

Analysis of variation in inorganic contaminant concentration and distribution in a landfill leachate plume: Astrolabe Park, Sydney, Australia

Author:

Jorstad, Lange B.

Publication Date:

2006

DOI:

<https://doi.org/10.26190/unsworks/23201>

License:

<https://creativecommons.org/licenses/by-nc-nd/3.0/au/>

Link to license to see what you are allowed to do with this resource.

Downloaded from <http://hdl.handle.net/1959.4/23427> in <https://unsworks.unsw.edu.au> on 2022-07-07

Analysis of variation in inorganic contaminant concentration and distribution in a landfill leachate plume: Astrolabe Park, Sydney, Australia

Lange B. Jorstad¹

A thesis submitted in fulfilment
of the requirements for the degree of
Doctor of Philosophy

UNSW Groundwater Group
School of Biological, Earth & Environmental Sciences
University of New South Wales
February 2006

¹ Copyright © Lange Jorstad, 2006

PLEASE TYPE

THE UNIVERSITY OF NEW SOUTH WALES
Thesis/Dissertation Sheet

Surname or Family name: Jorstad

First name: Lange

Other names: B.

Abbreviation for degree as given in the University Calendar: PhD

School: Biological, Earth and Environmental Sciences

Faculty: Science

Title: Analysis of variation in inorganic contaminant concentration and distribution in a landfill leachate plume: Astrolabe Park, Sydney, Australia

Abstract 350 words maximim: (PLEASE TYPE)

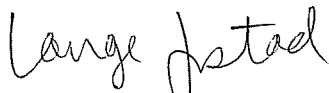
Spatial and temporal variation in inorganic contaminant concentration and distribution in a landfill leachate plume is examined to determine the mechanisms responsible for the observed variation, and to provide an assessment of the implications of this variation with respect to the interpretation of monitoring data, specifically with regards to its application to geochemical modelling. An integrated approach to field investigation was utilized, including soil and groundwater sample collection, geophysical investigation techniques, and measurement of vertical hydraulic head in the bundled piezometers. Nine groundwater sampling events were conducted over a 12 month period, with field measurement of water quality parameters and redox sensitive elements, and laboratory analysis for major and trace elements and stable isotopes. The vertical position of the centre of mass of the leachate plume shifted by up to two metres between monitoring events, and concentrations of key indicator parameters were observed to fluctuate by up to 160%. Surface resistivity images along a groundwater flow path between the landfill and a downgradient pond suggest discrete pulses of concentrated leachate migrating conservatively between the landfill and the pond. It is hypothesized that these leachate slugs are flushed into the aquifer during sustained periods of rainfall. The most significant hydrogeochemical processes in the leachate-impacted aquifer included microbial degradation of organic waste, dissolution of inorganic waste, ion exchange, precipitation of sulfide and carbonate minerals, mixing with rainfall recharge, and redox transformation along the plume fringe. These processes are supported by hydrogeochemical data analysis, and generally agree with the results of inverse geochemical modelling. While analysis of detailed groundwater monitoring appears to provide plausible descriptions of the plume dynamics, the results of the electrical resistivity transects indicates a more varied and complex plume configuration than is suggested by the borehole data alone. This integration of investigation techniques underscores the inherent inadequacy of even a high-resolution monitoring well network to accurately describe the full extent of variation in time and space within a contaminant plume, even in a relatively simple aquifer environment, and accentuates the potentially significant limitations of site-scale hydrogeochemical interpretation based solely on borehole monitoring data.

Declaration relating to disposition of project thesis/dissertation

I hereby grant to the University of New South Wales or its agents the right to archive and to make available my thesis or dissertation in whole or in part in the University libraries in all forms of media, now or here after known, subject to the provisions of the Copyright Act 1968. I retain all property rights, such as patent rights. I also retain the right to use in future works (such as articles or books) all or part of this thesis or dissertation.

I also authorise University Microfilms to use the 350 word abstract of my thesis in Dissertation Abstracts International (this is applicable to doctoral theses only).

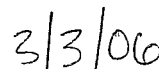
Signature:



Witness:



Date:



The University recognises that there may be exceptional circumstances requiring restrictions on copying or conditions on use. Requests for restriction for a period of up to 2 years must be made in writing to the Registrar. Requests for a longer period of restriction may be considered in exceptional circumstances if accompanied by a letter of support from the Supervisor or Head of School. Such requests must be submitted with the thesis/dissertation.

FOR OFFICE USE ONLY

Date of Completion of requirements for the Award:

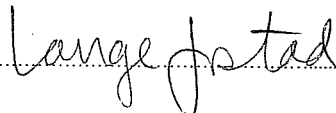
Registrar and Deputy Principal

THIS SHEET IS TO BE GLUED TO THE INSIDE FRONT COVER OF THE THESIS

ORIGINALITY STATEMENT

'I hereby declare that this submission is my own work and to the best of my knowledge it contains no materials previously published or written by another person, or substantial proportions of material which have been accepted for the award of any other degree or diploma at UNSW or any other educational institution, except where due acknowledgement is made in the thesis. Any contribution made to the research by others, with whom I have worked at UNSW or elsewhere, is explicitly acknowledged in the thesis. I also declare that the intellectual content of this thesis is the product of my own work, except to the extent that assistance from others in the project's design and conception or in style, presentation and linguistic expression is acknowledged.'

Signed

.....

Copyright and DAI Statement

'I hereby grant the University of New South Wales or its agents the right to archive and to make available my thesis or dissertation in whole or part in the University libraries in all forms of media, now or here after known, subject to the provisions of the Copyright Act 1968. I retain all proprietary rights, such as patent rights. I also retain the right to use in future works (such as articles or books) all or part of this thesis or dissertation.

I also authorise University Microfilms to use the 350 word abstract of my thesis in Dissertation Abstracts International (this is applicable to doctoral thesis only).

I have either used no substantial portions of copyright material in my thesis or I have obtained permission to use copyright material; where permission has not been granted I have applied/will apply for a partial restriction of the digital copy of my thesis or dissertation.'

Signature: Lange Jstad Date: 3/3/06

Authenticity Statement

'I certify that the Library deposit digital copy is a direct equivalent of the final officially approved version of my thesis. No emendation of content has occurred and if there are any minor variations in formatting, they are the result of the conversion to digital format.'

Signature: Lange Jstad Date: 3/3/06

To my beloved wife Julie,
who believed in me
through thick and thin.

To young Lachlan,
for showing us
the real meaning of life.

Abstract

Spatial and temporal variation in inorganic contaminant concentration and distribution in a landfill leachate plume is examined to determine the mechanisms responsible for the observed variation, and to provide an assessment of the implications of this variation with respect to the interpretation of monitoring data, specifically with regards to its application to geochemical modelling.

An integrated approach to field investigation was utilised in this study, including sample collection from a network of standard and bundled piezometers, surface and borehole geophysical investigation techniques, and a manometer board for the measurement of hydraulic head in bundled piezometers. Nine groundwater sampling events were conducted over a 12 month period, with sample analyses comprising field measurement of water quality parameters and redox sensitive elements, and laboratory analysis for major and trace elements and stable isotopes ($\delta^{18}\text{O}$, $\delta^2\text{H}$, $\delta^{13}\text{C-DIC}$, $\delta^{15}\text{N}$).

The vertical position of the centre of mass of the leachate plume was observed to vary up to 2 metres between monitoring events, and concentrations of key indicator parameters were observed to fluctuate by as much as 160%. The electrical images created by surface resistivity transects along a groundwater flow path between the landfill and a groundwater-fed pond a short distance downgradient suggest a plume configuration characterised by discrete pulses of concentrated leachate migrating in a conservative manner between the landfill and the pond. It is hypothesized that these leachate slugs are flushed into the aquifer during sustained periods of rainfall, presumed to be a significant driver of leachate mobilisation into the underlying aquifer.

The most significant hydrogeochemical processes affecting contaminant mobilisation, transport and attenuation in the leachate-impacted shallow aquifer included microbial degradation of organic waste, dissolution of inorganic waste, ion exchange, precipitation of sulfide and carbonate minerals, mixing with rainfall recharge along flow path, and redox transformations along the plume fringe. These processes are supported by hydrogeochemical data analysis, and generally agree with the results of inverse geochemical modelling.

While analysis of detailed groundwater monitoring appears to provide a plausible description of the plume dynamics, the results of the electrical resistivity transects indicates a more varied and complex plume configuration than is suggested by the borehole data alone. This integration of investigation techniques underscores the inherent inadequacy of even a high-resolution monitoring well network to accurately describe the full extent of variation in time and space within a contaminant plume, even in a relatively simple aquifer environment, and accentuates the potentially significant limitations of site-scale hydrogeochemical interpretation based solely on borehole monitoring data.

Acknowledgements

This thesis is the product of a significant amount of assistance, support, advice and encouragement, without which it would not be here. I would like to extend my deepest gratitude to the following individuals and organizations for their help with this project:

The Sydney Water Corporation, for the generous funding that made this research possible.

The Eastlakes Golf Course Greens Keepers, for their tolerance of my ongoing presence on the golf course (and the occasional provision of petrol for the generator).

Michael DeMol, for his help in surveying the elevations of my original monitoring well network.

Dorothy Yu, for her considerable skill in the analysis of my many water samples on the ICP-AES and ICP-MS machines.

The army of fellow students who assisted me with my fieldwork, selflessly risking life and limb in the face of errant golf balls. They are (in no particular order): Kelly-Jane Wallis, Ed Seeto, Wendy McLean, Karina Morgan, Bethany O'Shea, Sarah Groves, Jessica Northey, Kavita Gosavi and James Smith.

The crew at the UNSW Bookshop, who kept me afloat when the coffers were empty.

My parents, for their encouragement and support in this undertaking, and for understanding our desire to relocate ourselves to The Land Down Under.

Associate Professor Ian Acworth, for his considerable assistance with geophysical investigations, funding of the installation of shallow piezometers and surveying of the piezometers at the site, and insightful commentary and advice with regards to the development of this thesis.

Dr Jerzy Jankowski, my supervisor, for arranging the funding for this project, teaching me everything I know about aquatic chemistry, providing assistance with field and laboratory work, helping me make sense of massive spreadsheets full of data, and providing guidance and oversight for the development of this thesis.

*Empty your mind.
Be formless, shapeless, like water.
Now you put water in a cup, it becomes the cup,
You put water in a bottle, it becomes the bottle,
You put it in a teapot it becomes the teapot.
Water can flow or it can crash!
Be water my friend.*

Bruce Lee

Table of Contents

CHAPTER 1: INTRODUCTION	1
1.1 Landfill leachate in the environment.....	1
1.2 Evolution of landfill leachate research.....	2
1.3 Introduction to the research site.....	3
1.3.1 Previous investigations.....	5
1.4 Research objectives.....	6
 CHAPTER 2: REGIONAL ENVIRONMENTAL SETTING – BOTANY BASIN	 10
2.1 Location and description.....	10
2.2 Climate.....	13
2.2.1 Temperature and evaporation.....	14
2.2.2 Rainfall.....	14
2.2.3 Wind pattern.....	15
2.3 Surface topography and drainage.....	16
2.4 Geology.....	17
2.4.1 Bedrock geology.....	18
2.4.2 Quaternary stratigraphy.....	18
2.5 Hydrogeology of the Botany Basin.....	21
2.6 Hydrochemistry of the Botany Sands aquifer.....	23
2.7 Water contamination in the Botany Basin.....	28
 CHAPTER 3: LOCAL ENVIRONMENTAL SETTING – ASTROLABE PARK	 29
3.1 Location.....	29
3.2 Historical land use.....	30
3.2.1 Original environmental setting.....	30
3.2.2 Sand mining and landfilling activities.....	30
3.3 Local stratigraphy.....	36
3.3.1 David Phillips field and Astrolabe Park.....	36
3.3.1.1 Aeolian quartz sand.....	36
3.3.1.2 Waterloo Rock.....	37
3.3.1.3 Silt, clay and peat.....	37
3.3.2 Stratigraphic sequence.....	38
 CHAPTER 4: LANDFILLS AND LEACHATE	 42
4.1 Leachate production.....	42
4.1.1 landfill lifecycle.....	42
4.1.2 Physical processes.....	44
4.1.3 Chemical processes.....	45
4.1.4 Microbiological processes.....	45
4.2 Leachate composition.....	46
4.2.1 Major ions.....	47

4.2.1.1 Cations.....	47
4.2.1.2 Anions.....	49
4.2.2 Trace elements.....	50
4.2.3 Heavy metals.....	50
4.2.4 Dissolved organic matter.....	52
4.3 Redox processes and environments.....	52
4.3.1 Thermodynamic redox sequence.....	53
4.3.2 Concomitant redox reactions.....	55
4.3.3 Redox buffering.....	56
4.4 Carbonate system.....	57
4.4.1 Inorganic carbon - sources and sinks.....	58
4.4.2 pH buffering.....	59
4.5 Hydrogeochemical processes affecting leachate attenuation.....	60
4.5.1 Dilution.....	60
4.5.1.1 Mixing.....	60
4.5.1.2 Hydrodynamic Dispersion.....	60
4.5.2 Degradation.....	61
4.5.3 Mineral dissolution/precipitation.....	62
4.5.4 Complexation.....	62
4.5.5 Sorption reactions and element partitioning.....	63
4.5.5.1 Adsorption.....	63
4.5.5.2 Ion exchange.....	64
4.5.6 Redox transformation.....	65
4.6 Scales of variation in time and space.....	66

CHAPTER 5: METHODOLOGY

68

5.1 Piezometer construction and installation.....	68
5.1.1 Selection of piezometer installation locations.....	68
5.1.2 Design and construction of bundled piezometers.....	71
5.1.3 Design and construction of conventional piezometers.....	73
5.2 Hydrogeological investigation.....	76
5.2.1 Water level measurement.....	76
5.2.2 Manometer board.....	87
5.2.2.1 Standard piezometers.....	76
5.2.2.2 Bundled piezometers.....	76
5.2.3 Hydraulic conductivity – slug tests.....	80
5.3 Groundwater sample collection and analysis.....	81
5.3.1 Schedule and program design.....	81
5.3.2 General parameters and sample collection.....	83
5.3.3 Analysis of water samples.....	86
5.3.3.1 Field analyses.....	86
5.3.3.2 Laboratory analysis.....	88
5.3.3.3 Stable isotope sample collection and analysis.....	88
5.4 Soil sample collection and analysis.....	90
5.4.1 Sample locations.....	90
5.4.2 Soil sample collection.....	90
5.4.3 Soil sample preparation and analysis.....	91
5.5 Quality control.....	92
5.6 Geophysical investigations.....	94

5.6.1 Geophysical borehole logging.....	95
5.6.1.1 Bulk electrical conductivity.....	95
5.6.1.2 Gamma emission.....	96
5.6.2 Electrical resistivity methods.....	97
5.6.2.1 Surface electrical resistivity transects.....	97
5.6.2.2 Cross-borehole tomography.....	98
5.7 Data analysis methods.....	99

CHAPTER 6: HYDROGEOLOGY AND GEOPHYSICS **101**

6.1 Aquifer properties.....	101
6.1.1 Grain size analysis and porosity.....	104
6.1.2 Hydraulic conductivity and transmissivity.....	105
6.1.3 Storativity and specific yield.....	108
6.1.4 Hydraulic gradients and groundwater flow direction.....	109
6.1.4.1 Horizontal hydraulic gradients.....	109
6.1.4.2 Vertical hydraulic gradients.....	115
6.1.4.3 Hydraulic connection to Lachlan Pond No 5.....	117
6.1.5 Average linear groundwater flow velocity.....	120
6.2 Geophysical investigations.....	122
6.2.1 EM-39 borehole logging.....	122
6.2.1.1 Bulk electrical conductivity.....	123
6.2.1.2 Gamma emission.....	125
6.2.2 Electrical resistivity methods.....	127
6.2.2.1 Surface resistivity transects.....	129
6.2.2.2 Cross-borehole tomography.....	133

CHAPTER 7: SOIL CHEMISTRY **135**

7.1 Soil classification.....	135
7.2 Results of soil metals extracts.....	140
7.2.1 Piezometer installation boreholes.....	141
7.2.2 Shallow marsh cores.....	144
7.2.3 Quality control data.....	150
7.3 Comparison to regulatory criteria.....	150

CHAPTER 8: HYDROCHEMICAL CLASSIFICATION AND TREND ANALYSIS **154**

8.1 Chemical classification of water samples.....	154
8.1.1 Statistical analysis and classification.....	155
8.2 Uncontaminated regional groundwater.....	158
8.2.1 General parameters.....	159
8.2.2 Major elements.....	160
8.2.3 Trace elements.....	162
8.2.4 Redox sensitive elements.....	162
8.3 Landfill groundwater chemistry.....	163
8.3.1 General parameters.....	163
8.3.2 Major elements.....	166
8.3.3 Trace elements.....	167

8.3.4 Redox sensitive elements.....	167
8.4 Leachate plume chemistry.....	169
8.4.1 General parameters.....	169
8.4.2 Major elements.....	171
8.4.3 Trace elements.....	172
8.4.4 Redox sensitive elements.....	172
8.5 Marsh, pond and rainwater chemistry.....	173
8.5.1 General parameters.....	175
8.5.2 Major elements.....	175
8.5.3 Trace elements.....	176
8.5.4 Redox sensitive elements.....	176
8.6 Spatial and temporal chemical data trends.....	177
8.6.1 Vertical zonation of chemical constituents.....	177
8.6.2 Variation along flow path transects.....	186
8.6.3 Vertical dynamics of the leachate plume.....	189
8.6.3.1 Centre of mass.....	191
8.6.3.2 Maximum vertical extent.....	192
8.6.3.3 Chemistry variation.....	194

CHAPTER 9: HYDROGEOCHEMICAL PROCESSES

197

9.1 Dilution.....	197
9.1.1 Mixing.....	197
9.1.2 Dispersion.....	199
9.2 Redox reactions.....	201
9.2.1 Redox reactions in the landfill and leachate plume.....	201
9.2.2 Oxidation of pyrite in the peaty confining layer.....	204
9.2.3 Redox reactions at the upper margin of the plume.....	206
9.3 Dissolution/precipitation reactions.....	207
9.3.1 Dissolution of waste material and aquifer minerals.....	208
9.3.2 Mineral precipitation.....	212
9.4 Ion exchange.....	222
9.5 Environmental isotopes.....	226
9.5.1 Oxygen-18 and deuterium.....	229
9.5.2 Carbon-13.....	231
9.5.3 Nitrogen-15.....	236
9.6 Inverse hydrogeochemical modelling.....	238
9.6.1 Model selection.....	238
9.6.2 Mass balance modelling.....	239
9.6.2.1 Phases and constraints.....	240
9.6.3 Summary of modelling results.....	267

CHAPTER 10: CONCLUSIONS AND RECOMMENDATIONS

246

10.1 Conclusions.....	246
10.2 Recommendations.....	252

REFERENCES

253

APPENDICES

Appendix A – Hvorslev Analysis

Appendix B – Analytical Results Summary Tables

Appendix C – PHREEQC Saturation Indices Summary Tables

Appendix D – Research Publications

Tables

Table 2.1	Summary of chemical water types in the Botany Sands aquifer (Jankowski and Yu, 1998).....	26
Table 3.1	Chronology of sand mining and landfilling at Astrolabe Park and vicinity.....	35
Table 4.1	Concentration range of landfill leachate components (modified from Christensen <i>et al.</i> , 2001).....	48
Table 4.2	Prominent redox reactions that occur in landfill leachate plumes. Dissolved organic matter is represented by the generic form CH ₂ O (modified from Christensen <i>et al.</i> , 2001).....	56
Table 4.3	Carbonate species reactions, mass action equations, and approximate equilibrium constants at 25°C (modified from Appelo and Postma, 1999).....	58
Table 5.1	Construction details for bundled piezometers.....	72
Table 5.2	Construction details for conventional piezometers.....	75
Table 5.3	Schedule of full and reduced water sampling events.....	83
Table 5.4	Location and depth of piezometer borehole and marsh core soil samples.....	91
Table 6.1	Hydrogeological parameters determined from various regional studies.....	102
Table 6.2	Hydrogeological parameters in and around Astrolabe Park.....	104
Table 6.3	Hydraulic conductivity results from Hvorslev analysis of slug test data.....	107
Table 6.4	Vertical hydraulic head values from July 2002.....	111
Table 6.5	Water level data for the semi-confined portion of the aquifer below Astrolabe Park, Eastlakes Golf Course and Area A.....	116
Table 6.6	Water level data for the shallow, unconfined portion of the aquifer below Astrolabe Park and Eastlakes Golf Course.....	118

Table 7.1	Location and depth of piezometer borehole and marsh core soil samples.....	136
Table 7.2	Classification of soil samples from piezometer installation boreholes according to the Unified Soil Classification System (ASTM, 2003)...	136
Table 7.3	Lithological classification of the shallow marsh cores according to the Unified Soil Classification System (ASTM, 2003).....	137
Table 7.4	Metals concentrations in soil samples collected from the piezometer installation bores.....	142
Table 7.5	Metals concentrations in soil samples collected from the shallow marsh cores (mg/kg).....	145
Table 7.6	Quality control data for the soil analyses, expressed as the relative percent difference (RPD) ¹ between original and duplicate samples ($\pm 25\%$ is considered acceptable).....	150
Table 7.7	Summary of samples exceeding EILs for each metal.....	152
Table 8.1	Mean parameter values and element concentrations for water groups determined by hierarchal cluster analysis (concentrations in mg/L)...	156
Table 8.2	Statistical parameter values and element concentrations for uncontaminated groundwater from Astrolabe Park and David Phillips Field.....	159
Table 8.3	Statistical parameter values and element concentrations for landfill groundwater from Astrolabe Park (concentrations in mg/L unless otherwise noted).....	164
Table 8.4	Statistical data for parameter values and element concentrations for leachate - impacted groundwater (concentrations in mg/L; n = 259)...	170
Table 8.5	Mean parameter values and element concentrations for pond, marsh and rainwater samples (concentrations in mg/L).....	174
Table 9.1	Change in maximum chloride concentration (mM) along a groundwater flow path between upgradient (10, 30) and downgradient (3, 4A) piezometers.....	200
Table 9.2	Statistical parameters for saturation indices calculated using PHREEQC (n = 498).....	200

Table 9.3	Results of stable isotope analyses (including associated parameters used for interpretation of the results).....	227
Table 9.4	Summary of inverse geochemical modelling results along a central flow path between piezometers 10 and 3 for each major sampling event.....	242

Figures

Figure 1.1	Location of Astrolabe Park and the extent of the Botany Sands Aquifer.....	4
Figure 1.2	Site plan with locations of all previous piezometers, CPT locations and cross-sections.....	8
Figure 1.3	Site plan with locations of all previous geophysical investigations....	9
Figure 2.1	Location aerial extent and relationship between the Sydney and Botany Basins.....	11
Figure 2.2	Location, geology and aerial extent of the Botany Basin.....	13
Figure 2.3	Climate zones of Australia based on a modified Köppen classification system (Bureau of Meteorology, 2002).....	9
Figure 2.4	Long term temperature data for Sydney (Bureau of Meteorology).....	16
Figure 2.5	Long-term rainfall data for Sydney, recorded from 1929–2001 (Bureau of Meteorology, 2002).....	17
Figure 2.6	Quaternary stratigraphy of the Botany Basin (Albani, 1981).....	18
Figure 2.7	Detailed hydrochemical zonation of the Botany Sands aquifer (Jankowski and Yu, 1998).....	25
Figure 2.8	Detailed hydrochemical zonation of the Botany Sands aquifer (Jankowski and Yu, 1998).....	26
Figure 3.1	Astrolabe Park research site and environs.....	29
Figure 3.2	Aerial photograph of South Sydney and the study site in 1930, including the current boundary of Astrolabe Park (DLWC, 1930)....	31
Figure 3.3	History of sandmining and landfilling at Astrolabe Park and vicinity.....	34
Figure 3.4	Stratigraphic cross-section through David Phillips field (modified from Webb & Watson, 1979).....	40
Figure 3.5	Stratigraphic cross-section between Astrolabe Park and Lachlan Pond No. 5.....	41

Figure 4.1	Evolution of landfill leachate and gas over the lifespan of a landfill (modified from Christensen & Kjeldsen, 1989).....	43
Figure 4.2	Theoretical redox zonation along a flow path downgradient from a landfill, and associated distribution of redox-sensitive species (modified from Christensen <i>et al.</i> , 2001).....	54
Figure 5.1	Location of piezometers and pond sample points used in conjunction with this investigation.....	69
Figure 5.2	Location of mini-piezometers and soil samples in marsh.....	69
Figure 5.3	Photographs of a multi-level piezometer prior to installation; (a) view of sample point construction, and (b) lengthwise view of piezometer sections with sample tubes attached.....	74
Figure 5.4	Multi-level piezometer installation; (a) installing piezometer sections through hollow-stem augers, and (b) augers ready for retraction.....	75
Figure 5.5	Schematic diagram of the multi-channel manometer board.....	77
Figure 5.6	Photographs of the manometer board (a) equilibrating at piezometer 3, and (b) demonstration of the correlation between water levels for the PVC casing and adjacent sample point at piezometer 10.....	78
Figure 5.7	Purging sample point and measurement of general parameters.....	84
Figure 5.8	Field equipment for analysis of time-sensitive elements.....	85
Figure 6.1	Potentiometric maps for Astrolabe Park and Eastlakes Golf Course: July, September, December 2000 and January 2001.....	113
Figure 6.2	Potentiometric maps for Astrolabe Park and Eastlakes Golf Course: March, May, July and August 2001.....	114
Figure 6.3	Potentiometric maps for the shallow, unconfined aquifer at Astrolabe Park and Eastlakes Golf Course: November 2001 and February, April and July 2002.....	116
Figure 6.4	Vertical head and EC values for piezometers 3 and 10, July 2002...	119
Figure 6.5	Vertical hydraulic head distribution, based on manometer board measurements in July 2002.....	122
Figure 6.6	Borehole geophysical logs for piezometer 3.....	124

Figure 6.7	Borehole geophysical logs for piezometer 4.....	124
Figure 6.8	Borehole geophysical logs for piezometer 4A.....	126
Figure 6.9	Borehole geophysical logs for piezometer 6.....	126
Figure 6.10	Borehole geophysical logs for piezometer 4.....	128
Figure 6.11	Comparison of bulk and fluid EC for piezometer 10.....	128
Figure 6.12	Location of geophysical transects.....	129
Figure 6.13	Surface electrical resistivity transects; (A) Normal to groundwater flow, and (B) Parallel to groundwater flow.....	129
Figure 6.14	Conductivity section produced using cross-borehole tomography...	130
Figure 7.1	Location of shallow marsh cores.....	138
Figure 7.2	Lithological logs of the marsh soil cores.....	139
Figure 7.3	Metals concentrations in soil samples from landfill and pond shoreline piezometer installation.....	143
Figure 7.4	Maximum metals concentrations for each sample location, with interim urban Ecological Investigation Levels (EILs) provided, where available (NEPC, 1999).....	147
Figure 7.5	Metals concentrations at 10cm intervals in Marsh cores 1, 2 and 3.....	161
Figure 7.6	Metals concentrations at 10cm intervals for Marsh cores 4, 5 and 6.....	162
Figure 8.1	Piper diagram displaying the approximate plot areas for various water types present at the site.....	157
Figure 8.2	Site plan depicting piezometer locations used during this research	178
Figure 8.3	Longitudinal cross section depicting hydrogeology and water quality zonation in the subsurface between the landfill and the pond.....	180
Figure 8.4	Vertical distribution of general parameters for piezometers 3, 4 and 4A; July 2001 sample event.....	181

Figure 8.5	Vertical distribution of general parameters for piezometers 6, 10 (July 2001) and 30 (February 2002).....	181
Figure 8.6	Vertical distribution of major and trace elements for piezometers 3, 4 and 4A; July 2001.....	182
Figure 8.7	Vertical distribution of major and trace elements for piezometers 6, 10 (July 2001) and 30 (February 2002).....	183
Figure 8.8	Vertical distribution of redox sensitive elements for piezometers 3, 4 and 4A; July 2001.....	184
Figure 8.9	Vertical distribution of redox sensitive elements for piezometers 6, 10 (July 2001) and 30 (February 2002).....	185
Figure 8.10	Variation in the distribution of selected groundwater parameters and elements along a groundwater flow path between piezometers 10 and 3 (adapted from Jorstad <i>et al.</i> , 2004).....	187
Figure 8.11	Variation in the distribution of selected groundwater parameters and elements along a groundwater flow path between piezometers 30 and 4A (adapted from Jorstad <i>et al.</i> , 2004).....	188
Figure 8.12	Vertical profiles of electrical conductivity during all sampling events for piezometers 4 and 10.....	191
Figure 8.13	Elevation of plume centre of mass (A), maximum plume concentration expressed as fluid electrical conductivity (B), and average daily rainfall (C) in the vicinity of the study site for piezometers 3, 4, 4A, 6 and 10 for all regular sampling events.....	193
Figure 9.1	Redox sensitive elements vs. EC, indicating the distribution of redox species with increasing leachate impact (data grouped according to results of hierarchical cluster analysis; n = 498).....	202
Figure 9.2	Concentration of reduced iron and manganese with increasing leachate impact (n = 498).....	203
Figure 9.3	Relationship between DIC and alkalinity demonstrated in a “Deffeyes” plot, with indicators of trends produced by various redox reactions (“ideal” redox trends from Herczeg <i>et al</i> [1991])...	203
Figure 9.4	Relationship between A) reduced sulfur and reduced iron in groundwater; and B) reduced sulfur and pH in groundwater.....	205
Figure 9.5	Sulphate concentrations in shallow groundwater at piezometer 3...	206
Figure 9.6	Concentrations of major elements with increasing leachate impact (n=498).....	237

Figure 9.7	Major and trace elements associated with carbonate dissolution....	212
Figure 9.8	Saturation indices for iron oxide/hydroxide minerals versus the degree of leachate impact (represented by fluid EC values).....	214
Figure 9.9	Saturation indices for carbonate minerals versus the degree of leachate impact (represented by fluid EC values).....	217
Figure 9.10	Saturation indices for iron sulfide minerals versus the degree of leachate impact (represented by fluid EC values).....	218
Figure 9.11	Vertical profiles of saturation indices for iron sulfides, carbonates and oxide/hydroxide for piezometer 3, 4 and 4A.....	220
Figure 9.12	Vertical profiles of saturation indices for iron sulfides, carbonates and oxide/hydroxide for piezometer 6, 10 and 30.....	220
Figure 9.13	Relationship of (A) Na^+ and Cl^- in groundwater at the research site, and (B) the Na:Cl ratio relative to leachate impact (represented by fluid EC values); molar ratios for seawater and halite dissolution provided for reference (See Figure 8.18 for symbol description)....	223
Figure 9.14	Relationship of (A) Ca^{2+} to Na^+ in groundwater at the research site, and (B) the Ca:Na ratio relative to leachate impact (represented by fluid EC values; see Figure 8.18 for symbol description).....	224
Figure 9.15	Molar relationship between NH_4^+ and (A) $\text{Ca}^{2+} + \text{Mg}^{2+}$; (B) Na^+ ; and (C) K^+ in water samples at the site (See Figure 8.18 for symbol description).....	225
Figure 9.16	Cation exchange analysis for total groundwater samples.....	225
Figure 9.17	Bivariate plot of δD (‰ VSMOW) vs. $\delta^{18}\text{O}$ (‰ VSMOW) for groundwater, surface water and rainwater (Global meteoric water line [GMWL] provided for reference [Rozanski <i>et al.</i> , 199.....	230
Figure 9.18	Dissolved inorganic carbon (DIC) vs $\delta^{13}\text{C}$ for groundwater, marsh and pond water samples, depicting potential DIC sources based on the $\delta^{13}\text{C}$ signature (the low DIC and $\delta^{13}\text{C}$ samples are essentially unaffected by leachate impact).....	232
Figure 9.19	$\delta^{13}\text{C}$ of DIC vs the reciprocal of the DIC concentration. Note the steep regression line indicating ^{13}C enrichment associated with methanogenesis in the landfill, and the shallow regression line associated with oxidation of organic matter.....	233

Figure 9.20	Plot of δD values vs $\delta^{13}C$ values for all water samples. Note the unusually depleted δD signature of the leachate-impacted water samples.....	235
Figure 9.21	Relationship between $\delta^{15}N$ and total nitrogen concentration, with $\delta^{15}N$ enrichment due to denitrification indicated.....	237
Figure 9.22	Results of inverse geochemical modelling using PHREEQCv2.6, displaying the phase mass balance, exchange mass balance and electron mass balance for each sampling event (missing data points indicate that the given phase was not included in the modelling results for the sampling event).....	243
Figure 10.1	Conceptual site model of contaminant hydrology incorporating the results of various investigation techniques (note that plume fragmentation suggested by electrical resistivity transects not indicated in this figure).....	247

CHAPTER 1: INTRODUCTION

Landfilling of waste is an integral component of the solid waste management strategy for communities worldwide. While stringent legislation in most industrialised countries currently requires considerable measures to be undertaken to protect the environment from landfill emissions, there are numerous reported cases of old decommissioned (or abandoned) landfills, designed without engineered containment systems, that are acting as unregulated point-sources for the discharge of pollutants to the environment (e.g. Kimmel and Braids, 1974 and 1980; Zenone *et al.*, 1975; Baedecker and Back, 1979a,b; MacFarlane *et al.*, 1983; Baedecker and Apgar, 1984; Arneth *et al.*, 1989; Kehew and Passero, 1990; Lyngkilde and Christensen, 1992a,b; Mireki and Parks, 1994; van der Kamp *et al.*, 1994; Jankowski and Acworth, 1997; Basberg *et al.*, 1998; Kjeldsen *et al.*, 1998a,b; Cozzarelli *et al.*, 2000; van Breukelen *et al.*, 2003 and van Breukelen and Griffioen, 2004). Prior to the development of legislation governing the selection criteria for the siting of new landfills, it was common practice to dispose of municipal solid waste on property that was considered to be of low economic value, such as inundated swampland, or in abandoned sand mines and quarries with the intention of redeveloping the landscape. These practices have resulted in numerous cases of uncontrolled contaminant migration directly into the aquifer systems beneath the landfilled waste with minimal contaminant attenuation.

1.1 LANDFILL LEACHATE IN THE ENVIRONMENT

Landfill leachate can be loosely defined as a water-based solution of compounds derived from the waste (Christensen *et al.*, 1994, 2001). Leachate is generated when water infiltrates into the waste, either in the form of precipitation or a shallow groundwater table, instigating a combination of physical, chemical and microbial processes that act as mass transfer mechanisms between the waste and the infiltrating water (Andreottola and Cannas, 1992; Christensen *et al.*, 1994, 2001). While the composition of landfill leachate can vary significantly between different landfills, it is generally characterised by elevated concentrations of dissolved organic matter, major ions, and a variety of trace elements and synthetic organic contaminants (Chian and

DeWalle, 1977; Baedeker and Back, 1979a,b; Ehrig, 1983, 1988; Andreotolla and Cannas, 1992; Christensen, 1992; Christensen *et al.*, 2001).

Of the well documented landfill leachate plumes in published literature, evidence of landfill leachate impact is rarely observed beyond 1 km from the source landfill, suggesting that the aquifer environments have an intrinsic capacity to attenuate the leachate constituents. Additionally, whilst most of the researched landfills were established in the 1960's, none has demonstrated complete attenuation in the environment, attesting to the refractory, long-term nature of landfills as contaminant sources.

1.2 EVOLUTION OF LANDFILL LEACHATE RESEARCH

A significant body of published research has evolved over the past 30 years reflecting the growing awareness and concern with regards to the occurrence, composition and behaviour of landfill leachate in the environment. Early studies generally focused on the macroscopic behaviour of leachate plumes in groundwater, with description of the extent and composition of the plumes with respect to various hydrogeological settings (eg Kimmel and Braids, 1974 and 1980; Zenone *et al.*, 1975). The first significant research towards defining the hydrochemical zonation and governing hydrogeochemical processes associated with landfill leachate impacts was undertaken by Baedeker and Back (1979a,b), whose seminal publications introduced the concept of redox zonation within the plume by relating it to the analogous natural redox processes observed in marine sediments. Significantly, this was also some of the earliest research to utilise environmental isotopes to assist with the identification of hydrochemical processes occurring in the leachate plume. In 1983, researchers from the University of Waterloo published a landmark case study of a landfill leachate plume investigation, comprising a series of publications that documented the hydrogeology and hydrogeochemistry of a painstakingly researched leachate plume at a Defence site in Canada (including MacFarlane *et al.*, 1983, and Nicholson *et al.*, 1983). This study advanced the concept of integrated approaches to investigations in contaminant hydrogeology, and introduced the use of closely spaced vertical sampling to provide high-resolution profiles of contaminant distribution within a plume. The same year, Ehrig (1983) published the

first comprehensive assessment of the evolution of leachate quality over time. In the 1990's, a body of research from the Technical University of Denmark significantly advanced the understanding of contaminant transport and attenuation in landfill leachate plumes. This research was conducted at a pair of Danish landfills (Grinsted and Vejen), and focused on a range of topics, including: redox zonation and processes (Lyngkilde and Christensen, 1992 a,b; Bjerg *et al.*, 1995; Ludvigsen *et al.*, 1998, 1999; Jakobsen *et al.*, 1998), landfill source characterisation (Kjeldsen, 1993; Kjeldsen *et al.*, 1998a,b), leachate composition (Kjeldsen and Christensen, 2001; Kjeldsen *et al.*, 2002), influence of aquifer sediment geochemistry (Heron and Christensen, 1994 a,b, 1995; Heron *et al.*, 1998), dissolved organic carbon and metals mobilisation (Christensen *et al.*, 1996, 1998, 1999; Christensen and Christensen, 1999, 2000), and leachate attenuation (Christensen, 1992, Christensen *et al.*, 1994, 2001). The most comprehensive integrated hydrogeochemical study to date was published by van Breukelen *et al.* (2003) and van Breukelen and Griffioen (2004), which included detailed, high-resolution water sampling with a comprehensive analytical suite, and use of isotopes to identify the hydrogeochemical processes controlling contaminant transport and attenuation in the plume.

1.3 INTRODUCTION TO THE RESEARCH SITE

Astrolabe Park landfill is one of a number of old, unlined landfills in the Greater Sydney area that is locally degrading the quality of the groundwater resource. Prior to European redevelopment, the area that is now Astrolabe Park comprised swampland located at the confluence of the Lachlan stream draining the wetlands of Centennial Park, and Birds Gully stream draining a small catchment comprising present-day Kingsford, Kensington and Randwick. A portion of the land was used as a tip site for municipal waste from the late 1930's to mid-1940's. From the mid-1960's to mid-1970's the area was again subjected to the demands of urbanisation, with a program of sandmining and subsequent backfilling with waste on both the Astrolabe Park property and the south-adjacent 'Area A' property. Sandmining in these locations was advanced below the water table using a suction dredge, with the consequence that a significant portion of the waste disposed of in the pits is in permanent direct contact with shallow groundwater. Landfilling activities were terminated in 1975, when regulatory officials

observed leachate discharging into a pond located less than 100m down hydraulic gradient from the landfill. There was no engineered barrier installed in the floor of the sand mining pits prior to waste disposal, and the decommissioning process consisted of covering the waste with fine sand. As a result, the waste in Astrolabe Park and Area A has been in direct hydraulic contact with both shallow groundwater and rainfall recharge from the moment it was deposited.

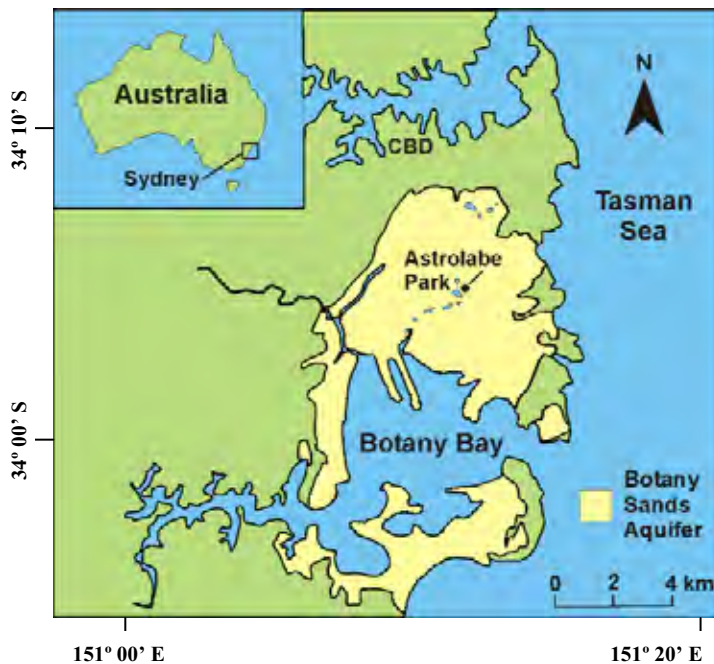


Figure 1.1 Location of Astrolabe Park, the Sydney Central Business District (CBD) and the extent of the Botany Sands Aquifer (Rickwood, 1998).

The aeolian quartz sand deposits that were targeted by the sand mining operations are part of a highly permeable regional aquifer system known as the Botany Sands Aquifer (Figure 1.1). This high yielding aquifer comprises a significant resource of generally good quality water in the immediate vicinity of Sydney and its local suburbs. Groundwater from the Botany Sands Aquifer is used for residential, irrigation and industrial purposes, and in its pristine state is generally considered potable without

treatment. However, since European settlement the aquifer has been subjected to the stresses associated with an industrialised urban centre. In a status report issued by the Department of Land and Water Conservation (DLWC, 2000), a variety of contaminant sources were identified as significant contributors to the degradation of water quality in the Botany Sands Aquifer. Amongst these were a number of industrial facilities in the Botany/Mascot/Banksmeadow area, fertiliser application to parks, golf courses and racetracks, and a few old, unlined landfills including Astrolabe Park. These contaminant sources were identified as the most critical issues to be resolved to enable the restoration and ongoing sustainability of the groundwater resource in the Botany Sands Aquifer.

It is worth noting that most previous international research of landfill leachate plumes has been undertaken in colder climates in the northern hemisphere, and as such the observed conditions at Astrolabe Park Landfill may provide some insight into potentially significant variations in the rate of waste degradation and leachate production over the lifespan of the landfill attributable to the warmer climate.

1.3.1 Previous investigations

Following decommissioning in 1975, the issue of groundwater contamination at Astrolabe Park was not addressed until 1990, when the first of a series of investigations was conducted to characterise the nature and extent of contamination associated with the old landfill. The first investigations into the leachate contamination at the site were conducted jointly by Jankowski and Knight (1991) and Hitchcock (1991). These initial investigations involved the integration of a monitoring well network (including several nested piezometers), cone penetrometer tests (CPTs), and various geophysical techniques to attempt to establish the water table depth, direction of groundwater flow, and the vertical and lateral extent of groundwater contamination (Figures 1.2 and 1.3). These investigations confirmed several fundamental conditions at the site, including:

- Groundwater generally flows in a westerly direction from Astrolabe Park towards Lachlan Pond No.5;
- Hitchcock (1991) performed a pump test at nearby David Phillips field to help determine the hydrogeologic properties of the aquifer;
- CPTs and geophysical logs identified the presence of a series of thin horizons of low permeability material (silt/clay/peat) between approximately 8 and 12 mbgs between the landfill and the pond, in an otherwise sandy profile;
- A leachate plume was issuing from Astrolabe Park and Area A, and had migrated towards Lachlan Pond No.5. An interesting finding from the investigation was that the leachate plume downgradient from Astrolabe Park was constrained to the shallow aquifer (approximately the upper 8 metres), but occurred at a greater depth downgradient from Area A; and
- The water in Lachlan Pond No.5 was slightly contaminated along the eastern shore of the pond, downgradient from Astrolabe Park, suggesting that at least a portion of the leachate plume was discharging into the pond.

Research by Smolski (1994) included installation of five more nested piezometers and analysis of groundwater samples to further refine the understanding of the spatial distribution and chemical characterisation of the leachate plume (Figure 1.2). Research by Acworth *et al.* (1994) in the same year employed borehole geophysical logging and surface resistivity transects to delineate the vertical extent of the leachate plume in several locations downgradient from Astrolabe Park and Area A (Figure 1.3). A number of additional sampling events have also been undertaken at the site in conjunction with undergraduate courses or projects. These research projects advanced the understanding of the spatial distribution of the leachate plume and the processes associated with contaminant transport, yet they comprised ‘snapshots’ of the plume in space and time. Little was known of the dynamics of contaminant transport on a short time scale, and the precise nature of the hydraulic connection between shallow groundwater and Lachlan Pond No.5 remained elusive.

In 1995 a bundled piezometer was installed at the eastern boundary of Astrolabe Park with sample points staggered at 1- to 2-metre vertical increments through the full thickness of the aquifer (piezometer 10, Figure 1.2). The bundled piezometer was sampled in 1995 and 1996, with concomitant geophysical logging, with the results published by Jankowski and Acworth (1997). Significantly, this study provided a detailed vertical profile of the leachate plume and demonstrated the variable nature of contaminant distribution within the plume over time.

1.4 RESEARCH OBJECTIVES

The resolution afforded by the bundled piezometer raised the possibility that solute transport within the plume was more complex and dynamic than previously assumed. Building on the dataset of the previous investigations, and utilising what remained of the monitoring infrastructure, an opportunity was provided to undertake a detailed study of the dynamic nature of contaminant transport within the leachate plume. Previous published research into the nature of landfill leachate plume development, transport and attenuation has focused primarily on redox zonation and contaminant attenuation along a groundwater flow path. The assessments have typically been based on one or two

‘snapshots’ of the leachate plume distribution, collected over lengthy time intervals. While considerable progress has been made towards a comprehensive understanding of the physical, chemical and (micro)biological processes governing the behaviour of contaminants within a landfill leachate plume, much less attention has been paid to the degree of spatial and temporal variation that occurs within this complex contaminated environment. The ramifications are potentially significant, as the underlying assumption in the detailed case studies in the published literature is that the systems can be described as steady-state over short time intervals, during which the chemical composition of the landfill leachate plume is evolutionary along a groundwater flow path, and that the variation along flow path can be attributed to a fixed set of processes. However, the timescale over which these assumptions are valid is unknown, as is the scale of spatial variation that may be present between the fixed monitoring locations.

Utilising a high resolution groundwater monitoring network comprised of bundled piezometers, in conjunction with various surficial and borehole geophysical investigation techniques and a novel new tool for measurement of hydraulic head in bundled piezometers, this study was undertaken to provide insight into the dynamic nature of contaminant transport within the complex environment of a landfill leachate plume with a relatively short flow path. The desired results of this research are:

1. To provide an enhanced understanding of the variability associated with contaminant mobilisation, transport and attenuation processes in a landfill leachate plume over short time intervals,
2. To attempt to identify the primary factors responsible for the observed variation and dynamics, and
3. To provide an assessment of the potential implications with regards to the interpretation of monitoring data and its application to geochemical models.

An associated outcome of this research was a detailed characterisation of the nature and extent of the leachate plume downgradient from Astrolabe Park, and its hydraulic relationship with Lachlan Pond No.5.

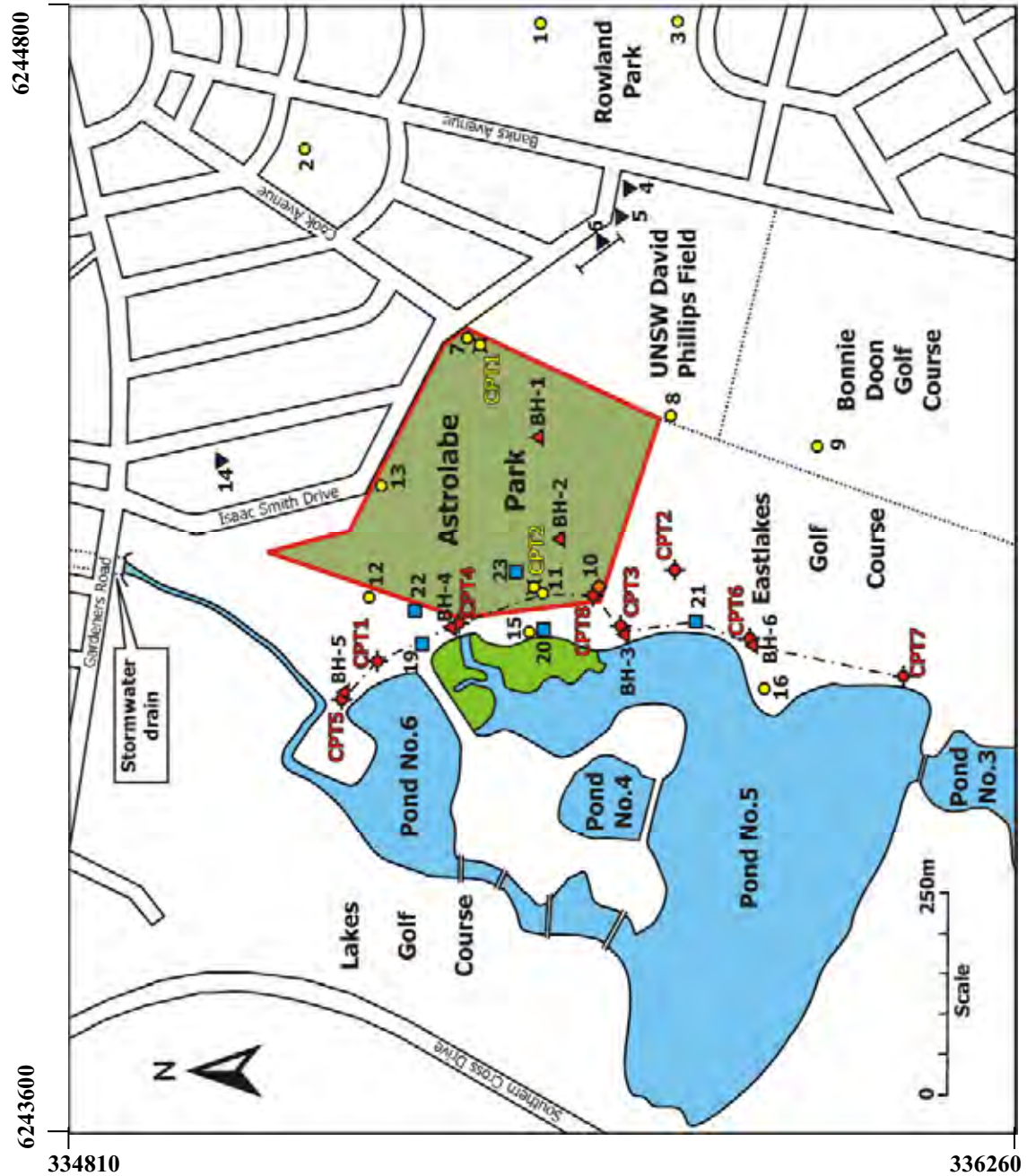


Figure 1.2 Site plan with locations of all previous piezometers, cone penetrometer test (CPT) locations and cross-sections (note that the ponds are all part of the Lachlan Ponds chain).

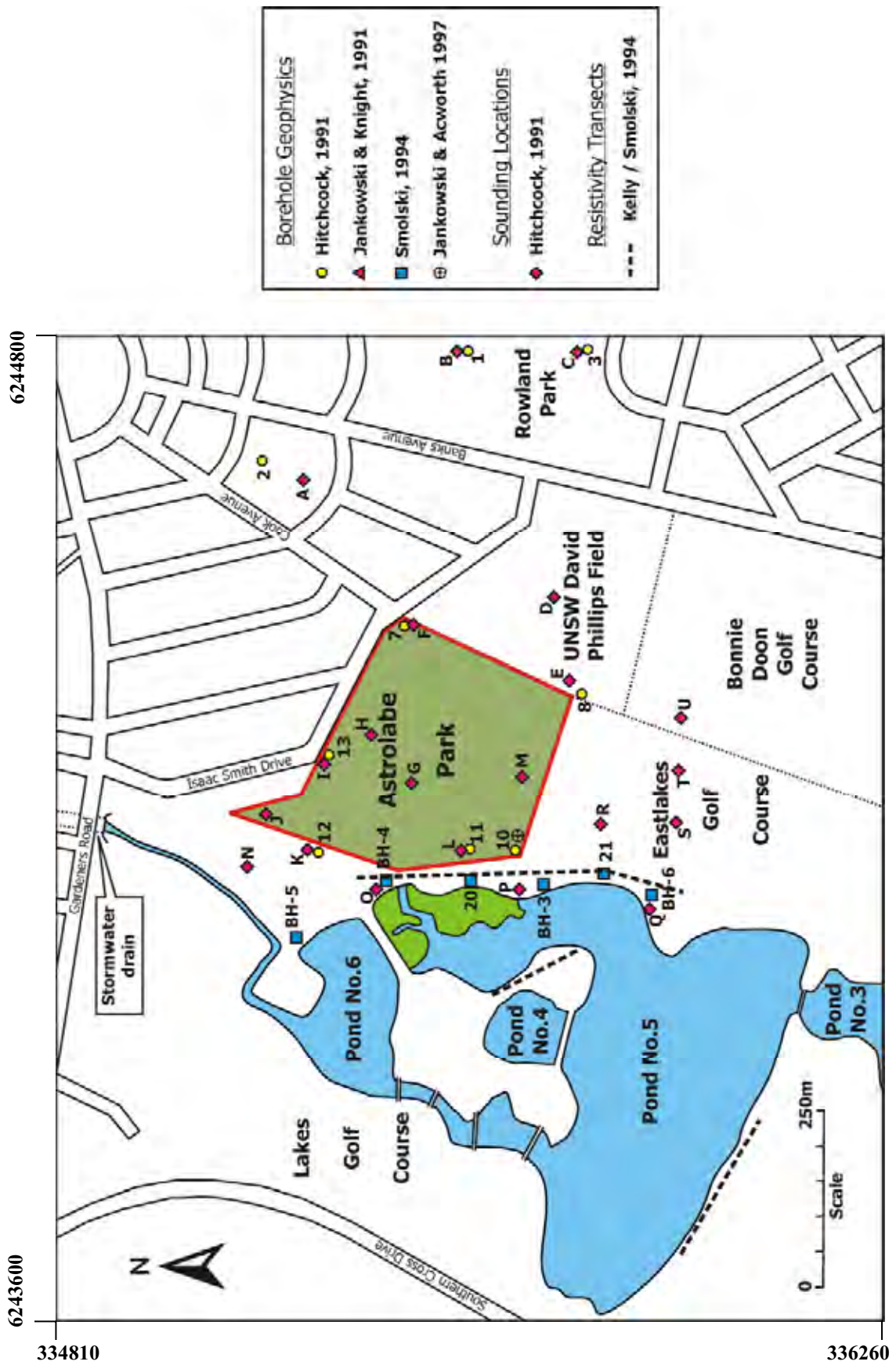


Figure 1.3 Site plan with locations of all previous geophysical investigations (note that the ponds are all part of the Lachlan Ponds chain).

CHAPTER 2: REGIONAL ENVIRONMENTAL SETTING – BOTANY BASIN

2.1 LOCATION AND DESCRIPTION

The study site is located along the eastern seaboard of New South Wales within a geological feature known as the Botany Basin, which in turn is located within a major regional sedimentary basin known as the Sydney Basin (Figure 2.1). The Botany Basin consists of an erosional depression in the Tertiary Hawkesbury Sandstone formation and Wianamatta Group bedrock (Ashfield Shale formation) that is filled with Pleistocene to recent sediments (Figure 2.2). Rickwood (1998) has defined its boundaries as following the crests of present-day outcropping bedrock ridges that surround the ancient buried drainage system, extending from Centennial Park in the north to Port Hacking in the south, west to Tempe, and just inland from the coastline to the east. It has an approximate aerial extent of 107 km² (Johnson, 1981). The stratigraphic units of the Botany Basin rest unconformably upon the Permo-Triassic sedimentary units of the Sydney Basin (Johnson, 1981).

The unconsolidated sediments that fill the Botany Basin comprise the Botany Sands aquifer (Griffin, 1963). The sediments range in thickness from a couple metres at the northern boundary of the Botany Basin to approximately 60 metres close to Botany Bay (Griffin, 1963; Acworth, 1998). The aquifer is recharged in the northern portion of the Botany Basin and discharges into Botany Bay (Acworth and Jankowski, 1993).

The Astrolabe Park landfill is located in the centre of the northern portion of the Botany Basin, approximately 6 kilometres south of the Sydney central business district (CBD).

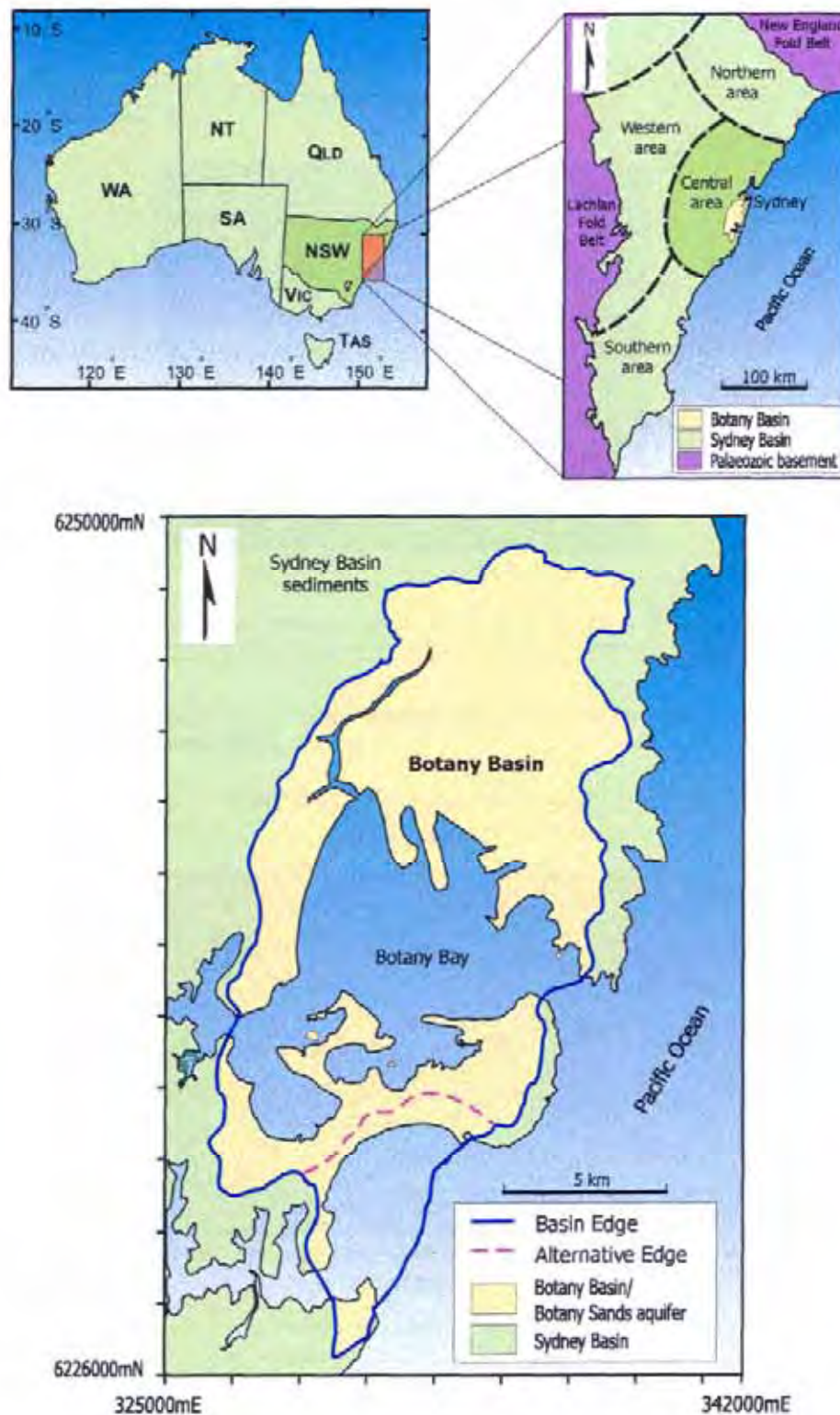


Figure 2.1 Location, aerial extent and relationship between the Sydney and Botany Basins.

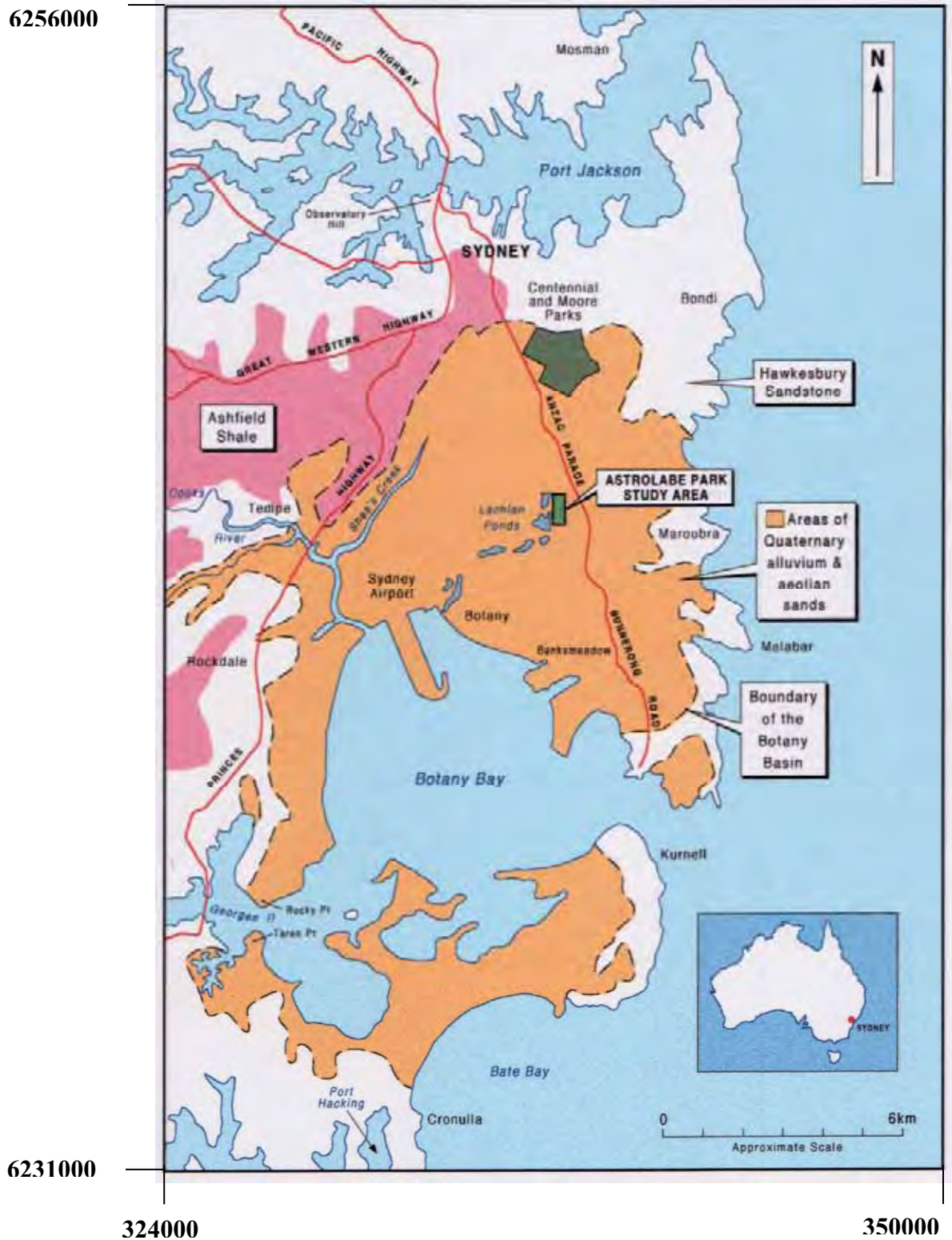


Figure 2.2 Location, geology and aerial extent of the Botany Basin.

2.2 CLIMATE

The Bureau of Meteorology collects and stores climate data from a network of gauging stations across Australia. This data provides a record of the general climatic environments encountered throughout Australia, including much information that is helpful in the interpretation of the general environmental setting of the Botany Basin. The following climate descriptions for the Sydney/Botany Basin region are taken from the Bureau of Meteorology website (www.bom.gov.au), and primarily from gauging station 66037 located at Sydney Airport (but also including gauging stations 66073, 66052 and 66160, located in Randwick and Centennial Park, to assess spatial variability in rainfall). Data collected from this gauging station can be considered to be a close approximation of the climate for the Botany Basin as a whole.

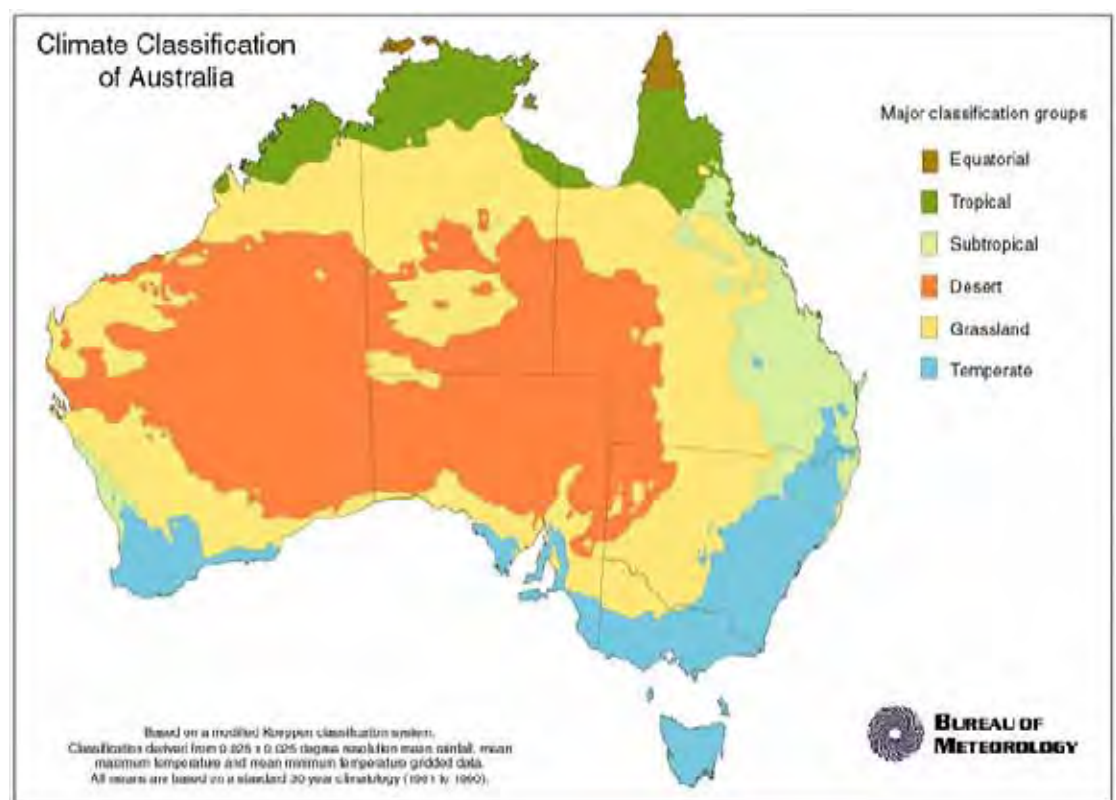


Figure 2.3 Climate zones of Australia based on a modified Köppen classification system (Bureau of Meteorology, 2002).

2.2.1 Temperature and Evaporation

The Botany Basin falls within a temperate climate zone characterised by warm summers and cool winters, and on average is free of extreme heat and cold (Figure 2.3). Data collected from gauging station 66037 from 1929 to 2001 (extending through the research period) indicate that the mean maximum and minimum temperature range falls between 18.9 to 26.3°C in summer (February) and 6.9 to 16.9°C in winter (July) (Figure 2.4). Mean daily evaporation data indicates a monthly mean evaporation range of 77 to 229 mm/month (June and December, respectively). The greatest evaporation occurs from October to February, and a sharp decline is observed from May to July.

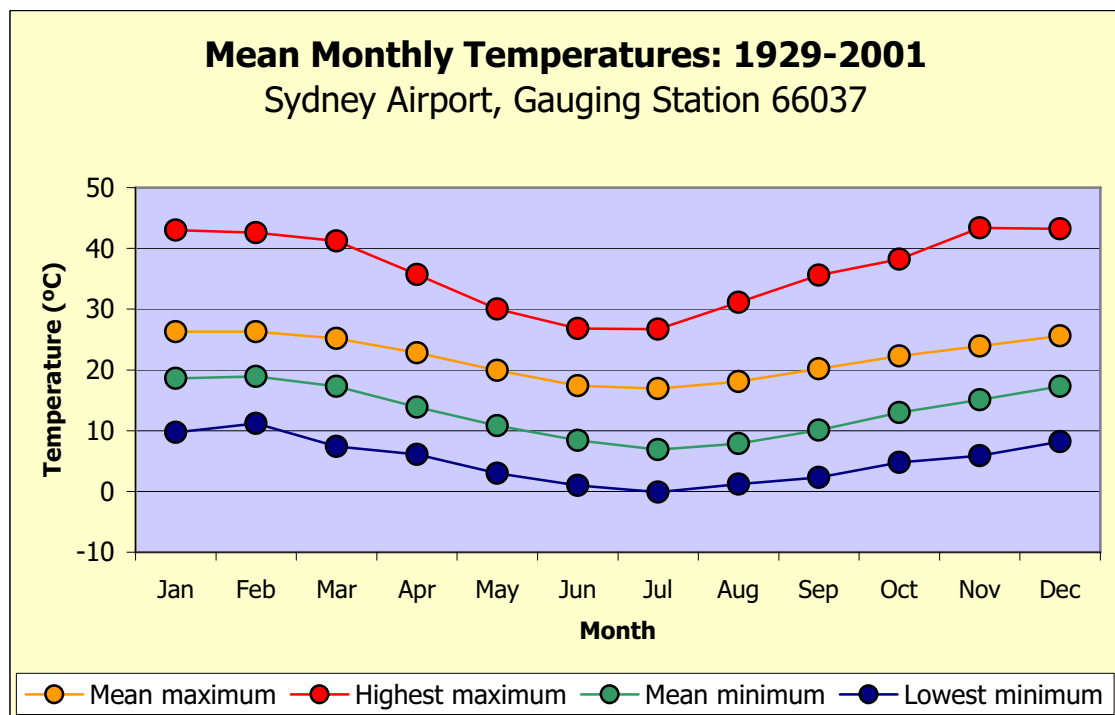


Figure 2.4 Long-term temperature data for Sydney, recorded from 1929-2001 (Bureau of Meteorology, 2002).

2.2.2 Rainfall

Sydney and the Botany Basin have a temperate to subtropical rainfall pattern, without a distinct dry season. The total precipitation varies across the basin depending on the local topography. As presented in Figure 2.5, the greatest amount of rainfall typically occurs in summer and autumn, with winter and spring being relatively drier. The number of rain days is fairly consistent from month to month, indicating that the intensity of the storm events is greatest in summer and autumn. June has the highest median monthly

rainfall at 100.4 mm; the lowest occurs in August at 44.6 mm. However, storm events can occur any time of year in Sydney; a maximum monthly rainfall total between 249.4 and 596.9 mm has been recorded for each month of the year at some point within the past 70 years of climate monitoring. An assessment of rainfall at three proximal gauging stations indicates that while minor spatial variations in rainfall do occur, the overall trends are sufficiently consistent that the data from station 66037 can be considered representative of conditions at the study site.

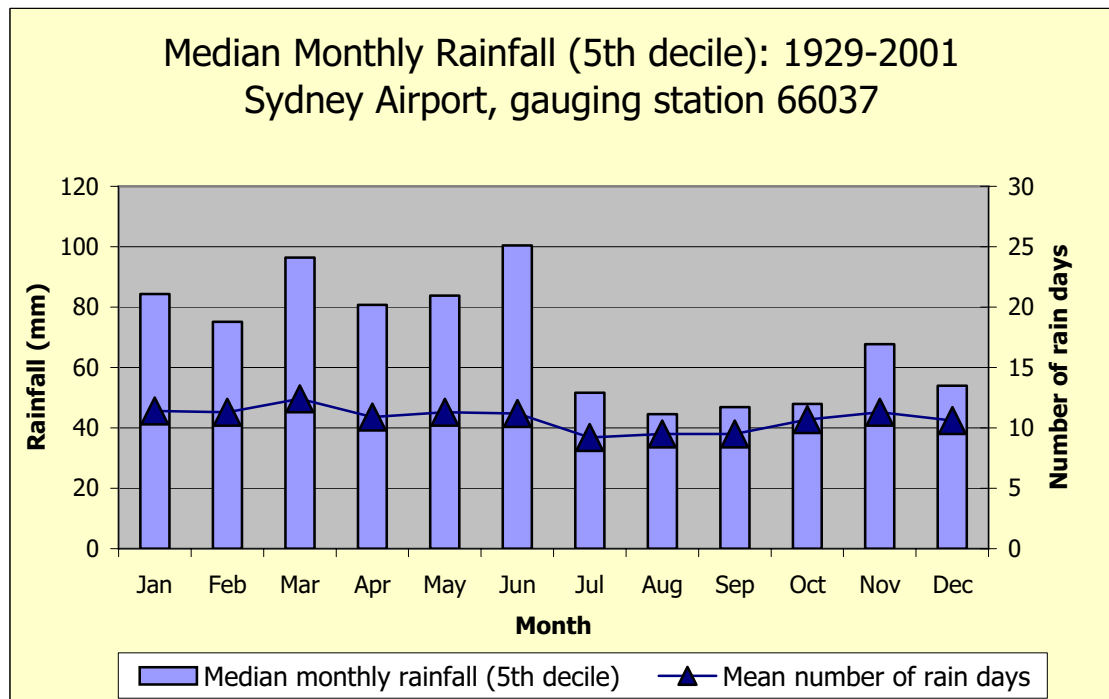


Figure 2.5 Long-term rainfall data for Sydney, recorded from 1929–2001 (Bureau of Meteorology, 2002).

2.2.3 Wind Patterns

Sydney and the Botany Basin experience a wind pattern typical of coastal environments, which is dictated by the relatively buffered temperature range of the ocean with respect to the land. In the morning, winds are predominantly from the west (inland) as the ground temperatures climb above ocean temperatures. In the afternoon, as the ground temperatures cool, ocean breezes blow inland from the east. As the temperature range is fairly mild in the Sydney area, there is little annual variation in wind patterns. However, storm systems associated with barometric pressure troughs are occasionally associated with strong winds from the south.

2.3 SURFACE TOPOGRAPHY AND DRAINAGE

The surface topography of the Botany Basin is that of a shallow depression between Port Jackson and Port Hacking. The northern, western and southern boundaries are defined by ridges of bedrock comprised of both Hawkesbury Sandstone and overlying shales from the Wianamatta Group. The elevation of the outcropping bedrock ridge in Centennial Park is approximately 40 m Australian Height Datum (AHD). Botany Bay lies in the centre of the basin; a shallow bay surrounded by beaches and associated belts in the south. Botany Bay is the ultimate discharge point for two major drainages: the Georges River in the southwest and the Cooks River in the west. However, the drainage areas of these rivers are not generally considered as being within the boundaries of the Botany Basin.

Although surface drainage in the northern portion of the basin is limited by the highly permeable nature of the surface sediments, there is a natural chain of swamps and ponds that begin in Centennial Park and ultimately discharge into Botany Bay. The ponds used to form a continuous watercourse between Centennial Park and Botany Bay, but now the ponds in Centennial Park are connected to the present day Lachlan Ponds chain by a stormwater drain. The Lachlan Ponds are currently comprised of 11 ponds separated by a series of dams and weirs. These ponds receive stormwater runoff from the surrounding suburbs, and have a direct hydraulic relationship to the groundwater system. Research by Johnson (1981) demonstrated that the water chemistry in the ponds in Centennial Park is nearly identical to that of groundwater adjacent to the ponds, suggesting that the ponds are a surface expression of the shallow groundwater system. Later studies by Knight *et al.* (1991) and Butler (1990) supported a slightly more complex surface water/groundwater relationship in the Lachlan Ponds system, asserting that some of the ponds both receive shallow groundwater discharge and contribute to groundwater recharge (flow-through ponds), while others only receive groundwater discharge without contributing recharge to the groundwater system. The chemical composition of the pond waters was described by Knight *et al.* (1991), which was reported to be similar to uncontaminated groundwater adjacent to the pond during dry periods and significantly diluted by rainfall and stormwater runoff during wet periods.

In a technical report to the Sydney Water Board, Jankowski and Knight (1991) grouped the Lachlan Ponds into three zones based on the hydrochemistry of the pond water and the relationship with the shallow groundwater system. Zone 1 (Ponds No 5 and 6) is characterised by a mixture of slightly polluted discharging groundwater and surface water entering from stormwater drains. Zone 2 (Pond Nos 3, 3A, 2, 1, 1A and the northern portion of Mill Pond) is characterised by a mixture of slightly polluted discharging regional groundwater and unpolluted local groundwater (from adjacent sand dunes). Zone 3 (southern portion of Mill Pond and Engine Ponds 1 and 2) is comprised of a mixture of slightly polluted regional groundwater discharge and heavily polluted local groundwater discharge attributed to heavy industrialisation in this area.

The Lachlan Ponds and associated wetlands currently represent one of the few remaining remnant ecosystems in the Botany Basin (Benson and Howell, 1990). Remnant ecosystems are comprised of natural habitats that have remained relatively unaltered since European colonisation. The health of these remnant ecosystems is considered to be a direct reflection of the environmental health of the Botany Basin as a whole.

2.4 GEOLOGY

The Botany Basin is set within a larger structural basin known as the Sydney Basin (Figure 2.1). The Sydney Basin is the main structural basin in New South Wales, and extends from Newcastle in the north to Batemans Bay in the south, and west to Lithgow (Herbert and Helby, 1980). The Sydney Basin formed as a result of a marine transgression at the end of the Late Palaeozoic glaciation, followed by a marine regression during the late Permian and Triassic (Herbert, 1979). During the Triassic, river-borne sediments were deposited as a deltaic sequence within a portion of the Sydney Basin (Johnson, 1981). These sediments comprise the Hawkesbury Sandstone, the predominant sedimentary unit beneath the Botany Basin.

2.4.1 Bedrock Geology

The Hawkesbury Sandstone Formation is the predominant bedrock unit beneath the Botany Basin, although the Ashfield Shale Formation, the lower-most unit of the Wianamatta Group, occurs beneath the north- and south-western edges of the basin (Roy, 1983). The Hawkesbury Sandstone was deposited during the Middle Triassic and is up to 290 m thick (Herbert, 1979). It consists primarily of quartz, feldspar and mica in a kaolinitic clay matrix and is cemented by secondary quartz and siderite (Standard, 1969). Sandstone beds are highly lenticular and strongly cross-bedded, however massive beds are present.

The Ashfield Shale is the lower-most unit of the Wianamatta Group, and was deposited conformably upon the Hawkesbury Sandstone as a result of later marine incursions and the development of brackish backswamps (Griffin, 1963). This unit consists of dark grey to black sideritic claystone-siltstone, grading upwards into a fine sandstone-siltstone laminite (Herbert and Helby, 1980).

A period of uplift in the Tertiary instigated the rejuvenation of many river systems, which incised deep, steep-sided valleys into the Triassic Bedrock (Albani, 1981; Griffin, 1963; Johnson *et al.*, 1977). One of the longest of these channels was the Lakes Valley, which begins in Centennial Park to the north and plunges south to a depth of ~80 m at Banksmeadow where it enters Botany Bay (Griffin, 1963; Jeffery, 1985). The present day Lachlan Ponds chain approximately follows the course of this buried valley. This erosional process also effectively removed the Wianamatta Group from within the boundaries of the present day Botany Basin, with the exception of the basal Ashfield Shale present at the north- and south-western margin (Figure 2.2).

2.4.2 Quaternary Stratigraphy

The Triassic bedrock units of the Botany Basin are unconformably overlain by Quaternary sediments, which represent a depositional sequence that includes marine, beach, dune and river facies (Albani, 1981; Roy, 1983). Albani (1981) reported that this deposition occurred in conjunction with a series of marine transgressions and

regressions, as evidenced by the presence of erosional surfaces that unconformably separate the various sedimentary units. These deposits generally thicken towards the centre of Botany Bay (Goodwin, 1971 and 1976), and range from a few metres thick along the margins of the basin to a maximum thickness of approximately 80 m close to Botany Bay (Griffin, 1963). Whilst the Botany Sands aquifer is comprised of all sand units described, the dune sands of Unit 3 comprise the most significant water bearing unit in the sequence, in terms of yield and water quality.

In general, the stratigraphy grades from marine and estuarine deposits at the base through brackish, freshwater, aeolian and finally recent marine deposits (Albani, 1981). This sedimentary sequence has been subdivided into several units, which have been identified by previous investigators (Albani, 1981; Evans, 1993; Griffin, 1963; Roy, 1983; Smart, 1974; Yu, 1993 and 1994). A description of a four-unit subdivision is provided based on Albani's comprehensive research (1981; Figure 2.6), and incorporates modifications from Beck and Jankowski (1998):

AGE	LITHOLOGY	UNIT
Holocene	Marine Sands	4
	Dune Sands	3
Upper Pleistocene	Fluvial Sands fresh to brackish water deposits	2
	Fluvial Clay fresh to brackish water deposits	
	Estuarine to Marine Sands	1
	Estuarine to Marine Clay	
	Estuarine to Marine Gravel	
Triassic	Bedrock	

Figure 2.6 Quaternary stratigraphy of the Botany Basin (Albani, 1981).

- *Unit 1*: The basal sedimentary unit, comprised of gravel, sand and clay deposited in a marine environment. The presence of benthic and pelagic foraminifera is characteristic of modern-day open tidal environments (Albani, 1981). The

sediments of this unit are found at the base of most of the incised bedrock channels.

- *Unit 2:* This unit is comprised primarily of estuarine clay and clayey sand deposits, with occasional organic sand and peat layers present. Concentrations of shell fragments and the lack of abundant benthic foraminifera suggest fresh to brackish depositional environment (Albani, 1981). This is analogous to modern-day lagoons, separated from the open ocean by sand barriers. Estimates of the thickness of this unit range between 5 and 15 m.
- *Unit 3:* This unit is characterised by aeolian deposits of well-rounded, medium quartz sand grains that were deposited in a dune environment (Albani, 1981; Smart, 1974), and forms the most significant water-bearing unit of the Botany Sands aquifer. Also present are shell debris of aeolian origin, as well as lenses of peat, silt and clay. Another unusual feature of this unit is the presence of iron-cemented horizons within the upper portion of the aeolian deposits, known as Waterloo Rock. Albani postulated that these horizons might represent ancient soil horizons that were temporarily established within the dune environment (as reported in Evans, 1993). The average thickness of this unit is 30 – 32 metres throughout the basin.
- *Unit 4:* This unit represents the most recent fluvial and marine deposits and is comprised primarily of sand, with silt, clay and peat present in calmer depositional environments (Albani, 1981). This unit ranges in thickness from a few metres to 20-30 m at the entrance to Botany Bay (Albani, 1998).

While the units in this classification scheme are generally representative of the Botany Basin as a whole, each unit is not necessarily present at all locations within the basin. In particular, Unit 4 is most evident in the vicinity of Botany Bay and current watercourses, and is often absent around the margins of the basin (Albani, 1981). This research was conducted in deposits of dune sands representative of Unit 3.

2.5 HYDROGEOLOGY OF THE BOTANY SANDS AQUIFER

Groundwater in the Botany Basin is stored and transported in both the bedrock units and the unconsolidated Quaternary sediments. In particular, the unconsolidated sediments constitute a significant reservoir for groundwater and are referred to in aggregate as the Botany Sands aquifer (Acworth and Jankowski, 1993). The general characteristics of these three geologic units with respect to the storage and transmission of groundwater are described below:

- Groundwater in the Hawkesbury Sandstone is stored and transmitted in joints and fractures within the rock, with little contribution from matrix porosity. Water from this unit has a relatively low salinity, due to the predominantly quartz composition of the sandstone, but bores installed in this unit are generally low yielding. Thus, this unit is generally considered to be an aquitard with respect to groundwater movement in the overlaying unconsolidated sediments.
- As an aquifer, the Ashfield Shale is characterised by low yields, limited storage and poor water quality (McNally and Branagan, 1998). The total dissolved solids load typically ranges from 10,000 – 12,000 mg/L, but can be as high as 40,000 mg/L (Wallis, 1967).
- The unconsolidated sediments of the Botany Sands aquifer comprise the primary groundwater resource in the Botany Basin. As the name suggests, the aeolian sand deposits comprise the most significant water-bearing unit within the sedimentary profile.

The primary fresh water resources associated with the Botany Sands aquifer occur north of Botany Bay. This portion of the Botany Sands aquifer is bounded by outcropping Hawkesbury Sandstone to the east and west, by Ashfield Shale to the northwest, and by the fresh/saline water interface along the Botany Bay shoreline to the south (Webb and Watson, 1979) (Figure 2.5). On a regional scale the aquifer is unconfined, but the presence of silt, clay and peat lenses create locally confined and semi-confined conditions. The average saturated thickness across the basin is approximately 15 metres (Jankowski and Knight, 1991).

The Botany Sands aquifer receives recharge primarily from the direct infiltration of precipitation into the sand beds in a number of open spaces in the northern Botany Basin, the largest of which are Centennial and Moore Parks, Randwick Racecourse, and five golf courses located in the vicinity of the Lachlan Ponds chain (Figure 2.6; Dudgeon, 1993). According to calculations by Nieweglowski (1988) it would take 10 years of average rainfall to fill the Botany Sands aquifer to capacity. Thus, the Botany Sands aquifer would be flushed with one pore volume approximately every ten years.

Another source of recharge to the Botany Sands aquifer is surface and stormwater run off from the sealed urban areas that cover a large portion of the Botany Basin. This water discharges into the ponds of Centennial Park and the Lachlan Ponds system via stormwater drains; ultimately a portion of this water infiltrates into the groundwater system, depending on the hydraulic groundwater/surface water relationship of the ponds.

A final source of recharge to the Botany Sands aquifer is seepage from joints, fractures and bedding planes within the Hawkesbury Sandstone (Johnson, 1981). Rainfall enters the Hawkesbury Sandstone through joints and fractures in the elevated outcrops and then migrates through the sandstone until it discharges into the unconsolidated sediment, typically where the joints and fractures intersect bedding planes. However, the amount of recharge from this source is negligible when compared with the contribution from rainfall (Jankowski and Knight, 1991). Merrick (1994) estimated that the total recharge quantity to the northern portion of the Botany Basin is from 22 to 44 ML/day.

Water that enters the Botany Sands aquifer in the northern portion of the basin migrates approximately south to southwest towards Botany Bay, following the Lakes Valley watercourse, the predominant feature of the bedrock topography north of Botany Bay. During development of a model to simulate groundwater flow in the Botany Basin, Merrick (1994) defined four major processes through which groundwater discharges from the Botany Sands aquifer:

- Discharge of shallow groundwater into the Botany Wetlands in the central and southern portion of the Basin (estimated to be about 8 ML/day during wet climatic conditions, with surface water infiltrating back into the aquifer during dry spells when the water table is depressed),
- Discharge of shallow groundwater into the Alexandria Canal which then drains into the Cooks River and ultimately into Botany Bay (estimated at 9-12 ML/day),
- A minor component of downward percolation into the underlying Triassic bedrock,
- Deep regional groundwater discharges directly into the ocean through the floor of Botany Bay.

A final discharge mechanism is through groundwater abstraction for commercial and industrial purposes. In a status report issued by the DLWC (2000), the annual abstraction rate for 1989 was estimated to be 9745 ML for the northern portion of the Botany Basin, and Merrick (1998) reported an abstraction volume of 10,940 ML in 1992.

In a study of historical water level fluctuations (1942 – 1988) within the Botany Sands aquifer north of Botany Bay, Merrick (1998) noted a couple of prominent trends. A pronounced rise of approximately 4 m in the north-central portion of the aquifer was attributed to a decrease in abstraction in this region, along with a cyclic increase in rainfall and irrigation of recreation areas (notably Moore Park and Kensington Golf Course). A significant decrease in water levels in the southeastern portion of the aquifer (-8 m) was attributed to a significant increase in industrial abstraction in this area, particularly in the 1950's and 1960's.

2.6 HYDROCHEMISTRY OF THE BOTANY SANDS AQUIFER

Groundwater chemistry in the Botany Sands aquifer is influenced by the chemical composition of the rainfall that recharges the aquifer, interaction with minerals and organic matter present in the aquifer, and contamination sources affecting water quality in the aquifer (Jankowski and Knight, 1991). Regional, macro-scale variations are

observed along flow path between the recharge and discharge zone of the aquifer. Within these broad regions local variations are observed as a result of heterogeneities within in the aquifer and anthropogenic impacts.

The hydrochemistry of the Botany Sands aquifer was first investigated by Sheil (1942) who concluded that, although the water was largely unpolluted, there was considerable variation in the chemical composition of samples collected only a few metres apart. Later studies by Griffin (1963) supported Sheil's findings, highlighting the influence of micro-scale heterogeneities on water quality in the aquifer. Following Griffin's work, the groundwater chemistry of the Botany Sands aquifer has been the focus of dozens of academic and commercial investigations.

Today an extensive network of monitoring bores exists within the Botany Sands aquifer, and a comprehensive review of groundwater chemistry data from these bores has been used to create an overall hydrochemical classification scheme for groundwater in the northern Botany Basin. Three proposed classification schemes have been reported in literature (Johnson, 1981; Acworth and Jankowski, 1993; Jankowski and Yu, 1998). Johnson defined three subdivisions within the northern Botany Basin, based on the relative concentrations of ions in groundwater samples. Acworth and Jankowski (1993) provided the first detailed classification of the Botany Sands aquifer along the Lakes Valley watercourse, dividing it into a northern, middle and southern zone based on the hydrogeochemical processes occurring in each zone. Jankowski and Yu (1998) also provided a three-zone classification scheme for the aquifer, but then divided each zone into a number of subgroups based on locally significant hydrogeochemical processes affecting the groundwater chemistry. The classification scheme developed by Jankowski and Yu (1998) is summarized below (Figure 2.7):

- **Zone A:** This zone extends from Centennial Park to the southern end of the Lachlan Ponds system, and represents the main body of uncontaminated to lightly contaminated groundwater in the aquifer. This zone is subdivided into four groups:

- **A-1:** Groundwater in Centennial and Moore Park with a chemical signature similar to the precipitation that recharges the aquifer (Cl-SO₄-Na-Ca). This water is characterised by low salinity, slightly acidic pH, and is locally affected by the influence of storm water runoff discharging into the ponds of Centennial Park.
- **A-2:** Groundwater between Centennial Park and the northern end of the Lachlan Ponds system. This water is characterised by slightly higher salinity as a result of fertilizer contamination from Randwick Racecourse, and a relative increase in Ca and HCO₃ from the dissolution of shell fragments in the aquifer.
- **A-3:** This group spans the region in the vicinity of the Lachlan Ponds system, and represents a mix of regional groundwater and extremely fresh groundwater associated with recharge from sand dunes.
- **A-4:** Groundwater associated with the leachate contamination from the Astrolabe Park landfill. This group is characterised by high salinity, elevated pH, and low redox potential as a result of chemical reactions occurring between the landfill leachate and the aquifer.
- **Zone B:** This zone represents the highly contaminated groundwater associated with the industrialised north shore of Botany Bay. Water quality in this zone has been impacted by both organic and inorganic contaminants related to Kingsford-Smith Airport, the Orica site (formerly ICI), and a variety of petrochemical and other industrial facilities. In addition, this region has been subject to seawater intrusion as a result of groundwater abstraction by industrial sites close to the Botany Bay margin. This zone is divided into two distinct groups:
 - **B-1:** Represents the major industrial centre around Banksmeadow. Groundwater in this group is characterised by very high salinity, a slightly acidic pH (6.5).
 - **B-2:** This group represents groundwater in the vicinity of the airport, and is characterised by very high salinity and alkaline pH (8.4).

- **Zone C:** This zone represents groundwater in the Alexandria area. It is characterised by high salinity and elevated pH (8.2). This water is enriched in HCO_3^- and Ca^{2+} compared to the other zones. Groundwater quality in this zone has been affected by leachate from the landfilling of the defunct brick pits in the St Peters area.

The representative water types for each of these zones are summarised in Table 2.1 (elements in brackets represent alternative water types present within the group). In general, groundwater in the northern recharge zone of the Botany Sands aquifer is the highest quality, chemically similar to the rainfall that recharges the aquifer; the water quality is variably degraded along flow path as a result of interactions with aquifer minerals and organic matter, and especially as a result of contamination sources impacting the aquifer.

Table 2.1 Summary of chemical water types in the Botany Sands aquifer (Jankowski and Yu, 1998).

Zone	Group	EC (avg)	pH (avg)	Water types
A	A-1	168	>6.6	Cl-SO ₄ -Na-Ca (HCO ₃ -Cl-Na-Ca, Cl-HCO ₃ -Ca-Mg)
	A-2	213	6.6	Cl-SO ₄ -HCO ₃ (or HCO ₃ -SO ₄)-Ca-Na
	A-3	158	5.8	Cl-SO ₄ -Na (or -Na-Ca, or -Na-Ca-Mg)
	A-4	1130	8.2	HCO ₃ -Cl-Ca-Na
B	B-1	1718	6.5	Cl-SO ₄ -Na (or -Na-Mg), Cl-Na
	B-2	1699	8.4	HCO ₃ -SO ₄ -Na-Ca (or -Ca-Na) HCO ₃ -Cl-Ca-Na
C	--	605	8.2	HCO ₃ -(Cl or SO ₄)-Ca-Na

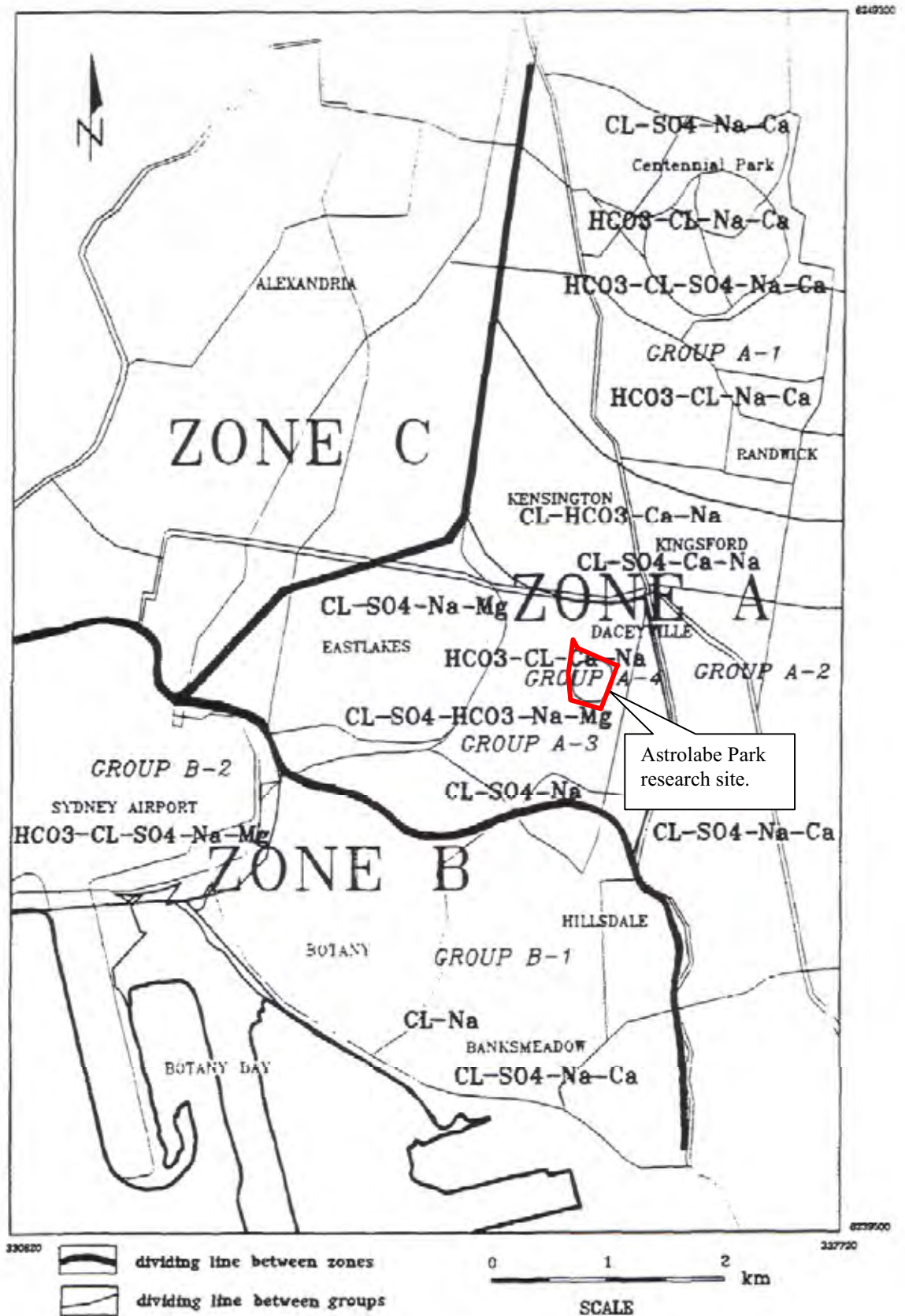


Figure 2.7 Detailed hydrochemical zonation of the Botany Sands aquifer (Jankowski and Yu, 1998).

2.7 WATER CONTAMINATION IN THE BOTANY BASIN

Water quality in the Botany Sands aquifer has deteriorated over the past 100 years at the hands of industrialisation and the stresses associated with urbanisation. It is unavoidable that an aquifer, upon which is situated a major international capital with a population in the millions, will be subjected to various sources of contamination. Groundwater contamination has been the focus of numerous investigations in the Botany Sands aquifer, although details of many of the commercial investigations are confidential due to their legal sensitivity. As early as 1942, Sheil (1942) had identified sewage contamination in the northern and eastern portions of the aquifer. Another survey conducted by the Geological Survey of NSW in 1973 (Hawke, 1973) identified several significant sources of pollution to groundwater throughout the Botany Sands aquifer. Both the East Botany – Banksmeadow industrial area and the Astrolabe Park landfill have been the subject of regulatory scrutiny as a result of their impact to water quality in the Botany Sands aquifer.

Some of the major contaminants of concern across the Botany Basin include leachate plumes from a number of old, unlined landfills, urban stormwater run off that percolates into the groundwater system, leaking sewer mains, nutrient loading from the use of fertilizers in parks and golf courses, as well as manure from Randwick Racecourse, seawater encroachment from excessive groundwater abstraction close to the coast, and a variety of organic and inorganic contaminants associated with the highly industrialised suburbs of Banksmeadow and Botany (Lavitt *et al.*, 1997; DLWC, 2000). That said, there has been progress towards the rehabilitation of water quality in the Botany Sands aquifer over the past three decades, as an understanding and general awareness of sources and behaviour of groundwater contaminants has increased. In a climate such as that of Australia, which is subject to recurring cycles of drought, the availability of an emergency reservoir of good quality water in the immediate vicinity of Australia's largest urban centre is a valuable asset.

CHAPTER 3: LOCAL ENVIRONMENTAL SETTING – ASTROLABE PARK

3.1 LOCATION

The research site is located in Daceyville, an urban suburb of Sydney, Australia (Figure 2.5). It consists of the Astrolabe Park landfill, which was completed as sports field upon decommissioning in 1975, as well as a portion of the adjacent Eastlakes Golf Course and a portion of the eastern shore of Lachlan Pond No. 5, which is adjacent to the golf course (Figure 3.1). It is located approximately 6 km south of the Sydney central business district (CBD), and 4 km north of Botany Bay (Figure 2.5). The site is situated adjacent to the start of the Lachlan Ponds system, the major surface drainage feature of the northern Botany Basin. The Astrolabe Park Landfill is bounded by Eastlakes Golf Course to the west, by a residential neighbourhood to the north, by the UNSW David Phillips sports fields to the east, and by an open grass field to the south that is commonly used as a golf driving range and materials (sand and earth) stockpile area by Eastlakes Golf Course.

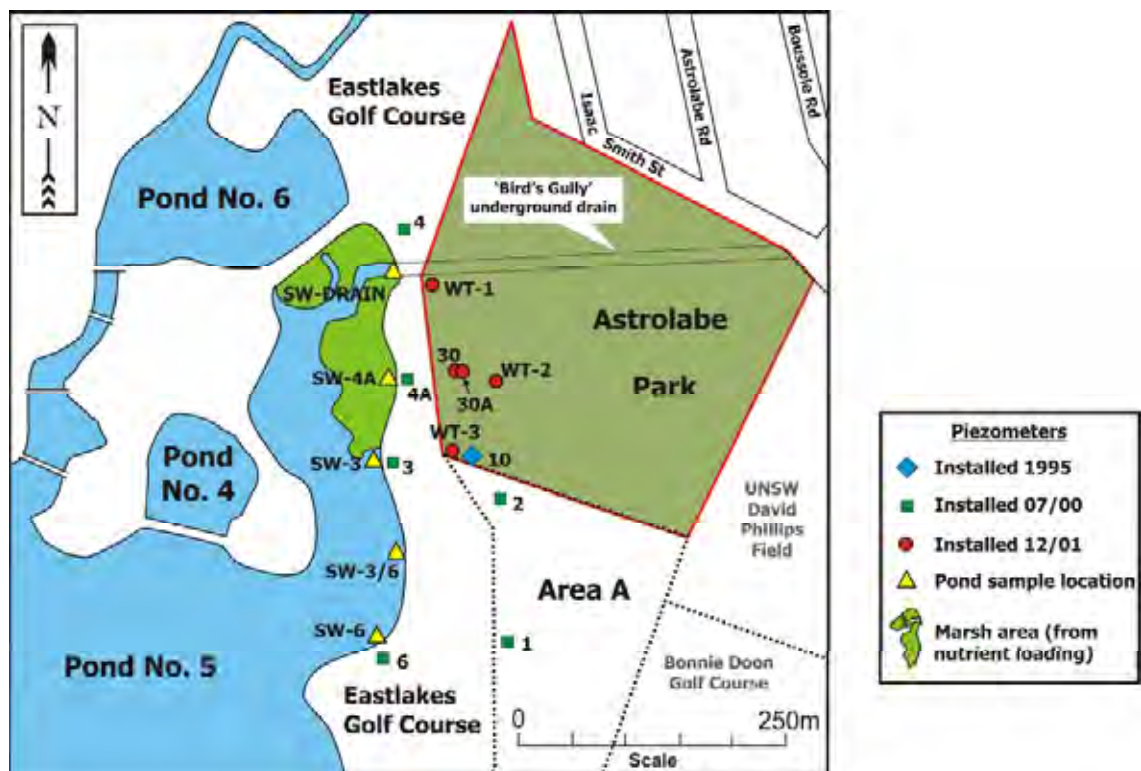


Figure 3.1 Astrolabe Park research site and environs.

3.2 HISTORICAL LAND USE

The Lachlan Ponds (or swamps) have been the target of development since the early 1800's, when various mills were established along the banks of the 'Botany Stream' (Annable and Cable, 1995). In the mid-1800's, the Botany Swamps Supply Scheme was instigated to provide an alternative fresh water supply for Sydney, resulting in the construction of many of the dams and weirs that separate the Lachlan Ponds today. The area that is now Astrolabe Park remained relatively intact until its initial use as a rubbish tip in the early 1900's. From the early 1960's to the mid 1970's a comprehensive program of sand mining and landfilling of waste in Astrolabe Park and the surrounding environs resulted in a major physical transformation, and environmental impact, to the site.

3.2.1 Original environmental setting

Prior to European settlement the site of Astrolabe Park was a flat, brushy field, with the Bird's Gully drainage flowing diagonally across the site from northeast to southwest. The Bird's Gully drainage originated in Randwick and terminated in a low-lying swamp at the confluence with the Botany Swamps (Thorpe, 1953). An aerial photograph of the site from 1930 depicts the Bird's Gully swamp as a relatively large pond covering a significant portion of the research site (Figure 3.2; Department of Land and Water Conservation, 1930). In aerial photos from the 1960's and early 1970's the Bird's Gully has been reduced to a creek, and is finally replaced by an underground concrete drain upon decommissioning of the Astrolabe Park landfill.

3.2.2 Sand mining and landfilling activities

The source of the information for this section comes from a thorough review of Botany Council records as reported by Hitchcock (1991). Because of the scarcity of information regarding the waste disposal activities at the Astrolabe Park landfill, the Botany Council records comprise the sole source for this information. Unless otherwise noted, the information in this section is based entirely upon Hitchcock's review. A chronological summary of the sand mining and landfilling at Astrolabe Park and surrounding

properties is presented in Table 3.1 (Hitchcock, 1991). Figure 3.4 outlines the various regions that were sand mined and used for waste disposal at the site.



Figure 3.2 Aerial photograph of South Sydney and the study site in 1930, including the current boundary of Astrolabe Park (modified from DLWC, 1930).

The histories of landfilling and sand mining at Astrolabe Park roughly parallel each other, and in some cases are intimately related. Prior to the commencement of sand mining at the site, two episodes of landfilling occurred at Astrolabe Park:

- 1938 – 1945: The Botany Council allocates two acres of swampland in Astrolabe Park for waste disposal (Area C, Figure 3.3). Sometime around 1945 the municipal waste disposal is routed to an alternate tip site at Sir Josef Banks Park in Botany.
- Early 1964 to mid-1965: The tip at Sir Josef Banks Park is full, and landfilling resumes at Astrolabe Park (Portion of Area D, Figure 3.3). Upon receiving

permission from the Botany Council in May 1965, waste disposal switches to backfill for the sand mining pits in Area A.

In 1962 sand mining commenced on the property to the south of Astrolabe Park, in the dunes of Area A (Figure 3.3). The resulting water-filled sand pits were backfilled with solid and liquid waste of domestic and commercial origin, a process that was repeated in Astrolabe Park between 1970 and 1975. An attempt was made to mine Area D (Figure 3.3) in 1970, but was abandoned shortly after starting due to difficulties with peat beds that were encountered. Mining of the remainder of Astrolabe Park commenced in 1971, with the following provisions: no mining was allowed in the areas previously used for waste disposal, or within a 30-foot corridor on either side of the Bird's Gully creek running across the site. The general sand mining lease requirements, as set forth by the Lands Department, were as follows:

- Excavations are not to exceed 20 feet (6.1 metres) below ground surface (bgs). This restriction was later relaxed to allow excavations to 17 metres bgs. Excavations are required to be backfilled with suitable material within four weeks; backfill material is to be compacted into layers not exceeding 4 feet (1.2 metres) thick.
- Excavation faces must be graded to a 1:3 slope.
- A dragline operation must be used for removal of sand below the water table.
- No mining is allowed within 30 feet (9.2 metres) of either side of the Bird's Gully creek at Astrolabe Park.

The total volume of waste deposited was calculated by Hitchcock to be 204,000 m³, based on an extrapolation of the reported disposal rate of 4,000 m³ per month over the 51 months of landfilling in the sand mined portion of Astrolabe Park (Hitchcock, 1991). The lateral extent of the landfill was estimated to be 4.7 ha, based upon examination of a series of aerial photographs, resulting in a calculated average waste thickness of approximately 4.35 m across the site. A survey conducted by the Municipal Waste Disposal Authority in January 1974 concluded that each ton of waste occupied an average volume of 3.7 m³; this density ratio suggests that little or no waste compaction was applied during waste disposal at Astrolabe Park (Hitchcock, 1991).

Sand mining and waste disposal at Astrolabe Park concluded in 1973 and 1975, respectively. One week after waste disposal concluded, the Department of Lands reported that leachate was leaking into Pond No. 6 via a stormwater drain on the western edge of the landfill (Hitchcock, 1991). In 1978 the entire site was covered with approximately 500 mm of fine sand as part of the decommissioning process, and artificial hills were graded along the western edge of Astrolabe Park to prevent surface run off from the landfill from draining onto Eastlakes Golf Course. It is presumed that this is the same time period when the Bird's Gully creek was replaced by an underground concrete drain that runs beneath Astrolabe Park and Eastlakes Golf Course and discharges directly into Pond No. 5.

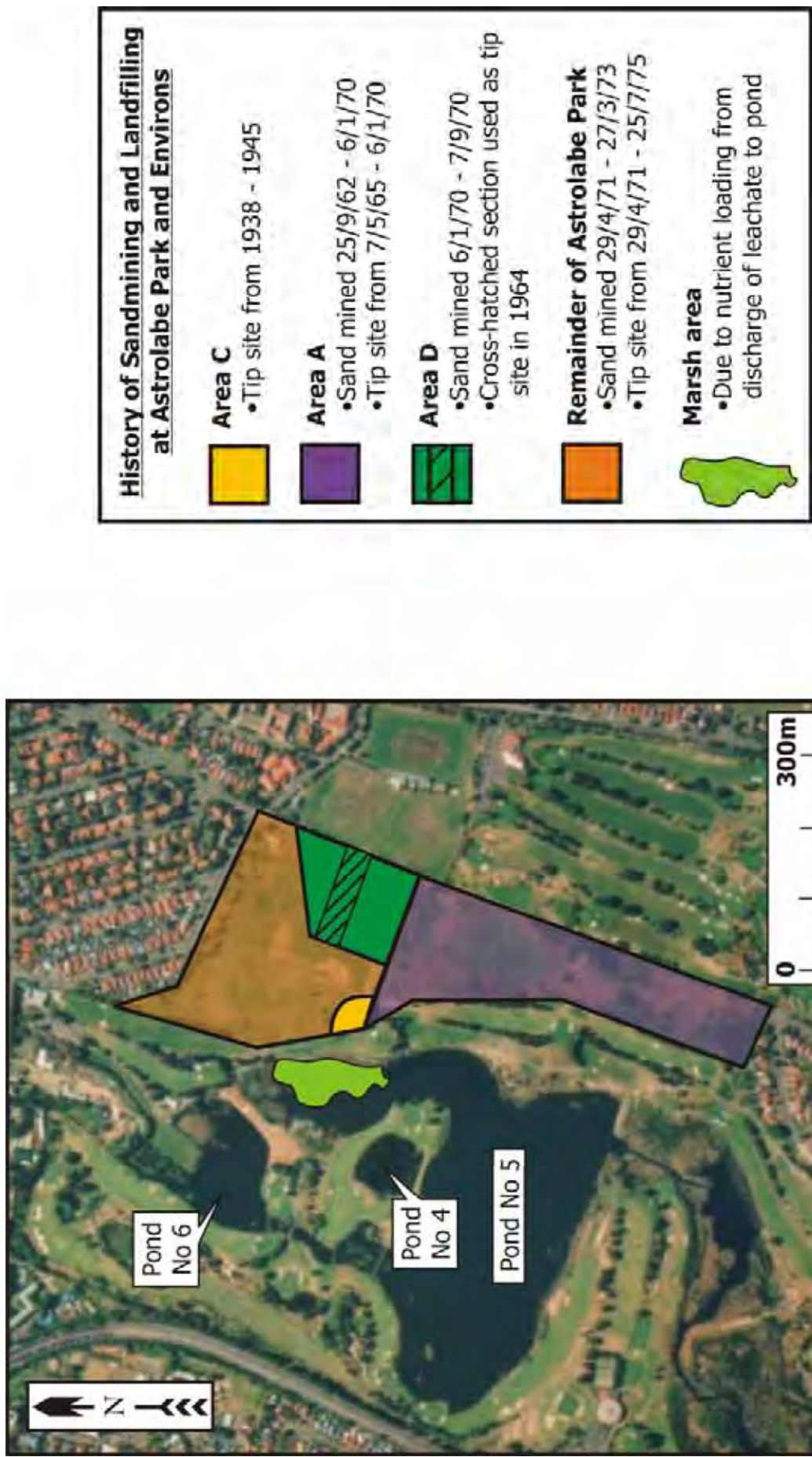


Figure 3.3 History of sandmining and landfilling at Astrolabe Park and vicinity (summarised from Botany Council records, as reported by Hitchcock [1991]).

Table 3.1 Chronology of sand mining and landfilling at Astrolabe Park and vicinity (summarised from Botany Council records, as reported by Hitchcock [1991]).

Date Range	Activities Undertaken
February 2, 1938 to approximately 1945	Botany Council uses two acres of swampland at Astrolabe Park (area C, Figure 3.1) for waste disposal.
September 25, 1962	N.S.W. Mining Company Pty Ltd begins sand mining in area A (Figure 3.2).
Early 1964 to May 7, 1965	Botany Council uses area D at Astrolabe Park for waste disposal (Figure 3.1).
May 7, 1965	N.S.W. Mining Company Pty Ltd allowed to backfill sand mining area A with waste from Botany Council (Figure 3.1). Subsequently Botany Council stops using area D for waste disposal.
January 6, 1970	N.S.W. Mining Company Pty Ltd starts to sand mine area D at Astrolabe Park (Figure 3.2).
January 6, 1970 to September 7, 1970	N.S.W. Mining Company Pty Ltd has difficulty mining area D due to the presence of shallow peat layers.
April 29, 1971	N.S.W. Mining Company Pty Ltd begins mining the rest of Astrolabe Park and backfills with municipal waste (area A', Figure 3.1).
March 27, 1973	Sand mining and waste disposal by N.S.W. Mining Company Pty Ltd ceases.
March 27, 1973 to July 25, 1975	Waste disposal by Botany Council continues at Astrolabe Park (area A', Figure 3.1).
July 25, 1975	Botany Council terminates waste disposal operations at Astrolabe Park.
July 31, 1975	The Department of Lands reports that leachate is entering Pond No. 6 via a rough drain along the western edge of the landfill that discharges into the stormwater drain.
1978	Botany Council is in compliance with the decommissioning surface grading requirements set forth by the Lands Department and the Metropolitan Waste Disposal Authority.

3.3 LOCAL STRATIGRAPHY

The Quaternary stratigraphy of the research site is generally interpreted to belong to Unit 3 of the Botany Basin stratigraphic sequence, which is characterised by thick deposits of aeolian quartz sand with local occurrences of discontinuous clay and peat lenses (see chapter 2 for unit description). The stratigraphy of the site has been investigated in conjunction with several previous research programs (Hitchcock, 1991; Jankowski and Knight, 1991; Smolski, 1994). Previous research was also conducted by Webb and Watson (1979) at the east-adjacent UNSW David Phillips field, located hydraulically upgradient from Astrolabe Park (Figure 3.1), and also south of Astrolabe Park at the East Lakes Experimental (ELE) site (Evans, 1993; Jankowski et al, 1997; Beck, 2000; Jankowski and Beck, 2000). A review of bore logs, dutch cone penetrometer tests, borehole geophysical logs and resistivity soundings indicate a high degree of heterogeneity with respect to the vertical and lateral distribution of site-scale sedimentary features (Jankowski, 1997). However, several stratigraphic units are consistently present in and around the research site, and there appears to be a general correlation between a couple prominent stratigraphic features observed both within and in the vicinity of David Phillips field and Astrolabe Park.

3.3.1 Description of stratigraphic units

The Quaternary stratigraphy of the research site can generally be described as being comprised of four major sediment types: aeolian sand, silt/clay, peat, and cemented ferruginous horizons known as Waterloo Rock. However, it should be noted that, as is often the case with lithologic interpretations, the composition of these units tends to vary and overlap, and the gradational and discontinuous nature of these units often precludes a distinct lithologic classification. Bearing that in mind, commonly reported variants of each of these units will be discussed.

3.3.1.1 *Aeolian quartz sand*

By far the most abundant sedimentary unit is aeolian sand. This unit is the primary component of the Botany Sands aquifer, and has the highest values of porosity, hydraulic conductivity and dispersivity (Evans, 1993; Yu *et al.*, 1993; Yu, 1994). It is

predominantly uniform, well-rounded medium quartz sand (greater than 99%), with minor heavy minerals and silt (Yu, 1994). The XRD analyses indicate an absence of clay minerals, and iron oxide has been observed under a microscope coating grains and filling microscopic pits and cracks on the surface of the grains (Beck and Jankowski, 1998).

3.3.1.2 *Waterloo Rock*

The colloquially termed Waterloo Rock occurs as dense, cemented ferruginous horizons within the upper three metres of the sand profile. These layers are generally between 0.1 and 0.2 m thick (Hitchcock, 1991; Jankowski and Knight, 1991; Evans, 1993), but have been observed by Hitchcock to be as thick as 110 cm east of the site. Evans (1993) noted that in cross-section these layers are parallel to present dune surfaces, suggesting that they may be related to previous soil horizons. Albani postulated in a verbal communication to Evans (1993) that these might represent ancient soil horizons that may have been temporarily established in intra-dune basins during brief hiatuses in aeolian deposition and dune migration.

3.3.1.3 *Silt, clay and peat*

Embedded within the quartz sand are regionally discontinuous layers that contain varying percentages of silt, clay and peat. These can generally be subdivided into layers that are primarily sandy with a minor component of silt, clay and peat, and layers that are dominated by silt, clay and peat. However, there is an obvious gradational relationship between the two, and the particular classification scheme seems to vary between different researchers. Two commonly occurring lithologies are thin silty/organic layers, interpreted as being ancient soil horizons (Evans, 1993), and clay layers, sometimes containing a significant peat component. These layers appear to vary in thickness from a few centimetres to over a metre; the thin silty/organic layers appear to be laterally discontinuous as they often show little correlation between boreholes. These layers have been variously reported as silty/clayey sand, organic silty sand, peaty sand, sandy peat, sandy clay, organic/peaty clay, peat and clay. They have much lower values of hydraulic conductivity than the sand unit, and exert a significant control over the local hydrochemistry and groundwater flow system, particularly with respect to solute transport, as will be discussed in detail in later chapters.

3.3.2 Stratigraphic sequence

While it appears that many of the heterogeneities present in the sandy aquifer are discontinuous, there are a couple features that have been correlated between several borehole cores, geophysical logs, and cone penetrometer test sites in the vicinity of Astrolabe Park. In David Phillips field, Webb and Watson (1979) were able to divide the unconsolidated sediments into three hydrogeologic units separated by locally continuous low-permeability layers located at depths of approximately 7 and 17 m (Figure 3.4). The upper low-permeability layer is reportedly comprised of sandy peat, and varies in thickness from just a few centimetres to over a meter. The lower low-permeability layer is comprised primarily of peaty clay, and has an average thickness of between 1 and 2 m. These layers were correlated between five boreholes along a 72 metre transect; several other discontinuous fine-grained layers were also recorded in these boreholes. The Hawkesbury Sandstone bedrock was encountered between 30 and 35 metres below ground surface in all bores.

A stratigraphic cross-section along a transect that runs N-S between the western edge of Astrolabe Park and the eastern shore of Lachlan Pond No.5 is presented in Figure 3.5, based on a review of bore logs, geophysical surveys and cone penetrometer tests performed by Hitchcock (1991), Jankowski and Knight (1991) and Smolski (1994). The cross-section indicates four major discontinuous clay/organic clay layers present at depths of 7, 8, 10 and 13 metres between the landfill and the pond, although only the basal layer was detected towards the northern end of Astrolabe Park. It's possible that one or more of these layers are the same as the ones observed in David Phillips field, particularly since they fall in the depth range of the upper confining unit, but a lack of stratigraphic information from the central portion of Astrolabe Park makes this correlation difficult.

However, it is reasonable to assume these layers represent a brief change in the depositional environment at the site, from aeolian deposition of quartz sand to swampy terrain that would result in the co-deposition of fine sediments and organic material. Further, it is not unreasonable to assume that this change in depositional environments

would have affected the area that is now Astrolabe Park as well as the adjacent David Phillips field, particularly since both lay approximately at the confluence of the Botany Swamps and Birds Gully creek that drains the low topography of Randwick and Kingsford. In the 1930 aerial photo of the site (Figure 3.2), it is apparent that the extent of the swamp area associated with the Birds Gully creek in Astrolabe Park is much larger than just the commonly defined southwest corner of the site. It is plausible that a previous inundation of Astrolabe Park and David Phillips Field by this drainage system due to rising sea levels in the Pleistocene could have resulted in the deposition of peaty clay layers that are contiguous between the two sites (Acworth, 1998). The site-scale continuity of these regionally ‘discontinuous’ clay layers constitutes a significant constraint on the hydrogeology and solute transport at the research site.

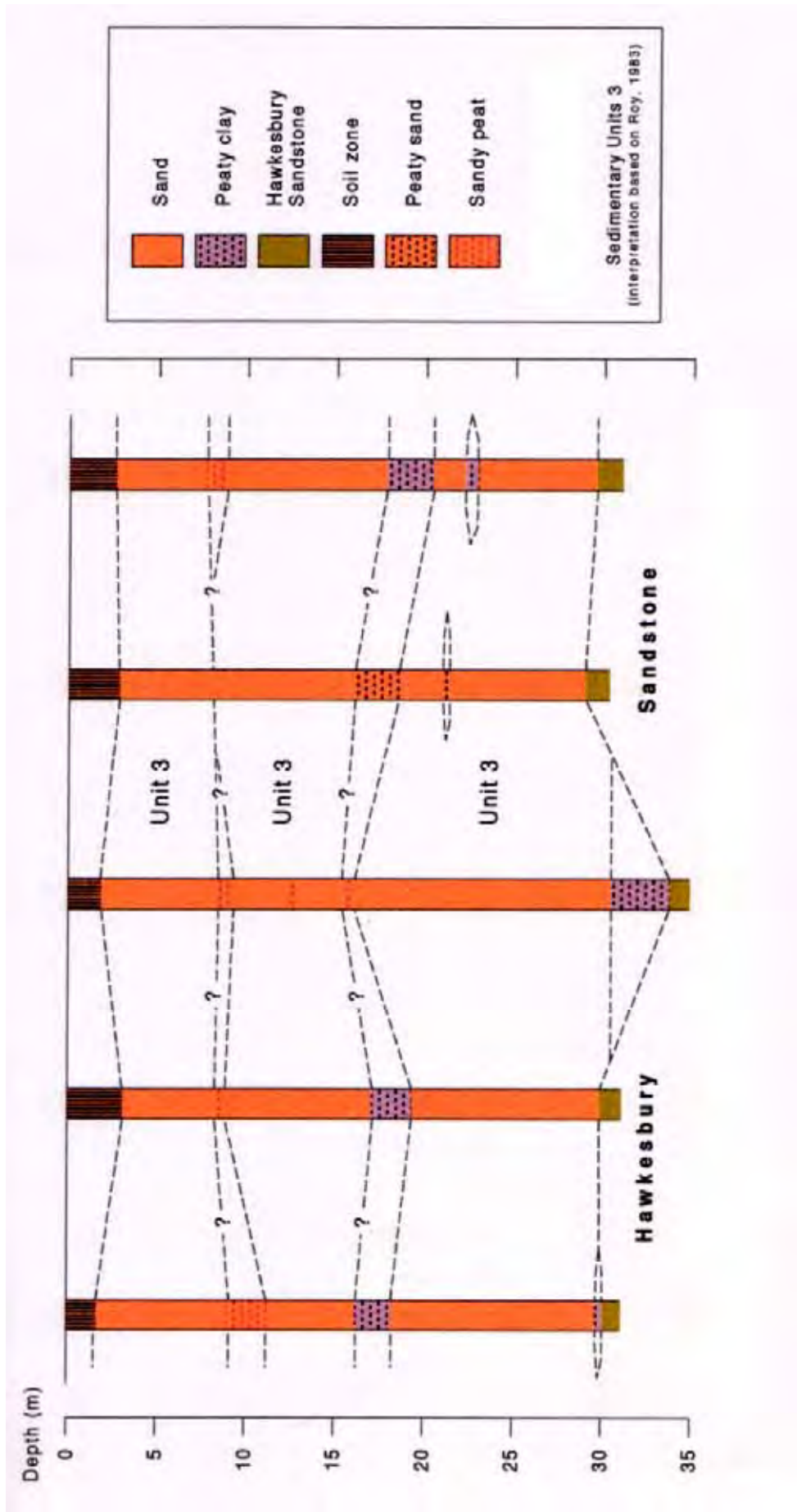


Figure 3.4 Stratigraphic cross-section through David Phillips field (modified from Webb and Watson, 1979). Reference Figure 1.2 for the cross-section location.

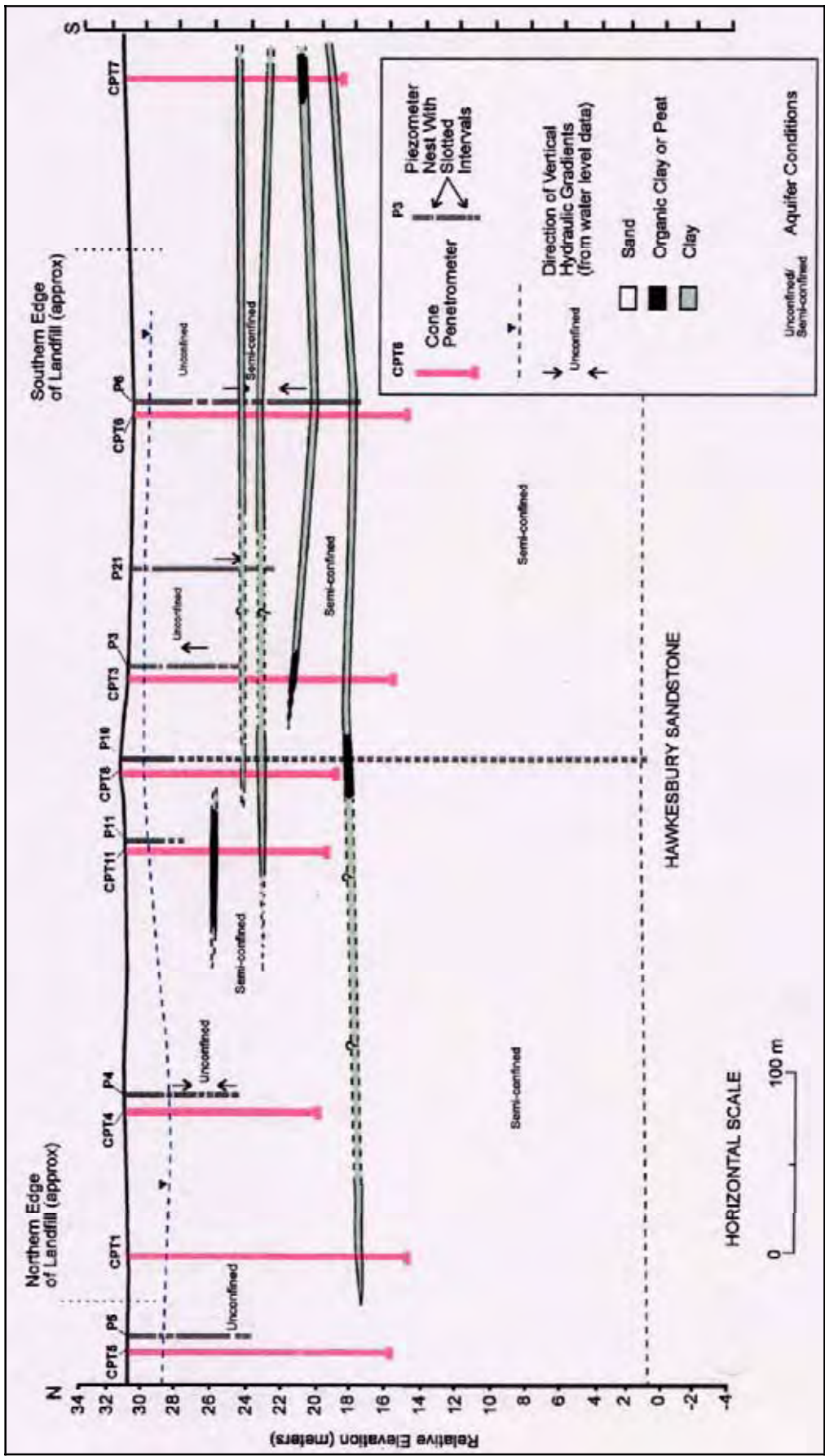


Figure 3.5 Stratigraphic cross-section between Astrolabe Park and Lachlan Pond No. 5 (Roberts, 1996). Reference Figure 1.2 for the cross-section location.

CHAPTER 4: LANDFILLS AND LEACHATE

A substantial amount of research has been conducted over the past several decades with regards to formation of leachate in landfills, and the transport and attenuation of leachate contaminants in the environment. This chapter provides an overview of the mechanisms governing leachate formation, transport and attenuation, and finishes by addressing the data gap that constitutes the primary focus of this research project. A considerable acknowledgement is made to the comprehensive review of current literature by Christensen *et al.* (2001).

4.1 LEACHATE PRODUCTION

Landfill leachate can be defined as a water-based solution of compounds derived from the waste (Andreottola and Cannas, 1992; Christensen *et al.*, 1994 and 2001). Landfill leachate is generated when water infiltrates into the waste layers, instigating a number of physical, chemical and biological processes that act as mass transfer mechanisms for contaminants between the waste and the leachate. The quality and quantity of leachate produced depends on many factors, including: the composition and volume of the landfilled waste, the degree of compaction of the waste, the moisture content of the waste, the total flux of water into the waste layers, and the age (or maturity) of the waste. Waste disposed of in modern sanitary landfills contains a large percentage of organic material (Bennett and Doyle, 1997; Bell, 1998); the degradation of the organic matter results in the formation of a sequence of redox environments that play a dominant role in establishing the overall hydrochemical framework for the composition of the leachate produced.

4.1.1 Landfill lifecycle

The objective of a sanitary landfill is to act as a repository for waste until it has had the chance to completely degrade into simple, environmentally benign end products, at which point the waste is considered to have ‘stabilised’. Waste stabilisation occurs as the result of four major physical, chemical and biological processes occurring within the landfill: hydrolysis of solid waste, biological degradation of organic waste,

solubilization of soluble salts contained in the waste, and transport of waste as colloids or particulate matter (Andreottola and Cannas, 1992). The degradation process begins immediately after waste is buried in the landfill. Research from a large number of landfills of varying ages has resulted in the identification of several distinct phases of waste decomposition that can be characterised by the composition of the leachate and landfill gas that is generated (Figure 4.1). While the subdivision of these phases vary depending on the researcher, they can generally be categorised as a primary aerobic phase followed by a number of anaerobic phases, with the potential for an ultimate return to a terminal aerobic phase. A summary of these phases is provided, as reported by Andreottola and Cannas (1992):

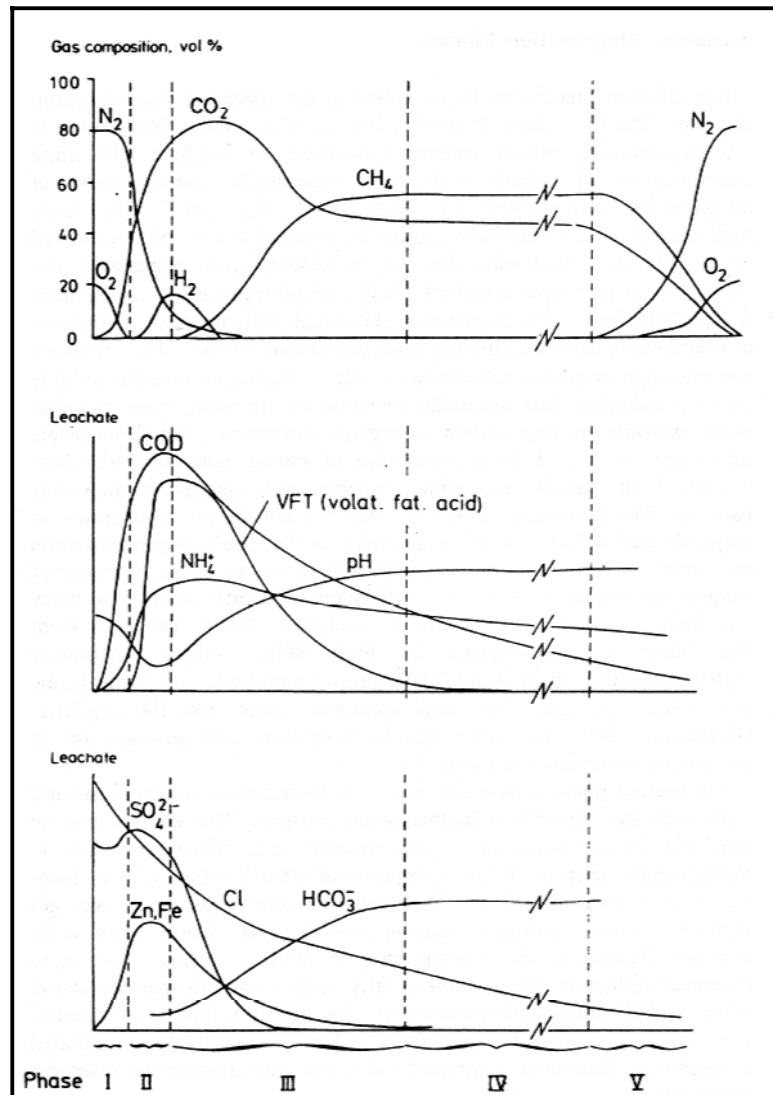


Figure 4.1 Evolution of landfill leachate and gas over the lifespan of a landfill (modified from Christensen and Kjeldsen, 1989).

- ***Initial Aerobic Phase:*** This phase is characterised by high-intensity aerobic degradation of organic waste, resulting in leachate generation characterised by acidic conditions and an elevated solute load. This phase is generally short lived, due to the high biological and chemical oxygen demand in the waste layers.
- ***Anaerobic Phases:*** Once the free oxygen in the waste is consumed, the waste degradation proceeds through anaerobic pathways. The changes in the primary waste degradation processes result in a general evolution in leachate composition from acidic with a high solute load, to progressively lower pH values with a diminished solute load. Labile organic carbon compounds are consumed during microbial respiration, resulting in an increased percentage of refractory organic compounds in leachate. Progressive depletion of electron acceptors results in the eventual onset of methanogenesis. This phase of waste degradation will continue indefinitely until the waste is ultimately *stabilised*.
- ***Terminal Aerobic Phase:*** Theoretically, the waste in the landfill will ultimately degrade to such a degree that oxygen will begin to diffuse into the waste from the atmosphere. Whilst published evidence of this stage of landfill evolution is lacking, researchers have predicted that some recalcitrant compounds may continue to affect landfill leachate for centuries (Ehrig, 1983 and 1988; Belevi and Baccini, 1992).

4.1.2 Physical processes

The main physical process contributing to leachate formation in landfills is the advective entrainment of colloids and particulate matter in the water that infiltrates into the waste, including bacteria, long-chain dissolved organic carbon molecules, and rock or mineral fragments or precipitates (Freeze and Cherry, 1979; Domenico and Schwartz, 1998). These tiny solid particles have been found to be a significant component of landfill leachate (eg. Gounaris *et al.*, 1993), and can have substantial effects on the transport of hydrophobic leachate contaminants through adsorption and surface complexation reactions.

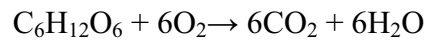
4.1.3 Chemical processes

There are a number of chemical reactions that facilitate the mass transfer of contaminants from waste to leachate in a landfill, including: dissolution/precipitation of inorganic components due to changes in pH and ionic strength of the water, hydrolysis of organic compounds, and ion exchange and sorption reactions. Dissolution of inorganic waste components is especially prominent in the acid fermentation phase of waste decomposition due to the low pH of the leachate. Organic compounds are broken down and solubilized through both hydrolysis and redox reactions; however, in a landfill setting redox reactions are microbially mediated and are discussed in the next section. While ion exchange plays a more important role with regards to the exchange of leachate components with ions present on clay surfaces in an aquifer setting, ions can also be exchanged within the landfill if the appropriate exchange surfaces are present. As mentioned in the previous section, adsorption or complexation of hydrophobic organic molecules and metals onto organic colloids present in the leachate can greatly increase the flux of certain contaminants that might otherwise be considered immobile.

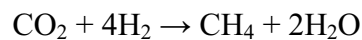
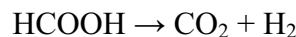
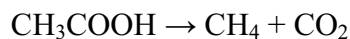
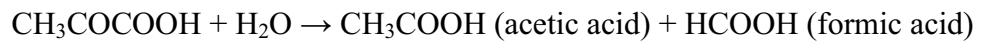
4.1.4 Microbiological processes

By far the most important processes with regards to leachate generation and composition in landfills are microbially mediated oxidation-reduction (redox) reactions. Redox reactions can be defined as reactions in which electrons are transferred between products and reactants; the element that loses electrons is oxidized and the element that gains electrons is reduced. In practical terms, these are reactions in which organic matter is decomposed (or oxidized) by bacteria and other microbes through the utilization of oxidants present in the water or waste material. The decomposition of organic matter can result in the generation of a high concentration of intermediate organic acid compounds, dissolved hydrogen gas, and ultimately CO₂, which can significantly lower the pH of the leachate and increase its capacity to solubilize inorganic waste components. The decomposition of organic matter, using glucose (C₆H₁₂O₆) as a generic term for organic matter, generally proceeds as follows (Fetter, 1999):

Aerobic decomposition



Anaerobic decomposition



The utilization of oxidants results in the depletion or transformation of elements found in the infiltrating water and waste material. The transformation of elements involved in redox processes can either increase or decrease their solubility, significantly affecting the leachate composition and contaminant mobility. Due to the large number of elements that are susceptible to redox transformations, and the significant influence that redox environments exert on other chemical reactions, the microbially mediated decomposition of organic waste in landfills is considered to be the most important process with respect to leachate generation and quality.

4.2 LEACHATE COMPOSITION

Landfill leachate is a highly contaminated solution composed of both organic and inorganic constituents derived from the waste material. Leachate has been reported to contain larger contaminant loads than raw sewage and many industrial wastes (Qasim and Chiang, 1994). The presence and concentration of specific constituents in landfill leachate generated at a given landfill will vary depending on a complex set of interrelated factors, but regardless of concentration there are certain constituents that are common to nearly all landfills (Table 4.1). The following sections describe the organic and inorganic components that are found at various concentrations in landfill leachates. As this research focuses on inorganic compounds, discussion of organic constituents is limited.

4.2.1 Major ions

The major ions comprise a significant portion of the inorganic component of landfill leachates. While some of these ions are present in the surface or groundwater that infiltrates into the waste layers, the majority of the ionic load is derived from chemical and biological reactions in the landfilled waste. In general, the major ions found in leachates are comprised of the cations Ca^{2+} , Mg^{2+} , Na^+ , K^+ , Fe^{2+} , Mn^{2+} and NH_4^+ , and the anions HCO_3^- , Cl^- , and SO_4^{2-} (Baedecker and Back, 1979a,b; Nicholson *et al.*, 1983; Baedecker and Apgar, 1984; Jankowski and Knight, 1991; Mireki and Parks, 1994; Jankowski, 1997; Jankowski and Acworth, 1997; Christensen *et al.*, 2001). The relative abundances of these ions in leachate is highly variable, and is dependent on a number of factors including waste composition, degree of waste stabilisation, and the volume and initial composition of the infiltrating water. If leachate is released to the environment, interactions between the leachate and uncontaminated regional water and aquifer minerals will have a significant impact on the concentration and distribution of major ions in the leachate.

4.2.1.1 Cations

The major cations typically present in landfill leachates are Ca^{2+} , Mg^{2+} , Na^+ , K^+ , Fe^{2+} , Mn^{2+} and NH_4^+ . With regards to leachate formation, the vast majority of these constituents are derived from the waste material through mass transfer processes. The concentration of these cations in leachate is specific to the waste composition and stabilisation phase of the landfill. In the acid fermentation stabilisation phase, the low pH environment promotes dissolution of soluble salts and metal compounds present in the waste, and results in a dramatically elevated cation load. The reducing conditions characteristic of anaerobic decomposition has a dual effect on the cation composition of the leachate: Fe and Mn will be transformed to their reduced states through microbial degradation processes and therefore become more soluble, and under methanogenic conditions, the fermentative formation of CH_4 and NH_4^+ results in a significant increase in NH_4^+ concentration and a concomitant increase in pH, which ultimately reduces the solubility of the waste material and may promote the precipitation of Ca- and Fe-carbonate and sulfide minerals.

Ammonium constitutes one of the primary contaminants of concern in landfill leachates. It is formed by the fermentative decomposition of protein-rich organic compounds in the waste (Nicholson *et al.*, 1983; Andreottola and Cannas, 1992; Christensen *et al.*, 2001), although direct formation via reduction of NO_3^- present in water may also account for a portion of the total. Ammonium is unique as a contaminant in landfill leachate for the following reasons: 1) it is considered a priority contaminant in legislation governing water quality criteria in many countries due to its ecological toxicity; 2) it is often present at significantly higher concentrations than most of the other toxic components of landfill leachate (Erskine, 2000; Kjeldsen *et al.*, 2002); and estimates based on studies of element mass transfer and long-term emissions from landfills indicate that N-species are likely to be persistent for centuries (Ehrig, 1988; Belevi and Baccini, 1992).

Once leachate is released to an aquifer environment, several factors will affect the cation concentrations within the leachate plume. The reducing capacity of the leachate may reduce and solubilize mineral oxides and hydroxides present in the aquifer matrix (most notably Fe- and Mn- oxides and hydroxides), resulting in an increase in Fe and Mn concentrations. The high partial

Table 4.1 Concentration range of landfill leachate components (modified from Christensen *et al.*, 2001)^{1,2}

Parameters	Range
pH (units)	4.5 – 9.0
Specific conductivity ($\mu\text{S}/\text{cm}$)	2500-35,000
Total solids	2000-60,000
<i>Organic Matter</i>	
Total organic carbon (TOC)	30-29,000
Biological oxygen demand (BOD_5)	20-57,000
Chemical oxygen demand (COD)	140-152,000
BOD_5/COD (ratio)	0.02-0.80
Organic nitrogen	14-2500
<i>Inorganic macrocomponents</i>	
Total phosphorus	0.1-23
Chloride	150-4500
Sulphate	8-7750
Bicarbonate	610-7320
Sodium	70-7700
Potassium	50-3700
Ammonium-N	50-2200
Calcium	10-7200
Magnesium	30-15,000
Iron	3-5500
Manganese	0.03-1400
Silica	4-70
<i>Inorganic trace elements</i>	
Arsenic	0.01-1
Cadmium	0.0001-0.4
Chromium	0.02-1.5
Cobalt	0.005-1.5
Copper	0.005-10
Lead	0.001-5
Mercury	0.00005-0.16
Nickel	0.015-13
Zinc	0.03-1000

¹ See Christensen *et al.* (2001) for complete list of referenced works.

² All units in milligrams per litre (mg/L) unless otherwise noted.

pressure of CO_2 in leachate also promotes the dissolution of calcium carbonate minerals present in the aquifer, with a resultant increase in the calcium concentration. As the leachate mixes with the fresh regional groundwater it will become progressively more diluted with respect to all leachate components, and the related changes in saturation and redox conditions may result in the precipitation of carbonate and oxide/hydroxide minerals that affect the concentrations of Ca, Mg, Fe and Mn.

Finally, ion exchange between the leachate components and aquifer minerals will affect the cation concentrations. Potassium and NH_4 have a high affinity for cation exchange and will be retarded along groundwater flow path, and Ca will typically increase, as it is often the natural saturating ion on exchange surfaces and will be expelled. Na, Mg, Fe and Mn are less susceptible to exchange processes.

4.2.1.2 Anions

The major anions present in landfill leachate are HCO_3^- , Cl^- , and SO_4^{2-} , but their relative concentrations are dependent on the waste composition and stabilisation phase of the landfill. The concentration of HCO_3^- in acid phase leachate is typically low, as the majority of the CO_2 generated through organic decomposition reactions will exist as carbonic acid (H_2CO_3), as well as other organic acids, at low pH values. As the waste stabilisation of the landfill progresses and methanogenic conditions are established, the CO_2 generated will be converted to HCO_3^- at the higher pH values and will provide a significant pH buffering capacity to the leachate. The role of the carbonate system in landfill leachate is discussed in more detail in Section 4.4.

Concentrations of SO_4^{2-} can be elevated in acid phase leachate if there is a significant source present in the waste (i.e. plaster board constructed of gypsum [CaSO_4]). Elevated SO_4^{2-} concentrations have been reported by several researchers (Kimmel and Braids, 1980; Nicholson *et al.*, 1983), but in the absence of a significant source of SO_4^{2-} in the waste material the concentration is expected to be low. This is especially true when the landfill enters the anaerobic degradation phases; SO_4^{2-} is utilized as an oxidant for organic decomposition reactions and ultimately reduced to S^{2-} . If dissolved Fe is present in the leachate, S^{2-} will typically precipitate out of solution as $\text{FeS}_{(s)}$ or $\text{FeS}_{2(s)}$, resulting

in very low concentrations or total removal from the leachate (Appelo and Postma, 1999; Heron and Christensen, 1995).

Chloride behaves conservatively in natural environments and is relatively unaffected by the chemical and biological reactions associated with leachate generation. The initial concentration of chloride in landfill leachate is dependent the presence of soluble salts in the waste material. The availability of chloride as a leachate component slowly decreases over time due to continuous, long-term ‘flushing’ of the waste by infiltrating water (Ehrig, 1983; Andreottola and Cannas, 1992; Christensen *et al.*, 2001).

4.2.2 Trace elements

There are a large number of organic and inorganic elements present in landfill leachate at trace concentrations (operationally defined as less than 1 mg/L [Drever, 1988]) some of which are governed by stringent water quality criteria due to their toxicity. The presence of trace elements in leachate is dependent on the composition of the waste material in the landfill, although some trace elements may also be mobilised from aquifer solids in the reducing environment of a leachate plume. Some of the more commonly reported inorganic trace elements in landfill leachate include the heavy metals Cd, Cr, Cu, Ni, Pb, and Zn. Depending on the hydrogeochemical environment, some of the major ions are occasionally only present in trace concentrations as well, particularly the reduced forms of Fe, Mn and S.

4.2.3 Heavy metals

As discussed in the previous section, the heavy metals typically encountered in landfill leachates are Cd, Cr, Cu, Ni, Pb, and Zn (Ehrig, 1983; Christensen *et al.*, 2001). The metalloids As and Se are rare but have been occasionally reported (e.g. Kimmel and Braids, 1974). Heavy metals are typically present at extremely low concentrations or absent in landfill leachates, with very little difference observed between leachates from different waste decomposition phases (Ehrig, 1983 and 1988; Andreottola and Cannas, 1992; Kjeldsen and Christophersen, 2001; Christensen *et al.*, 2001; Kjeldsen *et al.*, 2002). While a significant quantity of metal waste may be present in the landfill, it is

estimated that only a small fraction is leached even after several decades (Baccini *et al.*, 1987; Ehrig, 1998).

There are several factors that influence this observed trend: the steadily rising pH of leachates from aging landfills is not conducive to the aggressive dissolution of metals, and actually promotes adsorption and precipitation of aqueous metal cations both within landfills and in aquifer settings if released to the environment. Metals in solution will readily precipitate as solid metal sulfides, carbonates, or (oxy)hydroxides in the presence of S^{2-} , HCO_3^- or hydroxyl groups, depending on the pH and redox state of the leachate plume. Metals also have a strong affinity for adsorption onto organic matter surfaces that comprises a large portion of the waste present in landfills, especially at the elevated pH values of methanogenic phase leachate (US EPA, 1992).

A growing body of research indicates an increase in the mobility of metals in a landfill leachate environment due to complexation with organic and inorganic ligands, as well as adsorption onto organic and inorganic colloids (US EPA, 1992; Gounaris *et al.*, 1993; Christensen *et al.*, 1996; Jensen and Christensen, 1999; Jensen *et al.*, 1999; Christensen *et al.*, 1999; Christensen and Christensen, 1999 and 2000). Several of these studies have indicated a significant, although highly variable, correlation between metals and colloidal fractions in leachate samples. Complexation with organic acid anions and inorganic ions (e.g. SO_4^{2-} , Cl^- , OH^- , PO_4^{3-} , NO_3^- and CO_3^{2-}) was also noted through laboratory experiments and speciation modelling. It should be noted that standard sample filtration procedures using a 0.45 μm filter would allow nearly all of the complexed and colloid-associated metals to pass, and subsequently to be operationally quantified as 'dissolved' metals (Horowitz *et al.*, 1996). This potential over-estimation of the dissolved fraction of metals species in leachate could have significant consequences with regards to contaminant transport calculations, since metal-complexes and colloid-bound metals may be much more mobile in an aquifer environment than ionic species that are prone to attenuation by adsorption and precipitation processes.

4.2.4 Dissolved organic matter

An enormous range of dissolved organic species have been reported in landfill leachate studies, ranging from volatile fatty acids (VFAs) to the more refractory humic- and fulvic-like compounds (Chian and DeWalle, 1977; Harmsen, 1983; Barker *et al.*, 1986; Weiss *et al.*, 1989; Christensen *et al.*, 1998; Kang *et al.*, 2002). These species represent intermediate degradation products of organic waste in the landfill.

Dissolved and colloidal matter in landfill leachate can have a significant influence on the mobilisation and attenuation of various contaminants. It acts as a substrate for microbially-mediated redox reactions, and may increase the mobility of heavy metals in solution through complexation with organic ligands and sorption onto organic colloids.

In addition to natural organic matter, a large range of anthropogenic organic chemicals have been reported in landfill leachates. Although these compounds comprise only a small fraction of the total DOC load in a typical leachate, they receive a disproportionate amount of attention due to their toxicity at low concentrations. Whilst these compounds have been the focus of substantial research, they are beyond the scope of this research and a review of literature has therefore not been undertaken.

4.3 REDOX PROCESSES AND ENVIRONMENTS

The dominant processes affecting the quality of leachate generated in a landfill are oxidation and reduction (redox) reactions. In the context of a landfill, these are primarily microbially mediated reactions in which organic matter is oxidized with the concomitant reduction of an 'oxidized' species (abiotic redox reactions, whilst present, are generally considered insignificant in comparison). The progression of redox processes that occurs as a landfill matures is used to define the phase of waste stabilisation. Furthermore, the specific redox environment present within a landfill acts as a fundamental control over all other chemical reactions associated with waste degradation and leachate generation. The following sections provide a general framework for the mechanics of redox reactions, and a more specific discussion of their role in the formation, evolution and environmental impact of landfill leachate.

4.3.1 Thermodynamic redox sequence

Redox reactions are reactions in which a reductant (or electron donor) is oxidized and an oxidant (or electron acceptor) is reduced through a transfer of electrons. In the context of a landfill, this process is facilitated by bacterial respiration, whereby the energy to support their physiological processes is sequestered from the flow of electrons (Bedient *et al.*, 1994; Chapelle *et al.*, 1995; Christensen *et al.*, 2000). Organic matter is the most abundant reducing agent in a landfill environment (and in most natural environments as well), while the distribution of oxidants is site specific.

Thermodynamic equilibrium dictates that, in a closed environment in which a surplus of reductant and a variety of oxidants are present, redox reactions will proceed in a sequence determined by the Gibbs free energy for the reactions (ΔG_r), from greatest to smallest energy yield (Stumm and Morgan, 1996). Assuming that the organic reductants involved are similar, the Gibbs free energy of the reactions, and thus the sequence of the reactions, is determined by the oxidant involved. Theoretically, if all the relevant oxidants are available, they will be preferentially consumed in the following order:



Once all of the oxidants listed above have been depleted, organic matter reduction may proceed by direct fermentation processes, including methanogenesis and nitrogen fixation, with end products of CH_4 and NH_4^+ , respectively. Table 4.2 lists the most prominent redox reactions that occur in landfill leachate plumes and their associated Gibbs free energy values at standard conditions of $pH = 7$ and $T = 25^\circ C$ (ΔG_r^0). In a closed, equilibrium system, the reactions with numerically larger values of ΔG_r^0 will proceed preferentially to those with numerically smaller values; the negative sign indicates that the reactions are exothermic.

Table 4.2 Prominent redox reactions that occur in landfill leachate plumes. Dissolved organic matter is represented by the generic form CH_2O (modified from Christensen *et al.*, 2001).

Process	Reaction	ΔG_r° (kcal/mol) ¹
Aerobic respiration	$\text{CH}_2\text{O} + \text{O}_2 \rightarrow \text{CO}_2 + \text{H}_2\text{O}$	-120
Denitrification	$5\text{CH}_2\text{O} + 4\text{NO}_3^- + 4\text{H}^+ \rightarrow \text{CO}_2 + 2\text{N}_2 + 7\text{H}_2\text{O}$	-114
Manganese reduction	$\text{CH}_2\text{O} + 2\text{MnO}_2 + 4\text{H}^+ \rightarrow \text{CO}_2 + 2\text{Mn}^{2+} + 3\text{H}_2\text{O}$	-81
Iron reduction	$\text{CH}_2\text{O} + 4\text{Fe}(\text{OH})_3 + 8\text{H}^+ \rightarrow \text{CO}_2 + 4\text{Fe}^{2+} + 11\text{H}_2\text{O}$	-28
Sulfate reduction	$2\text{CH}_2\text{O} + \text{SO}_4^{2-} + \text{H}^+ \rightarrow 2\text{CO}_2 + \text{HS}^- + 2\text{H}_2\text{O}$	-25
Methanogenesis	$2\text{CH}_2\text{O} \rightarrow \text{CH}_3\text{COOH} \rightarrow \text{CH}_4 + \text{CO}_2$	-22

¹ Gibbs free energy values taken from Champ *et al.* (1979).

The series of redox reactions that occur following the introduction of reduced leachate to an aerobic aquifer can be compared to a series of redox titrations, where the leachate is the titrator and the aquifer is the titrant (Christensen *et al.*, 2001). The aquifer is progressively depleted of electron acceptors, according to their availability and concentration in the aquifer and the reducing capacity of the leachate. Methanogenic conditions are established beneath and immediately downgradient from the landfill, characterised by elevated concentrations of CH_4 and NH_4 , followed by a SO_4^{2-} -reducing zone, an Fe(III)-reducing zone, an Mn(IV)-reducing zone, a NO_3^- -reducing zone, and finally a return to aerobic conditions when the concentration of DOC substrate in the plume is insufficient to cause the depletion of the free O_2 in groundwater (Figure 4.2) (Baedecker and Back, 1979b; Lyngkilde and Christensen, 1992a; Christensen *et al.*, 2001).

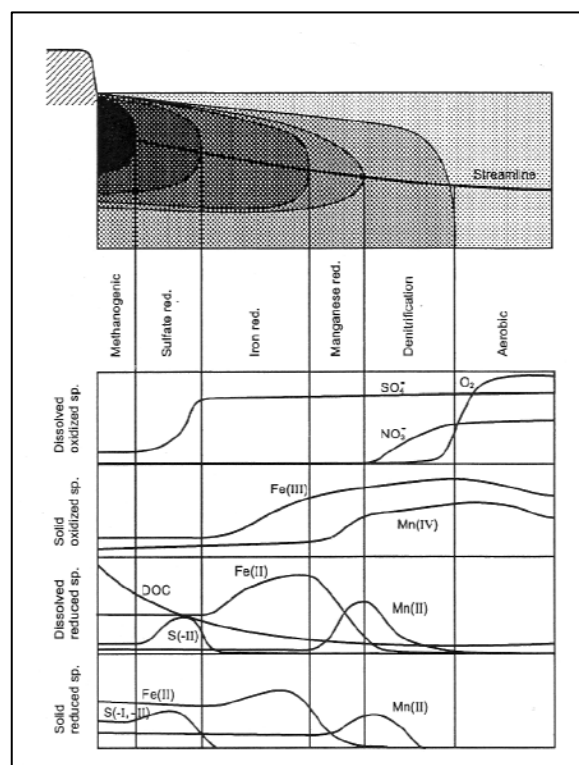


Figure 4.2 Theoretical redox zonation along a flow path downgradient from a landfill, and associated distribution of redox-sensitive species (modified from Christensen *et al.*, 2001).

