

Assessment of the Potential for Well Clogging Associated with Salt Water Interception and Deep Injection at Chowilla, SA

Paul Pavelic, Joanne Vanderzalm, Peter Dillon,
Andrew Herczeg, Karen Barry, Kerry Levett, Joao
Mimoso and Paul Magarey*

* Department of Water, Land and Biodiversity and Conservation

October 2007

Final Report to Department of Water, Land, Biodiversity and
Conservation



Water for a Healthy Country Flagship Report Series

ISSN: ISSN: 1835-095X

Copyright and Disclaimer

© 2007 CSIRO To the extent permitted by law, all rights are reserved and no part of this publication covered by copyright may be reproduced or copied in any form or by any means except with the written permission of CSIRO.

Important Disclaimer

CSIRO advises that the information contained in this publication comprises general statements based on scientific research. The reader is advised and needs to be aware that such information may be incomplete or unable to be used in any specific situation. No reliance or actions must therefore be made on that information without seeking prior expert professional, scientific and technical advice. To the extent permitted by law, CSIRO (including its employees and consultants) excludes all liability to any person for any consequences, including but not limited to all losses, damages, costs, expenses and any other compensation, arising directly or indirectly from using this publication (in part or in whole) and any information or material contained in it.

Cover Photograph:

Title: Backwaters along Chowilla Creek, upstream from Renmark, SA 1993

From: CSIRO Land and Water Image Gallery <http://www.clw.csiro.au/ImageGallery/>

File: PDC00411_006.jpg

Photographer: Willem van Aken

CONTENTS

- EXECUTIVE SUMMARY 2**
- INTRODUCTION 3**
- LITERATURE REVIEW 3**
 - International..... 4**
 - National 4**
 - Hydrogeology of the Target Aquifer 5**
 - Sampling and Analysis Methods 5**
 - Physical Clogging Assessment 8**
 - Turbidity, TSS and VSS data 8**
 - MFI data 8**
 - Particle size data..... 8**
 - Biological Clogging Assessment 9**
 - Geochemical Assessment..... 9**
 - Mineral phases 10**
 - Aeration of the source water 11**
 - Mixing and temperature changes..... 11**
 - Reaction with minerals in the storage zone 12**
 - Ion exchange..... 13**
 - Management of Clogging 14**
- CONCLUSIONS 14**
- ACKNOWLEDGEMENTS 15**
- REFERENCES 15**
- APPENDIX A. MINERALOGICAL AND CHEMICAL DATA FOR CUTTING
SAMPLES COLLECTED FROM THE RG INJECTION WELL (CH1)..... 18**
- APPENDIX B. PARTICLE SIZE ANALYSIS OF MS GROUNDWATER..... 20**

EXECUTIVE SUMMARY

Interception of shallow saline groundwater from the Monoman Sands (MS) aquifer on the floodplain of the River Murray at Chowilla prior to it reaching the River Murray and then injecting it into the deeper Renmark Group (RG) aquifer is proposed as part of a major long-term initiative to control salinity levels in the river and lower the watertable beneath the floodplain.

This report addresses the feasibility of injecting MS groundwater into the RG aquifer from the viewpoint of assessing the risks from well clogging in proceeding with an injection pilot trial at Chowilla.

The quality of the groundwater in the MS aquifer was determined from sampling of three MS wells in 2005 and a further four wells in 2007 for a range of physico-chemical parameters. In January 2007 a 540 m deep well was drilled by the Department of Water, Land and Biodiversity and Conservation (DWLBC) into the RG aquifer and cutting samples collected at regular intervals during drilling. Following well development sampling was performed to determine the quality of the ambient groundwater. Geochemical modelling with PHREEQC using the water quality and mineralogical data enabled an assessment of the potential for geochemical reactions to impact on aquifer permeability.

The results demonstrate that injection into the RG would not be entirely without risk as a result of clogging due to physical, chemical or microbial processes. These risks in proceeding with a trial should be manageable to within acceptable limits by ensuring:

- a) that the quality of the source water from the MS aquifer is improved by pre-treatment prior to injection to achieve low particulate levels (notionally <10 mg/L TSS)
- b) that aeration of the MS source water during storage be prevented or at least minimized
- c) redevelopment be performed before a 20% reduction in injection rate is observed and recovered water evaluated for composition of particulates present
- d) the turbidity and oxygen status of the injectant as well as the injection rates and pressures should be carefully monitored during the trial to inform the operational performance and to enable fine-tuning in subsequent phases of project development
- e) additional parameters should also be evaluated in the injectant and at observation wells directly influenced by breakthrough of recharge water. Parameters would include total and dissolved iron and manganese, electrical conductivity, pH, redox potential, dissolved oxygen, temperature, major ions, nitrogen species (nitrate, ammonium and total Kjeldahl Nitrogen), phosphorus (total and soluble reactive P) and dissolved organic carbon.

INTRODUCTION

A proposal to intercept shallow saline groundwater on the floodplain of the River Murray at Chowilla prior to it reaching the River Murray is part of a major long-term initiative to control salinity levels in the River. It is proposed that saline groundwater from the shallow unconfined Monoman Sands (MS) aquifer would be pumped from multiple wells situated close to the River and injected into the deep confined Renmark Group (RG) aquifer, thereby minimizing the footprint typically associated with surface basin disposal of more conventional Salt Interception Schemes (Rammers *et al*, 2005).

Virtually all well injection operations experience some degree of well clogging that can critically limit the quantity of water that is stored within the aquifer and/or lead to large increases in pressures within the well that may necessitate costly pre-treatment and maintenance procedures, or even project abandonment in extreme cases.

One of the key considerations for the success of the proposed project is the extent of clogging which could occur when the MS groundwater mixes with the RG groundwater as a result of the injection process. Risks associated with well clogging include:

- irrecoverable accumulation of particulate matter that is present within the MS source water
- precipitation of minerals due to chemical or bacterial processes
- swelling or dispersion of reactive clays that may be present in the aquifer
- production of bacterial biomass and polysaccharide 'slime' growth around the well due to nutrients present in the source water
- degassing during injection leading to gas binding

The objective of this study is to determine the feasibility of the proposed pilot injection trial on the Chowilla floodplain from a clogging perspective, taking into account each of the above-noted risks. The assessment consists of:

- a literature review on clogging issues associated with well injection of saline waters
- an evaluation of the physical, biological and geochemical clogging processes likely to occur due to mixing and water-matrix interactions at the Chowilla site

LITERATURE REVIEW

The injection of saline water into deeper saline groundwater aquifers for the purpose of intercepting and disposing of salt is not a common practice, however within the petroleum resources industry, this is relatively common. For example, in the USA, where large quantities of saline formation water are produced as a by-product of extracting hydrocarbons from deep reservoirs, around 8GL/day is reinjected into suitable formations.

Several international and national case studies were reviewed as summarized below. National studies are abstracted from previous work by Rammers *et al*, (2005):

International

In the Gulf Coast region of the USA, brine injection has been reported to cause the clogging of injection wells (Raber *et al*, 1981). Ultra-filtration methods were used to remove particulates and large organic molecules that led to reduced clogging potential. Multi-media filtration combined with chemical pre-treatment by coagulation produced a high quality feed water which also produced satisfactory results. Injection testing without any form of pre-treatment proved unsuccessful.

