Lessons Drawn from Attempts to Unclog an ASR Well in an Unconsolidated Sand Aquifer

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Cover Photograph

Description: Urrbrae Wetland ASR operation, Allin Hodson contemplating

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EXECUTIVE SUMMARY

To date, Australian experience in ASR has largely focussed on limestone or fractured rock aquifers, with little attention given to ASR in siliceous aquifers.

An ASR trial was initiated at the Urrbrae Wetland site in metropolitan Adelaide to investigate the viability of injecting wetland-treated urban stormwater into an unconsolidated fine-grained siliceous aquifer for inter-seasonal storage so that the recovered water could be used for landscape irrigation of adjacent school grounds.

The trial was shut-down six weeks after operation commenced due to excessive clogging of the ASR well. This report describes the initial injection period and the attempts made to restore well efficiency through intermittent pumping, chlorination and surging. Water quality information from injected water and purged water were recorded; pumping tests to define changes in specific capacity of the well, and down-hole flow meter and camera logs were recorded to assess progress with rehabilitation. While the initial clogging reduced specific capacity to 20% of pre-trial values, the three methods combined only raised specific capacity to 40% of pre-trial values. Turbidity and bio-available nutrients in the injectant were considered the prime causes of physical and biological clogging respectively, but mobilization of drilling fluid or aquifer fines into the formation may have compounded this, along with evidence of vandalism of the ASR well.

Target values for injectant water quality parameters for controlling clogging have been estimated for this site. Research is continuing aimed at identifying passive pre-treatment processes which will achieve those water quality targets in preparation for a further trial at the site.

1. INTRODUCTION

The aquifer storage and recovery (ASR) literature contains many case studies that demonstrate the success of field trials and established operating schemes and generally promote the positive aspects of ASR (eg. Pyne, 1995 and the references cited therein). Documented evidence of ASR failures, and the underlying causes for failure, have been far less common. In a review of the ASR literature, Pavelic and Dillon, (1997) provide two specific examples of failure; one involving excessive well clogging due to injection of wastewater into a fractured rock aquifer (Lakey, 1978), and the other due to rupturing of a clay layer overlying the target aquifer resulting from injection of surface water (Ramnarong, 1989). Hesitance to report on negative aspects of ASR may lead to the false perception that ASR is a fail-safe technology under all circumstances. Only through the dissemination of both positive and negative ASR outcomes can the issues and failures of the past be avoided.

ASR operations in Australia have largely focussed on limestone or fractured rock aquifers and the results have generally been successful (Martin and Dillon, 2002; Hodgkin, 2004). From a well clogging perspective, limestone aquifers are the more tolerant of poorer source water quality due to the offsetting effect of matrix dissolution. Although fractured rock aquifers are more complex to characterize in terms of their permeability structure and storativity, detailed studies have not yet been conducted, apparently because existing sites have been operating successfully (Hodgkin, 2004).

Unconsolidated fine-grained aquifers present challenges to maintain adequate rates of injection in ASR wells. In Australia and elsewhere, opportunities to enhance groundwater resources through ASR have been foregone due to lack of knowledge of water quality requirements for injection into unconsolidated alluvial aquifers.

In 1997 an ASR trial was initiated at the Urrbrae Wetland site in metropolitan Adelaide, South Australia to test the viability of injecting wetland-treated urban stormwater into an unconsolidated siliceous aquifer so that the recovered wetland water could be used for landscape irrigation of adjacent school grounds. The trial failed due to irreversible clogging of the ASR well. This report documents the main outcomes and lessons learnt during the ASR trial and attempts to remediate the ASR well after all viable options were exhausted. Although the ASR trial did not succeed and was suspended, this examination of the causative factors of failure may be helpful for proponents contemplating ASR under similar circumstances.

2. LOCAL HYDROGEOLOGY

The regional hydrogeology of the Adelaide Plains is comprised of a Quaternary alluvial sequence of low yielding aquifers, overlying a Tertiary limestone sequence of higher yielding aquifers and Pre-Cambrian bedrock, with a combined thickness of up to 500 metres in the western part of the Plain (Figure 1) (Gerges, 1996; 1999).

The late Quaternary/Tertiary aquifers targeted for ASR at the trial site consist of inter-fingered marine sands (Carisbrooke Sand and the lateral equivalent of the Port Willunga Formation) that probably represent the margins of extensive sandy deposits common along the eastern margins of the Adelaide Plains that form the intermediate (trough) zone between the hard-rock aquifers of the Mount Lofty Ranges and the Tertiary sequence of the Adelaide Plains proper. Groundwater flow direction is generally westward, towards the coast. Isotopic data from a well completed in the Tertiary within two

kilometres of the study site suggest groundwater velocities of around 1-2 m/year and a carbon-14 age of around 3000 years (Dighton *et al.*, 1994).

Local drilling at the trial site identified the upper 63 m to be Hindmarsh Clay, a fluvial Quaternary unit comprised of stiff clay inter-bedded with thin aquifers. This was underlain by around 8 m of Carisbrooke Sand, the oldest Quaternary deposits, then by 23 m of Port Willunga Formation, which is of the Tertiary period (John Botting and Associates and Lisdon Associates, 1998; Gerges, 1999). The Carisbrooke Sand and the Port Willunga Formation are the most productive aquifers and were targeted for ASR. The Carisbrooke consists of medium- to fine- grained calcareous sand with some ferruginous and possibly some inter-bedded silt layers. The Port Willunga formation consists of coarse sands and gravels with varying lignitic content.

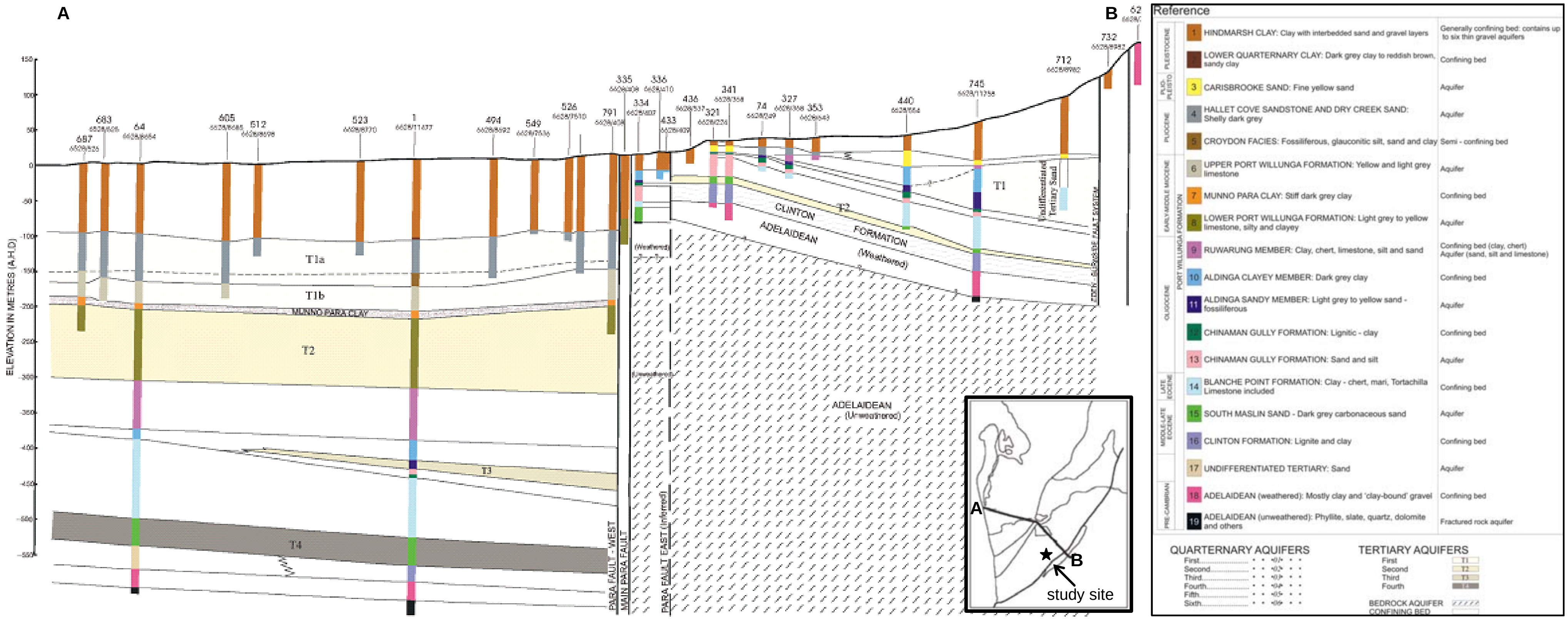

Figure 1. Hydrogeological transect across the Adelaide Plains (from Gerges, 1996)

3. ASR SYSTEM DESIGN

In July 1997 the ASR well (Unit Number 6628-18576) was drilled to a depth of 93.6 m using the rotary mud drilling method. Drilling ceased at this depth due to increasing lignite content and the well was later backfilled to 84.7 m (John Botting and Associates and Lisdon Associates, 1998; Appendix 1). The original pilot hole was reamed to a diameter of 229 mm (9-inch) then the well was cased in 203 mm (8-inch) UPVC and cement grouted to the surface, with a larger, 298 mm (12-inch) UPVC collar casing in the top 5 m. A 152 mm (6-inch) wire-wrapped stainless steel screen assembly was installed on the basis of geophysical logs and a limited amount of sample cuttings. Screen was installed over three intervals representing the most productive zones, with blanks fitted to avoid the more lignitic layers (Figure 2). Interpretation of the hydrogeological log presented in Figure 2 can be found within the report by John Botting and Associates and Lisdon Associates, (1998) provided in Appendix 1. The uppermost screen aperture was 0.5 mm for the finer-textured Carisbrooke sand and the lower two screens were 1.0 mm for the coarser textured Port Willunga Formation. The well was extensively airlifted and backwashed to dislodge residual drilling muds and develop a natural gravel pack. The airlift yield of the well was 3 L/s and discharge testing with the pump positioned immediately above the screens at 63.5 m (>30m below the standing water level) led to cavitation of the pump at a flow rate of 4.3 L/s. The anticipated long-term yield based upon well coefficients derived from step testing and the 33 m of available drawdown was estimated to be 3 L/s. The combined transmissivity of the aquifers was estimated to be around 6 m^2/day. The ASR well is situated on the south-western corner of the holding pond, within close proximity to the source water and necessary power supply (Figure 3).

The components of the ASR system include the ASR well fitted with a submersible pump (Calpeda MXS 204) positioned at 80 m depth, an 8 m^3 ferro-cement storage tank, two rapid sand media filters (Yamit 600 series), on-line cumulative flow meter, manual flow control valves, electrical control system and cement footing for the sand filter and proposed irrigation pumps (Figure 4).

Water was pumped from the holding pond through the sand filters and into the holding tank at a rate of 1.25 L/s using a submersible pump mounted on a float positioned just below the pond surface. This water was then gravity-fed into the ASR well. The sand filters were programmed to backwash every two hours for five minutes, with the waste stream returned to the main lagoon. A recharge line, composed of 20 mm UPVC pipe, was installed to a depth just below the standing water level to control clogging by aeration. The depth to standing water level, which ranged from 30-34 m below ground surface (bgs), provided latent storage capacity and was at a sufficient depth that the tank provided a good driving head.