While redox zonation is most often reported along flow paths downgradient from landfills, distinct vertical zonation has also been observed. An approximation of this sequence has been reported in literature for both natural and contaminated settings (Baedecker and Back, 1979a,b; Kimmel and Braids, 1980; Nicholson *et al.*, 1983; Baedecker and Apgar, 1984; Kehew and Passero, 1990; Lyngkilde and Christensen, 1992; Bjerg *et al.*, 1995; Jankowski and Acworth, 1997). However, it should be noted that the establishment of a specific redox zone is highly dependent on the availability of the specific electron acceptor in the aquifer (i.e. a discrete sulfate-reducing zone will not be established in the absence of available sulfate). Also, the physical distribution of redox zones may be affected by fluctuating water tables (vertical smearing) and variations in groundwater flow directions (MacFarlane *et al.*, 1983; Kjeldsen, 1993; Kjeldsen *et al.*, 1998b).

4.3.2 Concomitant redox reactions

In the case of microbially mediated redox reactions, redox zonation is thought to result from the dominance of a particular microbe that is best suited to the specific redox environment (based on its ability to most efficiently utilize the energy yield associated with specific redox reaction that it mediates), resulting in the competitive exclusion of all other microbial populations. For example, NO_3^- -reducing bacteria are dominant until the available NO_3^- has been depleted, at which point Mn(IV)- or Fe(III)-reducing bacteria proliferate. However, several studies have reported a significant overlap of redox zones, where evidence for concomitant methanogenesis and Fe(III) and SO_4^{2-} reduction have been observed (Bjerg *et al.*, 1995; Postma and Jakobsen, 1996; Jakobsen *et al.*, 1998; Jakobsen and Postma, 1999). It has been argued that redox reactions are best described as being in a state of partial equilibrium (Postma and Jakobsen, 1996; Jakobsen *et al.*, 1998; Jakobsen and Postma, 1999). The degradation of organic matter generally occurs in a series of steps, including: the hydrolyzation of organic matter, fermentation into simpler organic compounds, and finally consumption of these simpler compounds in terminal electron accepting processes (TEAPs) (Christensen *et al.*, 2000). One of the more important intermediate degradation products that is utilized during TEAPs is dissolved H_2 . The concentration of H_2 in groundwater is typically extremely low compared to the concentrations of other reactants that are involved in the TEAPs,

which has lead researchers to conclude that the fermentative step is rate limiting for these reactions, while the TEAPs controlling the H_2 concentration can be described as equilibrium reactions (and thus can be examined quantitatively using thermodynamic calculations; Postma and Jakobsen, 1996).

A recent research suggests that the ΔG_r yield for various TEAPs is highly dependent on the concentration and reactivity of the electron acceptors involved (as well as temperature and pH of the system); thermodynamic calculations have shown that concomitant occurrence of TEAPs associated with the oxidation of fermentation by-products (specifically H_2) can be favourable in a range of conditions (Postma and Jakobsen, 1996; Jakobsen *et al.*, 1998; Jakobsen and Postma, 1999). Another explanation is that end-member products from one redox zone may have migrated into an adjacent zone, and thus are not actually the result of simultaneously occurring processes (i.e. advective transport of CH_4 into a SO_4^{2-} reducing zone; Cozzarelli *et al.*, 2000). These factors must be given careful consideration when interpreting the results of groundwater samples from a leachate plume.

4.3.3 Redox buffering

The redox buffering capacity of an aquifer is best described by the titration analogy in section 4.5.1. The potential for leachate to donate electrons (reduction capacity, RDC) is balanced against the capacity for the aquifer system to accept electrons (oxidation capacity, OXC) (Barcelona and Holm, 1991; Heron *et al.*, 1994a,b; Heron and Christensen, 1995). The continued infiltration of leachate into the aquifer system will successively deplete electron acceptors from groundwater and aquifer solids, aiding in the formation of redox zones.

The RDC of landfill leachate is a function of the DOC concentration, although CH_4 and NH_4 may comprise a limited inorganic reductant source. In many environmental settings, the OXC of an aquifer is dominated by iron (and to a lesser extent manganese) oxides and hydroxides, as the abundance of these minerals per unit volume of aquifer is typically much greater than the concentration of other electron acceptors present in groundwater (Christensen *et al.*, 2001). It should be noted, however, that the availability

of Fe minerals as electron acceptors is dependent on the reactivity of the specific mineral; amorphous Fe-oxides are extremely reactive whereas Fe minerals with a crystalline structure such as magnetite and hematite are essentially non-reactive as electron acceptors (Postma, 1993; Heron, 1994b).

The aquifer mineralogy will be progressively altered due to the redistribution of Fe minerals involved in redox reactions (Heron and Christensen, 1995; Heron *et al.*, 1998). In the strongly reducing zones close to the landfill, the concentration of reactive Fe oxides and hydroxides is depleted as they are utilized in TEAPs. The resulting reduced iron (Fe^{2+}) goes into solution, and either migrates downgradient or precipitates as a reduced Fe mineral such as FeS , FeS_2 , or FeCO_3 , which increases the RDC of the aquifer solids in the vicinity of the landfill (Heron *et al.*, 1994b; Heron and Christensen, 1995). Soluble Fe^{2+} that does not precipitate in the reduced zone is transported downgradient until it encounters more oxidized zones, where it precipitates as highly reactive amorphous Fe hydroxides. This precipitation of Fe hydroxides at the distal end of the plume increases the OXC of the aquifer and may ultimately limit the size of the plume through redox buffering processes.

Research focusing on the quantification of both the OXC and RDC of aquifers in natural and contaminated settings indicates that the aquifer solids account for the vast majority of the oxidation or reduction capacity, while the contribution of oxidants and reductants in groundwater is relatively minor. This highlights the importance of Fe oxides and hydroxides as controls over the development of a leachate plume.

4.4 CARBONATE SYSTEM

The equilibrium between carbonate species in groundwater provides an important control over the pH of the system. This system is especially significant in the context of landfills since a large volume of CO_2 can be generated through the degradation of organic matter, which in turn has a significant effect on the solubility of carbonate minerals and precipitates. The carbonate species involved in the equilibrium relations are dissolved CO_2 , which by convention is quantified as carbonic acid (H_2CO_3), HCO_3^- , and CO_3^{2-} . Carbon dioxide that is introduced to the groundwater system, either through

contact with atmospheric CO₂ or from the oxidation of organic matter, will associate with water molecules to a certain extent to form H₂CO₃. Since H₂CO₃ is an acid it will tend to dissociate into HCO₃⁻, which in turn will dissociate into CO₃²⁻ under certain conditions. Table 4.3 lists these formation and dissociation reactions, their mass balance equations, and approximate equilibrium constants at 25°C (Appelo and Postma, 1999).

Table 4.3 Carbonate species reactions, mass action equations, and approximate equilibrium constants at 25°C (modified from Appelo and Postma, 1999).

$\text{CO}_{2(\text{aq})} + \text{H}_2\text{O} \rightarrow \text{H}_2\text{CO}_3$	$K_H = [\text{H}_2\text{CO}_3]/P_{\text{CO}_2}$	$K_H = 10^{-1.5}$
$\text{H}_2\text{CO}_3 \rightarrow \text{H}^+ + \text{HCO}_3^-$	$K_1 = [\text{H}^+][\text{HCO}_3^-]/[\text{H}_2\text{CO}_3]$	$K_1 = 10^{-6.3}$
$\text{HCO}_3^- \rightarrow \text{H}^+ + \text{CO}_3^{2-}$	$K_2 = [\text{H}^+][\text{CO}_3^{2-}]/[\text{HCO}_3^-]$	$K_2 = 10^{-10.3}$

In a landfill setting where CO₂ is being added to a closed system, the ratios of the various carbonate species remain constant as long as the pH remains constant. However, many anaerobic degradation processes consume H⁺ ions, which will in turn affect the distribution of carbonate species. In an anaerobic environment with a progressively increasing pH, HCO₃⁻ will become increasingly dominant while the P_{CO2} decreases. At 25°C, HCO₃⁻ is the dominant carbonate species between a pH of approximately 6.3 and 10.3. Only in rare cases will the pH increase to the point that CO₃²⁻ becomes significant.

4.4.1 Inorganic carbon - sources and sinks

In most uncontaminated aquifer systems, the primary sources of inorganic carbon are precipitation and soil water with elevated P_{CO2}, or the dissolution of carbonate minerals present in the aquifer. However, in a landfill setting the decomposition and mineralisation of organic carbon results in a significant increase in the total inorganic carbon (TIC) of the system, assuming that the landfill represents an essentially closed system where CO₂ and/or CH₄ does not off-gas. Depending on the pH of the system, this inorganic carbon will exist in solution as either dissolved carbonate species or CH₄, or will precipitate out of solution as carbonate minerals. At low pH values, an elevated P_{CO2} may result in aggressive dissolution of carbonate minerals present in the aquifer, increasing the HCO₃⁻ concentration.

4.4.2 pH buffering

As the TIC of a solution increases, so does its ability to buffer either the addition of a strong acid or base. The ability of a solution to buffer the addition of a strong acid is defined as *alkalinity*, while the ability of a solution to buffer the addition of a strong base is defined as *acidity*. In most groundwater system, the alkalinity is directly attributable to the HCO_3^- concentration, while the acidity is attributable to the P_{CO_2} . As the total concentration of these species increases, so does the alkalinity or acidity of the solution.

Leachate-impacted groundwater is often reported to have a pH of approximately 6.5, where the activity of H_2CO_3 and HCO_3^- are approximately equal. This situation occurs when the increased P_{CO_2} produced through organic decomposition is balanced by its conversion to HCO_3^- . This means that the surplus H^+ ions supplied through the dissociation of H_2CO_3 must be consumed through other hydrogeochemical processes active in the system. In many cases this is accomplished through dissolution of carbonate minerals present in the aquifer; in aquifers with relatively minor components of carbonate minerals, anaerobic oxidation of organic matter may be the primary buffering reaction. The strong buffering potential of landfill leachate is a function of both elevated TIC and chemical and biological processes capable of maintaining a balance between H_2CO_3 and HCO_3^- activities.

4.5 HYDROGEOCHEMICAL PROCESSES AFFECTING LEACHATE ATTENUATION

Once landfill leachate is released to an aquifer system, a series of physical, chemical and biological processes occur which act to attenuate the contaminant concentration in groundwater along a flow path. The following sections provide a review of the fate of landfill leachate constituents following release to the environment, with emphasis placed on significant processes in this research.

4.5.1 Dilution

Dilution is a general term for the reduction of contaminant concentration through the distribution of contaminants throughout larger volumes of water. This occurs as leachate flow interacts with groundwater flow in an aquifer, forming a leachate plume that develops in three dimensions as it migrates away from the landfill. While the process of dilution ultimately decreases the concentration of contaminants in groundwater, the result is a larger volume of impacted groundwater, albeit to a lesser extent. All contaminants present in a leachate plume are subjected to dilution as they migrate from the landfill; for conservative elements such as chloride, dilution constitutes the only attenuation mechanism.

4.5.1.1 *Mixing*

The simplest dilution mechanism is mixing of leachate flow from the landfill with groundwater flow in the aquifer. The most significant factors affecting the degree of mixing of leachate and groundwater include fluctuations in local groundwater gradients in and around landfills due to groundwater mounding (MacFarlane *et al.*, 1983; Jankowski and Knight, 1991; Kjeldsen, 1993; Kjeldsen *et al.*, 1998b), density differences between leachate and groundwater (Kimmel and Braids, 1980; MacFarlane *et al.*, 1983; Kjeldsen *et al.*, 1998b), and heterogeneities present in the aquifer (Back and Baedecker, 1989; Christensen *et al.*, 2001). For simple mixing, the relative ratio of the two solution types being mixed (i.e. leachate and groundwater) will determine the degree of dilution of the contaminant.

4.5.1.2 *Hydrodynamic Dispersion*

Hydrodynamic dispersion is an attenuation process that occurs as a result of a solution moving through a porous medium, and is comprised of both mechanical dispersion and diffusion (Fetter, 1994). Mechanical dispersion results from variations in the velocity of solute transport, due to variations in actual flow path lengths, friction effects on grain surfaces, and pore throat diameters. Various large-scale tracer tests have been undertaken to assess the effects of dispersivity on solute transport in a contaminant plume (eg. Sudicky *et al.*, 1983; Garabedian *et al.*, 1991; Gelhar *et al.*, 1992), and have found a significant scale dependency (i.e. plume length). The process of dispersion is

largely responsible for the development of the ‘plume’ shape of contaminants in groundwater that are released from a concentrated point source.

Diffusion is defined as the movement of dissolved chemical constituents from areas of higher concentrations to areas of lower concentrations based solely on the concentration gradient (Fetter, 1994). This migration of chemical species is due to random movement caused by the thermal kinetic energy of the solute, and does not require the physical movement of groundwater to occur. Because of the slow kinetics of diffusion in a porous medium, it is normally insignificant compared to dispersion in terms of solute mass transport.

The attenuation value of dispersion (dispersivity) has been determined through various field studies to vary with plume length (eg. Gehlar et al. 1992; Xu and Eckstein, 1995), and has been demonstrated to vary between 2 to 3 orders of magnitude for a given plume length. With respect to the leachate plume at the Astrolabe Park landfill, the relatively short groundwater flow path between the landfill and Lachlan Pond No.5 means that contaminant attenuation through mechanical dispersion will only have a limited influence before the leachate reaches the pond. Likewise, attenuation of contaminants by diffusion is considered to be insignificant due to the relatively large average groundwater velocity.

4.5.2 Degradation

In the context of a landfill leachate plume, degradation refers to the microbially mediated oxidation of dissolved or particulate organic matter into benign end members. As discussed earlier, this is the process that is responsible for establishing redox zones within the leachate plume, and has a significant influence on many other attenuation mechanisms.

The Astrolabe Park landfill, which has been decommissioned for nearly 30 years, is well into the methanogenic phase of waste stabilisation. The DOC content of the landfill leachate would consist of the refractory organic carbon compounds characteristic of methanogenic phase leachate, and it is unlikely that substantial degradation occurs along

the short flow path between the landfill and Lachlan Pond No.5. The most significant effect of the DOC content of the leachate plume, in conjunction with the naturally occurring peat deposits and potential anaerobic oxidation of CH_4 , would be to provide an adequate substrate to maintain a strongly reducing redox environment between the landfill and the pond.

4.5.3 Mineral dissolution/precipitation

The dissolution of aquifer minerals and precipitation of aqueous species are intimately related to the redox processes occurring in a leachate plume. Dissolution and precipitation reactions are important mechanisms for both the mobilization and attenuation of major cations and anions from solution, and are particularly important with respect to Ca^{2+} , Mg^{2+} , Fe^{2+} , Mn^{2+} , HCO_3^- , and S^{2-} . The elevated P_{CO_2} of the landfill leachate promotes enhanced dissolution of carbonate minerals present in the aquifer, resulting in an increase in Ca^{2+} , Mg^{2+} and HCO_3^- concentrations in solution. Reactive iron (and to a lesser extent Mn^{2+}) (oxy)hydroxides present as grain coatings in the aquifer material are important electron acceptors in microbially mediated degradation reactions and are highly susceptible to reductive dissolution within the leachate plume.

The distribution of aqueous redox species in the leachate plume at the Astrolabe Park landfill suggests that both iron and sulphate reduction is occurring, and general absence of heavy metals in water samples (and substantial presence in soil samples) suggests that precipitation as metal sulfides and carbonates is an important process in the plume.

4.5.4 Complexation

Complexation can act as either a mobilisation or an attenuation process, depending on the type of complex formed. Complexation is the formation of complexes through the combination of cations, anions and dissolved organic molecules (US EPA, 1992). The main impact of complex formation in the context of a contaminant plume is the increased mobility of metals, as metal complexes can remain soluble in conditions that normally would result in their precipitation. The involvement of major cations in

complexation reactions can also decrease the saturation indices for some minerals, enabling enhanced dissolution of the minerals.

The one situation where complexation acts as an attenuation mechanism is in the case of surface complexation of soluble cations onto aquifer solids. This process is included in the broader discussion of sorption reactions in the following section.

4.5.5 Sorption reactions and element partitioning

Sorption processes refer to a number of surface-related physical and chemical reactions responsible for binding dissolved species to solid phase minerals and organic matter, including: adsorption, absorption, chemisorption, surface complexation, and ion exchange (Fetter, 1999). For lack of a method to separate the individual contributions of each type, adsorption, absorption, chemisorption and surface complexation will be referred to under the blanket term adsorption, and ion exchange will be discussed separately.

4.5.5.1 Adsorption

The dissolved elements most affected by sorption processes include the heavy metals (Cd, Cr, Cu, Pb, Ni, Zn) and DOC (both as degradation intermediates from the waste material and trace organic contaminants). In general, dissolved species have a much higher affinity for sorption onto solid organic matter than mineral surfaces, and will partition almost exclusively onto organic carbon surfaces if the organic carbon fraction of the aquifer material is greater than 1% by weight (US EPA, 1992; Fetter, 1999). While adsorption can provide a significant attenuation mechanism for heavy metals, adsorption onto the surfaces of organic colloids can actually enhance the mobility of metals in solution by eliminating their susceptibility to precipitation or adsorption onto immobile aquifer organic matter (Jensen and Christensen, 1999; Jensen *et al.*, 1999). Evidence for this transport mechanism appears to be present at the Astrolabe Park landfill, as supported by the elevated concentrations of metals in the organic sediments along the discharge zone of the leachate plume into Lachlan Pond No.5 in contrast to the nearly undetectable concentrations in the leachate plume.

Sorption processes are enhanced at higher pH values, and there has been speculation that sorbed metals that have accumulated on organic and mineral surfaces in the near neutral pH range of many leachate plumes may eventually re-mobilize if environmental conditions change and the system becomes more acidic. However, because there are no reported cases of leachate plumes that have attenuated beyond the methanogenic phase of waste stabilisation there is no field evidence to support this idea.

4.5.5.2 Ion exchange

Ion exchange operates on the premise that the weak electrical charge on the surfaces of clay minerals attracts and binds ions of the opposite charge. Because clay surfaces typically exhibit a negative electrical charge in the near neutral pH range of most natural waters, ion exchange pertains almost exclusively to cations, although anion exchange is also possible in acidic conditions (US EPA, 1992; Thornton *et al.*, 2000). Christensen *et al.* (2001) calculated that in a pristine aquifer, cations associated with exchange sites generally comprise 80% of the total cation distribution per volume of aquifer, while in the presence of high ionic strength leachate the percentage is closer to 40% (calculated from data published by Nicholson *et al.*, 1983).

In a natural setting in the absence of drastic water chemistry fluctuations, the ion exchange sites will typically be saturated with whichever ion is dominant for that system (quite often Ca^{2+} or Na^+). As a rule of thumb, the greater the charge on the ion, the greater the affinity for exchange, with the generally accepted pecking order as follows (Domenico and Schwartz, 1998):



The introduction of landfill leachate to a pristine aquifer drastically alters the chemical composition of the system, and typically results in the chromatographic displacement of the base saturating cations in the pristine material and replacement with a preferential exchanger from the leachate. The two major cations typically introduced in landfill leachate plumes that display the greatest affinity for exchange sites are K^+ and NH_4^+ (eg Lyngkilde and Christensen, 1992a; Erskine, 2000; Thornton *et al.*, 2000). Potassium is generally only present in low concentrations to begin with, and is not considered a

serious contamination concern, but NH_4^+ is considered a critical long-term contaminant in landfill leachate plumes and cation exchange can provide a significant attenuation mechanism (Erskine, 2000; Thornton *et al.*, 2000). At the Astrolabe Park landfill, the short migration path, and thus residence time in the aquifer, between the landfill and Lachlan Pond No.5 precludes significant attenuation of NH_4^+ via anaerobic oxidation. The presence of heterogeneous clay lenses dispersed throughout the sand deposits may comprise a more significant attenuation mechanism for NH_4^+ in this environmental setting.

4.5.6 Redox transformation

Redox reactions associated with the degradation of dissolved organic compounds in a leachate plume result in the transformation of electron acceptors involved in the reactions. The major elements that are typically involved in these reactions are: $\text{NO}_3^-/\text{NH}_4^+$, Mn(IV)-oxide/ Mn^{2+} , Fe(III)-oxide/ Fe^{2+} , $\text{SO}_4^{2-}/\text{S}^{2-}$, and CO_2/CH_4 . Redox transformations can result in both the mobilisation and attenuation of leachate contaminants, depending on the behaviour of the contaminants in different redox environments. Many of the heavy metals that are commonly associated with immobile mineral oxide phases in pristine oxidised aquifers will become soluble in the strongly reduced environment of a leachate plume. Nitrate present in the natural aquifer may be reduced to NH_4^+ through redox transformations and contribute to the total nutrient load of the plume. In terms of attenuation, the presence of reduced S in groundwater will typically precipitate all heavy metals as sulfide minerals. Ferrous iron present in groundwater may be transported to less reducing zones of the plume and get oxidized to highly reactive amorphous iron oxide, thus increasing the oxidation capacity of the aquifer. Another important transformation is the mineralisation of organic carbon, although this process most accurately falls under the degradation section. The progressive mineralisation of organic carbon in the leachate plume is responsible for the attenuation of one of the most persistent contaminants associated with landfill leachate.

With respect to contaminant attenuation (or at least retardation) in the leachate plume at Astrolabe Park, the most significant element transformation is probably the reduction of SO_4^{2-} to S^{2-} , which enables the precipitation of heavy metals from solution as metal

sulfides. On the fringes of the plume, denitrification and subsequent escape of N_2 gas may facilitate a slight reduction in the NH_4^+ concentration, but the majority of the plume remains strongly reducing throughout the entire flow path between the landfill and Lachlan Pond No.5.

4.6 Scales of variation in time and space

The detailed research of contaminant plumes in groundwater over the past few decades has underscored the importance of the scale of investigation (eg. Back and Baedecker, 1989; Back et al, 1993; Cozzarelli et al., 1999). Contaminant plumes are dynamic entities in time and space, with variation observed in contaminant concentration and distribution in response to fluctuating hydrogeologic conditions and hydrogeochemical environments. To appropriately address the dynamic nature of the processes affecting solute transport in a contaminant plume, it is necessary to design research programs on a physical scale that is adequate to elucidate the intricate variations produced by physical, chemical and biological processes that are active within the plume (Ronen et al, 1987; Back and Baedecker, 1989; Smith *et al.*, 1991b; Bjerg and Christensen, 1992; Thompson, 1993; Thompson and Jackson, 1996; Thompson et al., 1996; Webb and Anderson, 1996).

Heterogeneities in a vertical profile are especially significant in a sedimentary aquifer due to the horizontal disposition of the sedimentary deposits and limited transverse dispersivity associated with advective solute transport (Sudicky, 1986; Bjerg and Christensen, 1992; Bjerg et al., 1992; Thompson, 1993; Thompson and Jackson, 1996; Thompson et al., 1996). Steep vertical chemical gradients have been observed in heterogeneous aquifers on the order of a few to tens of centimetres (Baedecker and Back, 1979b; Ronen et al., 1987; Bjerg and Christensen, 1992; Bjerg et al., 1992; Jorstad et al, 2004, Acworth and Jorstad, 2006). Examples of small-scale vertical zonation of groundwater chemistry have been well documented in the Botany Sands aquifer (Jankowski and Acworth, 1997; Jankowski et al., 1997; Lavitt et al., 1997; Jankowski and Beck, 1998; Jankowski and Beck, 2000, Jorstad et. al, 2004).

It is important to note that even the most detailed sampling event only represents a 'snapshot in time' of the current plume configuration, and that the true nature of

variation occurring within the plume requires detailed data collection over small increments of time as well as space. A number of detailed, large-scale research investigations have been performed at various unlined landfills in North America and Europe to determine the mechanics of contaminant transport and attenuation for leachate that has been released to the environment; an excellent review of the most prominent research investigations is provided by Christensen *et al.* (1994, 2001). The level of detail undertaken by each of these investigations varies greatly, reflecting differences in project scope, budget, and advances in analytical techniques and general understanding of contaminant transport and attenuation mechanisms related to landfill leachate. However, nearly all of the investigations constitute isolated ‘snapshots in time’ of leachate plume configurations in the subsurface, and have not adequately addressed the potential variations in plume configuration across relatively short time intervals. In some cases follow up sampling events have been conducted, but on time scales that are only appropriate for monitoring long-term trends. A rigorous understanding of the potential physical and chemical fluctuations within a leachate plume over short time intervals is fundamental to many aspects of an environmental investigation, including: demonstration of the knowledge of the maximum lateral and vertical extent of the plume boundaries; understanding of the physical, chemical and biological processes associated with the natural attenuation of landfill leachate pollutants, and how these processes vary over time; and a knowledge and possible prediction of potential fluctuations in contaminant concentrations in order to optimise remediation activities or determine the risk to sensitive receptors posed by fluctuations in the quality and quantity of leachate in the plume.

A significant objective of the research at the Astrolabe Park landfill has been to assess the variation that occurs within the leachate plume over relatively short timescales, and to determine which hydrogeochemical processes are primarily responsible for the observed variations. The use of a landfill leachate plume for this research is particularly useful because of the broad range of constituents typically present (eg. elevated inorganic and organic elements, heavy metals, nutrients), and the variety of hydrogeochemical processes that affect the mobilisation and attenuation of these constituents.

CHAPTER 5: METHODOLOGY

In general terms, the research associated with this project can be divided into the categories of fieldwork, laboratory work, and data analysis. Each of these broad categories is comprised of numerous tasks; each task was conducted in accordance with a developed set of methods to ensure the highest quality results were achieved within the practical and financial constraints of the project. The adopted methodology for each phase of the research is described in detail in the following sections.

5.1 PIEZOMETER CONSTRUCTION AND INSTALLATION

A network of newly installed piezometers was used to perform this research, as well as the existing bundled piezometer 10 located towards the southwest corner of the Astrolabe Park landfill (Figure 5.1). The new piezometers include both conventional and bundled designs, as described below.

5.1.1 Selection of piezometer installation locations

The locations of the new piezometers were chosen to provide data coverage both within and downgradient from the landfill, with an emphasis placed on water quality along the shore of Lachlan Pond No. 5 to aid in the determination of the hydraulic relationship between the groundwater and surface water. All of the piezometers were installed within the aeolian sand aquifer, to various depths depending on the specific function of the piezometer.

In June 2000 the initial suite of piezometers was installed. Two conventionally designed piezometers (piezometers 1 and 2) were installed directly in the landfill to provide source concentration data for the leachate plume. Both were installed in Area A of the study area (Figure 5.1); piezometer 1 was installed towards the southern end of the study site, and piezometer 2 was installed close to the southern boundary of Astrolabe Park. The piezometers were installed to a depth of 9 mbgs, to intercept the centre of mass of the landfill leachate plume.

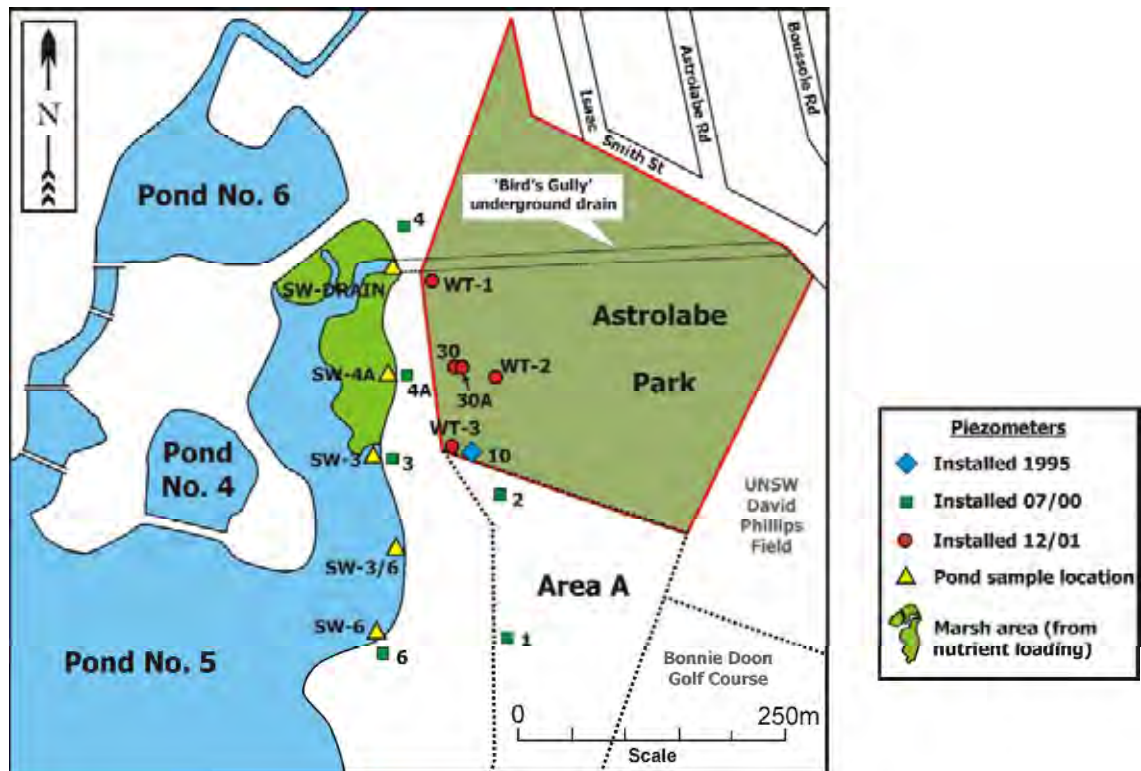


Figure 5.1 Location of piezometers and pond sample points used in conjunction with this investigation.

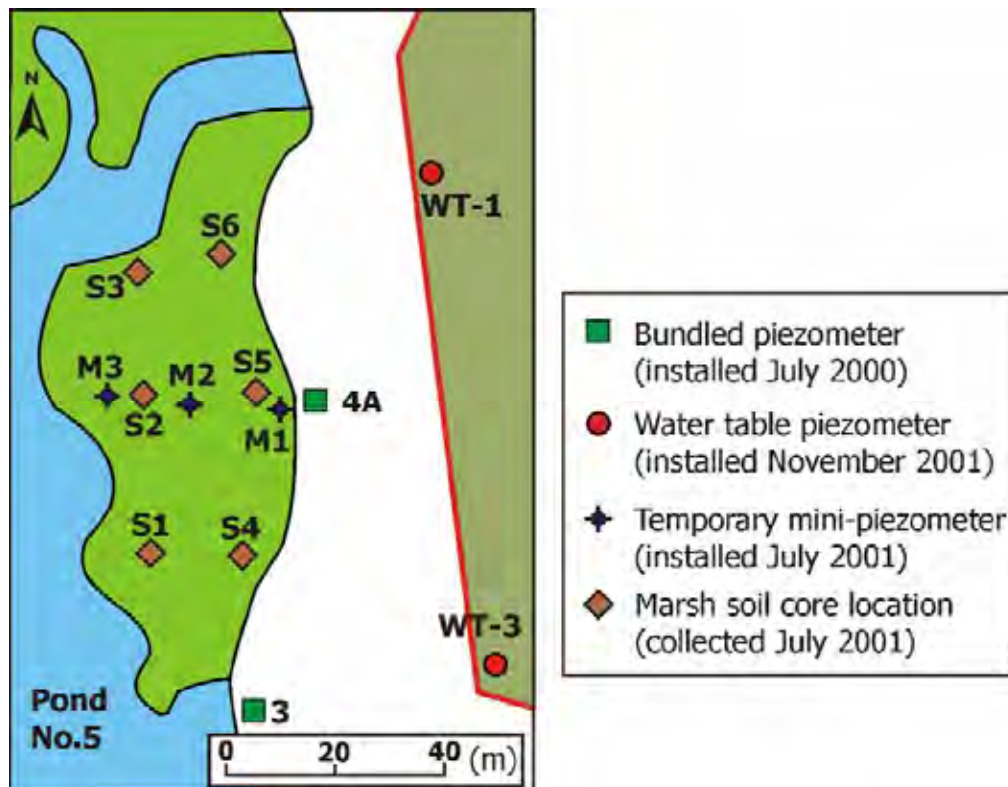


Figure 5.2 Location of mini-piezometers and soil samples in marsh.

Four bundled piezometers were installed along the shore of Lachlan Pond No. 5, spanning the stretch of shoreline that is down hydraulic gradient from Astrolabe Park and Area A (Figure 5.1). These piezometers were installed, from south to north, as follows: piezometer 6, at the far southern end of the research site; piezometer 3, downgradient from the southern end of Astrolabe Park; piezometer 4A, downgradient from the mid-section of Astrolabe Park; and piezometer 4, downgradient from the northern portion of Astrolabe Park. These bundled piezometers were installed at locations where nested piezometers from previous research had been installed and subsequently were buried and lost, and also where cone penetrometer tests had been conducted in conjunction with previous research, in order to provide the maximum amount of background data necessary to optimise the design and construction of the new bundled piezometers. The piezometers were installed to a depth of 20 mbgs, such that they fully penetrated the leachate plume and extended below the plume into uncontaminated regional groundwater.

In December 2001 an additional eight bores were installed in the southwestern corner of Astrolabe Park to provide a field training facility for a groundwater short course. Some of these bores were also used to collect data for this research, and are summarized as follows (the bores from this installation event that were used for the investigation are included in Figure 5.1):

- One bundled piezometer (30) located approximately 50 m north of piezometer 10, installed to 15 mbgs to target a historical area of elevated leachate concentration,
- Two piezometers installed four metres to the north and south of piezometer 30, to a depth of 15 mbgs, equipped with electrodes and leads for performing cross-borehole geophysical surveys,
- One conventional piezometer (30A) installed one meter east of piezometer 30 to a depth of 9 mbgs, to provide a means for demonstration of pumping influences on head values in the bundled piezometer,
- One conventional piezometer located approximately half way between piezometer 10 and 30 for borehole geophysical surveys, and

- Three shallow conventional piezometers arrayed as a triangle around the research site, installed to a depth of 6 mbgs for the determination of the water table depth and the hydraulic gradient between the landfill and the pond.

In addition, three temporary mini-piezometers were installed in the marsh along a transect perpendicular to the shoreline, starting just west of piezometer 4A and extending to the far edge of the marsh area (M-1,2,3: Figure 5.2). The mini-piezometers consisted of 40mm diameter, 1.5m long PVC casing with slots sawed into the bottom 30cm. The screened interval was covered with nylon mesh to keep sediment from entering the casing. The piezometers were installed to a total depth of one metre using a hand auger and allowing the hole to collapse back onto the piezometer following installation. These temporary mini-piezometers were used during the July 2001 sample event to determine whether contaminated groundwater was discharging into the marsh.

The transects defined by piezometers 10 and 3, and piezometers 30 and 4A, comprise two parallel groundwater flow paths between the edge of the Astrolabe Park landfill and the shore of Lachlan Pond No. 5; data from these piezometers are used to assess the migration of contaminants between the landfill and the pond. It should be noted that local groundwater flow at the research site (west-southwest) varies from the regional groundwater flow direction (south-southeast) due to the influence of surface topography and the hydraulic interconnection with the Lachlan Ponds. Refer to Figures 6.1 to 6.3 in Chapter 6 for a detailed assessment of local groundwater flow at the research site.

5.1.2 Design and construction of bundled piezometers

The bundled piezometers installed at the research site are based on a design outlined by Cherry (1983). This piezometer design is intended to provide detailed, depth-discrete groundwater data across small vertical increments along a profile. The design consists of a central 50mm diameter polyvinyl chloride (PVC) casing to which 26 sample collection tubes are attached (Figure 5.3). A 0.5m section of 0.020-inch (0.5mm) factory slotted PVC well screen is attached to the bottom of the casing to facilitate the piezometer installation and allow for monitoring of hydraulic head at depth. The sample

tubes are constructed from flexible PVC tubing (3mm ID, 5mm OD) measured out to a predetermined depth, that terminate in a sample point.

Table 5.1 Construction details for bundled piezometers.

Piezometer ID:	3	4	4A	6	10	30
TOC¹ elevation (m,AHD²)	15.826	17.654	15.871	15.929	16.628	15.452
Sample Point Depths (mbgs³)	1.90	1.84	2.15	1.68	3.00	1.50
	2.40	2.34	2.65	2.68	4.00	2.00
	2.90	2.84	3.15	3.68	5.00	3.00
	3.40	3.34	3.65	4.68	6.00	3.50
	3.90	3.84	4.15	5.18	7.00	4.00
	4.40	4.34	4.65	5.68	8.00	4.50
	4.90	4.84	5.15	6.18	9.00	5.00
	5.40	5.34	5.65	6.68	10.00	5.50
	5.90	5.84	6.15	7.18	11.00	6.00
	6.40	6.34	6.65	7.68	12.00	6.50
	6.90	6.84	7.15	8.18	13.00	7.00
	7.40	7.34	7.65	8.68	14.00	7.50
	7.90	7.84	8.15	9.18	15.00	8.00
	8.40	8.34	8.65	9.68	16.00	8.50
	8.90	8.84	9.15	10.18	18.00	9.00
	9.40	9.34	9.65	10.68	20.00	9.50
	10.40	9.84	10.15	11.18	22.00	10.00
	11.40	10.34	11.15	11.68	24.00	10.50
	12.40	10.84	12.15	12.18	26.00	11.00
	13.40	11.34	13.15	12.68	28.00	11.50
	14.40	12.34	14.15	13.18	30.00	12.00
	15.40	13.34	15.15	13.68		12.50
	16.40	14.34	16.15	14.18		13.00
	17.40	15.34	17.15	15.18		14.00
	18.40	16.34	18.15	16.18		15.00
	19.40	18.34	19.15	18.18		

Notes:

1. TOC = top of casing
2. AHD = Australian height datum
3. mbgs = metres below ground surface

The sample points are constructed by perforating a 5cm interval at the down-hole end of the sample collection tube, inserting synthetic filter material (in this case either nylon

aquarium filter or strips of nylon scrubbing pads) into the perforated portion of the tube, and finally covering the perforations with a sleeve of synthetic mesh material (in this case sections of nylon shoe laces) to minimize the potential for clogging by entrainment of fine grained particles in the tubes. The terminal depths for each of the sample points was decided prior to installation through a review of previous water quality data for each of the piezometer installation locations. Sample points were spaced at 0.5m increments across the vertical section thought to define the maximum vertical extent of the leachate plume within the aquifer, and then at 1.0m intervals for approximately 10m below the plume in the fresh regional groundwater. The sample collection tubes were attached to the central PVC pipe at the pre-defined intervals using electrical tape prior to installation.

Engineering Exploration Pty Ltd was contracted to perform the piezometer installations using a drill rig equipped with hollow stem augers (Figure 5.4). The boreholes were advanced to approximately 20m bgs; in the saturated zone water was pumped through the hollow stem of the augers to keep the flowing formation sand from entering the augers. Once the desired depth was reached the piezometers were installed through the centre of the hollow stem augers, the augers were retracted, and the formation was allowed to collapse around the sample points. Any remaining annular space around the piezometer at the top of the boreholes was backfilled with sand to a minimum of 1m above the shallowest sample point; the boreholes were then sealed with both bentonite slurry and grout, and a locking steel monument was installed over the top of the piezometers as the surface completion.

Following installation of the piezometers, the top of the well casing of each piezometer was surveyed to millimetre accuracy for use in potentiometric analysis and to aid in correlation of water chemistry data between piezometers (Table 5.1).

5.1.3 Design and construction of conventional piezometers

The conventional piezometers are constructed from 50mm diameter PVC casing with a pre-determined length of 0.010-inch (0.25mm) factory slotted PVC well screen attached to the bottom. Piezometers 1 and 2 were installed to 9m bgs using a hollow stem auger

rig (as described in the previous section), with a screened interval from 6-9m bgs. The shallow water table piezometers (WT-1,2,3) were installed to 6m bgs, with a screened interval from 4-6m bgs; these well screens are designed to intercept or be just below the water table. The surface of the piezometer boreholes were sealed with bentonite slurry and grout, and were completed with steel well boxes flush to ground surface. The conventional piezometers were used both to collect groundwater samples and for potentiometric analysis.

Following installation of the piezometers, the top of the well casing of each piezometer was surveyed to millimetre accuracy for use in potentiometric analysis and to aid in correlation of water chemistry data between piezometers (Table 5.2).

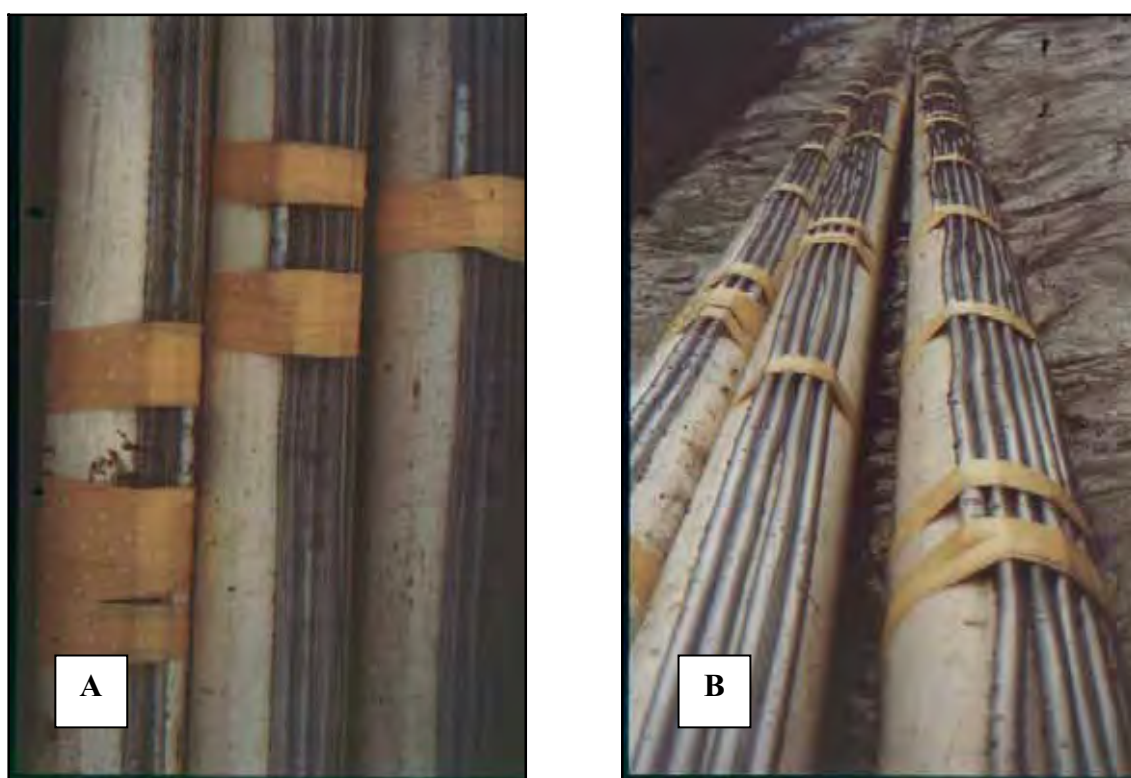


Figure 5.3 Bundled piezometers prior to installation; (a) sample point construction, and (b) piezometer sections with sample tubes attached.

Table 5.2 Construction details for conventional piezometers

Bore ID	Total depth of casing (m)	TOC ¹ Elevation (m, AHD ²)	Screened Interval (mbgs ³)	Screened Interval (m, AHD)
WT-1	6	17.459	4 to 6	13.46 to 11.46
WT-2	6	17.624	4 to 6	13.62 to 11.62
WT-3	6	16.412	4 to 6	12.41 to 10.41
30A	9	15.506	8 to 9	7.51 to 6.51
1	9	18.160	6 to 9	12.16 to 9.16
2	9	18.458	6 to 9	12.46 to 9.46

Notes:

1. TOC = top of casing
2. AHD = Australian Height Datum
3. mbgs = metres below ground surface

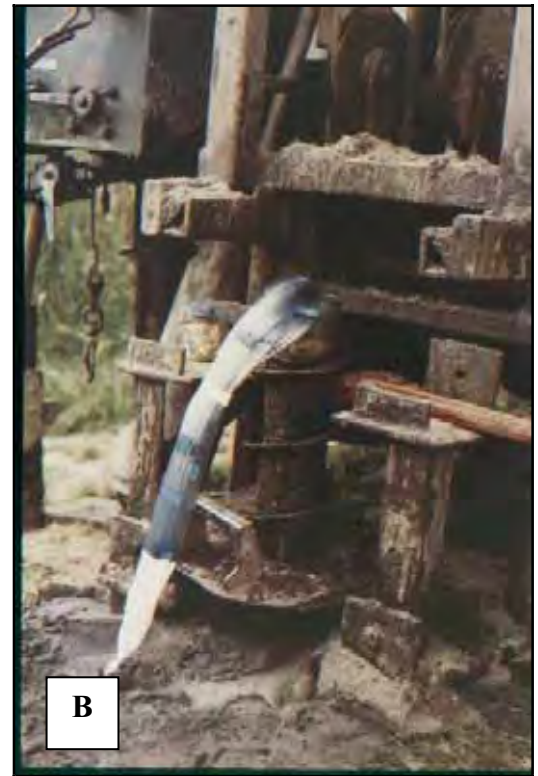


Figure 5.4 Bundled piezometer installation; (a) installing piezometer sections through hollow-stem augers, and (b) augers ready for retraction.

5.2 HYDROGEOLOGICAL INVESTIGATION

Throughout the research period, hydrogeological data were collected in conjunction with water chemistry data to aid in the determination of the relative influence of both physical and chemical process with respect to contaminant transport from the landfill, and especially to investigate the hydraulic connection between groundwater and surface water along the eastern shore of Pond No.5. The hydrogeological investigation consisted of slug tests to determine the hydraulic conductivity of the aquifer material at various depths, standard water level measurements to monitor changes in the potentiometric surface over time, and use of an innovative approach for measuring hydraulic head in the small sample tubes of the bundled piezometers.

5.2.1 Water level measurement

Water level measurements were collected during each of the monitoring events to assess the spatial and temporal variation in hydraulic head distribution (both lateral and vertical). The methodology for water level measurements for both standard and bundled piezometers differs significantly, as described in the following sections.

5.2.1.1 *Standard piezometers*

The depth to water was measured in each of the standard piezometers during each sampling event using an electric sounder. The depth to water measurements were normalised to a common datum by subtracting from the surveyed ‘top of casing’ elevations to translate the data to hydraulic head values, which can then be used to assess groundwater flow directions and vertical gradients across the site.

5.2.1.2 *Bundled piezometers*

Conventional water level measurements were not possible for the small-diameter sample tubes of bundled piezometers. However, it is desirable to perform this type of measurement as it would provide highly detailed hydraulic head information, particularly if correlated with hydraulic head data from a proximal bundled piezometer. Measurements of this type have been made using homemade electric sounders consisting of just two wires that fit inside the small diameter sample tubes. However,

these measurements are prone to ‘false alarms’, which may be caused by small water droplets that rise above the standing water level in the sample tubes due to capillary action, or the presence of trapped gas bubbles that result in a broken water column in the small tubes. These types of ‘false alarms’ can result in errors on the order of several centimetres, which can be extremely significant for sites with naturally subtle hydraulic gradients.

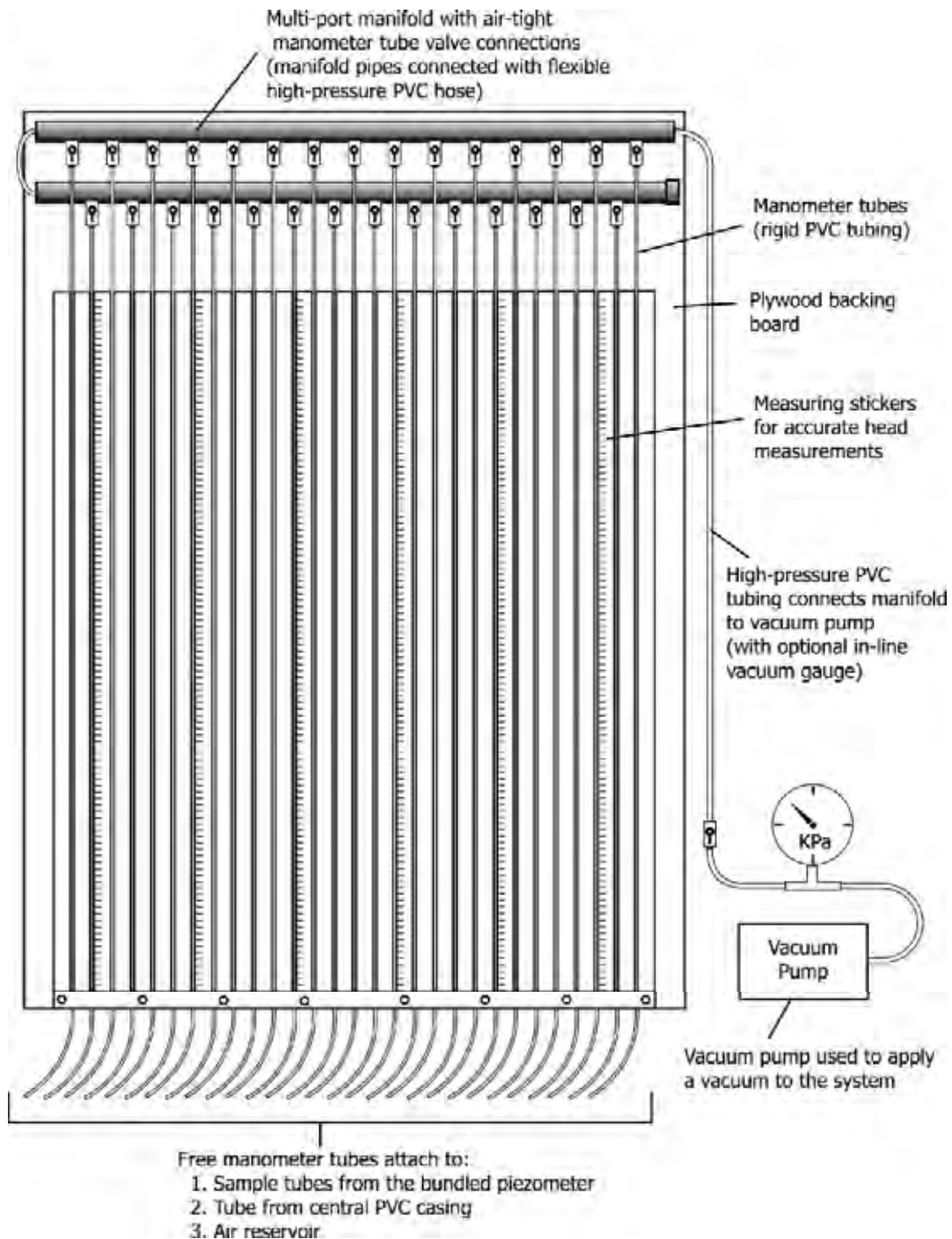


Figure 5.5 Schematic diagram of the multi-channel manometer board.

To resolve this technical problem, a multi-channel manometer board was used which was developed by researchers working in the UNSW Water Research Lab in Manly Vale, NSW. The manometer board consists of a series of small-diameter, rigid plastic tubes that are mounted next to each other on a backing board, with measuring tape stickers placed between the tubes for measurement of head to millimetre accuracy (Figure 5.5). These small tubes are all connected to an airtight multi-port manifold at the top of the board, which in turn is attached to a vacuum pump.

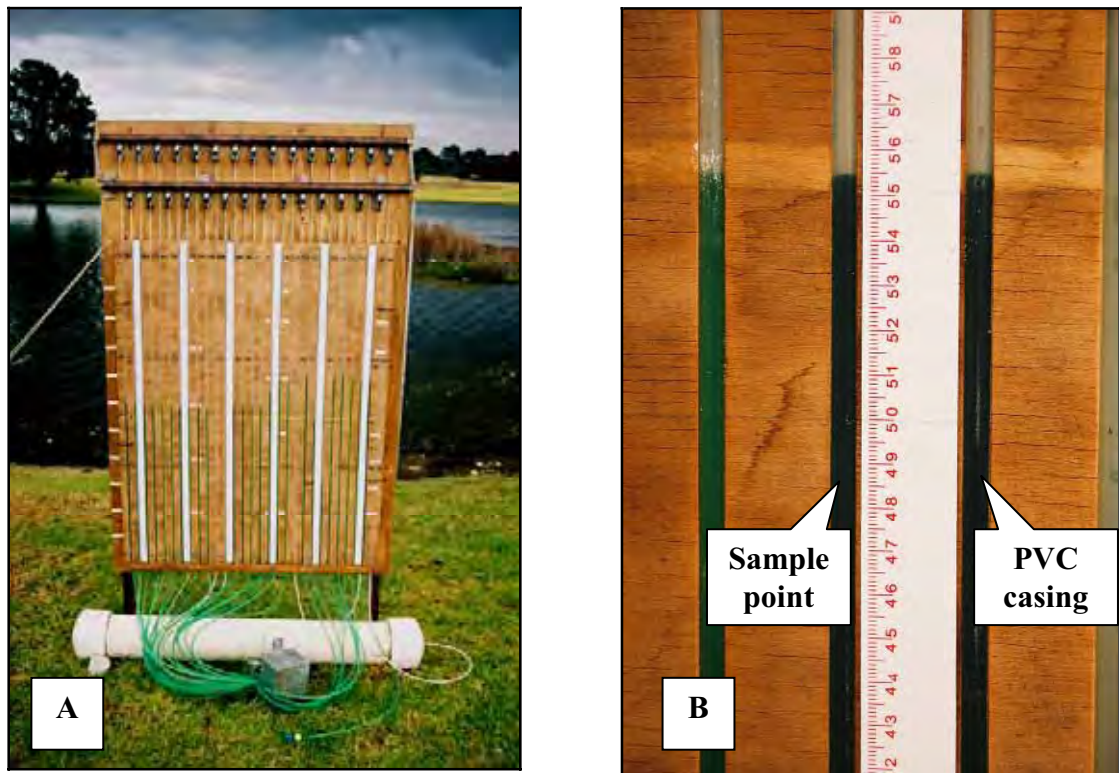


Figure 5.6 Photographs of the manometer board (a) equilibrating at piezometer 3, and (b) demonstration of the correlation between water levels for the PVC casing and adjacent sample point at piezometer 10.

The sample tubes from the bundled piezometer are attached to the tubes on the manometer board (in consecutive order of increasing depth for ease of comparison) and a vacuum is applied to the closed system. The vacuum is evenly distributed across all sample points and the water rises through the tubes and onto the manometer board (green food colouring was introduced in the sample tubes to make the water more easily visible in the opaque tubes on the manometer board). The amount that each water level raises in response to the induced vacuum is directly proportional to the head at the

intake of the sample point at depth. This provides a relative head profile across each of the sample points.

To translate the relative head values into an elevation head that can be compared directly to other piezometers, a tube is attached to one of the channels of the manometer board with the free end placed below the water level in the central 50mm-diameter PVC casing comprising the core of the bundled piezometer. The head value was calculated for the PVC casing by the conventional method using an electric sounder as described previously. Once this value was known, the head values for each of the sample points was calculated relative to the head value in the PVC casing as the difference between the heights of the water levels of the sample tubes and the PVC casing as they appear on the manometer board. The validity of the relationship between the central casing and sample tube measurements is demonstrated by comparison of the water levels, as they appear on the manometer board, of the PVC casing and the deepest sample tube which terminates adjacent to the screened interval of the PVC casing (Figure 5.6). In each case these water levels were identical to millimetre accuracy, indicating that the head values calculated for the sample tubes are equivalent to the head values calculated by the conventional, universally accepted method.

One technical complication that arose in the use of the manometer board is related to the increase in pressure within the closed system that occurs as the water levels rise in the sample tubes. In particular, a change in the water level in one tube was observed to result in significant reciprocal changes in the water levels in the other tubes. In order to dampen the influence of these small pressure changes in the closed system, a large-volume air reservoir was connected to the manifold of the manometer board. The reservoir was constructed from a 150mm diameter by 1m long PVC pipe with sealed end caps and an airtight coupling port for attachment to the manometer board. With the reservoir connected, any change in pressure in the closed system caused by a change in the water level in one or more of the tubes was dampened by the compressibility of the relatively large volume of air in the reservoir. This experiment was repeated with both two and three reservoirs attached to spare manometer tubes, and no discernible difference was observed in the relative variation between water levels (even where the difference in water levels was limited to several millimetres). Thus, it was concluded

that use of a single reservoir was sufficient to allow the system to equilibrate without the influence of increased pressure within the closed system.

5.2.3 Hydraulic conductivity – slug tests

The determination of the hydraulic conductivity of aquifer sediments present at the research site is an essential component of the hydrogeologic investigation. A slug test method was chosen for this investigation, due to the technical simplicity of the method and the high-quality data that are collected. The field apparatus used for the slug tests consists of an airtight collar that forms a seal on the top of the PVC casing of the piezometer. The collar is equipped with an injection port connected to an air tank, a depressurisation valve, and a pressure transducer connected to an external data logger. The collar is fitted onto the top of the PVC casing, ensuring that the pressure transducer is lowered deep enough into the piezometer that it will stay submerged throughout the entire procedure. Then a ‘slug’ of air is introduced (in this case, the piezometers were pressurised to 10 kPa as measured on an in-line pressure gauge), the system is allowed to return to equilibrium under the increased pressure environment, and then the pressure valve is released. The pressure transducer records, in 0.25s time increments, the initial water pressure in the piezometer, the pressure change and subsequent equilibration due to the introduction of the slug, and finally the pressure change and subsequent equilibration to static conditions due to the pressure release.

In practice this is a very simple procedure, and typically the process was repeated 2-3 times for each piezometer to verify the precision of the method. Piezometers with screened intervals at different depth ranges were selected for this procedure to provide hydraulic conductivity data across several hydrogeologic subunits and sediment types. The resulting data were plotted to determine the value of hydraulic conductivity according to the Hvorslev method (Hvorslev, 1951) as described in Chapter 6.

Note that this methodology requires depression of the water column in the well under increased air pressure, and as such care must be taken to avoid introducing air into the screened intervals of wells that will be used for sample collection. The slug tests were conducted on shallow wells WT-1, WT-2, WT-3 and 30A, which were not used for

sample collection, and on the central casing of bundled piezometers 3, 4 and 4A, which are screened at 20 mbgs where there was no risk of air being introduced into the screened interval.

5.3 GROUNDWATER SAMPLE COLLECTION AND ANALYSIS

The primary focus of the research at the study site revolved around collection and analysis of water samples from the site. Water samples were collected from conventional piezometers, bundled piezometers and from set locations along the shore of Lachlan Pond No.5. They were analysed for physical parameters and chemical constituents both in the field and in an analytical laboratory at the university. The sampling schedule, sample collection protocols, and field and laboratory analyses are described in the following sections.

5.3.1 Schedule and program design

One of the primary focuses of this research project was to document the changes occurring within the leachate plume over relatively short time intervals. Each sampling event entailed the collection and analysis of samples from a large number of sample points at various locations within the leachate plume. The piezometer locations represent different hydrogeochemical zones along the short groundwater flow path between the landfill and the pond, as described below (Figure 5.1):

- Samples collected from piezometers 1 and 2, located within the landfilled portion of Area A, represent source leachate concentration data,
- Piezometers 10 and 30 represent post-landfill waters, located directly on the terminal edge of landfilling activities,
- Piezometers 3, 4, 4A and 6 represent post-landfill water approximately 50-100m downgradient from Astrolabe Park and Area A, and provide a cross-sectional image of the leachate plume directly along the shore of Lachlan Pond No.5.
- The pond water sampling locations were used to determine whether water quality in the pond is affected by mixing with the discharge of leachate-impacted groundwater. Pond sample locations 3, 4A and 6 are directly adjacent

to piezometers 3, 4A and 6 to aid in the correlation of groundwater and pond water chemistry, although location 4A was only sampled once because it is located within a zone of dense vegetation along the shoreline (Fig 5.1) and surface water was only available following storm events. Sample location 3/6 was included for better coverage downgradient from Area A, and the drain sample is used to assess the impact to pond water quality from discharge water from the underground Bird's Gully drain, into which leachate is believed to be leaking.

The sampling efforts were scheduled at closely spaced time intervals to provide a detailed documentation of variations within the leachate plume in both space and time (Table 5.3). Variations within the plume at each piezometer were monitored by several criteria, including: the maximum 'concentration' of the plume as reflected by selected indicator parameters, the total vertical extent of the plume, and the vertical distribution of constituents within the plume. The last two criteria are only applicable to the bundled piezometers, for which vertical profiles of groundwater chemistry are available. General trends observed in a piezometer can then be compared to the rest of the piezometers to assess the lateral variations within the plume.

Due to budget and timing constraints, it was impractical to sample every available sample point during each sampling event; especially since approximately a third of the sample points were below the leachate plume in fresh, regional groundwater of relatively uniform chemical composition. In order to maintain a closely spaced sampling schedule it was decided to alternate between full sample events, in which samples were analysed for the full suite of field and laboratory chemical analyses, and reduced sample events in which only selected water quality parameters were measured. Further, with regards to the full sample events, only the July 2000 and July 2001 sample events included all available sample points, while only the sample points that define the maximum vertical extent of the leachate plume were used for the December 2000, and March and May 2001 sample events.

Data from two additional sampling events in February and July of 2002 were also incorporated into this study. These sample events were undertaken to address specific issues with regards to the research, and are described in detail in a later chapter.

Table 5.3 Schedule of full and reduced water sampling events.

Sample Schedule	2000						2001						
	Jul	Aug	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul
Full	X					X			X		X		X
Reduced			X				X			X		X	

5.3.2 Field parameters and sample collection

Water samples were collected from conventional and bundled piezometers, as well as from the shore of Lachlan Pond No.5. Due to the constraints imposed by the size of the sample tubes, a Geopump 2 peristaltic pump was used to purge and collect samples from the bundled piezometers. For the sake of consistency (and the desire to not irreparably contaminate the submersible electric pump) the peristaltic pump was also used to purge and sample the conventional leachate piezometers, using a length of dedicated sample tube (5mm diameter flexible PVC tubing) with the intake end positioned in the middle of the screened interval. Pond water samples were collected using a plastic beaker attached to the end of a pole, in an effort to collect water far enough from the shore to avoid potential contamination from rubbish that had collected on the shoreline.

All samples were collected employing low flow sampling techniques (Puls and Barcelona, 1996; ASTM, 2002). For bundled piezometers, the peristaltic pump was connected directly to the small diameter sample point tubing, and purged at a rate of ~100 ml/min until the field water quality parameters were observed to stabilise (typically ~1L purged in total). Validation testing was not conducted to assess the potential for the zones of influence for the sampling points to overlap and interfere with each other; the sampling rate was considered to be sufficiently low, and the aquifer transmissivity sufficiently high, to minimise the potential of this occurring.

For the standard piezometers, fresh disposable small-diameter tubing was lowered to the midpoint of the screened interval, and purging was conducted as for the bundled piezometers. Water levels were measured during purging from standard piezometers to confirm that the purge rate was sufficiently low as to not draw down the stagnant water column.



Figure 5.7 Purging sample point and measurement of field water quality parameters.

Water quality parameters were measured as the piezometers and sample points were purged to ensure that the collected sample was representative of water in the formation and that stagnant water residing in the sample tubes had been removed (Figure 5.7). The measured water quality parameters included: pH, redox potential (E_h), electrical conductivity (EC), dissolved oxygen (DO) and temperature ($^{\circ}\text{C}$). The pH and E_h measurements were made using an Orion model 290A portable pH/concentration meter. A glass electrode with an Ag/AgCl reference was used for the pH measurements, calibrated daily against standard buffer solutions of 4.01, 7.00 and 10.01. The E_h measurements were made using a platinum electrode with an Ag/AgCl reference, calibrated against either Zobell's solution (Nordstrom, 1977; Zobell, 1946) or Orion ORP reference solution No.967961; calibration values were corrected to provide field measurement results relative to a normal hydrogen electrode. Temperature and EC were measured using Orion models 130A and 135A conductivity meters (which only vary in

their datalogging capabilities). The conductivity meters were calibrated against Orion reference solutions of 1413 and 12,900 $\mu\text{S}/\text{cm}$ (Orion Cat. No. 011007 and 011006, respectively). Dissolved oxygen was measured using an Orion model 810Aplus dissolved oxygen meter. Two meters each were used to measure pH, Eh and EC/temp to monitor instrument drift or probe contamination; the final value was recorded as the average of the two meter readings. Meters that were suspected of drifting from the last calibration were immediately checked against a fresh standard in the field and recalibrated if necessary. These parameters were monitored until the values stabilised, after which the final values were recorded and the samples were collected.

To ensure that high sample integrity was maintained, all equipment used in the handling and collection of the water samples was soaked for 24 hours in a dilute nitric acid solution and thoroughly rinsed with deionized water prior to each sampling event. Field equipment that was re-used for each sample was thoroughly rinsed with deionized water and dried between each sample, and then rinsed with the sample water prior to sample collection, handling, processing or analysis.



Figure 5.8 Field equipment for analysis of redox sensitive elements.

Samples for field analysis of NO_3^- , NH_4^+ , Fe^{2+} , S^{2-} and PO_4^{3-} were field filtered using a 0.45 μm Millipore™ cellulose acetate membrane filter prior to analysis; samples for

alkalinity and acidity titrations were not filtered to prevent potential removal of inorganic carbon species. Refer to Section 5.3.3.1 for a detailed discussion of field analytical methods.

Samples for laboratory analysis were collected following the field analyses. Prior to collection, all samples were field filtered using a 0.45 µm Millipore™ cellulose acetate membrane filter, and the sample bottles were rinsed with filtered sample water prior to collection. Laboratory samples consisted of one 60ml sample acidified using 0.4 ml of concentrated analytical grade nitric acid (HNO₃) for analysis of major and minor elements, and one 120 ml unacidified sample for chloride analysis. Following collection, the samples were sealed, labelled and stored in an ice-filled cooler, and transferred to a laboratory refrigerator upon return from the field. All laboratory samples were analysed within the required technical holding limits.

5.3.3 Analysis of water samples

5.3.3.1 *Field analyses*

Field analyses were performed for the chemical constituents whose concentrations are susceptible to rapid change under atmospheric conditions. These analyses included titrations for alkalinity (recalculated as bicarbonate) and acidity (recalculated as free CO₂), and spectrophotometric analysis for the redox sensitive species Fe²⁺, S²⁻, NH₄⁺, NO₃⁻, and PO₄³⁻ (Figure 5.8).

Alkalinity titrations were performed immediately following sample collection using a 0.01M HCl titrant with a bromocresol green indicator (APHA,1998); each titration was performed twice to minimise the potential for analytical error, and the final value was reported as the average of the two titrations. Alkalinity was recalculated as bicarbonate using the following equation (APHA, 1998):

$$\text{HCO}_3^- \text{ (mg/L)} = 0.01 \times 61.012 \times V_{\text{acid}} \text{ (ml)} \times 1000/25 \quad (5.1)$$

It should be noted that in environments with elevated concentrations of dissolved organic material, such as landfills, ionically charged organic acids can contribute

significantly to the total alkalinity (Baedecker and Back, 1979a; Kehew and Passero, 1990; Baedecker and Cozzarelli, 1992). However, organic acids are highly reactive and are preferentially metabolised during microbial processes in the subsurface; thus they tend to degrade a short distance downgradient from landfills (Baedecker and Apgar, 1984; Rügge *et al.*, 1995). Further, the presence of organic acids in significant concentrations in leachate is primarily associated with the early acetogenic phase of waste decomposition, whereas leachate from mature landfills in the methanogenic phase is dominated by refractory, non-polar organic compounds (Chian and DeWalle, 1976 and 1977; Ehrig, 1983; Lu *et. al.*, 1984; Andreottola and Cannas, 1992). Taking into consideration the relative age and maturity of the Astrolabe Park landfill, total alkalinity has been calculated as HCO_3^- with the recognition that an unidentified minor percentage may be contributed by organic acid anions.

Acidity was determined by titrating a 25 ml aliquot of sample with a 0.025M NaOH titrant in the presence of a phenolphthalein indicator (APHA, 1998). Each titration was performed twice to minimise the potential for analytical error, and the final value was reported as the average of the two titrations. The acidity was recalculated as free CO_2 by the following equation (APHA, 1998):

$$\text{CO}_2 \text{ (mg/L)} = 0.025 \times 44.009 \times V_{\text{NaOH}} \times 1000/25 \quad (5.2)$$

Spectrophotometric analyses for Fe^{2+} , S^{2-} , NH_4^+ , NO_3^- , and PO_4^{3-} were performed in the field immediately following sample collection using a Hach DR/2010 spectrophotometer. In general, the procedures for these analyses involve the fixation of the target analyte by the addition of a chemical compound that reacts with the analyte to form a compound with a distinctive colour. The reaction results in the formation of a distinctive colour in solution that changes intensity in proportion with the concentration of the analyte. The concentration is then determined by directing a light beam of known wavelength through the sample vial and measuring the absorbance. A detailed explanation of each individual analysis can be found in the Hach procedures manual (Hach, 2000). Refer to Section 5.5 for a description of the quality control and error calculation methods associated with these analyses.

5.3.3.2 Laboratory analysis

Two samples were collected from each sample point for laboratory analysis. The 60 ml acidified sample was analysed for major and minor elements by inductively coupled plasma atomic emission spectrometry (ICP-AES) in the UNSW School of Biological, Earth and Environmental Sciences analytical laboratory. For the February 2002 sample event, the samples were further analysed for a large suite of trace elements using inductively coupled plasma mass spectrometry (ICP-MS) through the UNSW School of Chemistry analytical laboratory. A complete description of the methods can be found in Standard Methods for the Examination of Water and Waste Water, 20th Edition (APHA, 1998).

The 120 ml unacidified sample was analysed for chloride by an argentometric procedure. This method involves titration of a 25 ml aliquot of sample with a 0.0141M silver nitrate (AgNO_3) solution in the presence of a potassium chromate (K_2CrO_4) indicator (APHA, 1998). The chloride titrations were performed in duplicate to minimize the potential for error, and the final value was taken as the average of the two titrations. Chloride concentrations were calculated using the following equation (APHA, 1998):

$$\text{Cl (mg/L)} = 0.0141 \times 0.3545 \times V_{\text{AgNO}_3} \times 1000/25 \quad (5.3)$$

5.3.3.3 Stable isotope sample collection and analysis

Samples for analysis of stable isotopes were collected during the July 2001 sample event. Samples for analysis of $\delta^{13}\text{C}$, $\delta^{18}\text{O}$, $\delta^2\text{H}$ and $\delta^{15}\text{N}$ were collected from selected sample points from each bundled piezometer, one landfill piezometer, the three temporary piezometers installed in the marsh area, and from the pond sample points.

The samples for $\delta^{13}\text{C}$ analysis were collected in 1 L plastic bottles that were filled by placing the discharge tube at the bottom of the bottle to minimize atmospheric contamination. Twenty-five ml of SrCl_2 (saturated solution) and 2 ml of 11M NaOH solution was added to the sample to precipitate all dissolved inorganic carbon as SrCO_3 (Barnes *et al.*, 1978; Bishop, 1990). The carbon precipitate was prepared by filtering the sample and then drying the filter and precipitate at 48°C for 24 hours. The dried

precipitate was transferred into glass sample bottles and sent to the Commonwealth Scientific and Industrial Research Organization (CSIRO) Centre for Isotope Studies (CSIRO-CIS) in North Ryde, New South Wales, for analysis. The $\delta^{13}\text{C}$ values were determined on the CO_2 precipitate using a mass spectrometer following production of gaseous CO_2 , with a precision of 0.1‰ (McCrea, 1950; Craig, 1953), and reported relative to the Vienna Pee Dee Belemnite (VPDB) standard (Coplen, 1994).

The $\delta^{18}\text{O}$ and $\delta^2\text{H}$ samples were collected in two 40 ml glass bottles with Teflon-lined caps by submerging the bottles in a beaker filled with sample water and screwing the caps on underwater to eliminate all headspace in the bottles. The samples were then sealed, labelled, and stored in a dark environment pending shipment to the CSIRO Isotope Analytical Laboratory in Urrbrae, South Australia, for analysis. The $\delta^{18}\text{O}$ values were determined by mass spectrometry following equilibration with CO_2 gas, and have a precision of 0.15‰ (Epstein and Mayeda, 1953). The values of $\delta^2\text{H}$ were determined by mass spectrometry following conversion to hydrogen gas, and have a precision of 1‰ (Coleman et. al., 1982). Both $\delta^{18}\text{O}$ and $\delta^2\text{H}$ values were reported relative to Vienna Standard Mean Ocean Water (VSMOW) (Coplen, 1994).

The $\delta^{15}\text{N}$ samples were collected according to the CSIRO isotope laboratory (Urrbrae, SA) protocols (Leaney *et al.*, 2001). Samples were field filtered using a 0.45 μm cellulose acetate membrane filter, and the sample bottles were rinsed with filtered sample water prior to collection. The filtered samples were collected in thick-walled 500 ml plastic bottles, with approximately 20% headspace left to accommodate freezing of the sample. The sample bottles were sealed, labelled, placed in individual plastic bags and immediately frozen in a cooler filled with dry ice. The frozen samples were shipped in coolers packed with dry ice to the CSIRO Isotope Analytical Laboratory in Urrbrae, South Australia, for analysis. The $\delta^{15}\text{N}$ values were determined using mass spectrometry following conversion to elemental nitrogen, with reference to an atmospheric air standard (Leaney, pers. comm.).

5.4 SOIL SAMPLE COLLECTION AND ANALYSIS

A limited number of soil samples were collected during the installation of the piezometers at the request of Sydney Water for analysis of pesticides, polychlorinated biphenyls (PCBs) and selected metals. Soil samples were also collected within the marsh area to in an effort to assess the potential accumulation of metals in the soil of the groundwater discharge zone. While most metals were present at extremely low concentrations in the water samples, they have the potential to form complexes with dissolved organic carbon ligands often present in landfill leachate-impacted groundwater, which increases their mobility by reducing their capacity for adsorption or precipitation (Christensen *et al.*, 1996; Jensen and Christensen, 1999; Jensen *et al.*, 1999). The metal complexes have the potential to accumulate at the groundwater discharge zone and thus pose a threat to the environment.

5.4.1 Sample locations

As stated previously, a limited number of soil samples were collected from each of the boreholes during the installation of piezometers 1, 2, 3, 4, 4A, and 6. The marsh soil samples were collected from shallow cores from six locations within the marsh, spaced to provide an even spread of data coverage (Figure 5.4). Table 5.4 outlines the location and depth interval of each soil sample collected.

5.4.2 Soil sample collection

For the piezometer installation soil samples, two to three samples were collected from each borehole at pre-determined depths. Each sample was homogenized in the field by mixing with a trowel in a stainless steel bowl, a small volume was placed into a laboratory-provided glass jar and the samples were shipped to Australian Laboratory Services (ALS) for analysis. Sample splits were also retained in plastic bags for additional analysis at the university.

The marsh soil samples were collected using a 50mm diameter, 1m long section of PVC casing that is split lengthwise up the middle. The two halves are taped together, an end

cap is placed on one end, and the casing is hammered into the ground. Once the desired depth is reached the casing is retracted, the two halves are split apart, and the soil core is exposed. Soil core recovery varied between 50 and 90 cm at the six locations. The recovered cores were divided into 10cm increments, collected in individual plastic bags and stored in a dark, cool environment pending analysis.

The PVC soil core collector was washed and rinsed with deionized water between each core collection to prevent cross-core contamination.

Table 5.4 Location and depth of piezometer borehole and marsh core soil samples.

Core ID	Marsh core soil sample depth interval (cm, bgs)								
	0-10	10-20	20-30	30-40	40-50	50-60	60-70	70-80	80-90
1	X	X	X	X	X	X			
2	X	X	X	X	X	X	X		
3	X	X	X	X	X	X	X	X	X
4	X	X	X	X	X	X	X	X	X
5	X	X	X	X	X	X			
6	X	X	X	X	X				
Borehole ID	Borehole sample depth (m, bgs)								
1	2.5	5.0	7.5						
2	2.5	5.0	7.5						
3	3.0	5.8							
4	3.5	5.5	10.5						
4A	2.5	7.5							
6	6.5	12.5							

5.4.3 Soil sample preparation and analysis

The soil samples collected from the piezometer boreholes were initially analysed for pesticides, polychlorinated biphenyls (PCBs) and selected metals (Cd, Cu, Mn, Ni, Pb, Zn) by Australian Laboratory Services Pty Ltd (ALS). The retained sample splits were stored frozen until the marsh core soil sampling program was performed.

The marsh cores were collected to determine whether metals were accumulating in the shallow soil profile due to the discharge of leachate-impacted groundwater. The

moisture content of the marsh core and piezometer borehole soil samples was determined by drying a portion of the samples at 105°C for 24 hours and determining the change in mass. The dried samples were then homogenized by grinding to pass through a 0.25mm sieve. The ground and sieved samples were then subjected to a hot acid extraction according to U.S. EPA method 3050B (U.S. EPA, 2001). Following the acid extraction, the samples were allowed to sit for 48 hours to allow the suspended matter to settle. The extractant was then taken to the UNSW School of Biological, Earth and Environmental Sciences analytical laboratory for analysis of selected metals (Cd, Cr, Cu, Fe, Mn, Ni, Pb, Se, Zn) by ICP-AES.

The results from the analysis of the soil extracts were recalculated from mg/L to mg/kg soil using the known volume of the sample extracts (30ml) to determine the total mass of each extracted metal, and then using the pre-recorded oven-dried mass of each soil sample used in the extractions to recalculate for the concentration of metals present per kilogram of soil:

$$\text{Total metal mass (mg)} = \text{Extract conc. (mg/L)} \times [\text{Extract vol. (ml)}/1000] \quad (5.4)$$

$$\text{Soil metal conc. (mg/kg)} = \text{Total metal mass (mg)} \times [1000/\text{soil sample mass (mg)}] \quad (5.5)$$

5.5 QUALITY CONTROL

Throughout the course of the research for this project, a very large number of environmental samples were collected. A rigorous system of quality control protocols was followed to monitor and validate the quality of the data generated. The quality control measures discussed in this section are those related to sample collection strategies, field and especially laboratory chemical analyses.

The quality control measures undertaken for this research were designed to assess both the accuracy and precision of the analytical procedures. Accuracy refers to the agreement between a measurement and a known value, in this case determined by analytical measurement of reagent standards of a set concentration. Monitoring for accuracy is typically standard procedure in an analytical laboratory to assess the

instrument drift. Precision refers to the repeatability of a measurement, and can be monitored by including 'blind' duplicate samples in the sample collection program for comparison to the original sample values. The sample collection protocol for this project was to collect one blind duplicate for each 10 samples collected.

Both the accuracy and precision of the analytical procedures were assessed by calculating the *relative percent difference* (RPD) either between the standard concentration and the measured value, or between the measured values for a sample and its blind duplicate (ISO, 1993; U.S. EPA, 2000). The RPD is calculated as:

$$\text{RPD} = (C_2 - C_1) / [(C_2 + C_1)/2] \times 100\% \quad (5.5)$$

An acceptable RPD tolerance should be chosen at the outset of the project, to ensure that the data are being analysed in a consistent manner, and that any discrepancies can be immediately addressed. For this project an RPD tolerance of $\pm 10\%$ was considered acceptable for water samples, and an RPD tolerance of $\pm 25\%$ was considered acceptable for soil samples (due to the increased heterogeneity associated with the sample matrix). Any values falling outside this range were re-analysed to determine the source of the error, and if the error was persistent the data were disregarded.

Quality control measures in the field included the use of duplicate water quality meters for the measurement of pH, E_h , and EC/Temp to identify instrument drift. Also, all titrations were performed in duplicate to monitor the precision of the measured values; all titration measurements fell within an RPD tolerance range of $\pm 10\%$ (and usually much better). Due to time constraints and the cost of analytical reagents, the spectrophotometer measurements were not routinely performed in duplicate. Any unusual or suspect values were reanalysed for confirmation, and the spatial and temporal trends identified by the spectrophotometer analyses closely matched those observed through the other analytical techniques.

In general, the RPD results for water samples analysed by ICP were $<5\%$, the RPD results for re-analysed samples using the Hach kit were $<10\%$, and titration RPD was

generally less than 2% RPD. Any data failing the adopted QA/QC criteria were excluded from further consideration.

A final, and important, quality control measure used to assess the validity of the data was the calculation of charge balance error (CBE) for each of the samples. This calculation operates on the premise of electroneutrality of aqueous solutions: any solutions containing dissolved ions must have an equal number of cations and anions when quantified as milliequivalents per litre (meq/L) (Freeze and Cherry, 1979). The CBE was calculated according to the following equation:

$$\text{CBE (\%)} = [\Sigma \text{ cations} - \Sigma \text{ anions}] / [\Sigma \text{ cations} + \Sigma \text{ anions}] \times 100\% \quad (5.6)$$

This calculation is especially useful as it integrates data from the alkalinity and chloride titrations, the spectrophotometer analyses, and the ICP-AES analyses. A CBE tolerance of $\pm 5\%$ was considered acceptable for this project (Freeze and Cherry, 1979). The CBE for the leachate-impacted water samples was generally very low; the few samples outside the tolerance range came from the low-TDS, fresh regional groundwater, where a small degree of analytical error could result in a significant effect to the CBE. These samples were rectified by a minute adjustment to the bicarbonate value.

5.6 GEOPHYSICAL INVESTIGATIONS

Several geophysical techniques were used to complement the hydrogeochemical investigation at the site. Geophysical investigation techniques have the substantial benefit of being relatively inexpensive to perform, and provide a large volume of high quality data in a short amount of time. The integration of geophysical techniques to the groundwater investigation was very useful in delineating the location of heterogeneities in the aquifer matrix, and was especially useful for adding an additional degree confidence to the groundwater chemistry results. Perhaps the greatest benefit was the generation of cross-sectional images of the leachate plume both parallel and perpendicular to the direction of groundwater flow, which provided details about the distribution of plume between piezometers that would otherwise have remained

unresolved. A description of the geophysical methods is provided in the following sections.

5.6.1 Geophysical borehole logging

A borehole geophysical investigation program was conducted at the Astrolabe Park landfill in conjunction with a groundwater sampling event to provide an additional degree of confidence to the interpretation of groundwater chemistry and site-scale stratigraphy. The geophysical investigation program consisted of bulk electrical conductivity and gamma emission logs for piezometers 3, 4, 4A, 6 and 10. The bulk conductivity logs are used to provide a detailed assessment of the leachate plume location and the gamma logs are intended to aid in the delineation of the subsurface stratigraphy, as described in detail in the following sections. In addition, the current borehole geophysical logs were compared to previous logs by Kelly (1994), in the locations of current wells 4 and 4A, and Jankowski and Acworth (1997) in existing bundled piezometer 10 (refer to Chapter 6, Section 6.2 for discussion).

5.6.1.1 *Bulk electrical conductivity*

The generation of borehole logs of bulk electrical conductivity is a method that was developed specifically for groundwater contamination studies (McNeill, 1990). This method is designed to measure the bulk electrical conductivity of the formation surrounding a borehole. The bulk electrical conductivity of the formation is determined by the electrical properties of both the aquifer minerals and the pore fluid (Archie, 1942; Bussian, 1983); distinct changes in the bulk conductivity in the absence of major lithological changes can therefore be interpreted to result from changes in the fluid EC of the pore water. This type of investigation is particularly useful in the context of a landfill leachate plume released to low TDS groundwater environment, because the elevated solute load of the leachate significantly increases its conductivity. This results in a notable increase in the bulk conductivity of the formation in the zone of contamination, which clearly contrasts with the bulk conductivity signature of the uncontaminated groundwater with a low solute load (Mack, 1993). The presence of clay material in the subsurface, which has a weak electrical surface charge, can also affect the bulk electrical conductivity of the formation; a gamma log run in conjunction with

the bulk electrical conductivity log is used to investigate the presence of clay layers to aid in the overall interpretation.

The field gear consists of a Geonics EM39 geophysical logging system, which includes an electrically powered variable speed winch with 100m of cable. The cable passes over a tripod-mounted pulley system that comes equipped with a shaft encoder to measure the logging speed. The individual sondes used for performing the borehole measurements are attached to the end of the cable and lowered down the borehole at a predetermined speed. The signal from the sonde is recorded using a DOS program on a portable computer, which allows for a real time plot of the data to be generated (Acworth, 1998).

The EM-39 sonde used for borehole measurement of bulk electrical conductivity consists of a three-coil induction system. A transmitter coil in the sonde induces eddy currents in the formation surrounding the bore, which in turn generate a magnetic field that is measured by a coaxial receiver coil in the sonde. The intensity of the generated magnetic field is directly proportional to the electrical conductivity of the formation. A more detailed discussion is provided by McNeill (1986).

As mentioned above, bulk electrical conductivity logs were generated for bundled piezometers 3, 4, 4A, 6 and 10 in conjunction with the April 2001 sampling event. The fluid EC data from the sampling event are compared with the continuous bulk electrical conductivity profile for each piezometer to establish how well the groundwater chemistry data represents the actual plume distribution.

5.6.1.2 *Gamma emission*

Gamma-ray emission logs are used in general to determine the subsurface distribution of clayey material. The technical function of gamma-ray investigations is to measure the emission of gamma photons released during the decay of naturally occurring radioactive isotopes (Keys, 1989). Radioactive isotopes of uranium and thorium occur in natural environments and may contribute to gamma emissions, but in most hydrogeologic studies potassium-40 is the most abundant isotope. Potassium is an abundant component of feldspars, which weather to produce clays, and uranium and thorium can concentrate

in clays through adsorption and ion exchange. Thus, gamma anomalies are often interpreted as increases in clay concentration in the subsurface.

The gamma logs were collected using the same Geonics equipment described in the bulk electrical conductivity section, only using a gamma detecting sonde. These logs were collected in conjunction with the bulk conductivity logs to help determine if observed changes in the bulk conductivity properties of the formation are associated with changes in pore fluid conductivity, aquifer matrix conductivity or both. The gamma logs also aid in the interpretation of the site stratigraphy, particularly with the identification of confining layers.

5.6.2 Electrical resistivity methods

The use of electrical techniques in the investigation of this leachate plume were particularly useful as the high ionic load of the contaminated groundwater results in an enhanced capacity to conduct an electrical current, giving it a very distinct signature in contrast to the uncontaminated, low TDS groundwater. The electrical resistivity transects provide a two-dimensional contour image depicting the resistance of the subsurface material to conduct an electrical current. As with the borehole conductivity measurements, the resistivity of the formation is a function of the aquifer mineralogy, degree of saturation and pore water composition (Archie, 1942, Bussian, 1983). As with all geophysical investigation techniques, there must be a measurable anomaly in the target parameter to produce an effective result.

5.6.2.1 *Surface electrical resistivity transects*

The equipment used to perform the resistivity transects is comprised of 64 steel electrodes pounded into the ground at 2.5m intervals along a 160m transect. The electrodes are connected by leads to take-outs on a multi-core cable, which is in turn connected to an ABEM SAS4000 resistance meter linked with LUND ES 464 electrode controller. There are four electrodes involved in each resistivity measurement arranged according to a Wenner array; the electrode controller directs a current between the first and last electrode in the circuit and the resistivity is measured at the two central electrodes. The measured value is an apparent resistivity, and corresponds to a point

midway between the first and last electrode at a depth equal to half the distance between the first and last electrode. The spacing between electrodes is incrementally increased until the maximum or desired depth range is reached. The surface elevation of electrodes was determined by levelling so that topographic corrections could be made to the image data. The resultant apparent resistivity pseudosection is subjected to an automated inversion routine, which uses a least squares inversion to create an inverse model resistivity section. Some software packages will also convert the resistivity image into a reciprocal conductivity image to aid in correlation with fluid EC or borehole bulk electrical conductivity data. Electrical imaging methods are described in detail by Loke (2001).

Two electrical resistivity transects were performed in conjunction with this research: one parallel to the edge of the landfill (perpendicular to groundwater flow) and the other along a groundwater flow path between the landfill and the edge of Lachlan Pond No.5 (Refer to Figure 6.12 in the geophysical results discussion for transect locations). Both fluid EC values and detailed vertical head data were used to constrain the resistivity section between the landfill and the pond.

5.6.2.2 *Cross-borehole tomography*

The measurement of bulk electrical resistivity data using cross-borehole tomography provides a higher resolution vertical image than surface resistivity techniques, although it is constrained to a relatively narrow vertical section defined by the two boreholes. This study employed the use of a pair of custom-built tomography bores installed 8m apart, with bundled piezometer 30 located precisely at the midpoint between the two tomography bores (Figure 5.9). The bores are constructed of 25mm – diameter PVC casing with gold-plated electrodes connected to electrical leads attached at 1m intervals to a total depth of 15m. The electrical leads are connected to an ABEM SAS4000 resistance meter linked with LUND ES 464 electrode controller. An electrical pulse is discharged into the formation, and the electrical current is conducted through the subsurface according to the bulk electrical properties of the formation matrix. As each electrical pulse is discharged, the electrical potential is measured simultaneously at each of the electrodes on the opposite bore, providing an apparent resistivity measurement of the formation in a range of directions from the electrode. Once the full cycle of

electrodes has been utilized, the data are compiled and subjected to an automated inversion routine, which uses a least squares inversion to create an inverse model resistivity section. Note that the resistivity image produced for this experiment has been inverted to a conductivity image to provide a more convenient method of comparison with the surface transects and the fluid EC data. Loke (2001) provides a review of borehole tomography applications.

Because each electrical pulse is measured simultaneously from various depths and directions of propagation, cross-borehole tomography provides a more accurate, higher-resolution depiction of the electrical properties of the formation with depth than surface resistivity techniques. Used in conjunction, the two techniques provide an excellent method for detecting and delineating electrical anomalies in the subsurface, such as the presence of a landfill leachate plume in an otherwise pristine aquifer.

5.7 DATA ANALYSIS METHODS

The research involved with this project resulted in the generation of a large volume of numerical data, primarily water chemistry data. The data vary over both space and time, and need to be analysed in such a way that meaningful interpretations can be made about the lateral and vertical distribution of chemical components, and the hydrogeochemical processes controlling the observed distribution. The data analysis process included statistical, graphical and computer modelling methods.

The first step was to characterise the data from each individual sample round. Once the data were validated according to the quality control protocols outlined in section 5.5, it was plotted using graphical methods, including: vertical profiles to determine the spatial distribution of major components, bivariate plots to assess the relationships between various components and to help identify significant hydrogeochemical processes, and trilinear Piper diagrams to assess different water types. The characterisation of various chemical water types was also performed using the statistical software package SPSSv10.0 (2000). An heirarchal cluster analysis was used to classify distinct water types based on the relative concentrations of the major ions and redox sensitive elements (eg. Colby, 1993; Suk and Lee, 1999; McGuire et al., 2005; Zeng and

Rasmussen, 2005). The clusters were determined according to a Ward's (1963) method calculation, using Euclidian squared distance as a measure of variable proximity to the corresponding cluster centre. The water samples were ultimately classified into five groups that can be used to correlate water chemistry data between piezometers; these groups are indicated on the bivariate plots to determine which hydrogeochemical processes are associated with the formation of each group.

Once the water chemistry data were characterised for each sample round, temporal variations between the sample rounds were assessed. In particular, the fluid EC values measured during the full and limited sampling events were used as an indicator parameter to track changes in the plume configuration in time. Variations in time of the total concentration of various leachate components was also analysed to determine how the attenuation processes evolve between sampling events.

The computer modelling code PHREEQCv2.6 (Parkhurst and Appelo, 1999) was used to calculate the speciation and saturation indices for select minerals (including metal oxides, hydroxides, sulfides and carbonates) to aid in the identification of active hydrogeochemical processes affecting the different water types. PHREEQCv2.6 was also used to perform inverse modelling calculations for end-member waters along a common flow path between the edge of the landfill and the shore of Lachlan Pond No.5.

CHAPTER 6: HYDROGEOLOGY AND GEOPHYSICS

Due to its proximity to a major urban centre, the Botany Basin has been the subject of a large number of both commercial and research-oriented hydrogeological investigations (Cotton, 1937; Sheil, 1942; Mulholland, 1942; Griffin, 1963; Swann, 1971; Smart, 1974; Kalf and McKibbin, 1976; Webb and Watson, 1979; Brodie, 1988; Nieweglowski, 1988; Jankowski and Knight, 1991; Hitchcock, 1991; Dudgeon, 1993; Evans, 1993; Acworth and Jankowski, 1994; Merrick, 1994 and 1998; Lavitt *et al.*, 1997; Yu *et al.*, 1993, Yu, 1994; Jankowski *et al.*, 1997; Jewell, 1997; Maunsell, 1997; Beck, 2000; Jankowski and Beck, 2000). Several geophysical investigation methods have been employed to study the aquifer properties and groundwater quality of the Botany Sands aquifer; many of these studies are associated with localised contamination investigations, but a number of regional geophysical surveys have been carried out in the Botany Basin as well (e.g. Acworth, 1998). The results of previous hydrogeological investigations in the Botany Basin, with an emphasis on the Astrolabe Park vicinity, are summarized in the following sections, including the new information from this research project. The results of the site-specific geophysical surveys are discussed at the conclusion of the chapter.

6.1 AQUIFER PROPERTIES

The assessment and quantification of the physical properties of an aquifer is essential to both regional and site-scale hydrogeological investigations. In particular, the transport of contaminants in groundwater is often greatly influenced by small-scale heterogeneities in the subsurface that create preferential flow paths. Regional identification of hydrogeologic properties of an aquifer can act as a baseline for the various hydraulic units that may be present at a site, but it is necessary to both delineate the distribution of the units (in this case sedimentary) on a site-scale, and to verify the hydrogeologic properties at the site to determine whether a direct correlation to regional values is appropriate.

Table 6.1 Hydrogeological parameters determined from various regional studies.

Research Project	Location	Test type	Porosity (%)	Hydraulic gradient	K (m/d)	T (m ² /d)	Specific yield	Storativity
Cotton (1937)	Botany	Pump Test			85		0.24	0.24
Mulholland (1942)			33					
Sheil (1942)	Matraville	Various			70			
Griffin (1963)	Tomago sand beds (Newcastle)	Pump Test		0.004	80		0.26	0.4-0.0022
Smart (1974)	Banksmeadow (ICI)	Pump Test	42		28.9-36.8			0.024-0.26
Brodie (1988)	lab test	Grain size/falling head			12-28			
Nieweglowski (1988)	Griffin X-section				21	629		
Dudgeon (1993)	Centennial Park			0.004	20-35			
Merrick (1994)	Moore Park East Botany Alexandria Canal				20 28 14			0.2 0.03 0.075
Lavitt (1994)	Banksmeadow	Lab test	25-31		20			
Acworth and Jankowski (1994)	Kensington (UNSW)	Slug test		0.007	1.4-20.6			
Yu (1994)	Various	Various lab tests	32-42		0.001-21.50			0.001-0.003
Maunsell (1997)	Various				15-85			
Jewell (1997)	Centennial Park	Pump test		0.008	40	400		
	E.S. Marks Field				17	140		
	Moore Park				18-80		0.17	
	Zetland				11-47		0.2	

Table 6.2 Hydrogeological parameters in and around Astrolabe Park.

Research Project	Location	Test type	Porosity (%)	Hydraulic gradient	K (m/d)	T (m ² /d)	Storativity
Swan (1971)	David Phillips Park	Pump test				230	0.0004-0.0022
Kalf and McKibbin (1976)	David Phillips Park	Pump test				300	0.003
Webb and Watson (1979)	David Phillips Park	Pump test			20-30	280	0.001-0.003
Hitchcock (1991)	Astrolabe Park	Slug test		0.0045-0.0052	5.20-10.04	250-301	0.024-0.0058
Jankowski and Knight (1991)	Astrolabe Park	Slug test			1.85-8.61		
Evans (1993)	ELE	Constant head test			12.9-38		
Smolski (1994)	Astrolabe Park			0.0052			
Yu (1994)	Astrolabe Park	Various	36		14.69		
Jankowski <i>et al.</i> (1997)	ELE	Falling head		0.003-0.009	1.8-50		

The Botany Basin has been the subject of numerous hydrogeological investigations, some dating back to over a century ago when the early European settlers were looking for fresh sources of drinking water. The results of a number of regional studies are presented in Table 6.1, while studies conducted in the vicinity of Astrolabe Park are summarized in Table 6.2. The various aquifer properties reported for these studies are discussed below, with an emphasis on those conducted in the vicinity of the study site.

6.1.1 Grain size analysis and porosity

Grain size analyses are a useful tool for characterising sedimentary units based on the relative fractions of different grain sizes. These fractions can be accurately quantified by passing the soil sample through a series of progressively finer sieves and weighing the fraction retained by each sieve. Yu (1994) performed 50 grain size analyses for soil samples collected from different locations and depths around the northern portion of the Botany Basin. His results indicate an overwhelming abundance of medium grained quartz sand, with minor percentages of fines. Beck (2000) expanded on his work by compiling the unpublished results of a further 55 grain size analyses; his results indicate a greater overall percentage of fine-grained material but include a large number of samples collected along the margin of Botany Bay.

Grain size analyses were carried out on three samples from Astrolabe Park (Jankowski and Knight, 1991) and on 22 samples collected at the ELE site south of Astrolabe Park (Evans, 1993; Beck, 2000; Jankowski and Beck, 2000). The results indicate that sediments at the site are comprised predominantly medium grained sand, and the various sedimentary units are distinguished by the composition and relative percentages of fines, which appear to vary from <5-12%. The layers with higher percentages of fines appear to affect the flow of groundwater at the site by preferentially diverting groundwater flow to more coarse-grained units, resulting in the establishment of intricate, complex preferential flow paths (Evans, 1993; Beck, 2000; Jankowski and Beck, 2000). A sample of sandy peat from the floor of Lachlan Pond No.5 analysed by Jankowski and Knight (1991) indicates a significantly elevated fraction of fines, but is still predominantly medium grained sand.

Because of the relative uniformity of the grain size of the aeolian sand deposits, the porosity is relatively high. Porosity values in the Botany Basin have been reported between 25 and 40%, and a value of 36% is commonly accepted as a regional average (Mulholland, 1942; Smart, 1974; Lavitt *et al.*, 1997; Yu, 1994), which is within the range reported from other reliable studies (Peyton *et al.*, 1986). Yu (1994) broadly classified the sediments of the Botany Basin into three categories based on their hydrogeologic properties. The units with progressively greater proportions of fines were thought to have greater values of porosity, although the decrease in interconnected pore spaces results in a general decrease in permeability.

6.1.2 Hydraulic conductivity and transmissivity

A broad range of hydraulic conductivity values (0.01 to 80 m/day) have been reported for studies in the northern Botany Basin, no doubt reflecting the presence or absence of silty, clayey or peaty lenses in the aeolian sand unit. The highest values come from wells completed in pure sand profiles, as is common in the upper portion of the Botany Basin where thick aeolian sand deposits accumulated in the absence of wetlands, watercourses and near shore coastal environments. Fine-grained deposits in the upper sand profile are more common in the immediate vicinity of the Botany wetlands and Botany Bay. Values between 20 and 30 m/day are generally used to describe the hydraulic conductivity of the sand unit on a regional scale.

Several comprehensive studies have been undertaken at Astrolabe Park, David Phillips Field and the Eastlakes Experimental (ELE) site to determine hydraulic conductivity values of the various sediment types present in the aquifer (Table 6.2). Webb and Watson (1979) found that the aquifer was essentially divided into three layers separated by locally continuous low permeability layers at 7 and 17 mbgs. The sandy aquifers were found to have hydraulic conductivity values between 20 and 30 m/day, while the low permeability layer at 17 mbgs was determined to have a vertical hydraulic conductivity value of 0.001 m/day. These results were confirmed by similar tests by Hitchcock (1991) and Yu (1994). At the ELE site, Evans (1993) reported hydraulic conductivity values between 19 and 38 m/day using a constant head permeameter test on five shallow injection wells, and a range of <2-50 m/day using a modified falling head test on bundled piezometer sampling tubes. The

modified falling head test used by Evans (1993) is not considered a highly accurate field method for quantifying hydraulic conductivity due to the very small volume of formation that is influenced by the tests and the potential for significant bias from minor variations in construction methods of the small diameter sample tubes, which results in a potentially significant and unquantifiable uncertainty in the test results. However, in a relative sense the large range of values and steep vertical gradients observed testify to the heterogenous nature of the sediments on a microscale. The constant head permeameter tests were conducted on standard piezometers, and therefore test a greater volume of formation material and are less sensitive to minor variations in well construction. Within Astrolabe Park and the adjacent Eastlakes Golf Course, the average values of hydraulic conductivity were found to be slightly lower with values reported between 1.85 and 14.69 m/day by Hitchcock (1991), Jankowski and Knight (1991) and Yu (1994).

Slug tests were also performed on seven of the piezometers installed for this research project, with three piezometers tested in the shallow, unconfined aquifer (WT-1,2,3), three piezometers tested in the deeper semi-confined portion aquifer (3, 4, 4A), and one piezometer (30A) screened in the low permeability zone acting as an aquitard between the previous two layers (Appendix A). Due to the rapid response times, the slug tests were repeated 2 to 3 times for each piezometer (except for piezometer 30A) to verify the precision of the results. The Hvorslev method (Hvorslev, 1951) was used to calculate values of hydraulic conductivity from the slug test recovery data. The recovery data are plotted as the ratio of the head recovery (H_0-h) at time t to the maximum head displacement (H_0) due to the removal of the slug. From this plot the value of T_0 is determined as the time when $(H_0-h)/H_0 = 0.37$; the hydraulic conductivity is then calculated according to the following equation (Hvorslev, 1951):

$$K = \frac{r^2 \ln(L/R)}{2LT_0} \quad (6.1)$$

where:

K is hydraulic conductivity

r is the radius of the casing

R is the radius of the screen

L is the length of the screened interval

T_0 is the time at which $(H_0-h)/H_0 = 0.37$

Table 6.3 Hydraulic conductivity results from Hvorslev analysis of slug test data.

Test #	Screen interval (m, bgs)	Screen interval (m, AHD)	Screen length (L, m)	Screen radius (R, m)	Casing radius (r, m)	T ₀ (sec)	K (m/sec)	K (m/day)
Piezometer WT-1								
1	4-6	13.46 to 11.46	2	0.025	0.025	4.17	1.64E-04	14.2
2	4-6	13.46 to 11.46	2	0.025	0.025	4.95	1.38E-04	12.0
avg =								13.1
Piezometer WT-2								
1	4-6	13.62 to 11.62	2	0.025	0.025	39.61	1.73E-05	1.5
2	4-6	13.62 to 11.62	2	0.025	0.025	45.32	1.51E-05	1.3
avg =								1.4
Piezometer WT-3								
1	4-6	12.41 to 10.41	2	0.025	0.025	4.79	1.43E-04	12.3
2	4-6	12.41 to 10.41	2	0.025	0.025	5.05	1.36E-04	11.7
3	4-6	12.41 to 10.41	2	0.025	0.025	5.12	1.34E-04	11.6
avg =								11.9
Piezometer 30A*								
1	8-9	7.50 to 6.50	0.5	0.025	0.025	108	1.74E-05	1.5
Piezometer 3								
1	19.9-20.4	-4.09 to -4.59	0.5	0.025	0.025	10.09	1.84E-04	16.0
2	19.9-20.4	-4.09 to -4.59	0.5	0.025	0.025	9.99	1.87E-04	16.2
avg =								16.1
Piezometer 4								
1	19.8-20.3	-2.48 to -2.98	0.5	0.025	0.025	9.54	1.96E-04	17.0
2	19.8-20.3	-2.48 to -2.98	0.5	0.025	0.025	9.41	1.99E-04	17.2
3	19.8-20.3	-2.48 to -2.98	0.5	0.025	0.025	10.25	1.83E-04	15.8
avg =								16.7
Piezometer 4A								
1	19.7-20.2	-4.38 to -4.88	0.5	0.025	0.025	13.63	1.37E-04	11.9
2	19.7-20.2	-4.38 to -4.88	0.5	0.025	0.025	13.66	1.37E-04	11.8
avg =								11.9

The values used for these parameters and the calculated hydraulic conductivity values are presented in Table 6.3. Most of the hydraulic conductivity values for both the unconfined and semi-confined portions of the aquifer are generally between 10 and 20 m/day. Piezometers WT-1 to WT-3 are screened within the upper unconfined aquifer unit, and suggest an average hydraulic conductivity value of 12.5 m/day (with the exception of piezometer WT-2, which was installed within the shallow waste layers of the landfill, thus accounting for the unusually low hydraulic conductivity value of 1.4 m/day). Piezometers 3, 4, and 4A are representative of the lower semi-confined aquifer unit, and the slug test data indicates an average hydraulic

conductivity value of 14.9 m/day, which is sufficiently close to the upper unit to represent a negligible difference in the ability to transmit groundwater. The hydraulic conductivity value for piezometer 30A is 1.5 m/day, which is sufficiently high to suggest that the sediments represented at this depth can act as a leaky confining layer between the shallow and deeper portions of the aquifer, rather than a total hydraulic barrier.

The hydraulic conductivity values determined from this research are similar to those reported by Hitchcock (1991), Jankowski and Knight (1991) and Yu (1994), but are slightly lower than the maximum values reported in David Phillips Field (Webb and Watson, 1979) and the ELE site (Evans, 1993). This may be due to an increased percentage of fines in the sedimentary profile at Astrolabe Park and the adjacent Eastlakes Golf Course, which are located at the confluence of the historical Bird's Gully drainage and the Lachlan Pond chain and have been subjected to periodic inundation as sea levels have fluctuated over the Pleistocene.

Transmissivity can be calculated by analysis of time-drawdown data from pump tests or as the product of the hydraulic conductivity and the saturated thickness of the aquifer. Hydraulic conductivity can be determined quite easily by the use of slug tests, and transmissivity is most commonly calculated by estimating the saturated thickness of the aquifer. Merrick (1994) has reported that the average saturated thickness of the aquifer is 15-20 m, although in some areas a saturated thickness of 30 m is more realistic (Yu, 1994). Values of transmissivity have been reported between 140 and 630 m²/day, depending on the stratigraphy at the particular test site.

6.1.3 Storativity and specific yield

Storativity and specific yield are typically determined from analysis of time-drawdown data from pump tests. Specific yield has been reported between 17 and 26% in the Botany Basin (Cotton, 1937; Griffin, 1963; Smart, 1974; Jewell, 1997), which is characteristic of sediments ranging from silt to medium sand (Fetter, 1994). Storativity has been reported anywhere from 0.0004-0.4 across the Botany Basin (Cotton, 1937; Griffin, 1963; Swann, 1971; Smart, 1974; Kalf and McKibbin, 1976; Webb and Watson, 1979). In the development of his groundwater flow model for the

basin, Merrick (1994) assigned values of 0.2, 0.03 and 0.075 for storativity in different zones of the northern Botany Basin.

Reported values of storativity close to Astrolabe Park (David Phillips Field) range from 0.0004 to 0.003 (Swann, 1971; Smart, 1974; Kalf and McKibbin, 1976; Webb and Watson, 1979). There are no reported values of specific yield in the vicinity of Astrolabe Park, but it is probably safe to assume a value of 0.26 for the aeolian sands as has been reported elsewhere.

6.1.4 Hydraulic gradients and groundwater flow direction

6.1.4.1 *Horizontal hydraulic gradients*

The regional groundwater flow direction is approximately south-southwest, from the northernmost recharge zone in Centennial Park towards Botany Bay. A regional horizontal hydraulic gradient of 0.004 has been reported in several cases (Griffin, 1963; Dudgeon, 1993), but this value varies locally depending on the occurrence of low hydraulic conductivity materials (i.e. peat lenses, increased clay fraction, organic waste material).

In the vicinity of Astrolabe Park, previous studies have established that the local groundwater flow direction has a strong westward component, with groundwater flowing west to southwest from David Phillips Field to Astrolabe Park and then on towards Lachlan Pond No.5 (Hitchcock, 1991; Jankowski and Knight, 1991; Smolski, 1994). This local variation in groundwater flow direction is likely influenced by several of factors, including the local bedrock topography and the fact that the Lachlan Ponds act as groundwater discharge points.

The horizontal gradient in David Phillips Field appears to be on the order of 0.003 to 0.004, but then increases to 0.008 to 0.01 in Astrolabe Park. This steepening of the hydraulic gradient is most likely due to the decreased permeability of the waste material in the landfill with respect to the surrounding sands. Both Jankowski and Knight (1991) and Smolski (1994) report distinct groundwater mounds within Astrolabe Park, a phenomenon that has been documented for several landfill

investigations in literature (MacFarlane *et al.*, 1983; Kjeldsen, 1993; Kjeldsen *et al.*, 1998), and is believed to promote the flux of leachate from the landfill into the underlying formation.

The horizontal groundwater flow direction was confirmed between Astrolabe Park and Lachlan Pond No.5. Unfortunately, an exhaustive site search determined that the majority of the monitoring network used for the previous studies has been lost due to landscaping and grading activities within Astrolabe Park, Eastlakes Golf Course and Area A, so the current potentiometric analysis was confined to the area between Astrolabe Park and Lachlan Pond No.5 where the new piezometers are installed. The groundwater level data are presented in Tables 6.4 and 6.5 for the semi-confined and unconfined portions of the aquifer, respectively. The potentiometric maps presented in Figures 6.1 and 6.2 are based on data from piezometers screened in the semi-confined portion of the aquifer, and the potentiometric maps in Figure 6.3 are based on the more recently installed piezometers that are screened in the shallow, unconfined portion of the aquifer. The potentiometric analysis was created using head data input into the graphical display software Surfer (version 7; 1999), using kriging to create the potentiometric contours (in conjunction with a 'reality check' of the contour results).

It is clear from Figures 6.1, 6.2 and 6.3 that the groundwater flow direction in both the unconfined and semi-confined portions of the aquifer is consistently west-southwest from Astrolabe Park towards the Lachlan ponds. There are slight variations between the potentiometric maps for the unconfined and semi-confined aquifers; this may be due to differences in the piezometer locations used, or may reflect the influence of surface topography in the shallow, unconfined aquifer. The horizontal hydraulic gradient observed between the landfill and the pond ranges from 0.004 to 0.006 in the unconfined aquifer, and from 0.008 to 0.012 in the confined aquifer. The lesser gradient observed in the shallow aquifer may be influenced by the lack of shallow head data in the middle of the landfill where the greatest degree of groundwater mounding was observed in previous studies (Jankowski and Knight, 1991; Smolski, 1994).

Table 6.4 Water level data for the semi-confined portion of the aquifer below Astrolabe Park, Eastlakes Golf Course and Area A.

Piezometer ID	Screened interval (m, bgs)	Top of casing elevation (m, AHD)	Date	Depth to water (metres below top of casing)	Hydraulic head (m, AHD)
1	6-9	18.160	14-Jul-00	2.648	15.512
			13-Sep-00	2.760	15.400
			12-Dec-00	2.985	15.175
			23-Jan-01	3.145	15.015
			15-Mar-01	2.905	15.255
			13-Apr-01	2.480	15.680
			21-Aug-01	2.265	15.895
2	6-9	18.458	14-Jul-00	2.882	15.576
			13-Sep-00	2.995	15.463
			12-Dec-00	3.225	15.233
			23-Jan-01	3.345	15.113
			15-Mar-01	3.100	15.358
			13-Apr-01	2.640	15.818
			20-Jul-01	2.595	15.863
3	19.9-20.4	15.826	21-Aug-01	2.640	15.818
			14-Jul-00	1.435	14.391
			13-Sep-00	1.550	14.276
			12-Dec-00	1.770	14.056
			22-Jan-01	1.915	13.911
			15-Mar-01	1.670	14.156
			13-Apr-01	1.285	14.541
			16-May-01	1.030	14.796
			18-Jun-01	1.195	14.631
4	19.8-20.3	17.654	20-Jul-01	1.270	14.556
			21-Aug-01	1.285	14.541
			14-Jul-00	2.780	14.874
			13-Sep-00	2.900	14.754
			12-Dec-00	3.120	14.534
			23-Jan-01	3.260	14.394
			15-Mar-01	2.920	14.734
			13-Apr-01	2.520	15.134
			16-May-01	2.265	15.389
4A	19.6-20.1	15.877	18-Jun-01	2.415	15.239
			20-Jul-01	2.500	15.154
			21-Aug-01	2.520	15.134
			14-Jul-00	1.360	14.517
			13-Sep-00	1.485	14.392
			12-Dec-00	1.700	14.177
			22-Jan-01	1.845	14.032
			15-Mar-01	1.550	14.327
			13-Apr-01	1.170	14.707
			16-May-01	0.950	14.927
			18-Jun-01	1.070	14.807
			20-Jul-01	1.165	14.712
			21-Aug-01	1.170	14.707

Table 6.4 Water level data for the semi-confined portion of the aquifer below Astrolabe Park, Eastlakes Golf Course and Area A.

Piezometer ID	Screened interval (m, bgs)	Top of casing elevation (m, AHD)	Date	Depth to water (metres below top of casing)	Hydraulic head (m, AHD)
6	19.6-20.1	15.929	14-Jul-00	1.780	14.149
			13-Sep-00	1.880	14.049
			12-Dec-00	2.070	13.859
			22-Jan-01	2.200	13.729
			15-Mar-01	1.985	13.944
			13-Apr-01	1.645	14.284
			16-May-01	1.420	14.509
			18-Jun-01	1.590	14.339
			20-Jul-01	1.645	14.284
			21-Aug-01	1.645	14.284
10	29.5-30.0	16.628	14-Jul-00	2.020	14.608
			13-Sep-00	2.145	14.483
			12-Dec-00	2.340	14.288
			23-Jan-01	2.490	14.138
			15-Mar-01	2.250	14.378
			13-Apr-01	1.910	14.718
			16-May-01	1.690	14.938
			18-Jun-01	1.725	14.903
			20-Jul-01	1.790	14.838
			21-Aug-01	1.785	14.843

Piezometers 1 and 2, which were used for the potentiometric analysis in the semi-confined aquifer, are installed directly into the waste layers and therefore may reflect influence from groundwater mounding in the Area A landfill, thus resulting in a steeper gradient. It should be noted that the shallow gradient in particular changes even along the short transect between the landfill and the pond; between piezometers 10 and 3, the horizontal gradient is closer to 0.002.

The current groundwater flow direction and horizontal gradient data confirm the results of previous studies in the area between Astrolabe Park and Lachlan Pond No.5. The discrepancies in the horizontal gradient values are most likely due to the differences in the existing monitoring networks used in the potentiometric analyses. However, even in the absence of groundwater mounding data in the centre of Astrolabe Park the hydraulic gradients in the area between the landfill and the pond are very similar to those measured in the current study, and thus the current gradient data are used for groundwater velocity and contaminant transport calculations.

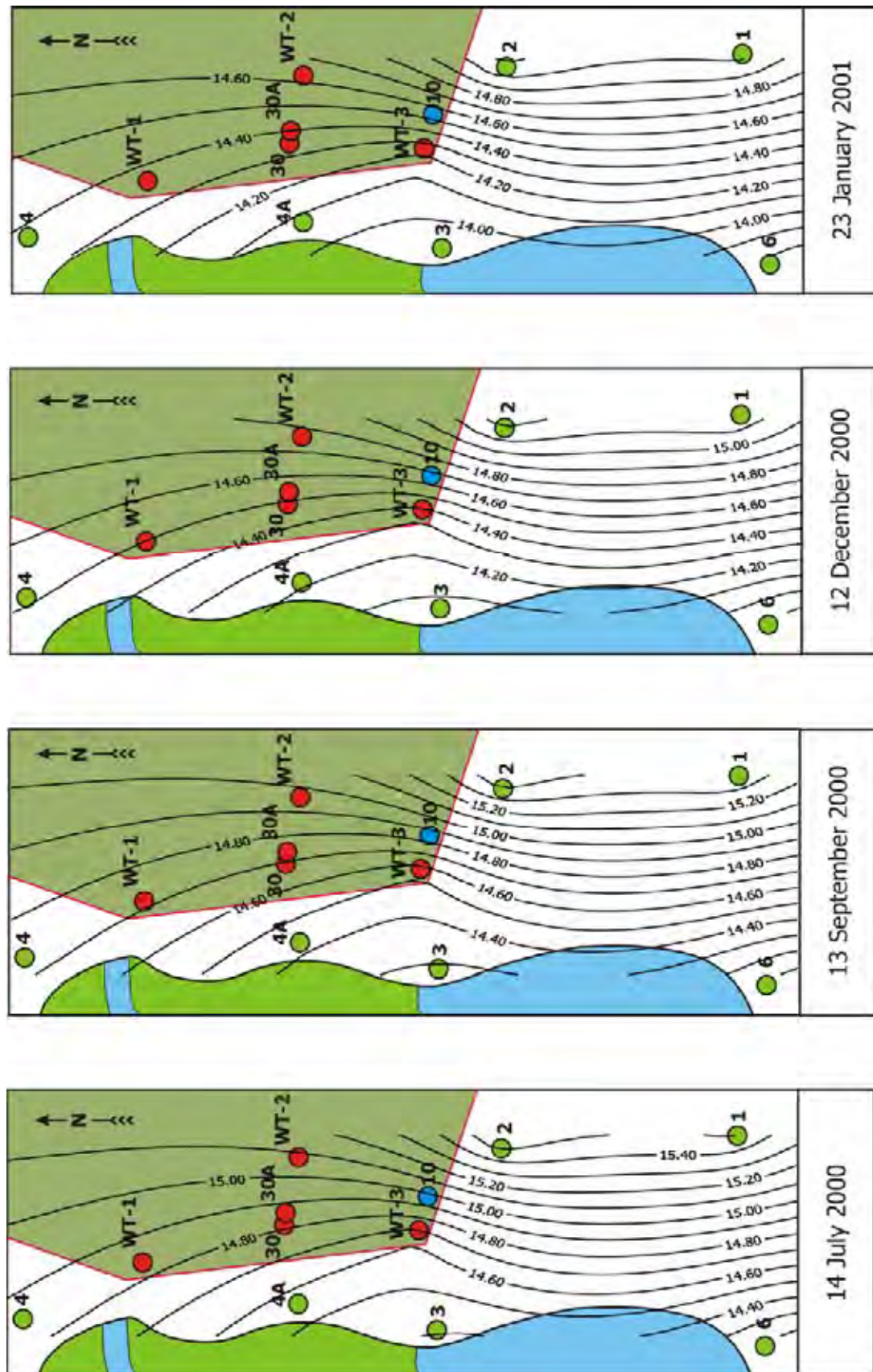


Figure 6.1 Potentiometric maps for Astrolabe Park and Eastlakes Golf Course: July, September, December 2000 and January 2001 (refer to Table 6.4 for a summary of the hydraulic head values at each piezometer).

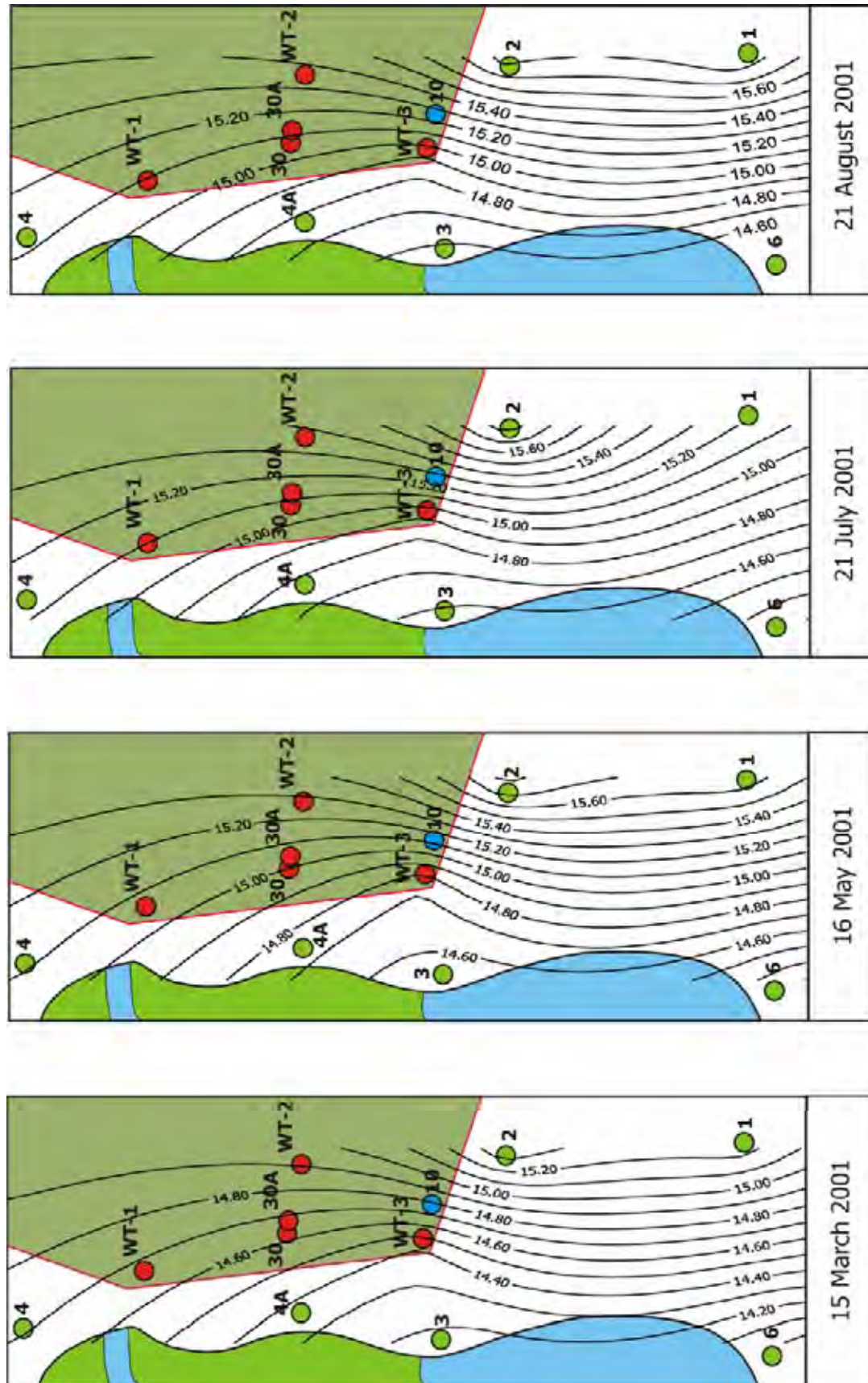


Figure 6.2 Potentiometric maps for Astrolabe Park and Eastlakes Golf Course: March, May, July and August 2001. (refer to Table 6.4 for a summary of the hydraulic head values at each piezometer).

Table 6.5 Water level data for the shallow, unconfined portion of the aquifer below Astrolabe Park and Eastlakes Golf Course.

Piezometer ID	Screened interval (m, AHD)	Top of casing elevation (m, AHD)	Date	Depth to Water (metres below top of casing)	Hydraulic Head (m, AHD)
WT-1	13.46-11.46	17.459	27-Nov-01	2.940	14.519
			14-Feb-02	2.935	14.524
			24-Apr-02	2.865	14.594
			29-Jul-02	2.960	14.499
			30-Jul-02	2.881	14.578
WT-2	13.62-11.62	17.624	27-Nov-01	2.990	14.634
			14-Feb-02	2.975	14.649
			24-Apr-02	2.895	14.729
			29-Jul-02	2.996	14.628
			30-Jul-02	2.997	14.627
WT-3	12.41-10.41	16.412	27-Nov-01	2.000	14.412
			14-Feb-02	2.000	14.412
			24-Apr-02	1.930	14.482
			29-Jul-02	2.020	14.392
			30-Jul-02	2.000	14.412
30A	7.51-6.51	15.506	27-Nov-01	0.945	14.561
			14-Feb-02	0.910	14.596
			24-Apr-02	--	--
			29-Jul-02	0.940	14.566
			30-Jul-02	0.928	14.578
Pond	--	16.256	27-Nov-01	2.035	14.221
			14-Feb-02	2.030	14.226
			24-Apr-02	2.010	14.246
			29-Jul-02	2.035	14.221
			30-Jul-02	1.980	14.276

6.1.4.2 Vertical hydraulic gradients

The vertical hydraulic gradients were measured along a groundwater flow path transect between the landfill and the pond using a multi-channel manometer board at piezometers 10 and 3. The measurements were made in July 2002 as part of an experiment in integrated investigation techniques to establish the connection between the groundwater and Lachlan Pond No.5; the results are presented in Table 6.6, and are plotted in Figures 6.4 and 6.5 with fluid EC values to demonstrate the relationship between vertical head distribution and plume occurrence. In general, a strong upward hydraulic gradient was observed in both piezometers. The calculated vertical hydraulic gradients at piezometers 3 and 10 is 0.0053 and 0.0039, respectively, while horizontal hydraulic gradients between the piezometers, calculated for zones of equal elevation, range from 0.0009 to 0.0021, with a mean value of 0.0018.

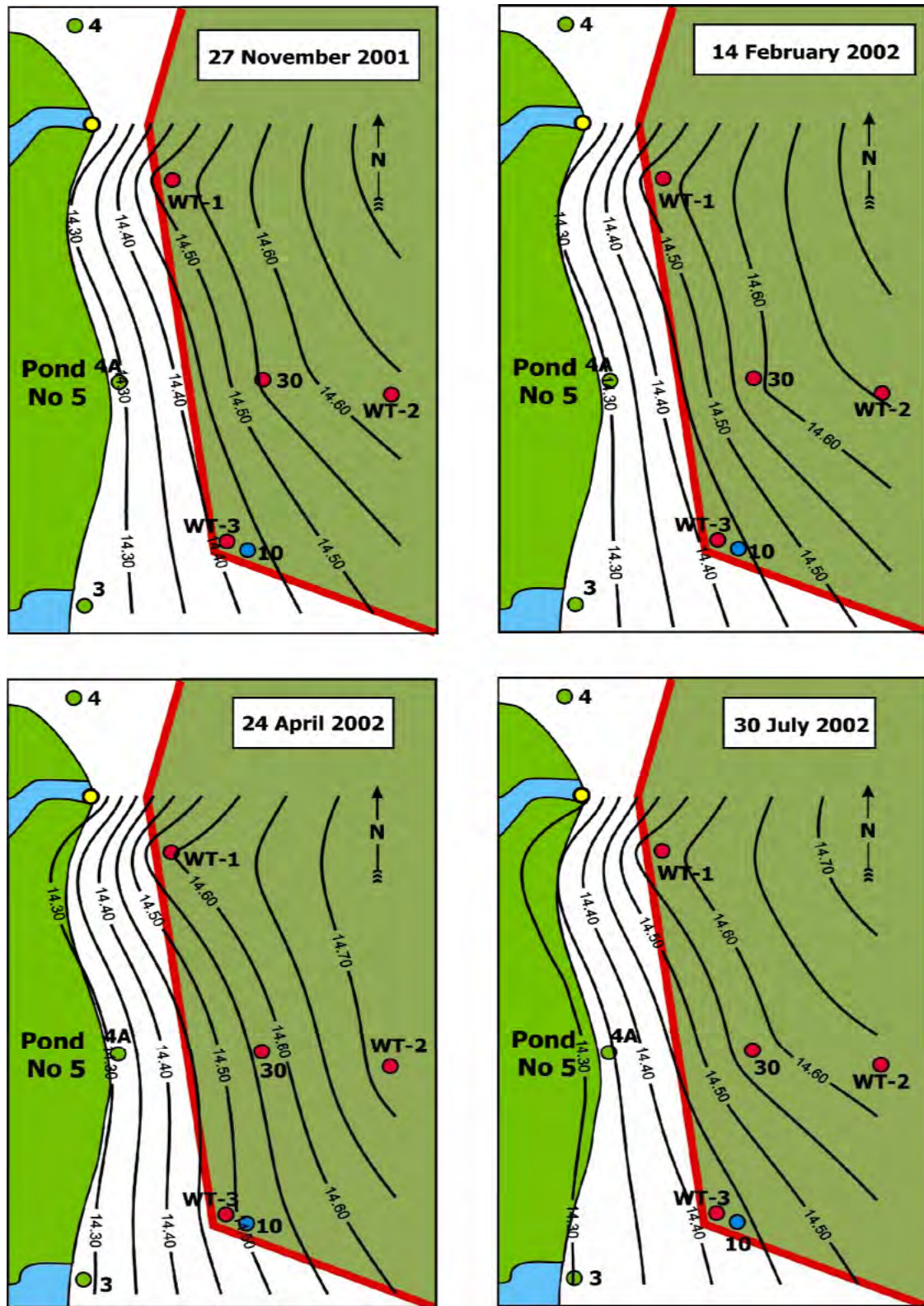


Figure 6.3 Potentiometric maps for the shallow, unconfined aquifer at Astrolabe Park and Eastlakes Golf Course: November 2001 and February, April and July 2002 (refer to Table 6.5 for a summary of the hydraulic head values at each piezometer).

The data presented in Figures 6.4 and 6.5 indicate that the majority of the vertical head change at piezometer 3 occurs across a thin vertical interval between 4.5 and 6.5 m AHD, which correlates well with cone penetrometer test (CPT) data for this location that indicate the presence of a clayey lens from approximately 5 to 6 m AHD (Jankowski and Knight, 1991). Another sharp change in head occurs between 2.5 and 3.5 m AHD, which correlates to another clayey layer detected by the CPT at approximately 3.5 m AHD. Similarly, the most significant vertical head change at piezometer 10 is observed between 7.5 and 9.5 m AHD, which correlates well with CPT data for this location indicating a clayey lens at approximately 8 to 9 m AHD (Jankowski and Knight, 1991). A second sharp change in head is observed between approximately 2.5 and 5.5 m AHD at piezometer 10. This is below the final depth of the CPT log for this location, but would have a good correlation to the depth range of the lower clayey unit observed in the deeper CPT log at piezometer 3, which may be continuous between the two piezometers.

The strong correlation between the occurrence of clayey lenses and steep vertical hydraulic gradients illustrates the semi-confining effect of the subsurface heterogeneities present in the sandy profile.

6.1.4.3 Hydraulic connection to Lachlan Pond No 5

The horizontal groundwater flow direction between the Astrolabe Park landfill and Lachlan Pond No.5 has been well documented in this and previous studies (Hitchcock, 1991; Jankowski and Knight, 1991; Smolski, 1994). However, while it was speculated that the leachate plume was at least partially discharging into Lachlan Pond No.5, the precise nature of the groundwater/surface water interaction remained elusive due to a lack of detailed vertical hydraulic head data close to the discharge zone at the shore of the pond. The results of the vertical head measurements clearly demonstrate that the vertical gradient close to the edge of the groundwater-fed pond has a significantly (factor of 2-3) greater impact on the groundwater flux than the horizontal gradient.

Table 6.6 Vertical hydraulic head values from July 2002

Piezometer ID	Sample ID	Sample elevation (m,AHD)	Manometer board value (m)	Δ PVC casing (m)	Hydraulic head (m,AHD)
3	1.5	13.914	0.594	-0.092	14.321
3	2.0	13.414	0.600	-0.086	14.327
3	2.5	12.914	0.592	-0.094	14.319
3	3.0	12.414	0.589	-0.097	14.316
3	3.5	11.914	0.585	-0.101	14.312
3	4.0	11.414	0.584	-0.102	14.311
3	4.5	10.914	0.584	-0.102	14.311
3	5.0	10.414	0.583	-0.103	14.310
3	5.5	9.914	0.583	-0.103	14.310
3	6.0	9.414	0.584	-0.102	14.311
3	6.5	8.914	dry		
3	7.0	8.414	dry		
3	7.5	7.914	0.589	-0.097	14.316
3	8.0	7.414	0.590	-0.096	14.317
3	8.5	6.914	0.599	-0.087	14.326
3	9.0	6.414	0.593	-0.093	14.320
3	10.0	5.414	0.644	-0.042	14.371
3	11.0	4.414	0.685	-0.001	14.412
3	12.0	3.414	0.670	-0.016	14.397
3	13.0	2.414	0.704	0.018	14.431
3	14.0	1.414	0.554	-0.132	14.281
3	15.0	0.414	0.213	-0.473	13.940
3	16.0	-0.586	0.696	0.010	14.423
3	17.0	-1.586	0.706	0.020	14.433
3	18.0	-2.586	0.704	0.018	14.431
3	19.0	-3.586	0.686	0.000	14.413
3	casing	--	0.686	0.000	14.413
10	3.0	13.628	0.449	-0.106	14.401
10	4.0	12.628	0.448	-0.107	14.400
10	5.0	11.628	0.450	-0.105	14.402
10	6.0	10.628	0.453	-0.102	14.405
10	7.0	9.628	0.461	-0.094	14.413
10	8.0	8.628	0.482	-0.073	14.434
10	9.0	7.628	0.507	-0.048	14.459
10	10.0	6.628	0.508	-0.047	14.460
10	11.0	5.628	0.510	-0.045	14.462
10	12.0	4.628	dry		
10	13.0	3.628	0.526	-0.029	14.478
10	14.0	2.628	0.539	-0.016	14.491
10	15.0	1.628	0.543	-0.012	14.495
10	16.0	0.628	0.544	-0.011	14.496
10	18.0	-1.372	0.545	-0.010	14.497
10	20.0	-3.372	0.548	-0.007	14.500
10	22.0	-5.372	0.550	-0.005	14.502
10	24.0	-7.372	0.551	-0.004	14.503
10	26.0	-9.372	0.551	-0.004	14.503
10	28.0	-11.372	0.553	-0.002	14.505
10	30.0	-13.372	0.555	0.000	14.507
10	casing	--	0.555	0.000	14.507

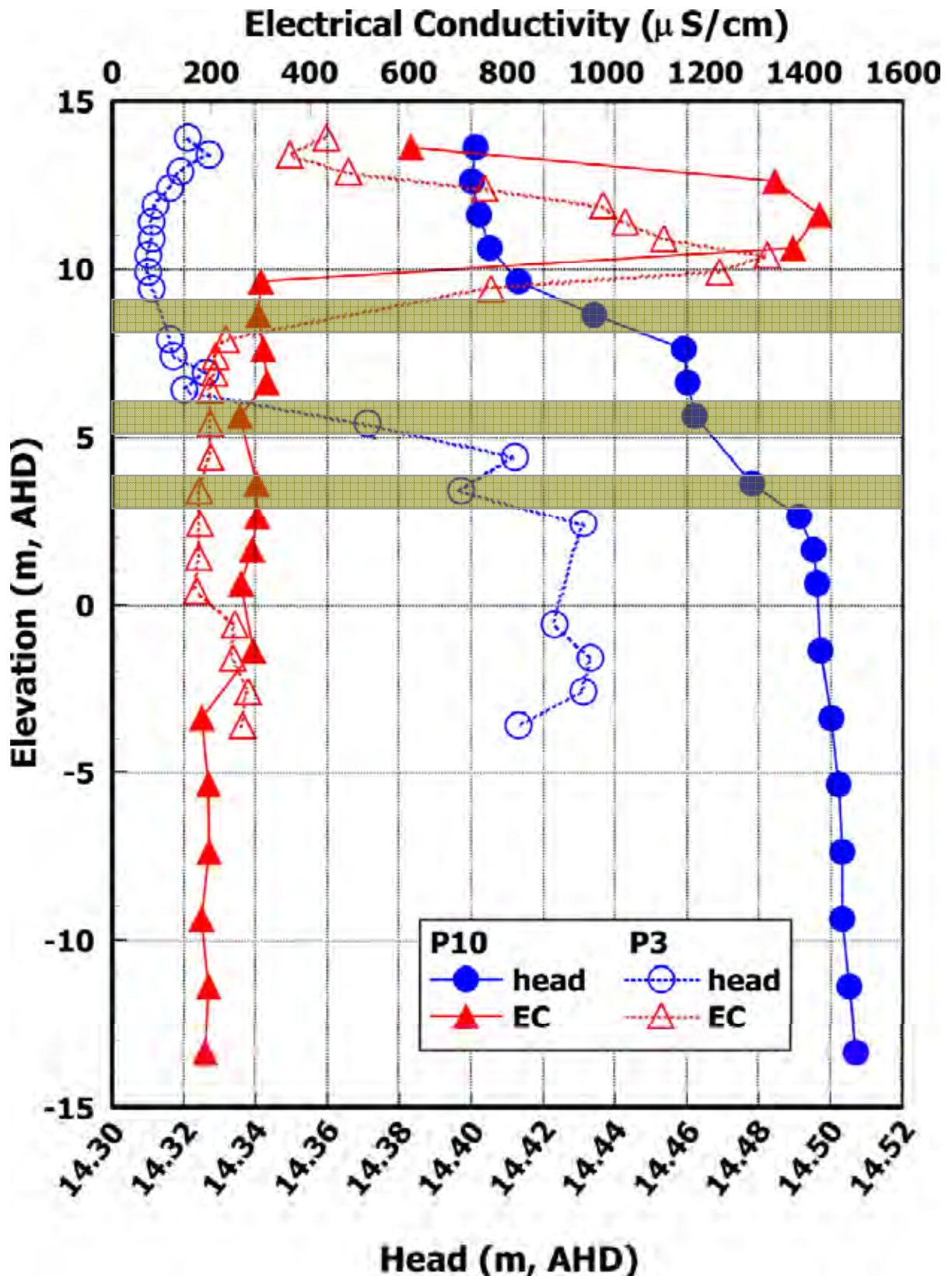


Figure 6.4 Vertical head and EC values for piezometers 3 and 10, July 2002 (shaded bars indicate the locations of fine-grained layers identified through CPT investigations (Jankowski and Knight, 1991).

The magnitude of the upward hydraulic gradient supports the assertion that a significant portion of the shallow groundwater is discharging into Lachlan Pond No.5 along with the landfill leachate plume, which is constrained to the shallow, unconfined portion of the aquifer at this location (Figure 6.5). This assertion is further supported by analysis of groundwater chemical types collected from within the vegetation mass, as will be discussed in a later chapter.

6.1.5 Average linear groundwater flow velocity

The velocity of groundwater flow is obviously an important consideration in the analysis of contaminant transport through an aquifer. The groundwater flow velocity is dependent on the hydraulic conductivity and effective porosity of the aquifer material and the groundwater flow gradient, as expressed by the following equation (Fetter, 1994):

$$V_x = -K/n_e dh/dl \quad (6.2)$$

Where:

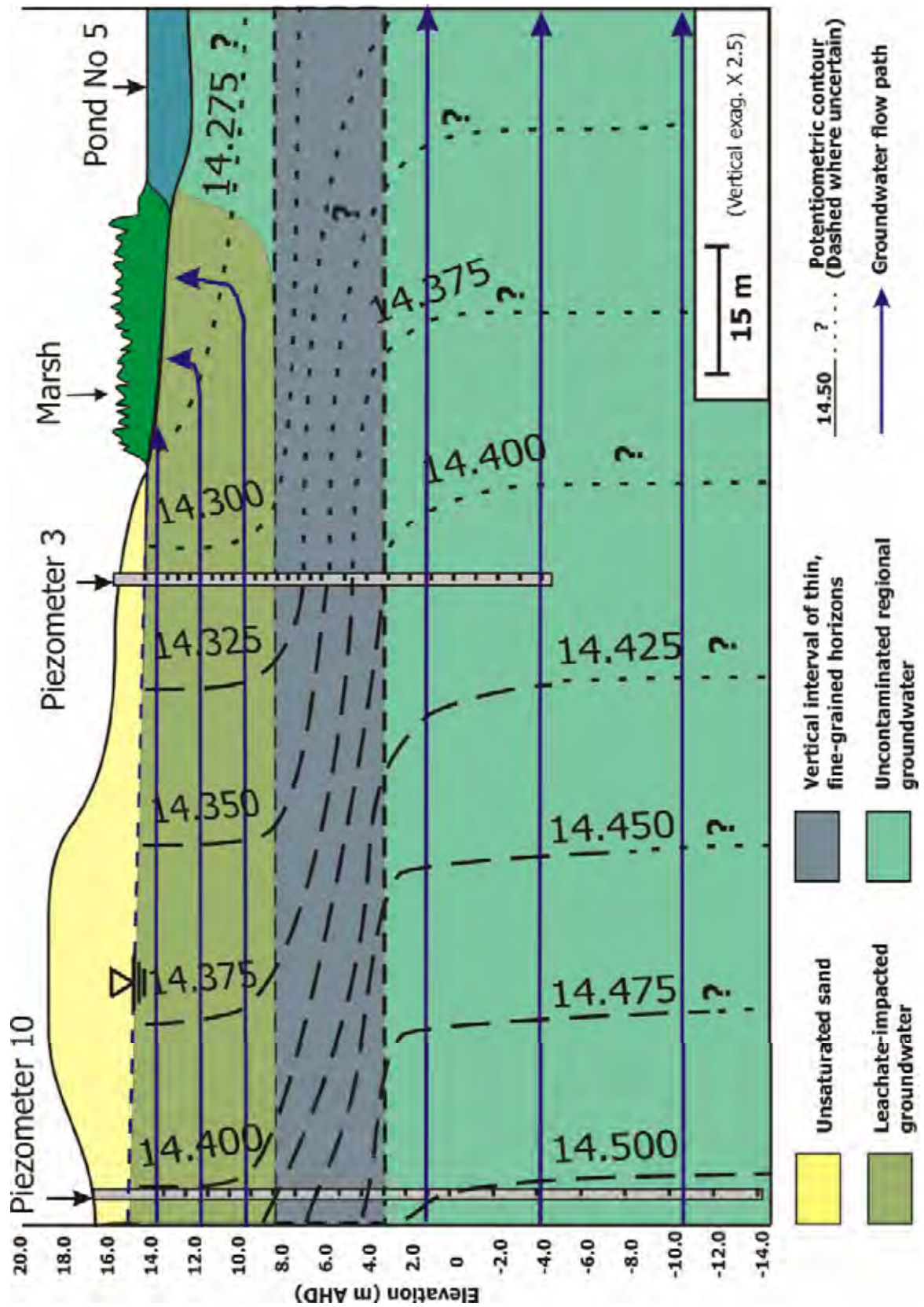
V_x is the average linear velocity (m/day)

K is the hydraulic conductivity of the aquifer material (m/day)

n_e is the effective porosity

dh/dl is the hydraulic gradient

The leachate plume between Astrolabe Park and Lachlan Pond No.5 is effectively confined to the shallow, unconfined portion of the aquifer. The average value of hydraulic conductivity for this portion of the aquifer is 12.5 m/day, expressed as the average of the K values determined by slug tests in piezometers WT-1 and WT-3 (Table 6.3; WT-2 was ignored as it is assumed that the anomalously low K value is due to its being installed into waste layers at the edge of the landfill). Although the horizontal hydraulic gradient is variable along flow path between the landfill and the pond, an average value of 0.005 is used, which is consistent with values determined for this area in previous studies (Table 6.3, Hitchcock, 1991; Jankowski and Knight, 1991; Smolski, 1994). Finally, an effective porosity value of 0.36 is used, based on a report indicating the effective porosity in unconsolidated sediments is essentially equal to the total porosity in relatively homogenous, well sorted aquifer materials (Peyton, 1986).



Based on these values, the average linear velocity of the groundwater in the shallow, unconfined portion of the aquifer between Astrolabe Park and Lachlan Pond No.5 is calculated as follows:

$$\begin{aligned}V_x &= (12.5 \text{ m/day}) / (0.36 \times 0.005) \\&= \mathbf{0.1736 \text{ m/day}} \\&= \mathbf{\text{approx. 65 m/year}}\end{aligned}$$

Considering that the average groundwater flow path between the landfill and the pond is <75 m, the residence time for leachate-impacted groundwater migrating towards the pond is approximately one year. Note that whilst the above value is considered to be sufficiently representative of the average linear velocity within the sandy aquifer unit, the lower hydraulic conductivity (1.4 m/day; WT-2) and steeper hydraulic gradient within the waste layers at the edge of the landfill suggests an average linear velocity of approximately 15 m/year. However, as the focus of this research is on the migration of leachate through the aquifer after it has discharged from the landfill, groundwater velocity within the landfill is not considered further.

6.2 GEOPHYSICAL INVESTIGATIONS

The use of geophysical techniques designed to identify electrical anomalies in the subsurface has been successfully employed for similar landfill research sites (eg. McNeill et al. 1988; McNeil, 1989; Mack, 1993; Lucius and Bisdorf, 1995), and has confirmed fluid EC data collected from the bundled piezometers, and also has identified an unusual contaminant plume structure that could not be identified by groundwater data alone (Jorstad et al., 2004; Acworth and Jorstad, 2006). The gamma-ray emission logs had mixed success in identifying peat and clay lenses previously identified in CPT logs from the site (Jankowski and Knight, 1991), perhaps because some of these thin layers approach the minimum thickness that can be effectively identified using gamma emission techniques.

6.2.1 EM-39 borehole logging

The EM-39 geophysical logging system is a convenient tool for performing a variety of geophysical borehole investigation methods by exchanging the sondes. The

borehole geophysical methods for this study include bulk electrical conductivity logging to delineate the extent of the electrical anomaly created by the presence of the leachate plume, and gamma emission logging to attempt to confirm the location of clayey and peaty lenses. Borehole logs were generated for piezometers 3, 4, 4A, 6 and 10, and are presented in Figures 6.6 – 6.11.

6.2.1.1 Bulk electrical conductivity

In general, the electrical anomaly created by the presence of the leachate plume is clearly identified in the bulk conductivity logs. The spike in the EC value observed at the top of the conductivity logs represents the presence of either the steel standpipes (piezometers 3, 4, 4A and 6) or metal debris in the landfill (piezometer 10). In most cases fluid EC data measured shortly before the borehole logging event are also provided for comparison to the bulk EC data; the borehole geophysical data for piezometer 10 come from several previous research (Jankowski and Acworth, 1997), and fluid EC data from the previous research and the current study are provided. In all cases there is close agreement between the plume location as identified by both the bulk and fluid EC measurements, which adds further evidence that the electrical anomaly is primarily the result of the leachate plume and not lithological changes. The groundwater EC values are considerably higher than the bulk EC values, due to the presence of the low conductivity quartz sand matrix.

The multiple bulk conductivity logs for piezometer 10 exhibit a consistent vertical plume configuration, with variations in the fluid EC of the leachate plume reflected in the bulk EC logs (Figures 6.10 and 6.11). The main electrical anomaly indicates a distinct trough at 10m AHD, which correlates well with the lower boundary of the leachate plume as defined by recent groundwater EC data from this piezometer. Two lesser peaks are observed below the main anomaly at, 7 and 9m AHD, in the bulk EC logs from 1996; only the 9m AHD peak is present in the 1995 bulk EC log. This is generally reflected in the fluid EC profiles from 1995 and 1996 in which a significant portion of the leachate plume is present between 6 and 10m AHD, although the bulk EC trough between these peaks is not represented in the fluid EC profiles.

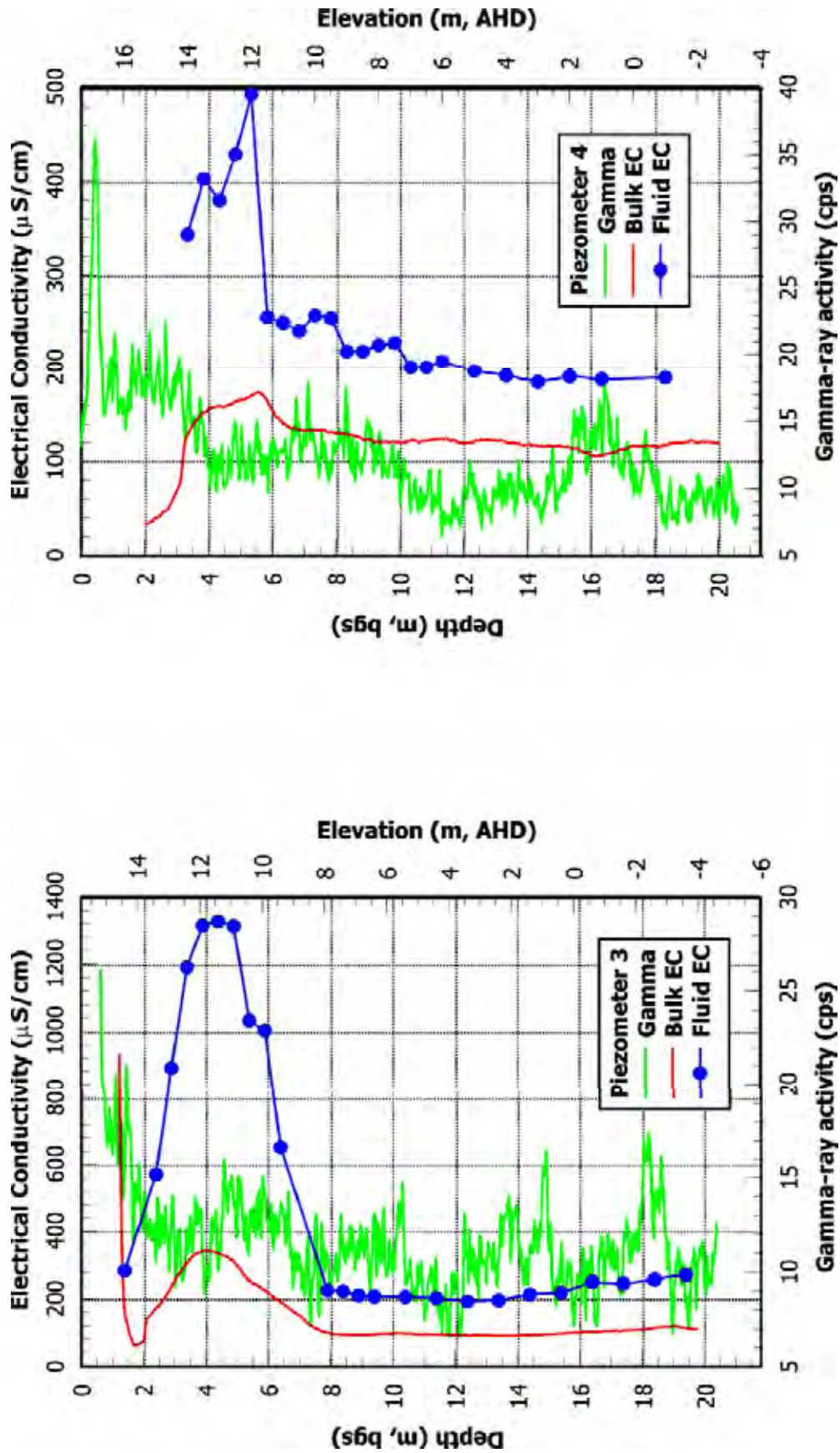


Figure 6.6 Borehole geophysical logs for piezometer 3.

Figure 6.7 Borehole geophysical logs for piezometer 4.

The concomitant fluctuation of the bulk and fluid EC profiles adds further evidence that the electrical anomaly detected by the various geophysical methods is indeed a result of the presence of the leachate plume, and imparts a higher degree of confidence to the validity of the surface resistivity techniques as an accurate method for determining the plume configuration and relative concentration (as discussed in section 6.3.2).

6.2.1.2 *Gamma emission*

The gamma ray emission logs identified very few pronounced anomalies, but several minor anomalies appear to correlate reasonably well with the lower boundary of the leachate plume observed in the groundwater and bulk EC data. Also, some of these anomalies occur at the same elevation at several piezometer locations, suggesting that some of the clayey/peaty lenses may constitute locally continuous confining layers. The gamma emission logs were correlated primarily against CPT logs sited adjacent to current borehole locations (Jankowski and Knight, 1991), with additional reference to previous geophysical assays undertaken at the site (Kelly, 1994; Acworth, 1998).

Three clayey/peaty lenses were detected at piezometer 3 at depths of 10.0, 15.0 and 18.0 mbgs (6.0, 1.0 and -2.5 m AHD) (Figure 6.6). The layer at 6.0 m AHD was also present in a CPT log from this location; however, there is no clear indication in the gamma log of a clayey layer detected at 3.5 m AHD in the same CPT log (Jankowski and Knight, 1991). The layer at 6.0 m AHD is also present at piezometers 4A and possibly 10; the layer at 1.0m AHD was also detected at piezometers 4 and 4A.

There are three extremely minor clayey/peaty layers detected at piezometer 4 at depths of 6.5, 7.0 and 8.5 mbgs (11.0, 10.5 and 9.0 m AHD), in addition to a more substantial layer detected at 16.5 mbgs (1.0m AHD) (Figure 6.7). However minor, the three upper layers appear to correlate with the lower boundary of the main plume and the upper and lower boundaries of a lesser, lower plume identified in the groundwater EC profile (Figure 6.6). However, only the layer at 1.0 m AHD appears to have any correlation to other piezometers.

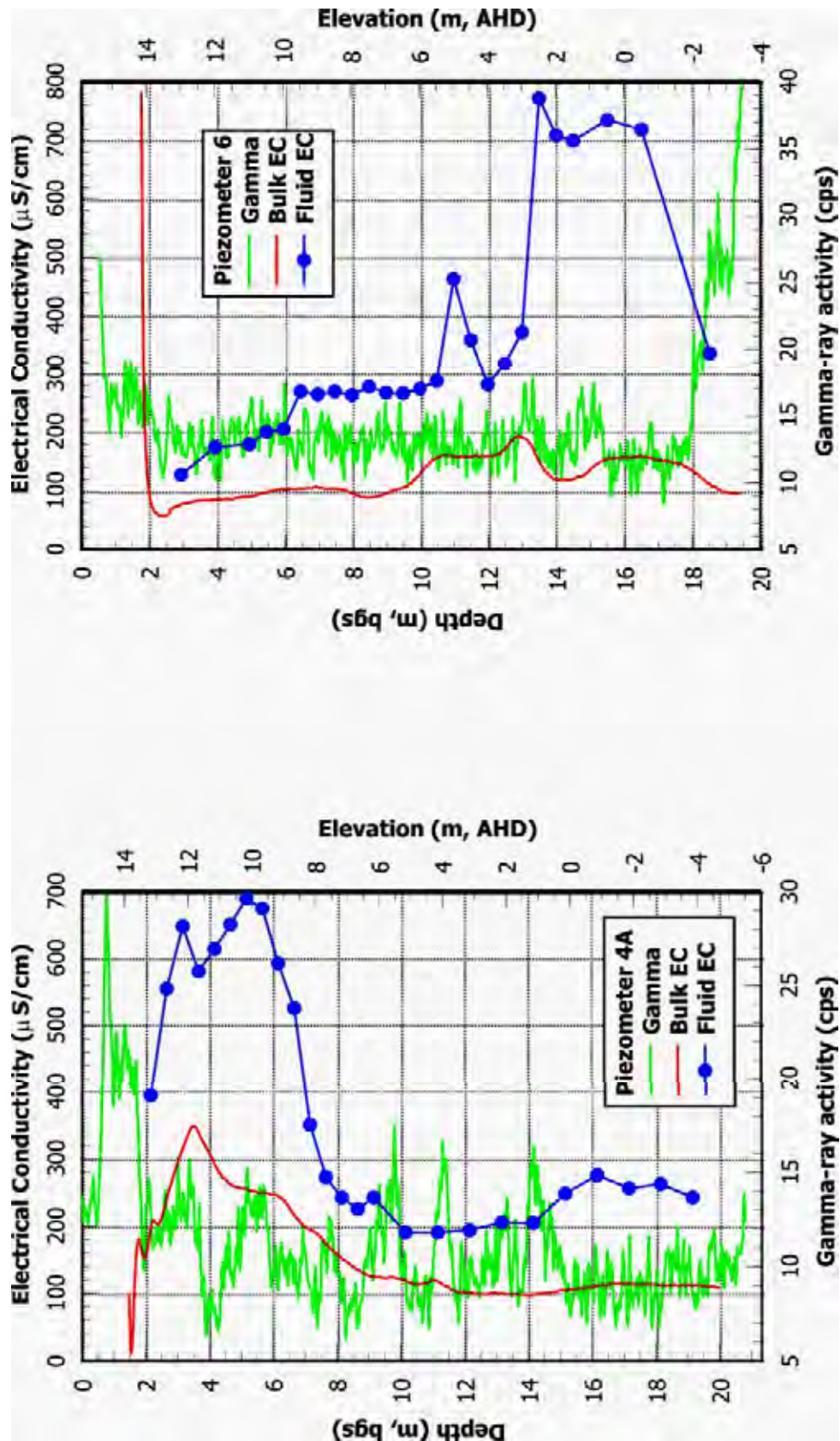


Figure 6.8 Borehole geophysical logs for piezometer 4A.

Figure 6.9 Borehole geophysical logs for piezometer 6.

