Hydraulic evaluation of aquifer storage and recovery (ASR) with urban stormwater in a brackish limestone aquifer

Paul Pavelic · Peter J. Dillon · Karen E. Barry · Nabil Z. Gerges

Abstract A 5-year aquifer storage and recovery trial at Andrews Farm in South Australia involving the injection of more than 250 ML (250,000 m^3) of fresh but turbid stormwater into a brackish limestone aquifer over 4 years and recovery of 150 ML in the fifth provided the opportunity to evaluate rates of clogging and unclogging and the potential to recover water suitable for irrigation supplies. Results reveal there is some clogging by injected sediment, but only to a relatively small degree considering the high suspended solid concentrations and moderate aquifer transmissivity. This clogging was offset by increased matrix porosity through calcite dissolution and by routine well redevelopments after each 40 ML of injection. No significant microbial clogging occurred. Breakthrough responses at three observation wells and the proportion of injectant in the recovered water were determined from chloride data. Temperature and caliper profiles clearly indicate the heterogeneous nature of the aquifer that is attributed, in part, to sand removal during the initial well development. The recovery efficiency was greater than 60%. The trial demonstrates that urban stormwater containing high and variable particulate levels, which receives only passive pre-treatment and is not disinfected, can be used to freshen a heterogeneous brackish aquifer to create a useful water resource.

Résumé Un essai d'injection et de récuperation a été réalisé sur cinq ans sur le site de la ferme Andrews dans le sud de l'Australie. Plus de 250 ML (250,000 m^3) d'eau pluviale d'orage, douce mais turbide, ont été injectés dans un aquifère calcaire saumâtre, sur une période de quatre ans, suivie d'une récupération de 150 ML la cinquième année. L'essai a permis d'évaluer les taux de colmatage et de décolmatage ainsi que la possibilité de disposer d'eau pouvant être utilisée pour l'irrigation. Les résultats montrent un colmatage par les sédiments injectés, cependant à un faible degré si on considère les concentrations élevées de matières en suspension et la transmissivité moyenne de l'aquifère. Ce colmatage était compensé par l'augmentation de la porosité de la matrice due à la dissolution du calcaire et par des développements réguliers de l'ouvrage après chaque injection de 40ML. Aucun colmatage microbien important n'a été observé. La réaction aux trois puits d'observation et le taux de dilution avec la nappe naturelle ont été déterminés grâce aux chlorures. Les profils de la température et du diamètre du puits montrent clairement la nature hétérogène de l'aquifère, laquelle s'explique en partie par un désensablement lors du développement initial de l'ouvrage. Le rendement de la récupération fut supérieure à 60%. L'essai démontre que de l'eau pluviale urbaine, présentant des niveaux élevés et variables de particules, après un traitement passif et non désaffectée, peut être utilisée pour adoucir un aquifère saumâtre hétérogène et constituer une importante ressource en eau.

Resumen Una prueba de abastamiento y recuperación de cinco años en un acuífero en Andrews Farm en South Australia que involucró la inyección de más de 250 ML de agua pluvial fresca pero túrbida a un acuífero de caliza de agua salina por cuatro años y la recuperación de 150 ML en el quinto año proporciona la oportunidad de evaluar tasas de obstrucción y desobstrucciòn y el potencial de recuperar agua adecuada para irrigación. Los resultados revelan que hay alguna obstrucción por el sedimento inyectado, pero sólo en un grado menor, considerando las concentraciones altas de sólidos suspendidos y la transmisividad moderada del acuífero. Esta obstrucción se balanceó por un incremento en la porosidad de la matriz debido a la disolución de calcita y por el redesarrollo rutinario de pozos después de cada 40 ML de inyección. No ocurrió ninguna obstrucción de microbios importante durante la prueba. La respuesta de breakthrough en tres pozos de observación y la proporción de agua inyectada en el agua recuperada se determinó por los datos de cloro. Los perfiles de temperatura y caliper demuestran claramente la naturaleza heterogénea del acuífero, que se atribuye en parte a la arena extraída durante el desarrollo

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de pozos inicial. La eficacia de recuperación es mayor a 60%. La prueba demuestra que las aguas pluviales urbanas que contienen niveles de partículas altas y variables que reciben solamente pretratamiento pasivo y que no se desinfectan, pueden ser utilizadas para incrementar la frescura de un acuífero con agua salina heterogéneo para crear un recurso útil de agua.

Keywords Carbonate rocks · Clogging · Recovery efficiency · Urban groundwater · Water banking

Introduction

Adelaide is a city of just over 1 million people with a semi-arid climate (500 mm/year average rainfall; 1,800 mm/year potential evaporation) that relies on surface water resources that are vulnerable due to rainfall variability. Owing to the limited opportunities for further expansion of traditional water resources, new approaches to water management have been considered. With the average annual volumes of urban stormwater runoff generated in metropolitan Adelaide estimated to be comparable to the potable demand, there is potential to harvest some of the water in winter months to meet part of the large non-potable (irrigation) water requirements of the city during the drier summer months, if it can be effectively stored. Stormwater would be captured on-site and passively treated using constructed wetlands and injected into confined aquifers for subsequent recovery from the same well for non-potable supplies in a practice that has become known as aquifer storage and recovery (ASR; after Pyne 1995). To capitalise on this concept, an ASR demonstration project commenced at the Andrews Farm site in Adelaide, South Australia in the early 1990s to evaluate the technical feasibility, environmental sustainability and economic viability of stormwater ASR (Dillon and Pavelic 1996; Dillon et al. 1997).

A host of technical issues may arise when fresh, particle-laden stormwater is injected into a brackish carbonate aquifer (Dillon et al. 1997; Pyne 1995). Clogging of injection wells is a commonly reported problem, particularly when waters of impaired quality are used (Olsthoorn 1982; Pérez-Paricio and Carrera 1999). The few field studies that have been undertaken on clogging have tended to be of relatively brief duration, and those that are more extended rarely take the measurements necessary to quantify the rates of clogging or the success of any remedial or preventative measures. Where the receiving groundwater quality is too saline to meet the intended use of the recovered water, the quantity of water that can be usefully recovered, defined as the 'recovery efficiency' (RE) becomes an important consideration. Recovery efficiency is known to vary enormously between ASR schemes due to a range of hydrogeological and operational factors (Pavelic et al. 2002). Determining the reasons why wells clog and the estimation of recovery efficiencies in heterogeneous environments continues to remain a challenge for practitioners of ASR.

This paper reports on the hydraulic data collected during the 5-year ASR trial at the Andrews Farm site with the aim of: (1) establishing the nature and extent of well clogging and success of unclogging procedures, and (2) evaluating the fate of the injectant in the aquifer and the degree of mixing between the injectant and the ambient groundwater. Geochemical aspects of the study were previously reported by Herczeg et al. (2004).