Saline injection operations in the Texas East and Permian Basins have experienced clogging problems due to chemical precipitation (so-called 'scaling'), clay mobilization and filter cake development which has placed an upper limit on the injection rate that may be sustained. Site-specific solutions have been found such as injecting acids to treat the scaling and providing pre-treatment such as the addition of chemicals to source waters to stabilize clays.

Also in Texas, proposals for the injection of brackish concentrate from desalination plants into depleted oil or gas reservoirs have considered clogging risks and demonstrated the concept is an environmentally and economically attractive option (Nicot and Chowdhury, 2005).

In the Tongonan Geothermal Well Field in the Philippines, a 50% reduction in injection rates within four months of operation was attributed to the presence of particulates within the source water composed of polymerized silica formed from the reaction between magnetite and silica. Improving the level of solids removal through bifurcation traps were shown to found to rectify the problem (Villa *et al*, 2004).

In the Zueta Well Field in Venezuela, clogging by high total suspended solids (TSS) and organics content (hydrocarbons at ppm levels) was identified from declines in injectivity. Initial trials with various chemicals added to the injectant produced only short-lived benefits. Horizontal wells were trialled, but did not prove useful. Further work on removing residual organics and drilling new wells that would allow injection above formation fracturing pressures has been proposed (Briceno *et al*, 2003).

National

A feasibility study to investigate the potential for deep aquifer disposal was undertaken at Noora Evaporation Basin (near Berri in the Murray Basin) in the 1970s (Forth and Reed, 1979). The aim of the study was to determine a target aquifer for deep aquifer disposal of 200,000 mg/L TDS brine at 1.5 ML/day or bitterns (brine following NaCl removal) at 0.8 ML/day over a period of at least 50 years. A field investigation into aquifer hydraulic properties supported by a study of the possible chemical reactions resulting from mixing of injected water with aquifer water was proposed to enable a design to be finalised. The project has yet to proceed.

Sinclair Knight Merz (SKM), on behalf of Goulburn-Murray Water, undertook investigations to assess the feasibility of deep groundwater injection into the Renmark Group aquifer as a means of brine disposal in the Kerang Lakes district of Victoria (SKM, 2003; 2004). A staged approach involving sampling of wells and lakes; investigation of the characteristics of the target aquifer; prediction of impacts, costing of conceptual design and pilot trials was proposed. The salinities of the Renmark Group in Kerang district are typically around 40,000 mg/L, which are about twice the observed values in the Chowilla region. PHREEQC modelling identified the potential for chemical clogging to occur when high calcium lake water combines with high iron groundwater in a reducing environment. Recommendations from the report included investigating potential of biological clogging, and assessing the hydraulic properties of the Loxton-Parilla Sands and Renmark Group formations and completing a more comprehensive sampling programme.

Australian Water Environments (AWE) were engaged by the Murray-Darling Basin Commission (MDBC) to explore the potential for deep injection into the Renmark Group at the Stockyard Plain Disposal Basin to address future disposal options required in the Woolpunda-Waikerie district. Investigations thus far have focussed on the identification of a suitable target aquifer (AWE, 2005).

Hydrogeology of the Target Aquifer

The target aquifer for injection is the Warina Formation that represents the lower zone of the Renmark Group. The Warina Formation comprises pale grey to pale brown, medium to coarse quartz sand with minor carbonaceous fine silt, clay and minor pebble conglomerate (Brown and Stephenson, 1991). The formation is weakly consolidated, loose and friable in places, occasionally lithified in part by siliceous cement, intergranular clay, carbonaceous material, and or quartz pyrite aggregates. In some bores these aggregates have been oxidised to form ferruginous cement. The formation is characterised by clean sand, high porosity, good permeability and is a high yielding confined aquifer.

In January a 540 m deep pilot well was drilled by DWLBC into the Warina Formation (Renmark Group) (CH1; Unit No. 7030-809) at the Chowilla study site. The Warina Formation contained layers of coarse, poorly sorted sand and gravel, grading to sandy clay. Although aquifer pumping tests are yet to be conducted in the test well, an ambient discharge rate of 8 L/sec was recorded by DWLBC during a 95 minute test due to an ambient artesian head of approximately 11m above ground surface (K. Osei-Bonsu, pers. comm.).

Sampling and Analysis Methods

The inherent spatial variations in the quality of the MS groundwater across the Chowilla floodplain required the sampling of multiple wells to gain reasonable insight of the quality likely to be injected in a pilot trial, whereby multiple MS wells would be expected to supply a single RG injection well.

Groundwater samples were collected in April 2007 by DWLBC personnel from CH1 well (Unit No. 7030-809) completed in the RG formation and from four nearby MS monitoring wells identified as wells CH2-CH5 (Unit Nos. 7030- 712, 756, 759 and 765) (**Figure 1**). Field parameters, temperature, pH, oxidation-reduction potential (ORP), electrical conductivity and dissolved oxygen were measured *in-situ* using a YSI 556 multi-parameter instrument. Additional physio-chemical and microbiological analyses (**Table 1**) were undertaken at the Australian Water Quality Centre (AWQC) and at CSIRO Land and Water laboratories according to standard methods (APHA, 1999).

An earlier investigation that targeted the Murray Group Limestone (overlying the Renmark Group) was undertaken in the Gum Flat area of Chowilla during 2005, which included drilling and coring of an injection well and subsequent aquifer testing. While this investigation found the Murray Group Limestone aquifer unsuitable for injection of MS groundwater, it gave additional useful information on source water quality from the MS. Groundwater data from MS observation wells (Unit Nos. 7030-577, 7030-695 and 7130-56) sampled in 2005 was also used in this assessment.

Cutting samples provided by DWLC from the recent drilling of RG well CH1 (Unit No. 7030-809) were examined and six representative subsamples were selected on the basis of the geological and geophysical log and particle size data in addition to one sample of drilling mud and one sample of fine material collected during well redevelopment for physico-chemical and mineralogical analyses (results in **Appendix A**).

Organic and inorganic carbon, exchangeable cations and cation exchange capacity (CEC) were determined by the analytical services group of CSIRO Land and Water, Adelaide using standard methods (**Appendix A**).

Mineralogical determinations on oven-dried and ground subsamples were performed by X-Ray Diffraction with a Philips PW1800 diffractometer by the mineralogical services group of CSIRO Land and Water, Adelaide. Subsamples were sieved to <math><200\mu\text{m}</math> to enhance the detection of the non-quartz components.

Two of the MS groundwater samples collected in May 2005 were analysed for Membrane Filtration Index (MFI) at CSIRO Land and Water (Dillon *et al.*, 2001). The MFI is a measure of the potential for a particular water to clog wells by filtration and development of a filter cake. The test procedure involves measuring the reduction in the rate of flow through a $0.45\mu\text{m}$ filter at normalised temperature and operating pressure.

Particle size analysis was also undertaken on one of the 2005 samples (Unit No. 7030-577) by CSIRO Minerals.