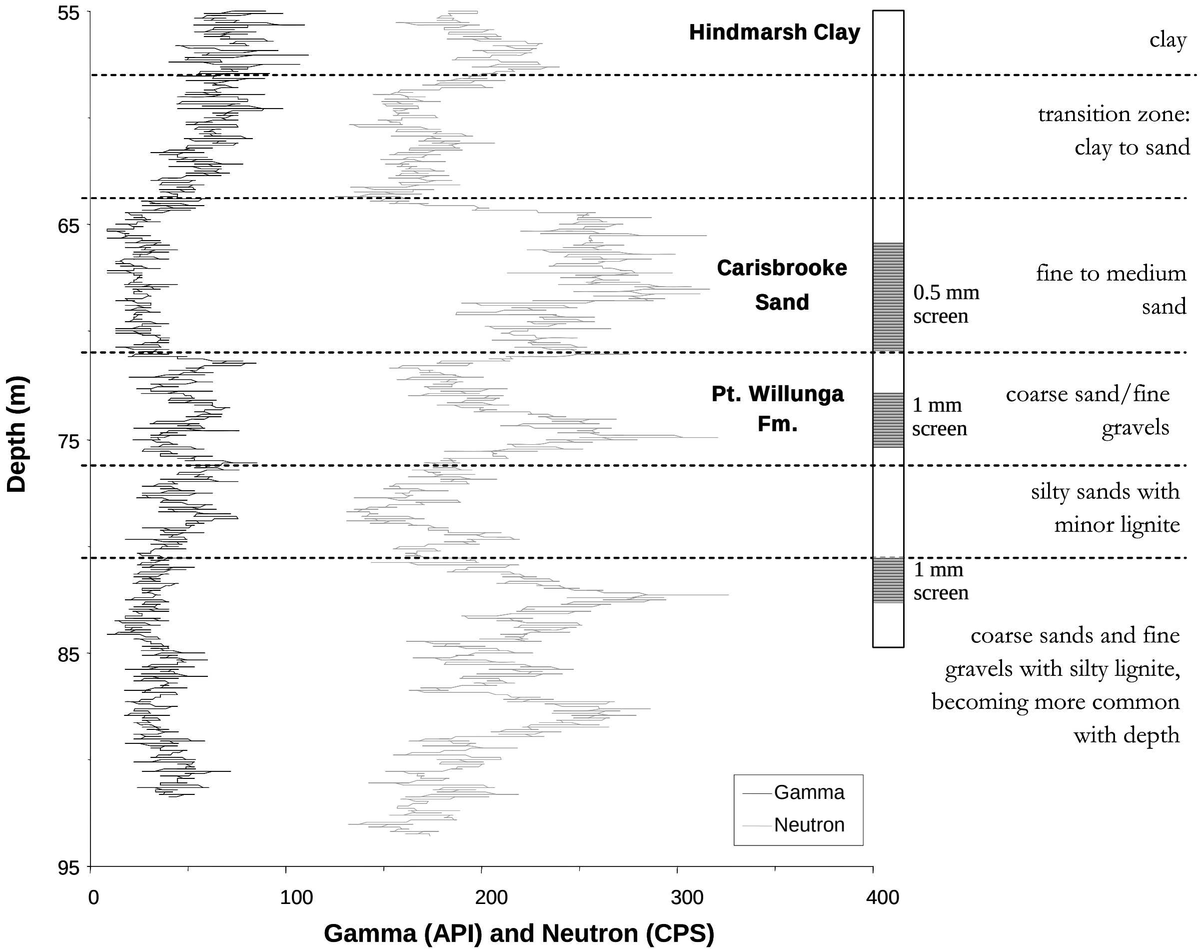

Figure 2. Geophysical logs, hydrogeology and completion arrangement of the Urrbrae ASR well (from John Botting and Associates and Lisdon Associates, 1998)

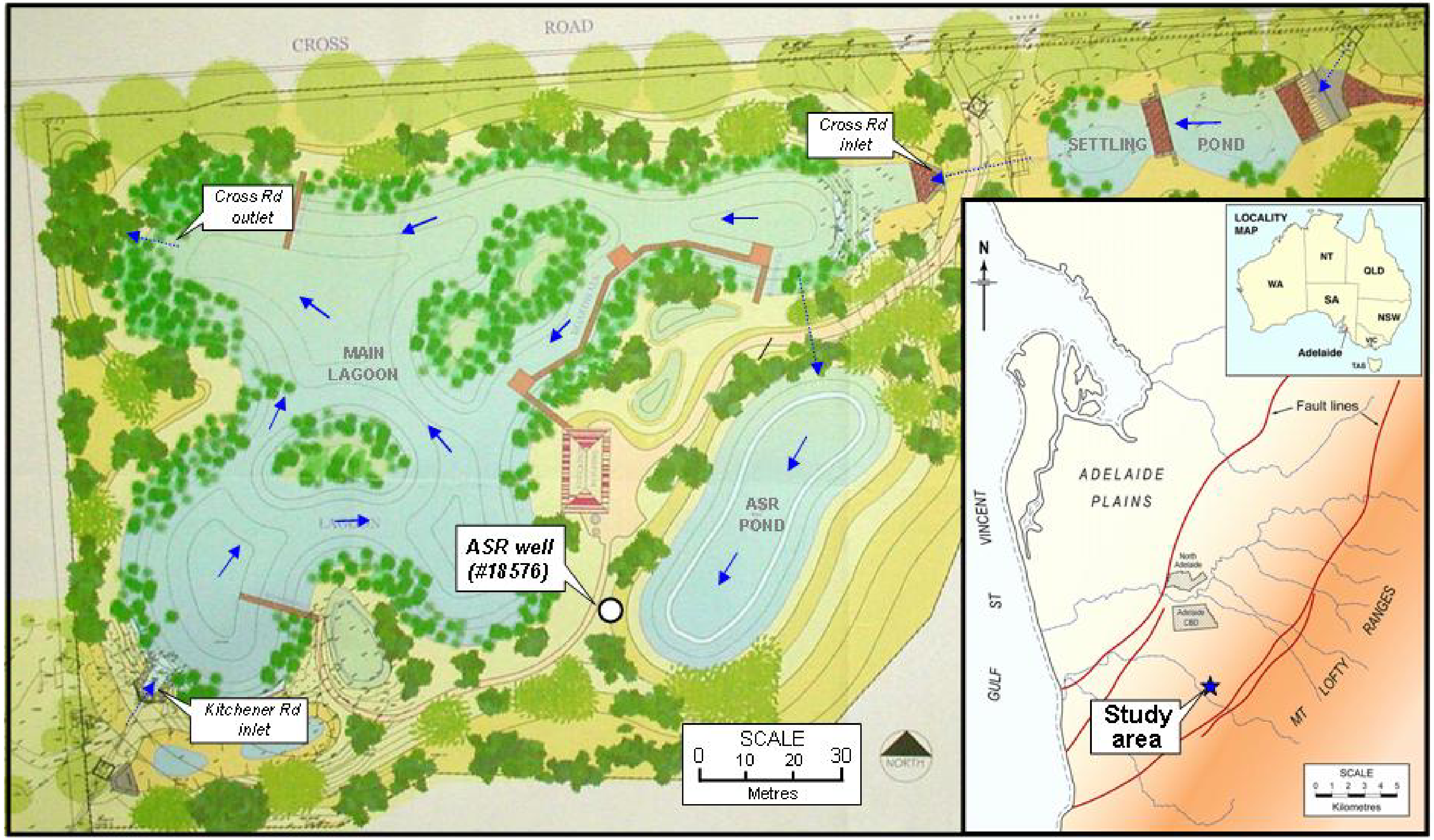

Figure 3. Site map showing location of the ASR well in relation to the holding pond (source of regional map: Bob Schuster, CSIRO). The arrows indicate the inferred direction of flow during stormwater inflow and well injection (from Lin et al., 2006). Note the pre-settling pond was established in 2003; several years after the conclusion of this study.

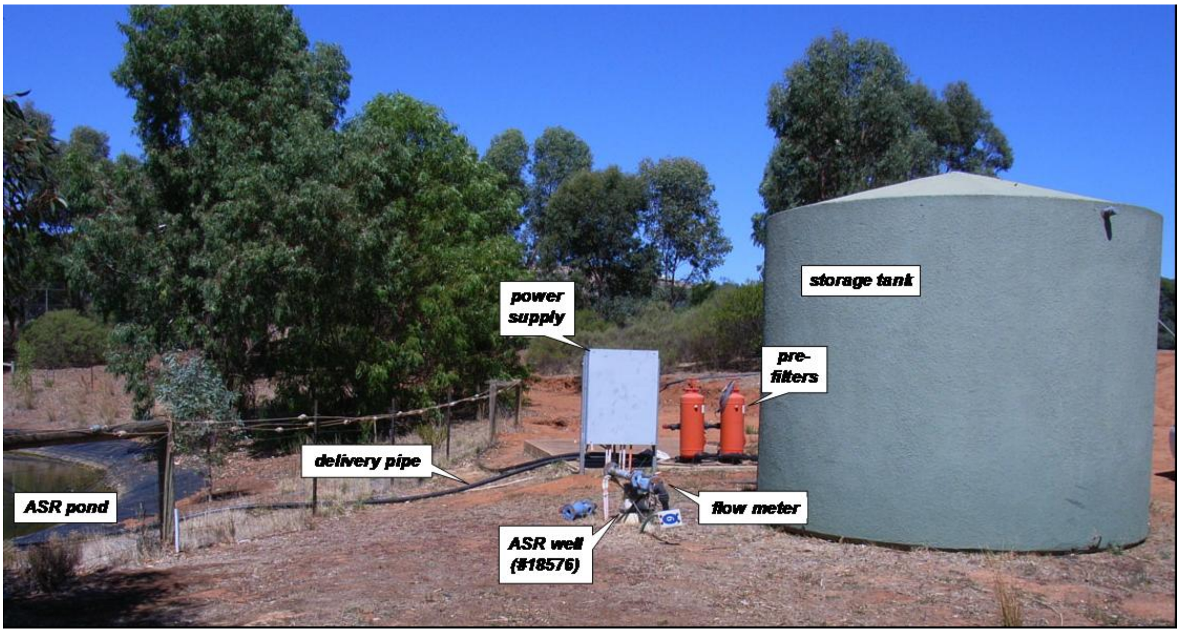

Figure 4. Study site at Urrbrae, South Australia indicating the main components of the ASR system

4. CATCHMENT AND WETLAND CHARACTERISTICS

The 375 hectare (Ha) catchment is comprised of two sub-catchments (Cross Road and Kitchener Street). Each sub-catchment takes in the margins of the Mount Lofty Ranges and adjacent Adelaide Plains (Figure 5). The catchment contains a mix of landuses including agriculture (mainly non-irrigated), residential, agricultural education and research facilities. The catchment contains virtually no industry and little development of commercial property (Hodson, 1999).

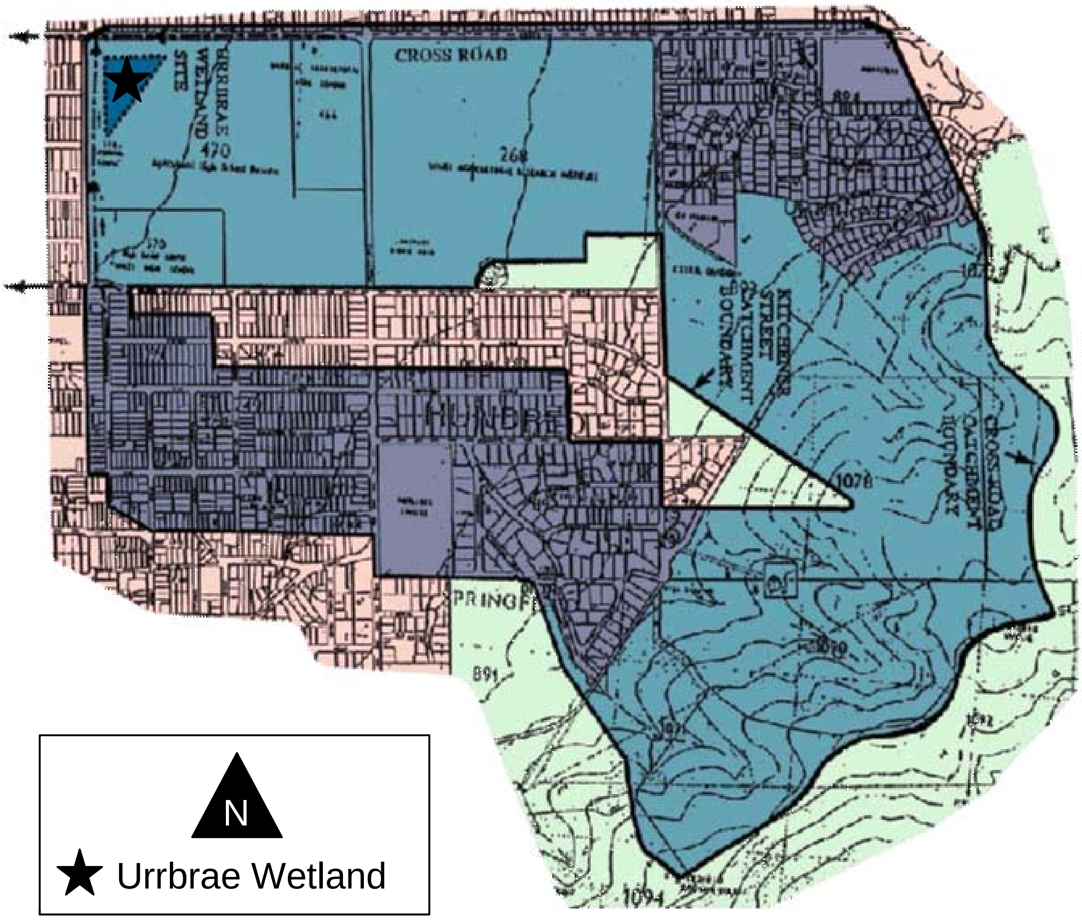

Figure 5. Surface water catchment for the Urrbrae Wetland comprising of the Cross Road and Kitchener Street sub-catchments (from Hodson, 1999)