Site description and experimental conditions

The aquifer targeted for storage, known locally as the "T2" aquifer, is the second of a series of Tertiary marine sediments overlain by 9 m of highly plastic clay. The T2 is intersected locally at a depth of 105 m and consists of interbedded sequences of variably cemented limestone and sand. A bulk aquifer transmissivity of 180 m^2/day and storage coefficient of 5×10^{-4} have been determined through aquifer pump testing (Gerges et al. 1995). The direction of regional groundwater flow is approximately to the west, with an average hydraulic gradient of ~0.002.

The injection/recovery (ASR) well was completed without casing in the upper sequence (108–127 m depth). Although the well initially extended to a depth of 140 m, the base of the formation is subject to collapse due to poor consolidation and the bottom 13 m backfilled soon after drilling. Three observation wells were drilled along a transect at distances of 25, 65 and 325 m down-gradient of the ASR well to monitor piezometric heads and water quality (Figs. 1 and 2).

The T2 aquifer is heterogeneous in nature. Investigations at a nearby ASR site suggest a layered stratigraphy and at least two orders of magnitude variation in the hydraulic conductivity, with the most silicious and least cemented zones generally being the most permeable (Pavelic et al. 2006). The efficiency in the vicinity of the ASR well was significantly increased by extensive well development in an effort to enhance the recharge capacity. The high variability in cementation of the aquifer resulted in significant quantities of sand being recovered from the least cemented parts of the aquifer (Gerges et al. 1995). Similar occurrences have been reported at other ASR sites in limestone formations (e.g. Campbell et al. 1997). Caliper log data before and after well-development demonstrate that sand recovery significantly enhanced the well diameter over the interval from 120–123 m (Fig. 3). Flow conduits were thus created, resulting in direct hydraulic communication between the ASR and the 25 m observation well; and consequently, minimal piezometric head-difference and rapid solute breakthrough occurred, as will be demonstrated later. The connectivity between these two wells was verified at the 25-m well within minutes of the start of injection with downhole video camera footage showing colloidal-rich waters, easily identifiable by their red-brown colour as characteristic of the injectant.

All surface flow from the 55-km^2 surface water catchment is routed through three interconnected storm-

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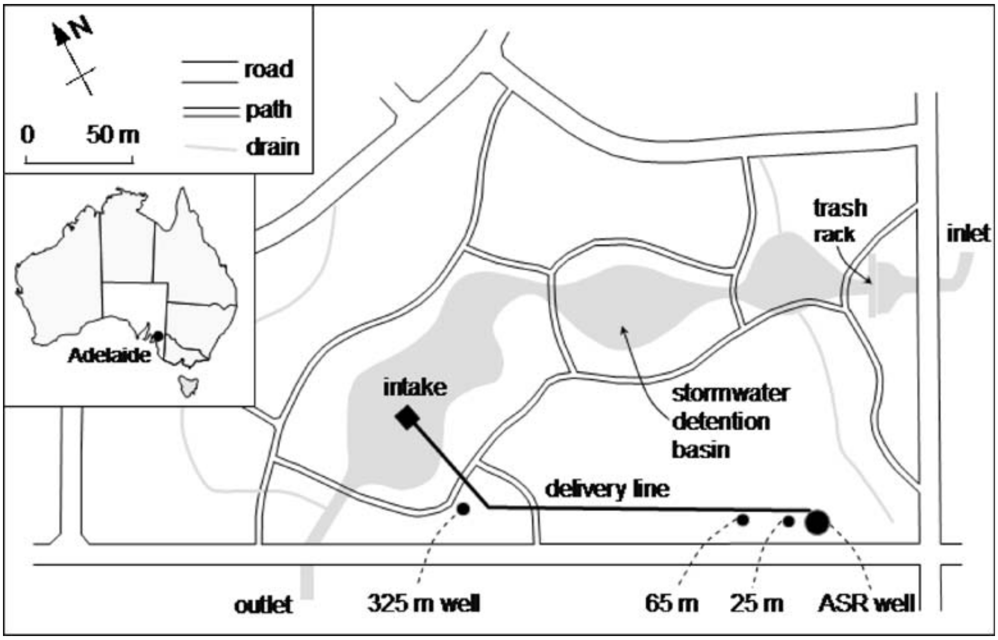

Fig. 1 Location map of the Andrews Farm site

water detention basins that serve to mitigate flooding and provide passive pre-treatment of the stormwater prior to injection. The first injection was in August 1993 (a small test with treated potable water), and then stormwater was injected later in that same year and intermittently during the winter and spring periods over the next 3 years until October 1996 (Fig. 4). A net total of 256 ML (256,000 m^3) of water was injected over 4 years at average rates of injection of between 1,300 and 1,700 m^3/day (15–20 L/s; Barry et al. 2002). Progressively larger volumes in each year reflect the succession of winter rainfall amounts and the capacity of the active storage of the detention system. The rationale for allowing 4 years of injection prior to recovery was to create a sufficient freshwater storage zone to achieve a reasonable recovery efficiency. A major recovery phase was conducted in the following year (July 1997 to July 1998) when a total volume of 150.8 ML was extracted (Fig. 4). Aside from pre-treatment within the basin and passage through a 100-µm stainless steel filter and geotextile filter

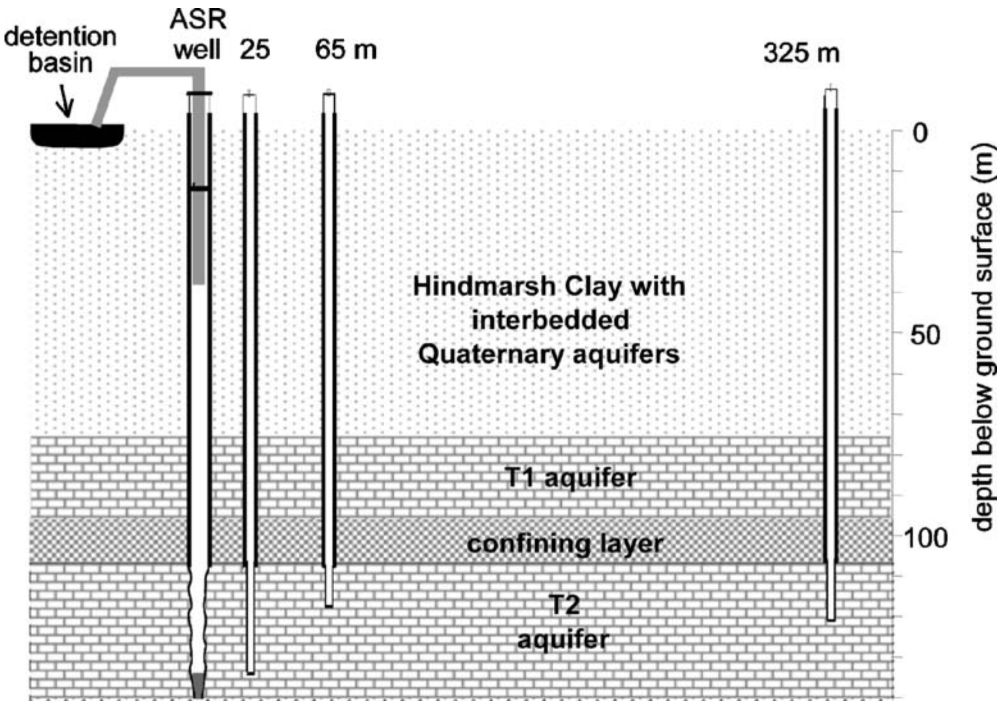

Fig. 2 Schematic vertical cross-section of the Andrews Farm trial site showing the ASR well, three observation wells and stormwater detention basin