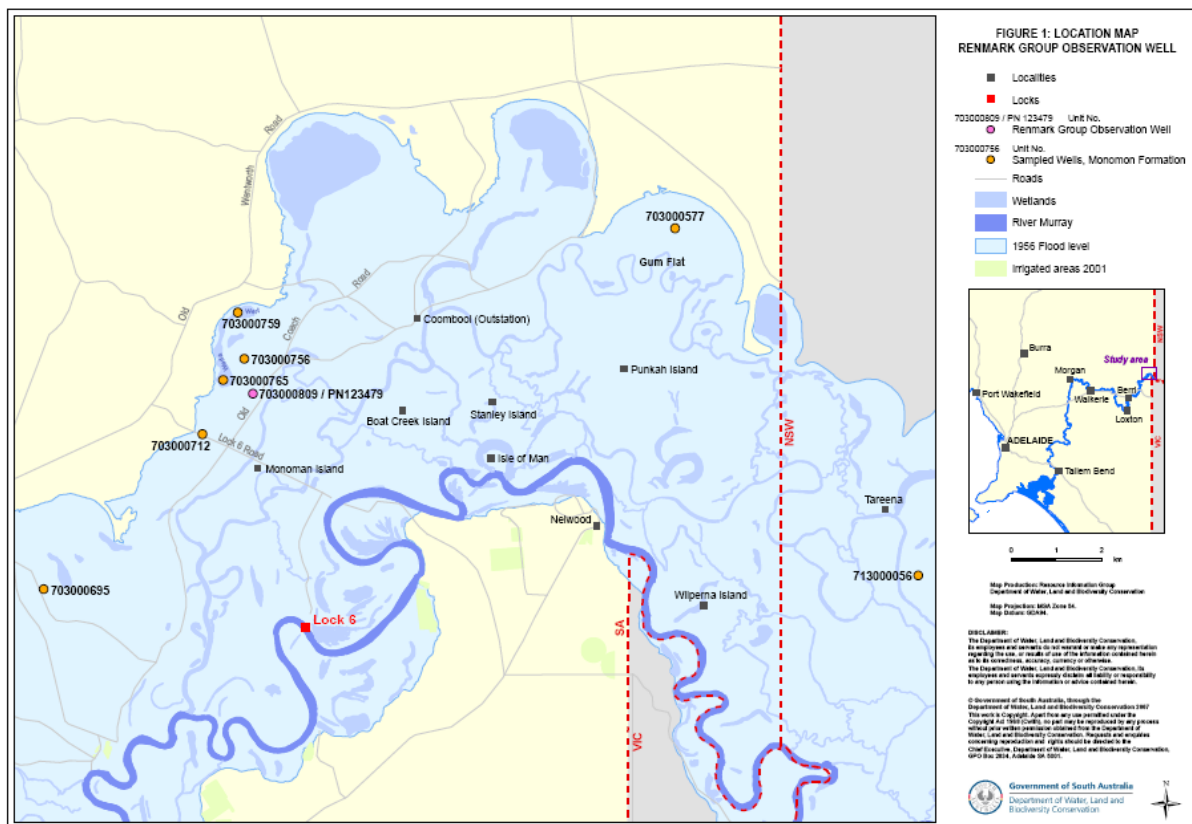


Figure 1. Location map of observation well 7030-809 (CH1) completed in the Renmark Group and wells 7030-712 (CH2), 7030-759 (CH3), 7030-765 (CH4) and 7030-756 (CH5) in the Monoman Sands. This assessment also used groundwater quality data from additional Monoman Sands observation wells 7030-577 (located at Gum Flat), 7030-695 and 7130-56. *Figure provided by DWLBC.*

Table 1. Groundwater quality data for the Renmark Group and the Monoman Sands

Aquifer Well name/description Unit No. Date Sampled	RG and MS observation wells sampled in 2007					MS obs. wells sampled in 2005		
	RG CH1	MS CH2	MS CH3	MS CH4	MS CH5	MS Gum Flat 7030-577	MS 7030-695	MS 7130-56
Temp (°C)	36.08	21.6	19.5	20.15	23.26	20.3	21	22
pH (-)	6.83	6.29	6.46	6.39	6.75	6.6		6.7
DO (sat %)	0.8	1.9	1	0.9	1.9			
ORP (mV) †	-270.5	-62.4	-122.9	-157.4	-85.8			
TDS (g/L)	21.74	33.89	45.99	33.64	32.72			
Electrical Conductivity (mSiemens/cm)	40.53	48.75	63.31	46.95	48.66	75	42.9	48.2
Turbidity (NTU)	1.3	10	41	27	140	6.2	14	17
Suspended solids (mg/L)	6	16	30	15	36	73	55	56
Volatile suspended solids (mg/L)	2	4	6	2	7			
Alkalinity as CaCO ₃ (mg/L)	351	242	263	284	551	200		
Calcium (mg/L)	529	596	627	576	492	469	630	501
Magnesium (mg/L)	578	1440	2040	1460	1270	2330	1560	1830
Potassium (mg/L)	75.1	161	213	145	148	213	177	121
Sodium (mg/L)	6530	11200	16200	11100	10700			
Bicarbonate (mg/L)	428	296	321	364	672	244	237	213
Chloride (mg/L)	4860	18100	26500	7600	7050	32900	20300	28200
Sulphate (mg/L)	2120	6300	9480	6390	5430	8950	6340	6570
Bromide (mg/L)	<0.1	52.5	<0.1	52.1	51.5			
Aluminium (mg/L)	<0.01	<0.01	<0.01	<0.01	0.012	0.021	0.518	0.178
Iron (mg/L)	0.271	1.4	2.97	2.06	9.88	3.61	8.15	5.23
Manganese (mg/L)	0.764	0.018	0.056	0.081	0.29	0.013	0.072	0.155
Manganese –soluble (mg/L)	0.569	0.016	0.032	0.056				
Phosphorus (mg/L)	0.169	0.084	0.198	0.078	0.66	0.071	0.066	0.413
TKN as N (mg/L)	4.61	2.44	1.1	2.29	1.83	0.83	1.49	0.5
Ammonia as N (mg/L)	4.338	0.11	0.069	0.095	<0.005	0.331	0.867	0.102
Nitrate + Nitrite as N (mg/L)	<0.005	<0.005	<0.005	<0.005	0.007	<0.005	<0.005	0.006
Nitrate as N (mg/L)	<0.005	<0.005	<0.005	<0.005	0.002			
Nitrite as N (mg/L)	<0.005	<0.005	<0.005	<0.005	<0.005			
Silica – Reactive as SiO ₂ (mg/L)	14	14	13	16	13	6	15	19
Total organic carbon (TOC) (mg/L)	2.1	0.5	1.2	0.7	2.5	0.5	0.7	0.7
Dissolved organic carbon (DOC) (mg/L)	2.3	0.5	1.1	0.7	2			
DOC – Biodegradable (mg/L)		<0.2	<0.2	<0.2	<0.2			
MFI (sec/L ²)						66.1		208.9
Aerobic colony count (20°C) (/mL)		1	210	3	34			
Aerobic colony count (35°C) (/mL)		5	220	1	13	210		
Heterotrophic Iron Bacteria (/mL)		<10	10	<10	<10	20	500	10
Radon-222 (Bq/L)	1.43±0.11	16.9±0.5	21.4±0.5	39.6±0.8	3.3±0.16			

† as measured in the field

Physical Clogging Assessment

The potential for the injection of MS water to lead to physical clogging can be assessed from examining the levels of indicator parameters such as MFI, total suspended solids (TSS), volatile suspended solids (VSS) and turbidity.