The Urrbrae Wetland ASR site is located at the Urrbrae Agricultural High School adjacent to Cross Road, Urrbrae, South Australia. The major features of the wetland include the main lagoon and rubber-lined holding pond (Figure 3). The wetland was built in 1996 primarily to mitigate local flooding and was engineered to handle peak stormwater flows associated with a 1-in-5-year storm event. Estimated mean annual volumetric flow through the wetland is around $350x10^3$ m^3 (Hodson, 1999).

Water depth in the main lagoon is typically greater than 1.5 m for the majority of the year (Hodson, 1999). The bottom is clay-lined and needs to be kept full during the dry season to protect the lining from shrinkage and cracking. The bottom of the holding pond is lined with welded polythene sheeting and filled during the wet season from the main lagoon through a subsurface pipe located between the Cross Road inlet and eastern extent of the observation deck. During the wet season, flow from the main lagoon into the holding pond is minimal due to the high surface water elevation maintained in the holding pond after initial storms. During the dry season, water in the holding pond is used to replenish water lost to evaporation in the main lagoon. Source water for ASR is pumped directly from the holding pond. The anticipated direction of surface water flow during injection is indicated in Figure 3 and suggests that the injectant will be derived from stormwater entering the main lagoon via the Cross Road inlet and overflowing into the holding pond and a small component from direct rainfall. Residence time of water in the holding pond is likely to be higher than the main lagoon.

5. CLOGGING AND UNCLOGGING

5.1 Clogging processes

Clogging is one of the most serious operational problems in ASR since it restricts the volume of water injected, thereby increasing the effective unit price of stored water. Clogging develops with time as a result of the interaction between the source water (including its constituents), and the native groundwater and the porous media, which can lead to a reduction in the permeability at the well screen, gravel pack or surrounding aquifer. Clogging-induced permeability reductions cause a decline in injection rate and/or hydraulic head increase.

The following physical, chemical and biological processes are known to cause clogging:

- filtration of suspended solids
- microbial growth
- chemical precipitation
- clay swelling and dispersion
- air entrapment and gaseous binding
- particulate rearrangement and mobilisation of aquifer fines

Comprehensive reviews of these processes are provided by Olsthoorn, (1982) and Pérez-Paricio and Carrera, (1999) and only a brief summary is provided below.

Clogging by filtration results from the filling-in of the aquifer pore space with injected particulates of a comparable size, which results in the formation of a filter-cake layer that undergoes compression with increased hydraulic head build-up within the well. The extent of clogging is dependent on the relationship between the nature, size, velocity and loading of the particulates in the source water relative to the physical dimensions of the porous media.

Introduced or indigenous bacteria may grow or multiply in porous media under aerobic or anaerobic conditions where sufficient organic matter and nutrients are present. Microbial activity tends to be concentrated around the ASR well where substrate materials are filtered out. The microbes create a biofilm of extracellular polymers (polysaccharides) that reduce aquifer permeability. Unlike particulate clogging which is instantaneous, microbial clogging can develop over time frames ranging from days to weeks. High levels of iron or manganese in the presence of oxygen can stimulate bacteria such as *Gallionella* to produce precipitates that lead to clogging. Microbial growths have been most evident where nutrient-rich waters are used.

Clogging by air entrainment can occur if water is allowed to cascade into the well and bubbles that are produced block pore spaces and restrict flow. Dissolved gases may also be released from solution due to temperature changes (eg. where cool source waters meet warm groundwater) or geochemical reactions.

Injection of waters incompatible with groundwater or aquifer materials can cause chemical reactions which alter the hydraulic properties of the porous media. These reactions may include dissolution, precipitation, ion-exchange, ion-adsorption and oxidation-reduction. Geochemical reactions that lead to clogging are not widely reported as they are difficult to characterise or take long periods of time to develop. Other geochemical reactions, such as dissolution, have the opposite effect to clogging by

increasing permeability (eg. where calcite cement dissolves), leading to mobilisation of remnant materials and potential for well instability.

One of the most commonly reported geochemical reactions is ion exchange between cations in solution and those associated with clays within the aquifer. This can lead to either swelling or dispersion. Dispersion is possibly the more serious, as it results in the physical movement of the clays, and is therefore more difficult to remediate. Clay swelling is most prevalent where reactive clays are present (eg. montmorillonite), and where there is a large decrease in the electrolytic concentrations of the injectant as compared to the native groundwater.

Changes in flow direction caused by repeated injection and recovery can lead to mobilisation, movement and redeposition of fines that may be present in the aquifer.

Multiple forms of clogging can occur over similar or different intervals of time and space. In many cases the processes responsible for clogging are difficult to discern, and often conclusions are drawn from indirect evidence.

5.2 Prevention and remediation of clogging

Clogging is an intrinsic but manageable part of most ASR operations. Although this can be problematic and an expensive issue in some cases, it can be avoided or remediated by appropriate management, particularly with respect to the pretreatment of injectant or by regular backwashing of the ASR well.

The aim of redevelopment is to return the well to its prior state by restoring the hydraulic properties of the aquifer. A variety of mechanical and chemical techniques can be employed. Mechanical methods rely on physical agitation of the porous media, and include pumping, jetting and surging. Chemical methods include the addition of acids, flocculants and disinfectants. The frequency of redevelopment varies, and may be as often as daily to annually, depending on how quickly clogging develops.

Often a trade-off exists between the cost of pretreatment and the type and frequency of redevelopment. Generally, the higher the quality of the source water, the lower the level of clogging. Although improving the quality of the source water is possibly the most effective means of dealing with clogging, there are situations, however, where this cannot be justified on economic grounds. The composition of source water would typically be characterized in terms of the levels of suspended solids (TSS, turbidity, MFI), nutrients (N, P), organic matter, iron, manganese, sodium adsorption ratio (SAR) and microorganisms.

6. WETLAND WATER QUALITY

The principal gross pollutants entering the wetland are associated with the extensive vegetation cover within the catchment, which, when combined with the high runoff velocities due to the moderate slopes produce a significant influx of leaf and other organic debris throughout the year. The large contribution of organic matter results in elevated TOC concentrations causing periodic oxygen depletion within the wetland. Inorganic fines and colloidal matter are generally a second-order phenomenon (Table 1), except during periods of building construction within the catchment.

It is recognised that there are inherent temporal variations in the quality of water in the wetland due to stormwater runoff and algal growth in the shallow, nutrient-rich water. The variability in the composition of the wetland water with respect to water quality parameters indicative of clogging are presented in Table 1. Total suspended solids (TSS) and volatile suspended solids (VSS) data reveal that most of the suspended solids in the wetland are organic in nature. Only in samples collected near the inlet-end of the main lagoon during runoff events (eg. 19 Oct. 05) are the majority of particles inorganic in nature. Table 1 shows that particulate concentrations in the wetland water are highly variable, as anticipated of urban stormwater (eg. turbidity values range from 0.8 to 55 NTU). The physical clogging potential of the wetland water according to Membrane Filtration Index (MFI) data is high relative to the levels of particulates due to the predominantly organic nature of suspended particles in the stormwater which more easily compress and clog the filter paper pores than rigid inorganic particles (Dillon *et al.,* 2001). The MFI of the water from the detention basin is higher per unit TSS than the main lagoon due to the higher organics content (Lin *et al.,* 2006).

Bacterial regrowth potential (BRP) concentrations range from 39 to 331 acetate carbon equivalent (ACE) μg/L. The assimilable organic carbon (AOC) threshold for biologically stable waters is 40 ACE μg/L (Werner and Hambsch, 1986), and the maximum permissible level for AOC in the Netherlands for injection into fine sandy aquifers is 10 μg/L (Hijnen and van der Kooij, 1992). Unfortunately AOC and BRP relate to different components of labile organic carbon and are therefore incomparable indices of nutrient bioavailability.

Electrical conductivity (EC) values of the recharge water during the winter-spring period when the greatest opportunity for injection exist are typically <300 μS/cm, and during the summer-autumn period are highest at 300 to 500 μS/cm (Figure 6). This figure also shows that the temperature of the recharge water was likely to have been in the range of 10 to 25 ^{o}C.

Table 1. Physico-chemical characteristics of the wetland water from 9 sets of analyses between March 1999 and October 2005

Parameter	4 Mar. 99 [A] (MLG)	8 Mar. 99 [A] (MLG)	25 Jun.99 (MLG)	16 Jul.99 (MLG)	8 Nov.99 (HP)	8 Nov.99 (ST)	5 Apr.01 (HP)	17 Oct. 05 [B] (HP)	19 Oct. 05 [B] (MLG)
MFI (s/L^2)	170	345	389	213	323	90	123	170	207
d_{50} (μm)	88	195	34	15	127	-	-	130	10
Turbidity (NTU)	3.7	7.5	55	36	6.4	0.81	5.9	10.5	33.9
TSS (mg/L)	4	10	33	20	10	<1	-	11	30
VSS (mg/L)	3	10	-	-	10	<1	-	11	11
TOC (mg/L)	10.2	12.6	4.3	3.9	6.6	4.6	-	4.6	6.8
BRP (μg/L)	88	258	331	39	-	-	-	190	293

'-' = not analysed; MLG = main lagoon; HP = holding pond; ST = storage tank; BRP = bacterial regrowth potential (acetate carbon equivalents); [A] reported in Massmann *et al.,* (1999); [B] reported in Lin *et al.,* (2006)