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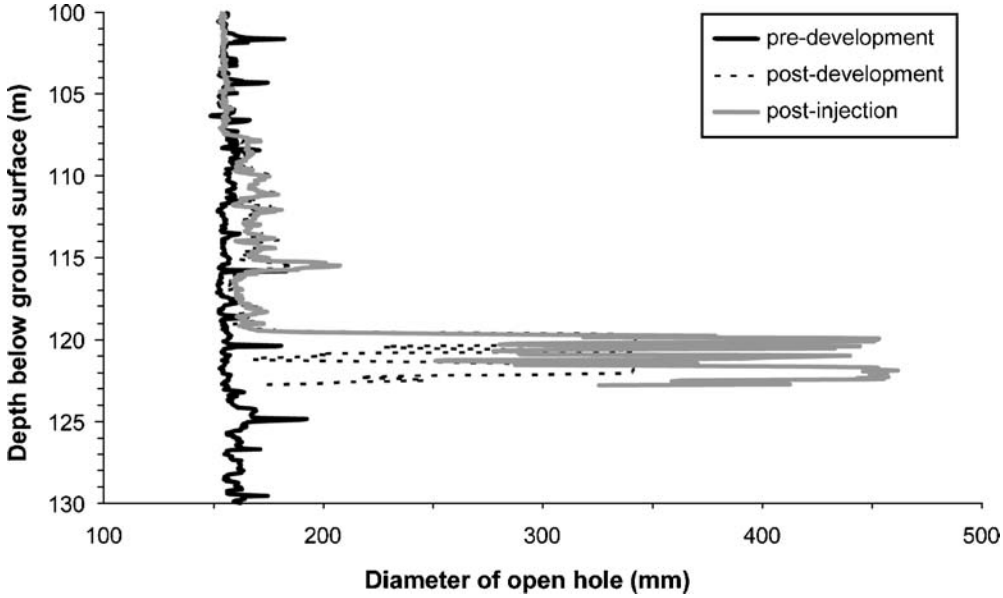

Fig. 3 Caliper log profiles over the open interval of the ASR well on three occasions: *pre-development* (before casing and development); *post-development* (after casing and development); and *post-injection* (after 4 years of injection). Note the limited span (340 mm) of the tool used for the *post-development* profile

fabric to screen out gross pollutants, there was no other treatment or disinfection prior to injection.

Characteristically, the stormwater is colder, has higher concentrations of dissolved oxygen, suspended solids, nitrogen, organic carbon, coliforms and higher pH and turbidity than the brackish ambient groundwater, but has lower concentrations of iron and total dissolved solids (Table 1).

Well clogging and unclogging

Calculation of hydraulic conductivity changes

Rates of clogging are quantified in terms of the temporal changes in hydraulic conductivity (K). For the periods of time where the hydraulic response to injection has reached quasi-steady state, the average K of the aquifer materials between the ASR and observation wells (i.e. 0–25, 25–65, 65–325-m) can also be determined by rearranging the well-known Thiem equation:

$$K_{(r_1 \leftrightarrow r_2)} = \frac{-Q_i(\ln r_2 - \ln r_1)}{2\pi b(h_{r1} - h_{r2})} \quad (1)$$

where Q_i is the rate of injection into the ASR well, b is the thickness of the aquifer, r_2 and r_1 are the radial distances of the two observation wells from the ASR well, such that $r_2 > r_1$, and $h_{r1,r2}$ are the piezometric heads measured concurrently at the wells. It is assumed that the effects of temperature and salinity differences between observation wells on the viscosity and density of the groundwater are small and can be neglected.

Time to reach quasi steady-state, calculated from the equation reported by Harvey et al. (1994) and using the aquifer parameters determined from the pumping test occurs after 1.0 h at 25 m, 7.4 h at 65 m, and 7.7 days at

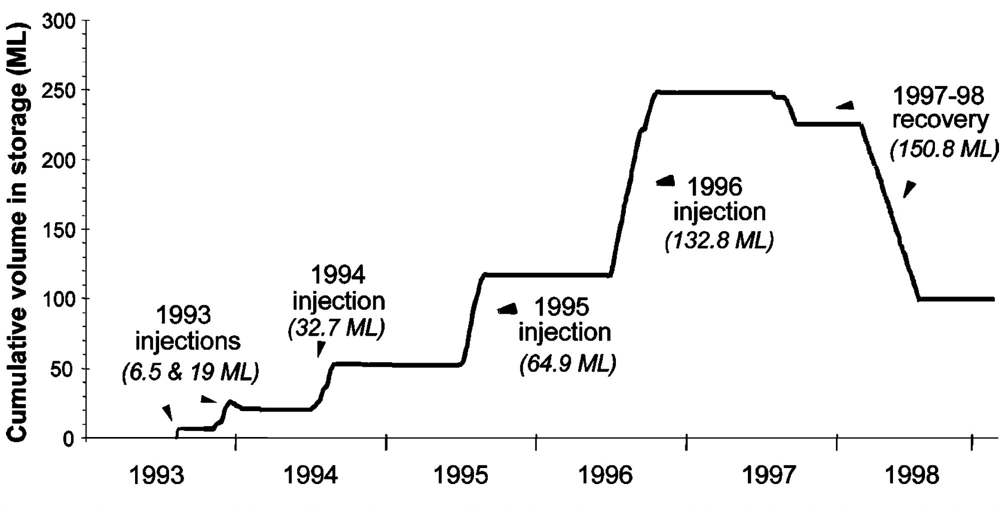

Fig. 4 Cumulative net volume of water injected into the T2 aquifer during the 5-year trial period. Injections took place between 1993 and 1996 followed by recovery of around 60% of the injected volume between 1997 and 1998

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325 m. Declines in injection rate are relatively minor over these time-scales and create minimal error in the calculated value of K.

Hydrologic response and water quality

During the first 12 months of stormwater injection (1993 and 1994), operational problems were experienced and operational procedures adapted to overcome them. The most significant was caused by zooplankton which bloomed within the basins that caused extensive blockage of the 100-μm screen and ASR well and required extended redevelopment. Shielding the pump intake with geotextile fabric avoided further recurrence. Because of these difficulties as well as some data losses, the initial potable water injection test is compared with stormwater injections in 1995 and 1996.

The history of flow rates, piezometric head increase at the ASR well, hydraulic conductivities (from Eq. (1)) and the suspended solids (SS) content of the injectant over the course of each event/year is shown in Fig. 5. Flow rates were maintained at a constant 1,340 m^3/day for the brief potable water test. Flow rates during the 1995 (65 ML) stormwater injection steadily declined by 250 m^3/day (15%) from the initial value of 1,660 m^3/day and during 1996 (133 ML) by 270 m^3/day (17%) from the initial value of 1,590 m^3/day.