Turbidity, TSS and VSS data

Levels of particulate matter in the MS water, as characterized by the levels of turbidity, ranged from 10 to 140 NTU during the April 2007 sampling. This compares with values of 6 to 17 NTU in the previous (2005) sampling. TSS levels in the MS water ranged from 15 to 36 mg/L in 2007 and from 55 to 73 mg/L in 2005. The low VSS contribution to TSS indicates that only 10 to 30% of the particulate content is combustible at high temperature (ie. organic matter). By difference, the remaining 70 to 90% must be comprised of inorganic clay and silt-sized particles. The levels of all particulate indicators were lower in the deeper RG water than in the MS water.

MFI data

The Membrane Filtration Index (MFI) is an index of physical clogging potential that better accounts for the effect of particle size and composition than TSS or turbidity. MFI is a laboratory-based measure of the potential for physical clogging of a 0.45 μm membrane filter (Dillon *et al.*, 2001). The greater the retention of particles on the filter, the greater the MFI value (reported in units of sec/L^2) and hence also the rate of physical clogging. Note that chemically and biologically derived forms of clogging are not accounted for due to the nature and brevity of the test procedure.

MFI analyses conducted on two samples from the 2005 sampling were 66 and 209 sec/L^2 for turbidities of 6 and 17 NTU respectively. The estimated values for CH2-CH5 from 2007 sampling (using turbidity as the scaling factor) range from around 170 to 430 sec/L^2 .

The values are high as compared to the range of 3-5 sec/L^2 generally accepted as an upper limit by Dutch water utilities for injection into fine-textured dune sands (Olsthoorn, 1982). By contrast, values of MFI as high as 400 to 4200 sec/L^2 were estimated to have been successfully injected into a sandy limestone aquifer with significant secondary porosity in the near-well zone during four successive years with routine backwash redevelopment every 40 ML (Pavelic *et al.*, 2006). These two examples illustrate that: a) the clogging potential of any given source water is highly dependent on the hydraulic characteristics of the target formation; and b) there is a trade off between the source water quality and the extent of clogging and hence the degree of redevelopment needed to sustain injection rates.

Particle size data

The particle size distribution curve was analyzed for one sample of MS water (2005 sample with MFI value of 66 sec/L^2). The size distribution curve given in **Appendix B** shows that particle sizes vary by almost two orders of magnitude, with a median size in the order of 22-26 microns. The majority of the size fraction is within the range that would be anticipated to be easily taken-out within settling tanks (Wegelin, 1996).

Biological Clogging Assessment

Microbial growth, defined as the collective increase in the number of bacterial cells and the extracellular polymeric materials (slimes) that they secrete, occurs where sufficient organic and inorganic substrates are present in the source water. In their own right, bacteria occupy little space at concentrations typically found in source waters and aquifers and it is only when they are given the opportunity to attach and grow on surfaces that they can cause significant practical problems. Bacteria reproduce by fission, and their rate of replication with time is exponential where the availability of substrate is not limiting. Unlike physical clogging, which is rapid, microbial clogging develops over time-scales of days to weeks (Rinck-Pfeiffer *et al.*, 2000; Marsden, 2001).

In the absence of microbial inhibitors such as chemical disinfectants, the extent to which bioclogging of the well-screen and adjacent porous media can occur depends on the amount of organic matter or other key nutrients that are available to support bacterial metabolism.

The potential of the MS water to lead to microbial clogging can be determined by examining source water parameters including organic carbon, nitrogen and phosphorus. Levels of dissolved organic carbon (DOC) in the MS water were low and ranged from 0.5-2 mg/L. Levels of total nitrogen (1.1-2.4 mg/L) and total phosphorus (0.1-0.7 mg/L) were also low. Nutrient levels were higher in the ambient groundwater than the MS water in all cases apart from CH5 (Unit No. 7030-756). This well has the shallowest completion interval and may represent younger groundwater thereby limiting the time available for natural attenuation of reactive constituents.

Biodegradable organic carbon (BDOC) provides an indicator of the bio-availability of the organic carbon in water samples that may be analogous to microbial clogging of injection wells (Hijnen and van der Kooij, 1992; Pavelic *et al.*, 2007). BDOC concentrations in all MS wells were below detection (<0.2 mg/L). The data suggests that the MS groundwater is bio-stabilized, ie. offers little opportunity for growth of microbial slimes.

Microbially induced clogging may also occur due to the precipitation of iron hydroxides (Forward, 1994) or aluminium hydroxides (James-Smith *et al.*, 2005). Iron precipitates can occur if iron-bearing minerals such as pyrite present within the aquifer are oxidized by oxygen and nitrate. Levels of iron in the ambient groundwater were low and were not indicative of a potential issue, however higher levels are present in the source water. The presence of low numbers of filamentous and non-filamentous iron bacteria in some of the MS wells lend support that field trials would need to consider this as a possible issue. Clogging due to iron precipitation is further discussed below.

Geochemical Assessment

The potential for geochemical reactions resulting from the injection of saline groundwater from the Monoman Sands (MS) aquifer into the deeper Renmark Group (RG) formation on the Chowilla Floodplain was assessed using the PHREEQC code (Parkhurst and Appelo, 1999).

The assessment was based on groundwater samples collected from the RG well (CH1) and four nearby MS monitoring wells (CH2-5) (**Table 1**). Unfortunately the cutting samples collected from the RG well were contaminated by trace amounts of drilling mud and could not be used to indicate the nature of the reactive mineral phases in the storage zone.

The PHREEQC modelling was used to:

- Determine the mineral phases in equilibrium with the ambient groundwater from the RG and the source water from the MS
- Examine the potential for mineral precipitation or gas formation due to:
 - reaction between the source water and the mineral phases present in the storage zone using the EQUILIBRIUM PHASES subroutine
 - mixing between the injectant and the groundwater using the MIX subroutine
 - the warmer temperature in the storage zone
 - aeration of the source water during storage prior to injection using equilibration with oxygen and carbon dioxide at atmospheric partial pressures in the GAS PHASE subroutine
- Evaluate the potential for clay swelling due to changes in the surface site composition through ion exchange using the EQUILIBRIUM subroutine.

Electron activity (pe), calculated from Eh measurements, was used to describe the redox state of all solutions, according to $Eh = 0.059pe$ (Appelo and Postma, 1999). Eh (mV SHE) of samples was estimated by adding the theoretical value for the Zobell Solution to the reported oxidation-reduction potential (ORP) value (mV) and assumes the YSI556 Multiparameter ORP probe was operating within the acceptable error range. The sensitivity of modelling results to redox state was considered and is discussed later in the report. It is recommended that future ORP measurements included a calibration check with measurement of a calibration (Zobell) solution to allow reporting against the standard hydrogen electrode (SHE).