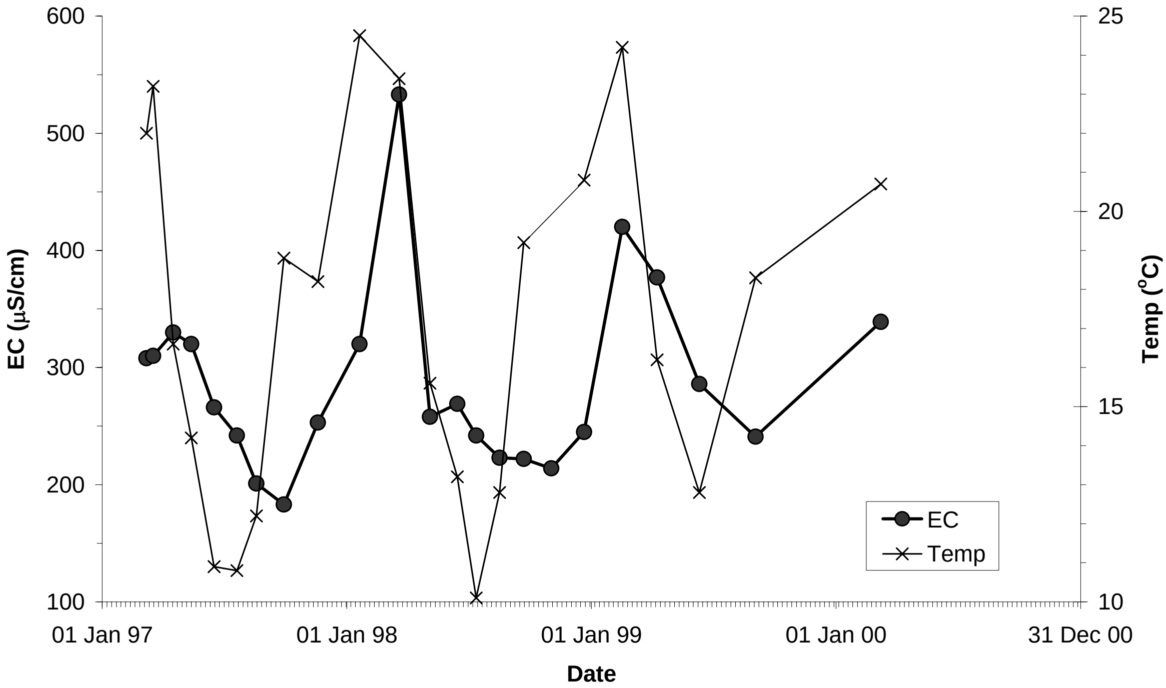

Figure 6. Electrical conductivity (EC) and temperature variations in holding pond between March 1997 and March 2000

6.1 Scanning electron microscopy

Scanning electron microscopy (SEM) images reported by Lin et al., (2006) reveal mostly inorganic and organic particle assemblages containing large organisms, some smaller organic remnants, diatoms, and bacteria in the main lagoon (Figure 7). Particle sizes range from 10 to 100 µm. Energy Dispersive X-ray (EDX) spectra indicated aluminium-silicates, iron-oxides and organics. The holding pond water contains a diverse assortment of discrete particles (mostly macro-organisms) and complex organic and inorganic particle assemblages (or flocs) bound by organic mucilage (Figure 7). Indicative particle sizes range from 50 to 300 µm. Macro-organisms included algae, diatoms, amoebas, fungi and bacteria. Minerals included clay minerals, quartz, and iron-oxides. The abundant amorphous mucilage was reflected in EDX spectra by the high C, O, P, S and K, while aluminium-silicate peaks were associated with the minerals. Macro-organisms were much more common (some algae were observed) and flocs were more uniformly coated with mucilage indicating different biological population or environmental conditions. SEM images previously reported by Massmann et al., (1999) show similar characteristics.

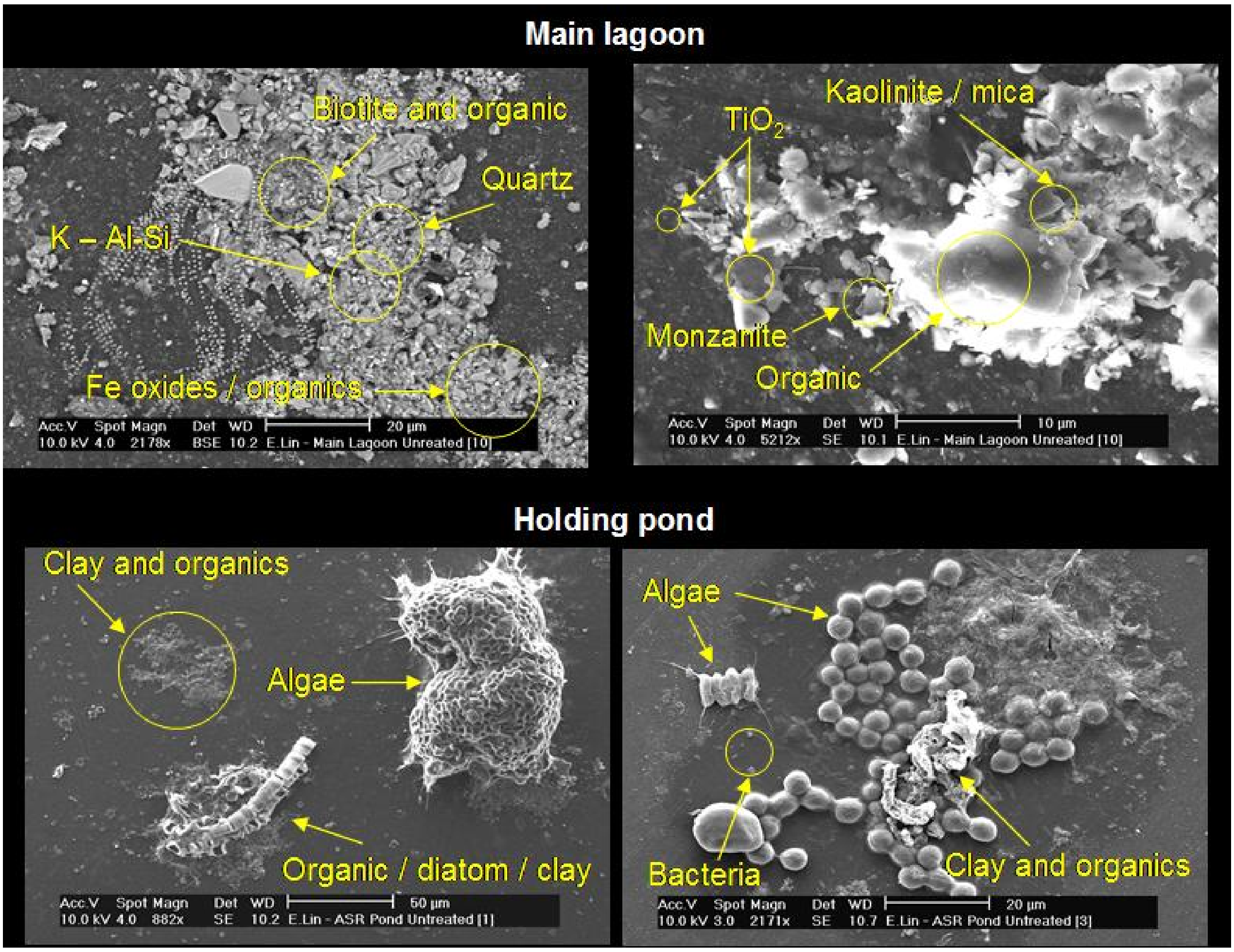

Figure 7. SEM micrographs showing typical particles in main lagoon during stormwater inflow on 19 Oct. 2005 (top row) and holding pond water from 17 Oct. 2005 (bottom row) (from Lin et al., 2006).

7. INJECTION PHASE OF THE TRIAL

The trial was operated during the spring of 1999 and about 4 x10^3 m^3 was injected over a 6 week period. This amount was only one-fifth of the target volume of 20 x10^3 m^3 over winter. Initial injection rates of around 3 L/s were reduced to a final value 0.5 L/s. Injection was halted by 5 November 1999 due to the unacceptably low flow rate. The decline was noted to have occurred over the injection period, although actual changes over time were not recorded. Unfortunately injection commenced approximately one month before the pump was installed in the well. Periodic backwashing of the well upon installation of the recovery system failed to stop the decline in injection rates. The small residual potentiometric head increase following injection indicated that the storage capacity of the aquifer was not a constraint. After modifying the headworks by installing the injection line, air entrainment in the injected water was eliminated, also removing this as a potential cause of clogging. In addition, there was also at least one input of engine oil from the stormwater catchment and black staining on the tank water level gauge indicate that traces of oil had breached the sand filter and entered the ASR well.

The severity of clogging is indicated by the results of pumping tests conducted before and after injection (Figure 8). This figure shows, for example, that post-injection a 28 m drawdown was achieved 9 times faster with pumping rates 2-4 times lower than pre-injection.

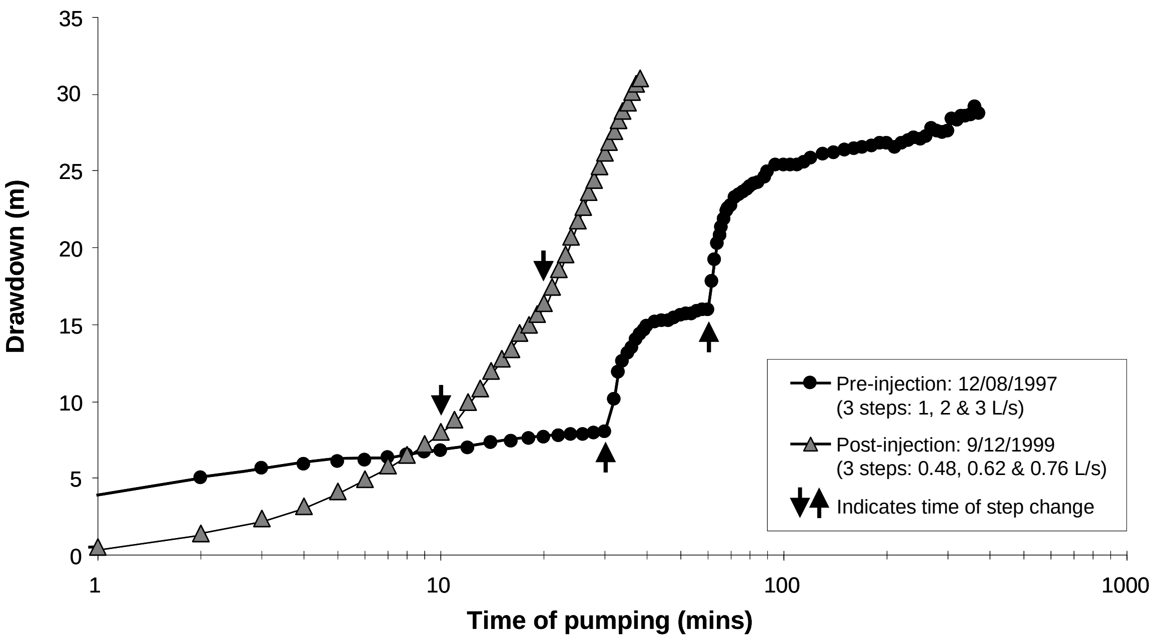

Figure 8. Semi-log plot of drawdown in ASR well versus time of pumping for pre- and post- injection pumping tests

Rapid clogging occurred despite pre-treatment of the injectant by rapid sand filtration. Particulate matter present in the stormwater was found not to be substantially reduced by the rapid sand filter. Rather, the large, complex flocs evident in Figure 7 were broken-up into smaller flocs due to high shear stresses within the sand filter. Measures of particles/clogging parameters (turbidity, TSS, MFI etc) were reduced by 10-30%, but still remained high (Table 2). The sand filter, with an effective particle size (d_{50}) of 0.95 mm and high uniformity, proved ineffective in removing an adequate proportion of particulates from the stormwater (note the uniformity coefficient, u = d_{60}/d_{10} = 1.14).

SEM imaging of the backwash water revealed that all of the particles were significantly smaller than the injected particles, with dimensions less than 5 to 10 µm (Figure 9). Lin *et al.*, (2006) demonstrated that the passage of the Urrbrae Wetland water through a roughing filter pre-treatment system also reduced the effective size of particles in the treated water. The majority of particles evident in Figure 9 are inorganic, whereas a much higher organic content was present in the injectant (Figure 7).

The potential causes of clogging included: suspended solids or hydrocarbons entering the well; biofilm production on the well screens and surrounding natural gravel pack; and remobilisation of drilling muds or fines from the aquifer. Chemical precipitation and gas binding by entrained or evolved gases from the injectant were eliminated. The sodium adsorption ratio of the injectant was lower than ambient groundwater and unlikely to disperse clays in the aquifer as a result of reducing groundwater salinity. The next step was to identify the most appropriate techniques for restoring the injection rate and maintaining it in the long term.