The time-varying hydraulic conductivities for the 0–25, 25–65 and 65–325-m interval are shown in Fig. 5. K data for the 25–65-m and 65–325-m interval show there was no change in K beyond the variance associated with measurement errors. For the 0–25-m interval, K values were elevated with respect to the other intervals as expected due to the well development, but no significant decline was observed over time. Gradual rises in piezometric head and associated falls in injection rate are suggestive of hydraulic conductivity reductions due to gradual clogging, but were not observed in the K data due to diminishing headloss over time in the 0–25-m interval. The high degree of preferential flow may suggest the Thiem equation to be inappropriate in this interval.

Table 1 Mean quality of injected stormwater in 1994 and ambient groundwater at the 65-m observation well

Parameter[a]	Stormwater[b]	Groundwater[d]
pH (-)	8.2	7.7
Temperature (°C)	10.1	24.0
Dissolved oxygen	9.7	0.7
Turbidity (NTU)	50	13
Suspended solids	29	13
Total dissolved solids	190	2340
Chloride	39.8	1152
Bicarbonate	118	228
Calcium	23.6	237
Fluoride	0.31	0.18
Iron-total	0.16	0.45
Nitrogen-total	0.77	0.37
Dissolved organic carbon	4.3	1.2
Total coliforms (cfu/100 mL)	620[c]	0

[a] mg/L unless otherwise stated

[b] Mean concentration in 1994

[c] Geometric mean concentration in 1994

[d] 65-m observation well (pre-injection)

The daily variations in SS concentrations presented in Fig. 5 showed that SS concentrations were typically below 100 mg/L in 1995 and around 50–200 mg/L in 1996. The practice of using open earth channels for drainage combined with soil erosion in the upland areas of the catchment can result in highly turbid runoff waters. SS concentrations in the injectant were up to 200 mg/L in 1995 and 600 mg/L in 1996. On average, SS concentrations were around 2.5 times higher in 1996 than in 1995, whilst the mass flux was around 5 times higher. Annual averages ranged from <0.1 mg/L for the potable water injection to between 29 and 169 mg/L for the stormwater injections in 1994 and 1996 respectively (Table 2).

Considering the hydraulic data in light of the particulate data shows that the relationship between the hydrologic response and the quality of the injectant is relatively subdued. One notable exception occurred at around day 24 in 1995, when a clearly defined slug of more turbid water was injected that caused heads in the ASR well to increase by several metres. Interestingly, this type of occurrence was not as apparent in 1996, even though concentrations generally exceeded those of the previous year.

Redevelopment to unclog the ASR well was undertaken whenever flow rates were considered insufficient or when piezometric heads approached the ground surface. The method of redevelopment used involved jetting the well with compressed air and is referred to as 'airlifting'. There were a total of six airlifts during the 4 years of injection, or on average, one airlift per 40 ML. Two smaller redevelopments occurred during the course of injection in the latter 2 years, and the more substantive redevelopment occurred at the end of each year of injection. It is clear from the Q_i data (Fig. 5) that injection rates were partially restored by mid-injection airlifts in 1995 and 1996, and fully restored by the end of year airlifts.

Membrane filtration index (MFI) measurements indicate the potential of a given water to clog wells through filtration of particulate material and the development of a filter cake. The higher the MFI value, the greater the risk of clogging (Schippers and Verdouw 1980; Dillon et al. 2001). The MFI value measured on a stormwater sample in 1999 (i.e. after the trial had concluded) was 440 s/L^2 and the corresponding SS content was only 20 mg/L, which was lower than the annual average of the stormwater injections. MFI values during the 4 years of injection are estimated to have ranged from 400 to 2,600 s/L^2 based upon turbidity and TOC data using a regression equation presented by Dillon et al. (2001). These values are several orders of magnitude greater than for ASR studies in the Netherlands or the USA and the generally accepted limit of 3 s/L^2 by water utilities in the Netherlands for injection into dune sands (van Duijvenbode and Olsthoorn 1998). Hutchinson and Randall (1995) clearly demonstrate the dependence of both aquifer permeability and MFI of recharge water on the rate of physical clogging of ASR wells. This implies that higher values of MFI were

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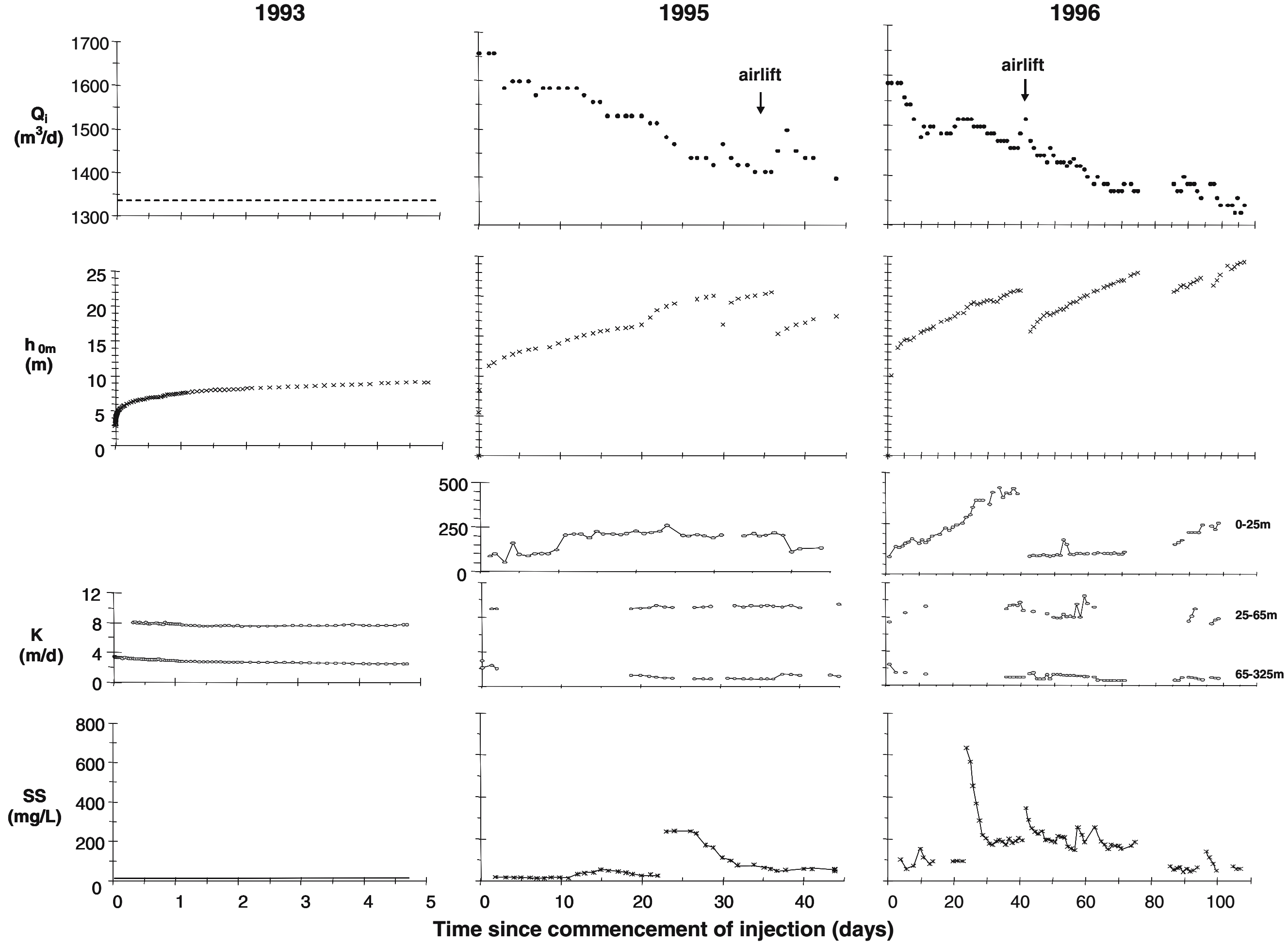