Mineral phases

The background water quality for the RG (based on one sample) is near equilibrium with respect to calcite, quartz, illite and montmorillonite; oversaturated with respect to dolomite, kaolinite and K-mica and undersaturated in K-feldspar, albite, iron hydroxides ($Fe(OH)_3$) and aluminium hydroxides ($Al(OH)_3$) (**Table 2**). Therefore, calcite, quartz, illite and montmorillonite appear to be the major mineral phases influencing groundwater quality in the proposed storage zone. The salinity of the source water is controlled by evapotranspiration during recharge processes.

Regionally, the Warina Formation of the RG consists of medium to coarse quartz sand with minor carbonaceous fine silt, clay and minor pebble conglomerates (Rammers *et al.*, 2005). While pyrite is also reported regionally (Rammers *et al.*, 2005), it was not calculated as an equilibrium phase due to the absence of sulphide in the input data, and not observed in the borehole log for RG well (CH1). The borehole log confirms the presence of quartz and clay but does not report any carbonaceous material in the Warina Formation.

Most samples of the source water from the MS indicate near equilibrium with quartz and albite; oversaturation with respect to illite, montmorillonite, kaolinite, K-mica and K-feldspar, and undersaturation in calcite, dolomite, $Fe(OH)_3$ and $Al(OH)_3$. The water quality sample from CH5 has higher alkalinity and iron concentrations than the other observation wells (CH2-CH4) indicating dissolution of calcite and $Fe(OH)_3$.

To test the sensitivity to uncertainty in redox estimates, mineral saturation index (SI) calculations were undertaken under varying redox states for two of the end-member samples (CH1 and CH2); from pe of -1.5 to 1.5 mV for CH1 (RG) and 0 to 4 mV for CH2 (MS). While the saturation index for $Fe(OH)_3$ was affected, this is not considered a major influence on groundwater quality locally. $Fe(OH)_3$ precipitation becomes important if the MS is exposed to aeration prior to injection and is discussed later in this report.

Table 2. Mineral saturation indices calculated from groundwater samples from the RG (CH1) and the MS (CH2-5)

Mineral Phase	Saturation Index				
	Groundwater (CH1)	Injectant (CH2)	Injectant (CH3)	Injectant (CH4)	Injectant (CH5)
Calcite	0.24	-0.70	-0.54	-0.58	0.05
Dolomite	0.99	-0.68	-0.25	-0.43	0.88
Quartz	0.25	0.50	0.55	0.58	0.43
Illite	-0.03	2.00	2.71	2.45	1.92
Ca-montmorillonite	0.24	2.55	3.05	2.97	2.12
Kaolinite	2.02	4.06	4.41	4.32	3.62
K-mica	5.58	8.19	8.97	8.62	7.99
K-feldspar	-0.76	0.35	0.87	0.66	0.44
Albite	-0.99	-0.13	0.43	0.21	-0.01
Fe(OH) ₃ (a)	-4.12	-1.63	-1.89	-2.70	0.22
Al(OH) ₃ (a)	-2.27	-1.61	-1.48	-1.56	-1.74

Aeration of the source water

If the groundwater from the MS were to be held within a balancing storage prior to injection, equilibration with the air would occur. As a result, the source water becomes oxygen-rich and the concentration of soluble iron decreases while the tendency for precipitation of Fe(OH)₃ increases (**Table 3**). Once the soluble Fe(II) present in the groundwater from the Monoman Sands aquifer is oxidised to Fe(III) it will precipitate as insoluble amorphous iron hydroxide. This precipitate should be removed prior to injection to minimise aquifer clogging. Removal can be achieved through natural settling and in-line filters prior to injection. Aeration also leads to loss of carbon dioxide through degassing, which marginally reduces the potential for calcite dissolution. Whilst calcite was detected in all aquifer samples in small amounts, this was in part due to contamination from drilling muds.

Mixing and temperature changes

Mixing between the two end-member waters (RG and MS) was examined for the groundwater samples from CH2 (MS) and CH1 (RG) (**Table 4**) and does not suggest any reaction processes that will cause clogging. The temperature increases as the portion of groundwater increases or residence times increase (due to reestablishment of the native geothermal gradient), which reduces the solubility of carbon dioxide slightly. The effect of temperature on the source water can be seen when the CH2 groundwater sample normally at 22°C is placed under the warmer temperature of the storage zone (36°C) and the aqueous carbon dioxide concentration is reduced (**Table 3**). However, the increased pressure during storage will more than compensate for the effect of temperature on solubility.

Table 3. Comparison of MS source water (CH2-CH5) before and after aeration and also with elevated temperatures typical in the storage zone (CH2)

	CH2		CH3		CH4		CH5		CH2 unaerated @36°C
	unaerated	aerated	unaerated	aerated	unaerated	aerated	unaerated	aerated	
pH	6.3	6.4	6.5	6.5	6.4	6.5	6.8	6.8	6.3
pe (mV)	2.9	14.7	1.9	14.7	1.4	14.7	2.5	14.2	2.9
Ca (mg/L)	619	619	664	664	592	592	505	505	619
Fe(II) (mg/L)	1.4	0	3.0	0	2.1	0	9.9	0	1.4
CO _{2(aq)} (mg/L)	121	99	80	68	114	96	95	85	103
TIC (mg/L)	93	87	88	84	101	95	161	155	88
SI _{Calcite}	-0.70	-0.63	-0.54	-0.49	-0.58	-0.51	0.05	0.08	-0.52
SI _{Quartz}	0.50	0.50	0.55	0.55	0.58	0.58	0.43	0.43	0.30
SI _{Fe(OH)₃ (a)}	-1.63	2.51	-1.89	3.09	-2.70	2.82	0.22	3.60	-1.31

Table 4. Mixing between the MS source water (CH2) and groundwater (CH1)

	100% CH2	75% CH2	50% CH2	25% CH2	0% CH2
	0% CH1	25% CH1	50% CH1	75% CH1	100% CH1
Temp (°C)	22	25	28	32	36
pH	6.3	6.4	6.5	6.6	6.8
pe (mV)	2.9	2.4	1.8	1.1	-0.9
Ca (mg/L)	619	599	578	558	537
Fe(II) (mg/L)	1.4	1.2	0.87	0.57	0.27
CO _{2(aq)} (mg/L)	121	104	87	71	51
TIC (mg/L)	93	94	96	97	99
SI _{Calcite}	-0.70	-0.50	-0.29	-0.06	0.24
SI _{Quartz}	0.50	0.44	0.37	0.31	0.24
SI _{Fe(OH)₃ (a)}	-1.63	-1.83	-2.07	-2.44	-4.12

Reaction with minerals in the storage zone

As the MS source water is undersaturated in calcite, injection is expected to lead to dissolution of any calcite present. Silicate weathering is much slower and clay minerals are more likely to influence water quality through ion exchange than mineral dissolution and precipitation (Appelo and Postma, 1999). Calcite dissolution is not considered to present a risk to this project as it causes minimal increases to calcium concentrations (**Table 5**), can act to alleviate clogging and would not be likely to lead to stability concerns given calcite is a negligible or minor constituent of the target aquifer.

