Chávez, 2004). This is the largest known case of indirect water reuse for human consumption in the world. This study considers only one spring, the Cerro Colorado spring which currently produces 0.6 m^3/s of potable water. Treatment after extraction consists only of chlorination to inactivate pathogens. Furrow irrigation of untreated effluent occurs within 20 m of the spring. A wall surrounds the spring to ensure there is no direct surface discharge of effluent into the spring.

The Parafield aquifer storage transfer and recovery (ASTR) site is located in a northern suburb of Adelaide, South Australia approximately 17 km from the city centre. Urban stormwater from a mixed residential and industrial catchment is passed through a constructed wetland prior to recharge via injection wells into a confined limestone aquifer. Water is recovered via separate wells after a mean residence time in the aquifer of 241 days (Kremer et al., 2008). Currently, the site is managed as a trial to determine the suitability of the

Table 1 – Description of case study sites.

General information	Tula Valley, Mexico	Atlantis, South Africa	Parafield, Australia	Torreele/St-André, Belgium
Town population	72,500 (Cerro Colorado region)	250,000	122,000 (Salisbury region)	61,000
Mean annual rainfall (mm)/Mean annual evaporation	550	450	450	700
Source water	Reclaimed effluent	Reclaimed domestic effluent/Stormwater	Stormwater	Reclaimed effluent
Mean temperature of source water (°C)	21	21	18	15
Redox status of recharge water	Anaerobic	Anoxic to Anaerobic	Aerobic	Aerobic
Year of recharge commencement	1986	1976	2006	2002
Annual recharge volume (m^3/year)	$>788 \times 10^6$ for the Tula Region	Stormwater: 2200 summer 72,000 winter Reclaimed water: 5×10^6	0.25×10^6	2.5×10^6
Annual extraction volume (m^3/year)	12.6×10^6 (Cerro Colorado region)	4.6×10^6	0.25×10^6	3.5×10^6
Average aquifer residence time (days)	20	365	241	55
Minimum flow path length (days)	Unknown	182	100	30
Mean temperature of aquifer (°C)	30	20	23	11
Redox status of aquifer	Iron reducing	Nitrate reducing	Nitrate reducing	Nitrate reducing
% Recharged water recovered from aquifer	100	40	90	70
MAR system components				
1. Capture Zone	Reclaimed effluent	Residential stormwater catchment and reclaimed effluent	Residential stormwater catchment	Reclaimed effluent
2. Pre-treatment	Primary sedimentation in storage dams	Activated sludge, maturation ponds, constructed wetland	Constructed reedbed	Activated sludge, ultrafiltration, reverse osmosis, UV disinfection
3. Recharge	Infiltration from storage canals and reservoirs and irrigation areas	Recharge basins	Recharge wells	Recharge basins
4. Subsurface storage	Partially confined basaltic aquifer with some volcanic ash and lava intervals	Unconfined sandy aquifer	Confined limestone aquifer	Unconfined sandy aquifer
5. Recovery	Spring discharge	Extraction wells	Extraction wells	Extraction wells
6. Post-treatment	Chlorination	Softening, chlorination	Aeration tank, Chlorination, UV disinfection	Aeration, rapid sand filtration, UV disinfection
7. End use	Drinking water	Drinking water	Drinking water	Drinking water

recovered water for drinking supplies. Post-treatment options are still being considered and may include UV and chlorine disinfection prior to entering the drinking water distribution system. Further details of the hydrogeology (Pavelic et al., 2004; Kremer et al., 2008) as well as the development of the risk assessment and management plan (Swierc et al., 2005; Page et al., 2008, 2009) have been reported.

The Atlantis site is located 40 km north of Cape Town near the semiarid southwest coast of South Africa. Secondary treated reclaimed water, together with wetland-treated urban stormwater from a residential catchment is recharged to an unconfined sandy aquifer. Pre-treatment includes secondary wastewater treatment (activated sludge) prior to blending with urban stormwater flows and passing through a wetland. Water is infiltrated by means of two recharge basins, has a residence time in the aquifer of approximately one year prior to recovery by means of two well fields. Poor quality stormwater from industrial zones is pumped into a coastal recharge basin which also forms a barrier between the extraction well fields and the sea to prevent saline intrusion. Post treatment involves water softening and chlorination before water is blended with Cape Town supplied mains water entering the drinking water distribution system.

The Torreele/St-André site is located at the Flemish coast in Belgium, and it has been developed to establish sustainable groundwater management. The drinking water supply can be augmented, environmental functions maintained and sea water intrusion is prevented. Tertiary (reverse osmosis) treated effluent is recharged to an unconfined sandy aquifer via an infiltration basin and recovered via a series of extraction wells after a residence time of minimum 30 and on average 55 days. Post treatment includes aeration, rapid sand filtration and UV disinfection prior to supply to the drinking water network. Details of the Torreele/St-Andre facility (Van Houtte and Verbauwhede, 2008) and the infiltration system (Van Houtte and Verbauwhede, 2005) have been reported.

3. Methods for risk assessment and valuing aquifer treatment

The microbial risk assessment methodology used follows the approach outlined in WHO (2004) and NRMMC-EPHC-AHMC (2006). The traditional approach to identifying tolerable risk has been to define maximum levels of infection or disease. However, this approach fails to consider the varying severity of outcomes associated with different hazards. This shortcoming can be overcome by measuring severity in terms of disability adjusted life years (DALYs). DALYs have been used extensively by agencies such as the World Health Organisation (WHO) to assess disease burdens (WHO, 2004) and is the approach adopted in this study. Three representative pathogens; rotavirus, *Cryptosporidium* and *Campylobacter*, were used to assess the risk of viruses, protozoa and bacteria as described in WHO (2004) and EPHC-NHMRC-NRMMC (2008). As the risk estimates are probability distribution functions, the mean, median and 95th percentile were routinely calculated for each pathogen risk. The tolerable mean risk adopted is 10^{-6} DALYs per person per year (WHO, 2004).