Figure 9. SEM micrographs showing particles in backwash water from ASR well on 10 Dec. 1999

Table 2. Performance of the rapid sand filter during sampling on 8 Nov. 1999

Parameter	Pre-filter	Post-filter
MFI (s/L^2)	323	229
d_{50} (µm)	127	104
Turbidity (NTU)	6.4	5.2
TSS (mg/L)	10	9
VSS (mg/L)	10	9
TOC (mg/L)	6.6	6.3

8. EFFORTS TO REMEDIATE THE CLOGGED ASR WELL

Over the period from December 1999 to November 2000 a series of activities involving various types of inspection methods and restoration approaches were undertaken with the aim of diagnosing the cause of the problem and remediating the ASR well. These were punctuated by a series of short, single- or multiple- step drawdown tests as a basis for assessing the change in hydraulic performance of the well. The well efficiency is defined here in terms of its specific capacity (at a specified time) as it is particularly sensitive to the well-loss component of drawdown. A summary of the main activities during the trial are given in Table 3. Details on the remediation activities are given below.

Table 3. Inventory of major activities during the Urrbrae Wetland ASR trial

Date	**Event**
5 Aug. 1997	First downhole camera survey of the well
12 Aug. 1997	First pre-injection aquifer pump test (3-step, 370 mins.)
14 Jul. 1998	Second pre-injection aquifer pump test (1-step, 360 mins.)
9 Mar. 1999	Brief injection-recovery test (3-cycles, 360 mins.)
Spring 1999	Start of injection (3 L/s)
5 Nov. 1999	Injection stopped due to >80% reduction in flow rate ($4x10^3$ m^3 injected)
9 Dec. 1999	First post-injection aquifer pump test (3-step, 38 mins.)
9-14 Dec. 1999	Intermittent backwashing of well
14 Dec. 1999	Second post-injection aquifer pump test (1-step, 14 mins.)
19-21 Jan. 2000	Injection of disinfection agent
1 Feb. 2000	Third post-injection aquifer pump test (3-step, 26 mins.)
8 Mar. 2000	Fourth post-injection aquifer pump test (1-step, 61 mins.)
14 Mar. 2000	Second downhole camera survey of the well
5-6 Apr. 2000	Surging of the upper screens and partial removal of sand accumulated around bottom screen
30 May 2000	Third downhole camera survey of the well
14 Jul. 2000	Downhole EM flowmeter survey of well under ambient and pumped conditions (fifth post-injection pump test, 1-step, 74 mins.)
3 Nov. 2000	Sixth post-injection aquifer pump test (3-step, 40 mins.)

8.1 Intermittent backwashing of ASR well

The first approach involved intermittent backwashing over a 5-day period (9-14 December 1999). Pumping events were scheduled on the hour for 3-5 minutes each at rates of 1.8-2.8 L/s. Using the pump control system, automatic aquifer pump tests were conducted before and after the backwashing to gauge the success of the approach.

As Figure 10a shows, the turbidity of the recovered water initially peaked at 3500 NTU and declined exponentially to a final value of 21 NTU after 22 m^3 had been pumped. The final turbidity was similar to the average injectant turbidity (Table 1). There was no visual evidence of oil residue in the backwash waters. Recurrent pumping and the demonstrated removal of at least some of the clogging agents from around the well failed to produce a measurable improvement in well efficiency. The specific capacity remained unchanged at 3-5 m^3/d/m, far lower than the pre-injection values of 11-13 m^3/d/m (Figure 11). The evidence seemed to suggest that only a small fraction of the most easily-dislodged clogging agents had been recovered from around the well-screens.

8.2 Chlorination of ASR well followed by intermittent backwashing

As backwashing alone had proven ineffective, a slug of chlorine solution was introduced into the well to oxidize the organics prior to further backwashing. Here, 34 m^3 of potable quality water was dosed with standard pool chlorine (calcium hypochlorite containing 65% available chlorine) to an average concentration of around 300 mg/L and injected into the well over a 2 day period (19-21 Jan. 2000). The chlorinated slug remained within the gravel pack of the aquifer for a further 6 days before 58 m^3 was pumped over a 4 day period (27-31 Jan. 2000). The aggressive character of the chlorinated water was confirmed by observed etching of the plastic coating of the pressure transducer that had been resident within the well. Dark, slimy material was clearly evident in March 2000 upon recovery of the pump column and the small-diameter UPVC access pipe for the transducer that had been placed within the ASR well in December 1999.

During pumping the initial peak in turbidity was 400 NTU and declined exponentially to reach a final value of 15 NTU (Figure 10b). Surprisingly, this peak value was almost an order of magnitude lower than the highest concentration measured in December 1999. Once again, pump testing on 1 Feb. and 8 Mar. 2000 revealed little or no improvement in the specific capacity of the well (Figure 11).

Pérez-Paricio and Carrera, (1999) noted the inadequacy of chlorine treatment in cases where high concentrations of soluble iron strongly reacts with an oxidant (levels of iron in this study were high as will later be shown). Repeated bursts of chlorination at higher concentrations than used here, followed by acidisation to remove chemical precipitates often associated with the biofilm, may have been more successful (eg. Driscoll, 1986 recommended chlorine concentrations of 500-2000 mg/L).

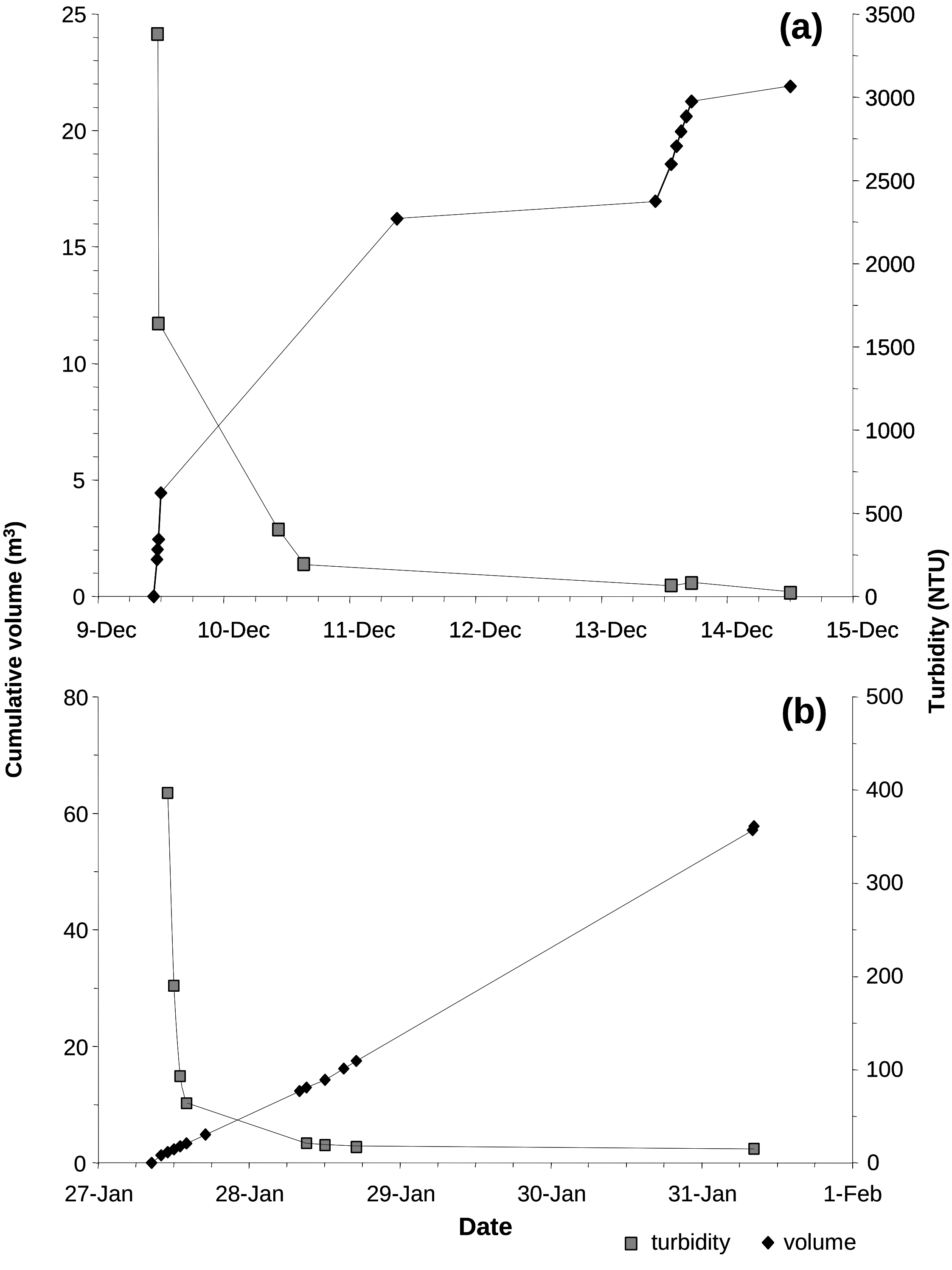

Figure 10. Changes in turbidity and the cumulative volume of water pumped during redevelopment events. The upper plot (a) is before rehabilitation in December 1999; the lower plot (b) is after well chlorination in January 2000.

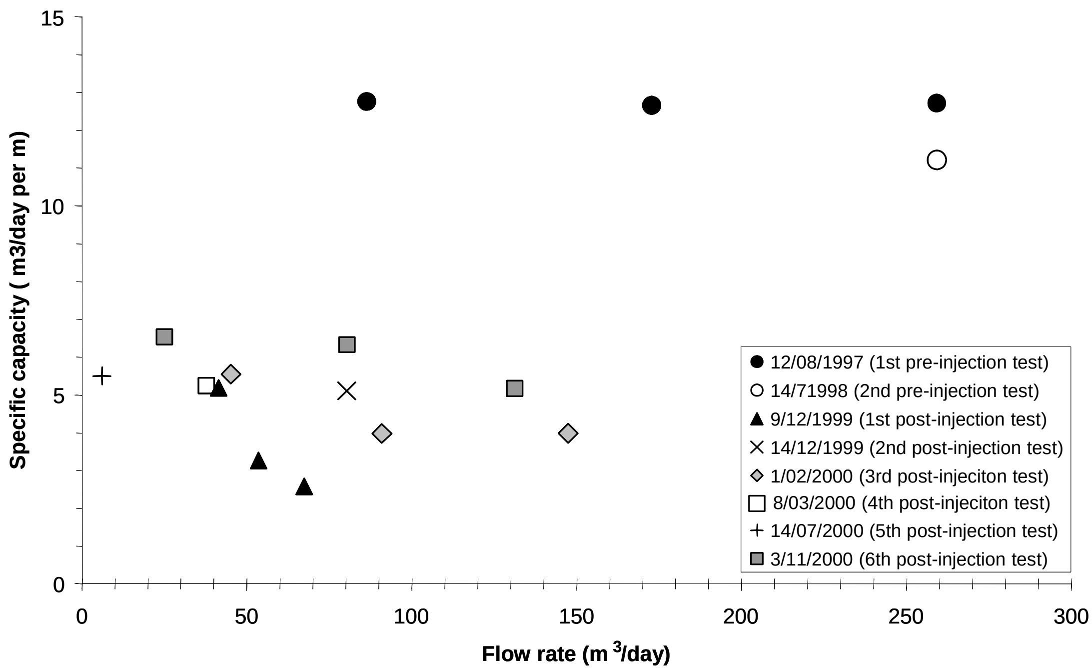

Figure 11. Plot of specific capacity (at t=10mins) versus pumping rate at different stages of the trial

8.3 Downhole camera surveys

A downhole video camera survey on 14 Mar. 2000 revealed heterogeneous discolorations on the well-screens symptomatic of persistent fouling of the screens. Such discolorations were not observed in the camera survey prior to injection (5 Aug. 1997). Rubbing of the centralizing arms of the camera on the walls of the casing and screens stripped some of the coating material that appeared to be composed of large dark coloured organic flocs, as had been seen on the pump column, and presumably the result of excessive microbial activity. There was no observed evidence of lignite protrusion through the screens.