At piezometer 4A, three fairly distinct clayey/peaty layers were detected at depths of 10.0, 11.5 and 14.0 mbgs (6.5, 4.0 and 1.5m AHD) (Figure 6.8). The layer at 6.5 m AHD appears to correlate reasonably well with the lower boundary of the leachate plume. This same layer appears to occur at piezometer 3 and possibly piezometer 10, while the layer at 1.5 m AHD also appears in piezometers 3 and 4.

Piezometer 10 displays a number of minor peaks in the gamma log, although one reasonably distinct peak at 6.5 mbgs (10 m AHD) corresponds exactly to the lower boundary of the leachate plume observed in the bulk EC logs (Figure 6.10) and groundwater EC profiles (Figure 6.11). This layer was also detected in a CPT log from this location (Jankowski and Knight, 1991). There are several other minor peaks below this one that may correlate with clayey/peaty lenses detected at neighbouring piezometers, but the connection is somewhat tenuous.

Piezometer 6 only displays one significant gamma anomaly at a depth of 18 mbgs (-2 m AHD) (Figure 6.9). The lack of shallow confining layers may explain why the plume occurs at a greater depth at this location, although it has yet to be established whether the plume detected at piezometer 6 is related to the plume issuing from Astrolabe Park. The significant anomaly at -2 m AHD occurs at the depth where the drill rig experienced auger refusal during the installation of this piezometer, and may possibly represent the weathered fringe of bedrock at this location; a significantly shallower depth than was encountered at Astrolabe Park or David Phillips field.

The gamma emission logs provide evidence supporting the notion of locally continuous clayey and peaty lenses acting to confine the leachate plume to the shallow aquifer downgradient from Astrolabe Park. The absence of detectable gamma anomalies in the shallow aquifer at piezometer 6 may help to explain the occurrence of the plume at a much lower elevation than in observed at the other bores.

6.2.2 Electrical resistivity methods

Both the surface resistivity transects and the cross-borehole tomography survey identified the electrical anomaly created by the leachate plume. The spatial distribution of the plume in the conductivity images correlates well with the fluid EC

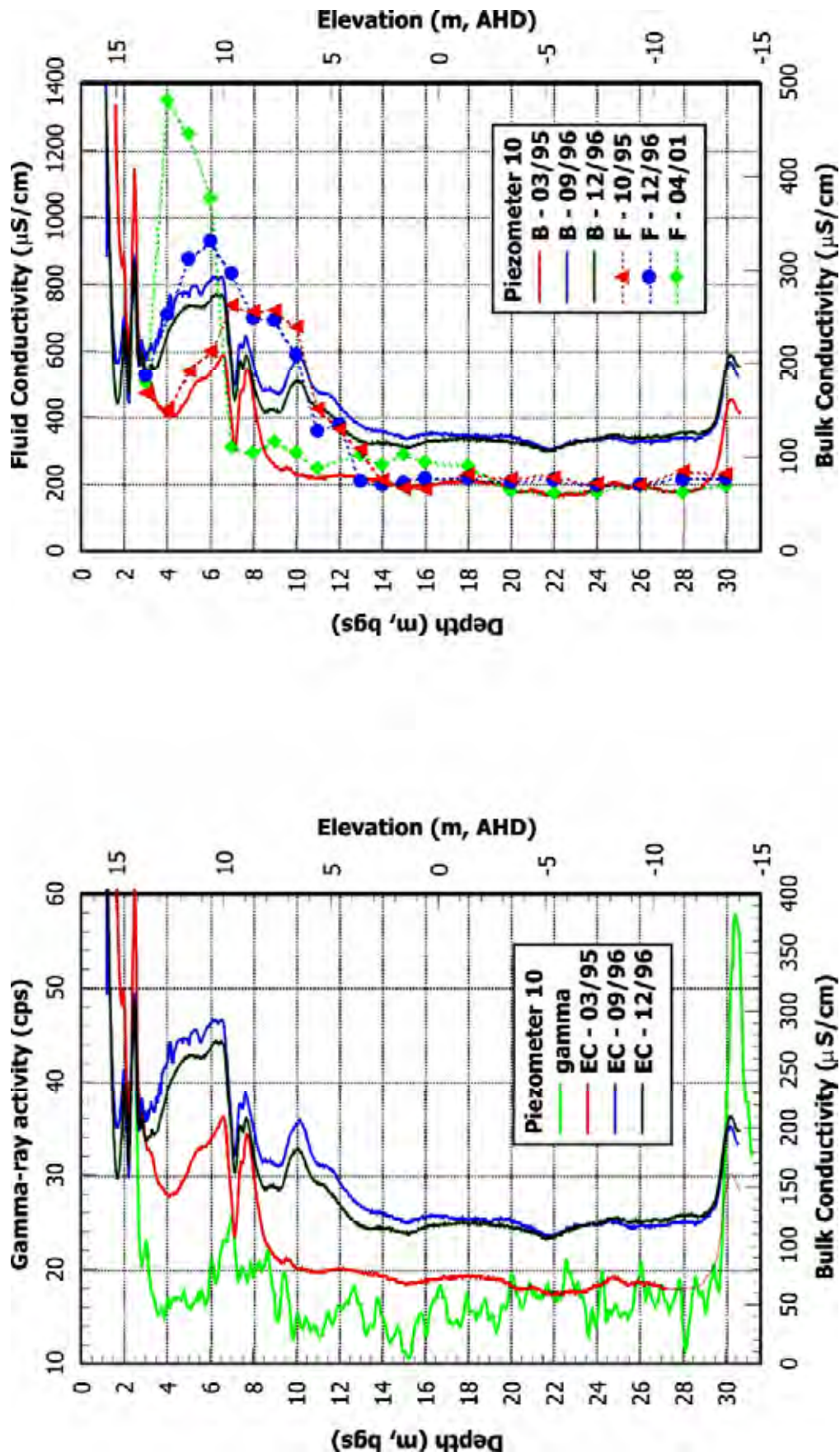


Figure 6.10 Borehole geophysical logs for piezometer 10.

Figure 6.11 Comparison of bulk and fluid EC for piezometer 10.

data where the transect intersects a bundled piezometer location. However, the transect images provide some interesting results with respect to the nature of contaminant transport between the landfill and the pond, which have the potential to significantly complicate the use of inverse geochemical modelling at this site.

6.2.2.1 *Surface resistivity transects*

The surface electrical resistivity transects clearly identify the subsurface electrical anomaly created by the presence of the leachate plume. The locations of the surface resistivity transects are presented in Figure 6.12, and the images generated from resistivity transects are presented in Figure 6.13. Note that the resistivity data have been inverted, resulting in images depicting the distribution of apparent *conductivity* as opposed to apparent *resistivity*, to provide a more convenient method of comparison to the fluid EC and bulk electrical conductivity data.

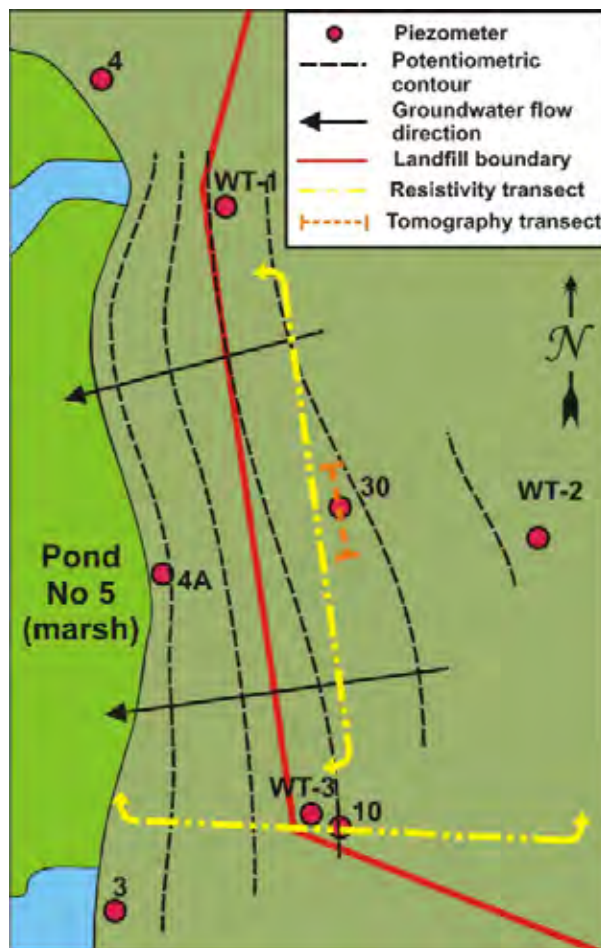


Figure 6.12 Location of geophysical transects.

The transect normal to groundwater flow indicates a relatively consistent plume thickness, with the bulk of the plume confined to the depth interval between 10 and 5 m AHD. A distinct change in the vertical apparent conductivity gradient can be observed at approximately 8m AHD, the same approximate elevation at which the groundwater EC values decrease and a dramatic increase in vertical head variations are observed.

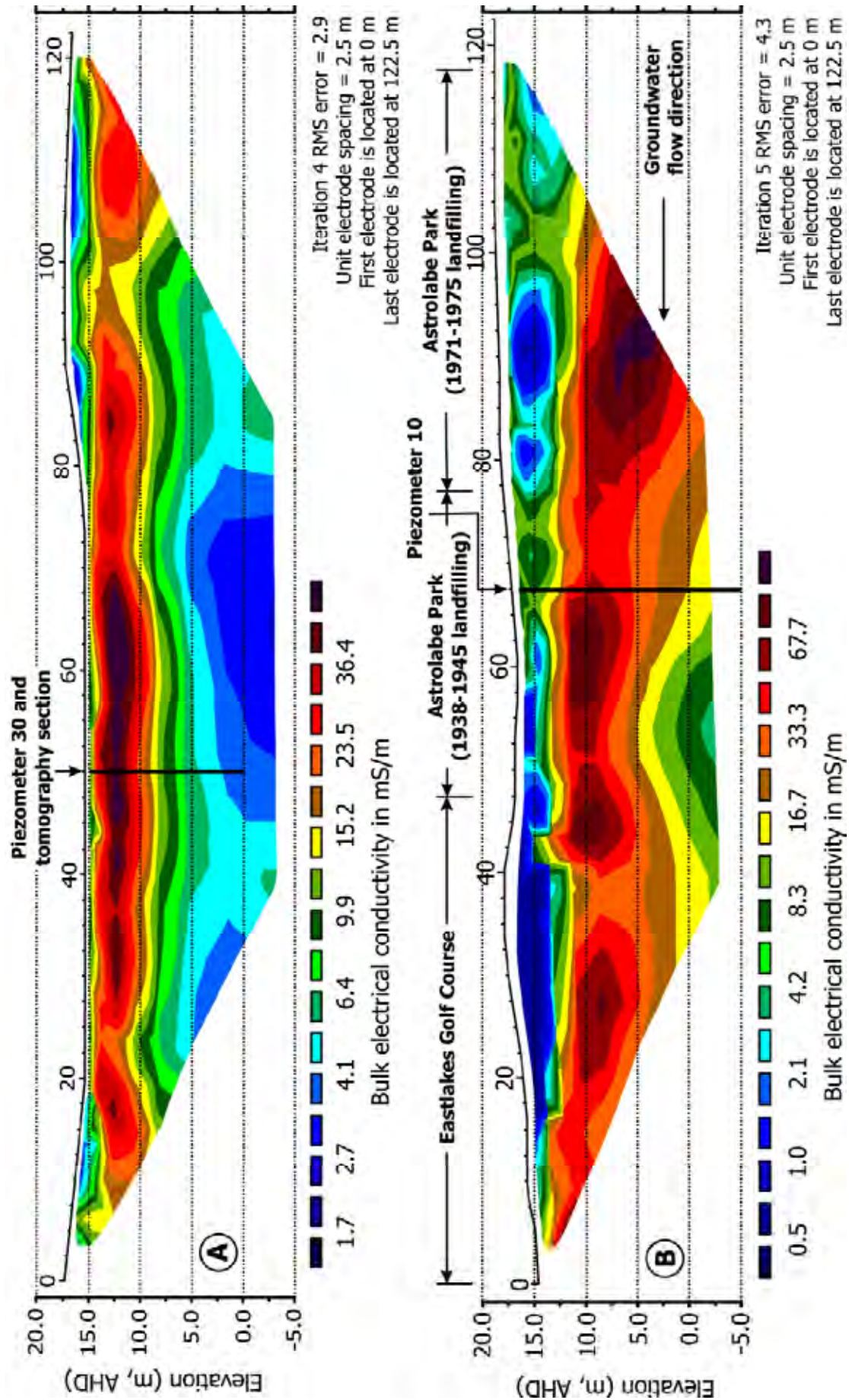


Figure 6.13 Surface electrical resistivity transects; (A) Normal to groundwater flow, and (B) Parallel to groundwater flow (note that the resistivity images have been inverted to conductivity images for ease of comparison with bulk and fluid EC data).

The conductivity image does depict a slightly more gradational transition zone between the leachate plume and fresh regional groundwater than is suggested by the groundwater EC profile; this may be due to an increased clay/peat fraction in the sediments directly beneath the plume that would result in an increased bulk conductivity value for the formation that is not reflected in the groundwater data alone. This may also be attributable to the nature of the surficial investigation technique as well, as the resolution of the data tends to decrease with depth. Unfortunately the diameter of the central casing of piezometer 30 and the tomography bores is too small to allow the use of the EM39 borehole gamma sonde to investigate the clay/peat content of the formation at this location. It is interesting to note the segmentation of the plume in cross-section. It is uncertain whether this is attributable to lateral variations in hydraulic conductivity or whether it reflects the varied nature of waste distribution in the landfill. Regardless of the mechanism responsible for the segmentation, this image provides an example of the large-scale plume structure revealed by this geophysical investigation technique that is not revealed by the use of piezometers alone.

Surface resistivity transects were carried out along a similar transect line, normal to groundwater flow, both by Smolski (1994) and Acworth *et al.* (1994) (Figure 1.3). It is interesting to note that, while electrical anomalies representing leachate-impacted groundwater were detected in both studies, the anomalies were markedly limited in lateral extent in comparison to the current study, with the anomalies constrained to an approximately 60 m long section of the transect in the vicinity of current piezometer 4A (Figure 3.1). Unfortunately it is difficult to ascertain whether the difference between the previous and current anomalies represents temporal variability in contaminant transport from the landfill, or is attributable to the significant advances in the sensitivity and resolution of the investigation technique available for the current study.

The transect along groundwater flow path between the landfill and the pond displays considerably more variation in plume structure than is observed in the normal transect. As with the normal transect, a significant degree of plume segmentation is evident, with at least four distinct contaminant slugs visible. A distinct linear feature is observed at approximately 14.5m AHD, which represents the water table. Between

70 and 100 m from the shore of the pond, the leachate plume is much thicker and deeper than has been observed elsewhere, extending from 2 – 10m AHD. This portion of the transect overlaps the edge of the main waste disposal zone, thus the plume is directly beneath the waste layers in this location. From the shore of the pond to 70m along transect, the conductivity image suggests that the leachate plume becomes significantly narrower, with the centre of mass constrained between 8 and 12 m AHD. Unfortunately the image tapers towards the edge of the pond and the plume image is partially hidden, but the visible portion of the plume appears to be rising towards the surface, which would be consistent with a groundwater discharge zone. The distinct contaminant slugs observed in the surface resistivity transects may suggest that contaminant transport between the landfill and the pond is greatly influenced by rainfall events, which may provide a flux of oxygen-rich recharge water to the landfill stimulating a limited period of high intensity aerobic degradation, and potentially flushing out a slug of concentrated leachate that was retained as the saturated moisture content of the waste layers during dry spells. These mechanisms will be investigated in more detail in later chapters.

Piezometer 10 is located at 67.5m along transect (Figure 6.13), and the vertical groundwater EC profile exhibits a strong correlation with the conductivity distribution in the transect image at that location. Because the conductivity image tapers at the ends, it is not possible to make a direct comparison of the vertical groundwater EC data from piezometer 3 to the plume distribution on the conductivity image. However, it is clear that a simple linear interpolation of observed conductivity values between piezometers 10 and 3 would yield significantly different results in terms of leachate plume configuration than is indicated by the irregular, segmented nature of the conductivity image. This could have significant ramifications with respect to the interpretation of active hydrogeochemical processes along flow path between the piezometers, and may limit the value of results based on inverse geochemical modelling alone. It may also help to explain several occasions where the concentrations of certain plume constituents were observed to increase along flow path.

6.2.2.2 *Cross-borehole tomography*

The conductivity distribution of a vertical section that passes through the most highly contaminated portion of the leachate plume is presented in Figure 6.14 (refer to Figure 6.12 for the location of the tomography section). It is evident from the image that this geophysical technique provides a much higher resolution depiction of the distribution of formation electrical properties than is provided by the surface methods. A significant electrical anomaly is present in the upper portion of the image between 7 and 14 m AHD, with the centre of mass concentrated between 13 and 14 m AHD.

A second smaller anomaly is present between 9 and 10 m AHD. The conductivity distribution observed in this image is closely supported by groundwater chemistry data collected in February 2002 from bundled piezometer 30, located precisely at the midpoint between the two tomography bores (Jorstad and Acworth, 2006). However, the smaller, lower electrical anomaly is not reflected in the groundwater EC profile, which may indicate a variation in the plume distribution in the three months between the time of the sampling event and the cross-borehole tomography survey, or may just reflect the degradation of resolving power with depth associated with surface derived geophysical measurements.

The tomography image is located approximately at the midpoint of the surface resistivity transect normal to groundwater flow (refer to Figures 6.12 and 6.13). In general, the image generated using the cross-borehole technique supports the anomaly distribution identified in the surface transect, with the centre of mass of the plume located approximately between 12 and 14 m AHD, and extending to a depth of 7m AHD in this location. The tomography image can be easily compared to the surface transect image to verify the plume distribution, but is of a sufficient resolution to also provide a detailed comparison with the depth discrete fluid EC data from piezometer 30.

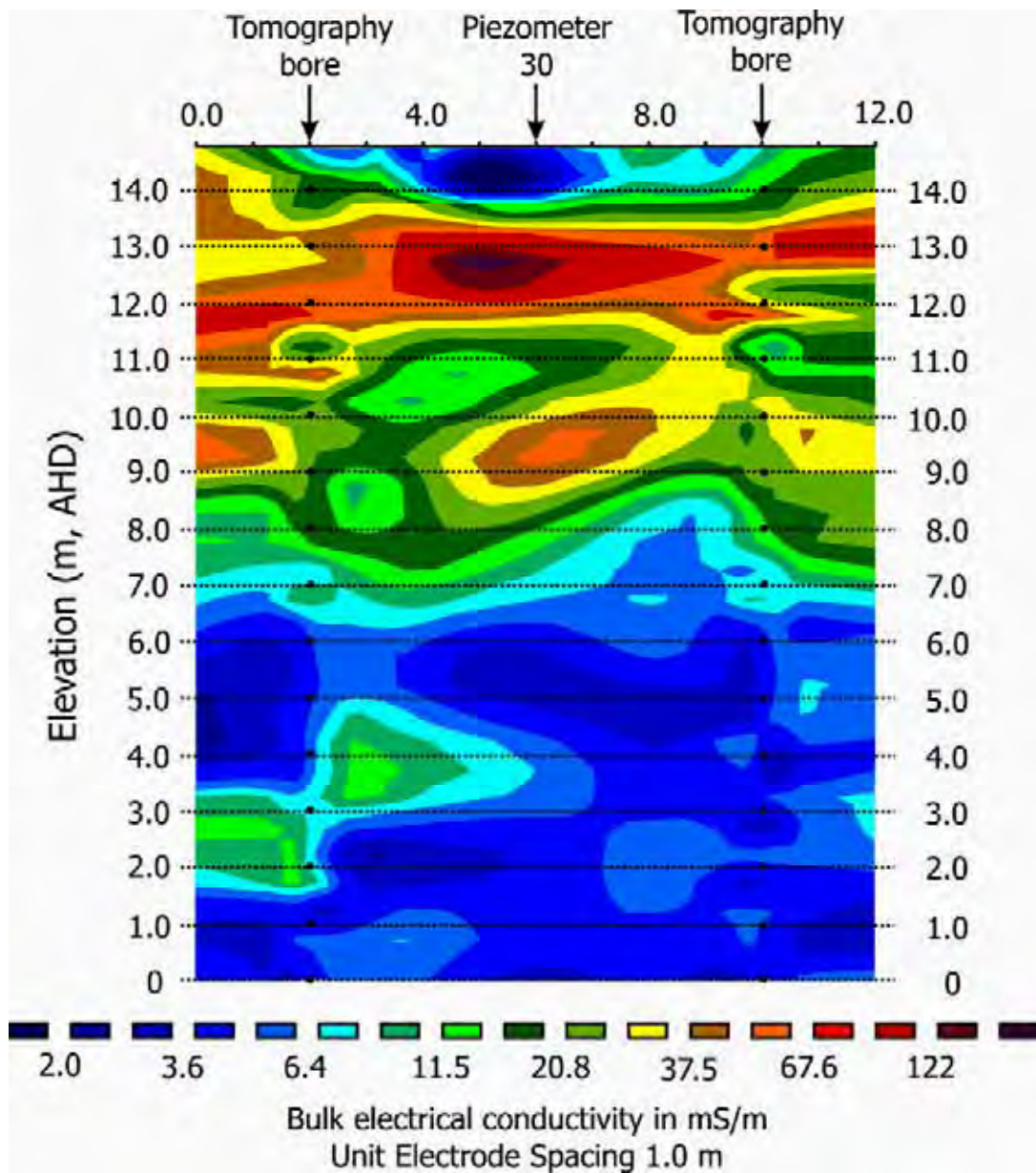


Figure 6.14 Conductivity section produced using cross-borehole tomography.

CHAPTER 7: SOIL CHEMISTRY

Soil samples were collected and analysed from both the piezometer installation boreholes and the marsh on the eastern shore of Lachlan Pond No.5, downgradient from Astrolabe Park, to determine whether trace metals were being transported in the leachate plume and accumulating at the groundwater discharge zone. While the presence of reduced sulfur and elevated concentrations of carbonate species in the leachate plume would normally result in the precipitation of dissolved metal species as insoluble metal sulfides and carbonates, there is a growing body of evidence that metals are readily sorbed onto organic and inorganic colloids, or complexed by organic and inorganic ligands, and can be mobilised in environments where they normally would be precipitated (US EPA, 1992; Gounaris, 1993; Christensen *et al.*, 1996 and 1999; Christensen and Christensen, 1999). This phenomenon is thought to be especially relevant to landfill leachate plumes, which are rich in colloidal material and dissolved organic ligands (Chian and DeWalle, 1977; Christensen *et al.*, 1998). The accumulation of heavy metals in the shallow sediments of an aquatic environment could pose a significant threat to the health of the ecosystem. The results of the soil investigation are discussed in the following sections.

7.1 SOIL CLASSIFICATION

The soil samples collected from the piezometer installation bores and shallow marsh cores were described in the field using a general field classification system based on grain size composition, colour and presence of organic material. A more detailed laboratory classification was not undertaken, as the objective of the soil sampling program was simply to investigate the potential accumulation of metals in the groundwater discharge zone, rather than determine the specific adsorptive association between the metals and soil types. A summary of the soil samples collected and analysed is presented in Table 7.1. The lithologic descriptions of the piezometer borehole soil samples and shallow marsh cores are presented in Tables 7.2 and 7.3, and lithologic logs for the shallow marsh cores are graphically presented in Figure 7.2.

Table 7.1 Location and depth of piezometer borehole and marsh core soil samples.

Core ID	<u>Marsh core soil sample depth interval (cm, bgs)</u>								
	0-10	10-20	20-30	30-40	40-50	50-60	60-70	70-80	80-90
M1	X	X	X	X	X	X			
M2	X	X	X	X	X	X	X		
M3	X	X	X	X	X	X	X	X	X
M4	X	X	X	X	X	X	X	X	X
M5	X	X	X	X	X	X			
M6	X	X	X	X	X				
Borehole ID	Borehole sample depth (m, bgs/AHD)								
1	2.5/18.2	5.0/15.7	7.5/13.2						
2	2.5/16.0	5.0/13.5	7.5/11.0						
3	3.0/12.8	5.8/10.0							
4	3.5/13.8	5.5/11.8	10.5/6.8						
4A	2.5/12.8	7.5/7.8							
6	6.5/9.2	12.5/3.2							

Table 7.2 Classification of soil samples from piezometer installation boreholes according to the Unified Soil Classification System (ASTM, 2003).

Sample ID	Soil type	Soil colour ¹		Description
1/2.5	CL	10YR6/4	Light yellowish brown	Sandy lean clay w/ pebbles and debris
1/5.0	CL	10YR6/4	Light yellowish brown	Sandy lean clay w/ pebbles and debris
1/7.5	CL	5YR6/4	Light reddish brown	Sandy lean clay
2/2.5	SP-SM	10YR6/4	Light yellowish brown	Well sorted fine sand with silt
2/5.0	SP	10YR7/4	Very pale brown	Well sorted fine sand
2/7.5	SP	10YR5/2	Greyish brown	Well sorted fine sand
3/3.0	SP	10YR6/4	Light yellowish brown	Well sorted fine sand
3/5.8	SP	10YR6/3	Pale brown	Well sorted fine sand
4/3.5	SP	2.5Y7/4	Pale yellow	Well sorted fine sand
4/5.5	SP	10YR7/2	Light gray	Well sorted fine sand
4/10.5	SP	10YR6/2	Light brownish gray	Well sorted fine sand
4A/2.5	SP-SM	10YR3/2	Very dark grayish brown	Well sorted fine sand with silt
4A/7.5	SP-SM	10YR2/2	Very dark brown	Well sorted fine sand with silt
6/6.5	SP	10YR6/3	Pale brown	Well sorted fine sand
6/12.5	SP-SM	10YR2/2	Very dark brown	Well sorted fine sand with silt

1. Soil colour determined using Munsell soil colour charts.

Table 7.3 Lithological classification of the shallow marsh cores according to the Unified Soil Classification System (ASTM, 2003).

Unified Soil Classification System (USCS, 2008).

Core ID	Soil type	Soil colour ¹		Description
<u>Core M1</u>				
0-15	OL	10YR2/1	Black	Organic clay
15-35	SP	5YR4/4	Reddish brown	Well sorted fine sand
35-60	SP	10YR5/2	Grayish brown	Well sorted fine sand
<u>Core M2</u>				
0-10	OL	10YR2/1	Black	Organic clay
10-20	SP	5YR4/4	Reddish brown	Well sorted fine sand
20-50	SP-SM	10YR5/2	Grayish brown	Well sorted fine sand with silt
50-70	SP	5YR5/3	Reddish brown	Well sorted fine sand
<u>Core M3</u>				
0-10	OL	10YR2/1	Black	Organic clay
10-40	SP	5YR4/4	Reddish brown	Well sorted fine sand
40-50	SP-SC	10YR2/2	Very dark brown	Well sorted fine sand with clay
50-90	CL	10YR2/2	Very dark brown	Lean clay
<u>Core M4</u>				
0-30	OL	10YR2/1	Black	Organic clay
30-50	SP	10YR5/2	Grayish brown	Well sorted fine sand
50-60	SP	10YR6/2	Light brownish gray	Well sorted fine sand
60-85	SP-SM	10YR4/1	Dark gray	Well sorted fine sand with silt
<u>Core M5</u>				
0-25	OL	10YR2/1	Black	Organic clay
25-60	SP-SC	10YR3/2	Very dark grayish brown	Well sorted fine sand with clay lenses
<u>Core M6</u>				
0-10	OL	10YR2/1	Black	Organic clay
10-40	SP	10YR5/2	Grayish brown	Well sorted fine sand
40-50	SP-SM	10YR2/2	Very dark brown	Well sorted fine sand with silt

1. Soil colour determined using Munsell soil colour charts.

Nearly all of the piezometer borehole soil samples are comprised of fine sand of varying colours, with very minor components of silt, clay or organic material. A notable exception is the piezometer 1 soil samples, which are comprised primarily of clay mixed with waste debris. This significant clay fraction may help to explain the elevated metals concentrations observed for these samples.

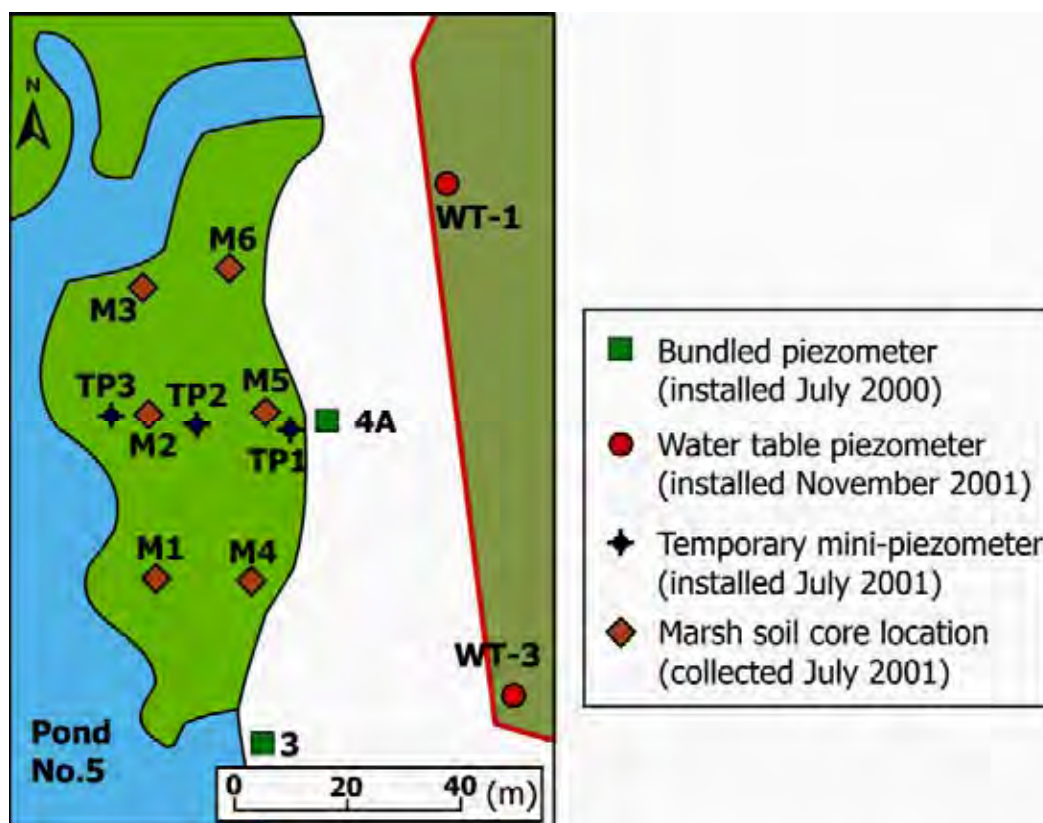


Figure 7.1 Location of shallow marsh cores.

The marsh cores are comprised of four distinct lithologic units, some of which occur in a range of colours. The units are: (1) black decaying organic matter, which comprises the surface interval of each core; (2) fine to medium sand in a variety of colours; (3) a brown fine silty sand, and (4) dark brown clay with evidence of organic material.

The predominant lithologic unit in these cores is the fine to medium sand unit, which occurs at various depths and thicknesses. While there doesn't appear to be a defining trend for the occurrence of the various sand colours, it is notable that the iron-stained sands only appear in the cores furthest from the pond shoreline (M1 to M3; Figure 7.1). This may result from the mixing of strongly reduced, leachate-impacted groundwater, containing an elevated concentration of reduced iron (as observed immediately adjacent to the marsh at piezometer 4A), with oxidised pond water, resulting in the precipitation of iron (oxy)hydroxides. Similar accumulations of iron (oxy)hydroxide precipitates were observed in Denmark at the interface of a leachate plume with oxidised regional groundwater (Heron and Christensen, 1995; Heron et al., 1998), and were interpreted as natural redox buffering zones between the leachate plume and the surrounding

uncontaminated environment. In fact, groundwater samples collected from the three shallow marsh piezometers identify progressively more oxidising groundwater conditions along a transect between the shore and the edge of the vegetation mass. Mixing of leachate-impacted groundwater and pond water was further demonstrated with a Piper plot of groundwater, marsh water and pond water (refer to Figure 8.1).

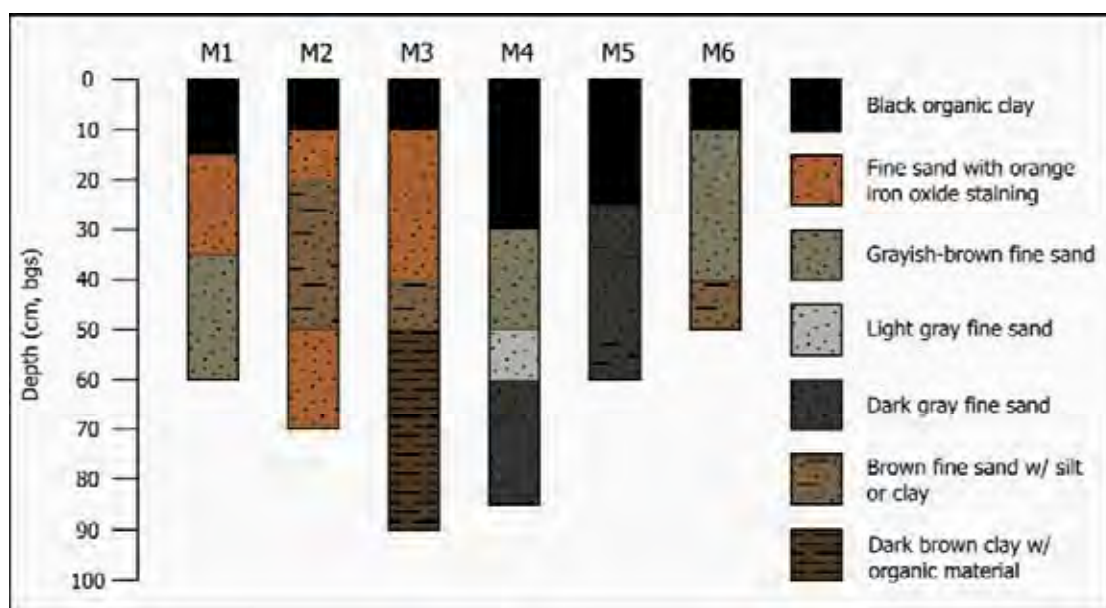


Figure 7.2 Lithological logs of the marsh soil cores.

With regards to correlation of soil lithology with geophysical (gamma) logs, the soil samples for the piezometer installation boreholes were collected from the augers of the drill rig rather than from a core-barrel. As such, the representative depth range of these soil samples is approximate at best and is not suitable for correlation with the more accurate borehole geophysical results. The marsh cores were collected with a hand-held coring device and as such provide exact representations of the surface lithology in the marsh, but geophysical logging was not undertaken in the marsh for correlation.

The results of the metals extracts indicate that iron is still present at considerable concentrations in the cores closest to the shore (M4 to M6; Figure 7.1), suggesting that the absence of oxidised iron in these cores is due to a lack of mixing with oxidised water as opposed to the absence of iron in the soil pore water. Indeed, the greyish colours of the sands in cores M4 to M6 may indicate the precipitation of FeS in a strongly reducing environment (FeS is black, but when dispersed amongst quartz sand

would appear grey). The abundant (oxy)hydroxide precipitates observed in the outer marsh sediments likely represent a zone of redox buffering at the interface of the leachate-impacted groundwater and the oxidised pond waters, with the oxidised pond waters providing a pool of electron acceptors sufficient to counter-balance the reducing capacity of the leachate plume.

The decaying organic matter that forms the upper interval of each core is presumably the result of dead and charred vegetation within the marsh (the vegetation is annually burned back to control the proliferation of 'exotic' plant species), and varies in thickness from 10 to 30 cm. The establishment of strongly reducing environments in wetlands is a well-published phenomenon (eg. Whitcar et al., 1986; Hornibrook et al., 2000), and this layer appears to provide a dual effect of maintaining a reducing environment in the subsurface, as evidenced by the strongly reducing Eh values of groundwater in the shallow marsh piezometers, and is acting as a medium for the adsorption and long-term accumulation of metals that are discharging with the leachate plume.

The brown fine silty sand unit is present in cores M2, M3 and M6. It occurs in identical intervals in cores M3 and M6, which have nearly identical lithologies for the upper 50cm, and as a thicker layer in M2. The results of the soil extracts indicate that the adsorptive capacity of these silty intervals is slightly elevated compared to the sand intervals, but not nearly to the degree of the organic layers.

The dark brown organic clay layer only appears at the bottom of core M3, although the lithologic similarity of this core to M6 and the close proximity of these sample locations may suggest that this unit underlays the sediments recovered in core M6 as well. This unit is characterised by generally elevated metals concentrations, in some cases higher than those of the upper organic layer.

The lithologic sequences observed in the shallow marsh cores are varied, but two important observations are evident: (1) the iron oxide/hydroxide stained sand intervals only occur in the outer portion of the marsh sediments, where the reducing groundwater of the leachate plume encounters the oxidised pond water, and (2) there appears to be a

correlation between the metals accumulation and the fine-grained units, especially the organic matter and the clay layer, as will be discussed further in the following sections.

7.2 RESULTS OF SOIL METALS EXTRACTION

The soil samples collected during the installation of the new bundled piezometers and from the shallow marsh cores were subjected to a hot acid extraction process according to EPA Method 3050B, as described in Chapter 5, to determine the total extractable concentrations of selected metals within the landfill (piezometers 1 and 2), along the shore of Lachlan Pond No.5 immediately before the discharge zone (piezometers 3, 4, 4A and 6), and within the groundwater discharge zone (M1 – M6). The extracts were analysed for nine metals, including Cd, Cr, Cu, Fe, Mn, Ni, Pb, Se and Zn (Se is technically a metalloid) using the ICP-AES method. The metals concentrations in soil were calculated by multiplying the concentrations in the extract by the volume of the extract to obtain the total metals extracted, and then dividing by the *oven-dried* mass of the sample used in the extraction process to calculate the metals concentrations in milligrams per kilogram (mg/kg) of soil.

7.2.1 Piezometer installation boreholes

The results of the metals extracts from the landfill and pond shoreline bore locations are provided in Table 7.4, and are presented graphically in Figures 7.3 and 7.4. In most cases, the highest metals concentrations were encountered in the soil samples collected from the installation borehole for piezometer 1, with the samples exhibiting the highest concentrations for seven of the nine metals analysed (the two exceptions are chromium, which occurs at an anomalously high concentration in one sample from piezometer 6, and selenium, which was not detected in the landfill boreholes). This is not surprising; as the drilling at this bore location was hindered by the presence of an excessive amount of metal scrap in the waste layers. Also, the soil samples collected at piezometer 1 are comprised primarily of clay, which has an elevated adsorptive capacity. In general, the metals concentrations in the samples from the piezometer 2 borehole were equal to or slightly greater than those from samples collected from the pond shoreline boreholes. This distribution highlights the inherent variability in waste disposal patterns in

landfills, and underscores the difficulty in determining ‘representative’ metals concentrations for the landfill without a large number of samples.

Table 7.4 Metals concentrations in soil samples collected from the piezometer installation bores (mg/kg).

Sample ID	Cd	Cr	Cu	Fe	Mn	Ni	Pb	Se	Zn
HILs ¹	40	200	2000	NA	3000	600	600	NA	14000
EILs ²	3	400	100	NA	500	60	600	NA	200
1/2.5	0.54	10.31	42.34	21406	78.72	9.20	51.85	<1.0	76.40
1/5.0	0.37	8.43	26.64	18794	55.67	4.80	196.08	<1.0	137.75
1/75	0.44	10.25	22.25	32852	23.36	2.24	32.87	<1.0	28.76
1/7.5 (D) ³	0.52	9.66	20.84	32007	22.26	2.23	34.96	<1.0	26.11
2/2.5	0.28	2.74	3.13	1874	9.24	1.58	30.73	<1.0	14.83
2/5.0	0.27	3.01	1.96	1386	2.07	1.37	3.92	<1.0	4.36
2/7.5	0.31	8.67	4.55	3211	25.03	1.97	21.67	<1.0	15.51
3/3.0	0.28	3.25	1.62	1116	1.44	0.95	4.19	0.98	4.55
3/3.0 (D) ³	0.27	3.11	1.50	1150	1.44	1.06	3.44	0.95	4.40
3/5.8	0.26	3.22	1.84	811	2.89	0.81	3.73	<1.0	4.70
4/3.5	0.25	2.49	1.10	605	7.85	0.75	1.98	<1.0	7.38
4/5.5	0.27	2.06	1.13	414	1.04	0.73	1.41	1.39	2.86
4/10.5	0.26	2.52	4.21	1000	4.14	1.06	5.13	0.95	6.77
4A/2.5	0.34	3.13	11.75	1457	9.69	1.28	73.99	<1.0	45.43
4A/7.5	0.24	2.49	2.00	385	3.78	1.15	6.69	1.67	5.51
6/6.5	0.27	11.24	1.52	658	1.20	0.86	3.74	1.09	2.18
6/12.5	0.24	3.86	1.10	359	1.17	0.60	1.42	1.44	1.78

1. HILs = NEPC (1999) Health Investigation Levels, human exposure setting ‘E’: parks, recreational open space and playing fields (Table 5-A, Soil Investigation Levels).

2. EILs = NEPC (1999) interim urban Ecological Investigation Levels (Table 5-A, Soil Investigation Levels).

3. (D) = duplicate sample, used as a quality control measure for the laboratory analysis procedure.

The results from the pond shoreline soil samples are relatively uniform, characterised by low concentrations of all analysed metals as would be expected in aquifer material with a low percentage of clay and organic matter, although a couple exceptions are present. The soil sample collected from 2.5 m depth at piezometer 4A has the highest concentrations amongst the pond shoreline samples for seven of the nine metals analysed. This may be due to the location of this piezometer, directly adjacent to the vegetation mass that defines the dominant groundwater discharge zone along the shore of Lachlan Pond No.5.

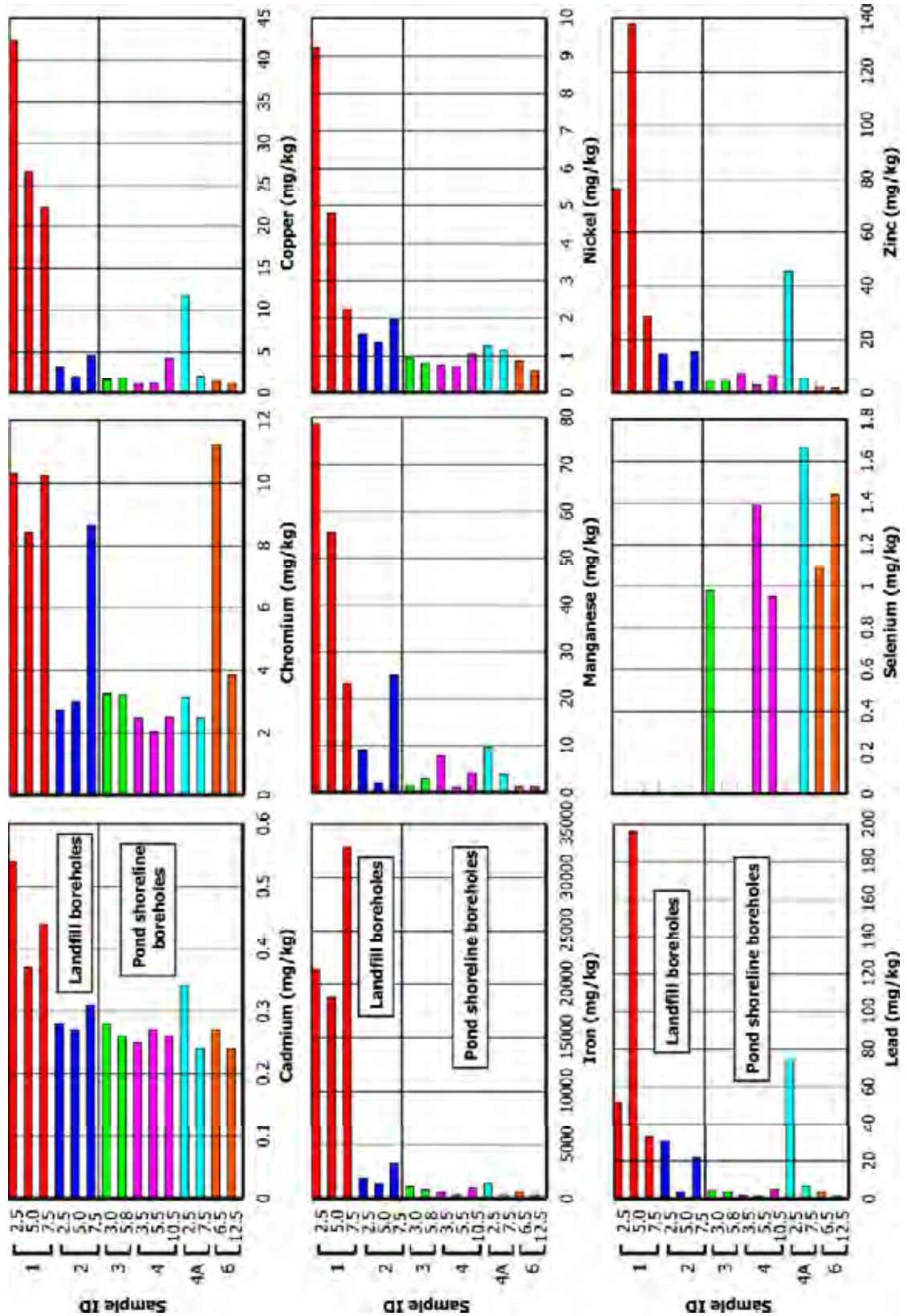


Figure 7.3 Metals concentrations in soil samples from landfill and pond shoreline piezometer installation boreholes (colours used to differentiate data between different boreholes).

It is of interest to note that selenium was present at elevated concentrations in soil samples from the pond shoreline, but was absent in landfill soil samples. This may be due to the fact that selenium is a redox-sensitive metalloid (such as arsenic), and a potential change in redox conditions at the pond boundary may have induced a change in redox state that resulted in increased adsorption. However, selenium speciation was not undertaken and as such a robust assessment of the reason for its accumulation at the shoreline and absence in the landfill is beyond the scope of the current research.

7.2.2 Shallow marsh cores

Shallow soil cores were collected from six locations within the marsh area along the shore of Lachlan Pond No.5, believed to be a discharge zone for leachate-impacted groundwater (Figure 7.1). The results of the marsh core analyses are summarized in Table 7.5, and presented graphically in Figures 7.5 and 7.6. In general, the maximum metals concentrations in the marsh cores are significantly higher than those in either the landfill or pond shoreline soil samples (Figure 7.3).

Table 7.5 Metals concentrations in soil samples collected from the shallow marsh cores (mg/kg).

Sample ID	Cd	Cr	Cu	Fe	Mn	Ni	Pb	Se	Zn
HILs ¹	40	200	2000	NA	3000	600	600	NA	14000
EILs ²	3	400	100	NA	500	60	600	NA	200
<i>M1</i>									
0-10	3.26	31.43	193.28	22749	1629.05	22.96	573.44	<1.0	863.26
10-20	0.58	21.44	51.34	34476	215.43	22.91	112.74	<1.0	230.46
20-30	0.47	29.63	65.82	45506	316.33	35.52	185.95	<1.0	317.07
30-40	0.37	23.80	48.05	26930	204.70	27.57	111.32	<1.0	210.75
40-50	0.57	15.25	48.30	15691	125.00	16.15	134.10	<1.0	220.46
50-60	0.56	10.43	30.69	10449	104.43	9.48	110.11	<1.0	242.74
<i>M2</i>									
0-10	2.59	13.27	120.97	11304	293.69	13.57	300.46	<1.0	433.46
10-20	0.48	7.14	31.81	7820	61.16	9.24	195.04	<1.0	193.57
20-30	0.55	6.97	29.96	5741	44.86	6.77	194.60	<1.0	155.96
30-40	0.76	8.99	42.48	7571	60.24	7.69	317.00	<1.0	225.57
40-50	0.64	7.50	38.54	11203	100.00	9.27	221.93	<1.0	219.07
50-60	0.73	16.06	49.34	10629	131.84	9.31	400.64	<1.0	268.33
60-70	0.54	9.39	34.40	7823	70.19	7.47	221.45	<1.0	181.27
<i>M3</i>									
0-10	2.20	16.13	98.30	16186	269.74	14.46	365.04	<1.0	431.26
10-20	0.58	11.40	48.72	13268	72.46	10.99	355.63	<1.0	283.52
20-30	0.61	8.80	32.76	7441	53.27	7.98	261.87	<1.0	219.10

Table 7.5 Metals concentrations in soil samples collected from the shallow marsh cores (mg/kg).

Sample ID	Cd	Cr	Cu	Fe	Mn	Ni	Pb	Se	Zn
HILs ¹	40	200	2000	NA	3000	600	600	NA	14000
EILs ²	3	400	100	NA	500	60	600	NA	200
30-40	0.58	12.13	62.96	9944	110.51	10.69	219.32	<1.0	252.86
40-50	0.79	22.36	54.26	10087	87.94	9.62	372.56	<1.0	260.48
50-60	1.60	48.80	82.90	16090	73.51	16.54	411.43	<1.0	398.29
60-70	1.68	18.25	70.01	6395	19.80	7.89	206.58	<1.0	561.72
70-80	2.61	21.48	188.69	12986	44.58	12.45	393.52	<1.0	918.62
80-90	1.33	26.29	106.03	15689	67.71	11.02	191.03	<1.0	261.06
<i>M4</i>									
0-10	2.93	11.21	95.15	12355	100.97	9.52	314.42	<1.0	306.27
10-20	2.59	16.17	111.22	12367	128.88	9.58	430.30	<1.0	260.74
20-30	1.12	18.75	105.92	12121	125.77	5.52	400.82	<1.0	166.13
30-40	0.27	1.22	5.00	436	3.15	0.33	10.57	1.19	5.92
40-50	0.31	1.10	2.74	829	4.45	0.32	17.87	1.12	5.73
50-60	0.26	0.82	1.41	394	<1.0	0.49	9.43	<1.0	2.10
60-70	0.33	1.01	1.49	1735	10.06	0.47	2.99	<1.0	11.41
70-80	0.31	0.94	1.26	1036	4.77	0.76	1.64	<1.0	1.53
80-85	0.28	0.95	1.79	473	3.55	0.44	4.26	0.92	4.72
<i>M5</i>									
0-10	1.92	22.43	67.53	62523	493.78	19.76	217.05	<1.0	513.60
10-20	2.21	43.26	98.36	31343	82.78	18.76	525.02	<1.0	739.67
20-30	1.02	8.80	41.47	6324	28.81	4.91	191.75	<1.0	228.59
30-40	1.89	19.46	144.72	12387	57.13	12.48	497.31	<1.0	596.43
40-50	0.96	9.20	57.88	6490	35.62	6.68	189.03	<1.0	231.55
50-60	1.79	14.85	120.06	12174	73.65	11.58	290.41	<1.0	585.86
<i>M6</i>									
0-10	2.11	61.22	176.89	18375	52.17	12.98	658.64	<1.0	547.28
10-20	0.36	3.21	10.21	1590	3.98	0.66	30.71	0.95	24.13
20-30	0.38	1.67	3.77	1180	1.48	0.42	10.36	1.01	14.59
30-40	0.30	2.52	1.08	3783	2.17	0.36	2.28	<1.0	7.34
40-50	0.39	10.60	1.36	11832	11.96	2.21	6.09	<1.0	3.66

1. HILs = NEPC (1999) Health Investigation Levels, human exposure setting 'E': parks, recreational open space and playing fields (Table 5-A, Soil Investigation Levels).

2. EILs = NEPC (1999) interim urban Ecological Investigation Levels (Table 5-A, Soil Investigation Levels).

3. (D) = duplicate sample, used as a quality control measure for the laboratory analysis procedure.

Most of the soil cores exhibit the highest metals concentrations near the ground surface, although elevated concentrations are also present at discrete depths in the profiles of some cores. Based on a qualitative assessment using the lithological logging results, it appears that there is a strong correlation between the presence of organic material and clay in the sediment and accumulation of metals (particularly at the interfaces of different lithologies). The upper 10 to 20 cm of each core is comprised almost completely of decaying organic matter, which has a strong tendency to for metals sorption (US EPA, 1992; Christensen et al., 2001). Metals accumulation observed at

discrete depths in the profiles coincide with the presence of fine (silt or clay) lenses within the core (Table 7.3; Figures 7.2, 7.5 7.6), which have been demonstrated to have significant adsorptive capacities through previous research at this site (Jankowski and Knight, 1991). There also appears to be a correlation between elevated metals concentrations in zones of observed iron oxide/hydroxide precipitation (identified qualitatively by orange staining in sand profiles).

In contrast, the distinctly lower concentrations of metals in soil samples collected both in the landfill and from the shore of the pond suggests that metals are mobilised through the aquifer with the leachate plume, likely either as complexes or adsorbed to colloids (eg. Gounaris et al., 1993; Christensen et al., 1999; Christensen and Christensen, 1999; Christensen and Christensen, 2000), and then accumulating in the marsh sediments as the groundwater discharges into Lachlan Pond No.5.

It is important to note that deposition of heavy metals associated with stormwater runoff discharging into the pond during high intensity rainfall events likely provides an additional, unquantifiable contribution to the metals accumulating in the shallow marsh deposits, and differentiating between potential surface and groundwater contributions is beyond the scope of this research. However, considering that measured hydraulic head values one meter below ground level (observed in the temporary marsh piezometers) were higher than ground level during a period of limited rainfall, and the presence of dissolved metals in shallow groundwater at the edge of the pond was confirmed through trace analysis using ICP-MS at piezometer 4A (refer to Appendix B), there is sufficient qualitative evidence to suggest that the marsh sediments comprise a sink for metals present in shallow, leachate-impacted groundwater.

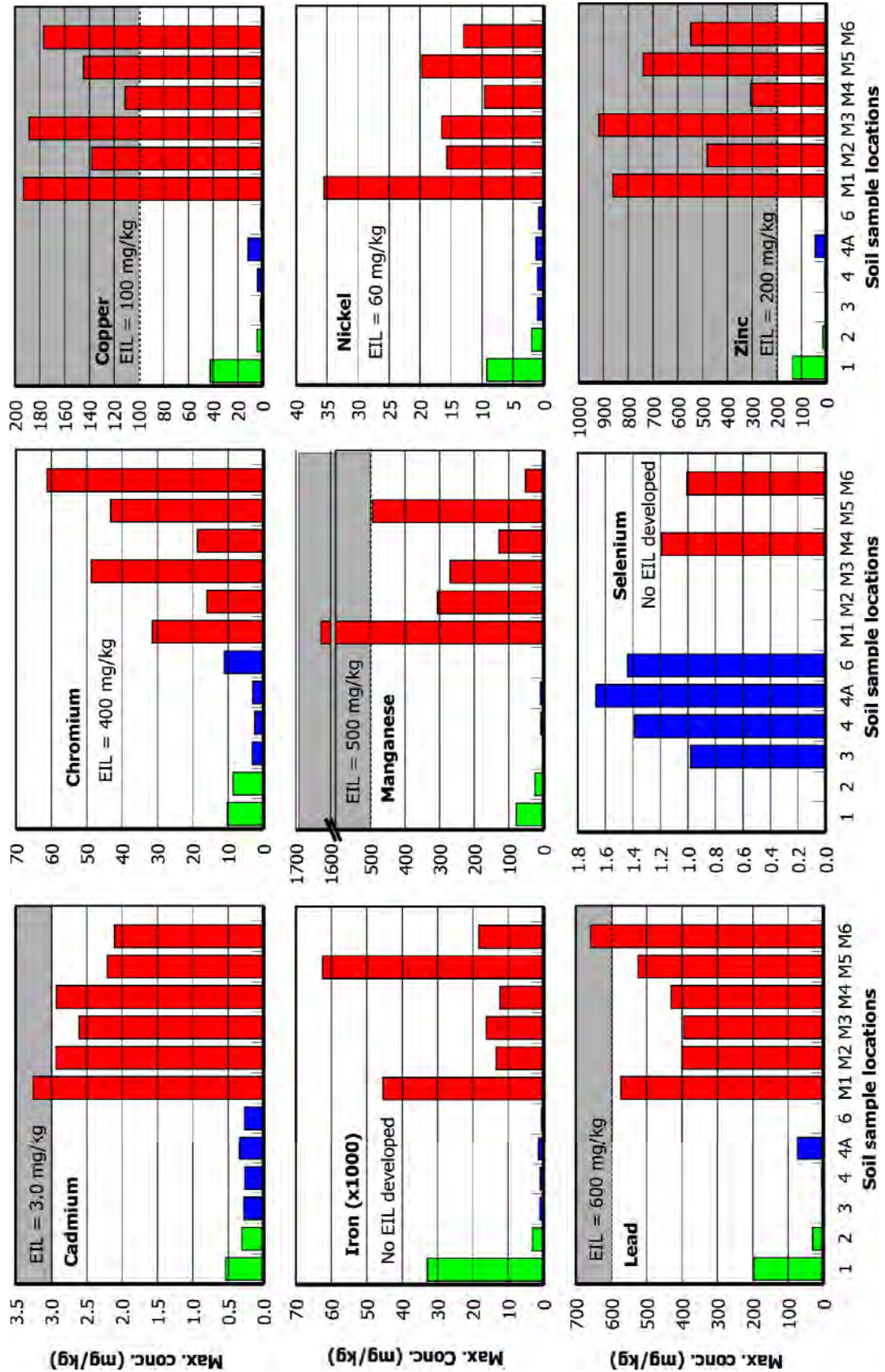


Figure 7.4 Maximum metals concentrations for each borehole (irrespective of depth), with interim urban Ecological Investigation Levels (EILs) provided, where available (NEPC, 1999). Refer to section 7.2.3 for a discussion of analytical error (colours used to differentiate data between different boreholes).

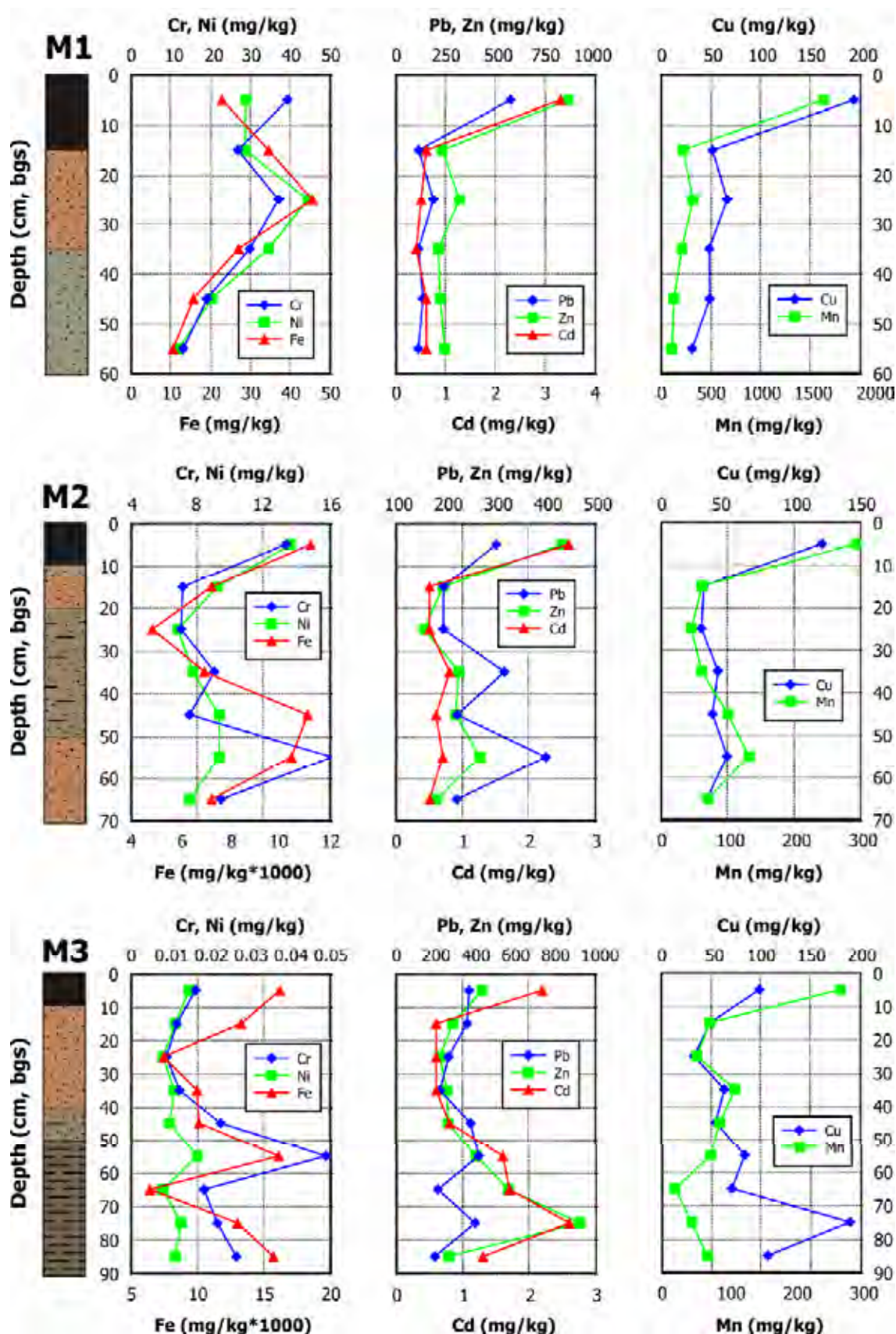


Figure 7.5 Metals concentrations at 10 cm intervals in marsh cores M1, M2 and M3 (Refer to section 7.2.3 for a discussion of analytical error).

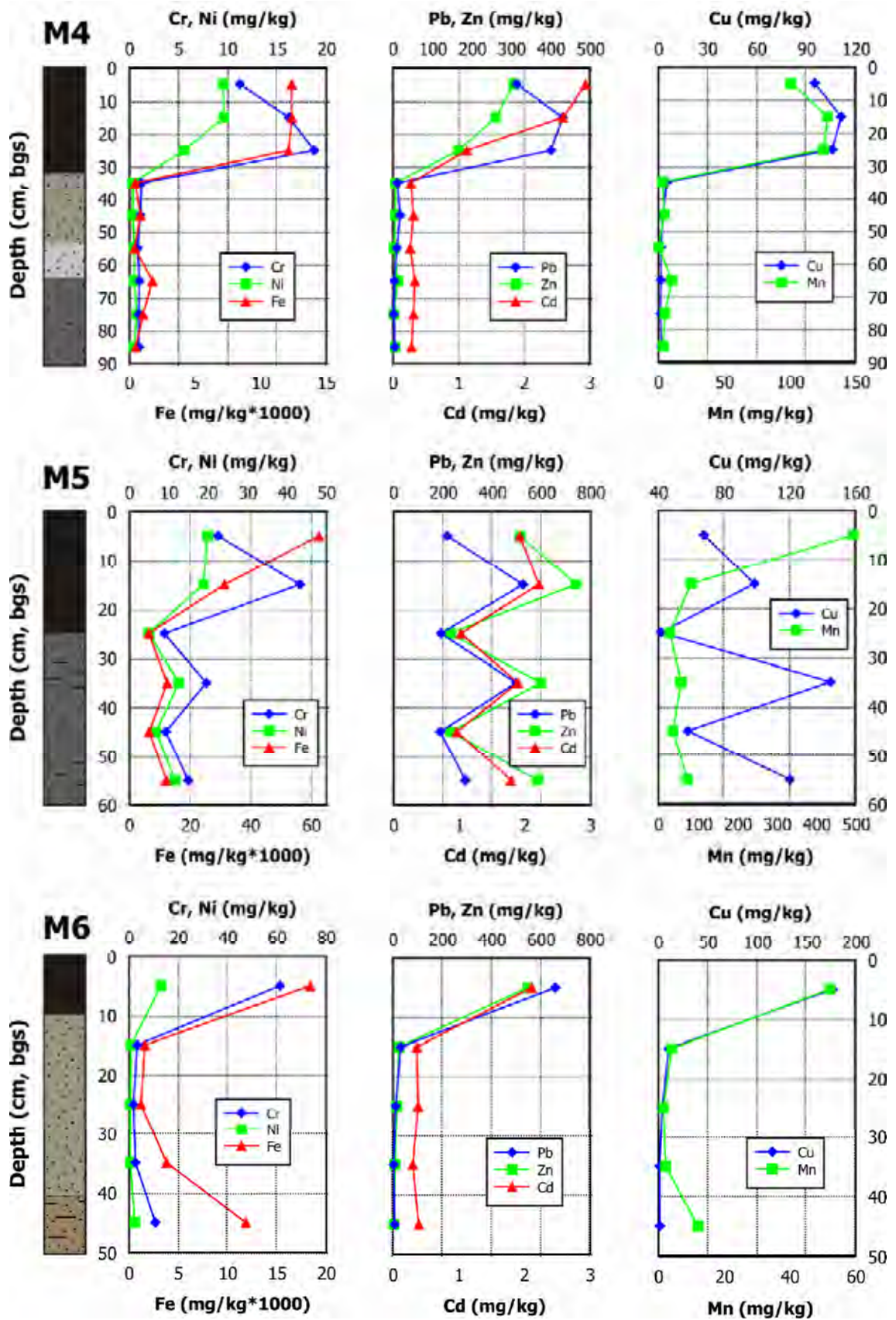


Figure 7.6 Metals concentrations at 10 cm intervals for marsh cores M4, M5 and M6 (Refer to section 7.2.3 for a discussion of analytical error).

7.2.3 Quality control data

The quality control assessment for the duplicate sample analyses is presented in Table 7.6. Duplicate samples were prepared and analysed simultaneously to assess the precision of the analytical method, and to determine how representative each of the soil sub-samples were for a single master sample. The relative percent differences were calculated from the results of the duplicate samples; due to the inherent heterogeneity in the soil sample matrix a tolerance limit of $\pm 25\%$ was considered acceptable for this study.

All of the quality control data fell within the tolerance limits except for the Ni data from sample 4/40-50. However, the detected concentration of Ni in these samples was very close to the method detection limit, and although the RPD error is large it only represents an absolute difference of 0.53 mg/kg between the two samples. This error margin would still see the concentrations below the regulatory threshold for Ni, and is therefore considered inconsequential with respect to the validity of the sample data as a whole.

Table 7.6 Quality control data for the soil analyses, expressed as the relative percent difference (RPD)¹ between original and duplicate samples ($\pm 25\%$ is considered acceptable).

Sample ID	Cd	Cr	Cu	Fe	Mn	Ni	Pb	Se	Zn
1/50-60	3.5	11.1	10.4	13.5	12.3	11.0	7.2	ND ²	6.6
2/0-10	12.7	15.2	13.3	16.8	4.4	14.6	8.8	ND	10.5
3/70-80	1.0	0.7	0.7	1.4	1.4	0.2	0.6	ND	3.1
4/40-50	11.2	9.6	9.0	9.4	11.6	91.1	17.7	3.7	13.2
1/7.5	16.3	5.9	6.5	2.6	4.8	0.5	6.2	ND	9.7
3/3.0	3.6	4.4	7.7	3.0	0.4	11.6	19.8	3.2	3.5

1. $RPD = (C2 - C1) / [(C2 + C1) / 2] * 100$

2. ND = Analyte not detected at or above method detection limits.

7.3 COMPARISON TO REGULATORY CRITERIA

The metals concentrations for the soil samples are compared to regulatory threshold values to determine the relative risk posed by these metals to both human health and the environment. The regulatory criteria for the soil contaminants are taken from the National Environmental Protection Measure (NEPM; NEPC, 1999), which lists ‘investigation levels’ for a number of contaminants commonly found in soil. The Health

Investigation Levels (HILs) are subdivided into six categories based on different land use/exposure scenarios; the category ‘E’ criteria have been used here, which are intended for ‘parks, recreational open space and playing fields’. Due to the ecological diversity exhibited throughout Australia, the Ecological Investigation Levels (EILs) are intended to be developed on a regional basis to address regional differences in naturally occurring metals concentrations and soil types. However, the regional criteria are still under development, and ‘interim urban’ EILs are provided until the regional values become available. The values for both the ‘E’ HILs and the interim urban EILs for the analysed metals are included in Tables 7.4 and 7.5 (*note that iron and selenium are not included in the NEPM soil criteria list*).

The category E HILs are only slightly higher than the category A ‘standard residential’ levels, which takes into account the potential for consumption of home-grown garden produce and direct dermal contact with garden soil (and are also applicable for children’s day- care centres, kindergartens, preschools and primary schools). The category E HILs also take into account the potential for direct dermal contact with soil, but no consumption of produce grown on the property and relatively limited exposure times. Even with these conservative HILs, all of the metals concentrations for all of the soil samples are below the threshold values.

The interim urban EILs are considerably more conservative than the HILs, as they consider chronic exposure of flora and fauna to the contaminants of concern. This is particularly applicable to an aquatic ecosystem, where contaminants may directly enter the food chain and then bioaccumulate in the aquatic organisms with detrimental consequences for the health of the ecosystem as a whole. The applicable EILs are indicated on Figure 7.4, which indicates that the EILs are exceeded for five of the seven analysed metals for which EILs are available. The specific metals concentrations in excess of the EILs for each sample are highlighted in Table 7.5, and Table 7.8 summarizes the total number of samples from each core that exceed the EILs for each metal.

The first significant observation is that none of the piezometer installation soil samples exceed either the EILs or HILs for any of the metals. This is particularly surprising for

the landfill soil samples considering the amount of metal scrap that was encountered during the installation of these piezometers. This is most likely an indication that the near neutral pH range of the leachate-impacted groundwater in the landfill is not conducive to the aggressive dissolution of metal waste in the landfill. Another possible explanation is that free metal ions in the leachate adsorb onto colloidal matter or complex in the presence of organic and inorganic ligands in preference to bonding to soil surfaces in the landfill. Regardless of the specific mechanism responsible, the concentrations of most of the metals in the landfill soil samples are only slightly elevated with respect to those from the pond shoreline soil samples, which are well below even the average ‘background’ ranges for metals in the environment listed in the NEPM (NEPC, 1999).

Table 7.7 Summary of samples exceeding EILs for each metal.

Core ID	Cd	Cr	Cu	Fe	Mn	Ni	Pb	Se	Zn
M1	1	--	1	N/A ¹	1	--	--	N/A	7 ^{2,3}
M2	--	--	2 ²	N/A	--	--	--	N/A	5
M3	--	--	3 ²	N/A	--	--	--	N/A	10 ^{2,3}
M4	--	--	2	N/A	--	--	--	N/A	2
M5	--	--	2	N/A	--	--	--	N/A	6 ²
M6	--	--	1	N/A	--	--	1	N/A	1

1. N/A = Not applicable; no existing EIL for this element.

2. Includes one set of duplicate samples from the same depth interval.

3. Indicates all samples from this core are above EIL threshold for the selected metal.

Table 7.7 clearly indicates that copper and zinc are the primary metals of concern in the marsh cores, with the concentrations in excess of the EILs in at least one soil sample from each of the cores. Zinc is especially elevated, with most or all of the core samples well in excess of the EIL in four of the six cores. The surface sample from core M1 has a Cd concentration just above the EIL, and what appears to be an anomalously high concentration of Mn (over 3 times greater than the next closest concentration). The surface sample from core M6 has a Pb concentration just above the EIL.

It is interesting to note that the bulk of the metals concentrations in excess of the EILs in marsh cores M1, M2, M4 and M6 occur in the surface sample. Cores M2, M3 and M5 also have discrete intervals of elevated metals concentrations below the surface. This poses the question of whether all of these metals originate in the landfill or if a certain percentage is derived from urban stormwater runoff that discharges into Lachlan Pond

No.5, inundating the marsh area during high-intensity storm events. It is likely that a combination of discharging leachate-impacted groundwater and flooding by urban stormwater runoff are contributing to the metals concentrations in the surficial organic layer in the marsh, while the discharging leachate plume is responsible for the accumulation of metals in discrete intervals below the surface. Note that the quality of stormwater runoff entering the pond was not assessed during this research, and as such a robust argument cannot be put forth for this metals deposition mechanism, but there are several lines of strong qualitative evidence to support the occurrence of the groundwater deposition mechanism (refer to discussion in Section 7.2.2, p.146).

This investigation demonstrates that metals are accumulating in the sediments of the marsh, where leachate-impacted groundwater discharges into Lachlan Pond No.5. Metals concentrations are significantly higher in the surficial marsh sediments than in soil samples collected from the landfill and from within the leachate plume along the shore of the pond. The metals are primarily associated with the surficial layer of humic material in the marsh, but can also be correlated with subsurface silt and clay lenses and iron precipitates (especially at the interfaces between different lithologies). The metals may be introduced to the marsh by the discharging leachate plume and periodic flooding by urban stormwater runoff, but it is beyond the scope of the current research to differentiate or quantify the relative contributions of the two mechanisms. Copper and zinc both appear to exceed EILs throughout the marsh, an issue that will have to be addressed for the long-term welfare of the marsh and the pond. Two additional samples also had concentrations of Cd, Mn and Pb in excess of EILs, which may merit further soil investigation to confirm the results

CHAPTER 8: HYDROCHEMICAL CLASSIFICATION AND TREND ANALYSIS

The introduction of landfill leachate to the relatively pristine Botany Sands aquifer has resulted in profound changes to the chemical composition of the groundwater below and downgradient from the landfill. These changes play a defining role in the transport and transformation of elements between the landfill and Lachlan Pond No.5. The long-term uncontrolled release of leachate has resulted in the establishment of a permanent zone of strongly reduced groundwater between the landfill and the pond in which the electron acceptors have been exhausted, which facilitates the migration of reduced species between the landfill and the pond with little attenuation along flow path from oxidation processes. This chapter focuses on two issues: 1) A description of the chemical composition of various water types at the site (uncontaminated, landfill, leachate plume, marsh, pond and rainfall) and classification of water types based on both a major elements method and statistical analysis; and 2) Discussion of the vertical and lateral distribution of leachate elements at the site, including analysis of spatial and temporal variations observed within the leachate plume between sampling events. The following chapter presents an analysis of the hydrogeochemical process contributing to the mobilisation, transport and retardation/attenuation of leachate elements between the landfill and the pond. The entire chemical data set is presented in Appendix B, while limited excerpts and statistical tables are included in this chapter to accommodate space limitations.

8.1 CHEMICAL CLASSIFICATION OF WATER SAMPLES

A total of 498 water samples were collected over six field efforts for full chemical analysis. These include 465 groundwater samples collected from the bundled piezometers within and below the leachate plume, seven groundwater samples collected from the landfill piezometers, three groundwater samples collected from the temporary piezometers installed in the marsh on the shore of Lachlan Pond No.5, 21 pond water samples, and two rainwater samples collected during the July 2001 sample event. The water samples were categorised using the statistical software package SPSS 10.0 to take into account the redox sensitive elements as well as the major elements (SPSS, 2000).

8.1.1 Statistical analysis and classification

The statistical analysis software package SPSS 10.0 was used to define groups with similar chemical characteristics for all of the samples collected during the ‘full’ sampling events. An heirarchal cluster analysis was used to classify distinct water types based on the relative concentrations of the major ions and redox sensitive elements (eg. Colby, 1993; Suk and Lee, 1999; McGuire et al., 2005; Zeng and Rasmussen, 2005). Refer to Chapter 5 (Section 5.7) for a more detailed discussion of this methodology.

The cluster analysis identified five major water types, including four groups with various degrees of leachate impact and one group representing uncontaminated groundwater. The mean values for the elements in each group are presented in Table 8.1. The groups range from highly contaminated leachate plume samples (Group 1) to uncontaminated water samples (Group 5), with a gradational transition of most elements in the intermediate groups. The mean TDS, EC and pH values decrease from Group 1 to Group 5, reflecting various degrees of contamination, while the Eh/pe trends are more ambiguous with respect to the group progression. With respect to the major elements, there is a trend of decreasing concentration from Group 1 to Group 5, and the major ionic ratios generally change to reflect the progression from Na-Cl dominance in uncontaminated water, to Ca-HCO₃⁻ dominance in leachate-impacted water. The trace elements B, Sr²⁺ and Zn²⁺ mirror the trend of the major elements, suggesting a correlation to the presence of landfill leachate, while the other trace elements exhibit more ambiguous patterns with respect to the group progression. Finally, there is a general increase in the concentrations of oxidised elements with a concomitant decrease in reduced element concentrations moving from Group 1 to 5. The major exception is S²⁻, which increases concomitantly with SO₄²⁻. This may be due to a decrease in the concentration of dissolved metals in the more oxidising groups, which form highly insoluble precipitates in the presence of S²⁻, or to an additional source of SO₄²⁻ at specific locations within the aquifer (eg. oxidation of FeS present in peaty layers). Whilst the total concentrations of most elements are lower than those reported in the bulk of international literature (refer to summary table in Christensen et al., 2001), the relative abundances exhibit a similar pattern.

Table 8.1 Mean parameter values and element concentrations for water groups determined by hierarchal cluster analysis (concentrations in mg/L).

Group: n =	1 39	2 72	3 57	4 108	5 222
<i>General parameters</i>					
EC ($\mu\text{S}/\text{cm}$)	1331	826	624	402	209
TDS	1157	692	492	307	141
pH	6.511	6.197	5.973	5.951	5.404
Eh	-129.5	-159.2	-157.5	-137.8	-117.1
pe	-2.227	-2.741	-2.714	-2.367	-2.015
<i>Major elements</i>					
Ca^{2+}	115.9	67.0	35.0	27.2	11.3
Na^+	77.1	45.8	43.0	28.9	19.1
Mg^{2+}	21.5	14.6	9.7	5.9	3.3
K^+	26.1	17.9	16.2	9.0	5.5
HCO_3^-	761.9	462.0	287.3	182.7	40.9
CO_3^{2-}	1.39	0.46	0.00	0.00	0.14
SO_4^{2-}	11.3	10.3	12.9	17.4	18.9
Cl^-	66.9	54.4	53.4	44.5	33.8
<i>Trace elements</i>					
Al^{3+}	0.140	0.062	0.077	0.060	0.118
B	0.607	0.486	0.417	0.266	0.087
Cd^{2+}	0.001	0.001	0.001	0	0
Cu^{2+}	0.007	0.009	0.008	0.010	0.013
Li^+	0.023	0.037	0.038	0.025	0.020
Ni	0.001	0	0	0	0
Pb^{2+}	0.006	0.003	0.006	0.006	0.005
PO_4^{3-}	0.438	0.591	0.468	0.384	0.300
Sr^{2+}	0.556	0.304	0.197	0.121	0.043
Zn^{2+}	0.060	0.014	0.013	0.012	0.04
<i>Redox sensitive elements</i>					
O_2	0.8	1.8	1.7	2.4	2.1
CO_2	150.3	97.3	95.1	66.6	34.5
Fe^{2+}	5.19	4.42	3.75	2.32	0.802
Mn^{2+}	0.695	0.374	0.107	0.080	0.029
NH_4^+	29.9	22.0	18.9	9.23	1.99
NO_3^-	7.5	8.4	11.0	13.0	13.9
SO_4^{2-}	11.3	10.3	12.9	17.4	18.9
S^{2-}	0.023	0.018	0.028	0.122	0.206

The statistically defined groups are not completely exclusive in terms of element ranges; there is a degree of overlap of the element maxima and minima between adjacent groups. Water samples are classified according to the group whose range best defines the *majority* of element concentrations in a given sample, and therefore a degree of overlap for some element ranges is inevitable. However, this grouping system appears to provide a valid and rational approach to categorising the large number of water samples collected at the site, and will be used for analysis of plume dynamics and total contaminant load calculations in the following chapters.

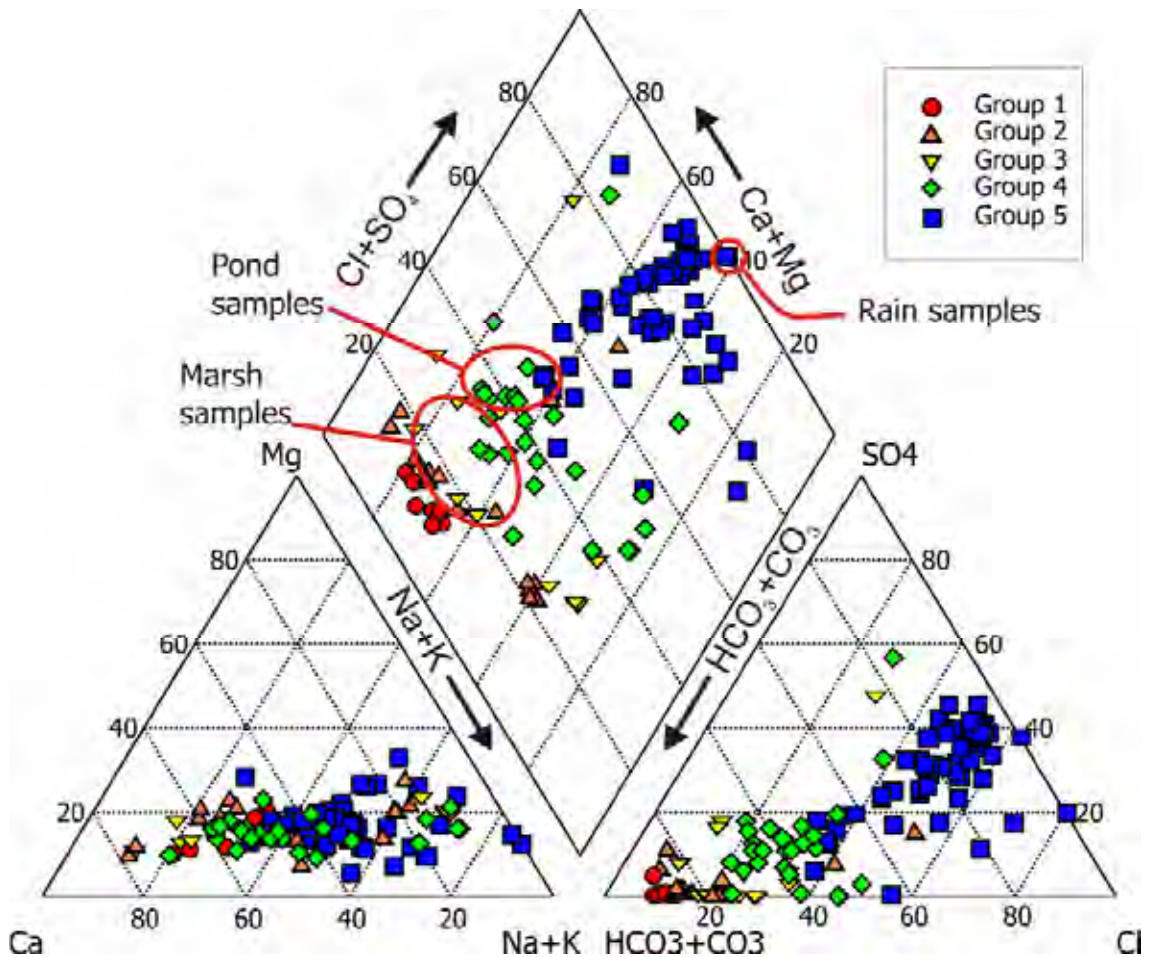


Figure 8.1 Piper diagram displaying the major ion data from the July/August 2001 monitoring event (approximate plot areas of specific water types labelled for reference).

Figure 8.1 presents a Piper diagram plot of the major ion data collected during the July/August 2001 monitoring event, with samples categorised based on their statistically derived cluster analysis groups (data from the other monitoring events plot in a similar manner). In terms of the major ions, the degree of leachate contamination can generally

be defined by the relative abundance of HCO_3^- in the sample, with the landfill samples plotting close to the HCO_3^- corner, and uncontaminated samples plotting closer to the Cl^- and SO_4^{2-} corners. The waters with varying degrees of leachate impact plot in between these zones, depending on the relative abundances of HCO_3^- , Ca^{2+} and Na^+ . In general, Ca^{2+} is dominant in the more contaminated samples and Na^+ is dominant in the uncontaminated samples, but the difference is subtle and is of relatively minor importance in terms of water classification.

It is interesting to note that the marsh and pond water samples plot well within the zone of leachate-impacted groundwater on the Piper diagram (in fact, the marsh samples plot next to the most highly contaminated leachate plume and landfill samples). This is compelling evidence to demonstrate the impact that the leachate plume has on water quality in Lachlan Pond No.5.

8.2 UNCONTAMINATED REGIONAL GROUNDWATER

Of the 465 groundwater samples collected between the landfill and the pond, 206 represent groundwater unaffected by landfill leachate. These samples represent groundwater that was collected beneath the lower confining layer that constrains the plume depth (refer to Figure 8.3). The results of these analyses are summarised and briefly discussed in the following sections, together with previous data from a bundled piezometer upgradient from Astrolabe Park (David Phillips Field [DPF]; Groves, 1999), to highlight the contrast between the leachate plume samples and the regional uncontaminated groundwater.

8.2.1 General parameters

The pH of the uncontaminated groundwater at DPF ranges from 4.01 to 5.77, with an average value of 4.78, and below the leachate plume in Astrolabe Park the pH ranges from 4.12 to 6.62, with an average value of 5.34 (Table 8.2). This is significantly more acidic than the leachate-contaminated groundwater, and reflects both the slightly acidity of local precipitation (Acworth and Jankowski, 1993) and the limited buffering capacity

Table 8.2 Statistical parameter values and element concentrations for uncontaminated groundwater from Astrolabe Park and David Phillips Field.

	Astrolabe (current; n=206)				David Phillips (Groves, 1998; n=29)			
	Min	Max	Mean	Std. Dev.	Min	Max	Mean	Std. Dev.
<i>General parameters</i>								
T(°C)	13.9	27.3	19.6	2.67	16.5	19.3	18.3	0.7
EC (µS/cm)	52.4	301	208	41.13	155	301	210	40.06
TDS	48.39	256.7	142.0	32.36	96.70	206.0	133.0	27.35
pH	4.12	6.62	5.34	0.68	4.01	5.77	4.78	0.50
Eh (mV)	-246	+106.5	-135	96.36	+8	+278	+194	61.08
pe	-4.216	1.811	-2.336	1.656	0.138	4.819	3.353	1.055
<i>Major elements</i>								
Ca ²⁺	0.039	25.60	10.4	5.82	8.96	31.04	15.40	5.57
Na ⁺	4.56	36.60	19.4	4.82	11.48	30.67	15.90	4.01
Mg ²⁺	0.221	7.19	3.24	1.32	1.86	4.29	2.97	0.720
K ⁺	1.99	11.50	5.60	1.76	1.70	4.12	2.92	0.710
HCO ₃ ⁻	1.69	104.9	31.5	23.92	1.59	46.36	15.23	14.56
SO ₄ ²⁻	0.537	43.79	19.4	9.47	14.06	42.38	26.51	6.76
Cl ⁻	10.04	57.98	34.2	6.47	16.02	32.05	22.54	3.72
<i>Trace elements</i>								
Al ³⁺	0.004	1.35	0.123	0.173	0.003	0.845	0.178	0.229
B	0.008	0.348	0.091	0.066	0.002	0.346	0.110	0.104
Cd ²⁺	0.001	0.002	0.001	0.0003	0.001	0.007	0.003	0.003
Cu ²⁺	0.002	0.369	0.014	0.036	0.001	0.020	0.007	0.006
Li ⁺	0.002	0.099	0.020	0.022	0.001	0.007	0.003	0.002
Ni	0.004	0.004	0.004	0.001	0.001	0.013	0.006	0.006
Pb ²⁺	0.004	0.019	0.007	0.004	0.005	0.048	0.021	0.023
PO ₄ ³⁻	0	4.35	0.293	0.649	0.010	0.100	0.022	0.023
Sr ²⁺	0.001	0.218	0.035	0.041	0.031	0.136	0.064	0.034
Zn ²⁺	0.002	0.256	0.036	0.083	ND	ND	ND	ND
<i>Redox sensitive elements</i>								
O ₂	0.050	7.70	1.66	2.35	0.60	8.10	3.89	2.03
CO ₂	5.28	149.6	33.9	27.25	23.8	82.4	45.4	17.73
Fe ²⁺	0	8.25	0.851	1.11	0.005*	0.179*	0.067*	0.056*
Fe ³⁺	0	4.53	0.200	0.612	ND	ND	ND	ND
Mn ²⁺	0.001	0.093	0.026	0.021	0.001	0.013	0.005	0.003
NH ₄ ⁺	0	13.5	2.16	2.57	0.000	0.030	0.009	0.006
NO ₃ ⁻	0	54.12	14.5	9.11	18	45.3	26.5	7.95
SO ₄ ²⁻	0.537	43.8	19.4	9.47	14.1	42.4	26.5	6.76
S ²⁻	0	2.25	0.223	0.405	ND	ND	ND	ND

Notes:

ND = Not detected (element below laboratory reporting limits);

* = Measured as Fe total only, but attributed to Fe²⁺.

of the aquifer sediments in the vicinity of Astrolabe Park. This pH range is typical of uncontaminated groundwater close to the Lachlan Ponds, which represents a mix of regional groundwater and fresh recharge associated with sand dunes (Acworth and Jankowski, 1993; Jankowski and Yu, 1998).

The Eh exhibits a large variation both upgradient from Astrolabe Park and beneath the leachate plume, with generally more reducing values beneath the leachate plume. Values at DPF range from +8 to +278 mV with an average value of +194 mV, and at Astrolabe Park range from -246 to +107 mV, with an average of -135 mV (Table 8.3). Note that all Eh values presented have been corrected to the standard hydrogen electrode (SHE) by entering a manual offset during calibration of the ORP probe. The oxidised environment upgradient from Astrolabe Park is typical of an uncontaminated aquifer containing little or no organic matter. A slightly more reducing environment would be expected in the uncontaminated zone beneath the leachate plume due to the depletion of oxidised elements in the landfill and plume and limited contact with fresh oxidising recharge waters, and due to the increased fraction of aquifer organic matter in the vicinity of Lachlan Pond No.5. As expected, these average Eh values are less reducing than for the leachate plume.

The mean EC and TDS values for uncontaminated water at DPF and Astrolabe Park are very similar (208 and 210 $\mu\text{S}/\text{cm}$, 142 and 133 mg/L, respectively), and are significantly lower than those of leachate-contaminated groundwater (Table 8.2). This is due to the large load of soluble salts provided by the waste in the landfill, in contrast with the relative lack of soluble mineral phases in the uncontaminated aquifer environment. The average EC values are very close to the value of 213 $\mu\text{S}/\text{cm}$ reported for uncontaminated regional groundwater upgradient from the Lachlan Ponds (Acworth and Jankowski, 1993; Jankowski and Yu, 1998).

8.2.2 Major elements

The major elements comprise the cations Ca^{2+} , Mg^{2+} , Na^+ and K^+ and the anions HCO_3^- , CO_3^{2-} , SO_4^{2-} and Cl^- ; statistical data for these elements are presented in Table 8.2. The concentrations of the major elements of the uncontaminated waters are relatively low in

both the DPF and Astrolabe Park samples, although the Astrolabe Park samples reflect a slightly higher solute load and a slightly more reducing groundwater environment. The concentrations of most of the major elements are considerably lower than in the leachate plume, due to the lack of an abundant soluble mineral source in the uncontaminated sandy aquifer. The most pronounced variation is observed with HCO_3^- , which is generated in excess in the leachate plume as a product of both microbial degradation of organic matter and dissolution of concrete rubble.

The only major element present in higher concentrations in the uncontaminated groundwater samples is SO_4^{2-} , which is depleted in the reducing environment of the leachate plume due to the consumption of electron acceptors during microbial degradation processes.

It should be noted that Jankowski and Beck (2000) previously reported significant localised variations in the chemical composition of uncontaminated groundwater in this aquifer across small vertical increments. These variations were attributed to fine-scale heterogeneities in the aquifer material, notably lenses of silty/clayey/peaty material containing shell fragments, which significantly influenced the major ion chemistry within discrete aquifer horizons. Specifically, an increase in Ca^{2+} , Mg^{2+} and HCO_3^- concentrations were attributed to dissolution of shell material (and potentially degradation of natural organic material), and an increase in SO_4^{2-} and decrease in pH was attributed to oxidation of iron sulfide minerals present in the peaty lenses. However, the maximum contributions of major ions from these processes were still orders of magnitude lower than those observed within the leachate plume.

8.2.3 Trace elements

The trace elements included in the groundwater analyses are Al^{3+} , B, Cd^{2+} , Cu^{2+} , Li^+ , Ni, Pb^{2+} , PO_4^{3-} , Sr^{2+} and Zn^{2+} , and are presented in Table 8.2. The trace elements are present in relatively similar concentrations in both the DPF and Astrolabe Park uncontaminated groundwater samples with the exception of PO_4^{3-} , for which the average value is about an order of magnitude higher beneath the leachate plume. The majority of trace elements also have similar concentrations in both the leachate plume

and uncontaminated groundwater. The heavy metals Cd^{2+} , Pb^{2+} and Zn^{2+} are present in similar concentrations in fresh and contaminated groundwater, and Cu^{2+} and Ni^{+} are actually slightly elevated in the uncontaminated samples with respect to the plume. This doesn't necessarily indicate that there is not an additional source for these elements in the landfill; attenuation processes could be capable of reducing metals concentrations in the plume to be equal to or less than background levels.

The elements B, PO_4^{3-} and Sr^{2+} are present at considerably lower concentrations in the uncontaminated groundwater, suggesting that there is an additional source for these elements in the landfill. A strong correlation between B, Sr^{2+} and the other major leachate plume indicators supports this assertion. In groundwater unaffected by landfill leachate, B and Sr^{2+} may derive from the dissolution of shell fragments, and PO_4^{3-} may be contributed through transport of fertilizers or degradation of organic material present within the aquifer.

8.2.4 Redox sensitive elements

The redox elements taken into consideration for this study are O_2 , $\text{CO}_{2(\text{aq})}$, Fe^{2+} , Fe^{3+} , Mn^{2+} , NH_4^+ , NO_3^- , SO_4^{2-} and S^{2-} ; statistical data for these elements are presented in Table 8.3. Dissolved O_2 concentrations are relatively depleted in the uncontaminated groundwater beneath the leachate plume, and are somewhat higher at DPF. This is likely due to an increase in aquifer organic material in the sandy profile close to the Lachlan Ponds, as evidenced in borehole gamma logs by Acworth (1998), Groves (1998) and Jorstad *et al.* (2004), since the landfill leachate impact is almost exclusively confined to the shallow aquifer (Jorstad and Acworth, 2006). The total $\text{CO}_{2(\text{aq})}$ concentration in the uncontaminated groundwater is lower than in the leachate plume, but it is slightly elevated with respect to HCO_3^- in the uncontaminated groundwater. This is due to the low pH buffering capacity of the uncontaminated aquifer, and contributes to the observed lower pH value.

As expected, the uncontaminated groundwater is characterised by an abundance of oxidised elements (NO_3^- and SO_4^{2-}) with respect to reduced elements (Fe^{2+} , Mn^{2+} , NH_4^+). One exception is S^{2-} , which may be present in uncontaminated groundwater due

to the oxidation of pyrite-bearing peat lenses and subsequent reduction of the resultant SO_4^{2-} . In the leachate plume, S^{2-} produced via SO_4^{2-} reduction is mostly removed from solution by precipitation as metal sulfides. In addition, the statistical correlation method has defined certain samples with slightly elevated NH_4^+ values as being uncontaminated (Table 8.2). These comprise a small number of samples from the outer fringe of the leachate plume that otherwise exhibit the characteristics of uncontaminated groundwater, and which have skewed the statistical parameters for the current study.

8.3 LANDFILL GROUNDWATER CHEMISTRY

Groundwater samples were collected from piezometers 1 and 2 to determine the water quality at the discharging edge of the landfill. The data are coupled with previous data from piezometers installed in the centre of Astrolabe Park landfill (Jankowski and Knight, 1991) as these piezometers could not be located for the current research. As expected, samples from these piezometers had the highest concentrations of leachate elements. Note that access to these piezometers was occasionally restricted due to sand and debris being stockpiled over the wells, and therefore were not sampled during each sample event. It is also interesting to note that the groundwater chemistry at the two piezometers is markedly different (Appendix B). The chemistry at piezometer 2 is similar to the centre of the leachate plume downgradient from the landfill, while piezometer 1 appears to be unique amongst the samples. Also note that the majority of research for this study is focused on the leachate plume between the landfill and the pond, and the data from these relatively few samples only serve as a rough estimate of the water quality within the landfill. Landfills are notoriously variable as contaminant sources, and a significantly enhanced monitoring network would be required to accurately characterise the variations in groundwater quality across the entire landfill (eg. Kjeldsen *et al.*, 1998b).

8.3.1 General parameters

Statistical data for the landfill groundwater samples are presented in Table 8.3. The general parameters for Jankowski and Knight (1991), Smolski (1994) and the current study are both indicative of a mature, methanogenic-phase leachate, with some subtle

Table 8.3 Statistical parameter values and element concentrations for landfill groundwater from Astrolabe Park (concentrations in mg/L unless otherwise noted).

	Current research (n=7)				Previous studies ¹ (n=8)			
	Min	Max	Mean	Std. Dev.	Min	Max	Mean	Std. Dev.
<i>General parameters</i>								
EC (µS/cm)	896	1853	1430	337	964	3830	2153	897.7
TDS	797	1570	1157	244	938.1	3361	1821	865.9
pH	6.34	9.07	7.28	0.84	5.89	7.45	6.74	0.52
Eh (mV)	-312.1	-69.2	-189.2	87.8	-196	-88	-120.4	40.2
pe	-5.401	-1.186	-3.260	1.519	-3.358	-1.513	-2.065	0.688
<i>Major elements</i>								
Ca ²⁺	15.0	128	66.3	48.0	92.1	169	123	30.5
Na ⁺	38.5	369	160	130	35.1	220	142	74.6
Mg ²⁺	0.90	29.7	18.6	12.0	9.45	67.8	28.1	18.4
K ⁺	7.56	36.9	23.2	13.1	13.2	140	54.8	42.3
HCO ₃ ⁻	97.0	1101	620	374	559	2460	1110	646
SO ₄ ²⁻	1.87	28.9	7.70	9.48	0.11	141	21.6	35.6
Cl ⁻	52.5	318	114	97.7	22.8	382	170	160
<i>Trace elements</i>								
Al ³⁺	0.008	1.30	0.541	0.618	NA	NA	NA	NA
B	0.307	0.921	0.658	0.222	NA	NA	NA	NA
Cd ²⁺	0	0.002	0.001	0.001	0.001	0.002	0.001	0.0003
Cu ²⁺	0.007	0.014	0.010	0.003	0.003	0.017	0.008	0.007
Li ⁺	0.002	0.017	0.008	0.007	NA	NA	NA	NA
Ni	ND	ND	ND	ND	NA	NA	NA	NA
Pb ²⁺	0	0.005	0.002	0.004	0.010	0.030	0.019	0.009
PO ₄ ³⁻	0.021	2.69	0.891	1.06	3.00	6.00	4.50	2.12
Sr ²⁺	0.050	0.584	0.303	0.245	NA	NA	NA	NA
Zn ²⁺	0.005	0.022	0.014	0.009	0.018	0.069	0.043	0.021
<i>Redox sensitive elements</i>								
O ₂	0.0	0.5	0.2	0.2	0.5	0.8	0.7	0.1
CO ₂	0	195	92.4	88.39	10.5	440	145	203
Fe ²⁺	0.06	12.7	4.39	4.70	0.010*	76.4*	20.1*	17.1*
Fe ³⁺	0.24	27.6	13.3	12.63	NA	NA	NA	NA
Mn ²⁺	0.004	0.353	0.170	0.155	0.022	0.415	0.173	0.173
NH ₄ ⁺	3.35	63.9	26.7	24.37	0.800	187	60.5	72.4
NO ₃ ⁻	0	13.2	3.96	5.53	0.050	0.070	1.09	0.010
SO ₄ ²⁻	1.87	28.9	7.70	9.48	0.110	141	21.6	35.6
S ²⁻	0.006	0.101	0.029	0.035	NA	NA	NA	NA

Notes:

1. Data from nested piezometers 1-1,2 and 2-1,2 (Jankowski and Knight, 1991; and Smolski, 1994)

NA = element not analysed; ND = element not detected; * = Analysis for Fe total only

variations in individual parameters. The pH is slightly elevated and the Eh/pe values are more reducing for the current research, but that is due primarily to the influence of the unique site data from piezometer 1 (Appendix B). The pH at piezometer 1 ranged from 8.2 to 9.1, with an average value of 8.7. Piezometer 2 on the other hand was well buffered, with a narrow pH range of 6.3 to 6.5 and an average value of 6.4. Ordinarily the leachate at piezometer 1 would be interpreted as representing a more advanced state of degradation than at piezometer 2, with a more active consortium of methanogens reducing the free $\text{CO}_{2(\text{aq})}$ resulting in a subsequent rise in pH. However, waste deposition at these two locations (both in Area A) would have occurred within a maximum interval of five years (1965-1970), which at this date would suggest that the waste is in a similar state of degradation at these two locations. Most likely the difference in pH is due to a difference in the composition of waste deposited at the two locations. The pH range for the groundwater samples collected during previous studies was 5.89 to 7.45, with an average value of 6.7. This is very similar to the results for piezometer 2 in the current study.

The groundwater Eh at piezometer 1 is significantly lower than at piezometer 2, ranging between -312 and -205 mV (pe: -5.4 to -3.5), with an average value of -247 mV (pe: -4.3). The groundwater Eh at piezometer 2 ranges from -69 to -250 mV (pe: -1.2 to -4.3), with an average value of -146 mV (-2.9). This would suggest a generally more reducing environment in the vicinity of piezometer 1, which is in accordance with the differences in pH at the two piezometers. Eh values from the previous studies range from -88 to -196 mV, with an average value of -120 mV. These values are also very similar to those from piezometer 2 in the current study.

The EC and TDS values from both the current and previous studies are relatively low for groundwater collected from directly within the waste layers of a landfill (Christensen *et al.*, 1994, 2001). The values from the previous studies that come from the centre of the landfill are higher than the current values collected at the downgradient boundary of Area A. The EC values at piezometer 1 range from 896 to 1770 $\mu\text{S}/\text{cm}$, with an average value of 1224 $\mu\text{S}/\text{cm}$. EC values at piezometer 2 range from 1212 to 1853 $\mu\text{S}/\text{cm}$, with an average value of 1468 $\mu\text{S}/\text{cm}$. Likewise, TDS at piezometer 1 ranges from 330 to 846 mg/L, with an average value of 764 mg/L, and TDS at

piezometer 2 ranges from 405 to 1570 mg/L, with an average value of 881 mg/L. For comparison, the minimum values of EC and TDS reported in a comprehensive literature review by Christensen et al. (2001) are 2500 $\mu\text{S}/\text{cm}$ and 2000 mg/L, respectively. This suggests that the Astrolabe Park and Area A landfills currently produce a relatively dilute leachate, a fact that can perhaps be attributed to the degree of flushing that the waste has been subjected to since the start of landfilling operations in the highly permeable sand beds of the Botany Sands aquifer. The EC values from the previous studies range from 964 to 3830 $\mu\text{S}/\text{cm}$, with an average value of 2153 $\mu\text{S}/\text{cm}$. These values are higher than those observed in the current study, and indicate a more concentrated leachate in the centre of the landfill.

8.3.2 Major elements

In general, the concentrations of major elements are low compared to the average methanogenic leachate values listed in Table 4.1 (Christensen *et al.*, 2001). The concentration of major elements is elevated in the Jankowski and Knight (1991) and Smolski (1994) studies with respect to the current data, which may reflect the difference in sample locations or perhaps a decrease in leachate concentration in the 10+ years between these studies. It is interesting to note that the cation distribution at piezometer 1 is dominated by Na^+ , as all other highly contaminated water types are dominated by Ca^{2+} at the site. This variation can likely be attributed to differences in the composition of waste at piezometer 1 as the cation distribution at piezometer 2 is similar to the data from the previous studies.

As expected in a mature landfill, the anions are dominated by HCO_3^- , which is a product of organic degradation. A couple of elevated SO_4^{2-} concentrations were observed from the Jankowski and Knight (1991) data, and for piezometer 1 in the current study, which are considerably higher than the SO_4^{2-} results for leachate-impacted groundwater elsewhere on the site. However, these concentrations the concentration range for leachate-impacted groundwater reported by Christensen et al. (2001) is 10 to 420 mg/L, so these concentrations are not without precedent. There may be a SO_4^{2-} source in the waste, such as gypsum boards used in construction, which is providing an unusually large influx of SO_4^{2-} to the system. The data for piezometer 1 (Appendix B) indicate that

there was an anomalously high SO_4^{2-} concentration during the first sampling event that may be the result of oxygen introduced to the system during installation of the piezometers, which only occurred several weeks before the first sampling event. The Cl^- concentrations for the historical and current data sets are also on the low end of the range listed in Table 4.1 (Christensen *et al.*, 2001). The Cl^- concentration at piezometer 1 is significantly greater than at piezometer 2 (factor of 3) for the current study, suggesting that a localised source of Cl^- is present in the waste near piezometer 1 (this may also be associated with the elevated Na^+ concentration at this piezometer).

8.3.3 Trace elements

Trace elements can be operationally defined as elements present at concentrations <1 mg/L; PO_4^{3-} is present above 1 mg/L in some samples but is sufficiently low to be included under this heading. While they comprise only a small fraction of the total dissolved solids of landfill leachate, trace elements can be useful indicators of leachate concentration and mixing. Trace metals can be toxic at low concentrations, and even a low concentration of PO_4^{3-} can greatly increase the likelihood of eutrophication in surface waters. While all of the metals are present at low concentrations, Cd^{2+} , Cu^{2+} , Pb^{2+} and Zn^{2+} have average concentrations that exceed the Australia New Zealand Environment and Conservation Council (ANZECC) water quality guidelines for aquatic ecosystems (95% protection level; ANZECC, 2000). PO_4^{3-} is present at elevated concentrations in the Jankowski and Knight (1991) data and at piezometer 1 in the current data set.

8.3.4 Redox sensitive elements

The redox sensitive elements are particularly important with regards to landfill leachate studies as they can be used as indicators of the state of aerobic or anaerobic degradation that is occurring within the plume. It should be noted that the Fe^{3+} concentrations are unusually high for such a strongly reducing environment. It is probable that Fe-colloids present in the leachate may have passed through the filter during sample collection, and were erroneously quantified as dissolved Fe upon acidification of the sample.

Interpretation of redox processes at the site will only consider Fe^{2+} concentrations, which were unaffected by the presence of Fe-colloids and are considered valid.

Oxygen was present at very low concentrations (<1 mg/L) for the current and previous studies, and taking into account the inherent inaccuracies of field O_2 measurements using a probe can be considered anoxic. Carbon dioxide is abundantly present in the previous studies and at piezometer 2 in the current study (Table 8.3; Appendix B). The elevated pH values at piezometer 1 indicate that nearly all of the $\text{CO}_{2(\text{aq})}$ has been converted through buffering processes to HCO_3^- and CO_3^{2-} .

The reduced species Fe^{2+} and Mn^{2+} were present in groundwater during all investigations; the mean Mn^{2+} concentrations are equivalent for both studies (0.17 $\mu\text{g/L}$). The concentrations of both elements are much lower at piezometer 1 than 2 for the current study, which is likely attributable to the abundance of SO_4^{2-} available for reduction at piezometer 1 and the subsequently elevated concentrations of S^{2-} present in groundwater (maximum value of 0.1 mg/L at piezometer 1). Both Mn and Fe form insoluble precipitates with S^{2-} in strongly reducing groundwater environments (Appelo and Postma, 1999) (note the significantly lower mean Eh values for groundwater at piezometer 1 compared to piezometer 2 [-247 mV and -146 mV, respectively]).

The oxidised species NO_3^- and SO_4^{2-} are depleted at both piezometers in the current study relative to uncontaminated groundwater (mean concentrations of 4.0 mg/L and 7.7 mg/L, respectively in landfill groundwater, and 14.5 mg/L and 19.4 mg/L, respectively, in uncontaminated groundwater), while NH_4^+ is present at elevated concentrations (mean concentration of 26.7 mg/L in landfill groundwater, and 2.2 mg/L in uncontaminated groundwater). The mean SO_4^{2-} concentration from the Jankowski and Knight (1991) investigation is anomalously high for a strongly reducing environment such as a landfill, suggesting a local source of SO_4^{2-} (such as gypsum board) in the waste. The S^{2-} concentrations in the current study are low, as S^{2-} forms insoluble precipitates in the presence of most metals (note that S^{2-} concentrations are slightly higher at piezometer 2, which agrees with the slightly more alkaline pH at this location). Therefore its presence at even these low concentrations is significant. The presence of NH_4^+ indicates that fermentative degradation of organic waste material is

occurring in the landfill (Andreottola and Cannas, 1992). Overall, the redox chemistry for past and current studies is indicative of a strongly reducing groundwater environment as is expected in a mature landfill.

8.4 LEACHATE PLUME CHEMISTRY

In the following sections the average composition of the leachate-impacted groundwater is discussed, with a more detailed discussion of spatial and temporal trends later in the chapter. The data included in this section comprise all of the samples collected from piezometers 3, 4, 4A, 6, 10 and 30 ($n = 259$) that were statistically determined to have at least a minimal impact from landfill leachate (classified in Groups 1 to 4 as determined by the hierarchical cluster analysis).

8.4.1 General parameters

The statistical values for the general parameters are presented in Table 8.4. The average pH values range from 5.11 to 6.93, with a total average of 6.07. This value is slightly lower than the average pH values within the landfill (6.4 at piezometer 2, 6.7 from previous research [Jankowski and Knight, 1990; Smolski, 1994]), most likely reflecting dispersion and mixing with the local low-pH groundwater. However, it is within the pH buffering range of the $\text{CO}_{2(\text{aq})}/\text{HCO}_3^-$ system, suggesting that chemical reactions within the plume are capable of buffering the addition of weak carbonic acid present in the uncontaminated groundwater above and below the leachate plume, and $\text{CO}_{2(\text{aq})}$ produced through the degradation of dissolved organic carbon compounds present in the leachate plume.

There is a large variation in Eh/pe values, with Eh values ranging from +7.7 to -269 (pe: 0.13 to -2.51), and an average value of -146 (pe: -2.51). It is difficult to determine exactly which processes are responsible for the fluctuation, since Eh measurements represent a mixed potential based on all redox reactions occurring within the aquifer, but the trend generally indicates a less reducing environment than was recorded within the landfill during the current investigation.

Table 8.4 Statistical data for parameter values and element concentrations for leachate - impacted groundwater (concentrations in mg/L; n = 259).

	Min	Max	Mean	Std. Dev.
<i>General parameters</i>				
EC ($\mu\text{S}/\text{cm}$)	284	1660	671	306
TDS	159	1428	555	288
pH	5.11	6.93	6.07	0.37
Eh (mV)	-269	+7.70	-146	62.2
pe	-4.62	0.13	-2.51	1.075
<i>Major elements</i>				
Ca^{2+}	1.82	204	52.5	42.4
Na^+	8.44	98.8	40.0	16.2
Mg^{2+}	1.83	32.1	11.2	6.66
K^+	2.40	39.4	15.2	7.58
HCO_3^-	31.7	1008	331	217
SO_4^{2-}	0.350	155	13.5	19.8
Cl^-	14.9	177	50.4	14.7
<i>Trace elements</i>				
Al^{3+}	0.005	0.371	0.055	0.114
B	0.029	1.54	0.403	0.246
Cd^{2+}	0	0.002	0.001	0.0002
Cu^{2+}	0.002	0.035	0.008	0.005
Li^+	0.001	0.095	0.031	0.030
Ni	0	0.002	0.001	0.0004
Pb^{2+}	0	0.017	0.007	0.003
PO_4^{3-}	0	4.35	0.472	0.855
Sr^{2+}	0.011	1.08	0.252	0.218
Zn^{2+}	0.002	0.458	0.030	0.060
<i>Redox sensitive elements</i>				
O_2	0	9.10*	1.69	1.96
CO_2	5.28	356	96.8	64.9
Fe^{2+}	0	16.7	3.6	2.59
Fe^{3+}	0	65.0	5.8	9.87
Mn^{2+}	0.003	2.16	0.281	0.359
NH_4^+	0	63.5	17.5	12.9
NO_3^-	0	92.0	10.9	10.8
SO_4^{2-}	0.350	155	13.5	19.8
S^{2-}	0	2.52	0.061	0.195

* Erroneously high dissolved O_2 concentrations due to faulty field equipment during one sampling event.

EC values varied between 284 and 1660 $\mu\text{S}/\text{cm}$, with an average value of 671 $\mu\text{S}/\text{cm}$. This is significantly lower than the average values for the landfill piezometers (2500 to 35,000 [Christensen et al., 2001]), although the maximum EC values in the centre of the plume are equivalent to or only slightly lower than the landfill values (Table 8.2). While these average EC values are relatively low compared to other documented landfill leachate research, they are still considerably higher than background groundwater levels (~ 213 $\mu\text{S}/\text{cm}$; Acworth and Jankowski, 1993), which creates an easily identifiable electrical anomaly that can be used to track changes in the general intensity and distribution of the plume.

TDS values in the leachate plume range from 159 to 1428 mg/L, with an average value of 555 mg/L. Again, this is considerably lower than the values at the landfill piezometers (Table 8.2), but the centre of plume concentration is much closer to the landfill values. However, this TDS concentration is indicative of a generally dilute leachate plume (typical TDS range = 2000 to 60,000 mg/L [Christensen et al., 2001]).

8.4.2 Major elements

The major cations Ca^{2+} , Mg^{2+} , Na^{+} and K^{+} are present in similar proportions to landfill piezometer 2, with Ca^{2+} present at a slightly greater concentration (in mg/L) than Na^{+} , and Mg^{2+} and K^{+} both at much lower concentrations (Table 8.4). The average concentrations for these elements are approximately half the values of the average concentrations from landfill piezometer 2, but once again the maximum concentrations from the centre of the leachate plume are considerably closer to the landfill piezometer values (eg. average Ca^{2+} concentration of 101 mg/L in piezometer 2, and 110 mg/L from the central plume sampling point at piezometer 10). In terms of molar concentrations, Na^{+} is the dominant cation in all but a handful of the most highly contaminated groundwater samples from the concentrated centre of the leachate plume.

The major anions are also present in a similar distribution to landfill piezometer 2, with HCO_3^{-} clearly dominant, Cl^{-} present at slightly elevated concentrations with respect to unaffected background water, and SO_4^{2-} almost completely depleted in the centre of the plume (<1 mg/L). The anion concentrations in the centre of the plume vary only slightly

from those in the landfill, while the averages for most constituents throughout the plume are up to 50% lower than the landfill values.

8.4.3 Trace elements

The most abundant trace elements, B, PO_4^{3-} and Sr^{2+} , all average less than 0.5 mg/L (Table 8.4), although variations in their concentrations follow the same pattern as the major elements and EC values. This suggests a source for these elements in the landfill, although the exact source is unknown. The metals are present at very low concentrations (generally <0.05 mg/L), and most of them were below the laboratory reporting limit during each sampling event. The average concentrations of these elements are generally lower than in the landfill samples, although the plume centre samples exhibit only slight attenuation.

While it is one of the more abundant trace elements, PO_4^{3-} is still present at sufficiently low concentrations to preclude any imminent threat of eutrophication to Lachlan Pond No.5. Some of the PO_4^{3-} may be bound in metal complexes and therefore not detected during spectrophotometric analysis, as will be assessed in a later chapter, but leachate plumes are not generally characterised by elevated PO_4^{3-} concentrations (Christensen et al., 2001).

8.4.4 Redox sensitive elements

Dissolved O_2 is depleted within the leachate plume, although a couple of anomalous measurements have skewed the total average upwards. For the remaining sampling events, the O_2 concentrations averaged less than 1 mg/L, and in the centre of the plume the values were less than 0.5 mg/L. Considering the inherent inaccuracy involved in the field measurement of exceedingly low O_2 concentrations using a probe, the accepted convention is to consider groundwater samples with O_2 concentrations below 0.5 mg/L anoxic.

The $\text{CO}_{2(\text{aq})}$ levels are elevated with respect to uncontaminated groundwater, although slightly less concentrated than at landfill piezometers. The elevated concentrations are

the result of organic matter degradation within the plume, although the pH range indicates buffering between $\text{CO}_{2(\text{aq})}$ and HCO_3^- .

The reduced elements Fe^{2+} , Mn^{2+} , S^{2-} and especially NH_4^+ are present at elevated concentrations in the plume (max. conc. of 16.7, 2.16, 2.52 and 63.5 mg/L, respectively), which is to be expected in the strongly reducing environment. The presence of Mn^{2+} , Fe^{2+} and S^{2-} (and depletion of SO_4^{2-}) indicate that manganese, iron and sulphate reducing zones are present either within the landfill or in the plume itself, and the elevated concentration of NH_4^+ is the result of fermentation processes within the landfill. The oxidised elements NO_3^- and SO_4^{2-} are depleted in the centre of the plume (less than 0.5 mg/L), with gradually increasing concentrations towards the upper and lower margins. Once again, the Fe^{3+} concentrations appear to be too high for a groundwater environment in which sulphate reduction is occurring; most likely these concentrations represent Fe-colloids that passed through the filters during sample collection and were subsequently quantified as dissolved iron during the ICP analysis. This conclusion is supported by the fact that the highest Fe^{3+} concentrations correspond to the most highly contaminated portion of the plume, where colloidal iron precipitates are most likely to form.

8.5 MARSH, POND AND RAINWATER CHEMISTRY

Pond water samples were collected from four locations along the eastern shore of Lachlan Pond No.5 during each full sampling event. Sample SW-4A was only collected during the June 2000 sampling event as it came from a ephemeral puddle in the marsh that was only present after sustained rainfall. Three shallow groundwater samples were collected during the July 2001 monitoring event from temporary piezometers installed in the marsh at 15 m intervals along a transect perpendicular to the shoreline (reference Figure 5.4). Finally, two rainwater samples were collected at the site during two separate rainfall events in September 2001. The chemical signature of the marsh and pond water samples was similar to that of samples collected from the margins of the leachate plume (Figure 8.1), providing evidence of the impact to pond water quality from the discharging leachate plume. The average values for the marsh, pond and rain samples are presented in Table 8.5, along with data from uncontaminated pond water

Table 8.5 Mean parameter values and element concentrations for pond, marsh and rainwater samples (concentrations in mg/L).

	Previous ¹	SW-3	SW-3/6	SW-6	SW-drain	SW-4A	Marsh	Rain
<i>n</i> =	6	5	5	5	5	1	3	2
<i>General parameters</i>								
EC (μS/cm)	138	255	222	220	367	568	808	60.4
TDS	88.0	176	156	160	279	439	686	26.8
pH	6.99	7.08	7.02	6.87	6.86	6.54	6.41	4.85
Eh (mV)	-100	+ 49.2	+ 85.1	+ 88.2	+ 73.3	-139	-105	+ 47.9
pe		+ 0.82	+ 1.44	+ 1.50	+ 1.23	-2.42	-1.81	+ 0.82
<i>Major elements</i>								
Ca ²⁺	10.5	20.7	17.9	27.3	36.7	18.2	85.6	2.36
Na ⁺	11.1	17.0	15.5	19.4	23.2	16.3	49.6	4.53
Mg ²⁺	3.61	3.55	3.20	5.30	6.73	3.78	15.8	0.359
K ⁺	3.52	4.48	3.73	6.82	6.26	4.48	13.2	0.412
HCO ₃ ⁻	24.5	67.3	56.0	56.3	130	242	421	ND
SO ₄ ²⁻	10.7	13.0	12.2	14.0	20.6	22.4	25.2	4.20
Cl ⁻	24.1	33.7	31.6	33.3	39.5	53.5	48.0	7.25
<i>Trace elements</i>								
Al ³⁺	NA ²	0.059	0.057	0.050	0.032	<0.050	0.023	0.025
B	NA	0.053	0.039	0.065	0.091	0.041	0.374	0.007
Cd ²⁺	NA	0.001	0.001	0.001	0.001	0.001	<0.0003	0.002
Cu ²⁺	NA	0.016	0.009	0.007	0.012	<0.007	<0.0008	0.009
Li ⁺	NA	0.031	0.031	0.031	0.031	<0.070	<0.001	0.002
Ni	NA	<0.001	<0.001	<0.001	<0.001	<0.03	<0.020	0.001
Pb ²⁺	NA	<0.003	0.011	0.017	<0.003	<0.060	<0.003	<0.003
PO ₄ ³⁻	NA	0.554	0.204	0.250	0.732	0.280	0.713	ND
Sr ²⁺	NA	0.125	0.130	0.144	0.280	0.091	0.457	0.014
Zn ²⁺	30	0.008	0.009	0.011	0.030	<0.070	<0.001	0.341
<i>Redox sensitive elements</i>								
O ₂	NA	8.0	8.0	8.1	7.0	1.9	1.1	6.7
CO ₂	10.5	9.15	5.81	6.51	11.8	23.8	58.4	4.84
Fe ²⁺	0.490	0.085	0.033	0.028	0.033	0.480	4.87	0.010
Fe ³⁺	NA	0.938	0.623	3.80	1.41	4.00	9.30	0.005
Mn ²⁺	0.040	0.024	0.016	0.192	0.074	0.016	0.667	0.002
NH ₄ ⁺	0.023	1.63	0.436	0.704	2.82	2.90	5.38	0.432
NO ₃ ⁻	NA	9.50	9.15	7.00	11.6	6.16	5.28	6.86
SO ₄ ²⁻	10.7	13.0	12.2	14.0	20.6	22.4	25.2	4.20
S ²⁻	NA	0.008	0.001	0.005	0.003	0.004	0.017	ND

Notes:

1. Previous data is averaged uncontaminated Pond No.5 water data from Knight *et al.* (1990)
2. NA = Not analysed during previous study

samples collected from Pond No.5 during a previous study (Knight *et al.*, 1991) for comparison.

8.5.1 General parameters

The EC and TDS values for pond samples SW-3, SW-3/6, and SW-6 are close to the average values for uncontaminated groundwater ($\sim 210 \mu\text{S/cm}$) in the region (Table 8.5). Sample SW-drain is collected from the mouth of a stormwater drain that discharges into the pond. This drain passes through the buried waste in the landfill, and leachate is believed to be leaking into it, which is reflected in the slightly elevated EC and TDS values ($\sim 150 \mu\text{S/cm}$ and 120 mg/L higher than other pond samples, respectively). Sample SW-4A is similar to the marsh samples, with elevated EC and TDS values that reflect the impact from the discharging leachate plume. The rainwater has extremely low EC and TDS values, as would be expected. The EC values from the uncontaminated pond samples from Knight *et al.* (1991) are slightly lower than those collected during the current study. In general, the water quality in Lachlan Pond No.5 reflects a mixture of shallow groundwater and stormwater runoff that discharges into them.

The average pH values of all of the pond samples are close to neutral, with Eh values that indicate an oxidised environment (Table 8.5), whilst the pH and Eh values of SW-4A and the marsh samples (pH = 6.5 and 6.4; Eh = -139 and -105 , respectively) are closer to those of the leachate plume. The average pH value of the rainwater samples is 4.9, which is slightly more acidic than uncontaminated groundwater. As expected, the Eh value for the rainwater samples is slightly oxidising. The Eh value for uncontaminated pond water from Knight *et al.* (1991) is surprisingly low, and may reflect a period of limited stormwater discharge through the pond system.

8.5.2 Major elements

The major elements are all present at relatively low concentrations in the pond water samples, with a slight increase observed in the sample SW-drain (Table 8.5). A significant increase in major element concentrations is observed in sample SW-4A and the marsh samples, reflecting the increased solute load of the discharging leachate

plume. In particular, the HCO_3^- concentrations are significantly higher in the marsh samples. All element concentrations are exceedingly low in the rainwater samples, and correlate well with the average rainwater concentrations reported by Acworth and Jankowski (1993). The concentrations of major elements in the uncontaminated pond water samples from Knight et al. (1991) are slightly lower than those from the current study, suggesting an increase in the influence from the discharging leachate plume.

8.5.3 Trace elements and metals

The trace elements are all present at very low concentrations in the pond water samples. The two trace elements present at the highest concentrations are Sr^{2+} and PO_4^{3-} , which may be a cause for concern with regards to the risk of eutrophication in the pond. A slight increase in Sr^{2+} and PO_4^{3-} concentrations is observed in the SW-drain sample, but there is no discernable change in the other trace element concentrations. Most of the trace metals in the marsh samples were below detection limits, suggesting that they are removed by attenuation processes within the marsh (as discussed in Chapter 7). Of the remaining trace elements, only B and Sr^{2+} are slightly elevated with respect to the pond water samples. All of the trace elements are present at very low concentrations in the rainwater with exception of Zn^{2+} , which is unusually elevated. It is unknown whether this actually represents atmospheric pollution, or whether sample contamination may have occurred during collection or analysis.

8.5.4 Redox sensitive elements

As expected, O_2 and the oxidised elements NO_3^- and SO_4^{2-} are elevated in the pond water samples with respect to the reduced elements (Mn^{2+} , Fe^{2+} , NH_4^+ , S^{2-} ; Table 8.5). A slight increase in NH_4^+ , NO_3^- and SO_4^{2-} concentrations is observed in SW-drain, reflecting the influence of landfill leachate coupled with the subsequent oxidation of reduced species in the pond. Dissolved oxygen is mostly depleted in SW-4A and the marsh samples, with a commensurate rise in the concentrations of reduced elements. Sulphate is slightly elevated in the marsh samples, most likely due to oxidation of reduced sulfur by diffusion of atmospheric O_2 . Slightly elevated concentrations of NO_3^- ,

NH_4^+ and SO_4^{2-} in the rainwater samples are due to atmospheric pollution by nitrogen and sulfur-based industrial emissions.

8.6 SPATIAL AND TEMPORAL CHEMICAL DATA TRENDS

The introduction of the concentrated, strongly reducing landfill leachate to the essentially pristine Botany Sands aquifer has resulted in significant chemical zonation within the affected portion of the aquifer. There is a distinct pattern of vertical chemical zonation within the plume, and variations in the chemical signature and concentration of the chemical zones are observed over short time intervals. A description of the vertical chemical zonation of groundwater at the site is provided, followed by an analysis of the spatial and temporal variation.

8.6.1 Vertical zonation of chemical constituents

Zonation of groundwater chemistry is a commonly observed phenomenon associated with the release of landfill leachate into an aquifer environment (Baedecker and Back, 1979b; Nicholson et al., 1983; Baedecker and Apgar, 1984; Kehew and Passero, 1990; Lyngkilde and Christensen, 1992a; Bjerg et al., 1995; Jankowski, 1997; Jankowski and Acworth, 1997; Basberg et al., 1998). Significant research has been conducted into the delineation of redox zones downgradient from unlined landfills (Bjerg et al., 1995; Chapelle et al., 1996; Basberg et al., 1998; Ludvigsen et al., 1998 and 1999; Carey et al., 2003), created by a series of redox buffering reactions between the strongly reducing leachate and the oxidised groundwater and aquifer minerals. Typically redox zones are investigated along a groundwater flow path, but in the case of Astrolabe Park the distance between the landfill and the pond is too short to accommodate the formation of multiple redox zones along flow path. However, a degree of vertical redox zonation is observed, which varies in accordance with fluctuations in the overall concentration and centre of mass of the leachate plume. In addition, a clear zonation of general parameters, and major and trace elements is observed along a vertical profile through the plume.

Plots of the vertical distribution of field water quality parameters (Figures 8.4 and 8.5), major and trace ions (Figures 8.6 and 8.7) and redox sensitive elements (Figures 8.8 and

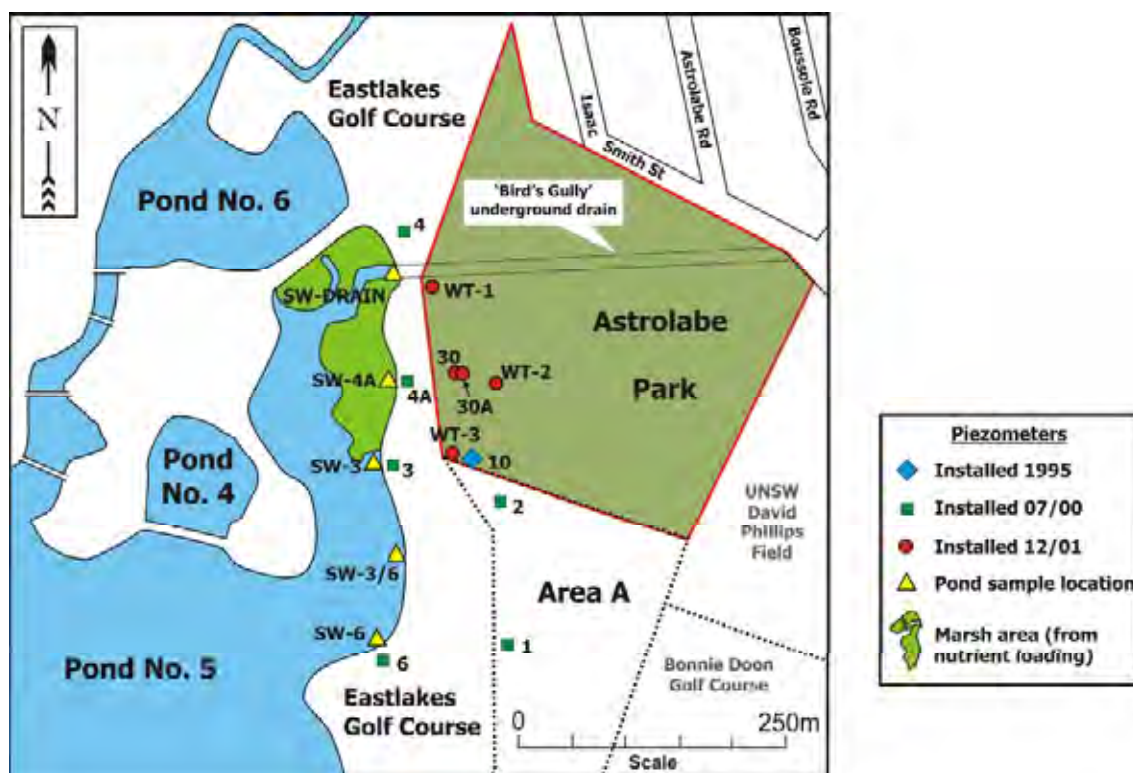


Figure 8.2 Site plan depicting piezometer locations used during this research.

8.9) for the July/August 2001 monitoring event are presented, along with previous Figures 3.1 and 6.5 repeated here (Figs 8.2 and 8.3) to provide context with regards to piezometer location and the conceptual site model for subsurface hydrogeology and leachate impact to groundwater. The vertical distribution of field water quality parameters, major ions, trace elements and redox sensitive elements in the aquifer is intimately related to the presence of the leachate plume at Astrolabe Park, according to the following general characteristics:

- Elevated concentrations relative to unaffected groundwater of most major ions (Ca^{2+} , Mg^{2+} , Na^+ , K^+ , HCO_3^- and Cl^-) and the trace elements Sr and B, attributable to the loading of soluble salts derived from the waste in the landfill. This trend is mirrored in the distribution of EC and TDS in the plume, which is significantly greater than that of the relatively dilute unaffected groundwater;
- Elevated concentrations of the reduced redox constituents NH_4^+ and Fe^{2+} , along with more negative Eh values, which attest to a strongly reducing environment within the plume; and

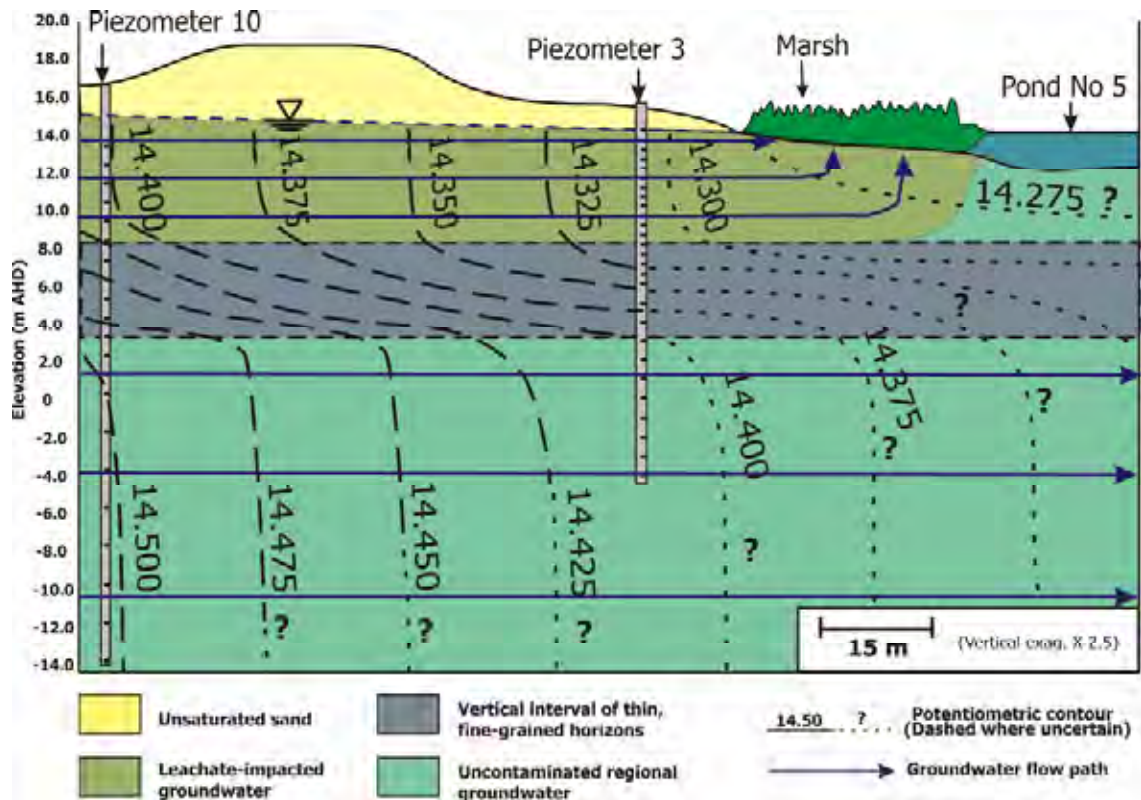


Figure 8.3 Longitudinal cross section depicting the hydrogeology and water quality zonation in the subsurface between the landfill and the pond.

- Depleted concentrations of the electron acceptors dissolved oxygen, nitrate and sulfate relative to unaffected groundwater, which are consumed during microbial degradation of dissolved organic carbon in the plume; and
- The maximum concentrations of the redox sensitive compounds occur marginally lower in the profile than the max concentrations of the major ions (~0.5 m), which suggests that redox processes are influencing the distribution, which do not directly affect the major ions. Oxidation of the upper NH_4^+ fringe may potentially be occurring due to the presence of electron acceptors in the shallow aquifer (eg. dissolved oxygen, nitrate, sulfate), but more intensive research of the plume fringe would be required to confirm this. It is worth noting that the distribution of NH_4^+ and ferrous iron are very similar, likely as a result of similar redox processes along the top fringe of the plume.

Whilst this distribution is valid for most of the groundwater chemical constituents, the following exceptions to the general trends are noted:

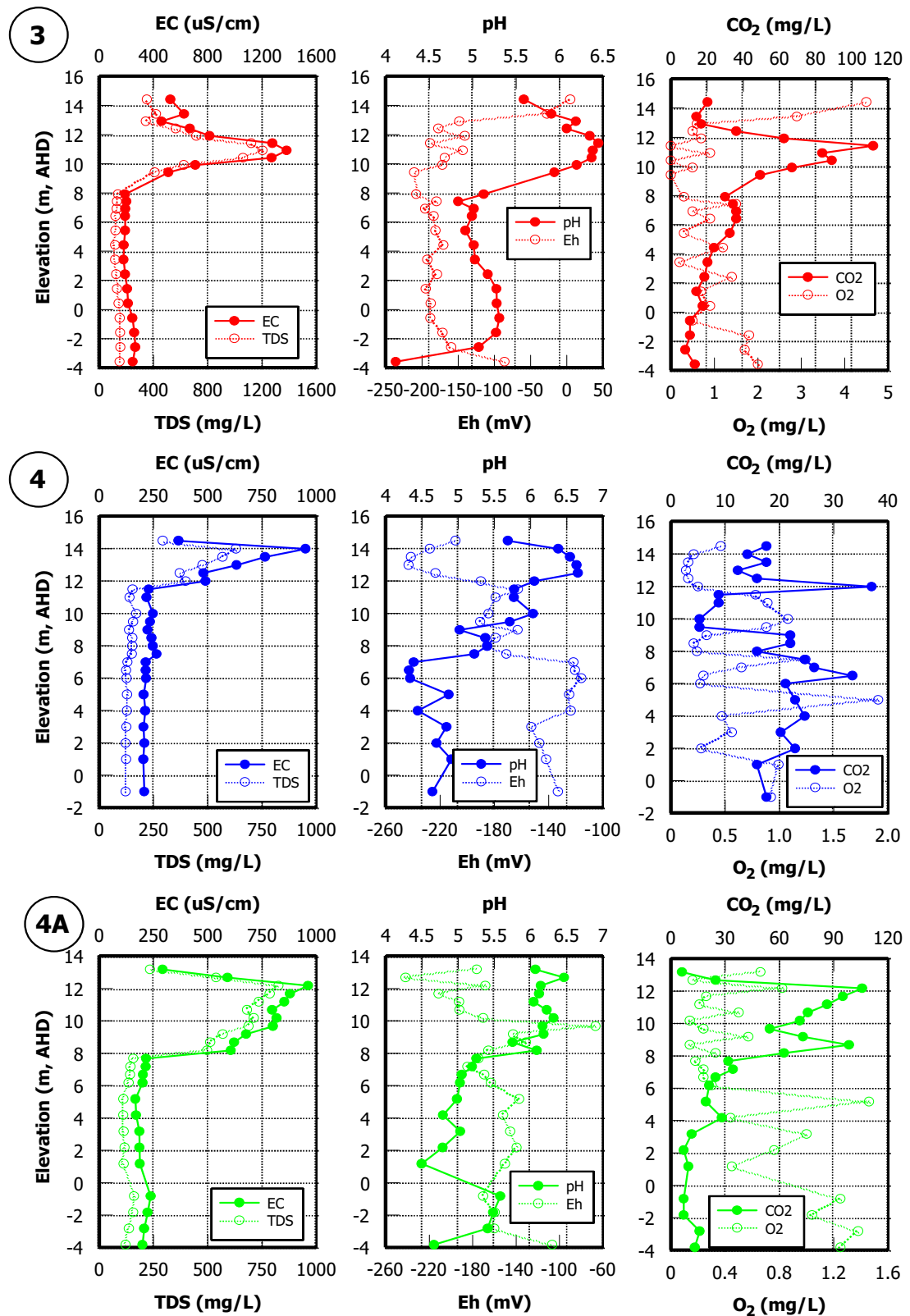


Figure 8.4 Vertical distribution of general parameters for piezometers 3, 4 and 4A; July 2001 sample event.

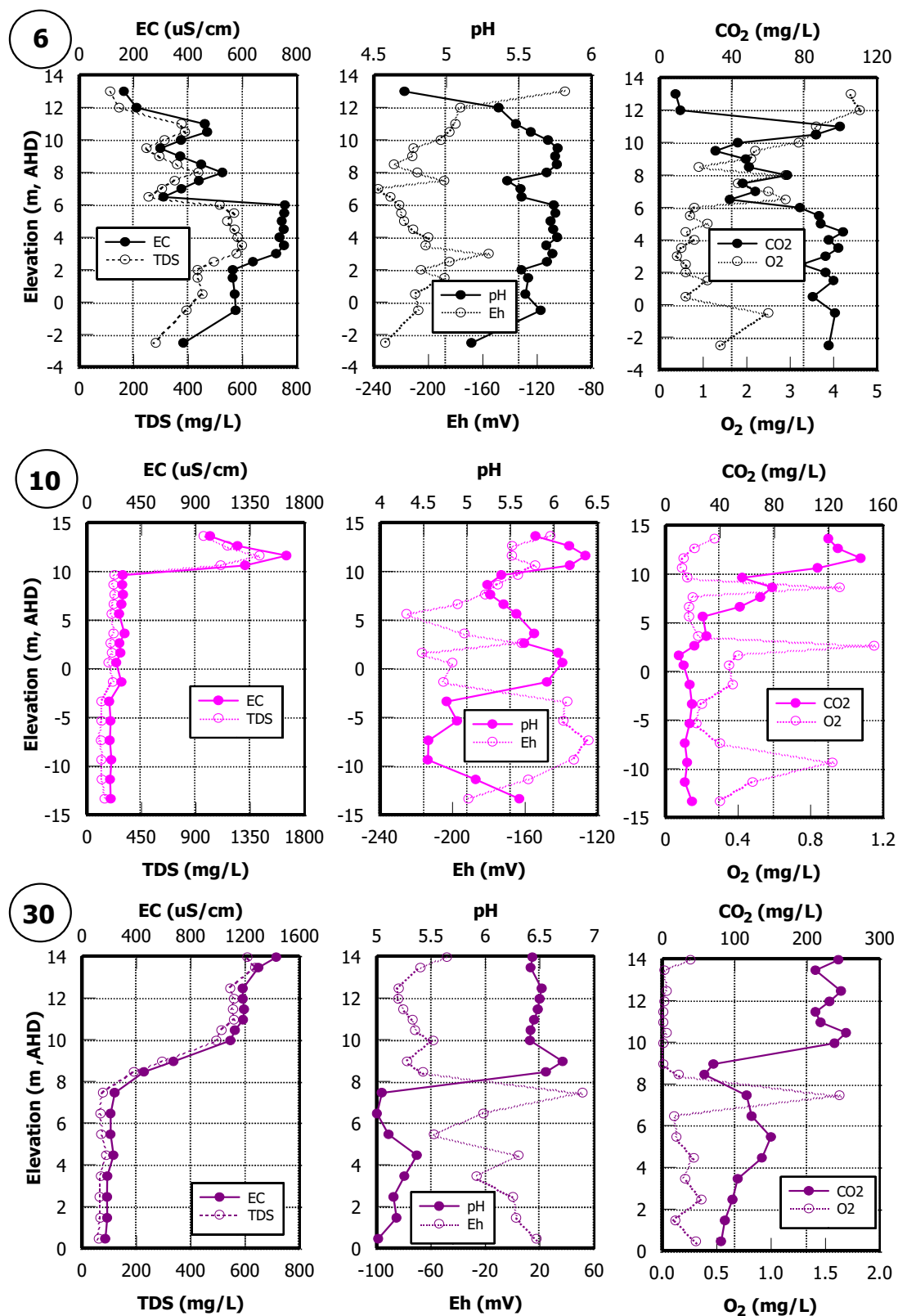


Figure 8.5 Vertical distribution of general parameters for piezometers 6, 10 (July 2001) and 30 (February 2002).

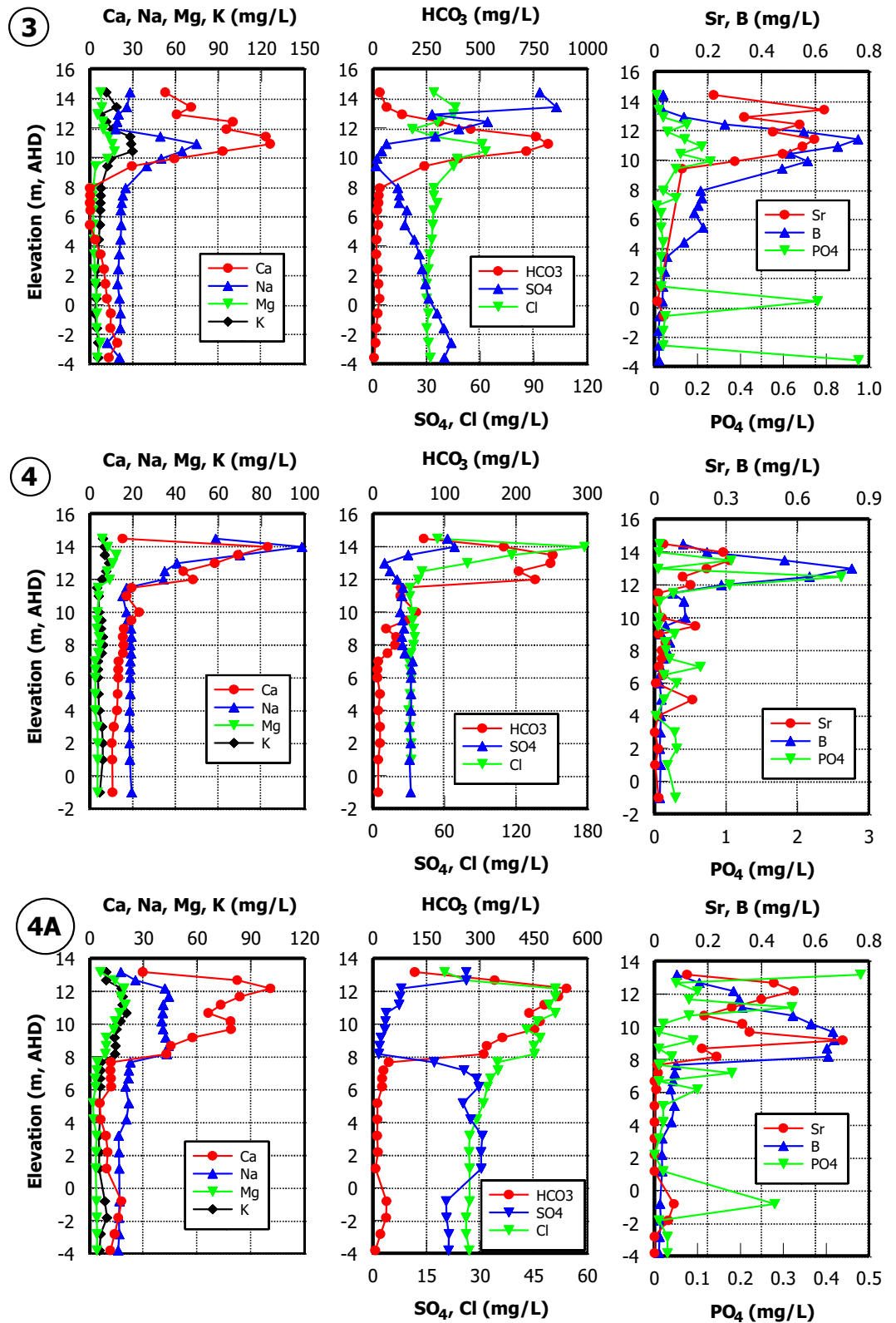


Figure 8.6 Vertical distribution of major and trace elements for piezometers 3, 4 and 4A; July 2001.

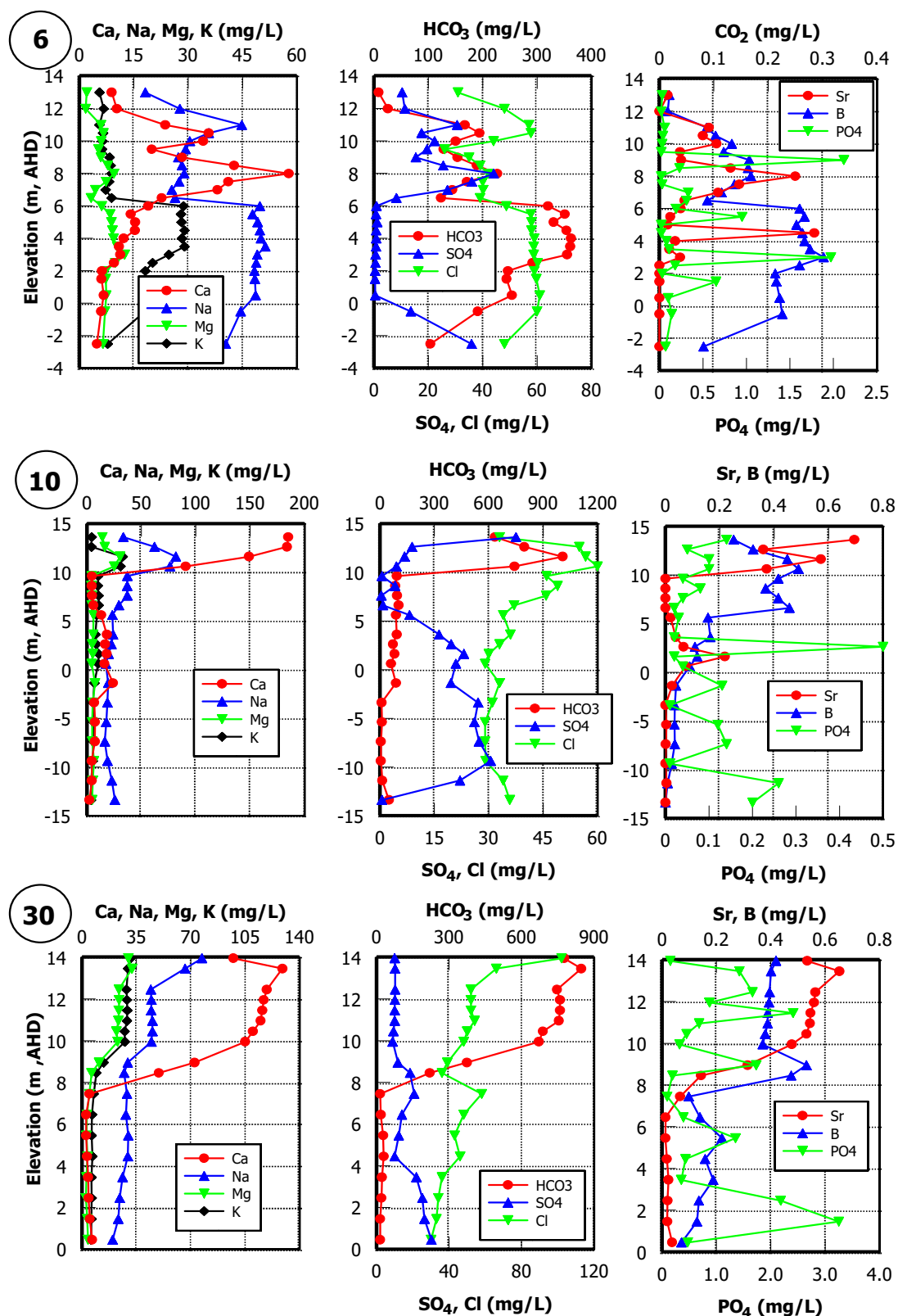


Figure 8.7 Vertical distribution of major and trace elements for piezometers 6, 10 (July 2001) and 30 (February 2002).

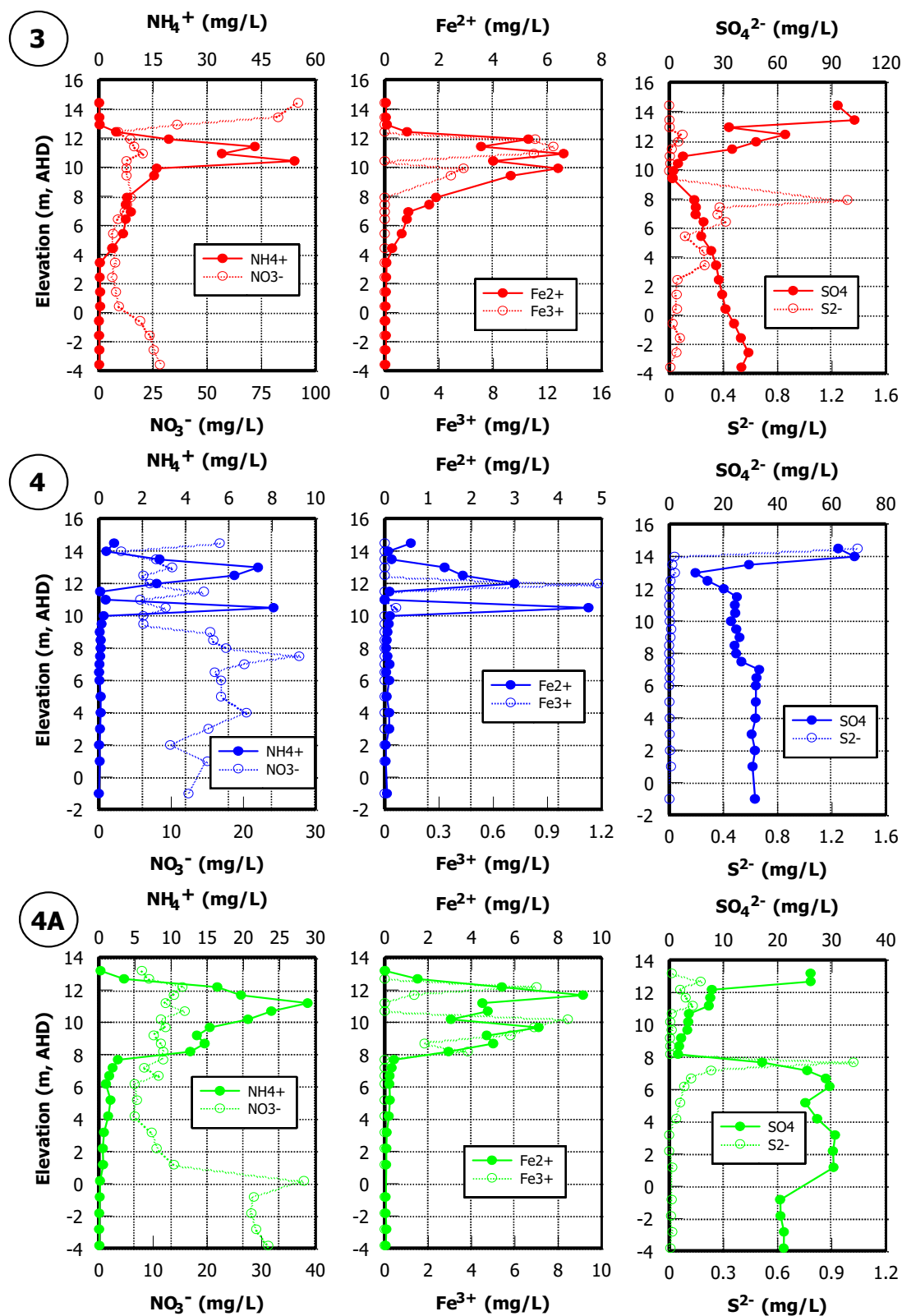


Figure 8.8 Vertical distribution of redox sensitive elements for piezometers 3, 4 and 4A; July 2001.