Injection of the oxygen-rich source water into the deeper Renmark Formation will also oxidise any Fe(II) present, either in the groundwater or within reduced minerals such as pyrite. The end result will be the formation of iron hydroxide flocs near the point of injection and aquifer clogging. There is approximately 0.3 mg/L reduced iron in the RG groundwater available for reaction with oxygen. However if pyrite is present there is a far greater pool of reduced iron in the sediments, and reaction of aerated MS source water with pyrite releases an average of approximately 18 mg/L iron (**Table 5**), which will precipitate under oxic conditions.

Pyrite oxidation can also release trace species such as arsenic into the groundwater, which would be of concern if the groundwater were to be recovered as a drinking water supply.

Similar geochemical evaluations have been undertaken for ASR sites at Waruwi in the Northern Territory for injection of shallow groundwater into a deeper sandstone aquifer (Pavelic *et al.*, 2001), and Rosedale in Melbourne for injection of urban stormwater into a fractured rock aquifer (Pavelic *et al.*, 2006). Both sites had iron in the ambient groundwater from 0.2-0.8 mg/L (*c.f.* 0.27 mg/L in RG), pyrite confirmed within the storage zone and identified iron oxide precipitation as the predominant clogging concern.

Table 5. Calcium and iron concentrations when MS groundwater (unaerated and aerated) is allowed to reach equilibrium with calcite and pyrite

mg/L	CH2		CH3		CH4		CH5	
	unaerated	aerated	unaerated	aerated	unaerated	aerated	unaerated	aerated
Ca	527	532	532	539	497	504	393	399
Fe(II) †	1.4	19	3.1	20	2.1	20	10.6	28

† Fe(II) expected to precipitate as Fe(III) under oxic conditions

Ion exchange

Clay swelling occurs when the diffuse double layer around the clay particles extends and can be caused by increasing the proportion of monovalent cations (eg Na⁺) on the solid surface or by freshening the storage zone (Appelo and Postma, 1999), causing a deterioration in the hydraulic conductivity of the aquifer.

In this project, freshening is not expected as the MS source water (34-45 g/L TDS) is more saline than the groundwater in the RG (22 g/L TDS). The more saline MS injectant also has a higher sodium concentration than the RG groundwater.

In the absence of reliable measured data, the exchange site composition was estimated by assuming equilibrium with the groundwater (CH1). This indicated the exchange sites were dominated by sodium (49% based on charge), calcium (24% of charge) and magnesium (26% of charge), with a minor contribution from potassium. When this exchange composition reacts with the more saline MS source water, sodium from the source water displaces calcium from the exchange sites resulting in more calcium and less sodium in the MS water in storage (**Table 6**), which also alters the calcite saturation index slightly from -0.70 to -0.42.

Table 6. Cation concentrations in MS source water (CH2) before and after cation exchange

mg/L	Initial	After exchange
Na	11644	10789
Ca	619	1275
Mg	1496	1567
K	167	115

Following exchange there is a slight increase in sodium on the solid phase (4%) but this is not expected to lead to a clogging issue, especially if the content of clay minerals is low.

Management of Clogging

Upper limits on the levels of total suspended solids and turbidity in the MS injectant to minimize clogging risks are difficult to specify due to the dependence of physical properties of selected target aquifers. Whilst the hydraulic conductivity of the RG aquifer has yet to be determined by aquifer pump testing, preliminary estimates from the particle size distribution of the aquifer are suggestive high values, although it is recognised that the sampling method was preferentially biased towards the collection of coarser particles. Given the uncertainty in the actual particle size distribution in the aquifer, TSS values <10 mg/L should enable sustained operations that eliminates the need for an excessive degree of backwashing (Pérez-Paricio and Carrera, 1999).

Options for minimizing the risk of excessive clogging include:

- pre-treatment of the MS water to reduce particulate levels prior to injection
- implementing periodic redevelopment of the injection well when head buildup and/or injection rates reach unacceptable levels

Techniques to prevent clogging and to redevelop clogged wells are described in detail by Olsthoorn (1982), Driscoll, (1986); Cullimore (1993) and Pérez-Paricio and Carrera, (1999) and Segalen *et al*, (2005). Briefly, methods include hydraulic methods, such as pumping, surging and juttering; chemical methods such as chemical oxidants, such as adding chlorine and hydrogen peroxide, to reduce bioclogging and inactivate bacterial growth, and polyphosphates to reduce physical clogging by reducing the stability of retained particles, which enhances the efficiency of detachment.

The rate of clogging of injection and ASR wells in unconsolidated formations is highly dependent on the choice of drilling technique, the quality of the drilling, well design and completion, as well as redevelopment methods. For example, it is known that wells drilled with cable tool significantly outperform reverse circulation rotary; using biodegradable mud gives rise to less clogging than when bentonite-based mud is used; residual mud on or in the vicinity of the borehole wall severely limits recharge capacity; and completion with wire wrapped screens and natural gravel pack gives significantly higher performance than wells with slotted casing and emplaced gravel pack (Segalen *et al*, 2005).

If clogging is allowed to proceed then clogging layers can become compacted which limits the efficiency of the redevelopment. As a general 'rule of thumb' redevelopment should be initiated before a 20% reduction in injection rate is observed.

CONCLUSIONS

The untreated MS groundwater contains sufficient particulate matter to lead to some degree of physical clogging. Pre-treatment of source water using settling and/or filtration methods should be adequate. Note that during sustained pumping operations from MS wells it is likely that particulate levels would diverge from those measured during these investigations. Variations in the physical properties of the aquifer, the design and method of completion of the monitoring wells, the rate of pumping would affect particulate concentrations in the source water.

TSS values in the source water for injection of <10 mg/L should enable sustained operations that eliminates the need for an excessive degree of backwashing.

The nutrient status of the MS water is sufficiently low as to suggest a minimal risk of clogging from biofilm production.

Geochemical modelling shows that injection of cooler and more saline MS groundwater into the warmer RG aquifer will result in some degassing of carbon dioxide at higher temperature during injection, dissolution of calcite present in the storage zone and some ion exchange. The water quality changes resulting from calcite dissolution and ion exchange are not likely to inhibit injection rates.

Clay swelling may occur when sodium displaces calcium on exchange sites but this should not be a significant problem if clay minerals are a minor constituent of the aquifer. Similarly calcite dissolution would not be expected to lead to any instability concerns if present in small amounts. Due to the collection of cuttings rather than intact core samples, there is some uncertainty on the physico-chemical properties of the aquifer, and hence on the results from the geochemical modelling.

Effort should be taken to minimise degassing during injection as gas binding could result in clogging.

If the MS source water is to be retained within balancing/settling tanks prior to injection then care should be taken to prevent, or at least limit, exposure to atmospheric oxygen. Oxygenation would convert soluble Fe(II) in the source water to particulate Fe(III), which will need to be removed prior to injection by appropriate pre-treatment. In addition, adding oxygen to the storage zone will oxidise any Fe(II) present in the near well zone, leading to well clogging. This is expected to be manageable by regular backwashing of the injection well.

Any observed clogging should not be allowed to become too advanced before initiating well redevelopment.