The camera footage also showed that a metal fence post (also known as a ‘star-dropper’) had lodged on the casing shoe as a result of an incident of vandalism at some point between August 1997 and March 2000.

The bottom of the hole was reached at a depth of 77.2 metres, or 7 m less than the drilled depth, indicating that there had been significant in-filling of the well with sand. Although it was theoretically conceivable that the sand had entered the well via the screens, damage was noted to the rubber seal (the so called ‘J-latch’) set between the narrower 152 mm (6-inch) telescopic screen assembly and the wider 203 mm (8-inch). Evidence derived from flow metering (presented below) would reveal this had significantly exacerbated sand entry.

The first camera survey revealed that the screen assembly was positioned off-centre. It was considered that the long assembly had flexed under its own weight during installation when sitting on fill-material at the bottom of the hole. The misalignment between casing and screen had caused the J-latch to intrude which caused difficulty in lowering of the submersible pump beyond the top of the

screen assembly (so as to gain additional drawdown and maximise pumping rate and encourage flow from the lowest screen).

Clearly the J-latch could have been damaged by repeated raising and lowering of the pump and/or the star-dropper incident. The detection of a 100 mm diameter plastic pipe buried under 3 m sand and recognized to be the pump shroud, reinforced the view that the J-latch had been stressed by the pump.

8.4 Bailing, airlifting and final camera survey

Attempts to remove the sediment from the base of the well through bailing and surging operations were thwarted by the pump shroud. The upper two screens were briefly surged with a rubber flange system and the bottom re-bailed. Unfortunately the problem of sand ingress could not be overcome. A plan to later inject a clay dispersing agent into the well was cancelled due to the sand ingress.

The final video camera survey on 30 May 2000 revealed that the bailing cleared most of the sand apart from the bottom metre of screen (bottom of hole at 82.1 m). The screens were significantly cleaner than in March. The star dropper was not recovered.

Pump testing in July and November 2000 clearly showed that there was a slight improvement in well performance as compared to the situation in December 1999 situation, but still well below the initial conditions (Figure 11).

8.5 EM flowmeter survey

An electromagnetic (EM) flowmeter survey of the ASR well was conducted on 14 Jul. 2000 to determine the flow contributions from each of the three screened intervals. Positions immediately above the screens were selected and all of the flow through the cross-sectional area of the well channelled through the flowmeter using a circumferential rubber flange fitting.

The flowmeter survey was performed under pumped conditions, where the tested well was simultaneously pumped at a constant flow rate of 4.2 L/minute and the flow distribution determined after the drawdown had stabilized. Before any pumping had occurred the ambient rate of flow within the well was determined to assess the net differential flow. Details on the test procedure are given by Molz et al., (1994). Changes in flow rate between adjacent depths implied there was flow into or out of the well over that particular interval.

The survey revealed that the bulk of the flow contribution (58%) was derived from the perforated junction (Table 4). The remainder (42%) was derived from the screened intervals, with each interval contributing in approximate accordance with the screen length. The flowmeter data implied that the in-filling of the lowest screened interval had not been the cause of the substantial decline in injection rate.

Table 4. Summary of flow meter survey results

Well zone	Percent contribution	Anticipated percent contribution [A]
Above screens [B]	58	0
Upper screen	18	52
Middle screen	11	26
Lower screen	13	22

[A] assuming uniform flow contribution per unit length of screen
[B] interval above 63.5m bgs and includes casing/screen junction

8.6 Water quality monitoring

All of the available water quality monitoring data apart from that already presented in Table 1 and Figure 6 is given in Table 5. This table provides information on ambient groundwater in the ASR well, an indication of source water quality from the main lagoon (two seasons prior to injection), and groundwater from the ASR well during initial backwash redevelopment as well as prior to-, soon after- and in latter stages of- well chlorination. The following points can be drawn from the data:

- wetland water is significantly fresher than the marginally brackish ambient groundwater (by a factor of six in terms of the chloride concentration).
- marginally elevated groundwater EC with respect to the wetland water on 10 Dec. 99 and slightly higher again on 19 Jan. 01 suggest some residual ambient groundwater in backwash water, perhaps owing to incomplete flushing caused by aquifer heterogeneity exacerbated by the small volume of water injected or boundary effects from the multi-aquifer well completion.
- high iron content in the initial backwash water relative to the injectant (3-17 mg/L *cf.* 0.5 mg/L) and detection of low levels of heterotrophic iron bacteria in the groundwater is suggestive of the dissolution of iron-bearing minerals due to the injection of oxygenated injectant into a partially reduced groundwater (due to the absence of data on iron levels in ambient groundwater other mechanisms may also be possible).
- no active algal cells were observed in the backwash waters, eliminating the possibility of growth of algal species that do not rely on light for their metabolism.
- the source water contains sufficient particulate matter to reduce pore-space of the media close to the well screens and sufficient organic matter and other key nutrients to promote biofilm production as was previously noted.

Table 5. Composition of wetland water, ambient groundwater, and backwash water at various stages of the remediation program

Parameter	Units	Wetland (MLG) *(8 Mar.97)*	Ambient Groundwater [(1)] *(12 Aug.97)*	Intermittent backwash [(1)] *(10 Dec.99)*	Pre-chlorination [(1)] *(19 Jan.00)*	Initial post-chlorine backwash [(1)] *(27 Jan.00)*	Final post-chlorine backwash [(1)] *(31 Jan.00)*
Suspended solids	mg/L			518	88	413	15
Turbidity	NTU		1.6	310	59	397	15
TDS	mg/L		1110		256	628	
Conductivity	μS/cm	330		421	458	1177	526
pH	-	6.39		7.63	6.98	7.14	7.08
Dissolved oxygen	mg/L			2.1 [(2)]			
Alkalinity	mg/L			168	169	141	93
Bicarbonate	mg/L	46.8	511	205	206	172	114
Bromide	mg/L			0.13	0.15	0.34	0.21
Calcium	mg/L	25.2	41	44.5	44.3	55.8	31
Chloride	mg/L	63.47	383	32	40	243	91
Fluoride	mg/L			0.19	0.2	0.39	0.27
Magnesium	mg/L	6	48.2	12.3	10.3	23	9.4
Potassium	mg/L	6.2	7.1	5.3	4.3	7.6	5.5
Silica (reactive)	mg/L			8	10	8	7
Sodium	mg/L	29	317	36.2	31.7	126	60.2
Sulphate	mg/L	14.1	65.1	7.6	9.1	78.2	27.1
Iron (total)	mg/L	0.5		17.1	5.5	7.45	2.91
Manganese (total)	mg/L	0.03		0.668	0.44	0.515	0.191
Phosphorus (total)	mg/L	0.1		0.443	0.472	0.272	0.285
TKN as N	mg/L			3.04	4.27	5.04	1.92
Ammonia as N	mg/L	<0.4		2.48	3.75	1.8	1.6
Nitrate + nitrite as N	mg/L	<0.4		0.026	0.014	0.039	0.032
Dissolved organic carbon	mg/L	17		8	8	17.6	4.2
Total organic carbon	mg/L			10	8	18.6	4.5
Biochemical oxygen demand	mg/L				13	22	5
Chemical oxygen demand	mg/L				26	71	25
Algae (total)	cells/mL				ND [(3)]	ND [(3)]	ND [(3)]
Total heterotrophic count (20^{o}C)	cells/mL			18000	500 – 5000 [(4)]	50000	515
Total coliforms	cells/50mL				0	0	0
E. coli	cells/100mL			<10			
Heterotrophic iron bacteria	cells / mL			900			
Pseudomonas spp.	cells/100mL			1800	40	D [(5)]	0

[(1)] sampled from ASR well; [(2)] from sampling on 14 Dec. 99; [(3)] ND = not detected; [(4)] value lower than expected, may be due to use of non-optimal medium and/or minor contamination of well with chlorinated water (well was purged and pumped dry after this event before sampling); [(5)] D = detected

9. WATER QUALITY CONTRASTS WITH TWO OTHER ASR SYSTEMS

Experience drawn from other case studies over long periods of time have shown that higher levels of pre-treatment than was provided to the Urrbrae Wetland water is required to avoid excessive well clogging problems. A review of the literature reveals that in the Netherlands, with aquifers of similar mineralogical characteristics, sites operate using source waters treated to a level such that the MFI value is <(3-5) s/L^2 and AOC is <10 μg/L, even though aquifer transmissivity can be up to two orders of magnitude higher than at Urrbrae (Table 6). While the quality of water required to avoid clogging will depend on the aquifer, it is thought that the Netherlands experience sets a target for sustainable operations in low to moderate transmissivity silicious aquifers. These parameter values are significantly lower than the values that were injected at Urrbrae. The poor quality of source water is considered to be largely responsible for the failure of the Urrbrae trial.

Corresponding values for water quality parameters for the Bolivar ASR site are considerably higher than in the Netherlands (and largely within the range measured at Urrbrae) owing to the higher aquifer transmissivity and calcite content of the aquifer, which serves to offset physical and microbial clogging if injectant is undersaturated in calcite. It is interesting to note that, from a clogging viewpoint at least, the Urrbrae water may have been acceptable for injection at Bolivar. This convincingly illustrates the point that water quality criteria cannot be considered in isolation, but must also consider the nature of the receiving formation.

Table 6. The quality of recharge water at Urrbrae compared with two contrasting case studies where comprehensive investigations have shown the viability of ASR

Location:	**Urrbrae, SA** [A]	**Bolivar, SA** [B]	**Netherlands** [C]
Source water type:	*Stormwater*	*Reclaimed water*	*Treated river water*
Target aquifer type:	*Siliceous*	*Carbonaceous*	*Siliceous*
Transmissivity (m^2/day):	*6*	*150*	*80-1800*
MFI (s/L^2)	90-389	<100	<(3-5)
Turbidity (NTU)	0.8-55	<3	very low
AOC/BRP (μg/L) [D]	39-331 (as BRP)	1000 (as BRP)	<10 (as AOC)
Total Nitrogen (mg/L)	0.9 [E]	<10	very low

[A] this study' [B] reported in Pavelic *et al.,* (2007); [C] from Olsthoorn, (1982) and Hijnen and van der Kooij, (1992); **[D]** AOC and BRP values are not directly comparable (Page and Dillon, 2007); **[E]** from Lin *et al.,* (2006) (since data in Table 5 does not include the organic-N component)

10. HYPOTHESES FOR FAILURE

The reason why the performance of the ASR well at Urrbrae was not restored significantly by the three different mechanical and chemical techniques is interesting when one considers that these techniques have consistently been successful in a variety of other ASR studies (eg. Olsthoorn, 1982;

Pyne, 1995; Pérez-Paricio and Carrera, 1999; Pavelic *et al.,* 2007). Whilst we have noted that there was room for improvement in some of the approaches used (eg. through the use of acid following chlorine), the literature suggests that the outcome should have been far more successful if the deterioration of the well was due to just particle filtration and biofilm growth.

However, it is also conceivable that the efficiency of unclogging was limited by the low permeability of the aquifer, in that chlorine was not easily able to access the biofilm occupying the small pore spaces of the aquifer. Since the volume of water recovered by pumping was probably small compared to the volume previously injected (Table 3), most of the introduced nutrients would have been retained within the aquifer and the conditions for supporting the biomass maintained. Data on changes in aquifer permeability close to a reclaimed water ASR well suggest that biomass can persist within porous media for periods in excess of a year (Pavelic *et al.*, 2007).

Consideration must also be given to the prospect that residual drilling mud had invaded the formation or that fine textured aquifer particles had been mobilized and redeposited. Clearly there is inadequate direct evidence to confirm this, although the proposition is rendered likely in part by the lack of direct evidence on some other potential causative factors.

It was possible that on initial redevelopment of the well only a small fraction of the mud cake was removed, and injection caused blockage of the remaining unclogged parts of the formation. Segalen *et al.,* (2005) offers evidence that the choice of drilling technique, the quality of the drilling, well completion and design, have a very significant effect on the performance of ASR wells in unconsolidated aquifers.