Fig. 5 Temporal changes in injection rate (Q_i), piezometric head at the ASR well (h_{0m}), hydraulic conductivity (K) in the 0–25, 25–65 and 65–325-m intervals, and suspended solids content (SS) of injectant during the potable water injection in 1993 and stormwater injections in 1995 and 1996. The timing of the mid-injection airlifts in 1995 and 1996 are indicated

tolerated at Andrews Farm as a result of the higher permeability of the limestone aquifer.

Unclogging effects by carbonate reactions

From the perspective of well clogging in a carbonaceous aquifer, the most important chemical reaction to consider when oxygenated low-salinity stormwater containing organic matter is injected into an anaerobic brackish aquifer, is calcite dissolution or precipitation. This can be described by the reaction:

$$CaCO_3 + CO_2 + H_2O \leftrightarrow Ca^{2+} + 2HCO_3^- \qquad (2)$$

Table 2 Mass balance of particulates added to the T2 aquifer through injection, and removed from the aquifer through well redevelopments, recovery events and dissolution of calcite for the period from 1993 to 1998

Year	Mass injected (kg)	Mass recovered or dissolved (kg)					
		Redevelopment		Recovery	Net loss of solids	Dissolution	Total aquifer loss
		Mid-	Post-				
1993[a]	<1	1	1	–	1	NA	1
1993	2,620	NA	NA	–	−2,620	610	−2,010
1994	950	–	990	–	40	1,050	1,090
1995	4,280	460	1,820	–	−2,000	2,080	80
1996	22,400	10,150	4,930	–	−7,320	4,250	−3,070
1997–98				14,400	14,400		14,400
Total	30,250	10,610	7,740	14,400	2,500	7,990	10,490

[a] Potable water injection

[b] NA = not available

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The direction this reaction takes can either alleviate (right) or contribute (left) to clogging (Vanderzalm et al. 2006).

Estimates of the calcite added to, or subtracted from, the groundwaters at the ASR and 25 m wells as a result of geochemical reactions through measurement of calcium, magnesium and bicarbonate concentrations were derived from the work of Herczeg et al. (2004). Time-series data show there was generally more calcium, magnesium and bicarbonate in the groundwater around the ASR well than expected by conservative mixing between low-salinity stormwater and the higher-salinity-receiving groundwater. This is to be expected given the consistent undersaturation of the source water with respect to calcite ($SI_{calcite}$ values ranging from −0.1 to −0.6). Evidence presented by Herczeg et al. (2004) shows that dissolution tends to be greatest during and immediately following injection, and is accentuated by production of carbon dioxide from oxidation of organic matter. When the groundwater returns to an anaerobic condition, dissolution rates steadily decline. On average, the stoichiometric estimate of calcium plus magnesium produced by reactions was ~2 meq/L (Herczeg et al. 2004). Assuming an equi-molal ratio of calcium and magnesium, the quantity of calcite minerals dissolved from the aquifer in the different years has been estimated to range from 610 kg in 1993 to 4,300 kg in 1996. The total quantity of calcite dissolved in the ~250 ML of stormwater injected is around 8 tonnes. Given that calcite makes up approximately 20% of the aquifer material, and if a bulk density of 1.5 g/cm^3 is assumed, then approximately 4 m^3, or <0.01% of the aquifer mass was dissolved within the injection zone.

Dissolving calcite, which is present as a cementing agent for the primary minerals such as quartz (the co-dominant mineral in the aquifer), has the potential to impact on ASR in at least two ways. Firstly, the porosity, and hence the permeability of the aquifer would probably increase, offsetting the clogging that occurs by filtration of injected particulates. Secondly, mobilisation of sand can cause problems with sand production placing stresses on pumps, the stability of the well, and possibly, in time, the overlying aquitard. In fact, the persistence of coarse particles in the water extracted during the recovery phase lead to breakdown of a submersible pump within days of pumping. Extended airlifting was required to remove the mobile sediment from around the well face and the replacement pump operated satisfactorily for the remainder of the recovery period. Although the average increase in the effective porosity over the storage zone would have been extremely small (<0.01%), its significance depends on the spatial distribution of dissolution in the aquifer.

Sediment mass balance

The mass of particulate matter injected into, and extracted from, the ASR well was determined in each year, with the exception of the recovery in the first year (due to the operational difficulties previously outlined). Results are presented in Table 2. The mass flux of injected sediment ranged from an assumed <0.1 kg during the potable water injection, up to an estimated 22,400 kg for stormwater injected in 1996. The quantities of sediment extracted by well redevelopment were similar or lower than that injected, but by the end of the trial there was a net loss of solids from the system of 2,500 kg, and a further loss due to calcite dissolution of 8,000 kg. The maintenance of injection efficiencies at the commencement of each year in spite of the high particulate loadings entering the well indicates that well redevelopments combined with calcite dissolution are effective in managing clogging. Net loss of aquifer material may, however, be an issue for well and aquitard stability in the longer term.

The sediment in backwash water is derived both from the stormwater and the aquifer matrix (as is apparent from visual inspection of samples). Consequently, an attempt was made to distinguish between these two different sources by examining their characteristic particle size distributions. This was feasible to do in 1994 because the two sources are clearly different (Fig. 6). The cumulative particle size distribution curves for the aquifer and stormwater differ markedly between the smaller stormwater particles and the larger aquifer particles, with median particle sizes of ~4 μm and ~150 μm, respectively. Airlift data shows that the particle size distribution after 11 min of pumping, when SS concentrations had peaked at 12,500 mg/L, consists of a combination of both stormwater and aquifer derived sediments, but by 352 min is derived exclusively from the aquifer. About 99% of the recovered sediments are derived from the aquifer with only ~1% of sediments derived from stormwater. This suggests that the majority of the stormwater sediments do not accumulate around the well, but penetrate further into the aquifer to distances where airlifting is ineffective in their removal. As a consequence, much of the injected sediments are retained within the aquifer where they are deposited within interstitial pore-spaces.