ACKNOWLEDGEMENTS

The assistance of Kwadwo Osei-Bonsu, Stephen Howles and Zoe Marsden (DWLBC) with data acquisitions is gratefully acknowledged.

Mr Mark Raven of the mineralogical services group of CSIRO Land and Water, Adelaide and Mr Adrian Beech of analytical services group of CSIRO Land and Water, Adelaide are acknowledged for dutifully performing analyses on aquifer samples.

REFERENCES

APHA (1999). Standard methods for the examination of water and wastewater 20th edition. APHA, USA.

Appelo, C. A. J. and Postma, C. (1999). Geochemistry, groundwater and pollution. A.A. Balkema, Netherlands.

AWE (2005) Stockyard Plain Disposal Basin – Assessment of Renmark Group Injection Potential. Unpublished Report for MDBC.

Briceno, M.C. et al. (2003) Horizontal and deviated wells water disposal injection experiences in a Venezuelan heavy oil reservoir in the Orinoco belt and future injection practices. Society of Petroleum Engineers Annual Technical Conference and Exhibition, Denver, Colorado, USA, 5-8 Oct. 2003.

Brown C.M. and Stephenson, (1991) Geology of the Murray Basin, Southeastern Australia. Bureau of Mineral Resources, Geology and Geophysics. Bulletin 235.

Cullimore, D.R. (1993) Practical manual of groundwater microbiology. Lewis Publishers, USA.

Dillon, P., Pavelic, P., Massmann, G., Barry, K. and Correll, R. (2001) Enhancement of the membrane filtration index (MFI) method for determining the clogging potential of turbid urban stormwater and reclaimed water used for aquifer storage and recovery. *Desalination* 140(2):153-165.

Driscoll, F.G. (1986) Groundwater and Wells. *Published by Johnson Filtration Systems Inc.*, St. Paul, MI, USA.

Forth J. and Reed J. (1979) Noora Evaporation Basin Deep Aquifer Disposal Study. E&WS Report No. 79/8.

Forward P.D. (1994) Control of iron biofouling in submersible pumps in the Woolpunda Salt Interception Scheme in South Australia. *Proc. of Water Down Under 94*, The Institution of Engineers, Australia, Vol. 2, pp.169-174.

Hijnen, W.A.M. and van der Kooij, D., 1992. The effect of low concentrations of assimilable organic carbon (AOC) in water on biological clogging of sand beds. *Water Research* 26(7):963-972.

James-Smith, J., Shand, P., Hodgkin, T., Hill, T., Stadter, M. and Love, A. (2005) Aluminium hydroxide clogging of wells, Bookpurnong, South Australia: Conceptual model and Environmental risks. South Australia. Department of Water, Land and Biodiversity Conservation. DWLBC Report 2005/40.

Marsden, Z. (2001) Biochemical clogging processes during aquifer storage and recovery (ASR) with recycled water: laboratory experiment, South Australia. Flinders University of South Australia, School of Chemistry, Physics and Earth Sciences, Hons. Thesis.

Nicot, J-P. and Chowdhury, A.H. (2005) Disposal of brackish water concentrate into depleted oil and gas fields: a Texas study. *Desalination* 181:61-74.

Olsthoorn, T.N. (1982) Clogging of recharge wells, main subjects. KIWA-Communications 72, 150p.

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

Pavelic, P., Dillon, P.J., Barry, K.E., Vanderzalm, J.L., Correll, R.L. and Rinck-Pfeiffer, S.M. (2007) Water quality effects on clogging rates during reclaimed water ASR in a carbonate aquifer. *Journal of Hydrology* 334:1-16.

Pavelic, P., Barber, C., Dillon, P., Toze, S. and Yin Foo, D. (2001) Water banking trial at Waruwu, South Goulburn Island, NT. Stage 1 Report to Department of Land Planning and Environment, Northern Territory. Centre for Groundwater Studies Report No. 98.

Pavelic, P., Page, D., Vanderzalm, J., Barry, K., Dillon, P. and Parsons, S. (2006). Rosedale Golf Club ASR Trial: Preliminary Water Quality Assessment. Unpublished report to Steering Committee of the *Developing Aquifer Storage and Recovery Opportunities in Melbourne Project*.

Pérez-Paricio, A. and Carrera, J. (1999) Clogging Handbook. *EU Project on Artificial Recharge of Groundwater, Research Program on Environment and Climate*, Contract ENV-CT95-0071.

Raber, E., Thompson, R.E., Smith, F.H. (1981) Improving the injectability of high-salinity brines for disposal or water flooding operations. Society of Petroleum Engineers Annual Conference, San Antonio, Texas, USA, 4-7 Oct. 1981.

Rammers, N., Hill, T. and Yan, W. (2005). Regional disposal strategy – Renmark Group deep injection: Phase 1 Desktop study. South Australia Department of Water, Land and Biodiversity Conservation. DWLBC Report 2005/29.

Rinck-Pfeiffer, S.M., Ragusa, S.R., Sztajn bok, P. and Vandavelde, T. (2000) Interrelationships between biological, chemical and physical processes as an analog to clogging in Aquifer Storage and Recovery (ASR) wells. *Water Research* 34(7):2110-2118.

Segalen, A.-S., Pavelic, P. and Dillon, P.J. (2005) Review of drilling, completion and remediation methods for ASR wells in unconsolidated aquifers. CSIRO Land and Water Technical Report 04/05.

SKM (2003) Lake Tutchewop Complex Disposal Options: Scoping of Deep Injection Option. Unpublished report for Goulburn - Murray.

SKM (2004) Disposal of Saline Water into Deep Aquifers in the Kerang Lakes District Project Feasibility Report for Goulburn - Murray.

Villa, R. R. et al. (2004) Improved Efficiency of Solid Trap in Brine Injection Pipeline of Tongonan Geothermal Field, Leyte, Philippines. *Geothermal Resources Council Transactions*, Vol. 28, pp.573-577.

Wegelin, M. (1996) Surface water treatment by roughing filters. A design, construction and operation manual, Swiss Federal Institute for Environmental Science and Technology (EAWAG) and Department Water and Sanitation in Developing Countries (SANDEC).

APPENDIX A. MINERALOGICAL AND CHEMICAL DATA FOR CUTTING SAMPLES COLLECTED FROM THE RG INJECTION WELL (CH1)

Table A.1 Physico-chemistry data for the 6 sub-samples from well CH1 (Unit No. 7030-809), 1 sample of fine fraction from well development and 1 sample of drilling mud.