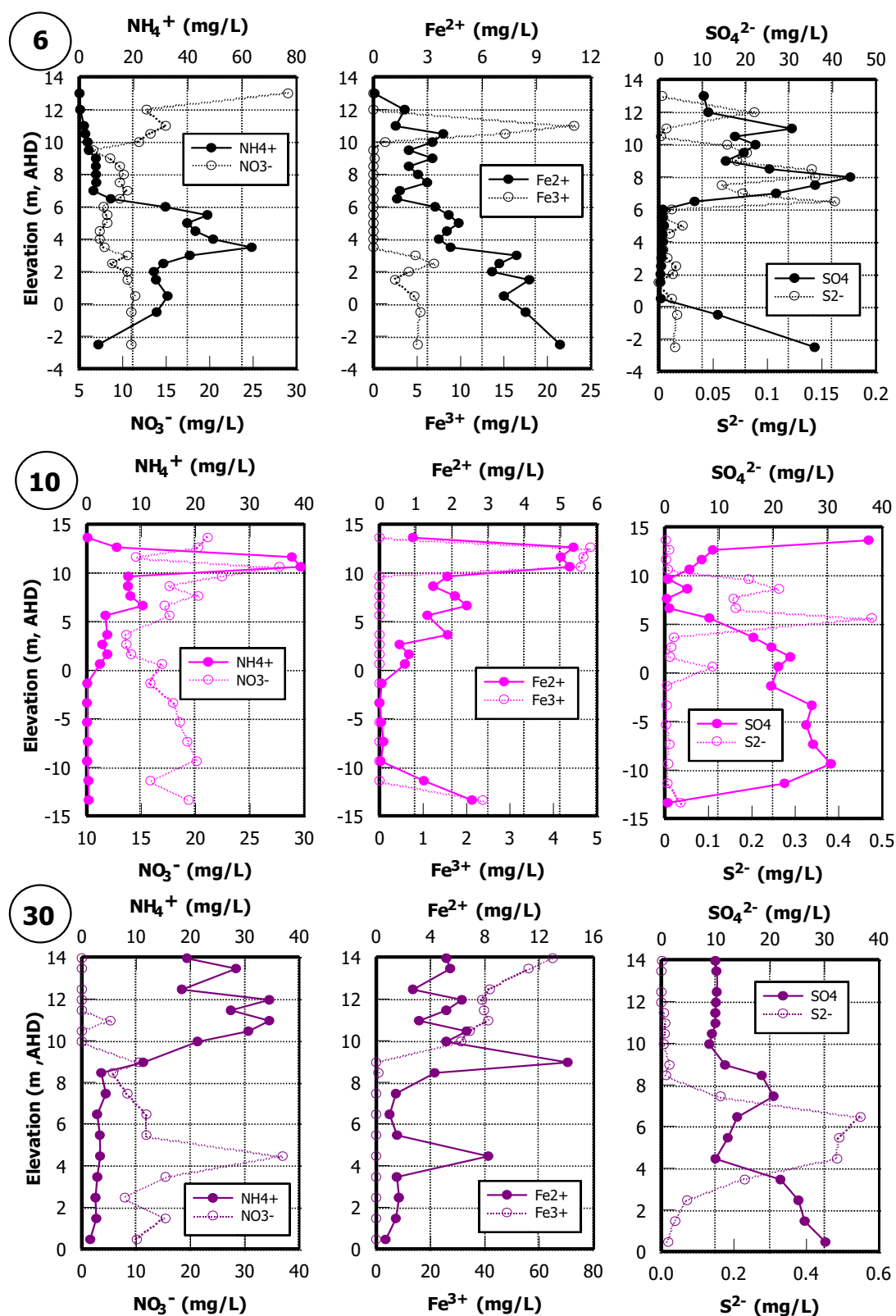


Figure 8.9 Vertical distribution of redox sensitive elements for piezometers 6, 10 (July 2001) and 30 (February 2002).

- Depletion of sulfate within the plume is not matched by a concomitant increase in sulfide. However, sulfide forms insoluble precipitates in the presence of dissolved iron according to a kinetically rapid reaction (Appelo and Postma, 1993), and where the concentration of ferrous iron is elevated sulfide will typically be removed entirely from groundwater;
- An interesting pattern is observed at piezometers 3, 4A and 10, where a spike in the S^{2-} concentrations is observed directly below the leachate plume, which correlates well with the presence of the thin peaty layer that constrains the plume to the shallow aquifer. An excess of S^{2-} is generated in these locations both from oxidation of iron sulfides within the confining layer and reduction of SO_4^{2-} that is advectively transported into the low permeability lens by means of the strong upwards hydraulic gradient. The hydraulic conductivity value of the confining layer was determined through slug tests to be approximately 1.5 m/d; low enough to create semi-confined conditions in the lower aquifer and constrain the leachate plume to the shallow aquifer, but still permeable enough to allow a small component of upwards flux of uncontaminated groundwater into the confining layer. As the iron sulfide minerals are oxidised, the increased S^{2-} concentration results in precipitation of Fe^{2+} out of solution; and
- The distribution leachate constituents at piezometer 6 varies significantly from the other piezometers, with the concentrated plume occurring at depth rather than in the shallow aquifer. This is due to the absence of the low permeability confining layer at this location, which has resulted in the plume sinking through the aquifer. The reason for the absence of the shallow confining layer close to piezometer 6 is unknown, although is likely due to the limited lateral extent of the marsh in which the confining layers would have originally been deposited.

8.6.2 Variation along flow path transects

The variations in chemistry along groundwater flow path are presented in Figures 8.10 and 8.11 (from Jorstad et al., 2004), utilising ‘snapshots’ of vertical chemistry profiles along two parallel groundwater flow paths defined by piezometers 10 and 3, and 30 and 4A, with the data collected in July 2001 and February 2002, respectively. These two flow paths are only separated by approximately 50 m, and there are both similarities and

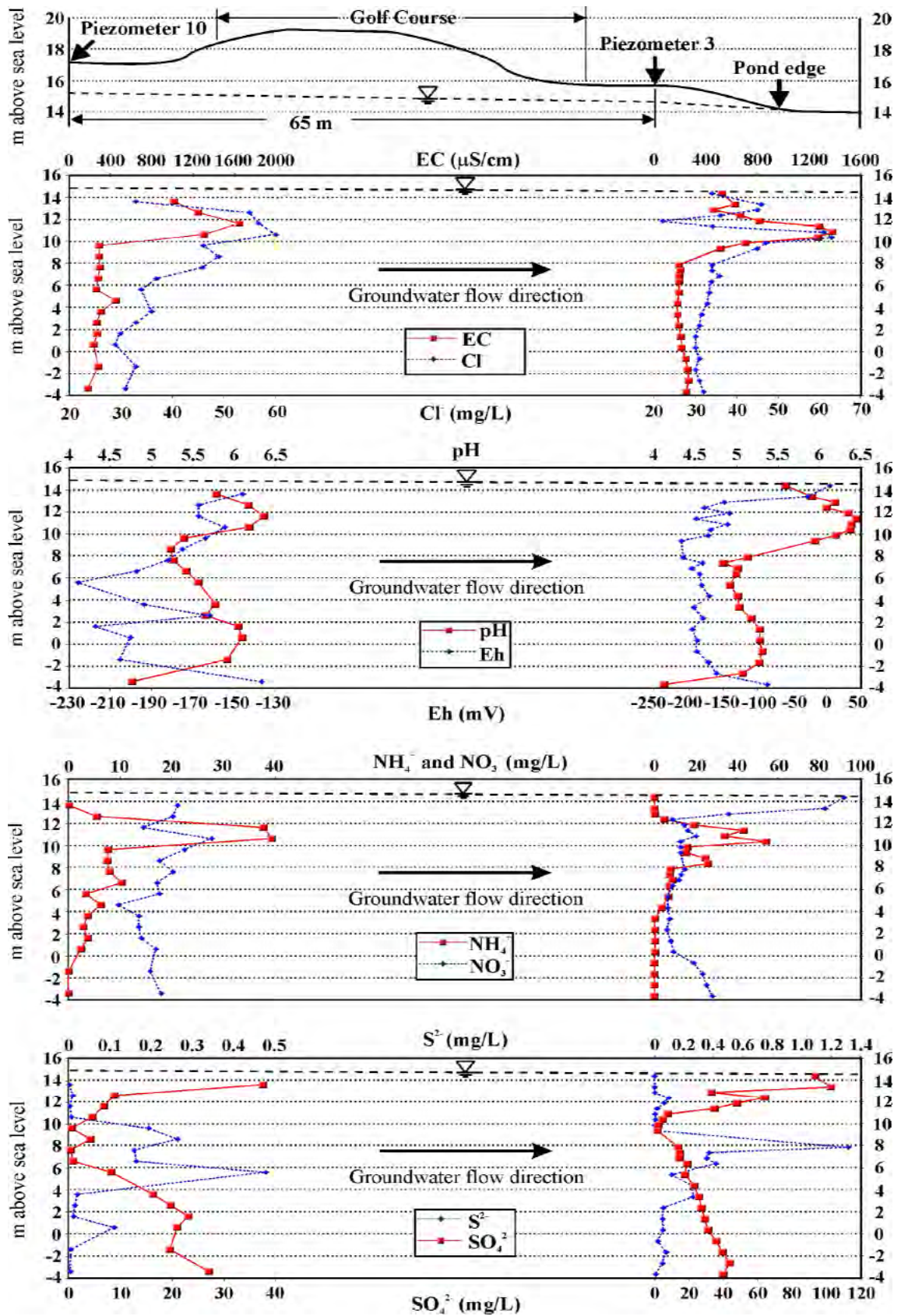


Figure 8.10 Variation in the distribution of selected groundwater parameters and elements along a groundwater flow path between piezometers 10 and 3 (from Jorstad et al., 2004).

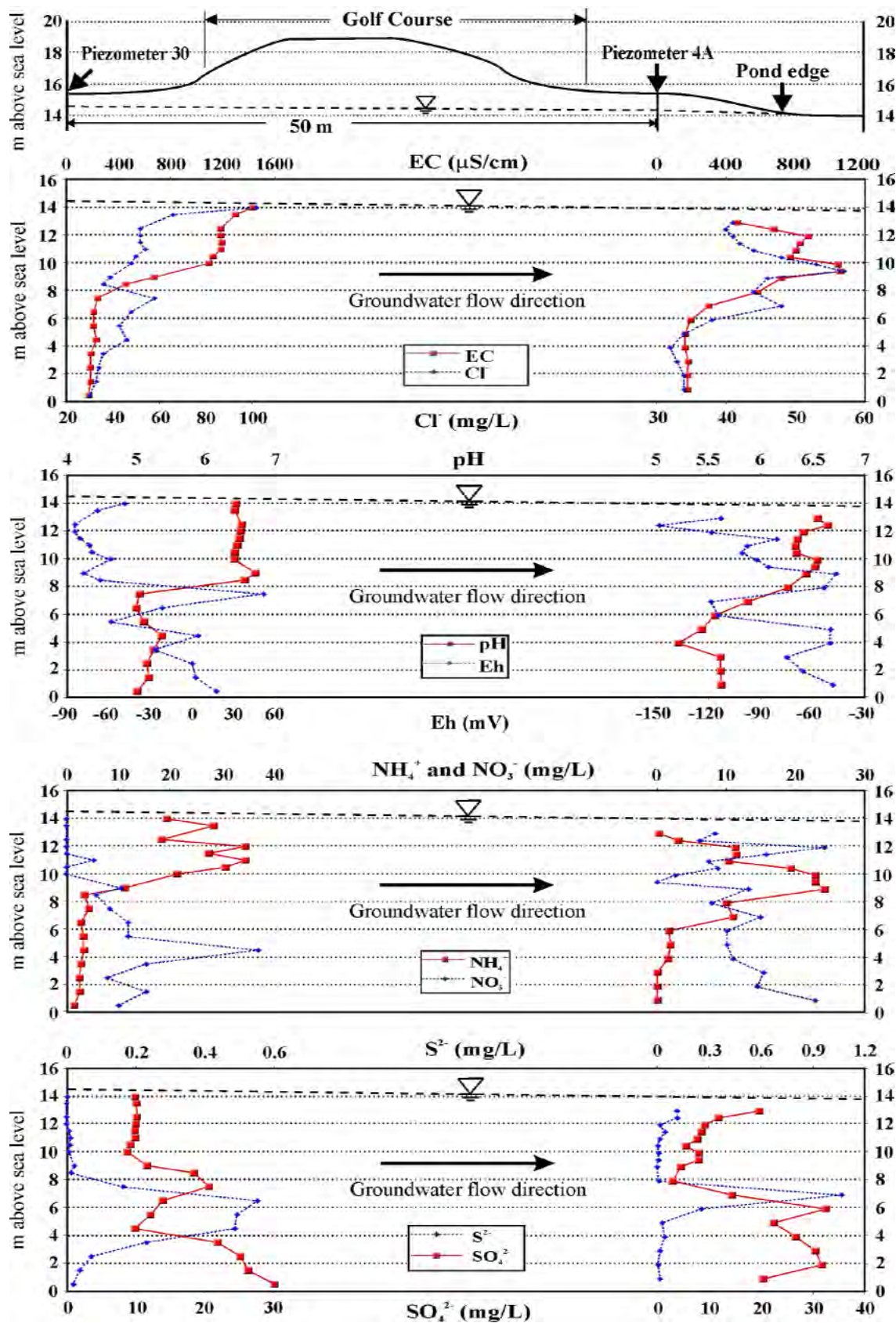


Figure 8.11 Variation in the distribution of selected groundwater parameters and elements along a groundwater flow path between piezometers 30 and 4A (adapted from Jorstad *et al.*, 2004).

significant differences in the distribution of elements at the beginning and end of the two transects, underscoring the variable nature of the plume in space and time.

- The centre and thickness of the leachate plume is similar for both transects, with the highest concentrations constrained within a vertical section between approximately 10 and 12.5 m AHD. The plume geometry varies slightly at piezometer 30, with elevated concentrations of leachate indicator parameters from 10 to 14 m AHD, with the maximum concentrations directly at the water table. Maximum EC values close to the landfill are of similar magnitude (1650 $\mu\text{S}/\text{cm}$ at piezometer 10, and 1429 $\mu\text{S}/\text{cm}$ at piezometer 30);
- The peak EC values decrease along flow path for both transects (30% decrease between piezometers 10 and 3, and 16% decrease between piezometers 30 and 4A). Unexpectedly, the peak concentration of chloride (a non-reactive tracer) remains approximately stable along flow path, and the peak concentrations of NH_4^+ , NO_3^- and SO_4^{2-} increase along flow path between piezometers 10 and 3. This observation lends further support to the notion of ‘slugs’ of concentrated leachate migrating through the aquifer, with discrete groundwater monitoring events representing snapshots of the relative locations of the wells with regards to the slugs at any given time. The peak values of the same elements decrease along flow path as expected between piezometers 30 and 4A, but variable rates of attenuation (Cl^- decreased 17% along flow path, and NH_4^+ decreased 34% along flow path) suggest that different processes are influencing the mobilisation of different chemical constituents. With regards to NH_4^+ , both ion exchange and oxidation may be contributing attenuation mechanisms (this will be explored later in the thesis);
- As noted in the discussion of the vertical profiles, the upper and lower bounds of the leachate plume are characterised by steep concentration gradients, and this pattern holds true along both flow path transects; and
- Large spikes of S^{2-} are present between 6 and 8 m AHD at all wells, which is interpreted to represent oxidation of iron sulfide minerals in the peaty layer immediately beneath the plume. As such, transport along flow path has little bearing on this trend.

8.6.3 Vertical dynamics of the leachate plume

In addition to the vertical zonation of leachate constituents observed for the ‘snapshot’ data set in Section 8.6.1, significant changes were also observed in the shape, location and total concentration of the plume over time. The degree of variation differed between the various piezometers, although some similar temporal trends were reflected in data from all of the piezometers. The following sections will focus on piezometers 4 and 10, located on the northern fringe of the plume and in the lateral plume centre, respectively, to illustrate the changes in the vertical extent of the leachate plume, the location of the centre of mass of the plume, and variations in the total concentration of the plume between subsequent sampling events.

8.6.3.1 *Centre of mass*

The vertical profiles of electrical conductivity at piezometers 4 and 10 are presented in Figure 8.12. It is clear that the centre of mass of the plume at both locations is subject to significant vertical displacement during the duration of the research period. Piezometer 10, located close to the downgradient edge of Astrolabe Park landfill, exhibits a prominent, narrow plume centre at approximately 12.5 m AHD from July to December 2000, which then drops one metre to 11.5 m AHD from January to April 2001, then lowers a further metre to approximately 10.5 m AHD during May and June 2001, and finally raises a metre back to 11.5 m AHD during the July 2001 sampling event.

The centre of mass of the leachate plume at piezometer 4 is generally constrained between 12 and 13 m AHD throughout the sampling period, with both upwards and downwards fluctuations of approximately 0.5 m between sampling events. The main exception is between the June and July 2001 sampling events when a shallow flux of concentrated leachate reaches piezometer 4, causing the centre of mass of the plume to raise approximately 1.5 metres from 12.5 to 14 m AHD. Another variation observed at piezometer 4 is the presence of a lower, less concentrated plume at 10 m AHD, which slowly decreases in concentration from July 2000 to January 2001, and is undetectable by the March 2001 sampling event. The dissection of the plume at this location is likely caused by the minor lenses of low permeability material that were detected in the gamma logs for piezometer 4 (see section 6.3.1.2 and Figure 6.6).

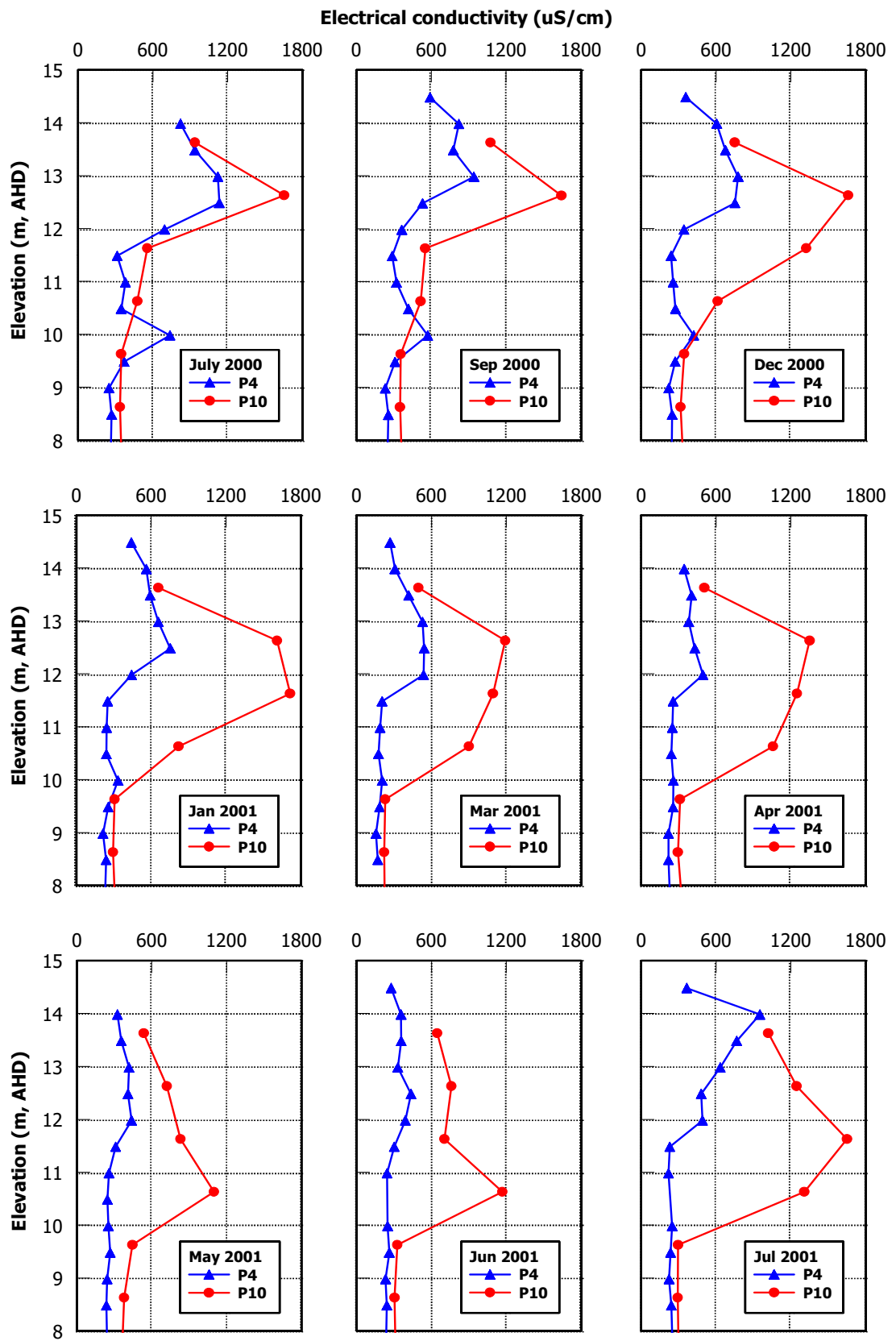


Figure 8.12 Vertical profiles of electrical conductivity during all sampling events for piezometers 4 and 10.

The relationship between the vertical location of the plume centre, the plume concentration and local rainfall records is presented in Figure 8.13. It appears that at piezometers 3, 4, 4A and 10, where the leachate plume is constrained to the shallow aquifer, the vertical disposition of the leachate plume is strongly influenced by the intensity and especially the duration of rainfall. The inference is that the additional head created by the infiltrating rainwater forces the plume downwards as the system attempts to regain equilibrium. A clear decrease in the elevation of the centre of the plume is observed at piezometers 3, 4 and 10 following the high intensity precipitation event in mid-January. A slight rebound is observed in the following mostly dry period, followed by another plume depression that correlates with a sustained wet period through April to mid-May, and finally ending with another rebound under relatively dry conditions through the end of the sampling period.

Piezometer 4A shows a curious reciprocal trend to the other piezometers downgradient from Astrolabe Park (Figure 8.13). This can be attributed to the fact that this piezometer is located in a topographical depression close to Lachlan Pond No.5; the vertical location of the plume here is controlled by the relative elevation of the pond, which rises in response to rainfall events. Thus the vertical plume fluctuations at piezometer 4A are the opposite of those observed in the other piezometers downgradient from Astrolabe Park. Because the plume at piezometer 6 occurs at a significantly greater depth than the rest of the piezometers it is less sensitive to recharge events at the surface. This is reflected in the stability of the plume throughout the majority of the sampling program. The specific mechanism responsible for the steep rise in the plume centre elevation during the last two sampling events is unknown, but may be due to the arrival of a slug of concentrated leachate that was flushed into the aquifer from Area A during the wet periods in the preceding months.

8.6.3.2 *Maximum vertical extent*

The maximum vertical extent, or thickness, of the plume also varies at piezometer 4 and 10 (Figure 8.12). While the most concentrated portion of the plume is typically limited to an approximately one metre thick section close to the centre of mass of the plume, the full vertical thickness varies between sampling events. At piezometer 10, the maximum plume thickness is approximately two metres during the July and September 2000

sampling events, then increases to three metres by December 2001, then increases to four metres from January to July 2001. The actual thickness of the concentrated centre of the plume increases from one to two metres during the January, March and July 2001 sampling events. The location of the centre of mass of the plume fluctuates within the vertical plume boundaries during this time, moving from the lower boundary to the centre to close to the water table during various sampling events.

At piezometer 4 the leachate plume retains a fairly consistent thickness of 2.5 metres from July 2000 until approximately January 2001, at which point the decreasing plume concentration causes the plume extent to retract. During this same time interval, the secondary plume is present in a one metre vertical interval beneath the primary plume, although it is undetectable by March 2001. In March 2001 the vertical extent of the primary plume has retracted to approximately two metres, then to one metre from April to June 2001, and finally increases dramatically to three metres by the July 2001 sampling event. This is indicative of the arrival of a highly concentrated slug of leachate-contaminated groundwater, which may have been flushed into the aquifer during the preceding wet period.

8.6.3.3 *Chemistry variation*

In addition to the vertical displacement of the centre of mass of the leachate plume and the expansion and contraction of its maximum vertical extent, the overall concentration of inorganic constituents within the plume has been observed to fluctuate throughout the research period. To illustrate this point, the fluid EC within the plume is used as an indicator parameter for the relative concentration of major ions in the plume. The fluctuation in fluid EC values is presented in Figures 8.12 and 8.13.

There is a large difference in the relative plume concentration at each piezometer throughout the sampling period. The variation over the sampling period with respect to the minimum observed fluid EC values at each piezometer ranges from 53 to 160%, with the piezometers closest to the landfill exhibiting the least overall variation and the downgradient piezometers exhibiting the greatest variation.

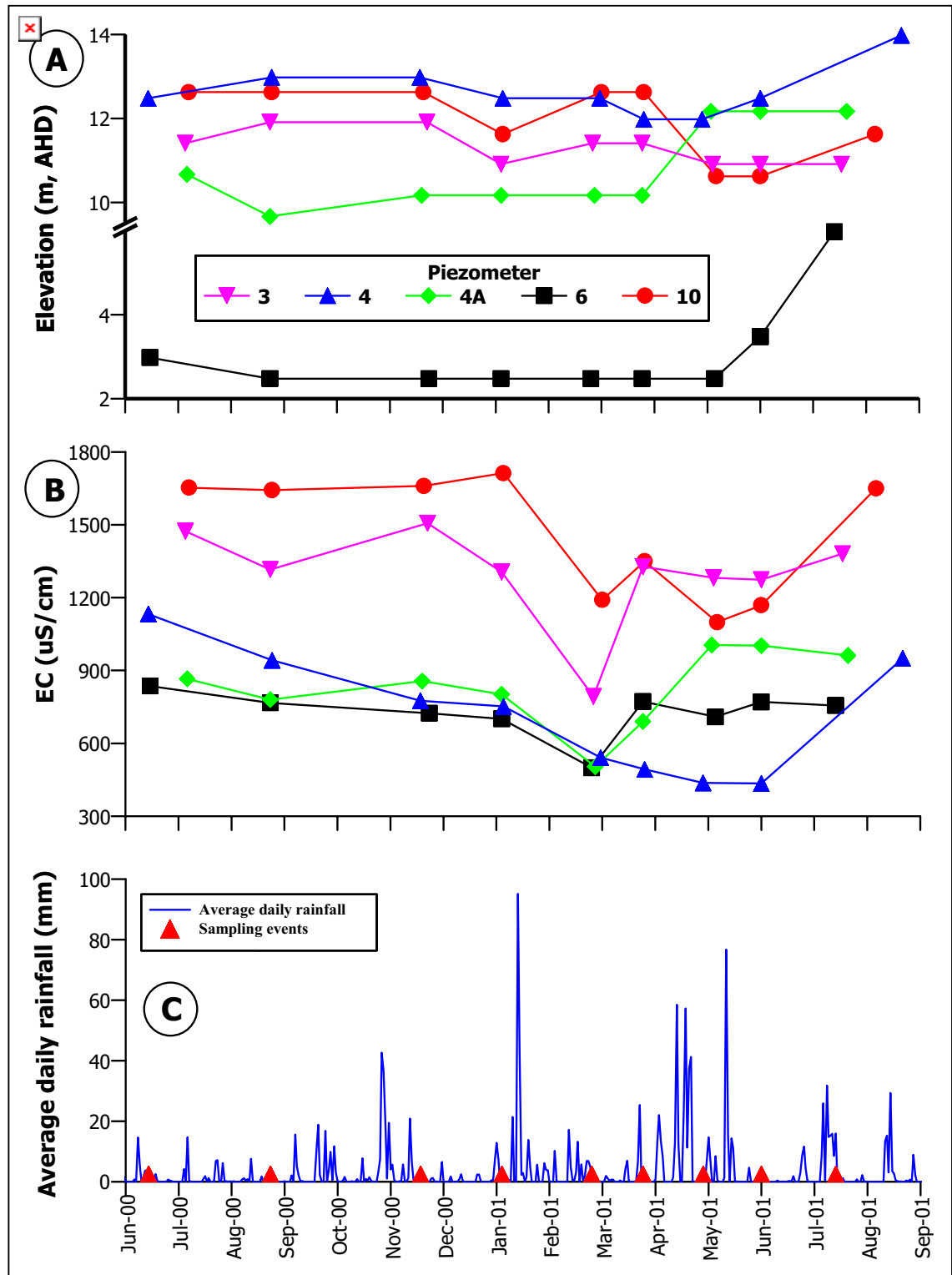


Figure 8.13 Elevation of plume centre of mass (A), maximum plume concentration expressed as fluid electrical conductivity (B), and average daily rainfall (C) in the vicinity of the study site for piezometers 3, 4, 4A, 6 and 10 for all regular sampling events.

Landfill piezometer 2 and bundled piezometer 10 exhibited a total variation of 53% and 56%, respectively. Downgradient piezometers 3 and 4A had a total variation of 90% and 110%, respectively. The greatest observed variation (160%) occurred at piezometer 4, which being located near the northern fringe of the plume is susceptible to both vertical and lateral fluctuations in plume distribution. The total variation at piezometer 6 was 67%, which indicates that the plume concentration at this location is less impacted by the processes responsible for the observed variation at piezometers 3 and 4A. It may well be that as the plume sinks further through the aquifer at piezometer six, dispersion plays a more prominent role in 'homogenising' the concentrated leachate slugs into a more uniform concentration range. Significant variations in leachate quality at a landfill in Spain were also reported by Vadillo *et al.* (1999), with a strong temporal correlation to rainfall events.

With a few exceptions, the fluctuations in maximum fluid EC measured at each piezometer during each sampling event follow a generally consistent trend. A fairly stable trend from July 2000 to January 2001 is observed (Figure 8.11), followed by a sudden decrease in plume concentration in March 2001. A subsequent increase in plume concentrations in April 2001 is observed at piezometers 3, 4A, 6 and 10, while EC values at piezometer 4 continue to slowly decrease in concentration until June 2001. It is interesting to note that between April and July 2001, the maximum EC temporal trend at piezometers 4 and 10 are relatively similar, as are the trends at piezometers 3 and 4A, although the two trends differ from one another. This may be because piezometers 4 and 10 are located at similar elevations on the potentiometric surface, and fluxes of leachate-impacted groundwater from the landfill may reach the two piezometers at approximately the same time following rainfall events. Likewise for piezometers 3 and 4A, which are both located approximately the same distance from the landfill.

The sudden decrease in fluid EC concentrations observed at all piezometers during the March 2001 sampling event is somewhat unusual, as it suggests some common mechanism or event that affected all of the piezometers simultaneously. It's possible that a significant amount of plume dilution occurred following the high intensity rainfall period in mid-January that was reflected in the groundwater chemistry in March, although the trend is not repeated following the next significant wet period in April to

mid-May. Perhaps because the March 2001 rainfall event was the first significant rainfall after several months of relatively dry conditions it had a larger diluting effect on the relatively weak leachate plume, and by the time the next significant rainfall events occurred another slug of concentrated leachate had been flushed from the landfill, which counterbalanced the dilution effects of the rainfall recharge. Regardless of the exact order of events, it appears that there is at least a loose correlation between leachate plume concentration and rainfall. It would be beneficial to assess the variations in fluid EC using a datalogger in a well screened within the plume to provide a continuous data set for temporal comparison to rainfall trends.

The expansion and contraction of the vertical extent of the leachate plume, considered in correlation with the vertical displacement of the centre of mass of the plume and fluctuations in the concentration of the plume (Figures 8.12 and 8.13), could be explained by the migration of discrete slugs of concentrated, leachate-impacted groundwater past the fixed monitoring points defined by the bundled piezometers, as suggested by the electrical images produced by surface resistivity (Figure 6.13). The evidence for the processes affecting the plume structure and chemical composition are explored in further detail in the following chapter.

CHAPTER 9: HYDROGEOCHEMICAL PROCESSES

In the previous sections the vertical and flow path distribution of elements was explored without providing a detailed analysis of the processes responsible for the mobilisation, transport and attenuation of the leachate plume contaminants. Evidence for the processes occurring at the site will be presented in the following sections, followed by an analysis of the relative contribution of the various processes to the observed distribution of elements at the site. The objectives of this section are to identify the hydrogeochemical processes with the greatest influence over the flux of *inorganic* constituents in the leachate plume, and to assess how these processes vary over time.

9.1 Dilution

Dilution of leachate plume contaminants is often identified as a significant factor affecting contaminant concentrations along flow path (Christensen *et al.*, 2001). Dilution occurs naturally along flow path as the landfill leachate mixes with groundwater in the aquifer. The effects of dilution are most easily demonstrated using conservative ion tracers such as chloride, since the concentrations of these ions are not affected by most other reactions. It is important to remember that the process of dilution simply spreads the solute load across a larger volume of water, and does not result in degradation of contaminant mass. Dilution occurs both by the processes of dispersion along flow path and mixing at Astrolabe Park, as described below.

9.1.1 Mixing

Simple mixing of contaminated and uncontaminated groundwater occurs via three mechanisms downgradient from Astrolabe Park: aquifer recharge following rainfall events, aquifer recharge following daily watering of the fairways of Eastlakes Golf Course, and mixing with pond water at the groundwater/surface water interface along the shore of Lachlan Pond No.5. The recharge from significant rainfall events has several results:

1. Rainfall recharge infiltrating above the low permeability waste layers would result in a water table mound as the excess head at the margins of the landfill equilibrates more quickly through the sandy aquifer matrix. The best evidence of water table mounding in the landfill was presented by Jankowski and Knight (1991), when a monitoring network was present throughout the landfill. This phenomenon was also reported by Kjeldsen et al. (1998b) in a detailed study of landfill hydrology at a Danish landfill (Grinsted). The head differential created by the perched water table results in the expulsion of concentrated leachate into the aquifer, that would otherwise be retained as the moisture content of the waste layers during dry periods;
2. Infiltrating rainwater mixes with the upper margin of the leachate plume, resulting in dilution of the upper plume margin and a possible depression of the plume centre. Evidence of this process can be seen in the vertical chemical profiles in the previous chapter, with significantly lower concentrations of the major ions and EC/TDS in the upper metre of each piezometer (Figures 8.4 to 8.7). Evidence of the vertical mobility of the plume is presented in Figure 8.13, notably for the January and May 2001 monitoring events, which were both undertaken following significant rainfall events and both demonstrated a downward shift in the plume centre by up to 1.5 m (piezometer 10). In addition, there is ample evidence of the presence of oxidising conditions in the top one metre intervals in most of the piezometers, including the presence of dissolved oxygen, nitrate and sulfate, which could only have been derived from rainfall recharge above the plume (Figures 8.4, 8.5, 8.8 and 8.9). This may promote the oxidation of DOC and reduced species such as CH_4 and NH_4^+ in the plume, creating a reactive attenuation zone along the fringe of the plume (van Breukelen et al., 2003 and 2004). The elevated concentration of oxidised species near the water table and the observed steep chemical gradients with respect to reduced species immediately beneath the water table support this assertion;
3. For rainfall events large enough to contribute a significant volume of stormwater runoff to the Lachlan Ponds and raise the pond surface elevation, the hydraulic gradient between the pond and the aquifer system may temporarily reverse at the edge of the pond (Acworth, pers. comm.), and pond water mixes with the discharging front of the leachate plume; and
4. The introduction of oxygen-enriched recharge water into the landfill during rainfall events may instigate a limited period of aerobic degradation in the landfill,

generating a concentrated slug of leachate within the waste layers. Evidence of this process was observed at piezometer 10 (closest to the landfill), where elevated DO concentrations (5.5 mg/L in the plume centre) were observed following significant rainfall prior to the May 2001 monitoring event (Figure 8.11), followed by a doubling of the plume concentration (measured as EC), and near absence of DO, by the subsequent monitoring event. Due to its residence time in direct contact with the waste material, this leachate is highly concentrated, and enters the aquifer as a concentrated slug once the moisture content of the waste layers is exceeded (Figure 6.13).

9.1.2 Dispersion

As a solute plume moves through a porous medium the plume constituents are naturally diluted by dispersion, or the variable advective transport rates of solutes due to micro-scale differences in flow paths through pores, variations in pore diameters, and differences in flow rates due to the surface friction effects of the aquifer matrix (Fetter, 1994). This process is clearly demonstrated by tracer tests, where a pre-defined volume of tracer fluid is introduced into an aquifer and subsequently spreads out longitudinally and laterally and is diluted due to dispersion as it is transported along flow path (Mackay et al., 1986; Garabedian et al., 1991; LeBlanc et al., 1991; Boggs et al., 1992; Jensen et al., 1993; Beck and Jankowski, 1998; Jankowski and Beck, 2000). Plume dilution due to dispersion is most pronounced at the leading edge of a contaminant plume, where the contaminated groundwater mixes directly with uncontaminated groundwater. Also, it is well established that the effects of dispersion are scale dependent, becoming more pronounced with distance along flow path (Gehlar et al., 1992).

At Astrolabe Park the effects of dispersion are diminished for two reasons. First, the leachate that is currently generated in the waste layers is flushed into a shallow aquifer that is already strongly contaminated. The leachate plume that formed shortly after the commencement of landfilling activities in the former sand mining pits would have resulted in the complete contamination of the shallow aquifer zone between the landfill and the pond within the first couple of years. Thus, there is only a subtle difference in

concentration between the leachate that is flushed into the aquifer and the contaminated groundwater in the shallow aquifer. Second, the distance between the downgradient edge of Astrolabe Park landfill and the shore of Lachlan Pond No.5 represents a relatively short flow path. Further, the vertical extent of the plume is constrained by the water table and the low permeability confining layer underneath, limiting the potential for vertical transverse dispersion. Thus, there was very little room for the effects of dispersion to develop between the landfill and the pond.

Table 9.1 Change in maximum chloride concentration (mM) along a groundwater flow path between piezometers 10 and 3, and piezometers 30 and 4A.

Sample date	Jul-00	Dec-00	Mar-01	May-01	Jul-01	Feb-02
10	2.133	2.341	2.313	1.629	1.692	--
3	1.678	2.200	1.974	1.861	1.777	--
30	--	--	--	--	--	2.877
4A	--	--	--	--	--	1.607
% change	-21.3	-6.0	-14.7	+ 14.2	+ 5.0	- 44.1

To analyse the effects of dispersion between the landfill and the pond, the maximum chloride concentrations between groundwater flow path piezometers 10 and 3, and 30 and 4A, are presented in Table 9.1. Changes in the chloride concentration along groundwater flow path are generally assumed to be the result of dispersion and/or mixing, as chloride generally is non-reactive. However, there is considerable variation in both the maximum chloride concentrations and the relative change in chloride concentrations along flow path between sampling events. During May and July 2001, the chloride concentrations in downgradient piezometer 3 are actually higher than in upgradient piezometer 10. While the difference in maximum chloride concentrations can be explained by the variable influx of leachate into the aquifer, if dispersion provided a significant control over the plume concentration then a consistent relative change in concentration along flow path would be expected for each sampling event. The seemingly random variation along flow path between sampling events is more indicative of a groundwater system in which discrete pulses of concentrated leachate are migrating through the shallow aquifer, with the measured chloride concentrations dependent on the relative positions of the concentrated leachate slugs with respect to the piezometers when the samples are collected. This conceptual model suggests that a simple comparison of the upgradient and downgradient Cl^- concentrations between the

landfill and the pond is insufficient to quantify the effect of dispersion on the leachate plume. Ultimately, it appears that the only significant dilution of the leachate plume constituents occurs when the plume discharges into the pond.

9.2 Redox reactions

Redox reactions are involved in both the mobilisation and attenuation of landfill leachate contaminants. Redox reactions are associated with the microbially mediated degradation of organic matter in the landfill and leachate plume, oxidation of pyrite in peaty lenses in the aquifer, and oxidation and reduction of redox sensitive species at the perimeter of the plume. Each of these processes is discussed in the following sections.

9.2.1 Redox reactions in the landfill and leachate plume

The typical redox reactions associated with the degradation of organic matter in a landfill environment are outlined in Table 4.3. Evidence of the occurrence of these processes comprises a strongly reducing groundwater environment (reflected in low Eh values), a significant increase in HCO_3^- concentrations, depletion of the available electron acceptors (O_2 , NO_3^- , Fe- and Mn- oxides, and SO_4^{2-}), and the presence of the reduced species in groundwater (NH_4^+ , Fe^{2+} , Mn^{2+} , and S^{2-}). While a detailed geochemical assessment of the presence of Fe- and Mn-oxides was not included in this research, the relationship between the aqueous redox sensitive species with increasing EC (as an indicator of the degree of leachate contamination) is presented in Figure 9.1; the data are included in Appendix B.

Figures 9.1A, C and E clearly demonstrate a trend of depleted electron acceptors with increasing leachate impact. Figures 9.1B and D demonstrate an increase in HCO_3^- and NH_4^+ with increasing leachate impact. Both are products of organic degradation,

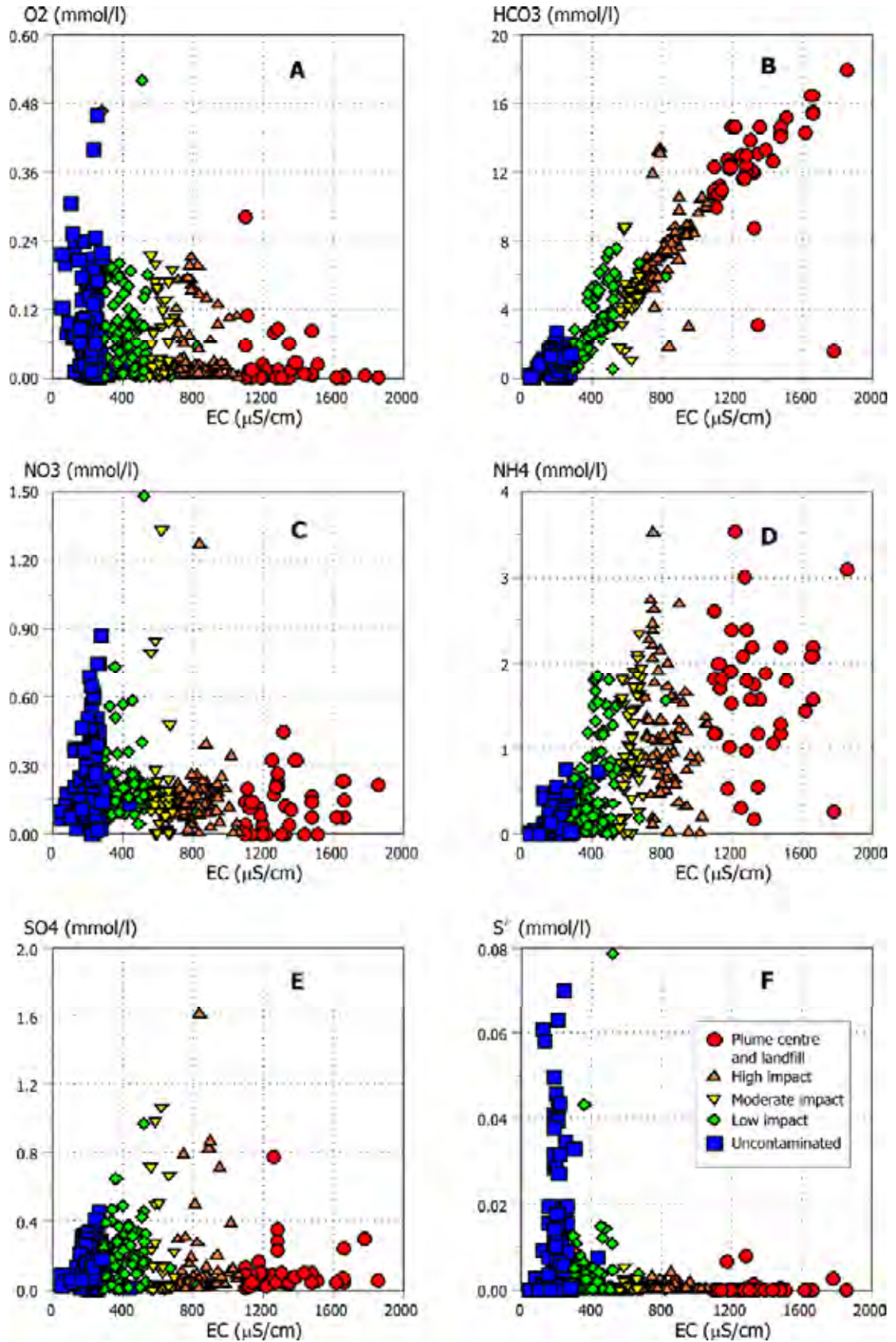


Figure 9.1 Redox sensitive elements vs. EC, indicating the distribution of redox species with increasing leachate impact (data grouped according to results of hierarchal cluster analysis; n = 498).

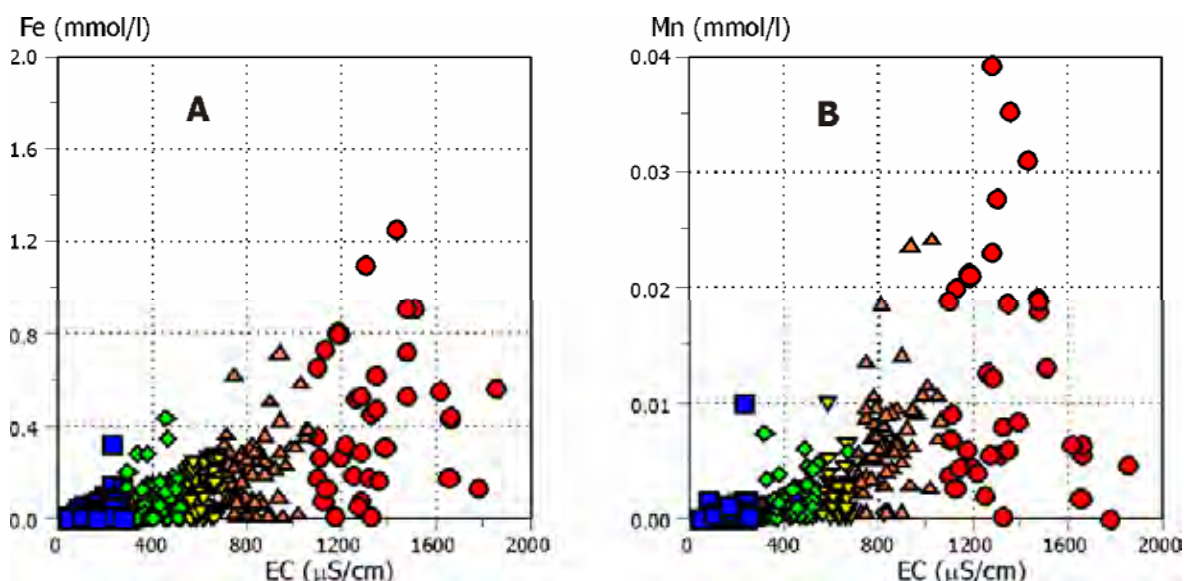


Figure 9.2 Concentration of reduced iron and manganese with increasing leachate impact (n = 498; see Figure 9.1 for symbol description).

resulting from mineralisation of organic C and N in the waste material. A fraction of the NH_4^+ , which is the most significant leachate contaminant generated through redox processes, may also be contributed by the reduction of NO_3^- present in the

uncontaminated groundwater.

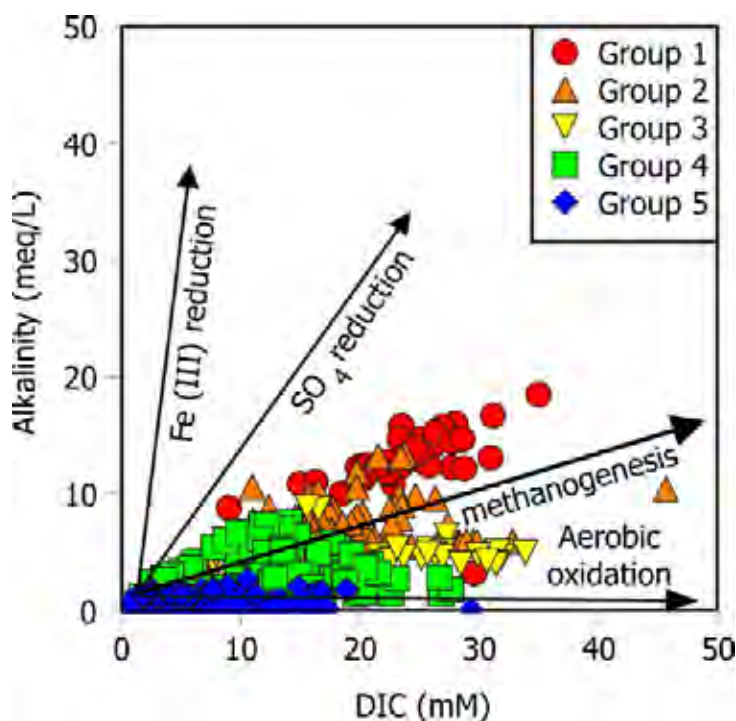


Figure 9.3 Relationship between DIC and alkalinity demonstrated in a “Deffeyes” plot, with indicators of trends produced by various redox reactions (“ideal” redox trends from Herczeg et al [1991]).

A clear increase in the concentrations of the reduced species Fe^{2+} and Mn^{2+} is observed with increasing leachate impact (Figure 9.2), although iron is present at much higher concentrations than manganese. The reduction of Fe(III) consumes a significant amount of H^+ , and in addition to oxidising dissolved organic matter it also provides an important control with regards to pH buffering observed within leachate-

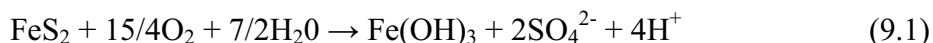
impacted groundwater environments (eg Kehew and Passero, 1990).

The various redox reactions that involve aerobic or anaerobic oxidation of organic substrates produce different relative amounts of both dissolved inorganic carbon (DIC) and alkalinity (Herczeg *et al.*, 1991). These reactions either result in a net production or consumption of acidity, which may comprise a significant control on the pH of the groundwater system. If it is assumed that, under ideal conditions, individual redox reactions produce a set ratio of DIC to alkalinity (as determined through stoichiometric relationships), then a plot of DIC vs alkalinity values for the groundwater samples may provide insight into the dominant redox reaction(s) contributing to the observed groundwater chemistry (Deffeyes, 1965). A “Deffeyes” plot comprising all samples from the current study is provided in Figure 9.3, along with indicators of the evolutionary pathways of the various redox reactions (based on stoichiometric ratios reported in Herczeg *et al.* [1991]). This plot suggests that both methanogenesis and sulphate reduction influence the evolution of groundwater chemistry in the leachate plume. The least squares regression line for the Group 1 samples (landfill and centre of leachate plume) is almost exactly equivalent to the slope of the methanogenesis line. In addition, the regression line for the Group 5 samples (uncontaminated) is equivalent to the aerobic oxidation line, suggesting that aerobic oxidation is a significant process in the uncontaminated groundwater system.

9.2.2 Oxidation of pyrite in the peaty confining layer

Figure 9.1F exhibits an interesting trend, with the highest concentrations of the reduced sulphur species clearly associated with uncontaminated (or low contamination) samples, which is contrary to the other reduced species. The reduced sulphur species resulting from SO_4^{2-} reduction within the leachate plume would rapidly precipitate as iron sulfide in the presence of elevated Fe^{2+} concentrations in the plume (Figure 9.4A). This would support the conceptual model that the occurrence of elevated concentrations of S^{2-} in the system is associated with the natural oxidation of pyrite in peaty lenses. The conceptual model suggests that oxidised uncontaminated groundwater infiltrates into the peaty confining layer from beneath under a strong upwards hydraulic gradient (Figure 8.2). The pyrite in the peat lens is then oxidised by the oxygen (or potentially other electron

acceptors) in the infiltrating water, according to the following reaction (Appelo and Postma, 1999):



The SO_4^{2-} generated is subject to advective transported upwards through the peaty lens, and subsequently reduced again through microbial degradation of the peat material or the dissolved organic carbon fraction within the leachate plume, according to equation 9.2. The acidity generated by the pyrite oxidation is sufficient to overcome the buffering capacity of the leachate plume, and may account for the sharp decrease in pH values observed at the same depth interval as the elevated S^{2-} concentrations (Figures 8.4 and 8.5).

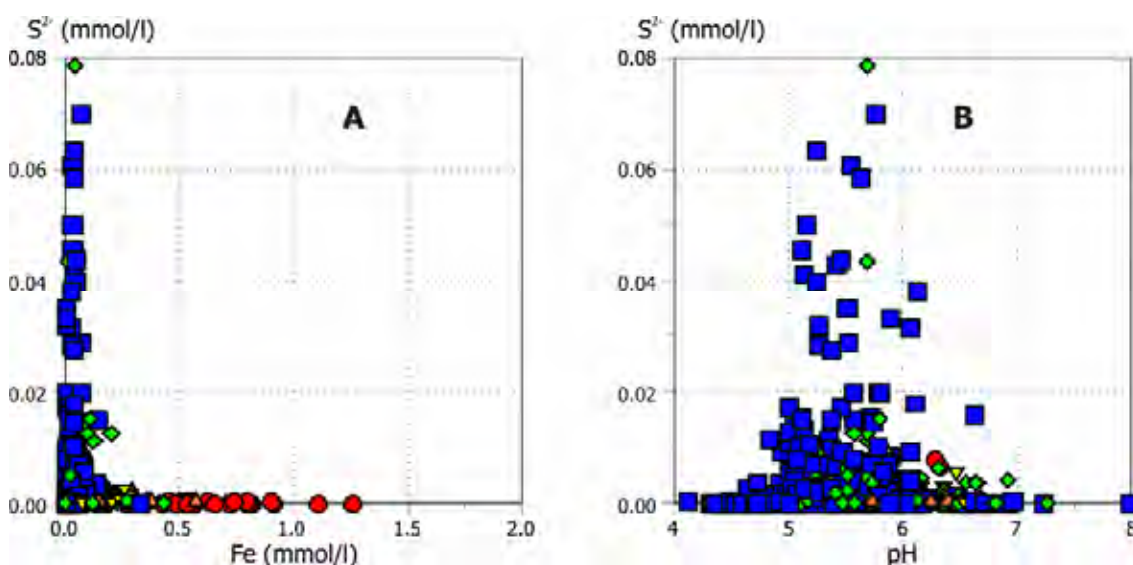


Figure 9.4 Relationship between A) reduced sulfur and reduced iron in groundwater; and B) reduced sulfur and pH in groundwater (see Figure 9.3 for symbol description).

This relationship is further demonstrated by Figure 9.4B, where the elevated S^{2-} is associated with a mean pH value of 5.5. The additional influx of SO_4^{2-} into the lower boundary of the leachate plume provides sulphate-reducing bacteria with a surplus of electron acceptors to use in the degradation of dissolved organic carbon in the plume. The resultant production of S^{2-} acts to precipitate reduced iron and other trace metals out of solution as highly insoluble metal sulfides, providing a key attenuation mechanism along the lower boundary of the plume.

9.2.3 Redox reactions at the upper margin of the plume

The final significant redox process occurring between the landfill and the pond is the oxidation of reduced species along the upper plume fringe due to 1) mixing with oxidised recharge water, and 2) fluctuations in the water table. The best evidence of this process is from observations of SO_4^{2-} concentrations at the shallow sample points of the piezometers located within Eastlakes Golf Course (3, 4, 4A, 6), where the daily watering of the golf course provides a continuous supply of oxygen-enriched recharge water to the upper margin of the leachate plume, but the effects are also observed at all piezometers where sample collection followed periods of heavy rainfall (Figures 8.4, 8.5, 8.8 and 8.9; Appendix B).

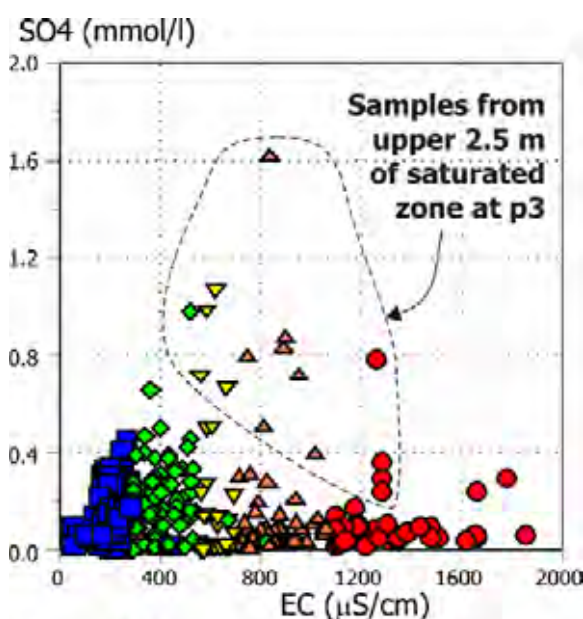


Figure 9.5 Sulphate concentrations in shallow groundwater at piezometer 3 (see Figure 9.1 for symbol description).

The SO_4^{2-} profile at piezometer 3 from the July 2001 sampling event provides a good example of the effects of water table fluctuation on the redox processes along the upper margin of the plume. The SO_4^{2-} concentration above the leachate plume reaches 103 mg/L, well above background concentrations and nearly three times greater than the SO_4^{2-} concentration at the water table at upgradient piezometer 10 (Figure 8.10). A similar distribution was reported by Ulrich et al. (2003) in a study of SO_4^{2-} sources at Norman Landfill in Oklahoma,

USA. In their study, they found elevated concentrations of SO_4^{2-} and low concentrations of iron sulfides at the water table, despite high SO_4^{2-} reduction activity in the sediment slurries for this zone. In batch incubations in which sediments were amended with various electron acceptors, Ulrich et al. (2003) found much higher rates of iron sulfide oxidation in the aerobic slurries as compared to the anaerobic slurries amended with other common electron acceptors (Mn-oxide, Fe-oxide, NO_3^-). The rate of oxidation of

iron sulfide under aerobic conditions near the water table was determined to be greater than the rate of SO_4^{2-} reduction, resulting in the observed elevated SO_4^{2-} concentrations. Fluctuations in the water table at piezometer 3 are greatly influenced by a dynamic response to the water level of the pond, which is prone to rapid changes in response to storm events as it receives a large volume of stormwater runoff. During such fluctuations, the hydraulic gradient has even been observed to reverse for a limited time (Acworth, pers. comm.), causing oxygen-enriched pond water to recharge into the aquifer near the bank of the pond. This influx of oxygen may well instigate a period of high intensity aerobic oxidation of iron sulfide precipitates that were formed within the leachate plume, resulting the elevated SO_4^{2-} concentrations observed near the water table at piezometer 3 (Figure 9.5).

Degradation of organic waste in the landfill, and dissolved organic carbon in the leachate plume, through redox processes results in a strongly reducing groundwater environment. The reducing capacity of the plume is buffered by an influx of electron acceptors above and below the plume, as well as at the dynamic zone of groundwater/surface water interaction at the shore of Lachlan Pond No.5. The combined influence of advective transport of electron acceptors from below the plume, rainfall recharge, lawn watering and surface water/groundwater interactions along the shore of the pond provides important redox buffering mechanisms with regards to the leachate plume, and results in the vertical redox zonation observed through the plume.

9.3 Dissolution/precipitation reactions

The stability of an element between the solid and dissolved phase in an aquifer environment is a function of a complex, interrelated set of factors, including the chemical composition, ionic strength, temperature, pH and redox conditions of the groundwater, and the composition and solubility of the minerals that comprise the aquifer matrix. An induced variation in one or more of these factors, as is caused by the release of concentrated landfill leachate into an aquifer, can promote dissolution of the aquifer minerals, or cause minerals to precipitate from solution.

9.3.1 Dissolution of waste material and aquifer minerals

The dissolution of waste material in the landfill results in the elevated solute loads characteristic of landfill leachates. Several mechanisms are involved, including equilibrium reactions between soluble waste components and relatively fresh, acidic groundwater, slow hydrolysis of refractory waste components, and reductive dissolution of various elements (particularly metals) as the groundwater environment becomes increasingly reducing (Andreottola and Cannas, 1992). Dissolution occurs most rapidly in young landfills where the production of organic acids lowers the pH and increases the chemical aggressiveness of the groundwater towards inorganic waste. With the onset of methanogenesis in the landfill the pH typically rises towards neutral and the total solute load decreases relative to the aerobic and acetogenic phases of waste decomposition. However, the ongoing infiltration of uncontaminated groundwater and rainfall recharge with a low solute load that comes into prolonged contact with the waste material in the low permeability waste layers continues to generate a concentrated leachate characterised by elevated concentrations of major and trace elements relative to regional uncontaminated groundwater. The release of strongly reducing landfill leachate into an oxidising aquifer may then cause stable aquifer minerals (notably Fe- and Mn-oxides/hydroxides) to become soluble through reductive dissolution, and results in the precipitation (or at least saturation) of reduced mineral species such as FeS (ppt), pyrite (FeS_2), siderite (FeCO_3) and rhodocrosite (MnCO_3).

The concentrations of major elements with increasing leachate impact (indicated by increasing EC values) are presented in Figure 9.6. A clear trend of increasing concentrations with increased leachate impact is evident for all elements except for chloride. This suggests that, as expected, chloride behaves conservatively and concentrations do not change dramatically as a result of leachate impact. Sulphate is another exception for the major elements as it is a redox sensitive element and subsequently exhibits a decreasing trend with increasing leachate impact (Figure 9.1E). The increase in Fe^{2+} and Mn^{2+} concentrations with increasing leachate impact is due to reductive dissolution of aquifer minerals, facilitated by the strongly reducing conditions within the leachate plume (Figure 9.2). A few outliers are present in each plot, which indicate either localised anomalies in the system or analytical error.

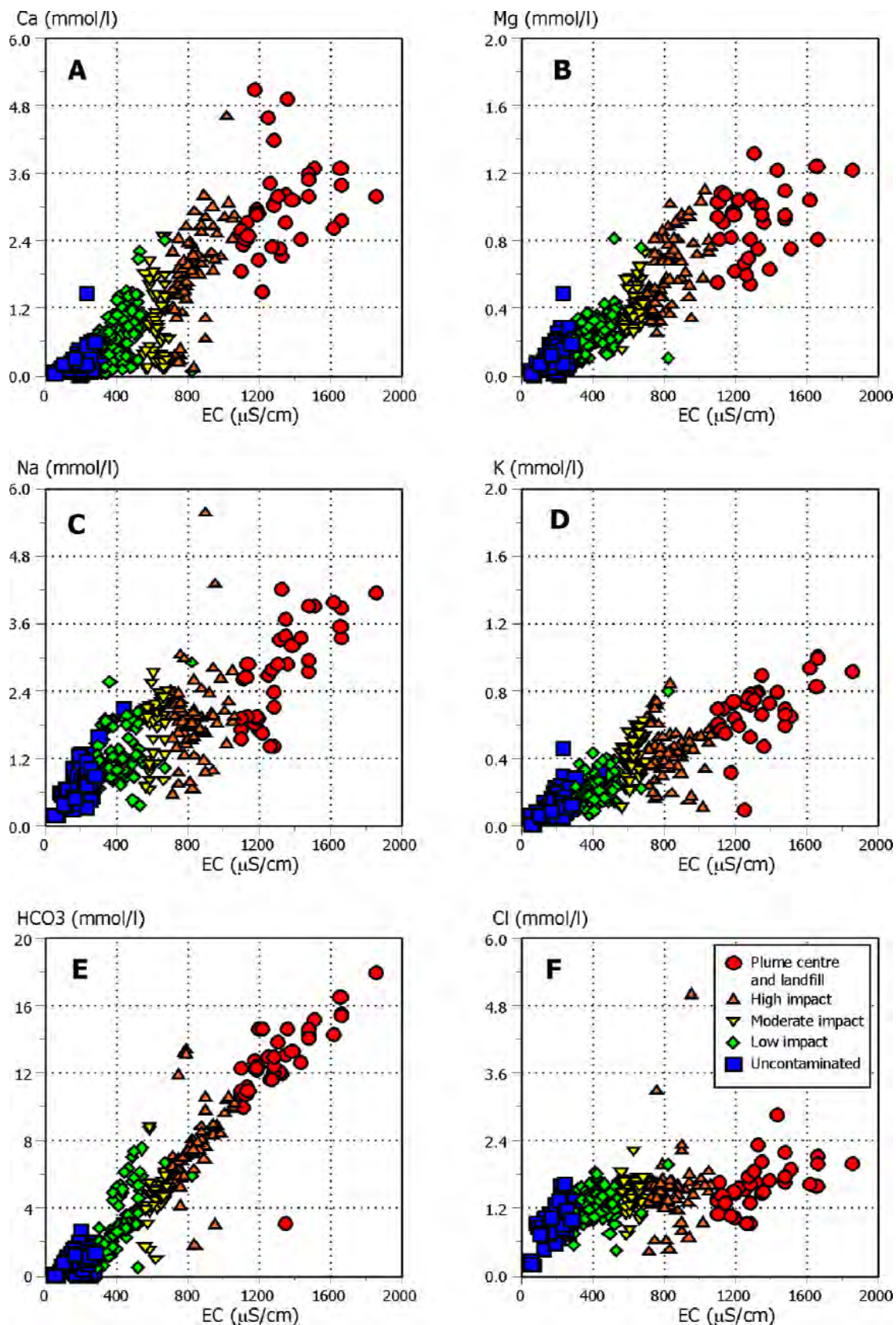
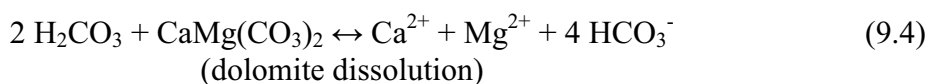


Figure 9.6 Concentrations of major elements with increasing leachate impact (n=498).

A substantial component of the major elements in the landfill leachate appear to be derived from the dissolution of carbonate-based waste material. Dissolution of naturally occurring carbonate minerals in the aquifer, in the form of shell fragments deposited in discrete horizons within the sandy aquifer, also contributes to the chemical signature of the groundwater in this area (Jankowski and Beck, 2000). However, the concentrations of the elements associated with carbonate dissolution (Ca^{2+} , Mg^{2+} , Sr^{2+} , HCO_3^-) in the leachate plume are significantly higher than are found in uncontaminated groundwater, indicating that waste material is the most likely source. Carbonate dissolution occurs according to the following reactions:



Equation 9.3 results in a Ca:HCO₃ molar ratio of 1:2, and equation 9.4 results in a Ca:Mg:HCO₃ molar ratio of 1:1:4. These would be the expected molar ratios if dissolution of carbonate minerals alone were responsible for these elements in groundwater.

Plots of the carbonate elements are presented in Figure 9.7. There is a strong positive correlation between some of the elements commonly associated with carbonate minerals (Figures 9.7A,B) suggesting a common source for at least the Ca^{2+} , Mg^{2+} and Sr^{2+} . In a study of chemical processes in uncontaminated groundwater a short distance from Astrolabe Park, Jankowski and Beck (2000) presented evidence of natural carbonate dissolution from shell fragments, but only reported maximum concentrations for Ca^{2+} and HCO_3^- of 1.89 mmol/L and 3.04 mmol/L, respectively, as opposed to the maximum concentrations of 5.09 and 18.0 mmol/L within the leachate plume. The large excess of HCO_3^- with respect to Ca^{2+} and Mg^{2+} within the leachate plume indicates that mineralisation of organic carbon is the primary source of aqueous carbonate species in the leachate plume. Figures 9.7C and D demonstrate the relationship of the molar ratio of Ca:HCO₃ with increasing EC, with lines representing the molar ratios of carbonate dissolution included. Figure 9.7C indicates a surplus of Ca^{2+} with respect to HCO_3^- in the less contaminated samples, which rapidly becomes a deficit in the more

contaminated samples. Figure 9.7D presents the same data on a logarithmic scale, which makes it easier to determine the average Ca:HCO₃ ratio within the leachate plume (approximately 0.2). The concentrations of the major elements are also influenced by ion exchange reactions, as will be discussed in the following section, but the strong correlation between Ca²⁺, Mg²⁺ and Sr²⁺ provide ample evidence of dissolution of carbonate waste in the landfill.

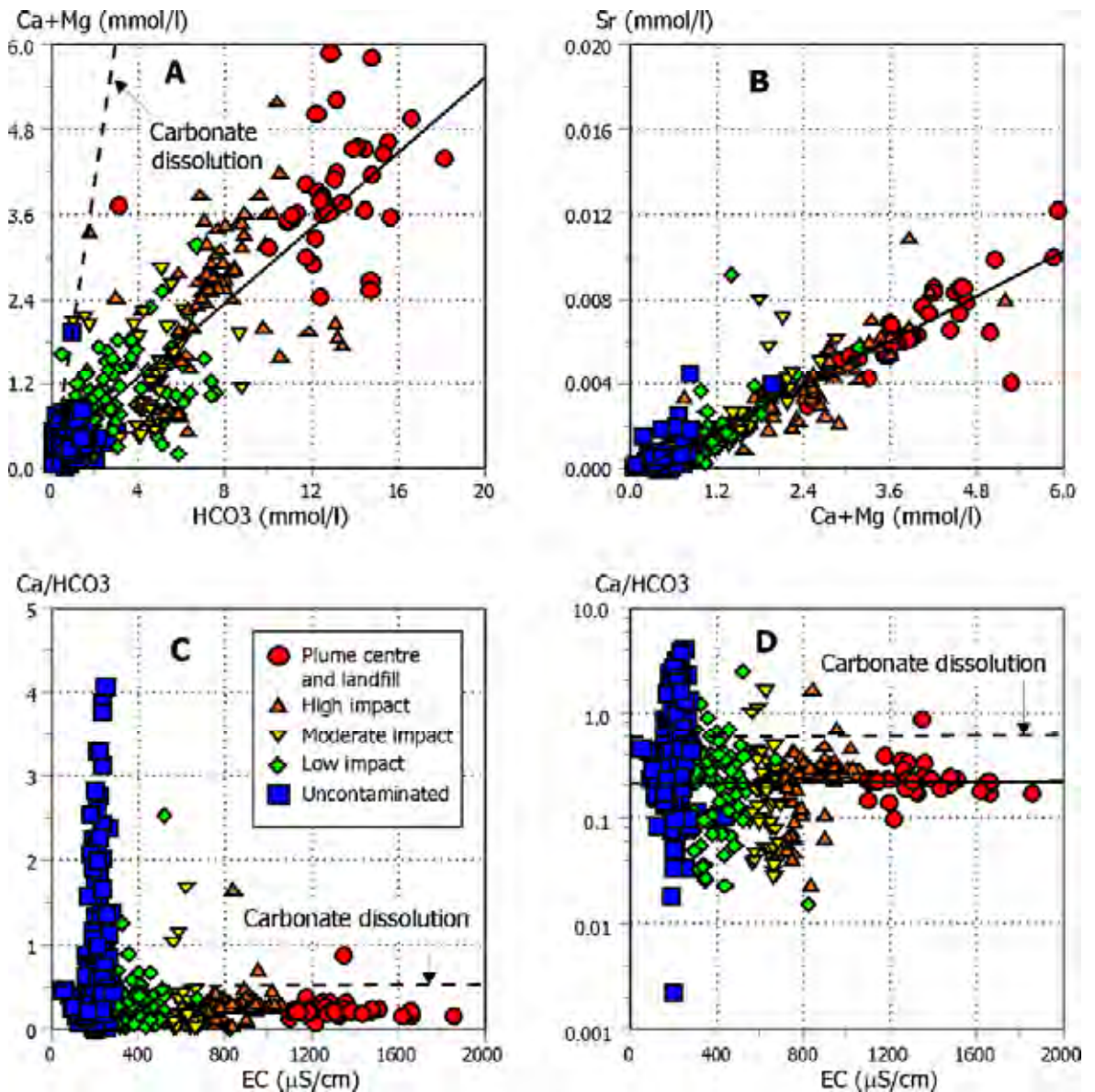


Figure 9.7 Major and trace elements associated with carbonate dissolution.

9.3.2 Mineral precipitation

The strongly reducing environment of the leachate plume promotes the reductive dissolution of metal oxides and hydroxides in the aquifer matrix, as well as the concomitant precipitation of reduced mineral species, including sulfide and carbonate minerals. The interaction between minerals and groundwater can be expressed the saturation index (SI); a negative SI value indicates that the groundwater is undersaturated with respect to the given mineral and it will tend to dissolve, while a positive SI value indicates that the groundwater is supersaturated with respect to the mineral phase and it will tend to precipitate. Neutral SI values (0 ± 0.5) indicate that the solution is in equilibrium with respect to the given mineral.

It should be noted that the precipitation kinetics varies for different minerals, with particularly slow precipitation kinetics reported for siderite (FeCO_3) and rhodochrosite (MnCO_3) (Jensen *et al.*, 2002), and pyrite (Appelo and Postma, 1999). Thus, the SI values for minerals with slow precipitation kinetics can become quite elevated, while the actual concentrations of the mineral elements in groundwater are controlled by more kinetically rapid reactions. In the cases of minerals with slow precipitation (or dissolution) kinetics, the SI value indicates the direction that the reaction is likely to proceed, and does not necessarily reflect equilibrium conditions in groundwater. Summary tables for the calculated saturation indices are presented in Appendix C.

The saturation indices for the minerals determined to be the most relevant to the leachate-impacted aquifer system are presented in Figures 9.8 to 9.10, with the statistical parameters for each mineral summarised in Table 9.2. In addition, vertical profiles of SI values for the target mineral phases are presented in Figures 9.11 and 9.12 to assist with correlating mineral saturation with location within the leachate plume. The target mineral phases (iron oxides/hydroxides, carbonates and iron sulfides) were selected on the basis of their association with elements present at elevated concentrations within the leachate plume, especially the mineral phases that precipitate under reducing conditions (iron sulfides and carbonates). The SI values were modelled using PHREEQC, and speciation modelling results are most sensitive to pH, pe and the

Table 9.2 Statistical parameters for saturation indices calculated using PHREEQC ($n = 498$).

	Max	Mean	Min	Std Dev
<i>Iron Oxides/Hydroxides</i>				
Amorphous $\text{Fe}(\text{OH})_3$	1.39	-8.03	-14.82	2.61
Goethite ($\alpha\text{-FeOOH}$)	7.20	-2.33	-9.15	2.64
Hematite (Fe_2O_3)	16.41	-2.68	-16.32	5.28
<i>Carbonates</i>				
Calcite (CaCO_3)	1.15	-2.36	-6.58	1.48
Dolomite ($\text{CaMg}(\text{CO}_3)_2$)	1.89	-4.97	-11.27	2.83
Siderite (FeCO_3)	1.00	-1.58	-5.89	1.66
Rhodocrosite (MnCO_3)	0.29	-2.10	-5.63	1.38
<i>Iron Sulfides</i>				
FeS (ppt)	0.22	-2.74	-8.16	1.72
Mackinawite (Fe_{1+x}S)	0.95	-2.00	-7.43	1.72
Pyrite (FeS_2)	20.02	5.54	-3.85	3.65

concentrations of the elements associated with the modelled phases (in near-surface environments, variations in temperature are negligible).

The range of SI values presented in Table 9.2 reflects the reducing environment of the leachate plume, with several reduced mineral

phases reaching saturation (and slight supersaturation). The maximum SI values for the mineral phases calcite, siderite, rhodochrosite, FeS (ppt) and mackinawite are all close to saturation, which indicates that precipitation of these minerals in the leachate impacted aquifer is a controlling process for the elements associated with those mineral phases. As expected, the diagenetically mature mineral phases hematite and pyrite are supersaturated for nearly all samples, suggesting that it is the more reactive mineral phases that are of significance within the plume.

Figure 9.8 presents the SI values for three the iron oxide/hydroxide minerals, including amorphous $\text{Fe}(\text{OH})_3$, goethite ($\alpha\text{-FeOOH}$) and hematite (Fe_2O_3). Iron oxides and hydroxides are commonly found in oxidised aquifer systems, or in areas where reducing groundwater containing ferrous iron comes into contact with oxygen (such as near a fluctuating groundwater table, mixing zones for oxidised and reduced groundwater, or groundwater discharge zones). Of the three, amorphous $\text{Fe}(\text{OH})_3$ is considered to be the most reactive mineral phase and hematite the least. The reactivity of these minerals is reflected in the SI values, with all but a few of the samples undersaturated with respect to amorphous $\text{Fe}(\text{OH})_3$, approximately half of the samples supersaturated with respect to FeOOH , and just over a third of the samples supersaturated with respect to Fe_2O_3 .

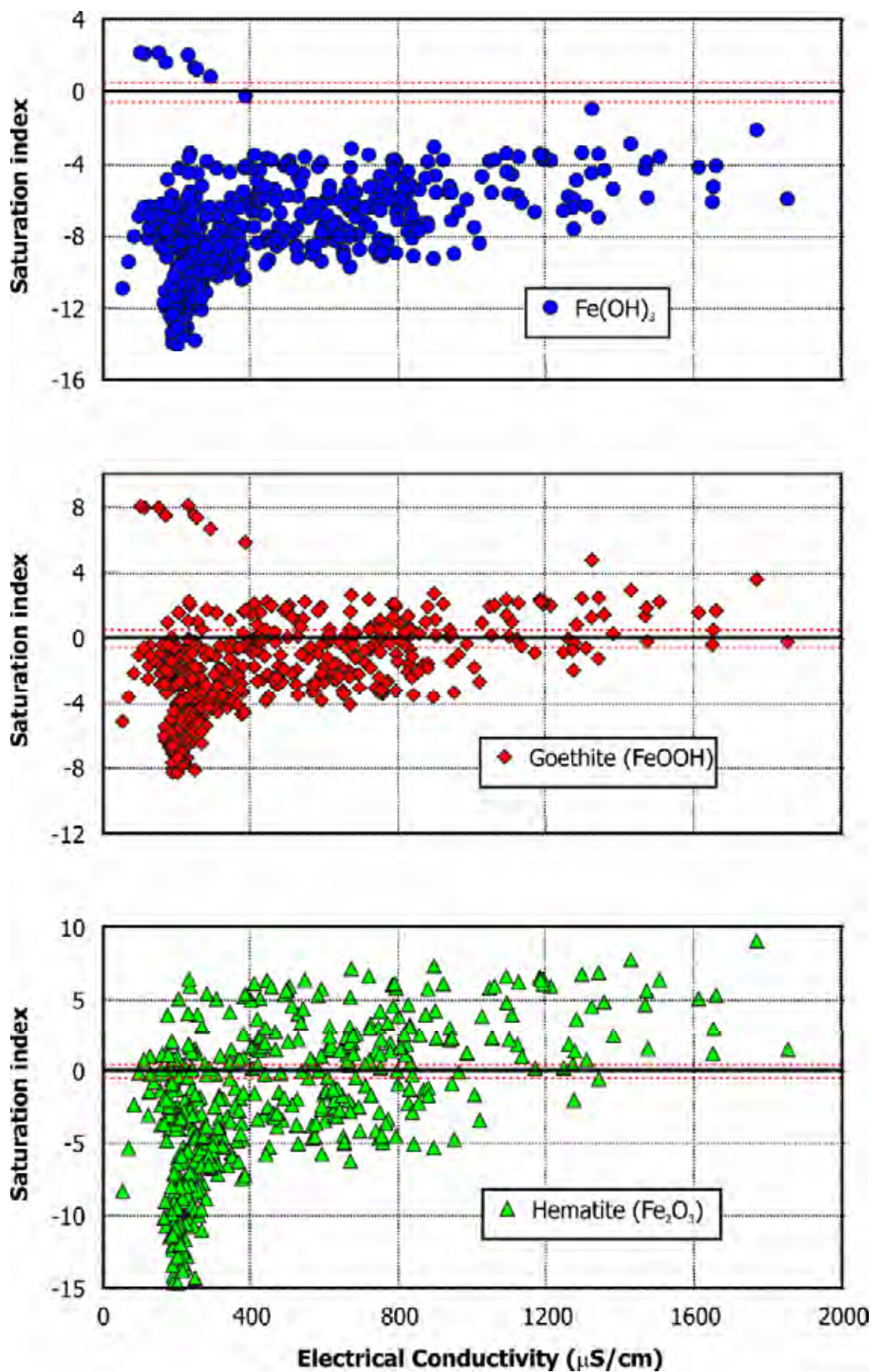


Figure 9.8 Saturation indices for iron oxide/hydroxide minerals versus the degree of leachate impact (represented by fluid EC values).

The variation in the SI values for these minerals is a function of the iron concentration, the presence of atmospheric oxygen, and the pH and redox condition of the water samples. The majority of the low EC water samples that are undersaturated with respect to all of these minerals; these represent the uncontaminated water samples, in which the iron concentrations are extremely low, the pH values are more acidic, and the redox conditions are naturally slightly reducing. For both goethite and hematite, however, saturated and supersaturated conditions are present in groundwater samples across the EC spectrum. It might be expected that Fe oxides and hydroxides would precipitate close to the water table, or in the groundwater discharge zone in the pond, but not within the strongly reducing environment of the leachate plume. These SI values should be interpreted to indicate that amorphous $\text{Fe}(\text{OH})_3$ is reactive within the plume, while goethite and hematite are relatively stable, with much slower dissolution kinetics.

Figure 9.9 presents the SI values for the major carbonate minerals that may be influencing the leachate plume chemistry, including calcite (CaCO_3), dolomite ($\text{CaMg}(\text{CO}_3)_2$), siderite (FeCO_3) and rhodochrosite (MnCO_3). As organic matter in the landfill waste is degraded, the organic carbon is mineralised, either in the form of CH_4 or $\text{CO}_{2(\text{aq})}$. This large flux of mineralised carbon into groundwater, coupled with an influx of Ca^{2+} and Mg^{2+} from waste dissolution and ion exchange processes and Fe^{2+} and Mn^{2+} from reductive dissolution of aquifer minerals, causes the leachate-impacted groundwater to become saturated with respect to the carbonate minerals. The saturation and supersaturation of carbonate minerals has been reported in several published landfill studies (Nicholson *et al.*, 1983; Kehew and Passero, 1990; Bjerg *et al.*, 1995; Amirbahman *et al.*, 1998; Jensen *et al.*, 2002), and precipitation of carbonate minerals from landfill leachate was identified as a significant control over the leachate plume composition (van Breukelen *et al.*, 2003). Manning (2001) estimated calcite precipitation to be a sink for up to 45% of the carbon flux resulting from mineralisation of organic carbon in a landfill.

Clear trends of increasing carbonate mineral equilibrium and supersaturation are evident with increasing leachate impact (represented by fluid EC values) (Figure 9.9). In particular, calcite and siderite precipitation appears to constitute a significant control on

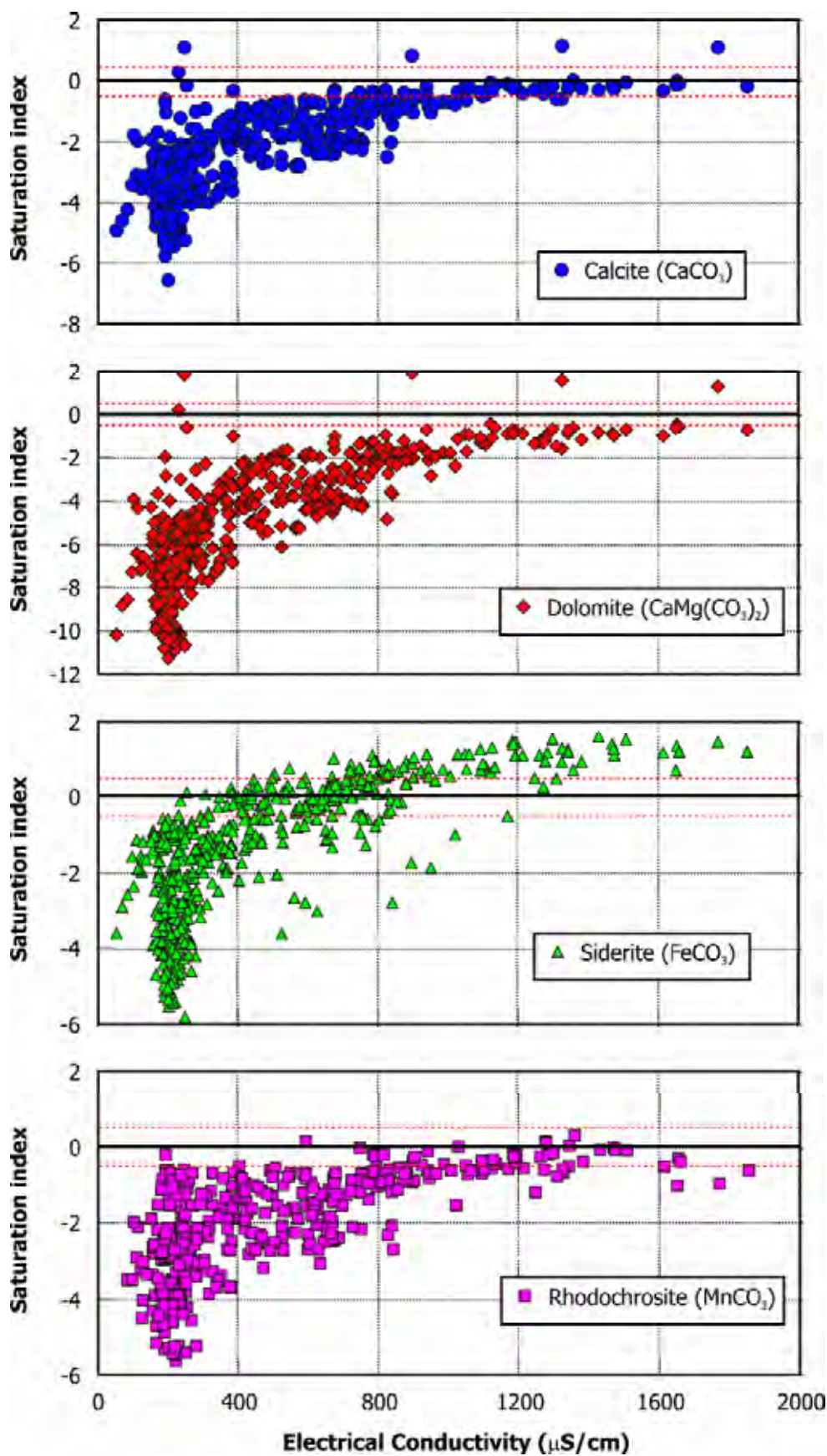


Figure 9.9 Saturation indices for carbonate minerals versus the degree of leachate impact (represented by fluid EC values).

leachate plume chemistry with increasing leachate impact, while dolomite and rhodochrosite are less influential due to lower concentrations of Mg^{2+} and Mn^{2+} in the aquifer system. Physical evidence of calcite precipitation from landfill leachate has previously been observed through laboratory experiments, analysis of filtered precipitate, and analysis of precipitates clogging leachate drainage systems (Fleming *et al.*, 1999; Manning and Robinson, 1999; Cooke *et al.*, 2001; Manning, 2001; Rittmann *et al.*, 2003). Calcite precipitation likely represents a stronger control over the leachate plume chemistry than siderite precipitation due to the higher concentration of Ca^{2+} in the plume and the reported slow kinetics of siderite dissolution in laboratory experiments (Jensen *et al.*, 2002). The slow precipitation kinetics of siderite have been suggested to account for the slight supersaturation of some leachate-impacted groundwater with respect to siderite (Kehew and Passero, 1990; Jensen *et al.*, 2002).

Iron sulfide minerals are highly insoluble mineral phases that readily precipitate from solution in reducing groundwater environments containing both ferrous iron and sulfide ions. The saturation indices of iron sulfide minerals are presented in Figure 9.10, including amorphous FeS precipitate, Mackinawite (Fe_{1+x}S) and Pyrite (FeS_2). Amorphous FeS precipitate is the first iron sulfide phase formed by the reaction of ferrous iron and sulfide in groundwater by a kinetically fast reaction (Appelo and Postma, 1999), while mackinawite is described as a 'meta-stable iron-sulphur phase important as a precursor to pyrite formation in sedimentary and hydrothermal systems' (Lennie *et al.*, 1995). Pyrite is the most stable iron sulfide phase, and forms primarily through diagenesis of the less stable iron sulfide phases. The precipitation kinetics of pyrite are described as sluggish (Appelo and Postma, 1999), which accounts for amorphous FeS being the rate-controlling precipitation phase for iron sulfide in groundwater.

Iron sulfide minerals are highly insoluble mineral phases that readily precipitate from solution in reducing groundwater environments containing both ferrous iron and sulfide ions. The saturation indices of iron sulfide minerals are presented in Figure 9.10, including amorphous FeS precipitate, Mackinawite (Fe_{1+x}S) and Pyrite (FeS_2). Amorphous FeS precipitate is the first iron sulfide phase formed by the reaction of ferrous iron and sulfide in groundwater by a kinetically fast reaction (Appelo and

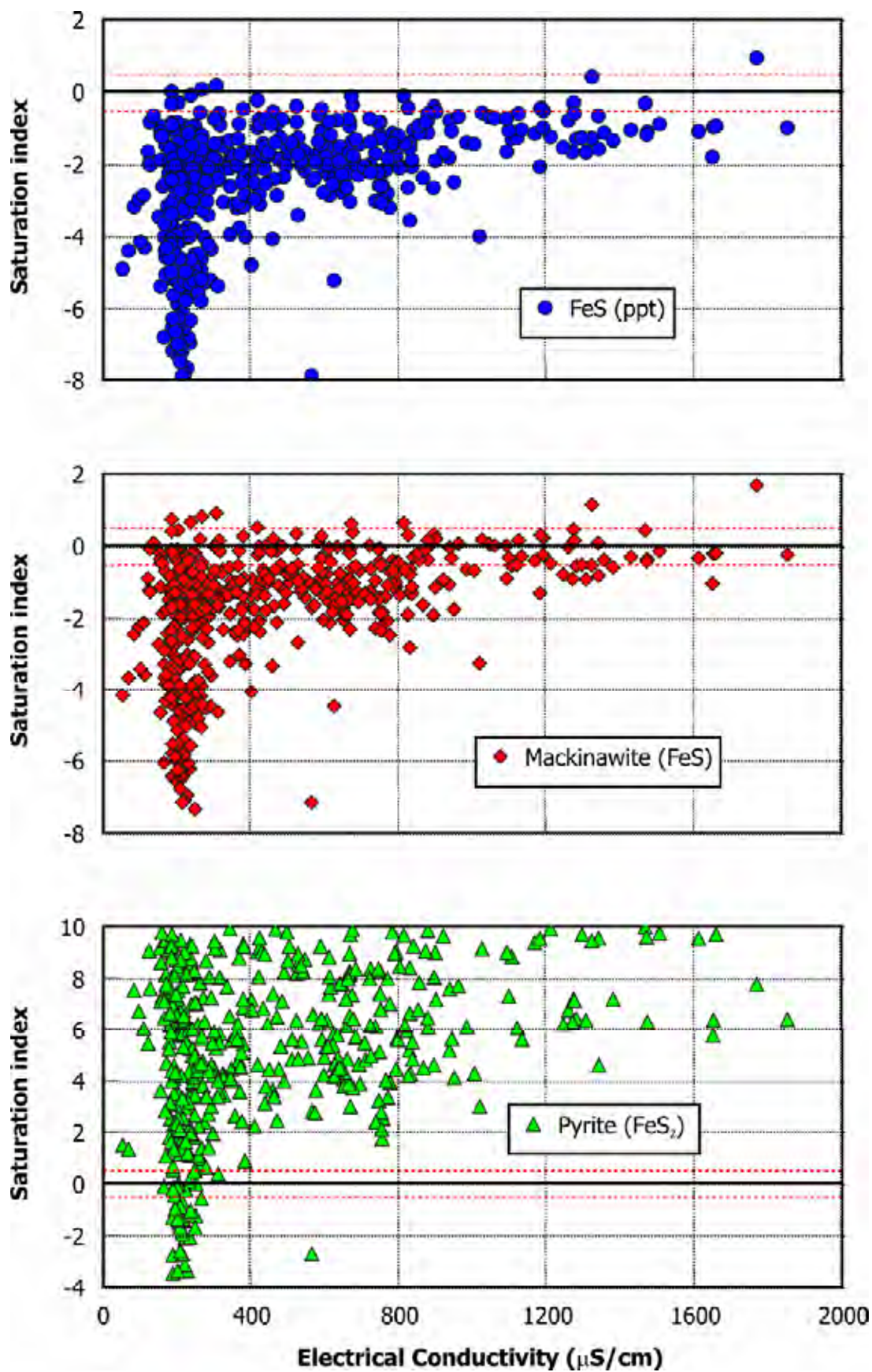


Figure 9.10 Saturation indices for iron sulfide minerals versus the degree of leachate impact (represented by fluid EC values).

Postma, 1999), while mackinawite is described as a ‘meta-stable iron-sulphur phase important as a precursor to pyrite formation in sedimentary and hydrothermal systems’ (Lennie *et al.*, 1995). Pyrite is the most stable iron sulfide phase, and forms primarily through diagenesis of the less stable iron sulfide phases. The precipitation kinetics of pyrite are described as sluggish (Appelo and Postma, 1999), which accounts for amorphous FeS being the rate-controlling precipitation phase for iron sulfide in groundwater.

Figure 9.10 indicates that precipitation of amorphous FeS and mackinawite comprises a strong control over the iron and sulfide concentrations in groundwater. It is interesting to note that in general the more highly leachate-impacted groundwater samples are undersaturated with respect to FeS and mackinawite, while the most saturated samples occur in the low EC groundwaters. The centre of the leachate plume is characterised by elevated Fe^{2+} concentrations, depleted SO_4^{2-} concentrations and little or no S^{2-} . Without a readily available source of SO_4^{2-} , the dissolved S^{2-} is precipitated completely out of solution while excess Fe^{2+} remains. The saturated low EC samples represent pore water in the lower confining layer, in which pyrite oxidation and subsequent SO_4^{2-} reduction and FeS precipitation occurs along the lower plume boundary. This excess sulfur flux into the system results in the pore water reaching saturation with respect to FeS and mackinawite. Note that the rapid kinetics of amorphous FeS precipitation preclude the water samples reaching supersaturation, while a few water samples exhibit slight supersaturation with respect to the slightly more stable mackinawite phase (Figure 9.10). The sluggish precipitation kinetics of pyrite are clearly demonstrated, with the vast majority of samples exhibiting significant supersaturation with respect to this mineral phase.

The vertical profiles of SI values for the July/August 2001 monitoring event (Feb. 2002 for piezometer 30) are presented in Figures 9.11 and 9.12. The following trends are noted with regards to the vertical distribution of SI values:

- Saturation of most mineral phases is associated with the centre of the leachate plume, where the highest concentrations of bicarbonate, iron and calcium are present;

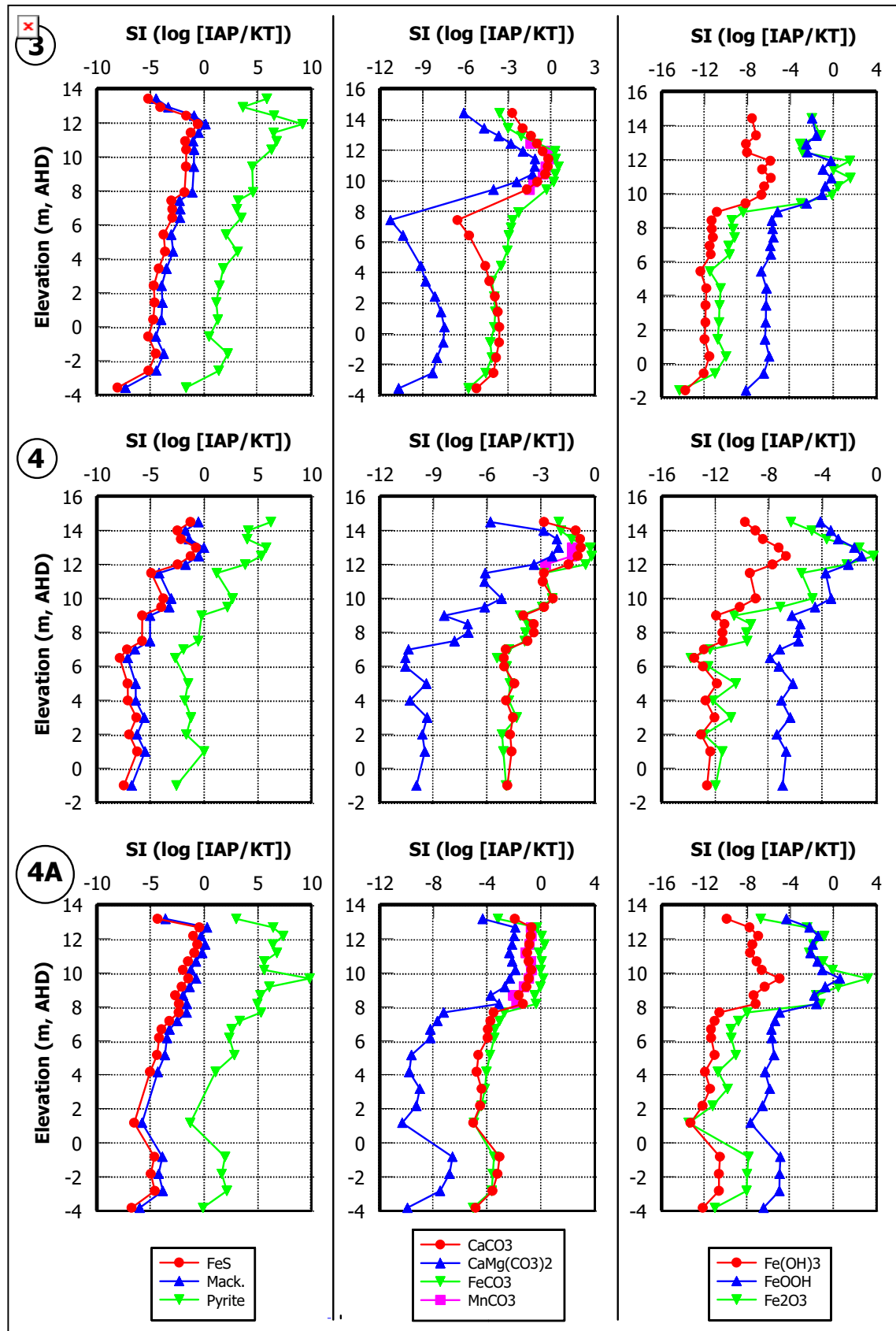


Figure 9.11 Vertical profiles of saturation indices for iron sulfides, carbonates and iron oxide/hydroxides for piezometers 3, 4 and 4A.

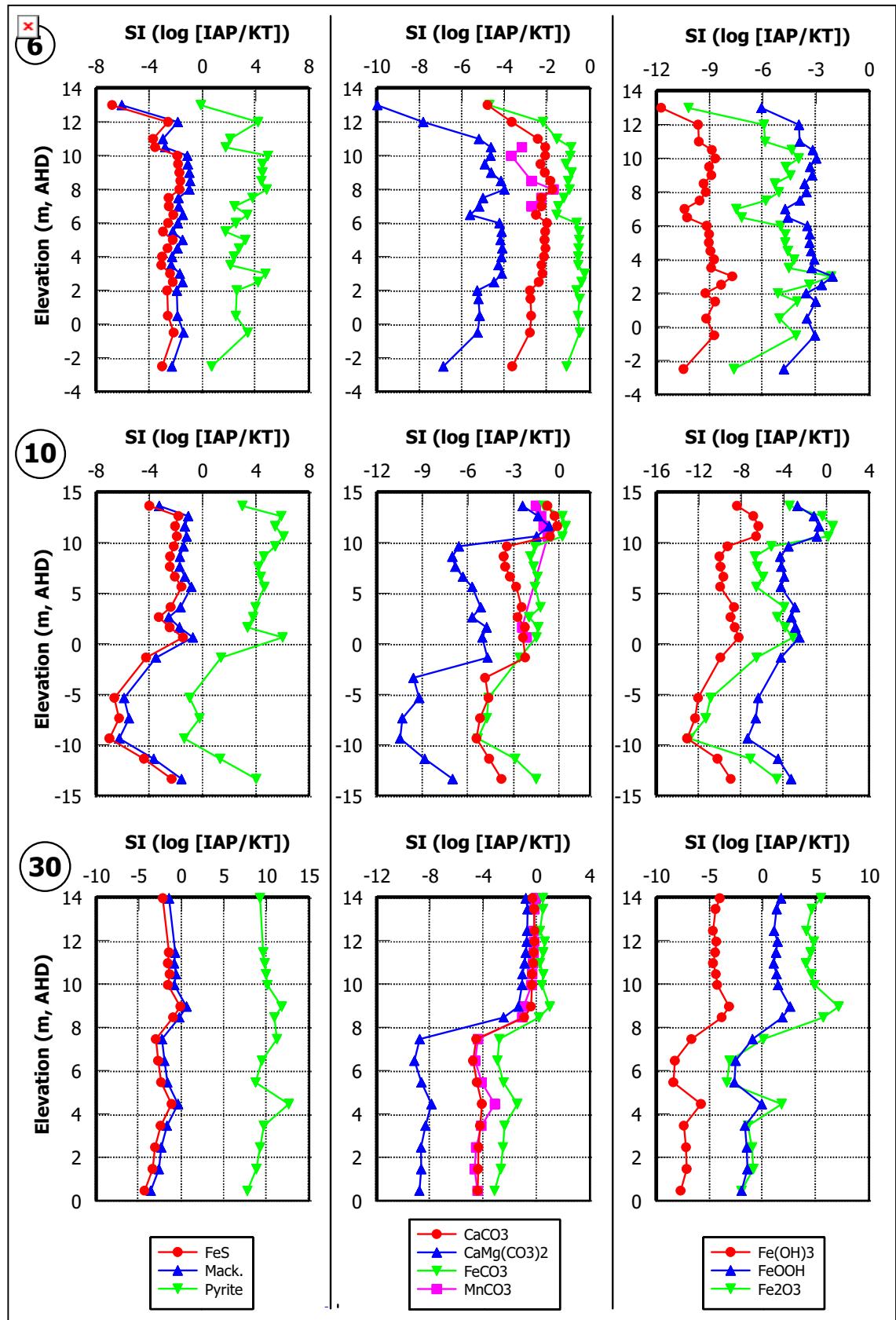


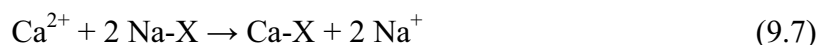
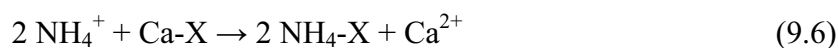
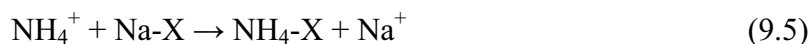
Figure 9.12 Vertical profiles of saturation indices for iron sulfides, carbonates and iron oxide/hydroxides for piezometers 6, 10 and 30.

- Both FeS (ppt) and mackinawite exhibit a similar SI trend, whilst pyrite is supersaturated in nearly every sample, indicating that this mineral phase contributes little control over this system;
- Calcite, siderite and rhodochrosite all exhibit a similar SI distribution, with all phases at saturation within the plume. Dolomite is less saturated due to the relatively low concentrations of Mg^{2+} in the aquifer system; and
- Both goethite and hematite reach saturation within the plume, while the more reactive $\text{Fe}(\text{OH})_3$ is undersaturated in all samples.

9.4 Ion exchange

Ion exchange reactions comprise another important control over the distribution of elements in the leachate plume. Research has demonstrated exchange sites can comprise a significant reservoir for cations in the aquifer system, with a considerably greater total concentration of the saturating element(s) bound to exchange sites than are present as free ions in groundwater (Thornton et al., 2000a; Christensen *et al.*, 2001). Ion exchange capacities typically range from 0.2 to 5 meq per 100 grams (eg. Bjerg and Christensen, 1993; Heron et al., 1998), which is enough to be significant with regards to the major ion chemistry of a leachate plume.

Landfill leachate typically contains elevated concentrations of constituents (specifically NH_4^+) that have a higher affinity for the exchange sites than the saturating elements in the pristine aquifer (Erskine, 2000; Thornton et al., 2000a). The preferential exchange of bound ions for more favourable ions in solution has the dual effect of providing a buffering mechanism for the more favourable exchange species and reintroducing to solution an elevated concentration of previously bound ions. The effects of ion exchange can be observed as chemical trends in groundwater with increasing leachate impact. A decrease in concentration along flow path of the preferential exchanger is expected, and the concentration in groundwater of elements typically associated with exchange sites in natural systems (ie Ca^{2+} , Mg^{2+} and Na^+) would be expected to increase (eg. Kjeldsen and Christensen, 1984). An example of some potential ion exchange reactions occurring within the leachate plume is as follows:



where the X denotes the exchange site.

Figures 9.13 through 9.16 are bivariate plots of the major elements associated with ion exchange processes. It is assumed that in pristine conditions the exchange sites in this portion of the Botany Sands aquifer would be saturated with Na^+ , since the relative paucity of carbonate minerals in the aquifer matrix results in the uncontaminated groundwater retaining the Na-Cl chemical signature of rainfall recharge. The relationship between Na^+ and Cl^- is presented in Figure 9.13A, and the ratio of Na:Cl with increasing leachate impact (represented by fluid EC values) is presented in Figure 9.13B. The primary source of Na^+ to groundwater, in the absence of Na-bearing minerals (albite, halite, etc), is rainfall. The rainfall recharge to shallow coastal aquifers will often retain an Na:Cl molar ratio close to that of seawater (0.87), and any changes in the Na^+ concentration relative to Cl^- is most likely attributable to ion exchange processes. Figure 9.13A demonstrates that Na^+ becomes enriched with respect to Cl^- at higher concentrations, and Figure 9.13B further demonstrates this trend with respect to increasing leachate impact, suggesting displacement from ion exchange sites by more favourable exchangers, notably NH_4^+ .

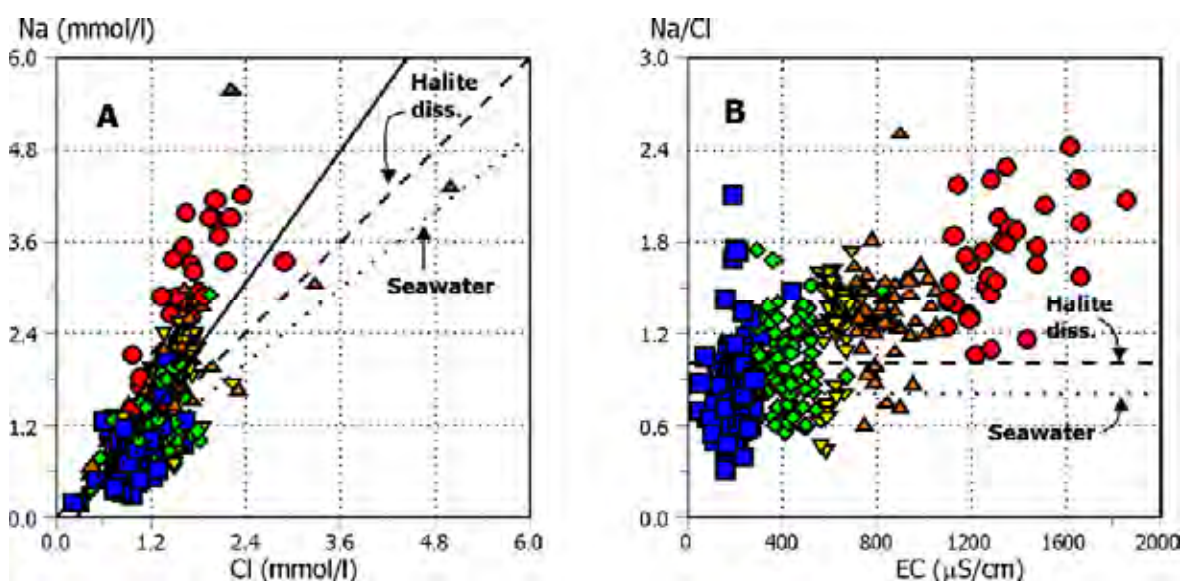


Figure 9.13 Relationship of (A) Na^+ and Cl^- in groundwater at the research site, and (B) the Na:Cl ratio relative to leachate impact (represented by fluid EC values); molar

ratios for seawater and halite dissolution provided for reference (See Figure 9.7 for symbol description).

The molar ratios of Ca^{2+} and Na^+ are presented in Figure 9.14. In many natural settings these two elements are the primary components of ion exchange processes. Magnesium behaves similarly to Ca^{2+} in ion exchange processes, but the low Mg^{2+} concentrations in this aquifer system make it a less important component. Figure 9.14A indicates a generally positive correlation between Ca^{2+} and Na^+ , although the correlation coefficient is only moderate ($r^2 = 0.53$). If one element were being exchanged for the other then a negative correlation would be expected; this trend suggests that both Ca^{2+} and Na^+ are being expelled by a preferential exchanger (most likely NH_4^+). Figure 9.14B indicates enrichment in Ca^{2+} relative to Na^+ with increasing leachate impact. Dissolution of carbonate waste in the landfill is responsible for part of the increased Ca^{2+} concentration, but there may also be an input Ca^{2+} ions bound to exchange sites that are preferentially exchanged by NH_4^+ . Calcium may be the saturating ion in discrete horizons where shell fragments have been previously observed (Jankowski and Beck, 2000). However, it is difficult to quantify the relative contributions of the two potential Ca^{2+} sources from the total concentrations in groundwater, and inverse geochemical modelling will be necessary to assess the relative contributions from both of these sources.

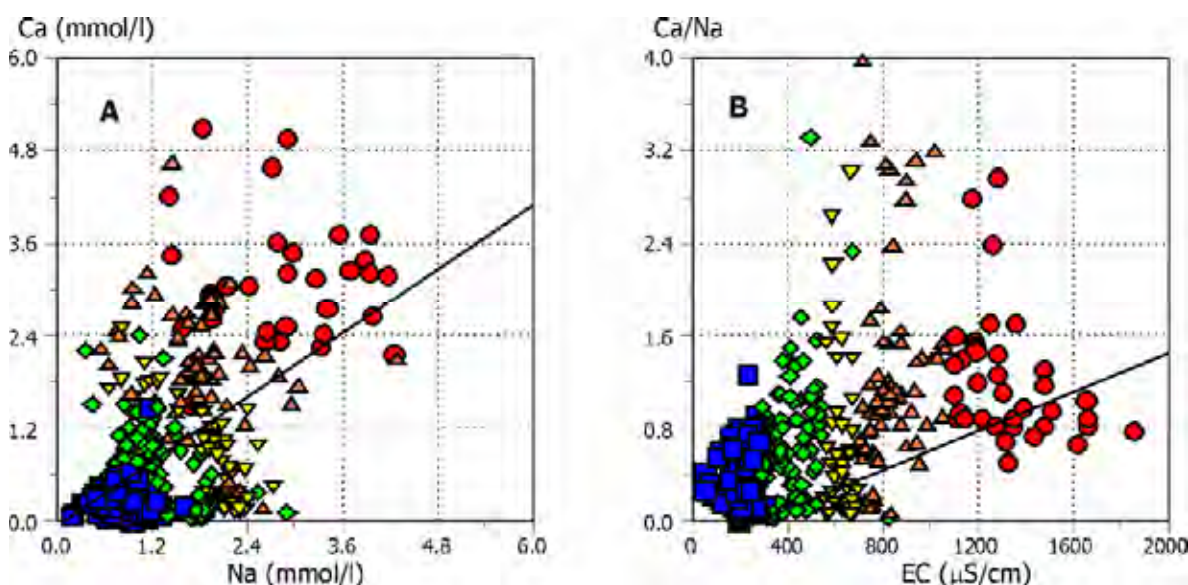


Figure 9.14 Relationship of (A) Ca^{2+} to Na^+ in groundwater at the research site, and (B) the Ca:Na ratio relative to leachate impact (represented by fluid EC values; see Figure 9.7 for symbol description).

The relationships between Ca^{2+} , Mg^{2+} , Na^+ , K^+ and NH_4^+ are presented in Figure 9.15. There is a general positive correlation between the major cations with respect to NH_4^+ . Ordinarily a negative correlation would be expected between exchanged species, but the elevated concentration of NH_4^+ produced in the landfill over several decades is enough to saturate the available exchange sites and still retain an elevated concentration in groundwater. There is a relatively strong correlation between NH_4^+ and K^+ ($r^2 = 0.85$), which underscores their similar affinity for cation exchange sites.

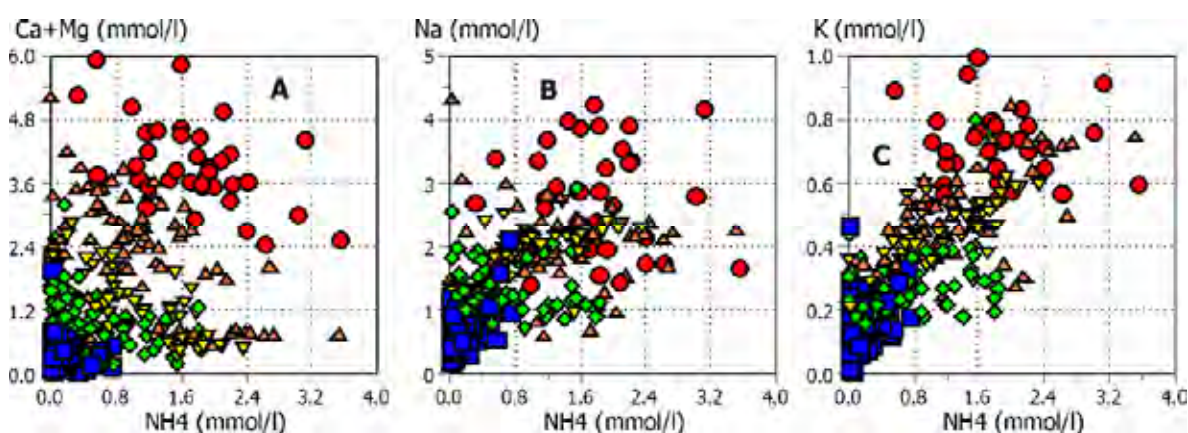


Figure 9.15 Molar relationship between NH_4^+ and (A) $\text{Ca}^{2+} + \text{Mg}^{2+}$; (B) Na^+ ; and (C) K^+ in water samples at the site (See Figure 9.7 for symbol description).

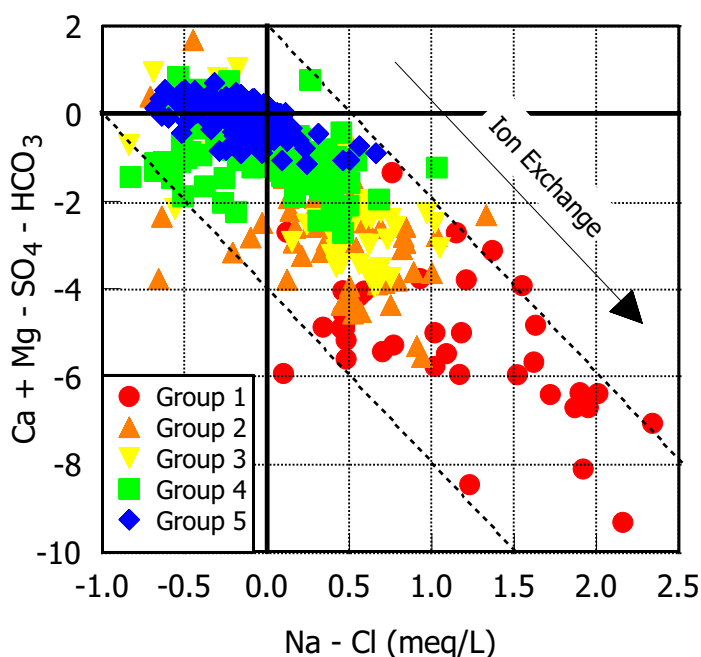


Figure 9.16 Cation exchange analysis for total groundwater samples.

Figure 9.16 is a specialised bivariate plot that is used to assess the relative influence of ion exchange processes on the major cations Ca^{2+} , Mg^{2+} and Na^+ (Fisher and Mullican, 1997). The predominant input of Na^+ to a shallow sandy aquifer such as this one is rainfall recharge, since reactive Na^+ -bearing minerals are absent in the aquifer matrix. Changes in the ratio of Na^+

to Cl^- are driven by ion exchange reactions, and are represented on the x-axis. The concentrations of SO_4^{2-} and HCO_3^- are subtracted from Ca^{2+} and Mg^{2+} to remove the contributions from calcite, dolomite and gypsum dissolution. If these elements are derived solely from mineral dissolution, then the water samples would plot close to zero on both axes; if ion exchange plays a dominant influence, then a best fit line to the data should have a slope of -1 . In figure 9.14 it is clear that the water samples form a negative linear trend, suggesting ion exchange is active in the aquifer system. However, this type of plot was developed to assess ion exchange in natural systems, where dissolution of carbonate minerals is the primary source of HCO_3^- in groundwater. The best-fit line has a slope of -2.8 ($r^2 = 0.65$), which results from the elevated input of HCO_3^- to the system from the redox processes associated with waste degradation in the landfill. Taking into account the effect of leachate impact on the slope of the linear trend line, this plot still provides evidence that ion exchange is occurring in the leachate-impacted groundwater, and a degree of reverse ion exchange (exchange of Ca^{2+} for Na^+) has occurred in the uncontaminated aquifer zone (positive y-values and negative x-values).

8.7.5 Environmental isotopes

During the July 2001 sampling event, additional groundwater, surface water and rainwater samples were collected for analysis of the stable environmental isotopes ^{18}O , ^2H , ^{13}C , and ^{15}N . These elements are important components of both water and landfill leachate constituents, including H_2O , HCO_3^- , CH_4 , NH_4^+ and NO_3^- . The results of the isotopic analyses are presented in Table 9.3, along with several additional parameters that are used in the interpretation of the isotopic results.

The various hydrogeochemical processes occurring both within the landfill and within the leachate plume in the aquifer result in a distinct groundwater isotopic signature. Analysis of the isotopic signatures of the leachate components can be used to identify the likely processes contributing to their generation (Games and Hayes, 1977; Baedeker and Back, 1979a; Hackley *et al.*, 1996; Clark and Fritz, 1997; Hornibrook *et al.*, 2000; Grossman *et al.*, 2002; van Breukelen *et al.*, 2003). The variation in isotopic signatures is analysed along groundwater flow path and along vertical transects through

the leachate plume to help constrain the primary geochemical processes active within the plume.

A very brief review of isotopic data assessment is provided. Isotopic data are expressed as a ratio of two isotopes of a single element (eg. $^{18}\text{O}/^{16}\text{O}$ or $^{13}\text{C}/^{12}\text{C}$). These isotopic ratios are in turn expressed relative to the isotopic ratio of an internationally accepted standard reference material, in order to provide a globally consistent method for analysis and interpretation of isotopic data. Isotopes of oxygen and hydrogen are assessed relative to what is called Vienna Standard Mean Ocean Water (VSMOW) (Hoefs, 1980) as originally identified by Craig (1961a); ^{13}C relative to a carbonate formation in the United States called the Pee Dee Belemnite (PDB) (Craig, 1953 and 1957); and ^{15}N relative to atmospheric nitrogen, which is reported to be constant (Junk and Svec, 1958; Clark and Fritz, 1997).

Table 9.3 Results of stable isotope analyses (including associated parameters used for interpretation of the results).

Sample ID	$\delta^{18}\text{O}$ ¹ (‰)	$\delta^2\text{H}$ ¹ (‰)	$\delta^{13}\text{C}$ ² (‰)	$\delta^{15}\text{N}$ ³ (‰)		EC ($\mu\text{S}/\text{cm}$)	Alkalinity as HCO_3^- (mM)	DIC (mM)	$\text{NH}_4\text{-N}$ (mM)	$\text{NO}_3\text{-N}$ (mM)
2	-5.13	-22.9	4.125	5.3	as NH_4^+	1853	18.037	35.01	2.767	0.214
3_1	-2.73	-10.0	-16.377	4.3	as NO_3^-	524	0.520	3.60	0.006	1.492
3_2.5	-3.34	-12.4	-15.401	9.6	as NO_3^-	458	2.220	5.48	0.011	0.585
3_3.5	-3.20	-10.6	-15.824	7.6	as NH_4^+	814	7.460	14.77	1.071	0.236
3_4.5	-5.01	-24.3	-3.438	8.7	as NH_4^+	1381	13.380	25.17	1.885	0.328
3_5.5	-4.97	-25.1	-1.960	1.5	as NH_4^+	707	6.500	15.53	0.885	0.207
3_8	-5.04	-26.9	-13.375	8.9	as NH_4^+	200	0.444	16.47	0.414	0.243
4_2.5	-5.28	-23.3	-19.680	--	--	365	1.160	6.59	0.011	0.564
4_4	-4.91	-24.3	-15.354	9.1	as NH_4^+	633	4.071	6.16	0.019	0.178
4_5	-5.03	-25.5	-9.580	3.2	as NH_4^+	489	3.720	11.10	0.207	0.157
4_6	-4.93	-26.9	-18.810	23.4	as NO_3^-	218	0.631	3.15	0.021	0.557
4_7	-4.83	-25.5	-19.021	39.5	as NO_3^-	247	0.987	3.12	0.003	0.535
4_8	-4.75	-24.6	-19.335	8.7	as NO_3^-	223	0.300	7.24	0.001	0.328
4a_2	-5.42	-24.1	-16.555	18.9	as NO_3^-	292	1.920	5.83	0.012	0.129
4a_3	-4.75	-22.0	-8.388	7.6	as NH_4^+	962	8.880	22.93	0.907	0.250
4a_4	-4.77	-21.0	-9.242	8.1	as NH_4^+	853	7.860	23.42	1.599	0.200

Table 9.3 Results of stable isotope analyses (including associated parameters used for interpretation of the results).

Sample ID	$\delta^{18}\text{O}$ ¹ (‰)	$\delta^2\text{H}$ ¹ (‰)	$\delta^{13}\text{C}$ ² (‰)	$\delta^{15}\text{N}$ ³ (‰)	EC ($\mu\text{S}/\text{cm}$)	Alkalinity as HCO_3^- (mM)	DIC (mM)	$\text{NH}_4\text{-N}$ (mM)	$\text{NO}_3\text{-N}$ (mM)
4a_5	-4.80	-23.4	-2.280	7.1 as NH_4^+	817	7.660	15.66	1.142	0.186
4a_6	-4.77	-24.4	-1.583	8.4 as NH_4^+	677	5.940	14.64	0.750	0.164
4a_7	-4.83	-24.5	-1.950	9.2 as NH_4^+	607	5.080	14.39	0.700	0.193
4a_8	-5.09	-25.7	-9.977	-- --	214	0.480	7.76	0.105	0.136
6_2.5	-4.09	-17.6	-18.748	4.0 as NO_3^-	164	0.140	7.48	0.004	0.471
6_4.5	-5.16	-25.4	-16.569	-- --	462	2.740	23.12	0.099	0.243
6_8	-4.34	-19.8	-17.618	8.3 as NH_4^+	440	2.800	26.70	0.346	0.157
6_11	-4.89	-25.0	0.056	8.4 as NH_4^+	752	5.802	29.53	1.871	0.136
6_12	-4.98	-24.5	--	8.8 as NH_4^+	753	5.920		3.513	0.129
6_13	-4.95	-25.5	3.271	8.2 as NH_4^+	639	4.780	26.32	1.713	0.143
6_15	-4.83	-24.7	0.891	8.8 as NH_4^+	572	4.160	30.67	3.598	0.186
6_18	-4.96	-25.1	-5.193	13.6 as NH_4^+	383	1.702	27.67	0.389	0.178
10_3	-5.41	-27.4	-15.804	-- --	1019	10.400	45.71	0.006	0.343
10_5	-5.27	-25.6	-4.689	8.7 as NH_4^+	1650	16.516	31.18	2.088	0.236
10_6	-5.04	-24.4	-5.634	7.9 as NH_4^+	1308	12.160	28.77	2.178	0.450
10_7	-5.06	-26.8	-3.017	8.0 as NH_4^+	295	1.520	15.34	0.421	0.364
10_9	-5.07	-27.0	-3.016	9.4 as NH_4^+	298	1.540	20.42	0.443	0.328
10_10	-5.04	-26.1	-5.247	8.1 as NH_4^+	285	1.680	16.10	0.568	0.278
10_11	-4.77	-24.3	-9.439	6.5 as NH_4^+	265	1.440	10.25	0.189	0.286
SW-3	-3.54	-15.9	-4.480	6.7 as NO_3^-	343	1.940	3.45	0.150	0.214
SW-3/6	-3.53	-16.0	-6.054	7.4 as NO_3^-	279	1.320	1.51	0.024	0.193
SW-6	-3.55	-15.3	-17.456	6.7 as NO_3^-	280	1.400	2.22	0.030	0.143
SW-drain	-4.06	-20.8	-10.956	10.3 as NH_4^+	409	2.640	3.57	0.203	0.200
Marsh-1	-4.98	-22.4	-15.925	-- --	612	4.240	6.54	0.019	0.100
Marsh-2	-4.52	-20.8	-12.474	-- --	1026	10.560	19.73	0.189	0.000
Marsh-3	-4.65	-21.9	-5.373	9.3 as NH_4^+	785	5.900	13.54	0.685	0.157
RW-1	-3.67	-11.8	--	-- --	68.4	0.000	--	0.046	0.121
RW-2	-4.29	-15.7	--	-- --	52.4	0.000	--	0.001	0.150

Notes:

1. $\delta^{18}\text{O}$ and $\delta^2\text{H}$ values relative to Vienna Standard Mean Ocean Water (VSMOW).
2. $\delta^{13}\text{C}$ values relative to the Pee Dee Belemnite (PDB).
3. $\delta^{15}\text{N}$ values relative to atmospheric nitrogen.

For ease of expression, the isotopic data from a water sample are expressed as the ratio of the isotopic ratio of the sample to the isotopic ratio of the international standard, according to the following equation (using carbon as an example):

$$\delta^{13}\text{C} (\text{‰}) = [({}^{13}\text{C}/{}^{12}\text{C} \text{ sample})/({}^{13}\text{C}/{}^{12}\text{C} \text{ standard})] * 1000 \quad (9.8)$$

9.5.1 Oxygen-18 and deuterium

Oxygen-18 and deuterium are the most widely used isotopes in hydrologic studies, and traditionally have been used to determine the source and flow paths of water in aquifers, as interactions with aquifer minerals results in an isotopic signature that is distinct from meteoric water. These isotopes are also commonly used to quantify the degree of mixing between two distinct end-member waters, and to determine whether a variety of hydrogeochemical reactions may be occurring along a groundwater flow path.

A plot of $\delta^{18}\text{O}$ and $\delta^2\text{H}$ data from around the world has defined what is known as the Global Meteoric Water Line (GMWL), which generally describes the global isotopic evolution of rainwater from equatorial ocean regions to inland polar regions (Rozanski *et al.*, 1993). The $\delta^{18}\text{O}$ and $\delta^2\text{H}$ data from groundwater are typically assessed relative to the meteoric water line, and systematic departures from this line are used to identify potential hydrogeochemical processes occurring the aquifer. As fractionation effects for $\delta^{18}\text{O}$ and $\delta^2\text{H}$ result in departures from the GMWL, it is preferable to reference a locally established MWL, known as a local meteoric water line (LMWL), to facilitate a more relevant assessment.

With regards to deuterium, previous research established that the fractionation is due to formation of microbial methane within the closed system of the landfill, where methanogens preferentially utilise the lighter hydrogen isotope with a resultant enrichment of the heavier isotope in the leachate water (Fritz *et al.*, 1976; Baedeker and Back, 1979b; Hackley *et al.*, 1996; Clark and Fritz, 1997). Deuterium enrichment has also been observed in association with biodegradation of petroleum hydrocarbon (e.g. Mancini *et al.*, 2002). Oxygen-18 may be affected by a similar process, with the

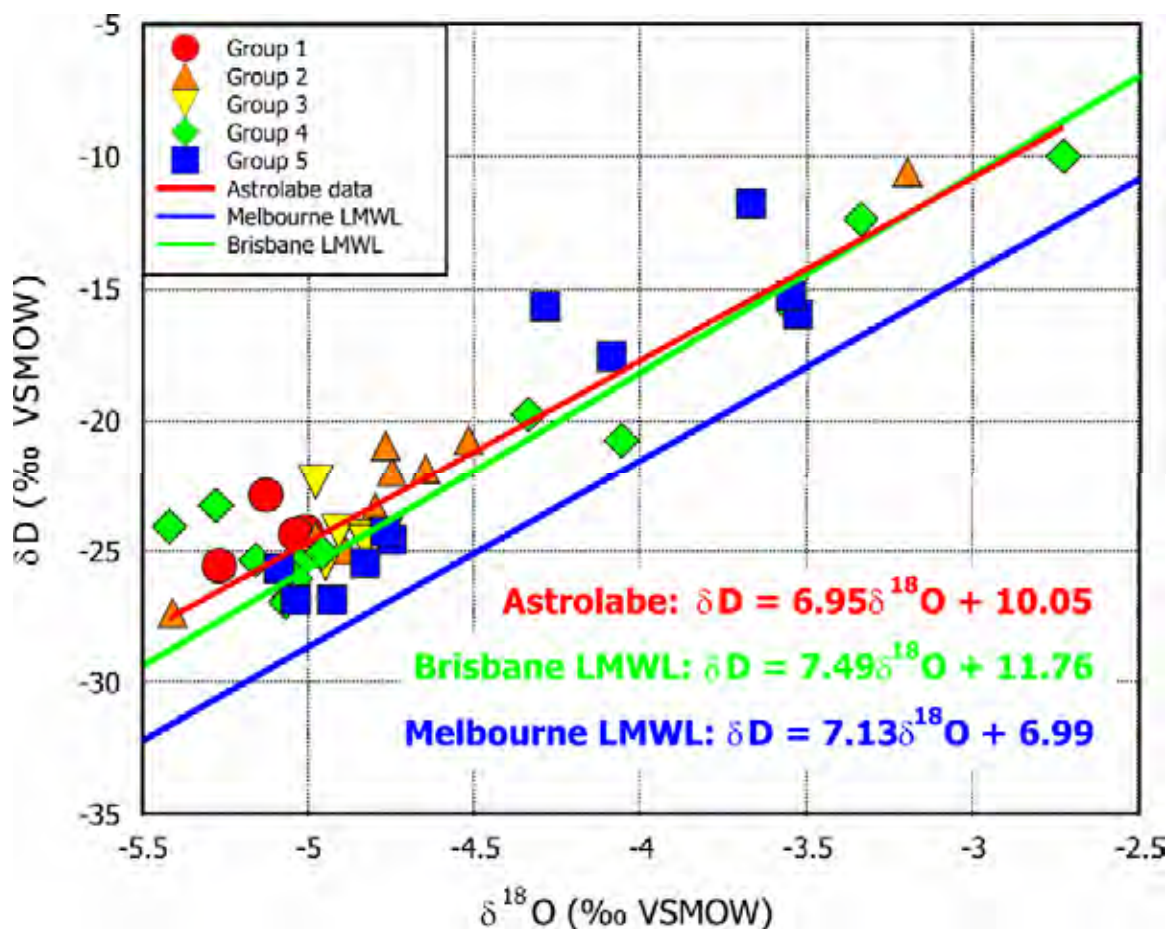


Figure 9.17 Bivariate plot of δD (‰ VSMOW) vs. $\delta^{18}O$ (‰ VSMOW) for groundwater, surface water and rainwater (Global meteoric water line [GMWL] provided for reference [Rozanski *et al.*, 1993]).

lighter oxygen isotopes being preferentially utilised in the formation of $CO_{2(aq)}$ during microbial oxidation of organic matter in the landfill.

A plot of $\delta^{18}O$ versus δ^2H (or δD) values from Astrolabe Park is presented in Figure 9.17. The Astrolabe $\delta^{18}O$ values range from -5.42 to -2.73 ‰, and the δD values range from -27.4 to -10.0 ‰. In addition, the LMWL for Melbourne and Brisbane have been included (insufficient publicly available data for Sydney) for comparison.

The data presented in Figure 9.17 do not suggest a significant departure from precipitation. As demonstrated in this figure, the data from Astrolabe Park define a trend line that is nearly identical to the LMWL for Brisbane. The groundwater samples from the centre of the plume are marginally more enriched in both δD and $\delta^{18}O$ than the samples from the plume fringe (approximately $+3$ ‰ and $+1$ ‰, respectively), a

characteristic that has been reported for other landfills (Fritz *et al.*, 1976; Baedeker and Back, 1979b; Hackley *et al.*, 1996). However, the degree of δD enrichment at Astrolabe Park is far less than that reported by Fritz *et al.* (1976) and Hackley *et al.* (1996), which is in the range of 30 to 60‰. Also, there is no clear distinction between where the various statistical groups plot, and in comparison with the LMWL from Brisbane the data do not present compelling evidence that either $\delta^{18}O$ or δD are being influenced by microbial processes in the leachate plume.

9.5.2 Carbon-13

Analysis of the stable carbon isotope (^{13}C) has found wide application in the research of both natural and contaminated environments to study the evolution of carbon phases in aquifer systems. Applications of carbon isotopic data include: determination of isotopic signatures for various carbon sources in natural environments (e.g., Whiticar *et al.*, 1986; Chapelle *et al.*, 1988; Hornibrook *et al.*, 2000), source assessment for CO_2 and CH_4 in aquifer systems (e.g. Barker and Fritz, 1981; Coleman *et al.*, 1988), and assessment of hydrogeochemical processes related to carbon-based contaminants such as petroleum hydrocarbons (e.g. Hunkeler *et al.*, 2001; Mancini *et al.*, 2002), organic solvents (e.g. Hunkeler *et al.*, 2002), landfill leachate (e.g., Games and Hayes, 1977; Baedeker and Back, 1979b; Hackley *et al.*, 1996; Desrocher and Lollar, 1998; Grossman *et al.*, 2002; van Breukelen *et al.*, 2003) and other organic waste streams (e.g. Herczeg *et al.*, 1991).

In uncontaminated aquifers, the two most significant sources of dissolved inorganic carbon (DIC) in groundwater are dissolution of carbonate minerals and oxidation of organic matter (either in the soil zone or in subsurface organic deposits). Each has a distinct $\delta^{13}C$ signature, with DIC produced from dissolution of carbonate minerals having a more enriched $\delta^{13}C$ value ($\sim 0\text{‰}$) and DIC produced from oxidation of organic matter having a more depleted $\delta^{13}C$ value ($\sim -25\text{‰}$) (Deines *et al.*, 1974; Baedeker and Back, 1979b; Hornibrook *et al.*, 2000). In strongly anaerobic environments, where the organic substrate is more abundant than the available oxidants, decomposition via fermentation and methane production occurs.

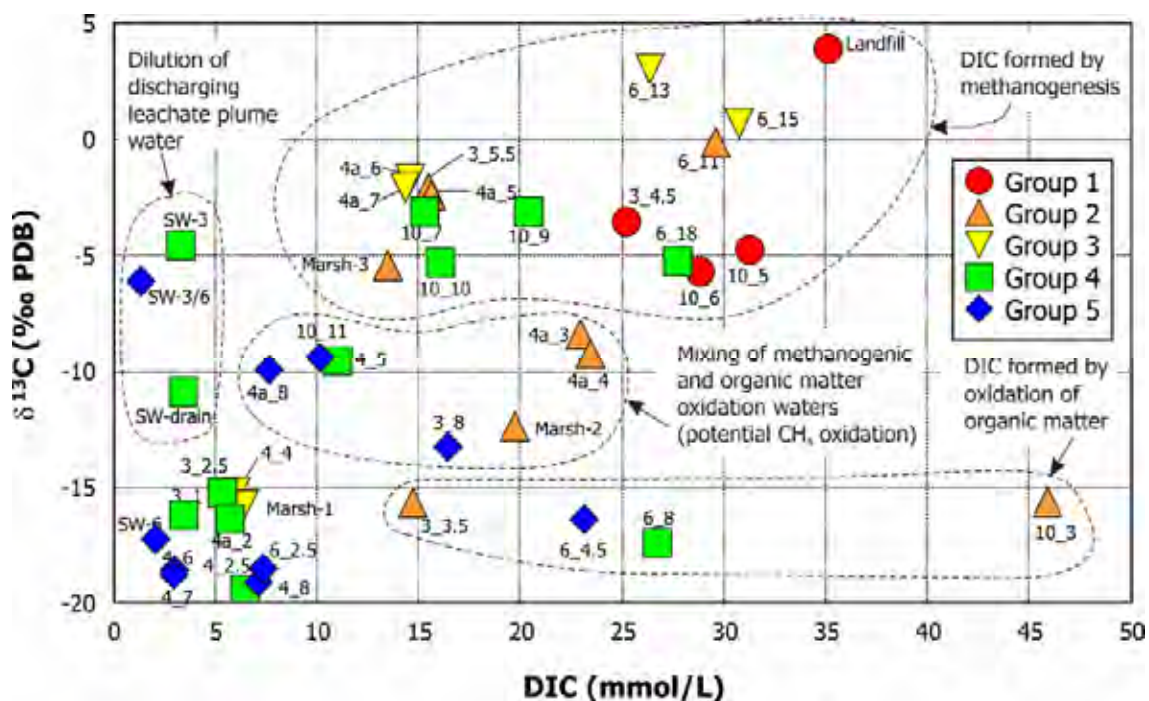


Figure 9.18 Dissolved inorganic carbon (DIC) vs $\delta^{13}\text{C}$ for groundwater, marsh and pond water samples, depicting potential DIC sources based on the $\delta^{13}\text{C}$ signature (the low DIC and $\delta^{13}\text{C}$ samples are essentially unaffected by leachate impact).

This is the case in most landfills, where the ratio of labile organic matter to available oxidants is very high (Hornibrook *et al.*, 2000). The methanogenic fermentation of organic matter in landfills generally results in production of CH_4 gas with a depleted $\delta^{13}\text{C}$ signature and strongly enriched DIC in pore water, as a result of voluminous CH_4 production in a closed system (Games and Hayes, 1977; Baedeker and Back, 1979b, Hackley *et al.*, 1996). Several researchers have also observed a switch to isotopically ‘heavier’ CH_4 and ‘lighter’ DIC down flow path from the contaminant source, which has been interpreted as resulting from oxidation of CH_4 (Hackley *et al.*, 1996; Grossman *et al.*, 2002; van Breukelen, 2003). The ^{12}C - CH_4 is preferentially oxidised resulting in progressive enrichment in ^{13}C - CH_4 and depletion in ^{13}C -DIC, and a resultant $\delta^{13}\text{C}$ -DIC value of -10‰ has been reported for this process.

The $\delta^{13}\text{C}$ values of the water samples range from -19.68‰ (sample 4_2.5) to +4.125‰ (landfill), with the plume centre samples generally the most enriched in ^{13}C and the lateral and vertical plume fringes the most depleted. The $\delta^{13}\text{C}$ value for the landfill sample is approximately 5-10‰ less enriched than those reported for other old, methanogenic phase landfills (e.g. Games and Hayes, 1977; Baedeker and Back, 1979b; Hackley *et al.*, 1996), and may represent either a small degree of mixing

between the leachate and inflowing groundwater that is permanently in contact with the waste, or a control on $\delta^{13}\text{C}$ composition via precipitation of carbonate minerals in the aquifer. The water samples with limited leachate impact are characterised by low DIC concentrations and $\delta^{13}\text{C}$ values between -20‰ and -15‰ ; this is similar to the 'background' $\delta^{13}\text{C}$ -DIC signature reported by other authors (e.g. Games and Hayes, 1977; Baedeker and Back, 1979b).

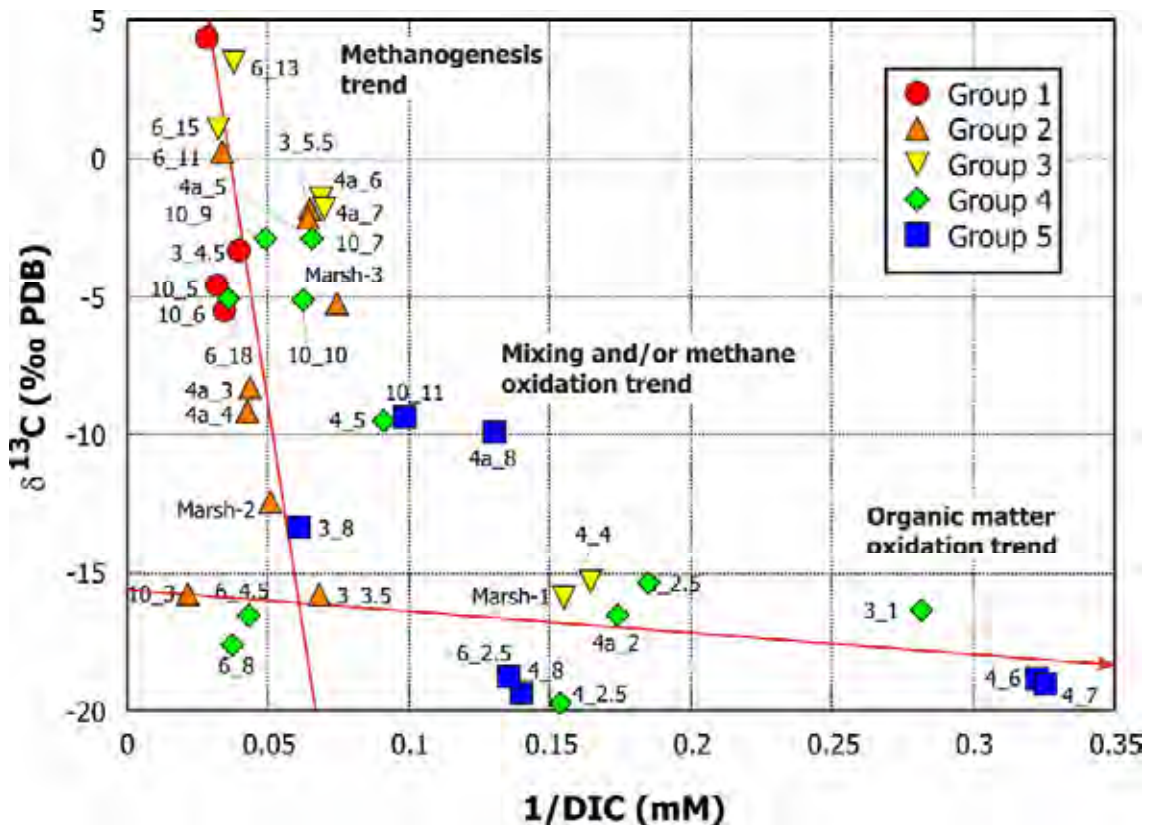


Figure 9.19 $\delta^{13}\text{C}$ of DIC vs the reciprocal of the DIC concentration. Note the steep regression line indicating ^{13}C enrichment associated with methanogenesis in the landfill, and the shallow regression line associated with oxidation of organic matter.

A plot of DIC concentration vs $\delta^{13}\text{C}$ values is used to assess the isotopic signature of the various water samples (Figure 9.18). Two major evolutionary trends are evident: (1) an increase in DIC concentration with concomitant enrichment in $\delta^{13}\text{C}$ values, and (2) an increase in DIC concentration with little or no enrichment in $\delta^{13}\text{C}$ values. The first case could be explained by DIC production associated either with methane production or, to a certain extent, dissolution of carbonate minerals. However, the relative influence of the latter would be negligible considering the minor DIC contribution from carbonate content in the aeolian sand aquifer (refer to Section 9.3.1 for discussion). The second

case indicates DIC production through oxidation of organic matter. A number of groundwater samples from the fringe of the plume plot in an intermediate position, with a $\delta^{13}\text{C}$ value of approximately -10‰ . This $\delta^{13}\text{C}$ value is likely associated with oxidation of methane along the plume fringe (Herczeg et al., 1991, Grossman et al., 2002). The $\delta^{13}\text{C}$ signature for samples 10_3, 6_4.5 and 6_8 are unusually depleted with regards to the DIC concentrations in these samples. These samples were collected from above the leachate plume, and the $\delta^{13}\text{C}$ values are similar to the samples collected from above the leachate plume at the other piezometers. This pattern suggests that the DIC in these samples is influenced by oxidation of dissolved organic carbon under relatively oxidising conditions, resulting in the depleted $\delta^{13}\text{C}$ signature.

A plot of $\delta^{13}\text{C}$ values vs the reciprocal of the DIC concentration is presented in Figure 9.20. According to Grossman et al. (1989), changes in isotopic concentration due to mixing of different DIC pools will result in $\delta^{13}\text{C}$ values that are linearly proportional to the inverse of the DIC values. This plot indicates a steep change in $\delta^{13}\text{C}$ values as a function of the $1/\text{DIC}$ value, which is similar to results reported by Grossman et al. (1989) and Herczeg et al. (1991). The two major carbon isotope evolutionary pathways are readily apparent in this plot, with the steep regression line indicative of methanogenesis, and the flatter regression line indicative of oxidation of organic matter. Note the slight depletion in $\delta^{13}\text{C}$ values in the oxidation of organic matter trend, which results from the preferential mineralisation of the lighter ^{12}C isotope by heterotrophs.

The copious production of methane in older, methanogenic phase landfills has been reported to produce enrichment in both the $\delta^{13}\text{C}$ and δD signature of the resultant DIC in groundwater (e.g. Baedeker and Back, 1979b; Hackley et al., 1996). To assess this process at this site, a plot of $\delta^{13}\text{C}$ and δD is presented in Figure 9.20. It was previously established that $\delta^{13}\text{C}$ enrichment is associated with greater leachate impact (Figures 9.18 and 9.19), but without the expected enrichment in δD values with respect to precipitation (Figure 9.17). Unusually, the δD signature of the leachate-impacted samples only varies within an approximately 5‰ range, and does not exhibit a large degree of enrichment in the more impacted samples.

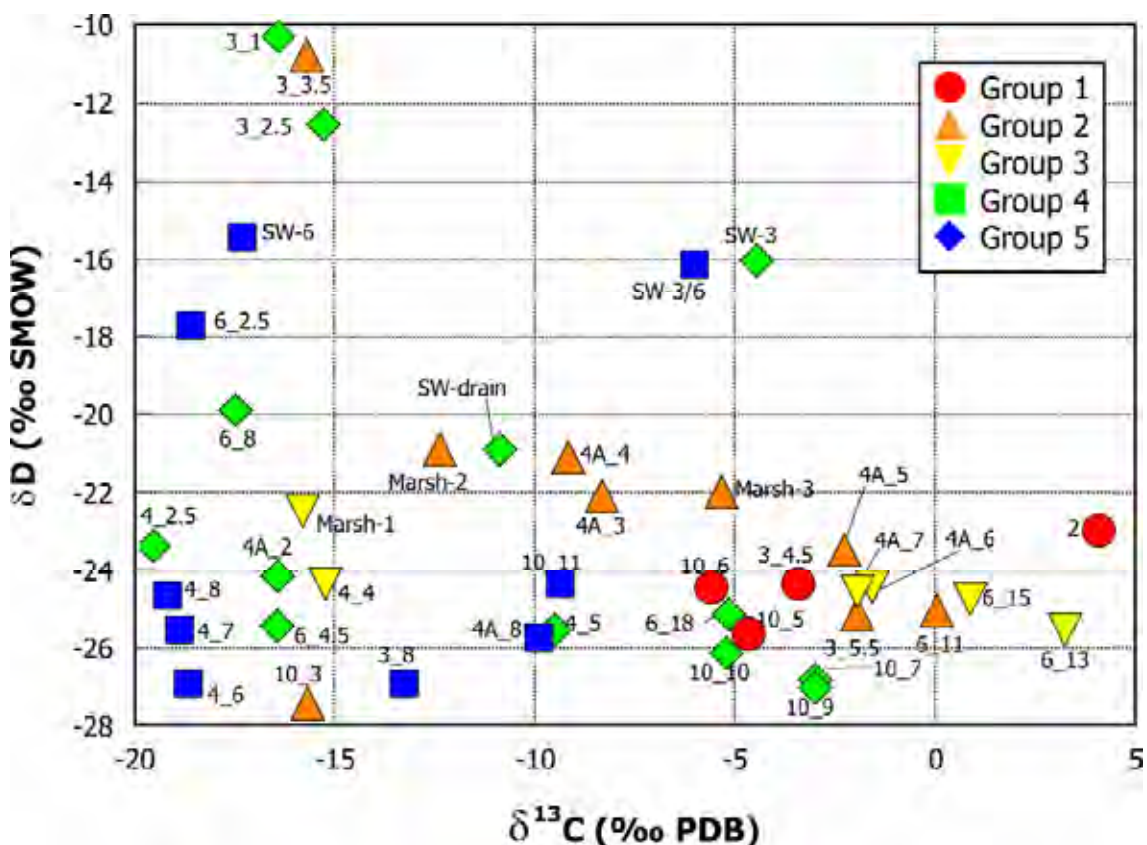


Figure 9.20 Plot of δD values vs $\delta^{13}C$ values for all water samples. Note the unusually depleted δD signature of the leachate-impacted water samples.

Enrichment in δD values of 30 to 60‰ was reported by Fritz et al. (1976), with similar results reported by Hackley et al. (1996). In fact, the greatest δD enrichment occurs in a handful of relatively uncontaminated groundwater samples from piezometers 3 and 6, and the marsh and pond water samples, which suggests evaporation as a more likely enrichment process. The relatively depleted δD signature of the leachate-impacted samples suggests that the rate of methanogenesis at Astrolabe landfill is less than that observed at other researched landfills, which is in agreement with the relatively dilute nature of the leachate plume.

In general, enrichment in $\delta^{13}C$ values is associated with increasing leachate impact in the water samples from the site. The degree of $\delta^{13}C$ enrichment associated with the landfill and plume centre samples suggest that the majority of DIC is being produced via methane fermentation within the landfill, which supports the results of the Deffeyes plot (Figure 8.15). Groundwater samples from the plume fringe with a $\delta^{13}C$ value of approximately -10‰ may indicate the occurrence of methane oxidation. The somewhat lower values for $\delta^{13}C$ at this landfill, as compared with other published studies, may

result from lower rate of methane production within the landfill with respect to published studies, methane oxidation at the plume fringe re-introducing depleted $\delta^{13}\text{C}$ to groundwater, or via precipitation of carbonate minerals in the carbonate-saturated environment of the leachate plume.

9.5.3 Nitrogen-15

The isotopes of nitrogen ($\delta^{14}\text{N}$ and $\delta^{15}\text{N}$) have received little attention relative to the other stable isotopes in terms of landfill leachate contamination studies. Nitrogen-15 data, in conjunction with $\delta^{18}\text{O}$ data, have primarily been used to identify sources of NO_3^- (e.g. Kreitler *et al.*, 1978; Williams *et al.*, 1998) in groundwater, or to elucidate the specific processes affecting NO_3^- concentrations (denitrification, assimilatory and dissimilatory nitrate reduction, abiotic nitrate reduction) (Clark and Fritz, 1997; Kendall and Aravena, 2000). Previous research has been heavily weighted towards contamination studies related to agricultural processes (e.g. Mengis *et al.*, 2001), fertilizer application (e.g. Iqbal *et al.*, 1997) and sewage leaks (e.g. Smith *et al.*, 1991a; Aravena *et al.*, 1993; Aravena and Robertson, 1998), with more recent work in petroleum hydrocarbon degradation (Schürmann *et al.*, 2003). To date, I am aware of only two publications (van Breukelen *et al.*, 2003; and North *et al.*, 2004) that make any mention of $\delta^{15}\text{N}$ in conjunction with a landfill leachate study.

The majority of nitrogen present in landfill leachate is the result of microbial fixation of organic nitrogen during decomposition of organic waste (Clark and Fritz, 1997). In the strongly reducing environment of the landfill, the fixation of organic nitrogen is reflected by elevated concentrations NH_4^+ . The NH_4^+ produced in the landfill will bear a similar ^{15}N signature to the waste from which it was derived (Létolle, 1980; Hübner, 1986; Kendall and Aravena, 2000), and any departures from this signature should indicate a secondary process effecting the NH_4^+ or NO_3^- in the plume.

The $\delta^{15}\text{N}$ values versus total nitrogen concentration are plotted in Figure 9.21. It appears that the NH_4^+ produced in the landfill has a distinct $\delta^{15}\text{N}$ signature, with an average value of $8 \pm 2\text{‰}$. While there is little published evidence indicating a characteristic $\delta^{15}\text{N}$ range associated with a landfill waste source, this value is within the range specified for

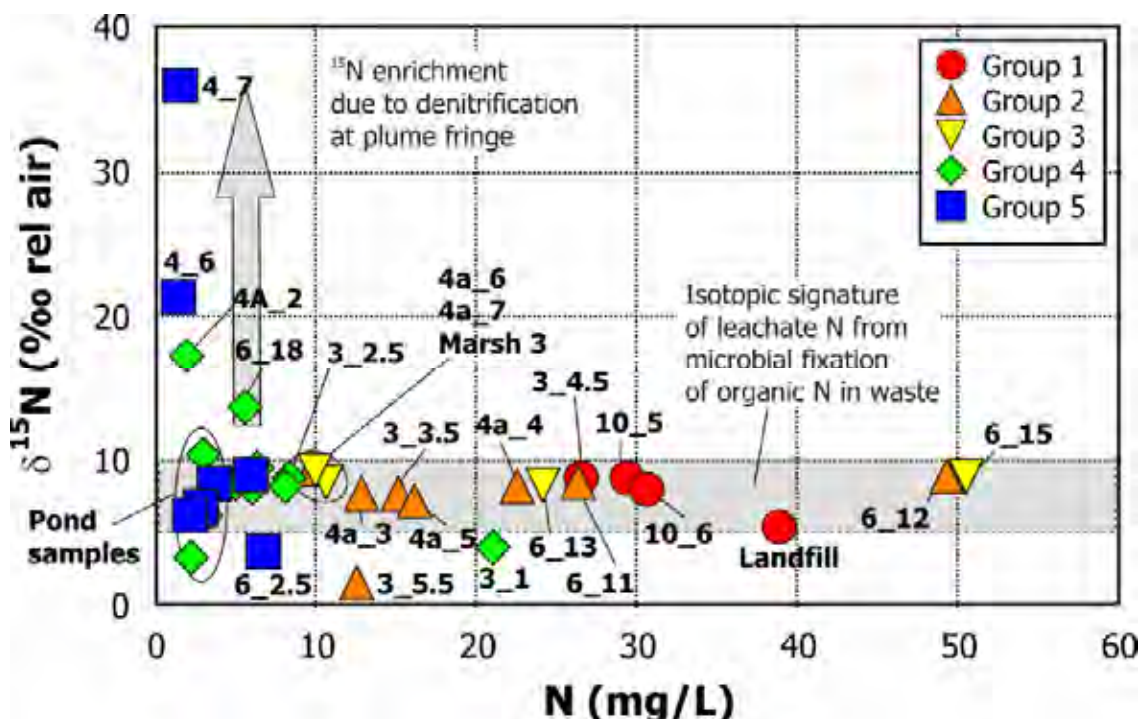


Figure 9.21 Relationship between $\delta^{15}\text{N}$ and total nitrogen concentration, with $\delta^{15}\text{N}$ enrichment due to denitrification indicated.

sewage and manure ($\leq 8\text{‰}$), and greater than the average ranges for organic soil N (2 to 6‰) and fertilizer ($\geq 2\text{‰}$) (Hübner, 1986; Aravena et al., 1993; Williams et al., 1998, Clark and Fritz, 1997; Aravena and Robertson, 1998). The total N concentration doesn't influence the $\delta^{15}\text{N}$ values, it appears to be the location of the sample that determines the potential for fractionation (Kendall and Aravena, 2000). In the strongly reducing core of the leachate plume (Group 1 and 2 samples), the NH_4^+ is relatively unaffected by microbial or abiotic transformations, thus very little isotopic fractionation is observed. In an aquifer setting NH_4^+ is most susceptible to adsorption and cation exchange, which result in a change in concentration but no fractionation effect. The most obvious departure from the $\delta^{15}\text{N}$ signature of the leachate is the enrichment in samples 4_6, 4_7, 4A_2 and 6_18. These samples are from the fringe of the leachate plume (lateral and vertical), and the $\delta^{15}\text{N}$ enrichment is most likely due to denitrification or anaerobic oxidation involving nitrate. Anaerobic oxidation via denitrification results in generation of isotopically light N_2 gas, leading to enrichment of $\delta^{15}\text{N}$ in the remaining dissolved NO_3^- pool. The maximum $\delta^{15}\text{N}$ fractionation factor of 21.5‰ (sample 4_7) is consistent with enrichment from denitrification as reported in other studies (Mariotti et al., 1988; Smith et al., 1991; Desimone and Howes, 1996). Denitrification via anaerobic oxidation

at these sample locations is further supported by the depleted $\delta^{13}\text{C}$ – DIC values of these samples (Table 9.3).

There is also a subtler trend of ^{15}N depletion associated with NO_3^- in the shallow samples from two piezometers. This would suggest an NO_3^- source from NH_4^+ nitrification along the upper margin of the leachate plume, in which isotopically ‘light’ NO_3^- is produced. However, it is difficult to determine whether fertilizer application on the golf course, which would introduce a relatively depleted ^{15}N source, might also be influencing the $\delta^{15}\text{N}$ – NO_3^- values in these shallow samples.

The ^{15}N data ultimately indicate that there is very little attenuation of leachate NH_4^+ due to microbial or abiotic transformations. It is likely that nitrification of NH_4^+ and subsequent denitrification is occurring at some locations along the fringe of the plume, as reflected by enriched $\delta^{15}\text{N}$ values. In other locations, the rate of nitrification may be outpacing the rate of denitrification, resulting in a depleted $\delta^{15}\text{N}$ signature. In general, the data suggest that N transformations may be active along the plume fringe, but do not comprise a significant NH_4^+ attenuation mechanism.

9.6 INVERSE HYDROGEOCHEMICAL MODELLING

Inverse modelling was performed to lend support to the hydrogeochemical interpretations in the previous sections, and to quantify the relative influence of the various hydrogeochemical processes along flow path between the landfill and the pond. For this exercise, piezometers 10 and 3 were used to define a typical groundwater flow path between the landfill and the pond, utilising the sample depth that corresponds with the centre of the plume. Models were developed for each of the major sampling events to assess the variations in active hydrogeochemical processes over relatively short time intervals, as interpreted by the model.

9.6.1 Model Selection

The inverse modelling function of the geochemical modelling code PHREEQCv2.6 (Parkhurst and Appelo, 1999) was used to develop the models. This modelling code

functions in much the same manner as the popular inverse modelling code NETPATH (Plummer *et al.*, 1994), but incorporates a significantly more user-friendly interface and offers the added functionality of being able to assign uncertainty limits to the chemical data to reduce the degree of trial-and-error required to establish a valid model. A comparison between modelling results for PHREEQC and NETPATH is provided as a worked example in the PHREEQC user manual (Parkhurst and Appelo, 1999). The mass balance calculations were based on the stoichiometric relationships as defined in the WATEQ4F database (Ball and Nordstrom, 1991) that is distributed with the PHREEQC software.

9.6.2 Mass balance modelling

Inverse geochemical modelling operates on the premise of mass balance. The assumption inherent in the model is that the water samples have an evolutionary relationship, such that the water at the downgradient sample location originated from the location of the upgradient sample. Assuming that the water samples are representative of a common groundwater flow path, the inverse geochemical model attempts to reconcile the differences in the chemical compositions of the samples by solving a set of linear equations to account for the mol balance of individual elements. Factors influencing the concentrations of elements in solution may include dissolution and precipitation of minerals, mixing of distinct water types, dilution and evaporation, ion exchange and redox processes; all of these can be assessed using the mass balance approach of an inverse geochemical model.

Inverse geochemical originally found application to assess geochemical reactions controlling water quality in natural water systems (eg. Garrels and Mackenzie, 1967; Plummer *et al.*, 1990; Parkhurst and Plummer, 1993). With regards to groundwater, the initial focus was on the evolution of groundwater chemistry along regional-scale flow paths resulting from water-rock interactions. More recently, inverse geochemical modelling has been used to elucidate processes occurring in contaminated groundwater environments, with recent examples including application to hydrocarbon plumes (Essaid *et al.*, 2003) and a single published example, that the author is aware of, for application to a landfill leachate plume (van Breukelen *et al.*, 2003).

While inverse geochemical models comprise a powerful tool for analysing the evolution of groundwater along a flow path, there is a significant burden of responsibility for the modeller to have a strong fundamental understanding of the various components present in the aquifer system in order to properly constrain the model. Further, the results of inverse modelling provide an overall accounting of mole balance changes between the evolutionary water samples, but do not explicitly identify all of the variations in solute concentrations or distribution occurring between the sample locations. It is important to remember that the water samples used in an inverse model represent a snapshot in time of the solute concentrations at two specific locations along a flow path, and the model simply accounts for the bulk total differences between these samples. However, reliance on inverse modelling for a solute plume that exhibits a large degree of variation in relatively short distances along a flow path may result in an erroneous interpretation of processes occurring in the aquifer, and may not be appropriate for use in this environment.

9.6.2.1 *Phases and constraints*

Inverse geochemical models are constructed through the definition of phases and constraints that are present in the within a modelled system. Constraints are the fundamental chemical elements that are significant to the aquifer system, and phases comprise the various minerals and reactions that influence the distribution of the constraints. While NETPATH requires manual selection of constraints for model development, PHREEQC references the input groundwater chemistry data to determine the relevant constraints.

The phases were chosen based on the aquifer mineralogy and the previous hydrogeochemical assessment of the bulk groundwater data, and were refined based on the results of the initial modelling trials to attempt to limit the total number of valid models that were generated. The phases included precipitation/dissolution of calcite, FeS (ppt), Goethite (dissolution only), siderite, and pyrite; ion exchange of Ca^{2+} , Mg^{2+} , Na^+ , K^+ and NH_4^+ ; and exchange of $\text{CO}_{2(\text{aq})}$. In addition, $\delta^{13}\text{C}$ constraints were placed on the calcite, siderite and $\text{CO}_{2(\text{aq})}$. In addition, the decision was made to adopt the field measured Eh/pe to define the redox environment at each end-member water sample, as

equilibrium with respect to a single redox couple in a contaminated environment is rarely, if ever, observed, and the mixed potential measured by the Eh probe was considered to be as representative as arbitrarily assigning a redox couple to define the redox environment. These phases were considered sufficient to address the major ion chemistry of the aquifer, including redox sensitive species, and to realistically represent the processes active within the leachate impacted groundwater.

The initial set of phases was established using the data from piezometers 10 and 3 (centre of plume) from the July 2001 sampling event. The data were selected for the initial model development because of the isotope data collected during this event, which can be used to further constrain the model and refine the phase selection. Once a valid model was developed using this data, the same phases were applied to the data from other sampling events to assess the inverse model interpretation of the observed variation in the plume. The results of the modelling effort are discussed in the following section.

9.6.3 Summary of modelling results

The inverse models for each sampling event were refined until the most feasible results within the lowest uncertainty limits were achieved. The model input data were taken from the most concentrated water samples at the centre of the plume at piezometers 10 and 3 during each sampling event. Several phases were selected based on their previously determined saturation indices, including calcite, siderite, FeS (ppt) and pyrite (the carbonate and sulfide minerals are considered to be important sinks for leachate constituents). Based on the elevated iron concentrations in groundwater, goethite was included to represent a pool of reactive iron oxyhydroxide along the flow path. Finally, the ion exchange functions for Ca^{2+} , Mg^{2+} , Na^+ , K^+ and NH_4^+ were included as possible element sources and sinks along flow path. While exchange of Mg^{2+} and K^+ are relatively minor processes, they were included because PHREEQC requires that every constraint is accounted for in at least one phase. Water was also selected as a phase to simulate solute concentration or dilution along the flow path, which is expressed as 'relative concentration' in Table 8.9.

To provide further constraint to the model runs, a maximum element variation tolerance of $\pm 5\%$ was assigned. In addition, a requirement for chloride balance ($\pm 5\%$) and pH balance (± 0.1 pH units) was incorporated as a pre-condition of model validation.

The modelling results listed in Table 9.4, and presented graphically in Figure 9.22, are presented as molar mass balance of mineral and exchange phases along the flow path (a positive value means the phase entered groundwater, a negative value means it was removed from groundwater), relative concentration (or water mass balance), and electron balance to quantify the relative influence of various redox processes (expressed as moles of relevant mineral phase involved in electron transfer [redox] processes).

Table 9.4 Summary of inverse geochemical modelling results along a central flow path between piezometers 10 and 3 for each major sampling event.

Phase	Jul-00	Dec-00	Mar-01	May-01	Jul-01
<u>Phase mass balance (mmol/L)</u>					
Calcite	-1.903	-0.467	3.771	-1.640	-1.496
Siderite	--	-1.367	-1.542	-10.140	-4.568
Goethite	--	--	--	6.858	3.156
FeS (ppt)	0.106	3.858	3.241	6.575	3.099
Pyrite	-0.061	-2.025	-1.600	-3.271	-1.546
<u>Exchange mass balance (mmol/L)</u>					
CaX2	1.680	0.661	-0.880	1.336	0.792
MgX2	--	-0.486	--	-0.455	-0.610
NaX	-1.074	--	--	-0.688	--
KX	-0.545	-0.351	0.244	--	-0.103
NH4X	-1.742	--	-1.058	-1.075	-0.263
Rel. conc.	+17%	no change	+21%	+31%	+0.1%
<u>Electron balance (redox)</u>					
Fe(3)	--	--	--	6.858	3.156
N(-3)	--	-0.315	-0.440	0.006	3.39E-07
S(-2)	-0.015	-0.192	-0.040	0.033	0.081

-- = Phase not included in the final model.

Rel. conc. = Relative concentration of groundwater along flow path.

The results suggest a large degree of variation in the relative magnitude of the phases during each sampling event. In general the precipitation of calcite, siderite and pyrite represents significant sink for Ca^{2+} , Fe^{2+} , HCO_3^- and S^{2-} along flow path, while the goethite and FeS (ppt) were identified as sources of elements to groundwater along flow path. The identical phase and electron balances for goethite indicate that this phase was introduced to groundwater via reductive dissolution along flow path. The increase in goethite reduction is in turn matched by an increase in siderite precipitation. Both

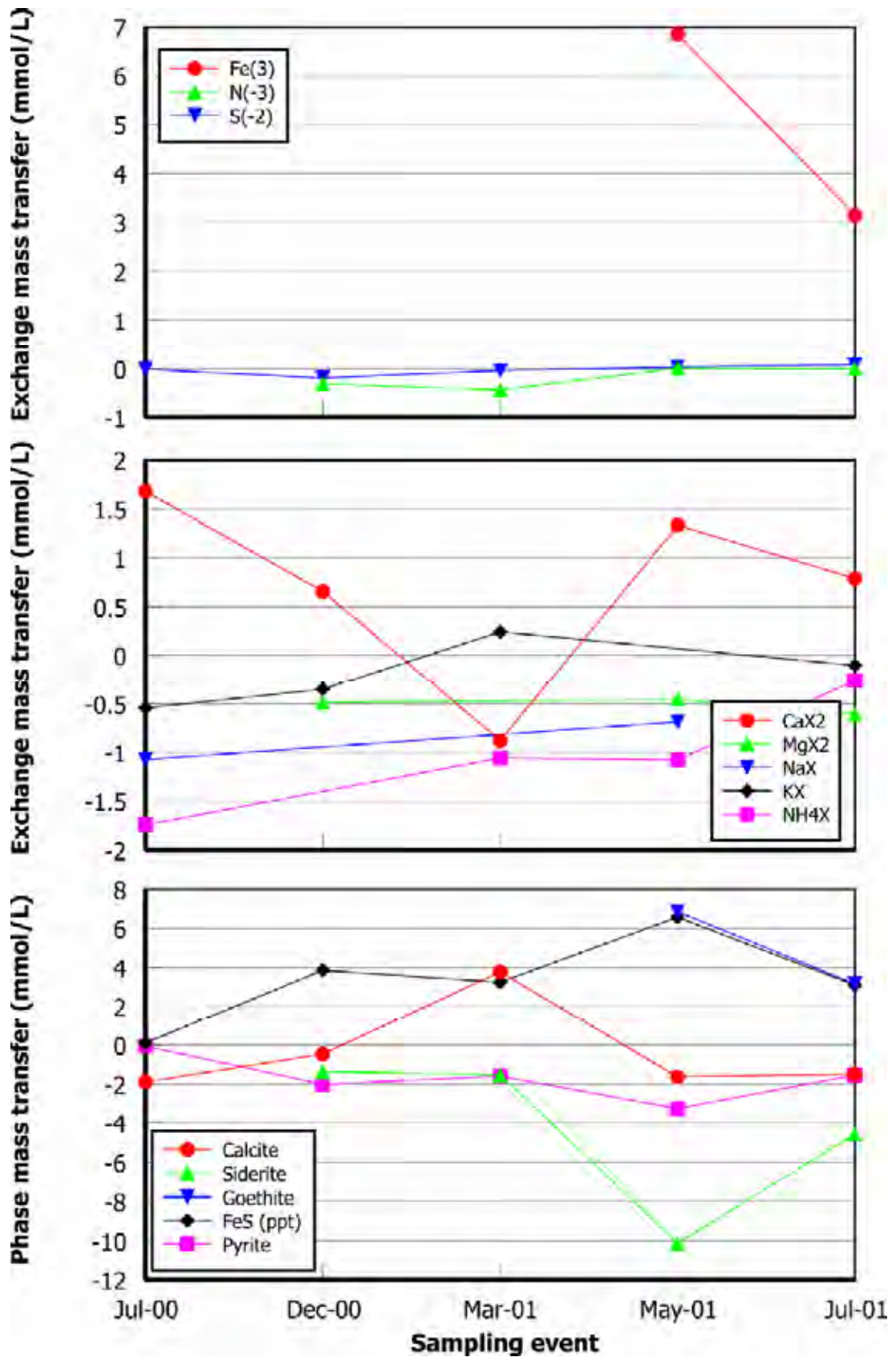


Figure 9.22 Results of inverse geochemical modelling using PHREEQCv2.6, displaying the phase mass balance, exchange mass balance and electron mass balance for each sampling event (missing data points indicate that the given phase was not included in the modelling results for the sampling event).

calcite and pyrite precipitation are considered element sinks during each sampling event. Calcite precipitation and dissolution appears to fluctuate in response to Ca^{2+} exchange, which influences the degree of calcite saturation in groundwater.

The transfer of elements to and from exchange sites is considered a significant process for both Ca^{2+} and NH_4^+ , while a lesser effect is exhibited for Mg^{2+} , Na^+ and K^+ . The influx of Ca^{2+} from exchange sites appears to increase the calcite saturation in groundwater, reflected by a good correlation in calcite precipitation. Both NH_4^+ and S^{2-} appear to have very little influence over the electron balance in all sampling events, with iron providing the dominant influence when it is included.

The inverse modelling results from PHREEQC were also checked against a simulation run with NETPATH to assess whether similar results were obtained. The NETPATH model was developed by attempting to match as closely as possible the phases and constraints used in the PHREEQC model runs. Whilst similar patterns of dissolution, precipitation and exchange were observed for similar phases, subtle variations were evident in the mass balance ratios for most phases. The departures in total flux concentrations may stem from slight differences in the data input procedures, calculations, and databases for the two programs, but the general consensus with the primary phases suggested that the PHREEQC results were valid.

The results of the water balance component indicate that the relative solute concentration in groundwater increases along flow path during July 2000, March and May 2001, and is stable along flow path during December 2000 and July 2001 (Table 8.9). This is a fairly unusual result, since solute concentrations in a contaminant plume are generally expected to become more dilute along flow path.

The sets of modelling results would appear to be plausible for individual sampling events, but are suspect when considered together. Why, for example, would iron reduction only occur along flow path during the final two sampling events, and what is causing such a dramatic increase in siderite precipitation across relatively short time intervals? What causes the relative solute concentrations in groundwater to switch

between becoming more concentrated and more dilute along flow path within a couple of months?

The distribution of the plume as determined using surface electrical methods (Figure 6.12) might offer an explanation to the unusual inverse modelling results. The electrical image of the plume reveals an unexpected segmentation along flow path, with pulses of concentrated leachate separated by relatively more dilute leachate zones. It was postulated in Chapter 8 that leachate mobilisation occurs during periods of high rainfall, where the increased infiltration and groundwater mounding above the waste layers forces concentrated leachate out of the waste and into the aquifer. During dry spells, a lesser baseline volume of leachate is introduced to the aquifer where waste is buried below the water table.

The electrical image of the plume in Figure 6.12 appears to indicate that the pulses of concentrated leachate migrate in a relatively conservative manner between the landfill and the pond, retaining a distinct 'segmented' distribution throughout the flow path. If this is the case, then even detailed vertical sampling along a flow path will do little to reveal the degree of plume heterogeneity between the two piezometer locations. These sampling events comprise snapshots of the vertical plume distribution at two specific locations, and the relative change in solute concentrations between the piezometers may simply be a function of the distribution of the concentrated leachate slugs relative to the piezometers during the sampling event. There may be multiple peaks and troughs in the concentrations of leachate constituents between the piezometers at any given time, but the inverse modelling will interpret the data as variations in active hydrogeochemical processes rather than the relative positions of concentrated leachate pulses migrating in a conservative manner between the landfill and the pond. The movement of concentrated leachate pulses through the aquifer also helps to explain the observed temporal variations in the vertical extent of the plume, the fluctuations of the centre of mass of the plume, and the temporal fluctuations in the relative concentration of the plume at the upgradient and downgradient sampling locations.

CHAPTER 10: CONCLUSIONS AND RECOMMENDATIONS

10.1 CONCLUSIONS

A detailed groundwater investigation was conducted to provide an assessment of the dynamics of leachate transport between an old, unlined landfill and a shallow groundwater-fed pond located less than 100 m down hydraulic gradient from the landfill. The investigation incorporated a network of bundled piezometers to provide detailed vertical profiles of the leachate plume chemistry and hydraulic head utilising a newly developed manometer board. In contrast to all other significant published landfill investigations to date (eg. Kimmel and Braids, 1974 and 1980; Zenone *et al.*, 1975; Baedecker and Back, 1979a,b; MacFarlane *et al.*, 1983; Baedecker and Apgar, 1984; Arneth *et al.*, 1989; Kehew and Passero, 1990; Lyngkilde and Christensen, 1992a,b; Mireki and Parks, 1994; van der Kamp *et al.*, 1994; Basberg *et al.*, 1998; Kjeldsen *et al.*, 1998a,b; Cozzarelli *et al.*, 2000; van Breukelen *et al.*, 2003 and van Breukelen and Griffioen, 2004), groundwater data were collected across short time intervals to provide an assessment of the degree of temporal variation in the plume distribution. Surface and borehole geophysical techniques were also employed to aid in the delineation of the sedimentary features of the aquifer and to provide a useful cross-sectional depiction of the distribution of the leachate plume between the landfill and the pond.

A conceptual site model of the plume occurrence and behaviour is included as Figure 10.1. The initial results of the investigation indicated that the leachate plume downgradient from Astrolabe Park was constrained to the upper approximately 8 to 10 m of the aquifer by a series of locally continuous lenses of low permeability material. These lenses effectively isolate the shallow, unconfined portion of the aquifer from the deeper, semi-confined portion of the aquifer. An analysis of vertical head distribution using the manometer board provided strong evidence to suggest that a strong upwards vertical hydraulic gradient was causing the leachate plume to discharge into Lachlan Pond No.5. This assertion was further supported by the chemical signature of water samples collected along the shore of the pond and from a series of temporary shallow

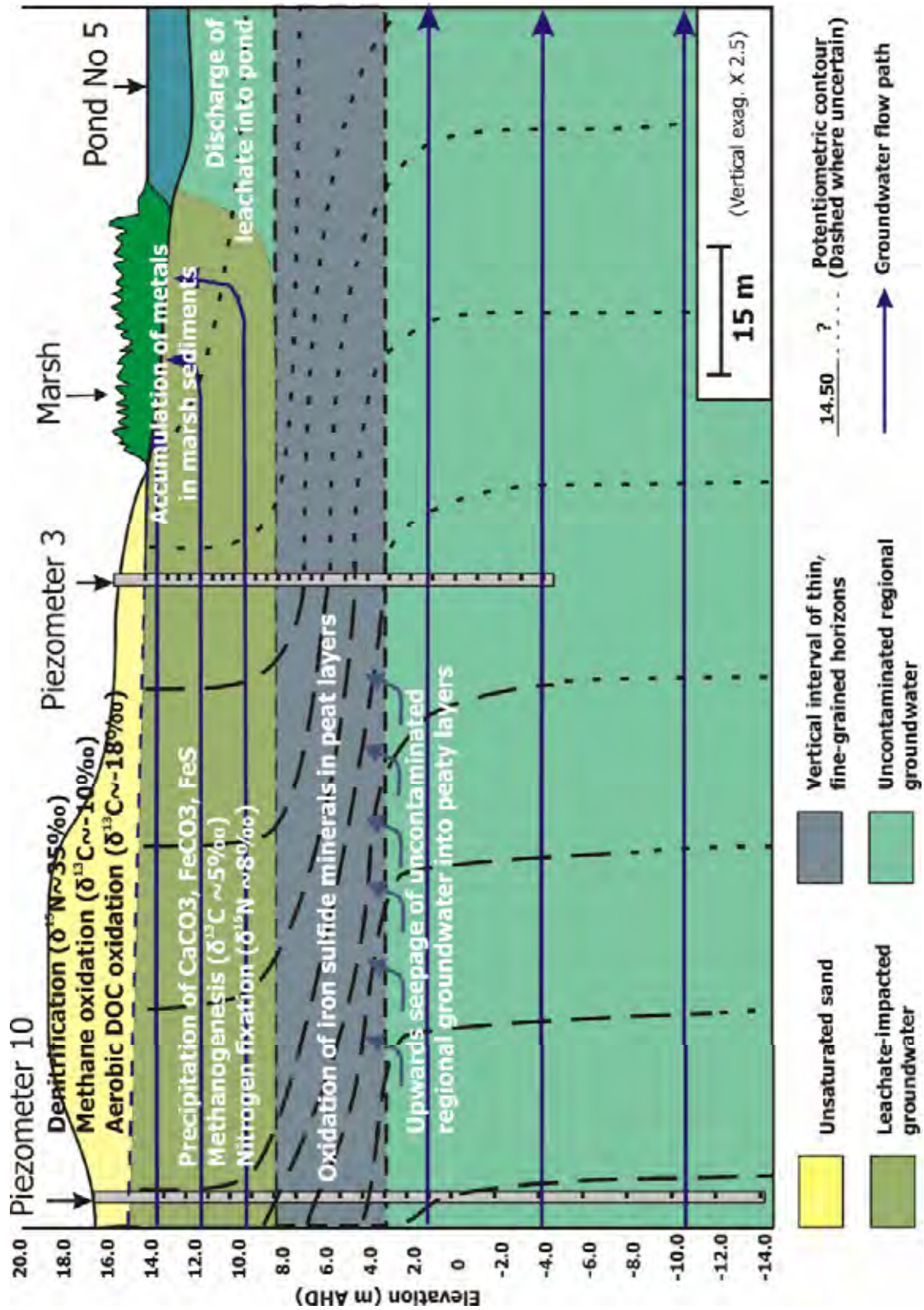


Figure 10.1 Conceptual site model of contaminant hydrology incorporating the results of various investigation techniques (note that plume fragmentation suggested by electrical resistivity transects not indicated).

piezometers installed in the marsh area, which was similar to the chemical signature of leachate-impacted groundwater.

Surface electrical resistivity imaging along transects parallel and perpendicular to groundwater flow provided significant insight into the nature of contaminant transport within the leachate plume. This process has seen limited published application to the definition of plume occurrence and geometry at landfills, despite the fact that old landfills were commonly sited in sand mines and quarries in contact with relatively fresh groundwater, in which the leachate comprises a significant electrical anomaly. The image perpendicular to groundwater flow indicated a relatively uniform plume thickness and distribution, while the image parallel to groundwater flow exhibited a segmented plume characterised by discrete ‘pulses’ of concentrated leachate separated by zones of relatively dilute leachate. These pulses appeared to retain their form along the short flow path between the landfill and the pond, suggesting a contaminant transport regime that is driven by mobilisation of distinct pulses of leachate, most likely flushed from the waste layers following extended periods of rainfall and the subsequent formation of a groundwater mound over the landfill. This conceptual model contrasts with those included in the intensely researched landfills in Denmark (eg. Lyngkilde and Christensen, 1992a,b; Bjerg et al., 1995; Kjeldsen *et al.*, 1998a,b) in which a progressive series of evolutionary redox zones were described along flow path away from the landfills.

An assessment of the groundwater chemical data collected over short time intervals from the bundled piezometers indicated that a significant degree of variation was occurring in the plume over space and time. The vertical extent of the leachate plume was observed to fluctuate by more than a metre within a month, as well as the centre of mass of the plume. The concentration of the plume centre was observed to vary by 100% between consecutive monitoring events. The vertical chemical profiles through the plume were invariably characterised by steep chemical gradients along the upper and lower plume margins, with a rapid depletion of electron acceptors (NO_3^- , SO_4^{2-}) and a concomitant increase in reduced elements within the plume (Mn^{2+} , Fe^{2+} , NH_4^+). Redox processes appeared to be most significant along the plume margin, with the plume centre remaining strongly reducing along the entire flow path between the landfill and

the pond. These results are similar to the findings of van Breukelen et al. (2003), and van Breukelen and Griffioen (2004).

Statistical analysis software (SPSSv10) was used to perform a hierarchical cluster analysis resulting in the classification of five water types based on the relative concentrations of major, trace and redox sensitive elements. The water chemical data were plotted according to these water types to determine the primary hydrogeochemical processes associated with the leachate contamination. The geochemical modelling code PHREEQCv2.6 was used to perform speciation and saturation index calculations for the water samples. Based on these calculations, it was clear that increasing leachate impact to groundwater was associated with saturation or supersaturation of several carbonate (calcite, siderite) and sulfide ($\text{FeS}_{(\text{ppt})}$, mackinawite, pyrite) mineral phases, which has been observed in previous research (eg. Kehew and Passero, 1990; Manning, 2001; Jensen et al, 2002; van Breukelen, 2003). Bivariate plots of the groundwater data indicated that a significant component of the inorganic solute load in the landfill leachate was derived from dissolution of waste material in the landfill, and that ion exchange along flow path constitutes a significant control over the chemical composition of the leachate plume. A considerable degree of pH buffering between H_2CO_3 and HCO_3^- was observed within the plume, with pH values consistently averaging 6.4 in the centre of the plume.

The stable environmental isotopes $\delta^{18}\text{O}$, $\delta^2\text{H}$, $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ were analysed during the July/August monitoring event to assist with identification of the processes controlling the plume composition. Unlike previous research (Fritz et al., 1976; Hackley et al., 1996), very little, if any, enrichment of deuterium was observed within the plume, which is typically a characteristic of fractionation associated with methane production. However, significant fractionation of $\delta^{13}\text{C}$ was observed, with two distinct process trends suggested by the data: elevated DIC coupled with $\delta^{13}\text{C}$ enrichment of up to ~20‰ due to methanogenesis, and elevated DIC with no enrichment of $\delta^{13}\text{C}$ interpreted to be associated with oxidation of organic matter (Figure 10.1). The latter process was characteristic of samples collected above the upper plume fringe, where the presence of oxidants would have precluded the establishment of methanogenic conditions. A distinct $\delta^{15}\text{N}$ signature was observed for leachate-impacted groundwater, with an

average value of $8 \pm 2\%$. In addition, enrichment of $\delta^{15}\text{N}$ was observed in plume fringe samples, which was interpreted to be due to oxidation of NH_4^+ and subsequent denitrification (Figure 10.1). To the author's knowledge, this represents one of only three applications of $\delta^{15}\text{N}$ to a landfill environment.

PHREEQCv2.6 was also used to perform inverse geochemical modelling for each major sampling event between piezometers 10 and 3, which roughly define an evolutionary groundwater flow path between Astrolabe Park and Lachlan Pond No.5. Groundwater data from the July 2001 sampling event, which included stable isotope data, were used to define the set of plausible phases that best described the changes observed between the two piezometers. These phases included precipitation and/or dissolution of calcite, siderite, $\text{FeS}_{(\text{ppt})}$ and pyrite, reductive dissolution of goethite, and ion exchange of Ca^{2+} , Mg^{2+} , Na^+ , K^+ and NH_4^+ , as well as a water component representing dilution/concentration along flow path. These phases were then applied to the datasets for the remaining sampling events. The predominant processes identified included ion exchange (especially with regards to Ca^{2+} and NH_4^+) and precipitation of calcite and siderite (similar to the findings from van Breukelen et al., 2003). The modelling results indicated a significant degree of variation in active hydrogeochemical processes within the plume between sampling events. Some processes disappeared entirely during certain sampling events, while others varied dramatically in magnitude. Considered individually, the modelling results would appear plausible for the given sampling event. However, considering all of the sampling events as a whole, the changes in hydrogeochemical processes between subsequent sampling events suggested by the modelling results appear unlikely.

An alternative explanation to the modelling results for the observed spatial and temporal variation in the leachate plume is the notion of pulses of concentrated leachate migrating in a conservative manner between the landfill and the pond. While certain hydrogeochemical processes are undoubtedly influencing the transport of contaminants in the plume, it is important to remember that the aquifer matrix is almost exclusively comprised of quartz sand with a very minor silt/clay fraction (except in discrete lenses in the sandy profile). As such, this aquifer constitutes a relatively non-reactive medium for the transport of leachate between the landfill and the pond. The migration of pulses

of concentrated leachate through the aquifer would help to explain the observed variation in the vertical extent of the plume, the vertical location of the plume centre, the relative concentration of the plume centre, and the observed switching between leachate concentration and dilution along flow path.

This fundamental difference in the interpretation of contaminant transport mechanisms between the landfill and the pond has significant implications for the interpretation of even detailed groundwater monitoring data. Performing inverse modelling of the data from any of these individual sampling events would indicate a significantly different set of active hydrogeochemical processes in the aquifer, none of which suggests the possibility of relatively conservative migration of leachate pulses between the landfill and the pond. These individual modelling results could potentially provide a basis for erroneous management decisions for this (or similar) groundwater contamination scenarios, particularly with the increasingly popular use of monitored natural attenuation as a remedial option, which requires a thorough understanding of the hydrogeochemical processes active within the aquifer system. It is my hope that this evaluation of the degree of site-scale spatial, and especially temporal, variation present in a contaminant plume will underscore the efficacy of an integrated approach to contamination investigations to provide a more robust understanding of the plume dynamics and behaviour over time.

10.2 RECOMMENDATIONS

Whilst this research provided significant insight into the unexpected nature of contaminant transport at this site, it would benefit from additional investigation into a number of issues. An assessment of even shorter timescales of plume variations could be undertaken by instrumenting one or two piezometers with dataloggers incorporating pressure transducers and EC probes to determine exactly how rapidly conditions change within the plume, particularly in response to storm events. It would also be helpful to collect a series of electrical resistivity images at the same location along the groundwater flow path at set time intervals to see if it is possible to physically track the movement of the pulses of leachate through the aquifer, and to observe any changes to the shape and relative intensity of the pulses along flow path. Had the efficacy of this

technique been realised earlier in the field program, it would certainly have been repeated in conjunction with every monitoring event conducted.

Further, it would have been very useful to have a bundled piezometer installed directly within the heart of the landfill help provide a better assessment of the mobilisation mechanisms of landfill leachate, particularly the vertical head distribution following periods of extended rainfall.

With regards to the future management of the leachate contamination, three recommendations are provided. The first would be to analyse a set of groundwater samples for organic contaminants, which is something that has received scant attention at this site and may comprise a significant issue with regards to assigning remediation objectives. The second would be to conduct an investigation into the potential generation and transport of landfill gas in the vicinity of the landfill, particularly with residential neighbourhoods developed right up to the landfill boundary. Finally, it would be worthwhile conducting an ecological assessment of the flora and fauna in the aquatic ecosystem of the pond that comprises the receiving environment for the discharging leachate plume, to assess how tolerant this ecosystem is to the leachate plume constituents. This would also provide valuable information for the future management of similar contamination issues along different stretches of the Lachlan Pond chain, and could be used to develop site-specific regulatory assessment criteria for the various leachate components.

Whilst this plume has now been characterised to minute detail, it now provides an excellent basis for the ongoing research of waste stabilisation within an old, unlined landfill. It would be useful to continue revisiting this site at periodic intervals to attempt to track long-term trends in leachate quality and transport, especially with the excellent monitoring network in place. With any luck it will continue to provide a training ground for future environmental researchers.

REFERENCES

- Acworth, R.I., 1998. Electromagnetic induction logs from selected bores in the Botany Sands aquifer. *In: McNally G. H. and Jankowski, J. (Eds.), Environmental Geology of the Botany Basin, Collected Case Studies in Engineering Geology, Fourth Series, Environmental, Engineering and Hydrogeology Specialist Group (EEHSG) of the Geological Society of Australia*, 143-159.
- Acworth, R.I. and Jankowski, J., 1993. Hydrogeochemical zonation of groundwater in the Botany Sands aquifer. *AGSO Journal of Australian Geology and Geophysics*, **14 (2/3)**, 193-199.
- Acworth, R.I. and Jankowski, J., 1994. Groundwater survey at the Kensington Campus. Technical Report, Water Research Laboratory, No. 94/18, 60 pp.
- Acworth, R.I., Kelly, B. and Jankowski, J., 1994. Improved contamination assessment of a domestic landfill using integrated geophysical, geochemical and hydrogeological methods. *In: Water Down Under '94 (conference proceedings)*, Adelaide, Australia, 21-25 November 1994, 235-240.
- Acworth, R.I. and Jorstad, L.B., 2006. Integration of multi-channel piezometry and electrical tomography to better define chemical heterogeneity in a landfill leachate plume within a sand aquifer. *Journal of Contaminant Hydrology*, **83**, 200-220.
- Albani, A.D., 1981. Sedimentary environments and Pleistocene chronology of the Botany Basin, NSW. *Geo-Marine Letters*, **1**, 163-167.
- Albani, A.D., 1998. The Botany Basin: Its bedrock topography and recent geological history. *In: McNally G. H. and Jankowski, J. (Eds.), Environmental Geology of the Botany Basin, Collected Case Studies in Engineering Geology, Fourth Series, Environmental, Engineering and Hydrogeology Specialist Group (EEHSG) of the Geological Society of Australia*, 190-196.
- American Society for Testing and Materials (ASTM), 2002. Standard Practice for Low Flow Purging and Sampling for Wells and Devices Used for Ground Water Quality Investigations. ASTM D-6771-02.
- American Public Health Association (APHA), 1998. *Standard Methods for the Examination of Water and Wastewater: (20th Edition)*, APHA-AWWA-WET, Washington DC.
- Amirbahman, A., Schonenberger, R., Johnson, C.A. and Sigg, L., 1998. Aqueous- and solid-phase biogeochemistry of a calcareous aquifer system downgradient from a municipal solid waste landfill (Winterthur, Switzerland). *Environmental Science and Technology*, **32(13)**, 1933-1940.

- Andreottola, G. and Cannas, P., 1992. Chemical and Biological Characteristics of Landfill Leachate. In: Christensen, T.H., Cossu, R. and Stegmann, R. (eds.) *Landfilling of Waste: Leachate*, Chapter 2.4. E and FN Spon, London, 65-88.
- Annable, R. and Cable K., 1995. South Sydney Heritage Study for South Sydney City Council. Volume 2 - Historical Material, 2-416.
- ANZECC, 2000. Guidelines for Fresh and Marine Water Quality, Australia New Zealand Environment and Conservation Council.
- Appelo, C.A.J. and Postma, D., 1999. Geochemistry, groundwater and pollution. Balkema, Rotterdam, NL, 536 pp.
- Aravena, R., Evans, M.L. and Cherry, J.A., 1993. Stable isotopes of oxygen and nitrogen in source identification of nitrate from septic tanks. *Ground Water*, **31**(2), 207-212.
- Aravena, R. and Robertson, W.D., 1998. Use of multiple isotope tracers to evaluate denitrification in ground water: Study of nitrate from a large-flux septic system plume. *Ground Water*, **36**(6), 975-982.
- Archie, G.E., 1942. The electrical resistivity log as an aid in determining some reservoir characteristics. *Transactions of the American Institute of Mining and Metallurgical Engineers*, **146**, 54-62.
- Arnth, J.-D., Milde, G., Kerndorff, H. and Schleyer, R., 1989. Waste deposit influences on groundwater quality as a tool for waste type and site selection for final storage quality. In: Baccini, P. (ed.), *The Landfill* vol. 20 (Lecture Notes in Earth Sciences), Springer, Berlin, 399-424.
- ASTM International (ASTM), 2003. Standard classification of soils for engineering purposes (Unified Soil Classification System), ASTM Standard D2487-00, Annual Book of ASTM Standards, Vol. 4.08.
- Baccini, P., Henseler, G., Figi, R., and Belevi, H., 1987. Water and element balances of municipal solid waste landfills. *Waste Management and Research*, **5**, 483-499.
- Back, W. and Baedeker, M.J., 1989. Chemical hydrogeology in natural and contaminated environments. *Journal of Hydrology*, **106**, 1-28.
- Back, W., Baedeker, M.J. and Wood, W.W., 1993. Scales in chemical hydrogeology: a historical perspective. In: Alley, W.M. (ed.) *Regional Groundwater Quality*, Van Nostrand Reinhold, New York, 111-129.
- Baedeker, M.J. and Back, W., 1979a. Modern marine sediments as a natural analog to the chemically stressed environment of a landfill. In: W. Back and D.A. Stephenson (Eds.), *Contemporary Hydrogeology – The George Burke Maxey Memorial Volume*, *Journal of Hydrology*, **43**, 393-414.

- Baedecker, M.J. and Back, W., 1979b. Hydrogeological processes and chemical reactions at a landfill. *Ground Water*, **17(5)**, 429-437.
- Baedecker, M.J. and Apgar, M.A., 1984. Hydrogeochemical studies at a landfill in Delaware. In: *Groundwater Contamination, Studies in Geophysics*, Chapter 10. National Academy Press, Washington D.C., 127-138.
- Baedecker, M.J. and Cozzarelli, I.M., 1992. The determination and fate of unstable constituents of contaminated groundwater. In: S. Lesage and R.E. Jackson (eds.) *Groundwater Contamination and Analysis at Hazardous Waste Sites*, Marcel Dekker, Inc., New York, 425-461.
- Ball, J.W. and Nordstrom, D.K., 1991, WATEQ4F--User's manual with revised thermodynamic data base and test cases for calculating speciation of major, trace and redox elements in natural waters: *U.S. Geological Survey Open-File Report 90-129*, 185 p.
- Barcelona, M.J. and Holm, T.R., 1991. Oxidation-reduction capacities of aquifer solids. *Environmental Science and Technology*, **25(9)**, 1565-1572.
- Barker, J.F. and Fritz, P., 1981. The occurrence and origin of methane in some groundwater flow systems. *Canadian Journal of Earth Science*, **18**, 1802-1816.
- Barker, J.F., Tessmann, J.S., Plotz, P.E. and Reinhard, M., 1986. The organic geochemistry of a sanitary landfill leachate plume. *Journal of Contaminant Hydrology*, **1**, 171-189.
- Barnes, I., Downes, C.J. and Hulston, J.R., 1978. Warm springs, South Island, New Zealand, and their potentials to yield laumontite. *American Journal of Science*, **278**, 1412-1427.
- Basberg, L., Banks, D. and Sæther, O.M., 1998. Redox processes in groundwater impacted by landfill leachate. *Aquatic Geochemistry*, **4(2)**, 253-272.
- Beck, P., 2000. Transport of conservative and reactive elements in the saturated part of a heterogenous sand aquifer, Botany Basin, Sydney, Australia. PhD thesis, School of Geology, University of New South Wales (unpublished).
- Beck, P. and Jankowski, J., 1998. A review of the geology of the Botany Basin: Bedrock topography and grainsize distribution in the Botany Sands aquifer. In: McNally G. H. and Jankowski, J. (Eds.), *Environmental Geology of the Botany Basin, Collected Case Studies in Engineering Geology, Fourth Series, Environmental, Engineering and Hydrogeology Specialist Group (EEHSG) of the Geological Society of Australia*, 179-189.
- Bedient, P.B., Rifai, H.S. and Newell, C.J., 1994. Ground Water Contamination, Prentice Hall, New Jersey, 350 pp.

- Belevi, H. and Baccini, P., 1992. Long-Term Leachate Emissions from Municipal Solid Waste Landfills. In: Christensen, T.H., Cossu, R. and Stegmann, R. (eds.) *Landfilling of Waste: Leachate*, Chapter 4.1, E and FN Spon, London, 431-440.
- Bell, F.G., 1998. Environmental Geology, Principles and Practice, Chapter 14: Waste and its disposal. Blackwell Science, Oxford, 459-486.
- Bennett, M.R. and Doyle, P., 1997. Environmental Geology, Geology and the Human Environment, Chapter 9: Waste and Pollution Management. John Wiley & Sons, Chichester, 288-312.
- Benson, D. and J. Howell, Taken for Granted, The Bushland of Sydney and its Suburbs, Kangaroo Press, 1990.
- Bishop, P.K., 1990. Precipitation of dissolved carbonate species from natural waters for $\delta^{13}\text{C}$ analysis – a critical appraisal. *Isotope Geoscience*, **80**, 251-259.
- Bjerg, P.L. and Christensen, T.H., 1992. Spatial and temporal small-scale variation in groundwater quality of a shallow sandy aquifer. *Journal of Hydrology*, **131**, 133-149.
- Bjerg, P.L., Hinsby, K., Christensen, T.H. and Gravesen, P., 1992. Spatial variability of hydraulic conductivity of an unconfined sandy aquifer determined by a mini slug test. *Journal of Hydrology*, **136**, 107-122.
- Bjerg, P.L. and Christensen, T.H., 1993. A field experiment on cation exchange affected multicomponent solute transport in a sandy aquifer. *Journal of Contaminant Hydrology*, **12**, 269-290.
- Bjerg, P.L., Rugge, K., Pederson, J.K. and Christensen, T.H., 1995. Distribution of Redox-Sensitive Groundwater Quality Parameters Downgradient of a Landfill (Grinsted, Denmark). *Environmental Science and Technology*, **29(5)**, 1387-1394.
- Brodie, R., 1988. A laboratory investigation into the acid treatment of carbonate incrustated boreholes. M.App.Sc. thesis, University of New South Wales (unpublished).
- Boggs, J.M., Young, S.C., Beard, L.M., Gelhar, L.W., Rehfeldt, K.R. and Adams, E.E., 1992. Field study of dispersion in a heterogeneous aquifer, 1. Overview and site description. *Water Resources Research*, **28(12)**, 3281-3291.
- Bureau of Meteorology, 2002. Average temperature and rainfall data for Sydney, collected from gauging station 66037, Sydney Airport, from 1929-2001. *Bureau of Meteorology web page* (www.bom.gov.au).
- Bussian, A.E., 1983. Electrical conductance in a porous medium. *Geophysics*, **48(9)**, 1258-1268.

- Butler, K.M., 1990. The hydrochemistry and hydrology of the Lachlan Lakes system in the Botany Wetlands, Sydney, New South Wales. M. App. Sc. Thesis, CGMH, University of New South Wales, 276 pp (unpublished).
- Carey, G.R., Van Geel, P.J., Wiedemeier, T.H. and McBean, E.A., 2003. A modified radial diagram approach for evaluating natural attenuation trends for chlorinated solvents and inorganic redox indicators. *Ground Water Monitoring and Remediation*, **23**(4), 75-84.
- Champ, D.R., Gulens, J., Jackson, R.E., 1979. Oxidation-reduction sequences in groundwater flow systems. *Canadian Journal of Earth Sciences*, **16**, 12-23.
- Chapelle, F.B., Morris, J.T., McMahon, P.B. and Zelibor, J.R. Jr., 1988. Bacterial metabolism and the $\delta^{13}\text{C}$ composition of groundwater, Floridan aquifer system, South Carolina. *Geology*, **16**, 117-121.
- Chapelle, F.H., McMahon, P.B., Dubrovsky, N.M., Fuji, R.F., Oaksford, E.T. and Vroblesky, D.A., 1995. Deducing the distribution of terminal electron-accepting processes in hydrologically diverse groundwater systems. *Water Resources Research*, **31**, 359-371.
- Chapelle, F.H., Haack, S.K., Adriaens, P., Henry, M.A. and Bradley, P.M., 1996. Comparison of Eh and H_2 measurements for delineating redox processes in a contaminated aquifer. *Environmental Science and Technology*, **30**(12), 3565-3569.
- Cherry, J.A., 1983. Piezometers and other permanently-installed devices for groundwater quality monitoring. In: *Proc. Seminar on Groundwater and Petroleum Hydrocarbons Protection, Detection, Restoration, Petroleum Assoc. for Conservation of the Canadian Environment, Ottawa*, p. IV-1 - IV-39.
- Chian, E.S.K. and DeWalle, F.B., 1976. Sanitary Landfill Leachates and Their Treatment. *Journal of the Environmental Engineering Division, ASCE*, 103(EE2), 411-431.
- Chian, E.S.K. and DeWalle, F.B., 1977. Characterization of soluble organic matter in leachate. *Environmental Science and Technology*, **11**(2), 158-163.
- Christensen, J.B., Jensen, D.L., and Christensen, T.H., 1996. Effect of dissolved organic carbon on the mobility of cadmium, nickel and zinc in leachate-polluted groundwater. *Water Research*, **30**(12), 3037-3049.
- Christensen, J.B., Jensen, D.L., Grøn, C., Filip, Z., and Christensen, T.H., 1998. Characterization of the dissolved organic carbon in landfill leachate-polluted groundwater. *Water Research*, **32**(1), 125-135.
- Christensen, J.B., Botma, J.J. and Christensen, T.H., 1999. Complexation of Cu and Pb by DOC in polluted groundwater: a comparison of experimental data and predictions by computer speciation models (WHAM and MINTEQA2). *Water Research*, **33**(15), 3231-3238.

- Christensen, J.B. and Christensen, T.H., 1999. Complexation of Cd, Ni and Zn by DOC in polluted groundwater: a comparison of approaches using resin exchange, aquifer material sorption, and computer speciation models (WHAM and MINTEQA2). *Environmental Science and Technology*, **33**(21), 3857-3863.
- Christensen, J.B. and Christensen, T.H., 2000. The effect of pH on the complexation of Cd, Ni and Zn by dissolved organic carbon from leachate-polluted groundwater. *Water Research*, **34**(15), 3743-3754.
- Christensen, T.H. and Kjeldsen, P., 1989. Basic biochemical processes in landfills. In: T.H. Christensen, R. Cossu and R. Stegmann (eds.) *Sanitary Landfilling: Process, Technology and Environmental Impact*. Academic Press, London, 592 pp.
- Christensen, T.H., 1992. Attenuation of Leachate Pollutants in Groundwater. In: Christensen, T.H., Cossu, R. and Stegmann, R. (eds.) *Landfilling of Waste: Leachate*, Chapter 4.21, E and FN Spon, London, 441-483.
- Christensen, T.H., Kjeldsen, P., Albrechtsen, H.-J., Heron, G., Nielsen, P.H., Bjerg, P.L. and Holm, P.E., 1994. Attenuation of landfill leachate pollutants in aquifers. *Critical Reviews in Environmental Science and Technology*, **24**(2), 119-202.
- Christensen, T.H., Bjerg, P.L., Banwart, S.A., Jakobsen, R., Heron, G. and Albrechtsen, H.-J., 2000. Characterization of redox conditions in groundwater contaminant plumes. *Journal of Contaminant Hydrology*, **45**, 165-241.
- Christensen, T.H., Kjeldsen, P., Bjerg, P.L., Jensen, D.L., Christensen, J.B., Baun, A., Albrechtsen, H.-J. and Heron, G., 2001. Biogeochemistry of landfill leachate plumes. *Applied Geochemistry*, **16**, 659-718.
- Clark, I.D. and Fritz, P., 1997. *Environmental Isotopes in Hydrogeology*, CRC Press LLC, Boca Raton, Florida.
- Colby, N.D., 1993. The use of 2-way cluster analysis as a tool for delineating trends in hydrogeologic units and development of a conceptual model. In: *Proceedings of the 1993 Ground Water Modeling Conference, International Ground Water Modeling Center, Golden, Colorado*, 91-100.
- Coleman, D.D., Liu, C.L., and Riley, K.M., 1988. Microbial methane in the shallow paleozoic sediments and glacial deposits of Illinois, USA. *Chemical Geology*, **71**, 23-40.
- Cooke, A.J., Rowe, R.K., Rittmann, B.E., vanGulck, J. and Millward, S., 2001. Biofilm growth and mineral precipitation in synthetic leachate columns. *Journal of Geotechnical and Geoenvironmental Engineering*, **127**(10), 849-856.
- Coplen, T.B., 1994. Reporting of stable hydrogen, carbon and oxygen isotopic abundances. *Pure and Applied Chemistry*, **66**, 273-276.
- Cotton, L.A., 1937. The underground water supply of the Botany district. University of Sydney (unpublished report).

- Cozzarelli, I.M., Suflita, J.M., Ulrich, G.A., Harris, S.H., Scholl, M.A., Schlottmann, J.L. and Jaeschke, J.B., 1999. Biogeochemical processes in a contaminant plume downgradient from a landfill, Norman, Oklahoma. U.S. Geological Survey Water Resources Investigations Report # 99-4018C, Reston, VA.
- Cozzarelli, I.M., Suflita, J.M., Ulrich, G.A., Harris, S.H., Scholl, M.A., Schlottmann, J.L. and Christenson, S., 2000. Geochemical and Microbiological Methods for Evaluating Anaerobic Processes in an Aquifer Contaminated by Landfill Leachate. *Environmental Science and Technology*, **34**(18), 4025-4033.
- Craig, H., 1953. The geochemistry of the stable carbon isotopes. *Geochimica et Cosmochimica Acta*, **4**, 53-92.
- Craig, H., 1957. Isotopic standards for carbon and oxygen and correction factors for mass-spectrometric analysis of carbon dioxide. *Geochimica et Cosmochimica Acta*, **12**, 133-149.
- Craig, H., 1961a. Standard for reporting concentrations of deuterium and oxygen-18 in natural waters. *Science*, **133**, 1833-1834.
- Craig, H., 1961b. Isotopic variations in meteoric waters. *Science*, **133**, 1702-1703.
- Deffeyes, K.S., 1965. Carbonate equilibria: a graphic and algebraic approach. *Limnology and Oceanography*, **10**, 412-426.
- Deines, P., Langmuir, D. and Harmon, R.S., 1974. Stable carbon isotope ratios and the existence of a gas phase in the evolution of carbonate waters. *Geochimica et Cosmochimica Acta*, **38**, 1147-1164.
- Department of Land and Water Conservation, Map Centre (1930 aerial photograph).
- Department of Land and Water Conservation (DLWC), 2000. Status Report No.2, Botany Sand Beds (GWMA 018), Botany Basin, NSW: Northern, Southern and Western Zones.
- Desimone, L.A. and Howes, B.L., 1996. Denitrification and nitrogen transport in a coastal aquifer receiving wastewater discharge. *Environmental Science and Technology*, **30**(4), 1152-1162.
- Desrocher, S. and Sherwood Lollar, B., 1998. Isotopic quantification of landfill CH₄ contamination – An innovative and cost effective methodology. *Ground Water*, **36**, 801-809.
- Domenico, P.A. and Schwartz, F.W., 1998. Physical and Chemical Hydrogeology, 2nd Edition. John Wiley and Sons, Inc., New York, 824p.
- Drever, J.I., 1988. The Geochemistry of Natural Waters, 2nd Edition. Prentice-Hall, Inc., New Jersey, 437p.

- Dudgeon, B., 1993. The hydrogeology, hydrology and hydrochemistry of the Botany Sands recharge zone in Centennial Park, Sydney. MEngSc thesis, University of New South Wales (unpublished).
- Ehrig, H.-J., 1983. Quality and quantity of sanitary landfill leachate. *Waste Management and Research*, **1**, 53-68.
- Ehrig, H.-J., 1988. Water and element mass balances of landfills. In: Baccini, P. (ed.), *The Landfill, Lecture Notes in Earth Sciences* v.20. Springer, Berlin, 83-116.
- Erskine, A.D., 2000. Transport of ammonium in aquifers: retardation and degradation. *Quarterly Journal of Engineering Geology and Hydrogeology*, **33**, 161-170.
- Essaid, H.I., Cozzarelli, I.M., Eganhouse, R.P., Herkelrath, W.N., Bekins, B.A., and Delin, G.N., 2003. Inverse modelling of BTEX dissolution and biodegradation at the Bemidji, MN crude-oil spill site. *Journal of Contaminant Hydrology*, **67**, 269-299.
- Evans, D.J., 1993. A physical and hydrochemical characterisation of a sand aquifer in Sydney, Australia. MappSc thesis, University of New South Wales, Sydney (unpublished).
- Fetter, C.W., 1994. Applied Hydrogeology. Macmillan College Publishing Company, Inc., New York, 691 pp.
- Fetter, C.W., 1999. Contaminant Hydrogeology. Prentice-Hall, Inc., New Jersey, 500 pp.
- Fisher, R.S. and Mullican III, W.F., 1997. Hydrochemical evolution of sodium-sulfate and sodium-chloride groundwater beneath the northern Chihuahuan Desert, Trans-Pecos, Texas, USA. *Hydrogeology Journal*, **5**, 4-16.
- Flemming, I.R., Rowe, R.K. and Cullimore, D.R., 1999. Field observations of clogging in a landfill leachate collection system. *Canadian Geotechnical Journal*, **36(4)**, 685-707.
- Freeze, R.A. and Cherry, J.A., 1979. Groundwater. Prentice-Hall, Inc., New Jersey, 604 pp.
- Fritz, P., Matthess, G., and Brown, R.M., 1976. Deuterium and oxygen-18 as indicators of leachwater movement from a sanitary landfill. In: *Interpretation of Environmental Isotope and Hydrochemical Data in Groundwater Hydrology*, International Atomic Energy Agency, Vienna, 131-142.
- Games, L.M. and Hayes, J.M., 1977. Carbon isotopic study of the fate of landfill leachate in groundwater. *Journal of Water Pollution Control Federation*, **49**, 668-677.

- Garabedian, S.P., LeBlanc, D.R., Gelhar, L.W. and Celia, M.A., 1991. Large-scale natural gradient tracer test in sand and gravel, Cape Cod, Massachusetts, 2. Analysis of spatial moments for a nonreactive tracer. *Water Resources Research*, **27(5)**, 911-924.
- Garrels, R.M. and Mackenzie, F.T., 1967. Origin of the chemical compositions of springs and lakes. *In: Equilibrium concepts in natural water systems*, American Chemical Society, Advances in Chemistry Series No. 67, 222-242.
- Gehlar, L.W., Welty, C. and Rehfeldt, K.R., 1992. A critical review of data on field-scale dispersion in aquifers. *Water Resources Research*, **28**, 1955-1974.
- Goodwin, R.H., 1971. Hydrodynamics and sedimentation in Botany Bay, Sydney. PhD thesis, University of Sydney.
- Goodwin, R.H., 1976. The surficial sediments of Botany Bay, Australia. *Pacific Geology*, **11**, 33-46.
- Gounaris, V., Anderson, P.R., and Holsen, T.M., 1993. Characteristics and environmental significance of colloids in landfill leachate. *Environmental Science and Technology*, **27(7)**, 1381-1387.
- Griffin, D.J., 1963. The Botany Basin. *Geological Survey of New South Wales Bulletin* No. **18**, 110pp.
- Grossman, E.L., Coffman, B.K., Fritz, S.J. and Wada, H., 1989. Bacterial production of methane and its influence on ground-water chemistry in east-central Texas aquifers. *Geology*, **17**, 495-499.
- Grossman, E.L., Cifuentes, L.A. and Cozzarelli, I.M., 2002. Anaerobic methane oxidation in a landfill-leachate plume. *Environmental Science and Technology*, **36(11)**, 2436-2442.
- Groves, Sarah, 1999. The influence of heterogeneity on groundwater chemistry in the Botany Sands Aquifer, Sydney. Honours thesis, UNSW, 148 pp.
- Hach, 2000. DR/2010 Spectrophotometer handbook – Procedures manual. *Hach Company*, Loveland, Colorado, 872 pp.
- Hackley, K.C., Liu, C.L. and Coleman, D.D., 1996. Environmental isotopes characteristics of landfill leachates and gases. *Ground Water*, **34(5)**, 827-836.
- Harmsen, J., 1983. Identification of organic compounds in leachate from a waste tip. *Water Research*, **17**, 699-705.
- Hawke, J. M., 1973. Pollution of Botany Sand Beds Final report. *Geological Survey of NSW, Dept of Mines: Sydney*, Report 1973/329.
- Herbert, C., 1979. The geology and resource potential of the Wianamatta Group. *Geological Survey of New South Wales*, **8**, 1-5.

- Herbert, C. and Helby, R., 1980. A guide to the Sydney Basin. *Geological Survey of New South Wales*, **26**, 1-6.
- Herczeg, A.L., Richardson, S.B. and Dillon, P.J., 1991. Importance of methanogenesis for organic carbon mineralisation in groundwater contaminated by liquid effluent, South Australia. *Applied Geochemistry*, **6**, 533-542.
- Heron, G., Christensen, T.H. and Tjell, J.C., 1994a. Oxidation capacity of aquifer sediments. *Environmental Science and Technology*, **28(1)**, 153-158.
- Heron, G., Crouzet, C., Bourg, A.C.M. and Christensen, T.H., 1994b. Speciation of Fe(II) and Fe(III) in contaminated aquifer sediments using chemical extraction techniques. *Environmental Science and Technology*, **28(9)**, 1698-1705.
- Heron, G. and Christensen, T.H., 1995. Impact of sediment-bound iron on redox buffering in a landfill leachate polluted aquifer (Vejen, Denmark). *Environmental Science and Technology*, **29**, 187-192.
- Heron, G., Bjerg, P.L., Gravesen, P., Ludvigsen, L. and Christensen, T.H., 1998. Geology and sediment geochemistry of a landfill leachate contaminated aquifer. *Journal of Contaminant Hydrology*, **29**, 301-317.
- Hitchcock, P.W., 1991. The hydrogeology, hydrochemistry and geophysical conditions of Astrolabe Park and environs. MAppSc thesis, CGMH, University of New South Wales, 325 pp (unpublished).
- Hoefs, J., 1980. *Stable Isotope Geochemistry*, 2nd Ed., Minerals and Rocks Series, v.9, Springer Verlag, Berlin.
- Hornibrook, E.R.C., Longstaffe, F.J. and Fyfe, W.S., 2000. Evolution of stable carbon isotope compositions for methane and carbon dioxide in freshwater wetlands and other anaerobic environments. *Geochimica et Cosmochimica Acta*, **64(6)**, 1013-1027.
- Horowitz, A.J., Lum, K.R., Garbarino, J.R., Hall G.E.M., Lemieux, C. and Demas, C.R., 1996. Problems associated with using filtration to define dissolved trace element concentrations in natural water samples. *Environmental Science and Technology*, **30**, 954-963.
- Hübner, H., 1986. Isotope effects of nitrogen in the soil and biosphere. In: Fritz, P. and J.-C. Fontes (eds.) *Handbook of Environmental Isotope Geochemistry: The Terrestrial Environment*, Vol. 2, Elsevier, Amsterdam, 361-425.
- Hunkeler, D., Andersen, N., Aravena, R., Bernasconi, S.M. and Butler, B.J., 2001. Hydrogen and Carbon Isotope Fractionation during Aerobic Biodegradation of Benzene. *Environmental Science and Technology*, **35(17)**, 3462-3467.

- Hunkeler, D., Aravena, R. and Cox, E., 2002. Carbon isotopes as a tool to evaluate the origin and fate of vinyl chloride: laboratory experiments and modelling of isotope evolution. *Environmental Science and Technology*, **36**, 3378-3384.
- Hvorslev, M.J., 1951. Time lag and soil permeability in ground water observations. *U.S. Army Corps of Engineers Waterway Experimentation Station*, Bulletin 36.
- Iqbal, M.Z., Krothe, N.C. and Spalding, R.F., 1997. Nitrogen isotope indicators of seasonal source variability to groundwater. *Environmental Geology*, **32(3)**, 210-218.
- International Organization for Standardization (ISO), 1993. Statistics – Vocabulary and Symbols – Part I: Probability and general statistical terms. ISO, Geneva, Switzerland, 1993.
- Jakobsen, R., Albrechtsen, H.-J., Rasmussen, M., Bay, H., Bjerg, P.L., and Christensen, T.H., 1998. H₂ concentrations in a landfill leachate plume (Grinsted, Denmark): In situ energetics of terminal electron acceptor processes. *Environmental Science and Technology*, **32(14)**, 2142-2148.
- Jakobsen, R. and Postma, D., 1999. Redox zoning, rates of sulfate reduction and interactions with Fe-reduction and methanogenesis in a shallow sandy aquifer, Rømø, Denmark. *Geochimica et Cosmochimica Acta*, **63(1)**, 137-151.
- Jankowski, J., 1997. Generation and migration of chemical constituents in a leachate plume: a case study from a municipal landfill at Astrolabe Park, Sydney. In: McNally G. H. (Ed.), *Collected Case Studies in Engineering Geology, Hydrogeology and Environmental Geology*, 3rd Vol., Geological Society of Australia, 181-205.
- Jankowski, J. and Knight, M.J., 1991. Astrolabe Park landfill leachate and its impact on the Botany wetlands. *Technical Report by Unisearch Ltd for Sydney Water Board* (unpublished).
- Jankowski, J. and Acworth, R.I., 1997. Development of a contaminant plume from a municipal landfill: Redox reactions and plume variability. In: Chilton J. et al. (Eds.), *Proceedings of the XXVIII Congress of the International Association of Hydrogeologists Congress on Groundwater in the Urban Environment: Problems, Processes and Management*, Nottingham, UK, 21-27 September 1997. Vol.1, 439-444. Balkema, Rotterdam.
- Jankowski, J., Acworth, R.I. and Beck, P., 1997. Vertical heterogeneity in the Botany Sands aquifer, Australia: Implications for chemical variations and contaminant plume delineation. In: *Proceedings of the XXVII Congress of the International Association of Hydrogeologists*, 21-27 September, Nottingham, UK.
- Jankowski, J. and Yu, X. W., 1998. Hydrogeochemistry of the Botany Sands aquifer. In: McNally G. H. and Jankowski, J. (Eds.), *Environmental Geology of the Botany Basin*, 37-57. Geological Society of Australia, EEHGS Collected Case Studies in Engineering Geology, Fourth Series.

- Jankowski, J. and Beck, P., 2000. Aquifer heterogeneity: hydrogeological and hydrochemical properties of the Botany Sands aquifer and their impact on contaminant transport. *Australian Journal of Earth Sciences*, **47**, 45-64.
- Jeffery, R.P., 1985. Botany Basin deposits. In Pells, P.J.N. (ed) *Engineering geology of the Sydney region*, Balkema, Rotterdam, 81-97.
- Jensen, D.L. and Christensen, T.H., 1999. Colloidal and dissolved metals in leachates from four Danish landfills. *Water Research*, **33(9)**, 2139-2147.
- Jensen, D.L., Ledin, A. and Christensen, T.H., 1999. Speciation of heavy metals in landfill-leachate polluted groundwater. *Water Research*, **33(11)**, 2642-2650.
- Jensen, D.L., Boddum, J.K., Tjell, J.C. and Christensen, T.H., 2002. The solubility of rhodocrosite (MnCO_3) and siderite (FeCO_3) in anaerobic aquatic environments. *Applied Geochemistry*, **17(4)**, 503-511.
- Jensen, K.H., Bitsch, K. and Bjerg, P.L., 1993. Large-scale dispersion experiments in a sandy aquifer in Denmark: observed tracer movements and numerical analysis. *Water Resources Research*, **29(3)**, 673-696.
- Jewell, C.M. and Associates Pty. Ltd., 1997. Groundwater Related Impacts of Proposed Eastern Distributor, Report No. AJ9707.27 (unpublished report).
- Johnson, B.D., Albani, A.D., Rickwood, P.C. and Tayton, J.W., 1977. The bedrock topography of the Botany Basin, NSW. *Journal of the Geological Society of Australia*, **24**, 403-408.
- Johnson, M., 1981. Hydrochemistry of the Botany Bay region. PhD thesis, University of New South Wales, 453 pp. (unpublished).
- Jorstad, L.B., Jankowski, J. and Acworth, R.I., 2004. Analysis of the distribution of inorganic constituents in a landfill leachate-contaminated aquifer: Astrolabe Park, Sydney, Australia. *Environmental Geology*, **46**, 263-272.
- Junk, G. and Svec, H., 1958. The absolute abundance of the nitrogen isotopes in the atmosphere and compressed gas from various sources. *Geochimica et Cosmochimica Acta*, **1**, 234-243.
- Kalf, F. and McKibbin, D., 1976. The Hydrogeological Effects of Sand Mining in the Botany Sand Beds, Water Conservation and Irrigation Commission, NSW. Hydrogeological Report No. 1976/13.
- Kang, K.-H., Shin, H.S. and Park, H., 2002. Characterization of humic substances present in landfill leachates with different landfill ages and its implications. *Water Research*, **36**, 4023-4032.
- Kehew, A.E. and Passero, R.N., 1990. pH and redox buffering mechanisms in a glacial drift aquifer contaminated by landfill leachate. *Ground Water*, **28(5)**, 728-737.

- Kendall, C. and Aravena, R., 2000. Nitrate isotopes in groundwater systems. *In*: Cook, P.G. and A.L. Herczeg (eds.) *Environmental Tracers in Subsurface Hydrology*, Chapter 9, Kluwer Academic Publishers, Dordrecht, Netherlands, 261-297.
- Keys, W., 1989. Borehole Geophysics Applied to Ground-Water Investigations. Dublin, Ohio: National Water Well Association.
- Kimmel, G.E. and Braids, O.C., 1974. Leachate plumes in a highly permeable aquifer. *Ground Water*, **12(6)**, 388-392.
- Kimmel, G.E. and Braids, O.C., 1980. Leachate plumes in groundwater from Babylon and Islip landfills, Long Island, New York. US Geological Survey, Professional Paper 1085, Washington D.C.
- Kjeldsen, P. and Christensen, T.H., 1984. Soil attenuation of acid phase landfill leachate. *Waste Management and Research*, **2**, 247-263.
- Kjeldsen, P., 1986. Attenuation of landfill leachate in soil and aquifer material. PhD Thesis, Technical University of Denmark (unpublished).
- Kjeldsen, P., 1993. Groundwater pollution source characterization of an old landfill. *Journal of Hydrology*, **142**, 349-371.
- Kjeldsen, P., Grundtvig, A., Winther, P. and Andersen, J.S., 1998a. Characterization of an old municipal landfill (Grindsted, Denmark) as a groundwater pollution source: landfill history and leachate composition. *Waste Management Research*, **16(1)**, 3-13.
- Kjeldsen, P., Bjerg, P.L., Rügge, K., Christensen, T.H. and Pedersen, J.K., 1998b. Characterization of an old municipal landfill (Grindsted, Denmark) as a groundwater pollution source: landfill hydrology and leachate migration. *Waste Management Research*, **16(1)**, 14-22.
- Kjeldsen, P. and Christensen, T.H., 2001. Composition of leachate from old landfills in Denmark. *Waste Management and Research*, **19**, 249-256.
- Kjeldsen, P., Barlaz, M.A., Rooker, A.P., Baun, A., Ledin, A., Christensen, T.H., 2002. Present and long-term composition of MSW landfill leachate: A review. *Critical Reviews in Environmental Science and Technology*, **32(4)**, 297-336.
- Knight, M.J., Jankowski, J. and Butler, K.M., 1991. Hydrochemistry of the Lachlan Lakes, Wetland System, Sydney, NSW. *CGMH, University of New South Wales, Res. Pub.* **90/1**, 142 pp (unpublished).
- Kreitler, C.W., Ragone, S.E. and Katz, B.G., 1978. $^{15}\text{N}/^{14}\text{N}$ ratios of groundwater nitrate, Long Island, New York. *Ground Water*, **16**, 404-409.
- Lavitt, N., Acworth, R.I., and Jankowski, J., 1997. Vertical hydrochemical zonation in a coastal section of the Botany Sands aquifer. *Hydrogeology Journal*, **5**, 64-74.

- Leaney, F., Dighton, J., and Lemontagne, S., 2001. Recommended protocol for the collection and shipping of water samples for $\delta^{15}\text{N}$ analysis of NH_4^+ and NO_3^- . CSIRO Isotope Laboratory, Adelaide, Australia, written communication.
- LeBlanc, D.R., Garabedian, S.P., Hess, K.M., Gelhar, L.W., Quadri, R.D., Stollenwerk, K.G. and Wood, W.W. Large-scale natural gradient tracer test in sand and gravel, Cape Cod, Massachusetts, 1. Experimental design and observed tracer movement. *Water Resources Research*, **27(5)**, 895-910.
- Lennie, A. R., England, K.E.R., and Vaughan D.J. (1995) Transformation of Synthetic Mackinawite to Hexagonal Pyrrhotite - a Kinetic-Study. *American Mineralogist*, **80(9-10)**, 960-967.
- Létolle, R., 1980. Nitrogen-15 in the natural environment. *In*: Fritz, P. and J.-C. Fontes (eds.) *Handbook of Environmental Isotope Geochemistry*, Elsevier, Amsterdam, 407-433.
- Loke M.H., 2001. Rapid 2D and 3D Resistivity and IP inversion using the least squares method – RES2DINV Ver 3.4 Manual. Published on the world wide web at www.geoelectrical.com.
- Lu, J.C.S., Eichenberger, B., and Stearns, R.J., 1984. Production and Management of Leachate from Municipal Landfills: Summary and Assessment. EPA-602/2-84-092, Municipal Environmental Laboratory, Cincinnati, Ohio: U.S. Environmental Protection Agency.
- Lucius, J.E. and Bisdorf, R.J., 1995. Results of geophysical investigations near the Norman, Oklahoma, Municipal Landfill. U.S. Geological Survey Open-File Report 95-825, 125 pp.
- Ludvigsen, L., Albrechtsen, H.-J., Heron, G., Bjerg, P.L., Christensen, T.H., 1998. Anaerobic microbial redox processes in a landfill leachate contaminated aquifer (Grinsted, Denmark). *Journal of Contaminant Hydrology*, **33**, 273-291.
- Ludvigsen, L., Albrechtsen, H.-J., Heron, G., Ringelberg, D.B., Ekelund, F. and Christensen, T.H., 1999. Composition and distribution of microbial biomass in a landfill leachate contaminated aquifer (Grinsted, Denmark). *Microbial Ecology*, **37**, 197-207.
- Lyngkilde, J. and Christensen, T.H., 1992a. Redox zones of a landfill leachate pollution plume (Vejen, Denmark). *Journal of Contaminant Hydrology*, **10**, 273-289.
- Lyngkilde, J. and Christensen, T.H., 1992b. Fate of organic contaminants in the redox zones of a landfill leachate pollution plume (Vejen, Denmark). *Journal of Contaminant Hydrology*, **10**, 291-307.
- MacFarlane, D.S., Cherry, J.A., Gillham, R.W., and Sudicky, E.A., 1983. Migration of contaminants in groundwater at a landfill: A case study. 1. Groundwater flow and plume delineation. *Journal of Hydrology*, **63**, 1-29.

- Mack, T.J., 1993. Detection of contaminant plumes by borehole geophysical logging. *Ground Water Monitoring Review*, **13(1)**, 107-114.
- Mackay, D.M., Freyberg, D.L., Roberts, P.V. and Cherry, J.A., 1986. A natural gradient experiment on solute transport in a sand aquifer, 1. Approach and overview of plume movement. *Water Resources Research*, **22(13)**, 2017-2029.
- Mancini, S.A., Lacrampe-Couloume, G., Jonker, H., Van Breukelen, B., Groen, J., Volkering, F. and Lollar, B.S., 2002. Hydrogen isotopic enrichment: an indicator of biodegradation at a petroleum hydrocarbon contaminated field site. *Environmental Science and Technology*, **36**, 2464-2470.
- Manning, D.A.C., 2001. Calcite precipitation in landfills: an essential product of waste stabilization. *Mineralogical Magazine*, **65(5)**, 603-610.
- Manning, D.A.C. and Robinson, N., 1999. Leachate-mineral reactions: implications for drainage system stability and clogging. In: (T.H. Christensen, R. Cossu and R. Stegmann, eds.) *Proceedings of the 7th International Waste Management and Landfill Symposium*, Sardinia, Italy, Vol. III, 269-276.]
- Mariotti, A., Landreau, A. and Simon, B., 1988. ¹⁵N isotope biogeochemistry and natural denitrification processes in groundwater: Application to the chalk aquifer in northern France. *Geochimica et Cosmochimica Acta*, **52(7)**, 1869-1878.
- Maunsell Pty. Ltd., 1997. Eastern Distributor: Review of Groundwater Related Issues for Parkway Proposal, Document No. n641/RP/CV/023A.
- McCrea, J.M., 1950. On the isotopic chemistry of carbonates and a paleo-temperature scale. *J. Chem. Phys.*, **18**, 849.
- McGuire, J.T., Long, D.T. and Hyndman, D.W., 2005. Analysis of recharge-induced geochemical change in a contaminated aquifer. *Ground Water*, **43(4)**, 518-530.
- McNally, G.H. and Branagan, D.F., 1998. The St Peters brickpits: their geology, operation and reclamation, and the adjacent quaternary shoreline. In: McNally G. H. and Jankowski, J. (Eds.), *Environmental Geology of the Botany Basin, Collected Case Studies in Engineering Geology, Fourth Series, Environmental, Engineering and Hydrogeology Specialist Group (EEHSG) of the Geological Society of Australia*, 93-111.
- McNeill, J.D., 1986. Geonics EM39 borehole conductivity meter – theory of operation. Geonics Limited, Technical note **TN-24**.
- McNeill, J.D., 1989. Application of Geonics EM39 borehole conductivity logger at two groundwater contamination sites. Geonics Limited, Technical note **TN-20**.
- McNeill, J.D., 1990. Use of electromagnetic methods for groundwater studies. In: Ward, S.H. (ed.), *Geotechnical and Environmental Geophysics. Vol.1: Review and Tutorial*. Tulsa: Society of Exploration Geophysicists, 191-218.

- McNeill, J.D., Bosnar, M., and Snelgrove, F.B., 1988. A borehole induction logger for monitoring groundwater contamination. Geonics Limited, Mississauga, Ontario, Canada.
- Mengis, M., Walther, U., Bernasconi, S.M. and Wehrli, B., 2001. Limitations of using $\delta^{18}\text{O}$ for the source identification of nitrate in agricultural soils. *Environmental Science and Technology*, **35**, 1840-1844.
- Merrick, N.P., 1994. A groundwater flow model of the Botany Basin. In: *Proceedings of the XXV Congress of the International Association of Hydrogeologists, Water Down Under*, Adelaide, Australia, November 1994. Vol. 2A, 113-118.
- Merrick, N.P., 1998. Glimpses of history: The evolution of groundwater levels and usage in the Botany Basin. In: McNally G. H. and Jankowski, J. (Eds.), *Environmental Geology of the Botany Basin, Collected Case Studies in Engineering Geology, Fourth Series, Environmental, Engineering and Hydrogeology Specialist Group (EEHSG) of the Geological Society of Australia*, 230-242.
- Mirecki, J.E. and Parks, W.S., 1994. Leachate geochemistry at a municipal landfill, Memphis, Tennessee. *Ground Water*, **32(3)**, 390-398.
- Mulholland, C.St.J., 1942. Emergency Water Supply. *Preliminary report in Sheil (1942): Investigation of Botany Sand Beds Emergency Water Supply*. Premiers Dept., Sydney (unpublished report).
- National Environmental Protection Council (NEPC), 1999a. National Environmental Protection (Assessment of Site Contamination) Measure, Schedule B(1): Guideline on the Investigation Levels for Soil and Groundwater.
- National Environmental Protection Council (NEPC), 1999b. National Environmental Protection (Assessment of Site Contamination) Measure, Schedule B(2): Guideline data collection, sample design and reporting.
- Nicholson, R.V., Cherry, J.A., and Reardon, E.J., 1983. Migration of contaminants in groundwater at a landfill: A case study. 6. Hydrogeochemistry. *Journal of Hydrology*, **63**, 131-176.
- Nieweglowski, P., 1988. A study of the hydrochemical changes occurring with the Botany Sands aquifer over the period 1960 to 1986. MAppSc thesis, CGMH, University of New South Wales, 271 pp (unpublished).
- Nordstrom, D.K., 1977. Thermochemical redox equilibria of Zobell's solution. *Geochimica et Cosmochimica Acta*, **41**, 1835-1841.
- North, J.C., Frew, R.D. and Peake, B.M., 2004. The use of carbon and nitrogen isotope ratios to identify landfill leachate contamination: Green Island Landfill, Dunedin, New Zealand. *Environment International*, **30**, 631-637.

- Parkhurst, D.L. and Plummer, L.N., 1993. Geochemical models. *In*, Alley W.M. (ed) *Regional Ground-water Quality, Ch.9*. Van Nostrand Reinhold, 199-225.
- Parkhurst, D.L., and Appelo, C.A.J., 1999, User's guide to PHREEQC (Version 2)--a computer program for speciation, batch-reaction, one-dimensional transport, and inverse geochemical calculations: U.S. Geological Survey Water-Resources Investigations Report 99-4259, 312 p.
- Peyton, G.R. et al., 1986. Effective porosity of geologic materials. *Proceedings of the Twelfth Annual Research Symposium*, U.S. Environmental Protection Agency. EPA/600/9-86:21-8.
- Plummer, L.N., Busby, J.F., Lee, R.W., and Hanshaw, B.B., 1990. Geochemical modelling of the Madison aquifer in parts of Montana, Wyoming, and South Dakota. *Water Resources Research*, **26**, 1981-2014.
- Plummer, L.N., Prestemon, E.C. and Parkhurst, D.L., 1994. An interactive code (NETPATH) for modeling net geochemical reactions along a flow path, Version 2.0. *U.S. Geological Survey Water-Resources Investigations Report 94-4169*.
- Postma, D., 1993. The reactivity of iron oxides in sediments: a kinetic approach. *Geochimica et Cosmochimica Acta*, **57**, 5027-5034.
- Postma, D. and Jakobsen, R., 1996. Redox zonation: Equilibrium constraints on the Fe(III)/SO₄-reduction interface. *Geochimica et Cosmochimica Acta*, **60(17)**, 3169-3175.
- Puls, R.W. and Barcelona, M.J., 1996. Low-Flow (Minimal Drawdown) Groundwater Sampling Procedures. US EPA Ground Water Issue publication EPA/540/S-95/504.
- Qasim, S.R., and Chiang, W., 1994. Sanitary Landfill Leachate: Generation, Control and Treatment. Technomic Publishing Company, Lancaster, Pennsylvania, 323pp.
- Rickwood, P. C., 1998. The Botany Basin: a convenient definition. *In*: McNally G. H. and Jankowski, J. (Eds.), *Environmental Geology of the Botany Basin, Collected Case Studies in Engineering Geology, Fourth Series, Environmental, Engineering and Hydrogeology Specialist Group (EEHSG) of the Geological Society of Australia*, 173-178.
- Rittmann, B.E., Banaszak, J.E., Cooke, A. and Rowe, R.K., 2003. Biogeochemical evaluation of mechanisms controlling CaCO_{3(s)} precipitation in landfill leachate-collection systems. *Journal of Environmental Engineering*, **129(8)**, 723-730.
- Roberts, P., 1996. Analysis of landfill leachate distribution at Astrolabe Park Landfill, Sydney, Australia. Honours Thesis, School of Geology, UNSW, pp. 159.
- Ronen, D., Magaritz, M., Gvirtzman, H. and Garner, W., 1987. Microscale chemical heterogeneity in groundwater. *Journal of Hydrology*, **92**, 173-178.

- Roy, P.S., 1983. Quaternary geology. In Herbert, C. (ed) *Geology of the Sydney 1:100,000 sheet 9130, New South Wales Geological Survey, Department of Mineral Resources*, 42-90.
- Rozanski, K., Araguás-Araguás, L. and Gonfiantini, R., 1993. Isotopic patterns in global precipitation, in P.K. Swart et al. (eds.) *Climate Change in Continental Isotopic Records*, Geophys. Monogr. Ser., 78, AGU Washington DC, 1-36.
- Rügge, K., Bjerg, P.L., and Christensen, T.H., 1995. Distribution of organic compounds from municipal solid waste in the groundwater downgradient of a landfill (Grindsted, Denmark). *Environmental Science and Technology*, **29**, 1395-1400.
- Schürmann, A., Schroth, M.H., Saurer, M., Bernasconi, S.M. and Zeyer, J., 2003. Nitrate-consuming processes in a petroleum-contaminated aquifer quantified using push-pull tests combined with ^{15}N isotope and acetylene-inhibition methods. *Journal of Contaminant Hydrology*, **66**, 59-77.
- Sheil, G., 1942. Investigation of the Botany Sand-Beds emergency water supply. Premier's Department, Sydney (unpublished).
- Smart, J.V., 1974. The geology, hydrogeology and groundwater chemistry of part of the Botany Basin, NSW. MSc thesis, University of Sydney.
- Smith, R.L., Howes, B.L. and Duff, J.L., 1991a. Denitrification in nitrate-contaminated groundwater: Occurrence in steep vertical geochemical gradients. *Geochimica et Cosmochimica Acta*, **55(76)**, 1815-1825.
- Smith, R.L., Harvey, R.W. and LeBlanc, D.R., 1991b. Importance of closely spaced vertical sampling in delineating chemical and microbiological gradients in groundwater studies. *Journal of Contaminant Hydrology*, **7**, 285-300.
- Smolski, B., 1994. Migration of landfill leachate through a sand aquifer at Astrolabe Park, Sydney. M.App.Sc. thesis, University of New South Wales (unpublished).
- SPSS, 2000. SPSS for Windows, student version, release 10.0.7. Copyright SPSS Inc.
- Standard, J.C., 1969. Hawkesbury Sandstone. *Journal of the Geological Society of Australia*, **16**, 407-415.
- Stumm, W. and Morgan, J.J., 1996. Aquatic Chemistry. John Wiley and Sons, Inc., New York, 1022pp.
- Suk, H. and Lee, K.K., 1999. Characterization of a ground water hydrochemical system through multivariate analysis: clustering into groundwater zones. *Ground Water*, **37(3)**, 358-366.

- Sudicky, E.A., Cherry, J.A. and Frind, E.O., 1983. Migration of contaminants in groundwater at a landfill: a case study, 4. A natural gradient tracer test. *Journal of Hydrology*, **63**, 81-108.
- Sudicky, E.A., 1986. A natural gradient experiment on solute transport in a sand aquifer: Spatial variability of hydraulic conductivity and its role in the dispersion process. *Water Resources Research*, **22**, 2069-2082.
- Surfer, 1999. Surface mapping system version 7.00. Copyright Golden Software Inc., Golden, Colorado.
- Swann, W.H.C., 1971. Pumping Test Report of Field Testing Well. Internal report, School of Civil Engineering, University of New South Wales.
- Thornton, S.F., Tellam, J.H. and Lerner, D.N., 2000. Attenuation of landfill leachate by UK Triassic sandstone aquifer materials, 1. Fate of inorganic pollutants in laboratory columns. *Journal of Contaminant Hydrology*, **43**, 327-354.
- Thorpe, N.J., 1953. The history of the Botany water supply. *Sydney Water Board Journal*, **3**, 74-86.
- Tompson, A.F.B., 1993. Numerical simulation of chemical migration in physically and chemically heterogeneous porous media. *Water Resources Research*, **29** (11), 3709-3726.
- Tompson, A.F.B., Schafer, A.L., Smith, R.W., 1996. Impacts of physical and chemical heterogeneity on contaminant transport in a sandy porous medium. *Water Resources Research*, **32** (4), 801-818.
- Tompson, A.F.B. and Jackson, K.J., 1996. Reactive transport in heterogeneous systems: An overview. In: P.C. Lichtner, C.I. Steefel, E.H. Oelkers (Eds.), P.H. Ribbe (Series Ed.) *Reviews in Mineralogy Vol. 34: Reactive Transport in Porous Media*, Chapter 6, Mineralogical Society of America, Washington DC, 269-310.
- Ulrich, G.A., Breit, G.N., Cozzarelli, I.M., and Suflita, J.M., 2003. Sources of sulfate supporting anaerobic metabolism in a contaminated aquifer. *Environmental Science and Technology*, **37**(6), 1093-1099.
- U.S. Environmental Protection Agency, 1992. Ground Water Issue: Behavior of Metals in Soils. EPA/540/S-92/018, Office of Research and Development; Office of SWER, Washington DC.
- U.S. Environmental Protection Agency, 2000. Guidance for Data Quality Assessment – Practical Methods for Data Analysis (QA/G-9), EPA/600/R-96/084, Office of Environmental Information.
- U.S. Environmental Protection Agency, 2001. Test Methods for Evaluating Solid Wastes, Third Edition. EPA SW-846, U.S. Environmental Protection Agency, Office of SWER, U.S. Government Printing Office: Washington D.C.

- Vadillo, I., Carrasco, F., Andreo, B., García de Torres, A., and Bosch, C., 1999. Chemical composition of landfill leachate in a karst area with a Mediterranean climate (Marbella, southern Spain). *Environmental Geology*, **4**, 326-332.
- van Breukelen, B.M., Röling, W.F.M., Groen, J., Griffioen, J. and van Verseveld, H.W., 2003. Biogeochemistry and isotope geochemistry of a landfill leachate plume. *Journal of Contaminant Hydrology*, **65(3-4)**, 245-268.
- van Breukelen, B.M. and Griffioen, J., 2004. Biogeochemical processes at the fringe of a landfill leachate pollution plume: potential for dissolved organic carbon, Fe(II), Mn(II), NH₄, CH₄ oxidation. *Journal of Contaminant Hydrology*, **73**, 181-205.
- van der Kamp, G., Luba, L.D., Cherry, J.A. and Maathuis, H., 1994. Field study of a long and very narrow contaminant plume. *Ground Water*, **32(6)**, 1008-1016.
- Wallis, G.R., 1967. Groundwater resources of the Sydney region. *New South Wales Geological Survey*, unpublished report GS **1967/126**.
- Ward, J.H., 1963. Hierarchal grouping to optimise an objective function. *Journal of the American Statistical Association*, **58(301)**, 236-244.
- Webb, E.K. and Anderson, M.P., 1996. Simulation of preferential flow in three-dimensional, heterogeneous conductivity fields with realistic internal architecture. *Water Resources Research*, **32 (3)**, 533-545.
- Webb, S.N. and Watson, K.K., 1979. Hydraulic behaviour of an unconfined aquifer. *Australian Water Resources Council*, **38**, 1-58.
- Weiss, M., Abbt-Braun, G., Frimmel, F.H., 1989. Humic-like substances from landfill leachates – characterization and comparison with terrestrial and aquatic humic substances. *Science of the Total Environment*, **81/82**, 343-352.
- Whiticar, M.J., Faber, E. and Schoell, M., 1986. Biogenic methane formation in marine and freshwater environments: CO₂ reduction vs acetate fermentation – isotope evidence. *Geochimica et Cosmochimica Acta*, **50**, 696-709.
- Williams, A.E., Lund, L.J., Johnson, A. and Kabala, Z.J., 1998. Natural and anthropogenic nitrate contamination of groundwater in a rural community, California. *Environmental Science and Technology*, **32**, 32-39.
- Xu, M. and Eckstein, Y., 1995. Use of Weighted Least-Squares Method in Evaluation of the Relationship Between Dispersivity and Scale. *Ground Water*, **33(6)**, 905-908.
- Yu, X.W., Jankowski, J. and Acworth, R.I., 1993. Hydrogeological properties of the Botany Sands aquifer, Sydney. In: *Proceedings of the 27th Newcastle Symposium on Advances in the Study of the Sydney Basin*, Newcastle, April 1993, 229-236.

- Yu, X.W., 1994. Study of physical and chemical properties of groundwater and surface water in the northern part of the Botany Basin, Sydney. PhD thesis, University of New South Wales (unpublished).
- Zeng, X. and Rasmussen, T.C., 2005. Lake Lanier water quality monitoring strategy. *Journal of Environmental Quality*, **34**, 1980-1991.
- Zenone, C., Donaldson, D.E., and Grunwaldt, J.J., 1975. Ground-water quality beneath solid-waste disposal sites at Anchorage, Alaska. *Ground Water*, **13**(2), 182-190.
- ZoBell, C.E., 1946. Studies on redox potential of marine sediments. *American Association of Petroleum Geologists Bulletin*, **30**, 447–450.

APPENDICES

Appendix A

Hvorslev Analysis

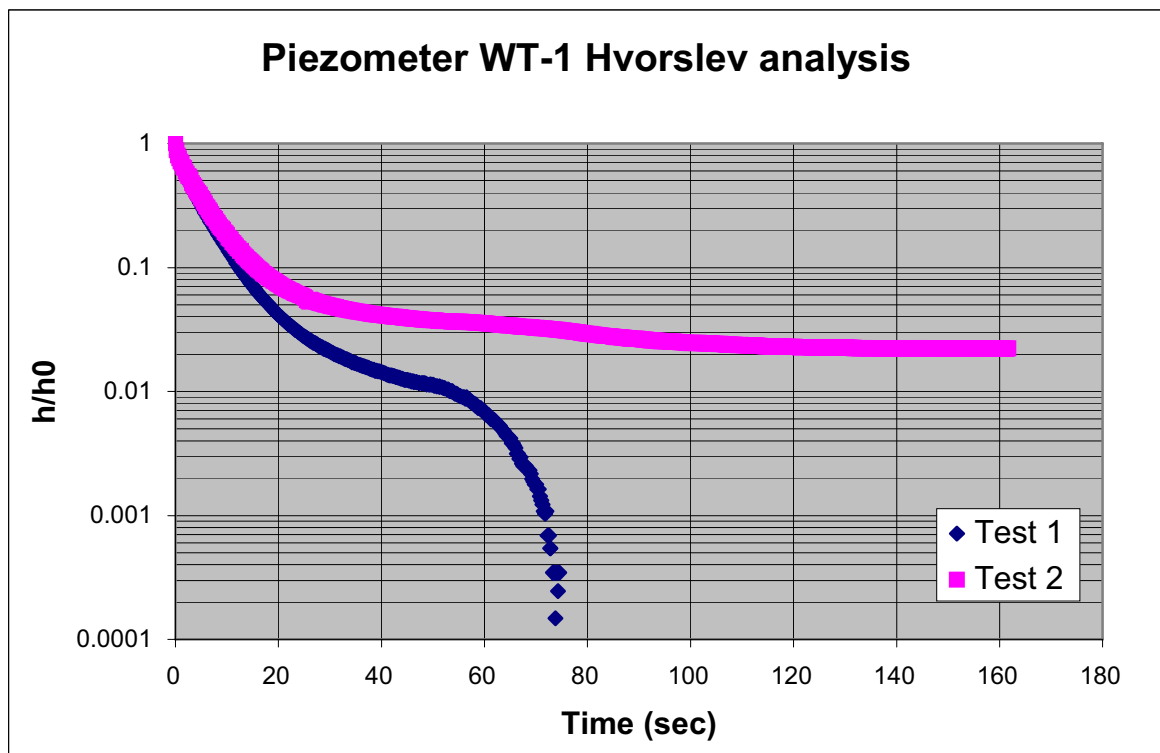
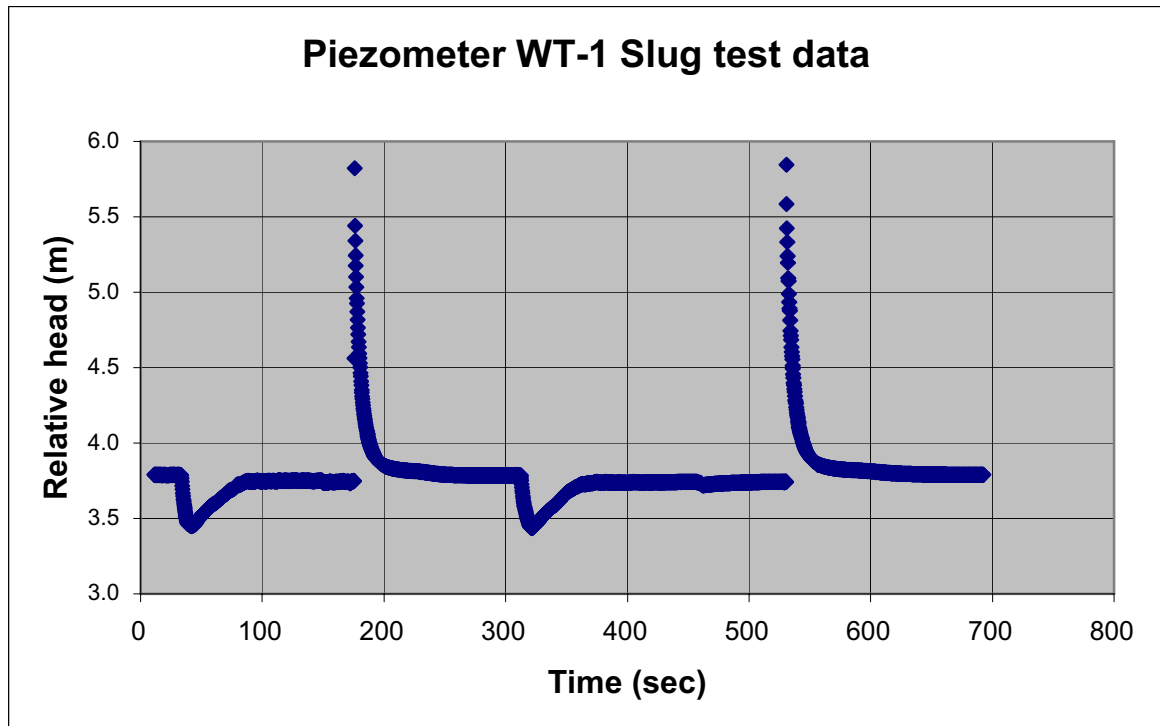


Figure 6.1 Slug test data and Hvorslev analysis for piezometer WT-1.

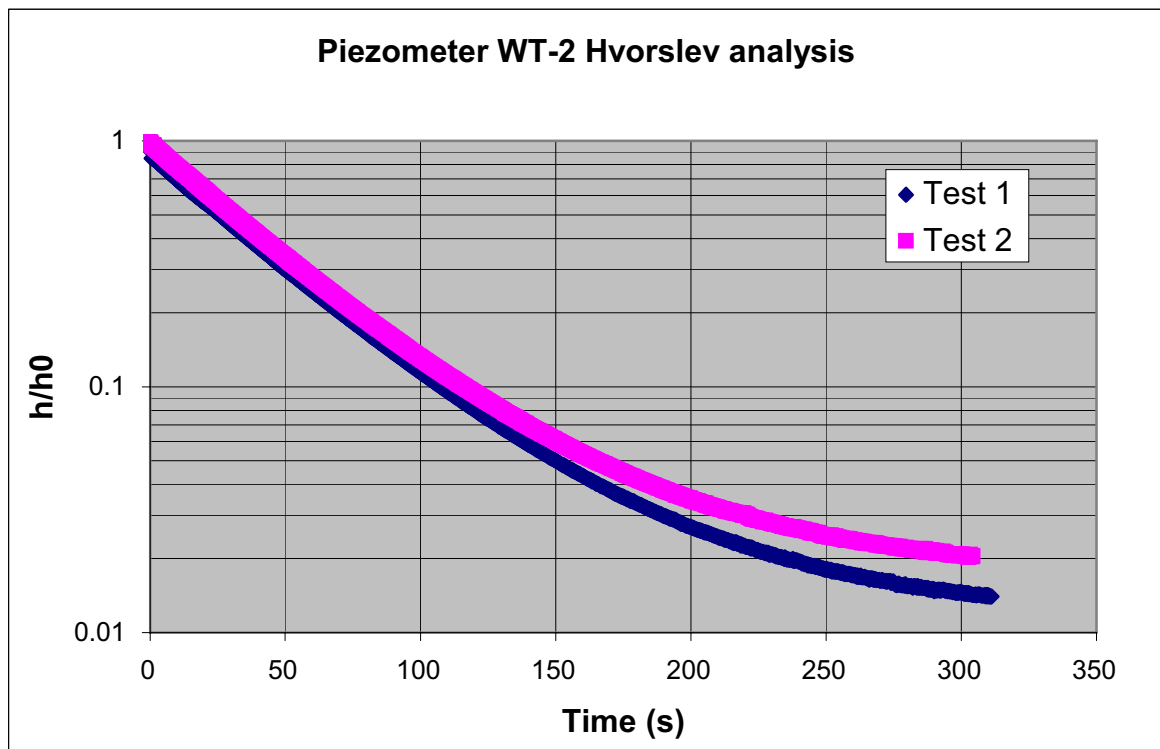
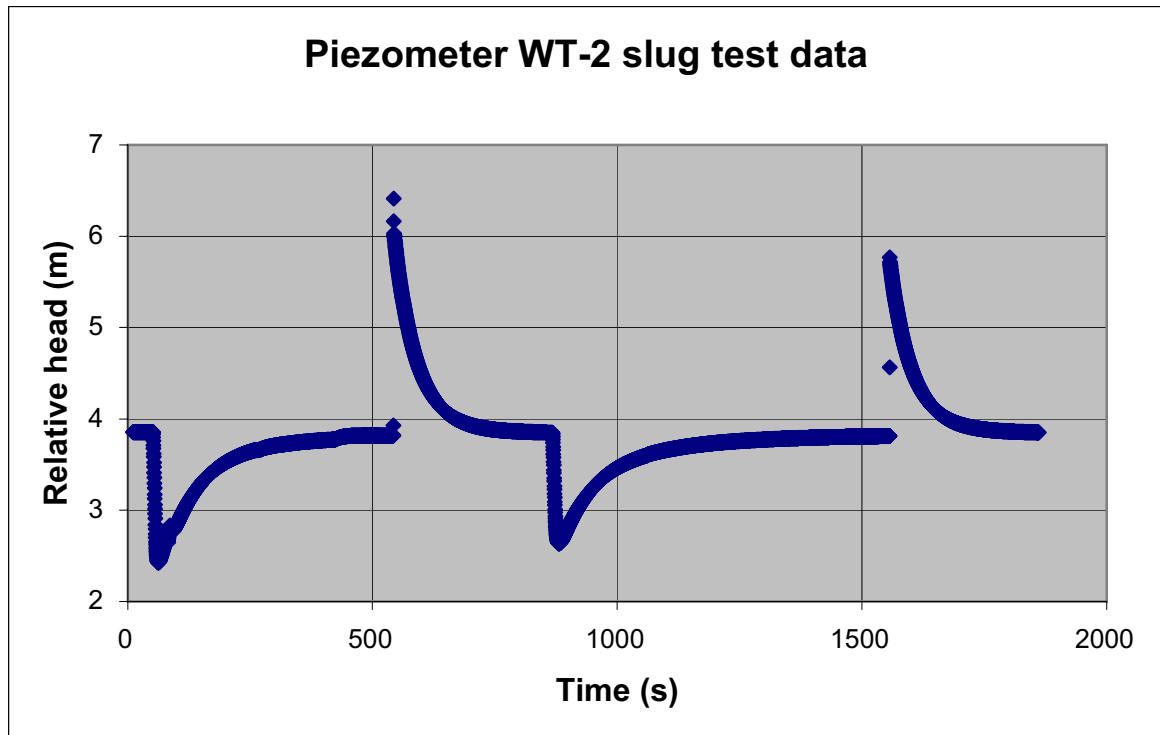


Figure 6.2 Slug test data and Hvorslev analysis for piezometer WT-2.

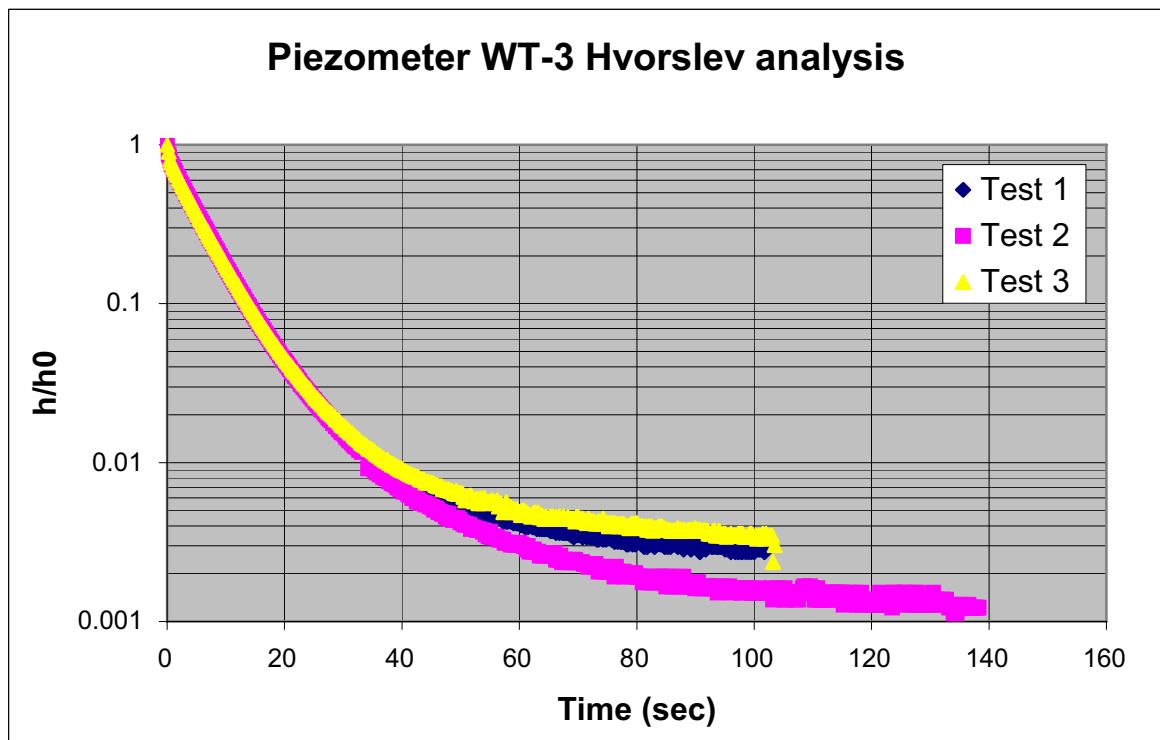
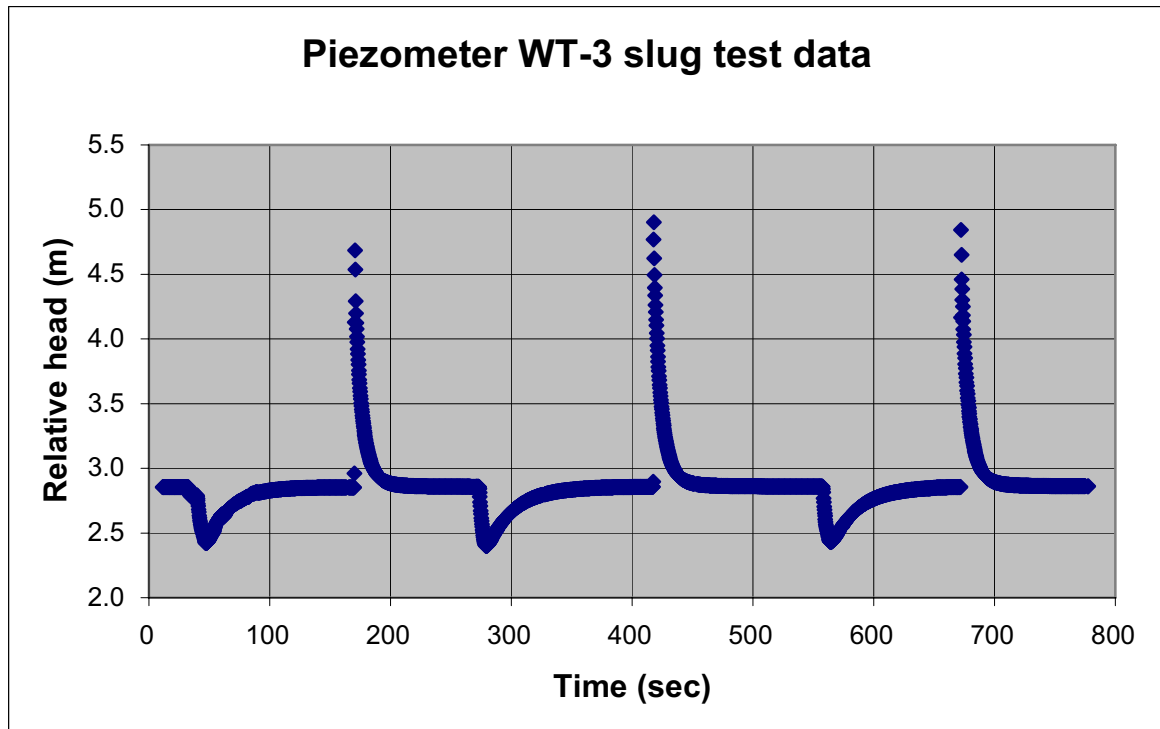


Figure 6.3 Slug test data and Hvorslev analysis for piezometer WT-3.

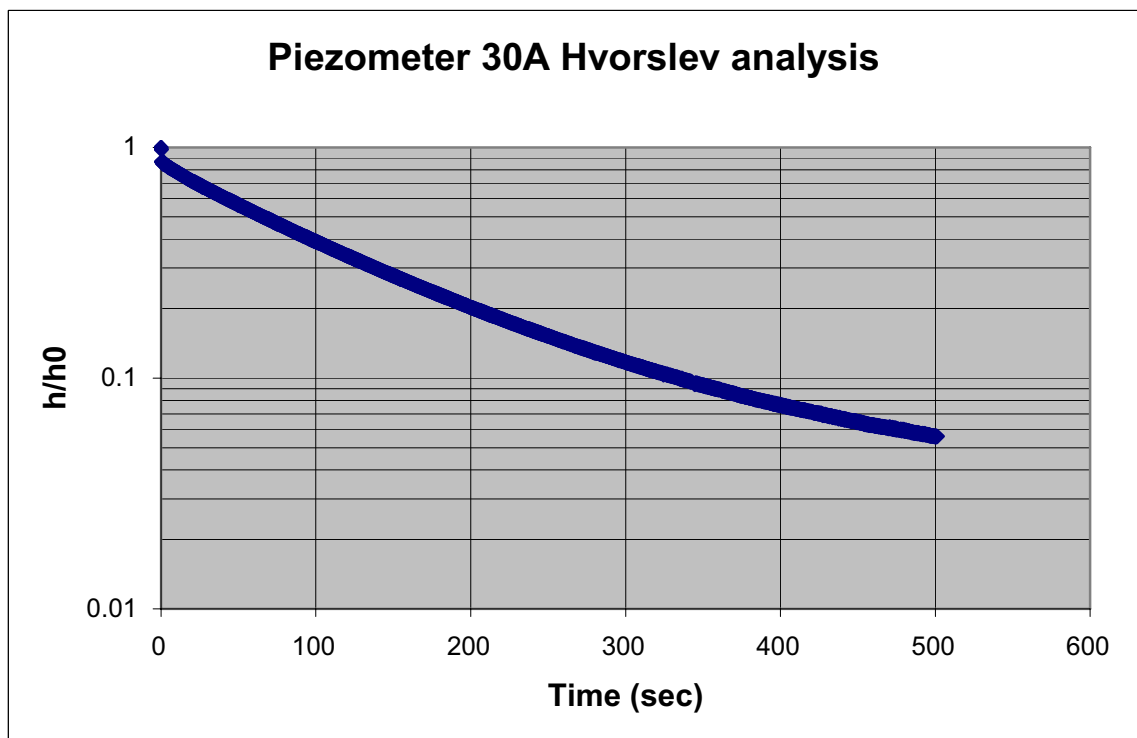
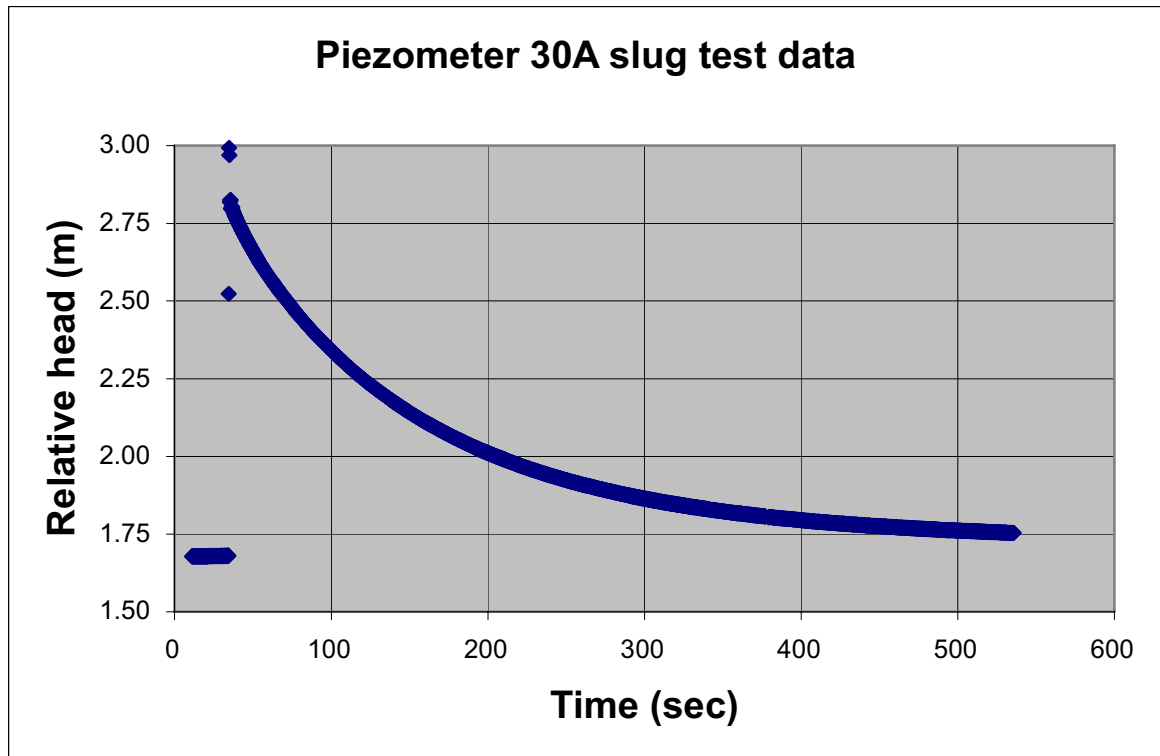


Figure 6.4 Slug test data and Hvorslev analysis for piezometer 30A.

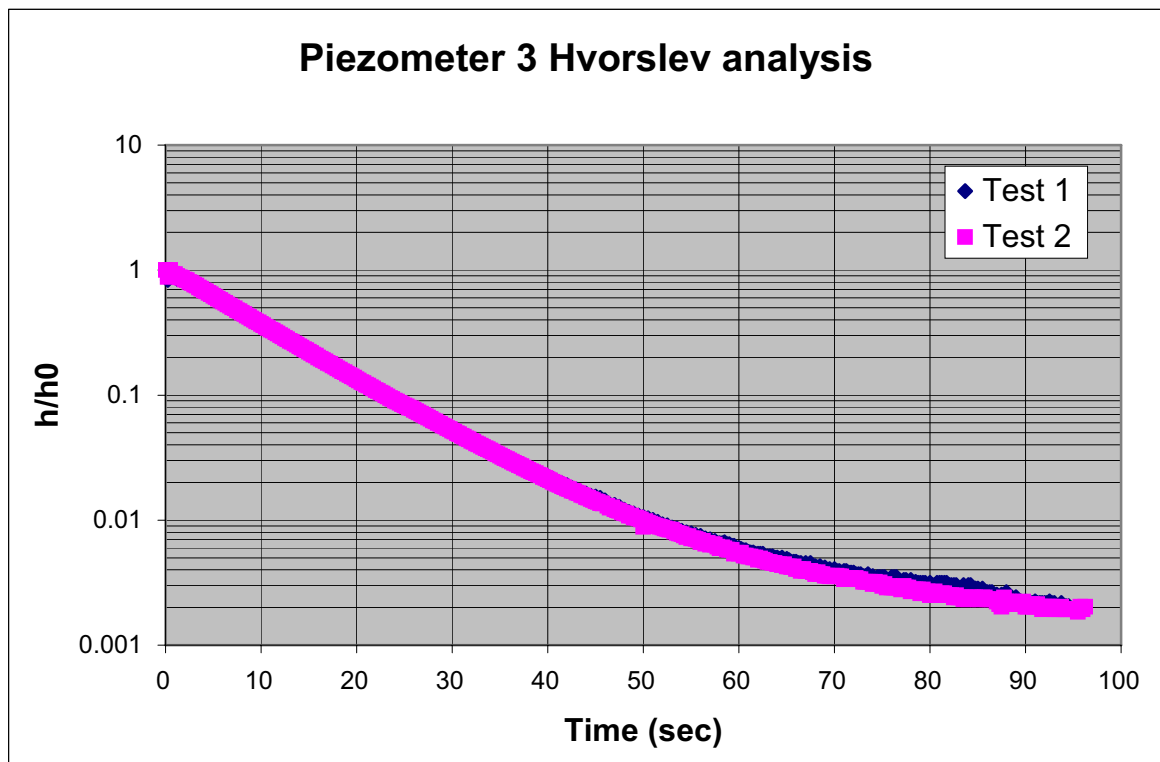
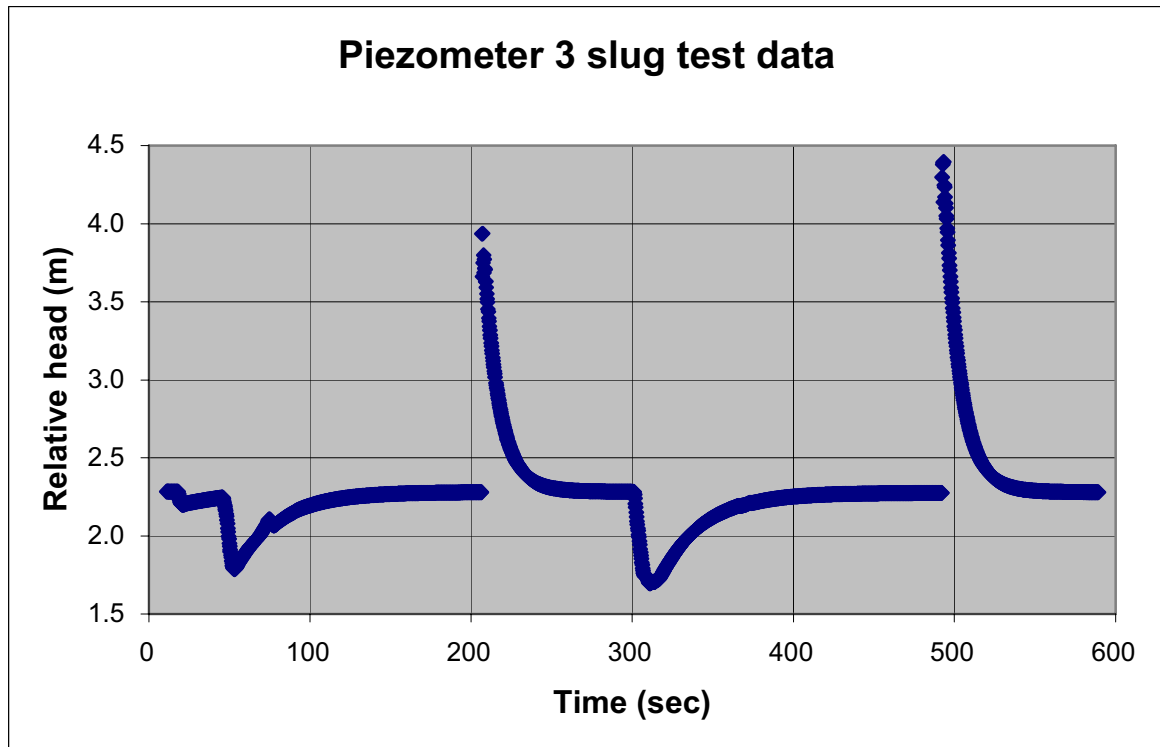


Figure 6.5 Slug test data and Hvorslev analysis for piezometer 3.

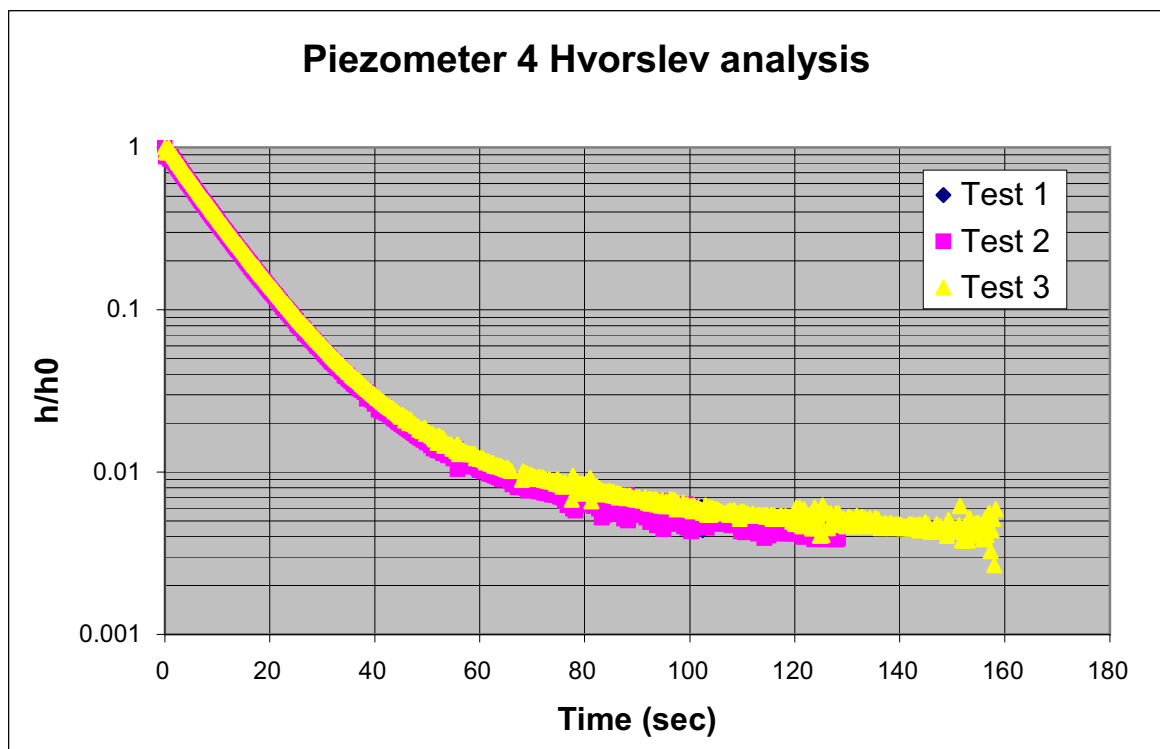
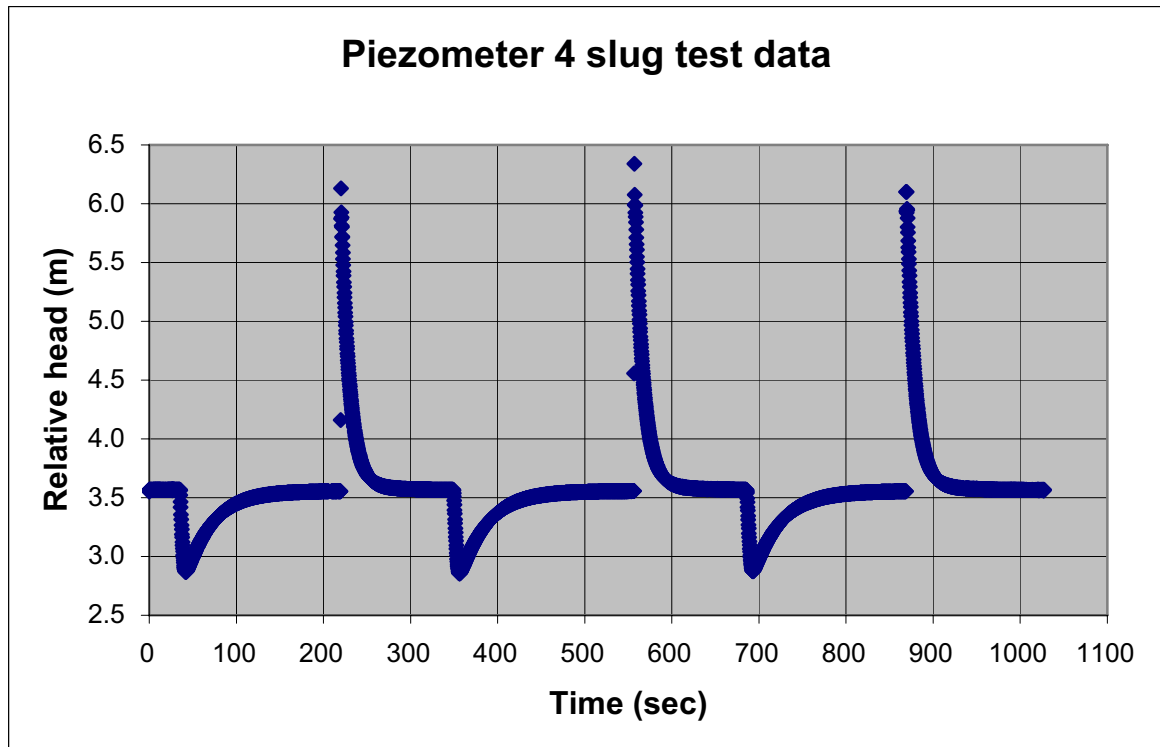


Figure 6.6 Slug test data and Hvorslev analysis for piezometer 4.

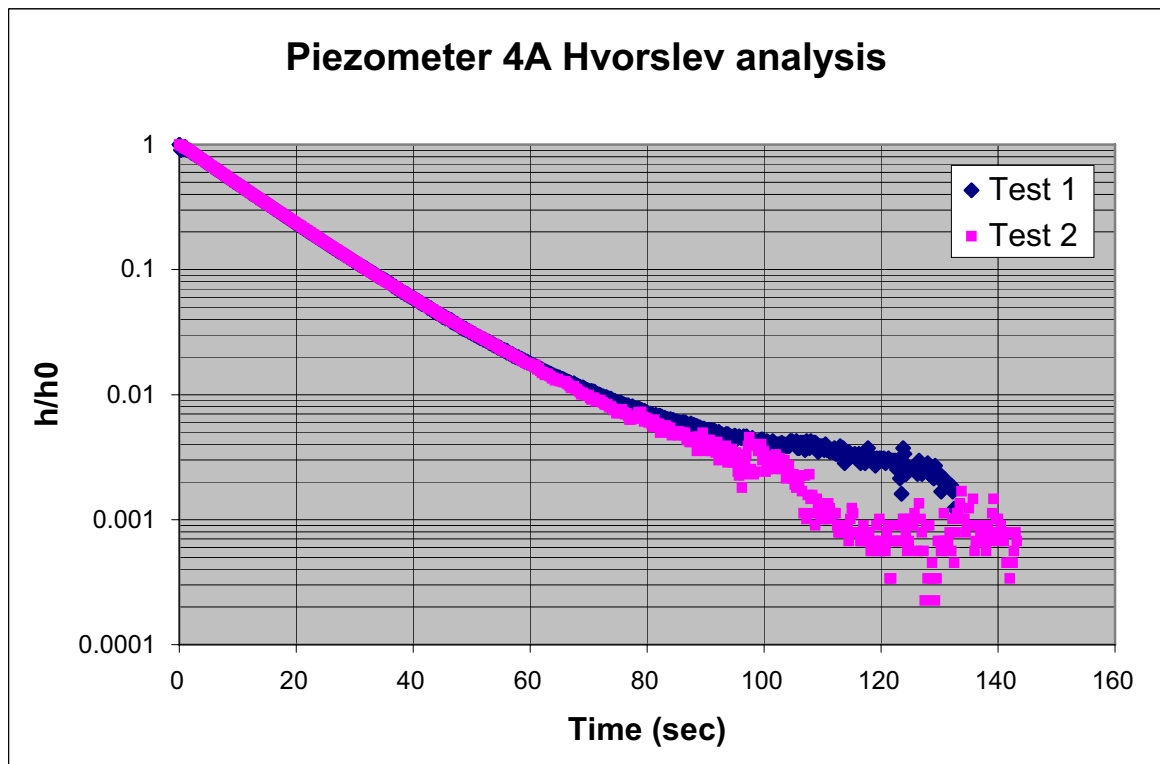
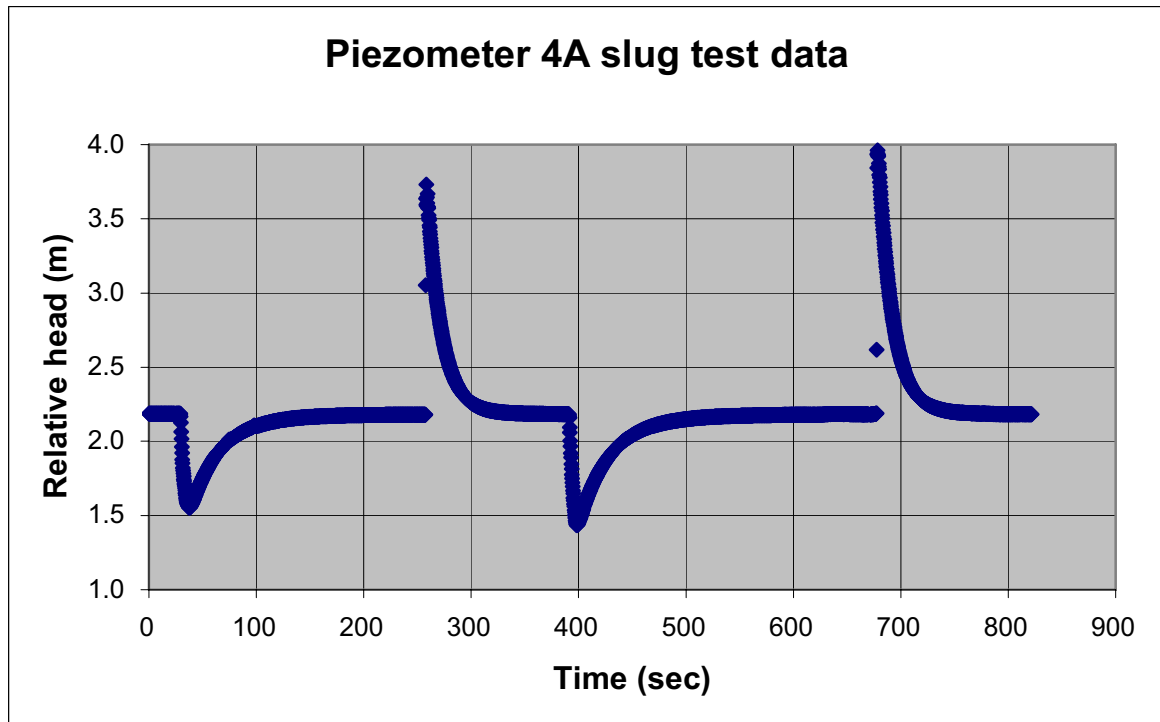


Figure 6.7 Slug test data and Hvorslev analysis for piezometer 4A.

Appendix B

Analytical Results Summary Tables

Table B1 Groundwater chemistry results for the bundled piezometers 3, 4, 4A, 6 and 10: July 2000 sampling event.

Bore ID	Depth (bgs ¹)	Elev. (AHD ²)	Sample Date	EC		Eh		Alk.			Cations														Anions						Uncharged			
				T(°C)	(μS/cm)	pH	(mV)	pe	O ₂	CO ₂	(meq/L)	Ca ²⁺	Na ⁺	Mg ²⁺	K ⁺	Fe ²⁺	Fe ³⁺	NH ₄ ⁺	Sr ²⁺	Mn ²⁺	Li ⁺	Zn ²⁺	Pb ²⁺	Cd ²⁺	Cu ²⁺	Al ³⁺	HCO ₃ ⁻	SO ₄ ²⁻	S ²⁻	Cl ⁻	NO ₃ ⁻	PO ₄ ³⁻	B	Ni
3	1.40	14.41	27-Jul-00	15.5	591	5.913	-157.3	-2.746	0.55	18.48	1.60	73.50	19.00	8.49	14.10	0.05	0.00	0.12	0.27	<0.002	<0.005	<0.07	<0.06	<0.005	0.01	<0.05	96.41	94.49	<0.001	39.99	52.36	0.27	0.05	<0.03
3	2.40	13.41	27-Jul-00	16.8	840	6.111	-161.9	-2.814	0.70	22.89	1.80	117.00	28.30	10.20	7.37	0.03	0.00	0.23	0.61	0.02	<0.005	<0.07	<0.06	<0.005	<0.007	<0.05	108.61	154.78	<0.001	59.48	78.76	0.58	0.12	<0.03
3	2.90	12.91	27-Jul-00	16.9	894	6.190	-205.2	-3.566	0.50	65.13	6.88	128.00	26.60	16.30	12.00	0.09	0.00	6.39	0.95	0.31	<0.005	<0.07	<0.06	<0.005	<0.007	<0.05	420.41	78.89	0.067	33.49	5.28	0.27	0.30	<0.03
3	3.40	12.41	27-Jul-00	17.2	1274	6.283	-220.5	-3.827	0.50	130.27	12.40	169.00	32.60	19.90	20.80	0.79	3.81	17.74	0.87	2.16	<0.005	<0.07	<0.06	<0.005	<0.007	<0.05	743.80	28.47	0.256	45.99	14.52	0.15	0.59	<0.03
3	3.90	11.91	27-Jul-00	17.4	1468	6.409	-109.2	-1.894	2.70	132.03	14.26	145.00	63.30	22.80	23.50	9.35	41.15	20.96	0.75	1.05	<0.005	<0.07	<0.06	<0.005	<0.007	<0.05	878.65	9.63	0.014	58.98	10.12	0.17	0.73	<0.03
3	4.40	11.41	27-Jul-00	18.3	1474	6.336	-176.0	-3.043	0.25	153.15	14.42	140.00	68.20	26.80	26.00	8.50	20.90	23.22	0.75	1.03	<0.005	<0.07	<0.06	<0.005	<0.007	<0.05	860.34	4.77	0.005	63.48	8.80	0.60	0.64	<0.03
3	4.90	10.91	27-Jul-00	DRY	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
3	5.40	10.41	27-Jul-00	15.8	1107	6.423	-99.4	-1.734	3.55	94.62	10.00	93.70	60.20	19.70	22.30	9.10	5.90	20.96	0.46	0.50	<0.005	<0.07	<0.06	<0.005	<0.007	<0.05	610.17	3.03	0.004	59.98	12.32	0.04	0.49	<0.03
3	5.90	9.91	27-Jul-00	17.7	984	6.307	-169.7	-2.940	0.65	83.62	8.38	85.10	60.20	17.40	20.50	9.30	7.50	15.16	0.37	0.58	<0.005	<0.07	<0.06	<0.005	<0.007	<0.05	513.76	2.85	0.004	56.48	11.88	0.19	0.54	<0.03
3	6.40	9.41	27-Jul-00	18.2	591	6.012	-211.6	-3.660	0.45	60.73	4.26	21.90	49.20	6.67	15.60	4.45	0.75	16.25	0.12	0.13	<0.005	<0.07	<0.06	<0.005	<0.007	0.08	255.66	1.93	0.067	48.39	10.56	0.12	0.47	<0.03
3	6.90	8.91	27-Jul-00	DRY	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
3	7.40	8.41	27-Jul-00	DRY	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
3	7.90	7.91	27-Jul-00	13.9	209	5.402	-181.2	-3.182	4.40	22.44	0.76	2.31	14.10	0.22	6.65	2.29	0.00	6.32	0.01	0.02	<0.005	<0.07	<0.06	<0.005	<0.007	0.10	46.37	10.68	1.380	31.99	24.64	0.11	0.16	<0.03
3	8.40	7.41	27-Jul-00	15.1	205	5.234	-185.3	-3.240	2.20	26.41	0.62	1.65	19.10	0.50	8.97	2.07	0.23	6.64	0.01	0.02	<0.005	<0.07	<0.06	<0.005	<0.007	0.06	37.22	19.83	1.275	30.49	16.72	1.22	0.22	<0.03
3	8.90	6.91	26-Jul-00	15.7	198	5.253	-171.9	-2.999	4.70	24.21	0.78	0.83	14.40	0.40	6.80	1.43	0.00	5.87	0.01	0.00	<0.005	<0.07	<0.06	<0.005	<0.007	<0.05	36.61	14.58	0.910	30.99	11.44	1.43	0.16	<0.03
3	9.40	6.41	26-Jul-00	17.4	194	5.129	-174.2	-3.022	2.80	20.24	0.44	0.94	17.40	0.57	7.52	0.75	0.17	5.61	0.01	<0.002	<0.005	<0.07	<0.06	<0.005	<0.007	<0.05	27.46	19.17	0.432	31.99	7.48	0.14	0.19	<0.03
3	10.40	5.41	26-Jul-00	18.3	199	4.991	-175.9	-3.042	1.00	24.65	0.44	0.87	19.60	0.63	8.50	0.44	0.12	0.45	0.01	<0.002	<0.005	<0.07	<0.06	<0.005	<0.007	<0.07	25.63	22.86	0.040	31.43	6.16	0.19	0.22	<0.03
3	11.40	4.41	26-Jul-00	16.9	192	4.897	-153.1	-2.661	2.15	21.56	0.30	0.97	18.60	0.77	7.18	0.63	0.17	5.68	0.01	<0.002	<0.005	<0.07	<0.06	<0.005	<0.007	0.07	18.92	23.34	0.051	32.36	6.16	0.24	0.18	<0.03
3	12.40	3.41	26-Jul-00	18.9	182	4.938	-194.1	-3.350	0.40	21.56	0.30	4.74	17.80	2.66	6.80	0.33	0.27	1.84	0.02	0.01	<0.005	<0.07	<0.06	<0.005	<0.007	<0.05	17.69	23.97	0.299	31.49	9.24	0.04	0.10	<0.03
3	13.40	2.41	26-Jul-00	18.9	181	4.996	-182.6	-3.151	1.10	19.36	0.28	6.60	18.00	3.08	6.44	0.18	0.19	0.34	0.03	0.02	<0.005	<0.07	<0.06	<0.005	<0.007	<0.05	17.08	25.35	0.181	31.49	7.04	0.04	0.08	<0.03
3	14.40	1.41	26-Jul-00	18.8	204	5.248	-192.9	-3.330	0.85	20.24	0.48	8.66	16.90	3.00	5.52	0.12	0.26	0.43	0.04	0.03	<0.005	<0.07	<0.06	<0.005	<0.007	<0.05	29.29	22.02	0.106	33.99	7.48	0.61	0.06	<0.03
3	15.40	0.41	26-Jul-00	17.3	209	5.360	-182.9	-3.173	1.00	15.84	0.52	10.20	19.90	3.40	5.71	0.04	0.13	0.53	0.05	0.04	<0.005	<0.07	<0.06	<0.005	<0.007	<0.05	31.73	24.00	0.027	33.92	9.68	0.34	0.07	<0.03
3	16.40	-0.59	25-Jul-00	18.8	199	5.507	-214.3	-3.699	0.40	17.16	0.52	8.09	19.60	2.82	3.54	0.03	0.16	0.01	0.05	0.01	<0.005	<0.07	<0.06	<0.005	<0.007	<0.05	31.73	19.17	0.002	30.99	8.36	0.63	0.04	<0.03
3	17.40	-1.59	25-Jul-00	18.6	215	5.436	-194.6	-3.361	2.65	14.52	0.36	8.94	21.60	3.24	5.85	0.04	0.12	<0.01	0.01	0.01	<0.005	<0.07	<0.06	<0.005	<0.007	<0.05	21.97	21.54	0.024	29.99	24.64	0.04	0.04	<0.03
3	18.40	-2.59	25-Jul-00	18.6	212	5.193	-188.1	-3.249	0.90	12.76	0.24	8.58	20.50	3.76	5.84	<0.01	0.43	0.01	0.02	0.01	<0.005	<0.07	<0.06	<0.005	0.01	<0.05	14.64	19.11	0.001	30.99	42.68	0.09	0.04	<0.03
3	19.40	-3.59	25-Jul-00	18.6	227	4.423	-137.2	-2.370	0.65	22.44	0.12	9.75	19.40	4.28	5.75	0.03	0.06	0.03	0.02	0.02	<0.005	<0.07	<0.06	<0.005	0.01	0.06	7.32	27.45	0.001	31.43	38.28	0.07	0.05	<0.03
3	20.40	-4.59	27-Jul-00	19.2	221	4.430	-151.0	-2.603	0.05	29.93	0.14	6.03	12.50	2.92	4.41	0.07	0.42	0.04	0.02	0.01	<0.005	<0.07	<0.06	<0.005	<0.007	0.08	9.76	15.39	0.003	32.99	35.20	0.06	0.04	<0.03
4	1.84	15.481	6-Jul-00	DRY	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
4	2.34	14.981	6-Jul-00	DRY	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
4	2.84	14.481	6-Jul-00	DRY	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--

Table B1 (cont.) Groundwater chemistry results for bundled piezometers 3, 4, 4A, 6 and 10: July 2000 sampling event.

Bore ID	Depth (bgs ¹)	Elev. (AHD ²)	Sample Date	EC		Eh		Alk.			Cations													Anions						Uncharged				
				T(°C)	(µS/cm)	pH	(mV)	pe	O ₂	CO ₂	(meq/L)	Ca ²⁺	Na ⁺	Mg ²⁺	K ⁺	Fe ²⁺	Fe ³⁺	NH ₄ ⁺	Sr ²⁺	Mn ²⁺	Li ⁺	Zn ²⁺	Pb ²⁺	Cd ²⁺	Cu ²⁺	Al ³⁺	HCO ₃ ⁻	SO ₄ ²⁻	S ²⁻	Cl ⁻	NO ₃ ⁻	PO ₄ ³⁻	B	Ni
4	14.34	2.981	5-Jul-00	17.4	216	5.092	-50.0	-0.867	3.25	19.80	0.17	9.40	17.40	3.43	6.88	0.19	0.00	0.05	0.03	0.00	<0.005	<0.07	<0.06	<0.005	<0.007	<0.05	10.98	26.13	<0.001	33.95	18.04	0.03	0.05	<0.03
4	15.34	1.981	5-Jul-00	18.7	210	4.862	-100.5	-1.736	0.75	22.89	0.18	7.54	15.30	3.27	6.35	0.04	0.12	0.06	0.03	0.00	<0.005	<0.07	<0.06	<0.005	<0.007	<0.05	12.20	28.80	0.007	32.60	11.00	0.05	0.05	<0.03
4	16.34	0.981	5-Jul-00	18.5	201	4.949	-56.9	-0.983	2.45	18.48	0.20	7.36	14.10	2.86	5.55	0.05	0.20	0.03	0.03	0.00	<0.005	<0.07	<0.06	<0.005	0.01	<0.05	11.59	24.75	0.002	33.59	15.40	0.12	0.04	<0.03
4	18.34	-1.019	5-Jul-00	18.7	219	4.670	-58.1	-1.003	2.15	14.52	0.14	10.50	18.50	3.31	5.76	0.01	0.18	0.10	0.04	0.00	<0.005	<0.07	<0.06	<0.005	0.01	0.23	8.54	29.61	0.002	35.00	27.28	0.09	0.05	<0.03
4	20.34	-3.019	27-Jul-00	19.1	220	4.344	-171.5	-2.957	0.05	32.13	0.10	6.03	12.70	2.30	2.70	0.01	0.26	<0.01	0.02	0.01	<0.005	<0.07	<0.06	<0.005	<0.007	0.37	6.10	18.06	<0.001	34.99	21.56	0.02	0.04	<0.03
4A	2.15	13.17	28-Jul-00	14.7	598	5.741	-172.6	-3.022	0.65	99.46	4.08	15.20	49.30	12.70	22.10	5.30	0.00	12.90	0.07	0.05	<0.005	<0.07	<0.06	<0.005	0.01	0.07	253.22	2.22	0.023	59.03	14.08	0.36	0.42	<0.03
4A	2.65	12.67	28-Jul-00	15.8	649	5.736	-187.3	-3.267	0.35	130.27	4.30	15.80	52.30	13.20	23.40	2.45	0.00	16.77	0.06	0.05	<0.005	<0.07	<0.06	<0.005	0.01	0.10	259.32	2.38	0.023	61.75	9.24	0.35	0.45	<0.03
4A	3.15	12.17	28-Jul-00	16.3	650	5.712	-184.1	-3.205	0.30	129.39	4.50	14.60	50.50	12.80	22.80	1.36	0.00	17.09	0.06	0.04	<0.005	<0.07	<0.06	<0.005	<0.007	0.11	278.24	2.41	0.022	61.62	7.48	1.05	0.41	<0.03
4A	3.65	11.67	28-Jul-00	16.8	668	5.589	-195.5	-3.398	0.35	124.99	4.62	14.80	42.80	11.70	19.80	1.09	0.00	16.13	0.06	0.04	<0.005	<0.07	<0.06	<0.005	<0.007	0.06	279.46	2.03	0.026	60.75	4.84	0.26	0.38	<0.03
4A	4.15	11.17	28-Jul-00	17.3	770	5.960	-191.9	-3.330	0.45	105.62	6.20	44.80	46.00	19.70	23.60	1.95	0.13	16.77	0.15	0.18	<0.005	<0.07	<0.06	<0.005	<0.007	<0.05	373.42	2.77	0.005	53.74	7.48	0.14	0.50	<0.03
4A	4.65	10.67	28-Jul-00	18.2	866	6.138	-178.4	-3.086	0.45	88.02	7.50	72.30	39.50	22.10	21.40	3.20	0.18	17.03	0.21	0.31	<0.005	<0.07	<0.06	<0.005	<0.007	<0.05	454.58	3.00	0.004	48.01	7.92	0.55	0.55	<0.03
4A	5.15	10.17	28-Jul-00	18.4	831	6.264	-181.2	-3.133	0.25	80.98	7.42	72.40	36.30	17.60	17.30	5.90	8.00	16.77	0.30	0.41	<0.005	<0.07	<0.06	<0.005	<0.007	<0.05	439.93	2.65	<0.001	44.19	11.44	0.31	0.57	<0.03
4A	5.65	9.67	28-Jul-00	18.4	828	5.896	-163.9	-2.833	0.50	68.66	7.36	74.10	42.00	16.40	16.70	6.35	4.05	15.80	0.32	0.43	<0.005	<0.07	<0.06	<0.005	<0.007	<0.05	444.20	2.56	0.008	47.49	10.12	1.50	0.66	<0.03
4A	6.15	9.17	28-Jul-00	19.2	723	6.176	-146.3	-2.522	3.05	69.54	5.98	46.90	44.50	7.83	13.90	4.50	0.34	14.51	0.18	0.17	<0.005	<0.07	<0.06	<0.005	<0.007	<0.05	367.32	1.73	0.005	49.39	8.36	0.37	0.66	<0.03
4A	6.65	8.67	28-Jul-00	18.7	624	6.019	-203.7	-3.517	0.15	73.06	4.70	31.50	48.00	9.77	15.90	2.03	0.01	12.58	0.12	0.10	<0.005	<0.07	<0.06	<0.005	<0.007	<0.05	280.07	1.72	0.015	49.48	5.28	0.27	0.71	<0.03
4A	7.15	8.17	28-Jul-00	18.8	634	6.164	-199.7	-3.447	0.90	65.13	5.04	42.10	45.70	11.50	15.90	3.00	0.30	14.84	0.16	0.15	<0.005	<0.07	<0.06	<0.005	<0.007	<0.05	303.87	1.87	0.012	49.98	7.92	0.09	0.70	<0.03
4A	7.65	7.67	28-Jul-00	18.5	237	5.754	-195.1	-3.371	3.50	23.77	1.06	14.20	28.40	2.64	8.36	2.70	1.40	3.19	0.07	0.09	<0.005	<0.07	<0.06	<0.005	0.01	0.07	65.29	21.96	2.250	36.99	9.68	0.07	0.14	<0.03
4A	8.15	7.17	28-Jul-00	18.4	220	5.447	-192.2	-3.323	1.05	24.65	0.58	7.66	22.10	2.63	6.45	2.21	0.26	2.30	0.04	0.04	<0.005	<0.07	<0.06	<0.005	0.01	<0.05	38.44	23.37	1.400	36.56	16.28	0.04	0.09	<0.03
4A	8.65	6.67	28-Jul-00	17.8	214	5.362	-188.0	-3.257	0.90	22.89	0.50	7.26	20.50	2.73	6.06	1.73	0.28	1.69	0.04	0.03	<0.005	<0.07	<0.06	<0.005	<0.007	0.05	31.12	25.29	0.875	32.99	12.32	0.03	0.08	<0.03
4A	9.15	6.17	31-Jul-00	17.0	206	5.113	-225.9	-3.924	0.40	31.69	0.42	6.42	20.10	2.61	6.01	1.07	0.04	1.44	0.03	0.01	<0.005	<0.07	<0.06	<0.005	0.01	<0.05	26.24	25.44	0.485	33.42	11.44	0.25	0.07	<0.03
4A	9.65	5.67	31-Jul-00	DRY	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
4A	10.15	5.17	31-Jul-00	14.9	188	4.642	-174.1	-3.046	1.30	34.33	0.20	2.20	18.70	1.18	5.13	0.27	0.00	2.09	0.01	<0.002	<0.005	<0.07	<0.06	<0.005	<0.007	<0.05	12.20	20.13	0.087	33.49	7.04	0.40	0.06	<0.03
4A	11.15	4.17	31-Jul-00	17.2	188	4.716	-188.9	-3.279	0.50	26.41	0.18	3.02	21.40	1.74	6.08	0.21	0.08	1.73	0.02	<0.002	<0.005	<0.07	<0.06	<0.005	<0.007	<0.05	10.98	25.47	0.120	32.99	4.40	0.10	0.07	<0.03
4A	12.15	3.17	31-Jul-00	15.7	188	5.098	-181.9	-3.174	1.70	27.29	0.32	8.82	16.20	3.46	4.51	0.19	0.01	0.15	0.03	0.01	<0.005	<0.07	<0.06	<0.005	<0.007	<0.05	18.31	25.26	0.321	29.43	7.92	0.10	0.05	<0.03
4A	13.15	2.17	31-Jul-00	17.2	188	4.786	-178.1	-3.091	0.60	21.12	0.18	7.83	16.80	3.18	5.31	0.01	0.00	0.05	0.03	0.01	<0.005	<0.07	<0.06	<0.005	<0.007	0.06	18.92	29.10	0.006	30.49	9.68	0.28	0.05	<0.03
4A	14.15	1.17	31-Jul-00	17.8	188	4.644	-187.2	-3.242	0.55	21.12	0.12	8.24	16.70	3.11	4.86	0.02	0.00	0.13	0.03	0.01	<0.005	<0.07	<0.06	<0.005	<0.007	0.08	7.93	28.14	0.005	30.31	12.76	0.04	0.05	<0.03
4A	15.15	0.17	31-Jul-00	16.9	228	5.294	-169.4	-2.943	3.60	17.60	0.32	14.30	16.30	4.10	3.45	0.16	0.04	0.15	0.03	0.01	<0.005	<0.07	<0.06	<0.005	<0.007	<0.05	23.19	22.83	0.075	31.49	37.84	0.06	0.05	<0.03
4A	16.15	-0.83	31-Jul-00	18.8	264	5.870	-198.5	-3.426	1.60	12.32	0.58	20.00	17.70	4.76	4.93	0.01	0.00	<0.01	0.08	<0.002	<0.005	<0.07	<0.06	<0.005	<0.007	<0.05	34.78	16.32	0.007	33.99	46.20	0.3		

Table B1 (cont.) Groundwater chemistry results for bundled piezometers 3, 4, 4A, 6 and 10: July 2000 sampling event.

Bore ID	Depth (bgs ¹)	Elev. (AHD ²)	Sample Date	EC		Eh		Alk.			Cations													Anions						Uncharged				
				T(°C)	(µS/cm)	pH	(mV)	pe	O ₂	CO ₂	(meq/L)	Ca ²⁺	Na ⁺	Mg ²⁺	K ⁺	Fe ²⁺	Fe ³⁺	NH ₄ ⁺	Sr ²⁺	Mn ²⁺	Li ⁺	Zn ²⁺	Pb ²⁺	Cd ²⁺	Cu ²⁺	Al ³⁺	HCO ₃ ⁻	SO ₄ ²⁻	S ²⁻	Cl ⁻	NO ₃ ⁻	PO ₄ ³⁻	B	Ni
6	11.18	4.48	7-Jul-00	17.9	484	5.757	-188.6	-3.266	1.95	292.23	3.10	6.60	43.90	3.04	15.40	4.35	0.66	27.09	0.03	0.03	<0.005	<0.07	<0.06	<0.005	<0.007	<0.05	189.15	9.54	0.464	51.45	14.96	0.94	0.15	<0.03
6	11.68	3.98	7-Jul-00	DRY	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
6	12.18	3.48	7-Jul-00	18.6	825	5.943	-171.7	-2.966	1.75	350.32	5.84	3.50	66.90	2.58	31.20	2.60	0.39	28.06	0.02	0.02	<0.005	<0.07	<0.06	<0.005	<0.007	<0.05	359.39	2.85	0.005	69.75	5.72	0.19	0.26	<0.03
6	12.68	2.98	7-Jul-00	20.5	836	6.154	-164.4	-2.822	1.55	274.62	6.26	5.65	59.60	8.80	32.80	3.21	0.47	35.80	0.03	0.02	<0.005	<0.07	<0.06	<0.005	<0.007	<0.05	383.80	1.30	0.006	59.79	5.28	3.50	0.28	<0.03
6	13.18	2.48	7-Jul-00	22.3	749	6.065	-145.2	-2.477	1.90	257.02	5.88	9.32	54.10	12.60	27.20	7.30	4.60	44.18	0.03	0.02	<0.005	<0.07	<0.06	<0.005	<0.007	<0.05	357.56	1.24	0.020	56.96	10.56	1.60	0.31	<0.03
6	13.68	1.98	7-Jul-00	19.0	675	5.896	-197.5	-3.407	0.40	320.39	4.88	5.74	53.20	8.38	23.40	6.40	1.58	42.25	0.03	0.02	<0.005	<0.07	<0.06	<0.005	<0.007	<0.05	298.37	1.50	0.006	58.92	10.12	1.19	0.26	<0.03
6	14.18	1.48	7-Jul-00	21.6	659	6.205	-157.7	-2.697	2.00	274.62	4.78	6.22	54.40	9.39	23.20	7.15	0.45	37.73	0.03	0.03	<0.005	<0.07	<0.06	<0.005	<0.007	<0.05	292.88	1.78	0.005	57.38	5.28	0.09	0.28	<0.03
6	15.18	0.48	7-Jul-00	19.5	664	5.899	-189.6	-3.265	0.75	355.60	4.74	5.40	52.00	9.92	22.40	8.15	0.00	33.54	0.03	0.04	<0.005	<0.07	<0.06	<0.005	<0.007	<0.05	289.83	1.48	0.005	58.06	7.92	0.08	0.25	<0.03
6	16.18	-0.52	7-Jul-00	17.4	667	6.534	-144.8	-2.512	3.15	246.45	4.80	7.03	52.80	9.24	23.20	6.95	1.84	37.09	0.04	0.05	<0.005	<0.07	<0.06	<0.005	<0.007	<0.05	291.66	2.29	0.009	57.18	9.68	2.50	0.27	<0.03
6	18.18	-2.52	7-Jul-00	18.2	300	5.574	-210.6	-3.643	0.35	159.14	1.58	3.10	32.50	3.79	7.92	6.35	5.05	8.77	0.02	0.03	<0.005	<0.07	<0.06	<0.005	<0.007	<0.05	96.41	17.31	0.400	40.91	12.76	0.06	0.13	<0.03
6	19.18	-3.52	27-Jul-00	18.8	296	5.556	-235.5	-4.066	0.00	85.38	0.54	2.52	26.30	3.10	7.91	3.29	4.25	8.13	0.02	0.04	<0.005	<0.07	<0.06	<0.005	0.01	0.05	33.56	23.79	0.014	37.41	7.92	0.22	0.12	<0.03
10	3.0	13.63	29-Jul-00	20.8	938	6.287	-149.6	-2.565	0.15	142.59	8.70	120.00	22.20	11.30	5.68	1.76	37.64	5.74	0.52	1.29	<0.005	<0.07	<0.06	<0.005	<0.007	<0.05	523.53	19.53	0.007	23.66	13.20	0.25	0.41	<0.03
10	4.0	12.63	29-Jul-00	20.9	1653	6.597	-185.5	-3.179	0.10	155.79	15.82	111.00	77.40	19.80	39.40	8.55	15.35	39.67	0.61	0.30	<0.005	<0.07	<0.06	<0.005	0.01	<0.05	948.21	5.19	0.003	75.62	9.24	0.40	0.63	<0.03
10	5.0	11.63	29-Jul-00	20.9	554	6.113	-190.4	-3.263	0.05	64.25	4.08	15.70	44.70	5.41	17.30	6.05	2.32	14.84	0.08	0.09	<0.005	<0.07	<0.06	<0.005	<0.007	0.08	245.29	3.63	0.006	42.88	10.56	0.59	0.66	<0.03
10	6.0	10.63	29-Jul-00	21.2	475	5.954	-194.6	-3.332	0.10	73.06	2.88	13.50	41.30	3.53	14.20	6.45	2.09	14.64	0.08	0.11	<0.005	<0.07	<0.06	<0.005	<0.007	0.07	185.49	1.18	0.008	46.43	9.24	0.54	0.54	<0.03
10	7.0	9.63	29-Jul-00	21.0	346	5.471	-206.1	-3.531	0.10	82.74	1.84	1.88	39.60	2.35	12.40	2.22	0.57	10.19	0.01	0.01	<0.005	<0.07	<0.06	<0.005	<0.007	<0.05	112.88	2.03	0.182	50.57	10.12	0.26	0.45	<0.03
10	8.0	8.63	29-Jul-00	20.6	336	5.432	-215.4	-3.695	0.05	89.78	1.60	1.82	39.20	1.98	12.00	1.74	0.19	9.42	0.01	0.01	<0.005	<0.07	<0.06	<0.005	0.01	<0.05	97.63	3.09	0.252	51.55	9.24	0.32	0.41	<0.03
10	9.0	7.63	29-Jul-00	20.7	349	5.525	-219.0	-3.756	0.10	75.70	1.90	2.03	40.40	2.47	12.80	2.21	0.30	11.80	0.01	0.01	<0.005	<0.07	<0.06	<0.005	<0.007	<0.05	113.49	1.96	0.149	49.00	9.68	0.19	0.48	<0.03
10	10.0	6.63	29-Jul-00	20.5	326	5.613	-221.8	-3.807	0.05	59.85	1.96	4.36	34.90	2.47	11.20	2.87	0.43	10.45	0.02	0.01	<0.005	<0.07	<0.06	<0.005	0.01	<0.05	120.20	1.45	0.052	42.23	8.80	0.31	0.55	<0.03
10	11.0	5.63	29-Jul-00	20.4	260	5.806	-245.6	-4.216	0.05	34.33	1.26	15.40	22.00	4.99	6.75	0.08	0.01	2.71	0.06	0.04	<0.005	<0.07	<0.06	<0.005	0.01	<0.05	79.32	14.13	0.638	37.96	9.68	0.84	0.14	<0.03
10	12.0	4.63	29-Jul-00	DRY	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	
10	13.0	3.63	29-Jul-00	20.3	313	5.915	-219.4	-3.768	0.10	40.49	1.62	16.50	25.30	6.02	8.11	1.92	0.06	3.55	0.06	0.05	<0.005	<0.07	<0.06	<0.005	<0.007	<0.05	98.24	13.41	0.018	40.50	5.72	0.27	0.17	<0.03
10	14.0	2.63	31-Jul-00	20.3	273	5.734	-227.8	-3.912	0.20	31.69	1.18	16.90	22.40	5.62	8.12	0.60	0.34	1.42	0.07	0.06	<0.005	<0.07	<0.06	<0.005	0.05	0.08	73.83	18.99	0.021	38.30	5.72	0.35	0.12	<0.03
10	15.0	1.63	31-Jul-00	20.3	324	6.164	-257.1	-4.416	0.20	25.53	1.98	20.40	22.80	7.39	14.10	0.65	0.00	2.86	0.11	0.18	<0.005	<0.07	<0.06	<0.005	<0.007	<0.05	109.83	17.85	0.110	35.98	3.96	0.57	0.14	<0.03
10	16.0	0.63	31-Jul-00	20.1	321	6.154	-258.7	-4.447	0.15	29.05	1.90	21.50	26.80	8.17	8.76	0.44	0.00	0.85	0.12	0.40	<0.005	<0.07	<0.06	<0.005	0.01	<0.05	117.15	18.78	0.147	34.36	5.72	0.05	0.10	<0.03
10	18.0	-1.37	31-Jul-00	19.9	234	5.615	-236.1	-4.060	0.20	7.92	0.28	15.70	15.00	4.57	6.17	0.02	0.00	0.03	0.06	0.01	<0.005	<0.07	<0.06	<0.005	<0.007	<0.05	18.92	25.23	0.002	32.47	29.92	0.84	0.05	<0.03
10	20.0	-3.37	1-Aug-00	19.8	190	4.906	-176.6	-3.038	0.15	17.60	0.18	4.20	19.20	4.43	5.78	0.03	0.00	0.09	0.03	0.02	<0.005	<0.07	<0.06	<0.005	<0.007	<0.05	12.20	25.02	0.035	33.52	7.48	0.12	0.05	<0.03
10	22.0	-5.37	1-Aug-00	19.9	193	4.994	-188.5	-3.242	0.15	13.20	0.16	7.45	19.00	3.83	5.70	0.02	0.15	0.06	0.04	0.02	<0.005	<0.07	<0.06	<0.005	<0.007	<0.05	10.37	27.72	0.011	34.24	11.00	0.19	0.07	<0.03
10	24.0	-7.37	1-Aug-00	19.8	201	4.682	-185.7	-3.195	0.15	9.68	0.14	6.86	19.00	3.68	6.20	0.01	0.00	0.21	0.05	0.01	<0.005	<0.07	<0.06	<0.005	<0.007	<0.05	7.93	26.13	0.006	34.44	19.36	0.07	0.06	<0.03
10	26.0	-9.37	1-Aug-00	20.3	216	4.442	-172.4	-2.961	0.25	14.08	0.10	4.06	18.60	7.13	5.52	0.02	0.12	0.01	0.04	0.02	<0.005	<0.07	<0.06	<0.005	<0.007	0.28	6.10	24.93	<0.001	35.91	19.80	0.07	0.05	<0.03
10	28.0	-11.37	1-Aug-00	20.1	208	5.058	-187.4	-3.221	0.10	16.72	0.22	4.13	20.90	5.42	4.79	0.11	0.13	1.03	0.04	0.02	<0.005	<0.07	<0.06	<0.005	<0.007	0.07	13.42	19.35	0.004	36.90	18.92	0.48	0.04	<0.03
10	30.0	-13.37	1-Aug-00	20.3	203	5.792	-230.3	-3.956	0.10	21.12	0.78	1.50	26.00	5.13	2.04	4.40	1.04	0.36	0.03	0.07	<0.005	<0.07	<0.06	<0.005	<0.007	0.05	48.20	1.43	0.025	43.29	7.92	0.18	<0.03	<0.03

1. bgs = below ground surface (metres)

2. AHD = Australian Height Datum (metres)

3. All concentrations in mg/L unless otherwise noted (except pH and pe)

Table B2 Groundwater chemistry results for bundled piezometers 3, 4, 4A, 6 and 10: December 2000 sampling event.

Bore ID	Depth (bgs ¹)	Elev. (AHD ²)	Sample Date	EC								Cations																Anions						Uncharged	
				T(°C)	µS/cm	pH	Eh (mV)	pe	O ₂	CO ₂	Alk. (meq/L)	Ca ²⁺	Na ⁺	Mg ²⁺	K ⁺	Fe ²⁺	Fe ³⁺	NH ₄ ⁺	Sr ²⁺	Mn ²⁺	Li ⁺	Zn ²⁺	Pb ²⁺	Cd ²⁺	Cu ²⁺	Al ³⁺	HCO ₃ ⁻	SO ₄ ²⁻	S ²⁻	Cl ⁻	NO ₃ ⁻	PO ₄ ³⁻	B	Ni	
3	1.40	14.414	11-Dec-00	23.0	401.0	5.866	7.7	0.131	1.53	28.17	1.63	44.20	18.40	6.54	16.90	0.12	0.026	0.30	0.169	<0.0003	<0.004	0.017	<0.02	0.001	0.010	0.085	99.45	47.69	0.001	34.26	35.20	0.49	0.066	<0.003	
3	2.40	13.414	11-Dec-00	24.5	527.0	5.879	-15.3	-0.259	1.80	91.54	4.57	83.90	31.10	5.04	5.42	0.43	0.169	0.57	0.384	0.010	<0.004	0.007	<0.02	0.001	0.007	0.014	278.82	22.02	0.006	31.86	8.36	0.04	0.176	<0.003	
3	2.90	12.914	11-Dec-00	21.1	1168.5	6.316	-130.6	-2.237	0.29	144.35	12.83	204.00	42.00	20.10	12.60	0.69	-0.005	9.74	1.080	0.333	<0.004	0.015	<0.02	0.001	0.006	0.009	782.78	16.86	0.217	37.84	7.04	0.09	0.533	<0.003	
3	3.40	12.414	11-Dec-00	21.1	1357.0	6.398	-73.2	-1.254	0.50	147.87	14.69	198.00	66.30	22.20	18.60	3.75	5.550	28.25	0.881	1.940	<0.004	0.009	<0.02	0.001	0.006	<0.009	896.27	6.84	0.022	53.77	6.60	0.24	0.677	<0.003	
3	3.90	11.914	11-Dec-00	21.8	1506.5	6.435	-81.0	-1.384	0.81	167.24	15.24	149.00	90.30	18.50	25.40	6.75	44.150	32.51	0.732	0.717	<0.004	0.028	<0.02	0.001	0.007	<0.009	929.82	5.43	0.006	68.16	ND	0.36	0.663	<0.003	
3	4.40	11.414	11-Dec-00	20.1	1472.0	6.331	-72.3	-1.243	0.16	156.67	14.68	129.00	90.20	23.40	27.40	6.20	34.100	39.35	0.752	0.988	<0.004	0.024	<0.02	0.001	0.005	0.019	895.66	4.95	0.005	77.98	4.84	0.16	0.614	<0.003	
3	4.90	10.914	11-Dec-00	24.4	1341.5	6.382	-52.0	-0.881	1.93	150.51	13.13	130.00	84.80	23.20	25.90	7.45	19.450	21.29	0.739	1.020	<0.004	0.023	<0.02	0.001	0.006	0.022	801.09	4.95	0.002	72.98	2.64	0.07	0.599	<0.003	
3	5.40	10.414	11-Dec-00	23.0	921.0	6.374	-52.0	-0.885	0.87	107.38	8.28	86.60	54.00	14.90	18.10	4.00	7.500	27.48	0.484	0.423	<0.004	0.012	<0.02	0.001	0.005	0.009	505.18	3.54	0.003	53.98	10.56	0.08	0.498	<0.003	
3	5.90	9.914	11-Dec-00	20.6	878.5	6.403	-72.2	-1.239	0.41	96.82	7.98	79.50	56.20	13.70	15.60	5.00	14.600	28.25	0.385	0.368	<0.004	0.014	<0.02	0.001	0.007	<0.009	486.88	3.21	0.007	56.98	13.20	0.04	0.609	<0.003	
3	6.40	9.414	11-Dec-00	21.6	668.5	6.224	-43.9	-0.751	0.54	80.98	5.60	47.60	45.70	10.70	14.90	3.22	5.150	30.83	0.224	0.358	<0.004	0.016	<0.02	0.001	0.008	0.014	341.67	2.42	0.005	47.99	29.48	0.02	0.534	<0.003	
3	6.90	8.914	11-Dec-00	DRY	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	
3	7.40	8.414	11-Dec-00	DRY	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	
3	7.90	7.914	11-Dec-00	23.7	184.9	5.520	-86.2	-1.463	1.37	42.25	0.74	7.20	20.70	0.37	7.58	2.43	1.350	5.90	0.009	0.014	<0.004	0.008	<0.02	<0.001	0.006	0.037	45.15	6.40	0.920	30.99	21.56	0.09	0.274	<0.003	
3	8.40	7.414	11-Dec-00	22.3	178.2	5.356	-117.5	-2.004	0.75	42.25	0.70	6.56	20.70	0.34	7.74	1.68	1.200	5.16	0.006	0.002	<0.004	0.008	<0.02	<0.001	0.004	0.119	42.71	8.99	0.206	28.13	23.32	0.10	0.269	<0.003	
4	1.84	15.481	7-Dec-00	DRY	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	
4	2.34	14.981	7-Dec-00	DRY	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	
4	2.84	14.481	7-Dec-00	19.6	357.0	6.369	-191.9	-3.304	0.60	20.24	2.31	26.80	23.50	7.12	12.80	0.46	0.318	8.06	0.167	0.053	<0.004	0.031	<0.02	0.002	0.007	0.152	140.94	15.51	0.046	39.99	5.72	0.15	0.585	<0.003	
4	3.34	13.981	7-Dec-00	19.1	604.5	6.535	-172.7	-2.979	0.12	24.65	5.38	58.00	35.60	14.30	10.60	2.29	1.020	13.22	0.315	0.145	<0.004	0.017	<0.02	0.002	0.007	0.096	328.24	2.80	0.011	40.82	2.64	0.02	0.970	<0.003	
4	3.84	13.481	7-Dec-00	19.0	673.5	6.541	-155.1	-2.675	0.16	24.65	6.18	65.40	35.50	14.10	12.50	1.87	1.080	13.22	0.330	0.242	<0.004	0.028	<0.02	0.001	0.011	0.061	377.05	2.40	0.025	42.82	ND	0.19	1.220	<0.003	
4	4.34	12.981	7-Dec-00	18.9	776.0	6.547	-158.1	-2.729	0.13	33.45	7.15	81.60	38.00	17.10	14.60	2.20	1.250	19.87	0.389	0.174	<0.004	0.010	<0.02	0.001	0.007	0.035	436.24	3.00	0.010	47.17	6.16	0.19	1.440	<0.003	
4	4.84	12.481	7-Dec-00	18.9	750.5	6.609	-170.6	-2.944	0.09	33.45	6.93	72.60	44.80	16.40	13.60	2.59	2.140	1.81	0.363	0.190	<0.004	0.005	<0.02	0.001	0.005	0.009	422.81	3.03	0.007	45.99	4.56	0.04	0.943	<0.003	
4	5.34	11.981	7-Dec-00	19.1	342.0	6.293	-170.9	-2.947	0.14	14.08	1.80	30.40	20.90	7.75	4.69	2.73	1.940	1.68	0.120	0.069	<0.004	0.006	<0.02	0.001	0.006	0.048	109.82	38.39	0.015	39.83	4.84	0.21	0.330	<0.003	
4	5.84	11.481	7-Dec-00	19.1	239.5	5.937	-11.0	-0.190	0.63	15.84	0.82	21.40	18.20	4.31	3.46	0.03	0.021	3.81	0.066	0.024	<0.004	0.015	<0.02	0.001	0.008	0.019	60.03	24.15	ND	31.99	2.70	0.02	0.311	<0.003	
4	6.34	10.981	7-Dec-00	19.6	255.5	5.887	-80.2	-1.381	0.75	17.60	0.97	24.30	20.20	4.43	3.49	0.04	0.055	4.39	0.081	0.028	<0.004	0.013	<0.02	0.001	0.012	<0.009	69.18	27.90	ND	32.99	1.44	0.08	0.214	<0.003	
4	6.84	10.481	7-Dec-00	20.5	273.5	6.167	-65.6	-1.126	2.34	12.32	1.01	25.60	18.10	4.89	3.67	0.05	0.062	5.42	0.048	0.008	<0.004	0.019	<0.02	0.001	0.009	<0.009	71.62	24.33	0.002	32.99	1.54	0.10	0.143	<0.003	
4	7.34	9.981	7-Dec-00	20.0	420.5	6.484	-157.6	-2.710	0.63	21.12	2.80	38.80	21.80	8.68	7.17	2.11	3.360	4.94	0.173	0.078	<0.004	0.019	<0.02	0.001	0.007	<0.009	170.83	25.80	0.036	36.99	13.64	0.04	0.599	<0.003	
4	7.84	9.481	7-Dec-00	19.8	271.0	5.894	-137.5	-2.366	0.42	19.36	1.00	21.90	20.80	4.96	4.86	0.06	0.032	3.23	0.091	0.051	<0.004	0.007	<0.02	0.001	0.006	<0.009	61.01	26.22	0.270	37.99	1.67	0.08	0.141	<0.003	
4	8.34	8.981	7-Dec-00	19.8	220.5	5.157	-102.3	-1.760	0.58	29.93	0.28	13.20	20.20	3.61	5.07	0.05	0.004	4.55	0.050	0.015	<0.004	0.008	<0.02	0.001	0.006	<0.009	37.08	27.69	0.016	36.99	0.09	0.01	0.067	<0.003	
4	8.84	8.481	7-Dec-00	19.5	247.5	5.402	-127.4	-2.194	0.31	33.45	0.56	13.30	22.80	4.72	7.36	0.04	0.031	2.58	0.057	0.025	<0.004	0.009	<0.02	0.001	0.005	0.016	34.17	29.73	0.023	37.99	1.80	0.03	0.093	<0.003	
4	9.34	7.981	7-Dec-00	20.0	244.5	5.476	-136.3	-2.343	0.66	26.41	0.56	14.40																							

Table B2 (cont.) Groundwater chemistry results for bundled piezometers 3, 4, 4A, 6 and 10: December 2000 sampling event.

Bore ID	Depth (bgs ¹)	Elev. (AHD ²)	Sample Date	EC								Cations																Anions						Uncharged	
				T(°C)	µS/cm	pH	Eh (mV)	pe	O ₂	CO ₂	Alk. (meq/L)	Ca ²⁺	Na ⁺	Mg ²⁺	K ⁺	Fe ²⁺	Fe ³⁺	NH ₄ ⁺	Sr ²⁺	Mn ²⁺	Li ⁺	Zn ²⁺	Pb ²⁺	Cd ²⁺	Cu ²⁺	Al ³⁺	HCO ₃ ⁻	SO ₄ ²⁻	S ²⁻	Cl ⁻	NO ₃ ⁻	PO ₄ ³⁻	B	Ni	
6	1.68	13.981	12-Dec-00	DRY	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	
6	2.68	12.981	12-Dec-00	22.7	161.0	5.060	-78.8	-1.342	0.29	33.45	0.31	7.83	19.90	1.82	4.97	1.74	0.340	0.53	0.035	0.043	<0.004	0.015	<0.02	0.001	0.005	0.139	18.91	20.61	0.255	30.99	10.12	0.03	0.044	<0.003	
6	3.68	11.981	12-Dec-00	22.9	154.4	5.570	-115.6	-1.968	0.70	25.53	0.57	6.97	24.00	2.50	4.60	2.13	1.560	1.42	0.024	0.028	<0.004	0.009	<0.02	0.001	0.005	0.890	34.78	21.09	0.636	25.99	14.52	0.03	0.064	<0.003	
6	4.68	10.981	12-Dec-00	21.1	196.9	5.780	-126.4	-2.165	0.29	28.17	1.10	8.95	29.60	2.34	3.28	1.51	0.530	0.86	0.030	0.026	<0.004	0.010	<0.02	<0.001	0.007	0.348	63.11	16.77	0.327	25.89	11.88	0.04	0.078	<0.003	
6	5.68	9.981	12-Dec-00	25.0	189.6	5.734	-113.6	-1.920	0.74	37.85	1.22	10.36	28.80	1.53	4.28	1.73	0.440	2.32	0.025	0.025	<0.004	0.030	<0.02	0.001	0.006	0.808	74.43	15.30	0.467	20.99	11.88	0.04	0.126	<0.003	
6	6.68	8.981	12-Dec-00	20.6	216.5	5.872	-113.5	-1.947	0.26	24.65	1.22	16.30	17.80	2.82	5.30	2.35	2.770	1.61	0.051	0.054	<0.004	0.007	<0.02	0.001	0.007	0.192	74.43	7.83	0.064	30.23	11.44	0.03	0.175	<0.003	
6	7.68	7.981	12-Dec-00	23.8	237.5	6.065	-111.6	-1.894	0.93	25.53	1.44	19.10	17.60	2.92	5.74	1.20	0.880	2.89	0.072	0.048	<0.004	0.006	<0.02	0.001	0.006	0.126	87.86	4.44	0.055	29.99	9.24	0.15	0.198	<0.003	
6	8.68	6.981	12-Dec-00	22.4	236.5	6.037	-129.5	-2.208	0.48	32.57	1.52	21.70	16.90	2.89	5.76	1.23	0.360	3.61	0.088	0.067	<0.004	0.006	<0.02	0.002	0.005	0.163	92.74	4.71	0.148	30.99	7.92	0.08	0.186	<0.003	
6	9.68	5.981	12-Dec-00	21.8	216.0	5.772	-106.6	-1.822	0.37	35.21	1.21	16.40	17.60	3.23	5.66	2.19	1.060	3.55	0.048	0.035	<0.004	0.006	<0.02	0.001	0.006	0.100	73.82	6.90	0.042	33.99	7.60	0.15	0.159	<0.003	
6	10.18	5.481	12-Dec-00	20.7	291.5	5.852	-128.4	-2.202	0.17	48.41	1.87	18.70	22.50	4.46	6.81	3.02	3.380	5.23	0.080	0.066	<0.004	0.005	<0.02	0.001	0.016	0.084	114.09	9.84	0.032	33.99	9.68	0.05	0.146	<0.003	
6	10.68	4.981	12-Dec-00	20.7	403.0	5.941	-139.2	-2.387	0.21	66.89	2.92	26.60	26.80	5.70	9.45	3.30	4.540	9.55	0.124	0.088	<0.004	<0.003	<0.02	0.001	0.006	0.097	178.16	14.10	0.015	38.14	12.32	0.03	0.174	<0.003	
6	11.18	4.481	12-Dec-00	23.0	367.5	5.921	-125.6	-2.137	0.79	65.13	2.75	25.40	26.30	5.35	8.93	2.82	4.210	8.84	0.116	0.085	<0.004	0.005	<0.02	0.001	0.007	0.040	167.78	12.90	0.114	37.84	10.12	0.12	0.168	<0.003	
6	11.68	3.981	12-Dec-00	26.6	316.0	5.879	-117.3	-1.972	1.13	51.05	2.03	20.10	23.30	4.88	7.12	2.86	1.940	6.66	0.081	0.064	<0.004	0.009	<0.02	0.001	0.006	0.026	123.85	8.10	0.156	35.85	12.20	0.01	0.152	<0.003	
6	12.18	3.481	12-Dec-00	22.0	303.0	5.856	-128.1	-2.187	0.50	51.05	1.97	18.80	21.80	3.77	7.47	2.75	1.400	8.64	0.090	0.062	<0.004	0.005	<0.02	0.001	0.008	0.009	120.19	5.94	0.300	33.99	13.08	0.03	0.151	<0.003	
6	12.68	2.981	12-Dec-00	23.9	417.5	5.854	-116.7	-1.980	1.05	86.26	2.99	17.40	31.70	5.00	12.10	3.60	1.200	26.96	0.079	0.050	<0.004	0.008	<0.02	0.001	0.006	0.023	182.43	9.12	0.166	42.99	11.44	0.01	0.183	<0.003	
6	13.18	2.481	12-Dec-00	25.6	724.0	6.048	-99.7	-1.682	1.03	115.31	5.46	14.00	53.20	11.80	23.60	4.50	11.300	23.87	0.035	0.023	<0.004	0.011	<0.02	0.001	0.006	0.048	333.13	3.45	0.013	56.98	12.32	0.01	0.327	<0.003	
6	13.68	1.981	12-Dec-00	20.7	640.5	5.910	-110.7	-1.899	0.25	114.43	4.71	8.97	49.90	10.83	19.70	4.10	10.300	23.22	0.030	0.016	<0.004	<0.003	<0.02	0.001	0.006	0.093	287.37	2.92	0.015	56.98	11.44	0.01	0.295	<0.003	
6	14.18	1.481	12-Dec-00	24.1	684.0	5.984	-104.6	-1.773	0.66	109.14	4.88	9.61	52.00	10.80	21.10	3.95	11.650	32.90	0.030	0.016	<0.004	0.004	<0.02	0.001	0.005	0.106	297.74	2.51	0.017	57.75	10.12	0.07	0.313	<0.003	
6	15.18	0.481	12-Dec-00	21.6	666.5	5.967	-109.4	-1.871	0.60	116.19	4.90	9.71	50.70	10.70	20.80	2.40	13.300	29.03	0.031	0.017	<0.004	0.007	<0.02	0.001	0.008	0.110	298.96	2.68	0.051	55.98	14.52	0.04	0.309	<0.003	
6	16.18	-0.519	12-Dec-00	23.5	662.5	5.960	-150.0	-2.549	0.92	131.15	4.87	10.50	50.80	10.80	21.30	3.60	11.200	29.03	0.035	0.022	<0.004	0.016	<0.02	0.001	0.007	0.090	297.13	2.90	0.084	53.98	11.44	0.06	0.307	<0.003	
6	18.18	-2.519	12-Dec-00	21.5	278.0	5.459	-74.5	-1.274	0.56	63.37	1.30	4.40	26.60	4.69	5.90	2.99	4.530	7.09	0.025	0.015	<0.004	0.024	<0.02	0.001	0.010	<0.009	79.32	17.81	0.115	37.84	9.68	0.02	0.118	<0.003	
10	3.00	13.628	9-Dec-00	21.0	749.5	6.055	-74.9	-1.283	0.11	91.54	5.86	97.80	17.20	7.64	6.27	8.50	25.600	3.23	0.501	0.469	<0.004	0.095	<0.02	<0.001	0.007	0.013	357.53	75.89	0.001	21.99	11.44	0.25	0.377	<0.003	
10	4.00	12.628	9-Dec-00	20.9	1659.5	6.434	-89.3	-1.531	0.12	149.63	15.75	136.00	89.10	30.30	39.10	7.70	17.200	28.38	0.692	0.358	<0.004	0.012	<0.02	0.001	0.006	<0.009	945.94	23.82	0.010	70.98	4.40	0.14	0.781	<0.003	
10	5.00	11.628	9-Dec-00	20.7	1322.5	6.199	-75.0	-1.286	0.08	167.24	12.23	86.00	97.40	18.40	31.10	6.75	18.450	31.48	0.455	0.435	<0.004	0.014	<0.02	0.001	0.007	0.037	736.18	5.70	0.009	82.97	7.48	0.10	0.667	<0.003	
10	6.00	10.628	9-Dec-00	20.6	612.5	5.780	-29.2	-0.501	0.11	107.38	4.71	18.60	63.00	9.19	16.90	3.70	6.040	24.90	0.104	0.145	<0.004	0.010	<0.02	0.001	0.007	0.114	287.37	1.52	0.002	59.98	9.24	0.16	0.693	<0.003	
10	7.00	9.628	9-Dec-00	20.5	343.0	5.300	-29.7	-0.510	0.13	96.82	1.85	2.42	43.60	2.70	11.10	2.96	0.440	13.68	0.016	0.014	<0.004	0.014	<0.02	0.001	0.007	0.091	102.87	2.14	0.210	54.98	16.72	0.04	0.437	<0.003	
10	8.00	8.628	9-Dec-00	20.7	315.0	5.185	-80.7	-1.384	0.12	82.74	1.45	1.94	41.60	2.01	10.60	2.01	0.240	11.09	0.013	0.009	<0.004	0.008	<0.02	0.001	0.008	0.038	88.47	3.06	0.352	54.98	11.44	0.46	0.385	<0.003	
10	9.00	7.628	9-Dec-00	20.6	334.5	5.273	-97.2	-1.668	0.10	96.82	1.66	2.22	42.30	2.43	11.20	2.42	0.380	10.58	0.013	0.010	<0.004	0.006	<0.02	0.001	0.006	0.037	96.28	2.61	0.263	55.54	11.00	0.15	0.449	<0.003	
10	10.00	6.628	9-Dec-00	20.5	379.0	5.435	-89.0	-1.528	0.12	40.49	1.37	5.50	43.80	2.55	11.80	3.28	1.140	6.19	0.023	0.008	<0.004	<0.003	<0.02	0.001	0.006	0.021	83.59	6.72	0.040	48.60	11.68	0.20	0.588	<0.003	
10	11.00	5.628	9-Dec-00	20.4	253.0	5.508	-139.3	-2.649	0.05	83.62	1.10	14.90	25.60	5.02	6.07	0.33	ND	3.04	0.057	0.042	<0.004	0.005	<0.02	0.001	0.006	0.065	67.11	12.69	1.120	36.99	11.44	0.06	0.191	<0.003	

1. bgs = below ground surface (metres)

2. AHD = Australian Height Datum (metres)

3. All concentrations in mg/L unless otherwise noted (except pH and pe)

Table B3 Groundwater chemistry results for bundled piezometers 3, 4, 4A, 6 and 10: March 2001 sampling event.

Bore ID	Depth (bgs ¹)	Elev. (AHD ²)	Sample Date	T(°C)	EC		Eh		Alk.		Cations													Anions						Uncharged				
					(μS/cm)	pH	(mV)	pe	O ₂	CO ₂	(meq/L)	Ca ²⁺	Na ⁺	Mg ²⁺	K ⁺	Fe ²⁺	Fe ³⁺	NH ₄ ⁺	Sr ²⁺	Mn ²⁺	Li ⁺	Zn ²⁺	Pb ²⁺	Cd ²⁺	Cu ²⁺	Al ³⁺	HCO ₃ ⁻	SO ₄ ²⁻	S ²⁻	Cl ⁻	NO ₃ ⁻	PO ₄ ³⁻	B	Ni
3	1.40	14.41	15-Mar-01	23.2	232.0	6.149	106.5	1.811	5.7	14.96	0.84	18.27	18.90	4.11	6.33	0.09	0.00	0.15	0.053	0.001	0.002	0.007	<0.004	<0.001	0.006	0.023	36.66	5.10	0.008	50.98	12.24	2.47	0.024	<0.001
3	2.40	13.41	15-Mar-01	22.9	379.0	6.258	-26.6	-0.453	6.4	66.89	4.50	43.90	18.25	6.88	2.40	0.13	0.00	0.39	0.181	0.007	0.002	0.002	<0.004	<0.001	0.007	0.011	136.19	11.76	0.014	44.99	7.92	0.01	0.085	<0.001
3	2.90	12.91	15-Mar-01	22.4	589.0	6.357	-109.4	-1.865	5.4	98.58	8.68	65.17	22.30	7.27	11.40	0.40	0.00	27.48	0.508	0.182	0.002	0.007	0.008	<0.001	0.008	0.028	270.74	14.26	0.091	49.99	11.00	4.15	0.399	<0.001
3	3.40	12.41	15-Mar-01	22.2	745.0	6.489	-145.8	-2.488	5.5	176.04	11.86	64.81	21.50	7.77	10.40	2.98	5.28	36.77	0.286	0.731	0.001	0.002	0.004	<0.001	0.005	0.006	370.69	7.22	0.019	55.98	9.24	4.00	0.349	<0.001
3	3.90	11.91	15-Mar-01	22.5	783.0	6.510	-78.6	-1.340	6.3	146.11	13.20	61.36	28.80	7.72	13.20	2.80	12.93	34.19	0.312	0.439	0.002	0.002	0.005	<0.001	0.006	0.009	395.04	2.52	0.005	47.99	0.00	4.35	0.325	<0.001
3	4.40	11.41	15-Mar-01	21.4	793.0	6.428	-65.2	-1.116	5.5	149.63	13.40	54.72	34.70	8.82	11.60	4.30	9.97	38.70	0.287	0.387	0.002	0.002	<0.004	<0.001	0.005	0.021	401.45	2.07	0.000	60.98	3.52	3.15	0.276	<0.001
3	4.90	10.91	15-Mar-01	22.7	789.0	6.411	-57.1	-0.973	6.8	156.67	13.12	63.36	44.70	11.45	14.50	4.50	9.59	23.22	0.351	0.523	0.001	0.002	<0.004	<0.001	0.004	0.014	398.88	2.89	0.001	69.98	7.48	0.52	0.324	<0.001
3	5.40	10.41	15-Mar-01	22.1	591.0	6.437	-52.2	-0.891	6.4	87.14	8.82	35.81	24.10	6.28	8.83	3.15	2.47	32.25	0.182	0.216	0.002	0.002	0.004	<0.001	0.006	0.012	272.02	1.20	0.009	56.98	0.00	0.55	0.221	<0.001
3	5.90	9.91	15-Mar-01	21.8	546.0	6.384	-45.8	-0.783	5.8	116.19	7.54	33.63	26.50	5.44	12.60	5.10	7.54	26.96	0.227	0.250	0.001	0.009	0.014	<0.001	0.008	0.027	243.19	1.62	0.000	58.98	13.64	0.13	0.453	<0.001
3	6.40	9.41	15-Mar-01	22.0	381.0	6.222	-34.4	-0.587	5.8	77.46	4.96	20.63	20.30	4.56	8.34	3.27	2.05	21.16	0.115	0.212	0.002	0.006	<0.004	<0.001	0.008	0.019	137.47	0.94	0.000	47.99	10.12	4.00	0.351	<0.001
3	6.90	8.91	15-Mar-01	DRY	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
3	7.40	8.41	15-Mar-01	DRY	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
3	7.90	7.91	15-Mar-01	23.8	125.8	5.541	-24.5	-0.416	7.5	45.77	0.84	5.43	14.10	2.24	4.74	1.76	0.00	8.71	0.004	0.004	0.005	0.005	<0.004	<0.001	0.004	0.024	11.68	7.41	1.950	35.99	23.32	0.79	0.132	<0.001
3	8.40	7.41	15-Mar-01	23.4	124.4	5.466	-72.2	-1.227	7.0	47.53	0.68	5.43	13.40	2.29	5.61	1.18	0.00	7.42	0.004	0.002	0.004	0.004	0.007	<0.001	0.012	0.025	11.36	9.26	0.303	35.99	22.88	0.01	0.157	<0.001
4	1.84	15.48	19-Mar-01	DRY	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
4	2.34	14.98	19-Mar-01	DRY	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
4	2.84	14.48	19-Mar-01	21.7	267.0	6.620	-122.1	-2.087	1.7	16.72	1.74	16.82	16.80	3.59	5.70	0.79	0.00	2.27	0.080	0.058	0.003	0.077	0.019	0.001	0.017	0.227	64.43	12.40	0.510	31.99	6.16	0.21	0.232	0.004
4	3.34	13.98	19-Mar-01	21.0	306.0	6.925	-129.2	-2.214	0.7	17.60	2.78	24.27	15.30	4.35	4.41	1.48	0.00	6.64	0.089	0.038	0.002	0.015	0.013	<0.001	0.004	0.058	89.42	12.74	0.122	36.12	9.68	0.06	0.403	<0.001
4	3.84	13.48	19-Mar-01	20.7	417.0	6.650	-121.9	-2.091	1.0	25.53	4.14	32.57	17.90	7.64	6.28	1.70	0.00	9.48	0.127	0.115	0.002	0.006	0.008	<0.001	0.002	0.035	160.54	15.93	0.115	37.99	7.92	0.29	0.658	<0.001
4	4.34	12.98	19-Mar-01	20.4	527.0	6.686	-118.1	-2.027	0.8	36.09	5.38	40.91	28.60	10.51	8.65	2.04	0.00	12.45	0.160	0.083	0.001	0.010	0.005	<0.001	0.019	0.047	231.01	26.81	0.024	37.99	6.60	0.07	0.647	<0.001
4	4.84	12.48	19-Mar-01	20.3	541.0	6.661	-112.5	-1.932	0.9	44.01	5.98	44.27	25.90	9.91	9.43	2.05	0.95	13.74	0.203	0.136	0.002	0.045	0.009	<0.001	0.016	0.050	239.98	15.21	0.011	42.99	7.92	0.03	0.903	0.001
4	5.34	11.98	19-Mar-01	20.4	535.0	6.369	-85.2	-1.463	1.3	66.01	6.14	43.63	27.40	8.84	8.90	2.43	0.00	10.26	0.163	0.032	0.002	0.004	<0.004	<0.001	0.003	0.008	236.14	4.43	0.005	48.79	7.92	0.02	0.480	<0.001
4	5.84	11.48	19-Mar-01	20.4	203.0	5.919	-21.5	-0.369	1.4	17.60	1.04	18.64	15.80	4.90	4.06	0.03	0.01	0.09	0.050	0.022	0.003	0.015	0.017	<0.001	0.006	0.023	29.84	15.72	0.000	39.99	15.56	0.02	0.111	<0.001
4	6.34	10.98	19-Mar-01	21.7	187.3	6.094	5.2	0.089	3.1	14.08	0.86	18.73	13.30	3.46	2.85	0.03	0.02	0.05	0.036	0.034	0.003	0.008	<0.004	<0.001	0.021	0.028	26.15	12.16	0.000	38.14	13.76	0.04	0.107	<0.001
4	6.84	10.48	19-Mar-01	22.6	174.4	6.185	8.5	0.145	7.7	16.72	0.86	16.64	12.70	3.62	2.50	0.19	0.00	0.15	0.026	0.016	0.004	0.013	0.007	<0.001	0.014	0.012	23.11	12.61	0.022	37.99	12.12	0.24	0.054	<0.001
4	7.34	9.98	19-Mar-01	21.8	204.0	6.474	-36.4	-0.622	3.3	14.08	1.42	14.55	10.54	3.58	2.66	0.67	0.00	2.75	0.050	0.014	0.002	0.007	0.005	<0.001	0.004	0.009	30.07	10.49	0.040	35.99	7.48	1.16	0.186	<0.001
4	7.84	9.48	19-Mar-01	21.3	182.6	5.952	-9.7	-0.166	2.7	19.36	0.90	14.45	14.10	4.23	3.09	0.02	0.01	0.06	0.043	0.026	0.003	0.008	0.004	<0.001	0.009	0.014	25.04	12.32	0.001	38.68	10.32	0.11	0.066	<0.001
4	8.34	8.98	19-Mar-01	21.2	155.3	5.293	24.8	0.425	2.8	28.17	0.36	12.99	14.05	4.65	3.79	0.04	0.00	0.05	0.021	0.009	0.004	0.009	<0.004	<0.001	0.007	0.011	18.62	12.00	0.008	39.99	9.80	1.53	0.032	<0.001
4	8.84	8.48	19-Mar-01	20.1	168.7	5.561	21.1	0.363	1.0	28.17	0.66	11.56	12.90	4.51	4.87	0.03	0.09	0.46	0.035	0.020	0.003	0.014	0.008	<0.001	0.012	0.024	21.77	11.17	0.006	39.99	6.72	0.04	0.067	<0.001
4A	2.15	13.17	16-Mar-01	24.0	424.0	6.309	-9.3	-0.158	3.1	85.38	5.38	29.09	16.50	7.80	6.95	0.74	0.00	16.58	0.114	0.035	0.002	0.007	0.006	<0.001	0.005	0.046	165.02	3.71	0.205	43.98	10.52	0.05	0.086	<0.001
4A	2.65	12.67	16-Mar-01	22.9	424.0	5.938	-15.7	-0.267	2.3	124.99	5.30	16.27	24.30	7.44	11.30	0.59	0.00	29.93	0.050	0.070	0.002	0.012	<0.004	<0.001	0									

Table B3 (cont.) Groundwater chemistry results for bundled piezometers 3, 4, 4A, 6 and 10: March 2001 sampling event.

Bore ID	Depth (bgs ¹)	Elev. (AHD ²)	Sample Date	T(°C)	EC					Alk. (meq/L)	Cations															Anions					Uncharged			
					(µS/cm)	pH	Eh (mV)	pe	O ₂		CO ₂	Ca ²⁺	Na ⁺	Mg ²⁺	K ⁺	Fe ²⁺	Fe ³⁺	NH ₄ ⁺	Sr ²⁺	Mn ²⁺	Li ⁺	Zn ²⁺	Pb ²⁺	Cd ²⁺	Cu ²⁺	Al ³⁺	HCO ₃ ⁻	SO ₄ ²⁻	S ²⁻	Cl ⁻	NO ₃ ⁻	PO ₄ ³⁻	B	Ni
6	1.68	13.98	14-Mar-01	DRY	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
6	2.68	12.98	14-Mar-01	23.7	83.3	5.129	-44.6	-0.757	2.5	33.45	0.30	5.17	13.70	1.97	3.28	1.28	0.00	0.23	0.019	0.089	0.008	0.010	0.008	<0.001	0.021	0.171	1.69	4.83	0.087	32.99	11.00	0.10	0.048	<0.001
6	3.68	11.98	14-Mar-01	23.3	95.5	5.591	-74.0	-1.258	3.1	29.93	0.62	6.01	12.87	0.92	2.50	2.32	0.00	0.40	0.012	0.016	0.006	0.002	<0.004	<0.001	0.007	0.047	4.56	3.37	0.008	30.99	7.48	0.08	0.030	<0.001
6	4.68	10.98	14-Mar-01	22.2	108.6	5.877	-98.5	-1.681	2.3	24.65	0.80	5.35	15.30	1.42	2.48	2.17	0.65	0.80	0.025	0.025	0.006	0.022	0.012	<0.001	0.016	0.041	7.64	4.38	0.003	30.99	7.04	1.09	0.055	<0.001
6	5.18	10.48	14-Mar-01	22.0	127.2	5.823	-107.1	-1.829	2.4	26.41	1.14	3.94	15.60	0.92	1.99	2.04	0.00	0.97	0.013	0.016	0.005	0.002	<0.004	<0.001	0.006	0.072	12.01	6.05	0.036	27.99	6.16	0.19	0.043	<0.001
6	6.18	9.48	14-Mar-01	21.8	155.8	5.953	-122.4	-2.091	2.4	31.69	1.42	7.59	11.97	1.23	2.47	2.47	0.00	2.59	0.025	0.023	0.004	<0.001	<0.004	<0.001	0.003	0.033	18.74	5.31	0.102	25.89	9.24	3.00	0.060	<0.001
6	7.18	8.48	14-Mar-01	20.6	154.6	6.035	-235.5	-4.040	2.0	31.69	1.40	9.57	9.36	1.23	3.00	1.66	0.00	4.13	0.017	0.021	0.004	0.002	<0.004	<0.001	0.006	0.014	18.46	2.30	0.024	32.99	7.48	0.25	0.090	<0.001
6	8.18	7.48	14-Mar-01	21.7	168.1	6.069	-94.2	-1.610	2.9	35.21	1.66	11.45	9.86	2.48	2.94	1.15	0.00	4.00	0.042	0.047	0.003	<0.001	<0.004	<0.001	0.005	0.008	21.63	3.03	0.084	32.86	7.92	4.30	0.077	<0.001
6	9.18	6.48	14-Mar-01	21.8	159.1	6.056	-95.9	-1.639	2.8	24.65	1.58	8.95	7.03	2.93	2.84	1.34	0.00	4.19	0.024	0.020	0.003	0.002	<0.004	<0.001	0.005	0.007	19.51	1.53	0.123	33.99	9.28	0.05	0.051	<0.001
6	10.18	5.48	14-Mar-01	20.3	187.1	5.880	-88.8	-1.525	2.2	48.41	1.94	10.27	11.43	1.82	3.53	3.00	0.00	5.48	0.039	0.038	0.002	0.002	<0.004	<0.001	0.007	0.015	26.10	3.85	0.064	35.99	10.56	0.05	0.061	<0.001
6	11.18	4.48	14-Mar-01	21.4	200.0	5.869	-89.8	-1.536	3.1	54.57	2.66	10.36	12.20	2.75	5.53	3.70	0.00	10.13	0.061	0.056	0.003	0.005	<0.004	<0.001	0.012	0.009	29.13	6.61	0.040	42.99	12.32	0.07	0.086	<0.001
6	12.18	3.48	14-Mar-01	21.1	192.3	5.838	-86.6	-1.483	2.6	39.61	2.10	12.27	14.40	2.91	6.27	3.45	0.00	6.19	0.063	0.054	0.003	0.011	0.008	<0.001	0.008	0.019	27.32	3.82	0.027	44.99	18.48	0.05	0.106	<0.001
6	13.18	2.48	14-Mar-01	21.6	500.0	6.018	-94.3	-1.612	3.0	122.35	5.80	29.94	28.20	7.01	14.10	2.33	4.36	14.06	0.013	0.010	0.001	0.002	0.017	<0.001	0.006	0.017	213.72	1.48	0.025	52.74	9.24	0.02	0.146	<0.001
6	14.18	1.48	14-Mar-01	21.4	429.0	5.849	-74.3	-1.271	3.0	99.46	4.76	13.45	24.40	5.49	10.80	5.45	0.93	23.61	0.012	0.008	0.002	0.002	<0.004	<0.001	0.004	0.011	168.22	1.11	0.031	50.98	10.56	0.03	0.136	<0.001
6	15.18	0.48	14-Mar-01	21.3	437.0	5.961	-87.2	-1.492	2.4	105.62	4.90	8.70	25.40	4.91	10.40	4.10	2.51	33.02	0.013	0.009	0.002	0.002	0.004	<0.001	0.005	0.011	173.35	1.11	0.029	48.75	22.00	0.05	0.133	<0.001
6	16.18	-0.52	14-Mar-01	22.2	440.0	6.108	-92.3	-1.575	3.9	103.86	5.10	4.65	26.80	5.30	14.30	4.90	4.02	27.74	0.020	0.015	0.003	0.010	0.008	<0.001	0.008	0.026	175.27	1.85	0.079	47.98	12.76	0.01	0.183	<0.001
6	18.18	-2.52	14-Mar-01	21.1	176.7	5.576	-41.0	-0.702	1.9	52.81	0.74	4.75	20.00	2.20	3.30	2.49	1.84	3.48	0.012	0.009	0.003	0.005	<0.004	<0.001	0.013	0.006	23.65	14.02	0.038	35.99	12.32	0.06	0.051	<0.001
10	3.0	13.63	20-Mar-01	23.2	495.0	6.074	-14.3	-0.243	1.4	85.38	5.56	59.54	10.32	6.10	8.49	5.20	4.53	1.43	0.211	0.152	0.004	0.025	0.004	<0.001	0.015	0.012	210.51	32.12	0.006	26.99	2.48	0.79	0.147	0.002
10	4.0	12.63	20-Mar-01	22.1	1191.0	6.534	-76.8	-1.311	0.8	129.39	14.66	83.26	40.00	15.09	25.30	3.95	10.68	43.05	0.460	0.247	0.002	0.013	0.014	<0.001	0.005	0.034	656.45	3.72	0.012	36.99	2.64	0.56	0.657	<0.001
10	5.0	11.63	20-Mar-01	22.7	1093.0	6.351	-77.1	-1.313	1.9	145.23	12.32	75.45	39.70	13.73	22.20	3.14	6.59	47.05	0.257	0.212	0.002	0.012	0.007	<0.001	0.009	0.074	593.66	1.90	0.005	49.22	2.80	0.15	0.292	<0.001
10	6.0	10.63	20-Mar-01	21.2	899.0	6.136	-58.9	-1.008	0.6	166.36	9.80	41.09	37.90	23.18	19.30	6.60	5.85	48.38	0.203	0.259	0.003	0.006	0.005	<0.001	0.013	0.039	469.36	1.96	0.004	81.97	7.48	2.10	0.355	<0.001
10	7.0	9.63	20-Mar-01	21.0	229.0	5.365	-32.5	-0.557	0.8	107.38	1.78	7.85	22.20	1.93	4.70	2.47	0.00	9.29	0.005	0.006	0.002	0.002	<0.004	<0.001	0.005	0.037	35.95	0.76	0.258	56.98	8.80	0.10	0.184	<0.001
10	8.0	8.63	20-Mar-01	20.8	220.0	5.347	-69.2	-1.186	0.7	114.43	1.46	6.94	23.10	1.90	5.06	1.76	0.00	7.87	0.006	0.006	0.002	0.004	0.004	<0.001	0.009	0.030	33.83	1.88	0.365	56.98	7.04	0.11	0.192	<0.001
10	9.0	7.63	20-Mar-01	20.8	225.0	5.373	-96.6	-1.656	2.1	101.22	1.60	4.89	22.50	1.95	5.06	2.28	0.00	9.29	0.005	0.005	0.003	0.002	0.004	<0.001	0.009	0.033	35.01	1.05	0.310	51.98	8.36	0.08	0.214	<0.001
10	10.0	6.63	20-Mar-01	20.7	249.0	5.541	-102.9	-1.765	0.7	80.98	2.00	2.72	21.70	1.25	6.78	3.19	0.00	13.48	0.011	0.006	0.003	0.003	0.009	<0.001	0.005	0.034	40.65	0.77	0.047	51.98	7.48	0.08	0.345	<0.001
10	11.0	5.63	20-Mar-01	20.7	180.3	5.776	-143.3	-2.458	0.7	44.01	1.40	8.08	13.20	2.89	4.07	0.18	0.00	3.07	0.030	0.027	0.004	0.010	0.006	<0.001	0.009	0.036	24.50	4.07	0.633	33.99	6.60	0.14	0.092	<0.001

1. bgs = below ground surface (metres)

2. AHD = Australian Height Datum (metres)

3. All concentrations in mg/L unless otherwise noted (except pH and pe)

Table B4 Groundwater chemistry results for bundled piezometers 3, 4, 4A, 6 and 10: May 2001 sampling event.

Bore ID	Depth (bgs ¹)	Elev. (AHD ²)	Sample Date	EC								Cations																	Anions						Uncharged	
				T(°C)	µS/cm	pH	Eh (mV)	pe	O ₂	CO ₂	Alk. (meq/L)	Ca ²⁺	Na ⁺	Mg ²⁺	K ⁺	Fe ²⁺	Fe ³⁺	NH ₄ ⁺	Sr ²⁺	Mn ²⁺	Li ⁺	Zn ²⁺	Pb ²⁺	Cd ²⁺	Cu ²⁺	Al ³⁺	HCO ₃ ⁻	SO ₄ ²⁻	S ²⁻	Cl ⁻	NO ₃ ⁻	PO ₄ ³⁻	B	Ni		
3	1.40	14.41	22-May-01	19.5	234.5	5.410	19.2	0.331	6.7	7.92	0.64	10.30	27.80	1.070	11.50	0.02	0.00	0.01	0.007	<0.002	0.009	<0.003	<0.01	<0.001	0.031	0.263	39.05	9.75	0.000	48.58	7.48	0.04	0.029	<0.003		
3	2.40	13.41	22-May-01	19.5	289.5	5.808	-73.2	-1.261	6.6	13.20	1.04	27.70	25.30	1.83	5.09	0.00	0.00	0.00	0.134	<0.0001	0.072	<0.002	<0.009	<0.001	<0.001	0.146	63.45	20.58	0.000	47.99	8.80	0.06	0.060	<0.003		
3	2.90	12.91	22-May-01	19.9	560.0	6.151	-13.0	-0.224	6.9	19.36	1.74	73.00	27.00	5.85	4.45	0.03	0.00	1.33	0.339	0.041	0.071	<0.002	<0.009	<0.001	<0.001	0.089	106.16	68.99	0.000	65.98	49.28	0.02	0.105	<0.003		
3	3.40	12.41	22-May-01	20.0	583.0	6.308	-212.4	-3.651	2.7	38.73	3.84	69.00	15.00	8.27	8.77	3.15	8.45	9.09	0.315	0.552	0.071	<0.002	<0.009	<0.001	<0.001	0.078	234.29	47.99	0.025	52.98	17.16	0.04	0.397	<0.003		
3	3.90	11.91	22-May-01	20.1	901.0	6.408	-180.5	-3.102	2.1	60.73	7.66	113.00	22.10	14.00	16.50	5.00	23.10	21.93	0.509	0.765	0.077	<0.002	<0.009	<0.001	0.011	0.149	467.35	83.39	0.011	27.99	10.56	0.04	0.619	<0.003		
3	4.40	11.41	22-May-01	20.3	1260.0	6.352	-180.6	-3.102	2.6	102.10	11.7	138.00	33.10	14.60	28.80	4.25	24.85	37.41	0.682	0.690	0.076	<0.002	<0.009	<0.001	0.007	0.092	713.84	75.29	0.009	33.85	10.56	0.05	0.772	<0.003		
3	4.90	10.91	22-May-01	20.0	1281.5	6.379	-131.6	-2.263	2.8	108.26	13.04	122.00	55.20	26.00	30.60	3.65	26.15	32.25	0.651	0.670	0.077	<0.002	<0.009	<0.001	0.009	0.095	795.60	22.89	0.000	57.98	12.76	0.14	0.685	<0.003		
3	5.40	10.41	22-May-01	20.4	1052.0	6.181	-130.4	-2.239	3.5	101.22	9.84	97.60	63.20	17.90	22.10	3.25	15.95	24.38	0.508	0.368	0.077	<0.002	<0.009	<0.001	0.009	0.100	600.36	5.88	0.000	65.98	0.00	0.10	0.672	<0.003		
3	5.90	9.91	22-May-01	20.1	943.0	6.164	-133.3	-2.291	3.0	82.74	8.34	74.60	64.40	13.00	21.10	16.70	6.70	21.54	0.319	0.333	0.077	<0.002	<0.009	<0.001	0.009	0.108	508.84	4.38	0.000	63.73	11.00	0.07	0.640	<0.003		
3	6.40	9.41	22-May-01	20.2	718.5	5.963	-136.9	-2.352	2.5	78.34	6.24	49.00	54.60	9.22	18.20	15.60	0.10	20.12	0.199	0.310	0.078	<0.002	<0.009	<0.001	0.015	0.117	380.71	3.06	0.000	54.98	13.20	0.07	0.572	<0.003		
3	6.90	8.91	22-May-01	DRY	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--		
3	7.40	8.41	22-May-01	DRY	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--		
3	7.90	7.91	22-May-01	19.7	198.9	5.103	-148.3	-2.552	6.2	33.45	0.78	7.89	21.10	0.383	9.06	1.49	0.24	7.74	<0.003	<0.002	0.011	<0.003	<0.01	<0.001	0.029	0.206	47.59	10.68	1.465	33.99	15.84	0.29	0.222	<0.003		
3	8.40	7.41	22-May-01	20.4	190.0	5.038	-180.6	-3.101	5.1	36.97	0.62	5.56	20.90	0.577	8.92	1.36	0.02	7.29	0.135	<0.002	0.011	<0.003	<0.01	<0.001	0.032	0.369	37.83	12.03	0.445	37.99	11.44	0.04	0.215	<0.003		
4	3.34	13.98	16-May-01	20.0	324.5	6.561	-193.1	-3.320	6.1	5.28	0.6	29.90	23.90	3.06	3.98	0.05	0.02	0.19	0.059	<0.0001	0.071	<0.002	<0.009	<0.001	<0.001	0.263	36.61	19.47	0.004	44.99	34.76	0.08	0.082	<0.003		
4	3.84	13.48	16-May-01	20.2	355.5	6.687	-231.1	-3.971	5.6	7.04	1.18	42.60	22.40	3.10	3.76	0.08	0.00	0.28	0.099	<0.0001	0.071	<0.002	<0.009	<0.001	<0.001	0.229	71.99	17.22	0.002	53.55	45.32	0.12	0.139	<0.003		
4	4.34	12.98	16-May-01	20.1	417.0	6.375	-268.9	-4.622	5.4	14.08	2.62	56.50	21.70	5.75	4.24	1.18	0.08	0.34	0.177	<0.0001	0.071	<0.002	<0.009	<0.001	<0.001	0.162	159.85	27.45	0.000	52.98	11.00	0.14	0.173	<0.003		
4	4.84	12.48	16-May-01	20.2	408.5	6.432	-239.3	-4.112	5.4	13.20	2.52	53.70	24.90	4.64	3.51	0.99	1.02	0.85	0.136	<0.0001	0.071	<0.002	<0.009	<0.001	<0.001	0.110	153.75	30.00	0.001	52.98	10.12	0.79	0.172	<0.003		
4	5.34	11.98	16-May-01	20.3	437.0	6.450	-225.0	-3.865	5.2	19.36	2.86	48.60	20.90	8.61	6.38	2.34	2.33	3.74	0.173	0.056	0.071	<0.002	<0.009	<0.001	<0.001	0.073	174.49	37.20	0.001	42.82	9.68	0.39	0.454	<0.003		
4	5.84	11.48	16-May-01	20.5	310.0	5.599	-163.9	-2.813	5.6	28.17	1.26	32.30	22.00	5.52	4.82	0.09	0.00	0.58	0.020	<0.002	0.011	<0.003	<0.01	<0.001	0.028	0.185	70.88	37.50	0.001	42.82	21.96	0.16	0.102	<0.003		
4	6.34	10.98	16-May-01	22.3	257.0	5.456	-129.7	-2.213	5.8	17.60	0.52	23.00	19.60	4.99	4.54	0.10	0.00	0.37	0.042	<0.0001	0.071	<0.002	<0.009	<0.001	0.009	0.127	31.73	31.80	0.001	38.99	31.32	0.10	0.060	<0.003		
4	6.84	10.48	16-May-01	24.3	244.0	5.629	-139.7	-2.367	5.7	15.84	0.6	21.70	19.80	4.12	4.61	0.12	0.00	0.00	0.014	<0.0001	0.070	<0.002	<0.009	<0.001	<0.001	0.078	34.61	25.68	0.020	39.99	26.04	0.22	0.056	<0.003		
4	7.34	9.98	16-May-01	23.0	251.0	5.645	-129.3	-2.201	5.9	14.96	0.74	22.20	20.70	4.04	4.59	0.11	0.00	0.05	0.017	<0.0001	0.071	<0.002	<0.009	<0.001	<0.001	0.067	42.15	24.06	0.000	37.83	30.00	0.07	0.062	<0.003		
4	7.84	9.48	16-May-01	21.7	266.5	5.730	-148.8	-2.543	5.4	15.84	0.98	23.20	26.60	4.52	5.29	0.06	0.00	0.03	0.056	<0.0001	0.071	<0.002	<0.009	<0.001	0.018	0.076	59.79	25.68	0.000	37.14	27.28	0.05	0.051	<0.003		
4	8.34	8.98	16-May-01	20.9	241.5	5.426	-149.2	-2.557	5.8	22.89	0.62	19.00	19.60	4.06	5.96	0.07	0.00	0.03	0.026	<0.0001	0.071	<0.002	<0.009	<0.001	<0.001	0.067	35.83	25.02	0.000	36.99	20.24	0.10	0.041	<0.003		
4	8.84	8.48	16-May-01	19.8	235.5	5.159	-152.3	-2.620	6.0	21.12	0.48	13.80	22.20	4.32	6.61	0.07	0.00	0.31	0.010	<0.0001	0.071	<0.002	<0.009	<0.001	<0.001	0.081	29.29	28.65	0.001	37.84	15.84	0.05	0.080	<0.003		
4	9.34	7.98	16-May-01	20.3	239.5	5.301	-165.5	-2.843	5.9	24.65	0.56	14.50	20.50	4.53	6.10	0.06	0.00	0.31	0.009	<0.0001	0.071	<0.002	<0.009	<0.001	0.002	0.095	24.17	27.93	0.000	37.14	20.24	0.21	0.089	<0.003		
4A	2.15	13.17	21-May-01	18.5	367.0	6.150	-160.5	-2.774	5.9	11.44	1.6	37.60	19.90	8.99	7.53	0.19	0.00	0.54	0.147	<0.0001	0.071	<0.002	<0.009	<0.001	<0.001	0.134	97.62	27.33	0.009	50.98	21.56	0.12	0.104	<0.003		
4A	2.65	12.67	21-May-01	19.4	792.0	6.053	-231.3	-3.985	5.2	39.61	7.06	106.00	33.40	20.80	13.60	0.33	0.																			

Table B4 (cont.) Groundwater chemistry results for bundled piezometers 3, 4, 4A, 6 and 10: May 2001 sampling event.

Bore ID	Depth (bgs ¹)	Elev. (AHD ²)	Sample Date	EC							Cations													Anions						Uncharged				
				T(°C)	EC (µS/cm)	pH	Eh (mV)	pe	O ₂	CO ₂	Alk. (meq/L)	Ca ²⁺	Na ⁺	Mg ²⁺	K ⁺	Fe ²⁺	Fe ³⁺	NH ₄ ⁺	Sr ²⁺	Mn ²⁺	Li ⁺	Zn ²⁺	Pb ²⁺	Cd ²⁺	Cu ²⁺	Al ³⁺	HCO ₃ ⁻	SO ₄ ²⁻	S ²⁻	Cl ⁻	NO ₃ ⁻	PO ₄ ³⁻	B	Ni
6	1.68	13.98	23-May-01	DRY	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
6	2.68	12.98	23-May-01	20.4	52.4	4.907	-137.5	-2.361	4.0	17.60	0.18	3.37	4.56	0.81	2.77	0.26	0.12	0.52	<0.0003	<0.0001	0.073	<0.002	<0.009	<0.001	<0.001	1.350	10.98	9.42	0.013	10.04	3.96	0.16	<0.001	<0.003
6	3.68	11.98	23-May-01	21.1	170.6	5.261	-156.3	-2.677	3.3	28.17	0.7	7.80	17.90	2.10	5.93	2.93	0.84	0.63	<0.003	<0.002	0.01	<0.003	<0.01	<0.001	0.027	0.265	42.71	4.50	0.071	33.84	3.96	0.10	0.029	<0.003
6	4.68	10.98	23-May-01	21.2	181.4	5.602	-167.0	-2.860	3.3	22.89	0.98	12.41	20.40	2.13	3.06	2.58	1.37	0.86	<0.0003	<0.0001	0.071	<0.002	<0.009	<0.001	<0.001	0.157	59.79	4.77	0.125	31.99	4.84	0.05	0.044	<0.003
6	6.18	9.48	23-May-01	20.8	266.0	5.797	-194.3	-3.332	5.6	28.17	1.7	19.20	23.50	3.67	5.34	3.10	1.10	3.43	0.020	<0.0001	0.070	<0.002	<0.009	<0.001	<0.001	0.130	103.72	16.62	0.096	28.10	5.28	0.15	0.129	<0.003
6	8.18	7.48	23-May-01	20.4	305.0	5.739	-191.0	-3.280	4.8	31.69	1.9	27.80	21.90	3.56	6.57	1.33	0.00	4.45	0.071	<0.0001	0.071	<0.002	<0.009	<0.001	<0.001	0.082	115.92	15.45	0.116	33.99	7.48	0.22	0.133	<0.003
6	10.18	5.48	23-May-01	19.8	626.0	5.717	-173.3	-2.982	3.8	100.34	4.58	37.80	44.20	6.99	16.10	5.05	3.09	20.64	0.130	0.017	0.071	<0.002	<0.009	<0.001	<0.001	0.094	279.43	13.32	0.014	57.52	6.16	3.25	0.174	<0.003
6	11.18	4.48	23-May-01	20.0	631.0	5.613	-177.3	-3.048	3.8	88.02	4.76	35.10	46.80	6.41	17.60	3.35	5.96	20.00	0.119	0.009	0.073	<0.002	<0.009	<0.001	<0.001	0.371	290.42	12.75	0.019	58.51	6.16	1.98	0.184	<0.003
6	12.18	3.48	23-May-01	19.8	690.5	5.698	-166.0	-2.856	3.4	117.95	5.04	27.30	54.50	10.20	20.80	4.25	3.35	28.38	0.076	<0.0001	0.079	<0.002	<0.009	<0.001	0.009	0.124	307.50	7.32	0.012	62.23	6.60	0.30	0.211	<0.003
6	13.18	2.48	23-May-01	20.7	709.0	5.792	-165.2	-2.833	3.9	119.71	5.44	12.20	54.90	11.80	25.20	4.40	13.00	28.38	<0.0003	<0.0001	0.095	<0.002	<0.009	<0.001	0.025	0.195	331.91	1.96	0.006	60.74	5.28	0.21	0.324	<0.003
6	14.18	1.48	23-May-01	20.7	624.5	5.649	-165.4	-2.837	4.4	100.34	4.6	10.01	50.00	8.52	19.60	5.50	8.30	25.80	<0.0003	<0.0001	0.071	<0.002	<0.009	<0.001	<0.001	0.155	280.66	1.32	0.006	60.98	5.72	0.15	0.259	<0.003
6	15.18	0.48	23-May-01	19.9	645.5	5.707	-163.4	-2.810	4.4	90.66	4.86	10.40	54.60	9.74	23.40	4.70	10.50	34.19	<0.0003	<0.0001	0.080	<0.002	<0.009	<0.001	0.008	0.147	296.52	1.38	0.006	59.98	4.84	2.67	0.293	<0.003
6	16.18	-0.52	23-May-01	20.4	683.0	5.812	-146.1	-2.509	6.1	93.30	5.34	12.60	55.50	10.40	24.40	5.35	9.35	34.83	<0.0003	<0.0001	0.079	<0.002	<0.009	<0.001	0.009	0.122	325.80	1.98	0.008	61.98	5.72	0.36	0.306	<0.003
6	18.18	-2.52	23-May-01	19.5	343.5	5.107	-115.2	-1.984	2.8	74.82	1.44	5.61	42.70	6.32	6.70	4.50	10.70	5.35	<0.0003	<0.0001	0.071	<0.002	<0.009	<0.001	0.035	0.092	87.86	44.99	0.000	47.99	5.28	0.24	0.089	<0.003
10	3.00	13.63	24-May-01	21.3	535.5	5.974	-158.5	-2.713	4.8	58.97	5.08	88.30	8.44	7.69	5.46	4.70	6.00	0.85	0.271	<0.0001	0.074	<0.002	<0.009	<0.001	<0.001	0.084	309.94	31.50	0.006	16.00	10.12	0.15	0.209	<0.003
10	4.00	12.63	24-May-01	21.6	720.5	6.348	-148.8	-2.545	5.5	61.17	7.07	88.20	12.80	12.70	9.26	3.22	16.68	20.64	0.454	0.204	0.078	<0.002	<0.009	<0.001	0.006	0.105	431.35	28.53	0.012	14.94	11.44	0.09	0.495	<0.003
10	5.00	11.63	24-May-01	21.5	829.5	6.357	-143.5	-2.455	6.3	72.62	8.06	80.00	15.20	16.60	17.20	4.65	12.85	30.96	0.385	0.346	0.077	<0.002	<0.009	<0.001	0.008	0.140	491.76	26.22	0.002	15.93	4.84	0.13	0.628	<0.003
10	6.00	10.63	24-May-01	21.3	1098.5	6.264	-151.6	-2.595	9.1	101.22	10.82	100.00	35.90	25.10	23.80	4.35	15.65	32.90	0.475	0.379	0.078	<0.002	<0.009	<0.001	0.007	0.105	660.15	13.68	0.009	38.99	8.80	0.13	0.650	<0.003
10	7.00	9.63	24-May-01	21.0	444.5	5.574	-151.3	-2.593	1.2	97.70	3.18	19.40	43.40	6.32	12.30	5.00	1.85	12.13	0.024	<0.0001	0.071	<0.002	<0.009	<0.001	<0.001	0.118	194.02	1.86	0.068	55.51	9.24	0.12	0.429	<0.003
10	8.00	8.63	24-May-01	20.7	378.0	5.356	-160.3	-2.750	2.1	106.50	2.14	9.41	42.60	4.10	11.80	2.90	0.37	10.97	<0.0003	<0.0001	0.071	<0.002	<0.009	<0.001	<0.001	0.106	126.57	4.23	0.147	57.74	10.12	0.18	0.384	<0.003
10	9.00	7.63	24-May-01	20.5	362.0	5.375	-162.8	-2.794	1.6	110.90	1.99	7.21	42.70	3.97	13.00	2.53	0.27	10.45	<0.003	<0.002	0.012	<0.003	<0.01	<0.001	0.025	0.205	118.41	0.85	0.213	49.21	31.24	0.10	0.414	<0.003
10	10.00	6.63	24-May-01	20.7	284.0	5.357	-165.7	-2.842	6.8	68.66	1.6	6.27	33.70	2.53	9.87	2.51	0.43	7.22	<0.0003	<0.0001	0.070	<0.002	<0.009	<0.001	<0.001	0.123	95.62	0.81	0.008	39.99	7.48	0.98	0.570	<0.003
10	11.00	5.63	24-May-01	20.3	255.0	5.462	-188.7	-3.241	5.0	45.77	1.24	12.30	23.90	3.45	8.28	0.08	0.00	4.90	<0.003	<0.002	0.011	<0.003	<0.01	<0.001	0.026	0.219	73.65	10.35	0.556	37.83	3.52	0.03	0.225	<0.003
10	12.00	4.63	24-May-01	DRY	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
10	13.00	3.63	24-May-01	20.3	314.0	5.768	-144.1	-2.475	2.5	28.17	1.52	18.90	23.30	6.77	7.35	2.30	0.39	3.74	0.020	<0.0001	0.071	<0.002	<0.009	<0.001	<0.001	0.124	92.74	20.55	0.014	41.99	5.28	0.00	0.186	<0.003

1. bgs = below ground surface (metres)

2. AHD = Australian Height Datum (metres)

3. All concentrations in mg/L unless otherwise noted (except pH and pe)

Table B5 Groundwater chemistry results for bundled piezometers 3, 4, 4A, 6 and 10: July 2001 sampling event.

Bore ID	Depth (bgs ¹)	Elev. (AHD ²)	Sample Date	EC								Cations																Anions						Uncharged	
				T(°C)	µS/cm	pH	Eh (mV)	pe	O ₂	CO ₂	Alk. (meq/L)	Ca ²⁺	Na ⁺	Mg ²⁺	K ⁺	Fe ²⁺	Fe ³⁺	NH ₄ ⁺	Sr ²⁺	Mn ²⁺	Li ⁺	Zn ²⁺	Pb ²⁺	Cd ²⁺	Cu ²⁺	Al ³⁺	HCO ₃ ⁻	SO ₄ ²⁻	S ²⁻	Cl ⁻	NO ₃ ⁻	PO ₄ ³⁻	B	Ni	
3	1.40	14.41	03-Aug-01	17.0	524.0	5.590	4.85	0.084	4.50	20.24	0.52	52.60	28.00	7.42	11.40	0.04	0.00	0.10	0.220	<0.0001	0.017	<0.001	<0.003	<0.0003	<0.0008	0.017	31.73	93.29	0.000	33.99	91.96	0.01	0.032	<0.02	
3	2.40	13.41	03-Aug-01	17.6	624.5	5.908	-27.35	-0.474	2.90	14.08	1.04	70.70	25.70	8.37	18.50	0.04	0.00	0.13	0.633	<0.0001	0.017	<0.001	<0.003	<0.0003	<0.0008	0.010	63.45	102.59	0.001	45.99	82.72	0.02	0.029	<0.02	
3	2.90	12.91	03-Aug-01	17.7	458.0	6.189	-148.35	-2.570	0.60	16.72	2.22	60.60	19.80	5.28	7.17	0.08	0.00	0.21	0.334	<0.0001	0.017	<0.001	<0.003	<0.0003	<0.0008	0.015	135.45	33.00	0.002	44.99	36.08	0.04	0.109	<0.02	
3	3.40	12.41	03-Aug-01	17.9	667.0	6.084	-177.3	-3.070	0.50	36.09	5.06	100.00	19.00	8.94	11.40	0.82	0.00	4.71	0.541	0.140	0.017	<0.001	<0.003	<0.0003	<0.0008	0.005	308.72	64.19	0.097	35.99	8.80	0.15	0.262	<0.02	
3	3.90	11.91	03-Aug-01	18.0	813.5	6.348	-140.8	-2.438	0.70	62.49	7.46	95.50	17.90	9.21	15.70	5.30	11.10	19.35	0.441	1.010	0.017	<0.001	<0.003	<0.0003	<0.0008	<0.003	455.15	47.99	0.066	21.99	14.52	0.06	0.556	<0.02	
3	4.40	11.41	02-Aug-01	18.8	1275.5	6.446	-188.8	-3.260	0.00	111.78	12.62	123.00	49.10	13.20	28.10	3.55	12.45	43.22	0.596	1.270	0.016	<0.001	<0.003	<0.0003	<0.0008	<0.003	759.97	34.80	0.018	34.13	16.28	0.14	0.759	<0.02	
3	4.90	10.91	03-Aug-01	17.8	1381.0	6.384	-143.55	-2.486	0.90	83.62	13.56	126.00	74.50	15.60	28.70	6.60	11.00	34.06	0.552	0.458	0.016	<0.001	<0.003	<0.0003	<0.0008	0.006	816.32	7.56	0.004	60.98	20.24	0.22	0.683	<0.02	
3	5.40	10.41	02-Aug-01	19.0	1270.5	6.370	-168.4	-2.905	0.00	88.90	11.82	92.80	64.30	17.10	29.70	4.00	0.00	54.18	0.478	0.299	0.017	<0.001	<0.003	<0.0003	<0.0008	<0.003	715.16	4.77	0.008	62.98	12.76	0.12	0.506	<0.02	
3	5.90	9.91	03-Aug-01	18.5	707.0	6.197	-171.7	-2.967	0.50	66.89	6.50	59.00	49.70	12.30	14.20	6.40	5.80	16.00	0.299	0.132	0.017	<0.001	<0.003	<0.0003	<0.0008	0.013	396.58	2.18	0.000	46.99	12.76	0.26	0.571	<0.02	
3	6.40	9.41	02-Aug-01	19.2	509.5	5.942	-210.6	-3.631	0.00	49.29	3.92	29.30	39.60	3.92	12.20	4.65	4.88	15.29	0.102	0.162	0.017	<0.001	<0.003	<0.0003	<0.0008	0.024	239.17	1.69	0.025	44.99	12.76	0.10	0.476	<0.02	
3	6.90	8.91	02-Aug-01	DRY	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--		
3	7.40	8.41	02-Aug-01	DRY	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--		
3	7.90	7.91	02-Aug-01	18.7	191.2	5.127	-207.95	-3.591	0.30	29.93	0.54	<0.004	24.90	2.13	7.55	1.89	0.00	7.80	<0.0001	<0.0001	0.017	<0.001	<0.003	<0.0003	<0.0008	0.014	30.95	13.81	1.315	33.99	14.72	0.04	0.172	<0.02	
3	8.40	7.41	03-Aug-01	17.6	200.5	4.833	-179.85	-3.118	1.50	34.33	0.46	0.04	22.90	1.78	7.48	1.64	0.00	7.48	<0.0001	<0.0001	0.018	<0.001	<0.003	<0.0003	<0.0008	0.015	27.07	14.68	0.371	33.99	12.96	0.10	0.177	<0.02	
3	8.90	6.91	02-Aug-01	18.6	192.5	5.012	-195.65	-3.380	0.50	36.09	0.50	<0.004	22.20	1.70	7.31	0.87	0.00	8.84	<0.0001	<0.0001	0.017	<0.001	<0.003	<0.0003	<0.0008	0.006	25.51	14.50	0.355	35.83	11.88	0.01	0.163	<0.02	
3	9.40	6.41	02-Aug-01	18.1	191.5	4.992	-183.8	-3.181	0.90	36.09	0.32	0.24	21.50	1.99	7.10	0.81	0.00	7.42	<0.0001	<0.0001	0.017	<0.001	<0.003	<0.0003	<0.0008	<0.003	19.52	18.99	0.417	33.84	8.80	0.03	0.148	<0.02	
3	10.40	5.41	02-Aug-01	18.3	191.4	4.915	-181.4	-3.137	0.30	32.57	0.40	<0.004	21.80	2.36	6.92	0.62	0.00	6.71	<0.0001	<0.0001	0.017	<0.001	<0.003	<0.0003	<0.0008	0.006	24.40	17.70	0.117	33.35	6.60	0.03	0.181	<0.02	
3	11.40	4.41	02-Aug-01	17.3	180.7	5.013	-170.25	-2.954	1.20	23.77	0.28	3.76	21.30	2.85	6.18	0.27	0.00	3.68	<0.0001	<0.0001	0.018	<0.001	<0.003	<0.0003	<0.0008	0.012	17.08	23.16	0.257	32.83	6.60	0.04	0.110	<0.02	
3	12.40	3.41	02-Aug-01	18.5	179.2	5.026	-192.5	-3.326	0.20	20.24	0.26	7.38	20.40	3.11	5.84	0.06	0.00	0.27	<0.0001	<0.0001	0.018	<0.001	<0.003	<0.0003	<0.0008	0.012	15.86	25.90	0.261	31.49	7.48	0.03	0.046	<0.02	
3	13.40	2.41	02-Aug-01	17.3	190.7	5.173	-179.15	-3.108	1.40	18.48	0.36	9.75	19.80	3.26	5.18	0.05	0.00	0.26	<0.0001	<0.0001	0.017	<0.001	<0.003	<0.0003	<0.0008	0.006	21.96	27.43	0.060	30.99	6.16	0.03	0.038	<0.02	
3	14.40	1.41	02-Aug-01	17.5	206.0	5.275	-194.65	-3.375	0.70	14.08	0.42	10.80	19.20	3.80	4.59	0.04	0.00	0.35	0.022	<0.0001	0.017	<0.001	<0.003	<0.0003	<0.0008	0.102	25.63	29.30	0.055	29.99	7.92	0.03	0.031	<0.02	
3	15.40	0.41	02-Aug-01	17.2	212.0	5.278	-187.8	-3.260	0.90	17.60	0.50	11.90	20.60	4.40	4.38	0.03	0.00	0.36	0.010	<0.0001	0.017	<0.001	<0.003	<0.0003	<0.0008	<0.003	30.51	31.10	0.057	29.99	9.24	0.76	0.029	<0.02	
3	16.40	-0.59	02-Aug-01	17.7	244.5	5.308	-188.4	-3.265	0.50	10.56	0.36	14.60	21.40	4.78	3.15	0.02	0.00	0.05	0.034	<0.0001	0.017	<0.001	<0.003	<0.0003	<0.0008	<0.003	21.96	35.79	0.024	30.99	19.00	0.05	0.018	<0.02	
3	17.40	-1.59	02-Aug-01	16.4	257.5	5.270	-171.45	-2.985	1.80	10.56	0.26	14.30	21.20	4.55	4.80	0.04	0.00	0.06	<0.0001	<0.0001	0.017	<0.001	<0.003	<0.0003	<0.0008	<0.003	15.86	39.59	0.079	29.99	23.40	0.04	0.015	<0.02	
3	18.40	-2.59	02-Aug-01	16.5	265.5	5.070	-159.6	-2.777	1.70	7.92	0.20	19.10	12.00	7.19	5.75	0.03	0.00	0.13	<0.0001	<0.0001	0.017	<0.001	<0.003	<0.0003	<0.0008	<0.003	12.20	43.79	0.054	30.99	25.28	0.04	0.016	<0.02	
3	19.40	-3.59	02-Aug-01	18.0	247.5	4.115	-85.8	-1.485	2.00	13.20	0.08	13.10	20.60	4.90	5.73	0.02	0.00	0.09	<0.0001	<0.0001	0.017	<0.001	<0.003	<0.0003	<0.0008	0.150	4.88	39.89	0.008	31.99	28.16	0.95	0.016	<0.02	
4	1.84	15.48	06-Sep-01	DRY	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--		
4	2.34	14.98	06-Sep-01	DRY	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--		
4	2.84	14.48	06-Sep-01	17.9	365.0	5.687	-208.6	-3.613	0.46	17.																									

Table B5 (cont.) Groundwater chemistry results for bundled piezometers 3, 4, 4A, 6 and 10: July 2001 sampling event.

Bore Depth ID (bgs ¹)			Elev. (AHD ²)	Sample Date	EC							Cations											Anions						Uncharged								
					T(°C)	EC (µS/cm)	pH	Eh (mV)	pe	O ₂	CO ₂	Alk. (meq/L)	Ca ²⁺	Na ⁺	Mg ²⁺	K ⁺	Fe ²⁺	Fe ³⁺	NH ₄ ⁺	Sr ²⁺	Mn ²⁺	Li ⁺	Zn ²⁺	Pb ²⁺	Cd ²⁺	Cu ²⁺	Al ³⁺	HCO ₃ ⁻			SO ₄ ²⁻	S ²⁻	Cl ⁻	NO ₃ ⁻	PO ₄ ³⁻		
4	11.34	5.98	10-Sep-01	18.7	216.0	4.337	-115.9	-2.002	0.27	21.12	0.10	13.20	18.70	2.48	3.71	0.10	0.00	0.03	0.006	<0.0001	0.017	<0.001	<0.003	<0.0003	<0.0008	0.453	6.10	31.90	0.000	31.99	16.92	0.31	0.021	<0.02			
4	12.34	4.98	10-Sep-01	18.5	205.0	4.870	-124.8	-2.157	1.91	22.89	0.16	12.90	18.80	2.47	3.97	0.04	0.00	0.08	0.159	<0.0001	0.018	<0.001	<0.003	<0.0003	<0.0008	0.465	9.76	31.90	0.001	30.99	16.92	0.13	0.028	<0.02			
4	13.34	3.98	10-Sep-01	18.7	212.0	4.443	-123.7	-2.136	0.47	24.65	0.12	12.60	18.50	2.49	4.18	0.10	0.00	0.08	0.013	<0.0001	0.018	<0.001	<0.003	<0.0003	<0.0008	0.431	7.32	31.80	0.003	29.99	20.44	0.02	0.026	<0.02			
4	14.34	2.98	10-Sep-01	18.4	204.0	4.838	-152.7	-2.639	0.56	20.24	0.16	11.11	18.30	3.52	5.53	0.10	0.00	0.05	0.001	<0.0001	0.017	<0.001	<0.003	<0.0003	<0.0008	0.139	9.76	30.40	0.003	30.99	15.16	0.28	0.024	<0.02			
4	15.34	1.98	10-Sep-01	18.6	208.0	4.700	-146.8	-2.536	0.28	22.89	0.16	10.21	18.40	3.57	5.99	0.02	0.00	0.01	0.015	<0.0001	0.017	<0.001	<0.003	<0.0003	<0.0008	0.101	9.76	31.60	0.006	31.99	9.88	0.31	0.022	<0.02			
4	16.34	0.98	10-Sep-01	18.2	203.0	4.906	-141.8	-2.453	0.99	15.84	0.12	10.40	18.60	3.56	5.95	0.02	0.00	0.04	0.002	<0.0001	0.018	<0.001	<0.003	<0.0003	<0.0008	0.094	7.32	30.70	0.013	31.99	15.04	0.18	0.025	<0.02			
4	18.34	-1.02	10-Sep-01	18.0	207.5	4.645	-133.1	-2.304	0.92	17.60	0.12	10.53	19.40	3.50	4.58	0.05	0.00	0.00	0.017	<0.0001	0.017	<0.001	<0.003	<0.0003	<0.0008	0.271	7.32	31.60	0.001	31.99	12.40	0.29	0.022	<0.02			
4A	2.15	13.17	06-Aug-01	16.1	292.0	6.067	-176.3	-3.072	0.66	6.16	1.92	29.50	17.30	5.79	9.00	0.01	0.00	0.22	0.121	<0.0001	0.017	<0.001	<0.003	<0.0003	<0.0008	0.066	117.14	26.07	0.014	19.99	7.92	0.48	0.083	<0.02			
4A	2.65	12.67	06-Aug-01	16.9	591.0	6.463	-241.75	-4.201	0.16	24.65	5.58	82.40	25.40	13.90	9.07	1.51	0.00	3.47	0.445	0.291	0.017	<0.001	<0.003	<0.0003	<0.0008	0.041	340.45	26.10	0.174	25.99	9.24	0.05	0.167	<0.02			
4A	3.15	12.17	06-Aug-01	17.1	962.0	6.142	-168.45	-2.925	0.82	105.62	8.88	101.00	41.80	19.00	17.60	5.40	7.00	16.38	0.520	0.488	0.017	<0.001	<0.003	<0.0003	<0.0008	0.046	541.79	7.86	0.060	50.98	15.40	0.10	0.294	<0.02			
4A	3.65	11.67	06-Aug-01	17.9	878.5	6.119	-211.25	-3.658	0.26	95.06	8.66	83.80	44.10	17.30	18.30	9.15	1.35	19.67	0.398	0.358	0.017	<0.001	<0.003	<0.0003	<0.0008	0.047	518.36	7.55	0.091	50.98	13.84	0.08	0.315	<0.02			
4A	4.15	11.17	06-Aug-01	17.9	852.5	6.043	-192.5	-3.333	0.21	86.26	7.86	73.20	40.90	19.70	18.30	4.50	0.00	28.90	0.288	0.233	0.017	<0.001	<0.003	<0.0003	<0.0008	0.043	479.55	7.29	0.130	49.18	12.32	0.32	0.327	<0.02			
4A	4.65	10.67	06-Aug-01	18.5	796.5	6.225	-192.1	-3.319	0.50	75.70	7.16	66.20	40.70	16.60	20.40	4.75	0.00	23.87	0.185	0.381	0.017	<0.001	<0.003	<0.0003	<0.0008	0.029	436.85	3.63	0.014	50.98	15.84	0.08	0.516	<0.02			
4A	5.15	10.17	06-Aug-01	18.5	817.0	6.318	-170.35	-2.944	0.14	71.30	7.66	78.70	40.10	14.10	17.10	3.05	8.45	20.64	0.327	0.340	0.017	<0.001	<0.003	<0.0003	<0.0008	0.010	467.35	3.48	0.005	45.99	11.44	0.02	0.584	<0.02			
4A	5.65	9.67	23-Aug-01	17.5	800.0	6.166	-66.75	-1.158	0.24	54.57	7.40	78.90	40.80	13.30	14.80	7.10	6.90	15.35	0.354	0.319	0.017	<0.001	<0.003	<0.0003	<0.0008	0.010	451.49	3.33	0.013	42.99	12.32	0.01	0.665	<0.02			
4A	6.15	9.17	06-Aug-01	18.0	676.5	6.181	-142.7	-2.470	0.57	73.06	5.94	57.30	42.30	9.07	13.60	4.70	5.80	13.55	0.702	0.148	0.017	<0.001	<0.003	<0.0003	<0.0008	0.008	362.41	2.18	0.004	46.80	10.12	0.09	0.670	<0.02			
4A	6.65	8.67	23-Aug-01	18.2	621.0	5.752	-130.8	-2.262	0.14	98.41	5.22	45.20	43.90	9.02	14.30	5.00	1.83	14.64	0.177	0.067	0.017	<0.001	<0.003	<0.0003	<0.0008	0.009	318.48	1.80	0.006	44.99	11.44	0.01	0.643	<0.02			
4A	7.15	8.17	06-Aug-01	18.4	606.5	6.085	-165.4	-2.859	0.33	62.49	5.08	42.80	42.90	8.59	14.00	2.94	3.82	12.64	0.231	0.056	0.017	<0.001	<0.003	<0.0003	<0.0008	0.008	309.94	1.59	0.005	45.17	11.88	0.04	0.647	<0.02			
4A	7.65	7.67	23-Aug-01	17.5	216.0	5.252	-174.65	-3.029	0.18	31.69	0.72	11.67	22.60	4.12	6.19	0.42	0.00	2.63	0.010	<0.0001	0.017	<0.001	<0.003	<0.0003	<0.0008	0.035	43.93	17.13	1.020	34.84	11.88	0.01	0.080	<0.02			
4A	8.15	7.17	06-Aug-01	18.5	214.0	5.191	-184.75	-3.192	0.24	34.33	0.48	11.42	21.70	3.83	6.03	0.31	0.00	1.90	0.013	<0.0001	0.016	<0.001	<0.003	<0.0003	<0.0008	0.012	29.29	25.44	0.231	34.99	8.36	0.18	0.075	<0.02			
4A	8.65	6.67	23-Aug-01	17.9	201.1	5.048	-169.3	-2.932	0.24	24.65	0.42	11.69	21.50	2.95	5.71	0.22	0.00	1.43	<0.0001	<0.0001	0.017	<0.001	<0.003	<0.0003	<0.0008	0.018	25.63	28.89	0.121	32.85	11.00	0.01	0.069	<0.02			
4A	9.15	6.17	23-Aug-01	18.0	199.4	5.026	-163.15	-2.824	0.30	21.12	0.42	11.86	19.80	3.17	5.63	0.21	0.00	0.97	0.006	<0.0001	0.017	<0.001	<0.003	<0.0003	<0.0008	0.010	25.63	29.58	0.083	31.99	6.60	0.10	0.060	<0.02			
4A	9.65	5.67	23-Aug-01	DRY	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--			
4A	10.15	5.17	23-Aug-01	15.6	166.6	4.984	-137.05	-2.392	1.46	19.36	0.20	5.40	21.70	1.65	5.39	0.23	0.00	1.61	<0.0001	<0.0001	0.017	<0.001	<0.003	<0.0003	<0.0008	0.027	12.20	25.11	0.060	30.86	7.04	0.02	0.074	<0.02			
4A	11.15	4.17	23-Aug-01	17.1	169.4	4.791	-151.95	-2.639	0.44	28.17	0.20	5.97	20.40	1.97	5.63	0.18	0.00	1.29	<0.0001	<0.0001	0.018	<0.001	<0.003	<0.0003	<0.0008	0.039	12.20	27.30	0.039	28.99	6.60	0.02	0.063	<0.02			
4A	12.15	3.17	23-Aug-01	16.1	185.0	5.028	-145.4	-2.534	1.00	11.44	0.24	8.94	16.00	3.62	4.94	0.09	0.00	0.69	<0.0001	<0.0001	0.018	<0.001	<0.003	<0.0003	<0.0008	0.017	11.64	30.60	0.000	26.99	9.76	0.01	0.028	<0.02			
4A	13.15	2.17	23-Aug-01	16.8	185.3	4.786	-139.45	-2.424	0.76	7.04	0.36	9.88	16.20	3.40	5.15	0.07	0.00	0.54	<0.0001	<0.0001	0.017	<0.001	<0.003	<0.0003	<0.0008	0.069	13.96	30.20	0.000	26.87	10.64	0.00	0.027	<0.02			
4A	14.15	1.17	23-Aug-01	16.9	187.1	4.495	-150.15	-2.609	0.45	9.68	0.12	9.23	16.40	3.20	4.99	0.06	0.00	0.58	<0.0001	<0.0001	0.018	<0.001	<0.003	<0.0003	<0.0008	0.036	7.32	30.30	0.016	26.99	13.84	0.02	0.028	<0.02			
4A	15.15	0.17	23-Aug-01	DRY	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--				
4A	16.15	-0.83	23-Aug-01	17.6	237.0	5.583	-170.35	-2.953	1.25	7.04	0.62	17.60	16.50	3.59	8.26	0.04	0.00	0.12	0.072	<0.0001	0.016	<0.001	<0.003	<0.0003	<0.0008	0.004	37.83	20.47	0.013	26.99	28.60	0.28	0.020	<0.02			
4A	17.15	-1.83	23-Aug-01	18.2	223.0	5.484	-159.5	-2.759	1.04	7.04	0.60	15.80	16.20	3.80	9.52	0.04	0.00	0.05	0.051	<0.0001	0.018	<0.001	<0.003	<0.0003	<0.0008	<0.003	36.61	20.52	0.009	25.99	28.16	0.01	0.023	<0.02			
4A	18.15	-2.83	23-Aug-01	17.8	207.0	5.407	-160.15	-2.774	1.38	15.84	0.34	13.80	16.40	3.83	5.92	0.07	0.00	0.03	<0.0001	<0.0001	0.017	<0.001	<0.003	<0.0003	<0.0008	0.004	20.74	21.18	0.017	25.99	29.04	0.03	0.018	<0.02			
4A	19.15	-3.83	23-Aug-01	18.5	199.1	4.666	-106.7	-1.844	1.25	13.20	0.10	11.35	15.70	3.81	5.33	0.04	0.00	0.06	<0.0001	<0.0001	0.017	<0.001	<0.003	<0.0003	<0.0008	0.131	6.10	21.12	0.006	26.88	31.24	0.03	0.016	<0.02			
6	1.68	13.98	31-Jul-01	DRY	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--			
6	2.68	12.98	31-Jul-01	18.3	163.8	4.710	-99.55	-1.721	4.4	8.80	0.1	8.93	18.20	2.05	5.53	0.06	0.00	0.08	0.015	<0.0001	0.018	<0.001	<0.003	<0.0003	<0.0008	0.											

Table B5 (cont.) Groundwater chemistry results for bundled piezometers 3, 4, 4A, 6 and 10: July 2001 sampling event.

Bore Depth ID (bgs ¹)	Elev. (AHD ²)	Sample Date	EC								Cations													Anions						Uncharged				
			T(°C)	EC (µS/cm)	pH	Eh (mV)	pe	O ₂	CO ₂	Alk. (meq/L)	Ca ²⁺	Na ⁺	Mg ²⁺	K ⁺	Fe ²⁺	Fe ³⁺	NH ₄ ⁺	Sr ²⁺	Mn ²⁺	Li ⁺	Zn ²⁺	Pb ²⁺	Cd ²⁺	Cu ²⁺	Al ³⁺	HCO ₃ ⁻	SO ₄ ²⁻	S ²⁻	Cl ⁻	NO ₃ ⁻	PO ₄ ³⁻	B	Ni	
6	8.68	6.98	30-Jul-01	19.0	375.0	5.510	-236.9	-4.086	2.5	52.81	2.4	38.10	25.50	4.40	7.25	1.45	0.00	5.22	0.108	0.044	0.017	<0.001	<0.003	<0.0003	<0.0008	0.014	143.99	27.00	0.077	40.15	10.56	0.33	0.114	<0.02
6	9.18	6.48	30-Jul-01	18.9	309.0	5.518	-227.9	-3.933	2.9	38.73	2.0	22.70	26.40	3.21	8.91	1.30	0.00	11.61	0.046	<0.0001	0.018	<0.001	<0.003	<0.0003	<0.0008	0.016	123.24	8.25	0.162	38.99	9.68	0.31	0.088	<0.02
6	9.68	5.98	30-Jul-01	19.0	756.0	5.740	-221.35	-3.819	0.8	77.46	5.6	19.00	49.80	6.14	28.70	3.40	0.00	31.73	0.039	<0.0001	0.017	<0.001	<0.003	<0.0003	<0.0008	0.021	320.11	0.90	0.012	48.74	7.80	0.19	0.258	<0.02
6	10.18	5.48	30-Jul-01	19.1	754.0	5.751	-219.95	-3.793	0.7	88.02	5.8	14.20	47.80	8.48	28.00	4.15	0.00	47.21	0.020	<0.0001	0.017	<0.001	<0.003	<0.0003	<0.0008	0.058	351.31	0.75	0.004	57.74	8.24	0.95	0.267	<0.02
6	10.68	4.98	30-Jul-01	19.0	744.5	5.717	-217.85	-3.758	1.1	88.90	5.6	15.40	49.20	8.85	28.20	4.70	0.00	39.73	0.015	<0.0001	0.017	<0.001	<0.003	<0.0003	<0.0008	0.064	330.45	1.17	0.022	57.98	8.24	0.02	0.252	<0.02
6	11.18	4.48	30-Jul-01	18.8	751.5	5.735	-211.2	-3.646	0.6	101.22	5.9	15.30	49.80	9.12	29.00	4.05	0.00	42.80	0.285	<0.0001	0.017	<0.001	<0.003	<0.0003	<0.0008	0.050	353.97	0.84	0.010	57.98	7.36	0.02	0.263	<0.02
6	11.68	3.98	30-Jul-01	18.3	736.5	5.762	-200.15	-3.461	0.8	93.30	5.9	12.20	50.10	9.42	28.30	3.60	0.00	49.28	0.029	<0.0001	0.017	<0.001	<0.003	<0.0003	<0.0008	0.025	362.41	0.82	0.004	58.98	7.36	0.08	0.267	<0.02
6	12.18	3.48	30-Jul-01	18.8	753.0	5.688	-202.05	-3.488	0.5	98.58	5.9	10.90	51.30	10.40	29.00	4.25	0.00	63.47	0.018	<0.0001	0.017	<0.001	<0.003	<0.0003	<0.0008	0.013	361.19	0.75	0.004	58.75	7.92	0.11	0.277	<0.02
6	12.68	2.98	30-Jul-01	18.6	724.0	5.730	-155.6	-2.688	0.4	91.54	5.8	11.30	49.10	12.50	24.70	7.90	4.80	40.76	0.038	<0.0001	0.017	<0.001	<0.003	<0.0003	<0.0008	0.031	355.09	0.61	0.008	58.75	10.56	1.97	0.302	<0.02
6	13.18	2.48	25-Jul-01	18.9	639.0	5.693	-184.55	-3.185	0.6	78.34	4.8	9.56	48.80	9.91	20.20	6.95	6.95	30.96	<0.0001	<0.0001	0.017	<0.001	<0.003	<0.0003	<0.0008	0.051	291.64	0.55	0.016	60.22	8.80	0.18	0.258	<0.02
6	13.68	1.98	25-Jul-01	19.0	564.5	5.517	-205.55	-3.546	0.6	91.54	4.0	6.26	48.30	7.24	18.20	6.55	4.05	27.48	<0.0001	<0.0001	0.018	<0.001	<0.003	<0.0003	<0.0008	0.195	246.49	0.39	0.013	58.98	10.56	0.03	0.213	<0.02
6	14.18	1.48	25-Jul-01	18.8	564.0	5.563	-188.15	-3.248	1.1	95.94	4.0	6.11	48.40	7.25	18.20	8.60	2.50	28.25	<0.0001	<0.0001	0.018	<0.001	<0.003	<0.0003	<0.0008	0.042	244.05	0.37	0.000	59.74	10.56	0.65	0.215	<0.02
6	15.18	0.48	25-Jul-01	19.0	571.5	5.544	-209.8	-3.619	0.6	84.50	4.2	6.71	48.60	7.43	18.40	7.20	4.70	32.51	<0.0001	<0.0001	0.017	<0.001	<0.003	<0.0003	<0.0008	0.029	253.81	0.46	0.012	60.98	11.44	0.10	0.221	<0.02
6	16.18	-0.52	25-Jul-01	19.2	575.0	5.649	-207.4	-3.576	2.5	96.82	3.1	6.02	44.50	7.08	18.00	8.40	5.40	28.51	<0.0001	<0.0001	0.017	<0.001	<0.003	<0.0003	<0.0008	0.038	190.92	13.57	0.017	59.98	11.00	0.14	0.226	<0.02
6	18.18	-2.52	25-Jul-01	18.8	382.5	5.171	-231.7	-4.000	1.4	93.30	1.8	4.84	40.40	6.51	7.81	10.30	5.10	7.03	<0.0001	<0.0001	0.017	<0.001	<0.003	<0.0003	<0.0008	0.005	103.82	35.89	0.015	47.99	11.00	0.07	0.081	<0.02
10	3.00	13.63	22-Aug-01	19.4	1019.0	5.786	-146.0	-2.515	0.27	119.71	10.40	185.00	33.40	14.10	4.32	0.92	0.00	0.12	0.695	0.138	0.02	0.042	<0.003	<0.0003	<0.0008	<0.003	634.52	37.50	0.002	32.99	21.12	0.14	0.251	<0.02
10	4.00	12.63	22-Aug-01	19.7	1246.0	6.173	-167.1	-2.876	0.16	126.75	13.06	184.00	62.00	16.30	4.07	5.35	4.85	5.48	0.360	0.110	0.017	<0.001	<0.003	<0.0003	<0.0008	<0.003	796.82	8.82	0.010	54.98	20.24	0.05	0.322	<0.02
10	5.00	11.63	22-Aug-01	19.9	1649.5	6.359	-167.2	-2.875	0.10	143.47	16.68	149.00	81.60	30.40	32.70	5.00	4.69	37.73	0.572	0.098	0.018	<0.001	<0.003	<0.0003	<0.0008	<0.003	1007.68	6.78	0.003	56.70	14.52	0.10	0.448	<0.02
10	6.00	10.63	22-Aug-01	20.0	1307.5	6.181	-154.4	-2.655	0.09	111.78	12.16	90.70	76.20	24.90	30.70	5.25	4.63	39.35	0.372	0.302	0.019	<0.001	<0.003	<0.0003	<0.0008	0.006	741.91	4.53	0.007	59.98	27.72	0.10	0.489	<0.02
10	7.00	9.63	22-Aug-01	20.1	294.5	5.394	-163.9	-2.817	0.12	56.33	1.52	4.46	37.00	4.31	10.30	1.87	0.00	7.61	<0.0001	<0.0001	0.017	<0.001	<0.003	<0.0003	<0.0008	0.043	92.74	0.54	0.193	45.99	22.44	0.04	0.415	<0.02
10	8.00	8.63	22-Aug-01	20.0	291.5	5.233	-175.0	-3.009	0.96	78.34	1.38	4.37	36.90	4.23	10.00	1.48	0.00	7.55	<0.0001	<0.0001	0.018	<0.001	<0.003	<0.0003	<0.0008	0.024	84.20	4.14	0.263	48.98	17.60	0.08	0.368	<0.02
10	9.00	7.63	22-Aug-01	19.9	298.0	5.266	-182.0	-3.130	0.15	69.54	1.54	4.62	37.00	4.34	10.50	2.08	0.00	8.00	<0.0001	<0.0001	0.018	<0.001	<0.003	<0.0003	<0.0008	0.028	93.96	0.35	0.158	45.80	20.24	0.04	0.415	<0.02
10	10.00	6.63	22-Aug-01	20.1	285.0	5.419	-197.2	-3.389	0.13	54.57	1.68	5.98	29.30	4.28	10.70	2.41	0.00	10.26	<0.0001	<0.0001	0.017	<0.001	<0.003	<0.0003	<0.0008	0.027	102.50	0.84	0.163	36.99	17.16	0.02	0.456	<0.02
10	11.00	5.63	22-Aug-01	19.8	265.0	5.567	-225.4	-3.878	0.13	27.29	1.44	13.00	23.30	5.93	9.03	1.32	0.00	3.42	0.019	<0.0001	0.017	<0.001	<0.003	<0.0003	<0.0008	0.025	87.86	8.17	0.476	33.99	17.60	0.03	0.156	<0.02
10	12.00	4.63	22-Aug-01	DRY	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
10	13.00	3.63	22-Aug-01	19.9	310.0	5.772	-193.5	-3.328	0.18	29.93	1.84	18.30	23.90	5.70	7.70	1.88	0.00	3.74	0.038	<0.0001	0.017	<0.001	<0.003	<0.0003	<0.0008	0.015	92.26	16.29	0.021	35.99	13.64	0.02	0.165	<0.02
10	14.00	2.63	22-Aug-01	19.8	268.5	5.656	-161.8	-2.784	1.15	21.12	1.34	16.60	22.40	4.79	8.48	0.55	0.00	2.84	0.066	<0.0001	0.017	<0.001	<0.003	<0.0003	<0.0008	0.146	71.76	19.62	0.015	32.99	13.64	0.50	0.108	<0.02
10	15.00	1.63	22-Aug-01	19.6	276.5	6.048	-216.9	-3.734	0.40	9.68	1.48	18.10	19.90	5.29	10.70	0.81	0.00	3.74	0.218	0.039	0.016	<0.001	<0.003	<0.0003	<0.0008	0.027	80.30	23.10	0.012	29.99	14.08	0.02	0.117	<0.02
10	16.00	0.63	22-Aug-01	19.5	241.0	6.095	-200.1	-3.446	0.35	13.20	1.12	16.00	17.30	4.26	10.40																			

Table B6 Groundwater chemistry results for bundled piezometers 4A and 30: February 2002 sampling event.

Bore ID	Depth (bgs ¹)	Elev. (AHD ²)	Sample Date	EC								Cations															Anions						Uncharged	
				T(°C)	µS/cm	pH	Eh (mV)	pe	O ₂	CO ₂	Alk. (meq/L)	Ca ²⁺	Na ⁺	Mg ²⁺	K ⁺	Fe ²⁺	Fe ³⁺	NH ₄ ⁺	Sr ²⁺	Mn ²⁺	Li ⁺	Zn ²⁺	Pb ²⁺	Cd ²⁺	Cu ²⁺	Al ³⁺	HCO ₃ ⁻	SO ₄ ²⁻	S ²⁻	Cl ⁻	NO ₃ ⁻	PO ₄ ³⁻	B	Ni
4A	2.15	13.17	14-Feb-02	22.1	466	6.548	-112.9	-1.927	0.24	54.57	3.64	56.10	23.30	10.50	7.16	0.16	0.00	0.30	0.272	<0.0001	<0.0005	0.117	<0.004	<0.0003	0.006	0.046	222.08	19.67	0.114	40.99	8.36	0.44	0.120	<0.0009
4A	2.65	12.67	14-Feb-02	21.1	675	6.644	-148.4	-2.541	0.05	95.06	6.68	96.50	23.80	18.50	8.63	1.48	0.00	3.03	0.503	0.312	<0.0005	0.083	<0.004	<0.0003	0.007	0.053	407.56	11.84	0.116	39.99	6.16	0.15	0.190	<0.0009
4A	3.15	12.17	14-Feb-02	21.3	873	6.412	-118.3	-2.025	0.06	198.92	8.82	107.00	37.80	19.60	13.80	1.79	11.61	11.35	0.543	0.513	<0.0005	0.146	<0.004	<0.0003	0.008	0.078	538.13	9.20	0.017	40.99	24.20	0.08	0.297	0.001
4A	3.65	11.67	14-Feb-02	20.7	826	6.354	-80.7	-1.383	0.56	170.76	7.78	92.80	35.10	18.90	13.20	1.64	8.96	11.48	0.447	0.435	<0.0005	0.042	<0.004	<0.0003	0.007	0.056	474.67	8.49	0.046	41.99	15.84	1.19	0.290	<0.0009
4A	4.15	11.17	14-Feb-02	20.9	803	6.334	-97.8	-1.675	0.09	153.15	7.20	94.40	35.20	19.80	13.10	0.70	0.00	10.38	0.360	0.397	<0.0005	0.032	<0.004	<0.0003	0.006	0.049	439.29	7.70	0.016	43.99	7.48	0.77	0.287	<0.0009
4A	4.65	10.67	14-Feb-02	20.3	770	6.344	-100.6	-1.727	0.11	172.52	6.72	63.60	41.10	17.50	20.40	4.55	1.17	19.35	0.197	0.376	<0.0005	0.139	<0.004	<0.0003	0.007	0.056	410.00	5.54	0.004	47.99	8.80	3.10	0.479	<0.0009
4A	5.15	10.17	14-Feb-02	20.0	1047	6.545	-92.2	-1.584	0.07	146.11	9.92	112.00	42.60	19.50	19.20	5.85	14.95	22.90	0.470	0.573	<0.0005	0.081	<0.004	<0.0003	0.007	0.023	605.24	8.05	0.009	52.98	2.64	0.29	0.591	<0.0009
4A	5.65	9.67	14-Feb-02	20.4	1060	6.521	-85.5	-1.468	0.04	179.56	10.20	114.00	44.60	18.50	21.00	6.15	15.05	22.90	0.468	0.459	<0.0005	0.080	<0.004	<0.0003	0.007	0.022	622.32	8.01	0.008	56.98	0.00	0.16	0.556	<0.0009
4A	6.15	9.17	14-Feb-02	22.3	719	6.437	-46.5	-0.793	0.46	128.51	6.48	61.00	35.90	11.40	17.60	4.60	6.50	24.19	0.236	0.175	<0.0005	0.130	<0.004	<0.0003	0.006	0.031	395.36	4.53	0.000	45.99	13.20	1.12	0.561	<0.0009
4A	7.15	8.17	14-Feb-02	22.0	582	6.261	-53.5	-0.914	0.41	130.27	4.88	37.00	42.30	8.43	16.30	1.89	7.63	10.00	0.152	0.126	<0.0005	0.157	<0.004	<0.0003	0.006	0.023	297.74	3.01	0.010	43.99	7.92	0.47	0.591	<0.0009
4A	8.15	7.17	14-Feb-02	20.8	301	5.877	-118.5	-2.031	0.34	107.38	1.72	8.22	36.60	5.53	11.10	0.22	0.00	10.97	0.029	<0.0001	<0.0005	0.108	<0.004	<0.0003	0.007	0.057	104.94	14.39	1.065	47.99	14.96	0.22	0.348	<0.0009
4A	9.15	6.17	14-Feb-02	20.4	197	5.561	-114.4	-1.963	0.57	56.33	0.48	8.97	28.00	4.56	7.61	0.41	0.00	1.69	0.033	<0.0001	<0.0005	0.256	0.004	<0.0003	0.006	0.053	29.29	32.47	0.255	37.99	10.12	4.35	0.092	<0.0009
4A	10.15	5.17	14-Feb-02	24.9	165	5.434	-49.7	-0.840	1.81	51.05	0.24	5.51	23.30	2.87	6.29	0.29	0.00	1.91	0.016	<0.0001	<0.0005	0.124	<0.004	<0.0003	0.005	0.041	14.64	22.40	0.029	33.99	10.12	1.85	0.085	<0.0009
4A	11.15	4.17	14-Feb-02	23.4	165	5.210	-50.0	-0.849	0.65	47.53	0.16	5.10	22.80	2.54	6.28	0.22	0.00	1.57	0.059	<0.0001	<0.0005	0.185	0.004	<0.0003	0.005	0.055	9.76	26.75	0.044	31.99	11.00	1.22	0.074	<0.0009
4A	12.15	3.17	14-Feb-02	27.3	183	5.612	-74.8	-1.254	1.57	38.73	0.24	11.20	20.40	3.61	5.98	0.08	0.00	0.01	0.039	0.002	<0.0005	0.125	<0.004	<0.0003	0.005	0.040	14.64	30.45	0.017	32.99	15.40	0.79	0.049	<0.0009
4A	13.15	2.17	14-Feb-02	25.2	178	5.617	-65.7	-1.109	1.45	36.97	0.10	10.20	20.20	3.22	5.29	0.07	0.00	0.00	0.036	<0.0001	<0.0005	0.183	0.008	<0.0003	0.005	0.057	6.10	31.72	0.006	33.99	14.52	0.13	0.050	<0.0009
4A	14.15	1.17	14-Feb-02	24.6	181	5.621	-48.3	-0.818	2.52	44.89	0.12	10.00	18.40	3.34	5.13	0.06	0.00	0.01	0.033	<0.0001	<0.0005	0.088	0.004	<0.0003	0.006	0.026	7.32	20.46	0.016	33.99	22.88	0.25	0.050	<0.0009
30	1.55	13.95	15-Feb-02	21.5	1429	6.432	-48.3	-0.825	0.26	242.93	12.72	97.50	77.20	29.80	31.20	5.15	64.95	19.35	0.534	1.710	<0.0005	0.269	0.005	<0.0003	0.006	0.094	776.07	9.90	0.002	101.97	0.00	0.15	0.419	0.001
30	2.05	13.45	15-Feb-02	20.9	1297	6.415	-67.9	-1.164	0.02	211.25	13.88	129.00	66.30	32.10	29.60	5.45	56.05	28.38	0.652	1.520	<0.0005	0.458	0.008	<0.0003	0.006	0.045	846.85	10.12	0.000	65.98	0.00	1.42	0.401	0.001
30	3.05	12.45	15-Feb-02	20.7	1183	6.517	-83.8	-1.437	0.04	246.45	12.24	119.00	44.30	23.70	28.50	2.70	41.80	18.38	0.564	1.150	<0.0005	<0.01	0.005	<0.0003	0.004	0.045	746.79	10.16	0.000	51.98	0.00	1.66	0.395	<0.0009
30	3.55	11.95	15-Feb-02	21.3	1183	6.498	-84.1	-1.440	0.02	230.61	12.44	117.00	44.60	23.80	28.90	6.30	39.00	34.51	0.559	1.170	0.001	0.029	0.005	<0.0003	0.006	0.054	758.99	10.03	0.000	51.98	0.00	0.87	0.393	<0.0009
30	4.05	11.45	15-Feb-02	20.3	1192	6.481	-80.4	-1.381	0.01	211.25	12.44	116.00	44.50	23.60	29.10	5.15	39.65	27.41	0.546	1.160	<0.0005	0.024	0.005	<0.0003	0.005	0.051	758.99	9.90	0.007	51.98	0.00	2.41	0.388	0.001
30	4.55	10.95	15-Feb-02	19.8	1185	6.448	-73.5	-1.264	0.01	218.29	12.36	115.00	45.30	23.40	28.90	3.15	41.25	34.51	0.544	1.160	0.001	0.039	0.004	<0.0003	0.006	0.069	754.11	9.94	0.011	53.98	5.28	0.68	0.388	<0.0009
30	5.05	10.45	15-Feb-02	19.6	1125	6.415	-71.8	-1.235	0.04	253.50	11.28	110.00	45.30	22.10	27.40	6.65	34.35	30.64	0.531	1.090	<0.0005	0.059	0.011	<0.0003	0.007	0.067	688.22	9.28	0.009	49.98	0.00	0.45	0.379	<0.0009
30	5.55	9.95	15-Feb-02	20.1	1092	6.410	-58.4	-1.004	0.01	237.65	11.00	105.00	44.60	22.90	27.50	5.15	31.15	21.29	0.477	1.030	<0.0005	0.053	0.004	<0.0003	0.006	0.063	671.13	8.80	0.007	47.99	0.00	0.32	0.370	<0.0009
30	6.55	8.95	15-Feb-02	20.4	672	6.709	-77.6	-1.332	0.01	70.42	6.12	72.50	29.40	10.97	13.90	14.10	0.00	11.29	0.314	0.177	<0.0005	0.043	0.005	<0.0003	0.005	0.093	373.39	11.70	0.022	38.99	10.56	1.73	0.531	<0.0009
30	7.05	8.45	15-Feb-02	20.1	455	6.556	-65.9	-1.132	0.15	58.09	3.60	49.40	27.10	6.15	9.24	4.30	0.82	3.55	0.143	0.117	<0.0005	0.049	0.006	<0.0003	0.006	0.090	219.64	18.44	0.013	35.99	5.72	0.19	0.475	<0.0009
30	8.05	7.45	15-Feb-02	20.7	240	5.047	51.4	0.882	1.63	116.19	0.24	4.76	28.90	4.75	7.47	1.44	0.00	4.39	0.066	0.007	<0.0005	0.127	0.005	<0.0003	0.007	0.071	14.64	20.64	0.163	57.98	8.36	0.09	0.098	<0.0009
30	9.05	6.45	15-Feb-02	20.0	210	5.002	-21.5	-0.369	0.11	123.23	0.28	2.62	2																					

Table B6 Groundwater chemistry results for bundled piezometers 4A and 30: February 2002 sampling event.

Bore ID	Depth (bgs ¹)	Elev. (AHD ²)	Sample Date	Ag	Al	As	B	Ba	Be	Bi	Cd	Ce	Co	Cr	Cs	Cu	Dy	Er	Eu	Fe	Ga	Gd	Gd	Ge	Ho	La	Li	Lu	Mn	Mo	Nb	Nd	Ni
4A	2.15	13.17	14-Feb-02	0.073	50.376	1.397	123.006	97.509	0.052	<0.02	0.104	0.193	0.241	0.885	0.057	1.025	0.013	0.006	0.019	180.003	2.726	0.018	0.021	0.032	0.003	0.095	0.074	<0.003	5.939	1.343	<0.009	0.672	1.797
4A	2.65	12.67	14-Feb-02	0.016	53.441	1.356	182.751	107.899	0.029	<0.02	0.06	0.248	0.395	1.899	0.175	0.288	0.014	0.006	0.024	994.491	3.108	0.026	0.02	<0.02	<0.003	0.145	0.062	<0.003	147.657	0.959	<0.009	0.703	2.632
4A	3.15	12.17	14-Feb-02	0.126	78.528	0.533	242.362	228.004	0.057	<0.02	0.064	0.382	0.616	1.885	0.355	0.978	0.023	0.011	0.044	4449.024	6.53	0.04	0.038	<0.02	0.005	0.228	0.08	<0.003	232.483	0.343	<0.009	0.77	3.47
4A	3.65	11.67	14-Feb-02	0.029	47.625	0.439	242.844	122.758	0.027	<0.02	0.137	0.391	0.554	2.857	0.526	0.147	0.024	0.015	0.029	3469.766	3.539	0.043	0.038	<0.02	0.005	0.232	<0.03	<0.003	192.833	0.303	<0.009	0.791	1.95
4A	4.15	11.17	14-Feb-02	<0.02	51.714	0.386	267.875	36.366	0.017	<0.02	0.035	0.36	0.533	1.011	0.585	0.327	0.029	0.022	0.014	979.672	1.079	0.043	0.04	<0.02	0.006	0.202	0.031	0.005	183.294	0.435	<0.009	0.781	2.258
4A	4.65	10.67	14-Feb-02	0.015	52.464	0.503	432.901	44.887	0.047	<0.02	0.05	0.361	0.895	2.596	1.347	0.654	0.028	0.02	0.018	2079.541	1.373	0.043	0.042	0.041	0.005	0.207	0.038	0.004	174.329	0.199	<0.009	0.791	1.801
4A	5.15	10.17	14-Feb-02	0.094	37.31	1.568	496.187	125.121	0.032	<0.02	0.165	0.232	1.174	3.196	1.447	0.451	0.018	0.01	0.027	7355.404	3.704	0.028	0.029	0.115	0.003	0.131	<0.03	<0.003	275.904	0.627	<0.009	0.712	2.53
4A	5.65	9.67	14-Feb-02	0.299	26.65	1.911	420.983	130.705	0.03	<0.02	0.07	0.162	1.047	2.948	1.52	0.463	0.013	0.005	0.026	7084.813	3.89	0.019	0.02	0.09	<0.003	0.095	<0.03	<0.003	212.255	0.594	<0.009	0.677	2.268
4A	6.15	9.17	14-Feb-02	0.037	22.28	2.39	507.868	92.613	0.022	<0.02	0.052	0.145	0.605	1.89	1.468	0.389	0.017	0.006	0.021	3786.299	2.746	0.026	0.021	0.079	0.003	0.087	<0.03	<0.003	85.61	0.393	<0.009	0.705	1.564
4A	7.15	8.17	14-Feb-02	0.079	21.164	1.965	561.359	82.058	0.012	<0.02	0.66	0.209	0.416	1.942	1.281	0.142	0.016	0.007	0.02	3173.555	2.419	0.027	0.024	0.055	0.003	0.115	0.043	<0.003	132.893	0.651	<0.009	0.712	1.037
4A	8.15	7.17	14-Feb-02	0.076	23.755	1.288	376.389	41.183	0.057	<0.02	0.267	0.504	0.104	1.371	0.991	0.86	0.045	0.037	0.022	117.669	1.194	0.073	0.06	0.023	0.01	0.224	0.095	0.005	10.433	0.057	<0.009	0.903	0.726
4A	9.15	6.17	14-Feb-02	0.148	46.828	1.252	99.652	39.26	0.043	<0.02	0.956	0.186	0.098	1.164	0.275	0.555	0.015	0.014	0.009	290.443	1.153	0.021	0.026	<0.02	0.004	0.079	0.138	<0.003	9.03	0.087	0.013	0.697	0.776
4A	10.15	5.17	14-Feb-02	0.082	17.599	0.804	54.892	20.028	0.024	<0.02	0.113	0.146	0.111	0.411	0.283	<0.07	0.009	0.006	0.005	122.057	0.587	0.015	0.013	<0.02	<0.003	0.064	0.213	<0.003	3.627	0.071	0.013	0.657	<0.08
4A	11.15	4.17	14-Feb-02	0.114	25.182	0.795	37.183	28.754	0.068	<0.02	0.165	0.164	0.073	0.36	0.193	<0.07	0.008	0.003	0.006	64.644	0.859	0.014	0.014	<0.02	<0.003	0.083	0.132	<0.003	3.216	0.039	<0.009	0.656	<0.08
4A	12.15	3.17	14-Feb-02	0.214	18.783	0.252	53.993	34.798	0.032	<0.02	0.191	0.077	0.238	0.432	0.093	0.653	0.006	0.005	0.006	86.701	1.024	0.006	0.008	<0.02	<0.003	0.045	0.231	<0.003	15.543	0.061	<0.009	0.624	0.373
4A	13.15	2.17	14-Feb-02	0.309	50.619	0.138	54.138	40.102	0.052	<0.02	0.388	0.258	0.16	0.988	0.155	0.102	0.015	0.007	0.01	47.084	1.187	0.021	0.022	<0.02	0.003	0.166	0.326	<0.003	13.663	0.047	<0.009	0.685	0.248
4A	14.15	1.17	14-Feb-02	0.116	16.385	0.174	53.935	32.392	0.034	<0.02	0.203	0.156	0.104	0.932	0.132	0.333	0.011	0.008	0.008	44.378	0.942	0.016	0.014	<0.02	<0.003	0.085	0.239	<0.003	12.577	0.043	<0.009	0.661	0.168
30	1.55	13.95	15-Feb-02	0.166	77.678	15.316	205.174	658.493	0.045	<0.02	0.211	0.589	3.573	2.593	1.019	5.306	0.034	0.017	0.13	26477.341	18.697	0.058	0.054	0.361	0.006	0.409	0.224	0.003	825.338	2.93	0.023	0.844	4.129
30	2.05	13.45	15-Feb-02	0.202	59.976	8.47	210.653	630.363	0.069	<0.02	0.029	0.384	3.056	4.654	0.465	<0.07	0.025	0.013	0.12	21870.117	17.745	0.038	0.037	0.252	0.004	0.28	0.344	<0.003	691.422	1.17	0.018	0.759	3.294
30	3.05	12.45	15-Feb-02	0.048	44.381	0.812	245.352	532.469	0.053	<0.02	0.015	0.258	1.013	3.669	0.317	<0.07	0.023	0.014	0.099	15027.476	14.946	0.029	0.027	0.097	0.004	0.162	0.681	<0.003	515.932	0.068	<0.009	0.705	2.413
30	3.55	11.95	15-Feb-02	0.211	60.999	0.948	249.696	523.402	0.083	<0.02	0.027	0.289	1.076	3.595	0.341	0.864	0.025	0.012	0.099	15661.935	14.559	0.034	0.031	0.103	0.005	0.177	0.628	<0.003	534.565	0.175	<0.009	0.709	2.53
30	4.05	11.45	15-Feb-02	0.121	66.314	1.128	251.111	510.687	0.059	<0.02	0.022	0.301	1.067	4.632	0.412	<0.07	0.022	0.015	0.098	15661.957	14.675	0.033	0.031	0.091	0.005	0.189	0.41	<0.003	539.968	0.109	<0.009	0.727	2.362
30	4.55	10.95	15-Feb-02	0.357	46.402	1.118	242.15	486.861	0.059	<0.02	0.033	0.339	1.079	4.197	0.572	0.554	0.028	0.014	0.095	15217.512	14.015	0.037	0.036	0.097	0.005	0.211	0.359	<0.003	529.466	0.106	<0.009	0.736	2.435
30	5.05	10.45	15-Feb-02	0.054	57.725	1.11	245.835	450.682	0.024	<0.02	0.126	0.333	0.962	3.897	0.607	0.682	0.025	0.017	0.087	14341.55	12.856	0.039	0.04	0.1	0.006	0.204	0.331	<0.003	507.092	0.165	<0.009	0.736	2.492
30	5.55	9.95	15-Feb-02	0.133	52.545	1.425	249.439	392.513	0.04	<0.02	0.041	0.338	1.133	1.937	1.155	0.77	0.023	0.012	0.079	12561.05	11.333	0.033	0.031	0.12	0.004	0.206	0.177	<0.003	475.144	0.199	<0.009	0.735	2.529
30	6.55	8.95	15-Feb-02	0.126	81.229	1.063	500.671	94.339	0.037	<0.02	0.149	0.203	1.198	1.774	1.143	0.154	0.012	0.007	0.02	2227.191	2.72	0.02	0.023	0.208	0.003	0.106	0.049	<0.003	83.659	0.411	<0.009	0.684	2.428
30	7.05	8.45	15-Feb-02	0.417	61.927	1.647	482.386	51.675	0.041	<0.02	0.067	0.176	0.378	1.149	0.712	0.266	0.014	0.007	0.012	1888.164	1.433	0.02	0.021	0.117	<0.003	0.096	0.048	<0.003	129.27	0.223	<0.009	0.675	0.838
30	8.05	7.45	15-Feb-02	0.173	47.744	0.227	109.34	78.236	0.055	<0.02	0.093	0.087	0.084	0.945	0.383	0.34	0.009	0.005	0.013	1178.562	2.224	0.012	0.013	<0.02	<0.003	0.038	0.392	<0.003	20.063	0.032	<0.009	0.635	0.185
30	9.05	6.45	15-Feb-02	0.066	58.414	0.29	156.685	63.305	0.052	<0.02	0.16	0.146	0.09	1.096	0.374	<0.07	0.01	0.005	0.011	795.38	1.854	0.015	0.017	<0.02	<0.003	0.07	0.384	<0.003	11.663	<0.02	<0.009	0.654	<0.08
30	10.05	5.45	15-Feb-02	0.1	109.541	0.284	233.938	52.612	0.058	<0.02	0.594	0.633	0.054	1.291	0.358	<0.07	0.037	0.02	0.021	1159.132	1.55	0.06	0.052	<0.02	0.007	0.301	0.348	0.003	18.024	0.068	<0.009	0.874	0.216
30	11.05	4.45	15-Feb-02	0.05	87.791	0.205	152.384	47.765	0.037	<0.02	0.106	0.296	0.031	1.086	0.326	<0.07	0.02	0.012	0.013	2796.416	1.353	0.034	0.025	0.024	0.004	0.14	0.294	<0.003	95.656	0.024	<0.009	0.727	<0.08
30	12.05	3.45	15-Feb-02	0.134	69.135	3.301	197.904	41.877	0.024	<0.02	0.183	0.778	0.374	0.917	0.43	0.826	0.043	0.029	0.023	1406.578	1.241	0.074	0.065	<0.02	0.009	0.345	0.255	0.005	15.037	0.064	<0.009	0.953	0.575
30	13.05	2.45	15-Feb-02	0.873	32.759	4.981	136.192	41.215	0.032	<0.02	0.046	0.6	0.631	0.679	0.465	0.142	0.031	0.021	0.019	1297.95	1.227	0.057	0.048	<0.02	0.007	0.28	0.234	0.003	8.39	0.028	<0.009	0.862	<0.08
30	14.05	1.45	15-Feb-02	0.209	47.854	5.22	130.963	40.784	0.045	<0.02	0.112	0.872	0.667	0.776	0.474	<0.07	0.057	0.036	0.03	1398.505	1.214	0.098	0.095	<0.02	0.011	0.359	0.229	0.005	8.118	0.047	<0.009	1.086	<0.08
30	15.05	0.45	15-Feb-0																														

Table B6 Groundwater chemistry results for bundled piezometers 4A and 30: February 2002 sampling event.

Bore ID	Depth (bgs ¹)	Elev. (AHD ²)	Sample Date	P	Pb	Pr	Rb	Sb	Sc	Sm	Sn	Sr	Ta	Tb	Th	U	V	W	Y	Yb	Zn	Zr
4A	2.15	13.17	14-Feb-02	<5.0	3.875	0.025	7.182	2.018	1.409	0.017	0.862	136.288	<0.006	0.003	0.01	0.053	1.159	<0.01	0.078	0.005	7.224	0.134
4A	2.65	12.67	14-Feb-02	<5.0	4.742	0.031	13.091	0.25	1.211	0.02	0.31	259.434	<0.006	<0.003	0.008	0.009	1.218	<0.01	0.081	0.004	5.555	0.071
4A	3.15	12.17	14-Feb-02	<5.0	3.45	0.053	19.945	0.19	2.297	0.038	0.425	278.962	<0.006	0.004	0.025	0.008	1.445	<0.01	0.137	0.007	9.02	0.152
4A	3.65	11.67	14-Feb-02	<5.0	1.866	0.057	18.52	0.182	1.69	0.038	0.074	233.022	<0.006	0.005	0.008	0.004	1.451	<0.01	0.152	0.012	2.414	<0.05
4A	4.15	11.17	14-Feb-02	<5.0	2.424	0.054	18.551	0.155	0.993	0.037	0.356	182.499	<0.006	0.005	0.005	<0.003	2.002	<0.01	0.15	0.023	1.82	0.072
4A	4.65	10.67	14-Feb-02	<5.0	2.722	0.054	27.805	<0.02	0.181	0.041	0.236	103.582	<0.006	0.005	0.005	0.003	1.87	<0.01	0.137	0.025	8.744	<0.05
4A	5.15	10.17	14-Feb-02	<5.0	3.053	0.034	27.088	0.09	0.547	0.028	0.514	253.293	<0.006	0.003	0.005	0.009	1.238	<0.01	0.099	0.005	6.225	0.072
4A	5.65	9.67	14-Feb-02	<5.0	3.355	0.025	29.035	0.079	0.575	0.02	0.319	243.633	<0.006	<0.003	0.003	0.008	1.08	<0.01	0.071	0.005	3.801	0.117
4A	6.15	9.17	14-Feb-02	<5.0	2.303	0.031	27.371	<0.02	0.372	0.028	0.6	124.786	<0.006	0.003	<0.003	<0.003	1.254	<0.01	0.059	0.004	7.563	<0.05
4A	7.15	8.17	14-Feb-02	<5.0	4.208	0.034	28.148	<0.02	0.401	0.029	0.863	79.568	<0.006	<0.003	<0.003	<0.003	1.771	<0.01	0.074	0.006	10.069	0.06
4A	8.15	7.17	14-Feb-02	<5.0	3.895	0.084	38.385	<0.02	0.347	0.064	0.583	33.217	<0.006	0.007	<0.003	<0.003	2.975	<0.01	0.229	0.035	5.631	<0.05
4A	9.15	6.17	14-Feb-02	<5.0	7.987	0.029	20.419	<0.02	0.531	0.022	1.358	38.87	<0.006	<0.003	<0.003	<0.003	0.922	<0.01	0.09	0.015	15.283	<0.05
4A	10.15	5.17	14-Feb-02	<5.0	2.328	0.023	14.496	<0.02	0.293	0.019	0.432	12.031	<0.006	<0.003	<0.003	<0.003	0.424	0.1	0.052	0.003	3.878	<0.05
4A	11.15	4.17	14-Feb-02	<5.0	3.344	0.02	10.692	<0.02	0.213	0.017	0.265	46.55	<0.006	<0.003	<0.003	<0.003	0.362	0.044	0.046	<0.003	6.054	<0.05
4A	12.15	3.17	14-Feb-02	<5.0	2.422	0.009	11.171	<0.02	0.547	0.008	<0.03	42.407	<0.006	<0.003	<0.003	<0.003	1.44	<0.01	0.038	0.003	5.8	<0.05
4A	13.15	2.17	14-Feb-02	<5.0	8.667	0.029	11.28	<0.02	0.55	0.021	<0.03	41.587	<0.006	<0.003	<0.003	<0.003	3.611	0.025	0.102	0.006	10.106	<0.05
4A	14.15	1.17	14-Feb-02	<5.0	7.301	0.019	11.355	<0.02	0.486	0.012	0.616	37.868	<0.006	<0.003	<0.003	<0.003	1.878	<0.01	0.067	0.009	4.358	<0.05
30	1.55	13.95	15-Feb-02	25.8	5.238	0.07	41.808	0.894	3.48	0.049	0.152	311.467	<0.006	0.006	0.02	0.017	1.795	0.081	0.239	0.015	21.46	0.553
30	2.05	13.45	15-Feb-02	70.546	8.482	0.048	31.976	0.552	3.901	0.03	0.065	349.416	<0.006	0.004	0.006	0.036	1.292	0.107	0.173	0.011	26.608	0.46
30	3.05	12.45	15-Feb-02	472.812	3.764	0.033	30.624	0.389	4.831	0.031	0.06	299.616	<0.006	0.004	0.008	0.01	1.228	0.062	0.152	0.011	0.813	0.515
30	3.55	11.95	15-Feb-02	374.25	5.533	0.037	31.206	0.505	4.926	0.032	0.092	301.666	<0.006	0.003	0.011	0.012	1.325	0.071	0.154	0.012	3.122	0.54
30	4.05	11.45	15-Feb-02	167.669	5.977	0.037	31.356	0.4	5.045	0.027	0.07	300.624	<0.006	0.004	0.012	0.012	1.373	0.059	0.161	0.012	1.325	0.531
30	4.55	10.95	15-Feb-02	51.219	7.701	0.041	31.606	0.392	4.824	0.031	0.124	297.901	<0.006	0.004	0.013	0.013	1.418	0.053	0.173	0.012	4.118	0.589
30	5.05	10.45	15-Feb-02	39.601	7.811	0.043	29.561	0.405	4.583	0.037	0.075	279.674	<0.006	0.005	0.007	0.017	1.54	0.043	0.165	0.014	3.806	0.434
30	5.55	9.95	15-Feb-02	20.561	4.85	0.046	31.012	0.364	3.723	0.033	<0.03	256.597	<0.006	0.003	<0.003	0.044	1.24	<0.01	0.14	0.01	3.25	0.252
30	6.55	8.95	15-Feb-02	<5.0	9.139	0.027	18.759	<0.02	0.602	0.019	0.128	218.643	<0.006	<0.003	0.008	0.08	0.841	<0.01	0.07	0.004	6.145	<0.05
30	7.05	8.45	15-Feb-02	<5.0	6.641	0.027	14.326	<0.02	0.702	0.021	0.11	77.05	<0.006	<0.003	0.008	0.011	1.125	<0.01	0.061	0.004	1.982	0.082
30	8.05	7.45	15-Feb-02	7.991	5.098	0.012	35.541	<0.02	0.749	0.014	<0.03	68.494	<0.006	<0.003	<0.003	<0.003	2.445	<0.01	0.045	0.005	5.848	<0.05
30	9.05	6.45	15-Feb-02	<5.0	5.524	0.02	30.545	<0.02	0.724	0.013	<0.03	19.389	<0.006	<0.003	<0.003	<0.003	2.26	<0.01	0.061	0.005	2.102	<0.05
30	10.05	5.45	15-Feb-02	<5.0	4.405	0.079	28.89	<0.02	0.692	0.055	<0.03	18.446	<0.006	0.006	<0.003	<0.003	2.877	<0.01	0.222	0.015	5.237	<0.05
30	11.05	4.45	15-Feb-02	<5.0	3.522	0.036	31.108	<0.02	0.86	0.033	<0.03	23.365	<0.006	0.004	<0.003	<0.003	2.767	<0.01	0.13	0.013	2.764	<0.05
30	12.05	3.45	15-Feb-02	<5.0	4.583	0.097	23.831	<0.02	0.575	0.066	<0.03	34.671	<0.006	0.008	<0.003	<0.003	2.032	<0.01	0.26	0.029	10.124	<0.05
30	13.05	2.45	15-Feb-02	<5.0	2.624	0.076	22.402	<0.02	0.621	0.05	<0.03	25.557	<0.006	0.005	<0.003	<0.003	1.588	<0.01	0.203	0.024	2.645	<0.05
30	14.05	1.45	15-Feb-02	<5.0	6.141	0.122	22.682	<0.02	0.609	0.104	<0.03	24.571	<0.006	0.01	<0.003	<0.003	1.862	<0.01	0.327	0.034	4.608	<0.05
30	15.05	0.45	15-Feb-02	<5.0	4.732	0.04	15.544	<0.02	0.624	0.03	<0.03	41.681	<0.006	0.003	<0.003	<0.003	1.014	0.043	0.116	0.007	8.585	<0.05

1. bgs = below ground surface (metres)

2. AHD = Australian Height Datum (metres)

3. All concentrations in µg/L unless otherwise noted (except pH and pe)

Table B7 Groundwater parameters measured during interim ‘limited’ sample events.

Bore ID	Depth (bgs ¹)	Elev. (AHD ²)	13-14 September 2000			22-23 January 2001			12-13 April 2001			18 June 2001			29 July 2002 ⁴	
			pH	EC ³	T(°C)	pH	EC	T(°C)	pH	EC	T(°C)	pH	EC	T(°C)	EC	T(°C)
3	1.40	14.41	6.141	447	17.9	5.939	361.0	23.9	6.357	284	21.4	5.882	502	18.1	433.0	17.4
3	2.40	13.41	6.159	588	18.9	6.023	492.0	25.6	6.359	570	21.0	6.056	551	18.5	359.0	17.9
3	2.90	12.91	6.297	958	18.3	6.249	1031.0	23.1	6.555	890	21.1	6.292	529	18.7	477.5	17.7
3	3.40	12.41	6.429	1232	18.6	6.394	1144.0	22.9	6.645	1191	21.4	6.310	602	18.9	753.0	18.0
3	3.90	11.91	6.458	1316	19.4	6.411	1195.0	23.5	6.686	1316	20.8	6.382	922	19.0	993.0	18.2
3	4.40	11.41	6.400	1262	18.7	6.339	1232.0	21.6	6.576	1328	21.2	6.419	1201	19.2	1037.5	18.4
3	4.90	10.91	6.180	680	23.3	6.313	1306.0	25.5	6.553	1314	21.3	6.524	1274	18.9	1115.5	18.5
3	5.40	10.41	6.286	1136	20.0	6.217	757.0	24.3	6.563	1033	20.9	6.414	1181	19.1	1325.0	18.7
3	5.90	9.91	6.178	900	19.2	6.163	765.0	22.4	6.444	1003	20.8	6.335	1120	19.4	1228.0	18.6
3	6.40	9.41	5.901	578	19.8	6.010	581.0	24.9	6.170	651	20.8	6.021	590	19.1	766.0	18.6
3	6.90	8.91	DRY	-- ⁵	--	--	--	--	--	--	--	--	--	--	--	--
3	7.40	8.41	DRY	--	--	--	--	--	--	--	--	--	--	--	--	--
3	7.90	7.91	5.187	196	20.5	5.027	169.7	24.8	5.401	226	19.2	5.082	189	18.5	229.5	18.3
3	8.40	7.41	5.121	190	19.9	5.024	169.9	24.4	5.482	224	19.9	5.071	187	18.7	209.0	18.5
3	8.90	6.91	5.072	190	20.4	4.951	168.7	25.4	5.202	211	20.8	4.965	184	18.5	207.0	18.5
3	9.40	6.41	5.004	188	20.1	4.988	165.5	25.8	5.247	208	20.8	4.988	185	18.4	197.8	18.4
3	10.40	5.41	4.955	186	19.4	4.927	174.8	23.5	5.211	207	21	5.013	183	19	198.0	18.4
3	11.40	4.41	4.955	181	19.5	4.896	167.4	23.8	5.253	203	20.9	5.129	174	18.3	198.9	18.5
3	12.40	3.41	5.036	182	19.4	4.923	161.9	22.4	5.225	193	20.9	5.115	175	19.0	176.0	18.4
3	13.40	2.41	4.865	171	19.4	4.987	156.6	24.1	5.243	196	21.2	5.190	174	18.3	177.4	18.3
3	14.40	1.41	5.157	194	19.4	5.080	174.5	23.4	3.623	214	20.9	5.269	193	18.4	174.8	18.4
3	15.40	0.41	5.197	202	19.5	5.055	176.5	23.7	3.571	219	20.5	5.196	199	18.5	170.8	18.4
3	16.40	-0.59	5.388	193	19.4	5.260	189.0	22.1	5.569	251	20.3	5.308	228	18.7	248.0	16.8
3	17.40	-1.59	4.985	207	19.9	4.985	176.4	25.8	5.455	246	19.9	5.199	227	18.3	244.0	18.9
3	18.40	-2.59	5.103	197	19.3	5.145	191.3	23.9	5.426	258	20.3	5.049	246	18.4	274.5	17.7
3	19.40	-3.59	4.360	217	19.2	4.566	203.0	22.9	4.935	272	20.4	5.107	234	18.8	263.5	18.5
4	1.84	15.48	DRY	--	--	--	--	--	--	--	--	--	--	--	NA ⁶	NA
4	2.34	14.98	DRY	--	--	--	--	--	--	--	--	--	--	--	NA	NA
4	2.84	14.48	6.620	590	18.5	6.388	440.0	23.9	DRY	--	--	5.261	276	18.3	NA	NA
4	3.34	13.98	6.736	820	18.0	6.661	560.0	21.6	6.694	343	* ⁷	6.529	355	18.7	NA	NA
4	3.84	13.48	6.778	776	18.3	6.645	591.0	21.3	6.623	403	*	6.659	357	18.8	NA	NA
4	4.34	12.98	6.737	942	18.4	6.587	656.0	21.0	6.607	380	*	6.718	329	19.0	NA	NA
4	4.84	12.48	6.783	529	18.6	6.501	752.0	21.4	6.720	429	*	6.623	435	18.9	NA	NA
4	5.34	11.98	6.500	363	18.6	6.153	442.0	22.0	6.504	493	*	6.574	389	18.9	NA	NA
4	5.84	11.48	6.309	285	18.7	5.888	250.0	21.6	5.662	255	*	5.864	301	18.7	NA	NA
4	6.34	10.98	6.366	320	19.0	6.006	242.0	24.4	5.870	249	*	5.715	243	18.2	NA	NA
4	6.84	10.48	6.286	416	19.8	5.953	239.0	28.8	6.125	240	*				NA	NA
4	7.34	9.98	6.526	570	18.8	6.355	333.0	24.7	6.240	257	*	5.778	248	18.1	NA	NA
4	7.84	9.48	5.939	308	19.0	5.865	254.0	23.0	5.956	254	*	5.782	261	18.4	NA	NA
4	8.34	8.98	5.307	230	18.9	5.093	213.0	22.1	5.515	218	*	5.559	231	18.7	NA	NA
4	8.84	8.48	5.588	254	18.9	5.480	237.0	20.8	5.373	218	*	5.545	242	18.9	NA	NA
4	9.34	7.98	5.633	251	19.0	5.494	232.0	21.9	5.528	225	*	5.652	238	18.6	NA	NA
4	9.84	7.48	5.523	243	18.9	5.519	226.0	24.5	5.570	227	*	5.476	234	18.4	NA	NA
4	10.34	6.98	4.775	224	19.1	4.666	208.0	22.7	4.728	201	*	4.632	211	18.6	NA	NA
4	10.84	6.48	4.584	226	18.9	4.454	213.0	21.1	4.601	201	*	4.615	214	18.7	NA	NA
4	11.34	5.98	4.565	226	18.9	4.422	213.0	21.1	4.367	207	*	4.626	215	18.6	NA	NA

Table B7 (cont.) Groundwater parameters measured during interim 'limited' sample events.

Bore ID	Depth (bgs ¹)	Elev. (AHD ²)	13-14 September 2000			22-23 January 2001			12-13 April 2001			18 June 2001			29 July 2002 ⁴	
			pH	EC ³	T(°C)	pH	EC	T(°C)	pH	EC	T(°C)	pH	EC	T(°C)	EC	T(°C)
4	12.34	4.98	5.112	188	19.5	4.851	205.0	24.4	5.194	197	*	5.113	203	18.2	NA	NA
4	13.34	3.98	4.587	223	18.8	5.061	196.3	22.4	4.750	192	*	4.680	207	18.5	NA	NA
4	14.34	2.98	5.150	202	18.9	5.209	184.1	23.4	4.570	185	*	4.997	203	18.6	NA	NA
4	15.34	1.98	4.935	203	18.9	4.794	190.7	21.8	5.161	191	*	4.856	208	18.8	NA	NA
4	16.34	0.98	5.058	204	18.7	5.199	189.1	26.2	5.475	188	*	5.328	200	18.5	NA	NA
4	18.34	-1.02	4.940	184	18.9	4.921	193.6	25.2	5.298	190	*	4.891	203	18.7	NA	NA
4A	2.15	13.17	5.830	420	17.7	5.920	556.0	25.3	6.357	395	*	6.634	293	18.2	NA	NA
4A	2.65	12.67	5.900	610	16.6	5.735	744.0	22.5	6.224	555	*	6.515	971	18.3	NA	NA
4A	3.15	12.17	5.878	616	17.3	5.685	610.0	22.5	6.216	648	*	6.263	1002	18.7	NA	NA
4A	3.65	11.67	5.930	634	18.0	5.779	626.0	22.3	6.144	581	*	6.215	972	19.0	NA	NA
4A	4.15	11.17	6.130	742	18.2	5.941	695.0	22.1	6.113	614	*	6.243	915	19.3	NA	NA
4A	4.65	10.67	6.303	770	18.4	6.189	722.0	21.6	6.316	651	*	6.209	803	19.4	NA	NA
4A	5.15	10.17	6.383	703	18.4	6.388	802.0	20.9	6.478	690	*	6.338	863	19.4	NA	NA
4A	5.65	9.67	6.381	780	18.8	6.313	739.0	21.9	6.532	675	*	6.406	811	19.4	NA	NA
4A	6.15	9.17	6.249	666	19.3	6.025	605.0	24.8	4.000	593	17.7	6.236	680	19.4	NA	NA
4A	6.65	8.67	6.094	617	18.8	6.051	598.0	20.3	6.180	526	17.8	6.062	619	19.3	NA	NA
4A	7.15	8.17	6.077	607	19.1	5.942	562.0	24.2	6.152	351	19.3	6.011	591	19.2	NA	NA
4A	7.65	7.67	5.439	263	19.3	5.947	552.0	23.3	3.884	273	19.5	5.385	227	19.2	NA	NA
4A	8.15	7.17	5.250	202	19.1	5.899	540.0	22.7	3.727	243	19.2	5.205	207	19.1	NA	NA
4A	8.65	6.67	5.187	194	19.3	5.915	529.0	22.6	3.687	226	19.1	5.117	196	19.1	NA	NA
4A	9.15	6.17	5.182	179	19.5	5.865	536.0	22.5	3.799	243	18.9	5.076	193	19.2	NA	NA
4A	9.65	5.67	DRY	--	--	--	--	--	--	--	--	--	--	--	NA	NA
4A	10.15	5.17	4.706	184	20.0	4.820	156.4	26.6	5.369	191	19.1	5.128	160	19.7	NA	NA
4A	11.15	4.17	4.734	182	19.2	4.758	159.5	23.3	5.302	191	19.5	4.960	167	19.2	NA	NA
4A	12.15	3.17	4.717	188	20.7	4.919	156.7	27.7	5.463	194	19.3	5.286	168	19.3	NA	NA
4A	13.15	2.17	4.605	183	19.6	4.610	166.0	24.9	5.074	206	18.7	5.077	175	18.9	NA	NA
4A	14.15	1.17	4.659	184	19.9	4.652	165.3	25.2	5.110	205	19.1	5.111	182	18.7	NA	NA
4A	15.15	0.17	4.993	234	22.7	5.137	197.1	30.1	5.639	249	15.5	5.516	194	18.9	NA	NA
4A	16.15	-0.83	5.572	246	19.2	5.520	226.0	22.9	5.707	276	19.6	5.648	241	18.8	NA	NA
4A	17.15	-1.83	5.413	265	19.5	5.407	235.0	24.0	5.728	257	19.5	5.519	220	18.7	NA	NA
4A	18.15	-2.83	5.255	276	19.6	5.363	295.0	23.8	5.576	263	19.8	5.306	204	18.8	NA	NA
4A	19.15	-3.83	4.633	215	19.4	4.449	191.6	22.4	4.831	243	19.8	5.024	204	18.5	NA	NA
6	1.68	13.98	DRY	--	--	--	--	--	--	--	--	--	--	--	NA	NA
6	2.68	12.98	5.343	98	17.9	4.993	138.2	25.0	5.240	128	22.8	4.892	160	19.3	NA	NA
6	3.68	11.98	5.554	219	18.9	5.361	175.5	25.0	5.732	175	22.4	5.630	129	20.0	NA	NA
6	4.68	10.98	5.722	185	18.4	5.757	176.8	22.9	5.977	180	22.1	5.716	316	20.1	NA	NA
6	5.18	10.48	5.758	197	18.8	5.751	184.9	23.3	5.945	201	22.0	5.720	342	20.2	NA	NA
6	5.68	9.98	5.681	206	20.4	5.731	179.5	26.0	5.915	206	22.3	5.853	277	20.4	NA	NA
6	6.18	9.48	5.631	188	18.8	5.775	212.0	23.2	6.173	270	22.1	5.937	275	20.2	NA	NA
6	6.68	8.98	5.967	228	18.6	5.815	220.0	22.0	6.064	265	21.8	5.887	250	20.2	NA	NA
6	7.18	8.48	6.055	251	18.6	5.888	221.0	21.2	6.179	270	21.4	5.963	354	20.2	NA	NA
6	7.68	7.98	6.023	276	19.8	5.959	232.0	25.1	6.175	264	21.8	5.887	372	20.2	NA	NA
6	8.18	7.48	5.900	251	19.4	5.915	255.0	24.2	6.226	278	21.7	5.826	306	20.0	NA	NA
6	8.68	6.98	5.725	227	19.2	5.890	232.0	23.8	6.175	268	21.6	5.809	287	20.0	NA	NA
6	9.18	6.48	5.614	204	19.6	5.787	210.0	24.1	6.156	267	21.8	5.703	348	20.0	NA	NA

Table B7 (cont.) Groundwater parameters measured during interim 'limited' sample events.

Bore ID	Depth (bgs ¹)	Elev. (AHD ²)	13-14 September 2000			22-23 January 2001			12-13 April 2001			18 June 2001			29 July 2002 ⁴	
			pH	EC ³	T(°C)	pH	EC	T(°C)	pH	EC	T(°C)	pH	EC	T(°C)	EC	T(°C)
6	9.68	5.98	5.798	279	18.9	5.778	217.0	22.7	6.083	275	21.2	5.938	746	19.9	NA	NA
6	10.18	5.48	5.875	415	18.9	5.801	244.0	21.9	6.097	288	20.7	5.958	760	19.7	NA	NA
6	10.68	4.98	5.882	488	18.9	5.850	311.0	22.0	6.106	463	21.8	5.937	747	19.5	NA	NA
6	11.18	4.48	5.947	463	19.0	5.886	293.0	24.4	6.135	359	21.9	5.944	710	19.6	NA	NA
6	11.68	3.98	5.844	372	20.1	5.820	271.0	27.8	5.943	282	22.1	5.592	340	19.8	NA	NA
6	12.18	3.48	5.883	534	19.4	5.787	270.0	23.2	6.076	318	20.9	5.962	771	19.5	NA	NA
6	12.68	2.98	6.096	714	19.5	5.764	358.0	24.3	6.040	372	21.2	5.818	621	19.6	NA	NA
6	13.18	2.48	6.173	767	19.2	5.992	701.0	23.8	6.299	772	21.1	5.892	667	19.8	NA	NA
6	13.68	1.98	5.951	699	18.8	5.811	651.0	21.2	6.111	709	20.3	5.638	569	19.2	NA	NA
6	14.18	1.48	5.958	680	19.2	5.857	649.0	24.5	6.305	700	21.4	5.778	600	19.7	NA	NA
6	15.18	0.48	6.204	692	19.1	5.874	644.0	23.6	6.175	735	20.9	5.676	590	19.5	NA	NA
6	16.18	-0.52	5.966	731	18.6	5.944	620.0	25.9	6.261	719	21.2	5.966	643	19.6	NA	NA
6	18.18	-2.52	5.720	370	18.0	5.023	265.0	23.2	3.865	336	19.8	5.145	355	19.2	NA	NA
10	3.00	13.63	6.321	1075	21.2	6.037	657.0	22.6	5.920	505	*	5.761	646	20.2	604.0	20.6
10	4.00	12.63	6.558	1643	21.2	6.452	1608.0	22.7	6.409	1350	*	6.171	759	20.2	1340.5	21.1
10	5.00	11.63	6.102	551	21.1	6.436	1713.0	21.9	6.228	1249	*	6.093	703	20.45	1430.5	21.1
10	6.00	10.63	5.934	514	21.0	5.828	819.0	21.9	6.031	1057	*	6.353	1169	20.45	1377.0	21.1
10	7.00	9.63	5.462	354	21.0	5.296	306.0	22.1	5.292	310	*	5.744	325	20.25	300.5	20.8
10	8.00	8.63	5.442	349	20.7	5.263	293.0	21.5	5.101	293	*	5.551	304	20.15	295.5	20.6
10	9.00	7.63	5.557	362	21.0	5.360	310.0	22.0	5.370	327	*	5.549	311	20.1	305.5	20.7
10	10.00	6.63	5.621	397	20.8	5.614	274.0	21.7	5.493	294	*	5.632	308	20.05	312.5	20.7
10	11.00	5.63	5.574	269	20.8	5.669	250.0	21.3	5.648	247	*	5.718	275	19.95	258.0	20.7
10	12.00	4.63	DRY	--	--	--	--	--	--	--	--	--	--	--	--	--
10	13.00	3.63	5.700	303	20.8	5.741	288.0	22.0	5.547	287	*	5.686	304	19.85	292.0	20.8
10	14.00	2.63	5.485	264	20.8	5.550	235.0	21.9	5.589	258	*	5.584	253	19.85	293.0	20.3
10	15.00	1.63	5.888	314	20.8	6.028	303.0	21.5	5.569	289	*	5.869	310	19.75	282.5	20.9
10	16.00	0.63	5.870	311	20.7	6.110	253.0	21.8	5.594	266	*	5.831	300	19.85	260.5	20.6
10	18.00	-1.37	5.460	225	20.8	5.801	246.0	21.9	5.482	255	*	5.739	255	19.85	285.0	20.8
10	20.00	-3.37	5.124	196	20.6	4.877	163.6	21.5	4.969	179	*	4.873	189	19.75	181.4	20.7
10	22.00	-5.37	5.121	185	20.9	4.886	177.8	21.8	5.165	173	*	5.159	182	19.75	194.3	20.6
10	24.00	-7.37	5.222	175	20.5	5.018	177.7	22.6	5.148	175	*	4.942	186	19.65	195.8	20.6
10	26.00	-9.37	4.665	213	21.2	4.734	185.8	24.0	4.613	194	*	4.673	187	19.75	180.5	21.7
10	28.00	-11.37	5.159	199	20.5	5.901	311.0	21.2	4.678	175	*	5.251	174	19.7	194.5	20.4
10	30.00	-13.37	5.491	208	20.6	5.429	177.6	23.2	5.428	194	*	5.413	193	19.75	188.0	20.5

1. bgs = below ground surface (metres)

2. AHD = Australian Height Datum (metres)

3. EC expressed in µS/cm.

4. Data collected in conjunction with manometer board experiment (Acworth et. al., submitted; Jorstad & Acworth, submitted).

5. -- = Sample point dry during sample event.

6. NA = No sample collected from this sample point during this sample event.

7. * = Temperature probe malfunction, no temperature recorded.

Appendix C

PHREEQC Saturation Indices Summary Tables

Table C1. TIC, Log P_{CO2}, and saturation indices calculated using PHREEQC: July 2000 sampling event.

Well ID	Depth (m, bgs)	Elevation (m, AHD)	Sample Date	Saturation indices												
				TIC	log PCO2	Fe(OH) ₃	Goethite	Hematite	FeS(ppt)	Mackinawite	Pyrite	Calcite	Dolomite	Siderite	Rhodochrosite	
1	6-9	12.16 - 9.16	26-Jul-00	7.07	-3.53	-2.12	3.52	9.02	0.94	1.68	7.76	1.10	1.27	1.44	-0.98	
2	6-9	12.46 - 9.46	25-Jul-00	29.62	-0.45	-6.99	-1.29	-0.60	-0.66	0.07	4.57	-0.28	-0.92	1.24	-0.51	
3	1.40	14.41	27-Jul-00	6.09	-1.00	-9.43	-3.89	-5.81	NA	NA	NA	-1.85	-4.42	-2.80	NA	
3	2.40	13.41	27-Jul-00	4.86	-1.15	-9.13	-3.54	-5.11	NA	NA	NA	-1.43	-3.68	-2.80	-2.71	
3	2.90	12.91	27-Jul-00	16.71	-0.64	-9.27	-3.67	-5.37	-2.66	-1.93	4.59	-0.73	-2.11	-1.76	-0.94	
3	3.40	12.41	27-Jul-00	25.82	-0.49	-7.63	-2.02	-2.07	-0.30	0.44	7.18	-0.30	-1.28	0.19	0.14	
3	3.90	11.91	27-Jul-00	26.35	-0.55	-4.30	1.31	4.59	-0.32	0.42	9.98	-0.18	-0.92	1.39	-0.02	
3	4.40	11.41	27-Jul-00	27.73	-0.48	-5.88	-0.23	1.51	-1.11	-0.38	6.30	-0.26	-0.98	1.09	-0.08	
3	4.90	10.91	27-Jul-00	DRY	--	--	--	--	--	--	--	--	--	--	--	
3	5.40	10.41	27-Jul-00	18.45	-0.72	-4.59	0.96	3.89	-1.30	-0.57	8.83	-0.49	-1.43	0.79	-0.41	
3	5.90	9.91	27-Jul-00	17.45	-0.67	-6.01	-0.39	1.20	-1.40	-0.67	6.08	-0.68	-1.79	0.71	-0.47	
3	6.40	9.41	27-Jul-00	13.37	-0.65	-7.99	-2.35	-2.72	-1.11	-0.38	5.59	-1.77	-3.81	-0.24	-1.57	
3	6.90	8.91	27-Jul-00	DRY	--	--	--	--	--	--	--	--	--	--	--	
3	7.40	8.41	27-Jul-00	DRY	--	--	--	--	--	--	--	--	--	--	--	
3	7.90	7.91	27-Jul-00	8.42	-0.79	-9.67	-4.18	-6.41	-1.26	-0.53	6.58	-4.09	-9.01	-1.86	-3.64	
3	8.40	7.41	27-Jul-00	9.50	-0.71	-10.20	-4.67	-7.38	-1.61	-0.88	5.73	-4.48	-9.28	-2.11	-3.89	
3	8.90	6.91	26-Jul-00	8.94	-0.73	-10.08	-4.53	-7.10	-1.91	-1.18	5.80	-4.75	-9.59	-2.28	NA	
3	9.40	6.41	26-Jul-00	8.60	-0.72	-10.62	-5.01	-8.05	-2.64	-1.91	4.43	-4.92	-9.81	-2.68	NA	
3	10.40	5.41	26-Jul-00	10.78	-0.60	-11.25	-5.60	-9.23	-4.15	-3.42	1.56	-5.10	-10.09	-3.05	NA	
3	11.40	4.41	26-Jul-00	10.19	-0.64	-11.03	-5.44	-8.90	-4.10	-3.37	2.31	-5.30	-10.46	-3.13	NA	
3	12.40	3.41	26-Jul-00	8.41	-0.70	-11.67	-6.00	-10.03	-3.35	-2.61	2.51	-4.57	-9.13	-3.22	-4.73	
3	13.40	2.41	26-Jul-00	7.10	-0.77	-11.51	-5.84	-9.71	-3.66	-2.93	2.49	-4.39	-8.84	-3.39	-4.39	
3	14.40	1.41	26-Jul-00	6.86	-0.80	-10.93	-5.27	-8.56	-3.39	-2.66	2.67	-3.80	-7.79	-2.91	-3.75	
3	15.40	0.41	26-Jul-00	5.96	-0.89	-10.83	-5.22	-8.48	-4.14	-3.41	1.89	-3.61	-7.46	-3.15	-3.51	
3	16.40	-0.59	25-Jul-00	4.31	-1.03	-10.83	-5.16	-8.34	-4.90	-4.17	-0.79	-3.53	-7.26	-2.92	-3.93	
3	17.40	-1.59	25-Jul-00	3.47	-1.12	-10.78	-5.12	-8.26	-4.04	-3.31	1.69	-3.73	-7.63	-3.23	-4.17	

Table C1 (cont.) TIC, Log P_{CO2}, and saturation indices calculated using PHREEQC: July 2000 sampling event.

Well ID	Depth (m, bgs)	Elevation (m, AHD)	Sample Date			Saturation indices									
				TIC	log PCO2	Fe(OH) ₃	Goethite	Hematite	FeS(ppt)	Mackinawite	Pyrite	Calcite	Dolomite	Siderite	Rhodochrosite
3	18.40	-2.59	25-Jul-00	3.93	-1.05	-10.96	-5.30	-8.63	-5.47	-4.73	-1.38	-4.16	-8.41	-3.21	-4.57
3	19.40	-3.59	25-Jul-00	14.20	-0.46	-13.08	-7.42	-12.86	-7.68	-6.95	-3.37	-5.07	-10.23	-4.85	-5.24
3	20.40	-4.59	27-Jul-00	17.32	-0.37	-12.52	-6.84	-11.70	-6.43	-5.70	-2.10	-5.15	-10.33	-3.98	-5.41
4	1.84	15.48	6-Jul-00	DRY	--	--	--	--	--	--	--	--	--	--	--
4	2.34	14.98	6-Jul-00	DRY	--	--	--	--	--	--	--	--	--	--	--
4	2.84	14.48	6-Jul-00	DRY	--	--	--	--	--	--	--	--	--	--	--
4	3.34	13.98	6-Jul-00	11.05	-1.08	-5.75	-0.10	1.77	-1.41	-0.68	5.93	-0.32	-0.98	0.51	-0.71
4	3.84	13.48	6-Jul-00	12.36	-1.06	-5.59	0.06	2.10	-1.82	-1.09	5.15	-0.21	-0.74	0.61	-0.63
4	4.34	12.98	6-Jul-00	14.98	-0.99	-5.76	-0.10	1.77	-1.27	-0.54	6.05	-0.08	-0.46	0.63	-0.57
4	4.84	12.48	6-Jul-00	16.27	-0.89	-6.18	-0.52	0.94	-1.04	-0.30	5.59	-0.15	-0.62	0.77	-0.45
4	5.34	11.98	6-Jul-00	27.30	-0.29	-8.76	-3.09	-4.21	-1.44	-0.70	3.86	-1.40	-3.04	-0.04	-1.56
4	5.84	11.48	6-Jul-00	3.15	-1.41	-7.53	-1.86	-1.74	-0.59	0.14	5.83	-1.80	-3.90	-0.44	-1.96
4	6.34	10.98	6-Jul-00	3.96	-1.38	-6.21	-0.55	0.88	-0.47	0.26	6.84	-1.45	-3.13	0.22	-1.44
4	6.84	10.48	6-Jul-00	3.64	-1.45	-5.90	-0.27	1.44	-1.71	-0.98	5.56	-1.42	-3.24	-0.03	-1.63
4	7.34	9.98	6-Jul-00	10.52	-1.00	-5.98	-0.33	1.32	NA	NA	NA	-0.82	-2.05	0.56	-1.01
4	7.84	9.48	6-Jul-00	5.48	-1.05	-7.60	-1.95	-1.93	-3.77	-3.04	3.56	-2.05	-4.43	-1.47	-2.18
4	8.34	8.98	6-Jul-00	4.58	-1.00	-5.88	-0.24	1.50	NA	NA	NA	-3.48	-7.20	-2.87	-3.82
4	8.84	8.48	6-Jul-00	4.32	-1.05	-10.56	-4.91	-7.84	NA	NA	NA	-3.11	-6.41	-2.77	-3.16
4	9.34	7.98	5-Jul-00	3.59	-1.16	-9.94	-4.32	-6.68	-4.65	-3.91	0.74	-3.09	-6.43	-2.78	-3.18
4	9.84	7.48	5-Jul-00	5.28	-0.93	-10.27	-4.63	-7.28	-5.41	-4.68	-0.02	-3.69	-7.70	-3.04	-4.22
4	10.34	6.98	5-Jul-00	7.37	-0.76	-10.50	-4.87	-7.76	-6.96	-6.23	1.10	-4.90	-10.16	-4.42	NA
4	10.84	6.48	5-Jul-00	9.60	-0.64	-11.55	-5.89	-9.81	NA	NA	NA	-4.93	-10.22	-4.55	NA
4	11.34	5.98	5-Jul-00	8.35	-0.70	-11.61	-5.96	-9.94	NA	NA	NA	-5.02	-10.41	-4.70	NA
4	12.34	4.98	5-Jul-00	DRY	--	--	--	--	--	--	--	--	--	--	--
4	13.34	3.98	5-Jul-00	9.10	-0.66	-11.63	-5.98	-9.98	-6.82	-6.08	0.09	-5.08	-10.49	-4.65	NA
4	14.34	2.98	5-Jul-00	3.81	-1.07	-9.27	-3.65	-5.34	NA	NA	NA	-4.35	-8.90	-3.80	NA
4	15.34	1.98	5-Jul-00	7.08	-0.77	-10.87	-5.21	-8.43	-5.71	-4.97	1.59	-4.60	-9.30	-4.02	NA

Table C1 (cont.) TIC, Log P_{CO2}, and saturation indices calculated using PHREEQC: July 2000 sampling event.

Well ID	Depth (m, bgs)	Elevation (m, AHD)	Sample Date			Saturation indices									
				TIC	log PCO2	Fe(OH) ₃	Goethite	Hematite	FeS(ppt)	Mackinawite	Pyrite	Calcite	Dolomite	Siderite	Rhodochrosite
4	16.34	0.98	5-Jul-00	5.51	-0.89	-9.66	-4.01	-6.04	-5.88	-5.15	2.56	-4.55	-9.25	-3.77	NA
4	18.34	-1.02	5-Jul-00	8.16	-0.71	-10.64	-4.98	-7.98	-6.56	-5.83	1.28	-4.78	-9.80	-4.27	NA
4	20.34	-3.02	27-Jul-00	15.58	-0.42	-13.39	-7.72	-13.45	NA	NA	NA	-5.36	-10.88	-4.46	-5.63
4A	2.15	13.17	28-Jul-00	22.19	-0.41	-8.25	-2.74	-3.51	-2.14	-1.41	4.90	-2.26	-4.40	-0.57	-2.31
4A	2.65	12.67	28-Jul-00	22.58	-0.38	-8.82	-3.27	-4.57	-2.47	-1.74	4.05	-2.23	-4.31	-0.89	-2.30
4A	3.15	12.17	28-Jul-00	25.18	-0.33	-9.08	-3.51	-5.05	-2.80	-2.06	3.78	-2.25	-4.32	-1.14	-2.39
4A	3.65	11.67	28-Jul-00	31.94	-0.20	-9.72	-4.13	-6.30	-3.05	-2.32	2.97	-2.35	-4.57	-1.34	-2.50
4A	4.15	11.17	28-Jul-00	21.20	-0.45	-8.30	-2.69	-3.41	-2.81	-2.08	3.34	-1.40	-2.92	-0.61	-1.40
4A	4.65	10.67	28-Jul-00	19.27	-0.54	-7.33	-1.68	-1.39	-2.38	-1.65	4.48	-0.94	-2.14	-0.17	-0.93
4A	5.15	10.17	28-Jul-00	15.75	-0.68	-6.37	-0.72	0.54	NA	NA	NA	-0.83	-2.01	0.57	-0.69
4A	5.65	9.67	28-Jul-00	27.39	-0.31	-7.30	-1.65	-1.32	-2.05	-1.31	5.16	-1.18	-2.75	0.07	-1.03
4A	6.15	9.17	28-Jul-00	14.80	-0.66	-6.42	-0.74	0.50	-2.00	-1.27	6.14	-1.14	-2.78	0.00	-1.18
4A	6.65	8.67	28-Jul-00	14.39	-0.62	-8.24	-2.57	-3.17	-2.16	-1.43	4.18	-1.57	-3.39	-0.61	-1.65
4A	7.15	8.17	28-Jul-00	12.52	-0.73	-7.54	-1.87	-1.76	-1.79	-1.06	4.87	-1.28	-2.85	-0.24	-1.31
4A	7.65	7.67	28-Jul-00	5.37	-0.98	-8.47	-2.81	-3.65	-0.09	0.65	8.21	-2.77	-6.00	-1.08	-2.47
4A	8.15	7.17	28-Jul-00	5.88	-0.89	-9.54	-3.89	-5.80	-1.09	-0.35	6.50	-3.55	-7.31	-1.81	-3.33
4A	8.65	6.67	28-Jul-00	5.75	-0.90	-9.83	-4.20	-6.42	-1.54	-0.81	5.82	-3.76	-7.69	-2.07	-3.63
4A	9.15	6.17	31-Jul-00	8.51	-0.72	-11.51	-5.91	-9.86	-2.56	-1.83	2.73	-4.14	-8.43	-2.66	-4.43
4A	9.65	5.67	31-Jul-00	DRY	--	--	--	--	--	--	--	--	--	--	--
4A	10.15	5.17	31-Jul-00	13.03	-0.55	-12.69	-7.18	-12.39	-4.87	-4.14	0.52	-5.38	-10.82	-4.05	NA
4A	11.15	4.17	31-Jul-00	9.44	-0.66	-12.62	-7.02	-12.06	-4.52	-3.79	0.66	-5.19	-10.38	-3.96	NA
4A	12.15	3.17	31-Jul-00	6.34	-0.87	-11.58	-6.03	-10.09	-3.52	-2.79	3.08	-4.18	-8.56	-3.58	-4.61
4A	13.15	2.17	31-Jul-00	13.11	-0.52	-13.70	-8.09	-14.21	-7.16	-6.42	-2.76	-4.50	-9.16	-5.15	-4.88
4A	14.15	1.17	31-Jul-00	8.40	-0.70	-13.95	-8.32	-14.67	-7.20	-6.47	-3.48	-4.94	-10.05	-5.30	-5.33
4A	15.15	0.17	31-Jul-00	5.05	-0.96	-10.74	-5.14	-8.32	-3.76	-3.03	3.04	-3.67	-7.65	-3.28	-4.32
4A	16.15	-0.83	31-Jul-00	2.34	-1.36	-10.76	-5.09	-8.21	-4.94	-4.21	0.96	-2.76	-5.88	-3.82	NA
4A	17.15	-1.83	31-Jul-00	1.81	-1.44	-11.11	-5.45	-8.91	-5.27	-4.54	0.62	-3.23	-6.72	-4.23	NA

Table C1 (cont.) TIC, Log P_{CO2}, and saturation indices calculated using PHREEQC: July 2000 sampling event.

Well ID	Depth (m, bgs)	Elevation (m, AHD)	Sample Date			Saturation indices									
				TIC	log PCO2	Fe(OH) ₃	Goethite	Hematite	FeS(ppt)	Mackinawite	Pyrite	Calcite	Dolomite	Siderite	Rhodochrosite
4A	18.15	-2.83	31-Jul-00	2.75	-1.21	-10.90	-5.23	-8.49	-4.93	-4.20	0.54	-3.71	-7.67	-3.62	NA
4A	19.15	-3.83	31-Jul-00	10.68	-0.58	-13.48	-7.80	-13.62	-7.84	-7.10	-3.14	-4.94	-10.05	-5.42	-5.46
4A	20.15	-4.83	27-Jul-00	8.37	-0.69	-11.54	-5.86	-9.73	-6.42	-5.69	-0.68	-5.00	-10.03	-4.11	-5.43
6	1.68	13.98	14-Jul-00	DRY	--	--	--	--	--	--	--	--	--	--	--
6	2.68	12.98	14-Jul-00	1.60	-1.56	-8.16	-2.54	-3.12	-1.65	-0.92	5.43	-3.52	-7.23	-1.56	-3.27
6	3.68	11.98	14-Jul-00	3.18	-1.29	-7.56	-1.91	-1.85	-0.57	0.17	7.82	-2.97	-6.06	-0.97	-2.81
6	4.68	10.98	14-Jul-00	3.15	-1.31	-8.42	-2.76	-3.54	0.00	0.73	7.34	-2.93	-6.07	-1.03	-2.99
6	5.18	10.48	14-Jul-00	3.32	-1.28	-8.01	-2.36	-2.74	-0.30	0.43	7.50	-2.82	-5.91	-0.98	-2.71
6	5.68	9.98	12-Jul-00	4.33	-1.10	-7.88	-2.22	-2.47	-0.84	-0.11	7.18	-2.82	-6.01	-0.97	-2.69
6	6.18	9.48	12-Jul-00	4.54	-1.02	-8.53	-2.84	-3.69	-1.60	-0.86	5.49	-3.24	-6.80	-1.29	-2.97
6	6.68	8.98	12-Jul-00	2.92	-1.37	-7.69	-2.00	-2.02	-1.40	-0.66	5.48	-2.28	-4.92	-0.93	-2.19
6	7.18	8.48	12-Jul-00	5.19	-1.04	-8.73	-3.05	-4.12	-2.10	-1.36	3.93	-2.26	-5.01	-1.24	-2.25
6	7.68	7.98	12-Jul-00	4.09	-1.17	-7.34	-1.61	-1.22	-2.26	-1.53	4.54	-2.15	-4.79	-0.78	-2.11
6	8.18	7.48	12-Jul-00	4.86	-1.01	-7.90	-2.19	-2.40	-1.10	-0.37	6.75	-2.82	-5.96	-0.95	-2.66
6	8.68	6.98	12-Jul-00	5.71	-0.94	-8.41	-2.73	-3.48	-1.21	-0.48	6.36	-2.92	-6.15	-1.15	-2.82
6	9.18	6.48	12-Jul-00	5.66	-0.97	-7.91	-2.23	-2.49	-0.96	-0.22	6.98	-2.73	-5.70	-0.87	-2.65
6	9.68	5.98	12-Jul-00	12.92	-0.61	-8.12	-2.46	-2.95	-0.55	0.18	7.38	-2.35	-4.89	-0.51	-2.32
6	10.18	5.48	12-Jul-00	16.59	-0.48	-8.56	-2.90	-3.82	-0.83	-0.10	6.34	-2.49	-5.17	-0.52	-2.40
6	10.68	4.98	12-Jul-00	15.88	-0.50	-8.59	-2.95	-3.92	-0.89	-0.15	6.49	-2.71	-5.51	-0.64	-2.54
6	11.18	4.48	7-Jul-00	15.55	-0.52	-8.35	-2.72	-3.47	-0.75	-0.02	7.09	-2.67	-5.42	-0.61	-2.56
6	11.68	3.98	7-Jul-00	DRY	--	--	--	--	--	--	--	--	--	--	--
6	12.18	3.48	7-Jul-00	21.00	-0.43	-7.78	-2.12	-2.27	-2.65	-1.92	4.18	-2.50	-4.86	-0.44	-2.34
6	12.68	2.98	7-Jul-00	15.85	-0.61	-6.88	-1.16	-0.33	-2.07	-1.34	5.49	-2.03	-3.58	-0.10	-2.09
6	13.18	2.48	7-Jul-00	16.43	-0.54	-6.24	-0.45	1.10	-1.18	-0.44	7.40	-1.92	-3.39	0.32	-2.19
6	13.68	1.98	7-Jul-00	18.69	-0.47	-7.90	-2.23	-2.48	-2.21	-1.47	3.72	-2.40	-4.37	-0.12	-2.45
6	14.18	1.48	7-Jul-00	11.18	-0.77	-6.22	-0.45	1.09	-1.69	-0.95	6.12	-2.03	-3.57	0.21	-1.93
6	15.18	0.48	7-Jul-00	17.95	-0.48	-7.73	-2.03	-2.09	-2.26	-1.52	3.87	-2.43	-4.32	-0.10	-2.14

Table C1 (cont.) TIC, Log P_{CO2}, and saturation indices calculated using PHREEQC: July 2000 sampling event.

Well ID	Depth (m, bgs)	Elevation (m, AHD)	Sample Date	Saturation indices											
				TIC	log PCO2	Fe(OH) ₃	Goethite	Hematite	FeS(ppt)	Mackinawite	Pyrite	Calcite	Dolomite	Siderite	Rhodochrosite
6	16.18	-0.52	7-Jul-00	7.95	-1.12	-5.10	0.52	3.01	-0.83	-0.10	8.27	-1.71	-3.06	0.52	-1.45
6	18.18	-2.52	7-Jul-00	11.31	-0.63	-8.86	-3.21	-4.45	-0.75	-0.02	5.91	-3.45	-6.55	-0.67	-2.98
6	19.18	-3.52	27-Jul-00	4.08	-1.06	-9.46	-3.79	-5.61	-2.37	-1.64	1.94	-3.99	-7.62	-1.27	-3.28
10	3.0	13.63	29-Jul-00	17.77	-0.62	-5.25	0.48	2.96	-0.79	-0.06	7.59	-0.51	-1.75	1.11	-0.11
10	4.0	12.63	29-Jul-00	23.49	-0.68	-5.28	0.46	2.91	-0.97	-0.23	6.36	-0.02	-0.49	1.33	-0.31
10	5.0	11.63	29-Jul-00	10.73	-0.75	-7.01	-1.27	-0.55	-1.72	-0.99	4.88	-1.79	-3.75	0.10	-1.61
10	6.0	10.63	29-Jul-00	10.37	-0.71	-7.51	-1.75	-1.52	-1.85	-1.12	4.43	-2.12	-4.52	-0.13	-1.76
10	7.0	9.63	29-Jul-00	15.76	-0.44	-9.60	-3.85	-5.71	-1.87	-1.14	4.43	-3.64	-6.89	-1.26	-3.45
10	8.0	8.63	29-Jul-00	14.88	-0.46	-10.04	-4.31	-6.63	-1.96	-1.23	4.08	-3.76	-7.19	-1.52	-3.54
10	9.0	7.63	29-Jul-00	14.26	-0.49	-9.71	-3.98	-5.97	-1.90	-1.17	3.97	-3.56	-6.74	-1.26	-3.39
10	10.0	6.63	29-Jul-00	12.71	-0.56	-9.39	-3.66	-5.33	-2.07	-1.34	3.42	-3.12	-6.19	-1.03	-3.29
10	11.0	5.63	29-Jul-00	5.85	-0.93	-10.77	-5.05	-8.11	-2.17	-1.43	3.96	-2.56	-5.32	-2.57	-2.66
10	12.0	4.63	29-Jul-00	DRY	--	--	--	--	--	--	--	--	--	--	--
10	13.0	3.63	29-Jul-00	5.98	-0.95	-8.67	-2.95	-3.91	-2.17	-1.44	3.53	-2.34	-4.83	-1.04	-2.38
10	14.0	2.63	31-Jul-00	6.19	-0.89	-9.67	-3.95	-5.90	-2.76	-2.03	2.36	-2.63	-5.45	-1.66	-2.59
10	15.0	1.63	31-Jul-00	4.54	-1.15	-9.07	-3.35	-4.70	-1.41	-0.67	4.25	-1.96	-4.07	-1.24	-1.54
10	16.0	0.63	31-Jul-00	4.92	-1.11	-9.31	-3.59	-5.20	-1.49	-0.76	4.20	-1.92	-3.98	-1.40	-1.18
10	18.0	-1.37	31-Jul-00	2.03	-1.36	-11.82	-6.11	-10.24	-5.66	-4.93	-2.08	-3.36	-6.98	-4.01	-4.04
10	20.0	-3.37	1-Aug-00	6.28	-0.81	-12.74	-7.03	-12.08	-5.63	-4.90	-0.16	-4.80	-9.29	-4.69	-4.60
10	22.0	-5.37	1-Aug-00	4.36	-0.97	-11.92	-6.22	-10.45	-5.21	-4.47	-0.47	-4.53	-9.08	-3.92	-4.59
10	24.0	-7.37	1-Aug-00	7.39	-0.74	-14.04	-8.34	-14.70	-7.32	-6.59	-3.38	-4.96	-9.90	-5.54	-5.28
10	26.0	-9.37	1-Aug-00	11.37	-0.54	-13.37	-7.65	-13.32	NA	NA	NA	-5.46	-10.40	-4.67	-5.25
10	28.0	-11.37	1-Aug-00	4.76	-0.94	-11.55	-5.84	-9.70	-5.36	-4.63	-0.90	-4.62	-8.84	-3.60	-4.42
10	30.0	-13.37	1-Aug-00	3.66	-1.13	-8.74	-3.02	-4.05	-1.77	-1.04	3.45	-3.78	-6.73	-0.98	-2.61

Table C2. TIC, Log P_{CO2}, and saturation indices calculated using PHREEQC: December 2000 sampling event.

Well ID	Depth (m, bgs)	Elevation (m, AHD)	Sample Date			Saturation indices									
				TIC	log PCO2	Fe(OH) ₃	Goethite	Hematite	FeS(ppt)	Mackinawite	Pyrite	Calcite	Dolomite	Siderite	Rhodochrosite
1	6-9	12.16 - 9.16	26-Jul-00	9.01	-3.20	-0.95	4.71	11.41	0.41	1.14	10.50	1.15	1.56	1.27	-0.68
2	6-9	12.46 - 9.46	25-Jul-00	28.01	-0.45	-4.19	1.52	5.03	-1.09	-0.35	9.53	-0.34	-1.01	1.15	-0.52
3	1.40	14.414	11-Dec-00	6.29	-0.89	-6.02	-0.20	1.61	-4.80	-4.07	7.15	-1.96	-4.42	-2.22	NA
3	2.40	13.414	11-Dec-00	16.77	-0.45	-5.80	0.07	2.15	-3.42	-2.69	8.56	-1.23	-3.34	-1.21	-2.73
3	2.90	12.914	11-Dec-00	25.18	-0.49	-6.68	-0.93	0.13	-1.15	-0.42	9.38	-0.12	-0.93	-0.54	-0.58
3	3.40	12.414	11-Dec-00	26.32	-0.51	-4.35	1.40	4.79	-1.13	-0.40	10.27	0.00	-0.65	0.70	0.29
3	3.90	11.914	11-Dec-00	26.32	-0.52	-3.62	2.16	6.31	-0.89	-0.16	9.74	-0.07	-0.73	1.50	-0.09
3	4.40	11.414	11-Dec-00	28.64	-0.45	-3.92	1.79	5.57	-1.18	-0.44	9.59	-0.27	-0.99	1.26	-0.08
3	4.90	10.914	11-Dec-00	23.58	-0.52	-3.46	2.41	6.83	-1.58	-0.84	9.54	-0.19	-0.78	1.18	0.02
3	5.40	10.414	11-Dec-00	15.32	-0.71	-3.79	2.03	6.06	-1.69	-0.95	9.63	-0.53	-1.50	0.69	-0.48
3	5.90	9.914	11-Dec-00	14.57	-0.77	-3.88	1.86	5.70	-1.13	-0.40	9.85	-0.59	-1.64	0.91	-0.55
3	6.40	9.414	11-Dec-00	12.62	-0.73	-4.21	1.56	5.11	-1.86	-1.12	9.67	-1.09	-2.52	0.30	-0.81
3	6.90	8.914	11-Dec-00	DRY	--	--	--	--	--	--	--	--	--	--	--
3	7.40	8.414	11-Dec-00	DRY	--	--	--	--	--	--	--	--	--	--	--
3	7.90	7.914	11-Dec-00	5.56	-0.86	-7.14	-1.29	-0.58	-0.86	-0.13	10.33	-3.35	-7.65	-1.39	-3.56
3	8.40	7.414	11-Dec-00	7.48	-0.73	-8.32	-2.52	-3.05	-1.99	-1.25	7.15	-3.60	-8.16	-1.71	-4.61
4	1.84	15.481	7-Dec-00	DRY	--	--	--	--	--	--	--	--	--	--	--
4	2.34	14.981	7-Dec-00	DRY	--	--	--	--	--	--	--	--	--	--	--
4	2.84	14.481	7-Dec-00	4.48	-1.26	-7.30	-1.60	-1.22	-1.75	-1.02	5.75	-1.55	-3.40	-0.89	-1.79
4	3.34	13.981	7-Dec-00	8.78	-1.07	-5.96	-0.28	1.41	-1.53	-0.79	6.34	-0.74	-1.81	0.15	-0.94
4	3.84	13.481	7-Dec-00	10.04	-1.02	-5.71	-0.04	1.90	-1.36	-0.63	7.35	-0.63	-1.66	0.15	-0.67
4	4.34	12.981	7-Dec-00	11.52	-0.97	-5.71	-0.04	1.90	-1.46	-0.73	7.00	-0.49	-1.38	0.25	-0.78
4	4.84	12.481	7-Dec-00	10.62	-1.04	-5.59	0.07	2.13	-1.28	-0.55	6.78	-0.48	-1.34	0.44	-0.69
4	5.34	11.981	7-Dec-00	3.84	-1.29	-6.40	-0.72	0.54	-1.34	-0.61	6.51	-1.70	-3.71	-0.30	-1.86
4	5.84	11.481	7-Dec-00	3.58	-1.19	-6.62	-0.95	0.09	NA	NA	NA	-2.43	-5.28	-2.82	-2.88

Table C2 (cont.) TIC, Log P_{CO2}, and saturation indices calculated using PHREEQC: December 2000 sampling event.

Well ID	Depth (m, bgs)	Elevation (m, AHD)	Sample Date			Saturation indices									
				TIC	log PCO2	Fe(OH) ₃	Goethite	Hematite	FeS(ppt)	Mackinawite	Pyrite	Calcite	Dolomite	Siderite	Rhodochrosite
4	6.34	10.981	7-Dec-00	4.46	-1.08	-7.69	-2.00	-2.01	NA	NA	NA	-2.36	-5.19	-2.54	-2.81
4	6.84	10.481	7-Dec-00	2.95	-1.34	-6.50	-0.78	0.44	-4.04	-3.31	6.30	-2.03	-4.49	-2.16	-3.05
4	7.34	9.981	7-Dec-00	4.81	-1.29	-5.53	0.18	2.35	-0.77	-0.04	8.06	-1.20	-2.77	0.13	-1.45
4	7.84	9.481	7-Dec-00	3.87	-1.14	-8.66	-2.96	-3.93	-2.38	-1.65	7.24	-2.45	-5.27	-2.59	-2.59
4	8.34	8.981	7-Dec-00	10.34	-0.61	-10.48	-4.77	-7.56	-5.25	-4.52	2.92	-3.60	-7.49	-3.75	-4.04
4	8.84	8.481	7-Dec-00	5.65	-0.89	-10.06	-4.37	-6.76	-4.51	-3.78	3.42	-3.40	-6.98	-3.43	-3.62
4	9.34	7.981	7-Dec-00	4.82	-0.96	-9.88	-4.17	-6.36	-5.13	-4.39	1.79	-3.29	-6.78	-3.26	-3.42
4A	2.15	13.172	8-Dec-00	20.91	-0.34	-8.66	-2.89	-3.78	-0.40	0.33	8.18	-1.89	-3.71	-0.83	-2.09
4A	2.65	12.672	8-Dec-00	33.90	-0.13	-9.19	-3.48	-4.97	-2.61	-1.88	4.32	-2.06	-4.09	-1.12	-1.92
4A	3.15	12.172	8-Dec-00	30.64	-0.18	NA	NA	NA	NA	NA	NA	-2.05	-4.05	NA	-2.09
4A	3.65	11.672	8-Dec-00	31.58	-0.16	NA	NA	NA	NA	NA	NA	-1.97	-3.90	NA	-2.00
4A	4.15	11.172	8-Dec-00	22.58	-0.37	-8.18	-2.45	-2.91	-2.59	-1.86	5.12	-1.29	-2.74	-0.96	-1.21
4A	4.65	10.672	8-Dec-00	17.63	-0.56	-6.60	-0.90	0.18	-0.83	-0.09	8.27	-0.95	-2.26	0.05	-0.73
4A	5.15	10.172	8-Dec-00	16.70	-0.65	-5.45	0.24	2.47	-1.14	-0.41	7.83	-0.69	-1.79	0.66	-0.56
4A	5.65	9.672	8-Dec-00	15.87	-0.67	-5.10	0.60	3.19	NA	NA	NA	-0.72	-1.96	0.65	-0.59
4A	6.15	9.172	8-Dec-00	15.29	-0.58	-5.62	0.14	2.28	-1.96	-1.22	7.18	-1.24	-3.02	0.27	-1.13
4A	6.65	8.672	8-Dec-00	17.19	-0.51	-5.92	-0.24	1.51	-2.30	-1.57	7.37	-1.50	-3.44	-0.13	-1.47
4A	7.15	8.172	8-Dec-00	15.70	-0.56	-5.88	-0.13	1.73	NA	NA	NA	-1.41	-3.13	-0.15	-1.53
4A	7.65	7.672	8-Dec-00	18.36	-0.39	-8.33	-2.56	-3.12	-1.42	-0.69	6.81	-2.47	-5.01	-0.87	-2.27
4A	8.15	7.172	8-Dec-00	13.24	-0.49	NA	NA	NA	NA	NA	NA	-3.54	-7.18	NA	-3.61
4A	8.65	6.672	8-Dec-00	10.89	-0.58	NA	NA	NA	NA	NA	NA	-3.78	-7.62	NA	-3.97
4A	9.15	6.172	8-Dec-00	9.66	-0.63	NA	NA	NA	NA	NA	NA	-4.01	-8.08	NA	-4.52
6	1.68	13.981	12-Dec-00	DRY	--	--	--	--	--	--	--	--	--	--	--
6	2.68	12.981	12-Dec-00	6.32	-0.78	-8.67	-2.86	-3.72	-2.62	-1.88	7.34	-4.16	-8.63	-2.48	-3.91
6	3.68	11.981	12-Dec-00	3.58	-1.07	-7.52	-1.71	-1.41	-0.95	-0.22	9.17	-3.49	-7.09	-1.51	-3.38
6	4.68	10.981	12-Dec-00	4.76	-1.01	-7.41	-1.66	-1.32	-1.14	-0.41	8.72	-2.91	-6.10	-1.32	-2.95

Table C2 (cont.) TIC, Log P_{CO2}, and saturation indices calculated using PHREEQC: December 2000 sampling event.

Well ID	Depth (m, bgs)	Elevation (m, AHD)	Sample Date			Saturation indices									
				TIC	log PCO2	Fe(OH) ₃	Goethite	Hematite	FeS(ppt)	Mackinawite	Pyrite	Calcite	Dolomite	Siderite	Rhodochrosite
6	5.68	9.981	12-Dec-00	5.70	-0.88	-7.19	-1.29	-0.58	-1.02	-0.28	9.33	-2.77	-6.03	-1.22	-2.91
6	6.68	8.981	12-Dec-00	4.83	-1.03	-6.53	-0.80	0.39	-1.36	-0.63	8.34	-2.49	-5.45	-0.77	-2.48
6	7.68	7.981	12-Dec-00	4.05	-1.13	-6.22	-0.37	1.26	-1.44	-0.70	8.60	-2.11	-4.70	-0.85	-2.23
6	8.68	6.981	12-Dec-00	4.51	-1.09	-6.77	-0.98	0.04	-1.12	-0.39	8.77	-2.08	-4.72	-1.00	-2.12
6	9.68	5.981	12-Dec-00	5.67	-0.92	-6.88	-1.10	-0.21	-1.90	-1.17	7.68	-2.57	-5.54	-1.05	-2.76
6	10.18	5.481	12-Dec-00	7.67	-0.82	-6.78	-1.04	-0.10	-1.62	-0.89	7.24	-2.28	-4.89	-0.54	-2.26
6	10.68	4.981	12-Dec-00	10.21	-0.72	-6.65	-0.92	0.15	-1.66	-0.93	6.75	-1.87	-4.12	-0.21	-1.90
6	11.18	4.481	12-Dec-00	9.73	-0.71	-6.45	-0.63	0.74	-0.88	-0.15	8.79	-1.90	-4.15	-0.26	-1.92
6	11.68	3.981	12-Dec-00	7.49	-0.78	-6.46	-0.51	0.99	-0.91	-0.17	9.10	-2.10	-4.46	-0.52	-2.14
6	12.18	3.481	12-Dec-00	7.92	-0.79	-6.91	-1.13	-0.26	-0.75	-0.02	9.16	-2.23	-4.84	-0.68	-2.24
6	12.68	2.981	12-Dec-00	11.73	-0.60	-6.65	-0.79	0.42	-0.99	-0.26	9.03	-2.08	-4.37	-0.45	-2.18
6	13.18	2.481	12-Dec-00	15.31	-0.53	-5.28	0.63	3.27	-1.38	-0.65	8.36	-1.74	-3.19	0.46	-2.12
6	13.68	1.981	12-Dec-00	17.26	-0.48	-6.05	-0.31	1.36	-1.52	-0.78	7.80	-2.19	-4.00	0.17	-2.52
6	14.18	1.481	12-Dec-00	15.30	-0.52	-5.59	0.27	2.54	-1.31	-0.58	8.35	-2.03	-3.67	0.33	-2.40
6	15.18	0.481	12-Dec-00	16.19	-0.52	-5.80	-0.03	1.94	-0.94	-0.21	8.95	-2.07	-3.80	0.28	-2.42
6	16.18	-0.519	12-Dec-00	15.96	-0.50	-6.48	-0.64	0.72	-0.74	-0.01	7.97	-2.02	-3.70	0.28	-2.29
6	18.18	-2.519	12-Dec-00	11.28	-0.58	-6.92	-1.16	-0.32	-1.77	-1.04	8.68	-3.44	-6.54	-0.98	-3.42
10	3.00	13.628	9-Dec-00	16.62	-0.55	-4.68	1.07	4.13	-2.18	-1.45	7.38	-0.98	-2.77	0.71	-0.89
10	4.00	12.628	9-Dec-00	26.93	-0.52	-4.11	1.63	5.26	-0.94	-0.21	9.69	-0.11	-0.58	1.18	-0.40
10	5.00	11.628	9-Dec-00	27.73	-0.39	-4.51	1.23	4.45	-1.28	-0.55	9.44	-0.62	-1.61	0.91	-0.59
10	6.00	10.628	9-Dec-00	21.59	-0.36	-5.22	0.52	3.02	-2.87	-2.14	8.06	-2.01	-4.03	-0.14	-1.70
10	7.00	9.628	9-Dec-00	20.53	-0.31	-7.01	-1.29	-0.58	-2.11	-1.38	9.91	-3.76	-7.17	-1.40	-3.52
10	8.00	8.628	9-Dec-00	22.67	-0.26	-8.40	-2.66	-3.33	-2.26	-1.52	8.04	-4.02	-7.73	-1.74	-3.87
10	9.00	7.628	9-Dec-00	20.41	-0.31	-8.33	-2.60	-3.21	-2.12	-1.39	7.66	-3.84	-7.35	-1.53	-3.71
10	10.00	6.628	9-Dec-00	12.62	-0.54	-7.50	-1.77	-1.56	-2.41	-1.67	7.17	-3.35	-6.75	-1.23	-3.71
10	11.00	5.628	9-Dec-00	8.73	-0.70	-9.52	-3.80	-5.61	-1.93	-1.19	7.00	-2.94	-6.07	-2.37	-3.00

Table C3. TIC, Log P_{CO2}, and saturation indices calculated using PHREEQC: March 2001 sampling event.

Well ID	Depth (m, bgs)	Elevation (m, AHD)	Sample Date			Saturation indices									
				TIC	log PCO2	Fe(OH) ₃	Goethite	Hematite	FeS(ppt)	Mackinawite	Pyrite	Calcite	Dolomite	Siderite	Rhodochrosite
1	6-9	12.16 - 9.16	26-Jul-00	10.96	-2.43	-3.07	2.66	7.30	-0.41	0.32	8.04	0.83	1.89	0.87	-0.36
2	6-9	12.46 - 9.46	25-Jul-00	25.11	-0.56	-3.80	1.94	5.88	-1.24	-0.50	9.90	-0.43	-0.92	1.08	-0.57
3	1.40	14.41	15-Mar-01	2.11	-1.45	-3.63	2.19	6.38	-4.02	-3.29	12.22	-2.29	-4.90	-2.36	-4.06
3	2.40	13.41	15-Mar-01	9.72	-0.84	-5.54	0.27	2.54	-3.19	-2.46	9.32	-1.13	-2.74	-1.50	-2.51
3	2.90	12.91	15-Mar-01	16.54	-0.67	-6.30	-0.50	0.99	-1.64	-0.91	9.24	-0.64	-1.92	-0.77	-0.84
3	3.40	12.41	15-Mar-01	19.74	-0.67	-5.28	0.51	3.02	-0.83	-0.10	8.36	-0.40	-1.41	0.75	-0.04
3	3.90	11.91	15-Mar-01	21.52	-0.64	-3.80	2.00	6.00	-1.14	-0.41	9.78	-0.36	-1.31	1.08	-0.21
3	4.40	11.41	15-Mar-01	23.77	-0.56	-3.89	1.87	5.73	NA	NA	NA	-0.50	-1.50	0.94	-0.36
3	4.90	10.91	15-Mar-01	23.45	-0.54	-3.77	2.04	6.07	-1.94	-1.20	8.96	-0.45	-1.32	0.93	-0.24
3	5.40	10.41	15-Mar-01	15.57	-0.74	-3.94	1.85	5.70	-1.47	-0.74	10.37	-0.80	-2.04	0.48	-0.68
3	5.90	9.91	15-Mar-01	14.10	-0.75	-3.61	2.16	6.32	NA	NA	NA	-0.95	-2.38	0.73	-0.71
3	6.40	9.41	15-Mar-01	11.31	-0.76	-4.21	1.58	5.15	NA	NA	NA	-1.46	-3.26	0.09	-1.04
3	6.90	8.91	15-Mar-01	DRY	--	--	--	--	--	--	--	--	--	--	--
3	7.40	8.41	15-Mar-01	DRY	--	--	--	--	--	--	--	--	--	--	--
3	7.90	7.91	15-Mar-01	6.04	-0.83	-6.36	-0.51	0.97	-0.83	-0.09	12.83	-3.39	-6.84	-1.65	-4.04
3	8.40	7.41	15-Mar-01	5.74	-0.85	-7.57	-1.74	-1.48	-1.98	-1.25	9.06	-3.56	-7.17	-1.98	-4.50
4	1.84	15.48	19-Mar-01	DRY	--	--	--	--	--	--	--	--	--	--	--
4	2.34	14.98	19-Mar-01	DRY	--	--	--	--	--	--	--	--	--	--	--
4	2.84	14.48	19-Mar-01	2.60	-1.62	-5.24	0.53	3.06	0.07	0.81	11.74	-1.57	-3.51	-0.69	-1.56
4	3.34	13.98	19-Mar-01	3.49	-1.72	-4.24	1.51	5.00	0.17	0.90	11.46	-0.93	-2.31	0.04	-1.30
4	3.84	13.48	19-Mar-01	6.14	-1.28	-4.94	0.80	3.59	-0.23	0.50	10.91	-0.93	-2.20	-0.06	-0.97
4	4.34	12.98	19-Mar-01	7.76	-1.21	-4.73	0.99	3.97	-1.27	-0.54	8.93	-0.72	-1.73	0.12	-1.01
4	4.84	12.48	19-Mar-01	8.78	-1.14	-4.56	1.16	4.30	-1.62	-0.89	8.26	-0.67	-1.69	0.29	-0.79
4	5.34	11.98	19-Mar-01	11.81	-0.83	-5.05	0.67	3.33	-2.02	-1.28	8.49	-0.94	-2.29	-0.07	-1.69
4	5.84	11.48	19-Mar-01	3.84	-1.14	-6.93	-1.20	-0.42	NA	NA	NA	-2.46	-5.21	-2.90	-2.89

Table C3 (cont.) TIC, Log P_{CO2}, and saturation indices calculated using PHREEQC: May 2001 sampling event.

Well ID	Depth (m, bgs)	Elevation (m, AHD)	Sample Date			Saturation indices									
				TIC	log PCO2	Fe(OH) ₃	Goethite	Hematite	FeS(ppt)	Mackinawite	Pyrite	Calcite	Dolomite	Siderite	Rhodochrosite
4	6.34	10.98	19-Mar-01	2.38	-1.39	-5.80	-0.03	1.93	NA	NA	NA	-2.34	-5.10	-2.67	-2.58
4	6.84	10.48	19-Mar-01	2.08	-1.47	-4.87	0.93	3.86	-3.16	-2.43	10.29	-2.28	-4.91	-1.99	-2.81
4	7.34	9.98	19-Mar-01	2.46	-1.55	-4.27	1.51	5.01	-1.41	-0.68	11.74	-1.85	-4.00	-0.97	-2.39
4	7.84	9.48	19-Mar-01	3.13	-1.23	-6.72	-0.96	0.08	-5.15	-4.41	6.53	-2.57	-5.38	-3.02	-2.82
4	8.34	8.98	19-Mar-01	4.51	-0.96	-7.96	-2.21	-2.42	-5.39	-4.66	7.09	-3.67	-7.48	-3.93	-4.32
4	8.84	8.48	19-Mar-01	4.78	-0.98	-6.77	-1.06	-0.14	-4.59	-3.85	8.13	-3.21	-6.54	-2.95	-3.47
4A	2.15	13.17	16-Mar-01	10.85	-0.80	-4.33	1.52	5.05	-0.91	-0.18	13.70	-1.17	-2.56	-0.62	-1.69
4A	2.65	12.67	16-Mar-01	18.19	-0.45	-5.68	0.14	2.27	-3.12	-2.39	9.19	-1.81	-3.64	-1.11	-1.77
4A	3.15	12.17	16-Mar-01	17.69	-0.46	-6.07	-0.27	1.46	NA	NA	NA	-1.84	-3.87	-1.40	-1.96
4A	3.65	11.67	16-Mar-01	18.10	-0.46	-6.08	-0.30	1.40	-2.83	-2.09	9.56	-1.75	-3.54	-1.31	-1.68
4A	4.15	11.17	16-Mar-01	14.81	-0.64	-5.44	0.34	2.68	NA	NA	NA	-1.22	-2.73	-1.11	-1.17
4A	4.65	10.67	16-Mar-01	13.46	-0.77	-4.11	1.65	5.29	NA	NA	NA	-0.85	-2.09	0.32	-0.63
4A	5.15	10.17	16-Mar-01	11.77	-0.94	-3.80	1.94	5.87	-1.75	-1.02	9.24	-0.74	-1.93	0.58	-0.56
4A	5.65	9.67	16-Mar-01	12.19	-0.88	-3.95	1.83	5.65	-1.52	-0.79	9.78	-0.85	-2.16	0.49	-0.79
4A	6.15	9.17	16-Mar-01	11.64	-0.81	-3.67	2.15	6.29	-1.71	-0.98	10.23	-1.17	-2.50	0.45	-1.01
4A	6.65	8.67	16-Mar-01	13.37	-0.69	-4.26	1.45	4.89	-2.15	-1.42	10.13	-1.48	-3.25	-0.02	-1.38
4A	7.15	8.17	16-Mar-01	12.07	-0.71	-4.12	1.67	5.32	NA	NA	NA	-1.51	-3.41	-0.22	-1.63
4A	7.65	7.67	16-Mar-01	4.54	-1.10	-5.76	0.01	2.01	-0.33	0.40	12.56	-2.35	-4.74	-1.06	-2.24
4A	8.15	7.17	16-Mar-01	5.23	-0.95	-6.87	-1.12	-0.24	-1.10	-0.37	10.33	-2.95	-6.04	-1.41	-2.81
4A	8.65	6.67	16-Mar-01	3.98	-1.06	-7.14	-1.39	-0.80	-0.66	0.07	11.13	-3.34	-6.77	-1.67	-3.37
6	1.68	13.98	14-Mar-01	DRY	--	--	--	--	--	--	--	--	--	--	--
6	2.68	12.98	14-Mar-01	5.17	-0.86	-8.04	-2.20	-2.40	-3.19	-2.46	7.52	-4.25	-8.59	-2.60	-3.51
6	3.68	11.98	14-Mar-01	4.09	-1.01	-6.92	-1.09	-0.18	-2.98	-2.25	6.70	-3.42	-7.32	-1.59	-3.50
6	4.68	10.98	14-Mar-01	3.14	-1.20	-6.44	-0.65	0.70	-2.87	-2.14	6.03	-3.10	-6.45	-1.14	-2.93
6	5.18	10.48	14-Mar-01	4.93	-0.99	-6.91	-1.12	-0.25	-2.02	-1.29	7.60	-3.13	-6.59	-1.19	-3.04
6	6.18	9.48	14-Mar-01	4.91	-1.03	-6.72	-0.94	0.10	-1.22	-0.49	8.62	-2.64	-5.75	-0.91	-2.67

Table C3 (cont.) TIC, Log P_{CO2}, and saturation indices calculated using PHREEQC: May 2001 sampling event.

Well ID	Depth (m, bgs)	Elevation (m, AHD)	Sample Date			Saturation indices									
				TIC	log PCO2	Fe(OH) ₃	Goethite	Hematite	FeS(ppt)	Mackinawite	Pyrite	Calcite	Dolomite	Siderite	Rhodochrosite
6	7.18	8.48	14-Mar-01	4.31	-1.12	-8.62	-2.89	-3.79	-1.88	-1.15	3.59	-2.47	-5.55	-1.02	-2.65
6	8.18	7.48	14-Mar-01	4.78	-1.08	-6.24	-0.47	1.05	-1.46	-0.72	9.46	-2.28	-4.93	-1.08	-2.20
6	9.18	6.48	14-Mar-01	4.65	-1.08	-6.23	-0.45	1.09	-1.23	-0.49	9.77	-2.42	-5.01	-1.03	-2.59
6	10.18	5.48	14-Mar-01	7.69	-0.83	-6.34	-0.62	0.74	-1.58	-0.84	9.01	-2.48	-5.42	-0.81	-2.44
6	11.18	4.48	14-Mar-01	10.56	-0.68	-6.30	-0.54	0.91	-1.79	-1.06	8.46	-2.35	-4.97	-0.61	-2.16
6	12.18	3.48	14-Mar-01	8.83	-0.75	-6.36	-0.61	0.77	-2.05	-1.31	8.09	-2.41	-5.14	-0.76	-2.30
6	13.18	2.48	14-Mar-01	17.70	-0.50	-5.77	0.00	1.98	-1.53	-0.80	8.68	-1.45	-3.23	0.03	-2.53
6	14.18	1.48	14-Mar-01	19.31	-0.41	-5.93	-0.17	1.65	-1.70	-0.96	9.03	-2.04	-4.17	-0.22	-2.85
6	15.18	0.48	14-Mar-01	16.51	-0.51	-5.81	-0.05	1.89	-1.53	-0.79	8.92	-2.11	-4.17	-0.08	-2.68
6	16.18	-0.52	14-Mar-01	13.60	-0.64	-5.30	0.49	2.97	-0.74	-0.01	10.17	-2.20	-4.04	0.22	-2.29
6	18.18	-2.52	14-Mar-01	5.15	-0.94	-6.21	-0.46	1.06	-2.24	-1.51	9.10	-3.51	-7.06	-1.31	-3.74
10	3.0	13.63	20-Mar-01	15.24	-0.57	-4.04	1.78	5.57	-2.05	-1.32	10.18	-1.12	-2.90	0.24	-1.31
10	4.0	12.63	20-Mar-01	23.33	-0.63	-3.77	2.02	6.04	-0.99	-0.26	10.20	-0.19	-0.81	1.08	-0.43
10	5.0	11.63	20-Mar-01	23.41	-0.51	-4.44	1.37	4.74	-1.65	-0.92	9.00	-0.46	-1.34	0.69	-0.70
10	6.0	10.63	20-Mar-01	24.68	-0.40	-4.66	1.09	4.17	-1.90	-1.17	9.00	-1.05	-2.04	0.51	-0.90
10	7.0	9.63	20-Mar-01	18.87	-0.35	-6.99	-1.24	-0.49	-1.99	-1.26	10.19	-3.14	-6.59	-1.43	-3.78
10	8.0	8.63	20-Mar-01	16.16	-0.42	-7.81	-2.07	-2.15	-2.02	-1.29	9.01	-3.29	-6.86	-1.67	-3.88
10	9.0	7.63	20-Mar-01	16.78	-0.40	-8.09	-2.35	-2.72	-1.93	-1.20	8.14	-3.38	-6.86	-1.50	-3.89
10	10.0	6.63	20-Mar-01	14.87	-0.47	-7.57	-1.83	-1.68	-2.30	-1.57	7.05	-3.38	-6.80	-1.10	-3.57
10	11.0	5.63	20-Mar-01	6.67	-0.86	-8.78	-3.04	-4.10	-1.92	-1.18	7.66	-2.81	-5.77	-2.24	-2.80

Table C4. TIC, Log P_{CO2}, and saturation indices calculated using PHREEQC: May 2001 sampling event.

Well ID	Depth (m, bgs)	Elevation (m, AHD)	Sample Date			Saturation indices									
				TIC	log PCO2	Fe(OH) ₃	Goethite	Hematite	FeS(ppt)	Mackinawite	Pyrite	Calcite	Dolomite	Siderite	Rhodochrosite
3	1.40	14.41	22-May-01	6.28	-0.85	-8.05	-2.36	-2.74	NA	NA	NA	-3.44	-7.58	-3.91	NA
3	2.40	13.41	22-May-01	4.66	-1.04	NA	NA	NA	NA	NA	NA	-2.42	-5.75	NA	NA
3	2.90	12.91	22-May-01	4.39	-1.17	-6.30	-0.60	0.79	NA	NA	NA	-1.50	-3.82	-2.67	-2.27
3	3.40	12.41	22-May-01	7.89	-0.98	-6.72	-1.01	-0.03	-0.63	0.10	6.18	-1.04	-2.71	0.36	-0.69
3	3.90	11.91	22-May-01	13.88	-0.80	-5.59	0.12	2.23	-0.55	0.18	7.16	-0.49	-1.60	1.03	-0.27
3	4.40	11.41	22-May-01	22.36	-0.56	-5.81	-0.09	1.80	-0.79	-0.05	6.75	-0.30	-1.29	1.09	-0.26
3	4.90	10.91	22-May-01	24.22	-0.54	-4.91	0.80	3.59	NA	NA	NA	-0.28	-0.95	1.15	-0.22
3	5.40	10.41	22-May-01	23.29	-0.46	-5.60	0.13	2.24	NA	NA	NA	-0.65	-1.75	0.72	-0.73
3	5.90	9.91	22-May-01	20.26	-0.51	-5.59	0.13	2.24	NA	NA	NA	-0.85	-2.16	0.75	-0.83
3	6.40	9.41	22-May-01	20.62	-0.43	-6.37	-0.65	0.69	NA	NA	NA	-1.32	-3.07	0.32	-1.12
3	6.90	8.91	22-May-01	DRY	--	--	--	--	--	--	--	--	--	--	--
3	7.40	8.41	22-May-01	DRY	--	--	--	--	--	--	--	--	--	--	--
3	7.90	7.91	22-May-01	14.86	-0.45	-9.92	-4.22	-6.46	-1.89	-1.15	6.56	-3.76	-8.56	-2.19	NA
3	8.40	7.41	22-May-01	13.51	-0.48	-10.74	-5.01	-8.04	-2.61	-1.88	4.08	-4.07	-8.83	-2.43	NA
4	3.34	13.98	16-May-01	0.92	-2.05	-7.71	-2.00	-2.02	-3.06	-2.33	4.07	-1.89	-4.49	-2.28	NA
4	3.84	13.48	16-May-01	1.67	-1.87	-7.95	-2.24	-2.49	-3.12	-2.39	2.62	-1.32	-3.49	-1.82	NA
4	4.34	12.98	16-May-01	5.00	-1.21	-8.40	-2.69	-3.39	NA	NA	NA	-1.18	-3.08	-0.64	NA
4	4.84	12.48	16-May-01	4.53	-1.28	-7.51	-1.80	-1.61	-2.52	-1.78	2.20	-1.16	-3.10	-0.40	NA
4	5.34	11.98	16-May-01	5.05	-1.25	-6.86	-1.14	-0.29	-2.13	-1.40	3.11	-1.14	-2.74	0.03	-1.63
4	5.84	11.48	16-May-01	7.57	-0.78	-10.00	-4.28	-6.56	-5.38	-4.65	0.36	-2.53	-5.53	-2.85	NA
4	6.34	10.98	16-May-01	4.45	-0.97	-9.71	-3.92	-5.84	-5.56	-4.83	1.07	-3.12	-6.59	-3.23	NA
4	6.84	10.48	16-May-01	3.37	-1.09	-9.22	-3.35	-4.70	-3.81	-3.08	4.12	-2.90	-6.18	-2.91	NA
4	7.34	9.98	16-May-01	4.05	-1.03	-9.08	-3.26	-4.51	NA	NA	NA	-2.81	-6.03	-2.87	NA
4	7.84	9.48	16-May-01	4.95	-0.97	-9.47	-3.70	-5.40	NA	NA	NA	-2.58	-5.56	-2.93	NA
4	8.34	8.98	16-May-01	5.49	-0.89	-10.33	-4.58	-7.18	NA	NA	NA	-3.19	-6.75	-3.37	NA
4	8.84	8.48	16-May-01	8.12	-0.71	-11.21	-5.51	-9.04	-6.33	-5.59	-1.06	-3.69	-7.60	-3.74	NA

Table C4 (cont.) TIC, Log P_{CO2}, and saturation indices calculated using PHREEQC: May 2001 sampling event.

Well ID	Depth (m, bgs)	Elevation (m, AHD)	Sample Date			Saturation indices									
				TIC	log PCO2	Fe(OH) ₃	Goethite	Hematite	FeS(ppt)	Mackinawite	Pyrite	Calcite	Dolomite	Siderite	Rhodochrosite
4	9.34	7.98	16-May-01	4.88	-0.94	-11.06	-5.34	-8.70	NA	NA	NA	-3.60	-7.42	-3.74	NA
4A	2.15	13.17	21-May-01	4.13	-1.21	-8.05	-2.40	-2.82	-3.08	-2.35	4.78	-1.81	-3.97	-1.89	NA
4A	2.65	12.67	21-May-01	20.30	-0.48	-8.97	-3.29	-4.59	-1.76	-1.02	4.43	-0.88	-2.19	-0.78	-1.07
4A	3.15	12.17	21-May-01	26.36	-0.37	-7.54	-1.84	-1.70	-1.44	-0.70	4.27	-0.72	-1.82	0.54	-0.62
4A	3.65	11.67	21-May-01	23.56	-0.43	-7.16	-1.46	-0.94	-1.08	-0.35	5.61	-0.74	-1.85	0.52	-0.70
4A	4.15	11.17	21-May-01	19.75	-0.53	-7.30	-1.60	-1.21	-0.80	-0.06	6.07	-0.74	-1.87	0.37	-0.83
4A	4.65	10.67	21-May-01	22.69	-0.41	-8.15	-2.45	-2.91	-1.89	-1.15	4.81	-1.04	-2.28	-0.38	-1.20
4A	5.15	10.17	21-May-01	16.26	-0.64	-6.72	-1.02	-0.05	-1.22	-0.49	5.63	-0.84	-2.06	0.51	-0.84
4A	5.65	9.67	21-May-01	16.10	-0.63	-6.04	-0.34	1.31	NA	NA	NA	-0.86	-2.22	0.54	-0.89
4A	6.15	9.17	21-May-01	17.86	-0.50	-6.37	-0.65	0.68	-2.65	-1.92	4.81	-1.32	-3.00	0.18	-1.85
4A	6.65	8.67	21-May-01	18.67	-0.46	-7.50	-1.81	-1.63	-2.79	-2.06	4.20	-1.59	-3.51	-0.40	-2.71
4A	7.15	8.17	21-May-01	16.62	-0.52	-7.44	-1.73	-1.47	-1.80	-1.07	5.29	-1.51	-3.38	-0.14	-2.57
4A	7.65	7.67	21-May-01	12.73	-0.52	-10.39	-4.69	-7.41	-2.49	-1.76	4.34	-3.38	-7.15	-2.24	NA
4A	8.15	7.17	21-May-01	13.16	-0.50	-11.01	-5.32	-8.66	-3.34	-2.61	3.16	-3.78	-7.65	-2.81	NA
4A	8.65	6.67	21-May-01	13.29	-0.50	-11.22	-5.54	-9.10	-3.89	-3.16	2.37	-3.95	-7.93	-3.04	NA
6	1.68	13.98	23-May-01	DRY	--	--	--	--	--	--	--	--	--	--	--
6	2.68	12.98	23-May-01	5.01	-0.90	-10.90	-5.18	-8.37	-4.91	-4.18	1.48	-4.94	-10.21	-3.63	NA
6	3.68	11.98	23-May-01	9.24	-0.65	-9.18	-3.43	-4.88	-2.51	-1.78	4.67	-3.63	-7.53	-1.71	NA
6	4.68	10.98	23-May-01	6.39	-0.84	-8.34	-2.58	-3.18	-1.58	-0.85	6.15	-2.95	-6.37	-1.21	NA
6	6.18	9.48	23-May-01	7.67	-0.81	-8.25	-2.51	-3.03	-1.34	-0.60	5.72	-2.36	-5.15	-0.81	NA
6	8.18	7.48	23-May-01	9.56	-0.70	-8.89	-3.17	-4.35	-1.88	-1.15	5.26	-2.22	-5.05	-1.33	NA
6	10.18	5.48	23-May-01	23.69	-0.31	-7.99	-2.29	-2.59	-2.17	-1.43	4.62	-1.79	-4.04	-0.31	-2.72
6	11.18	4.48	23-May-01	29.91	-0.19	-8.31	-2.60	-3.21	-2.18	-1.45	4.40	-1.91	-4.28	-0.34	-3.08
6	12.18	3.48	23-May-01	27.01	-0.25	-7.96	-2.26	-2.53	-2.31	-1.58	4.62	-1.92	-3.98	-0.33	NA
6	13.18	2.48	23-May-01	24.33	-0.31	-7.28	-1.54	-1.09	-2.06	-1.33	4.79	-2.13	-3.97	0.17	NA

Table C4 (cont.) TIC, Log P_{CO2}, and saturation indices calculated using PHREEQC: May 2001 sampling event.

Well ID	Depth (m, bgs)	Elevation (m, AHD)	Sample Date			Saturation indices									
				TIC	log PCO2	Fe(OH) ₃	Goethite	Hematite	FeS(ppt)	Mackinawite	Pyrite	Calcite	Dolomite	Siderite	Rhodochrosite
6	14.18	1.48	23-May-01	26.95	-0.23	-7.78	-2.05	-2.11	-2.41	-1.68	4.15	-2.41	-4.60	-0.12	NA
6	15.18	0.48	23-May-01	25.71	-0.27	-7.57	-1.87	-1.75	-2.28	-1.55	4.46	-2.33	-4.41	-0.02	NA
6	16.18	-0.52	23-May-01	23.13	-0.34	-6.97	-1.25	-0.51	-1.97	-1.24	5.69	-2.10	-4.00	0.11	NA
6	18.18	-2.52	23-May-01	26.83	-0.19	-8.46	-2.77	-3.55	NA	NA	NA	-3.71	-7.08	-1.05	NA
10	3.00	13.63	24-May-01	16.39	-0.52	-6.81	-1.05	-0.11	-1.92	-1.19	5.51	-1.12	-3.00	0.11	NA
10	4.00	12.63	24-May-01	13.63	-0.75	-5.30	0.47	2.94	-0.74	0.00	7.98	-0.63	-1.79	0.84	-0.88
10	5.00	11.63	24-May-01	15.36	-0.71	-5.26	0.51	3.01	-1.55	-0.81	6.63	-0.61	-1.60	0.83	-0.61
10	6.00	10.63	24-May-01	22.81	-0.50	-5.68	0.08	2.14	-1.07	-0.34	7.29	-0.52	-1.33	0.86	-0.60
10	7.00	9.63	24-May-01	21.56	-0.32	-8.02	-2.27	-2.55	-1.76	-1.03	6.19	-2.33	-4.85	-0.60	NA
10	8.00	8.63	24-May-01	22.26	-0.28	-9.12	-3.38	-4.77	-2.14	-1.41	5.41	-3.03	-6.13	-1.29	NA
10	9.00	7.63	24-May-01	20.06	-0.33	-9.17	-3.44	-4.89	-2.03	-1.30	5.60	-3.15	-6.28	-1.36	NA
10	10.00	6.63	24-May-01	16.94	-0.40	-9.22	-3.49	-4.99	-3.42	-2.69	2.68	-3.30	-6.71	-1.42	NA
10	11.00	5.63	24-May-01	10.52	-0.62	-10.87	-5.15	-8.31	-2.96	-2.23	4.37	-3.03	-6.32	-2.99	NA
10	12.00	4.63	24-May-01	DRY	--	--	--	--	--	--	--	--	--	--	--
10	13.00	3.63	24-May-01	7.25	-0.83	-7.69	-1.97	-1.94	-2.43	-1.70	5.46	-2.46	-5.08	-1.09	NA

Table C5. TIC, Log P_{CO2}, and saturation indices calculated using PHREEQC: July 2001 sampling event.

Well ID	Depth (m, bgs)	Elevation (m, AHD)	Sample Date			Saturation indices									
				TIC	log PCO2	Fe(OH) ₃	Goethite	Hematite	FeS(ppt)	Mackinawite	Pyrite	Calcite	Dolomite	Siderite	Rhodochrosite
2	6-9	12.46 - 9.46	25-Jul-00	35.01	-0.37	-5.96	-0.26	1.45	-0.99	-0.26	6.39	-0.20	-0.75	1.18	-0.63
3	1.40	14.41	03-Aug-01	3.60	-1.14	-7.59	-1.99	-2.01	NA	NA	NA	-2.76	-6.14	-3.64	NA
3	2.40	13.41	03-Aug-01	3.93	-1.16	-7.21	-1.59	-1.20	-5.22	-4.48	5.84	-2.03	-4.73	-3.04	NA
3	2.90	12.91	03-Aug-01	5.48	-1.11	-8.16	-2.53	-3.09	-4.08	-3.35	3.62	-1.45	-3.70	-2.12	NA
3	3.40	12.41	03-Aug-01	14.25	-0.66	-8.05	-2.42	-2.86	-1.68	-0.94	6.51	-1.03	-2.85	-0.96	-1.46
3	3.90	11.91	03-Aug-01	14.77	-0.75	-5.37	0.26	2.50	-0.11	0.63	9.66	-0.63	-2.03	0.71	-0.22
3	4.40	11.41	02-Aug-01	21.86	-0.63	-5.99	-0.33	1.32	-0.60	0.14	7.12	-0.24	-1.17	0.93	0.10
3	4.90	10.91	03-Aug-01	25.17	-0.55	-5.40	0.23	2.43	-1.34	-0.61	7.18	-0.27	-1.20	0.91	-0.40
3	5.40	10.41	02-Aug-01	22.30	-0.58	-6.45	-0.77	0.43	-1.66	-0.93	6.28	-0.44	-1.34	0.25	-0.60
3	5.90	9.91	03-Aug-01	15.53	-0.66	-6.44	-0.78	0.41	NA	NA	NA	-1.01	-2.44	0.42	-1.27
3	6.40	9.41	02-Aug-01	13.87	-0.60	-7.88	-2.20	-2.41	-1.39	-0.65	4.80	-1.73	-4.06	-0.06	-1.56
3	6.90	8.91	02-Aug-01	DRY	--	--	--	--	--	--	--	--	--	--	--
3	7.40	8.41	02-Aug-01	DRY	--	--	--	--	--	--	--	--	--	--	--
3	7.90	7.91	02-Aug-01	9.45	-0.66	-10.86	-5.20	-8.42	-1.84	-1.10	4.56	NA	NA	-2.31	NA
3	8.40	7.41	03-Aug-01	16.47	-0.42	-11.35	-5.73	-9.49	-3.05	-2.31	3.17	-6.58	-11.27	-2.73	NA
3	8.90	6.91	02-Aug-01	10.17	-0.62	-11.33	-5.67	-9.36	-2.97	-2.23	3.05	NA	NA	-2.84	NA
3	9.40	6.41	02-Aug-01	8.27	-0.72	-11.23	-5.59	-9.21	-2.97	-2.24	3.48	-5.78	-10.39	-3.00	NA
3	10.40	5.41	02-Aug-01	12.22	-0.54	-11.53	-5.88	-9.79	-3.79	-3.06	2.04	NA	NA	-3.10	NA
3	11.40	4.41	02-Aug-01	7.01	-0.80	-11.44	-5.83	-9.69	-3.63	-2.90	3.12	-4.64	-9.15	-3.53	NA
3	12.40	3.41	02-Aug-01	6.20	-0.84	-12.40	-6.74	-11.51	-4.24	-3.51	1.78	-4.35	-8.81	-4.19	NA
3	13.40	2.41	02-Aug-01	6.23	-0.86	-11.85	-6.24	-10.52	-4.69	-3.96	1.44	-3.97	-8.17	-4.01	NA
3	14.40	1.41	02-Aug-01	5.74	-0.90	-11.91	-6.29	-10.62	-4.62	-3.89	1.14	-3.76	-7.74	-3.95	NA
3	15.40	0.41	02-Aug-01	6.83	-0.83	-11.93	-6.32	-10.67	-4.74	-4.00	1.28	-3.65	-7.49	-4.01	NA
3	16.40	-0.59	02-Aug-01	4.58	-1.00	-12.01	-6.38	-10.79	-5.22	-4.49	0.46	-3.67	-7.58	-4.29	NA
3	17.40	-1.59	02-Aug-01	3.69	-1.11	-11.57	-5.99	-10.02	-4.50	-3.76	2.20	-3.88	-8.03	-4.18	NA
3	18.40	-2.59	02-Aug-01	4.45	-1.01	-12.09	-6.51	-11.05	-5.19	-4.45	1.37	-4.06	-8.32	-4.62	NA
3	19.40	-3.59	02-Aug-01	29.34	-0.15	-13.80	-8.16	-14.35	-8.07	-7.34	-1.69	-5.26	-10.70	-5.82	NA

Table C5 (cont.) TIC, Log P_{CO2}, and saturation indices calculated using PHREEQC: July 2001 sampling event.

Well ID	Depth (m, bgs)	Elevation (m, AHD)	Sample Date	Saturation indices											
				TIC	log PCO2	Fe(OH) ₃	Goethite	Hematite	FeS(ppt)	Mackinawite	Pyrite	Calcite	Dolomite	Siderite	Rhodochrosite
4	1.84	15.48	06-Sep-01	DRY	--	--	--	--	--	--	--	--	--	--	--
4	2.34	14.98	06-Sep-01	DRY	--	--	--	--	--	--	--	--	--	--	--
4	2.84	14.48	06-Sep-01	6.59	-0.88	-9.79	-4.16	-6.35	-1.29	-0.56	6.20	-2.82	-5.81	-2.00	NA
4	3.34	13.98	06-Sep-01	5.72	-1.19	-9.01	-3.39	-4.80	-2.50	-1.77	4.10	-1.05	-2.85	-1.88	NA
4	3.84	13.48	06-Sep-01	6.70	-1.21	-8.46	-2.83	-3.69	-2.17	-1.43	4.00	-0.80	-2.11	-1.28	NA
4	4.34	12.98	06-Sep-01	6.16	-1.30	-7.27	-1.63	-1.28	-0.78	-0.04	5.76	-0.77	-2.02	-0.23	-1.25
4	4.84	12.48	06-Sep-01	4.98	-1.40	-6.73	-1.09	-0.21	-1.27	-0.54	5.31	-0.94	-2.37	-0.16	-1.25
4	5.34	11.98	06-Sep-01	11.10	-0.75	-7.59	-1.95	-1.93	-2.35	-1.62	3.96	-1.46	-3.38	-0.35	-2.74
4	5.84	11.48	06-Sep-01	3.16	-1.22	-9.42	-3.78	-5.58	-4.95	-4.21	1.16	-2.83	-6.12	-2.84	NA
4	6.34	10.98	06-Sep-01	3.15	-1.22	NA	NA	NA	NA	NA	NA	-2.89	-6.15	NA	NA
4	6.84	10.48	06-Sep-01	DRY	--	--	--	--	--	--	--	--	--	--	--
4	7.34	9.98	06-Sep-01	3.12	-1.29	-8.99	-3.37	-4.76	-3.82	-3.09	2.64	-2.32	-5.20	-2.37	NA
4	7.84	9.48	06-Sep-01	4.22	-1.09	-10.18	-4.57	-7.16	-4.01	-3.28	2.16	-2.82	-6.14	-2.92	NA
4	8.34	8.98	06-Sep-01	7.24	-0.78	-11.93	-6.29	-10.60	-5.78	-5.05	-0.26	-3.98	-8.41	-4.15	NA
4	8.84	8.48	06-Sep-01	5.78	-0.89	-11.32	-5.66	-9.35	NA	NA	NA	-3.40	-7.09	-3.74	NA
4	9.34	7.98	06-Sep-01	5.20	-0.95	-11.47	-5.83	-9.69	NA	NA	NA	-3.39	-7.06	-3.86	NA
4	9.84	7.48	10-Sep-01	5.07	-0.94	-11.45	-5.80	-9.63	-5.80	-5.07	-0.59	-3.75	-7.83	-3.91	NA
4	10.34	6.98	10-Sep-01	15.77	-0.42	-12.84	-7.19	-12.40	-7.20	-6.47	-1.95	-4.96	-10.40	-4.79	NA
4	10.84	6.48	10-Sep-01	16.76	-0.39	-13.57	-7.91	-13.84	-7.89	-7.15	-2.72	-5.06	-10.58	-5.45	NA
4	11.34	5.98	10-Sep-01	15.95	-0.41	-12.92	-7.26	-12.54	NA	NA	NA	-5.05	-10.57	-4.92	NA
4	12.34	4.98	10-Sep-01	5.48	-0.89	-11.88	-6.22	-10.47	-7.15	-6.41	-1.51	-4.48	-9.41	-4.73	NA
4	13.34	3.98	10-Sep-01	13.29	-0.49	-12.73	-7.07	-12.17	-7.12	-6.39	-1.82	-4.94	-10.32	-4.79	NA
4	14.34	2.98	10-Sep-01	6.07	-0.84	-12.06	-6.41	-10.84	-6.34	-5.60	-1.25	-4.56	-9.36	-4.35	NA
4	15.34	1.98	10-Sep-01	8.56	-0.69	-13.06	-7.41	-12.83	-7.01	-6.27	-1.69	-4.72	-9.63	-5.17	NA
4	16.34	0.98	10-Sep-01	3.97	-1.03	-12.37	-6.73	-11.48	-6.26	-5.53	-0.03	-4.64	-9.49	-5.10	NA
4	18.34	-1.02	10-Sep-01	7.72	-0.74	-12.61	-6.98	-11.98	-7.50	-6.77	-2.61	-4.87	-9.96	-4.94	NA

Table C5 (cont.) TIC, Log P_{CO2}, and saturation indices calculated using PHREEQC: July 2001 sampling event.

[illegible]

Table C5 (cont.) TIC, Log P_{CO2}, and saturation indices calculated using PHREEQC: July 2001 sampling event.

Well ID	Depth (m, bgs)	Elevation (m, AHD)	Sample Date			Saturation indices									
				TIC	log PCO2	Fe(OH) ₃	Goethite	Hematite	FeS(ppt)	Mackinawite	Pyrite	Calcite	Dolomite	Siderite	Rhodochrosite
6	2.68	12.98	31-Jul-01	7.48	-0.75	-11.73	-6.08	-10.19	-6.80	-6.06	-0.13	-4.80	-9.98	-4.72	NA
6	3.68	11.98	31-Jul-01	4.67	-0.97	-9.65	-3.96	-5.94	-2.59	-1.86	4.18	-3.66	-7.83	-2.20	NA
6	4.68	10.98	31-Jul-01	23.12	-0.29	-8.31	-2.63	-3.28	-2.41	-1.68	3.37	-2.46	-5.24	-0.25	NA
6	5.18	10.48	31-Jul-01	21.56	-0.33	-8.18	-2.48	-2.99	-2.87	-2.13	2.43	-2.10	-4.66	-0.19	-3.20
6	5.68	9.98	31-Jul-01	13.35	-0.56	-8.53	-2.83	-3.68	-1.73	-0.99	5.07	-2.10	-4.67	-0.77	-3.69
6	6.18	9.48	31-Jul-01	10.21	-0.69	-9.03	-3.34	-4.70	-1.84	-1.11	4.50	-2.31	-4.93	-1.12	NA
6	6.68	8.98	31-Jul-01	12.55	-0.60	-8.88	-3.19	-4.41	-1.71	-0.98	4.52	-2.11	-4.63	-0.84	NA
6	7.18	8.48	31-Jul-01	14.99	-0.53	-9.34	-3.65	-5.32	-1.66	-0.93	4.42	-1.85	-4.17	-1.01	-2.73
6	7.68	7.98	31-Jul-01	20.47	-0.38	-9.21	-3.54	-5.09	-1.73	-0.99	4.82	-1.75	-4.01	-0.95	-1.71
6	8.18	7.48	31-Jul-01	26.70	-0.23	-9.57	-3.91	-5.83	-2.54	-1.80	3.76	-2.27	-5.02	-1.22	-2.28
6	8.68	6.98	30-Jul-01	18.74	-0.39	-10.41	-4.74	-7.50	-2.52	-1.79	2.40	-2.26	-5.20	-1.48	-2.73
6	9.18	6.48	30-Jul-01	15.96	-0.46	-10.26	-4.59	-7.20	-2.20	-1.47	3.37	-2.52	-5.63	-1.56	NA
6	9.68	5.98	30-Jul-01	26.47	-0.28	-9.16	-3.49	-5.00	-2.59	-1.85	2.52	-2.01	-4.24	-0.62	NA
6	10.18	5.48	30-Jul-01	28.33	-0.25	-9.03	-3.35	-4.73	-2.97	-2.24	1.73	-2.09	-4.14	-0.50	NA
6	10.68	4.98	30-Jul-01	28.44	-0.24	-9.04	-3.36	-4.74	-2.23	-1.50	3.21	-2.12	-4.20	-0.50	NA
6	11.18	4.48	30-Jul-01	29.53	-0.23	-8.95	-3.28	-4.59	-2.62	-1.88	2.75	-2.08	-4.11	-0.53	NA
6	11.68	3.98	30-Jul-01	28.94	-0.25	-8.75	-3.10	-4.23	-3.03	-2.29	2.37	-2.15	-4.15	-0.55	NA
6	12.18	3.48	30-Jul-01	32.82	-0.18	-8.92	-3.25	-4.53	-3.09	-2.36	2.10	-2.27	-4.29	-0.55	NA
6	12.68	2.98	30-Jul-01	29.85	-0.23	-7.52	-1.86	-1.74	-2.23	-1.50	4.94	-2.22	-4.14	-0.05	NA
6	13.18	2.48	25-Jul-01	26.32	-0.27	-8.05	-2.38	-2.78	-1.93	-1.20	4.48	-2.40	-4.51	-0.09	NA
6	13.68	1.98	25-Jul-01	31.44	-0.17	-9.03	-3.35	-4.73	-2.46	-1.72	2.79	-2.82	-5.30	-0.43	NA
6	14.18	1.48	25-Jul-01	28.47	-0.22	-8.58	-2.91	-3.84	NA	NA	NA	-2.79	-5.23	-0.37	NA
6	15.18	0.48	25-Jul-01	30.67	-0.18	-8.97	-3.30	-4.62	-2.39	-1.66	2.73	-2.75	-5.19	-0.34	NA
6	16.18	-0.52	25-Jul-01	18.75	-0.41	-8.52	-2.84	-3.70	-1.94	-1.21	3.61	-2.81	-5.28	-0.27	NA
6	18.18	-2.52	25-Jul-01	27.67	-0.19	-10.30	-4.63	-7.29	-2.86	-2.12	0.86	-3.64	-6.89	-0.92	NA
10	3.00	13.63	22-Aug-01	45.71	-0.05	-8.42	-2.73	-3.48	-4.01	-3.27	2.98	-0.78	-2.41	-1.01	-1.55
10	4.00	12.63	22-Aug-01	30.99	-0.34	-6.61	-0.91	0.16	-1.54	-0.81	6.18	-0.31	-1.39	0.48	-1.21

Table C5 (cont.) TIC, Log P_{CO2}, and saturation indices calculated using PHREEQC: July 2001 sampling event.

Well ID	Depth (m, bgs)	Elevation (m, AHD)	Sample Date			Saturation indices									
				TIC	log PCO2	Fe(OH) ₃	Goethite	Hematite	FeS(ppt)	Mackinawite	Pyrite	Calcite	Dolomite	Siderite	Rhodochrosite
10	5.00	11.63	22-Aug-01	31.18	-0.43	-6.13	-0.42	1.15	-1.80	-1.06	5.74	-0.13	-0.67	0.69	-1.03
10	6.00	10.63	22-Aug-01	28.77	-0.37	-6.35	-0.64	0.70	-1.67	-0.94	6.35	-0.62	-1.51	0.47	-0.77
10	7.00	9.63	22-Aug-01	15.34	-0.45	-9.30	-3.58	-5.18	-2.17	-1.44	5.45	-3.44	-6.61	-1.60	NA
10	8.00	8.63	22-Aug-01	19.65	-0.33	-10.07	-4.36	-6.74	-2.45	-1.72	4.60	-3.65	-7.04	-1.90	NA
10	9.00	7.63	22-Aug-01	20.42	-0.32	-9.96	-4.25	-6.51	-2.47	-1.73	4.19	-3.55	-6.84	-1.68	NA
10	10.00	6.63	22-Aug-01	16.10	-0.43	-9.69	-3.98	-5.97	-2.09	-1.35	4.36	-3.24	-6.35	-1.43	NA
10	11.00	5.63	22-Aug-01	10.25	-0.65	-10.00	-4.30	-6.61	-1.59	-0.86	4.64	-2.83	-5.73	-1.61	NA
10	12.00	4.63	22-Aug-01	DRY	--	--	--	--	--	--	--	--	--	--	--
10	13.00	3.63	22-Aug-01	7.24	-0.84	-8.69	-2.98	-3.98	-2.40	-1.67	3.97	-2.47	-5.17	-1.24	NA
10	14.00	2.63	22-Aug-01	7.00	-0.83	-9.02	-3.31	-4.64	-3.30	-2.56	3.79	-2.73	-5.73	-1.99	NA
10	15.00	1.63	22-Aug-01	3.97	-1.17	-8.63	-2.94	-3.89	-2.48	-1.75	3.37	-2.26	-4.78	-1.39	-2.44
10	16.00	0.63	22-Aug-01	2.79	-1.34	-8.25	-2.56	-3.14	-1.48	-0.74	6.01	-2.39	-5.07	-1.51	-2.14
10	18.00	-1.37	22-Aug-01	5.40	-1.00	-9.97	-4.29	-6.59	-4.25	-3.52	1.38	-2.24	-4.72	-2.62	NA
10	20.00	-3.37	22-Aug-01	6.43	-0.80	NA	NA	NA	NA	NA	NA	-4.88	-9.60	NA	NA
10	22.00	-5.37	22-Aug-01	6.11	-0.84	-12.08	-6.44	-10.90	-6.64	-5.91	-0.98	-4.65	-9.22	-4.65	NA
10	24.00	-7.37	22-Aug-01	7.13	-0.76	-12.36	-6.67	-11.36	-6.28	-5.54	-0.24	-5.22	-10.33	-4.77	NA
10	26.00	-9.37	22-Aug-01	7.28	-0.75	-13.10	-7.43	-12.88	-7.00	-6.27	-1.39	-5.45	-10.48	-5.36	NA
10	28.00	-11.37	22-Aug-01	4.03	-1.02	-10.25	-4.57	-7.17	-4.41	-3.68	1.31	-4.61	-8.86	-2.91	NA
10	30.00	-13.37	22-Aug-01	5.28	-0.95	-8.72	-3.04	-4.10	-2.05	-1.31	4.30	-3.81	-7.02	-1.24	NA

Table C6. TIC, Log P_{CO2}, and saturation indices calculated using PHREEQC: February 2002 sampling event.

Well ID	Depth (m, bgs)	Elevation (m, AHD)	Sample Date			Saturation indices									
				TIC	log PCO2	Fe(OH) ₃	Goethite	Hematite	FeS(ppt)	Mackinawite	Pyrite	Calcite	Dolomite	Siderite	Rhodochrosite
4A	2.15	13.17	14-Feb-02	5.80	-1.23	-6.07	-0.28	1.44	-1.40	-0.67	9.88	-0.85	-2.11	-1.22	-2.39
4A	2.65	12.67	14-Feb-02	9.82	-1.08	-5.54	0.21	2.41	-0.36	0.37	9.88	-0.32	-1.05	-0.01	-0.75
4A	3.15	12.17	14-Feb-02	15.81	-0.73	-4.80	0.95	3.90	-0.83	-0.10	9.05	-0.41	-1.25	0.78	-0.71
4A	3.65	11.67	14-Feb-02	14.95	-0.73	-4.43	1.31	4.60	-0.44	0.29	11.22	-0.58	-1.55	0.59	-0.89
4A	4.15	11.17	14-Feb-02	14.13	-0.74	-5.94	-0.20	1.59	-2.11	-1.38	8.45	-0.61	-1.60	-0.63	-0.94
4A	4.65	10.67	14-Feb-02	13.14	-0.78	-5.06	0.67	3.32	-1.84	-1.11	7.98	-0.80	-1.87	0.27	-0.97
4A	5.15	10.17	14-Feb-02	15.78	-0.82	-3.83	1.88	5.75	-0.70	0.04	10.07	-0.24	-0.95	1.11	-0.50
4A	5.65	9.67	14-Feb-02	16.53	-0.78	-3.77	1.95	5.89	-0.73	0.00	10.22	-0.24	-0.97	1.11	-0.63
4A	6.15	9.17	14-Feb-02	11.36	-0.87	-3.50	2.30	6.59	NA	NA	NA	-0.71	-1.82	0.68	-1.17
4A	7.15	8.17	14-Feb-02	10.54	-0.81	-4.17	1.62	5.23	-1.37	-0.63	10.23	-1.19	-2.72	0.37	-1.23
4A	8.15	7.17	14-Feb-02	6.76	-0.88	-8.00	-2.26	-2.53	-1.43	-0.70	9.43	-2.65	-5.17	-2.01	-3.07
4A	9.15	6.17	14-Feb-02	3.43	-1.12	-8.58	-2.85	-3.72	-2.39	-1.66	7.35	-3.47	-6.95	-2.57	-3.96
4A	10.15	5.17	14-Feb-02	2.14	-1.25	-7.85	-1.96	-1.92	-3.64	-2.91	7.09	-4.02	-7.98	-3.05	-4.70
4A	11.15	4.17	14-Feb-02	2.38	-1.21	-8.68	-2.85	-3.70	-4.03	-3.30	6.44	-4.47	-8.91	-3.57	-5.16
4A	12.15	3.17	14-Feb-02	1.45	-1.42	-8.24	-2.27	-2.52	-4.09	-3.35	5.87	-3.52	-7.15	-3.41	-3.87
4A	13.15	2.17	14-Feb-02	0.61	-1.82	-8.18	-2.28	-2.56	-4.61	-3.88	5.22	-3.96	-8.08	-3.86	-4.33
4A	14.15	1.17	14-Feb-02	0.74	-1.74	-7.95	-2.08	-2.14	-4.24	-3.51	6.62	-3.88	-7.90	-3.84	-4.27
30	1.55	13.95	15-Feb-02	22.18	-0.60	-2.89	2.87	7.73	-1.05	-0.32	10.37	-0.32	-0.86	1.59	-0.08
30	2.05	13.45	15-Feb-02	24.61	-0.55	-3.38	2.37	6.72	NA	NA	NA	-0.20	-0.70	1.52	-0.16
30	3.05	12.45	15-Feb-02	19.84	-0.70	-3.46	2.28	6.54	NA	NA	NA	-0.16	-0.73	1.46	-0.21
30	3.55	11.95	15-Feb-02	20.40	-0.67	-3.50	2.26	6.50	NA	NA	NA	-0.18	-0.74	1.46	-0.20
30	4.05	11.45	15-Feb-02	20.86	-0.66	-3.52	2.20	6.38	-0.52	0.21	10.54	-0.21	-0.82	1.43	-0.22
30	4.55	10.95	15-Feb-02	21.48	-0.63	-3.52	2.18	6.35	-0.45	0.29	10.93	-0.26	-0.92	1.38	-0.27
30	5.05	10.45	15-Feb-02	20.31	-0.64	-3.61	2.09	6.16	-0.60	0.14	10.72	-0.34	-1.10	1.29	-0.34
30	5.55	9.95	15-Feb-02	19.87	-0.64	-3.43	2.29	6.56	-0.74	0.00	10.93	-0.37	-1.10	1.24	-0.37
30	6.55	8.95	15-Feb-02	8.60	-1.19	-3.16	2.57	7.12	-0.13	0.60	11.78	-0.41	-1.36	1.00	-0.97
30	7.05	8.45	15-Feb-02	5.76	-1.25	-3.78	1.93	5.84	-0.90	-0.17	11.01	-0.92	-2.47	0.26	-1.07

Table C6 (cont.) TIC, Log P_{CO2}, and saturation indices calculated using PHREEQC: February 2002 sampling event.

Well ID	Depth (m, bgs)	Elevation (m, AHD)	Sample Date			Saturation indices									
				TIC	log PCO2	Fe(OH) ₃	Goethite	Hematite	FeS(ppt)	Mackinawite	Pyrite	Calcite	Dolomite	Siderite	Rhodochrosite
30	8.05	7.45	15-Feb-02	5.23	-0.89	-6.70	-0.97	0.06	-3.01	-2.28	11.22	-4.53	-8.78	-2.80	-4.40
30	9.05	6.45	15-Feb-02	6.84	-0.78	-8.26	-2.55	-3.12	-2.73	-2.00	9.46	-4.77	-9.18	-2.94	-4.61
30	10.05	5.45	15-Feb-02	8.36	-0.70	-8.39	-2.68	-3.37	-2.39	-1.65	8.71	-4.48	-8.65	-2.47	-4.13
30	11.05	4.45	15-Feb-02	5.09	-0.93	-5.81	-0.10	1.78	-1.15	-0.41	12.61	-4.11	-7.90	-1.45	-3.13
30	12.05	3.45	15-Feb-02	4.95	-0.93	-7.41	-1.69	-1.39	-2.44	-1.71	9.68	-4.24	-8.35	-2.41	-4.15
30	13.05	2.45	15-Feb-02	5.53	-0.88	-7.21	-1.50	-1.02	-3.10	-2.37	9.24	-4.36	-8.68	-2.52	-4.55
30	14.05	1.45	15-Feb-02	3.91	-1.03	-7.15	-1.44	-0.90	-3.39	-2.65	8.81	-4.39	-8.67	-2.68	-4.66
30	15.05	0.45	15-Feb-02	5.73	-0.85	-7.71	-2.00	-2.01	-4.33	-3.60	7.75	-4.43	-8.81	-3.15	-4.44

Appendix D

Research Publications

ANALYSIS OF CHEMICAL DYNAMICS OF A LEACHATE PLUME IN A SANDY AQUIFER: ASTROLABE PARK, SYDNEY, N.S.W., AUSTRALIA

Lange B Jorstad¹ & Jerzy Jankowski²

¹PhD research student, University of New South Wales Groundwater Centre, Sydney, Australia

²Senior Lecturer, University of New South Wales Groundwater Centre, Sydney, Australia

Abstract

An investigation was conducted on a leachate plume migrating from the Astrolabe Park landfill, a decommissioned municipal landfill in Sydney, Australia, to determine the processes responsible for physical and chemical variations observed in the plume over time. A network of multi-level piezometers was used to collect discrete-depth groundwater samples at 0.5- to 1-meter intervals below ground surface to generate vertical profiles of the groundwater chemistry at various locations within the plume. Nine sampling rounds were conducted between July 2000 and September 2001: five rounds to measure general parameters (pH, redox potential (Eh), electrical conductivity (EC), dissolved oxygen (DO) and temperature), major and minor elements; and an additional four events to measure general parameters only. Analysis of the chemical data demonstrates variations in the spatial distribution of chemical species within the leachate plume between successive sampling events. Variations in EC and HCO_3^- as well as changes in Eh and redox couples ($\text{NO}_3^-/\text{NH}_4^+$, $\text{SO}_4^{2-}/\text{S}^{2-}$, $\text{Fe}^{3+}/\text{Fe}^{2+}$) have been used to accurately define the dynamic nature of the leachate plume over short time intervals and underscore the importance of frequent detailed vertical monitoring of landfill leachate in groundwater.

Key Words: groundwater, leachate plume, landfill, vertical monitoring, redox zonation

Introduction

Accurate monitoring of contaminant plumes in groundwater can be tricky business, and landfill leachate plumes are no exception. Leachate plumes have been observed to contain a broad range of potential contaminants, based on the type of waste disposed of in a particular landfill, and records regarding waste characterisation at older landfills are often poor or non-existent (Baedecker and Back, 1979a; Ehrig, 1982; Nicholson *et al.*, 1983; Christensen *et al.*, 1994). Hydrogeological investigations are further complicated by local heterogeneities typically present in the subsurface that may have important implications for both the physical transport of, and chemical interactions with, the contaminants in solution (Nicholson *et al.*, 1983; Bjerg *et al.*, 1995; Jankowski and Acworth, 1997). A significant burden of responsibility is placed on the investigator to design a carefully planned monitoring network and sampling strategy to account for all variables.

Although groundwater contaminant plumes are often depicted on a macroscopic scale as relatively static, symmetrical tongues slowly advancing through the aquifer, detailed vertical and lateral monitoring reveals a dynamic nature over relatively short time intervals. The spatial distribution of contaminants in a plume can be affected by both physical and chemical factors, including: the presence of preferential flow paths in the aquifer matrix, dispersion, ion exchange, dissolution/precipitation, redox and sorption reactions to name a few (Baedecker and Back, 1979a&b; Nicholson *et al.*, 1983; Kehew and Passero, 1990; Christensen *et al.*, 1994; Jankowski, 1997; Ludvigsen *et al.*, 1998). Detailed monitoring of leachate plumes is particularly important as the boundaries of the plume are often defined by one or more zones of microbially mediated oxidation/reduction processes, which have a tendency to fluctuate in time and space as the contaminant plume interacts with oxygen-rich regional groundwater and aquifer material (Baedecker and Back, 1979a,b; Christensen *et al.*, 1994; Bjerg *et al.*, 1995; Jankowski, 1997; Ludvigsen *et al.*, 1998). This discussion will focus

on several key indicators of plume variability, including pH, Eh, EC, HCO_3^- , and redox couples ($\text{NO}_3^-/\text{NH}_4^+$, $\text{SO}_4^{2-}/\text{S}^{2-}$, $\text{Fe}^{3+}/\text{Fe}^{2+}$), to demonstrate the dynamic nature of the leachate plume over small increments of time and to underscore the importance of detailed monitoring to accurately characterize the plume.

Environmental Setting

The Astrolabe Park landfill is located south of Sydney in the northern portion of the Botany Basin, an erosional depression in the Triassic Hawkesbury Sandstone bedrock (Albani, 1981; Griffin, 1963; Rickwood, 1998). Over the past approximately 200,000 years this depression was filled with Quaternary sedimentary deposits that comprise the Botany Sands aquifer (Figure 1).



Figure 1. Site location map and extent of the Botany Sands aquifer.

The Quaternary sediments that comprise the Botany Sands aquifer represent a sequence that ranges from marine to terrestrial deposits. From most recent to oldest, this sedimentary sequence has been divided into 4 units (Albani, 1981): recent marine deposits (Unit 1); aeolian deposits (Unit 2); fresh to brackish lagoon deposits (Unit 3); and marine/estuarine deposits (Unit 4). In the vicinity of the Astrolabe Park landfill, the aeolian deposits of Unit 2 comprise the most significant water bearing formation.

Site Description

The Astrolabe Park has been established over an abandoned landfill (Figure 2). The site is underlain by aeolian and alluvial deposits comprising the

upper part of the Quaternary Botany Sands Aquifer. The upper portion of the aquifer in this area is unconfined, although the occurrence of locally continuous silty, clayey and peaty lenses creates semi-confined conditions in the lower portion of the aquifer. The landfill is located up hydraulic gradient from a series of groundwater table ponds known as the Lachlan Ponds.



Figure 2. Location of multi-level piezometers and extent of the landfilling activities at Astrolabe Park.

Mining of heavy mineral sands occurred at this site using a floating suction dredge between 1964 and 1973 and this is believed to have extended to a maximum depth of 17m (Hitchcock, 1991). After the dredging, solid and liquid wastes of mostly domestic origin were deposited in the shallow water. The total volume of waste deposited was approximately $2 \times 10^5 \text{ m}^3$ (Hitchcock, 1991). As the lateral extent of the landfill was estimated to be 4.7 ha, based upon examination of a series of aerial photographs, the average thickness of landfill was approximately 4.35 m.

Methodology and Error Analysis

The data for this investigation was collected from a network of five multi-level piezometers: four of these were installed in a cross-sectional transect along the shore of Lachlan Pond No.5, approximately 100 m downgradient from the landfill; the fifth was installed closer to the landfill in conjunction with a previous investigation (Figure 2). Sample points were positioned at 0.5m intervals across the vertical section thought to define the maximum vertical extent of the leachate plume, and then at 1.0m intervals for an additional 10m below the plume in the fresh regional groundwater.

Sample points were purged using a peristaltic pump, and samples were collected once the general parameters had stabilised. The unstable chemistry was analysed in the field using a titration method for HCO_3^- and CO_2 determination, and analyte fixation and colorimetric analysis was used for determination of Fe^{2+} , S^{2-} , NO_3^- , NH_4^+ and PO_4^{3-} concentrations. One 60ml sample was filtered and preserved for ICP analysis of major and minor elements, and one 120ml sample was filtered and used for Cl^- determination using a titration method.

The accuracy and precision of the ICP analyses was assessed by analysis of spiked samples, deionized water blanks, and duplicate sample analyses as per standard laboratory protocols. All spiked samples, blanks and duplicates were within 5% relative percent difference (RPD), calculated as the difference in known and analysed concentrations (or first and second analysis results for duplicates) divided by the sum of these values, multiplied by 100.

Spectrophotometer and titration analyses were calibrated against laboratory prepared standards, with results from both methods falling within 5% RPD for each sampling event as calculated above. Duplicate spectrophotometer analyses were performed in the field following suspect results; the instability of the analytes following sample collection, as well as budget constraints, precluded direct, relevant comparison of the spectrophotometer results with other laboratory analysis techniques. However, the strong quality control results, coupled with the agreement in data trends observed across the three different analysis methods between subsequent sampling events, suggests that these analysis methods accurately identified spatial and temporal trends in chemical concentrations as they occurred within the plume.

Landfill Processes: A Brief Overview

A sequence of physical, chemical and microbial processes occurs in landfills when water, either infiltrating precipitation or an elevated groundwater table, comes in contact with the buried waste. The end result is a liquid that contains a high concentration of both simple organic carbon compounds and dissolved inorganic species known as leachate. The relative abundances of dissolved organic and inorganic species in the leachate depends on several factors including the composition and age of the landfilled waste, and the total flux and residence time of water in contact with the waste, but in general the pollutants present in landfill leachate fall into four

broad categories (Christensen et al., 1994): dissolved organic matter, inorganic species, heavy metals, and xenobiotic organic compounds.

Current waste landfilling technology incorporates a combination of a very low permeability clay or geotextile barrier installed on the floor of the landfill prior to waste disposal activities, and often a leachate drainage and collection system in order to protect the groundwater beneath the landfill from contamination. However, many older landfills were not designed with these protective measures. In these circumstances, the leachate generated in the landfill readily migrates beneath the landfill to the water table, creating a significant impact to water quality.

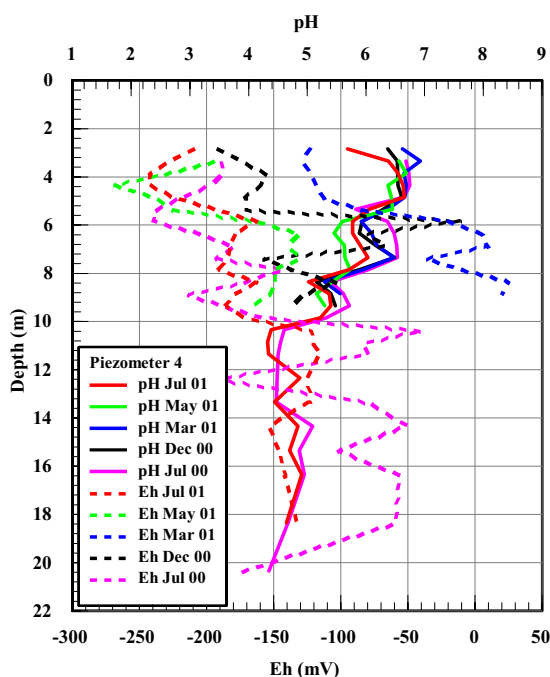
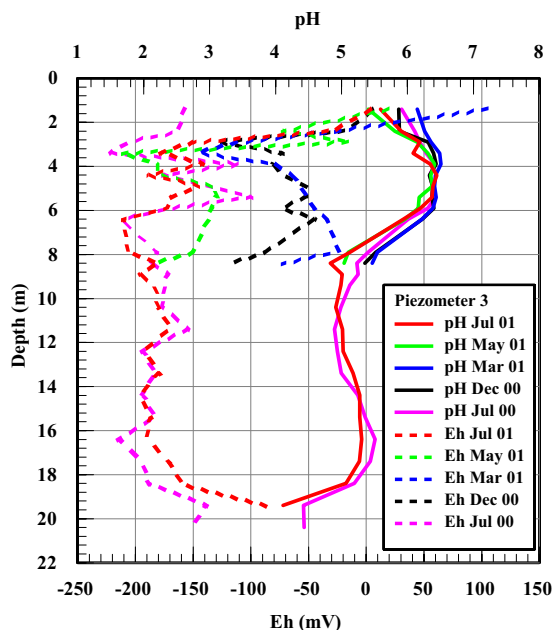
Leachate plumes entering the groundwater system often demonstrate a high degree of variation with respect to contaminant concentration and distribution. Heterogenous groundwater flow patterns, coupled with complex chemical processes occurring within the leachate plume create a state of dynamic solute flux through the aquifer. Two multi-level piezometers from the Astrolabe Park landfill site, piezometers 3 and 4, are used to provide evidence of this variation in time and space.

pH and Eh Measurements

The pH values for leachate plumes generally fall within a narrow range (between 6.0 and 7.0), as previously reported by several authors (Baedecker & Back, 1979a,b; Ehrig, 1983; Nicholson et al., 1983). This pH buffering can be the result of several processes, including production of CO_2 from redox reactions, the dissolution and precipitation of carbonate minerals, reduction of electron acceptors (i.e. SO_4^{2-} , $\text{Fe}(\text{OH})_3$), the reduction of NO_3^- to NH_4^+ (providing a large sink for H^+ ions), and the presence of organic acids (Baedecker and Back, 1979a,b; Kehew and Passero, 1990).

The leachate plume from Astrolabe Park landfill is characterised by high HCO_3^- , NH_4^+ , and $\text{CO}_{2(\text{aq})}$ concentrations (Jankowski and Acworth, 1997). In the presence of these elements, the pH values associated with the leachate plume will remain within a relatively narrow range regardless of the H^+ concentration of the regional groundwater. A simplified representation of the degradation of organic matter at Astrolabe Park landfill can be summarized as follows:





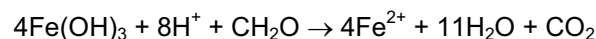
Figures 3a,b. Depth series of Eh and pH values for piezometers 3 and 4, over five sample rounds.

However, because the conditions in the landfill are anoxic, the oxygen in the above equation would have to come from the reduction of any number of electron acceptors, such as the following (note both the generation and consumption of H^+ in the following buffering reactions):

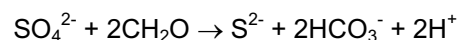
Nitrate reduction:



Fe reduction:



Sulfate reduction:



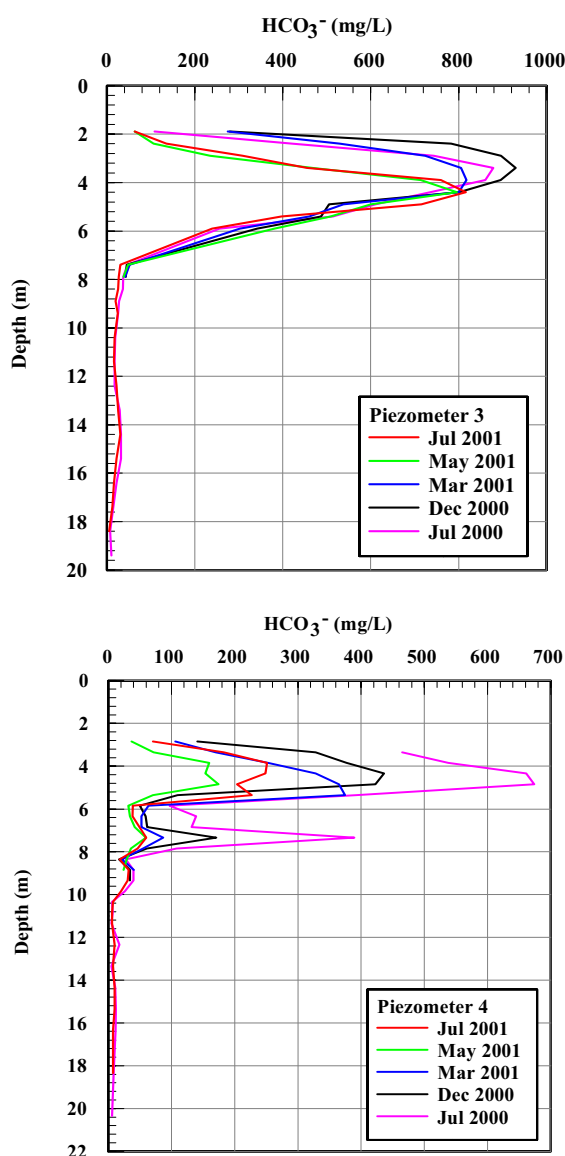
Figures 3a and 3b show pH values versus depth for two piezometers over five sampling events. The pH values corresponding to the plume remain relatively constant between 6.0-7.0, dropping to approximately 5.0 in the fresh regional groundwater. Then low pH of these natural waters is a result of the slight acidity of rainwater recharging the aquifer (Acworth and Jankowski, 1993). The relative paucity of calcareous shell material in the largely aeolian aquifer matrix precludes pH buffering of the acidic, regional groundwater.

In contrast to the well-buffered pH system in the aquifer, the Eh values show a greater degree of variation within the plume and especially between successive sampling events. This may be an indication that processes occurring within the groundwater system are affecting the overall redox potential of the system. Potential processes might include mixing of oxygen-enriched precipitation in the upper aquifer following large storm events, and the arrival of concentrated contaminant 'plugs' mobilised from the landfill during past rainfall events. A significant flux in the concentration of either electron acceptors or donors in the groundwater system might have the capacity to alter the dominant redox processes occurring in the plume. Also, the presence of organic lenses in the aquifer may act as substrates for localised redox processes occurring in discrete intervals that differ from those occurring in the bulk of the plume.

It should be noted that while field measurement of Eh values may give a relative indication of changing redox conditions within a groundwater system, they are not suitable for the determination of the specific redox processes occurring within the system (Baedecker and Cozzarelli, 1992).

Electrical Conductivity and Bicarbonate

Two good indicators of the position and extent of leachate impact to groundwater are the EC value and the HCO_3^- concentration. Within the leachate plume, both show a marked contrast to the respective values for the regional groundwater in the Botany Sands aquifer.



Figures 4a,b. Depth series of HCO_3^- concentrations for piezometers 3 and 4, over five sample rounds.

Figures 4a and 4b are vertical plots of HCO_3^- concentration versus depth for piezometers 3 and 4 at Astrolabe Park landfill. The elevated HCO_3^- values in the upper portion of the aquifer are a clear indication of the location of the centre of the leachate plume. Note the presence of a secondary, less concentrated plume beneath the primary plume in piezometer 4. This is most likely the result of local heterogeneities in the aquifer affecting groundwater flow patterns.

The high concentration of HCO_3^- in the leachate plume is due to the excess of CO_2 , which is formed in abundance as a by-product of the oxidation of organic matter present in landfills. In the typical pH range of a leachate plume (6.0 to 7.0; Baedeker and Back, 1979a; Ehrig, 1983;

Nicholson et al., 1983) this CO_2 is converted to HCO_3^- . As a result of the anaerobic decomposition of organic matter in the landfill and continuing in the plume, HCO_3^- concentrations easily reach several hundred milligrams per liter. This is in stark contrast to the regional groundwater in the Botany Sands aquifer, in which the HCO_3^- contribution comes from either CO_2 generated in the soil zone and transported to the water table with infiltrating precipitation, or dissolution of calcareous shell material in the aquifer matrix (Jankowski and Acworth, 1997). The average HCO_3^- concentration range for the regional groundwater is between 20-50 mg/L (Jankowski and Yu, 1998).

Electrical conductivity values also act as a good indicator of the position and relative concentration of the leachate plume. EC values are directly proportional to the concentration of ionic species present in the fluid. As a result of dissolution of waste material in the landfill, the leachate plume contains a high concentration of dissolved ionic species and thus has much higher EC values than the regional groundwater. EC values in the leachate plume were observed as high as $1800 \mu\text{S}/\text{cm}$, while regional values average around $250 \mu\text{S}/\text{cm}$ (Jankowski and Yu, 1998).

As the dominant ion present in the leachate-impacted groundwater, HCO_3^- concentrations have a significant influence over the EC value. Figure 5 shows a plot of EC values versus HCO_3^- concentrations for piezometer 4 over five sample rounds. The resulting equation for the relationship between EC and HCO_3^- for piezometer 4 is:

$$\text{HCO}_3^- = 0.7828\text{EC} - 131.96$$

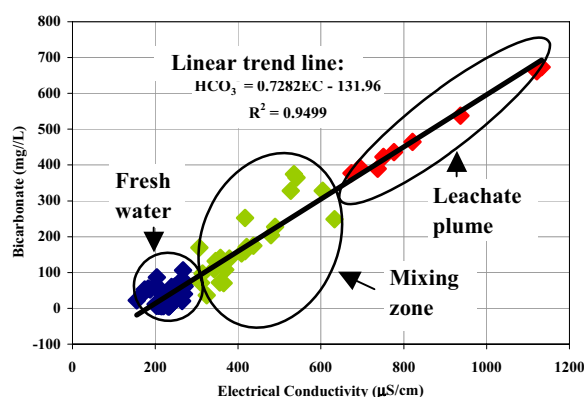
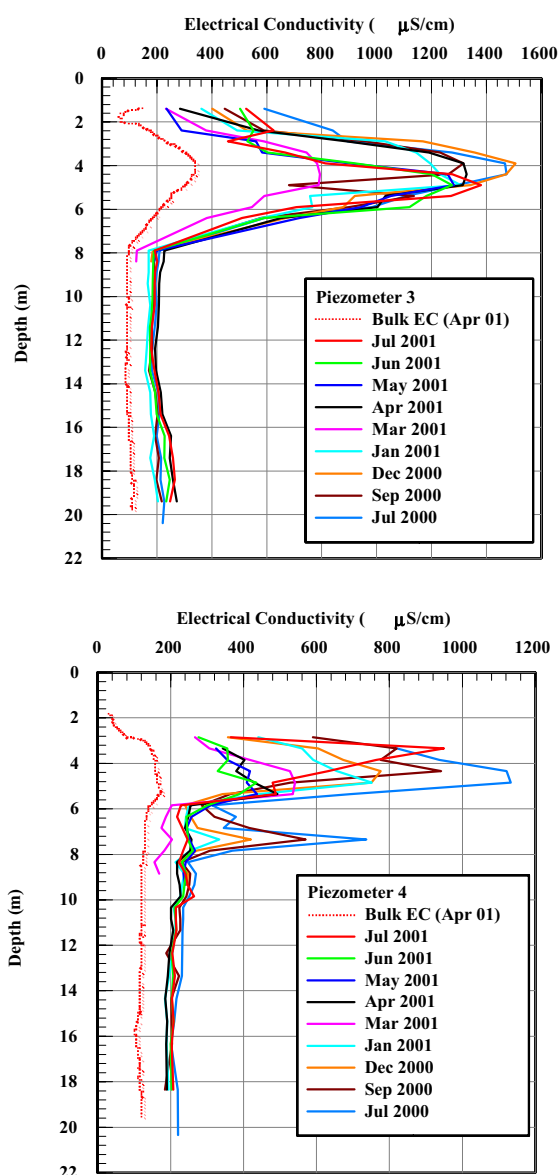


Figure 5. EC versus HCO_3^- for piezometer 4 over five sample rounds; identification of water type and calculation of linear relationship included.

There is an excellent linear correlation between the two values, with an R^2 value of 0.9499. Note that although the above equation works well for

the leachate plume and mixing zone waters, it needs to be recalculated to be applicable for water with an EC value less than approximately $250\mu\text{S/cm}$, reflecting the relatively diminished influence of HCO_3^- on EC values in the regional groundwater.

Figures 6a and 6b are vertical plots of EC versus depth for two piezometers at Astrolabe Park landfill. Fluid EC values were measured over nine rounds, and bulk EC was measured in April 2001 using a Geonics EM-39 geophysical logging system equipped with a bulk electrical conductivity sonde. Note both the evidence for a secondary plume beneath the primary plume, and the similarity in shape of the HCO_3^- and EC plots.



Figures 6a,b. Depth series of fluid and bulk electrical conductivity values for piezometers 3 and 4, over nine sample rounds.

Note that the absence of the 'secondary' plume in the bulk EC profile for piezometer 4 is in agreement with the groundwater EC profile measured at the same time in which the 'secondary' plume was not detected.

Vertical Redox Zonation

As highly reduced leachate enters the fresh, oxygen-rich regional groundwater system, a complex system of redox zones are created (Baedecker and Back, 1979a,b; Bjerg et al., 1995; Jankowski, 1997; Lyngkilde & Christensen, 1992). The extent to which various redox zones are formed depends on the relative abundance in the aquifer of both dissolved organic carbon and electron acceptor species. Theoretically, the thermodynamic sequence of redox zonation along a flow path from the contaminant source or vertically through a plume would follow this simplified progression: methanogenic $\rightarrow$ sulfate reduction $\rightarrow$ iron reduction $\rightarrow$ manganese reduction $\rightarrow$ denitrification $\rightarrow$ aerobic. The presence of each of these zones depends on the presence of the solid or dissolved electron acceptors in the system (a groundwater system containing negligible SO_4^{2-} will not develop a SO_4^{2-} reducing zone). This zonation can easily be complicated by localized sources of electron acceptors in the aquifer. An example would be the occurrence of pyrite-bearing lenses that are oxidized, mobilizing SO_4^{2-} in an otherwise sulfur-free system. If a reducing leachate plume then migrates into this area, it may create a local zone of sulfate reduction in a portion of the plume that is dominantly iron reducing.

Figures 7, 8 and 9 are plots of redox couples, or oxidized and reduced species of the same element ($\text{NO}_3^-/\text{NH}_4^+$, $\text{SO}_4^{2-}/\text{S}^{2-}$, $\text{Fe}^{3+}/\text{Fe}^{2+}$) versus depth over five successive sample rounds for piezometer 4.

N-species

There are a couple basic trends evident in these plots. Spikes in the NH_4^+ concentration in the upper part of the aquifer, from approximately 3 to 6 m below ground surface, indicate the presence of a reducing plume environment. This is matched by a reduction in the NO_3^- concentration. There is also similar evidence of a smaller secondary plume from 7 to 8 m below ground surface, although the NH_4^+ concentration is lower and it disappears completely during the May 2001 sample round. There is also a gradual decrease in NH_4^+ concentration between successive sampling rounds. In May 2001 the NH_4^+ concentration never exceeds the NO_3^- concentration, and only

rebounds slightly in July 2001. This piezometer is located close to the lateral boundary of the plume, and these trends may indicate a seasonal decrease in the plume extent or a change in groundwater flow direction, resulting in the encroachment of oxidized regional water. The oxidation of NH_4^+ might explain the relative increase in NO_3^- during the same interval.

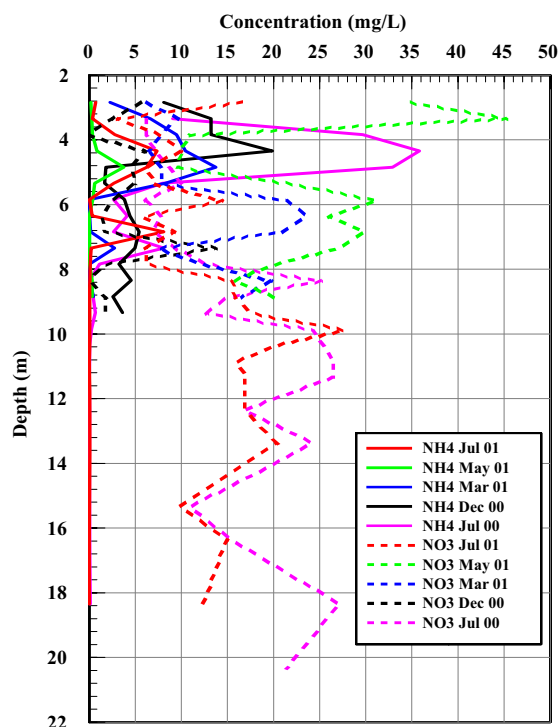


Figure 7. Depth series of NH_4^+ and NO_3^- concentrations over five sample rounds for piezometer 4.

S-species

The plots of sulfur species show the greatest variability in space and time (Figure 7). Between sampling rounds, the dominant sulfate reduction zone appears to switch between the bottom and the top of the plume. There are spikes in the S^{2-} concentrations at the very top of the water table during July 2000, December 2000, and July 2001, which are replaced by spikes between 7 to 8 meters in December 2000 and 6 to 7 meters in May 2001. There does not appear to be a strong correlation between S^{2-} and SO_4^{2-} concentrations with depth. SO_4^{2-} concentrations do appear to decrease in the zones where NH_4^+ increases, and S^{2-} concentrations show a reasonable correlation to Fe^{2+} concentrations. The relatively low concentration of S^{2-} in the system, as well as its spatial correlation to Fe^{2+} , may suggest that the two species are precipitating out of solution as $\text{FeS}_{(\text{ppt})}$, with the S^{2-} being almost completely depleted as the limiting reactant. The

concentration of S^{2-} also appears to decrease by an order of magnitude in May 2001, suggesting that the plume has achieved a higher redox potential during this event.

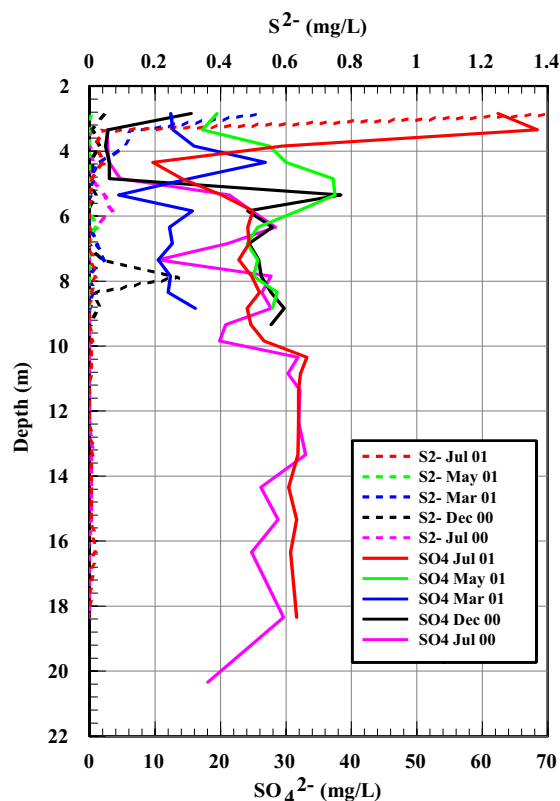


Figure 8. Depth series of S^{2-} and SO_4^{2-} concentrations over five sample rounds for piezometer 4.

Fe-species

Spikes in the concentration of Fe^{2+} in the upper portion of the aquifer coincide almost exactly with those from the N plots, adding further evidence of the plume location (Figure 8). Unusually, the Fe^{3+} distribution seems to mirror the Fe^{2+} distribution. This may be due to Fe^{2+} in solution becoming oxidized and entrained on colloidal material that passes through the filter membrane during sample collection, creating anomalous Fe^{3+} spikes coincident with the Fe^{2+} spikes. Below 8 m, there is no evidence of either Fe^{2+} or Fe^{3+} in the groundwater system. Once again, there is a gradual decline in the concentration of Fe^{2+} from July 2000 to May 2001, with a rebound observed in July 2001.

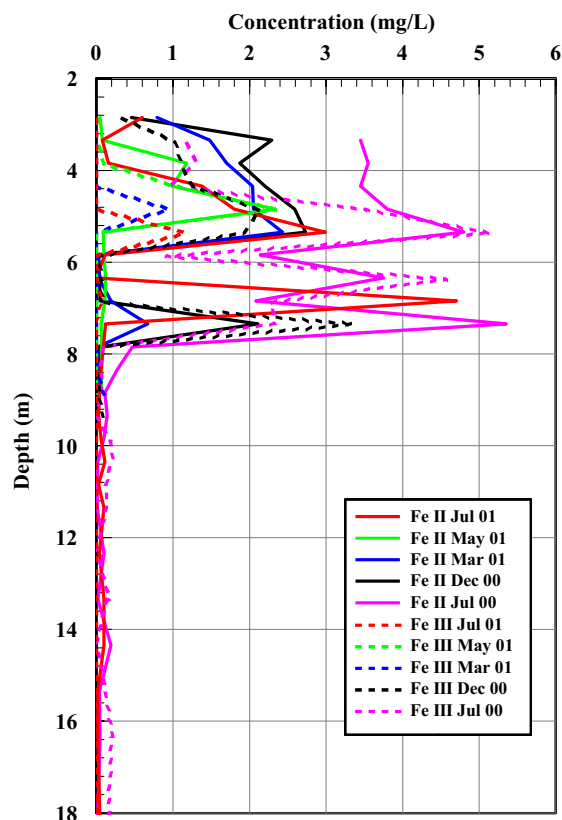


Figure 9. Depth series of Fe^{2+} and Fe^{3+} concentrations over five sample rounds for piezometer 4.

Conclusion

Landfilling of waste is the primary method of waste disposal in many industrialized countries. As waste decomposes inside a landfill, a sequence of physical, chemical and microbial reactions occurs which result in the formation of leachate, a water-based pollutant containing high concentrations of dissolved organic carbon and inorganic species, and to a lesser extent heavy metals and foreign organic contaminants (Christensen et al., 1994).

The impact to groundwater quality by leachate poses a significant threat to water resources in many urban settings. In order to effectively monitor the migration of leachate plumes, it is important to understand the processes affecting the distribution of contaminants in space and time.

The decomposition of waste in landfills results in the generation of CO_2 and HCO_3^- . The high concentrations of these species in groundwater buffer the pH range of the leachate plume, which has important implications for the attenuation of some contaminants. The consumption of oxidized

species in this process creates a reducing environment, lowering the Eh within the plume.

Leachate that enters an oxidized aquifer will tend to form a sequence of redox zones based on the relative abundance of oxygen or oxidized species present and the interaction of the plume with aquifer material. These zones define the boundary of the plume, but can fluctuate significantly over relatively small increments of time and may not conform to standard thermodynamic models. It is therefore critical to undertake detailed monitoring of the leachate plume in order to assess the variation in contaminant concentration and spatial distribution to be expected over time.

That said, while this type of investigation has merit in an academic setting, it is often beyond the scope of many public and private sector investigations. It is always advisable to conduct vertical, discrete-depth monitoring of landfill leachate plumes, but the total number of piezometers, sample points and analyses can be adjusted to conform to logistical and budget constraints. In many instances it may be acceptable to sample for the full suite of analytes once every year to 18 months to monitor long-term trends, relying on more inexpensive monitoring indicators such as EC, chloride, or other proprietary analytes to determine plume variations in between. Simple quarterly measurements of EC in vertical profiles, an extremely easy and inexpensive endeavour, will often greatly increase the knowledge of the plume dynamics for those with limited resources.

References

- Acworth, R.I. & Jankowski, J. (1993). "Hydrogeochemical zonation of groundwater in the Botany Sands aquifer", *AGSO Journal of Australian Geology and Geophysics*, 14, 193-199.
- Albani, A.D. (1981). "Sedimentary environments and Pleistocene chronology of the Botany Basin, NSW", *Geo-Marine Letters* 1, 163-167.
- Baedecker, M.J. & Back, W. (1979a). "Hydrogeological processes and chemical reactions at a landfill", *Ground Water*, 17, 5, 429-437.
- Baedecker, M.J. & Back, W. (1979b). "Modern marine sediments as a natural analog to the chemically stressed environments of a landfill", *Journal of Hydrology*, 43, 393-414.
- Baedecker, M.J. & Cozzarelli, I.M. (1992). "The determination and fate of unstable constituents of contaminated groundwater", In S. Lesage and R.E. Jackson (eds), *Groundwater contamination and*

analysis at hazardous waste sites, Marcel Dekker, Inc., New York, 425-461.

Bjerg, P.L., Rugge, K., Pedersen, J.K., and Christensen, T.H. (1995). "Distribution of redox-sensitive groundwater quality parameters downgradient of a landfill", *Environmental Science and Technology*, 29, 5, 1387-1394.

Christensen, T.H., Kjeldsen, P., Albrechtsen, H.-J., Heron, G., Nielsen, P.H., Bjerg, P.L. and Holm, P.E. (1994). Attenuation of landfill leachate pollutants in aquifers. *Critical Reviews in Environmental Science and Technology*, 24, 119-202.

Ehrig, H.-J. (1982). "Quality and quantity of sanitary landfill leachate", *Waste Management & Research*, 1, 53-68.

Griffin, R.J. (1963). *The Botany Basin*. Bulletin, Geological Survey of New South Wales, 18, 110pp.

Hitchcock, P.W. (1991). *The hydrogeology, hydrochemistry and geophysical conditions of Astrolabe Park and environs*. M.App.Sc. Thesis, CGMH, University of New South Wales, 325pp (unpublished).

Jankowski, J. (1997). "Generation and migration of chemical constituents in a leachate plume", In Greg McNally (ed) *Collected Case Studies in Engineering Geology, Hydrogeology, and Environmental Geology, Third Series*, Geological Society of Australia, 181-205.

Jankowski, J. & Acworth, R.I. (1997). "Development of a contaminant plume from a municipal landfill: Redox reactions and plume variability", In Chilton, J. et al. (eds) *Proceedings of the XXVII International Association of Hydrogeologists Congress on Groundwater in the Urban Environment*, Nottingham UK, September 1997. Balkema, Rotterdam, Vol.1, 439-444.

Jankowski, J. & Knight, M.J. (1991). *Astrolabe Park landfill leachate and its impact on the Botany Wetlands*. Technical report by Unisearch Ltd for Sydney Water Board (unpublished).

Jankowski, J. & Yu, X.W. (1998). "Hydrogeochemistry of the Botany Sands aquifer", In McNally, G.H. and Jankowski, J. (ed) *Environmental Geology of the Botany Basin, Fourth Series*, Geological Society of Australia, 37-57.

Jeffrey, R.P. (1985). "Botany Basin deposits", In Pells P.J.N. (ed) *Engineering geology of the Sydney region*. Balkema, Rotterdam, 81-97.

Johnson, B.D., Albani, A.D., Rickwood, P.C. & Tayton, J.W. (1977). "The bedrock topography of the Botany Basin, NSW", *Journal of the Geological Society of Australia* 24, 403-408.

Kehew, A.E. & Passero, R.N. (1990). "pH and redox buffering mechanisms in a glacial drift aquifer contaminated by landfill leachate", *Ground Water*, 28, 5, 728-737.

Ludvigsen, L., Albrechtsen, H.-J., Heron, G., Bjerg, P.L., Christensen, T.H. (1998). "Anaerobic microbial redox processes in a landfill leachate contaminated aquifer", *Journal of Contaminant Hydrology*, 33, 273-291.

Nicholson, R.V., Cherry, J.A. & Reardon, E.J. (1983). "Migration of contaminants in groundwater at a landfill: A case study. 6. Hydrogeochemistry. *Journal of Hydrology*, 63, 131-176.

Rickwood, P.C., (1998). "The Botany Basin: a convenient definition", In McNally, G.H. and Jankowski, J. (eds) *Environmental Geology of the Botany Basin, Fourth Series*, Geological Society of Australia, 173-178.

Yu, X. (1994). Study of physical and chemical properties of groundwater and surface water in the northern part of the Botany Basin, Sydney. Ph.D Thesis, University of New South Wales (unpublished).

Authors Biographies

Lange B Jorstad completed a BSc in geological sciences from the University of California, Santa Barbara in 1995. He was employed from 1997 to 2000 as a field technician and then staff geologist in the Environmental Science and Engineering department of Geomatrix Consultants, Inc., an environmental consulting firm based in San Francisco, California. His work there primarily focused on soil and groundwater contamination investigations. He is currently a PhD research student with the University of New South Wales in Sydney, Australia, researching the temporal and spatial variability of contaminant concentration and distribution in a landfill leachate plume.

Postal Address: Lange B Jorstad, University of New South Wales, School of Geology, Sydney, NSW 2052, Australia

E-mail: lange@student.unsw.edu.au

Jerzy Jankowski has a BSc in hydrology, an MSc in hydrology/hydrochemistry and a Ph.D. in hydrogeology/hydrogeochemistry, all from the University of Wroclaw, Poland. He has more than 20 years research experience in countries including Poland, Czechoslovakia, Bulgaria and Australia. He is author/co-author of over 100 research papers, chapters in books and conference proceedings. Areas of special research expertise include: water-rock interaction in different rock type environments, contamination of surface waters and groundwaters, chemical evolution of groundwaters in the arid zone, groundwater-sea water and groundwater-surface water interactions, dryland salinisation processes and hydrochemical and isotopic studies including environmental isotopes and hydrogeochemical modelling.

Postal Address: Dr Jerzy Jankowski, University of New South Wales, School of Geology, Sydney, NSW 2052, Australia

A review of environmental management solutions for an old, unlined landfill in Sydney, Australia

Lange Jorstad and Jerzy Jankowski

UNSW Groundwater Group, School of Biological, Earth & Environmental Sciences
University of New South Wales, Sydney, NSW 2052, Australia

Abstract: The decommissioned Astrolabe Park landfill, located in a suburb of Sydney, had an intermittent history of waste disposal dating back to 1938. The lack of an engineered barrier system to isolate the waste from the aquifer environment has resulted in the development of a leachate plume beneath and downgradient from the site. Starting shortly after waste disposal activities commenced, the leachate plume has been migrating from the landfill towards a shallow groundwater-fed pond (Lachlan Pond No.5), into which the contaminated groundwater discharges. Beginning in 1990 a number of groundwater investigations have been undertaken at the site to characterise the extent of contamination associated with the landfill and the flux of contaminants between the landfill and the pond. The presence of leachate contamination requires the development of an environmental management strategy, and the sensitivity of the aquatic ecosystem of Lachlan Pond No.5 requires the adoption of a stringent level of protection. Traditional pump-and-treat systems are proven and reliable, but come at a high cost over the long term. Permeable reactive barriers and monitored natural attenuation are innovative new approaches and more cost effective, but both lack a well-documented performance record with respect to leachate plumes. Dredging along the shore of the pond would remove the accumulated silt and decaying organic material, facilitating the dilution of the plume in the pond and the removal of accumulated contaminants in the sediments. However, a risk assessment would be required to determine the impact to the aquatic ecosystem and pond water quality from the infiltrating leachate plume. The degree of risk minimisation will ultimately strike a balance between the funding available to the site owners and the regulatory requirements imposed.

Keywords: Landfill, groundwater, leachate plume, aquatic ecosystem, remediation

1. INTRODUCTION

The Botany Sands aquifer constitutes a large reservoir of generally good quality water, which, in its pristine state, is suitable for residential, commercial and industrial use. Over the years the water quality in the Botany Sands aquifer has been progressively degraded due to the stresses imposed by urbanisation. Amongst other pollution sources, a number of old, unlined landfills such as Astrolabe Park have been identified as major point-source contributors to the degradation of water quality in the Botany Sands aquifer. While it is in the interest of stakeholders and regulatory officials to develop a management strategy to address this issue, there are very few viable options available within the technical, financial and political constraints inherent in the decision making process. Management options must take in to consideration the land use in the vicinity of the landfill, as well as the sensitive receptors that are affected by contaminants from the landfill, in this

case the aquatic ecosystem of Lachlan Pond No.5.

1.1 Landfills and leachate

The landfilling of waste is an integral component of solid waste management plans in countries around the world [Arneeth et al., 1989; USEPA, 1990; Qasim and Chiang, 1994]. Prior to the onset of stringent environmental legislation concerning the siting and operation of landfills in the mid-1960's, it was common practice to dispose of waste without employing engineered measures to isolate the waste from the surrounding environment. This was a particularly common scenario for sand mines and quarries, where backfilling of the mining pits with waste was considered to have the additional benefit of reconstructing the landscape.

The most significant environmental threat posed by old, unlined landfills is the uncontrolled release of a highly contaminated liquid called

leachate. Leachate can be loosely defined as a water-based solution of pollutants leached from the waste [Christensen et al., 2001]. Leachate is generated through the interaction of waste material in the landfill with infiltrating water, either as rainfall infiltration or a shallow water table that comes in direct contact with the waste. The composition of landfill leachate is highly dependent on factors such as: waste composition in the landfill, landfill age and associated waste degradation phase, the volume and composition of rainwater and/or groundwater infiltrating into the waste, and the residence time of moisture within the waste layers [Andreottola and Cannas, 1992]. Leachate has been reported to contain a variety of contaminants, including nutrients, heavy metals, synthetic organic chemicals, and pathogens [Christensen et al., 2001].

The introduction of concentrated, strongly reduced leachate into an otherwise pristine aquifer system has a profound impact on the chemistry of both the groundwater and the aquifer mineralogy. It is analogous to a redox titration, in which the continuous release of leachate progressively depletes the aquifer of electron acceptors along its flow path. A strongly reduced environment is established in the aquifer downgradient from the landfill, which minimizes the capacity of the aquifer to attenuate the leachate contaminants.

2 ASTROLABE PARK LANDFILL

2.1 Site Description

The Astrolabe Park landfill is an old, unlined landfill located approximately 7 km south of the Sydney CBD. The current research site consists of the decommissioned landfill, a portion of the Eastlakes Golf Course adjacent to Astrolabe Park, and a portion of the eastern shore of Lachlan Pond No.5 [Figure 1]. A network of piezometers has been installed at the site during various research projects in order to study groundwater flow and the nature of contaminant transport between the landfill and the pond

2.2 Environmental Setting

The Astrolabe Park landfill is situated within the Botany Sands aquifer, the primary source of groundwater within the Botany Basin. In the vicinity of the research site, the Botany Sands aquifer is comprised primarily of highly

permeable Quaternary aeolian quartz sand to a thickness of approximately 30m. Locally occurring lenses of silty, clayey and peaty material have resulted in the aquifer at the site being effectively separated into a shallow, unconfined portion and a deeper semi-confined portion.

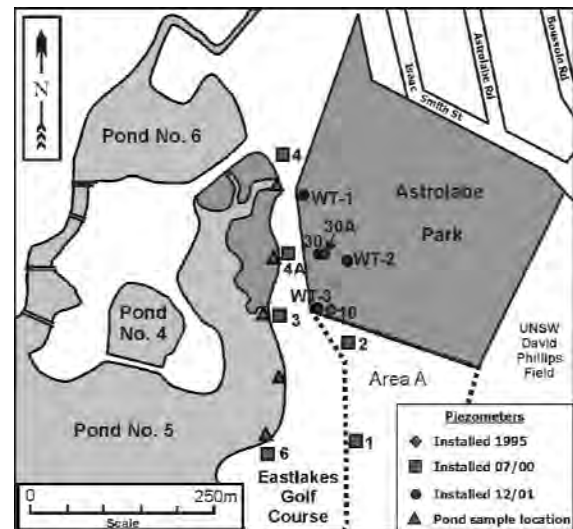


Figure 1. Astrolabe Park and environs.

Astrolabe Park is located less than 100m up hydraulic gradient from Lachlan Pond No.5. The Lachlan Ponds are a chain of interconnected ponds that act as the primary surface drainage feature for Sydney's southern suburbs, receiving a mixture of urban stormwater runoff and groundwater discharge before ultimately emptying into Botany Bay. The Lachlan Ponds chain constitutes a sensitive aquatic ecosystem that has been directly affected by the discharge of landfill leachate, which potentially dates back to the commencement of waste disposal activities at Astrolabe Park in the late 1930's.

2.3 History of landfilling

Astrolabe Park has an intermittent history of landfilling, beginning with the initial use of a 2-acre (0.8 hectare) portion of the southwestern corner of the site for municipal waste disposal from 1938 to 1945. The remainder of Astrolabe Park was used for waste disposal in conjunction with sand mining activities between 1965 and 1975. Heavy mineral sands were mined at the site to a maximum depth of 17m, and the pits were subsequently backfilled with municipal waste. The total volume of waste deposited at Astrolabe Park was estimated from council

records to be approximately 200,000 m³ [Hitchcock, 1991]. Waste disposal at the site was terminated in 1975, and the decommissioning process was completed in 1978 by covering the waste layers with approximately 10 cm of fine sand and converting Astrolabe Park to sports fields.

2.4 History of site investigations

The extent of impact from leachate infiltrating into the groundwater system was the subject of several commercial and research investigations beginning in 1991. The initial investigation conducted by Hitchcock [1991] was instrumental in defining the stratigraphy and some general hydrogeological parameters for the site, as well as identifying the presence of leachate-impacted groundwater between the landfill and Lachlan Pond No.5, but the monitoring network was insufficient to delineate the full vertical and lateral extent of the plume. Subsequent investigations by Jankowski and Knight [1991a,b], Kelly [1994], Smolski [1994], Jankowski and Acworth [1997] and Jorstad and Jankowski [2002a,b] included a considerable improvement in the monitoring network, and also employed a number of geophysical investigation techniques that resulted in a significantly enhanced understanding of the composition and distribution of the leachate plume, as well as the physical, chemical and biological processes affecting the plume chemistry.

An exercise conducted in July 2002 employed a combination of hydrogeological and geophysical investigation techniques to demonstrate that the eastern shore of Lachlan Pond No.5 downgradient from the landfill is a groundwater discharge zone, with an upward vertical hydraulic gradient approximately 2-3 times greater than the horizontal hydraulic gradient [unpublished]. The data suggest that the total volume of the shallow, leachate-impacted groundwater is discharging into the pond.

2.5 Extent of groundwater contamination

Environmental investigations at the Astrolabe Park landfill indicate that the uncontrolled release of leachate has resulted in a significant impact to groundwater quality between the landfill and the pond. Waste deposition at Astrolabe Park occurred without the use of an impermeable barrier lining the waste pits, resulting in a release of landfill leachate into the

aquifer that would have commenced shortly after the initial waste deposition. Because sand mining occurred beneath the water table, a certain portion of the waste deposited in the sand pits has been in direct contact with groundwater since waste deposition began.

Table 1. Range and mean for parameters and elements associated with landfill leachate impacted groundwater (all chemical concentrations in mg/L; *n* = 266 samples).

Parameter	Range	Mean
<i>General parameters</i>		
pH	5.1 - 9.1	6.1
Eh	-312.1 - 103.3	-145
Cond	192.3 - 1853	686
O ₂	0 - 8.1	0.6
<i>Inorganic macrocomponents</i>		
Ca	1.82 - 204	51.3
Na	8.44 - 369	42.9
Mg	0.9 - 32.1	11.1
K	2.4 - 39.4	15.1
HCO ₃	31.73 - 1100	356
CO ₃	0 - 54.01	0.31
Cl	14.9 - 318	52.0
SO ₄	0.35 - 155	13.8
H ₂ S	0 - 2.68	0.061
NO ₃	0 - 91.9	10.8
NH ₄	0 - 63.9	17.4
Fe	0 - 70.1	9.5
Mn	0.003 - 2.16	0.276
P	0 - 1.42	0.152
<i>Trace elements</i>		
Al	0.005 - 1.3	0.073
B	0.029 - 1.54	0.401
Cd	0 - 0.002	0.001
Cu	0.002 - 0.035	0.009
Li	0.001 - 0.095	0.03
Ni	0 - 0.002	0
Pb	0 - 0.017	0.005
Sr	0.01 - 1.08	0.25
Zn	0.002 - 0.46	0.022

Waste deposited above the water table also contributes to leachate discharge when the moisture content of the waste exceeds field capacity, typically following significant rainfall

events. Thus, a well-developed leachate plume exists below and downgradient from the landfill. A constant base-level contribution is sustained by groundwater passing through the lower waste layers, and discrete 'slugs' of concentrated leachate are flushed into the aquifer system following major rainfall events. Infiltration of rainwater likely has the dual effect of introducing oxygen to the waste, instigating a brief period of high-intensity aerobic degradation (and associated leachate production) within the landfill.

The ranges and mean values of groundwater parameters and element concentrations associated with the leachate plume are provided in Table 1. These values are calculated from 266 groundwater and pond water samples collected between July 2000 and February 2002, which were statistically determined to have a measurable impact from the leachate. The maximum element concentrations are modest, even for methanogenic phase leachate [Christensen et al., 2001], suggesting that the buried waste has been well flushed over the past three decades. However, many element concentrations are well above background levels and are addressed in section 3.1.

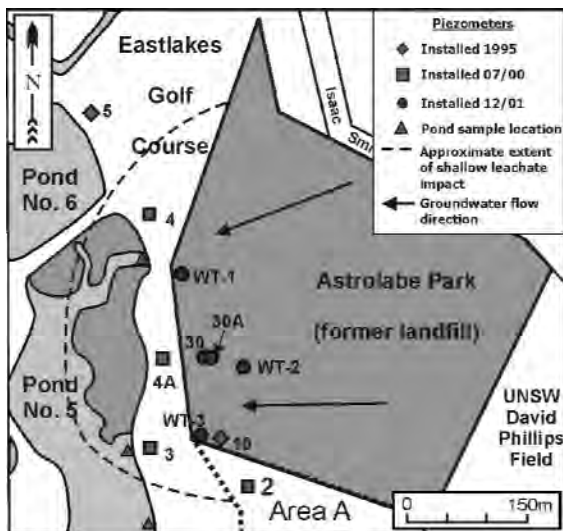


Figure 2. Groundwater flow direction and extent of leachate contamination to shallow groundwater and pond water downgradient from Astrolabe Park.

The approximate lateral extent of leachate impact to the shallow portion of the aquifer is illustrated in Figure 2. Past geophysical and hydrochemical analysis of piezometer 5 indicate

that it is free of impact from the leachate plume. There is evidence of leachate impact south of piezometer 3, but the plume occurs significantly deeper there, and the lack of proliferating vegetation along the shore of the pond suggests that the majority of the leachate plume south of piezometer 3 passes beneath the shallow Lachlan Pond No.5 with a minimal impact to pond water quality. The vegetation mass downgradient from Astrolabe Park is believed to have resulted from nutrient loading from the leachate discharging into the pond. Chemical analysis of pond water close to the vegetation indicates that it is similar in composition to moderately impacted groundwater along the fringes of the leachate plume, while pond samples just south of this zone have the chemical signature of uncontaminated regional groundwater [Jankowski and Knight, 1991a,b].

The vertical extent of the leachate plume downgradient from Astrolabe Park is controlled by the presence of silty/peaty lenses in the sand profile. Detailed hydrogeological investigations along the shore of the pond indicate steep chemical gradients at approximately 7-8 m below ground surface that coincide with fine-grained lenses identified by cone penetrometer tests and borehole geophysical techniques. No variation of the lower boundary of the plume was observed over the 18 months of monitoring, indicating that the leachate plume downgradient from Astrolabe Park is restricted to the upper 7-8 metres of the shallow, unconfined aquifer.

3 ENVIRONMENTAL MANAGEMENT STRATEGY

Environmental investigations at the Astrolabe Park landfill provide ample evidence that the site is acting as a point source of pollutants that is degrading the quality of both the groundwater in the Botany Sands aquifer and the pond water along the eastern shore of Lachlan Pond No. 5. When considering a methodology for the development of an environmental management plan, it is important to clearly define the extent of the impact and the sensitivity of local receptors to the contamination in order to determine the level of protection appropriate for the site [Tam and Byer, 2002]. This information can be used to determine the potential liability for the site owner and the potential exposure pathways for human or ecological receptors [Freeze et al., 1990]. A general guideline to the decision making process is presented in Figure 3.

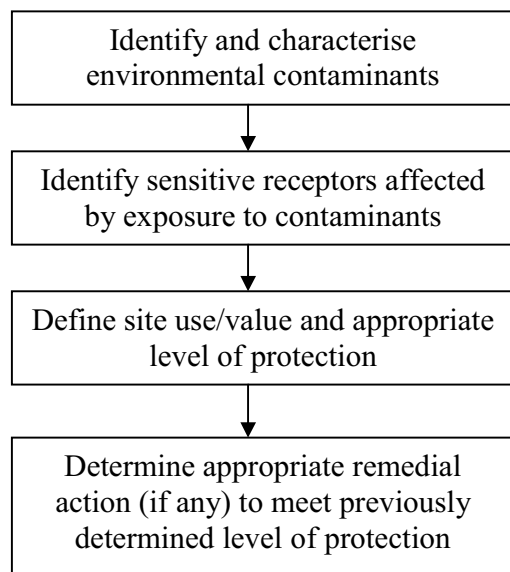


Figure 3. Flow chart depicting steps in a decision analysis method to develop an environmental management plan.

3.1 Contaminants of concern

Ammonium [NH_4^+] is a major concern in landfill leachate, both because it is toxic to aquatic organisms at elevated concentrations and can act as a source of nutrient loading [ANZECC, 2000]. Nutrient loading from NH_4^+ in the leachate plume into Lachlan Pond No.5 has resulted in the proliferation of vegetation along the shoreline downgradient from the landfill. When these plants die, they contribute a significant bulk of organic matter to the pond, causing depletion of oxygen in the pond water during decomposition and subsequent suffocation of the aquatic organisms living in the pond [Jankowski and Knight, 1991a,b]. Further, a high nutrient load is a powerful stimulant for algal blooms in lakes and ponds. Finally, NH_4^+ in landfill leachate stems from the hydrolysis of proteins in the waste material, and the slow kinetics of the reaction mean that NH_4^+ will be a critical long-term component of the leachate [Ehrig, 1988].

A number of toxic metals are also present in the leachate plume. The metals boron, cadmium, chromium, copper, lead and zinc are all present at slightly elevated concentrations, and soil analyses in the vegetation on the shore of Lachlan Pond No.5 indicate that some of these metals are accumulating in the organic sediments (data not shown). While several of

these metals are known to display acute toxicity towards aquatic organisms, the environmental conditions within a leachate plume often transform metals into less toxic or bioavailable forms [ANZECC, 2000]. The long-term accumulation of these metals in pond sediments is of greater concern than the concentrations in groundwater. It is recommended that a more detailed analysis with regards to metals speciation be conducted before basing any management decisions on these results.

3.2 Sensitive receptors

The Lachlan Ponds chain, an aquatic ecosystem that supports a variety of birds and aquatic life, constitutes the most sensitive receptor of contaminants released from the landfill. It also presents a special management issue: it is part of the Botany Wetlands, identified as one of the few remaining relict ecosystems in the urban confines of Sydney [Benson and Howell, 1990]. Relict ecosystems are areas of preserved bushland that remain largely unaltered from their original state prior to European colonisation. The preservation of this relict ecosystem will be a significant factor to take into account when making a decision on the level of environmental protection that is appropriate for the site.

3.3 Property value and designation of protection levels

It is difficult to quantify the value of a wetland environment within the framework of a cost-benefit analysis approach to site remediation, unlike a parcel of developable land that may one day return a tangible, quantifiable economic benefit [Freeze et al., 1990]. In the case of the protection of urban bushland, it can be argued that the land must be protected at all costs.

On the other hand, the current configuration of the Lachlan Ponds is the result of a series of dams and weirs that were constructed in the 1800's as part of a water supply scheme for Sydney. The Botany Wetlands have been subject to a long history of environmental impact from various industrial practices, and currently receive a considerable volume of contaminated urban stormwater runoff each year. Taking this history into consideration, it could be justifiably argued that the Botany Wetlands represents a disturbed environment, which may be held accountable to less stringent regulatory criteria than a pristine wetland ecosystem.

Within the framework of recently revised water quality guidelines, regulatory 'trigger values' are designated at four levels of protection for aquatic ecosystems, ranging from 99% to 80% of species protected [ANZECC, 2000]. The history of environmental impact to the Botany wetlands might prompt the argument that the low-end regulatory criteria are appropriate. However, the fact remains that the Botany Wetlands constitutes a significant ecological resource to Sydney, and in the interest of restoring the environmental health of the ecosystem the application of the 95% protection levels is a fair compromise.

3.4 Selection of remedial action

A variety of engineered remediation methods have been developed specifically to address contaminants in landfill leachate. However, a balance must be struck between the cost to implement and maintain the remediation system and the desired level of protection.

The Botany Wetlands are a valuable environmental feature in Sydney, but their value is primarily inherent and not economic. They do not generate revenue through development or recreation, and therefore the land use does not have the capacity to offset the capital or operating costs associated with a large-scale engineered remediation project.

3.4.1 Pump and treat methods

One option is to install a network of extraction wells along the western boundary of Astrolabe Park to intercept the leachate plume before it migrates off site. The contaminated water that is pumped out would then have to be treated to bring the ammonium concentrations into compliance with the pre-determined protection levels. Ammonium is most effectively removed from leachate through a combination of nitrification and subsequent denitrification using biological reactors, ultimately degassing as nitrogen gas [Jokela et al., 2002]. If elevated concentrations of trace metals were also determined to be an issue, the effluent would then have to be subjected to an additional treatment to remove the metals as well.

Pump-and-treat systems have the advantage of a long history of proven performance and regulatory approval on a site scale, and the

infrastructure for an ammonium removal system can be constructed relatively inexpensively. However, the long-term operation and maintenance [O&M] of a groundwater extraction system eventually become very cost prohibitive, especially taking into account the large volume of water extraction that would be required to contain the leachate plume in the highly permeable sand beds of the Botany Sands aquifer. The methodology would be suitable for achieving the contaminant reduction goals, and would almost certainly have the approval of regulatory officials, but would prove to be costly in the long term for the property owner.

3.4.2 Permeable reactive barrier

Permeable reactive barriers [PRBs] are a relatively new development in remediation technology. Operation involves the excavation of a trench to intercept the contaminant plume, filled with a permeable reactive material that either reacts chemically with the contaminant to reduce it to benign end products or adsorbs the contaminants. The majority of the cost is associated with the installation of the barrier, after which only groundwater monitoring is required to ensure the compliance downgradient from the barrier. However, the majority of the field-scale research associated with PRBs is limited to chlorinated organic contaminants, with very little field data available for other contaminants.

Laboratory scale experiments have highlighted the capacity of certain minerals for sorbing both NH_4^+ and metals from groundwater [Park et al., 2002]. In the long term this method might prove to be a cost effective solution for Astrolabe Park, especially since the contamination is restricted to the upper 7m of the aquifer, but has several drawbacks: a large initial investment to install the barrier; a scarcity of field scale data to suggest the method will actually perform as expected, and a potential subsequent reluctance for regulators to accept the technology.

3.4.3 Monitored natural attenuation

Monitored natural attenuation [MNA] is another recent approach towards contamination management. The premise is that naturally occurring reactions in the subsurface are sufficient to attenuate the contaminants before they can impact a receptor. This method requires a strong fundamental knowledge of the

processes contributing to the attenuation and a comprehensive monitoring network to verify the processes. While this method is beginning to gain regulatory acceptance for petroleum contamination sites, it is still conjectured that the complex set of processes associated with leachate attenuation are not yet understood well enough to make this method valid for leachate plumes [Christensen et al., 2000]. While the low cost associated with this method is compelling, the lack of industry acceptance with respect to landfills, coupled with the fact that leachate contamination is clearly already impacting the aquatic ecosystem of Lachlan Pond No.5, nullify MNA as a management option at Astrolabe Park.

3.4.4 Alternative strategies

A potential alternative to installing abstraction bores would be to excavate a drainage trench along the shore of the pond to intercept the discharging leachate plume, avoiding the long-term O&M costs associated with an abstraction network. This measure would protect the pond, but would still require a method for treatment and discharge of an estimated 200 m³/day of leachate-impacted groundwater.

One potential compromise solution may be to simply dredge the vegetation mass from the plume discharge zone and allow for the dilution of contaminants within the pond. The pond water samples collected directly adjacent to the plume discharge zone indicate only a mild impact from leachate contamination, and historical pond chemistry data indicates that the contaminants are diluted to background concentrations within a short distance from the shore [Jankowski and Knight, 1991a,b]. A deficiency of phosphorus in the leachate plume is likely preventing eutrophication of the pond from nutrient loading, as evidenced by the historical absence of algal blooms in otherwise favourable conditions. It appears that the vegetation along the shore of the pond poses the greatest threat to the pond water quality, both by providing a large mass of decaying organic matter that depletes the oxygen in the pond and by providing a medium for the accumulation of metals. Dredging of the vegetation and associated fine-grained sediments might permit a more efficient hydraulic connection between the aquifer and the pond, enhancing the dilution of the plume contaminants entering the pond.

An accompanying measure to dilute the leachate plume prior to off-site migration would be to excavate an infiltration trench along the western boundary of Astrolabe Park. The trench would be excavated to the shallowest confining layer [~7.5 m] and backfilled with highly permeable material such as pea gravel. Clean water could then be pumped into the trench from a nearby abstraction bore, reducing contaminant concentrations through dilution, precipitation of metal oxides/hydroxides and enhanced aerobic degradation of dissolved organic material. Obviously this would incur the additional cost of excavation, construction and maintenance of a water distribution system from the existing abstraction bore to the trench.

This approach would require an ecological risk assessment to determine the impact of the infiltrating leachate plume to the pond water quality and aquatic ecosystem in order to satisfy regulatory officials. The remobilisation of accumulated metals in the pond sediments during dredging would also have to be taken into consideration. The cost for dredging alone would be relatively modest compared to the other remediation options, and may provide a viable solution to a difficult management decision.

4 CONCLUSIONS

Addressing the discharge of leachate from an old, unlined landfill poses a difficult challenge for environmental managers and stakeholders. The problem is compounded by the ecological and political sensitivity of the affected environment. A solution must be reached that strikes an appropriate balance between providing an adequate level of protection for the aquatic ecosystem of Lachlan Pond No.5 and a cost effective long-term management strategy for the property owner.

Pump-and-treat systems have a proven record of accomplishment, but also incur a large price tag as a long-term solution. PRBs and MNA offer a more affordable long-term solution, but lack the documented success and regulatory acceptance of established methods. Dredging in the leachate plume discharge zone of the pond would achieve the goals of removing a significant source of decaying organic matter from the pond, removing accumulated metals from pond sediments and enhancing the hydraulic connection, and thus the dilution capacity, of the pond with respect to the discharging plume.

However, a thorough ecological risk assessment would be advisable to determine the risk posed to the aquatic ecosystem from the undiluted leachate plume as it enters the pond, as well as from the potential remobilisation of accumulated metals by dredging activities. Ultimately a balance must be struck between the funding available to the site owner and the level of protection imposed by the regulatory officials.

5 ACKNOWLEDGEMENTS

Significant funding for recent groundwater investigations at Astrolabe Park was provided by Sydney Water.

6 REFERENCES

- Andreottola, G. and P. Cannas, Chemical and biological characteristics of landfill leachate, In: Christensen, T.H., Cossu, R. and Stegmann, R., (eds.) Landfilling of Waste: Leachate, 65-88, 1992.
- ANZECC, Guidelines for Fresh & Marine Water Quality, Australia New Zealand Environment and Conservation Council, 2000.
- Benson, D. and J. Howell, Taken for Granted, The Bushland of Sydney and its Suburbs, Kangaroo Press, 1990.
- Christensen, T.H., P.L. Bjerg, and P. Kjeldsen, Natural attenuation: a feasible approach to remediation of ground water pollution at landfills?, *Ground Water Monitoring & Remediation*, 20(1), 69-77, 2000.
- Christensen, T.H., P. Kjeldsen, P.L. Bjerg, D.L. Jensen, J.B. Christensen, A. Baun, H.-J. Albrechtsen, and Heron, G., Biogeochemistry of landfill leachate plumes, *Applied Geochemistry*, 16, 659-718, 2001.
- Ehrig, H.-J., Water and element mass balances of landfills, In: Baccini, P. (ed.), *The Landfill, Lecture Notes in Earth Sciences v.20*. Springer, Berlin, 83-116, 1988.
- Freeze, R.A., J. Massman, L. Smith, T. Sperling, and B. James, Hydrogeological decision analysis: 1. a framework, *Ground Water*, 28(5), 738-766, 1990.
- Hitchcock, P.W., The hydrogeology, hydrochemistry and geophysical conditions of Astrolabe Park and environs, M.App.Sc. Thesis, CGMH, University of New South Wales, 1991.
- Jankowski, J. and M.J. Knight, Astrolabe Park landfill leachate and its impact on the Botany wetlands, *Technical Report by Unisearch Ltd for Sydney Water Board*, 1991a.
- Jankowski, J. and M.J. Knight, Impact of polluted discharging groundwaters on the water chemistry of wetlands, International Hydrology and Water Resources Symposium 1991, Perth, WA, October 2-4, 1991b.
- Jankowski, J. and Acworth, R.I., Development of a contaminant plume from a municipal landfill: Redox reactions and plume variability, In: Chilton J. et al. (Eds.), Proceedings of the XXVIII Congress of the International Association of Hydrogeologists, Nottingham, UK, Vol.1, pp. 439-444. Balkema, Rotterdam, 1997.
- Jokela, J.P.Y., R.H. Kettunen, K.M. Sormunen, and J.A. Rintala, Biological nitrogen removal from municipal landfill leachate: low-cost nitrification in biofilters and laboratory scale in-situ denitrification, *Water Research*, 36, 4079-4087, 2002.
- Jorstad, L. and J. Jankowski, Analysis of chemical dynamics of a leachate plume in a sandy aquifer: Astrolabe Park, Sydney, NSW, Australia, In: Proceedings of the 27th Hydrology and Water Resources Symposium, Melbourne, Australia, 9pp, Institution of Engineers Australia, Barton, ACT, 2002a.
- Jorstad, L. and J. Jankowski, Analysis of inorganic constituents in a landfill leachate-contaminated aquifer: Astrolabe Park, Sydney, Australia. In: E. Bocanegra et al. (Eds.) Proceedings of the XXXII International Association of Hydrogeologists Congress, Mar del Plata, Argentina, 369-378, 2002b.
- Kelly, B.F.J., Electrical properties of sediments and the geophysical detection of ground water contamination, PhD thesis, University of New South Wales, 1994.
- Park, J.-B., S.-H. Lee, J.-W. Lee, and C.-Y. Lee, Lab scale experiments for permeable reactive barriers against contaminated groundwater with ammonium and heavy metals using clinoptilolite (01-29B), *Journal of Hazardous Materials*, 65-79, 2002.
- Qasim, S.R., and W. Chiang, Sanitary Landfill Leachate: Generation, Control and Treatment, Technomic Publishing Company, 323pp., Lancaster, Pennsylvania, 1994.
- Smolski, B., Migration of landfill leachate through a sand aquifer at Astrolabe Park, Sydney, M.App.Sc. thesis, University of New South Wales, 1994.
- Tam, E.K.L. and P.H. Byer, Remediation decision methodology for site owners, *Journal of Environmental Management*, 64, 387-400, 2002.

ANALYSIS OF INORGANIC CONSTITUENTS IN A LANDFILL LEACHATE-CONTAMINATED AQUIFER: ASTROLABE PARK, SYDNEY, AUSTRALIA

*Lange B Jorstad & Jerzy Jankowski
UNSW Groundwater Centre, School of Biological, Earth & Environmental Science
University of New South Wales, Sydney, Australia*

Abstract. An investigation was conducted at the Astrolabe Park landfill, a decommissioned municipal landfill in Sydney, Australia, to assess the physical and chemical processes affecting the distribution of inorganic constituents in a leachate plume. The plume is migrating from the landfill towards a surface water pond into which leachate-impacted groundwater discharges. Depth-discrete groundwater samples were collected from four multi-level piezometers comprising two parallel groundwater flow path transects between the edge of the landfill and the shore of the pond. The relative concentrations and spatial distribution of several key indicator parameters (EC , HCO_3^- , pH , Eh , $\text{NH}_4^+/\text{NO}_3^-$, $\text{S}^{2-}/\text{SO}_4^{2-}$) are compared at the start and end of the flow paths to determine the efficiency of contaminant transport within the leachate plume for the various elements and the degree to which attenuation of the contaminants occurs, if at all. Where contaminant attenuation is observed, an effort is made to provide the most probable partitioning mechanism responsible for the observed changes along the flow paths.

Keywords: leachate plume, landfill, hydrogeochemistry, groundwater, vertical monitoring

INTRODUCTION

Disposal of wastes in landfills is an integral part of waste management strategies in much of the world. The production of leachate as a by-product of organic and inorganic decomposition in landfills poses a serious threat if released to the environment (Baedecker and Back, 1979; Kjeldsen, 1993; Jankowski, 1997). Christensen, et al., (1992) describe leachate pollution as the result of a mass transfer process between the waste and leaching water that has infiltrated into the waste layers. Physical, chemical and microbial processes transfer pollutants from the waste material to the infiltrating water, resulting in a contaminated liquid containing high concentrations of organic and inorganic contaminants (Christensen et al., 2001). If the leachate is released into the underlying aquifer, it forms a complex contaminant plume that fundamentally alters the chemical properties of the aquifer (Baedecker and Back, 1979; Bjerg, et al., 1995; Christensen, et al., 2001; Jankowski, 1997; Jankowski & Acworth, 1997; Ludvigsen, et al., 1998). For this study, groundwater samples were collected from four multi-level piezometers that define two parallel groundwater flow path transects within a leachate plume, between the Astrolabe Park landfill and a pond located hydraulically downgradient from the landfill. Several groundwater quality parameters and inorganic contaminants are examined to assess the variations in contaminant

concentrations and spatial distribution between the landfill and the pond.

ENVIRONMENTAL SETTING

The Astrolabe Park landfill is located south of Sydney in the northern portion of the Botany Basin, an erosional depression in the Triassic Hawkesbury Sandstone bedrock (Albani, 1981; Griffin, 1963; Rickwood, 1998). Over the past approximately 200,000 years this depression was filled with Quaternary sedimentary deposits that comprise the Botany Sands aquifer (Figure 1).



Figure 1. Site location and extent of the Botany Sands aquifer.

The sedimentary sequence observed in the Botany Sands aquifer represents depositional environments ranging from marine to terrestrial. From most recent to oldest, this sedimentary sequence has been divided into 4 units (Albani, 1981): recent marine deposits (Unit 1); aeolian deposits (Unit 2); fresh to brackish lagoon deposits (Unit 3); and marine/estuarine deposits (Unit 4). In the vicinity of the Astrolabe Park landfill, the aeolian deposits of Unit 2 comprise the most significant water bearing formation.

SITE DESCRIPTION

Astrolabe Park has been established over an abandoned landfill (Figure 2). The site is underlain by aeolian and alluvial deposits comprising the upper sequence of the Quaternary Botany Sands aquifer. The upper portion of the aquifer in this area is unconfined, although the occurrence of locally continuous silty, clayey and peaty lenses creates semi-confined conditions in the lower portion of the aquifer. The landfill is located up hydraulic gradient from a series of groundwater table ponds known as the Lachlan Ponds.

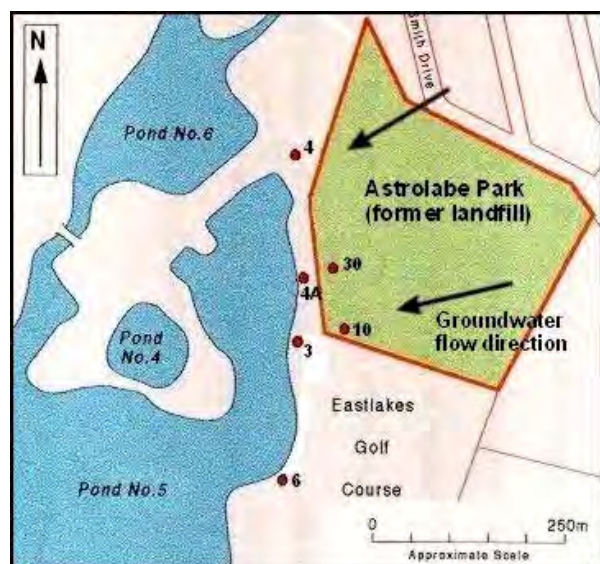


Figure 2. Location of multi-level piezometers in the vicinity of Astrolabe Park.

Mining of heavy mineral sands occurred at this site using a floating suction dredge between 1964 and 1973; this is believed to have extended to a maximum depth of 17m below the original ground surface (Hitchcock, 1991). Following the conclusion of the mining activities, the water-filled sand pits were backfilled with solid and liquid wastes of mostly domestic origin. The total

volume of waste deposited was approximately $2.04 \times 10^5 \text{ m}^3$ (Hitchcock, 1991). The lateral extent of the landfill was estimated to be 4.7 ha, based upon examination of a series of aerial photographs, resulting in a calculated average waste thickness of approximately 4.35 m.

The regional groundwater flow direction within the Botany Sands aquifer is generally towards the south. However, in the vicinity of Astrolabe Park the shallow, unconfined groundwater flows approximately in a west-southwest direction, as determined in this and previous investigations (Hitchcock, 1991; Roberts, 1996).

METHODOLOGY

The data for this investigation was collected from a network of multi-level piezometers: four of these were installed in two parallel flow path transects (30 → 4A and 10 → 3) between the edge of the landfill and the shore of Lachlan Pond No. 5, approximately 60m downgradient from the landfill (Figure 2). Sample points were positioned at 0.5 m intervals across the vertical section that defines the maximum vertical extent of the leachate plume, and then at 1.0 m intervals for an additional 5-10 m below the plume in the fresh regional groundwater. Piezometers 30 and 4A were sampled in February 2002, following the installation of piezometer 30 in late December. The results are compared to the most recent sample data from piezometers 10 and 3, collected in July 2001; it should be noted that variations in plume geometry and contaminant concentrations between the two transects may partially be a result of the discordant timing of the sampling events.

The sample points were purged using a Geopump 2 peristaltic pump, and groundwater samples were collected once the general parameters (pH, Eh, dissolved oxygen [DO], electrical conductivity [EC] and temperature) had stabilized. The unstable chemical constituents were analysed in the field immediately upon sample collection to minimize the analyte alteration. A titration method was used for HCO_3^- and CO_2 determination (APHA, 1998), and an analyte fixation and colorimetric analysis method was used for the field determination of Fe^{2+} , S^{2-} , NO_3^- , NH_4^+ and PO_4^{3-} concentrations. One 60ml sample was filtered using a $0.45 \mu\text{m}$ filter and preserved with HNO_3^- for analysis of major and

minor elements with an inductively coupled plasma spectrophotometer (ICP-AES), and one 120ml sample was filtered and used for Cl⁻ determination in the laboratory using a titration method (APHA, 1998).

It should be noted that in environments with elevated concentrations of dissolved organic material, such as landfill leachate, ionically charged organic acids can contribute significantly to the total alkalinity (Baedecker and Cozzarelli, 1992, Kehew and Passero, 1990). For the purpose of this study, total alkalinity has been calculated as HCO₃⁻ with the recognition that an unidentified percentage may be due to organic acid anions.

The accuracy and precision of the ICP analyses was assessed by analysis of spiked samples, deionized water blanks, and duplicate sample analyses as per standard laboratory protocols. All spiked samples, blanks and duplicates were within $\pm 5\%$ relative percent difference (RPD), calculated as: the difference in spiked concentration and analytical result (or first and second analysis results for duplicates) divided by the sum of these values, and multiplied by 100. Also, the charge balance error for each sample was calculated as the difference in cation and anion concentrations (as meq/L), divided by the sum of these values, and multiplied by 100; all samples were within the control range of $\pm 5\%$.

Duplicate spectrophotometer analyses were performed in the field following suspect results; the instability of the analytes following sample collection, as well as budget constraints, precluded direct, relevant comparison of the spectrophotometer results with alternative laboratory analysis techniques. However, the strong quality control results, coupled with the correlation in overall data trends observed across the three different analysis methods between six sampling events, suggests that these analysis methods accurately identified spatial and temporal trends in chemical concentrations as they occurred within the leachate plume.

LANDFILL PROCESSES AND LEACHATE PRODUCTION: A BRIEF OVERVIEW

Degradation and stabilization of waste in landfills has been observed to occur according to an ordered succession of physical, chemical and biological processes over time, as described in detail by Christensen, et al. (1992) and Ehrig

(1983). These include an initial, short-lived aerobic degradation phase followed by three phases of anaerobic degradation: (I) acid-fermentation, (II) intermediate anaerobiosis, and (III) methanogenic. Each stage represents a progression in the process and amount of decomposition of organic matter present in the wastes, and the resultant leachate produced can be identified by characteristics indicative of each stage (Christensen, et al., 1992; Ehrig, 1983).

In general, the organic waste material is ultimately reduced to CO_{2(aq)}, CH₄, and H₂O (Baedecker and Apgar, 1984; Christensen, et al. 1992; Ehrig, 1983). In the typical range of landfill leachate within the methanogenic phase (6.0 to 7.0; Baedecker and Back, 1979a; Ehrig, 1983; Nicholson et al., 1983) the CO_{2(aq)} is converted to HCO₃⁻. The amount of HCO₃⁻ introduced to the groundwater system by the degradation of organic matter can be very large; for leachate plumes in low salinity aquifers containing a low percentage of carbonate material, HCO₃⁻ is an excellent indicator of the plume location and relative concentration.

The various populations of microbes responsible for the degradation of organic matter in landfills utilize oxygen, or other oxygen-bearing species in the absence of free oxygen, to convert organic compounds to the simple end-members CO_{2(aq)}, CH₄ and H₂O (Baedecker and Apgar, 1984; Christensen, et al., 1992; Ehrig, 1983). As the microbes oxidize the organic matter, the chemical species that provide the oxygen are converted to their reduced forms, with a resultant decrease in the redox potential (measured in the field as Eh) of the groundwater. The oxidized and reduced species of a given element are referred to as redox couples. In this paper, the total concentrations and relative ratios of the redox couples for nitrogen (NH₄⁺/NO₃⁻) and sulfur (S²⁻/SO₄²⁻) are compared.

It should be noted that while field measurement of Eh values may give a relative indication of changing redox conditions within a groundwater system, they are not suitable for the determination of the specific redox processes occurring within the system (Baedecker and Cozzarelli, 1992; Lindberg and Runnells, 1984).

INTERPRETATION OF CHEMICAL DATA

Several indicator parameters have been used to assess the spatial distribution and relative attenuation of inorganic contaminants in groundwater between the Astrolabe Park landfill and Lachlan Pond No. 5, including: EC, HCO_3^- , pH, Eh, $\text{NH}_4^+/\text{NO}_3^-$, and $\text{S}^{2-}/\text{SO}_4^{2-}$. Vertical profiles of the concentrations of these indicator parameters are plotted graphically along groundwater flow paths according to sample point elevations between piezometers 30 and 4A and piezometers 10 and 3 (figures 4 and 5, respectively). These figures will be used in the following discussion to make direct comparisons of the relative concentrations and spatial distribution of the indicator parameters between the landfill and the pond.

EC and HCO_3^-

The relationship between EC and HCO_3^- for the samples used in this study is plotted in Figure 3. This plot indicates that there is an excellent correlation between these two values, and that both can be used characterise the relative degree of leachate impact to regional groundwater.

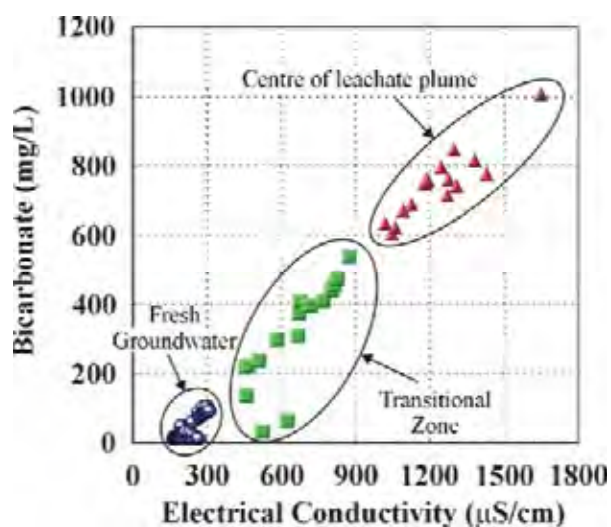


Figure 3. Plot of HCO_3^- versus EC in groundwater samples, and subsequent contamination characterisation.

Both EC and HCO_3^- show significant variations in maximum concentration and spatial distribution between piezometer 30 and 4A, located 50 m downgradient. Maximum EC values decrease from 1429 to 1060 $\mu\text{S}/\text{cm}$, a 26% decrease, and the elevation of the peak value dropped by approximately 4.4 m. Likewise, the

maximum HCO_3^- concentration decreased from 847 to 622 mg/L, a 27% decrease, with a vertical displacement nearly identical to that of the EC.

The decrease in both EC and HCO_3^- concentrations between these piezometers is most likely due to dilution of the leachate plume from both longitudinal and lateral dispersion. The decrease in elevation of the centre of mass may be due to a number of factors, including density differences between the fresh and contaminated water, the presence of preferential flow paths in the subsurface, or depression of the plume from infiltrating precipitation and lawn watering on the golf course between the landfill and the pond.

The observed variation between piezometers 10 and 3 is slightly different. The maximum EC value decreased from 1650 to 1381 $\mu\text{S}/\text{cm}$, a decrease of 16%, while the maximum concentration of HCO_3^- decreased from 1008 to 816 mg/L, a decrease of 19%. The elevation of the maximum concentration for both parameters only decreased by approximately 1 m. Assuming that the effects of dispersion and lawn watering infiltration on the leachate plume are similar in the two closely spaced transects, the most likely explanation for the discrepancy in variation between the 30 to 4A and 10 to 3 transects is the timing of the sampling events (February 2002 and August 2001, respectively) relative to rainfall events. Also, a slight increase in the fraction of fine sediment between piezometers 30 and 4A as compared to 10 and 3 may be responsible for increased attenuation of inorganic contaminants.

pH and Eh

Both transects display a similar pattern of increased pH values and decreased Eh values with respect to regional groundwater within the leachate plume, but the individual profiles vary significantly.

The pH of the contaminated groundwater at piezometer 30 is tightly constrained between values of 6.4 to 6.7 for approximately 6 m, below which a sharp increase is observed between an elevation of 8.5 to 7.5 m asl. A similar pattern is observed with Eh values, which rise dramatically within the same interval. The pH profile at piezometer 4A is also constrained between values of 6.4 to 6.7 in the contaminated zone, but then gradually returns to background levels between the elevations of 10 to 4 m asl.

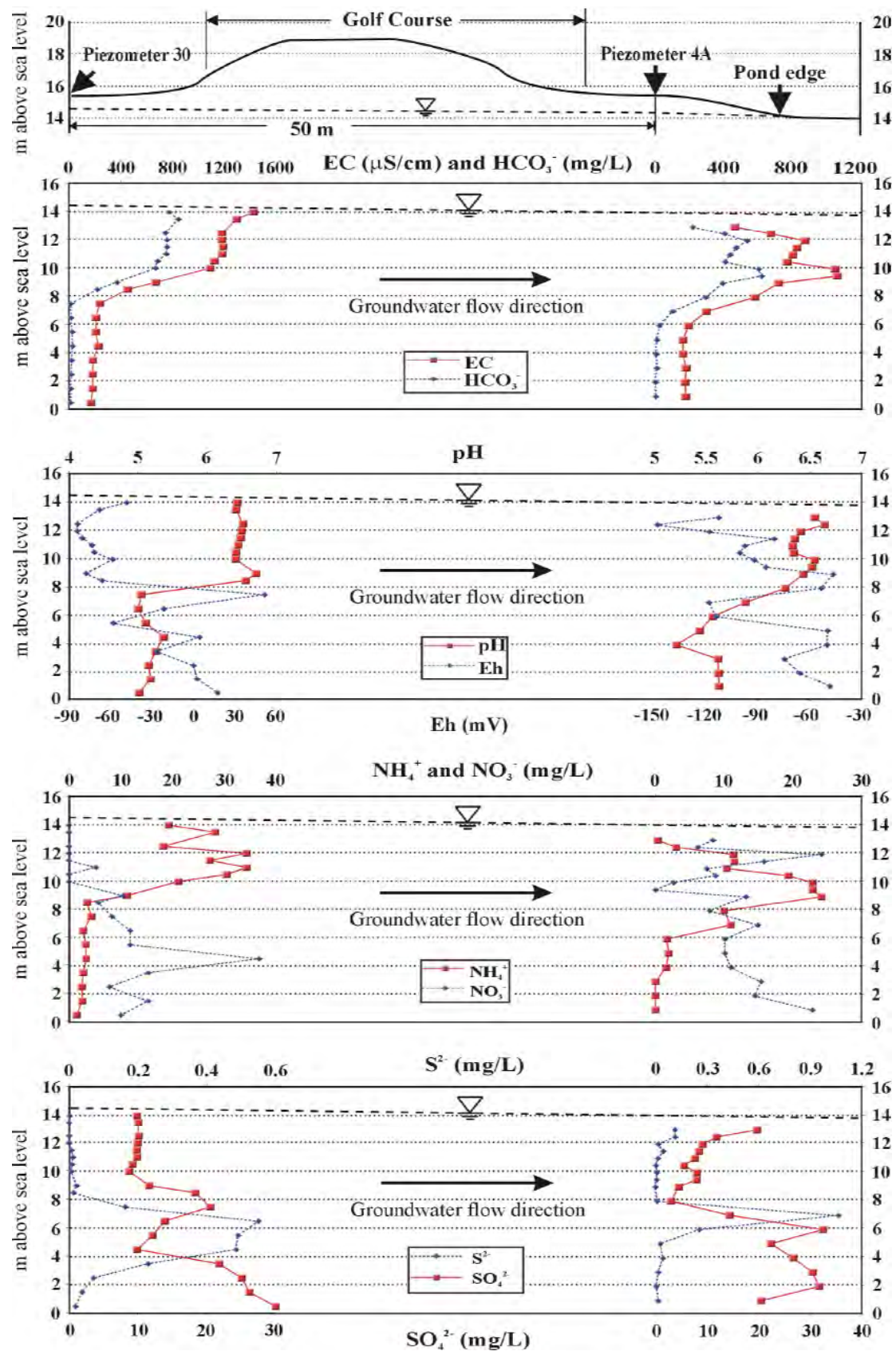


Figure 4. Vertical profiles of groundwater quality parameters and contaminants for multi-level piezometers 30 and 4A, located along a groundwater flow path between the Astrolabe Park landfill and Lachlan Pond No. 5.

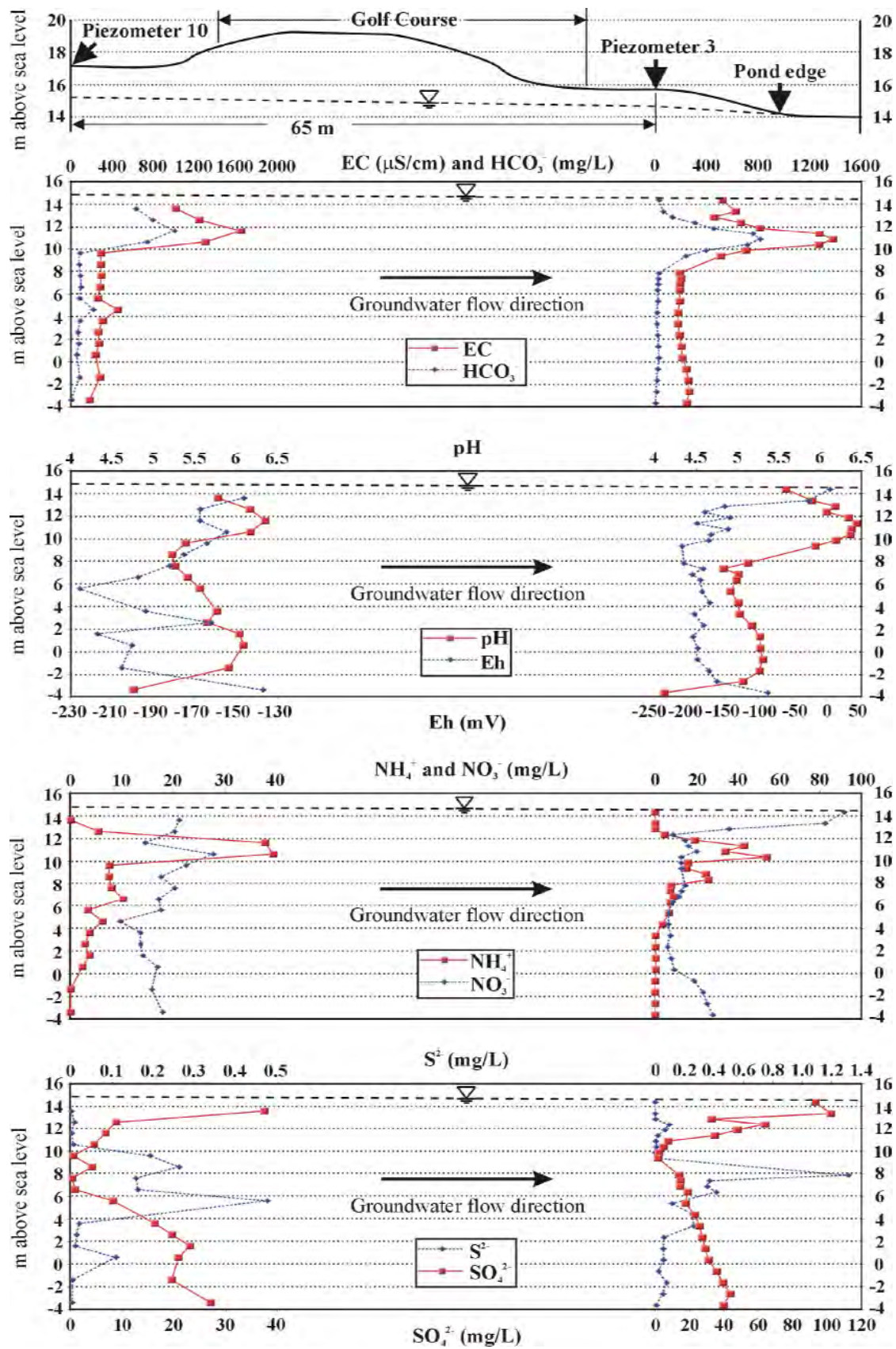


Figure 5. Vertical profiles of groundwater quality parameters and contaminants for multi-level piezometers 10 and 3, located along a groundwater flow path between the Astrolabe Park landfill and Lachlan Pond No. 5.

Likewise, the Eh profile at piezometer 4A displays a much narrower vertical section of highly reducing water with a more gradual transition to fresh water. The variation in the transitional zones may be due to vertical mixing between the two piezometers, although the severity of the transitional zone at piezometer 30 may also indicate the presence of a laterally discontinuous confining layer at an elevation of approximately 8 m asl. The maximum Eh value at piezometer 4A is lower than that of piezometer 30, this discrepancy is probably due to localized variations in the sedimentary composition of the aquifer, resulting in variable rates of microbiological activity.

The pH profiles at piezometers 10 and 3 also display pH values constrained between values of 6.0 to 6.5 within the leachate plume, although this zone appears to be approximately 1.5-2 m thicker at piezometer 3, most likely a function of vertical dispersion of the plume between the two piezometers. The Eh profiles between the two piezometers are slightly erratic, although both display a decrease in Eh values that corresponds to the zone of increased pH values. The variation in the shapes of the profiles is likely attributable to the presence of locally discontinuous peat lenses containing iron sulfides, which act as substrates for SO_4^{2-} -reducing bacteria. This may be responsible for the sharp drops in Eh observed at elevations of 6 m and 2 m asl at piezometer 10 (Jankowski and Acworth, 1998). Comparison of the shape of the Eh profiles to the $\text{S}^{2-}/\text{SO}_4^{2-}$ profiles appears to support this assertion.

NH_4^+ and NO_3^-

The profiles of N-species at all four piezometers clearly display the expected relationship between the redox couples both within and below the leachate plume.

Piezometer 30 displays a near absence of NO_3^- within the leachate plume, increasing with depth below 9 m asl. NH_4^+ concentrations display the opposite trend, with the highest concentrations observed within the plume, decreasing dramatically between the elevations of 10 to 8 m asl. A similar relationship is observed for piezometer 4A, although the maximum concentration of NH_4^+ decreases from 35 to 24 mg/L, a decrease of 30%. Likewise, the zone of maximum NH_4^+ concentration decreases from 12-

11 m asl to 10-9 m asl, and appears to narrow vertically by approximately 2 m. Also, a spike of high NO_3^- concentration is observed at piezometer 4A at an elevation of 12 m asl. This is likely due to denitrification along the edge of the plume as it reacts with oxygen-enriched recharge water, and perhaps by the application of nitrogen-based fertilizers on the golf course.

A much narrower vertical section of elevated NH_4^+ concentrations is observed in piezometers 10 and 3, although the maximum concentration values are nearly the same as that of piezometer 30. Surprisingly, the maximum concentration of NH_4^+ actually increases between piezometers 10 and 3, from 40 to 43 mg/L, an increase of 8%. There was no observed change in the elevation range of the maximum NH_4^+ concentration. Also, a spike of NO_3^- is observed at the top of the profile at piezometer 3; the most likely source of this additional nitrogen is from fertilizers used by the golf course. The high NO_3^- concentrations at the water table at piezometer 3 may be contributing to the increased NH_4^+ concentration within the leachate plume.

It seems likely that any potential contribution of nitrogen to groundwater from nitrogen-based fertilizers used by the golf course would be similar for the two transects as they are relatively close to each other, similar in length and cross the same portion of the golf course. By this assumption, the discrepancy between the observed NH_4^+ attenuation, or lack thereof, for the two transects may be due to a significantly increased percentage of fine-grained silt, clay and peat material present in the upper portion of the aquifer between piezometers 30 and 4A that is not present between piezometers 10 and 3. This may cause some attenuation of the NH_4^+ at exchange sites present on the clay surfaces.

S^{2-} and SO_4^{2-}

The SO_4^{2-} concentrations in all profiles display similar patterns of decreased concentrations within the leachate plume and elevated concentrations both above and below the plume. The increase in SO_4^{2-} concentrations below the plume is relatively sharp (10-30 mg/L increase in the first 2 meters below the plume) for all piezometers except for piezometer 3. The Eh values and the NO_3^- and SO_4^{2-} concentrations all recover much more slowly with depth beneath the plume than is observed in the other piezometers. It

is possible that the presence of lenses of organic material provide an additional substrate for microbial activity, which preserves the reducing environment to a greater depth than is observed in the other piezometers.

The concentration of S^{2-} in all groundwater samples is low compared to SO_4^{2-} , with only two samples exceeding 1 mg/L. Also, the S^{2-} is absent within the zone of greatest leachate contamination as defined by the EC, HCO_3^- and NH_4^+ profiles. Since SO_4^{2-} concentrations appear to be reduced within the plume, the expected result would be an increase in S^{2-} . The most plausible explanation for the absence of S^{2-} in the plume is that it is bonding with Fe^{2+} in solution and precipitating as FeS (Postma and Jakobsen, 1996).

The highest S^{2-} concentrations in all profiles occur immediately below the leachate plume (Jankowski and Acworth, 1997). This zone also happens to correspond to a relatively continuous layer of pyrite-bearing peat material present in the investigation area identified in previous research (Hitchcock, 1991). In contrast to the EC, HCO_3^- and NH_4^+ concentrations for piezometers 30 and 4A, which appear to have their source in the landfill and decrease in concentration and elevation as they are transported downgradient, the S^{2-} concentration increases downgradient from the landfill and does not display any vertical displacement. A similar trend is also present between piezometers 10 and 3. Further, the peaks in S^{2-} concentration occur at nearly the same elevation in all of the piezometers. This combined evidence suggests that the pyrite-bearing peat layer is the most likely source of the increased S^{2-} concentrations observed in the profiles (Jankowski and Beck, 2000).

CONCLUSIONS

In general, a decrease in the maximum value/concentration of EC and HCO_3^- was observed between the Astrolabe Park landfill and Lachlan Pond No. 5, as well as a decrease in the elevation at which the maxima were detected, although this was most pronounced in the piezometer 30 to 4A transect. As the groundwater is transported away from the landfill, and thus away from the source of the HCO_3^- production, it is assumed that EC and HCO_3^- concentrations decrease as a result of dispersion along the flow path and mixing with fresh regional groundwater. The relative differences in the concentration

decreases is likely due differences in precipitation at the time of the two sampling events.

The pH profiles for both transects display well buffered values within the leachate plume (Jankowski and Acworth, 1997; Kehew and Passero, 1990), which recover to background values below the plume. Once again, there is a greater observable change in the thickness of the buffered zone along the 30 to 4A transect than along the 10 to 3 transect, most likely attributable to different weather patterns due to the discordant timing of the sampling events. While generally displaying depressed values within the leachate plume, the Eh profiles varied significantly between each piezometer. Local variations in the sedimentary composition of the aquifer appear to have a significant impact on the variability of redox potentials with depth at each piezometer, which makes Eh values a difficult parameter from which to draw any significant conclusions regarding the specific effects of transport on leachate contaminants.

The profile for each piezometer displayed the expected relationship between NH_4^+ and NO_3^- within and below the leachate plume. The 30 to 4A transect showed a significant decrease in the maximum concentration of NH_4^+ along the flow path, most likely due to a combination of dispersion/dilution and attenuation of NH_4^+ on exchange site surfaces within the geological material of the aquifer. Unexpectedly, the maximum NH_4^+ concentration was increased between piezometer 10 and 3. The only possible cause for this occurrence is an additional influx of nitrogen along the flow path. The extremely high NO_3^- concentration immediately above the plume in piezometer 3 suggests that nitrogen may be flushed into the groundwater system from nitrogen-based fertilizers used on the golf course; nitrate-reducing bacteria may have reduced a portion of the NO_3^- to NH_4^+ , accounting for the observed increase in concentration along the flow path. Once again, the difference in the timing of the sample events may explain why this phenomena was not observed along the 30 to 4A transect.

SO_4^{2-} concentrations followed a similar pattern to NO_3^- : decreased concentrations within the leachate plume with an observed recovery below the plume. The expected resultant increase in S^{2-} concentrations within the plume was not observed; any S^{2-} produced within the plume as a

result of SO_4^{2-} reduction is most likely precipitated as FeS in the Fe^{2+} -enriched plume (Postma and Jakobsen, 1996).

In contrast, the highest concentrations of S^{2-} were observed immediately below the plume, where previous research has identified a locally continuous, pyrite-bearing peat layer (Jankowski and Acworth, 1997). The fact that the elevated S^{2-} concentrations occurred at approximately the same elevation in each profile, and the maximum concentrations increased along the flow path, suggests that the S^{2-} occurs naturally within the aquifer as a result of organic degradation occurring within the peat layer (Jankowski and Beck, 2000).

While dispersion, dilution and sorption processes along groundwater flow paths appear to be responsible for the decrease in concentrations of some components of the leachate plume, the influence of precipitation needs further investigation to determine whether the relative variations observed between the transects are indeed due to weather variations as a result of the discordant timing of the sampling events. Also, the relative nitrogen contribution from fertilizer along the flow path use needs to be investigated to explain the unexpected increase in NH_4^+ concentration between piezometers 10 and 3.

REFERENCES

Albani, A.D., 1981. Sedimentary environments and Pleistocene chronology of the Botany Basin, NSW. *Geo-Marine Letters*, 1, 163-167.

American Public Health Association (APHA), 1998. *Standard Methods for the Examination of Water and Wastewater: (20th Edition)*. APHA-AWWA-WET, Washington DC.

Baedecker, M.J. and Back, W., 1979. Hydrogeological processes and chemical reactions at a landfill. *Ground Water*, 17(5), 429-437.

Baedecker, M.J. and Apgar, M.A., 1984. Hydrogeochemical studies at a landfill in Delaware. **In:** *Groundwater Contamination, Studies in Geophysics*. National Academy Press, Washington D.C., 127-138.

Baedecker, M.J. & Cozzarelli, I.M., 1992. The determination and fate of unstable constituents of contaminated groundwater. **In:** S. Lesage and

R.E. Jackson (eds.) *Groundwater Contamination and Analysis at Hazardous Waste Sites*, Marcel Dekker, Inc., New York, 425-461.

Bjerg, P.L., Rugge, K., Pederson, J.K. and Christensen, T.H., 1995. Distribution of Redox-Sensitive Groundwater Quality Parameters Downgradient of a Landfill (Grinsted, Denmark). *Environmental Science & Technology*, 29, 1387-1394.

Christensen, T.H., Cossu, R. and Stegmann, R., 1992. *Landfilling of Waste: Leachate*. E & FN Spon, London.

Christensen, T.H., Kjeldsen, P., Bjerg, P.L., Jensen, D.L., Christensen, J.B., Baun, A., Albrechtsen, H.-J. and Heron, G., 2001. Biogeochemistry of landfill leachate plumes. *Applied Geochemistry*, 16, 659-718.

Ehrig, H.-J., 1983. Quality and quantity of sanitary landfill leachate. *Waste Management & Research*, 1, 53-68.

Griffin, R.J., 1963. The Botany Basin. *Bulletin, Geological Survey of New South Wales*, 18, 110pp.

Hitchcock, P.W., 1991. The hydrogeology, hydrochemistry and geophysical conditions of Astrolabe Park and environs. M.App.Sc. Thesis, CGMH, University of New South Wales, Australia.

Jankowski, J., 1997. Generation and migration of chemical constituents in a leachate plume. **In:** G.H. McNally (ed.) *Collected Case Studies in Engineering Geology, Hydrogeology, and Environmental Geology*, Geological Society of Australia, 181-205.

Jankowski, J. and Acworth, R.I., 1997. Development of a contaminant plume from a municipal landfill: Redox reactions and plume variability, **In:** J. Chilton, et al. (eds.) *Proceedings of the XXVII International Association of Hydrogeologists Congress on Groundwater in the Urban Environment*, Nottingham UK, September 1997. Balkema, Rotterdam, Vol.1, 439-444.

Jankowski, J. and Acworth, R.I., 1998. Surface water and ground-water interaction in the recharge zone of a coastal aquifer, Sydney, Australia. **In:** J. Van Brahana, et al. (eds.), *Proceedings of the*

XXVIII Congress of the International Association of Hydrogeologists, Las Vegas, Nevada, USA, 27 September - 2 October, 1998, 667 - 673.

Jankowski, J. and Beck, P., 2000. Aquifer heterogeneity: hydrogeological and hydrochemical properties of the Botany Sands aquifer and their impact on contaminant transport. *Australian Journal of Earth Sciences*, 47, 45-64.

Kehew, A.E. and Passero, R.N., 1990. pH and redox buffering mechanisms in a glacial drift aquifer contaminated by landfill leachate. *Ground Water*, 28(5), 728-737.

Kjeldsen, P., 1993. Groundwater pollution source characterisation of an old landfill. *Journal of Hydrology*, 142, 349-371.

Lindberg, R.D. and Runnells, D.D., 1984. Ground water redox reactions: An analysis of equilibrium state applied to Eh measurements and geochemical modeling. *Science*, 225, 925-927.

Ludvigsen, L., Albrechtsen, H.-J., Heron, G., Bjerg, P.L., Christensen, T.H., 1998. Anaerobic microbial redox processes in a landfill leachate contaminated aquifer (Grinsted, Denmark). *Journal of Contaminant Hydrology*, 33, 273-291.

Nicholson, R.V., Cherry, J.A. and Reardon, E.J., 1983. Migration of contaminants in groundwater at a landfill: A case study. 6. Hydrogeochemistry. *Journal of Hydrology*, 63, 131-176.

Postma, D. and Jakobsen, R., 1996. Redox zonation: Equilibrium constraints on the Fe(III)/SO₄-reduction interface. *Geochimica et Cosmochimica Acta*, 60(17), 3169-3175.

Rickwood, P.C., 1998. The Botany Basin: a convenient definition. **In:** G.H. McNally and J. Jankowski (eds.) *Environmental Geology of the Botany Basin*, Geological Society of Australia, 173-178.

Roberts, P., 1991. Hydrochemical trends within the contaminant plume at Astrolabe Park landfill, Sydney, Australia. M.App.Sc. Thesis, UNSW Groundwater Centre, University of New South Wales, Australia.



Integration of multi-channel piezometry and electrical tomography to better define chemical heterogeneity in a landfill leachate plume within a sand aquifer

R.I. Acworth ^{a,*}, L.B. Jorstad ^b

^a *Water Research Laboratory, School of Civil and Environmental Engineering, King Street,
Manly Vale 2093, NSW Australia*

^b *Environmental Resources Management Australia Pty Ltd., Building C, 33 Saunders Street, Pyrmont,
NSW 2009, Australia*

Received 19 December 2004; received in revised form 14 October 2005; accepted 10 November 2005

Abstract

The Hanai-Bruggeman effective medium theory is used to relate bulk electrical conductivity, measured by surface and cross-borehole images, to fluid electrical conductivity, surface conductance, porosity and the geometry factor, in a medium- to fine-grained sand deposit. The change in bulk EC is caused by the presence of a landfill leachate plume. Repeated electrical images over a period of 16 months indicate that various segments of the plume are moving. The chemical constituents of the leachate plume have been determined by sampling from a bundled piezometer located in the electrical image field. Very close agreement is demonstrated between the fluid EC anomaly and the presence of elevated bulk EC indicating that the electrical images can be used to map the plume geometry and to monitor the movement of the plume segments.

© 2005 Elsevier B.V. All rights reserved.

Keywords: Landfills; Leachate plume; Geophysical methods

1. Introduction

1.1. Landfilling of waste and leachate generation

Landfilling of waste is an integral component of solid waste management strategies for communities worldwide. While stringent legislation in most industrialised countries currently

* Corresponding author. Fax: +61 2 9949 4188.

E-mail address: i.acworth@unsw.edu.au (R.I. Acworth).

requires considerable measures to be undertaken to protect the environment from landfill emissions, there are numerous reported cases of old decommissioned (or abandoned) landfills, designed without engineered containment systems, that are still acting as unregulated point-sources for the discharge of pollutants to the environment (Kimmel and Braids, 1974; Baedecker and Back, 1979a; MacFarlane et al., 1983; Baedecker and Apgar, 1984; Kehew and Passero, 1990; Lyngkilde and Christensen, 1992; Mirecki and Parks, 1994; Jankowski and Acworth, 1997; Kjeldsen et al., 1998; Cozzarelli et al., 1999). Prior to legislation governing the selection criteria for the siting of new landfills, it was common practice to dispose municipal solid waste on property that was considered to be of low economic value. Seasonally inundated swampland, abandoned sand mines or quarries were used with the intention of later redeveloping the landscape. These practices resulted in numerous cases of uncontrolled leachate migration into adjacent aquifer systems.

Landfill leachate can be loosely defined as a water-based solution of compounds derived from the waste (Christensen et al., 2001). Leachate is generated when water infiltrates into the waste, either in the form of precipitation or as shallow groundwater, instigating a combination of physical, chemical and microbial processes that act as mass transfer mechanisms between the waste and the infiltrating water (Andreottola and Cannas, 1992; Christensen et al., 2001). While the composition of landfill leachate can vary significantly between different landfills, it is generally characterised by elevated concentrations of dissolved organic matter, major ions, and a variety of trace elements and synthetic organic contaminants (Chian and DeWalle, 1977; Baedecker and Back, 1979b; Ehrig, 1983, 1988; Andreottola and Cannas, 1992; Christensen, 1992).

1.2. Delineation of landfill leachate plumes in heterogenous aquifers

The introduction of landfill leachate into a formerly pristine aquifer system typically results in a complex set of reactions that profoundly alters the chemical composition of the groundwater and aquifer mineralogy (Baedecker and Back, 1979a; Baedecker and Apgar, 1984; Christensen, 1992; Cozzarelli et al., 1999). As with any release of contaminants to the environment, it is essential to undertake a thorough investigation that adequately addresses the mechanisms affecting the distribution, transport and attenuation of contaminants in the subsurface. The interpretation of the resulting data set can be considerably complicated by the heterogenous nature of many aquifers on a site scale, which results in significant variations in groundwater quality over short distances and the development of preferential groundwater flow paths that can be difficult to identify using conventional monitoring networks (Ronen et al., 1987; Back and Baedecker, 1989; Bjerg and Christensen, 1992; Jankowski and Beck, 2000). Vertical heterogeneity is especially significant in a sedimentary aquifer due to the horizontal disposition of the sedimentary deposits and limited transverse dispersivity associated with advective solute transport (Bjerg and Christensen, 1992). Steep vertical chemical gradients have been observed in heterogenous aquifers on the order of a few to tens of centimetres (Baedecker and Back, 1979b; Ronen et al., 1987; Bjerg and Christensen, 1992). Perhaps more surprisingly, examples of small-scale vertical zonation of groundwater chemistry in the homogeneous Botany Sands aquifer have also been identified (Jankowski and Acworth, 1997; Lavitt et al., 1997; Jankowski and Beck, 2000; Acworth, 2001). Smith et al. (1991) comment on the importance of providing closely spaced sampling points in the vertical dimension if chemical heterogeneity is to be satisfactorily resolved.

Geophysical investigation techniques often prove to be a valuable complement to a groundwater sampling program, provided that the chemical properties of the contaminant plume

result in a distinct geophysical anomaly with respect to the regional background water (Acworth and Dasey, 2003). They are particularly useful for the delineation of landfill leachate plumes, which are usually characterised by an elevated ionic load. The introduction of landfill leachate with enhanced electrical conductivity into an aquifer system containing groundwater with a naturally low electrical conductivity will result in a bulk electrical conductivity anomaly that is readily detectable using both surface, borehole or cross-borehole electrical resistivity imaging methods. Surface resistivity data can be effectively correlated with cross-borehole tomography data and depth-discrete groundwater electrical conductivity (Fluid EC) data measured from bundled piezometers, resulting in a continuous, high-resolution image of the distribution of the leachate plume as it appears in the subsurface. This integration of field investigation techniques provides a relatively fast and inexpensive method for delineating the extent of leachate contamination in an aquifer, and may provide valuable insight into the nature of solute distribution and transport within the plume. In this paper we compare the measured fluid electrical conductivities, obtained by sampling a bundled piezometer, with those estimated from the bulk electrical conductivity, derived from the inversion of cross-borehole tomography, using the Hanai-Bruggeman effective medium theory.

2. Site description

2.1. Environmental setting

The study site is the Astrolabe Park landfill located approximately 6 km south of the Sydney central business district in the suburb of Daceyville (Fig. 1). The site includes the former landfill, completed as the Astrolabe Park sports field upon decommissioning in 1978, a portion of the adjacent Eastlakes Golf Course, and a portion of the eastern shore of Lachlan Pond No. 5 (Fig. 2). The study site is located within the northern portion of the Botany Basin. The Botany Basin consists of three deep, steep-sided valleys incised into the Triassic Hawkesbury Sandstone bedrock that have been backfilled with a sequence of Quaternary sedimentary deposits associated with a series of marine transgressions and regressions. In the vicinity of the Astrolabe Park landfill, the unconsolidated sediments consist of an approximately 30 m thick deposit of aeolian sand with minor silt and peat horizons. The silt and peat deposits consist of organic staining around the sand grains and some minor silt. They represent former groundwater fed lake bottom deposits. The groundwater table moved vertically in response to sea level fluctuations throughout the Pleistocene (Lambeck and Nakada, 1990).

2.2. History of landfilling

The source of the historical landfilling information in, and in the vicinity of, Astrolabe Park is based on a thorough review of Botany Council records as reported by Philip Hitchcock (pers. comm.). Because of the scarcity of information regarding the waste disposal activities at the Astrolabe Park landfill, the Botany Council records comprise the sole source for this information. Unless otherwise noted, the information in this section is based entirely upon Hitchcock's review.

Astrolabe Park and the south-adjacent property, denoted 'Area A' in council records, were subjected to an intermittent history of sand mining and waste disposal between 1938 and 1975 (Fig. 2). A 2-acre portion of the southwest corner of Astrolabe Park was used as a municipal tip from 1938 to 1945, at which time waste disposal was routed to an alternate tip site. The tip was

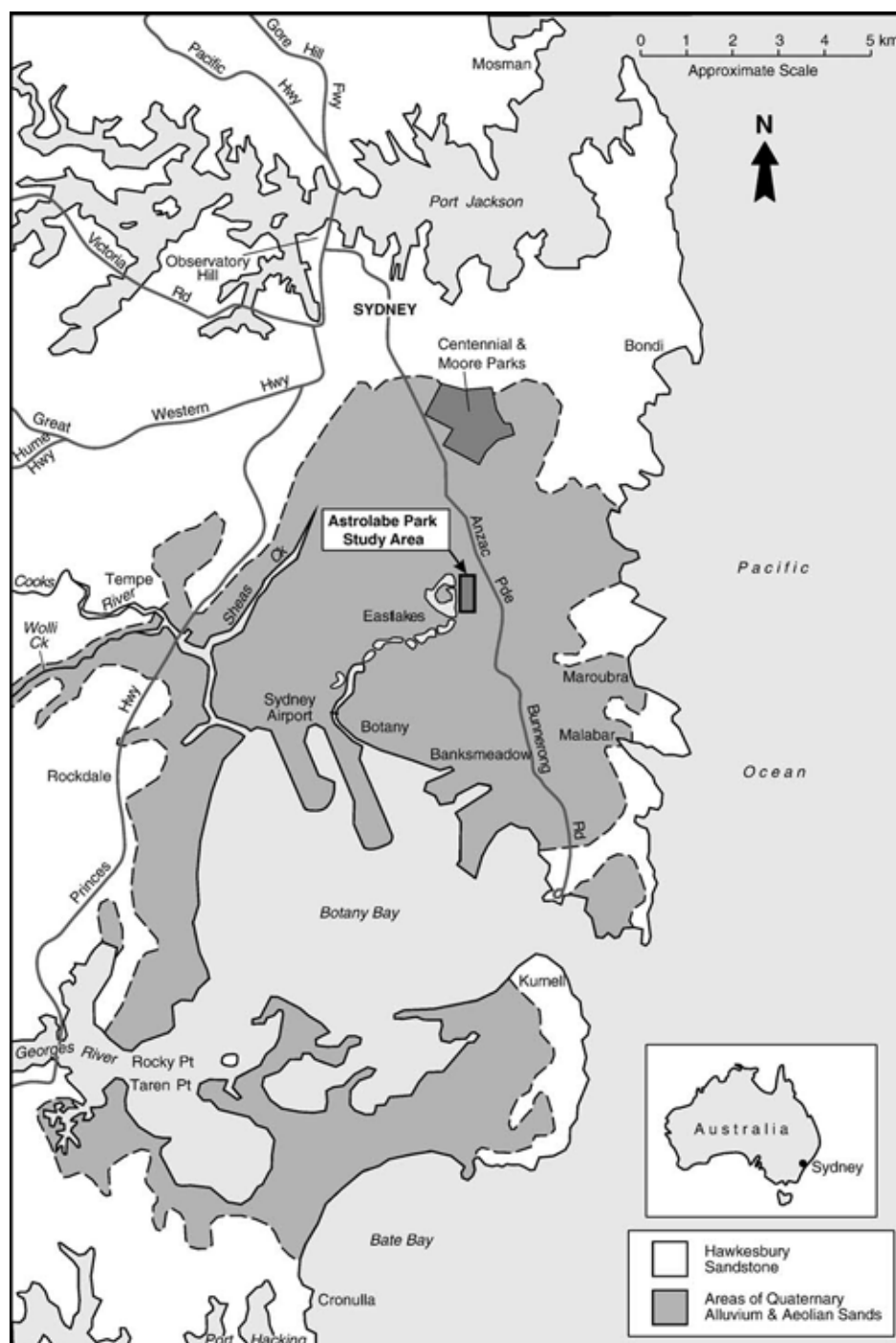


Fig. 1. Location of the Astrolabe Park landfill with respect to the greater Botany Basin.

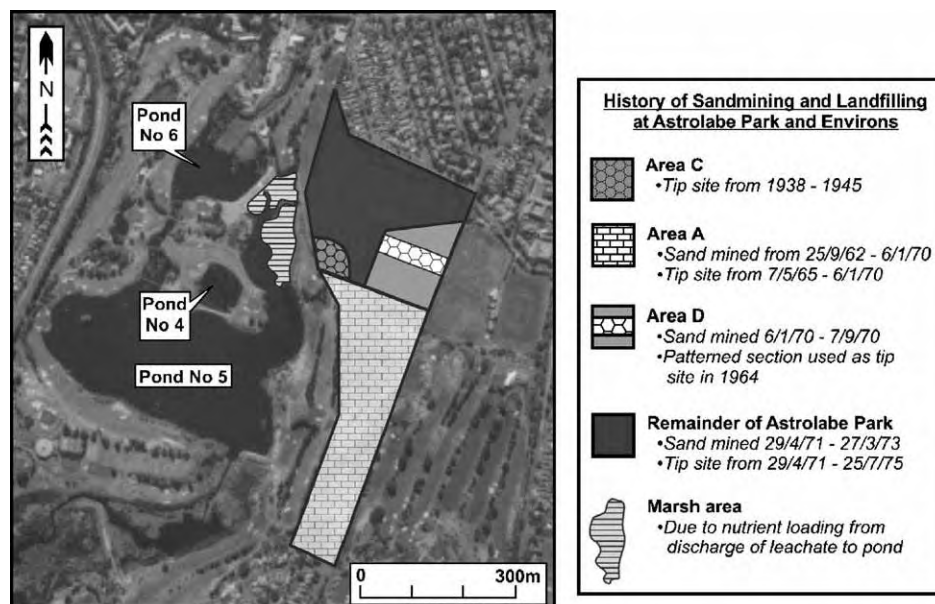


Fig. 2. History of sand mining and landfilling at Astrolabe Park and environs.

in a wetland area fed by an intermittent stream (the old Bird's Gully) that discharged into the lakes. A causeway had been built across the wetland and a pipe installed to allow drainage of the wetland.

Sand mining of the sand dunes was carried out in the 1950s and 1960s. The sand was used for glass manufacture. Council records show that sand mining was permitted using a dragline and that extraction was permitted to a maximum depth of 1.5 m below the water table. Standing water is seen in several air photographs of the site taken during the 1960s. Sand mining was not permitted in the area of the old tip or within a 10 m buffer on either side of the intermittent stream.

Waste disposal resumed in Astrolabe Park and Area A in 1964 as backfill into the sand mining pits. Waste disposal at Astrolabe Park concluded in 1975, when leachate was observed discharging into Lachlan Pond No. 5 via a stormwater drain on the western edge of the landfill. It is assumed that this was the original intermittent stream. The drain was relocated to a site further to the north by the construction of twin box culverts beneath the Park from the northeast corner of the site to the culvert entering Pond 5 (Fig. 3). The area was then landscaped. The decommissioning process was completed in 1978 by covering the entire site with approximately 500 mm of sand and completing it as a sports field.

The total volume of waste deposited in Astrolabe Park was calculated to be $2.04 \times 10^5 \text{ m}^3$, based on an extrapolation of the reported disposal rate of 4000 m^3 per month over the 51 months of landfilling in the sand mined portion of Astrolabe Park. The lateral extent of the landfill was estimated to be 4.7 ha, based upon examination of a series of aerial photographs, resulting in a calculated average waste thickness of approximately 4.35 m across the site. The sparse records regarding waste composition indicate a mix of domestic, commercial and possibly industrial solid waste. According to council requirements, only non-putrescible solid waste, including brick, concrete and steel scrap, was to be disposed of below the water table. Compliance with

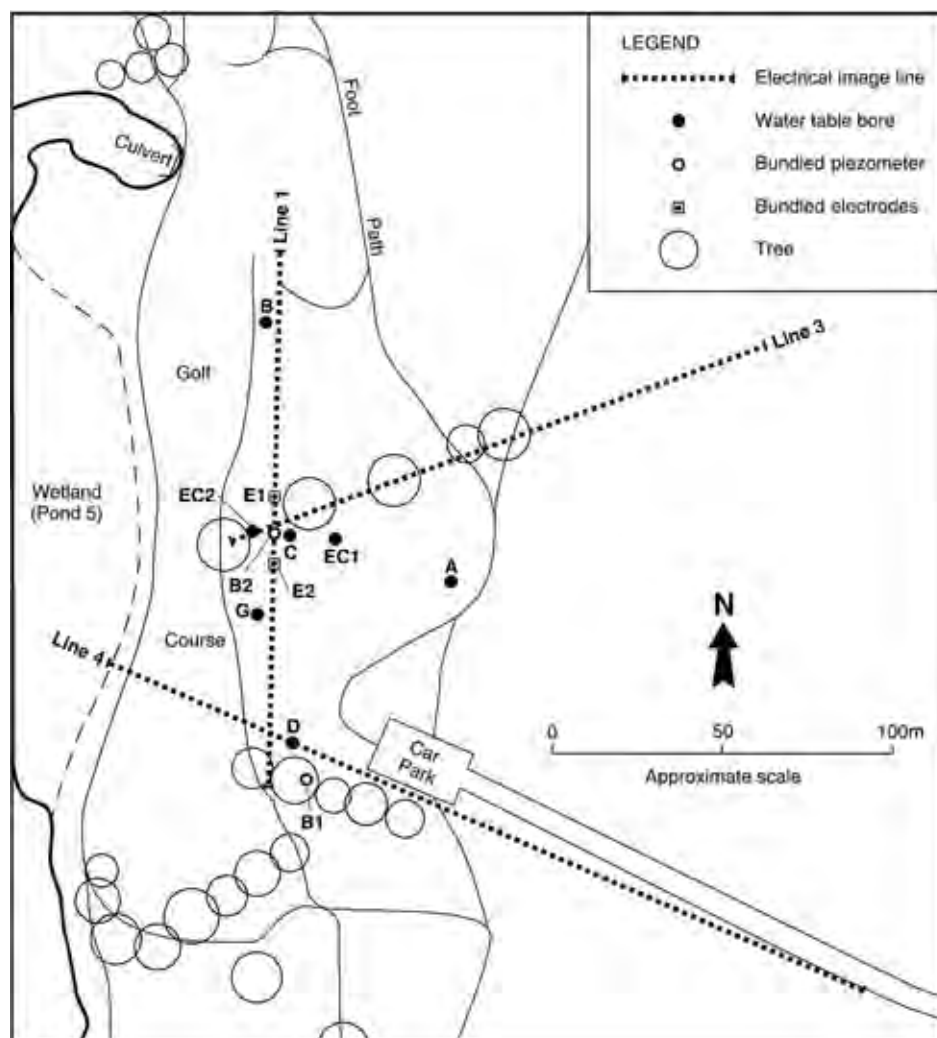


Fig. 3. Layout of the study site showing the individual and bundled piezometer locations, the electrical resistivity image lines and the location of the electrode bundles.

this requirement is unknown but old site photographs show that at least some putrescible waste was dumped. An approximate cross-section is shown through the investigation site in Fig. 4.

3. Field methods

A site for the investigation was selected in a low-lying area of the Park between the edge of the landfill and Pond No. 5 (Fig. 3). This was close to the original wetland area at the southwest of the site. The hydraulic gradient across the site was established by installing 4 shallow monitoring piezometers (A to D in Fig. 3) and measuring water level elevations. The water table depth varied between 1 and 2 m beneath the site, or between 13.7 and 15.1 m AHD (AHD is Australian Height Datum and is equivalent to mean sea level). Exploratory electrical imaging

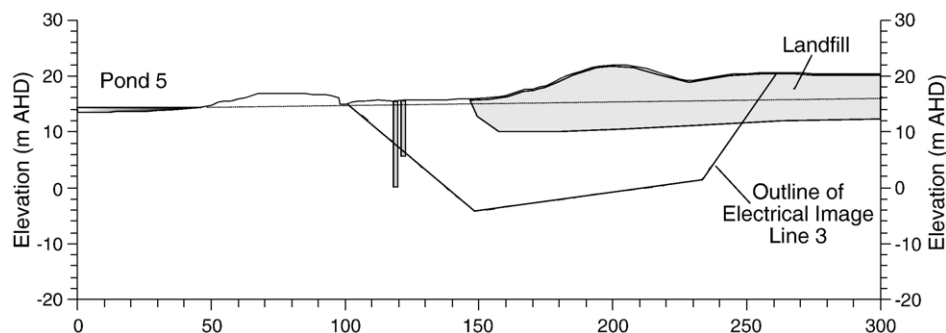


Fig. 4. Cross-section through the site along electrical image line 3.

(Line 1 in Fig. 3) and conductivity cone penetration tests (CCPT) confirmed that a contaminant plume existed in this area and that it was both shallow and of limited depth.

3.1. Electrical imaging

Electrical imaging was completed using 2 multi-core cables with takeouts every 2.5 m. The cables were connected to an ABEM LUND ES464 switching unit slaved to an ABEM SAS4000 Terameter. Standard configurations were used on the SAS4000 to acquire measurements of apparent resistivity using the Wenner equi-spaced electrode configuration. Data were uploaded to a laptop computer and inverted to produce a distribution of true resistivity using the RES2DINV software (Loke, 2001). Significant topography occurred along some lines and the elevation of the electrode positions was obtained using standard survey techniques to allow topographic corrections to the electrical image to be made (Loke, 2001). Images have been repeated over exactly the same line on 4 occasions to determine the possible change in position of segments of the contaminant plume.

3.2. Bundled electrode and piezometer construction

A bundled piezometer (B2 in Fig. 3) was installed to 15 m depth using hollow stem auger techniques. The bundled piezometer consisted of a central 50 mm diameter PVC casing with 5 mm diameter sample tubes strapped to the outside of the casing. Each sample tube terminated at a different depth with vertical spacing varying from 0.5 to 1 m. The intake area for the sample tube was formed from a 20 mm section at the base of the tube that had been perforated using a hole punch and wrapped around with a filter sock. The sample tubes were installed in the bore and the formation allowed to collapse back without any attempt to provide seals between samplers. A 50 mm piezometer (C) was installed to 9 m close (1 m) to the bundle and completed with 50 mm PVC casing. This piezometer was screened between 8 and 9 m.

Two 15 m strings of electrodes were prepared with gold-plated electrodes connected to multi-strand insulated electrical cable. Each string had 15 electrodes terminating at 1 m intervals. Individual electrodes were sealed to the cable insulation using heat shrink material to prevent ingress of aquifer water to the individual strands of the electrical cable that would result in corrosion. Individual electrodes were then taped to the outside of 25 mm diameter PVC tube (electrical conduit). The ends of all the wires were labelled and terminated in a junction box.

The electrode bundles were installed using hollow stem auger techniques and the augers withdrawn allowing the formation to collapse back against the electrodes. An electrode string was installed on either side of the bundled piezometer in a line approximately normal to the groundwater flow direction and 8 m apart (E1 and E2 in Fig. 3). Electrical tomography between the electrode strings could then be used to predict fluid electrical conductivity and the results compared to measured fluid EC obtained by pumping water from the individual piezometers in the bundle, located midway between the electrode strings. A 50 mm bore was installed to 18 m to provide access for geophysical logging. The location (G) is shown in Fig. 3.

The complete installation process took approximately 15 h but required prior construction of the bundles. A fluid EC and water level logger was deployed in piezometer D for 12 months, set to take hourly measurements. Details of the piezometer installations are given in Table 1.

3.3. Water chemistry determinations

Water samples were obtained for each of the bundled piezometer sampling points using a Geopump 2 peristaltic pump. The discharge from the pump was fed to a flow cell where determinations of fluid electrical conductivity (EC), temperature, pH and Eh were made. Purging was continued until the conductivity values were observed to stabilise, at which point the measurements were recorded. Measurements of all unstable parameters (bicarbonate and carbon dioxide) were carried out in the field at the time of sampling using titration methods (APHA, 1992). Samples were filtered using a 40 µm filter and preserved by adding 2 ml acid in the field for cation determination. Measurements were carried out using inductively coupled plasma spectrophotometry (ICP AES) in the laboratory. Samples for anion determinations were filtered in the field, and determination for chloride was carried out by titration (APHA, 1992) in the laboratory. Sulphate was determined as total sulphur by ICP.

3.4. Cross-borehole tomography

A combination of 15 surface electrodes, separated by 1 m, and the 2 sets of 15 borehole electrodes were used for the cross-hole tomography. Measurements were made using the standard Wenner configuration for the 15 surface electrodes. Borehole electrodes were then addressed with a current source (A) attached to the top electrode (1 m depth) in one bore and a current sink (B) in the top electrode in the second bore (Fig. 5). Potential measurements were then made between M1 and N1, M2 and N2, M3 and N3, M4 and N4 and M5 and N5 electrodes.

Table 1
Piezometer installation details

Piezometer	Construction details	Screen depth	Diameter (mm)	Surface elevation (m AHD)
B2	Bundled piezometer—takeouts every 0.5 m to 12 m and then every metre	20 mm screens at each piezometer	5 ID, 8 OD	
G	PVC	17–18	50	
A	PVC	4–5	50	17.624
B	PVC	4–5	50	17.454
C	PVC	4–5	50	15.506
D	PVC	4–5	50	16.412
Culvert outlet at Pond 5 (Fig. 3)	Top of pipe invert			16.256

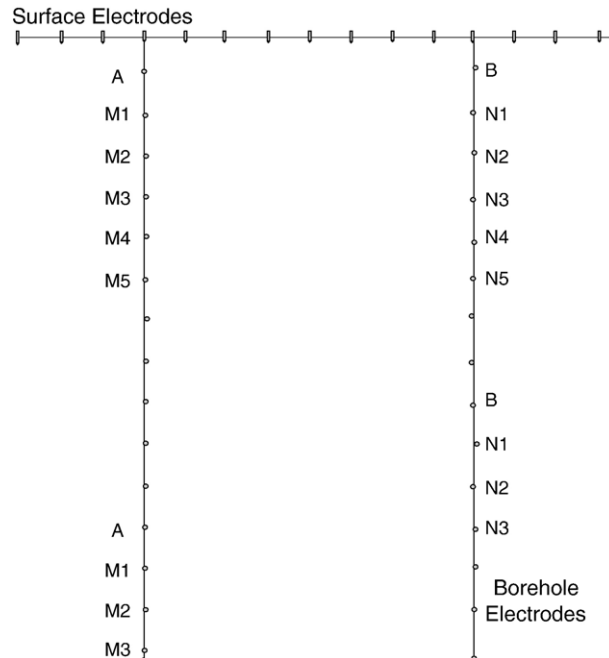


Fig. 5. Electrode configurations used for the cross-borehole tomography.

The current electrodes were then moved down one position and the process repeated until the base of the hole was reached. To maximise paths crossing the space between the 2 bores, measurements were repeated with the current source and sink offset from each other as shown in Fig. 5. No measurements were taken with all 4 electrodes connected to the same bore as these measurements are particularly susceptible to the disturbance of the aquifer represented by the wire and plastic pipe that form the electrode string. Measurements using these electrodes are therefore very noisy.

3.5. Gamma-ray logging

A GEOVISTA logging unit was used to measure gamma-ray activity in piezometers C and G. Measurements were taken every 10 mm at a logging speed of 0.6 m per minute.

4. Results

4.1. Electrical image data

Data for 4 electrical images taken along Line 3 (Fig. 3) are shown in Fig. 6. The period between May 2003 and July 2004 was dry in Sydney with increasing depth to the water table.

4.2. Fluid level and EC measurements in piezometer D

The record of water table depth and fluid EC variation is shown in Fig. 7 for the period March to December 2003. Measurements were logged at 30 min intervals.

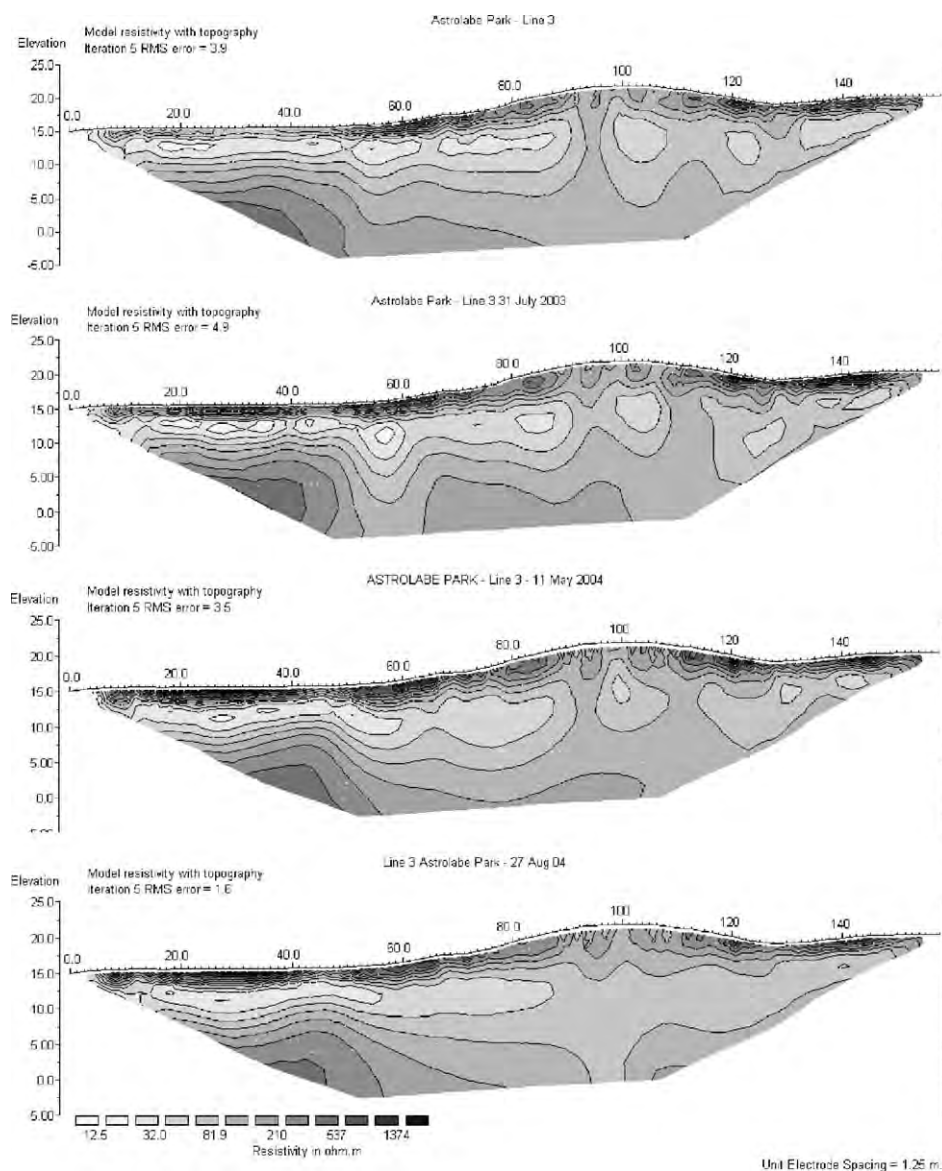


Fig. 6. Repeated electrical image results along line 3—measurements in April and July 2003 and May and August 2004. Note the logarithmic scale used.

4.3. Vertical hydraulic head measurements

A profile of fluid EC measurements at the bundled piezometer taken on 3 occasions is shown in Fig. 8.

4.4. Water chemistry data

Major cation and anion data for the bundled piezometer are presented in Tables 2 and 3.

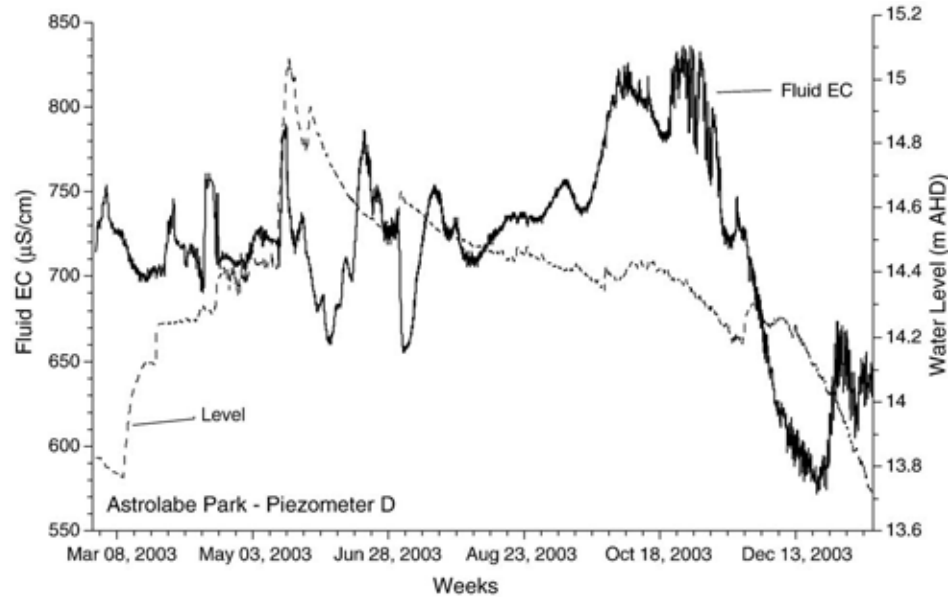


Fig. 7. Water level and fluid EC measurements for piezometer D.

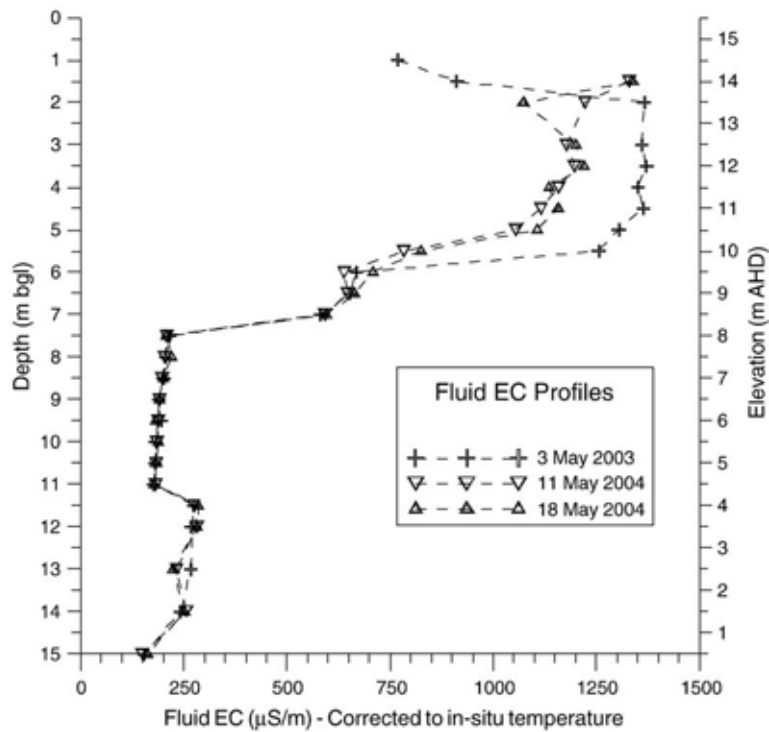


Fig. 8. Profiles of fluid EC measured in the bundled piezometer B2.

Table 2
Field chemistry for bundled piezometer B2 (February 2002)

ID	Depth (m)	Elevation (m AHD)	<i>T</i> (°C)	EC (μS/cm)	pH	Eh (mV)	O ₂ (mg/l)	CO ₂ (mg/l)
A	1.5	1.550	21.5	1429	6.432	−48.3	0.26	242.93
B	2.0	2.050	20.9	1297	6.415	−67.9	0.02	211.25
C	3.0	3.050	20.7	1183	6.517	−83.8	0.04	246.45
D	3.5	3.550	21.3	1183	6.498	−84.1	0.02	230.61
E	4.0	4.050	20.3	1192	6.481	−80.4	0.01	211.25
F	4.5	4.550	19.8	1185	6.448	−73.5	0.01	218.29
G	5.0	5.050	19.6	1125	6.415	−71.8	0.04	253.50
H	5.5	5.550	20.1	1092	6.410	−58.4	0.01	237.65
I	6.5	6.550	20.4	672	6.709	−77.6	0.01	70.42
J	7.0	7.050	20.1	455	6.556	−65.9	0.15	58.09
K	8.0	8.050	20.7	240	5.047	51.4	1.63	116.19
L	9.0	9.050	20.0	210	5.002	−21.5	0.11	123.23
M	10.0	10.050	19.9	209	5.108	−57.9	0.13	149.63
N	11.0	11.050	19.9	230	5.369	4.4	0.29	137.31
O	12.0	12.050	20.2	186	5.254	−26.5	0.21	677.75
P	13.0	13.050	20.0	184	5.154	0.2	0.36	96.82
Q	14.0	14.050	20.0	185	5.182	2.6	0.12	86.26
R	15.0	15.050	20.1	171	5.014	17.5	0.31	80.98

4.5. Geophysical logs

The gamma-ray activity log for piezometer C is shown in Fig. 9. The log of a similar bore drilled approximately 220 m south along the shore of the lake and bore G are also shown.

Table 3
Major ion chemistry for bundled piezometer B2 (February 2002)

ID	Na ⁺	Ca ²⁺	Mg ²⁺	K ⁺	Fe ²⁺	NH ₄ ⁺	HCO ₃ [−]	SO ₄ ^{2−}	Cl [−]	S ^{2−}	NO ₃ [−]
A	77.20	97.50	29.80	31.20	5.15	19.35	776.07	9.90	101.97	0.002	0.00
B	66.30	129.00	32.10	29.60	5.45	28.38	846.85	10.12	65.98	0.000	0.00
C	44.30	119.00	23.70	28.50	2.70	18.38	746.79	10.16	51.98	0.000	0.00
D	44.60	117.00	23.80	28.90	6.30	34.51	758.99	10.03	51.98	0.000	0.00
E	44.50	116.00	23.60	29.10	5.15	27.41	758.99	9.90	51.98	0.007	0.00
F	45.30	115.00	23.40	28.90	3.15	34.51	754.11	9.94	53.98	0.011	5.28
G	45.30	110.00	22.10	27.40	6.65	30.64	688.22	9.28	49.98	0.009	0.00
H	44.60	105.00	22.90	27.50	5.15	21.29	671.13	8.80	47.99	0.007	0.00
I	29.40	72.50	10.97	13.90	14.10	11.29	373.39	11.70	38.99	0.022	10.56
J	27.10	49.40	6.15	9.24	4.30	3.55	219.64	18.44	35.99	0.013	5.72
K	28.90	4.76	4.75	7.47	1.44	4.39	14.64	20.64	57.98	0.163	8.36
L	28.20	2.62	3.10	6.36	0.98	2.80	17.08	13.95	47.99	0.550	11.88
M	29.70	2.62	2.73	6.16	1.53	3.29	26.85	12.19	42.99	0.491	11.88
N	29.50	3.18	3.53	6.61	8.25	3.35	29.29	9.94	45.99	0.485	36.96
O	26.10	3.93	2.83	6.05	1.51	2.83	21.96	21.91	35.99	0.230	15.40
P	24.20	4.20	2.45	6.19	1.67	2.50	19.52	25.21	33.99	0.071	7.92
Q	23.30	4.96	3.38	6.08	1.44	2.63	14.64	26.36	32.99	0.038	15.40
R	19.70	6.53	3.81	5.77	0.69	1.52	14.64	30.18	29.99	0.019	10.12

All results in mg/l.

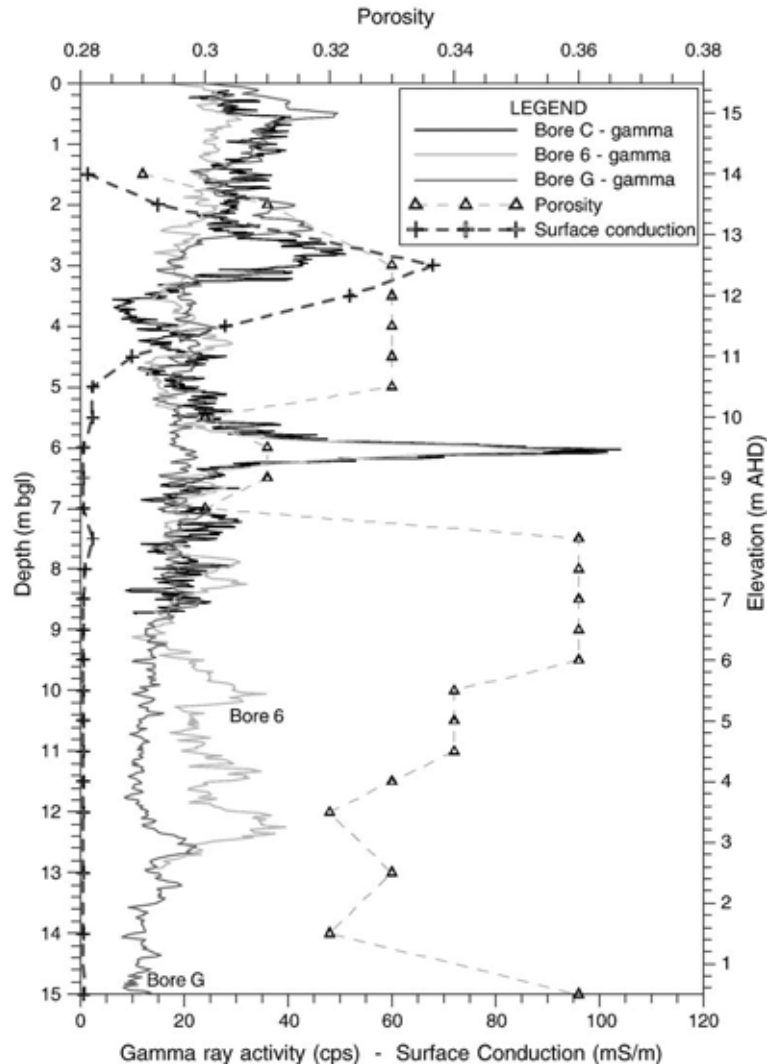


Fig. 9. Gamma-ray log for piezometers C and G. Data bore B6 (220m south along the shore of Pond 5) was included for comparison.

4.6. Cross-borehole tomography

Cross-borehole images for April 2003 and May 2004 are shown in Fig. 10.

5. Discussion and interpretation

The electrical image data (Fig. 6) clearly shows a low resistivity plume of contaminated water discharging from the landfill. The high point along the image (100 m) is close to the old tipping station where material was emptied from trucks and moved into the landfill. The edge of the landfill is difficult to determine as there is no longer any surface evidence. Based upon

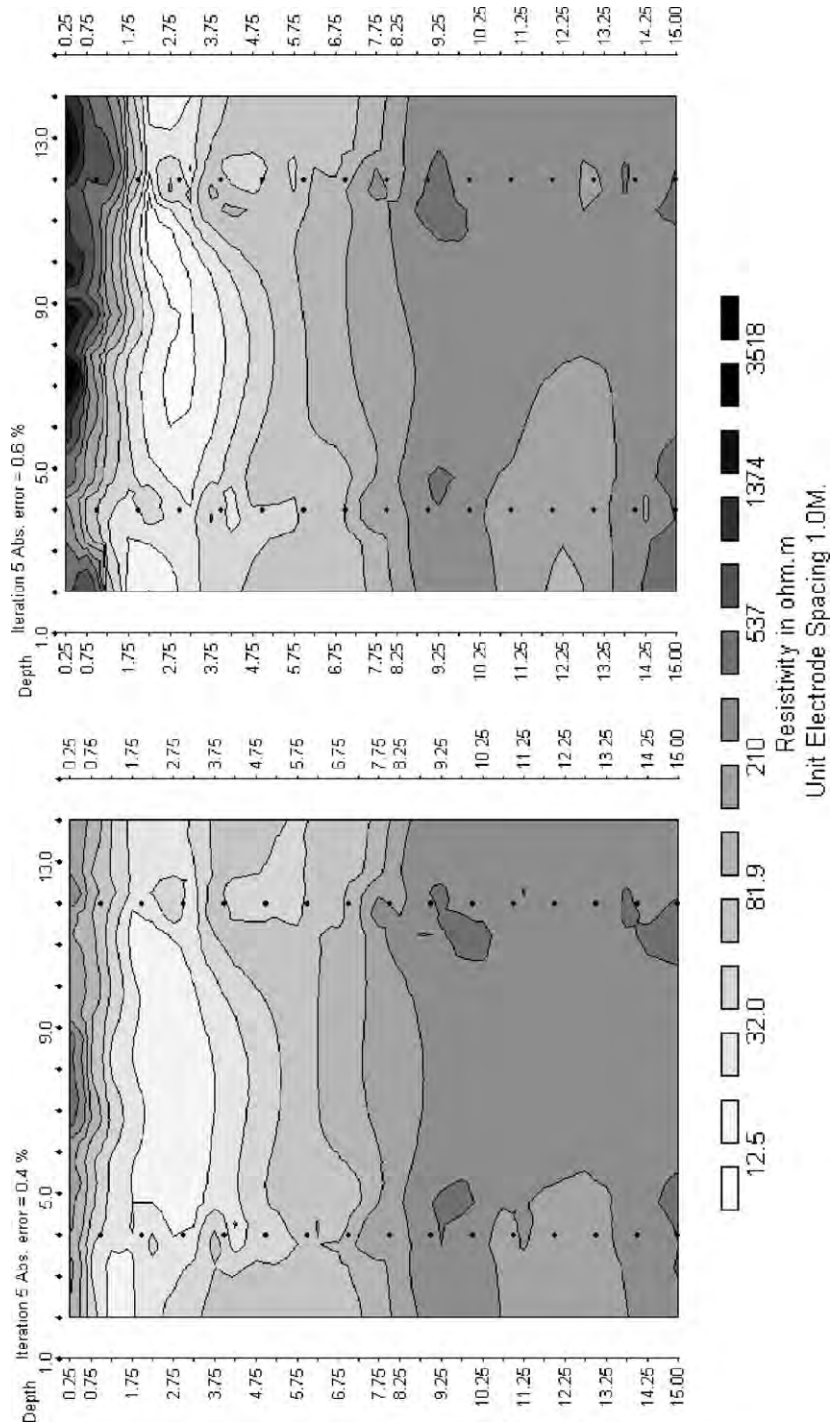


Fig. 10. Cross-borehole images between bores E1 and E2 for May 2003 and May 2004. Note the logarithmic scale used.

the indication of higher resistivity (clean groundwater) ending at approximately 50 m along the image line, it is expected that filling extended from 50 m to more than 160 m along the image line and to an approximate base of 12 m AHD. This would give a total thickness of landfill beneath the highest point of approximately 10 m, with the base of the landfill permanently saturated. The high resistivity close to the ground surface represents partially saturated sand/landfill above the water table. It is of interest to note that the low resistivity contaminant plume does not extend all the way under the landfill but appears to develop at the edge and to move into the uncontaminated part of the aquifer. Some segmentation of the plume is evident.

The cross-borehole image (Fig. 10) is located 18 m along the surface image line. It was not possible to extend the surface image further to the left due to the presence of the old culvert beneath the causeway. The causeway now forms part of a golf course. The cross-borehole images show a clear base to the zone of low resistivity material with high resistivity material beneath.

The fluid EC data obtained by pumping the bundled piezometer tubes on the same day as the cross-borehole images were measured is shown in Table 4. The field data is automatically corrected to 25 °C in many common field EC meters. The correction factor is 2.2% of fluid EC for every 1 °C of temperature. As this is significant, and the relationship between ground

Table 4
Resistivity model results

Depth (m bgl)	Field EC ($\mu\text{S}/\text{cm}$)	Bulk resistivity from x-bore image ($\Omega\text{ m}$)	Estimated EC using $\sigma_r=0.56$, $\phi=0.36$, and $m=1.51$ ($\mu\text{S}/\text{cm}$)	Modelled resistivity based on Eq. (3) ($\Omega\text{ m}$)	Surface conduction term, σ_r (mS/m)	Porosity, ϕ	m
1.0							
1.5	1338	75	593	75	1.5	0.29	2.0
2.0	1073	29	1581	29	15	0.31	1.51
2.5		17	2720				
3.0	1200	12	3865	12	68	0.33	1.51
3.5	1219	14	3310	14	52	0.33	1.51
4.0	1136	20	2308	20	28	0.33	1.51
4.5	1157	30	1528	30	10	0.33	1.51
5.0	1106	42	1083	42	2.4	0.33	1.51
5.5	825	61	736	61	2.4	0.30	1.51
6.0	708	82	539	82	0.56	0.31	1.51
6.5	664	101	432	101	0.56	0.31	1.70
7.0	594	125	344	127	0.56	0.30	1.77
7.5	208	150	281	150	2.4	0.36	1.51
8.0	220	177	234	178	0.8	0.36	1.51
8.5	200	205	198	203	0.56	0.36	1.51
9.0	192	228	175	227	0.56	0.36	1.61
9.5	183	245	161	244	0.56	0.36	1.65
10.0	188	253	155	253	0.56	0.34	1.63
10.5	183	252	156	253	0.56	0.34	1.60
11.0	179	242	163	242	0.56	0.34	1.52
11.5	285	232	172	232	0.56	0.33	1.91
12.0	281	220	183	220	0.56	0.32	1.77
13.0	225	217	186	217	0.56	0.33	1.58
14.0	251	225	178	224	0.56	0.32	1.68
15.0	162	235	169	236	0.68	0.36	1.51

conductivity and fluid conductivity must be estimated at the temperature of the ground, this correction is removed using Eq. (1).

$$\sigma_{\text{water}} = \sigma_{\text{water}25}(1 - 0.022(25 - T)) \quad (1)$$

where: T is the temperature of the water ($^{\circ}\text{C}$).

The relationship between bulk EC and fluid EC in a clay free sand has been investigated by many workers and good reviews are provided by several authors (Olhoeft, 1985; Kelly, 1994; and more recently by Friedman, 2005). Archie (1942) first investigated this relationship for clay free sand saturated with brine. Eq. (2) shows this relationship expressed in terms of conductivities.

$$\sigma_b = \sigma_w \phi^m \quad (2)$$

where: σ_b is the bulk electrical conductivity (mS/m), σ_w is the fluid electrical conductivity (mS/m), ϕ is the formation porosity and m is the geometry factor (dimensionless).

Model and field studies (Kelly, 1994; Acworth and Dasey, 2003) have shown that the Archie relationship is not valid in fine-grained well-sorted aeolian sands for fluid EC values of less than approximately 1000 $\mu\text{S}/\text{cm}$ because the conduction along grain surfaces (σ_r) becomes a more significant proportion of total conductivity at low fluid EC levels. The Hanai-Bruggeman effective medium theory (EMT) (Bussain, 1983) is used to accommodate the mixture of fluid and grain surface conduction. The EMT (Eq. (2)) collapses to Archie's relationship where the grain surface conduction approaches zero.

$$\sigma_b = \sigma_w \phi^m ((1 - \sigma_r/\sigma_w)/(1 - \sigma_r/\sigma_b))^m \quad (3)$$

Kelly (1994) carried out extensive laboratory studies of clean sands from the Botany Bay aquifer and found that the average terms in the EMT equation for Botany sand could be represented by a porosity (ϕ) of 0.36, a geometry factor (m) of 1.51 and a grain surface conduction (σ_r) of 0.56 mS/m. The samples were taken from a location close to Astrolabe Park on the far side of Pond 5. Kelly (1994) also experimented with mixtures of glass beads (of the same size as Botany Sand) and clay and found that the surface conduction term rises to 43 mS/m for a 10% montmorillonite clay mix with glass beads. The cementation factor (m) varied between 1.49 and 1.56 for the sands but increased to a maximum of 2.09 for a 10% kaolinite and glass bead mix.

The bulk resistivity data from the inversion of the cross-borehole image have been used in Eq. (3) to predict the fluid EC, using the average porosity, surface conduction and geometry factors reported by Kelly (1994). The measured fluid EC field data and the predicted data (using Eq. (3)) are shown in Table 4. It is clear that there is an over prediction of fluid EC in the area of the plume.

Examination of the gamma-ray activity log (Fig. 9) shows that the activity of the upper part of the profile is significantly higher than the background value of approximately 12 cps in the lower part of the aquifer. This background value is typical of uncontaminated Botany Sands measured in other bores in the area. Elevated gamma-ray activity towards the top of the bore is probably related to silt or organics transported from the landfill or possibly to the residue of sand washing during sand mining. The presence of this very fine-grained material with an enhanced surface area will increase the surface conduction term in Eq. (3).

Further solutions to Eq. (3) are shown in Table 4 where the fluid EC (σ_w) is maintained at the field sampled values and the bulk conductivity (σ_b) is held equal to the value derived from the cross-borehole tomography. Solutions have been obtained by varying the porosity (ϕ), the

surface conduction term (σ_r) and the geometry factor (m). Increasing either the porosity, or the surface conduction will increase the bulk conductivity whereas increasing the value of the geometry factor decreases the bulk conductivity. Some guidance on the range of possible values is available however. The porosity is unlikely to be greater than 0.36 or to be much less than 0.30 (Kelly, 1994). The surface conduction is not lower than 0.56 mS/m and will be significantly greater in silt and clay. The geometry factor will be close to 1.5 for well-rounded sand grains but increase to 2.0 for flat plate structures, typical of clays. The gamma-ray activity log also provides an indication of surface conduction. Typical clay free sands in the Botany aquifer have an activity of approximately 12 cps (Fig. 9). Values above this background value will normally (see below) indicate silt or clay. A trial and error approach yields a set of solutions for porosity, surface conduction and geometry factors as shown in Table 4 and Fig. 9. These are not unique solutions but as they are well constrained by field data and experimentation, they do indicate sensible ranges for the data described. A constrained non-linear minimum least-squares procedure could also have been used.

The sharp gamma anomaly shown at 6 m in Fig. 9 requires comment. The magnitude of this anomaly is much greater than that of silt/peat bands and it occurs at an elevation of 9 m AHD, which is deeper than the greatest depth of sand mining. Drilling has proven that there is no clay at this depth. The possibility exists that it represents a thin zone of heavy mineral sands with the gamma activity provided by the thorium in monzonite. Heavy mineral sands are mined at many locations along the New South Wales coast but have not been reported in Botany Bay as far as the authors are aware. The presence of mineral sands indicates an old beach environment, which, if correct, would indicate a high stand of sea level at the beginning of the Holocene. The log for another hole (bore 6) some 220 m along the lakeside is shown in Fig. 9 and it is seen that the anomalies overlap almost exactly. The anomaly is not present at the much closer bore G.

A water quality sample has been obtained by suction from the bundled piezometer at 1.5 m. Therefore the water table occurs between 1.0 and 1.5 m and the profile is fully saturated below this depth. Values of porosity, m and surface conduction (Table 4) have been derived for depths beneath the water table. Modelling has indicated a surface conduction of as high as 68 mS/m at 3.0 m. This is in agreement with the peak in gamma-ray activity at this depth (2.8 m). Further work is underway to determine the significance of this anomaly.

The fluid EC anomaly is clearly the result of increased salts in solution. Water at the base of the bundle (15 m) has fluid EC values of $<200 \mu\text{S/cm}$ (as clean as Sydney tap water). Water in the centre of the plume has fluid EC values of approximately $1300 \mu\text{S/cm}$. The chemical distribution of major ions is shown in Fig. 11 supporting the identification of a contaminant plume extending from 1 m to 7 m depth with a centre of concentration at approximately 3.5 m. This pattern very closely matches the distribution of bulk resistivity shown by the cross-hole images. The major ion chemistry from the bundled piezometer is shown on the fields of a Piper diagram (Domenico and Schwartz, 1997) in Fig. 12. The plume waters plot in a group dominated by calcium and bicarbonate as the principle cation and anion. These waters are distinctly different to the sodium and chloride rich water in the aquifer beneath the plume. The elevated bicarbonate is produced by the end products of microbial decay in the plume. The elevated calcium could be caused by old mortar from brickwork dumped in the landfill. The sodium chloride dominated regional groundwater beneath the plume is typical of Botany Sands water that is recharged by rainfall with significant contamination by sea spray (Jorstad et al., 2004).

The centres of lowest resistivity in the contaminant plume shown in Fig. 6 appear to change with time. This indicates that the fluid EC in the plume waxes and wanes rather than representing a constant concentration of ions in the water. As the landfill is not lined or capped and

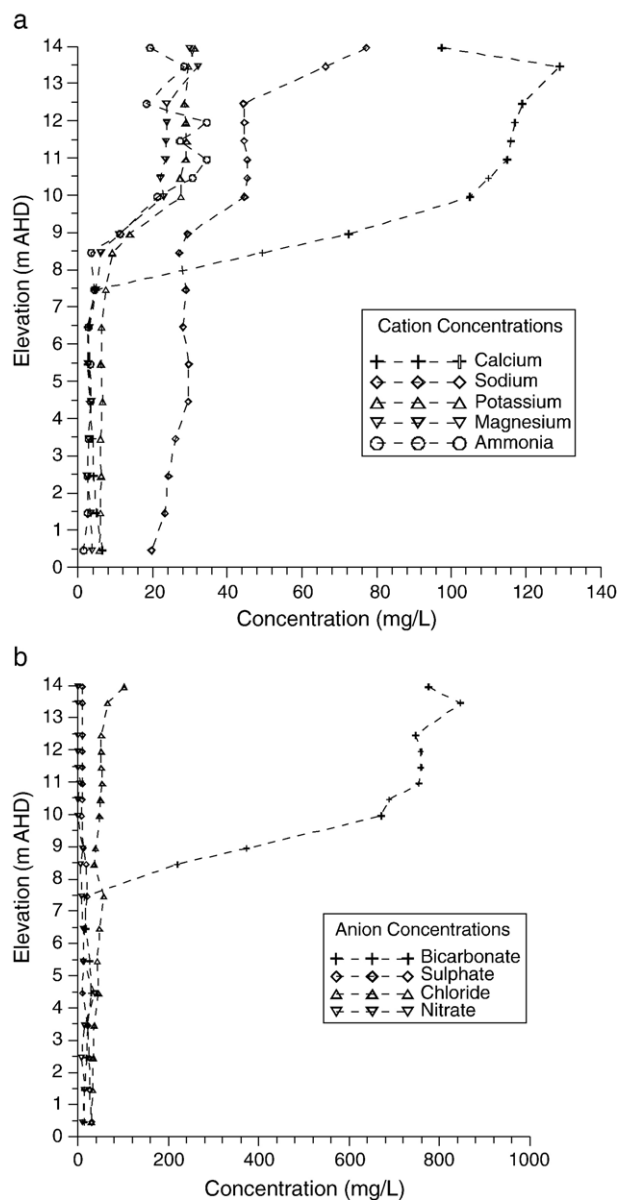


Fig. 11. Major ion chemistry for the bundled piezometer B2.

groundwater moves through the base of the landfill, it seems probable that the chemical and biological activity in the landfill may vary in intensity producing a variable source load to the plume. Monitoring of fluid EC at piezometer D (Fig. 7) shows a fluid EC change between 575 and 825 $\mu\text{S}/\text{cm}$. The fluid EC in Fig. 7 shows that rapid changes occur that are not directly linked to changes in hydraulic head. This would indicate separate bodies of water with different fluid EC being moved through the groundwater system. The variation in fluid EC is also noted at B2 where the bundled piezometers have been sampled at various times.

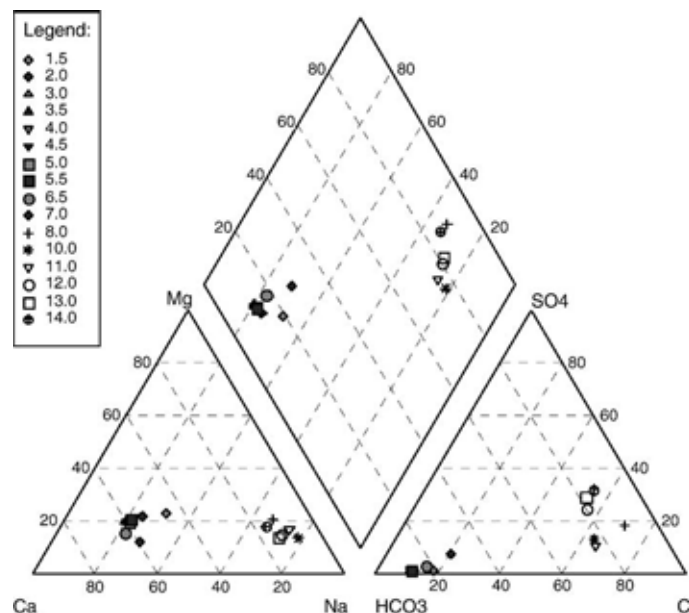


Fig. 12. Piper diagram showing the plume chemistry and the regional groundwater as two clearly different water types in bundled piezometer B2.

Given the standard values of ϕ (0.36) and σ_r (0.56), the fluid EC range noted at D would give a bulk resistivity variation of 77 to 54.6 Ω m. This change is similar to that seen in images conducted through that piezometer along Line 4 (not shown). Repeated monitoring is required to accurately identify the change with more regular repeats of the image carried out in the future to investigate this further.

6. Conclusions

The strong correlation between the complementary data afforded by the hydrochemical and geophysical investigation techniques facilitates the development of a detailed, well-constrained conceptual model of contaminant transport at the site. Measurement of groundwater electrical conductivity values at the bundled piezometers provides direct evidence of the presence and vertical extent of landfill leachate impacted groundwater in the otherwise pristine aquifer, with steep chemical gradients marking the upper and lower boundaries of the plume. The use of groundwater EC values as a general indicator parameter for the leachate plume distribution is supported by correlation with major cation and anion profiles for these piezometers.

Surface electrical resistivity surveys employing the Wenner array method confirm the presence of a distinct electrical anomaly in the shallow portion of the aquifer resulting from the presence of the leachate plume. The vertical distribution of formation electrical properties closely approximates the observed groundwater EC profiles, while on a broader scale the transect images provide significant insight into the irregular, segmented nature of contaminant distribution and transport between the landfill and the pond.

The results of the cross-borehole tomography survey show a strong correlation with the results of both the surface resistivity transect and the groundwater chemistry profiles from the

bundled piezometer, demonstrating the efficacy of this investigation technique to provide a basis of reference between detailed groundwater chemistry data and large-scale surface resistivity transects. The effective medium theory is shown to satisfactorily relate fluid EC to the bulk EC in low salinity waters. Solution of the EMT equation when constrained by known values of σ_b and σ_w allow reasonable estimates of ϕ and σ_r to be made.

The integration of hydrochemical and geophysical investigation techniques has resulted in a robust, rapid and relatively inexpensive methodology for the accurate delineation of a landfill leachate plume in a heterogeneous sandy aquifer, and has provided considerable insight into the nature of contaminant transport between the landfill and the pond.

Acknowledgements

Philip Hitchcock is acknowledged for his diligent search of Botany Council records that allowed the past record of sand mining and land filling to be detailed. Mark Groskops of the Water Research Laboratory in the School of Civil and Environmental Engineering at UNSW was responsible for the fabrication and installation of the piezometer and electrode bundles.

References

- Acworth, R.I., 2001. Physical and chemical properties of a DNAPL contaminated zone in a sand aquifer. *Quarterly Journal of Engineering Geology and Hydrogeology* 34.1, 85–98.
- Acworth, R.I., Dasey, G.R., 2003. Mapping of the hyporheic zone around a tidal creek using a combination of borehole logging, borehole electrical tomography and cross-creek electrical imaging. *Hydrogeology Journal* 11.3, 368–377.
- Andreottola, G., Cannas, P., 1992. Chemical and biological characteristics of landfill leachate. In: Christensen, T.H., Cossu, R., Stegmann, R. (Eds.), *Landfilling of Waste: Leachate*, Chapter 2.4. E and FN Spon, London, pp. 65–88.
- APHA, 1992. *Standard Methods for the Examination of Water and Wastewater*, 18th edn. American Public Health Association.
- Archie, G.E., 1942. The electrical resistivity log as an aid in determining some reservoir characteristics. *Transactions of the American Institute of Mining and Metallurgical Engineers* 146, 54–62.
- Back, W., Baedecker, M.J., 1989. Chemical hydrogeology in natural and contaminated environments. *Journal of Hydrology* 106, 1–28.
- Baedecker, M.J., Apgar, M.A., 1984. Hydrogeochemical studies at a landfill in Delaware. *Groundwater Contamination, Studies in Geophysics*, Chapter 10. National Academy Press, Washington, D.C., pp. 127–138.
- Baedecker, M.J., Back, W., 1979a. Hydrogeological processes and chemical reactions at a landfill. *Ground Water* 17 (5), 429–437.
- Baedecker, M.J., Back, W., 1979b. Modern marine sediments as a natural analog to the chemically stressed environment of a landfill. In: Back, W., Stephenson, D.A. (Eds.), *Contemporary Hydrogeology—The George Burke Maxey Memorial Volume*, *Journal of Hydrology*, vol. 43, pp. 393–414.
- Bjerg, P.L., Christensen, T.H., 1992. Spatial and temporal small-scale variation in groundwater quality of a shallow sandy aquifer. *Journal of Hydrology* 131, 133–149.
- Bussain, A.E., 1983. Electrical conductance in a porous medium. *Geophysics* 48 (9), 1258–1268.
- Chian, E.S.K., DeWalle, F.B., 1977. Characterization of soluble organic matter in leachate. *Environmental Science and Technology* 11 (2), 158–163.
- Christensen, T.H., 1992. Attenuation of leachate pollutants in groundwater. In: Christensen, T.H., Cossu, R., Stegmann, R. (Eds.), *Landfilling of Waste: Leachate*, Chapter 4.21. E and FN Spon, London, pp. 441–483.
- Christensen, T.H., Kjeldsen, P., Bjerg, P.L., Jensen, D.L., Christensen, J.B., Baun, A., Albrechtsen, H.-J., Heron, G., 2001. Biogeochemistry of landfill leachate plumes. *Applied Geochemistry* 16, 659–718.
- Cozzarelli, I.M., Suflita, J.M., Ulrich, G.A., Harris, S.H., Scholl, M.A., Schlottmann, J.L., Jaeschke, J.B., 1999. *Biogeochemical processes in a contaminant plume downgradient from a landfill*, Norman, Oklahoma. U.S. Geological Survey Water Resources Investigations Report # 99-4018C Reston, VA.
- Domenico, P.A., Schwartz, W., 1997. *Physical and Chemical Hydrogeology*, 2nd edition. John Wiley and Sons, Inc., New York.

- Ehrig, H.-J., 1983. Quality and quantity of sanitary landfill leachate. *Waste Management and Research* 1, 53–68.
- Ehrig, H.-J., 1988. Water and element mass balances of landfills. In: Baccini, P. (Ed.), *The Landfill*, vol. 20. Springer, Berlin, pp. 83–116. *Lecture Notes in Earth Sciences*.
- Friedman, S.P., 2005. Soil properties influencing apparent electrical conductivity: a review. *Computers and Electronics in Agriculture* 46, 45–70.
- Jankowski, J., Acworth, R.I., 1997. Development of a contaminant plume from a municipal landfill: redox reactions and plume variability. In: et al. (Eds.), *Proceedings of the XXVIII Congress of the International Association of Hydrogeologists Congress on Groundwater in the Urban Environment: Problems, Processes and Management*, Nottingham, UK, 21–27 September 1997, vol. 1. Balkema, Rotterdam, pp. 439–444.
- Jankowski, J., Beck, P., 2000. Aquifer heterogeneity: hydrogeological and hydrochemical properties of the Botany Sands aquifer and their impact on contaminant transport. *Australian Journal of Earth Sciences* 47, 45–64.
- Jorstad, L., Jankowski, J., Acworth, R.I., 2004. *Environmental Geology* 95 (2), 263–272.
- Kehew, A.E., Passero, R.N., 1990. pH and redox buffering mechanisms in a glacial drift aquifer contaminated by landfill leachate. *Ground Water* 28 (5), 728–737.
- Kelly, B., 1994. Electrical properties of sediments and the geophysical detection of ground water contamination. PhD thesis, School of Civil and Environmental Engineering, University of New South Wales. 146 pp. (unpublished).
- Kimmel, G.E., Braids, O.C., 1974. Leachate plumes in a highly permeable aquifer. *Ground Water* 12 (6), 388–392.
- Kjeldsen, P., Bjerg, P.L., Rügge, K., Christensen, T.H., Pedersen, J.K., 1998. Characterization of an old municipal landfill (Grindsted, Denmark) as a groundwater pollution source: landfill hydrology and leachate migration. *Waste Management Research* 16 (1), 14–22.
- Lambeck, K., Nakada, M., 1990. Late Pleistocene and Holocene sea-level change along the Australian coast. *Global and Planetary Change* 3, 143–176.
- Lavitt, N., Acworth, R.I., Jankowski, J., 1997. Vertical hydrochemical zonation in a coastal section of the Botany Sands aquifer. *Hydrogeology Journal* 5, 64–74.
- Loke, M.H., 2001. Rapid 2D and 3D Resistivity and IP inversion using the least squares method—RES2DINV Ver 3.4 Manual. Published on the world wide web at www.geoelectrical.com.
- Lyngkilde, J., Christensen, T.H., 1992. Redox zones of a landfill leachate pollution plume (Vejen, Denmark). *Journal of Contaminant Hydrology* 10, 273–289.
- MacFarlane, D.S., Cherry, J.A., Gillham, R.W., Sudicky, E.A., 1983. Migration of contaminants in groundwater at a landfill: a case study: 1. Groundwater flow and plume delineation. *Journal of Hydrology* 63, 1–29.
- Mirecki, J.E., Parks, W.S., 1994. Leachate geochemistry at a municipal landfill, Memphis, Tennessee. *Ground Water* 32 (3), 390–398.
- Olhoeft, G., 1985. Low frequency electrical properties. *Geophysics* 50.12, 2492–2503.
- Ronen, D., Magaritz, M., Gvirtzman, H., Garner, W., 1987. Microscale chemical heterogeneity in groundwater. *Journal of Hydrology* 92, 173–178.
- Smith, R.L., Harvey, R.W., Le Blanc, D.R., 1991. Importance of closely spaced vertical sampling in delineating chemical and microbiological gradients in groundwater studies. *Journal of Contaminant Hydrology* 7, 285–300.



Springer

Dear Author:

Please find attached the final pdf file of your contribution, which can be viewed using the Acrobat Reader, version 3.0 or higher. We would kindly like to draw your attention to the fact that copyright law is also valid for electronic products. This means especially that:

- You may print the file and distribute it amongst your colleagues in the scientific community for scientific and/or personal use.
- You may make your article published by Springer-Verlag available on your personal home page provided the source of the published article is cited and Springer-Verlag and/or other owner is mentioned as copyright holder. You are requested to create a link to the published article in Springer's internet service. The link must be accompanied by the following text: "The original publication is available at springerlink.com". Please use the appropriate DOI for the article. Articles disseminated via SpringerLink are indexed, abstracted and referenced by many abstracting and information services, bibliographic networks, subscription agencies, library networks and consortia.
- Without having asked Springer-Verlag for a separate permission your institute/your company is not allowed to place this file on its homepage.
- You may not alter the pdf file, as changes to the published contribution are prohibited by copyright law.
- Please address any queries to the production editor of the journal in question, giving your name, the journal title, volume and first page number.

Yours sincerely,

Springer-Verlag

Analysis of the distribution of inorganic constituents in a landfill leachate-contaminated aquifer Astrolabe Park, Sydney, Australia

L. B. Jorstad · J. Jankowski · R. I. Acworth

Abstract An investigation was conducted at Astrolabe Park landfill, a decommissioned municipal landfill in Sydney, Australia, to assess the physical and chemical processes affecting the distribution of inorganic constituents in the leachate plume. The plume is migrating from the landfill towards a groundwater-fed pond into which leachate-impacted groundwater discharges. Borehole geophysical logging and depth-discrete groundwater sampling were used to delineate the distribution of the leachate plume along two groundwater flow paths between the landfill and the shore of the pond. Borehole geophysical logs indicate a strong correlation between bulk and fluid electrical conductivity (EC) values, and help to identify small-scale heterogeneities that comprise a major constraint on contaminant transport within the aquifer. Variations in the distribution of several indicator parameters (EC, HCO_3^- , pH, Eh, NH_4^+ / NO_3^- , $\text{S}^{2-}/\text{SO}_4^{2-}$) are used to assess the dominant processes affecting contaminant distribution along the flow path, including mixing of fresh and contaminated groundwater, oxidation/reduction reactions and ion exchange.

Keywords Leachate plume · Landfill · Hydrogeochemistry · Vertical monitoring · Geophysics

Introduction

Disposal of waste in landfills is an integral part of waste management strategies around the world. The production of leachate as a by-product of organic and inorganic decomposition in landfills poses a serious threat if released to the environment (Kimmel and Braids 1974; Baedecker and Back 1979; Arneeth and others 1989; Kjeldsen 1993; Jankowski 1997). Andreottola and Cannas (1992) describe leachate pollution as the result of a mass transfer process between the waste and leaching water that has infiltrated into the waste layers. Physical, chemical and microbial processes transfer pollutants from the waste material to the infiltrating water, resulting in a contaminated liquid containing high concentrations of organic and inorganic contaminants. If the leachate is released into the underlying aquifer, it forms a complex contaminant plume that fundamentally alters the chemical properties of the aquifer (Baedecker and Back 1979; Nicholson and others 1983; Lyngkilde and Christensen 1992; Bjerg and others 1995; Jankowski 1997; Jankowski and Acworth 1997; Ludvigsen and others 1998; Cozzarelli and others 1999; Christensen and others 2001). For this study, groundwater samples were collected from four multi-level piezometers that comprise the endpoints of two parallel groundwater flow paths between the Astrolabe Park landfill and a groundwater-fed pond located hydraulically downgradient from the landfill (Lachlan Pond No.5). Several groundwater quality parameters and inorganic elements (EC, HCO_3^- , pH, Eh, $\text{NH}_4^+/\text{NO}_3^-$, $\text{S}^{2-}/\text{SO}_4^{2-}$) are examined to assess the variations in contaminant concentrations and spatial distribution between the landfill and the pond.

Environmental setting

The Astrolabe Park landfill is approximately 6 km south of the Sydney central business district (CBD) in the suburb of Daceyville (Fig. 1). The site is within the northern portion of the Botany Basin, an erosional depression in the Triassic Hawkesbury Sandstone bedrock (Albani 1981; Griffin 1963; Rickwood 1998). The Botany Basin consists of several deep, steep-sided paleochannels incised into the Triassic Hawkesbury Sandstone bedrock that were filled with a sequence of Quaternary sedimentary deposits that comprise the Botany Sands aquifer (Fig. 1).

Received: 24 August 2003 / Accepted: 16 September 2003
Published online: 24 February 2004
© Springer-Verlag 2004

L. B. Jorstad (✉) · J. Jankowski
UNSW Groundwater Group, School of Biological,
Earth and Environmental Sciences, University of New South Wales,
2052 Sydney, N.S.W., Australia
E-mail: lange.jorstad@erm.com
Tel.: +61-2-8584-8888
Fax: +61-2-8584-8800

R. I. Acworth
UNSW Groundwater Group, School of Civil and Environmental
Engineering, Water Research Laboratory, King Street,
2093 Manly Vale, N.S.W., Australia

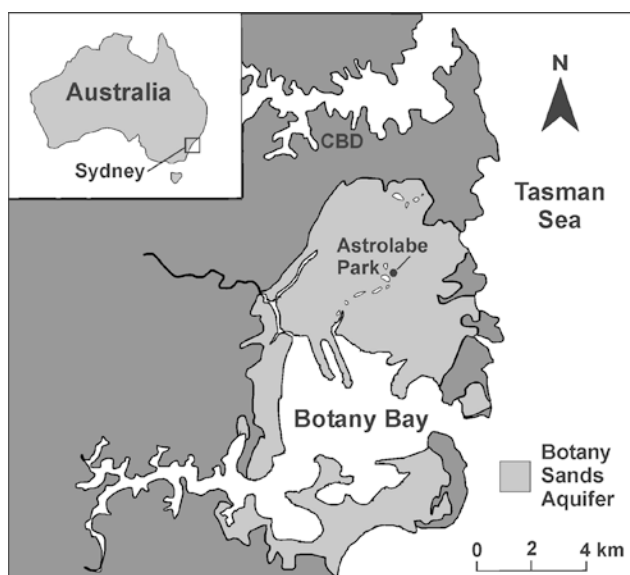


Fig. 1

Site location and extent of the Botany Sands aquifer

The sedimentary sequence observed in the Botany Sands aquifer represents depositional environments ranging from marine to terrestrial. From most recent to oldest, this sedimentary sequence has been divided into four units (Albani 1981): recent marine deposits (Unit 1); aeolian deposits (Unit 2); fresh to brackish lagoon deposits (Unit 3); and marine/estuarine deposits (Unit 4). In the vicinity of the Astrolabe Park landfill, the aeolian deposits of Unit 2 comprise the most significant water bearing formation.

Site description

The study site is comprised of the Astrolabe Park landfill (which was completed as a sports field upon decommissioning), a portion of the adjacent Eastlakes Golf Course, and a portion of the eastern shore of Lachlan Pond No. 5 (Fig. 2). The site is underlain by an approximately 30-m-thick deposit of predominantly aeolian quartz sand comprising the upper sequence of the Quaternary Botany Sands aquifer (Unit 2, discussed in the previous section). The upper portion of the aquifer in this area is unconfined, while the occurrence of silty, clayey and peaty lenses creates semi-confined conditions in the lower portion of the aquifer. On a regional scale these lenses are discontinuous, but they are locally extensive in the vicinity of Astrolabe Park and comprise a significant constraint on groundwater flow and solute transport in the shallow aquifer. The landfill is located up hydraulic gradient from a series of groundwater-fed ponds known as the Lachlan Ponds. Mining of heavy mineral sands occurred at this site using a floating suction dredge between 1964 and 1973; this is believed to have extended to a maximum depth of 17 m below the original ground surface (Hitchcock unpublished data 1991). Following the conclusion of the mining activities, the water-filled sand pits were backfilled with solid

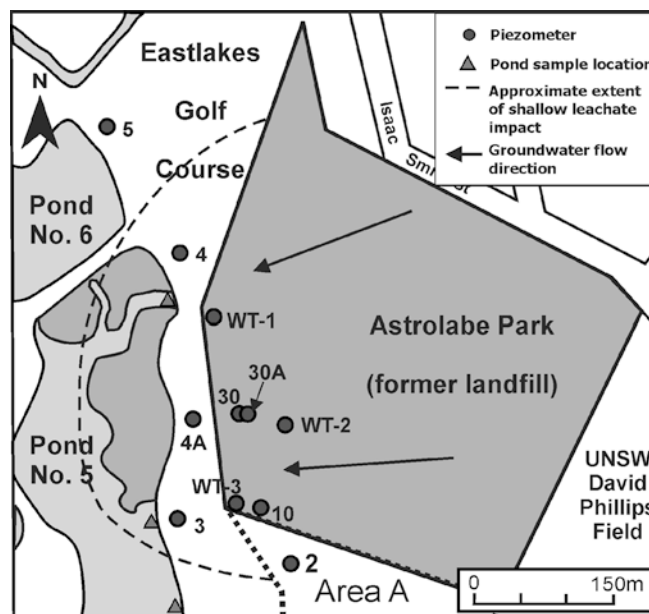


Fig. 2

Location of multi-level piezometers in the vicinity of Astrolabe Park

wastes of mostly domestic origin. The total volume of waste deposited was approximately $2.04 \times 10^5 \text{ m}^3$ (Hitchcock unpublished data 1991). The lateral extent of the landfill was estimated to be 4.7 ha, based upon examination of a series of aerial photographs, resulting in a calculated average waste thickness of approximately 4.35 m.

The regional groundwater flow direction within the Botany Sands aquifer is generally towards the south-southwest. However, in the vicinity of Astrolabe Park the shallow, unconfined groundwater flow is controlled by local topography and flows approximately in a west-southwest direction, as determined in this and previous investigations (Jankowski and Knight unpublished data 1991; Hitchcock unpublished data 1991).

Landfill processes and leachate production: a brief overview

Degradation and stabilization of waste in landfills has been observed to occur according to an ordered succession of physical, chemical and biological processes over time, as described in detail by Ehrig (1983), Andreottola and Cannas (1992) and Qasim and Chiang (1994). These include an initial, short-lived aerobic degradation phase followed by three phases of anaerobic degradation: (I) acid-fermentation, (II) intermediate anaerobiosis, and (III) methanogenesis. Each stage represents a progression in the process and rate of decomposition of organic matter present in the wastes, and the resultant leachate produced can be identified by characteristics indicative of each stage.

In general, the organic waste material is ultimately reduced to $\text{CO}_{2(\text{aq})}$, CH_4 , and H_2O (Baedecker and Apgar 1984; Christensen 1992; Ehrig 1983). In the typical pH range of 6.0–7.0 for methanogenic phase leachate the $\text{CO}_{2(\text{aq})}$ is converted to HCO_3^- (Baedecker and Back 1979; Ehrig 1983; Nicholson and others 1983). The amount of HCO_3^- introduced to the groundwater system through the degradation of organic matter can be very large; for leachate plumes in low salinity aquifers containing a low percentage of carbonate material, HCO_3^- is an excellent indicator of the plume location and relative concentration.

The various populations of microbes responsible for the degradation of organic matter in landfills utilize oxygen, or other oxygen-bearing species in the absence of free oxygen, to convert organic compounds to the simple end-members $\text{CO}_{2(\text{aq})}$, CH_4 and H_2O (Baedecker and Apgar 1984; Andreottola and Cannas 1992; Ehrig 1983). Anaerobic microbial oxidation of organic matter results in the depletion of solid and dissolved oxidants in the aquifer system and accumulation of soluble reduced redox species in groundwater, along with a resultant decrease in the redox potential (measured in the field as Eh) of the groundwater. A comparison of the distribution of oxidised and reduced redox species provides insight into the specific redox processes occurring within the landfill and leachate plume.

It should be noted that while field measurement of Eh values may provide a relative indication of changing redox conditions within a groundwater system, they are not suitable for the determination of the specific redox processes occurring within the system (Baedecker and Cozzarelli 1992; Lindberg and Runnells 1984).

Materials and methods

Borehole geophysical logging

Borehole geophysical investigation techniques were used to provide an additional degree of confidence to the interpretation of groundwater chemistry and site-scale stratigraphy. The geophysical program consisted of borehole bulk electrical conductivity and gamma emission logging. The bulk conductivity logs are used to provide a detailed assessment of the leachate plume location and the gamma logs are intended to aid in the delineation of the subsurface stratigraphy.

The field equipment consists of a Geonics EM39 geophysical logging system, which includes an electrically powered variable speed winch with 100 m of cable. The cable passes over a tripod-mounted pulley system that comes equipped with a shaft encoder to measure the logging speed. The individual sondes used for performing the borehole measurements are attached to the end of the cable and lowered down the borehole at a predetermined speed. The signal from the sonde is recorded using a DOS program on a portable computer, which allows for a real time plot of the data to be generated (McNeill 1990).

The EM-39 sonde used for borehole measurement of bulk electrical conductivity consists of a three-coil induction

system. A transmitter coil in the sonde induces eddy currents in the formation surrounding the bore, which in turn generate a magnetic field that is measured by a coaxial receiver coil in the sonde. The intensity of the generated magnetic field is directly proportional to the electrical conductivity of the formation. A more detailed discussion is provided by McNeill (1986).

The gamma logs were generated with the same equipment as the bulk conductivity logs, only using a gamma sonde. The gamma sonde detects and quantifies gamma emissions from the degradation of naturally occurring radioactive isotopes in the aquifer, especially Potassium-40. Potassium is an abundant component of feldspars, which weather to produce clays. Thus, gamma anomalies are often interpreted as increases in clay concentration in the subsurface.

The borehole geophysical logging occurred in April 2001, and fluid EC values measured concurrent to the logging program are provided for comparison to the bulk EC logs. The diameter of the central casing for piezometer 30 is too small to accommodate the geophysical logging equipment, and the geophysical logs for piezometer 10 were measured in 1995. Thus, the logs for piezometers 3 and 4A are provided as the most relevant to this study.

Groundwater sampling and analysis

The data for this investigation were collected from a network of multi-level piezometers: four of these were installed in two parallel flow path transects (30 → 4A and 10 → 3) between the edge of the landfill and the shore of Lachlan Pond No. 5, approximately 60m downgradient from the landfill (Fig. 2). Sample points on the multi-level piezometers were positioned at 0.5-m intervals across the vertical section that defines the maximum vertical extent of the leachate plume, and then at 1.0 m intervals for an additional 5–10 m below the plume in the uncontaminated regional groundwater. Piezometers 30 and 4A were sampled in February 2002, following the installation of piezometer 30 in December 2001. The results are compared to the most recent sample data from piezometers 10 and 3, collected in July 2001; it should be noted that variations in plume geometry and contaminant concentrations between the two transects are partially a result of the discordant timing of the sampling events.

The sample points were purged using a Geopump 2 peristaltic pump, and groundwater samples were collected once the general parameters (pH, Eh, dissolved oxygen [DO], electrical conductivity [EC] and temperature) had stabilized. The unstable chemical constituents were analysed in the field immediately upon sample collection to minimize the analyte alteration. An alkalinity titration method was used for the field determination of HCO_3^- using 0.01 M HCl as the titrant and a bromocresol green indicator (APHA 1998). Analyte fixation and spectrophotometric analysis methods were used for the field determination of Fe^{2+} , S^{2-} , NO_3^- , NH_4^+ and PO_4^{3-} concentrations using a HACH DR/2010 spectrophotometer (HACH 2000). One 60-mL sample was filtered using a 0.45- μm Millipore cellulose acetate membrane filter and preserved in the field by acidification using 0.4 mL of concentrated, analytical grade nitric acid for analysis of

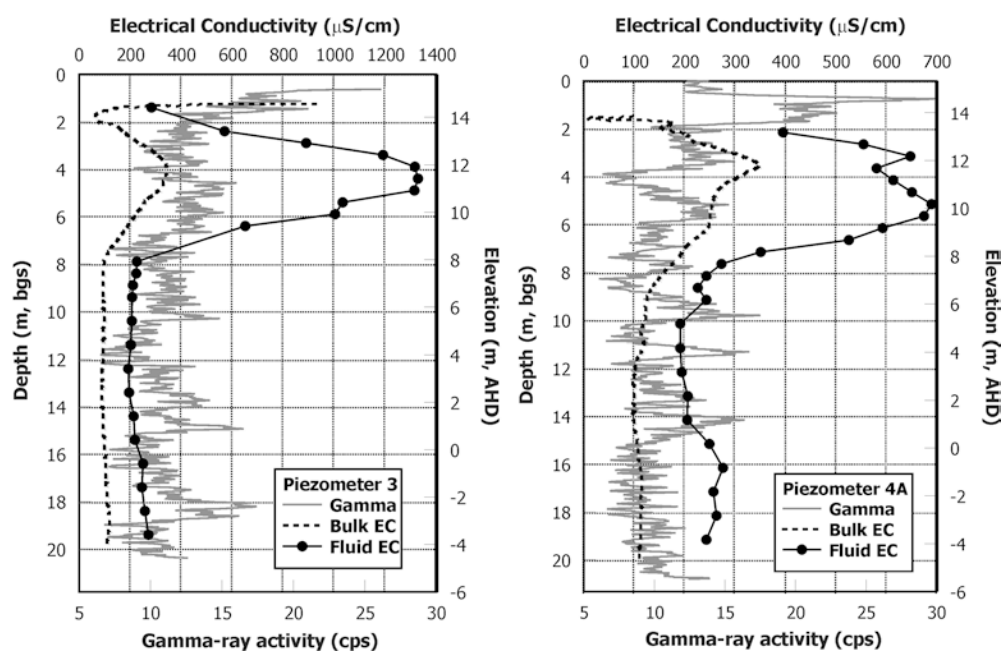


Fig. 3
Borehole geophysical logs for
piezometers 3 and 4A

major and minor elements by inductively coupled plasma atomic emission spectrometry (ICP-AES), and one 120-mL sample was filtered and used for Cl^- determination in the laboratory using an argentometric titration method (APHA 1998).

It should be noted that in environments with elevated concentrations of dissolved organic material, such as landfill leachate, ionically charged organic acids can contribute significantly to the total alkalinity (Kehew and Passero 1990; Baedecker and Cozzarelli 1992). However, the presence of organic acids in significant concentrations in leachate is primarily associated with the early acetogenic phase of waste decomposition, whereas leachate from mature landfills in the methanogenic phase is dominated by refractory, non-polar organic compounds (Chian and DeWalle 1977; Ehrig 1983; Andreottola and Cannas 1992). Taking into consideration the relative age and maturity of Astrolabe Park Landfill, total alkalinity has been calculated as HCO_3^- with the recognition that an unidentified minor percentage may be contributed by organic acid anions.

The accuracy and precision of the ICP analyses were assessed by analysis of spiked samples, deionized water blanks, and duplicate sample analyses as per standard laboratory protocols. All spiked samples, blanks and duplicates were within $\pm 10\%$ relative percent difference (RPD), calculated for sample spike recoveries duplicate analyses. Also, the charge balance error for each sample was calculated as the percent difference in cation and anion concentrations (as meq/L); all samples were within the control range of $\pm 5\%$.

Duplicate spectrophotometer analyses were performed in the field following suspect results; the instability of the analytes following sample collection, as well as budget constraints, precluded a direct, relevant comparison of the spectrophotometer results with alternative laboratory analysis techniques. However, the strong quality control

results, coupled with the correlation in overall data trends observed across the three different analytical methods between six sampling events, suggest that these analytical methods accurately identified spatial and temporal trends in chemical concentrations as they occurred within the leachate plume.

Results and discussion

Borehole geophysical logs

The results of the borehole geophysical logging program for piezometers 3 and 4A are presented in Fig. 3. The plots include the bulk electrical conductivity logs, the gamma logs, and fluid EC values measured concurrent to the geophysical logging for comparison to the bulk EC logs. A clear electrical anomaly is observed in the upper portion of the bulk and fluid EC logs for both piezometers. There is a strong correlation between the bulk and fluid EC profiles at each piezometer, indicating that the variations in bulk EC are primarily caused by presence of the leachate plume in groundwater as opposed to changes in the electrical properties of the aquifer sediments. The difference in the bulk and fluid EC values is explained by the additional electrical resistance of the aquifer sediments that is included in the bulk EC measurements. The electrical anomaly created by the presence of a leachate plume with an elevated solute load within an aquifer otherwise characterised by low conductivity groundwater underscores the value of bulk EC logging for delineation of the plume. Small peaks in the gamma logs indicate the presence of thin clayey horizons within the sandy aquifer. The peak at approximately 10 m below ground surface (bgs) at piezometer 4A exhibits a strong correlation with the lower boundary of the leachate plume. This provides a strong piece of evidence to suggest that solute transport in the

shallow aquifer is constrained by the presence of thin layers of low permeability material present in the sandy aquifer at this location. Several small peaks in the gamma log are also observed at piezometer 3, although there is no clear correlation with the changes in the bulk and fluid EC. However, a clayey horizon was detected at 8 m bgs at this location during a previous site investigation by cone penetrometer testing that correlates well with the lower margin of the leachate plume as defined by the bulk and fluid EC (Jankowski and Knight unpublished data 1991). The gamma logging equipment is most useful for detecting clayey layers that are greater than 1 m thick, and tend to lose sensitivity for thinner layers.

Groundwater chemistry

A statistical summary of the results of groundwater chemical analyses for selected indicator parameters is presented in Table 1. Several indicator parameters have been used to assess the spatial distribution and relative attenuation of inorganic contaminants in groundwater between the Astrolabe Park landfill and Lachlan Pond No. 5, including: EC, HCO_3^- , pH, Eh, $\text{NH}_4^+/\text{NO}_3^-$, and $\text{S}^{2-}/\text{SO}_4^{2-}$. Vertical profiles of the concentrations of these indicator parameters are plotted graphically along groundwater flow paths according to sample point elevations between piezometers 30 and 4A and piezometers 10 and 3 (Figs. 4 and 5, respectively). These figures will be used in the following discussion to make direct comparisons of the relative concentrations and spatial distribution of the indicator parameters between the landfill and the pond.

EC and HCO_3^-

The relationship between EC and HCO_3^- for all samples associated with this research project ($n=498$) is plotted in Fig. 6. This plot indicates that there is an excellent correlation between these two values ($r^2=0.9645$), and that both

can be used to characterise the relative degree of leachate impact to regional groundwater.

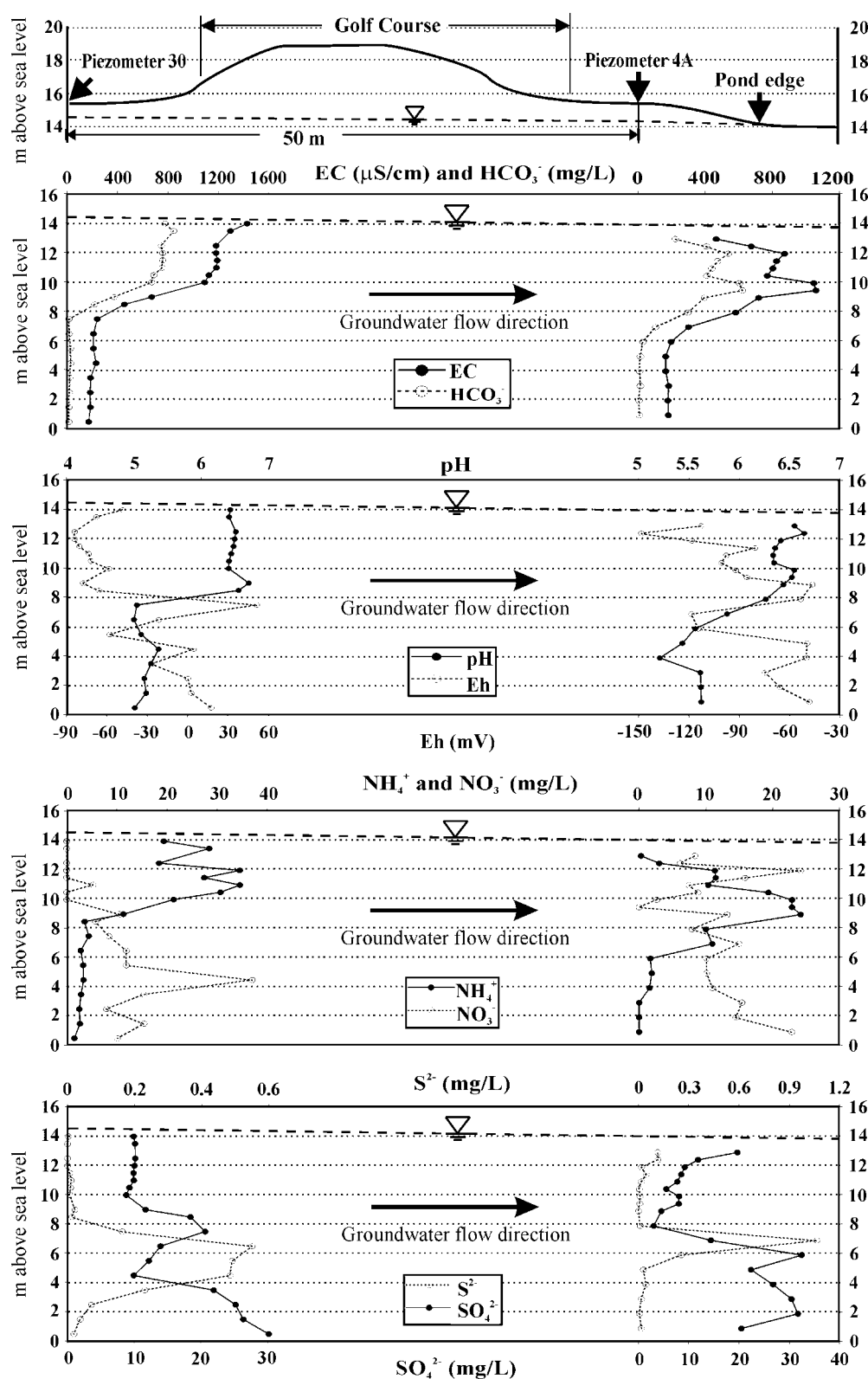
EC and HCO_3^- show significant variations in maximum concentration and spatial distribution between piezometer 30 and 4A (Fig. 4), 50 m downgradient. Maximum EC values decrease from 1,429 to 1,060 $\mu\text{S}/\text{cm}$, a 26% decrease, and the elevation of the peak value dropped by approximately 4.4 m between the piezometers. Likewise, the maximum HCO_3^- concentration decreased from 847 to 622 mg/L, a 27% decrease, with a vertical displacement nearly identical to that of the EC.

The decrease in EC and HCO_3^- concentrations between these piezometers is most likely due to dilution of the leachate plume from longitudinal and lateral dispersion. The decrease in elevation of the centre of mass may be due to a number of factors, including density differences between the fresh and contaminated water, the presence of preferential flow paths in the subsurface, or depression of the plume from infiltrating precipitation and lawn watering on the golf course between the landfill and the pond. The observed variation between piezometers 10 and 3 is slightly different. The maximum EC value decreased from 1,650 to 1,381 $\mu\text{S}/\text{cm}$, a decrease of 16%, while the maximum concentration of HCO_3^- decreased from 1,008 to 816 mg/L, a decrease of 19%. The elevation of the maximum concentration for both parameters only dropped by approximately 1 m. Assuming that the effects of dispersion and lawn watering infiltration on the leachate plume are similar in the two closely spaced transects, the most likely explanation for the discrepancy in variation between the 30 to 4A and 10 to 3 transects is the timing of the sampling events (February 2002 and July 2001, respectively) relative to rainfall events. Also, a slight increase in the fraction of fine sediment present between piezometers 30 and 4A as compared to 10 and 3 may be contributing to the increased attenuation of the inorganic contaminants.

Table 1

Statistical parameter values and element concentrations for contaminated and uncontaminated groundwater samples collected from piezometers 10 and 3 (July 2001) and 30 and 4A (February 2002)

	EC ($\mu\text{S}/\text{cm}$)	HCO_3^- (mg/L)	pH	Eh (mV)	NO_3^- (mg/L)	NH_4^+ (mg/L)	SO_4^{2-} (mg/L)	S^{2-} (mg/L)
Piezometers 3 and 10, July 2001 (n=44)								
<i>Leachate-impacted groundwater</i>								
Maximum	1,650	1,008	6.45	+4.9	92.0	54.2	103	0.163
Mean	783	397	6.02	-157	25.1	15.5	26.7	0.037
Minimum	241	31.7	5.27	-211	8.8	0.1	0.4	0
<i>Uncontaminated groundwater</i>								
Maximum	295	92.7	6.05	-85.8	28.2	8.8	43.8	1.315
Mean	220	32.0	5.11	-172	15.4	2.8	22.6	0.178
Minimum	179	4.9	4.12	-225	6.2	0.1	0.5	0.003
Piezometers 30 and 4A, February 2002 (n=35)								
<i>Leachate-impacted groundwater</i>								
Maximum	1,429	847	6.71	-46.5	24.2	34.5	19.7	0.116
Mean	931	550	6.46	-82.4	5.8	18.3	9.7	0.021
Minimum	455	220	6.26	-148.4	0	0.3	3.0	0.034
<i>Uncontaminated groundwater</i>								
Maximum	301	105	5.88	+51.4	37.0	11.0	32.5	1.065
Mean	199	23.0	5.34	-36.7	14.5	2.6	22.6	0.232
Minimum	165	6.1	5.00	-119	7.9	0	9.9	0.006

**Fig. 4**

Vertical profiles of groundwater quality parameters and contaminants for multi-level piezometers 30 and 4A, located along a groundwater flow path between the Astrolabe Park landfill and Lachlan Pond No. 5

pH and Eh

Both transects display a similar pattern of increased pH values and decreased Eh values with respect to regional groundwater within the leachate plume, but the individual profiles vary significantly.

The pH of the contaminated groundwater at piezometer 30 (Fig. 4) is tightly constrained between values of 6.4 to 6.7 for approximately 6 m, below which a sharp decrease is observed between an elevation of 8.5 to 7.5 m above sea level (a.s.l.). A similar pattern is observed with Eh values,

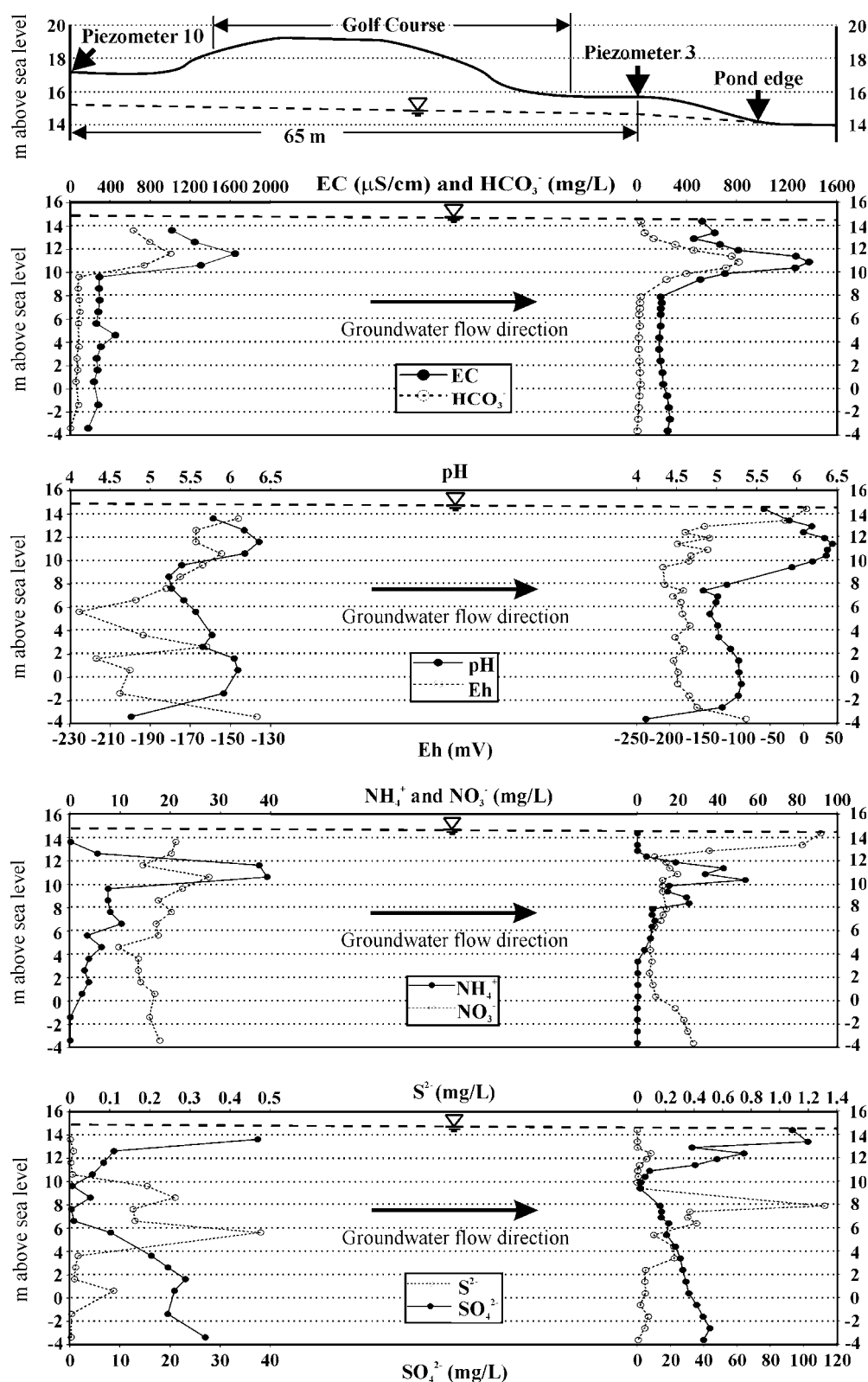


Fig. 5
Vertical profiles of groundwater quality parameters and contaminants for multi-level piezometers 10 and 3, located along a groundwater flow path between the Astrolabe Park landfill and Lachlan Pond No. 5

which rise dramatically within the same interval. The pH values at piezometer 4A (Fig. 4) are also constrained between 6.4 to 6.7 in the contaminated zone, and then gradually returns to background pH values of 5.0–5.5, which are characteristic for the uncontaminated part of the

aquifer (Jankowski and Beck 2000), within the 6-m vertical section between 10 to 4 m a.s.l.

Likewise, the Eh profile at piezometer 4A (Fig. 4) displays a much narrower vertical section of highly reducing water with a more gradual transition to fresh water. The

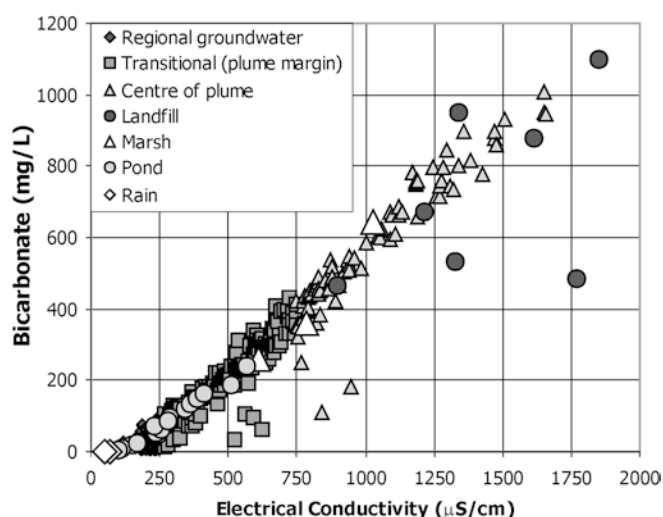


Fig. 6

Plot of HCO_3^- versus EC in groundwater samples, and subsequent contamination characterisation

variation in the transitional zones may be due to vertical mixing between the two piezometers, although the severity of the transitional zone at piezometer 30 may also indicate the presence of a laterally discontinuous confining layer at an elevation of approximately 8 m a.s.l. The maximum Eh value at piezometer 4A is lower than that of piezometer 30, this discrepancy is probably due to localized variations in the sedimentary composition of the aquifer, resulting in variable rates of microbiological activity.

The pH profiles at piezometers 10 and 3 (Fig. 5) also exhibit pH values constrained between values of 6.0 to 6.5 within the leachate plume, although this zone appears to be approximately 1.5–2 m thicker at piezometer 3, most likely as a function of vertical dispersion of the plume between the two piezometers. The Eh profiles between the two piezometers are slightly erratic, although both display a decrease in Eh values that corresponds to the zone of increased pH values. The variation in the shapes of the profiles is likely attributable to the presence of locally discontinuous peat lenses containing iron sulfides, which act as substrates for SO_4^{2-} -reducing bacteria. This may be responsible for the sharp drops in Eh observed at elevations of 6 and 2 m a.s.l. at piezometer 10 (Jankowski and Acworth 1997). Comparison of the shape of the Eh profiles to the $\text{S}^{2-}/\text{SO}_4^{2-}$ profiles appears to support this assertion.

NH_4^+ and NO_3^-

The profiles of N-species at all four piezometers clearly display the expected relationship between the redox couples within and below the leachate plume.

Piezometer 30 (Fig. 4) displays a near absence of NO_3^- within the leachate plume, but then gradually increases with depth below 9 m a.s.l. NH_4^+ concentrations exhibit the opposite trend, with the highest concentrations observed within the plume, decreasing along a steep vertical gradient between the elevations of 10 to 8 m a.s.l. A similar relationship is observed for piezometer 4A, although the maximum concentration of NH_4^+ decreases

from 35 to 24 mg/L, a decrease of 30%. Likewise, the zone of maximum NH_4^+ concentration drops from 12–11 to 10–9 m a.s.l., and appears to narrow vertically by approximately 2 m. Also, a spike of high NO_3^- concentration is observed at piezometer 4A at an elevation of 12 m a.s.l. This is likely due to ammonium oxidation along the edge of the plume as it reacts with oxygen-enriched recharge water, and perhaps by the application of nitrogen-based fertilizers on the golf course.

A much narrower vertical section of elevated NH_4^+ concentrations is observed in piezometers 10 and 3 (Fig. 5), although the maximum concentration values are nearly the same as that of piezometer 30. Surprisingly, the maximum concentration of NH_4^+ actually increases between piezometers 10 and 3, from 40 to 43 mg/L, an increase of 8%.

There was no observed change in the elevation range of the maximum NH_4^+ concentration. Also, a spike of NO_3^- is observed at the top of the profile at piezometer 3; the most likely source of this additional nitrogen is from fertilizers used by the golf course. The high NO_3^- concentrations at the water table at piezometer 3 may be contributing to the increased NH_4^+ concentration within the leachate plume. It seems likely that any potential contribution of nitrogen to groundwater from nitrogen-based fertilizers used by the golf course would be similar for the two transects as they are relatively close to each other, similar in length and cross the same portion of the golf course. By this assumption, the discrepancy between the observed NH_4^+ attenuation, or lack thereof, for the two transects may be due to a significantly increased percentage of fine-grained silt, clay and peat material present in the upper portion of the aquifer between piezometers 30 and 4A that is not present between piezometers 10 and 3. This may cause some attenuation of the NH_4^+ in groundwater due to ion exchange on the clay surfaces.

S^{2-} and SO_4^{2-}

The SO_4^{2-} concentrations in all profiles display similar patterns of decreased concentrations within the leachate plume and elevated concentrations above and below the leachate plume. The increase in SO_4^{2-} concentrations below the plume is relatively sharp (10–30 mg/L increase in the first 2 m below the plume) for all piezometers except for piezometer 3. The Eh values and the NO_3^- and SO_4^{2-} concentrations all recover much more slowly with depth beneath the plume at piezometer 3 than is observed in the other piezometers. It is possible that the presence of lenses of organic material provide an additional substrate for microbial activity, which preserves the reducing environment to a greater depth than is observed in the other piezometers.

It is interesting to note the elevated SO_4^{2-} concentration immediately above the plume at piezometer 3. Research by Ulrich and others (2003) at Norman Landfill in Oklahoma also observed this phenomenon, and suggested that it was due to microbial oxidation of iron sulfides in response to fluctuations in the water table. The water table at piezometer 3 is strongly controlled by the water level of Lachlan Pond No.5, which is subject to rapid fluctuations in response to storm events as it receives a significant volume

of stormwater discharge from local drainage networks. During storm events, the groundwater flow direction has been observed to temporarily reverse due to elevated water levels in the pond, causing pond water to infiltrate into the aquifer (Acworth personal communication 2003). The infiltration of oxygen-enriched pond water into the normally reducing zone of the leachate plume at piezometer 3 may instigate a brief period of aerobic oxidation of iron sulfides, thus generating excessive SO_4^{2-} concentrations. The groundwater sampling at piezometer 3 occurred immediately after 10 days of excessive rainfall. Sulfide is absent within the zone of greatest leachate contamination as defined by the EC, HCO_3^- and NH_4^+ profiles. As SO_4^{2-} concentrations appear to be reduced within the plume, the expected result would be a concomitant increase in S^{2-} . The most plausible explanation for the absence of S^{2-} in the plume is that it is rapidly precipitating as FeS in the presence of elevated concentrations of Fe^{2+} in the leachate plume (Postma and Jakobsen 1996).

The highest S^{2-} concentrations in all profiles occur immediately below the leachate plume (Jankowski and Acworth 1997). This zone also happens to correspond to a relatively continuous layer of pyrite-bearing peat material present in the investigation area identified in previous research. In contrast to the EC, HCO_3^- and NH_4^+ concentrations for piezometers 30 and 4A (Fig. 4), which appear to decrease in concentration and elevation as they are transported downgradient from the landfill, the S^{2-} concentration increases downgradient from the landfill and does not display any vertical displacement. A similar trend is also present between piezometers 10 and 3 (Fig. 5). Further, the peaks in S^{2-} concentration occur at nearly the same elevation in all of the piezometers. This combined evidence suggests that the pyrite-bearing peat layer is the most likely source of the increased S^{2-} concentrations observed in the profiles (Jankowski and Beck 2000). The excess acidity generated during the oxidation of these pyritic aquifer sediments is sufficient to overcome the pH buffering capacity of the leachate plume, and is likely responsible for the abrupt decrease in pH values observed at the same depth interval.

Conclusions

In general, a decrease in the maximum value/concentration of EC and HCO_3^- was observed between the Astrolabe Park landfill and Lachlan Pond No. 5, as well as a decrease in the elevation at which the maxima were detected, although this was most pronounced in the piezometer 30 to 4A transect. As the groundwater flows away from the landfill, and thus away from the source of the HCO_3^- production, it is assumed that EC values and HCO_3^- concentrations decrease as a result of dispersion along the flow path and mixing with fresh regional groundwater. The relative differences in the concentration decreases is likely due to differences in precipitation at the time of the two sampling events. The pH profiles for both transects display well buffered values within the leachate plume (Kehew and Passero

1990; Jankowski and Acworth 1997), which recover to background values below the plume. Once again, there is a greater observable change in the thickness of the buffered zone along the 30 to 4A transect than along the 10 to 3 transect, most likely attributable to different weather patterns due to the discordant timing of the sampling events. While generally displaying depressed values within the leachate plume, the Eh profiles varied significantly between each piezometer. Local variations in the sedimentary composition of the aquifer appear to have a significant impact on the variability of redox potentials with depth at each piezometer, which makes Eh values a difficult parameter from which to draw any significant conclusions regarding the specific effects of transport on leachate contaminants.

The profile for each piezometer displayed the expected relationship between NH_4^+ and NO_3^- within and below the leachate plume. The 30 to 4A transect showed a significant decrease in the maximum concentration of NH_4^+ along the flow path, most likely due to a combination of dispersion/dilution and attenuation of NH_4^+ on exchange site surfaces within the geological material of the aquifer. Unexpectedly, the maximum NH_4^+ concentration was increased between piezometers 10 and 3. The only possible cause for this occurrence is an additional influx of nitrogen along the flow path. The extremely high NO_3^- concentration immediately above the plume in piezometer 3 suggests that nitrogen may be flushed into the groundwater system from nitrogen-based fertilizers used on the golf course; and nitrate-reducing bacteria may have reduced a portion of the NO_3^- to NH_4^+ , accounting for the observed increase in concentration along the flow path. Once again, the difference in the timing of the sampling events may explain why this phenomena was not observed along the 30 to 4A transect.

The SO_4^{2-} concentrations followed a similar pattern to NO_3^- : decreased concentrations within the leachate plume with an observed recovery below the plume. The expected concomitant increase in S^{2-} concentrations within the plume was not observed; any S^{2-} produced within the plume as a result of SO_4^{2-} reduction is most likely precipitated as FeS in the Fe^{2+} -enriched plume (Postma and Jakobsen 1996). Elevated SO_4^{2-} concentrations at piezometer 3 are likely due to aerobic oxidation of iron sulfides in response to water table fluctuations following a significant storm event (Ulrich and others 2003).

In contrast, the highest concentrations of S^{2-} were observed immediately below the plume, where previous research has identified a locally extensive, pyrite-bearing peat layer (Jankowski and Acworth 1997). The fact that the elevated S^{2-} concentrations occurred at approximately the same elevation in each profile, and the maximum concentrations increased along the flow path, suggests that the S^{2-} is produced naturally within the aquifer as a by-product of microbial degradation occurring within the peaty horizon where oxidation of FeS_2 followed by subsequent reduction of SO_4^{2-} is occurring (Jankowski and Beck 2000). The excess acidity generated during the oxidation of the pyritic aquifer sediments may be

responsible for the abrupt decrease in pH values observed at the same depth intervals.

While dispersion, dilution and sorption processes along groundwater flow paths appear to be responsible for the decrease in concentrations of some components of the leachate plume, the influence of precipitation needs further investigation to determine whether the relative variations observed between the transects are indeed due to weather variations as a result of the discordant timing of the sampling events. Also, the relative nitrogen contribution from fertilizer along the flow path use needs to be investigated to explain the unexpected increase in NH_4^+ concentration between piezometers 10 and 3.

Acknowledgements This research was made possible through the financial assistance of Sydney Water. The authors thank Karina Morgan, Bethany O'Shea, Sarah Groves and Wendy McLean for their assistance during sample collection and Dorothy Yu for her help with chemical analysis in the laboratory.

References

- Albani AD (1981) Sedimentary environments and Pleistocene chronology of the Botany Basin, NSW. *Geo-Mar Lett* 1:163–167
- American Public Health Association (APHA) (1998) Standard methods for the examination of water and wastewater, 20th edn. APHA-AWWA-WET Washington, DC
- Andreottola G, Cannas P (1992) Chemical and biological characteristics of landfill leachate. In: Christensen TH, Cossu R, Stegmann R (eds) *Landfilling of waste: leachate*. E & FN Spon, London, pp 65–88
- Arneith J-D, Milde G, Kerndorff H, Schleyer R (1989) Waste deposit influences on groundwater quality as a tool for waste type and site selection for final storage quality. In: Baccini P (ed) *The landfill*, vol 20, (Lecture Notes in Earth Sciences). Springer, Berlin Heidelberg New York, pp 399–424
- Baedecker MJ, Apgar MA (1984) Hydrogeochemical studies at a landfill in Delaware. In: *Groundwater contamination, studies in geophysics*. National Academy Press, Washington, DC, pp 127–138
- Baedecker MJ, Back W (1979) Hydrogeological processes and chemical reactions at a landfill. *Ground Water* 17(5):429–437
- Baedecker MJ, Cozzarelli IM (1992) The determination and fate of unstable constituents of contaminated groundwater. In: Lesage S, Jackson RE (eds) *Groundwater contamination and analysis at hazardous waste sites*. Marcel Dekker, New York, pp 425–461
- Bjerg PL, Ruge K, Pederson JK, Christensen TH (1995) Distribution of redox-sensitive groundwater quality parameters downgradient of a landfill (Grinsted, Denmark). *Environ Sci Technol* 29:1387–1394
- Chian ESK, DeWalle FB (1977) Characterization of soluble organic matter in leachate. *Environ Sci Technol* 11(2):158–163
- Christensen TH (1992) Attenuation of leachate pollutants in groundwater. In: Christensen TH, Cossu R, Stegmann R (eds) *Landfilling of waste: leachate*, E & FN Spon, London, pp 441–483
- Christensen TH, Kjeldsen P, Bjerg PL, Jensen DL, Christensen JB, Baun A, Albrechtsen H-J, Heron G (2001) Biogeochemistry of landfill leachate plumes. *Appl Geochem* 16:659–718
- Cozzarelli IM, Suffita JM, Ulrich GA, Harris SH, Scholl MA, Schlottmann JL, Jaeschke JB (1999) Biogeochemical processes in a contaminant plume downgradient from a landfill, Norman, Oklahoma. U.S. Geological Survey Water Resources Investigations Report # 99–4018C, Reston, VA
- Ehrig H-J (1983) Quality and quantity of sanitary landfill leachate. *Waste Manage Res* 1:53–68
- Griffin RJ (1963) The Botany Basin. *Geol Surv New South Wales Bull* 18, 110 pp
- HACH (2000) DR/2010 Spectrophotometer handbook – Procedures manual. Hach Company, Loveland, Colorado, 872 pp
- Jankowski J (1997) Generation and migration of chemical constituents in a leachate plume. In: McNally GH (ed) *Collected case studies in engineering geology, hydrogeology, and environmental geology*, 3rd series, *Geol Soc Aust* pp 181–205
- Jankowski J, Acworth RI (1997) Development of a contaminant plume from a municipal landfill: Redox reactions and plume variability. In: Chilton J, Hiscock K, Morris B, Nash H, Tellam J, Hennings S (eds) *Proceedings of the XXVII Congress of the International Association of Hydrogeologists: Groundwater in the Urban Environment*, Nottingham, UK, 21–27 September 1997. Balkema, Rotterdam, pp 439–444
- Jankowski J, Beck P (2000) Aquifer heterogeneity: hydrogeological and hydrochemical properties of the Botany Sands aquifer and their impact on contaminant transport. *Aust J Earth Sci* 47:45–64
- Kehew AE, Passero RN (1990) pH and redox buffering mechanisms in a glacial drift aquifer contaminated by landfill leachate. *Ground Water* 28(5):728–737
- Kimmel GE, Braids OC (1974) Leachate plumes in a highly permeable aquifer. *Ground Water* 12(6):388–392
- Kjeldsen P (1993) Groundwater pollution source characterisation of an old landfill. *J Hydrol* 142:349–371
- Lindberg RD, Runnells DD (1984) Ground water redox reactions: an analysis of equilibrium state applied to Eh measurements and geochemical modeling. *Science* 225:925–927
- Ludvigsen L, Albrechtsen H-J, Heron G, Bjerg PL, Christensen TH (1998) Anaerobic microbial redox processes in a landfill leachate contaminated aquifer (Grinsted, Denmark). *J Contam Hydrol* 33:273–291
- Lyngkilde J, Christensen TH (1992) Redox zones of a landfill leachate pollution plume (Vejlen, Denmark). *J Contam Hydrol* 10:273–289
- McNeill JD (1986) Geonics EM39 borehole conductivity meter—theory of operation. Geonics Limited, Technical note TN-20
- McNeill JD (1990) Use of electromagnetic methods for groundwater studies. In: Ward SH (ed) *Geotechnical and environmental geophysics*, vol 1: review and tutorial. *Soc Explor Geophys Tulsa*, Oklahoma, pp 191–218
- Nicholson RV, Cherry JA, Reardon EJ (1983) Migration of contaminants in groundwater at a landfill: a case study. 6. *Hydrogeochemistry*. *J Hydrol* 63:131–176
- Postma D, Jakobsen R (1996) Redox zonation: equilibrium constraints on the Fe(III)/SO₄-reduction interface. *Geochim Cosmochim Acta* 60(17):3169–3175
- Qasim SR, Chiang W (1994) Sanitary landfill leachate: generation, control and treatment. Technomic Publishing Company, Lancaster, Pennsylvania 323 pp
- Rickwood PC (1998) The Botany Basin: a convenient definition. In: McNally GH, Jankowski J (eds) *Environmental geology of the Botany Basin*, collected case studies in engineering geology, hydrogeology and environmental geology, 4th series. *Geol Soc Aust* pp 173–178
- Ulrich GA, Breit GN, Cozzarelli IM, Suffita JM (2003) Sources of sulfate supporting anaerobic metabolism in a contaminated aquifer. *Environ Sci Technol* 37:1093–1099