Microbiological growth and decay

Microbial clogging occurs through the proliferation of bacteria and the polysaccharides (biofilms) they produce (Baveye et al. 1998; Pavelic et al. in press). The significance of microbial clogging under field conditions is typically investigated through the sampling of recovered waters and comparing this with conditions prior to injection (Rebhun and Schwarz 1968; Vecchioli 1970). A similar approach was chosen for this study, whereby total coliform bacteria were monitored during redevelopment events in 1994, 1995 and 1996, as previously discussed (Table 3).

Coliform numbers in the injectant during 1994 ranged from 160 to 5,800 colony-forming units (cfu) per 100 ml, with a geometric mean value of 620 cfu per 100 ml (Table 3). The redevelopment event which took place 28 days after the conclusion of injection, resulted in a peak of 17,000 cfu per 100 ml within a few minutes of pumping. By simple mass balance, the quantity of coliforms recovered is estimated at 1.7×10^9 cfu, or ~1%

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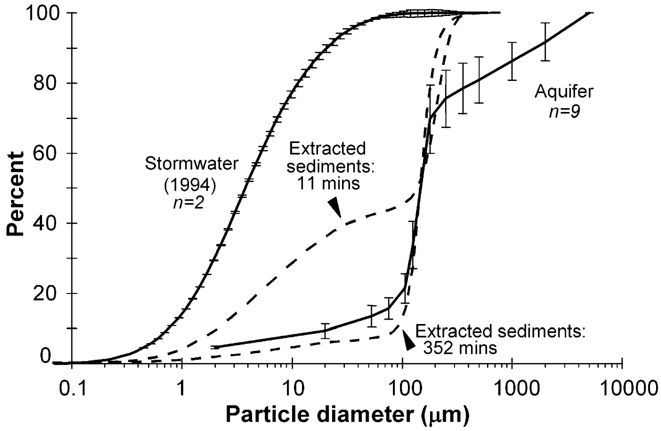

Fig. 6 Cumulative particle size distributions for the aquifer (from nine samples of core material), injected stormwater (two samples in 1994) and backwash water during redevelopment following 1994 injection (at 11 and 352 min into the airlift). Standard deviations are indicated by the *vertical bars*

of that introduced to groundwater. Coliform numbers were three orders of magnitude lower for the redevelopment event which took place 310 days after the 1995 injection had ended. For the final redevelopment, 293 days after the 1996 injection had ended, the numbers were again very low and peaked at only 85 cfu per 100 mL. The ratio of the redevelopment-recovered to injected coliforms following the 1994 to 1996 injections was 1×10^{-2}, 5×10^{-5} and 2×10^{-5}. This would suggest an absence of microbial growth around the ASR well. These results differ significantly from the case of the study by Vecchioli (1970), where many times the number of bacteria injected was subsequently recovered. In that case, the aquifer was a fine-medium sand and the injectant was tertiary treated sewage effluent (although still high in nutrients and colloidal organic matter). In our case, it is most likely that the flow regime around the well and the predominance of inorganic particles in the injectant do not allow for such a well-defined biofilm to develop.

Monitoring of the post-1996 injection redevelopment event included a greater diversity of microorganisms. Responses similar to that shown for coliforms were repeated for each of the organisms tested. *Pseudomonas* spp. counts were not sufficiently high ($<2\times10^4$ cfu per 100 mL) as to warrant concern from a clogging perspective. Heterotrophic iron bacteria were found to be present at concentrations of up to ~10^6 cfu per mL, but these levels were not greater than that found in the injectant or the ambient groundwater.

Fate of the injected water

0–25, 65 and 325-m breakthroughs

The history of chloride concentrations in groundwater at the ASR and observation wells for the duration of the experiment are shown in Fig. 7. Chloride is an ideal tracer of the injected water at this site due to the strong distinction in salinity between the two water types and the absence of evaporite minerals within the aquifer (Herczeg et al. 2004). The chloride concentration for the potable water injection was 93 mg/L and for the subsequent stormwater injections was between 25 and 40 mg/L. The chloride concentration of ambient groundwater ranged from 1,100 to 1,200 mg/L for all but the 325-m observation well, which had a lower concentration (~850 mg/L). This salinity gradient is significantly higher than the regional trend of the freshening of groundwater towards the west (Gerges 1999).

Over the 4 years of injection, the solute behaviour at each of the wells are distinctly different, apart from the ASR and 25 m wells, which were virtually identical due to their direct hydraulic connection. Permeable layers allow water to flow at higher velocities away from the well along discreet pathways and, therefore, breakthrough to the 25-m well is very rapid.

Mixing between the injectant and ambient groundwater occurs during the main storage periods. During the first 1–2 years of the trial, mixing was most apparent since the contrast in chloride between the injectant and the receiving groundwater was greatest. Whilst mixing also occurs in subsequent years, the effects appear to be less significant due to the replacement of ambient groundwater with injected stormwater. Only after the fourth year of injection (1996) was the ambient groundwater completely displaced in the 0–25-m zone and negligible change in chloride observed. Regional advective flow is insufficient to cause the changes observed (Pavelic et al. 2002), and therefore, the most likely explanation is that it arises from diffusive exchange with ambient groundwater entrained within the less permeable parts of the aquifer.

Table 3 Maximum and total number of coliform bacteria measured during injection and in backwash waters during redevelopment in relation to storage time in 1994, 1995 and 1996

Year	Injection		Redevelopment		Fractional recovery[a]	Storage time (days)[b]
	Max count (cfu/100 mL)	Total number (cfu)	Max count (cfu/100 mL)	Total number (cfu)		
1994	5,800	2.0×10^{11}	17,000	1.7×10^{9}	1×10^{-2}	28
1995	2,200	5.2×10^{11}	40	2.8×10^{7}	5×10^{-5}	310
1996	38,000	3.7×10^{12}	85	6.1×10^{7}	2×10^{-5}	293

[a] Ratio of the redevelopment-recovered to injected coliforms

[b] Time interval between the end of injection and start of redevelopment

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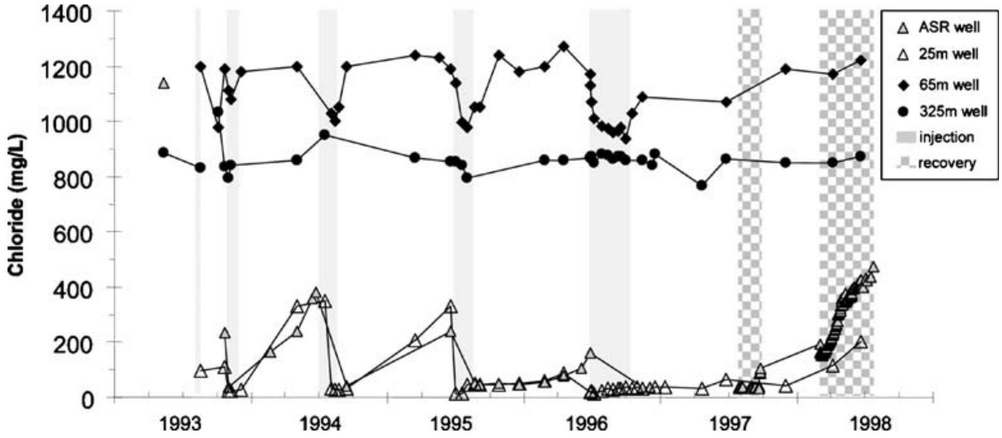
Fig. 7 Chloride concentrations at the ASR well and three observation wells from 1993 to 1998. Periods of injection and recovery are indicated by the *shaded* backgrounds



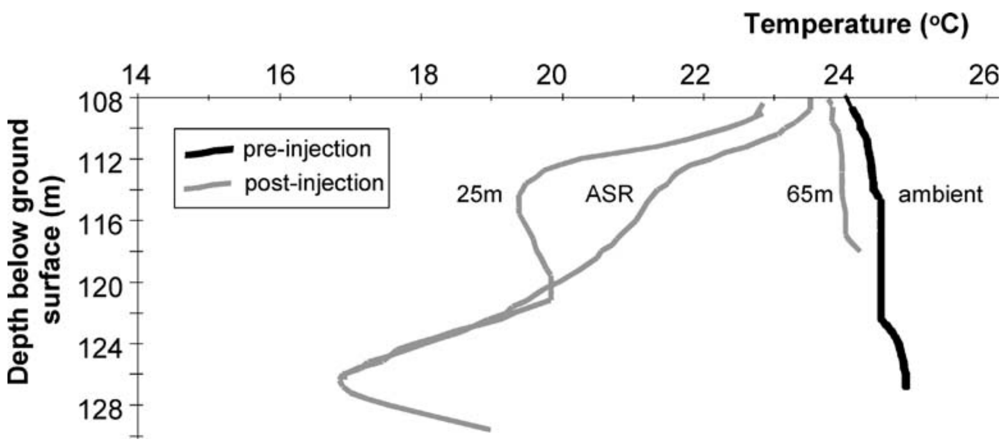
Water quality data are flow-weighted averages over the open interval of each well. Thus, breakthrough of injectant to observation wells depends on the permeability of the strata. The best evidence of a layered heterogeneity structure at this site comes from the temperature profiles collected before and after potable water injection. Figure 8 shows the impact of the 14°C injectant on the ambient groundwater at 24–25°C. Since the greater declines in temperature are indicative of higher rates of flow, the coldest water, found at a depth of around 127 m, must be associated with the layer of highest permeability. Temperatures fail to decline to that of the injected water due to heat loss from the solid phase. Responses for the ASR and 25-m wells are similar as expected.

A temperature decline in the order of ~0.5°C is observed at 65 m, which is equivalent to a mixing fraction (f) value of ~0.05 is similar to that found from chloride data following the 6.5 ML potable water injection event of 0.09 (refer to Fig. 7). The response at the 65-m well also exhibited dual-porosity characteristics. Partial breakthrough occurred within days of each successive injection (Fig. 9). The breakthrough response was virtually identical in each successive year and peak values of f increased only marginally from 0.18 to 0.22 between 1994 and 1996. Although the injectant is first detected within a few days of each year of injection, it is only in the largest event in 1996 that the breakthrough curve was most fully developed, although peak f values were still less than 0.25. Assuming this to be an upper limit for f (since the majority of the injected water failed to intersect this well), then the average travel time to 65-m is around 7 days. Each year, after injection had ceased, the solute concentration tended towards ambient levels. The rate at which this occurs was most rapid in the earliest years and declined in subsequent years, as was the case at the ASR and 25-m wells, possibly as a result of the larger injection volumes and accumulated effects of previous injection cycles.

No breakthrough of injectant was detected at the 325-m well, although perturbations were observed on several occasions, both above and below the mean chloride value (Fig. 7). No significant difference was found in the values during injection events as compared with other periods, indicating the variability is most likely the result of sampling or measurement error. Injection would have resulted in advection of ambient groundwater of higher salinity westward towards this well, potentially confounding interpretation of the chloride concentration changes.

Given that the temperature profiles revealed that the most permeable layers occur in the lower parts of the aquifer, it would appear that the partial penetration of the observation well at 65 m, which was completed to a depth of 118 m, would suggest that the majority of flow may have passed undetected beneath this well. The possibility of flow occurring beneath the 325-m well, which also only partially penetrated the aquifer, cannot be eliminated.

Fig. 8 Temperature profile at ASR, 25 and 65-m wells before and after 6.5 ML of potable water injection. The pre-injection profile was from 29 April 1993; the post-injection profile was from 6 September 1993, 3 weeks after injection had ceased



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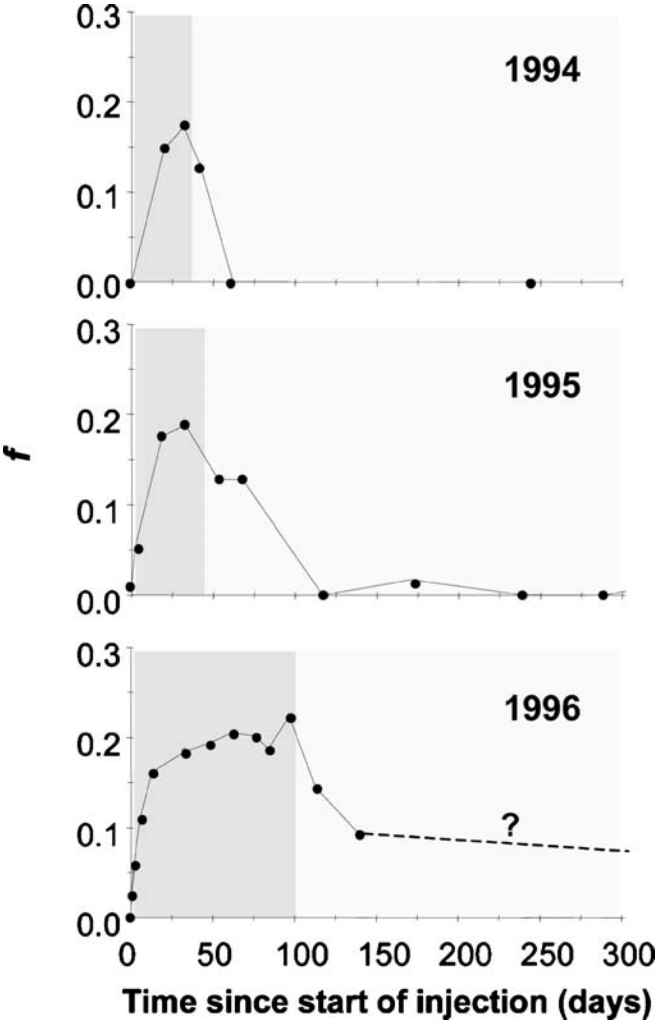


Fig. 9 Calculated mixing fractions (f) at the 65-m well as a function of the time since injection started in 1994, 1995 and 1996. Periods of injection are indicated by the darker shaded background

Recovery efficiency

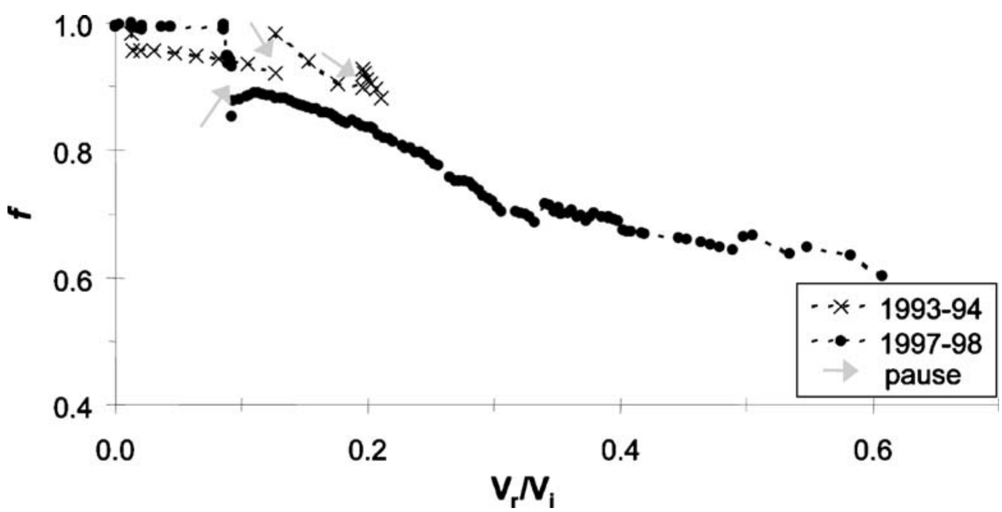
In Fig. 10 can be seen the changes in the quality of the water extracted during two recoveries: one, a relatively small event (5.4 ML) which followed the first year of injection during the summer of 1993–1994; the other, a larger event (150.8 ML) in 1997–1998 that concluded the trial. For comparative purposes, the volumetric data is presented in normalised form as the cumulative proportion of water recovered relative to the total previously injected (V_r / V_i) and the chloride data is expressed is terms of the mixing fraction (f).

During the larger event, values of f commence at ~1.0 (i.e. 100% injectant), and decline as pumping progresses to a minimum value of approximately 0.6 at the conclusion of the larger event. The step-like decrease at V_r / V_i=0.09 during the second event coincides with a period where extended airlifting was performed to control sand ingress. A 5-month pause between airlifting and the recommencement of recovery (shown in Fig. 10) produced only a slight decrease in f and the step-like change in concentration is predominantly the effect of the pumping rather than the pause during the intervening period. From V_r / V_i=0.09 to 0.60 a relatively steep decrease in f occurs up to V_r / V_i=0.3 followed by a reduced rate of change. Interestingly, two pauses in pumping during the smaller event caused water quality improvements (decreased Cl, increased f), whilst a pause during the larger event caused a water quality deterioration for reasons unknown.

The recovery efficiency is defined as the proportion of injected water that can be pumped at a quality that is suitable for its intended use (Pyne 1995). This definition is site-specific due to the influence of end-member concentrations and the maximum permissible concentration associated with a particular use of the water. For Andrews Farm, recovery efficiencies are defined in terms of the usefulness of the water for irrigation, and 1,500 mg/L TDS, a threshold which is considered to be the locally accepted criteria for irrigated agriculture, was used. This threshold corresponds to an f value of 0.40.

Figure 10 shows that during the entire withdrawal, water quality was acceptable for irrigation; therefore, the recovery efficiency at this site must be greater than 60% (since the value of f did not decline to 0.4). By integration of the chloride mass for the main recovery presented in Fig. 10, it was possible to determine that of the 150 ML of

Fig. 10 Calculated mixing fractions (f) of water pumped from an ASR well as a function of the cumulative proportion of water recovered relative to the total previously injected (V_r / V_i) during the small recovery event in 1993–1994 (5.4 ML) and the larger event in 1997–1998 (150.8 ML). *Arrows* indicate the pauses that occurred during recovery



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water recovered (60% of the volume injected), 77% was derived from stormwater (115 ML) and 23% was ambient groundwater (35 ML). If the principle of "last-in-first-out" is applied to the injected water, then none of the stormwater from the first 3 years of injection would have been recovered and a little over half of the injectant has been retained within the aquifer. The recovery efficiency estimated for the site is high considering that the minimum residence time of the recovered water was not less than 9 months and possibly up to 2 years, which is greater than the inter-seasonal time-frames that would occur under a typical operational scenario. Under such a scenario, where regular ASR cycling occurred, this would buffer against the more saline ambient groundwater in subsequent injections. Mirecki et al. (1998) demonstrated in field trials that where recovery efficiencies are initially insufficient, they can be improved by successive cycling.

Conclusions

The nature and extent of well clogging was studied using hydrological data in conjunction with a mass balance approach using physico–chemical and microbial data. Our results reveal that clogging takes place as stormwater injection proceeds; that this is, in part, reversed by porosity increases caused by dissolution of calcite due to oxidation of organic matter and by any redevelopment that may be undertaken during the course of the injection year; and finally, that the extended redevelopment that takes place at the end of each year fully restores the specific capacity of the ASR well to its former level. The high SS content of the injectant (yearly averages ranged from 29 to 169 mg/L), implies that the predominant cause of clogging is the filtration and accumulation of injected particles around the ASR well, although surprisingly, clogging rates are poorly correlated with sediment concentrations or flux. The degree of clogging is small considering the moderate aquifer transmissivity (180 m^2/day) and quality of the injectant. The recharge capacity at this site has been shown to be sustained over 4 years, but longer-term issues related to the fate of injected sediments and the stability of the ASR well and overlying aquitard due to sand loss remain unanswered.

Solute breakthrough curves provide firm evidence that flow around the ASR well is complex and has been exacerbated through procedures used to develop the well itself. Preferential flow paths appear to dominate the solute responses measured at two of the observation wells, and no breakthrough was observed at the third. Groundwater chloride data helped to explain aggregated effects of mixing mechanisms, but downhole temperature and caliper profiles in wells provide the clearest picture of aquifer heterogeneity. In spite of the heterogeneity evident and the large residence times of the injectant in the subsurface, the recovery efficiency was shown to be in excess of 60%. Over half the injected stormwater was not recovered and would serve to increase recovery efficiencies in subsequent cycles.

To our knowledge, successful ASR trials using water of this quality have not been previously reported. Before this trial, it was considered that injecting turbid water would quickly clog the well. It was also considered doubtful that the recovered blend of stormwater and ambient groundwater would be sufficiently fresh to be used for irrigation. The study has shared the lessons learnt about these two fundamental technical issues and has proved that they are manageable and provided the evidence for why this is so. Its success as a trial has since led to a series of stormwater ASR schemes being established in the Adelaide region. Other water-short cities may benefit by implementing conjunctive use systems that are appropriate and sustainable.

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