Sample #	Corrected Depth ^A m	Depth marked on samples ^B m	Percent <0.09mm ^E	EC	pH	pH	Cl	TC	TOC	CO ₃ as CaCO ₃	Exch. Cations					CEC
				(1:5 soil:water) dS/m		(0.01M CaCl ₂)	mg/kg	%	%	%	Ca	Mg	Na	K	Tot.	(NH ₄) cmol(+)/kg
1	406	410	0.7	0.83	8.9	8.1	591	1.7	1.6	1.3	1.7	1.7	2.4	0.24	6.0	6.2
2	418	422	0.1	0.58	9.6	8.6	336	0.76	0.5	2.4	1.0	0.75	1.5	0.25	3.5	2.4
3	422	426	0.3	0.71	9.6	8.5	430	1.1	0.8	2.2	1.2	0.85	1.6	0.29	4.0	2.9
4	432	436	1.5	0.60	9.6	8.6	350	0.65	0.4	2.1	1.0	0.62	1.1	0.21	2.9	1.9
5	444	448	3.3	0.80	9.3	8.3	440	2.2	1.9	2.2	2.2	1.9	3.0	0.45	7.5	6.5
6	456	460	2.5	0.60	9.3	8.5	399	1.0	0.9	0.9	1.8	1.3	1.8	0.27	5.1	4.6
7	Dev. ^C	NA	-	1.52	9.2	8.7	2040	0.38	0.2	1.6	0.9	0.70	0.94	0.14	2.7	2.1
8	Drill-Mud. ^D	NA	-	5.41	9.2	8.4	4810	3.8	1.5	19.9	4.5	0.69	29	1.1	35	26

^A 4 m difference between the geophysical logs and lithologs (geophysics is 4 m less than the litholog)

^B as marked on sample bags and from litholog (uncorrected for lag difference)

^C silt sample collected during well development 31/1/07

^D drilling mud

^E from PSD data provided by SA DWLBC (P. Magarey, pers. comm.)

Table A.2 Elemental data for the 6 sub-samples from well CH1 (Unit No. 7030-809), 1 sample of fine fraction from well development and 1 sample of drilling mud.

Sample #	Al	As	B	Ca	Cd	Co	Cr	Cu	Fe	K	Mg	Mn	Mo	Na	P	Pb	S	Se	Zn
	(mg/kg)																		
1	18300	<20	<20	5140	<20	<20	24	18	5170	3700	1680	80	<20	1090	70	<20	3150	<20	27
2	4150	<20	<20	9020	<20	<20	<20	21	3200	586	1040	76	<20	865	34	<20	3600	<20	7
3	6240	<20	<20	11400	<20	<20	<20	25	4500	845	1330	100	<20	1160	46	<20	4600	<20	10
4	2940	<20	<20	6550	<20	<20	<20	15	2400	423	748	53	<20	641	27	<20	2650	<20	8
5	13800	<20	<20	14800	<20	<20	24	34	6190	1590	2050	142	<20	1800	70	<20	5870	<20	30
6	13400	<20	<20	3960	<20	<20	<20	15	2460	1360	808	55	<20	729	44	<20	2340	<20	14
7	3190	<20	<20	8040	<20	<20	<20	22	4890	655	1260	106	<20	1560	64	<20	3680	<20	61
8	29000	<20	34	64300	<20	<20	30	127	21900	4240	7820	537	<20	10900	409	<20	6400	<20	47

Table A.3 Mineralogy data for the 6 sub-samples from well CH1 (Unit No. 7030-809), 1 sample of fine fraction from well development and 1 sample of drilling mud.

Sample #	Quartz	Barite	Calcite	Kaolin	Mica	Albite	Orthoclase	Smectite	Percentage analysed ^ж (<200µm)
1	68	1	1	16	13	-	1	-	17%
2	55	7	11	8	6	2	2	8	4%
3	36	9	18	11	10	2	4	9	3%
4	41	9	17	9	7	2	4	10	3%
5	53	5	8	15	9	2	2	5	7%
6	59	2	2	26	10	<1	<1	-	16%
7	70	4	5	6	5	3	7	-	15%
8	24	10	16	11	11	4	7	17	100%

^жThe quantitative analyses are reported on the <200µm fractions, with the exception of sample 8 which was analysed on the whole sample.

APPENDIX B. PARTICLE SIZE ANALYSIS OF MS GROUNDWATER

Analysis Report



Division of Minerals
Particle Analysis Service

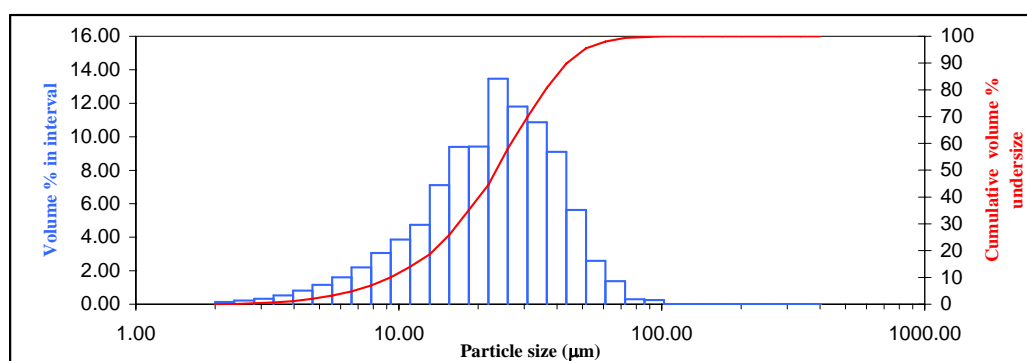
Client: Dept. Water, Land & Biodiversity Conservation
Sample: 1 Groundwater Sample
Batch No: R058829
PAS ID No: P47639

Analysis: Particle Counting by Hiac/Royco light extinction
Dispersant: Water
Additives: None

Sonication: None
Result units: Volume/Mass
Date: 31/08/07

Density: 2.65 g/cc (assumed value)
Calc. TSS : 3.64E+01 (mg/L, assuming spherical particles of homogeneous material)

Actual TSS: mg/L
Mass recovery: #DIV/0!



Min. size (µm)	Max. size (µm)	Volume % in interval	Min. size (µm)	Max. size (µm)	Volume % in interval	Min. size (µm)	Max. size (µm)	Volume % in interval
2.00	2.37	0.15	13.11	15.55	7.12	85.91	101.92	0.25
2.37	2.82	0.22	15.55	18.45	9.40	101.92	120.91	0.00
2.82	3.34	0.33	18.45	21.89	9.41	120.91	143.45	0.00
3.34	3.96	0.53	21.89	25.97	13.46	143.45	170.19	0.00
3.96	4.70	0.81	25.97	30.81	11.81	170.19	201.91	0.00
4.70	5.58	1.15	30.81	36.55	10.87	201.91	239.54	0.00
5.58	6.62	1.61	36.55	43.36	9.10	239.54	284.19	0.00
6.62	7.85	2.21	43.36	51.44	5.64	284.19	337.16	0.00
7.85	9.31	3.07	51.44	61.03	2.60	337.16	400.00	0.00
9.31	11.05	3.87	61.03	72.41	1.38			
11.05	13.11	4.74	72.41	85.91	0.30			

NOTE: This data is a calculated distribution based on the count distribution and the above assumptions.

Figure B.1 Particle size distribution of MS groundwater from well 7030-577.



Contact Us

Phone: 1300 363 400

+61 3 9545 2176

Email: enquiries@csiro.au

Web: www.csiro.au

Your CSIRO

Australia is founding its future on science and innovation. Its national science agency, CSIRO, is a powerhouse of ideas, technologies and skills for building prosperity, growth, health and sustainability. It serves governments, industries, business and communities across the nation.