Clay release due to the injection of low salinity water can result in rapid declines in permeability in brackish aquifers that contain reactive clays minerals. Since recharge causes divalent cations to substitute with monovalent cations, preconditioning the aquifer by initial flushing with $CaCl_2$ has had some success in alleviating clay dispersion (eg. Brown and Silvey, 1973). In addition, purely physical forces can mobilize fines. The propensity of the aquifer to erode and redeposit fine particles within pore throats is dependent on the pore water velocity, and on the grain size distribution and pore geometry of the aquifer (Nakai, 2006). This issue cannot be adequately resolved since the physical and mineralogical characteristics of the target zones are largely unknown.

11. SUMMARY OF LESSONS LEARNT AND CURRENT RESEARCH

This attempt to recharge passively-treated urban stormwater via a multi-completion ASR well that targeted confined, unconsolidated silicious aquifers at the Urrbrae Wetland site in the late 1990s, resulted in a significant decline in injection rates and the cessation of injection within the first year of operation. With the benefit of hindsight and the greater knowledge available at the present time, it is not surprising that clogging had occurred, particularly given the physico-chemical characteristics of the recharge water. Clogging was attributed to the high levels of suspended solids and bacterial growth fed by labile organic carbon and other nutrients in the wetland water. Mechanical and/or geochemical effects due to residual drilling muds or the mobilization and redepositing of aquifer fines possibly have an impact on clogging, however, this is extremely difficult to verify in practice. Further, a perforation of the well screen joint caused infilling of the screens with sand and reduced the effectiveness of procedures to unclog the ASR well.

Resolving the cause of clogging was initially considered a normal part of the ASR commissioning process. Three different attempts were made to restore the clogged ASR well. They included: repetitive backwashing; injection of chlorine disinfectant and backwashing; and bailing/surging to recover sand that had in-filled the lowest screened interval. These techniques proved to be ineffective in restoring the performance of the well. Restoration would ultimately require that the screen assembly be recovered and replaced, and the gravel pack re-established. Because the cost of retrofitting the well was similar to the cost of a new well, this was considerably non-viable in the absence of new funding sources.

The fundamental problem at the Urrbrae site was that the level of pre-treatment given to the recharge water was inadequate for the low transmissivity aquifer targeted, irrespective of the lack of success in restoring the injectivity of the ASR well. This was exacerbated by the absence of a nearby observation well and monitoring data during the injection phase of the trial. Both elements were initially intended but omitted due to budgetary constraints. We consider that a more focussed well restoration program would have ensued had this baseline information been collected. At least one nearby monitoring well is recommended in all situations where clogging is a potential issue. A confounding problem was premature injection of water before a pump was installed to allow redevelopment. This probably resulted in filter cake compression and made subsequent redevelopment much more difficult. Infilling of the well with sand was another confounding problem.

As a result of this experience, it is concluded that fine-grained unconsolidated aquifers are unacceptable targets for operational ASR systems with wetland-treated urban stormwater until further research is conducted to ensure sustainable injection.

Several research projects have commenced to address improving the design of ASR wells (Segalen *et al.*, 2005; Pavelic *et al.*, 2006) and on methods of pre-treatment including the use of roughing filtration (Lin *et al.*, 2006) and biofiltration (Page *et al.*, 2006). The studies on pre-treatment are aimed at removing colloidal matter and key bio-available nutrients from the recharge water. These methods have been selected for their simplicity, low cost and potentially low maintenance requirements, making them suitable for use in developing countries and in Australia for urban stormwater harvesting.

Consequently a stormwater treatment facility has been established at the Urrbrae Wetland adjacent to the ASR well with the aim of identifying appropriate methods to achieve suitable quality water for injection (Page *et al.*, 2007). At that stage it is proposed that a new ASR well be established within close proximity to the current well. The existing well would serve as an observation well allowing a rigorous assessment of clogging and its management.

12. ACKNOWLEDGEMENTS

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APPENDIX 1: THE JOHN BOTTING AND ASSOCIATES AND LISDON ASSOCIATES, (1998) REPORT

URRBRAE ENVIRONS WATER MANAGEMENT PLAN
Stage 2A -- Aquifer Injection Investigations

April 1998

Prepared for CSIRO, University of Adelaide, Urrbrae Agricultural High School, Unley High School, City of Mitcham, Patawalonga Catchment Management Board, Urban Stormwater Drainage Subsidy Scheme and SARDI

by John Botting and Associates Pty Ltd and Lisdon Associates.

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6. WATER QUALITY

The large catchment size, when compared with the relatively small wetland volume, means that stormwater that could be available for aquifer injection would only be retained within the wetland for a short period of time. This short retention time has implications with regard to the likely quality of the stormwater available for injection. In the short term, physical filtration will be necessary to remove coarse pollutants. However, this method will not remove soluble pollutants, or fine sediments and their associated pollutants.

Experience gained from the wetland site during periods of high inflows have shown that the existing gross pollutant traps are often overtopped.

Scope may exist on the site to construct an elongated filter bed that utilises natural biological processes to assist in the removal of both soluble and insoluble pollutants.

If any aquifer injection is proposed on a local scale, (e.g. for research and teaching purposes), then consideration should be given to the implementation of additional pollutant removal mechanisms.

7. CONCLUSIONS

The following major conclusions have been drawn from Stage 2A of this study.

1. Hydrogeological tests conducted on the trial well have revealed relatively low yield and even lower injection rates, which will limit the amount of water which can be extracted from the well.
2. Based on the revised yield and injection rates, it has been found to be uneconomical to proceed with a large scale aquifer injection and recovery scheme for this site.
3. It was also found that smaller scale aquifer injection and recovery schemes would not be economically feasible.
4. Additional stormwater treatment measures are considered necessary if any stormwater is to be injected into the well as part of a small scale localised scheme.

1. INTRODUCTION

A steering group was formed in 1996, consisting of representatives from CSIRO, University of Adelaide, Urrbrae Agricultural High School, Unley High School and the City of Mitcham, to develop a local water management plan for the research and education institutions in the environs of Urrbrae. A report[1] was released in February 1997 which discussed the water balance opportunities for the study area. The February report concluded that there would be sufficient stormwater runoff available from the catchment area to enable it's utilisation as an alternative source of irrigation water. The recommended method of using the stormwater runoff was by injecting treated stormwater from the Urrbrae Agricultural High School wetlands into an aquifer during winter, and then pumping the water out of the aquifer for irrigation use during summer.

Following consideration of the water balance report, the steering group proposed to conduct the further work in stages, with the continuation of the project being dependent on the successful outcome of each preceding stage. In broad terms, the proposed stages were;

Stage 1	Water Budget Investigations (completed)
Stage 2A	Feasibility Study and Preliminary Design
Stage 2B	Detailed Design
Stage 3	Construction and Implementation

This report summarises the investigations conducted as the second component of the preparation of an integrated water management plan for the Urrbrae environs.

2. STUDY OBJECTIVE

The principal objective of the Stage 2A investigation was to determine the suitability of the aquifer to enable an injection and recovery scheme to be implemented. Additional scoping of the distribution system were also intended to be carried out. Results obtained from the Stage 2A study were to provide sufficient information to allow the detailed design of the scheme to be undertaken.

The major tasks that were to be undertaken for Stage 2A were;

1. drill a bore on the wetland site at the most beneficial position,
2. assess the lithology and select a target aquifer,
3. complete the well as an injection and recovery well,
4. undertake a pilot scale injection trial,
5. assess the sizes of irrigated areas across the partner institutions,
6. determine the costs of reticulation to these areas,
7. develop a daily model of stormwater flow through the wetland,

[1] Urrbrae Environs Water Management Plan, Stage 1 -- Water Budget, February 1997

8. estimate the average annual amount of stormwater which could be harvested,
9. determine the active storage requirement for the target volume.

The investigations were coordinated by John Botting and Associates. Mr Don Armstrong (Lisdon Associates) was responsible for the hydrogeological component, which consisted of tasks 1 to 4 inclusive. The remaining tasks were to be undertaken by John Botting & Associates.

3. INJECTION WELL LOCATION

A suitable location for the trial injection well was found in the south west corner of the lined pond enclosure. This site, shown on Figure 1 was selected since it was adjacent to the likely source of injection water and within a short distance of the necessary power cables.

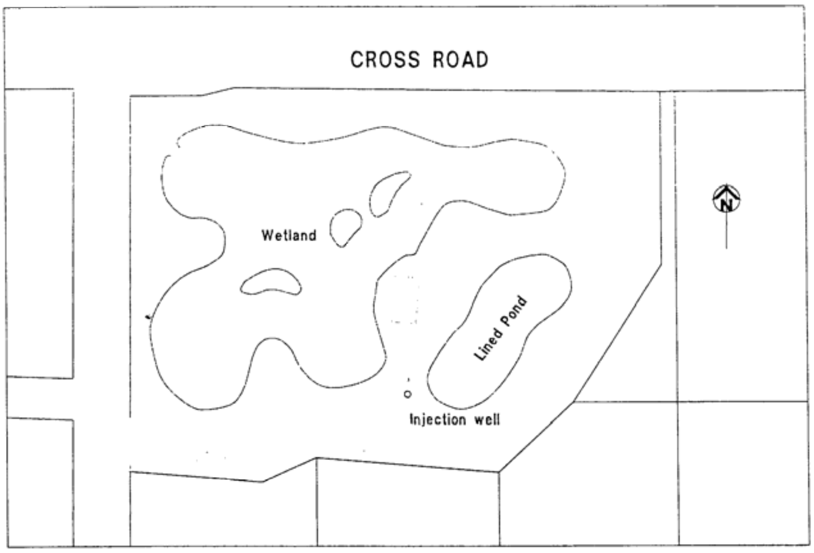

Figure 1 Injection Well Location

Prior to the commencement of any well drilling activities, a site management plan was prepared. This plan provided a clear indication of the respective responsibilities for the various environmental safeguards developed to ensure that the well drilling and development activities would only have a minimum impact on the wetland site.

4. HYDROGEOLOGICAL INVESTIGATIONS[2]

4.1 GENERAL

In order to evaluate the potential at the Urrbrae Wetlands site for a working ASR operation, a trial injection and recovery well was drilled. The operation was jointly funded by Mitcham City Council, Patawalonga Catchment Management Board, Urban Stormwater Drainage Subsidy Scheme and University of Adelaide, CSIRO, PISA, SARDI site management committee and represents Stage 2A of the Urrbrae Environs Water Management Plan.

The school~~s~~ associated with the project contributed by making the land available.

Drilling was commenced, by Mines & Energy SA (MESA), on Monday 7th July 1997 and by Wednesday 9th July had reached the depth of 93.6m at which point it was decided to terminate drilling and run a suite of electrical logs to provide detailed information on lithological boundaries as the basis for the design of the casing and well screen.

The drilling target was a unit of undifferentiated Tertiary sands believed to be a lateral equivalent of the Lower Port Willunga Formation. Previous drilling at Unley High School and WAITE Sports Ground demonstrated the presence of this aquifer beneath the Hindmarsh Clay and Carisbrooke Sands.

The well at Unley High School was drilled (in August 1968) by the cable-tool method and "running" sands prevented a successful completion. A depth to water of 18.3m and salinity of 1200mg/L were recorded at the time.

At the WAITE Sports Ground the well was completed (in October 1959) with a 3m length of 4" (100mm) diameter screen from 79.7 to 82.7m which yielded 1.6L/sec with a salinity of 1695mg/L on completion. A subsequent sampling in 1988 showed a salinity of 660mg/L and depth to water of 16.38m.

4.2 DRILLING

The hole (Permit Number 41793) was collared on Monday 7th July 1997 at the site shown in Figure 1, between the main wetland and the lined pond to the south of the teaching building.

5m of 298mm PVC collar casing was installed after drilling the first 5m with air and foam. Mud drilling commenced from 5m and a gravel was intersected immediately beneath the collar casing which caused some minor problems due to collapse throughout the first day of drilling. Hindmarsh Clay extended from 5m to 60m and provided the usual drilling problems associated with the formation of mud rings around the drill pipe which resulted in relatively slow progress through the clay. At close of work on Monday the depth was 24m.