For the case study sites discussed in this paper, qualitative residual risk assessments have been summarised as part of the RECLAIM WATER project (Ayuso-Gabella et al., 2007). In furthering the qualitative understanding of the pathogenic hazards at each site, a quantitative microbial risk assessment was performed to determine the residual risk for each case study site and the value of the aquifer treatment. The residual risks are risk probability estimates assuming normal operating conditions, i.e., where source waters are not exposed to unusual hazard inputs and treatment processes are operating according to specifications.

The risk models for simulating hazard reduction, consumption, infection and disease burden were constructed using MS Excel program [2003] enhanced with @Risk Industrial v. 4.5 (Palisade Corp., USA). The minimum value calculated was 1.0×10^{-10} DALYs per person per year.

A triangular probability distribution function (PDF) describing each engineered treatment barrier was adopted from literature for each pathogen (Smeets et al., 2006; NRMMC-EPHC–NHMRC, 2008). The triangular distribution was defined by a minimum, most likely and maximum $\log_{10}$ removal value (Smeets et al., 2006; NRMMC-EPHC–NHMRC, 2008) and is shown in Table 2. The risks associated with recontamination at each treatment barrier were not specifically assessed. For the aquifer treatment barrier, the product of two PDFs; the aquifer residence time and a daily pathogen decay rate (expressed in $\log_{10}$/day) were used to calculate the $\log_{10}$ removal value. Initial pathogen numbers in the stormwater and wastewater were derived from literature (Kocwa-Haluch and Zalewska (2002); NRMMC-EPHC–NHMRC (2009a); Robertson et al. (2006)). For the Atlantis case study, the two source waters were mixed to derive a final PDF of pathogen concentration for the injectant. Each of the PDFs (including pathogen numbers and aquifer and non-aquifer treatments) was subsequently used in the Monte Carlo simulations to calculate the residual risk.

Once the residual risks were calculated for each MAR scheme a sensitivity analysis was performed which standardises the factors which affect risk and is termed the factor sensitivity (FS) (Zwietering and van Gerwen, 2000). For each MAR scheme, the residual risk was then recalculated in the absence of each barrier in turn (such as the aquifer treatment barrier). The FS is a ratio calculated by dividing the revised residual risk estimate (in DALYs) when a factor (e.g. a treatment step) is removed from the treatment train (denoted $N(Barrier)$), by the baseline mean risk, $N(Mean)$ also in DALYs from the residual risk assessment and then $\log_{10}$ transforming the ratio:

$$FS = \log_{10}(N(Barrier)/N(Mean))$$

Higher FS values means the factor has a larger effect on risk. Following assessment of FS, a risk-based approach for determining suitable aquifer residence times for MAR schemes is proposed. Aquifer treatment uses the surrogate parameter, aquifer residence time to estimate the value of the aquifer treatment as part of the multiple barrier system. Simulations of changes in the aquifer residence time allow the aquifer barrier to be quantified and compared to the acceptable risk, 1.0×10^{-6} DALYs (WHO, 2004). This allows the determination of a required average residence time and associated monitoring can be utilised to manage this barrier within the treatment system.

4. Results

4.1. *Aquifer barrier treatment characterisation*

Aquifer treatment characteristics were derived from the PDFs of the residence time in the aquifer and the reported $\log_{10}$ decay rates for pathogens (Table 2) based on the work by Toze et al. (2009) at the Australian site. No data were available for pathogen attenuation rates at the other sites. The aquifer and engineered (non-aquifer) treatment barrier characteristics are reported as $\log_{10}$-removals (Table 3) which conveys the order of magnitude of the removal for each of the reference pathogens. Removal $\log_{10}$ values for each treatment barrier were considered additive and thus the Torreele/St-André scheme with multiple engineered barriers

Table 2 – Probability distribution functions used for the quantitative risk assessment.

Barrier	Atlantis			Parafield		
Pathogen	Rotavirus	*Cryptosporidium*	*Campylobacter*	Rotavirus	*Cryptosporidium*	*Campylobacter*
Pathogen source water number[a]	0.3, 0.6[b]/443, 220[c]	0.5, 1.2[b]/200, 100[d,e]	3.9, 9.8/10^1–10^4[b]	0.3, 0.6[b]	0.5, 1.2[b]	3.9, 9.8[b]
Artificial wetland[f]	0.0, 0.0,0.0	0.5, 0.5, 1.0	1.5,2,2.5	0.0, 0.0, 0.0	0.5, 0.5, 1.0	1.5, 2.0, 2.5
WWTP[f]	0.2,1.7,2.3	0.4, 1.8, 3.8	0.6, 1.4, 3.7			
UF[f]						
RO[f]						
UV[f]						
Subsurface storage (residence time days)		182, 365, 730[f]			241, 58[a]	
Pathogen decay rate 1-log (days)[d]	0.0055, 0.0036	0.012, 0.0030	5.6[g]	0.0055, 0.0036	0.012, 0.0030	5.6[g]
Recovery (% mixing)		0.4			0.9	
Rapid sand Filtration[f]						
UV[f]				2.0, 2.0, 3.0	2.0, 3.0, 4.0	2.0, 3.0, 4.0
Chlorination[f]	1.0, 2.0, 3.0	0.0,0.0,0.5	2.0, 4.0, 6.0	1.0, 2.0, 3.0	0.0, 0.0, 0.5	2.0, 4.0, 6.0
Barrier	Torreele/St-André			Tula Valley		
Pathogen	Rotavirus	*Cryptosporidium*	*Campylobacter*	Rotavirus	*Cryptosporidium*	*Campylobacter*
Pathogen source water number[a]	443, 220[d]	200, 100[b]	10^1–10^4	443, 220[c]	200, 100[b]	10^1–10^4
Artificial wetland[f]						
WWTP[f]	0.2, 1.7, 2.3	0.4, 1.8, 3.8	0.6, 1.4, 3.7			
UF[f]	4.0, 4.0, 6.5	3.0, 3.0, 7.0	4.0, 4.0, 7.0			
RO[f]	2.7, 3.0, 6.5	3.0, 3.0, 7.0	4.0, 4.0, 7.0			
UV[f]	2.0, 2.0, 3.0	2.0, 3.0, 4.0	2.0, 3.0, 4.0			
Subsurface storage (residence time days)		30, 35, 40[f]			20, 40	
Pathogen decay rate 1-log (days)[e]	0.0055, 0.0036	0.012, 0.0030	5.6[g]	0.0055, 0.0036	0.012, 0.0030	5.6[g]
Recovery (% mixing)		0.7			1	
Rapid sand Filtration[f]	0.1,0.5,3.9	0.8, 2.9, 5.4	0.8,1.5,3.3			
UV[f]	2.0, 2.0, 3.0	2.0, 3.0, 4.0	2.0, 3.0, 4.0			
Chlorination[f]				1.0, 2.0, 3.0	0.0, 0.0, 0.5	2.0, 4.0, 6.0

Engineered treatment efficacy $\log_{10}$ removal efficiencies come from Smeets et al. (2006); NRMMC-EPHC–NHMRC (2008); except Torreele/St-André from Ayuso-Gabella et al. (2007).

a Lognormal distribution mean, standard deviation.

b 95th Percentile as per Table A3.1 of the Guidelines for Stormwater Harvesting and Reuse: *Campylobacter* 15 n/L; *Cryptosporidium* 1.8 n/L; rotavirus 1 n/L (NRMMC-EPHC–NHMRC, 2009a).

c Robertson et al. (2006).

d Cited in Kocwa-Haluch and Zalewska (2002).

e Toze et al. (2009), normal distribution, mean, standard deviation.

f Triangular distributions: minimum, most likely, maximum.

g Single value only.

Table 3 – Calculated aquifer barrier and engineered treatment (non-aquifer) removal efficiency in log_{10} units.

Pathogen	Tula Valley		Atlantis		Parafield		Torreele/St-André	
	Aquifer	Non-aquifer	Aquifer	Non-aquifer	Aquifer	Non-aquifer	Aquifer	Non-aquifer
Rotavirus								
Min	0.0	1.0	0.0	1.2	0.0	3.0	0.0	8.3
Most likely	0.2	2.0	2.5	3.7	1.4	4.0	0.2	17.2
Max	0.8	3.0	> 6.0	5.3	> 6.0	6.0	0.7	25.2
Cryptosporidium								
Min	0.0	0.0	0.3	0.9	0.1	2.5	0.0	11.2
Most likely	0.4	0.0	5.0	2.3	2.8	3.5	0.4	16.7
Max	0.9	0.5	> 6.0	5.3	> 6.0	5.5	0.9	31.2
Campylobacter								
Min	> 6.0	2.0	> 6.0	4.1	> 6.0	5.5	> 6.0	13.4
Most likely	> 6.0	4.0	> 6.0	7.4	> 6.0	9.0	> 6.0	16.9
Max	> 6.0	6.0	> 6.0	12.2	> 6.0	12.5	> 6.0	29.0

resulted in the highest log_{10} removal for the 'non-aquifer' treatment component. All log_{10} removal values accredited to aquifers were capped at a maximum of 6.0 log_{10} consistent with the reported values for engineered treatments in NRMMC-EPHC–NHMRC (2008). Each of the MAR sites placed a different value on the aquifer removal characteristics compared to the engineered treatments. Tula Valley relied almost exclusively on the aquifer, where as Torreele/St-André had extensive redundancy in their system due to a long treatment train of engineered barriers and as such relied little on the aquifer. Each of the MAR sites was considered equally effective in removing *Campylobacter* (reported as >6.0 log_{10} units) but varied with respect to *Cryptosporidium* and rotavirus based on the differences in aquifer residence and storage times. Tula Valley and Torreele/St-André had the same calculated low log_{10} removal capabilities where as Parafield and Atlantis had greater calculated treatment capacities due to the longer residence times of water in the subsurface at these sites. Rotavirus removals were the lowest of the three pathogens studied at each site due to their low decay rates (Toze et al., 2009).

4.2. *Case study sites residual risk assessment*

The results in DALYs of the risk assessment are reported in Table 4. Tula Valley had the highest residual risk for rotavirus and *Cryptosporidium*. This can be attributed to the lack of pre-treatment and the low residence time of the reclaimed water in the aquifer (20 days average) prior to recovery. Atlantis had acceptable risk for *Campylobacter*, but higher risk for *Cryptosporidium* and rotavirus. Parafield had low risks for each of the pathogens. Torreele/St-André had a very low risk for each pathogen due to the large pre- and post-recovery treatment trains.

While the mean gives an assessment of the average risk, and the median it's central tendency, the 95th percentile gives an estimate of the variability and reasonable maximum of the risk. Where the 95th percentile was below the acceptable risk threshold, the risk assessment was considered to be robust. As such, the risk assessment from rotavirus for Parafield is not as robust and further work is required to reduce the uncertainty of this risk estimate or further treatment is required to reduce the risk. For the other case study sites, the median and

Table 4 – Mean, Median and 95th percentile residual risk assessment in DALYs.

Pathogen	Tula Valley	Atlantis	Parafield	Torreele/St-André
Cryptosporidium				
Mean	1.5×10^{-3}	7.0×10^{-6}	7.7×10^{-9}	$<1.0 \times 10^{-10}$
Median	1.5×10^{-3}	5.3×10^{-9}	2.0×10^{-10}	$<1.0 \times 10^{-10}$
95th	1.5×10^{-3}	1.2×10^{-5}	1.8×10^{-8}	$<1.0 \times 10^{-10}$
Rotavirus				
Mean	8.4×10^{-4}	2.3×10^{-4}	8.5×10^{-7}	$<1.0 \times 10^{-10}$
Median	8.4×10^{-4}	4.9×10^{-5}	5.0×10^{-8}	$<1.0 \times 10^{-10}$
95th	8.4×10^{-4}	8.3×10^{-4}	3.1×10^{-6}	$<1.0 \times 10^{-10}$
Campylobacter				
Mean	$<1.0 \times 10^{-10}$	$<1.0 \times 10^{-10}$	$<1.0 \times 10^{-10}$	$<1.0 \times 10^{-10}$
Median	$<1.0 \times 10^{-10}$	$<1.0 \times 10^{-10}$	$<1.0 \times 10^{-10}$	$<1.0 \times 10^{-10}$
95th	$<1.0 \times 10^{-10}$	$<1.0 \times 10^{-10}$	$<1.0 \times 10^{-10}$	$<1.0 \times 10^{-10}$

Bold value indicates value exceeds guideline of 1×10^{-6} DALYS; limit of calculation is 1×10^{-10} DALYS.

95th percentile are similar indicating that the assessments are generally robust.

4.3. *Valuing the aquifer barrier in MAR schemes*

A sensitivity analysis was performed for each barrier in the treatment train for each case study site and the factor sensitivity (FS) calculated. The FS calculation standardises the comparison between each of the water treatment barriers and the aquifer and thereby aids in valuing the aquifer as part of the larger treatment train. A value of 1.0 indicates a 10-fold increase in risk. Table 5 gives a comparison of the FS values for each of the treatment barriers across the MAR systems.

For Tula Valley, most of the FS scores were calculated to be zero as the calculated risk for the removal of a barrier, *N*(*Barrier*) was equal to the initial residual risk assessment, *N*(*Mean*). For example, the calculated risk for rotavirus was equal to 8.4×10^{-4} DALYs regardless if the chlorination barrier was in place, *N*(*Barrier*) = *N*(*Mean*). The exception was the aquifer treatment barrier for *Campylobacter* where there was >6 orders of magnitude increase in risk. For *Campylobacter*, the aquifer was the single most important barrier (compared to chlorination) in determining the residual risk.

For Atlantis, the FS analysis indicated that the aquifer was the single most important barrier in determining risk from all pathogens, where again > 6 orders of magnitude increase in risk would result if the aquifer was removed from the treatment train for *Campylobacter*. Like Tula Valley, if the aquifer barrier is in place at Atlantis then the other barriers have little influence in determining the residual risk from *Campylobacter*. For *Cryptosporidium*, the treatment train analysis was more complex with the secondary wastewater treatment plant having a slightly lower capacity to reduce the residual risk.

For Parafield, the aquifer barrier again dominated the risk from *Campylobacter*, resulting in over 10-fold increase in risk if it were not present. The aquifer was the third most important barrier with respect to rotavirus and second for *Cryptosporidium* risk.

For Torreele/St-André, the aquifer only played a measurable role in reducing residual risk for rotavirus. The most important barriers were ultrafiltration and reverse osmosis for each of the reference pathogens. Analogously to Tula Valley, the FS value of the aquifer could not be calculated for *Cryptosporidium* and *Campylobacter* as the revised risk in removing the barrier was equal to the initially calculated residual risk, < 1.0×10^{-10} DALYs.

From the FS analysis of Table 5, the subsurface treatment steps were identified as being highly variable in the treatment train in reducing the calculated residual risk. Fig. 2 shows the reduction in pathogen numbers of the injectant for each of the reference pathogens at each of the MAR sites; *Campylobacter* is not included due to its extremely rapid decay. The initial pathogen numbers in the water to be recharged for each MAR site is a function of the pre-treatment barriers. Torreele/St-André with its large pre-treatment train (average $\log_{10}$

Table 5 – Factor Sensitivity ratio – relative importance of barriers.

	Tula valley	Atlantis	Parafield	Torreele/ St-André
Rotavirus				
Constructed wetland	–	0.00[a]	0.00[a]	–
Secondary treatment	–	0.35	–	1.14
Ultrafiltration	–	–	–	4.51
Reverse osmosis	–	–	–	3.49
UV disinfection	–	–	–	2.23
Aquifer	0.00[b]	0.55	0.94	2.23
Rapid sand filtration	–	–	–	0.92
UV disinfection	–	–	1.94	2.23
Chlorination	0.00[b]	0.43	1.66	–
Cryptosporidium				
Constructed wetland	–	0.78	0.61	–
Secondary treatment	–	1.65	–	1.24
Ultrafiltration	–	–	–	3.48
Reverse Osmosis	–	–	–	3.48
UV disinfection	–	–	–	2.57
Aquifer	0.00[b]	1.93	2.03	0.00[b]
Rapid sand filtration	–	–	–	1.92
UV	–	–	2.78	2.57
Chlorination	0.00[b]	0.05	0.14	–
Campylobacter				
Constructed wetland	–	0.00[b]	0.00[b]	–
Secondary treatment	–	0.00[b]	–	0.00[b]
Ultrafiltration	–	–	–	0.00[b]
Reverse osmosis	–	–	–	0.00[b]
UV disinfection	–	–	–	0.00[b]
Aquifer	6.66	7.57	1.29	0.00[b]
Rapid sand filtration	–	–	–	0.00[b]
UV disinfection	–	–	0.00[b]	0.00[b]
Chlorination	0.00[b]	0.00[a]	0.00[a]	–

a Removal of viruses by constructed wetlands is considered to be negligible (NRMMC-EPHC-AHMC, 2006).
b FS score could not be calculated as the resultant risk was equal to the residual risk.

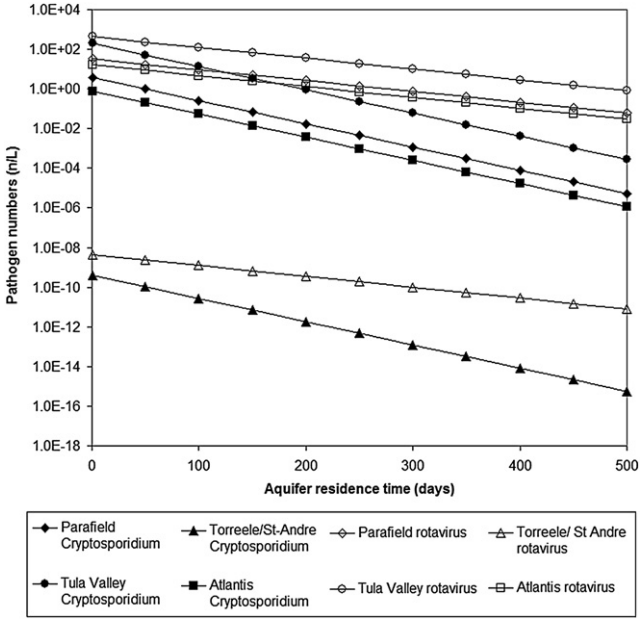

Fig. 2 – Decay in pathogen numbers as a function of residence time.

removals of 14.7, 10.8, 12.4 for rotavirus, *Cryptosporidium* and *Campylobacter*, respectively) begin with very low numbers of pathogens in the recharge water. This contrasts with Tula Valley which has no pre-treatment and hence high numbers of pathogens in the recharge water. Atlantis and Parafield sit in between Torreele/St-André and Tula Valley but Parafield has much lower numbers of pathogens than Atlantis as its recharge water was solely urban stormwater as opposed to reclaimed effluent with a minor component of stormwater. The pathogen numbers for each site steadily decreased as a function of the decay rate and the residence time in the aquifer reported in Table 2.

The probability of infection is then multiplied by the DALYs per infection to calculate the final residual risk of each MAR system in Table 4. The infection dose-response curve for each pathogen results in a conversion of the risk of infection to DALYs, which can be plotted as a function of aquifer residence time to examine the treatment role of the aquifer (Figs. 3 and 4). These figures show the DALYs per person per year for each of the MAR schemes as a function of aquifer residence time and pathogen decay rates. It is important to note that the decay rates are assumed to be linear and unchanging as a function of time. The change in DALYs from *Campylobacter* as a function of aquifer residence time is not shown as the risks from *Campylobacter* were not quantifiable for all sites. Torreele/St-André is not shown in Figs. 3 and 4 as the calculated risk was $<1.0 \times 10^{-10}$ annualised DALYs for each of the reference pathogens.

Fig. 3 shows the change in DALYs from rotavirus as a function of aquifer residence time for Tula Valley, Parafield and Atlantis. For Tula Valley and Atlantis, the risks from rotavirus remain high. At Parafield, the risk is reduced after approximately 200 days in the aquifer, comparable to the average aquifer residence time at this site.

Fig. 4 shows the change in DALYs from *Cryptosporidium* as a function of aquifer residence time for Tula Valley, Parafield and Atlantis. At Parafield, the risk is low due to the nature of the source water (stormwater). The Atlantis site requires over 500 days to reach the value of 1×10^{-6} DALYs, which is within the actual range of residence times for this site but exceeds the mean value. Tula Valley risks remain higher than other sites.

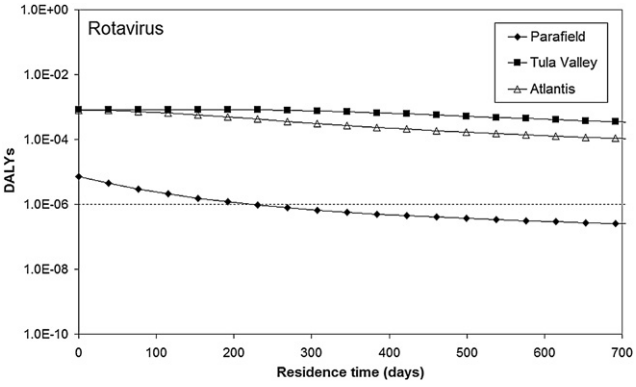

Fig. 3 – Changes in mean DALYs from rotavirus with increasing residence times in the aquifer.

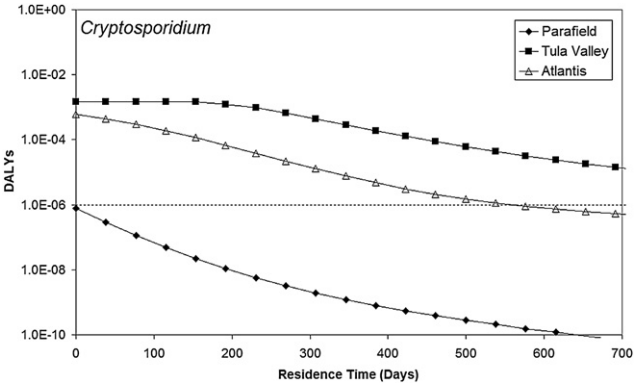

Fig. 4 – Changes in mean DALYs from *Cryptosporidium* with increasing residence times in the aquifer.

5. Discussion

5.1. *Characterisation of the value of aquifer treatment*

In order to provide safe drinking water with MAR, an integrated approach to managing risks needs to be adopted which includes characterisation of the aquifer treatment barrier. To date, there have been no reported case studies where the aquifer treatment barrier of a MAR scheme is accredited with $\log_{10}$ removals for pathogens much like in conventional drinking water treatment. In valuing the treatment capacity, integrity and independence of aquifers, MAR can be utilised in the same way as conventional engineered water treatment in an integrated water supply system.

The value of the aquifer barrier was determined by the relative $\log_{10}$ removal characteristics with respect to the reference pathogens (Table 3). The $\log_{10}$ removals for *Campylobacter* are potentially $> 6.0 \log_{10}$, a similar value attributed to other water treatment technologies such as reverse osmosis (NRMMC-EPHC-AHMC, 2006). For *Cryptosporidium*, the value of the aquifer was similar to primary treatment for Tula Valley and Torreele/St-André, ultrafiltration for Atlantis and dual media filtration with coagulation at Parafield depending upon residence time of the recharge water in the aquifer. Rotavirus had the poorest $\log_{10}$ removals in the aquifer (Table 3) due to the very low decay rates (Table 2).

Knowledge of both the aquifer residence time and the rate of decay are essential for enabling the treatment value of the aquifer to be determined (Table 3). The decay of pathogens in groundwater during MAR is influenced by a range of factors such as the activity of indigenous groundwater microorganisms, temperature, redox status, oxygen concentrations and organic carbon concentrations (Gordon and Toze, 2003; Toze et al., 2004). Research has shown that bacteria tend to survive for much shorter times in aquifers than enteric viruses and protozoa (Toze et al., 2004) but the relative times can be aquifer-dependent. Another issue relating to pathogen decay is that decay rate is not always linear. The decay of some pathogens, in particular the more resistant viruses have been observed to have a decline in decay rate with time. Thus, in these cases a broken stick model of decay with different rates of decay

may be more appropriate than a single rate of decay. The most appropriate decay rates to use will need to be verified in future risk assessments.

5.2. Risk assessment for the case study sites

To evaluate the risk from enteric pathogens during MAR, the potential presence of these pathogens and their numbers need to be determined. The major source of all enteric pathogens is faecal contamination, particularly from human faecal material. The largest number of enteric pathogens can be expected to be detected in untreated wastewater (Table 2) with numbers reducing through treatment processes (Table 3). The potential presence of enteric pathogens in the recharge water (Fig. 2) is directly linked to the potential of human faecal matter contaminating the water. Thus, in this study, the pathogen risk for Torreele/St-André was assessed to be very low due to the high level of treatment prior to MAR. Conversely, Tula Valley had the highest risk, due to a low level of engineered treatment which is reflected in the QMRA results (Table 4). The Atlantis scheme has less opportunity for the presence of microbial pathogens due to the blending of treated wastewater and stormwater, while the risk in the Parafield system is limited to the potential for sewer pump-station overflows and contamination from animal faeces.

A number of limitations to the QMRA approach with MAR systems have been previously identified by Toze et al. (2009), including factors such as variability in pathogen decay rates. An accurate risk assessment also requires the input of accurate pathogen numbers. The initial pathogen numbers in the recharge water (Fig. 2) are influenced by a range of factors such as disease burden of the local population and the level of treatment for the recharge water. The numbers of some pathogens is also less accurate due to the difficulties in detection. For example, the detection of *Cryptosporidium* oocysts and rotavirus is difficult due to the lack of suitable culture methods and the low numbers ($\leq$100 units) usually present in large volumes of water (>1 L). Numbers in river, canal and recreation water for *Cryptosporidium* oocysts have been quoted as between 5 and 240 oocysts per 10 litres (Schets et al., 2008, Plutzer et al., 2008, Mons et al., 2009). In comparison, rotavirus numbers in similar water types have been reported to be between 2 and 200 detectable units per litre (Mehnert et al., 1993, Lodder, et al. 2005).

In general, the risks evaluated for each of the MAR sites (Table 4) were in the order Tula Valley > Atlantis > Parafield > Torreele/St-André for *Cryptosporidium* and rotavirus but all had low risks for the bacterial pathogen, *Campylobacter*. Only Torreele/St-André and Parafield met the mean WHO guideline for all the reference pathogens (Table 4).

5.3. Standardisation of determining aquifer treatment

The factor sensitivity (FS) analysis method (Smeets et al., 2006; Zwietering and van Gerwen, 2000) was used to give an indication of the relative value of the aquifer (in terms of reducing human health risk) vis-à-vis the other barriers within the treatment train for each case study site. For the Tula Valley system (for *Campylobacter*), this was the maximum risk possible (4.6×10^{-3} reduced to $< 1 \times 10^{-10}$ DALYs) about a million-fold reduction. Conversely for Torreele/St-André the aquifer treatment effect was not measurable as the risk from *Campylobacter* was already $< 1 \times 10^{-10}$ DALYs (Table 5), resulting in the observed FS ratio of 0.00. At Torreele/St-André, there are multiple barriers that are effective in reducing the risk to an acceptable level and even if any one barrier fails the risk remains negligible. The Tula Valley site demonstrates the high value placed on the aquifer for mitigating the risk from pathogenic bacteria, it is the only barrier that significantly affects the risk from *Campylobacter*. Similarly, for the Atlantis and Parafield sites if the aquifer barrier is in place then the risks from pathogenic bacteria are negligible.

For Tula Valley, the role of the aquifer is not measureable for *Cryptosporidium* as the risk with removal of a barrier, *N*(*Barrier*) is the same as the residual risk (*N*(*Mean*), i.e., the maximum possible risk of 1.5×10^{-3} DALYs, Table 4). This contrasts to Atlantis where the aquifer is the single most important barrier (highest FS scores) influencing risk for each of the reference pathogens. For Parafield, the aquifer has the highest value in reducing the risk from *Campylobacter* and second highest value in reducing *Cryptosporidium* risk (UV slightly higher), but post-recovery UV and chlorine disinfection was each superior to the aquifer in reducing risk for rotavirus. For Torreele/St-André, the aquifer has little risk reduction value, most important are the ultrafiltration and reverse osmosis treatment barriers.

5.4. Integrating aquifer treatment with engineered treatments

To date, aquifer treatment has been slow to integrate into an engineered water treatment train due to the difficulty in measuring a quantifiable reduction in risk. This is in part due to the adoption of risk-based management systems, such as the Hazard Analysis and Critical Control Point (HACCP) approach. HACCP concepts have been adopted by the water industry and promoted as a more proactive approach to managing drinking water supplies (WHO, 2004; NRMMC-EPHC–NHMRC, 2008), as well as recycled water systems (NRMMC-EPHC-AHMC, 2006) and even MAR systems (NRMMC-EPHC–NHMRC, 2009b). Yet, aquifer treatment remains difficult to integrate as there are no easily identifiable critical limits and control points such as for the more common water treatment technologies such as chlorination which uses contact time, UV disinfection which uses UV-transmittance and membrane treatments which use pressure and electrical conductivity.

It is proposed that an extension of the FS sensitivity analysis could also be used to provide a means of generating evidence-based critical limits to manage critical control points. While there are no health-based targets for pathogen numbers (Fig. 2) QMRA can be used to address the issue of setting critical limits. This is done by treating the DALYs estimates as representing acceptable estimates of "absolute" risk and comparing them to the agreed international human health risk benchmarks, 1.0×10^{-6} DALYs (WHO, 2004). In this instance, the comparison of the Parafield risk estimate indicated that the residual risk was acceptable for *Campylobacter* when compared to this benchmark and this conclusion was robust as indicated by the 95th percentile being less than the

benchmark value. However, for rotavirus the assessment was less robust and the required aquifer residence time was just great enough for the scheme to support so additional post-recovery treatment could be required. An illustrative example for setting of critical limits for mean aquifer residence time comes from the *Cryptosporidium* for the Atlantis site, where the mean residence time needs to exceed ~550 days to achieve tolerable levels of risk. Again, this assumes that the pathogen decay rates of Toze et al. (2009) are linear and are representative of the processes occurring in the subsurface of this site. Use of the residence time critical limit could also be used to design infiltration and extraction pumping regimes to ensure the mean residence time in the aquifer is achieved. Where aquifer residence time is not accurately known, such as in the Atlantis and Tula Valley examples, it can be determined by use of suitable groundwater tracers. This can include both applied tracers, substances injected into the groundwater intentionally and thereby in controlled doses, time intervals and locations (such as SF_6) or natural tracers (such as the recharge water electrical conductivity if this has marked variation from the ambient groundwater). Knowledge of the residence time in the aquifer coupled with pathogen decay rates could then be used to fully appreciate the water treatment function of the subsurface and integrate the aquifer barrier with the engineered treatments in the provision of safe drinking water.

6. Conclusions

For the four MAR case study sites considered, the QMRA provides a means of quantifying the combined effects of aquifers and engineered treatments for reference pathogens in terms of $\log_{10}$ removal characteristics. For each site, the aquifer consistently had the highest $\log_{10}$ removal predicted for *Campylobacter* whilst rotavirus and *Cryptosporidium* had more variable removal rates depending upon the residence time in the aquifer. QMRA was found to be a useful tool in establishing the value of the aquifer within the treatment train and allowed the assessment of human health risk from pathogens in terms of DALYs. A sensitivity analysis was used to assess which of the treatment barriers was most important in each of the MAR systems. This approach allows the integration of the aquifer treatment characteristics into the larger engineered treatment train and could be used in the future to quantitatively assess the reduction of human health risk for MAR systems more generally.

Acknowledgements

The authors acknowledge support from the International Science Linkages Program established under the Australian Government's innovation statement. "*Backing Australia's Ability*". This study forms part of the RECLAIM WATER research project supported by the European Commission under the sixth research framework programme (Contract-No. 018309) and Salisbury stormwater ASTR Research Project which is partnered by United Water, SA Water, City of Salisbury, CSIRO, the Adelaide and Mt Lofty Ranges Natural Resources Management Board and the River Murray Natural Resources Management Board. The authors acknowledge support from the South Australian Premiers Science and Research Fund, CSIRO Water for a Healthy Country Program and the National Water Commission through the Raising National Water Standards Program.

REFERENCES

Ayuso-Gabella, M. N., Barry, K., Bixio, D., Dillon, P., Genthe, B., Jefferson, B., Jeffrey, P., Kopač, I., Page, D., Pavelic, P., Purdie, M., Regel, R., Rinck-Pfeiffer, S., Salgot, M., Van Houtte, E., Wintgens T. (2007) Deliverable D6.1 Report on case-study specific risk assessment. RECLAIM WATER.

Dillon, P., 2005. Future management of aquifer recharge. Hydrogeology Journal 13, 313–316.

Dillon, P., Toze, S., 2005. AWWARF project 2618, final Report. Water Quality Improvements during Aquifer Storage and Recovery. Compilation of Information from Ten Sites, vol. 2. AWWARF, p. 203.

Gordon, C., Toze, S., 2003. Influence of groundwater characteristics on the survival of enteric viruses. Journal of Applied Microbiology 95 (3), 536–544.

Jimenez, B., 2004. El Mezquital, Mexico: the biggest irrigation district that uses wastewater. In: Lazarova, V., Bahri, A. (Eds.), Water Reuse for Irrigation: Agriculture, Landscape and Turf Grass. CRC Press, pp. 535–562 (Chapter 12.3).

Jimenez, B., Asano, T., 2008. Water reuse around the world. In: Jiménez, B., Asano, T. (Eds.), Water Reuse: An International Survey of Current Practice, Issues and Needs. IWA Publishing, Inc, London (Chapter 2) 628 pp ISBN 156 670 6491.

Jimenez, B., Chávez, A., 2004. Quality assessment of an aquifer recharged with wastewater for its potential use as drinking source: "El Mezquital Valley" case. Water Science and Technology 50 (2), 269–273.

Kocwa-Haluch, R., Zalewska, B., 2002. Presence of rotavirus hominis in sewage and water. Polish Journal of Environmental Studies 11, 751–755.

Kopac I., Ayuso-Gabella M. N., Salgot M. Integrating Disability Adjusted Life-years (DALYs) as a Tool for Human Health Risk Assessment in the RECLAIM WATER. Proceedings of the 6th Conference on Wastewater Reclamation and Reuse for Sustainability. October 9–12, 2007, Antwerp, Belgium.

Kremer, S., Pavelic, P., Dillon, P., Barry, K., 2008. Flow and Solute Transport Observations and Modelling from the First Phase of Flushing Operations at the Salisbury ASTR Site. Available from:. CSIRO: Water for a Healthy Country National Research Flagship <www.csiro.au/org/HealthyCountry.html>.

Le Corre K., de Heyder B., Masciopinto C., Aharoni A., Cikurel H., Zhao X., Salgot M., Ayuso Gabella M. N., Saperas N., Cartmell E., Jefferson B., Jeffrey P. Preliminary Results of Managed Aquifer Recharge with Reclaimed Wastewater in Five Operational Case Studies from Around the World. Proceedings of the 6th Conference on Wastewater Reclamation and Reuse for Sustainability. October 9–12, 2007, Antwerp, Belgium.

Lodder, W.J., Husman, A.M.D., 2005. Presence of noroviruses and other enteric viruses in sewage and surface waters in The Netherlands. Applied and Environmental Microbiology 71 (3), 1453–1461.

Mehnert, D.U., Stewien, K.E., 1993. Detection and distribution of rotavirus in raw sewage and creeks in Sao Paulo, Brazil. Applied and Environmental Microbiology 59 (1), 140–143.

Mons, C., Dumetre, A., Gosselin, S., Galliot, C., Moulin, L., 2009. Monitoring of cryptosporidium and giardia river contamination in Paris area. Water Research 43 (1), 211–217.

Nasser, A.M., Oman, S.D., 1999. Quantitative assessment of the inactivation of pathogenic and indicator viruses in natural water sources. Water Research 33, 1748–1752.

NOM-014-CNA, 2003, Proposal of the Mexican Official Norm PROY-NOM-014-CNA-2003, requirements to recharge aquifers.

Natural Resource Management Ministerial Council, Environment Protection and Heritage Council and Australian Health Ministers Conference NRMMC-EPHC-AHMC, 2006. Australian guidelines for water recycling: managing health and environmental risks (phase1). Available from: <http://www.ephc.gov.au/sites/default/files/WQ_AGWR_GL__Managing_Health_Environmental_Risks_Phase1_Final_200611.pdf>.

NRMMC-EPHC–NHMRC, 2008. Australian Guidelines for Water Recycling: Managing Health and Environmental Risks. Phase 2A: Augmentation of Drinking Water Supplies. Available from: Natural Resource Management Ministerial Council, Environment Protection and Heritage Council, and National Health and Medical Research Council <www.ephc.gov.au/taxonomy/term/39>.

NRMMC-EPHC–NHMRC, 2009a. Australian Guidelines for Water Recycling: Managing Health and Environmental Risks. Phase 2B: Stormwater Harvesting and Reuse. Available from: Natural Resource Management Ministerial Council, Environment Protection and Heritage Council and National Health and Medical Research Council <www.ephc.gov.au/taxonomy/term/39>.

NRMMC-EPHC–NHMRC, 2009b. Australian Guidelines for Water Recycling: Managing Health and Environmental Risks. Phase 2C: Managed Aquifer Recharge. Available from:. Natural Resource Management Ministerial Council, Environment Protection and Heritage Council and National Health and Medical Research Council <www.ephc.gov.au/taxonomy/term/39>.

Page, D., Barry, K., Pavelic, P., Dillon, P., Chassagne, A., 2008. Preliminary Quantitative Risk Assessment for the Salisbury Stormwater ASTR Project. Available from: Water for a Healthy Country National Research Flagship Report CSIRO <http://www.clw.csiro.au/publications/waterforahealthycountry/2008/wfhc-Salisbury-ASTR-HACCP.pdf>.

Page, D.W., Vanderzalm, J., Barry, K., Levett, K., Kremer, S., Ayuso-Gabella, M.N., Dillon, P., Toze, S., Sidhu, J., Shackleton, M., Purdie, M., Regel, R., 2009. Operational Residual Risk Assessment for the Salisbury Stormwater ASTR Project. CSIRO: Water for a Healthy Country National Research Flagship. Available from: <http://www.csiro.au/files/files/psgp.pdf>.

Pavelic, P., Dillon, P., Robinson, N., 2004. Groundwater Modelling to Optimise Well-Field Design and Operation for the ASTR Trial at the Greenfield Railway Station Site. CSIRO, Salisbury, South Australia. CSIRO Land and water Technical Report No. 27/04.

Plutzer, J., Karanis, P., Domokos, K., Torokne, A., Marialigeti, K., 2008. Detection and characterisation of *Giardia* and *Cryptosporidium* in Hungarian raw, surface and sewage water samples by IFT, PCR and sequence analysis of the SSUrRNA and GDH genes. International Journal of Hygiene and Environmental Health 211 (5–6), 524–533.

Robertson, L.J., Hermansen, L., Gjerde, B.K., 2006. Occurrence of *Cryptosporidium* Oocysts and *Giardia* Cysts in sewage in Norway. Applied and Environmental Microbiology 72 (8), 5297–5303.

Schets, F.A., van Wijnen, J.H., Schijven, J.F., Schoon, A., de Husmant, A.M., 2008. Monitoring of waterborne pathogens in surface waters in Amsterdam, The Netherlands, and the potential health risk associated with exposure to *Cryptosporidium* and *Giardia* in these waters. Applied and Environmental Microbiology 74 (7), 2069–2078.

Smeets, P., Rietveld, L., Hijnem, W., Medema, G., Stenström, T.-A., 2006. Efficacy of Water Treatment Processes, MICRORISK Research Report. Available from: <http://www.microrisk.com/uploads/microrisk_efficacy_of_water_treatment_processes.pdf>.

Swierc, J., Page, D., Van Leeuwen, J., Dillon, P., 2005. Preliminary Hazard Analysis and Critical Control Points Plan (HACCP) – Salisbury Stormwater to Drinking Water Aquifer Storage Transfer and Recovery (ASTR) Project. CSIRO. CSIRO Land and Water Technical Report No. 20/05.

Toze, S., Hanna, J., Smith, T., Edmonds, L., McCrow, A., 2004. Determination of water quality improvements due to the artificial recharge of treated effluent. In: Wastewater Reuse and Groundwater Quality, vol. 285. IAHS Publication. 53–60.

Toze, S., Sidhu, J., Shackleton, M., Hodgers, L., 2009. Decay of Enteric Pathogens in Urban Stormwater Recharged to an Aquifer using Aquifer Storage, Transfer and Recovery. CSIRO: (Water for a Healthy Country National Research Flagship).

Van Houtte, E., Verbauwhede, J. Artificial recharge of treated wastewater effluent enables sustainable groundwater management of a dune aquifer in Flanders, Belgium. 5th International Symposium on Management of Aquifer Recharge, Berlin, Germany 11–16th June 2005.

Van Houtte, E., Verbauwhede, J., 2008. Operational experiences with indirect potable reuse at the Flemish coast. Desalination 218, 198–207.

WHO (World Health Organisation), 2004. Guidelines for Drinking-water Quality. In: Recommendations, third ed., vol. 1. World Health Organisation, Geneva.

WHO (World Health Organisation), 2005. Water Safety Plans – Managing Drinking-Water Quality from Catchment to Consumer. World Health Organisation, Geneva.

Yates, M.V., Sterzenbach, L.D., Gerba, C.P., Sinclair, N.A., 1990. The effect of indigenous bacteria on virus survival in ground water,. Journal of Environmental Science and Health 25, 81–100.

Zwietering, M.H., van Gerwen, S.J.C., 2000. Sensitivity analysis in quantitative microbial risk assessment. International Journal of Food Microbiology 58, 213–221.