[2] LISDON ASSOCIATES July 1997

Beneath the Hindmarsh Clay the Carisbrooke Sand was encountered. This was a white medium to fine grained calcareous sand with some ferruginous grains and possibly some interbeds of silt. Lithic fragments (quartzite and schist) appeared at around 69m where the colour of the sand changed to grey and the grain size increased to medium to coarse with some gravel. This grey sand/gravel represents the initial target zone which extended downwards to a lignitic[3] horizon around 78m. Beneath the lignitic horizon the sand was coarse, grey in colour, and quartz rich with lignite and some gravel sized material. At close of work on Tuesday the depth was 84m. The unit became darker in colour with increasing depth indicating that lignite was becoming more abundant although the sand was still coarse grained and quartzose. Drilling was stopped at 93.6m as the lignite content appeared to be increasing, probably due to the appearance of lignitic clay.

The lignitic material is believed (Gerges[4] pers comm.) to be a lateral equivalent of the Aldinga Member which is better developed in the southern part of the Metropolitan area Tertiary sequence.

4.3 GEOPHYSICAL LOGGING

MESA geophysical logging equipment arrived on site on Wednesday at 12.30pm but could not immediately log because of remnant mud rings which blocked the upper part of the hole. The hole was cleaned out and logging commenced at 4.00pm.

The following suite of geophysical downhole logs was run

- Gamma
- Spontaneous Potential (SP)
- Neutron
- Bulk Density
- Point Resistance
- Focused Resistivity
- Calliper

The quality of the logging data was excellent, enabling the different lithological units in the sequence to be accurately located and identified. The logs also facilitated the selection of the casing depth and well screen design.

A preliminary estimate of groundwater salinity, based on an assumed porosity of 0.25 and the focused resistivity value of 31ohm-m was >1500mg/L TDS. The lower part of the geophysical log is shown in Figure 2 together with the geological interpretation.

[3] brown coal of woody texture
[4] Nabil Gerges, MESA

4.4 CASING

After reaming from the pilot hole diameter of 229mm (9inch) to 280mm, the well was cased with 200mm ID Class 12 UPVC from the surface to 65m. The annulus between the casing and the wall of the hole were then grouted with a cement slurry introduced through the drill pipe at the foot of the casing. Cement grout was observed to return to the surface indicating a successful grouting operation.

4.5 RUNNING SCREEN ASSEMBLY

After renewing the mud, the aquifer section below the casing was cleaned out and the screen assembly was run in. Minor difficulties were encountered with the threads on the ends of the sub assemblies but the unit was eventually installed with the sump bottom within approximately 30cm of the design position.

The well completion details are illustrated in Figure 3.

4.6 WELL SCREEN

On the basis of the geophysical logs together with the limited amount of sample data the well screen was designed as follows.

66.0m to 71.0m (65.84 to 70.84)	slot aperture 0.5mm opposite medium to coarse sand of the lower part of the Carisbrook Sands
71.0m to 73.5m	zero aperture (blank screen) opposite finer grained and possibly lignitic material
73.5 to 76.0m (72.85 to 75.37)	1mm slot aperture opposite coarse sand/fine gravels of the Upper Pt. Willunga Formation
76.0 to 81.0m	Blank screen opposite lignitic interval
81.0 to 83.0m (80.57 to 82.67)	1mm slot aperture as above
83.0 to 85.0m (bottom at 84.7m)	Sump with end cap.

4.7 DEVELOPMENT

The mud was displaced into tanks and jetting of the lower screen commenced on Wednesday 16th July. A little fine sand and lignitic silt was produced by the jetting. The middle screen yielded fine silty sand and lignite fragments, some of which were fibrous.

The upper screen produced fine sand (approximately 0.3mm) typical of beach sand together with abundant drilling mud.

Air lifting was commenced at 2.00pm and by close (4.00pm) was still producing abundant "beach" sand but the amount appeared to be declining from an initial 10% to less than 5% solids. The complete absence of lignitic material at this stage suggested that the lower screens were buried in fine sand from the top screen. The sand disappeared from the airlift returns within one hour on the second day of development and the upper screen was considered to be developed.

The middle screen was then developed by lowering the airlift line into the blank screen at approximately 72m. Airlifting yielded abundant fine sand and silt together with some coarse grey sand, but very little lignite. After approximately 2 hours the air lift was lowered into the sump and the lowest screen was developed by both continuous airlifting and occasional surging. The returns were initially grey, due to fine lignitic particles but by close of work the water was becoming cleaner with less than 5% of very fine sand.

Development continued on Friday 18th July with airlifting and frequent backwashing.

The fine sand continued to appear at the start of each round of airlifting due to the high entry velocities developed at this time. Since it is not possible to airlift this particular well at a continuously high rate due to the inefficient airlift conditions, it was decided that final development should be by pumping.

The final airlift yield was 3L/sec and the salinity of the water was approximately 1300mg/L with a standing water level of 30.8m below surface. Less than 1% of fine (<0.3mm) sand was present in the final airlifted water.

The rig left the site at 4.00pm, Friday 18th July 1997.

The MESA pumping unit arrived on site on Tuesday 5th August and commenced development pumping at 3L/sec. The rate was gradually increased, with partial recovery between increases until a maximum rate in excess of 6L/sec was reached. After 5 minutes the rate had fallen to 4.5L/sec and the water level was clearly at the level of the pump intake. With the pump "forking" the yield was 4.25L/sec after 20 minutes.

An attempt was made to drop the pump into the screen assembly in order to gain additional head and, by allowing a higher pumping rate, to encourage flow from the lower screens. The pump could not be lowered beyond the top of the screen assembly therefore the MESA downhole TV camera was run to inspect the top of the assembly which was found to be lying off centre, presumably because the very long assembly had flexed under its own weight when standing on the fill at the bottom of the hole.

The TV camera revealed also that the sump was filled with sediment to approximately 83.43m below the top of the casing which would translate to approximately ~~83.9m~~ below ground, or 0.4m above the bottom of the sump at 84.7m below ground. 84.3

A tapered "pathfinder" was made up and attached to the bottom of the pump in the hope that it would allow the bottom of the pump to enter the top of the screen assembly. On Thursday 7th September, the pump was again installed and passed through the top of the screen assembly.

Over a three hour period the pump was operated at various levels within the screen assembly accompanied by surging. Up to 7L/sec was recorded at the start of a pumping cycle but this fell rapidly as drawdown developed. All screened intervals were pumped and very little sediment was produced from any interval indicating that the well is as completely developed as possible without resorting to further jetting of the lower screens. The final thirty minutes of pumping were carried out with the pump sitting at the depth of 63.5m, just above the screen assembly. The pump was forking at maximum discharge capacity which gradually declined to 4.25L/sec.

4.8 DISCHARGE TESTING

The MESA hydraulic pumping unit was returned to the site on Tuesday 12th August and a step drawdown test followed by a 5 hour constant discharge test was carried out.

The pump was set at 63.5m, just above the screen assembly, and pumping was carried out as follows.

- 30 mins. at 1.0L/sec
- 30 mins. at 2.0L/sec
- 410 mins at 3.0L/sec

The initial depth to water, below the test datum which was 0.35m above the top of casing, was 30.65m which equates to 30.30m below top of casing.

Drawdown versus time (log scale) is shown in Figure 4 from which it can be seen that well losses are a considerable part of total drawdown. The data were analysed to determine Transmissivity which was found to be very small at around $9m^2$/day.

A discharging boundary was noted at approximately 230 minutes into the test. This boundary causes the drawdown rate to double for a constant discharge. The source of the boundary effect is not known but a rough estimate places it within less than 500m of the pumping well.

The step drawdown part of the test was analysed by the graphical Eden Hazel method.

The resultant plot is shown in Figure 5.

The objective of carrying out a step drawdown test is to determine the constants a, b, and C, in the Well Equation:

$$st = aQ + b\left(\log_{10} tQ\right) = CQ^2$$

Given these constants it is possible to estimate the drawdown at time t due to any pumping rate Q, provided that no boundary effects occur within the time from $t = 0$ to t.

The constants determined for the Urrbrae well, which are valid for times less than 230 minutes, are :-

$a = 0.055$
$b = 0.023$
$C = 0$

The zero value for the C constant means that in this well, turbulent flow losses are negligible at the tested rates of discharge. The high well losses are due to the 'a' constant which is related to the very near well conditions including aquifer permeability. If the C value had been significant, the incremental drawdown at each change in discharge rate would have been greater and would have increased significantly as the discharge rate increased.

Given that there is only a limited head available (63.5-30.3m = 33.2m) between the likely position of a pump inlet and the water level, and the fact that some allowance must be made for seasonal fluctuations in water level in the aquifer, the most that can be expected from this well on a regular basis, is 3L/sec over an 8 to 10 hour pumping period.

This would yield only 86.4 (8 hrs) or 108m^3/day (10 hrs) over the pumping season.

If the discharging boundary is in fact another well which was pumping at the time of the test, it may be possible to increase the yield to 3.5L/sec by installing a shroud on the pump and setting the inlet to the pump shroud within the blank part of the screen assembly below 63.5 m thus increasing the available drawdown.

4.9 INJECTION TRIAL

The equipment shown diagrammatically in Figure 6 was set up on Friday 22nd August and a short trial was run to ensure that the equipment was functioning properly.

12.24m^3 of water was injected at a rate of 3.4L/sec over a one hour period, resulting in a water level rise of 25.1m.

The trial injection proper was started at 10 am Saturday 23rd August at 3.4L/sec. Head versus time is shown, from the beginning of the trial proper at 10 am Saturday in Figure 7, (linear t), Figure 8 (log t) and Figure 9 (log/log).

The injected water entered the aquifer under gravity alone until the casing became full after approximately 100 mins after which the system became pressurised and remained so for the duration of the trial.

Pressure build up was relatively slow with a head build up of 3.5m above the top of casing after 800 mins and 4.4m after one day of injection.

After 9 days of injection the head had risen to 9.7m above top of casing and the flow rate had stabilised at around 2.8L/sec. The total volume injected was recorded as 2,056,916L giving an average injection rate over the 9 days of 2.62L/sec which is less than any of the observed instantaneous flow rates. !! unsure which correct

Preliminary analysis of the data showed a logarithmic rate of increase in head of 5m/log cycle of time which implies that after 10 days the head should be 9.4m above top of casing and after 100 days 14.4m.

Injection rate varied from the initial 3.4L/sec to a minimum of 2.68L/sec at 10,000 mins, rising again to 2.8L/sec at the end of the injection period.

This rate of acceptance was considerably greater than would be predicted from the results of the step drawdown test, particularly in the light of the large temperature difference between the injected and aquifer waters of at least 8 degrees C.

It was considered possible that the lower screens were "blinded" by lignitic material under discharging conditions but the process of injection may have been forcing the lignitic material away from the screen apertures allowing water to penetrate the "blind".

A pump out test was considered necessary to determine whether the performance of the well had changed permanently or if it would revert to the initial condition, in which case any water injected into the lower screens may not be recoverable.

4.10 PUMP OUT TEST

An attempt was made on 18th November 1997 to repeat the initial pumping test at a rate of 3L/sec to aid in the assessment of the well performance following the injection trials. This assessment was carried out with respect to both drawdown and water quality.

The static water level before the commencement of pumping was 29.7m below the top of the casing.

The initial discharge water was slightly turbid and had a foetid aroma. After 5 minutes of pumping, the strength of the aroma increased and the colour became khaki-yellow with a great deal of suspended clay together with approximately 10% by volume of the fine sand previously produced from the top screened interval.

Sand and suspended clay were produced during the 5 hours of pumping regardless of the pumping rate which was varied between 1 and 4L/sec. It was estimated that at least 2 cubic metres of sand were produced during the 5 hours of pumping. This is considered to have significantly increased the dimensions of the annular cavity outside of the top screen.

It was apparent that the level of development attained prior to the injection phase (clean water at > 4.5L/sec) had been destroyed in the course of the injection. Two possible explanations for this setback are: