Recycled water for heavy industry and preventing sea water intrusion

A report of a study funded by the
Australian Water Recycling Centre of Excellence

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Recycled water for heavy industry and preventing sea water intrusion (Kwinana managed aquifer recharge)

Project Leader

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Partners

CSIRO Land and Water Flagship
Kwinana Industries Council
Western Trade Coast
Department of Water Western Australia Government
Department of Health Western Australia
Water Corporation

About the Australian Water Recycling Centre of Excellence

The mission of the Australian Water Recycling Centre of Excellence is to enhance management and use of water recycling through industry partnerships, build capacity and capability within the recycled water industry, and promote water recycling as a socially, environmentally and economically sustainable option for future water security.

The Australian Government has provided $20 million to the Centre through its National Urban Water and Desalination Plan to support applied research and development projects which meet water recycling challenges for Australia's irrigation, urban development, food processing, heavy industry and water utility sectors. This funding has levered an additional $40 million investment from more than 80 private and public organisations, in Australia and overseas.

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Recycled water for heavy industry and preventing sea water intrusion

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Shortened forms

AHD	Australian Height Datum
AMC	Australian Marine Complex
APET	Areal potential evapotranspiration
BAU	Business as usual
BOD	Biological oxygen demand
CBA	Cost benefit analysis
CBD	Central Business District
CGA	Cockburn Groundwater Area
CI	Confidence interval
CLW	CSIRO Land and Water
COD	Chemical oxygen demand
CMIP5	Fifth Coupled Model Intercomparison Project
CSIRO	Commonwealth Scientific and Industrial Research Organisation
CSMC	Cockburn Sound Management Council
DER	Department of Environmental Regulation
DEM	Digital elevation model
DoW	Department of Water
DPaW	Department of Parks and Wildlife
DTW	Depth to watertable
ET	Evapotranspiration
FIG	Floreat Infiltration Galleries
GCM	Global climate model
GDA	Geocentric Datum of Australia
GDE	Groundwater-dependent ecosystem
Geomorphic wetlands	A classification system for wetlands adopted by the Western Australian government which is based on both morphology and the degree of wetness
HPP	Hedonic Property Price
IPCC AR5	The fifth assessment report of the Intergovernmental Panel on Climate Change
IWSS	Integrated Water Supply Scheme
KIA	Kwinana Industrial Area
KWRP	Kwinana Water Reclamation Plant
KWWTP	Kwinana Wastewater Treatment Plant
LAI	Leaf area index

Landsat TM	Landsat Thematic Mapper – a type of satellite used for remote sensing
LGA	Local Government Area
LIDAR	Light Detection and Ranging – a method of measuring elevations using a laser
MAR	Managed aquifer recharge
MGA	Map Grid of Australia
MIP	Marginal implicit price
MODFLOW	A groundwater model which uses finite difference methods to estimate groundwater potentials (levels)
MSLP	Mean sea level pressure
NDVI	Normalised Difference Vegetation Index
NDWI	Normalised Difference Wetness Index
NPI	National Pollutant Inventory
NRMS	Normalised root mean squared
PRAMS	Perth Regional Aquifer Modelling System
RIZ	Rockingham Industrial Zone
RWR	Relative watertable rise
SDOOL	Sepia Depression Ocean Outfall Line
SGD	Submarine groundwater discharge
SWI	Salt water intrusion
SWSY	South West Sustainable Yield – a CSIRO project that estimated the impact of future climate on water yields
TAFE	Technical and Further Education
TEV	Total Economic Value
TEC	Threatened Ecological Community
TKN	Total Kjeldahl Nitrogen
TP	Total Phosphorus
TWW	Treated wastewater
VFM	Vertical Flux Model
WESROC	Western Suburbs Regional Organisation of Councils
WMP	Water Management Plan

Units

UNIT	DESCRIPTION
m^3/s	cubic metres per second or ‘cumecs’
L	litre
kL	kilolitre (1,000 litres)
ML	megalitre (1,000,000 litres)
GL	gigalitre (1,000,000,000 litres)

Executive summary

BACKGROUND TO THE STUDY

An investigation of the feasibility of recharging the unconfined aquifer in the Kwinana region south of Perth with treated wastewater arose out of several issues:

- The region has a history of falling groundwater levels and seawater intrusion as a result of a drying climate and extraction.
- This situation has increased concerns about the long-term ability of the aquifer to continue to meet over 60% of the water needs in the Kwinana Industrial Area (KIA).
- There is a lack of groundwater for new industry and recent projections have quantified future water demands which show an increasing gap between water supplies and demands.
- An abundance of highly treated wastewater which flows past the KIA to an ocean outfall with projections of increase quantities and quality
- Improved understanding of managed aquifer recharge (MAR) using treated wastewater including water quality, groundwater contamination, soil and aquifer clogging, preferred pathway flow in limestone aquifers and cost effectiveness.
- A long history of apparent safe disposal through infiltration ponds at wastewater treatment plants which lack an ocean outfall; rivers not being a feasible solution in south-western Australia.
- Increasing interest in water reuse, especially if it reduces the need for new water sources and overcomes or delays the need to expand ocean outfall disposal pipelines.

There are several advantages to developing managed aquifer recharge in the KIA using treated wastewater, including the large supply, sandy soils suitable for infiltration and treatment, aquifers with increasing amounts of storage, and the shallow groundwater that is not used for drinking, or very close to areas used for private irrigation. Being an industrial area it is possible to control access to infiltration ponds which are cheaper to install and maintain than buried galleries or injection bores.

There are, however, issues related to pockets of contaminated groundwater in the KIA and the need to keep nitrogen loads entering Cockburn Sound to a level below that likely to affect water quality and seagrass health.

PROJECT AIMS

The project was established to evaluate options for diverting locally available treated wastewater into the Superficial Aquifer in the Cockburn Sound catchment to raise groundwater levels, improve non-potable water supplies and security for heavy industry, reverse seawater intrusion, and improve the condition of groundwater-throughflow wetlands that have been seriously impacted by lower groundwater levels in recent decades.

While answering a local non-potable water need, the project aimed to help answer state-wide questions, thereby facilitating more widespread adoption of MAR in areas containing residential developments. These areas are losing throughflow wetlands of considerable economic, environmental and social value. They are also experiencing a loss of water for irrigating public open space as well as domestic gardens, the formation of acid sulphate soils as previously wet areas dry, and the gradual accumulation of salts in poorly flushed aquifers as is becoming evident as the climate dries and groundwater gradients fall in south west Western Australia.

PROJECT HISTORY AND PARTNERS

The partners that came together to develop and carry out the project were:

- The Australian Water Recycling Centre of Excellence provided funds and oversight through a Project Advisory Committee.
- The Kwinana Industries Council provided funds and identified the pending non-potable water shortage about ten years ago and has been investigating cost- effective solutions since.
- CSIRO provided funds and has been investigating MAR options for a number of years relating to chemical transformations that occur in the soil and aquifer as well as issues such as clogging, nitrogen removal, recovering throughflow wetlands and the value of wetlands in urban settings.
- The Western Australian Department of Water provided in-kind support and groundwater data. It has been carrying out a review of water demands and supply options for the Western Trade Coast area which includes the Kwinana Industrial Area as well as adjacent light industrial zones. The Department is refining its policies and governance related to MAR and used the project as a test ground for its ideas.
- The Water Corporation provided in-kind support and information on wastewater streams and groundwater in wastewater disposal areas. It is responsible for the wastewater plants and disposal licenses in most of Western Australia where a solution could be more broadly applied. The Corporation has a wastewater reuse policy of 20% reuse by 2020 and 30% by 2030.
- The Western Trade Coast provided chairing and secretarial support. It was a state government agency that coordinated development opportunities in the area around the KIA until it ceased in June 2015.
- The Western Australian Department of Health provided in kind-support to ensure that any public health risk associated with treated wastewater reuse are minimised.

The project ran over two years (2013 to 2015) and was coordinated by a Steering Committee comprising the partners listed above. Throughout the project people in the Cockburn Sound Management Council, Department of Environmental Regulation and Department of Parks and Wildlife were involved in workshops and field trips.

THE COCKBURN SOUND CATCHMENT STUDY AREA

The study area (Figure 1) is underlain by the unconfined Superficial Aquifer that occurs in a series of sand dunes that underlie most of the coastal plain in Greater Perth (population of 2 million in 2015). The aquifer supplies the majority of non-potable water for the Kwinana Industrial Area (KIA) that has an annual output worth more than AU$15 billion and directly employs almost 5000 people with a further induced employment of about 26,000 people. It also supports irrigation by peri-urban horticulturalists, local government and private bore owners. Groundwater throughflow wetlands occur where the watertable approaches the surface in inter-dunal swales.

Over 50 GL/yr of treated wastewater primarily from the Woodman Point wastewater treatment plant (WWTP) is discharged to the marine environmental via the Sepia Depression Ocean Outfall Line (SDOOL) located about 4 km south-west of Point Peron (Figure 1). About 1.7 GL/yr is infiltrated through ponds at the Kwinana WWTP. While the ponds were established to dispose of treated wastewater in 1975, they provide a long-term guide to how MAR may perform in the catchment.

While almost the entire catchment has sandy soils the western areas have highly transmissive aquifers associated with limestone. The south west is especially very transmissive. A chain of wetlands occurs in the eastern portion of the study area immediately east of the Kwinana WWTP, while another (western) chain lies within a few kilometres of the coast (Figure 1).

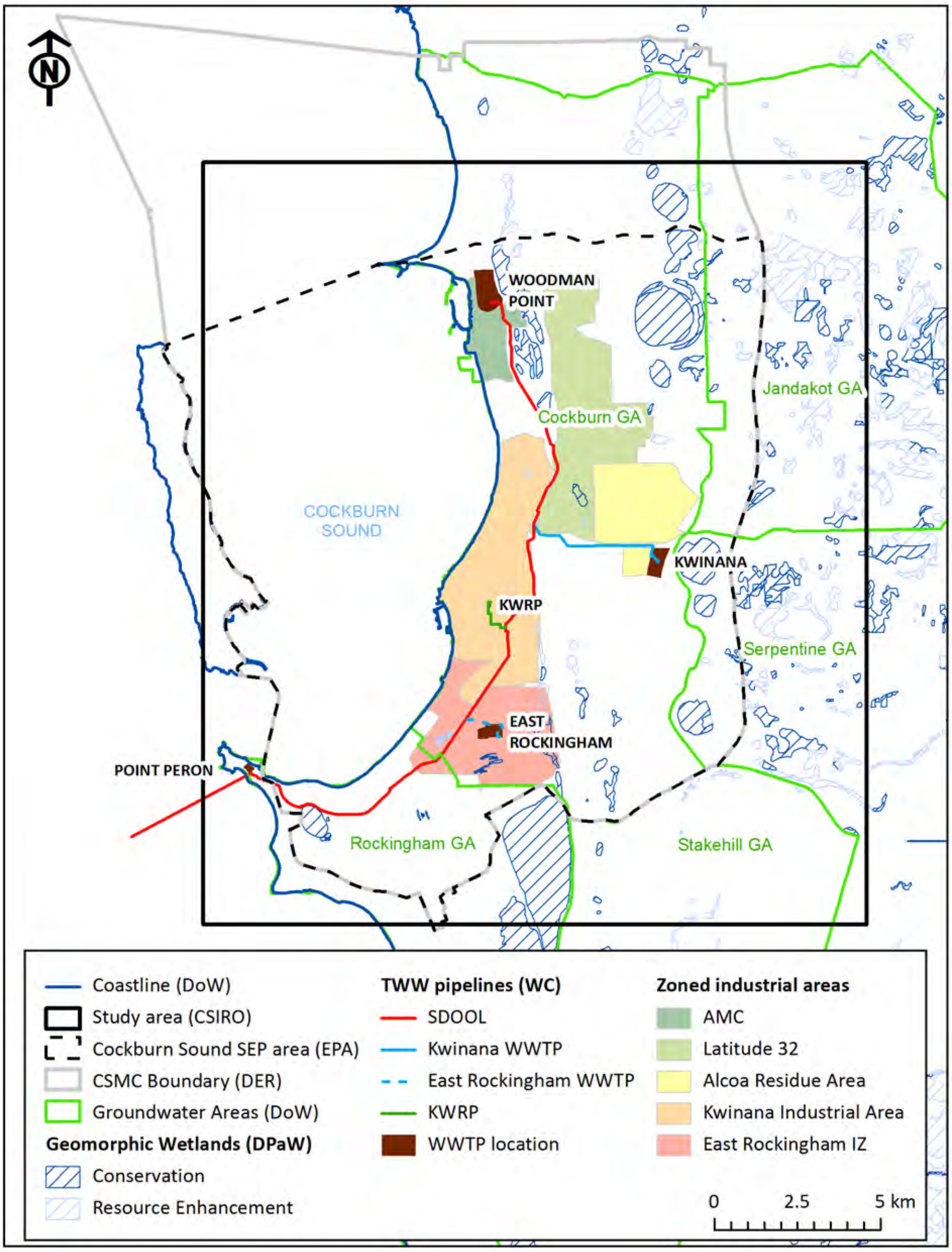

Figure 1. The study area in relation to wastewater treatment plants and pipelines (Water Corporation, WC), Cockburn Sound Management Council (CSMC; Department of Environment Regulation, DER) and Cockburn Sound State Environmental Policy (SEP; Environment Protection Agency, EPA) administrative boundaries, and groundwater management areas (Department of Water, DoW)

METHODS

A groundwater model that could simulate the effects of climate, land use changes, groundwater extraction and MAR on groundwater levels and the salt water interface was developed. Special attention was focused on investigating the Kwinana Wastewater Treatment Plant (WWTP) where increasing amounts of secondary treated wastewater has been infiltrated into the Superficial Aquifer through open basins since 1975. Groundwater levels were logged over a twelve month period and samples taken at two times for chemical analysis.

Engineering constraints and methods of extracting, treating and adding treated wastewater through open pits and galleries were investigated in a number of locations. The value that wetlands add to nearby house sale process was determined using available data. A series of workshops gathered expert input into options for MAR starting a sites spread throughout the catchment before settling on six for more investigation and finally three opportunities for which more detailed designs and cost benefit analyses were undertaken.

RESULTS

By 2031 the industrial water demand is expected to be 43 GL/yr, which is approximately 17 GL/yr higher than the current demand. About 17 GL/yr is extracted from local aquifers and this may need to reduce if the existing declining levels and increased salt water intrusion become more widespread in the next 16 years.

The 40-year history of infiltrating treated wastewater from ponds into the Superficial Aquifer at the Kwinana WWTP has indicated that MAR may be a realistic and safe option elsewhere in the catchment. It has appeared to have maintained levels in up-gradient wetlands in a drying climate over the same period. Upgrading the WWTP to oxidation ditch has reduced total nitrogen level by an order of magnitude resulting in lower levels than that coming from the mineralisation of peat in the wetland beds.

Groundwater modelling of MAR options has indicated that levels may be raised by more than 50 cm over 5 km from the site of infiltration, and the area that is affected may be 10 to 20 times larger than the area that receives inflow water (Figure 2). MAR site located within 3 km of the coast can have infiltrated water reach the Sound within the 20 year modelling period, especially in the south where transmissivities are highest. Some simulations show that the infiltrated water is intercepted by pumping bores (e.g. E1 examples) under the current pumping regime. Increased pumping is likely after MAR so such interception will be more likely than has been simulated.

Comparative unit costs estimates indicate groundwater is the cheapest option, followed by managed aquifer recharge (MAR) of recycled water into the Superficial Aquifer, then direct access by industry to treated wastewater from the Sepia Depression Ocean Outfall Line. With groundwater levels gradually falling, getting more groundwater is unlikely and existing supplies are becoming less secure. Therefore investigating the viability of MAR for specific parts of the Kwinana Industrial Area (KIA) seems worthwhile.

The cost of MAR varies between $0.40 and 1.44/kL with lower cost being for no pre-treatment and infiltration by ponds close to the wastewater pipeline (SDOOL) and higher costs being the converse. Pumping distance and the need to remove nitrogen increase costs the most. When compared to the price of Kwinana Water Reclamation Plant water ($2.00/kL) or the price of scheme water ($2.03/kL), the cost of MAR is competitive provided costs to include such as land purchase, extra treatment or very expensive monitoring.

A hedonic study found property prices are raised when they are located near major wetlands with recreational opportunities. For example, the Ramsar-listed Thomsons Lake and The Spectacles wetlands have increased property values by about $390 million. Correspondingly these values will be lost were the wetlands to dry completely. Raised or maintained groundwater levels can benefit communities through providing more or secure water for irrigating public open space, as well as water for peri-urban horticulturalists.

Groundwater modelling shows infiltration of between 1.7 and 3.5 GL/yr of recycled water could move the salt water wedge back several hundred metres making coastal bores feasible and reducing the risk posed for bores located up to 2 km inland. Nitrogen loads are not projected to rise in a drying climate even with these two levels of MAR because fluxes will be more substantially reduced. It is currently unclear whether nitrogen is adversely affecting seagrass health given the improved water quality in the Sound and the continuing deterioration in nearly Warnbro Sound which does not receive industrial water.

Groundwater modelling showed that the water regime in groundwater-throughflow wetlands could be improved or maintained in a drying climate through MAR, especially in the north and east of the catchment. These and community benefits were not included in the cost benefit analyses of MAR.

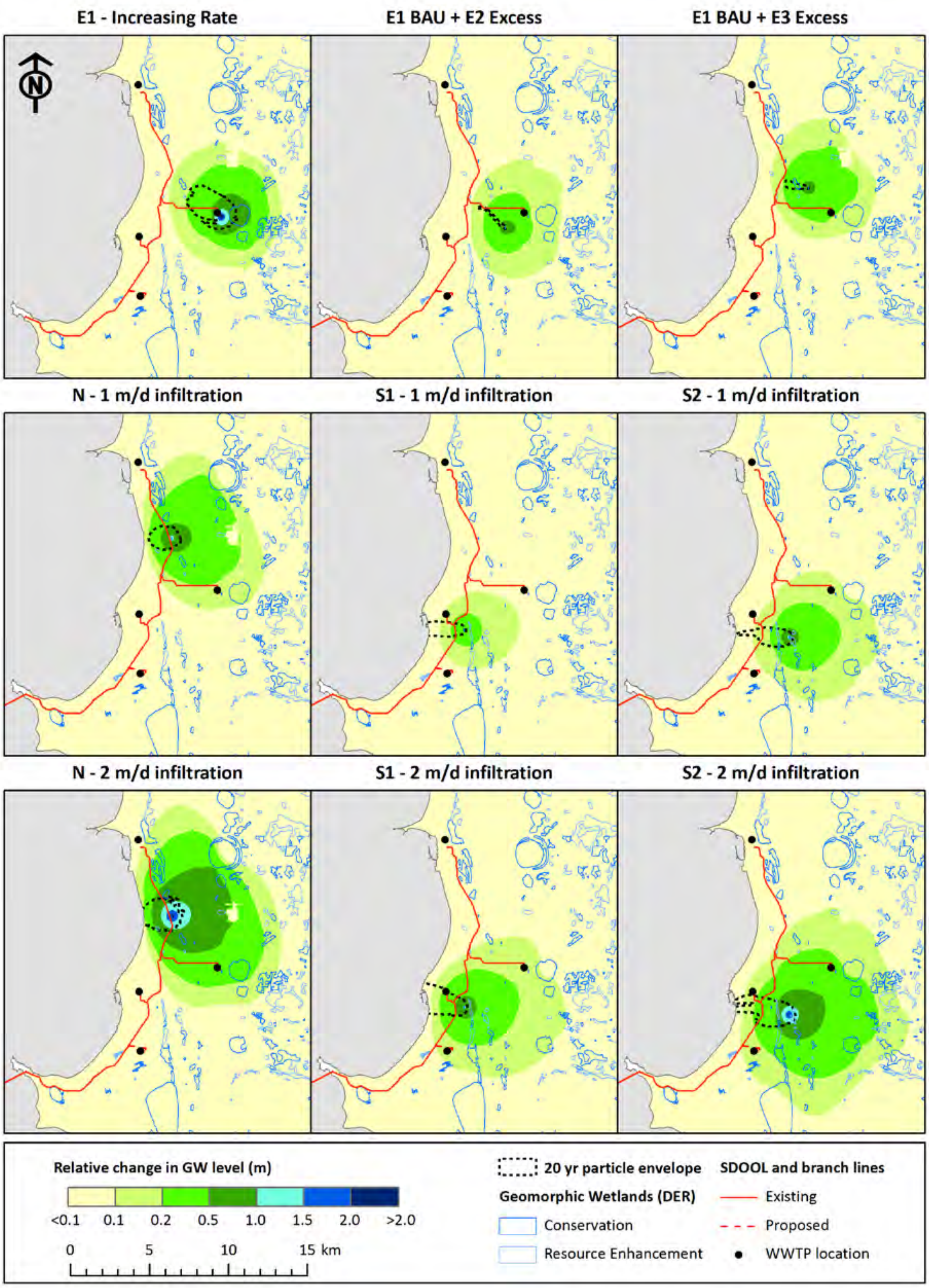

Figure 2. Relative change in predicted groundwater levels between each MAR scenario and the BAU (at the end of the 20 year simulation).The maximum areal extent of particle tracking pathways at the end of the 20 year simulation is superimposed on the maps.

CONCLUSIONS

The investigation reached the following main conclusions:

1. Managed aquifer recharge appears to be a cost-effective future non-potable water source for heavy industry in the Kwinana Industrial Area under the assumptions used in this investigation.
2. Discharging treated wastewater to the Superficial Aquifer at the Kwinana WWTP appears to have helped save up-gradient wetlands (The Spectacles) from drying as most other wetlands in the same chain have in recent decades. There is no evidence that the treated wastewater has contaminated the wetlands or down gradient groundwater which has flowed through the industrial area and eventually discharging into Cockburn Sound. Recent upgrades to the WWTP resulted in lower total nitrogen concentration water than comes from organic matter mineralisation in the wetland base.
3. Risks and management costs area associated with the following assumptions and site conditions;
    a. The degree of additional treatment. The treated wastewater may require particulate / solids removal (if added through galleries) or nitrogen removal (if loads pose a risk to downstream wetlands and Cockburn Sound)
    b. Distance to the treated wastewater access point. Long distances to the wastewater access point has a significant effect on the infrastructure cost of each MAR option.
    c. Whether MAR water mobilises or interacts with contaminated site water in a negative or positive way (e.g. by enabling more rapid reclamation)
    d. The fate of added water in terms of mixing with ambient groundwater, nutrient transformation or degradation process, uptake in bores or expression at the surface in wetlands or the Sound.
    e. Whether all added MAR water can be used or a proportion needs to be retained for environmental purposes in a drying climate
    f. Travelling time to the coast. The greater the distance from the coast the longer the residence time and hence the greater the potential for removal of nutrients within the aquifer by natural processes or by extraction
    g. Infiltration volume. High infiltration volumes reduce the average cost per kilolitre of the MAR project, as well as increase water supply reliability to end-users. Environmental benefits may also be realised should infiltration be lower than abstraction. However, these economic and environment benefits need to be balanced with the risk of increased nitrogen and phosphorus loads entering Cockburn Sound.
    h. How effective MAR water is at reversing seawater intrusion.

The findings support a number of prior investigations in terms of groundwater responses to MAR, flow rates through the aquifer, discharge to the Sound and nitrogen loads.

RECOMMENDATIONS

There remain areas requiring site-specific investigations to test these regional conclusions. One investigation already underway is on the importance of groundwater nitrogen loads compared with other sources of nitrogen in the water column (including bottom sediments) and whether nitrogen levels are no longer affecting water quality and seagrass health in Cockburn Sound.

The next steps in evaluating MAR may require site-specific investigations and the identification of a proponent to trial MAR so that remaining questions can be addressed in a timely manner before water shortages require an urgent solution be adopted.

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1 Introduction

Authors: Don McFarlane and Mike Donn

1.1 Background to the project

The population of Greater Perth exceeded 2 million in 2015 (ABS 2015). An unconfined aquifer occurs in a series of sand dunes that underlie most of this area. This 'Superficial Aquifer' forms two groundwater mounds, Gnangara north of the Swan Canning Estuary and Jandakot to the south (Figure 1.1).

The Superficial Aquifer has been a very important source of water for peri-urban horticulture, heavy and light industry, the irrigation of private and public parks and gardens and until recently, residential drinking water. The aquifer also forms many throughflow lakes where the watertable intersects the land surface, often in inter-dunal swales (Figure 1.1). The Jandakot Mound to the east of the study area provides 1 to 2 per cent of Perth's drinking water supplies.

Lower rainfall, likely to be the result of climate change, has affected the area since about 1975 and has caused groundwater levels to fall at the same time that demand for water has increased because of the reduced rainfall, slightly higher temperatures and an annual population growth of about 2.5%. As a result water allocations were reduced to some established users in the Gnangara Groundwater Management Area and this response may be extended to the Jandakot area following a review. The reductions have partly been made to protect the most important throughflow wetlands although these have continued to be lost resulting in much greater value being placed on those that remain.

In this backdrop, heavy industry in the Kwinana area in the southwestern part of Perth has had concerns for over ten years about the security of its existing process water supplies and the ability to provide water to new industrial entrants (Burns and Roe Worley 2006). A report for the Western Trade Coast identified the need to investigate the potential for Managed Aquifer Recharge (MAR) and for using water directly from the Sepia Depression Ocean Outfall Line (SDOOL) as low cost water supply options for industry as a high priority (SKM and REU 2013).

The Kwinana Industrial Area has an annual output worth AUD$15.77 billion and directly employs almost 5000 people with a further induced employment (multiplier effect) of about 26,000 people. About 60% of industrial process water (mainly cooling and dust suppression) used by the KIA comes directly from the Superficial Aquifer. An area in the northern part of the KIA has been affected by gradual seawater intrusion which limits groundwater extraction.

At the same time that recharge to the Superficial Aquifer has reduced, wastewater flows from the growing residential areas of Perth have been increasing annually by 2 to 3% (or 3 to 4 GL). Currently about 50 GL per annum of advanced secondary treated wastewater is piped past the KIA and disposed of into the Indian Ocean through the SDOOL (Figure 1.2). A further 1.7 GL/y is infiltrated into the Superficial Aquifer at the Kwinana wastewater treatment plant (WWTP).

To put these volumes into perspective, groundwater extraction from the study area is about 40 GL/y and it meets about 60% (18 GL/y) of the heavy industrial water demand of about 30 GL/y. As detailed in Chapter 4, groundwater also supplies about 3 GL/y to public open space irrigation and a further 5 GL/yr for irrigated agriculture in the Cockburn Groundwater Area (Figure 2).

This project was established to evaluate options for diverting more of the treated wastewater into the Superficial Aquifer to replenish depleted groundwater resources, provide more water to industry, stabilise or reverse seawater intrusion and possibly aid the wetlands in the area. At the same time the infiltration should not significantly increase nitrogen loads into the Cockburn Sound which has been affected by algal growth and seagrass loss, adversely mobilise contaminated sites or pollute the aquifer so that its utility would be reduced. The Cockburn Sound catchment is covered by an environmental protection policy managed through the Cockburn Sound Management Council.

The project was conceived to build upon recent investigations of managed aquifer recharge carried out in similar soil and aquifer conditions; the Floreat Infiltration Galleries (Bekele et al. 2011), Perry Lakes (McFarlane et al. 2009) and the MARRO project (Vanderzalm et al. 2015; Bekele et al. 2015).

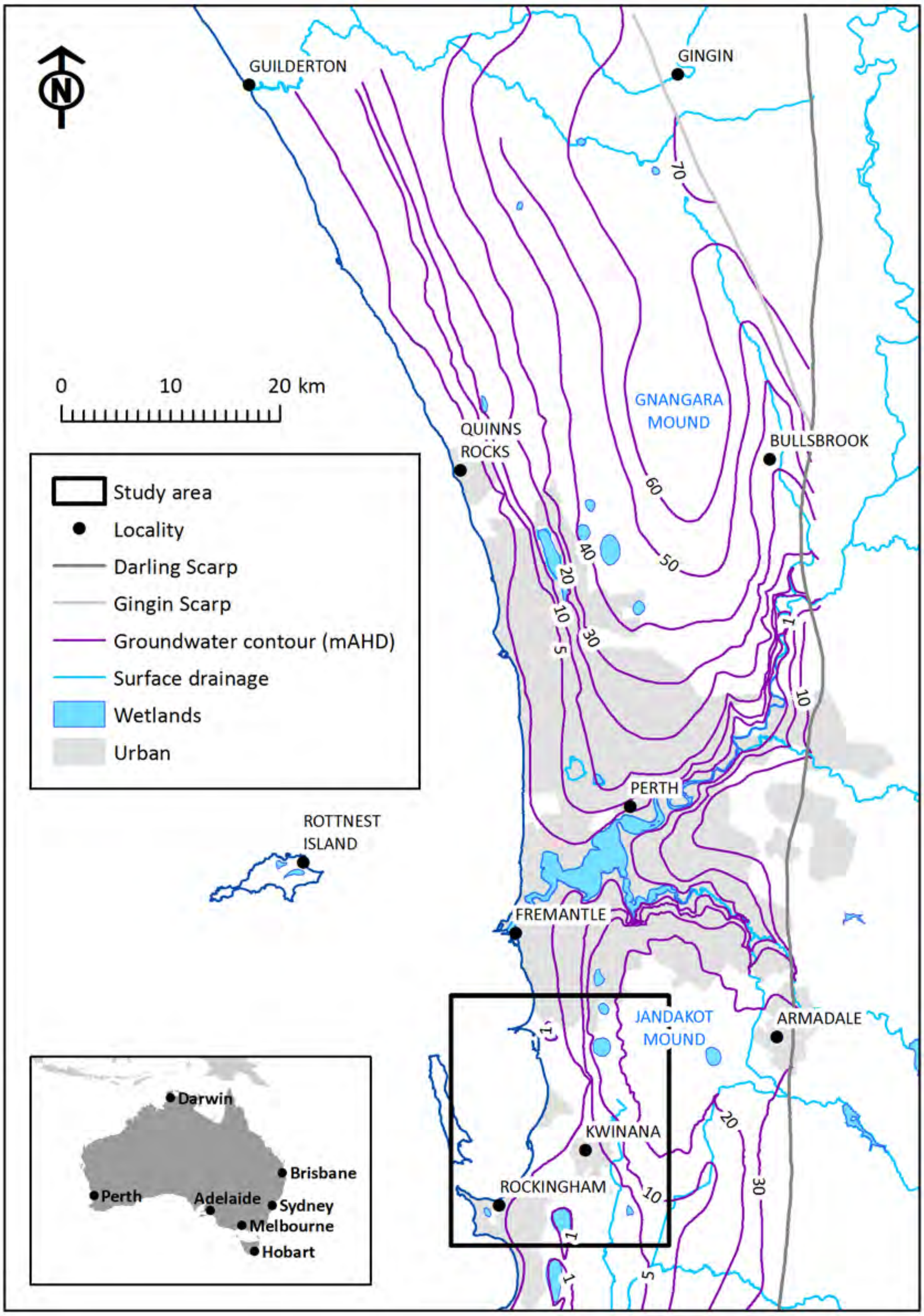

Figure 1.1 The Superficial Aquifer in the Perth area and relation to the study area

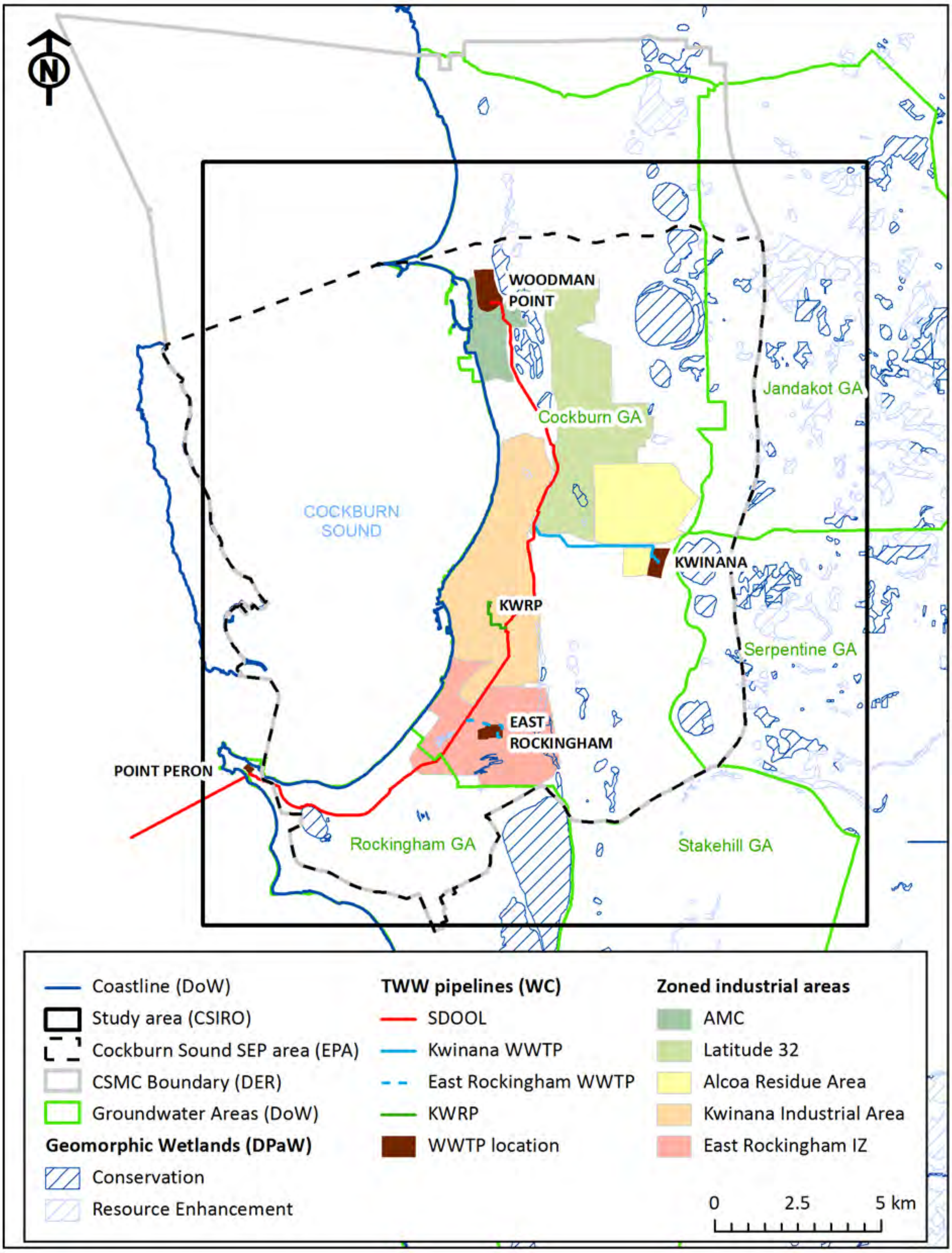

Figure 1.2. The study area in relation to wastewater treatment plants and pipelines (Water Corporation, WC), Cockburn Sound Management Council (CSMC; Department of Environment Regulation, DER) and Cockburn Sound State Environmental Policy (SEP; Environment Protection Agency, EPA) administrative boundaries, and groundwater management areas (Department of Water, DoW)

1.2 Objectives

The project aims to:

- Determine the cost effectiveness of heavy industry accessing appropriately treated recycled water via managed aquifer recharge (MAR) compared with the piped distribution of 1) reverse-osmosis treated wastewater, or 2) secondary treated wastewater;
- Examine options for additional benefits from recycling via MAR – preventing sea water intrusion, recovering throughflow wetlands and providing irrigation water for local government; and
- Assess methods for managing nitrogen pollution risks using additional pre-treatment, potential of nitrification–denitrification in the soil-aquifer system and plume interception.

1.3 Report structure

Chapter 2 provides an overview of the climate, physiography and landuses in the Cockburn Sound Catchment as it has been identified by the Cockburn Sound Management Council. The catchment's hydrogeology and groundwater management is described in Chapter 3 so that subsequent work on groundwater modelling and managed aquifer recharge can be placed into a larger context. Water demand and supply options are summarised in Chapter 4 to identify the emerging future water supply gap to industry. The nature of the growing treated wastewater resource is detailed in Chapter 5 while previous work on its addition to the Superficial Aquifer through managed aquifer recharge (MAR) is summarised in Chapter 6. The value of throughflow wetlands in the Perth area, including hedonic valuation methods are outlined in Chapter 7.

While some new material is presented in these first seven chapters, the main project results are presented from Chapter 8, starting with the local area groundwater model that was developed for the Cockburn Sound Catchment. Simulations are reported that identify areas in the catchment where MAR may be more effective as a result of the aquifer's hydrogeological properties, the depth of the watertable and the location of the salt water interface. Chapter 9 details environmental, economic, engineering and health constraints to MAR within the catchment.

Scenarios of the most promising MAR sites are detailed in Chapter 10 and Appendix A. These are discussed in Chapter 11 before recommendations are made (Chapter 12) and conclusions drawn (Chapter 13).

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2 Background

Authors: Don McFarlane, Mike Donn and Irina Emelyanova

Key points

- The study area (307 km^2) has sandy soils with chains of throughflow wetlands in inter-dunal swales
- The area has a Mediterranean climate with most areas experiencing lower rainfalls since about the mid 1970s as is common in south-west Western Australia
- Many lakes have dried but one series close to the Kwinana Wastewater Treatment Plant infiltration ponds have been largely unaffected
- The area has undergone rapid urbanisation since the mid 1990s
- Nutrient pollution in the 1980s caused eutrophication in Cockburn Sound and the loss of seagrass meadows
- Environmental responses have addressed many point sources pollution sources but the continued role of groundwater in contributing to nitrogen loads is unclear
- Seagrass health continues to decline despite the improvement in water quality in the Sound so the role of groundwater is an issue that this project partly addresses

2.1 Location

One of the potential constraints to adding treated wastewater to the Superficial Aquifer in the area around the Kwinana Industrial Area is the additional nitrogen load entering Cockburn Sound, a habitat that has already been adversely impacted by nitrogen discharges in the past (DAL 2001). A *State Environmental (Cockburn Sound) Policy 2005* was released by the Minister for Environment in January 2005 (EPA 2015). An Environmental Management Plan for Cockburn Sound and its Catchment was also released by the Cockburn Sound Management Council in 2005 (DoE 2005).

The study area was extended to cover the entire Cockburn Sound Management Plan area and adjacent areas that would affect groundwater levels and the value of housing around key wetlands (Figure 2.1). The area of the terrestrial part of the catchment is about 307 km^2.

The area falls within three local government authorities; the City of Cockburn, Town of Kwinana and the City of Rockingham. The collective industrial Zone is known as the Western Trade Coast which comprises the Kwinana Industrial Area (heavy industry), the Australian Marine Complex (AMC), Latitude 32, the Rockingham Industrial Zone (RIZ) and the ALCOA Residue Area (Figure 1.2). The AMC and Latitude 32 are light industrial zones while the RIZ can contain both light and heavy industry. Only the AMC at Henderson is approaching full development.

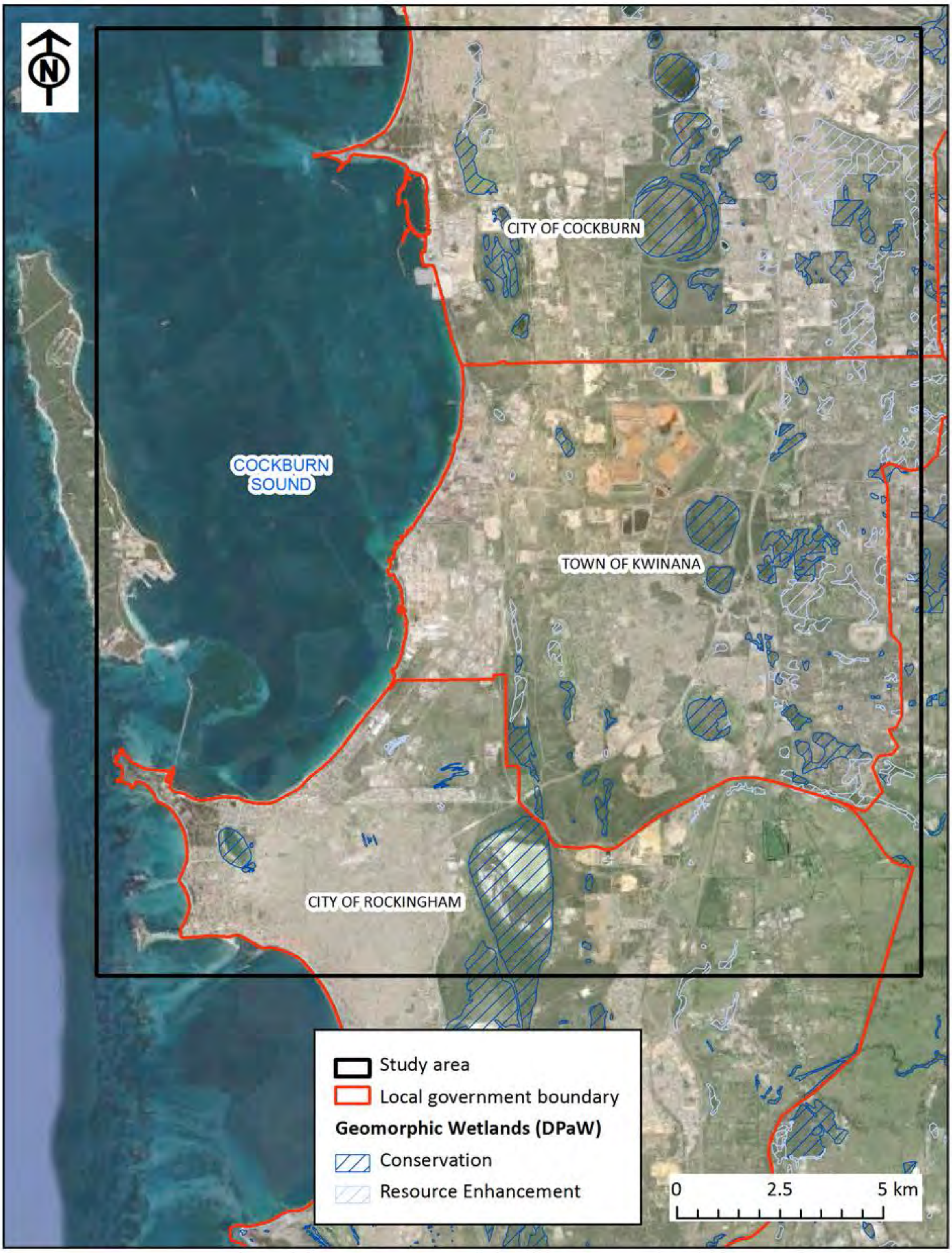

Figure 2.1 Study area in relation to local government authorities, industrial zones and natural features

2.2 Climate

The area has a Mediterranean climate with cool wet winters and hot dry summers. Climate statistics for the Kwinana BP Refinery (Site number 009064; 32.23^{0}S; 115.76^{0}E) for the period between 1955 and 2012 show a mean annual temperature of 23.2^{o}C, with a monthly maximum of 29.5^{o}C in February and minimums of 10.6^{o}C in both July and August (BoM 2015). The mean annual rainfall for this period has been 745mm with a mean monthly maximum of 157 mm in June and a mean monthly minimum of 9.1 mm in December (Figure 2.2). About 90% of the annual rainfall occurs in the seven months between April and October. The 30 year period rainfall between 1981 and 2010 is also shown in Figure 2.2. Like many areas in SW Western Australia, the April to July rainfall is now less and the annual average was 711 mm.

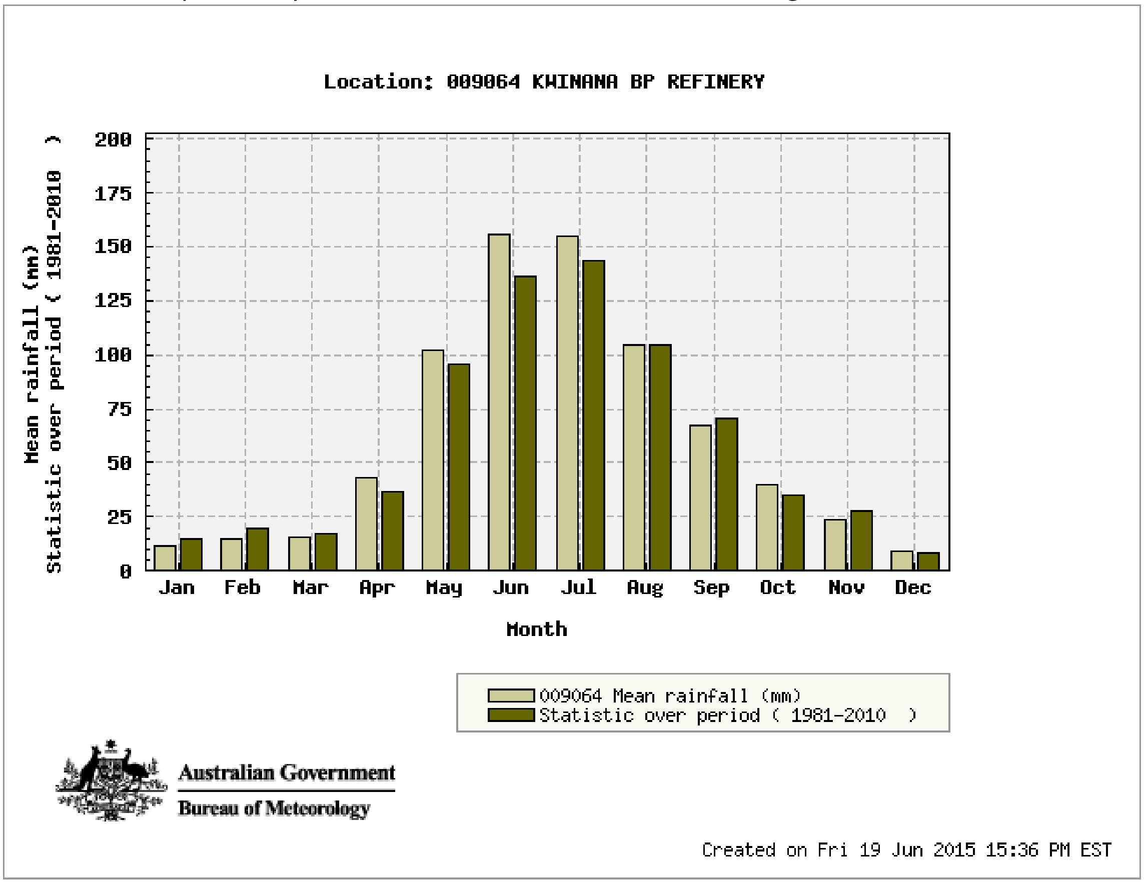

Figure 2.2 Monthly distribution of rainfall between 1955 and 2012 for the Kwinana BP refinery and the 30 year period 1981 – 2010 (Bureau of Meteorology website)

Similar statistics for the Jandakot Aero (Site number 009172; 32.10^{0}S; 115.88^{0}E) 18 km further north east are shown in Figure 2.3. At this station the 1955 to 2012 average rainfall was 819mm and the 1981 to 2010 mean rainfall was slightly higher at 832mm.

Climate records are much shorter at the Garden Island (Site number 009256; 32.24^{0}S; 115.68^{0}E) located 8km south of the BP refinery station (Figure 2.4). Average monthly rainfall in all months is less than 125mm and the annual average has been only 593mm during this period.

In summary, the degree of recent drying in the study area is variable. Historical rainfall data has been used for groundwater model calibration as described in Chapter 8. This chapter also defines the future climate that is used in scenarios.

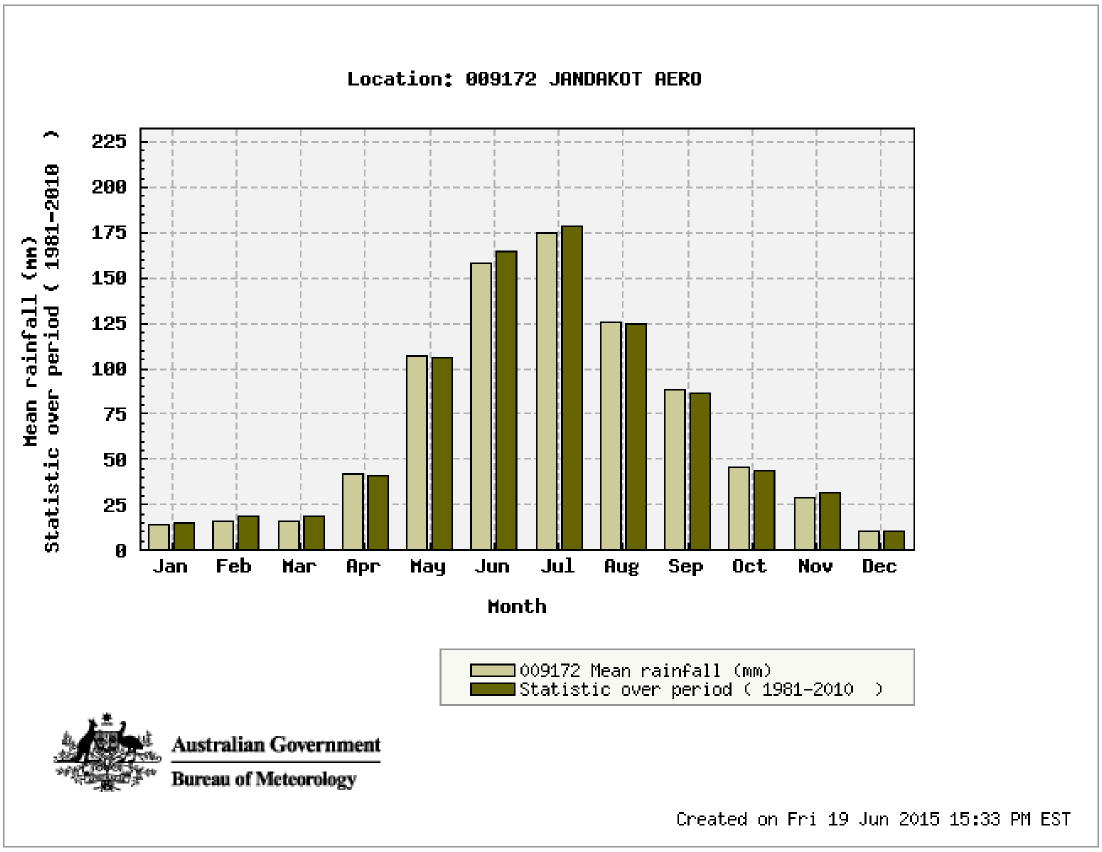

Figure 2.3 Monthly distribution of rainfall between 1955 and 2012 for the Jandakot airport and the 30 year period 1981 to 2010 (Bureau of Meteorology website)

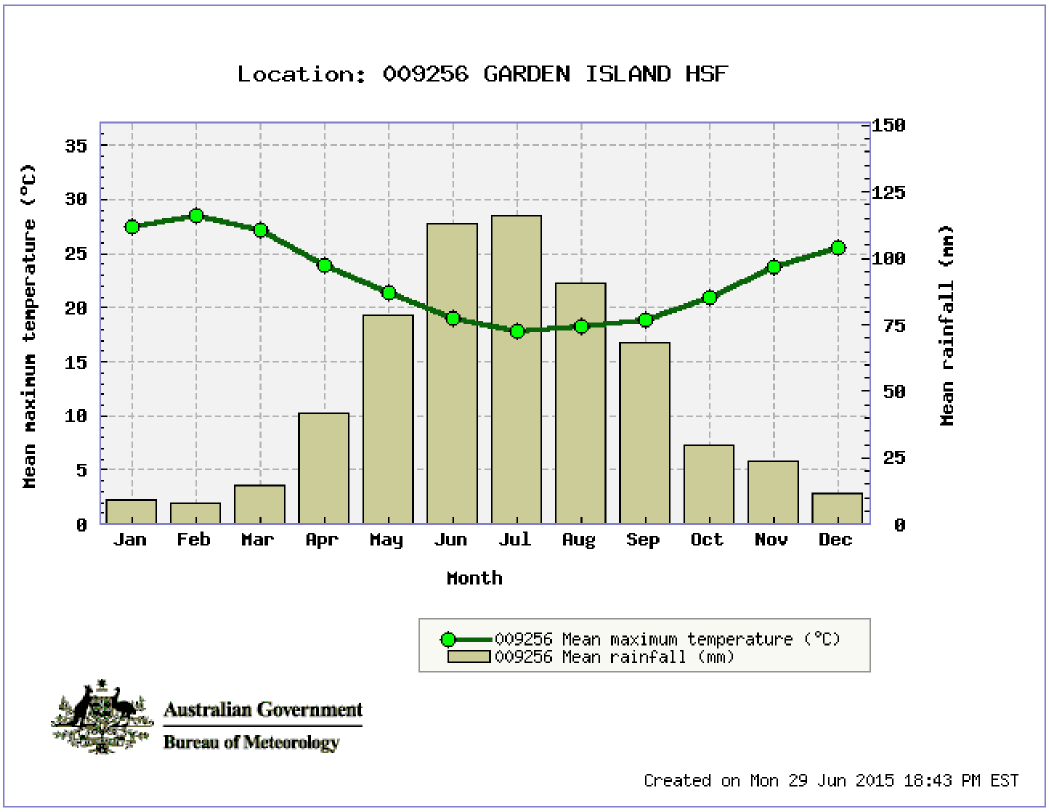

Figure 2.4 Rainfall and temperature at Garden Island between 2001 and 2015 (Bureau of Meteorology website)

The distribution of annual rainfall over the catchment during the baseline period of 1990 to 2013 as assessed using the SILO 5 x 5 km grid shows a slightly higher rainfall on the coast and in the east (around Jandakot) during this 24 year period (Figure 2.5). Annual evaporation rates are estimated to be slightly higher in those lower rainfall zones. It is not expected that this spatial variation would substantially affect modelled groundwater levels given the high transmissivity of the aquifer, especially in coastal areas.

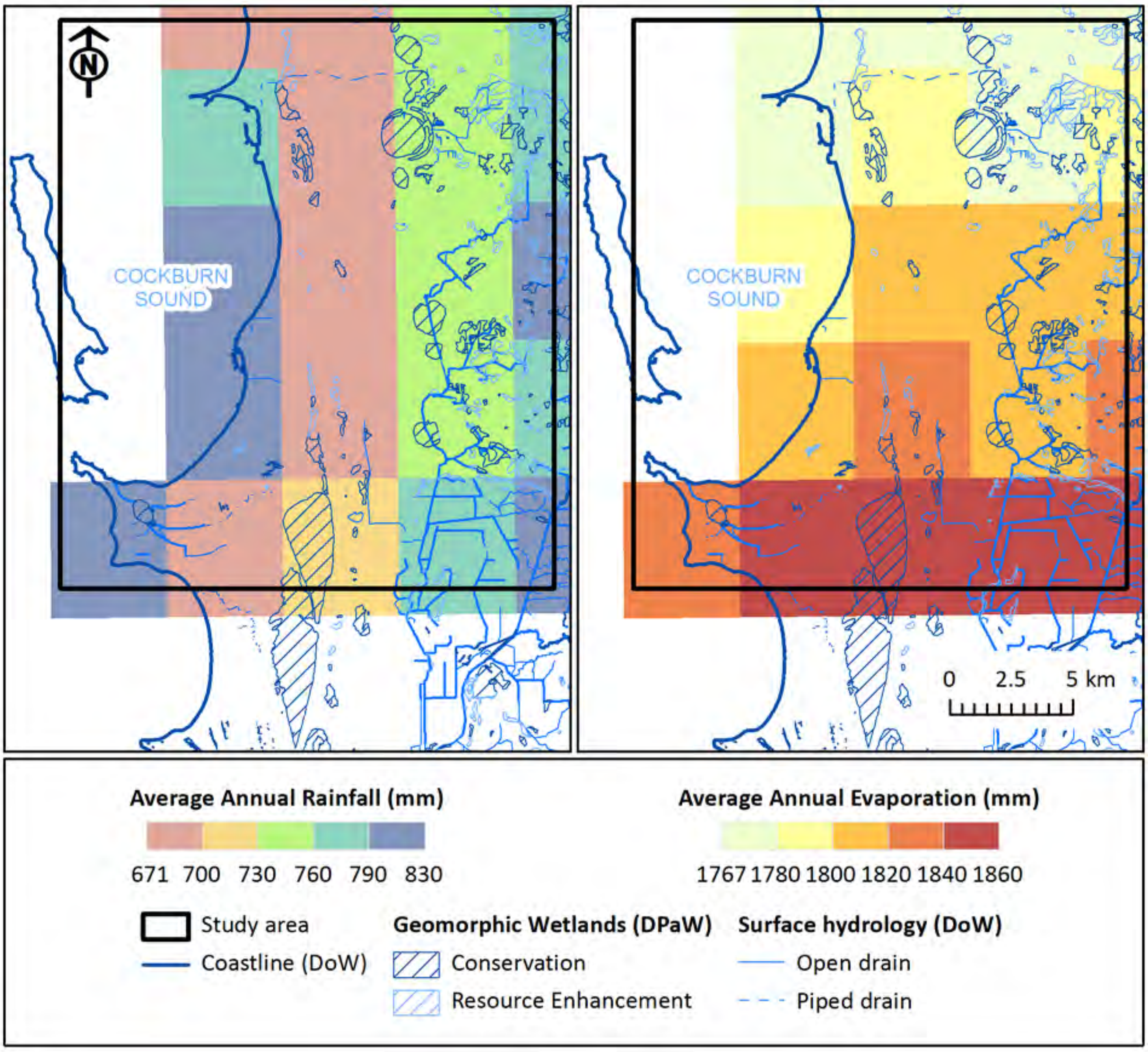

Figure 2.5 Distribution of annual rainfall and Class A pan evaporation over the catchment between 1990 and 2013 as estimated using SILO gridded data (Source: Queensland Government 2015)

2.3 Physiography

The sandy nature of soils over most of the catchment results in rainfall infiltrating close to where it falls and an absence of natural surface drainage lines. Artificial drains connect some wetland chains to the ocean and to the Serpentine River in the south (Figure 2.6). These were constructed to reduce the risk of inundation of nearby properties and to access more fertile peaty soils after settlement. Falling groundwater levels has resulted in these drains carrying much less water than in past wet periods such as the 1960s.

The highest elevations in the catchment are associated with two sets of Tamala Limestone dunes running north–south through the catchment (Figure 2.6). Two industrial disposal sites are amongst the highest elevations; the ALCOA alumina tailings storage (NW of Spectacles North) and a construction and demolition landfill site to the south west (2km south of Long Swamp). The south west of the catchment is characterised by low elevations (< 10m AHD) and beach ridges associated with the Safety Bay Sand Formation. The inter-dunal area between the two limestone ridges has a low elevation, as does an area in the south east around Bollard Bullrush Swamp. As is shown later these areas have a relatively low depth of the watertable and consequently seasonally wet areas.

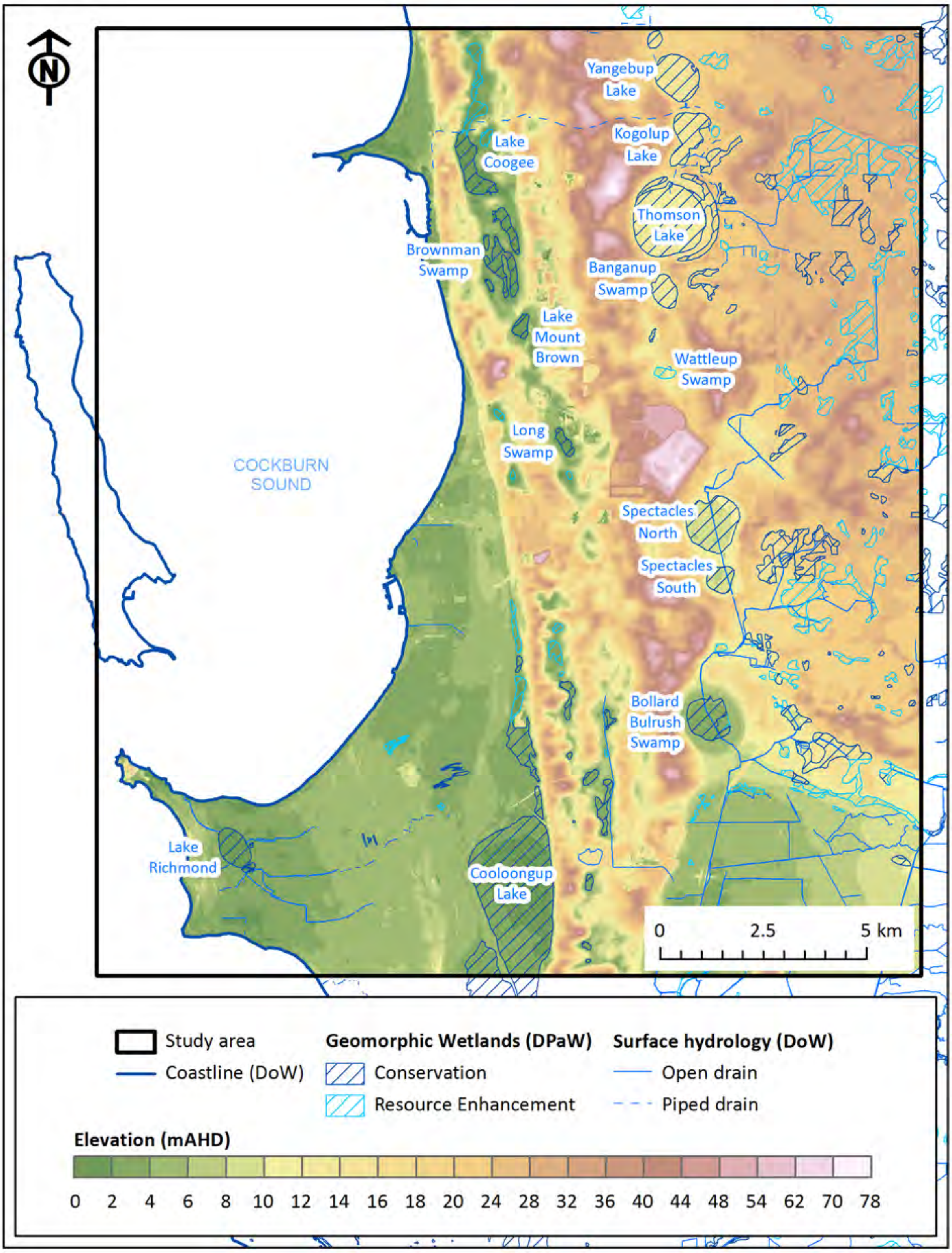

Figure 2.6 Elevation of the Cockburn Sound catchment showing constructed drains and wetlands

2.4 Wetlands

Two lines of inter-dunal wetlands within the Beeliar Regional Park extend into the catchment (CALM 2006). The western chain lies behind the coastal dunes about two kilometres from the coast and includes Lake Coogee, Brownman Swamps and Lake Mount Brown (Figure 2.6). The eastern chain lies 6 to 7 km from the coast and includes Ramsar-listed Thomson Lake in the north, Banganup Swamp and The Spectacles (so called because the pair of lakes is connected by a drain that make it resemble a pair of spectacles). Ballard Bullrush Swamp in an extension of this eastern chain but it lies outside the Beeliar Regional Park. Wattelup Swamp, Long Swamp and Bollard Bullrush Swamp have been identified as deserving formal protection by CALM (2006). Wetlands in Beeliar Park that lie within the catchment have been zoned for Conservation and Protection with the exception of Banganup Lake which has been designated for Special Use because it lies within the fenced-out Harry Waring Marsupial Reserve.

2.5 Landuse over time

The Cockburn Sound catchment is undergoing urbanisation but there are constraints to where residential development can occur because of the need for an adequate buffer around industry and areas of environmental value (e.g. Conservation Category Wetlands, Bush Forever sites). The changes affect the amount of recharge and discharge and therefore the groundwater model inputs outlined in Chapter 8.

Landsat TM images were developed for the main land use categories every second year between 1988 and 2002 and then every year until 2012. The results illustrated in Table 2.1 with examples from 1988, 2000 and 2012 shown in Figure 2.7, Figure 2.8 and Figure 2.9 respectively. Urban and commercial areas have expanded especially rapidly in the mid 1990s. This has been mainly at the expense of the three native vegetation classes which have different water use characteristics, and urbanisation of areas dryland agriculture. The density of vegetation can change between years depending upon the seasonal rainfall. The years 2001 and 2006 were especially dry and the area of low density vegetation increases in following years.

Table 2.1 Land use in the Cockburn Sound Catchment between 1988 and 2012 as determined from Landsat TM imagery (km^2)

Class / year	1988	1990	1992	1994	1996	1998	2000	2002	2004	2006	2008	2010	2012
Low density native vegetation	57	50	52	52	44	52	18	52	52	39	44	45	43
Medium density native vegetation	38	30	31	30	25	27	20	24	27	27	24	22	21
Medium to high density native vegetation	37	45	37	29	35	23	65	27	20	28	20	20	19
Dryland agriculture	150	145	140	137	124	123	117	114	114	115	118	114	114
Summer wet/ irrigated /seeps	2	3	3	3	3	4	6	5	4	5	3	3	4
Urban residential/ bare soil	14	22	32	41	50	52	54	56	60	63	67	69	72
Urban commercial	1	2	4	6	16	17	18	19	21	21	22	23	24
Water/estuaries	9	9	9	9	9	9	9	9	9	9	9	9	9

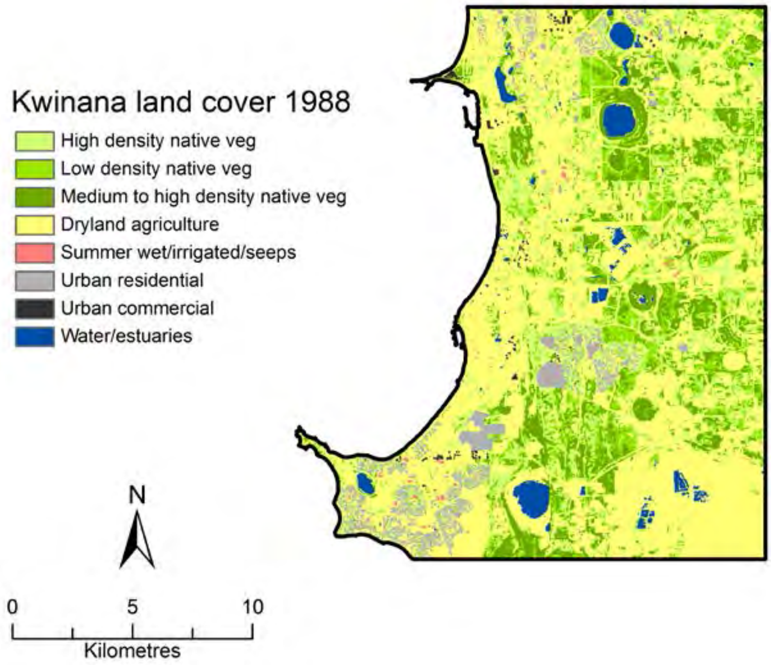

Figure 2.7. Land use in the Cockburn Sound Catchment in 1988 as determined by remote sensing analysis. The Urban commercial area is not shown on this figure

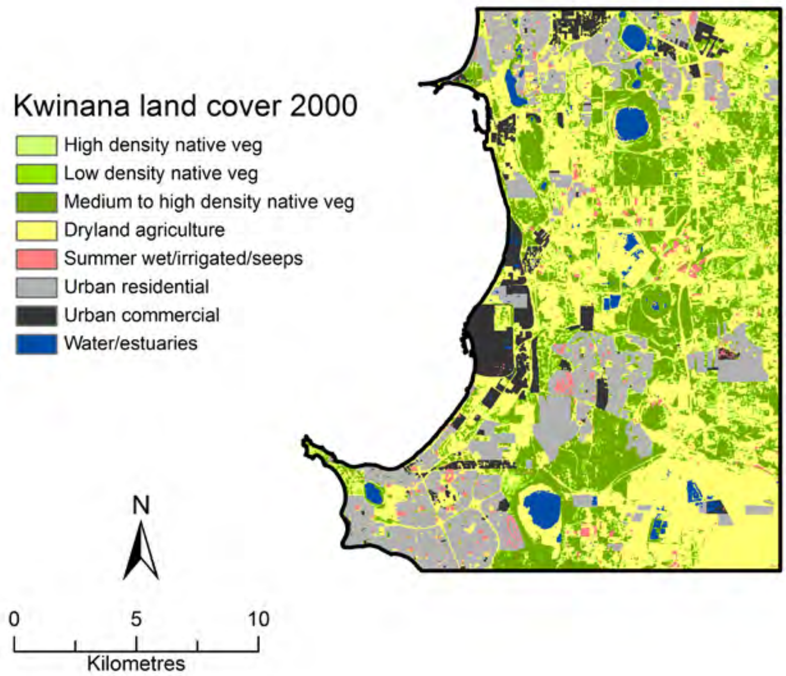

Figure 2.8 Land use in the Cockburn Sound Catchment in 2000 as determined by remote sensing analysis

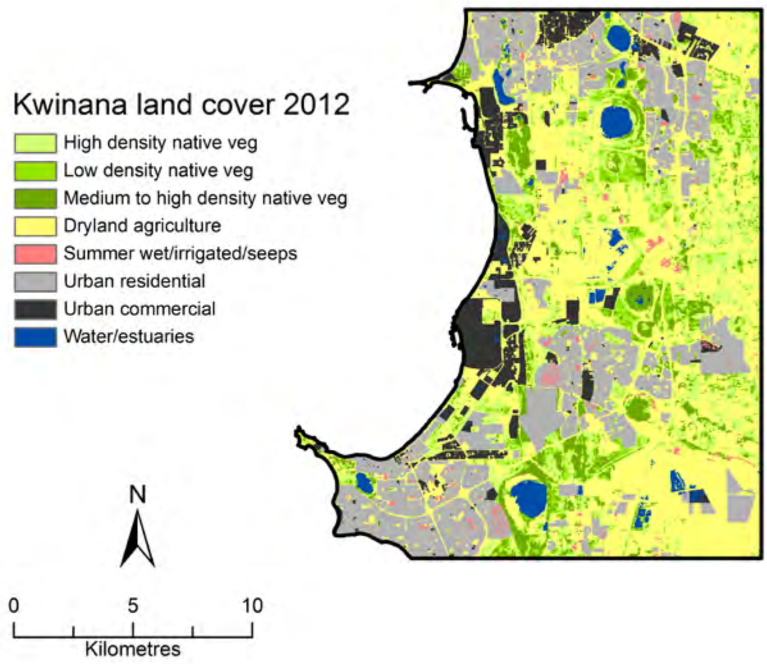

Figure 2.9 Land use in the Cockburn Sound Catchment in 2012 as determined by remote sensing analysis

There has been a substantial change in both vegetation (Normalised Difference Vegetation Index) and wetness (Normalised Difference Wetness Index) in the catchment in the twenty years between 1990 and 2010 (Figure 2.10). Urbanised areas have experienced a substantial reduction in greenness but there has also been a reduction in the degree of greenness in native vegetation areas, reflecting the statistics shown in Table 2.1. There are some areas of increased greenness associated with revegetation after urbanisation as lawns and gardens are established.

The wetness index shows many Beeliar chain lakes have decreased areas of open water between 1990 and 2012 (Figure 2.10). The exceptions are The Spectacles lakes and Ballard Bullrush Swamp, and to a lesser extent, Lake Richmond (see Figure 2.6 for lake locations). The former lakes are connected by a surface drain and the larger of The Spectacles is located within 400 m of the Kwinana WWTP infiltration ponds. Lake Richmond is a deep lake located in highly transmissive Tamala Limestone in an area where levels are affected by sea levels and tidal influences. Some areas have increase in both greenness and wetness and this can be associated with urban areas that have been revegetated and now receive summer irrigation. Summer wet areas also include horticultural crops that are being irrigated.

The results above have also been documented for the period between 1999 and 2011 by Tulbure and Broich (2013). They found that the low rainfall year 2010 had an especially large impact on the availability of open water in the study area.

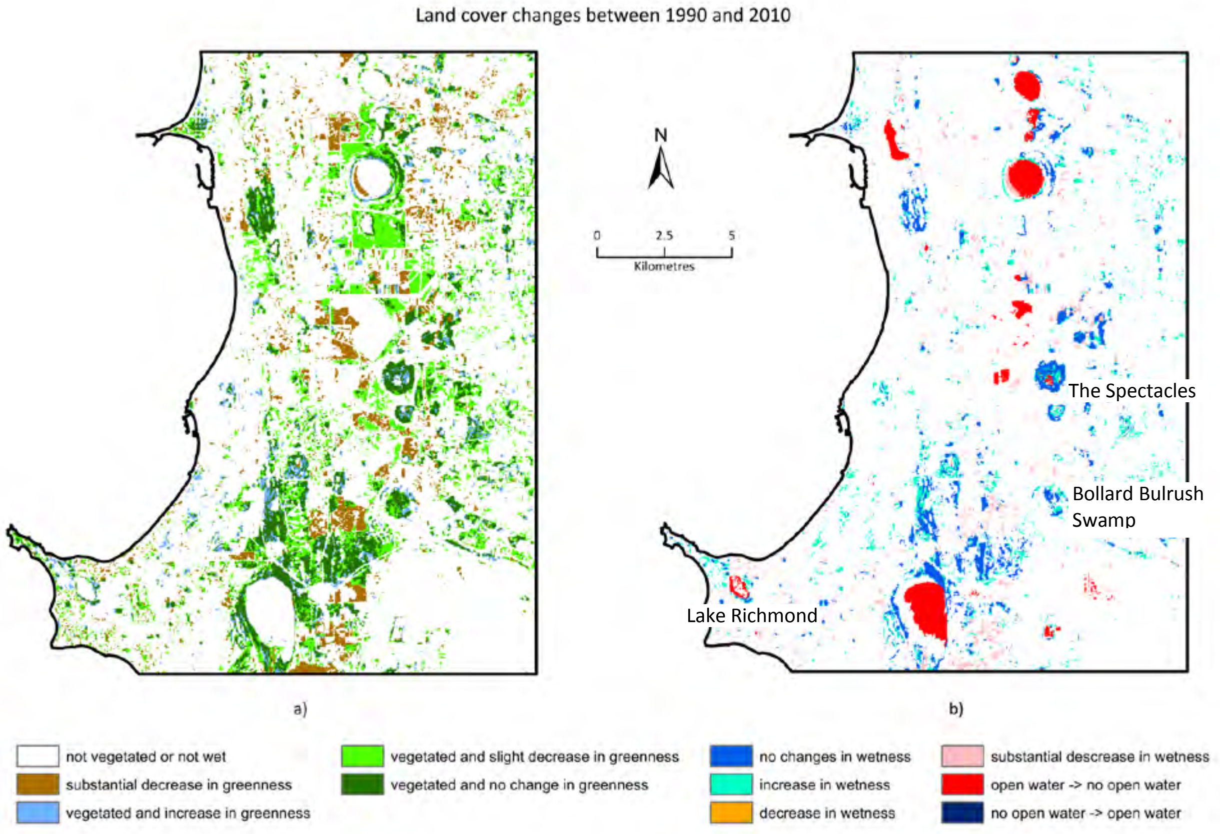

Figure 2.10 Change in vegetation (left) and in wetness (right) between 1990 and 2010

2.6 Cockburn Sound and nutrient loads

The Cockburn Sound forms the western boundary of the study area. It is approximately 16 km long and 9 km wide (Figure 2.1). It is protected from ocean swells by Garden Island and is ideal for recreation activities as well as providing port facilities for industries located on its shore (Cockburn Sound Management Council, 2001). In the past the disposal of municipal and industrial wastewaters into Cockburn Sound has adversely affected the ecology. It was estimated that 5000 kg/day of nitrogen and 3770 kg/day of phosphorus were discharged directly into the Sound in the late 1970s (Department of Conservation and Environment, 1979). These high nutrient loads into the Sound were linked with increased phytoplankton levels and a decline in the seagrass meadows.

Reduced point source inputs to the Sound from the Kwinana Nickel Company/CSBP and the Woodman Point WWTP resulted in improvements in water quality in the early 1980s (Cockburn Sound Management Council, 2001). However declines in water quality became evident in the late 1980s instigating the DEP Southern Metropolitan Coastal Waters Study (Department of Environmental Protection, 1996). This study found that nutrient-related water quality was only marginally improved in the early 1990s relative to the late 1970s. Due to decreased point source inputs, the nutrient loads to the Sound were shown to be associated largely with groundwater discharges, with approximately 70% of nitrogen inputs associated with groundwater discharge from two industrial sites Western Mining Corporation/CSBP and an area adjacent to Jervoise Bay.

In order to protect the environmental quality of Cockburn Sound the Western Australian government developed the State Environmental Policy for Cockburn Sound (SEP) which was enacted in January 2005 (Government of Western Australia, 2005). The nutrient related environmental quality criteria are based on chlorophyll a concentrations, light attenuation coefficient, phytoplankton biomass and seagrass root density (Environmental Protection Authority, 2005). While not set out in the SEP for the Sound it is still important to understand the potential sources of nutrient inputs, including groundwater discharge.

2.6.1 GROUNDWATER DISCHARGE AND NUTRIENT LOADS INTO COCKBURN SOUND

A number of studies have estimated the discharge of groundwater into Cockburn Sound and the associated nutrient loads. Smith and Nield (2003) summarised a number of studies conducted between 1991 and 2000 with submarine groundwater discharge (SGD) estimates ranging from 0.7 to 9.0 m^3/d/m of shoreline. These studies were conducted at various scales from local measurements to catchment based modelling. The modelling approach of Smith and Nield (2003) produced estimates of the spatially-averaged SGD to be 2.5 to 4.8 m^3/d/m of shoreline. However local variations resulted in point estimates of SGD under the two groundwater recharge scenarios (low and high) varying from 1.4 to 4.6 m^3/d/m and 2.4 to 7.9 m^3/d/m, respectively. More recently, Burnett et al. (2006) estimated SGD to be between 2.0 and 3.7 m^3/d/m using a number of different methods, seepage meters, radium isotopes and radon. Based on bulk ground electrical conductivity measurements Stieglitz et al. (2008) estimated SGD to be 2.9 m^3/d/m. While Loveless and Oldham (2010) produced estimates based on the hydraulic head difference along a single transect of between 2.1 and 2.7 m^3/d/m for a low hydraulic conductivity case (4.5 m/d) and 4.7 and 5.9 for a mid-range hydraulic conductivity case (10 m/d). Therefore regardless of the method of estimation used, SGD from these studies is in a similar range.

Smith et al. (2003) reviewed groundwater nitrogen loads to Cockburn Sound, including identifying the major nitrogen plumes the majority of which were dominated by ammonium (NH_4^+). The estimates of total N loads varied from 180 to 450 t-N/yr for the period 1978 to 2000. It was observed that most estimates prior to 2000 were in the order of 330 t-N/yr reducing to 220 t-N/yr in estimates by D.A. Lord in 2001. In turn Smith et al. (2003) estimated that the nitrogen load to Cockburn Sound was 234±88 t-N/yr, with the lower values corresponding to cases where SGD was determined based on a low groundwater recharge scenario. The nitrogen loads associated with SGD rely on measured groundwater or porewater concentrations. These are point measurements and as such extrapolating these spatially introduces errors into the load estimates, which contribute to the large uncertainty associated with the Smith et al. (2003) estimate. Another source of uncertainty arises from the potential for nitrogen transformation and removal as a result of biogeochemical processes along the flow path from the point source measurements. In a study of groundwater quality in the SGD zone, Loveless and Oldham (2010) showed that low oxygen conditions and high organic carbon availability in shallow groundwater were likely to contribute to the removal of nitrogen through coupled nitrification/denitrification reactions, thus potentially reducing the nitrogen load to Cockburn Sound.

Estimates of the phosphorus load to Cockburn Sound from SGD range between 2 t/yr (Appleyard (1994) in Smith et al., 2003) and between 1.3 and 2.5 t/yr depending upon the recharge scenario adopted by Smith et al. (2003).

Inputs of treated wastewater through managed aquifer recharge are likely to have a different speciation to the nitrogen and phosphorus already present in the aquifer and are discussed in Chapter 5. The impacts on Cockburn Sound will be dependent on the treated wastewater speciation as well as concentration, and the degradation within the aquifer before entering the Sound with SGD.

While SGD is an important source of nutrient inputs into the sound internal cycling of nutrient within the marine sediments may also be a relatively important source. In mesocosm experiments with sediments from Cockburn Sound, Li et al. (2013) observed that ammonium was released from the sediment as a result of organic matter degradation particularly at summer water temperatures.

Improvements in the quality of water in Cockburn Sound have been reported in recent traffic light reports made by the Cockburn Sound Management Council (CSMC) to the Environmental Protection Authority in accordance with the State Environmental (Cockburn Sound) Policy. However seagrass meadows have continued to decline and there appear to be factors affecting seagrass health other than high nitrogen levels (Hans Kemps pers. comm. 2015). As a result the CSMC have commissioned a study by UWA to identify what these factors may be because seagrass decline is also happening in Warnbro Sound which doesn't have identified nutrient discharges.

2.7 Conclusions

The Cockburn Sound Catchment has been the subject of investigations as a result of eutrophication affecting environmental and social values associated with seagrass loss in the Sound over the past 40 to 50 years. Some of the nutrients have come from groundwater containing nitrogen and other pollutants from industrial sources especially.

Before managed aquifer recharge with treated wastewater can be considered the continued contribution of nitrogen from groundwater needs to be placed into context which requires the catchment's physical and use characteristics to be understood. The slow progressive drying of the catchment over the same time period as industrialisation, urbanisation and increased groundwater use has contributed to groundwater levels declining (covered in future chapters) and the loss of throughflow wetlands of considerable environmental, social and cultural value.

Subsequent chapters detail the study area's hydrogeology, water uses and demands in some detail before exploring options associated with managed aquifer recharge to help overcome falling groundwater levels.

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3 Hydrogeology and groundwater management

Author: Elise Bekele

Key findings

- The study area catchment has a predominance of moderate- to highly-permeable strata in the Superficial Aquifer that can readily transmit both naturally- and managed aquifer-recharge water
- Declining water levels in the Superficial Aquifer in the Cockburn Groundwater Area suggests this aquifer is close to its sustainable limit
- Saltwater intrusion affects the water quality in areas of heavy groundwater abstraction in the Superficial Aquifer
- Groundwater discharge to Cockburn Sound and contaminant release from certain premises has been monitored in recent decades in response to the *State Environmental (Cockburn Sound) Policy*
- A review of groundwater pollutants indicates that nitrogen species are the dominant contaminant from industry, agriculture, landfills, septic tanks and urban development in the catchment
- Several industries and two landfills in the study area have been required to submit estimates of substance emissions to land under the National Pollution Inventory. Some increase in emissions to land have occurred for ammonia and for a range of metals and organic compounds
- There have been several previous groundwater models developed for areas that either encompass or overlap with the study area. A synthesis of conceptual models and approaches was considered in developing the local area groundwater model for this project

3.1 Hydrogeology of the catchment

3.1.1 GEOLOGICAL SETTING

The study area lies within the central onshore portion of the Perth Basin, an elongate structure referred to as the Dandaragan Trough, consisting of a succession of Permian to Recent sedimentary layers beneath the Swan Coastal Plain parallel to the N-S trending axis of the Daring Fault along the western margin of Western Australia (Figure 3.1; Davidson 1995, Timms *et al*. 2015). In the deepest part of the trough, the thickness of sedimentary layers overlying crystalline rock is up to 15 km based on a three-dimensional geological model of the Perth Basin by Timms *et al*. (2012).

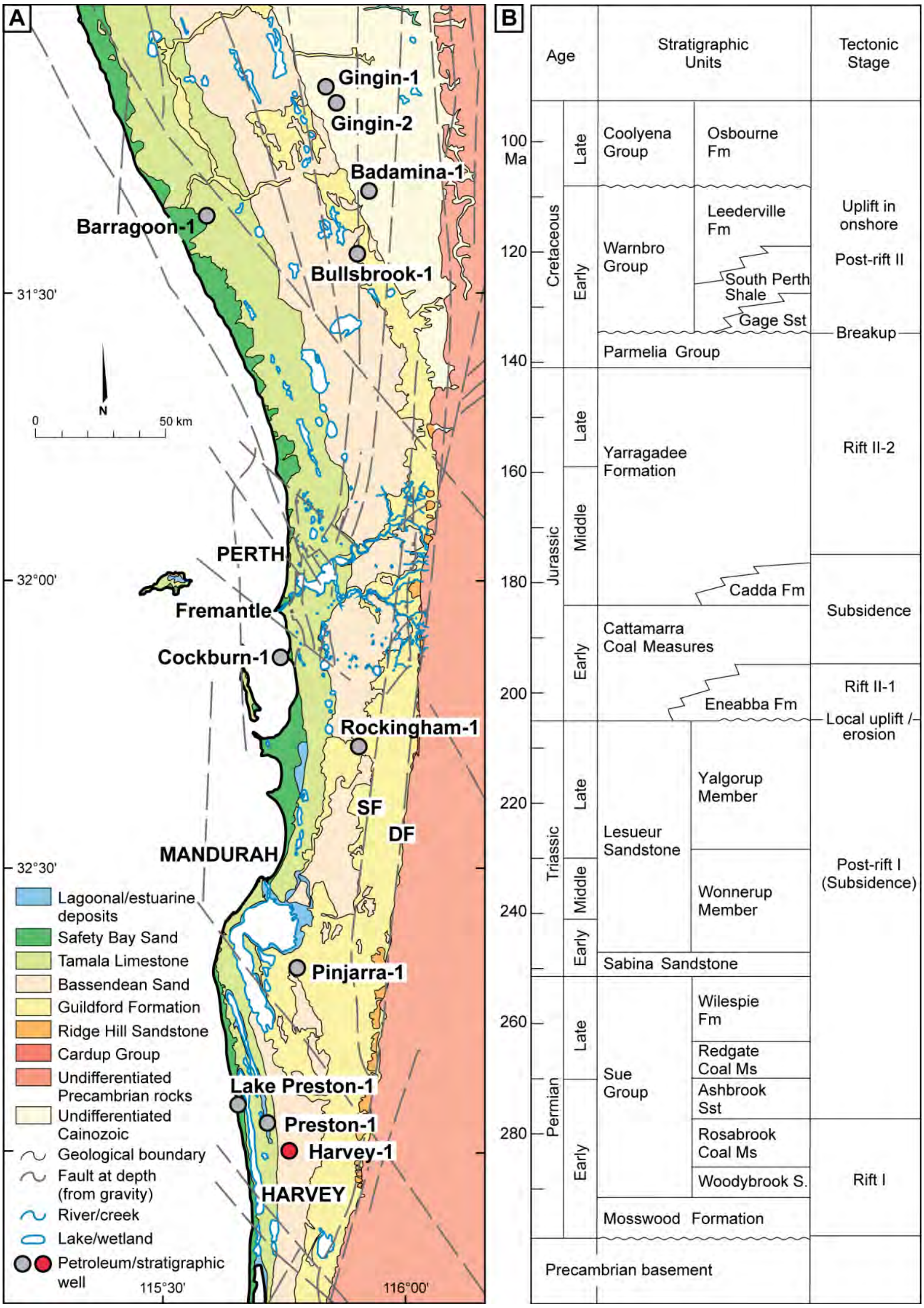

Figure 3.1 Geology of the central Perth Basin (Permian to Cretaceous units overlying Precambrian basement). Reprinted from "Facies-based rock properties distribution along the Harvey 1 stratigraphic well," by Delle Piane *et al*. (2013), CSIRO Report Number EP133710. Copyright 2012 by CSIRO. Reprinted with permission.

Details of the Cenozoic stratigraphy are given in Table 3.1. The stratigraphy in the Dandaragan Trough consists mainly of Permian to Cretaceous age formations, but overlying the Cretaceous formations are the Tertiary-Quaternary age formations referred to collectively as the superficial formations. The maximum total thickness of the superficial formations in the central onshore Perth Basin is approximately 65 m (Plate 49 in Davidson 1995).

The stratigraphic units in the Perth Basin have been grouped into different aquifers, named according to the main geological formation contributing to the aquifer as described in Table 7 in Davidson (1995).

As the focus of this study is on Cretaceous to Recent geological formations overlying the South Perth Shale (Figure 3.1 C; Table 3.1). Further details on the underlying stratigraphy are in De Silva *et al*. (2013).

Table 3.1 Stratigraphy in the Kwinana study area of the central Perth Basin (modified after Davidson (1995) and Timms *et al*. 2012), and estimated hydraulic properties of geological units from De Silva *et al*. (2013) and Smith *et al*. (2003); thickness variations from isopachs from Davidson and Yu (2008) and bore logs provided by the Department of Water; lithology from Davidson and Yu (2008)

AGE	GROUNDWATER	STRATIGRAPHIC UNITS	THICKNESS RANGE IN THE STUDY AREA (m)	LITHOLOGY	AVERAGE SPECIFIC YIELD FOR UNCONFINED UNITS; STORATIVITY (S) WHERE INDICATED	HORIZONTAL HYDRAULIC CONDUCTIVITY (m/d)
Late Tertiary - Quaternary	Superficial Aquifer	Superficial formations	30-65	Sand, silt, clay, limestone		
		Safety Bay Sand	0-20	Sand and shell fragments	0.2	15 (avg)
		Becher Sand	0-20	Sand, silt, clay and shell fragments	0.2	8 (avg)
		Tamala Limestone	0-65	Calcareous eolianite (limestone)	0.2-0.3	100-1000
		Bassendean Sand	0-50	Sand and minor silt and clay	0.2	10-50 (avg 15)
		Gnangara Sand	0-30	Sand, gravel and minor silt and clay	0.2	20
	Local confining bed	Guildford Clay	0-25	Clay with minor sand and gravel	0.05	0.1-1 (basal sandy units-up to 10)
	Superficial Aquifer	Ascot Formation	0-20	Limestone, sand, shells and clay	0.2	8
	Rockingham Aquifer	Rockingham Sand	0-70	Sand, silt and minor clay	0.2	20
Cretaceous	Aquitard	Kardinya Shale Member of the Osborne Formation	0-100	Shale, siltstone, minor sandstone		$10^{-4} – 10^{-6}$
	Leederville Aquifer	Henley Sandstone Member of the Osborne Formation	0-40	Sandstone and minor siltstone	S: $10^{-3} – 10^{-4}$	2-3
		Leederville Formation	150-320	Sandstone, siltstone and shale		
		Pinjar Member	0-70	Sandstone, siltstone and shale	S: $10^{-3} – 10^{-4}$	1
		Wanneroo	100-150	Sandstone, siltstone and	S: $10^{-3} – 10^{-4}$	1-10

AGE	GROUNDWATER	STRATIGRAPHIC UNITS	THICKNESS RANGE IN THE STUDY AREA (m)	LITHOLOGY	AVERAGE SPECIFIC YIELD FOR UNCONFINED UNITS; STORATIVITY (S) WHERE INDICATED	HORIZONTAL HYDRAULIC CONDUCTIVITY (m/d)
		Member		shale		
		Mariginiup Member	50-100	Sandstone, siltstone and shale	S: 10^{-3} – 10^{-4}	0.1-1

The watertable is located within the Superficial Aquifer across the Cockburn Groundwater Area. A major groundwater feature is the Jandakot Mound, a topographically elevated area that reaches a maximum saturated thickness of about 40 m in the northeast part of the study area and is mostly underlain by Bassendean Sand (Figure 3.2; Davidson 1995). The Safety Bay Mound in the southwest part of the study area has an average saturated thickness of 20 m and is mostly Safety Bay Sand, Becher Sand and Tamala Limestone (Davidson 1995). The watertable elevations are locally higher on these mounds because the rate of horizontal groundwater flow through the aquifer is less than the rate of vertical infiltration (Davidson 1995). The watertable configuration in the Cockburn Groundwater Area is largely controlled by the presence of the Jandakot Mound. The direction of groundwater flow is generally from east to west.

Figure 3.2 also depicts changes in hydraulic gradients (i.e. horizontal separation of the watertable contours) across the catchment. Along much of the wide expanse of Tamala limestone that outcrops along the coastal margin, hydraulic gradients are very low due to the extremely high hydraulic conductivity of the limestone. Further inland, east of a linear chain of lakes, the Superficial Aquifer transitions from Tamala Limestone into moderately permeable Bassendean Sand. Within this transition zone, there are large vertical and horizontal hydraulic gradients that are relatively steep (Nield 2004; Davidson and Yu 2008). Nield (1999; 2004) also attributes the steepening of hydraulic gradients to clay, which separates the Bassendean Sand from the extremely transmissive Tamala Limestone.

The hydrogeology of the Cockburn Sound Catchment was documented previously and is reviewed here using cross-sections and maps from Davidson 1995, Smith *et al*. (2003) and Trefry *et al*. (2006). Cross-sections through the hydrostratigraphy provide insight into lateral thickness variations and subcropping units (Figure 3.2). This study is confined to the onshore portion of the cross-sections in Figure 3.3 and extends farther east than the sections shown in Figure 3.3 . The eastward extent of the study area and a geological cross-section of the stratigraphy of the superficial formations are shown in Figure 3.4 (Davidson 1995).

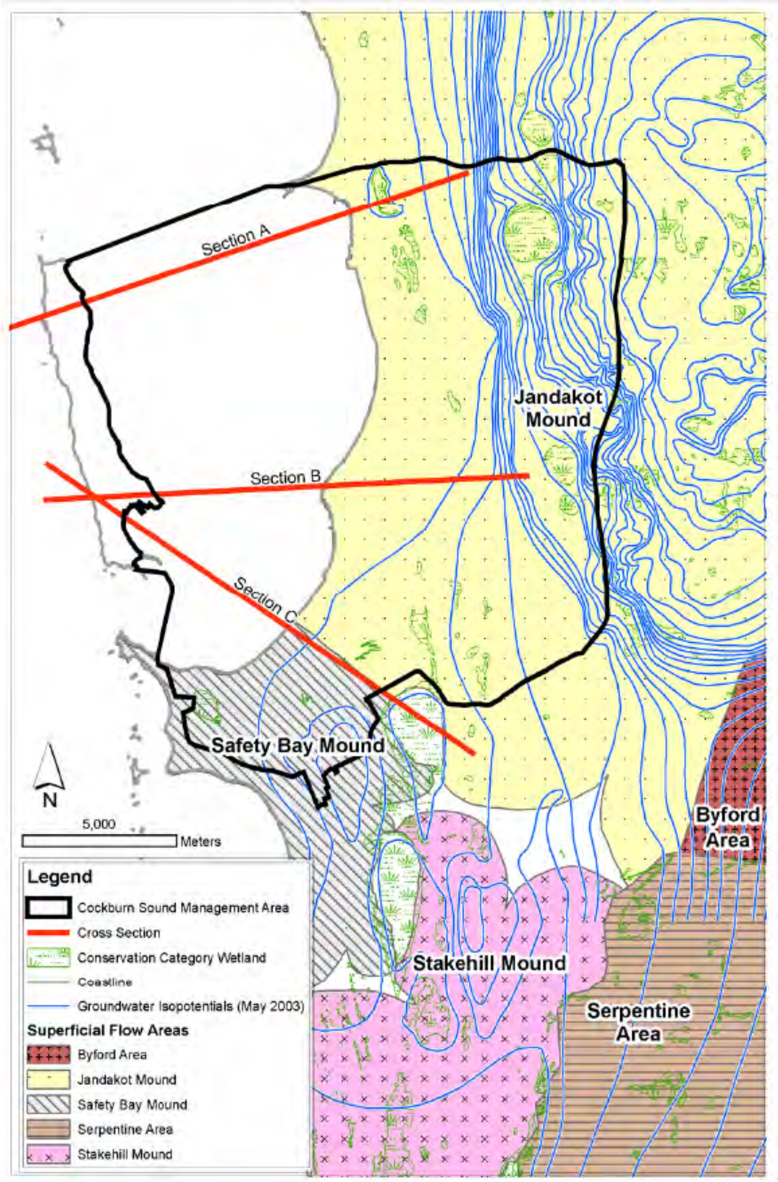

Figure 3.2 Features of the superficial groundwater system, watertable contours, and lines of section (A, B and C) depicted in Figure 3.3 based on data supplied by WA Department of Environment). The boundary of the management area of the Cockburn Sound catchment (CSC) is highlighted. Reprinted from "Status of the groundwater quality in the Cockburn Sound Catchment: Final report to Cockburn Sound Management Council," by Trefry *et al*. (2006), CSIRO Technical Report. Copyright 2006 by CSIRO. Reprinted with permission.

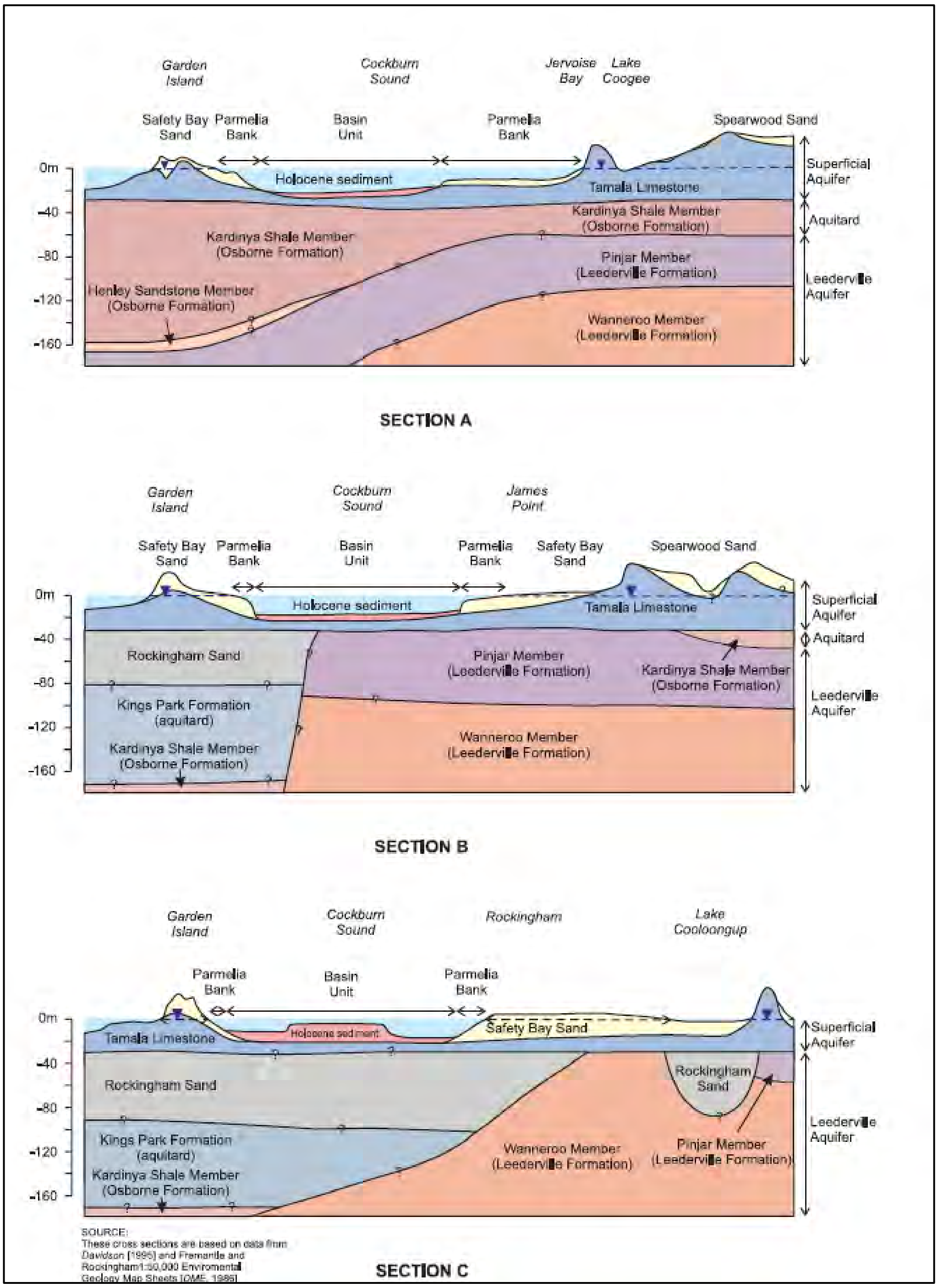

Figure 3.3 Hydrogeological cross-sections through the Cockburn Sound groundwater catchment. Reprinted from "Status of the groundwater quality in the Cockburn Sound Catchment: Final report to Cockburn Sound Management Council," by Trefry *et al*. (2006), CSIRO Technical Report. Copyright 2006 by CSIRO. Reprinted with permission.

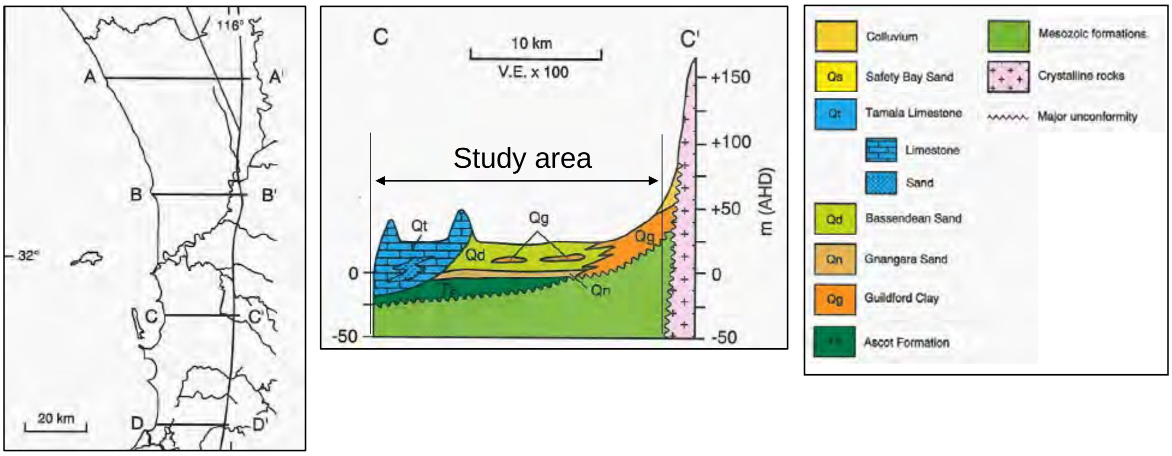

Figure 3.4 Geological section of the stratigraphy of the Cenozoic and Mesozoic. Note the extent of the study area labelled on the cross-section. Adapted from "Hydrogeology and groundwater resources of the Perth Region, Western Australia," by Davidson (1995), Western Australia Geological Survey, Bulletin 142. Copyright 1995 Government of Western Australia. Adapted with permission.

3.1.2 AQUIFER DESCRIPTIONS

In this chapter, the Cockburn Groundwater Area (CGA) refers to the groundwater management area with boundaries defined by the Department of Water as depicted in Figure 2 (Chapter 1). The CGA is largely encompassed by the Cockburn Sound catchment (CSC), but extends 5 km farther north. The CSC extends offshore to include Garden Island to the west and the Rockingham Groundwater Area to the south.

SUPERFICIAL AQUIFER

The unconfined Superficial Aquifer extends across the coastal plain of the Perth Basin and consist of Quaternary-Tertiary sediments, including the Safety Bay Sand, Becher Sand, Tamala Limestone, Bassendean Sand, Gnangara Sand and Ascot Formation (Davidson 1995). In the Cockburn Groundwater Area (CGA), the Superficial Aquifer has an average saturated thickness of approximately 30 m (DoW 2007).

<u>Safety Bay Sand</u>. The Safety Bay Sand is an eolian deposit along the coastal margin near Rockingham and consists of white, unlithified, calcareous fine- to medium-grained quartz sand and shell fragments with traces of fine-grained, black, heavy minerals (Davidson 1995). In the study area, the Safety Bay Sand reaches a maximum thickness of 20 m. The Safety Bay Sand unconformably overlies the Tamala Limestone and the Becher Sand.

The Safety Bay Sand is a moderately permeable unit, in contrast to the underlying Tamala Limestone, which reduces the overall transmissivity of the Superficial Aquifer and discharge to the ocean (Nield 1999; DoW 2007).

<u>Becher Sand</u>. The Becher Sand extends along the margin of the Swan Coastal Plain and consists of fine-to medium-grained quartz sand and lenses of silty calcareous clay rich in shell fragments (Davidson 1995). It was previously grouped with the Safety Bay Sand, but has been classified as a distinct unit due to its near-shore marine rather than eolian origin (Semeniuk and Searle 1985; Semeniuk *et al*. 1988). In the study area, the Becher Sand reaches a maximum thickness of 20 m.

There is a silty, calcareous clay containing shell fragments at the base of the Becher Sand (Semeniuk *et al*. 1988). A silty layer at the base of the unit impedes vertical exchange with the underlying Tamala Limestone. This silty layer is also known to locally affect the saltwater intrusion wedge, which rests on the base of the Tamala Limestone, extending up to 600 m inland (Nield 1999; DoW 2007). The lateral continuity of the basal silty layer has not been mapped and its lateral continuity remains uncertain (Nield 1999). The basal silty clay is in the order of 1-metre thick based on drill logs (Smith *et al*. 2003).

Bassendean Sand. The Bassendean Sand consist of pale grey to white, fine-to coarse-grained, but mainly medium-grained quartz sand. It also contains minor silt and clay. The thickness of the Bassendean Sand varies, depending on the topography (Davidson and Yu 2008). In the study area, the Bassendean Sand reaches a maximum thickness of 50 m. It inter-fingers to the east with the Guildford Clay and rests conformably on the Gnangara Sand. To the west, the Bassendean is unconformably overlain by the Tamala Limestone (Davidson 1995).

The Bassendean Sand is a moderately permeable unit with horizontal hydraulic conductivities between 10 and 50 m/d (DoW 2007; Davidson and Yu 2008).

Tamala Limestone. The Tamala Limestone is a highly transmissive calcareous eolianite with variable proportions of quartz sand, fine- to medium-grained shell fragments and minor clay lenses (Davidson and Yu 2008). It occurs along the coastal strip and comprises most of the western half of the Cockburn Sound catchment area. The thickness of the Tamala Limestone varies due to topography, but it reaches a maximum thickness of 65 m in the study area. It unconformably overlies Cretaceous sediments and Rockingham Sand in the study area.

Estimates of hydraulic conductivity of the Tamala Limestone range from 100 to 3000 m/d (Smith *et al*. 2011; De Silva *et al*. 2013). The high permeability of the Tamala Limestone is due to the presence of cavities and solution channels that are more common near the watertable (Davidson and Yu 2008). In some areas, the limestone has a relatively low permeability due to siliceous cement. According to an investigation of the geohydrology of the Tamala Limestone, there is predominantly dispersive flow within the aquifer, except where cavern development and large-scale conduct flow are present (Smith *et al*. 2011). Groundwater flow through the Tamala Limestone can be vertically restricted where the basal silty layer in the overlying Becher Sand is present.

Gnangara Sand. The Gnangara Sand consists of fine- to very coarse-grained quartz sand and abundant feldspar (Davidson 1995). In the study area, the Gnangara Sand reaches a maximum thickness of 30 m. It rests unconformably on the Ascot Formation and inter-fingers to the east with the Guildford Clay formation (Davidson 1995).

Ascot Formation. The Ascot Formation is a shallow marine deposit, consisting of calcarenite with thin beds of fine- to coarse-grained sand, bivalves and gastropods (Davidson 1995). There are thick beds of shell fragments, silty clay at the base of the formation (Davidson 1995). In the study area, the Ascot Formation is regionally extensive below the superficial formations and reaches a maximum thickness of 20 m. It lies unconformably on the Osborne or Leederville Formations except where it is absent due to erosion or non-deposition (Davidson 1995).

ROCKINGHAM SAND AQUIFER

Rockingham Sand. The Rockingham Sand consists of medium to coarse-grained quartz sand with some silt and subordinate clay. It has a maximum thickness of 70 m in the CGA (Dow 2007). The Rockingham Sand is located within an incised erosional channel in the Leederville Formation and is overlain by the superficial formations (Davidson 1995). The hydraulic conductivity of the Rockingham aquifer is similar to sands in the superficial formations (Smith *et al*. 2003; De Silva *et al*. 2013).

The Rockingham Sand aquifer is recharged via leakage from the overlying Superficial Aquifer and from the underlying Leederville aquifer (DoW 2007). Seawater is present at the base of the Rockingham aquifer (Davidson 1995) and groundwater discharge occurs adjacent to the coast and offshore (DoW 2007).

LEEDERVILLE AQUIFER

The Leederville aquifer is a regionally extensive, aquifer composed of Cretaceous Osborne Formation (Henley Sandstone Member) and Leederville Formation (Pinjar Member, Wanneroo Member and Mariginiup Member). It is mainly confined in the CGA by the overlying Kardinya Shale Member and overlies the South Perth Shale (DoW 2007). In the Rockingham area, Rockingham Sand is within an incised erosional channel and directly overlies the Leederville Formation (Figure 3.3). The Leederville aquifer consists of discontinuous, interbedded sandstones, siltstones and shale (Davidson 1995).

The Leederville aquifer is unconfined where it directly underlies the superficial formation and receives recharge by downward leakage, which mainly occurs where there are downward hydraulic gradients along the central and eastern portion of the coastal plain (DoW 2007; Davidson and Yu 2008). In areas where the Leederville aquifer is in direct contact with the overlying Superficial Aquifer and potentiometric heads increase with depth, there may be some upward discharge (DoW 2007). There is also groundwater discharge from the Leederville aquifer offshore (DoW 2007).

Henley Sandstone Member. The Henley Sandstone Member is predominantly a weakly consolidated sandstone, containing minor siltstone. It overlies the Pinjar Member of the Leederville Formation and overlain by the Kardinya Shale Member of the Osborne Formation (Davidson 1995). Within the study area, the Henley Sandstone member extends only over the northeast part where it reaches a maximum thickness of approximately 40 m.

Pinjar, Wanneroo and Mariginiup Members. These three units comprise the Leederville Formation and consist of discontinuous, interbedded sandstone, siltstone and shale. Their combined thickness across the CGA varies from approximately 150 to 300 m according to the isopach map in Davidson (1995; Plate 12).The Pinjar Member is the uppermost member of the Leederville Formation is overlain by the Kardinya Shale or superficial formations (Davidson and Yu 2008).

AQUITARDS

Guildford Clay. The Guildford Clay is a local confining unit among the superficial formations, consisting of silty and slightly sandy clay. The Guildford Clay inter-fingers to the west with the Bassendean Sand and the Gnangara sand and reaches a maximum thickness of 25 m along the east boundary of the study area based on bore log data from the DoW. It is commonly present along the Darling Scarp east of the study area.

Kardinya Shale Member. The Kardinya Shale Member consists mainly of consolidated, interbedded shale and siltstone. It also contains thin interbeds of mostly fine grained sandstone (Davidson 1995). The Kardinya Shale Member is overlain by the superficial formations and it overlies the Henley Sandstone and the Pinjar Member. The Kardinya Shale is absent in the southern part of the study area and increases in thickness to a maximum of 100 m in the northern part of the study area (Davidson 1995).

3.2 Seawater intrusion within the Superficial Aquifer

Seawater intrusion (SWI) is the movement of saline groundwater into coastal aquifers connected to the sea (Aitchison *et al*. 2003). Saltwater is denser than freshwater, hence where seawater intrudes into coastal aquifers, it typically underlies fresh groundwater. The encroachment of seawater into freshwater aquifers is exacerbated by pumping. The transition from fresh to salt water is commonly referred to as an ’interface‘, but is actually a diffuse zone of mixing rather than a distinct line. The salinity or concentration of total dissolved solids (TDS) in seawater is approximately 35,000 mg/L. A relatively inexpensive and convenient method to detect seawater intrusion is to monitor electrical conductivity (EC). The EC of seawater is about 48,000 µS/cm (Aitchison *et al*. 2003).

The groundwater quality in the Superficial Aquifer in the study area has been affected by seawater intrusion for many decades (Hazelgrove 1981). According to the DoW (2007), data collected from private monitoring and department data collated from several Cockburn Saltwater Interface (SWI) bores indicate the seawater interface extends 500 m inland at the base of the aquifer where there is high groundwater abstraction within the Kwinana industrial area. The DoW estimate of the SWI location is based on where salinity increases to a maximum of 35,000 mg/L TDS (seawater salinity).

There have been other observations in the Cockburn groundwater catchment showing that saltwater has intruded at least 2 kilometres inland from the coast referred to in Smith and Hick (2001) and Smith *et al*. (2003). This distance refers to saltwater and not seawater per se. As there were no accompanying salinity or EC data, it may refer to groundwater of lower salinity than seawater (e.g. salinity between 2000 and 5000 mg/L is considered moderately saline; DoW 2007).

Theoretical estimates of the SWI have been made based on the Ghyben-Herzberg approximation for calculating the thickness of the freshwater zone in a coastal water table aquifer. The Ghyben-Herzberg method predicts the position of the SWI based on the density difference between freshwater and seawater. It assumes a sharp interface exists between freshwater and saltwater, hydrostatic conditions within the aquifer and that the thickness of the freshwater zone is zero at the shore where the watertable has a zero elevation. This approximation typically over-estimates the inland extend of the SWI position because it assumes no flow through the aquifer (Aitchison *et al*. 2003). If a hydraulic gradient discharging groundwater to sea exists, then the SWI will be displaced toward the sea (Smith *et al*. 2005). Application of the Ghyben-Herzberg approximation to the Cockburn Sound catchment based on groundwater level data in 2004 in previous studies, showed the SWI located 2 km inland from the coast along most of the shoreline, excluding the Rockingham-Point Peron area where the SWI encroached much farther inland (Smith *et al*. 2005; Trefry *et al*. 2006).

3.3 Groundwater management

Over the past decade, there have been several key publications to address management of groundwater in the Cockburn Sound catchment (CSC) and the effects of submarine groundwater discharges to water quality in the Sound. The boundary of the CSC is highlighted in Figure 3.2. In 2005, the Government of Western Australia released the State's first *State Environmental (Cockburn Sound) Policy* (SECSP) for the protection of Cockburn Sound and its land catchment. It authorises the Cockburn Sound Management Council (CSMC), which was established in 2000, to report annually to Parliament on the 'State of the Cockburn Sound'. Whilst the policy focuses on monitoring water quality in the marine environment, if an environmental quality standard in the Sound is exceeded due to discharge from the catchment, the policy sets out a process to coordinate a management response through the CSMC. This includes establishing monitoring programs and implementing management actions to reduce contaminant inputs into the policy area (Government of Western Australia 2005).

With regard to the terrestrial environment, a review of the status of groundwater quality in the CSC by Trefry *et al*. (2006) identified gaps in the environmental management and offered recommendations, including the re-establishment of catchment-scale groundwater monitoring by the then Department of Environment and the establishment of a Proximate Vulnerability Zone along the Cockburn Sound shoreline whereby all premises within this Zone must monitor groundwater for a default suite of analytes (Trefry *et al*. 2006). Further details on groundwater pollution are provided in Section 3.4.

In 2008, the CSMC commissioned a survey of contaminants in marine water in Cockburn Sound (PB 2008), which revealed that the Sound was generally healthy, with many substances below laboratory reporting and detection limits, or at levels that were not found to pose an environmental threat (CSMC 2008). Nevertheless, it was recommended that sampling should be repeated in late winter or spring when groundwater levels were highest and likely to enter the Sound as this time would more likely detect contaminated groundwater plumes (CSMC 2008).

Several years prior to the release of the SECSP, CSIRO and Nield Consulting Pty Ltd conducted a study of nutrient discharges to the Sound, commissioned in part by the Kwinana Industries Council, Water and Rivers Commission of Western Australia, and the Natural Heritage Trust's Coast and Clean Seas Program. Their work estimated nutrient discharges to the Sound from diffuse sources in the catchment via groundwater discharge from the Superficial Aquifer. According to Smith and Nield (2003), uncertainty in groundwater recharge made it difficult to accurately estimate submarine groundwater discharges (SGD). Their work provided estimates of SGD from 24 km of shoreline, ranging from 2.5 to 4.8 ± 0.9 m^3/day/ m (2.2 to 4.2×10^7 kL/ year), corresponding to models of low and high recharge, respectively (Smith and Nield 2003). The modelled distribution of SGD was acknowledged as regional-scale estimates, having large uncertainty; they could not be confirmed with independent data, as none existed (Smith *et al*., 2003).

Nevertheless, this work demonstrated the significance of understanding the water balance in the CSC to aid coastal management of nutrient inputs to the Sound and prompted further studies on this topic.

An inter-comparison experiment involving the application of several types of SGD assessment approaches, including hydrogeological measurements, seepage meter readings, and tracer measurements was conducted over a 10-day period within the Northern Harbor area of Cockburn Sound by 20 international scientists (Burnett *et al*. 2006). The results indicate that SGD is occurring throughout the Sound, not just along the shoreline and there was fairly good agreement between different approaches (seepage meters: 2.5-3.7 m^3 /d/ m; radium isotopes: 3.2 m^3 /d/ m; radon: 2.0-2.7 m^3 /d/ m; Burnett *et al*. 2006). Seasonal variations in SGD to Cockburn Sound were investigated by Loveless *et al*. (2008), using radium isotopic signatures of marine and groundwater samples. Based on mass balance calculations of radium tracers in Cockburn Sound, SGD of freshwater ranged from 6.9 to 8.2 x 10^7 L/d, corresponding to early summer and end of winter, respectively (Loveless *et al*. 2008).

An audit of the environmental management practices of the Department of Environment and Conservation, the Office of the Environmental Protection Authority, and the Cockburn Sound Management Council in regard to the ecosystem health of Cockburn Sound was conducted by the West Australian government (OAG 2010). The review identified gaps in policy implementation and management oversight. A key finding was that total contaminant discharges into the Sound had not been monitored since 2001. Among the recommendations were periodic mapping of seagrass coverage as an indicator of ecosystem health, and monitoring of cumulative contaminant inputs to the Sound (OAG 2010). This prompted the EPA to develop an environmental quality management framework for the marine environment that includes spatially-defined environmental quality objectives to guide decision-making (EPA 2015).

In the most recent 'State of the Cockburn Sound' report to Parliament (2013), the CSMC reported results from a review of contaminant loads entering Cockburn Sound by GHD (2013). According to this work, the largest contributor of nitrogen (21-53 tonnes/year) and phosphorous (3-6 tonnes/year) in the CSC was from residential use and that the major pathways for contaminants entering the Sound were surface runoff via drains or groundwater from all land uses in the catchment (CSMC 2013). Moreover, there is now significantly less industrial inputs to the Sound due to improved treatment processes and because the disposal of industrial wastes has shifted from Cockburn Sound to the Sepia Depression Ocean Outfall Landline (SDOOL).

In 2007, the Department of Water developed a Water Management Plan (WMP) for the Cockburn Groundwater Area (CGA) that superseded the previous management plan developed in 1993 (DoW 2007). The WMP revised groundwater allocation limits due to declines in water levels for all aquifers in the CGA. The document recognised the importance of setting allocation limits to ensure that groundwater abstraction does not have unacceptable impacts on the groundwater quantity and quality, its dependent ecosystems and dependent social values (DoW 2007). The WMP provides allocation limits for three aquifers in the CGA: the Superficial, Leederville and Yarragadee aquifers. The DoW (2007) acknowledged that allocation limits for the Superficial Aquifer were close to the actual sustainable limit and that allocation limits may change with new information, whilst the Leederville and Yarragadee aquifers were over-allocated. At that time, the dominant land uses in the CGA were horticulture and heavy industry and it was noted in the WMP that the number of industrial subdivisions would likely increase in the future (DoW 2007).

The Kwinana Industries Council, an incorporated business association consisting of members from all the major industries and many of the smaller businesses adjacent to the Cockburn Sound, commissioned a study by Burns & Roe Worley to identify sustainable options for water supply, wastewater reuse and wastewater disposal for a 15 year planning horizon, 2006 to 2021 (KIA 2006; BRW 2006). The study by Burns & Roe Worley in 2006 did not describe temporal variations in water consumption due to seasonal patterns. As outlined in the next chapter, according to the KIA, industry water demand was estimated to be 32.7 GL/yr in 2006 and projected to increase to 48.9 GL/yr by 2021 for existing industries and 69.6 GL/yr by 2021 for new and existing industries (KIA 2006). Among the recommendations from BRW (2006) were preferred water sources and base source cost, which included groundwater (11.6 GL/yr), treated wastewater (56 GL/yr), aquifer recharge of South Jandakot stormwater (2.2 GL/yr), and industry synergies (KIA 2006).

3.4 Groundwater pollution

Groundwater contamination in the CSC derives from private and state-owned industries and corporations, population centres, residential areas, and agricultural activities according to a comprehensive study of the groundwater quality in the CSC conducted by Trefry *et al*. (2006). The study was based on data on diffuse sources of contamination and point source of nutrients, petroleum hydrocarbons, metals, pesticides and herbicides, phenols and solvents from 50 operators of industrial/commercial premises. Much of the data for this study was volunteered by industries near Cockburn Sound. Their review indicated that nearly all instances of groundwater contamination within the CSC were relatively long-standing and well-known, with the exception of one operator, and that most instances of contamination were directly associated within individual onsite activities (Trefry *et al*. 2006).

The acidic leachate from the oxidation of sulphide minerals in soils in marshy areas is a concern on the Swan Coastal Plain. A reactive-transport modelling study by Salmon *et al*. (2014) investigated the key controls on acidification on acidification in shallow groundwater with particular reference to the Bassendean Sand.

Acidification of groundwater is more likely to occur in carbonate-depleted sandy soils, such as the Bassendean Sand. A survey of the Bassendean Sand by Singh *et al*. (2012a) in the Perth metropolitan region revealed pervasively low pH values (less than 3 and in some cases as low as 1.8). The oxidation of low pH soils can lead to unacceptable levels of acidity that have the potential to impact surface water and groundwater resources (Singh *et al*. 2012b). The potential for acidification also depends on the sulphur content of the soils. The criterion for evaluating the potential for acidification assumes there is sufficient neutralising capacity within the soils to buffer minor acidity due to low sulphur contents (<0.03%; Singh *et al*. 2012b). Experimental studies conducted by Singh *et al*. (2012b) were used to test the applicability of this criterion for Bassendean sands containing less than 1% clay. The results from Singh *et al*. (2012b) indicate that poorly buffered sandy soils with low pH values coupled with sulphur content less than 0.03% have the potential to acidify when exposed to air (e.g. due to lowering of the water table).

An analysis of the data presented in Singh *et al*. (2012a) has revealed that of the 24 sites sampled in the CSC study area for this project, 17 were identified as having Bassendean Sand with low pH values (less than 3). Forty percent of these 17 sites had sulphur contents less than 0.03%. These sites are located along the eastern half of the study area (Figure 3.5). Along the narrow coastal strip to the west, the potential for acid sulfate soils to develop is less likely due to the higher carbonate content of the coastal sands and Tamala Limestone.

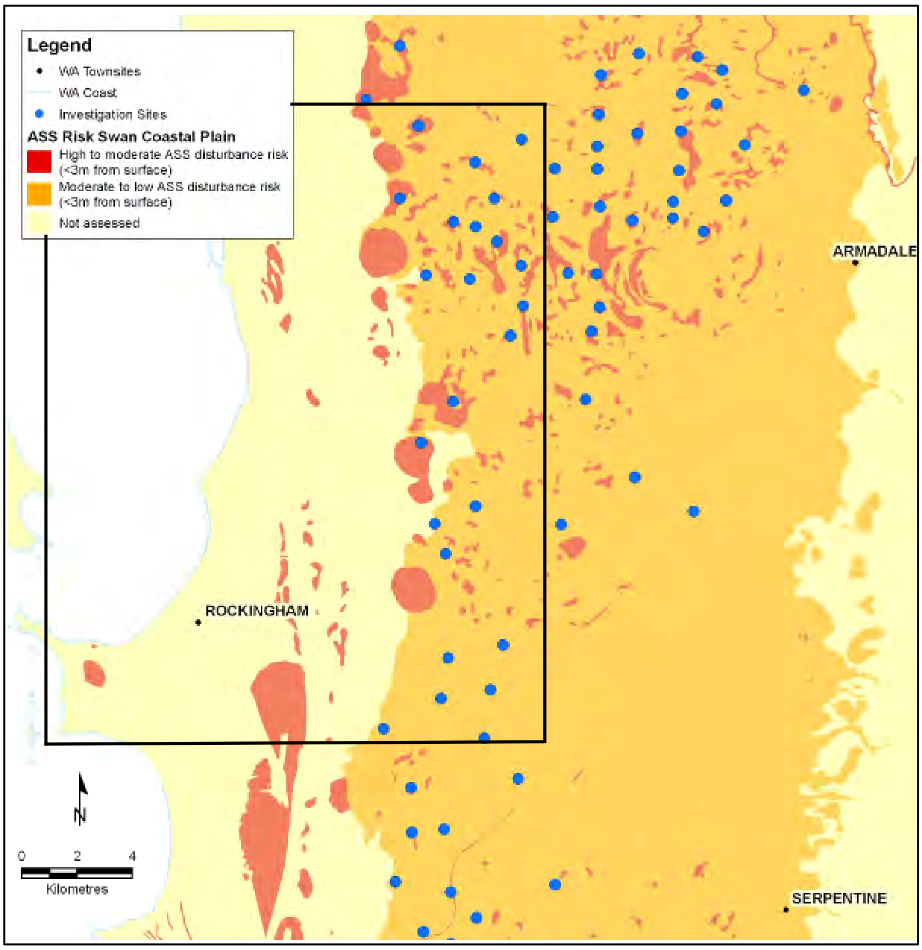

Figure 3.5 Acid sulfate soil risk mapping primarily in the Bassendean Sands located in the eastern half of the CSC study area. The CSC study area is the boxed area. Adapted from "Acid sulfate soil survey in Perth Metropolitan Region, Swan Coastal Plain WA," by Singh *et al*. (2012a), Department of Environment and Conservation, Government of Western Australia. Copyright 2012 by Government of Western Australia. Adapted with permission.

It has been suggested that implementing MAR schemes could be used to slow watertable decline and limit ongoing acidification on the Gnangara Mound (Appleyard and Cook 2009).

With regard to the effect of stormwater quality on groundwater quality, Trefry *et al*. (2006) acknowledged that there is potential for contamination from this pathway, but there is limited information about stormwater drainage inputs. There have been no detailed studies on this topic in the CSC area. Groundwater quality may be impacted by stormwater as it enters the subsurface through leaking compensation basins and wetlands.

Figure 3.6 indicates the location of 13 priority plumes from point sources that were identified in Trefry *et al*. (2006). These are only priority plumes measured within the Proximate Vulnerability Zone defined in Trefry *et al*. (2006). The dominant contaminant in these plumes was nitrogen species. The sources of nutrients include industry, agriculture, landfills, septic tanks and urban development. The presence of a regionally-high background concentration of nitrogen was noted by several operators. At some sites in the CSC it was difficult to discern the component of nitrogen contributed by onsite activities relative to background groundwater concentrations (Trefry *et al*. 2006).

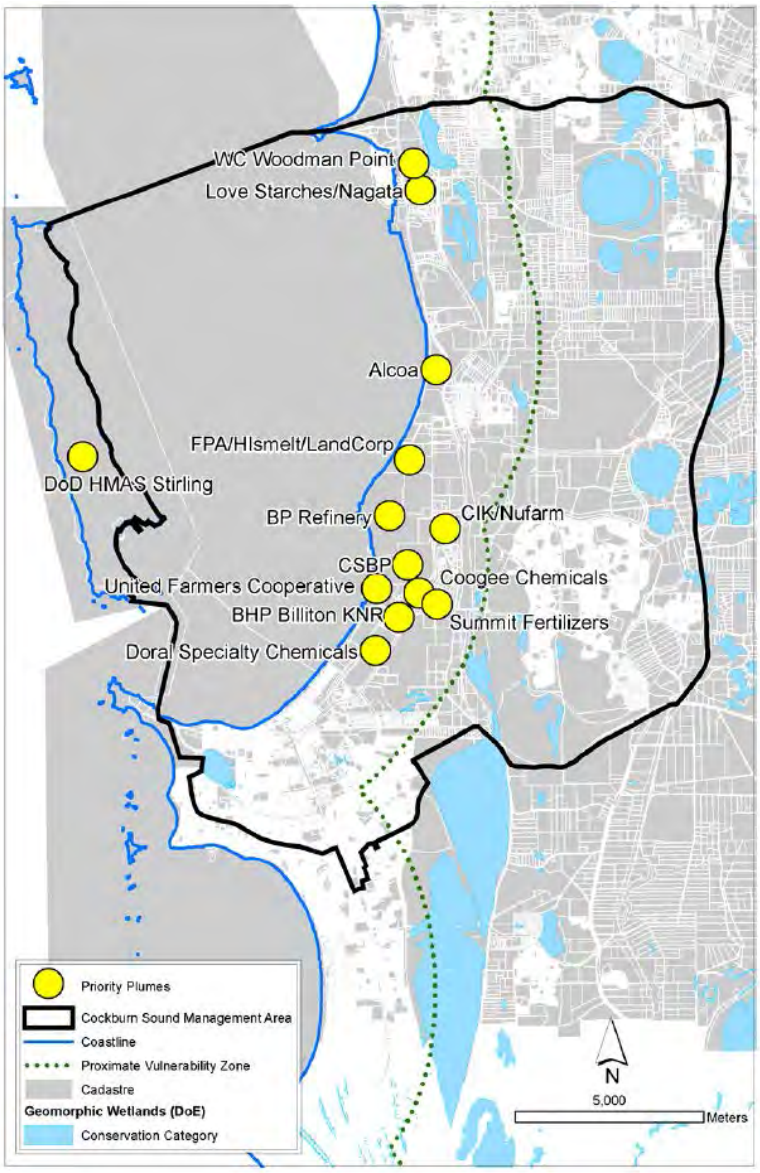

Figure 3.6 Distribution of the highest priority plumes from point sources based on data provided by companies and agencies. Note the location of the Proximate Vulnerability Zone wherein the priority plumes were identified from measurements. Reprinted from "Status of the groundwater quality in the Cockburn Sound Catchment: Final report to Cockburn Sound Management Council," by Trefry *et al*. (2006), CSIRO Technical Report. Copyright 2006 by CSIRO. Reprinted with permission.

A review of spatial variations in groundwater quality by Sarukkalige (2012) also highlighted significantly high ammonia levels in groundwater in the CSC due to the disposal of agricultural chemicals (i.e. pesticides, fertiliser). The study was based on groundwater quality samples from 2005 to 2011 collected by the Department of Water. Spatial analysis of trends in water quality and land use by Sarukkalige (2011) revealed that industrial areas of Kwinana have critical levels of nutrients, inorganic metals and heavy metals.

The abovementioned studies by Trefry *et al*. (2006) and Sarukkalige (2011, 2012) provide insight into the groundwater quality in the CSC over the time periods sampled for these investigations. However, another approach is to consider the potential for leaching to groundwater from reported emissions of pollutant substances to the land surface.

The National Pollution Inventory (NPI) is a database managed by the Department of Environment, which contains information about substance emissions from industries to the environment on an annual basis (*http://www.npi.gov.au*). The substances included in the NPI and the substance reporting thresholds were selected by consideration of the potential impacts of substances on human health and the environment. If an industry facility exceeds NPI reporting thresholds for a particular substance, it is required to lodge estimates of emissions for that substance to land, air and water. This includes releases to the environment from intentional activities and/or unintentional leaks and spills. It excludes on-site emissions that are contained in purpose-built, approved receiving facilities. The NPI database provides the source and location of substance emissions to land; however, there are a number of factors which control the harm to the environment and the exposure risk to humans.

Information about substance emissions to land reported to the NPI can be used to assess the potential for leaching of contaminants from soil into groundwater. Although this was not undertaken, a summary is given here of NPI data extracted for substance emissions to land reported by facilities within the CSC recently and a decade earlier. Industrial processing methods and management practices have changed within the catchment, thus it is useful to examine changes in the NPI data over time.

According to NPI data extracted for the CSC for the most recent reporting year (2013/2014), the estimate for emission of ammonia (total) to land was 74,000 kg, predominantly from the BHP Billiton Nickel Refinery in Kwinana and the Kwinana WWTP (KWWTP). The NPI annual report for 2005/2006 was selected for comparison with the most recent data. The 2005/2006 estimate for emission of ammonia (total) to land was 42,000 kg, mainly from the KWWTP[1]. There were only a few facilities that were required to report substance emissions to land in years 2005/2006 and 2013/2014. These were ALCOA's Kwinana Alumina Refinery, BP's refinery, BHP Billiton Nickel West Refinery, KWWTP, Tronox KMK Cogeneration Plant, The City of Rockingham Millar Road Landfill, and City of Cockburn Henderson Waste Recovery Park (previously Henderson Road Landfill). Emissions to land for NPI reported substances are given in Table 3.2. The majority (64%) of the reported substance emissions listed in Table 3.2 increased between the two reporting times and this was mainly for ammonia, and a range of metals and organic compounds reported for the two landfills (Henderson Road and Millar Road). The BP Refinery reported the greatest number of substances (18) with emissions that were lower in 2013/2014 compared with 2005/2006. These were various organic compounds and metals.

Table 3.2 Substance emissions to land in kg from the NPI database for facilities that exceeded NPI reporting thresholds for the substances indicated. The data shown are for two time periods: 2005/2006 and (2013/2014) in parentheses. NA indicates either the threshold was not exceeded or there was no emission to land for the substance.

SUBSTANCE	KWWTP	BHP BILLITON NICKEL WEST	HENDERSON RD LANDFILL	MILLAR RD LANDFILL	TRONOX KMK COGEN PLANT	ALCOA (KWINANA ALUMINA REFINERY)	BP REFINERY (KWINANA)
1,2-Dichloroethane	NA	NA	6.83E-4 (0.04)	NA (0.01)	NA	NA	NA
1,3-Butadiene (vinyl ethylene)	NA	NA	NA	NA	NA	NA	0.43 (NA)
Ammonia (total)	42000 (5100)	NA (67436)	14 (858)	NA (212)	NA	NA	85 (6.7)
Antimony & compounds	NA	NA	4.51E-03 (0.27)	NA (0.07)	NA	NA	4 (0.35)
Arsenic & compounds	NA	NA	9.57E-04 (0.06)	NA (0.01)	NA (5.40E-04)	140 (574)	11 (0.22)
Benzene	NA	NA	2.53E-03 (0.15)	NA (0.04)	NA	NA	25 (0.01)
Beryllium & compounds	NA	NA	3.28E-04 (0.02)	NA (4.86E-	NA	NA	NA (0.01)

[1] The KWWTP was upgraded to an oxidation ditch process in 2009 which greatly reduced N release to groundwater as is reported later

SUBSTANCE	KWWTP	BHP BILLITON NICKEL WEST	HENDERSON RD LANDFILL	MILLAR RD LANDFILL	TRONOX KMK COGEN PLANT	ALCOA (KWINANA ALUMINA REFINERY)	BP REFINERY (KWINANA)
				03)			
Cadmium & compounds	NA	NA	9.57E-04 (0.06)	NA (0.01)	NA (2.30E-05)	1.2 (0.89)	0.08 (0.95)
Chlorine & compounds	NA	NA	40.0 (2410)	NA (597)	NA	NA	NA
Chloroform (trichloromethane)	NA	NA	1.98E-03 (0.12)	NA (0.03)	NA	NA	NA
Chlorophenols (di, tri, tetra)	NA	NA	3.48E-05 (2.08E-03)	NA (5.16E-04)	NA	NA	NA
Chromium (III) compounds	NA	NA	2.87E-03 (0.25)	0.19 (0.06)	NA (1.10E-03)	19 (NA)	0.35 (0.23)
Chromium (VI) compounds	NA	NA	1.23E-03 (NA)	NA	NA	NA	0.06 (0.06)
Cobalt & compounds	NA	NA (2.76)	NA	NA	NA	NA	8 (0.37)
Copper & compounds	NA	NA (0.20)	3.69E-03 (0.22)	0.25 (0.055)	NA (6.40E-03)	8.6 (NA)	35 (11)
Cumene (1-methylethylbenzene)	NA	NA	NA	NA	NA	NA	15 (0.01)
Cyanide (inorganic) compounds	NA	NA	NA	NA	NA	NA	0.15 (0.35)
Cyclohexane	NA	NA	NA	NA	NA	NA	209 (0.01)
Dichloromethane	NA	NA	0.03 (1.80)	NA (0.45)	NA	NA	NA
Ethylbenzene	NA	NA	3.96E-03 (0.24)	NA (0.06)	NA	NA	58 (0.17)
Fluoride compounds	NA	NA	NA	NA (0.39)	NA (0.1)	3300 (NA)	1911 (NA)
Lead & compounds	NA	NA	4.30E-03 (0.26)	0.3 (0.064)	NA (5.90E-04)	0.94 (NA)	7 (5)
Manganese & compounds	NA	NA	NA	NA	NA	NA	274 (237)
Mercury & compounds	NA	NA	4.1E-05 (2.45E-03)	NA (6.10E-04)	NA (2.30E-05)	0.23 (NA)	6 (1.8)
n-Hexane	NA	NA	NA	NA	NA	NA	552 (0.01)
Nickel & compounds	NA	NA (390)	0.01 (0.69)	NA (0.17)	NA (3.40E-04)	NA (8.96)	12 (0.15)
Phenol	NA	NA	0.025 (1.55)	NA (0.38)	NA	NA	NA
Polychlorinated dioxins and furans (TEQ)	NA	NA	2.19E-08 (NA)	NA	NA	NA	NA
Polycyclic aromatic hydrocarbons (B[a]Peq)	NA	NA	1.71E-05 (NA)	NA	NA	NA	127 (0.11)

SUBSTANCE	KWWTP	BHP BILLITON NICKEL WEST	HENDERSON RD LANDFILL	MILLAR RD LANDFILL	TRONOX KMK COGEN PLANT	ALCOA (KWINANA ALUMINA REFINERY)	BP REFINERY (KWINANA)
Selenium & compounds	NA	NA	NA	NA	NA	120 (NA)	1.42 (3.25)
Styrene (ethenylbenzene)	NA	NA	NA	NA	NA	NA	1.51 (0.01)
Toluene (methylbenzene)	NA	NA	0.028 (1.67)	NA (0.41)	NA	NA	142 (0.01)
Total Volatile Organic Compounds	NA	NA	NA	NA	NA	NA	441 (NA)
Vinyl Chloride Monomer	NA	NA	2.73E-03 (0.16)	NA (0.04)	NA	NA	NA
Xylenes (individual or mixed isomers)	NA	NA	NA	NA	NA	NA	326 (0.01)
Zinc and compounds	NA	NA	0.046 (2.78)	3.1 (0.69)	NA	29 (38.1)	159 (510)

3.5 Prior groundwater modelling

As discussed in Section 3.3, there is a high priority to carefully manage groundwater resources in the Cockburn Groundwater area to meet the needs of industry, community and environmental values. The Department of Water (DoW) implements an adaptive management strategy toward groundwater, whereby they monitor water levels and water quality trends and set and review allocation limits (DoW 2007). The allocation limits set by the DoW are determined by considering the sustainable groundwater yield, which may be subject to change. One of the main tools used by the DoW for sustainable water resource management is the Perth Regional Aquifer Modelling System (PRAMS), which is discussed below. This section also describes several previous modelling studies conducted in the area to inform industry and government regulators.

PRAMS

The model domain for PRAMS extends from the Gingin Groundwater Area to the north to the Murray River system to the south, an area covering 9100 km^2 (De Silva *et al*. 2013). PRAMS is a coupled recharge and groundwater flow model that has been updated over time: Davidson and Yu (2008) documents the initial release of a fully reviewed version of the groundwater model (version 3.0). Subsequently, several versions of PRAMS have been released, leading up to the most recent publication by De Silva *et al*. (version 3.5; 2013). Table 3.3 summarises the most recent PRAMS versions, beginning with Davidson and Yu (2008). Note, VFM is the vertical flux model of recharge. PRAMS has grid elements that are 500 m by 500 m for a model domain that covers 217 by 107 km. Lakes and wetlands are not modelled explicitly, but were identified a priori from satellite images, then used in the classification of land use types for the model grid cells (Davidson and Yu 2008). The VFM is used to simulate recharge and discharge in the wetlands, whereas the groundwater model accounts for inflow to and outflow. If the VFM predicts a depth of the water table within 5 cm of the surface topography for a grid cell, the model will re-assign the land use type for the cell to that of a lake/wetland to account for seasonal inundation (Davidson and Yu 2008).

Table 3.3 PRAMS development (after De Silva 2013 and CyMod Systems 2009 a, b).

SATURATED MODEL	UNSATURATED MODEL	STATUS	DESCRIPTION
PRAMS 3.0	VFM 2.1.3-2.1.5	Released October 2003	Coupled model using 500 m by 500 m grid
PRAMS 3.0	VFM 2.1.6	Released August 2004	Change to plant root truncation algorithm and Modflow 2000 version
PRAMS 3.2	VFM 2.1.6	Released August 2008	13 layer model with an additional layer in the Superficial Aquifer. Updated Superficial Aquifer calibration. Updated landuse and groundwater monitoring data to 2008. Updated allocation database to 2007
PRAMS 3.3	VFM 2.1.6	Not Released*	Updated artesian calibration of PRAMS 3.2
PRAMS 3.4.1	VFM 2.1.6.3	Not Released	Reinterpreted geology. Improved private groundwater allocation estimates. Updated climate zonation. Improved VFM. Recalibrated and validated.
PRAMS 3.5 a and b	VFM 2.1.6.3	Documented in De Silva *et al*. 2013; Released after external review in 2014.	Layer geometry updated. Fault geometry updated. Range for aquifer parameters and faults revised.

*Neil Milligan of CyMod Systems; PowerPoint presentation: prams35final5.pdf

PRAMS 3.2 was obtained for use in this study as it was the most recent version available for release by the DoW at the start of project. PRAMS 3.5 was not available for general release until it was formally reviewed (personal communication J.P. Pigois, Department of Water, 10/1/2013). A suitable calibration of the model for the Superficial Aquifer was achieved in PRAMS 3.2; improved calibrations of the Leederville and Yarragadee aquifers were the focus of updates to PRAMS 3.3 (De Silva 2013). Most of the focus of these calibrations was near the Gnangara Mound (De Silva 2013). The most recent versions of PRAMS incorporates re-interpretations of geological faults and the compartmentalisation of layers by faults. There are two versions of the most recently released model: PRAMS v 3.5a is a model that has a few faults and the conceptual hydrogeology is effectively consistent with PRAMS 3.0; PRAMS v 3.5b has the same layer geometry, but is compartmentalised using faults (De Silva 2013).

The re-interpretation of layer geometries in the models subsequent to PRAMS v 3.2 pertain mainly to the central Perth Basin and do not overlap with the area modelled for the Cockburn catchment in this study.

<u>Lower Serpentine model</u>

Marillier *et al*. (2012) developed a regional surface water-groundwater model for the Lower Serpentine catchment to run different development, drainage and climate scenarios. The model overlaps with the southern half of the Cockburn catchment (Figure 3.7). It was constructed using Mike SHE 2011 modelling framework, using geological, hydrogeological, hydrological, soil and land use data (Marillier *et al*. 2012). It used a spatial resolution of 200 m with daily time steps for the simulation period 1970 to 2010.

Three computational layers with spatially varying hydraulic properties in each layer were included in the model. The three layers corresponded to the Superficial Aquifer, Pinjar Member and Kardinya Shale, and the Wanneroo (Wanneroo Member and Rockingham Member of the Leederville Formation). The processes simulated in the model were rainfall, evapotranspiration, flow through the saturated and unsaturated zones, channel flow, overland flow and abstraction (Marillier *et al*. 2012). The regional scale groundwater model was particularly well calibrated for the Superficial Aquifer, but Marillier *et al*. (2012) indicate that the model should not be used for fine-scale wetland, river and lake modelling, nor for abstraction or sustainable yield analysis in the Leederville and Rockingham aquifers. The report includes a hindcasting method for estimating groundwater abstraction based on groundwater allocation data.

Figure 3.7 Modelled area for the Lower Serpentine hydrological study. Reprinted from "Lower Serpentine hydrological studies – model construction and calibration report," by Marillier *et al*. (2012), Water Science Technical Series, Department of Water, Report 46. Copyright 2012 by Government of Western Australia. Reprinted with permission.

South-west Western Australia sustainable yields project (SWSY)

One of the model domains in the SWSY project used PRAMS (v. 3.2) and covered the Cockburn area (CSIRO 2009; Ali *et al*. 2010). The project estimated water yields for groundwater systems for several projections of climate and development scenarios. The project covered 37,200 km^2 of groundwater areas and used the coarse grid cell sizes of PRAMS (500 m spatial resolution). These models estimate recharge rates based on soil type, climate, land cover and water table depth (Ali *et al*. 2010).

Groundwater models for the Cockburn groundwater area by Nield Consulting

The steady-state groundwater model developed by Nield Consulting (1999) is the most comprehensive, publically-documented model for the Cockburn groundwater catchment. It is a saturated groundwater flow model for the Superficial Aquifer developed using the Modflow software package. It is a single layer model with a simplified representation of recharge based on land use and vegetation type (Nield 1999). It contains spatially varying hydraulic conductivities with a spatial resolution of 100 m. As there was large uncertainty in the model calibration, two versions were developed corresponding to low and high recharge estimates for the catchment. According to Nield (1999), the high recharge model calibration represented the best estimate of groundwater flow conditions at that time. The model results include water balance fluxes and groundwater levels under different scenarios.

As mentioned previously, Smith and Nield (2003) estimated the range of submarine groundwater discharge into Cockburn Sound based on the low and high groundwater recharge models. This work was also the basis for estimates of nutrient discharges as described in Smith *et al*. (2003).

The groundwater model for the CGA was further developed by Nield Consulting (2004) to investigate the impacts of future wastewater infiltration at the Kwinana Wastewater Treatment Plant. The model was similar to Nield (1999) with several important changes: hydraulic conductivities were revised near the KWWTP, artificial recharge was assigned at KWWTP and at a flyash disposal site in the catchment, and one transient solution was conducted (Nield 2004). Different model scenarios were conducted to investigate different options for relocating the infiltration lagoon area and increasing the volumes of wastewater disposed at the site.

3.6 Conclusions

There are a number of water-bearing sedimentary layers and aquitards within the CSC study area, an area that has been extensively reviewed by others in previous studies of the geology and hydrogeology of the central onshore portion of the Perth Basin. Declines in groundwater levels in all aquifers in the CGA have raised concerns that the Superficial Aquifer is close to its sustainable limit. Groundwater in the Leederville and Yarragadee aquifers is over-allocated. Projections for future development suggest that the number of industrial subdivisions will likely increase in the future (DoW 2007) and there have been estimates of industry water demand of 48.9 GL/yr by 2021 for existing industries and 69.6 GL/yr by 2021 for new and existing industries (KIA 2006). Groundwater modelling has been used to inform industry and government regulators to address these concerns. The quality of groundwater in the Superficial Aquifer is affected by seawater intrusion in some locations. Previous estimates of the saltwater interface vary and theexact location of the SWI along the full extent of the Swan Coastal Plain has not been determined based on measured data.

With regard to groundwater quality in the Superficial Aquifer, the most complete synthesis of information on pollutants in the catchment to date is by Trefry *et al*. (2006). Further insight into the potential for leaching of pollutants to groundwater was obtained by examining data extracted from the NPI database for substance emissions to land in the CSC study area for the most recent year reported and a decade earlier. Within the catchment, a number of priority plume locations have been identified (Trefry *et al*. 2006) and data from the NPI suggest substance emissions to land in the catchment have increased over the last decade. Analysis of the NPI data reveal the emission of ammonia (total) to land was 76% higher in the most recent year reported compared with a decade earlier. Two landfills in the study area reported higher emissions to land for a range of metals, organic compounds and ammonia in the most recent year reported compared with a decade earlier. It should be noted however, that there are a number of factors which control the harm to the environment and the exposure risk to humans. Although an evaluation of the potential for leaching of these substances to groundwater has not been undertaken, these data and the concerns about reaching sustainable limits for groundwater underscore the need to carefully manage groundwater resources in the CSC.

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4 Water demand and supplies

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Key points

- Water demand by heavy industry is currently about 28 GL/yr, about 60% (17 GL/yr) of which comes from groundwater, 5GL/y from the Kwinana Water Recycling Plants and the remainder from potable supplies and from on-site stormwater and recycling
- This is projected to increase by about 15 GL/yr by 2031 but could be much higher if economic growth rates increase or new industries enter the area
- Groundwater from the Superficial Aquifer is the cheapest source of water but declining levels and salt water intrusion are raising the possibility that future allocations may be reduced.
- Over 50 GL/yr of treated wastewater is disposed of to the ocean in the area
- An investigation of options has shown that scheme water, seawater or stormwater were unlikely to be viable options for meeting new large industry demands in the Western Trade Coast area (located in the western third of the study area)
- Comparative unit costs estimates indicated that groundwater was the least cost option (if it were available) followed by managed aquifer recharge of treated wastewater into the Superficial Aquifer and then direct access by industry to wastewater from the Sepia Depression Ocean Outfall Line.

4.1 Current water demand and supplies

4.1.1 HEAVY INDUSTRY

Water demand for heavy industry in the Western Trade Coast area is estimated to be around 28 GL/yr[2]. Most of the heavy industrial water use is within the Kwinana Industrial Area (KIA). The KIA is one of Western Australia's most important strategic heavy industrial precincts. It includes nickel, petroleum, titanium and alumina refining, power generation; and cement, chemical and fertiliser manufacturing. Water is used for a range of purposes including industrial processing, cooling towers, wash down, dust suppression, slurry transport and potable use. Water quality requirements for these different processes vary (Table 4.1).

[2] This includes water used by heavy industries within t he KIA, as well as Cockburn Cement to the north of the KIA. Estimated water demand is based on groundwater abstraction reporting by industry to the Department of Water; scheme water and KWRP water use information from Water Corporation; and information in BRW (2006) for other water sources.

Table 4.1 Water uses, sources and total dissolved solids levels in the KIA (GHD 2015)

WATER USE	APPLICATION	SOURCE	TOTAL DISSOLVED SOLIDS
Demineralised	Processed water	On-site water treatment plant (typically ion exchange, reverse osmosis or electrodialysis type plant)	< 10 mg/L
Cooling tower	Major water consumption in KIA	KWRP* IWSS** Groundwater SDOOL***	< 50 mg/L < 500 mg/L > 500 mg/L 900 mg/L
Potable	Potable water uses e.g. staff kitchens	IWSS	< 500 mg/L
Wash down water	Relatively small volumes, occasional use	IWSS Groundwater	< 500 mg/L > 500 mg/L
Process	Variable quality, can be low grade (depending on application)	KWRP IWSS Groundwater	< 50 mg/L < 500 mg/L > 500 mg/L
Dust suppression / slurry transport	Low grade (depending on application)	Groundwater SDOOL	< 1500 mg/L 900 mg/L

* Kwinana Water Reclamation Plant (tertiary-treated wastewater)
** Integrated Water Supply Scheme (drinking water)
** *Sepia Depression Ocean Outfall Line (secondary-treated wastewater)

Historically, industry has relied heavily on groundwater to meet its water needs. However, as groundwater has become more limited it has been supplemented by other water sources such as recycled wastewater.

Groundwater currently supplies around 60 per cent of heavy industrial water demand. Other water sources include recycled water from the Kwinana water reclamation plant (KWRP), reticulated or scheme water from the Integrated Water Supply Scheme (IWSS), on-site capture of stormwater and recycling of industry wastewater.

The KWRP was built in 2004 and provides up to 6 GL/yr of tertiary treated wastewater for industrial use. Treatment includes microfiltration, reverse osmosis and ultraviolet disinfection. Just over 5 GL/yr was used by industry in the 2013/14 year (Water Corporation 2014).

4.1.2 OTHER SECTORS

Water is also used for light industry, agriculture and urban purposes.

Most of the existing light industry is located within the Australian Marine Complex (AMC), which contains fabrication, technology and support industries that service the marine, defence and resource sectors. The AMC is serviced by scheme water from the IWSS and uses around 0.05 GL/yr. The AMC also includes some heavy industry in the form of ship building at Henderson.

There are existing urban areas within the northern and south-eastern portions of the Cockburn Groundwater Area. Water for household and commercial purposes is primarily supplied from the IWSS, with the exception of small rural land holdings to the north of ALCOA’s tailing ponds, which use groundwater.

Groundwater is the primary source of water for irrigating public open space and private recreation grounds, such as golf courses. Around 3.3 GL/yr of groundwater is licensed for use in the Cockburn Groundwater Area for public open space and private recreation, with approximately 2.8 GL/yr abstracted in 2014.

There is existing irrigated agriculture, including horticulture, turf farms and nurseries, on small rural land holdings to the north of ALCOA's tailings ponds. Groundwater is the primary water source for irrigated agriculture. Almost 4.6 GL/yr of groundwater is licensed for agricultural purposes in the Cockburn Groundwater Area.

4.2 Projected water demand

4.2.1 HEAVY INDUSTRY

Additional water supplies will be required to support the expansion of existing heavy industries or the establishment of new heavy industries. Discussions between the Department of Water and industry have indicated that a number of existing industries have plans to expand their operations over the next 5 to 10 years. The timing and magnitude of any expansions will be dependent on market conditions. Vacant land is still available within the KIA, and a large portion of the undeveloped Rockingham Industrial Zone (RIZ) south of the KIA has been designated for heavy industrial development.

Other drivers for new water supplies may include a potential decline in existing water supplies due to a drying climate or factors such as risks to throughflow wetlands, saltwater intrusion and salt concentration due to a lack of aquifer flushing or contamination.

Water demand for heavy industry is projected to increase by around 15 GL/yr by 2031 (Figure 4.1). This is based on an assumed growth rate of around 2 %/yr for the KIA[3] and partial development of the RIZ[4].The upper bound in Figure 4.1 represents a much higher growth in demand to 2021 (5.7 %/yr for the KIA) that could occur if a number of new large industries were to locate in the area.

It is important to note that whilst growth is shown as a linear trend, demand is actually likely to increase in step changes as a result of industry expansions or new industries locating into the area. The purpose of these projections is to provide an indication of the potential magnitude of demand increase over the medium-term. Water demand projections are currently being updated with the latest information by the Department of Water.

[3] Based on projected economic growth from the Monash-TERM economic model, high growth scenario (Resource Economics Unit 2008)
[4] Assumed to require 10 to 20 GL/yr at full development (EDAW 2009)

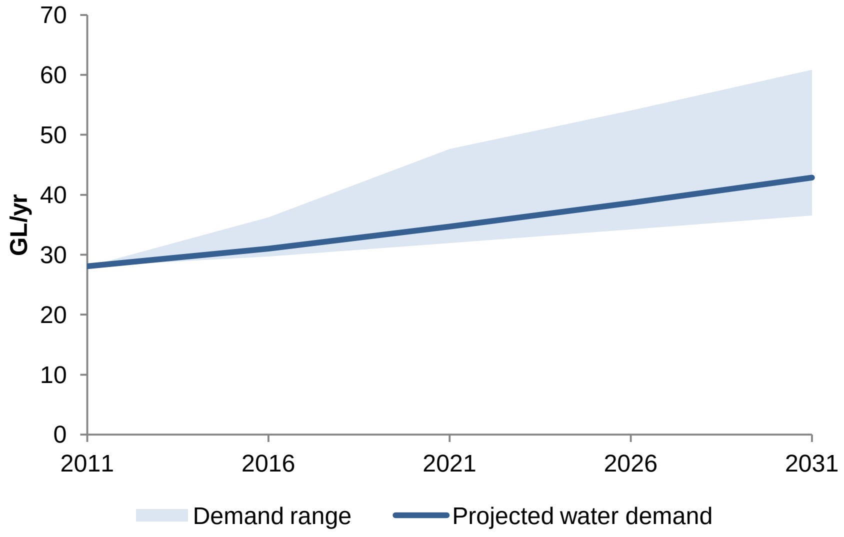

Figure 4.1 Projected water demand for heavy industry in the Western Trade Coast to 2031 (modified from DoW 2013)

4.2.2 OTHER SECTORS

The Australian Marine Complex (AMC) is close to being fully developed and so growth in water demand is not anticipated to be significant in this area. Any future growth can be met by the existing scheme water supply.

The Latitude 32 development to the east of the AMC consists of around 1400 ha of land planned to be redeveloped for light industrial purposes. This is a long-term project with staged development of the site over the next 20 to 30 years. Lots within the first stage of the development are currently on sale.

Water demand for Latitude 32 is estimated to be approximately 3 GL/yr when fully developed. However, actual demand will depend on the type of industries that are established, and water availability affects these to an extent. The timeframes for development will depend on market demand and other factors such as land tenure and land owner motivations (land ownership is very fragmented in some areas), the timing and delivery of transport infrastructure and the future staging and timing of existing quarrying activities (Landcorp 2010). The water demand projection in Figure 4.2 is an estimate based on current conceptual development timeframes. Scheme supply is proposed for the Latitude 32 development and is included within the Water Corporation’s planning. However, non-potable supplies may also be developed for select industries/precincts in response to industrial demands. This could include the use of groundwater or recycled wastewater. Around 2 GL/yr of groundwater is currently used for irrigated agriculture within this area. This water may become available as industry replaces agriculture over time provided allocation limits remain unchanged. In the longer-term, if the high-water-using Cockburn Cement site is redeveloped for smaller industries, groundwater availability is likely to increase.

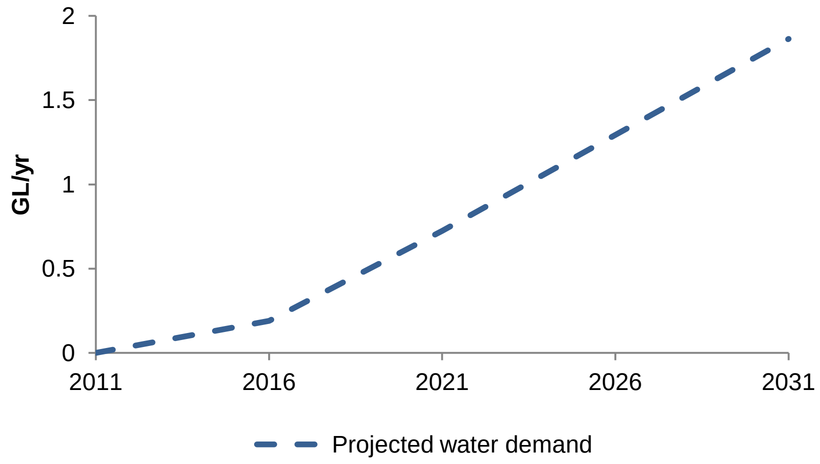

Figure 4.2 Projected water demand for new light industry for Latitude 32 to 2031 (DoW 2013)

The *Urban land development outlook* (WAPC 2014) identifies a number of undeveloped urban areas in the south-eastern and north-western parts of the Cockburn Groundwater Area. Future residential and commercial water needs will be met by scheme water from the IWSS. However, the developments will require non-potable water supplies for irrigating public open space. Groundwater licenses are already held by developers for a number of sites that are planned to be developed within the short-term. Additional water demand for all other sites is estimated to be approximately 0.15 GL/yr and 0.07 GL/yr in the south-eastern and north-western areas respectively[5].

Water demand for agriculture is expected to decrease within the Cockburn Groundwater Area, as existing rural land is redeveloped for industrial and urban purposes.

4.3 Future water supply

The *Kwinana industrial area water planning study 2006* (Burns and Roe Worley 2006) provided an assessment of potential water supply options, wastewater reuse and wastewater disposal options. Based on supply-demand and cost considerations, the study considered that the most promising options were groundwater, treated wastewater from Water Corporation's wastewater treatment plants, aquifer recharge of South Jandakot stormwater and 'industry synergies' as available options become economic and sustainable. Water supply options have been reviewed by the Department of Water (Department of Water 2013) and cost estimates have been updated (GHD 2015). Further information on the engineering costs is provided in Chapter 9.

4.3.1 GROUNDWATER

Cockburn Groundwater Area

Abstraction of local groundwater is generally the cheapest water supply option to meet future demand.

The Department of Water manages groundwater abstraction from the Cockburn Groundwater Area to ensure sustainable and productive use of the groundwater, and protection of ecosystems dependent on groundwater. The *Cockburn Groundwater Area water management plan* (Department of Water 2007) established allocation limits and the management approach for groundwater abstraction in this area. An allocation limit is the volume of water that can be abstracted from a resource annually.

[5] Based on an irrigation rate of 7500 kL/ha/yr and information on proposed public open space areas from structure planning (where available) or an assumption of 10% of the development area being used for public open space.

There is limited groundwater available for future use. Local confined aquifers are fully allocated and only limited volumes of water are currently available within the Superficial Aquifer. There can also be localised limitations on groundwater abstraction due to saltwater intrusion in coastal areas, impacts of pumping on groundwater dependent ecosystems (such as wetlands and bushland) or other users, and the presence of contamination plumes.

A number of superficial groundwater bores located close to the coast in the KIA have shown an increase in salinity due to saltwater intrusion as discussed in Section 3.2. Saltwater forms a wedge at the base of the Tamala Limestone, and where the Safety Bay Sand is present. Monitoring by industry shows that the main front of the wedge in the bottom half of the aquifer is about 890 m inland and the approximate position of the toe of the wedge is approximately 1 100 m inland. The wedge advanced approximately 400 m inland between 1998 and 2009.

Threatened ecological communities, conservation category wetlands and resource enhancement wetlands have been identified within the RIZ. Environmental approval conditions prohibit shallow groundwater abstraction during construction and for future industrial usage in a large portion of the RIZ (DSEWPC 2010).

Approximately 5 GL/yr of water currently licensed for industrial use was not abstracted in recent years. To optimise the use of groundwater, there may be opportunities for water trading or recouping under the *Rights in Water and Irrigation Act 1914,* if the water is no longer required by the licence holder.

Climate change projections predict that rainfall will continue to decline in this area, which will result in less recharge into aquifers. The Department of Water is currently reassessing the allocation limits for the Cockburn Groundwater Area to incorporate the latest water information and climate change projections. This may result in a reduction in current groundwater availability.

Groundwater from the Cockburn Groundwater Area will not be sufficient to meet the projected water demands for heavy industry shown in Figure 4.1.

Surrounding groundwater areas

Groundwater within 10 km of industry (includes subareas of the Jandakot, Rockingham, Stakehill and Serpentine groundwater areas) could potentially provide a cost-effective supply option (GHD 2015). However, there is currently limited water availability within these areas. Whilst water trading is permitted, existing water use in the surrounding groundwater areas is primarily made up of numerous small abstraction licenses. There are very few larger licenses that could make this a viable option for an industry supply.

Just under 2.5 GL/yr of groundwater is available within the Warnbro subarea of the Rockingham Groundwater Area. However, this is an urbanised area and so access to sufficient land for abstraction bores could be a constraint. Monitoring also indicates declining water levels.

4.3.2 WASTEWATER

Over 50 GL/yr of treated wastewater is currently disposed to the ocean via the Sepia Depression Ocean Outfall Line (SDOOL). The SDOOL provides for the disposal of wastewater from Woodman Point WWTP, Kwinana WWTP, Point Peron WWTP, KWRP-treatment brine, industrial wastewater from approved industries in Kwinana and the proposed East Rockingham WWTP. The relatively small volumes of industrial waste discharged into the SDOOL are insufficient to significantly alter the quality of the wastewater and therefore do not restrict where treated wastewater may be taken from the SDOOL.

The SDOOL passes through industrial areas within the Western Trade Coast. This provides significant opportunities for the treatment and recycling of wastewater for fit-for-purpose industrial use. Further detail on the availability and quality of wastewater is provided in Chapter 5.

Managed aquifer recharge of treated wastewater into the Superficial Aquifer to allow increased groundwater abstraction is one potential option to increase water supply for industry. Other wastewater recycling options include centralised tertiary treatment of wastewater and distribution to customers (similar to KWRP) or direct access by industry to wastewater from the SDOOL with onsite treatment for fit-for-purpose use.

4.3.3 OTHER WATER SOURCES

Scheme water and desalination of seawater are other potential water supply options. However, compared to groundwater these are high cost options (scheme water costs $2.032/kL on average for non-residential businesses in the metropolitan area) and the quality is likely to be higher than what is required for many industrial processes. Industries in the Western Trade Coast have indicated a preference to use fit-for-purpose water supplies for industrial processing, where cost-effective.

Current Water Corporation planning for the IWSS does not include additional water demand to meet unusually large industrial demands.

Some existing industries in the KIA collect, treat and store stormwater runoff from their sites. Whilst localised stormwater capture and reuse may continue to supplement other water sources, the volumes are generally relatively small. Larger stormwater volumes could be harvested from the Southern Lakes drainage system. However, availability is likely to decline with a drying climate. BRW (2006) found that direct use by industry of stormwater would not be feasible due to the variability in supply and high capital cost of infrastructure. There are also water quality issues with some of stormwater sources due to legacy contaminants from previous land uses and so for these sources costly water treatment could be required (GHD 2015).

4.3.4 SHORT-LISTED WATER SUPPLY OPTIONS

Based on supply-demand considerations, cost and/or environmental considerations; scheme water, seawater or stormwater were found to be unlikely to be viable options for meeting new large industry demands in the WTC (GHD 2015). Wastewater recycling, or a combination of wastewater recycling and groundwater are two potentially viable options. Groundwater availability is not sufficient to meet projected demand and so this is not an option in its own right.

Comparative unit costs by GHD (2015) indicate that the least cost supply option is groundwater, followed by managed aquifer recharge of treated wastewater into the Superficial Aquifer and then direct access by industry to wastewater from the SDOOL. Of the options investigated the highest cost option is centralised tertiary treatment of wastewater (MF/RO treatment as per the existing KWRP) and distribution to customers (GHD 2015). The cost of installing pipelines in the Kwinana Industrial Area is high. MAR provides a means of accessing water at the point of use.

4.3.5 ESTIMATING TOTAL DEMANDS AND SUPPLY UNDER A FUTURE CLIMATE SCENARIOS

CSIRO (2009) estimated high, median and low future water demand of 14 demand groups (potable, residential irrigation, industrial, horticulture etc) and compared them with groundwater supply options under wet, median and dry future climate scenarios. The results for the Cockburn Groundwater Management Area showed that demand would exceed supply in 2015 under a high demand scenario, by 2022 under a median demand scenario and be able to meet 2030 requirements if there was a low demand scenario (Figure 4.3). There was little discrimination in this area between climate scenarios, possibly because of the high transmissivity of the Tamala Limestone aquifer (resulting in only small changes in groundwater levels) and the fact that the planning horizon was short (22 years).

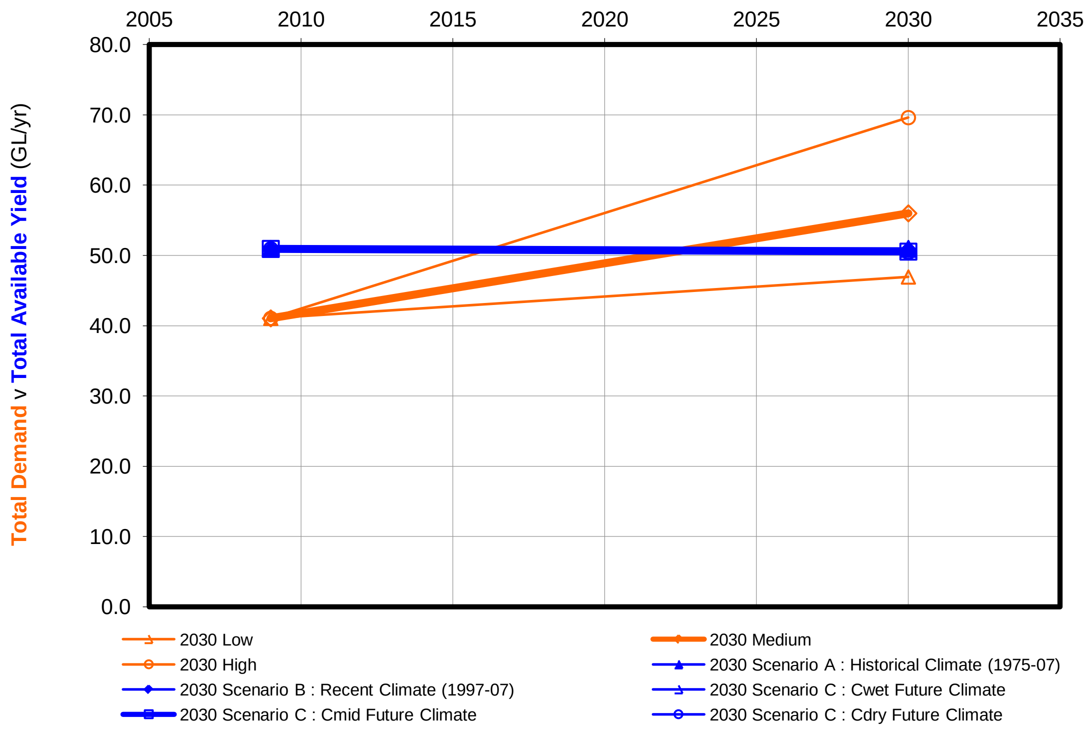

Figure 4.3 Supply and demand scenarios for the Cockburn Groundwater Management Area (CSIRO 2009)

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5 Wastewater availability and quality

Authors: Mike Donn and Don McFarlane

Key findings

- The Sepia Depression Ocean Outfall Line provides an abundant source of treated wastewater for Managed Aquifer Recharge. The smaller Kwinana WWTP is projected only to have modest increases in treated wastewater.
- Salinity and pH of treated wastewater is similar to the ambient groundwater and is within wetland ecosystem trigger values
- Treated wastewater nutrient concentrations generally exceed ambient groundwater and wetland ecosystem protection trigger values, however high ambient nitrogen concentrations exist as the result of anthropogenic and natural processes
- Treated wastewater is generally of higher quality from the Kwinana WWTP (and the soon to be commissioned East Rockingham WWTP) than Woodman Point WWTP and hence the Sepia Depression Ocean Outfall Line

5.1 Introduction

Water Corporation currently operates three wastewater treatment plants (WWTPs) in the Cockburn Catchment; Woodman Point, Kwinana and Point Peron (Figure 5.1). A fourth plant is currently being constructed in East Rockingham and is expected to commence operation in early 2016. Woodman Point and Kwinana WWTPs are of primary interest due to their location and current operation. All treatment plants use similar processes for the primary treatment of wastewater. Treated wastewater from the Point Peron plant is only primary treated (Water Corporation, 2014a) with the remaining plants utilising activated sludge technology for secondary treatment. Currently secondary treatment at the Woodman Point plant is based on a sequencing batch reactor (Water Corporation, 2009) with an upgrade to a continuous process, based on the Modified Ludzack-Ettinger process, proposed to occur in 2019 (source: Water Corporation). Treated wastewater quality data from the Beenyup WWTP was used to indicate the potential changes to future water quality as a result of future treatment process changes at the Woodman Point WWTP. The Beenyup WWTP services the northern suburbs of Perth (Figure 6.1). Both the Kwinana and new East Rockingham treatment plants utilise oxidation ditch technology for secondary treatment, thus water quality at East Rockingham WWTP is likely to be similar to Kwinana.

The majority of the treated wastewater (TWW) is disposed of via the Sepia Depression Ocean Outlet Landline (SDOOL) which transfers the TWW to an ocean outlet located in the Sepia Depression, approximately 4 km south-west of Point Peron (Figure 5.1). In 2013-2014, approximately 151 ML/d (55 GL/yr) of TWW was disposed of through the Sepia Depression ocean outlet (Water Corporation, 2014b). With respect to the TWW disposed through the SDOOL, a relatively small amount of TWW (~3%) is disposed of to land through infiltration basins. Based on the licence requirements, the Kwinana WWTP can dispose of up to 4.7 ML/d (~1.7 GL/yr) to land, with excess TWW disposed of through the SDOOL (DER, 2014).

Inputs to and off-takes from the SDOOL for 2013-14 are shown schematically in Figure 5.2; these have potential implications from quantity and quality of TWW available for managed aquifer recharge (MAR). While Figure 5.2 indicates the 2013-14 quantities there is the potential for greater recycling through the Kwinana Water Reclamation Plant (KWRP) and industry return and these are outlined below. Off-take of

TWW for KWRP of up to 24 ML/d in the current configuration (Department of Health, 2009) is used to produce high quality recycled water for industry by reverse osmosis (Water Corporation, 2014c). The disposal of industrial wastewater to the SDOOL of up to 30 ML/d (~11 GL/yr) is approved by Western Australian government (Ministerial statement 665, 2004). Industrial wastewater disposal is currently approved for KWRP concentrate (up to 7 ML/d, Department of Health, 2009) and wastewater from CSBP, BP and the Kwinana Cogeneration Plant (Water Corporation, 2014a).

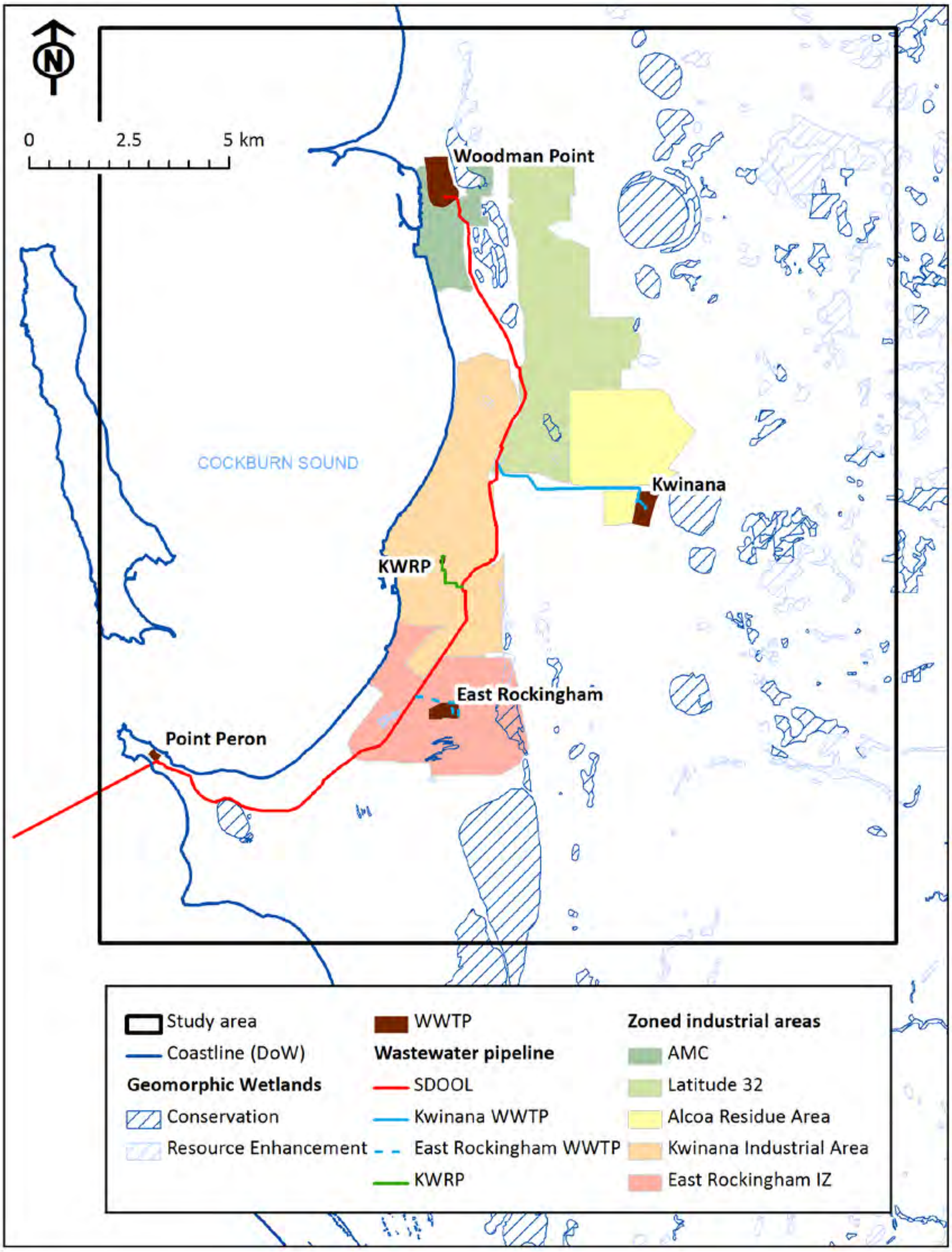

Figure 5.1 Location of wastewater treatment plants (WWTPs) and the Sepia Depression Ocean Outlet Landline (SDOOL) and associated treated wastewater pipeline connections

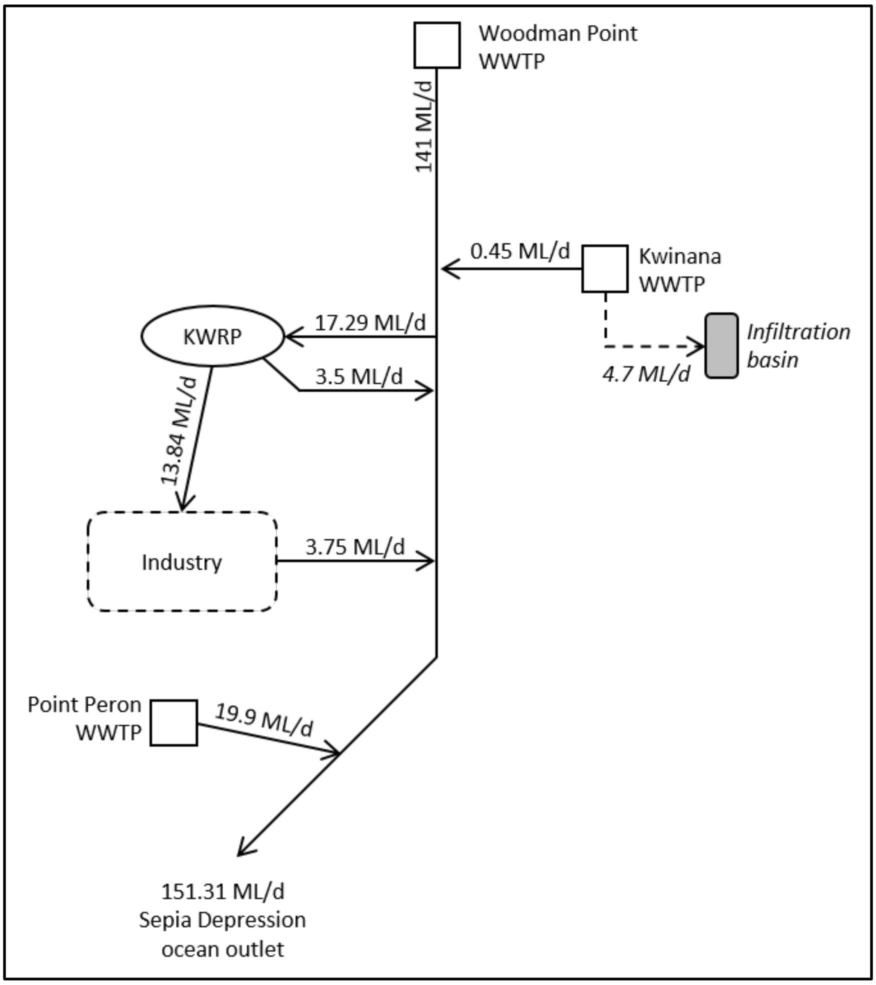

Figure 5.2 Schematic of the inputs to and off-takes from the Sepia Depression Ocean Outlet Landline (SDOOL) for the 2013/14 financial year. These include the three wastewater treatment plants (WWTPs), the Kwinana Water Reclamation Plant (KWRP) and selected industries. Data sourced from Water Corporation (2014b)

The disposal of TWW to the Sepia Depression has a nitrogen load limit of 1778 t/yr (equivalent to N load in 1994, Environmental Protection Authority, 2004), with appropriate licenced load limits set for industrial discharges to the SDOOL to ensure the total nitrogen loads do not exceed this value (e.g. 200 kg/d for CSBP (DER, 2013)). Relative to the inputs of total nitrogen from the Woodman Point WWTP (2072 kg/d in 2013) the input from industry is low. As a result the change in SDOOL water quality resulting from industry and KWRP wastewater additions are not considered in the analysis of MAR source water.

5.2 Wastewater availability

The main contributor to wastewater flow in the SDOOL is the Woodman Point WWTP (Figure 5.2 and Figure 5.3) representing >99% of current TWW inflow (excluding Point Peron WWTP which enters close to the ocean). Since the majority of the wastewater at the Kwinana WWTP is infiltrated (4.7 ML/d or 1.72 GL/yr) and the East Rockingham WWTP is not due for commissioning until 2016, it is unlikely that contributions are to alter substantially until 2025 when inflows to the East Rockingham WWTP are forecast to increase approximately 3.5 fold.

The maximum removal of wastewater from the SDOOL for recycling at KWRP is currently 17.29 ML/d (~6.3 GL/yr) with the potential to increase to 26.7 ML/d following an upgrade of the plant. This represents 12% and 19% of the TWW from the Woodman Point WWTP (141 ML/d in 2013-14). This is off-set by returns of industrial wastewater of up to 30 ML/d, though this was much smaller in 2013-14 (7.25 ML/d). Thus substantial volumes of wastewater remain available for potential managed aquifer recharge (MAR) schemes. At the typical MAR scheme sizes, 4.8 ML/d and 9.6 ML/d (1.75 GL/yr and 3.50 GL/yr) utilised for groundwater modelling and economic assessments, wastewater in the SDOOL represents a significant resource and is unlikely to be exhausted (Figure 5.2). The 52 GL/y of treated wastewater currently available compares with an estimated 39 GL/y of groundwater being extracted in the Cockburn Sound Catchment.

To put this in context with other wastewater recycling projects in Perth, Water Corporation are currently upgrading the Advanced Water Recycling Plant at Beenyup to a capacity of 14 GL/yr as part of the groundwater replenishment scheme with the potential to double this to 28 GL/yr. Should such a scheme be developed based at the Woodman Point WWTP, sufficient wastewater would remain in the SDOOL for MAR schemes designed around infiltration into the Superficial Aquifer to be viable.

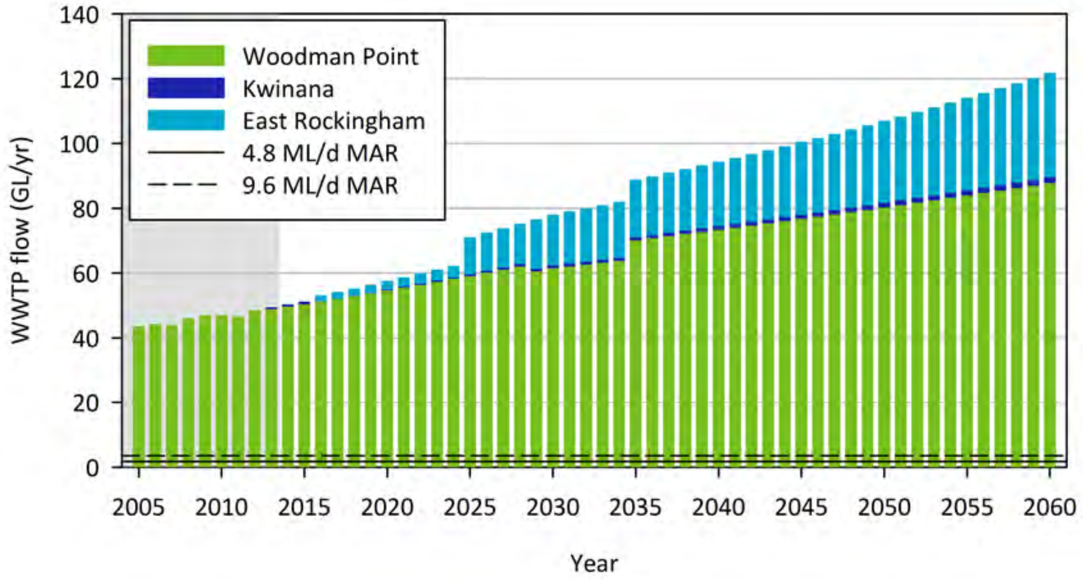

Figure 5.3 Annual total wastewater treatment plant (WWTP) measured (grey shaded area) and forecast flows to the Sepia Depression Ocean Outlet Landline (SDOOL) based on inflows. Note contributions from the Kwinana WWTP estimated assuming 4.7 ML/d (~1 m/d) is directed to infiltration (Inflow volumes supplied by Water Corporation). Typical volumes directed to a single managed aquifer recharge (MAR) site are also indicated by the lines at the bottom of the figure.

5.3 Treated wastewater quality

Treated wastewater quality is dependent on the treatment processes utilised within the plant. Given that two processes are utilised in the two plants of interest within the study area a comparison was undertaken using TWW quality data provided by Water Corporation for the Woodman Point WWTP (current operations), and Kwinana WWTP (which is also a surrogate for the East Rockingham plant). As the Woodman Point WWTP is due for upgrade in 2019 (Source: Water Corporation), TWW quality from the Beenyup WWTP was used as a surrogate for the future upgrades. The parameters included in the routine water quality monitoring of TWW by Water Corporation are shown in Table 5.1.

Due to changes in WWTP operations and treatment processes the TWW quality data between 2010 and 2013 was considered to be representative of the quality available for future MAR operations. The number of measurements of each parameter outlined in Table 5.1 vary between the WWTPs, therefore for comparison purposes box plots are used to show the variability in the available data. For comparison the median and inter-quartile range for ambient groundwater quality determined from data extracted from the Department of Water's Water Information Reporting database is provided in Table 5.2. It should be noted that the concentration of ambient groundwater nitrate + nitrite-N exceeds that of total nitrogen for the 75th percentile. This is due to the differences in the number of sites sampled for these parameters and the temporal variation in the parameters sets measured at each site.

Table 5.1 Water quality parameters measured on treated wastewater from the Woodman Point, Kwinana and Beenyup wastewater treatment plants (data supplied by Water Corporation)

PHYSICAL/CHEMICAL	NUTRIENT	HEAVY METALS
pH	Ammonia (NH_3-N)[1]	Arsenic (As)
Electrical conductivity (EC)	Nitrate (NO_3-N)[2]	Cadmium (Cd)
Total dissolved solids (TDS)	Nitrate (NO_2-N)	Chromium (Cr)
Total suspended solids (TSS)	Nitrate + nitrite (NO_x-N)	Cobalt (Co)
Total Alkalinity	Total Kjeldahl nitrogen (TKN)	Copper (Cu)
Biochemical oxygen demand (BOD)	Total nitrogen (TN)	Lead (Pb)
Chemical oxygen demand (COD)	Total phosphorus (TP)	Mercury (Hg)
		Nickel (Ni)
		Silver (Ag)
		Zinc (Zn)

1: No laboratory-based ammonia measurements at Kwinana WWTP, only in-situ measured ammonia was available. 2: In-situ nitrate also measured at the Kwinana WWTP

Salinity and pH for all TWW sources (Figure 5.3a and b) were generally within ambient groundwater values (Table 5.2). The salinity of the TWW was less than the 75th percentile of the ambient groundwater and generally less than the median concentration. The median salinity at the Kwinana WWTP was even below the 25th percentile of the ambient groundwater. This also applied for the trigger values for wetland aquatic ecosystems (south-west Australia, ANZECC-ARMCANZ, 2000) which represent a range within which there is a low risk that adverse biological effects will occur. The TWW pH was also generally within the interquartile range of the ambient groundwater measurements and the trigger values for wetland aquatic ecosystems. The data suggests that for these parameters the source water would not have an environmental impact on either groundwater in general or on wetlands. Certainly compared to the data extracted from the Department of Water WIR database, electrical conductivity of the TWW is lower than observed in Thomsons Lake (median 311 mS/m) and The Spectacles (north) (176 mS/m).

While median concentrations of total suspended solids were relatively low, there is a high degree of variability across all TWW sources (<1 to 630 mg/L; Figure 5.3c). The Kwinana TWW, however has a much narrower range (<1 to 27 mg/L) than the other two TWW sources. Due to the high total suspended solid concentrations the potential for clogging of infiltrating surfaces is high, especially for the Woodman Point and Beenyup TWW with 33% and 52% of samples >10 mg/L, the value suggested as posing high risk for clogging (NRMMC-EPHC-NHMRC, 2009), while only 10% of samples exceeded 10 mg/L for the Kwinana TWW. A more stringent TSS criteria of <5 mg/L would be required for gallery infiltration due to restricted access for remediating clogging. Thus for infiltration in galleries pre-treatment is likely to be required for all TWW sources as 78%, 94% and 59% of samples exceeded 5 mg/L for Woodman Point, Beenyup and Kwinana TWW, respectively.

As expected the concentration nitrogen species and phosphorus in the TWW exceeds the wetland aquatic ecosystem trigger values (the concentration that below which there is a low risk that adverse biological effects will occur; Figure 5.4) if added directly, however the nutrient concentrations are typical of treated wastewater in Australia (Table 4.10 in NRMMC-EPHC-NHMRC, 2006). Compared to the ambient groundwater, TWW concentrations are generally greater than the median values. However due to areas impacted by horticulture to the east of Lake Coogee (Department of Water, 2010) and nitrogen plumes within the industrial areas (Trefry et al., 2006), as well as natural nitrogen sources such as wetlands (see Section 6.4) TWW concentrations can be similar to the ambient groundwater.

Table 5.2 Ambient groundwater quality for study area (statistics based on average values at each sampling location)

PARAMETER	UNITS	NO OF SITES	MEDIAN	INTERQUARTILE RANGE
Electrical conductivity	mS/m	339	121	63 – 202
pH	-	529	7.2	6.4 – 7.7
Total N	mg/L	125	2.0	0.88 – 6.0
Ammonium-N	mg/L	239	0.18	0.06 – 0.61
Nitrate+nitrite-N	mg/L	281	1.1	0.05 – 18
Total P	mg/L	140	0.10	0.03 – 0.25
Soluble reactive P	mg/L	129	0.04	0.01 – 0.11

As with the physical parameters discussed above the Kwinana TWW had generally lower nitrogen concentrations than the other two plants, with the exception of ammonia-N which was lower in the Beenyup TWW (Figure 5.4a). Median total phosphorus (TP) concentrations were similar in Woodman Point and Kwinana TWW (4.4 and 4.6 mg/L, respectively) while TP concentration in the Beenyup TWW was approximately twice that of the other two (8.3 mg/L). The differences observed between the different TWW sources are likely to be related to (i) the treatment processes, and (ii) the differences in input to the plants arising from the differences in the sewerage catchment areas. Increasing amounts of drinking water are being sourced from seawater desalination plants as catchment runoff and Jandakot groundwater reduce in reliability and volumes. The TP in TWW is at least 10 times greater than observed in the ambient groundwater concentrations, however sorption capacity of the limestone/carbonate aquifers may be such that impacts from phosphorus may be limited to areas surrounding MAR sites.

Organic matter concentration in the TWW is monitored by measuring the Biological Oxygen Demand (BOD) and Chemical Oxygen Demand (COD). The BOD is representative of the biologically available organic matter and COD both the inorganic and organic matter subject to oxidation (APHA-AWWA-WEF, 2005). The BOD and COD values for the treatment plants are shown in Figure 5.5. The BOD represent a relatively small proportion of the COD (11 to 14%). In a comparison of BOD to bioavailable dissolved organic carbon, Krasner et al. (2009) showed that bioavailability decreased with decreasing BOD concentration. Therefore, the bioavailability of the organic carbon present in TWW from the Woodman Point, Beenyup and Kwinana WWTP is likely to be low, leading to the low potential for denitrification to occur without an additional source of organic carbon, either from natural sources (groundwater) or added to the TWW as an amendment.

A range of heavy metals is monitored in the TWW (Table 5.1 and Figure 5.6) with the majority of concentrations below the trigger values for wetland aquatic ecosystems (south-west Australia, ANZECC-ARMCANZ, 2000). Generally cobalt, mercury and silver were at or below the method detection limit. Potential elements of concern are copper and zinc where most samples exceed the trigger values for wetland aquatic ecosystems (south-west Australia, ANZECC-ARMCANZ, 2000).

While pollutants of emerging concern such as endocrine disrupting compounds (EDCs) have been measured in secondary treated wastewater in Western Australia (Leusch et al., 2014), no data was available for any of the TWW in the study area. Leusch et al. (2014) concluded that secondary treated wastewater from metropolitan WWTPs would require dilution prior to environmental discharge to remove the potential for any estrogenic effects. Further assessment is required to determine whether concentrations and biological responses observed by Leusch et al. {, 2014 #932} are observed in groundwater that has been impacted by infiltration of TWW, such as at the Kwinana WWTP. In laboratory column experiments, Patterson et al. {, 2011 #640} showed varying degrees of degradability of nine trace organic compounds (including EDCs) with factors such as sorption and redox conditions affecting the degradability of different compounds. The concentrations used in this study were generally higher than observed in wastewater, however it shows that site specific data is required to be able to assess these compounds.

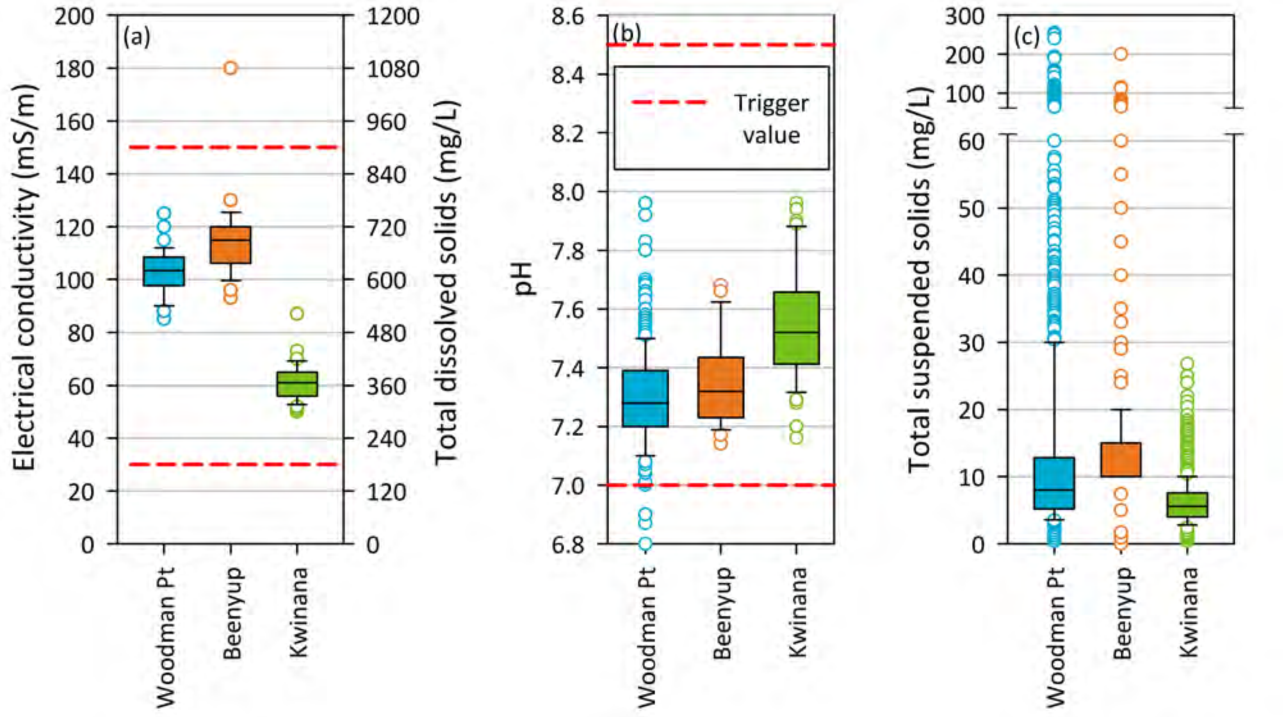

Figure 5.4 (a) Salinity (electrical conductivity and total dissolved solids), (b) pH and (c) total suspended solids in wastewater from the Woodman Point, Beenyup and Kwinana wastewater treatment plants (2010-2013). Total suspended solids calculated from electrical conductivity. The box represents the 25th and 75th percentiles with the 50th shown by the horizontal line within the box, the whiskers represent the 10th and 90th percentiles and the points the outliers. Trigger values represent the upper and lower limits for wetland aquatic ecosystems (south-west Australia, ANZECC-ARMCANZ, 2000)

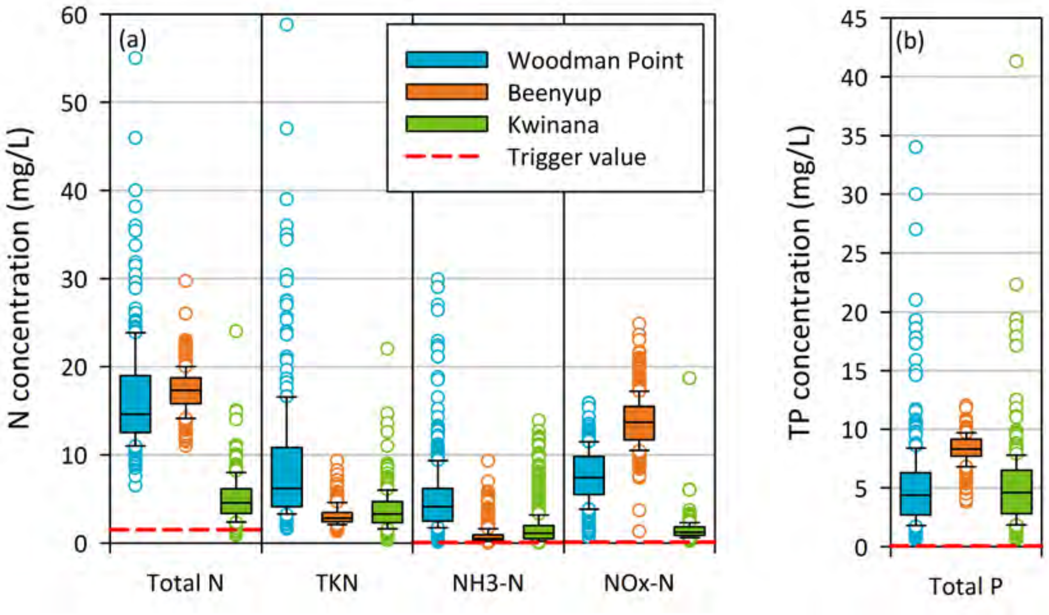

Figure 5.5 Nitrogen and phosphorus concentrations in treated wastewater from the Woodman Point, Beenyup and Kwinana wastewater treatment plants. TKN = total Kjeldahl nitrogen, NH3-N = ammonia-nitrogen and NOx-N =

nitrate+nitrite nitrogen. Box plots as described in Figure 5.3 and trigger values are for wetland aquatic ecosystems (south-west Australia, ANZECC-ARMCANZ, 2000)

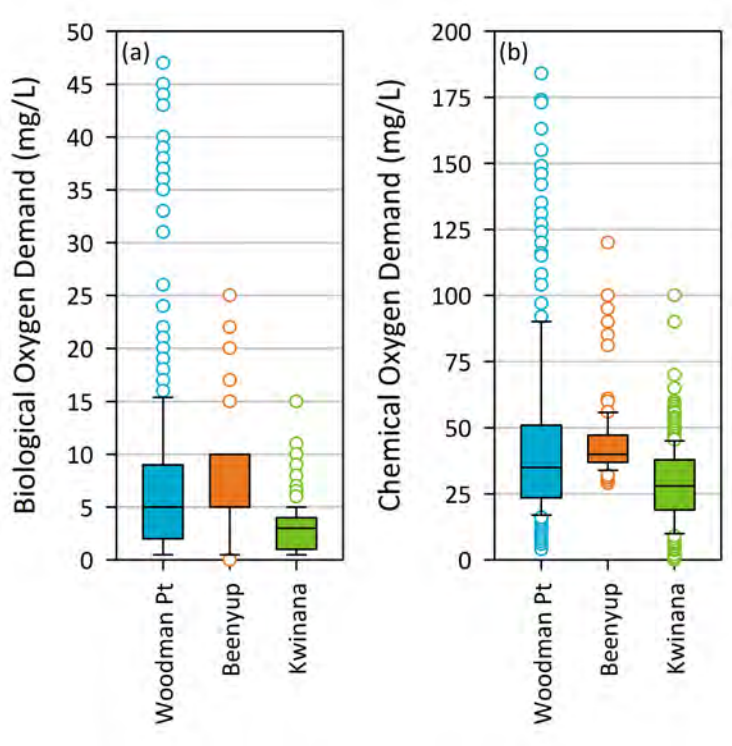

Figure 5.6 Biological and chemical oxygen demand in treated wastewater from the Woodman Point, Beenyup and Kwinana wastewater treatment plants. Box plots as described in Figure 5.3.

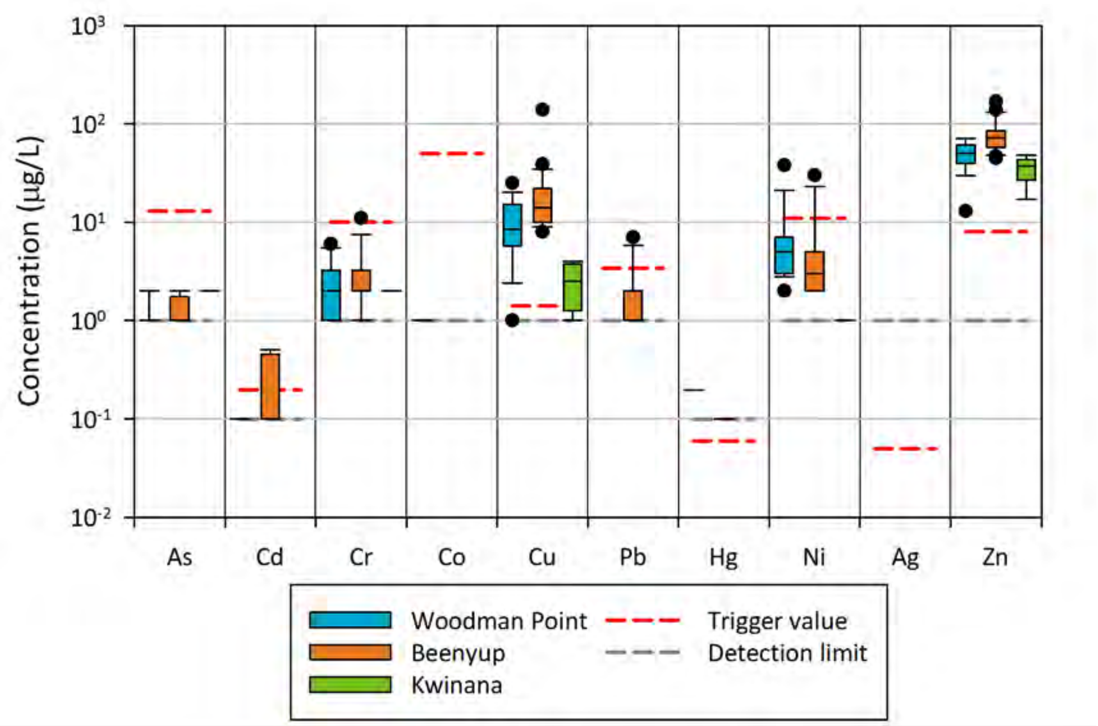

Figure 5.7 Heavy metal concentrations in treated wastewater from the Woodman Point, Beenyup and Kwinana wastewater treatment plants. Box plots as described in Figure 5.3, trigger values as described in Figure 5.4 and detection limits are the method lower reporting limits for each heavy metal.

5.4 Conclusions

The majority of treated wastewater (TWW) is produced at the Woodman Point WWTP (55 GL/yr in 2013/14) at the head of the SDOOL and this volumes will increase substantially in future. This provides an abundant source of TWW for MAR within the study area. At the smaller Kwinana WWTP most of the TWW produced is currently infiltrated with only modest increases in inflow to the WWTP projected in the future. Further to these WWTP the East Rockingham WWTP is due to be commissioned providing additional supply of TWW for MAR.

The TWW quality varies between the different WWTPs due to difference treatment processes used. The TWW is generally of higher quality from the Kwinana WWTP than the Woodman Point WWTP. The salinity and pH of all TWW sources is similar to the ambient groundwater and is within the wetland ecosystem trigger values, thus would be comparable to existing groundwater used for cooling following MAR. Given the inherent high nutrient concentrations in wastewater it is not surprising that TWW nutrient concentrations generally exceed groundwater and wetland ecosystem trigger values. However there are examples within the study area where high ambient nitrogen concentrations result from both anthropogenic and natural processes within the aquifer.

Trace metal analysis indicates that copper and zinc in TWW may potentially be of concern as both metals exceed wetland ecosystem trigger values. Data for trace organic compounds was not available for the WWTPs in the study area however literature data indicates that there may be some concern with some estrogenic compounds though further investigation is required for a wider range of compounds.

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6 Managed aquifer recharge for non-potable purposes

Authors: Elise Bekele, Mike Donn and Don McFarlane

Key findings

- Domestic wastewater volumes are increasing at almost 3% per annum in the Perth-Peel region
- The current disposal of treated wastewater by the Water Corporation in the Greater Perth area is by ocean outfall (94%), coastal infiltration basins (5%) and at the Groundwater Replenishment scheme (<1% but projected to grow)
- The Kwinana WWTP has increased the volume of treated wastewater infiltrated through ponds over the last four decades. Being inland it has mixed with ambient groundwater and travelled up to 6 km to the coast, thereby offering benefits to down-gradient water users and up-gradient lakes like The Spectacles. Maintenance costs of infiltration ponds have been low and likely to be lower following the upgrading of the plant. Nutrient loads entering the aquifer are now lower than from mineralisation of wetland peat. It was used as an analogy for MAR in this study.
- Two estimates of MAR suitability at a regional scale have shown a lot of potential in the study area. These assessments suggest there is potential to implement MAR across a large part of the Swan Coastal Plain.
- A review of MAR studies conducted in the region over recent decades demonstrates increasing confidence in its adoption for non-potable reuse purpose with a progression from desktop modelling studies to field investigations and trials.
- Where land values are a premium or land is fully developed in urban environments water can be added through infiltration galleries. This technology has been trialled at Halls Head and Floreat. Although infiltration galleries have clogged there is usually a simple reason (plant roots, insufficient filtering). Only minor clogging was experienced during a five month trial at the Floreat site at rates of 4 m/day.

6.1 Disposal of treated wastewater to the Superficial Aquifer

Domestic wastewater is a growing resource in a drying climate. The population of the Perth-Peel region (Greater Perth) has grown by about 3% per annum since 2006 (or about 50,000 people) and now exceeds 2 million. Domestic water use studies have shown that indoor water use (which ends up in the wastewater system) is relatively insensitive to income and season so it has grown at about the same 3% rate assuming in-house water use efficiencies are more than offset by reductions in out-door use as a result of smaller block sizes and outdoor efficiencies.

As a result of environmental issues associated with discharge to the Swan and Canning estuary, inland wastewater treatment plants were progressively closed last century and large plants on the coast established to enable excess treated wastewater to be discharged to the ocean via outfalls. The current distribution of these plants, their catchments, volumes and outfalls are shown in Table 6.1 and Figure 6.1.

Table 6.1 Treated wastewater volumes in 2013/14 and their disposal

	VOLUME OF WASTEWATER (GL/YR)
Ocean outfalls	
Alkimos	2.4
Ocean Reef	50.0
Swanbourne	22.3
Point Peron	58.4
Total ocean outfalls	**133.1**
Infiltration sites	
Yanchep	0.27
Kwinana	1.80
Gordon Rd	3.50
Halls Head	1.10
Caddadup	0.56
Total infiltration	**7.23**
Groundwater replenishment trial (Beenyup)	**1.2**
Total wastewater volumes	**141.53**

The 142 GL/yr collection of treated wastewater is about half of that delivered to households and industry in the Integrated Water Supply Scheme. Most of the remainder is used on irrigating lawns and gardens so is lost to the reticulation system but may add to the Superficial Aquifer when watering is excessive. About 94% is secondary treated and disposed of to the ocean, 5% is infiltrated as secondary treated wastewater and the remaining 1% is tertiary treated and injected into the Leederville aquifer for eventual use in the drinking water system. The Department of Water has approved a plan by the Water Corporation plan to inject 14 GL/yr into the Leederville aquifer by 2016.

The disposal of the 1.8 GL/yr at the Kwinana WWTP is a surrogate of MAR in the Cockburn Sound catchment and therefore was intensively investigated in this project with the results detailed in Section 6.4. By far the largest volume of treated wastewater is discharged from the Point Peron outfall (58 GL/yr or 44% of all ocean outfall), with most of this coming from the Woodman Point WWTP in the north of the study area. The Kwinana WWTP infiltration site is the second largest in the Greater Perth area, being exceeded only by Gordon Road which lies within the City of Mandurah. Infiltration in both cases started in the 1970s and has grown as the nearby residential areas have grown and been connected to the sewerage system.

This chapter looks at the history of managed aquifer recharge investigations at the regional and local scales before detailing the Kwinana WWTP experience and drawing some general conclusions.

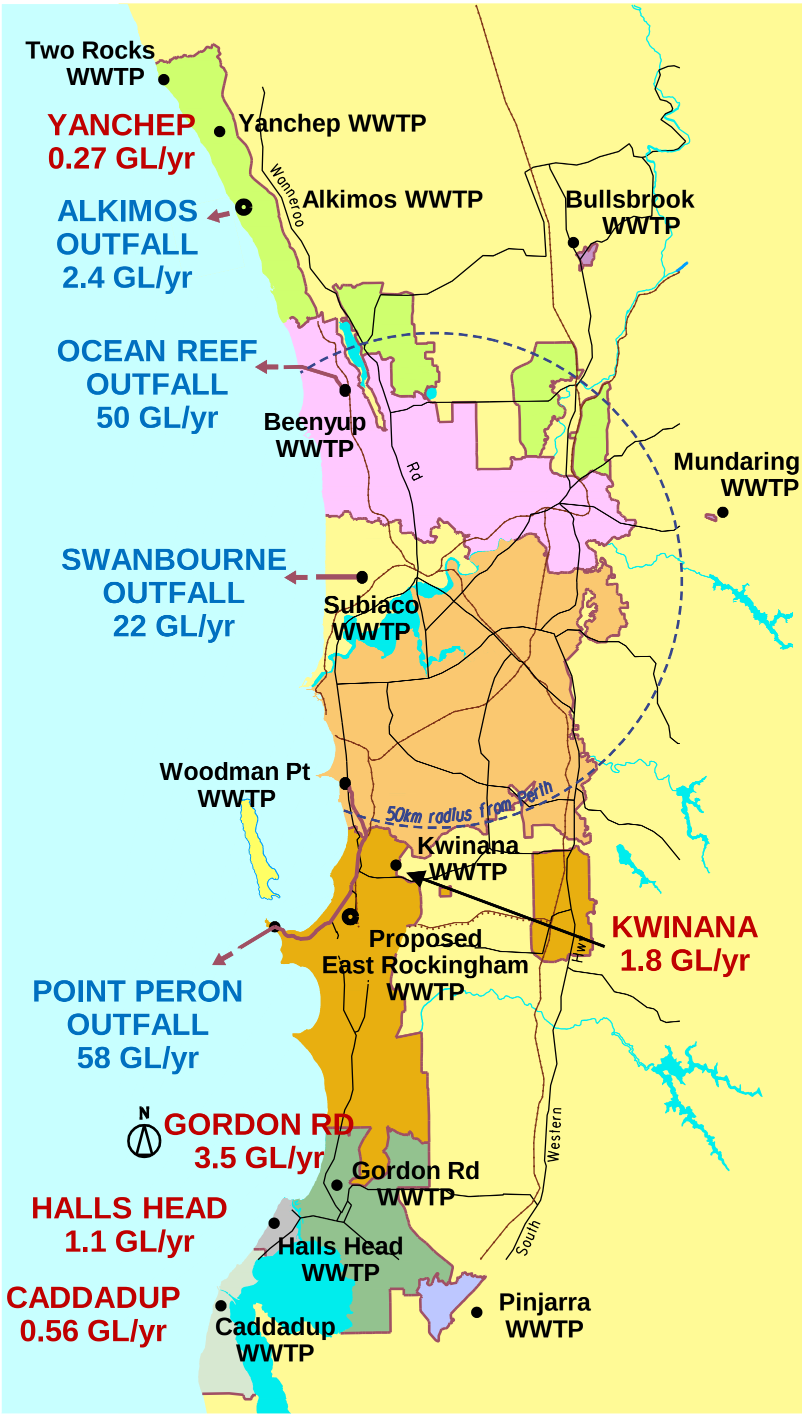

Figure 6.1 Location of wastewater treatment plants, catchment areas and ocean outfall in the Perth Peel area. Volumes of wastewater disposed via infiltration basins are highlighted in red. Volumes of wastewater discharge via ocean outfall are labelled in blue (Source: Water Corporation)

6.2 MAR suitability mapping at the regional scale

One of the first critical assessments of MAR in the Perth coastal area was in the Perth urban water balance study by Cargeeg et al. (1987a, b). This work involved the development of a regional groundwater flow model (a predecessor of PRAMS) to aid planning and management decisions. Due to a lack of information on variations in hydraulic properties, the regional model did not incorporate heterogeneity. The model was used to investigate scenarios for MAR with water of impaired quality (i.e. stormwater and treated wastewater). The study was triggered by an 18- month sprinkler ban and salt water intrusion causing shallow bores to fail in riverine peninsular areas and along the coast in the late 1970s and early 1980s. One of the key recommendations was to recharge the Superficial Aquifer with stormwater using infiltration basins. Cargeeg et al. (1987b) also proposed MAR with treated wastewater to raise the watertable near wetlands. The concept of placing a line of injection wells for MAR west of the Wanneroo chain of wetlands and similarly for the East Beeliar chain of wetlands was investigated. For the Beeliar chain of wetlands, Cargeeg et al. (1987b) proposed a line of recharge wells that would inject a total of nearly 3 ML annually to the Superficial Aquifer, resulting in about a 1.0 m rise in the watertable in the vicinity. This would be effective in protecting the East Beeliar wetlands from drying out; however, the well injection scheme was never trialled.

The feasibility of MAR in the Perth Basin at a regional scale was investigated by Scatena and Williamson (1999). Their report identified that heterogeneity and isotropy of hydraulic properties, aquifer thickness and areal extent are critical factors for assessing whether an aquifer is suitable for recharge. Scatena and Williams (1999) analysed different parameters for stratigraphic units in the Perth Basin and discovered that the Bassendean Sand and Tamala Limestone are the most suitable for recharge using both well injection and infiltration. Further evaluation of MAR suitability within these two aquifer units was mapped based on adequate depth to watertable and aquifer transmissivity (Figure 6.2). A minimum depth to the watertable of 6 m and a minimum transmissivity of 600 m^2/day were the criteria used to identify suitable areas for MAR. These criteria were deemed necessary to prevent excessive mounding. A greater depth to the watertable facilitates biodegradation in the unsaturated zone and a large transmissivity facilitates MAR injection and subsequent well yields (Scatena and Williamson 1999).

More recently, a regional study of the hydraulic feasibility of MAR in the Superficial Aquifer was conducted by Smith and Pollock (2010). The approach involved estimating the aquifer response to different recharge rates by basin infiltration and well injection. Analytical models were used to predict the growth in the watertable mound in relation to different hydraulic loads (Smith and Pollock 2010). The models incorporated spatial variations in aquifer transmissivity. In contrast to the study by Scatena and Williamson (1999), a fixed depth to the watertable was not imposed. Smith and Pollock (2010) mapped annual mean depth to groundwater and used these data to assess the limit on vertical growth of the predicated recharge mounds. The analysis did not consider potential effects of recharge water quality on physical, chemical and biological clogging. With regard to the modelling of basin infiltration, three hydraulic loads were tested, corresponding to small (1 ML/day), medium (5 ML/day), and large (10 ML/day) scales of MAR operation and applied over a 30-day period.

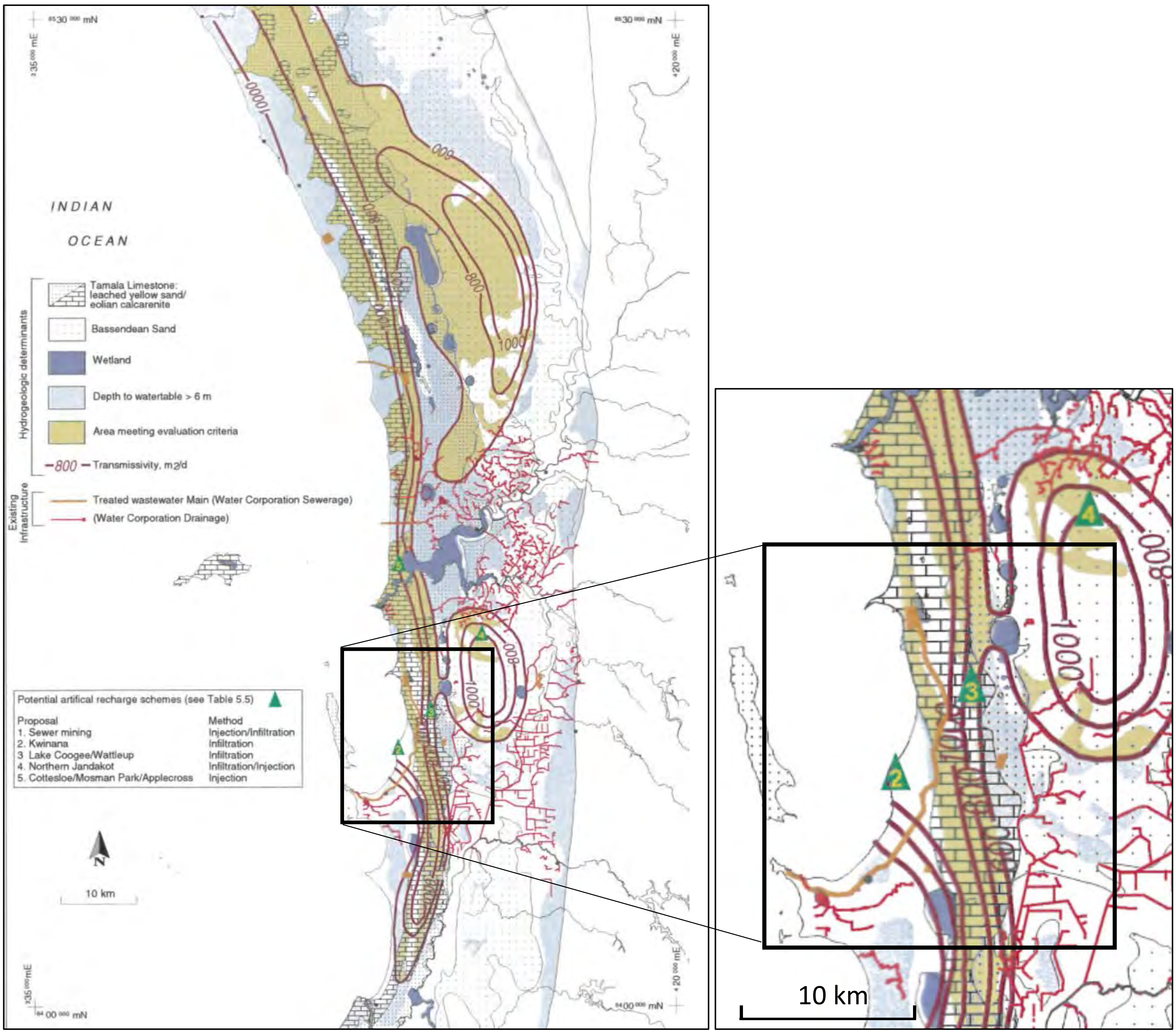

Figure 6.2 Suitability mapping of MAR in the Bassendean Sand and Tamala Limestone. The tan coloured areas meet the criteria for MAR suitability (i.e. a minimum depth to the watertable of 6 m and a minimum transmissivity of 600 m^2/day. A close up image of the CSC study area is shown on the right. Adapted from "A potential role for artificial recharge in the Perth Region: A pre-feasibility study," by M. Scatena and D. Williamson (1999), Centre for Groundwater Studies, Perth, Report No. 84. Copyright 1999 by Centre for Groundwater Studies. Adapted with permission.

Figure 6.3 shows the amounts of relative watertable rise (RWR), defined as the ratio of the predicted watertable mound to the depth to static watertable for the basin infiltration modelling (Smith and Pollock 2010). A value of RWR less than 1 indicates that the predicated recharge mound does not exceed ground surface elevation after 30 days, whereas a RWR value greater than or equal to 1 indicates that the predicted recharge mound exceeds the ground surface elevation before 30 days. RWR greater than or equal to 1 indicates the potential for waterlogging (Smith and Pollock 2012). Within the CSC study area, the results for the three hydraulic loads indicate a coastal strip of about 10 km wide, extending inland from the coastline that have RWR values mainly between 0 and 0.5 (Figure 6.3). The coastal strip was identified as potentially suitable for medium- to large-scale MAR due to the presence of sandy surface soils and moderate to large aquifer transmissivity (Smith and Pollock 2010). While it is preferable to not have recently infiltrated wastewater express at the soil surface, raising groundwater levels to recover throughflow wetlands could be a major benefit of MAR in cases such as Perry Lake and The Spectacles lakes as outlined below. Therefore areas with a RWR value of 1 or more should not be discarded without further investigation.

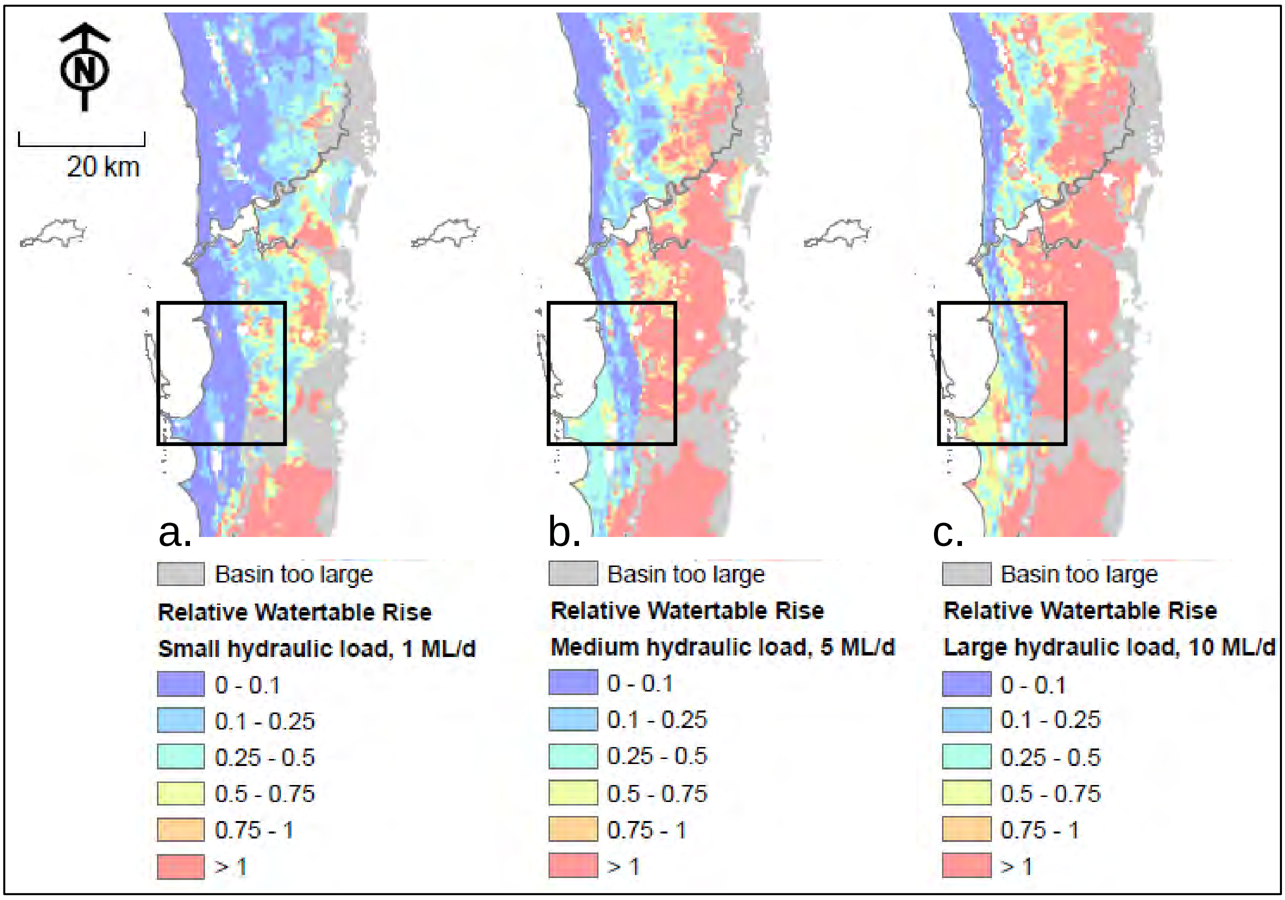

Figure 6.3 Relative watertable rise, defined as the ratio of the predicted watertable mound to the depth to static watertable for the basin infiltration modelling at 30 days for basin infiltration as the basis of MAR feasibility mapping. The boxed area is the CSC area for the present investigation. The images show the results from analytical models of basin infiltration using different hydraulic loads: (a.) small, (b) medium, and (c) large. Areas rendered in grey were excluded from the analysis because the basin size required to achieve the particular hydraulic load would exceed the criterion for maximum basin area (0.25 km^2) used by Smith and Pollock (2010). Adapted from "Artificial recharge potential of the Perth region superficial aquifer: Lake Preston to Moore River," by A. Smith and D. Pollock (2010), CSIRO Water for a Healthy Country National Research Flagship Report. Copyright 2010 by CSIRO. Adapted with permission.

6.3 Modelling and field studies of Managed Aquifer Recharge

6.3.1 MANAGED AQUIFER RECHARGE IN BASSENDEAN SAND

A recharge study was carried out near the Canning Vale WWTP to investigate the removal of contaminants and infiltration rates in Bassendean Sand recharged with secondary treated wastewater in infiltration basins with cyclic flooding and drying (Binnie and Partners 1976; Mathew et al. 1982). The results from the field study were that infiltration rates of 0.5 m/day could be maintained in the Bassendean Sands; however, nitrogen and long term phosphorous removal did not occur (Mathew et al. 1982). Typical values for the effluent quality sampled from the Canning Vale WWTP quality were as follows: total dissolved solids of 434 mg/L, suspended solids of 4 mg/L, total nitrogen of 17 mg/L, phosphorous of 4.6 mg/L and BOD5 of 8 mg/L as reported in Mathew et al. (1982) based on sampling by Binnie and Partners (1976). Significant removal of bacteria occurred within the upper part of the soil profile, and no bacteria were detected in bores more than 100 m from the infiltration basins (Mathew et al. 1982).

Wu (2003) conducted a modelling study of MAR by injection well in Bassendean Sand. The purpose was to determine the impact of spatial variability in the properties of Bassendean Sand on the effectiveness of MAR. Statistical models of aquifer heterogeneity with the characteristics of Bassendean sand were developed for the groundwater model. The effectiveness of MAR was assessed by investigating the success of sustaining wetlands that depend on the watertable being close to the ground surface. Wu (2003) compared the relative heights of the watertable mound simulated under conditions of low versus high rates of injection. At low injection rates, the mound spread more than rose, whereas the opposite occurs under high injection rates. The modelling by Wu (2003) demonstrated that the extent of increase in the rate of watertable rise relative to the radius of influence depends on the heterogeneity of the system. Aquifers with low spatial heterogeneity were found to generally be more suitable for MAR with the objective of sustaining wetlands (Wu 2003). According to Wu (2003), this is mainly caused by interconnected lenses of high hydraulic conductivity within heterogeneous systems generating greater lateral flow, which results in less rise in the watertable than in a homogeneous system (for the same injection rate).

6.3.2 MANAGED AQUIFER RECHARGE IN SPEARWOOD SAND OVERLYING TAMALA LIMESTONE

A study to assess the feasibility of using reclaimed wastewater to replenish groundwater in the Superficial Aquifer on the Mosman Peninsula in WA was initiated in 2003 by the Water Corporation to mitigate saltwater intrusion (Blair and Turner 2004). The peninsula is about 2 km wide and lies between the Indian Ocean and the Swan River (Figure 6.4; Rümmler et al. 2005). The thickness of Spearwood Sand overlying Tamala Limestone is variable and up to 20 m in some of the bore logs analysed by Rümmler et al. (2005).

Prior to the hydrogeological characterisation and groundwater quality assessment by Rümmler et al. (2005), the Water Corporation commissioned a study to model the groundwater system to allow preliminary assessment of MAR by well injection, and to predict likely impacts on groundwater abstraction and sensitive environments (ie. from the discharge of nutrient-rich water to the Swan River and Indian Ocean; Prommer et al. 2004). MODFLOW-SEAWAT was used to model variable-density groundwater flow, but as there were insufficient groundwater monitoring data available at the time of modelling, Prommer et al. (2004) were unable to properly calibrate a non-reactive transport model for the area.

Figure 6.5 shows initial model results of a four well injection scheme to recharge 1.5 GL/yr of recycled water, which is equivalent to the estimate of total groundwater abstraction on the Mosman Peninsula at that time (Blair and Turner 2004). In this worst case scenario, although a large proportion of groundwater users would benefit, the model predicts nutrient discharge to both the Swan River and the Indian Ocean (Prommer et al. 2004). This result was a consequence of assuming no natural dentrification and no additional pre-treatment of the source water to remove nitrogen (Blair and Turner 2004). Recent increases in groundwater salinity in the peninsula and a lack of available irrigation water has rekindled an interest in both direct and indirect use of treated wastewater to address emerging groundwater quantity and quality issues in WESROC councils (Roy Stone pers. comm.).

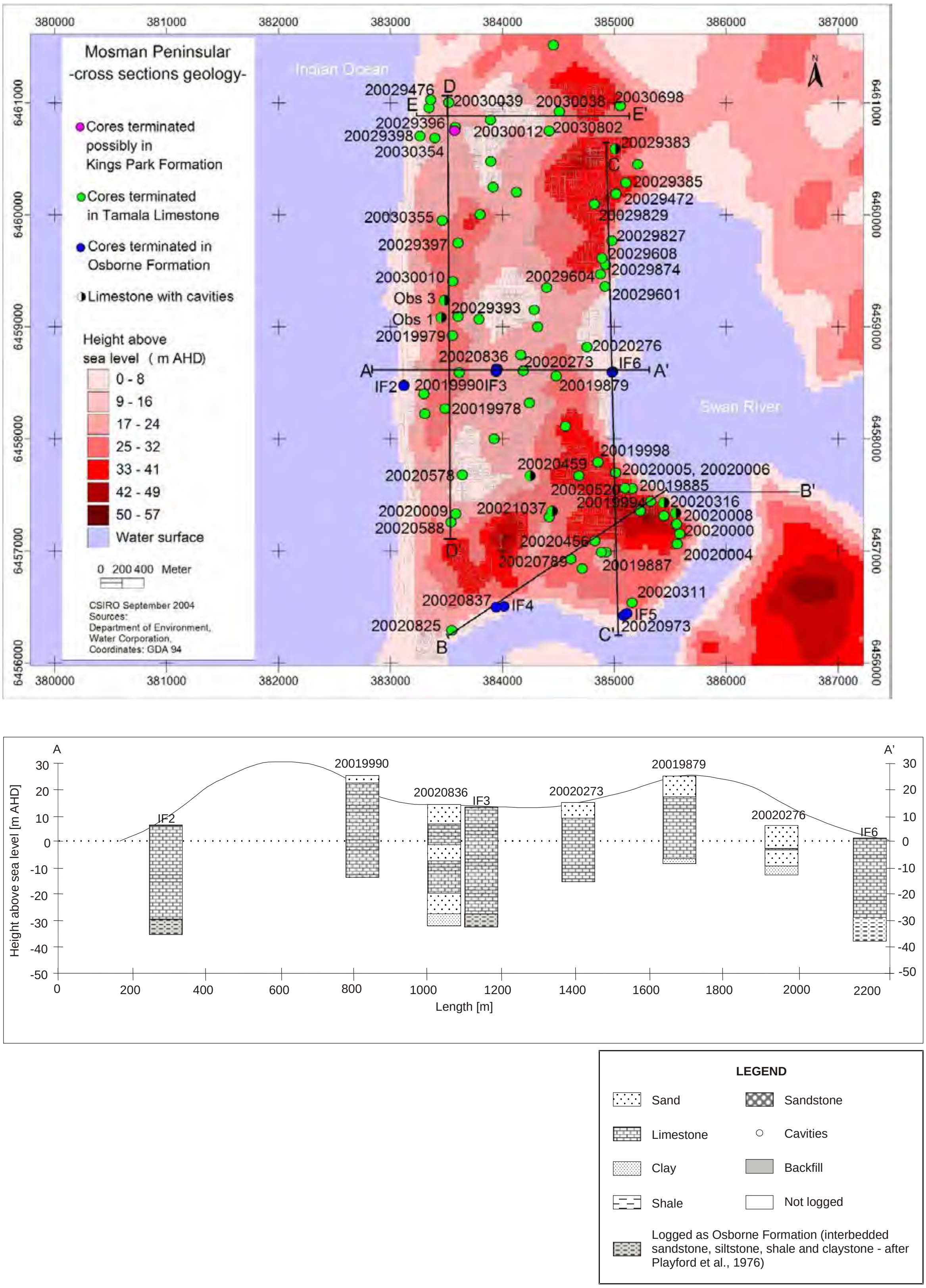

Figure 6.4 Map view of the Mosman peninsula showing the location of bores with logged stratigraphy and the location of cross-sections. The lower image is section A-A' (West to East). Adapted from "Hydrogeological characterisation and preliminary groundwater quality assessment for the Mosman Peninsula in Perth, Western Australia," by J. Rümmler et al. **(2005), CSIRO Water for a Healthy Country National Research Flagship Report. Copyright 2005 by CSIRO. Adapted with permission.**

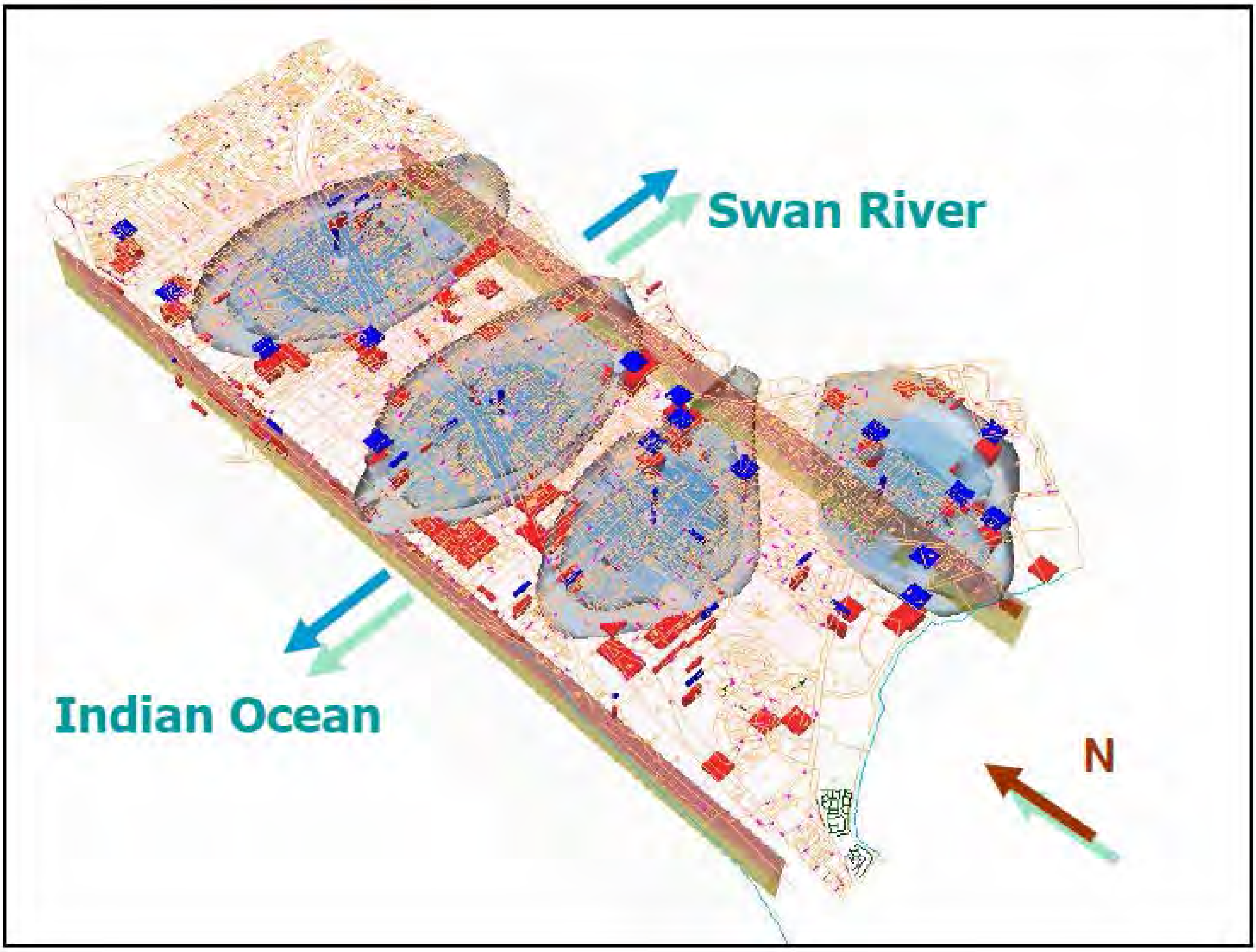

Figure 6.5 Simulated nitrogen plumes from a proposed MAR scheme on the Mosman Peninsula under a four-well injection scenario after seven years of recharge. Reprinted from "Groundwater - a crucial element of water recycling in Perth, Western Australia," by P. Blair and N. Turner (2004), Proceedings of International Conference on Water Sensitive Urban Design. Copyright 2004 by Engineers Australia. Reprinted with permission.

Pilot trials of MAR in Spearwood Sand overlying Tamala Limestone were conducted for a 3- year experiment, beginning in 2005 (Bekele et al. 2009), and a 1-year experiment in 2013/2014 (Bekele et al. 2015) at two separate installations at the Floreat Infiltration Galleries (FIG) site (Figure 6.6). While most of the documented trials of MAR on the Swan Coastal Plain involved infiltration basins, FIG was established to explore the potential for using a technique that involved covered percolation trenches (galleries), containing a medium or supporting structure with internal void spaces to facilitation infiltration. Both projects involved infiltration of secondary treated wastewater in galleries that were buried to a depth of 0.5 m below ground in Spearwood Sand[6]. The site consisted of about 7 m of Spearwood Sand grading into Tamala Limestone and a depth to watertable between 10 and 11 m below ground, depending on the season.

There are a number of advantages to using infiltration galleries for MAR over infiltration ponds, particularly in an urban setting. Buried galleries are designed to have a minimal surface footprint and minimal exposure of the community to wastewater. In addition, there can be multiple purposes for the land overlying a gallery (Bekele et al. 2011a). Exposed ponds of recycled water can pose risks to public health and safety, and there can be odour and pest issues (Vanderzalm et al. 2015). Moreover, there is potential for water loss through evaporation. Another concern is that the quality of wastewater quality can deteriorate during short-term storage in open storage ponds (Higgins et al. 2009). In a study of changes in recycled water

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[6] infiltration galleries were also trialled in Tamala Limestone at the Halls Head WWTP (See Section 6.3.3),

uality sampled from three open surface ponds with short-term storage, Higgins et al. (2009) observed increases in faecal coliforms, nutrients and chemical oxygen demand, most likely due to avian faecal contamination. According to Higgins et al. (2009), the one to two orders magnitude increase detected in faecal coliforms may limit the options for reuse under Australian water recycling guidelines. As infiltration galleries are buried and not exposed to sunlight, these systems are not likely to experience algal growth, which occurs on the basin floors of surface infiltration systems (NRMMC-EPHC-NHMRC 2009). The photosynthetic consumption of carbon dioxide may also enhance clogging by precipitating carbonate minerals (NRMMC-EPHC-NHMRC 2009).

Unlike infiltration basins which have been used widely to dispose of treated wastewater as outlined in Section 6.1, there have not been trials of sufficient duration of infiltration galleries for MAR in Spearwood Sand to determine the frequency and type of maintenance required. MAR via infiltration ponds or galleries is considered less expensive than well injection as there are costs associated with establishing wells and pre-treating water to a sufficient standard to minimize clogging (Dillon 2005). The economics of MAR options are more thoroughly discussed by Vanderzalm et al. (2015). Clogging in infiltration galleries is also a concern, but the criteria for water quality treatment to minimise clogging for galleries are generally less stringent than for well injection. Improperly designed well injection systems can clog within days or weeks, whereas basins may clog within weeks or months (NRMMC-EPHC-NHMRC 2009). The clogging potential for well injection is greater because a larger volume of water is injected over a smaller surface area compared with infiltration systems (galleries or basins); consequently, there is greater potential for interactions between the source water, its constituents and the porous media to lead to clogging (NRMMC-EPHC-NHMRC 2009).

Three-year MAR trial at the Floreat Infiltration Galleries site

In a comparative study of two gallery designs, the hydraulic performance of a gravel-filled trench was compared with a series of modular, lightweight, polypropylene crates referred to as the Atlantis Flo-Tank modules ® (Bekele et al. 2009; 2013). Both galleries received 25 kL of nutrient-rich, secondary treated wastewater per day. This equated to about 1 m/day of infiltration through the base of each gallery. The Atlantis gallery successfully infiltrated 17 ML of treated wastewater over three years. The slotted distribution pipe in the gravel gallery became clogged with plant roots after operating for one year. The infiltration capacity of the gravel gallery could not be restored despite high pressure cleaning, thus it was replaced with an Atlantis system (Bekele et al. 2009; 2011). Reduction in the infiltration capacity of the Atlantis system, indicative of clogging, was only observed when inflow was increased by about 3 fold for two months. The performance of the Atlantis system suggests it is superior to the gravel gallery, requiring less maintenance within at least the time frame of this study.

The 3-year FIG study also documented water quality changes resulting from passage of treated wastewater vertically through the Spearwood Sand (about 7 m thick) and laterally down-gradient through 50 m of Tamala Limestone (Bekele et al. 2009; 2011). A pumping bore operated continuously to abstract five times the infiltration rate. Results from a bromide tracer test indicated a minimum travel time of about four days for the recharged water to reach the watertable (Bekele et al. 2013).

The water quality improvements of the recycled water were based on changes in the chemistry and microbiology of (i) the recycled water prior to infiltration relative to (ii) groundwater immediately down-gradient from the infiltration gallery (Bekele et al. 2011a). A series of monitoring bores slotted over different depths were frequently sampled for a range of chemical and microbial constituents (Figure 6.6). Changes in the average concentrations of several constituents in the recycled water were identified with reductions of 30% for phosphorous, 66% for fluoride, 62% for iron and 51% for total organic carbon with average starting concentrations in recycled water of 4.71 mg/L for phosphorous, 0.73 mg/L for fluoride, 0.14 mg/L for iron and 7.78 mg/L for total organic carbon (Bekele et al. 2011a). The secondary treated wastewater was infiltrated at an applied rate of 17.5 L per minute with a residence time of approximately four days in the vadose zone and less than two days in the aquifer (Bekele et al. 2011a). Reductions were also noted for oxazepam and temazepam among the pharmaceuticals tested and for a range of microbial pathogens, but reductions were harder to quantify as their magnitudes in source water varied over time. Total nitrogen and carbamazepine persisted in groundwater down-gradient from the infiltration galleries

(Bekele et al. 2011a). Both the ambient groundwater and the recycled water were consistently aerobic during the 3-year FIG study.

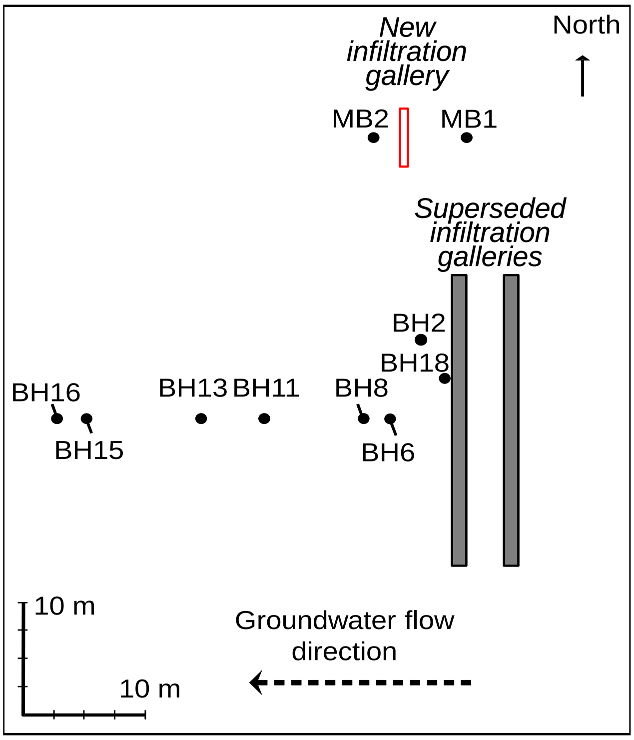

Figure 6.6 The Floreat MAR site showing the location of the pair of infiltration galleries to the south used in the study by Bekele et al. (2009), the newer gallery installed for the study by Bekele et al. (2015), and some of the monitoring bores (MB and BH). The bores were slotted within the Tamala Limestone, which underlies about 7 m of Spearwood Sand at the site. The north gallery was 4 m in length, whereas the southern pair of galleries were 25 m in length. Adapted from "Managed aquifer recharge and recycling options (MARRO): Understanding clogging processes and water quality impacts," by Bekele et al. (2015), Australian Water Recycling Centre of Excellence Report. Copyright 2015 by Australian Water Recycling Centre of Excellence. Adapted with permission.

Perry Lakes Managed Aquifer Recharge proposal

A MAR proposal was developed to replenish the Superficial Aquifer near Perry Lakes with secondary treated wastewater, thus providing ecological benefits to Perry Lakes, a flowthrough wetland located 500 m north of the Floreat MAR trial (Figure 6.7; McFarlane et al. 2009; Bekele et al. 2011b). The proposal also involved irrigating the parkland around the lakes, using existing groundwater abstraction bores, which are slotted towards the base of the aquifer (Bekele et al. 2011b). At this site, there is about 10 m of Spearwood Sand overlying Tamala Limestone (Figure 6.8).

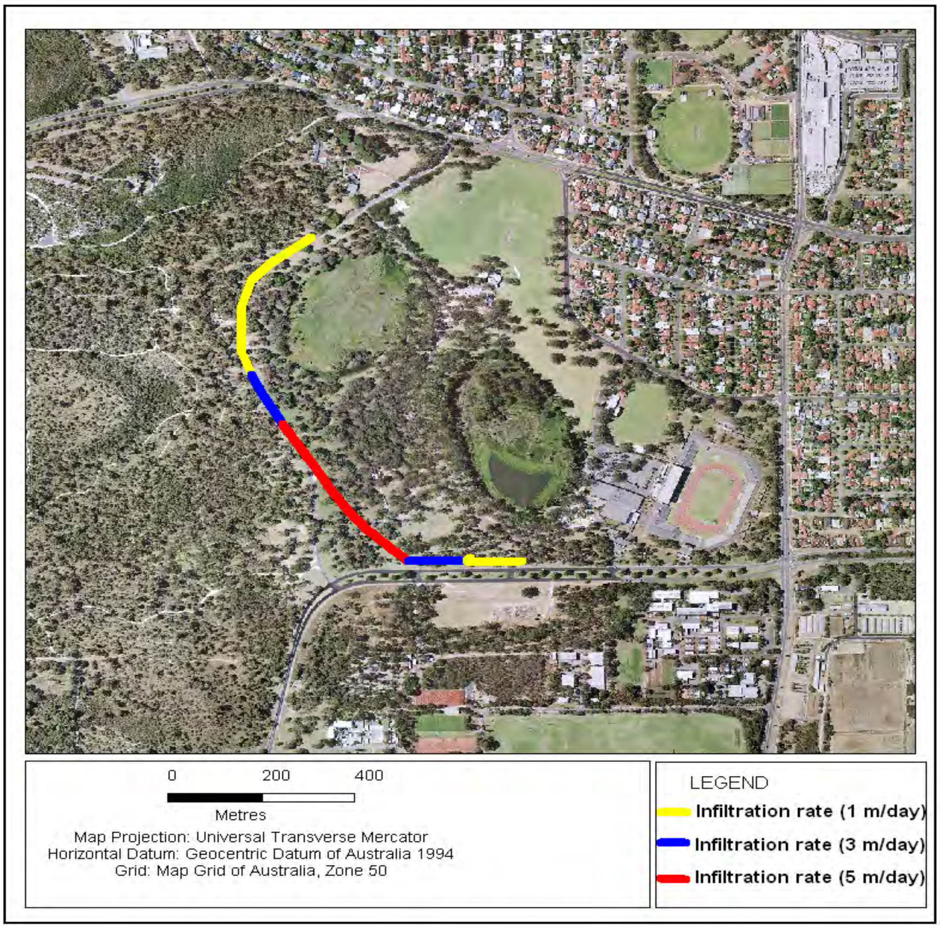

Figure 6.7 Aerial photo of Perry Lakes showing the recommended gallery alignment with spatially variable infiltration rates. Reprinted from "Design and costs effectiveness of infiltration galleries ay Perry Lakes," by Drummond et al**. (2011), AWA Ozwater Conference. Copyright 2011. Reprinted with permission.**

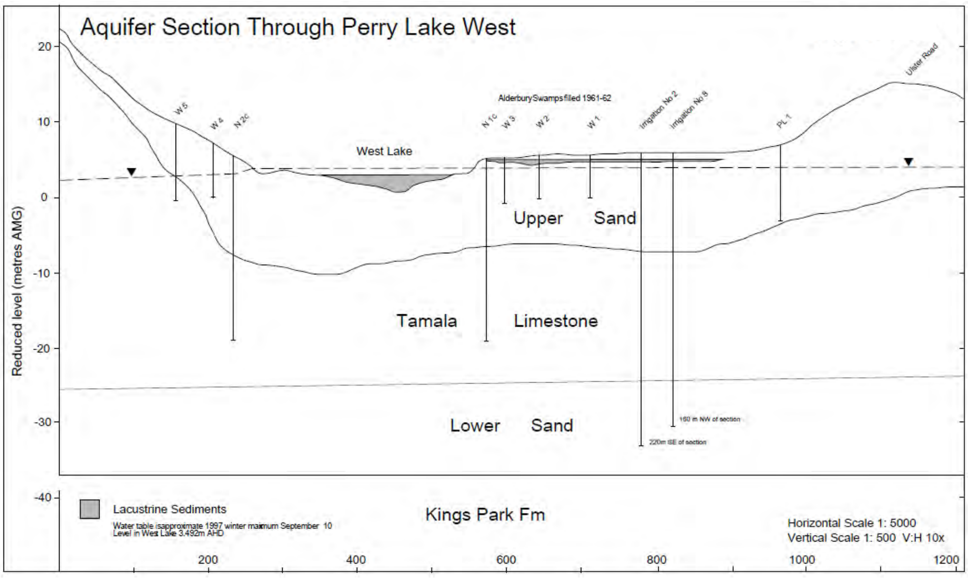

Figure 6.8 Aquifer section through Perry Lakes West oriented southwest-northeast, showing the stratigraphy and the maximum elevation of the watertable in winter September 1997. Reprinted from "Integrated mass, solute, isotopic and thermal balances of a coastal wetland," by J. Rich (2004), PhD thesis Murdoch University, Western Australia. Reprinted with permission.

The proposed MAR project was intended to produce a watertable mound by recharging the aquifer using a series of shallow infiltration galleries. The watertable mound would serve as a partial hydraulic dam that would raise the watertable beneath East and West Lakes in Perry Lakes Reserve and as far east as lakes Monger and Herdsman. The intent was for recharged water to not enter the lakes (Figure 6.9).

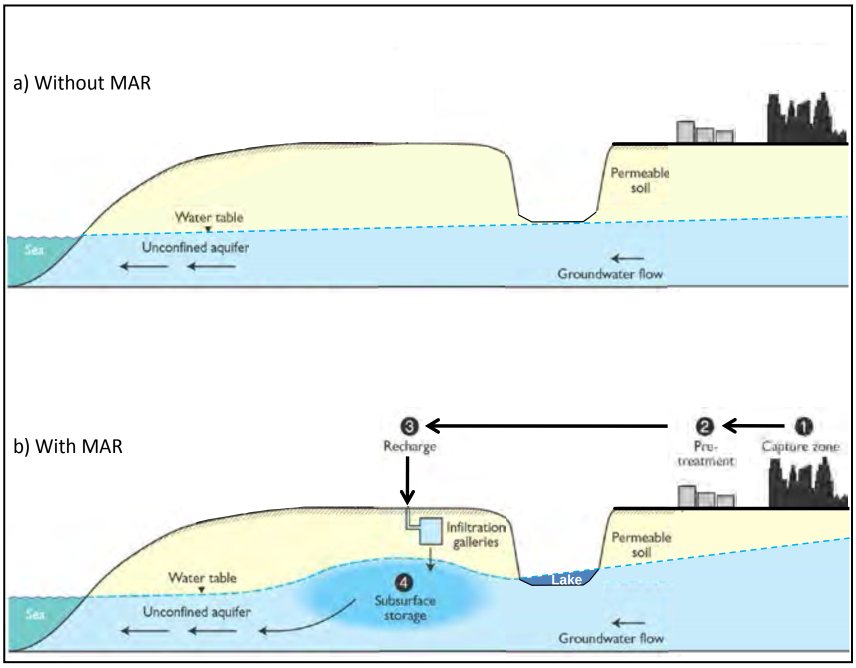

Figure 6.9 Perry Lakes aquifer replenishment schematic, depicting (a) the watertable gradient without MAR and drying of a lake, and (b) raising of the watertable and regional groundwater flowing beneath and into a lake in response to a watertable mound produced by recharge via infiltration galleries. Reprinted from "Application of the Australian guidelines for water recycling Phase 2 Managed aquifer recharge to Perry Lakes example," by Bekele et al. **(2011b), CSIRO Water for a Healthy Country National Research Flagship Report. Copyright 2011 by CSIRO. Reprinted with permission.**

An alignment of infiltration galleries adjacent to Perry Lakes was modelled, covering an infiltration area of 1,300 m^2 and the results from preliminary steady-state modelling by CSIRO suggested an infiltration rate of 4 m/day would raise groundwater levels under Perry Lakes by 1 m compared with current levels (McFarlane et al. 2009). According to the model, a mound height of between 0.5 and 1.0 m would be produced below ground, extending to 2 km radially from the infiltration site. The consulting services of GHD were engaged by the proponents to develop a concept design. According to the GHD study, which used a transient groundwater flow model to predict flow paths and water level changes, 1.2 m/day of infiltration through the base of the galleries covering a total infiltration area of 2,500 m^2 would restore the periodic presence of water in Perry Lakes (GHD 2011); however, this rate was adjusted to account for the operational efficiency of resting galleries to enhance aerobic conditions and a safety margin, hence a design flow rate of 5 ML/day, equivalent to an infiltration rate of 2 m/day through the base of the gallery was given in their final recommendation (GHD 2011).

The proposal for Perry Lakes did not proceed due to concerns about the long-term operating costs of the scheme. Specifically there was a lack of knowledge regarding the frequency of clogging and the effect on estimates of ongoing maintenance costs. This knowledge gap was the impetus for a study of clogging at the FIG site as discussed in the next section.

Clogging study at the Floreat Infiltration Galleries site

As identified in the Perry Lakes MAR proposal, while there is long-term experience with infiltration through open basins in wastewater treatment plant sites, there are relatively few well-documented, long-term field studies to provide guidance and confidence in infiltration galley techniques to recharge the Spearwood Sands with wastewater at high rates of infiltration. Prior experience from the 3-year trial at FIG involved wastewater infiltration rates of about 1 m/day and higher rates were only tested for the final few months.

The aim of the study by Bekele et al. (2015) was to understand the impacts of operating at higher infiltration rates (3.5 to 5.0 m/d) including clogging and nitrogen cycling associated with differing water qualities, such as unfiltered and filtered secondary treated wastewater. Infiltration rates of this order were recommended for the Perry Lakes proposal and MAR for other purposes (e.g. prevention of salt water intrusion; management of existing contaminant plumes) would likely require high rates of infiltration. Water quality monitoring was conducted in conjunction with the investigation of hydraulic performance to ascertain the water quality criteria that may impact clogging.

The research outcomes demonstrated that an infiltration gallery constructed of Atlantis Flo-Tank® modules in Spearwood sands can sustain recharge to the aquifer using secondary treated wastewater applied at an average rate of 4 m/day over a 5 month period (Figure 6.6; Bekele et al. 2015). A total of 750 kilolitres was recharged to the Tamala aquifer over this period (average rate of 6.7 kL/day), but the watertable elevation only changed in response to the seasonal pattern of rainfall instead of MAR due to the highly conductive Spearwood Sand and Tamala Limestone at this site (Bekele et al. 2015) and the relatively small volume of water added through the short gallery (Figure 6.6). Whilst high rates of recharge to the aquifer were sustained over the entire duration of the field experiment, changes occurred spatially in gallery wastewater levels and soil moisture contents surrounding the gallery within about 4 months. These observations support the theory that heterogeneous clogging developed locally within the gallery and promoted flow of wastewater through preferential flow paths and laterally away from the gallery (Bekele et al. 2015). To extrapolate results from this trial to new sites for MAR requires several caveats: it is anticipated that a gallery located in sands with similar hydraulic properties with no lateral restrictions to the outflow of wastewater could sustain recharge to the aquifer over a reasonable period of time. To ensure satisfactory hydraulic performance, filtration equipment should be frequently monitored to maintain a target level of total suspended solids of 5 mg/L. Based on this study, an infiltration gallery sized similarly and receiving wastewater of similar quality should provide satisfactory hydraulic performance for at least 6 months. Note that the maintenance of the galleries is yet to be tested.

6.3.3 MANAGED AQUIFER RECHARGE IN TAMALA LIMESTONE

As discussed in Section 6.1, there are several facilities operated by the Water Corporation that infiltrate a combined annual total of 7.23 GL of treated wastewater via infiltration basins on the Swan Coastal Plain. The disposal of wastewater at the Yanchep, Gordon Road, Halls Head and Caddadup WWTPs is principally in Tamala Limestone as the basins are lined with only a relatively thin layer of sand (Figure 6.10; Smith et al. 2012).

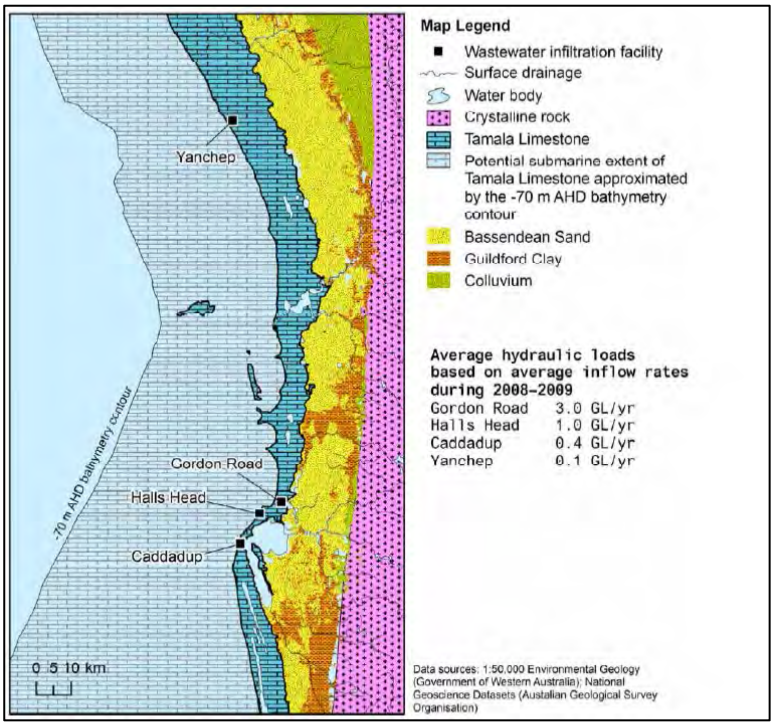

Figure 6.10 Locations of treated wastewater infiltration basins in Tamala Limestone. Reprinted from ") Geohydrology of the Tamala Limestone Formation in the Perth Region: origin and role of secondary porosity," by Smith et al**. (2012), CSIRO Water for a Healthy Country National Research Flagship Report. Copyright 2012 by CSIRO. Reprinted with permission.**

One of the first projects to document MAR in Tamala Limestone was conducted by Toze et al. (2002; 2004). It was a pilot indirect reuse scheme for the reuse of treated wastewater from the Halls Head WWTP for irrigation of public open space and road verges. Two recovery bores (SPB1 and SPB2) were installed at 80 m and 100 m from the infiltration ponds to abstract a mixture of ambient groundwater mixed with treated wastewater (Figure 6.11). This recycled water was then pumped to a storage tank for irrigation purposes. The results from the project showed improvements in recycled water quality (chemically and microbiologically) compared with the treated wastewater and that the recycled water is more suitable for irrigation than native groundwater (Toze et al. 2004).

To further investigate infiltration of wastewater at the Halls Head WWTP, a pair of infiltration galleries was installed in the Tamala Limestone, less than 200 m from the infiltration basins (Figure 6.11). The Halls Head galleries were intended to operate for several years beginning in 2005, but due to redevelopment at the WWTP, the study was terminated after only 22 months. A total of 8.5 ML of treated wastewater was infiltrated. The infiltration galleries received a daily supply of treated wastewater. Due to low flows of wastewater at various times at the WWTP (typically at night), the pump delivering wastewater to the galleries would frequently stop and require manual resetting. As there was no full time operational staff and the galleries were not monitored daily, a timer was installed to automatically shut off the pump at night.

The watertable depth below the Halls Head galleries was relatively shallow (2 m below ground on average). Groundwater levels in bore 2/84 (southeast of the infiltration galleries) mainly responded to tidal fluctuations, whereas groundwater levels recorded beneath the galleries were up to 10 cm higher and responded to both tidal fluctuations and the daily pulse of treated wastewater (Figure 6.12; Bekele et al. 2009). The delineation of groundwater flow directions in the aquifer was difficult due to the high transmissivity of the Tamala Limestone. Figure 6.11 shows a schematic model of the capture zones for two bores SPB1 and SPB2, which each pump at roughly 217 kL/day to recover water infiltrating below the ponds (Toze et al. 2002). There was presumably a watertable mound below the ponds that produced flow radially outward and towards the galleries, but the shape of the mound and flow directions could not be confirmed.

The chemical composition of groundwater was interpreted in relation to mixing with seawater within the aquifer and the impact of recharge from the adjacent wastewater ponds (Bekele et al. 2009).

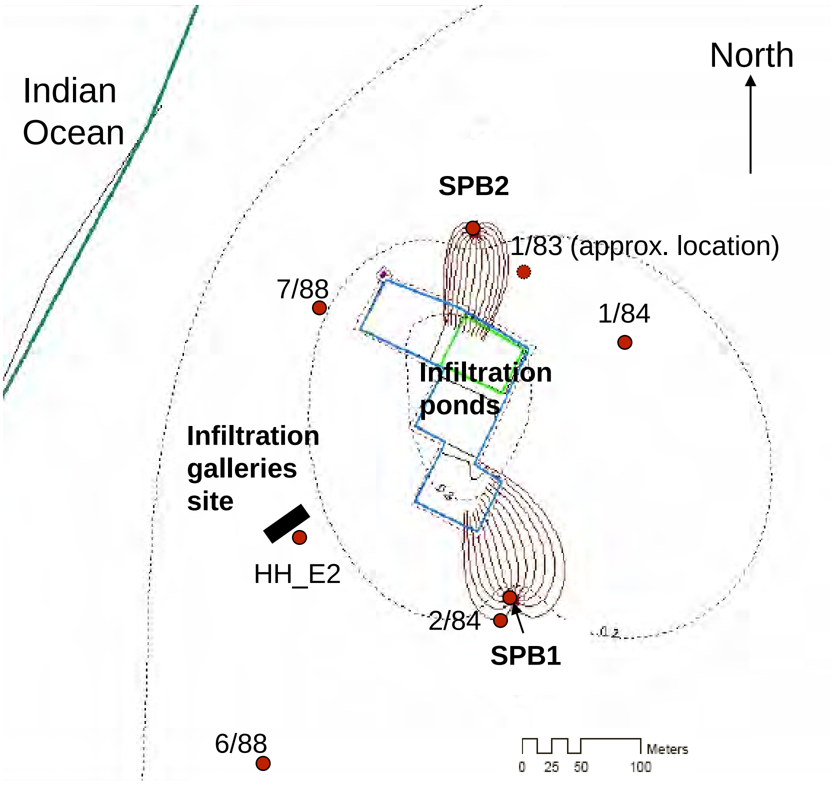

Figure 6.11 Map view of the infiltration galleries site at Halls Head relative to the ponds, monitoring bores and capture zones for recovery bores SPB1 and SPB2. Adapted from "Halls Head indirect treated wastewater reuse scheme," by Toze et al**. (2002), Client report to the Water Corporation. Copyright 2002. Adapted with permission.**

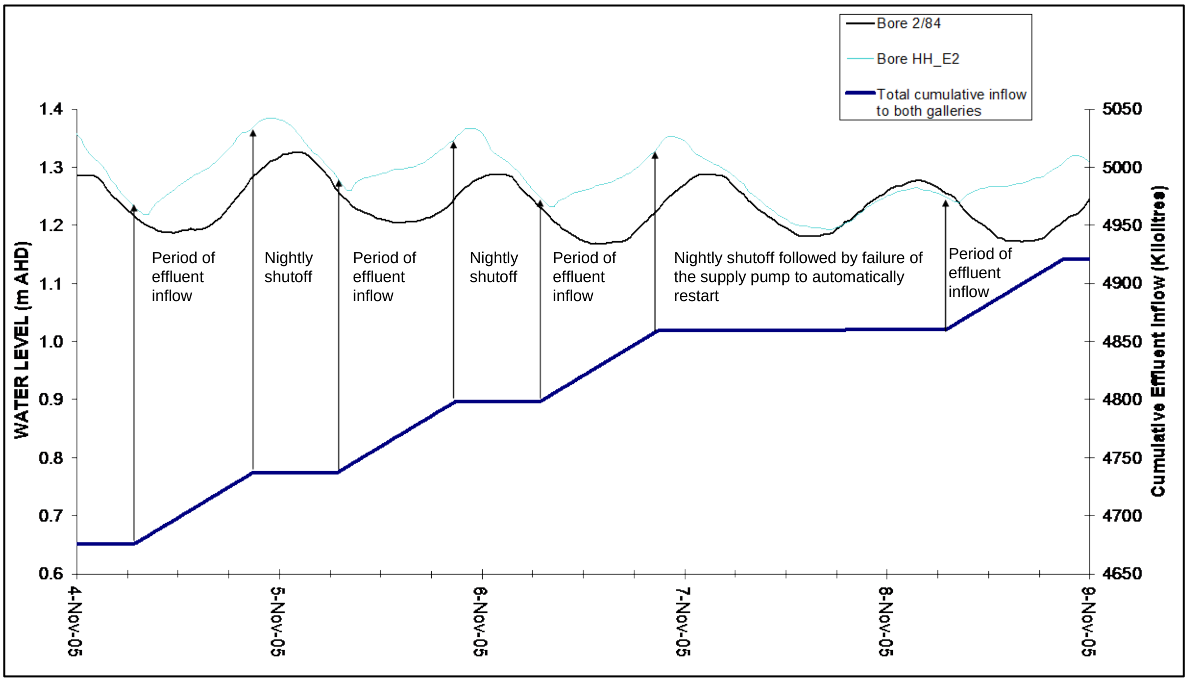

Figure 6.12 Groundwater response to MAR using infiltration galleries at the Halls Head WWTP (Bekele et al. 2009). The locations of bores HH_E2 and 2/84 are shown in Figure 6.11 relative to the sites of infiltration. Reprinted from "Design and operation of infiltration galleries and water quality guidelines, Chapter 1. In: Toze S, Bekele E (eds), Determining the requirements for managed aquifer recharge in Western Australia," by Bekele et al**. (2009), Water Foundation Report. Copyright 2009. Reprinted with permission.**

Preferential flow in the Tamala Limestone is a concern for MAR. As documented in a study of the hydrogeology of the Tamala Limestone (Smith et al. 2012), concerns have been raised about there being areas of cavern development and large-scale conduct flow in the Perth region, which could pose risks for MAR. There may be insufficient time for biodegradation to occur before the recycled water is abstracted for water supply or intercepted down-gradient for environmental benefits. However based on an extensive collection and analysis of considerable data for the Tamala Limestone, Smith et al. (2012) found that an appropriate conceptual model is one of dispersive flow through the formation pore system, rather than through large scale conduits, except where cavern development is known to be prevalent (e.g. Yanchep caves).

Bekele et al. (2014) conducted a MAR experiment and assessed aquifer travel times for treated wastewater in Tamala Limestone at the Floreat Infiltration Galleries site. Results from a three-dimensional solute transport model of the Tamala Limestone were compared with other tracer data. The study shows the limitation of relying on a single tracer to resolve residence times in the Tamala Limestone, and that heterogeneity can have a major influence on migration directions for recycled water plumes (Bekele et al. 2014).

6.4 Kwinana Wastewater Treatment Plant

The Kwinana WWTP is one of a number of treatment plants that dispose of treated wastewater (TWW) to land as discussed in Section 6.1. The treatment plant is located approximately 5.5 km from the coast of Cockburn Sound (Figure 6.1) and approximately 0.5 km to the west of the 'north eye' of The Spectacles wetland (Figure 6.13). The WWTP was commissioned in mid to late 1970 (Shams 2000) with a major upgrade in 2009 to the current configuration which utilises oxidation ditch followed by a clarifier for secondary treatment. The TWW disposed of under licence in infiltration basins located on-site (DER 2014). Treated wastewater volumes steadily increased since operation began until 2012 when the licenced disposal limited of 4.7 ML/d was reached (Figure 6.14). The resulting excess is disposed of to the ocean through the Sepia Depression Ocean Outfall Landline. With increased disposal volumes the infiltration area has undergone various upgrades until the current four basin configuration (Figure 6.13). The basins are operated in pairs to allow for maintenance with a maximum infiltration area of approximately 4,800 m^2 for each pair of basins.

As a result of the major upgrade to the plant in 2009 a marked improvement in the TWW quality occurred (Figure 6.15). The most significant change occurred with the nitrogen concentration with median total N concentration decreasing from 41 mg/L to 4.5 mg/L. Smaller changes occurred in total P concentration (median decreased from 6.2 to 4.2 mg/L), electrical conductivity (median decreased from 86 to 60 mS/m) and total suspended solid concentration (median decreased from 19 to 5.6 mg/L).

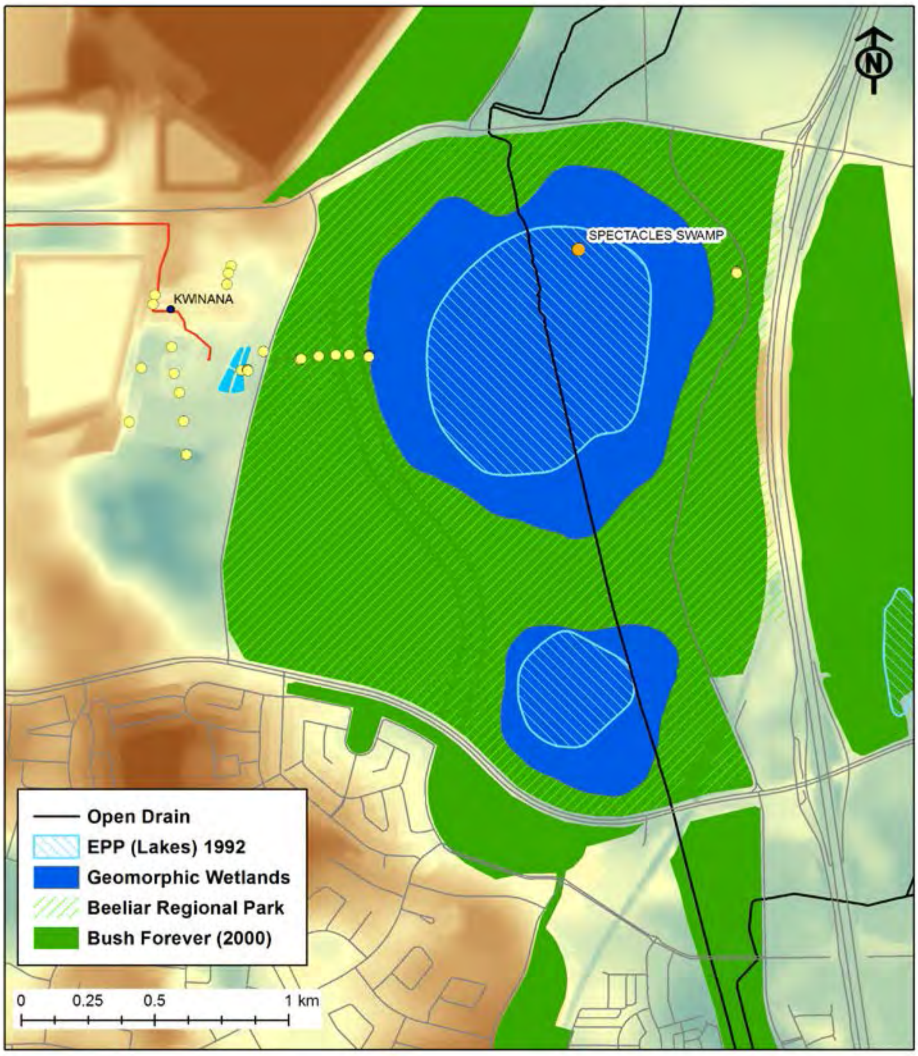

Figure 6.13 Location of the Kwinana WWTP and monitoring bores (yellow dots) relative to The Spectacles wetland

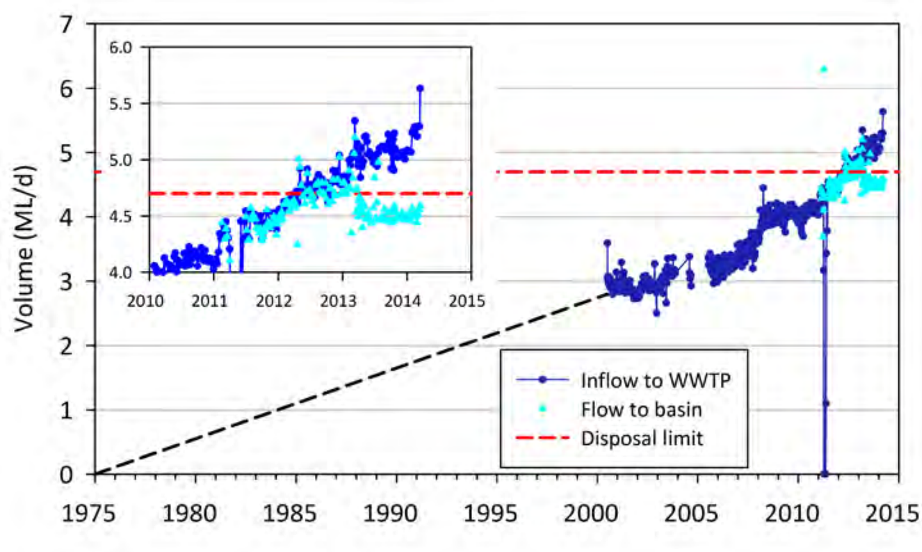

Figure 6.14 Weekly average inflow of wastewater to the Kwinana WWTP. It is assumed that inflows increased linearly from 1975 to 2000 (after Marillier et al., 2012). Data from provided by the Water Corporation

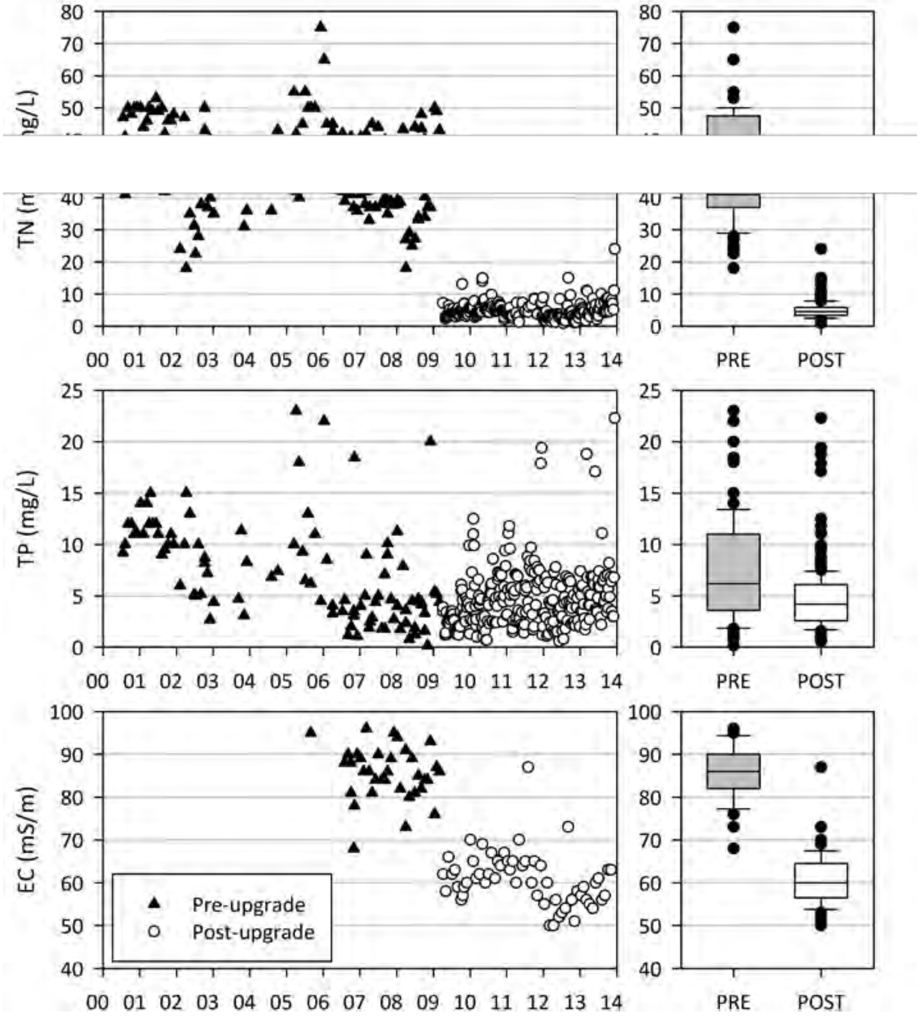

Figure 6.15 Treated wastewater quality showing the change in total nitrogen (TN) concentration, total phosphorus (TP) concentration and electrical conductivity (EC) following upgrade of the Kwinana WWTP

Due to the close proximity of the WWTP to The Spectacles wetland and as part of Water Corporation's license requirements, groundwater levels and quality have been monitored at up to 25 bore locations since the mid-1980s, though the duration of data collection varied from location to location. Data supplied by the Water Corporation and obtained from the Department of Water's Water Information Reporting database was used for the following analysis along with two sampling campaigns (April and November 2014) conducted as part of this project.

The infiltration of TWW at the Kwinana WWTP has resulted in the groundwater levels beneath the basins, and in the general vicinity, increasing. In April 2014 groundwater beneath the basins was approximately 4 m above the level at the edge of the groundwater levels at the edge of The Spectacles (SP1-1D, Figure 6.16). Groundwater levels have been increasing since the Water Corporation began records in the mid-1980s in response to increasing disposal of TWW (Figure 6.17). It appears that groundwater levels have stabilised since 2008 as the infiltration rate approached and was limited by the license conditions. The gradient towards The Spectacles created by the infiltration of the TWW suggests that wastewater may flow eastwards towards the wetland. This is also supported by the wetland water levels being lower than the groundwater at the western edge (Figure 6.17d). In an investigation of the hydrology and nutrient balance of the Spectacles, Shams (2000) concluded that groundwater was flowing eastwards from the infiltration ponds to the wetland and that nitrogen migrates towards the wetland. Subsequent to this study and following the installation of more bores between in the infiltration ponds and wetland, in a report to the Water Corporation, Woodward-Clyde (2000) concluded that while easterly flow of nitrogen from the WWTP occurs, a complex groundwater mixing zone exists between the infiltration ponds and the wetland. They identified three water types, (i) shallow low salinity, low nutrient groundwater at the western margin of the wetland, (ii) deeper high salinity groundwater beneath the lake flowing westwards and (iii) low salinity, high nutrient groundwater influenced by the infiltration at the WWTP.

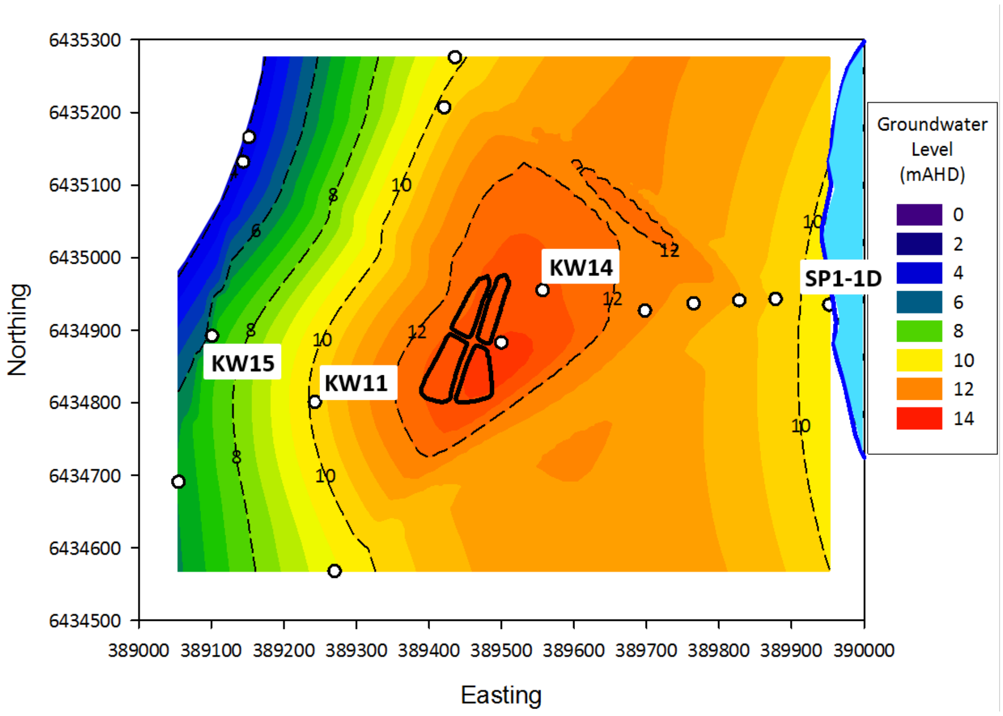

Figure 6.16 Extent of groundwater mound beneath the Kwinana WWTP infiltration basins in April 2014

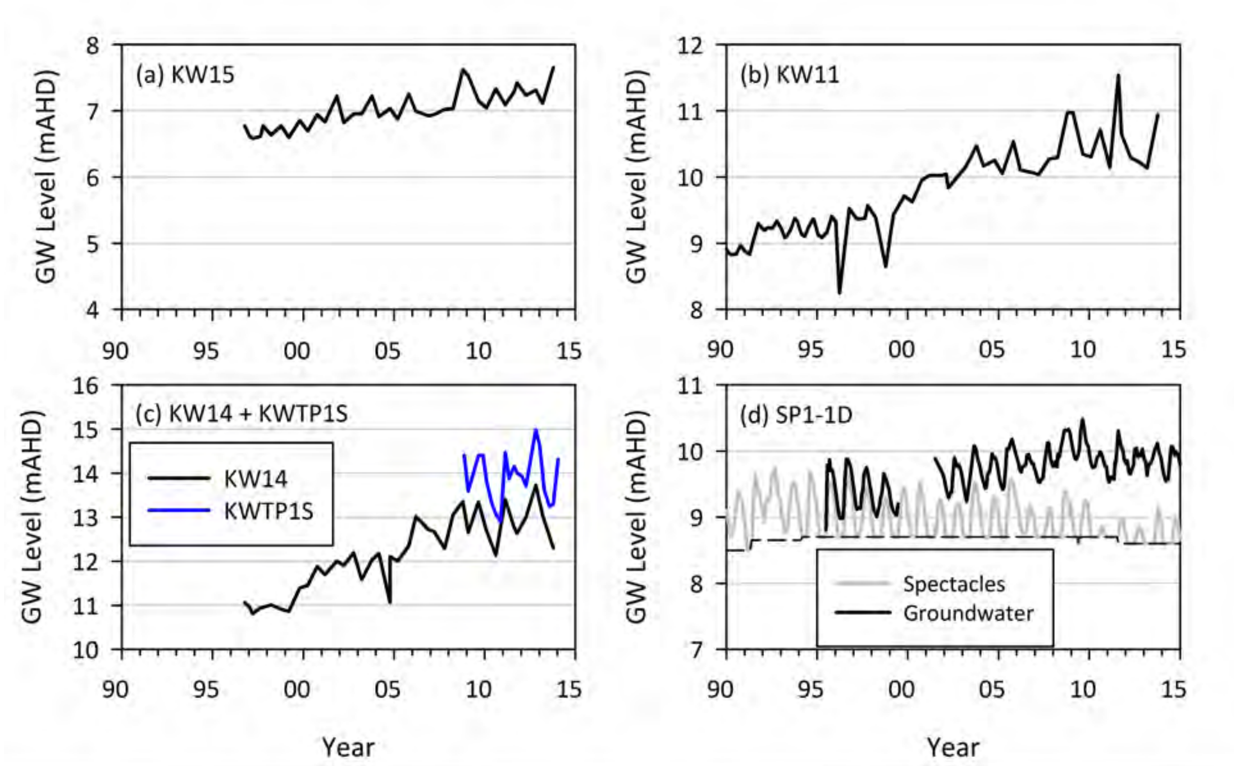

Figure 6.17 Temporal changes in groundwater level in response to increasing treated wastewater disposal at the Kwinana WWTP. These four bores form an east-west transect across the infiltration basins, the location of which is shown in Figure 6.16. The water level in the 'north eye' of The Spectacles also shown in (d) with the dashed line representing the ground level at the lake observation point (Figure 6.13)

Subsequent to the Woodward-Clyde study three additional shallow bores were installed between the infiltration basins and The Spectacles. While groundwater levels near the infiltration basin (KWTP1-S) always remain higher than close to the wetland bore SP1-1D, the slope of water table changes along this transect (Figure). The slope of the water table decreases between KWTP1-S and KWTP4 as would be expected as a result of mound development under the infiltration basins. However, the slope then increases between KWTP4 and SP1-1D. This may indicate that the connection between the infiltration basin and the wetland is not strong. This may be due to a rainfall recharge mound present in the sand dunes surrounding the wetland or the influence of the regional groundwater flow from east to west.

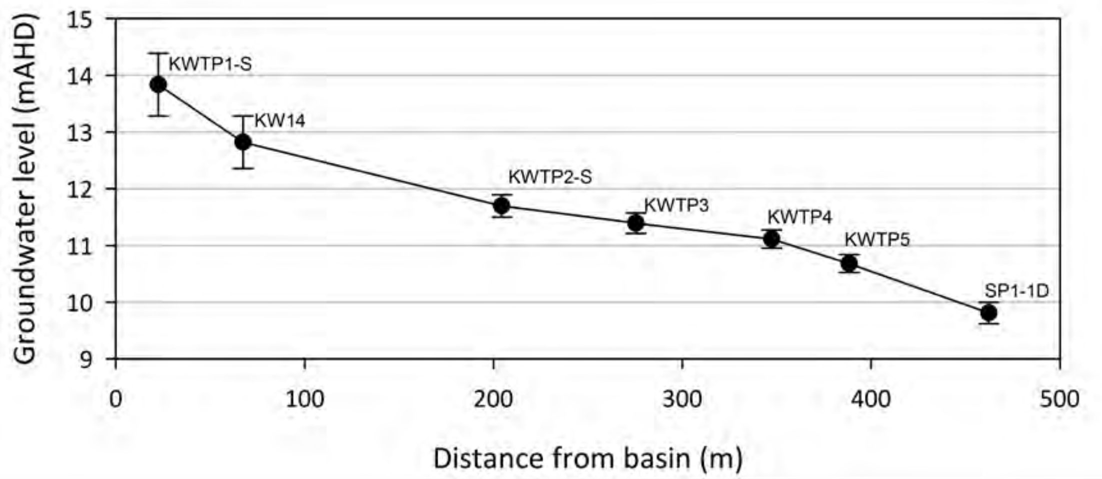

Figure 6.18 Average groundwater level (2010-2014) between the infiltration basins and the Spectacles (north) wetland. Error bars represent one standard deviation.

Results from the recent (April 2014) sampling campaign which also included bores on the eastern side of The Spectacles are shown in Figure 6.18, Figure 6.19 and Figure 6.21. The salinity, as indicated by the electrical conductivity, varied widely along a transect running through the wetland and infiltration basins (Figure 6.18a). To the east of the wetland groundwater is fresh relative to groundwater immediately to the west and comparable to the salinity of the TWW. A pocket of low salinity was also observed between the infiltration basins and the wetland similar to that observed by Woodward-Clyde (2000). Since the TWW has low salinity relative to the deep groundwater adjacent to the wetland, evaporation concentration within the wetland discharge to the groundwater may be occurring. The stable isotope data support this with the up-gradient groundwater showing similar isotopic composition to rainwater (Figure 6.19) and down-gradient groundwater showing highly enriched signature. This pattern is typical of flow-through lakes and wetlands described on the Swan Coastal Plain, including Thomson Lake to the north of The Spectacles (Turner and Townley 2006). Up-gradient groundwater discharges into the wetland where evaporation takes place followed by recharge on the down-gradient side resulting in groundwater with an evaporation signature, especially in the lower Superficial Aquifer due to the contrasting density with rainfall recharge. The stable isotope data and salinity data collected by Shams (2000) also support this conclusion with isotope composition of the groundwater immediately up-gradient and down-gradient showing similar to recent measurements (Figure 6.19). Therefore the groundwater chemistry of the three lower bores adjacent to the Spectacles relates to processes occurring within the wetland.

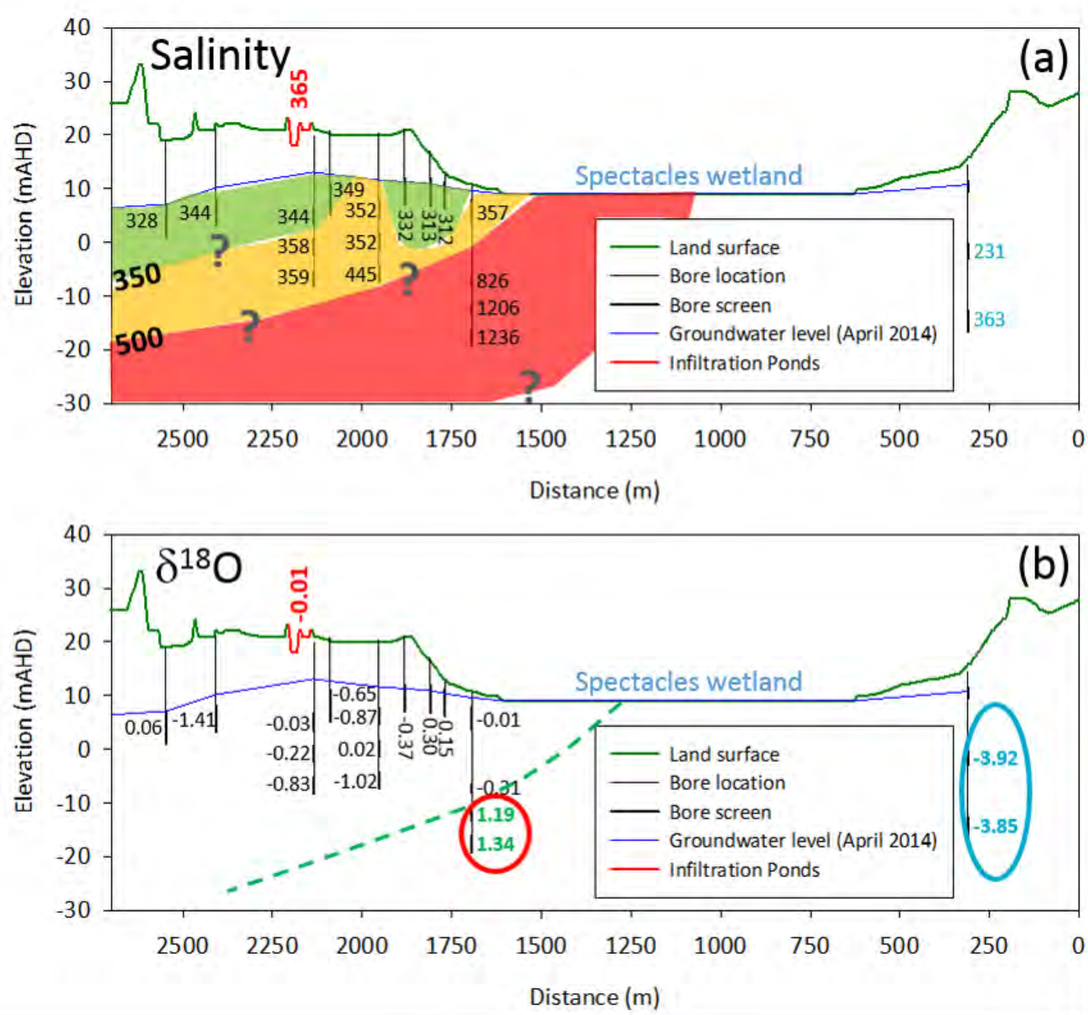

Figure 6.19 Cross-section showing the distribution of groundwater (a) salinity (mg/L) and (b) oxygen-18 ($\delta^{18}O$ in ‰) concentration along a transect across The Spectacles, through the infiltration basins and to the western edge of the WWTP for April 2014. The position of the infiltration basins are shown in red along with the basin EC and $\delta^{18}O$.

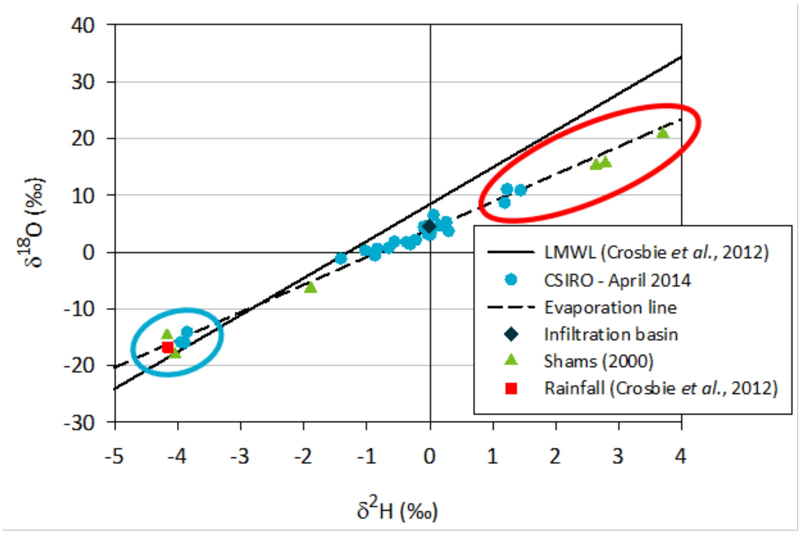

Figure 6.20 Stable isotope data from groundwater (this study and Shams 2000), treated wastewater (infiltration basin) and amount-weighted average rainwater for Perth (Crosbie et al. 2012) showing the local evaporation line (equation shown) in comparison to the local meteoric water line (LMWL, Crosbie et al. 2012). Blue and red ovals indicate the up-gradient and down-gradient bores indicated in Figure 6.19

The salinity and stable isotope measurements were not able to differentiate the shallow groundwater at SP1-1D from the groundwater impacted by wastewater. Therefore other tracers were investigated. Potassium concentrations have been used previously as a tracer for wastewater in groundwater (Wolf et al., 2004; Rueedi et al., 2009; Bekele et al., 2011a). Due to the strong salinity differences in groundwater in the vicinity of the WWTP the potassium concentrations were normalised by dividing by the chloride concentration and the spatial distribution along the east-west transect plotted in Figure 6.21. The ratio of K:Cl in the wastewater was approximately 10 times greater than the background groundwater (SP1-1A, SP1-1B, SP1-1C) and 5 to 10 times greater than in the Spectacles Swamp (0.022 – 0.040, WIR database). This suggests that the K:Cl ratio may also be a useful for the presence of wastewater. Close to the point of infiltration (e.g. KWTP1 and KW14) the groundwater K:Cl ratio is similar to wastewater and decreases with distance from the infiltration basins. As soil (Bassendean Sand) surrounding the infiltration basins is deficient in potassium (Coroneos et al., 1996) the infiltrated wastewater is the only likely source. Based on the K:Cl ratios shown in Figure 6.21 and the variability in background values, wastewater is expected to contribute 23 to 33% to the shallow groundwater close to the Spectacles north (SP1-1D).

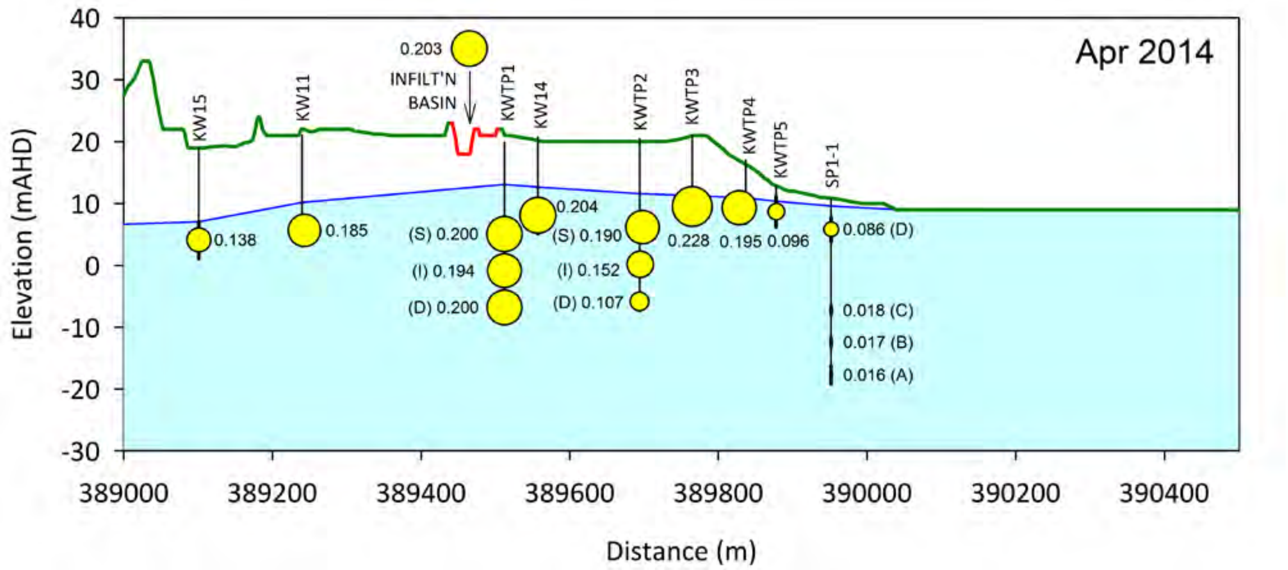

Figure 6.21 Potassium to chloride molar ratio in groundwater and wastewater collected in April 2014 near the Kwinana WWTP, values given next to bubbles. Surface elevation, infiltration basin (shown in red / arrow) and groundwater table are shown for reference

The high total N concentrations in the TWW prior to the WWTP upgrade result in high concentrations in the groundwater in the vicinity of the infiltration basins (Figure 6.20). Improvements to the TWW quality have resulted in a decrease in the groundwater total N with reductions occurring at a faster rate closer to the infiltration basin. This is to be expected as newly infiltrated TWW displaces groundwater and mixing occurs. Total N in groundwater close to the infiltration basins now has concentrations lower than the ANZECC (2000) trigger value of 1.5 mg/L. Due to a gap in the data for the shallowest bore close to the wetland (SP1-1D) it is unclear whether the high ammonium is related to the TWW infiltration or natural processes within the wetland. However the spatial distribution of groundwater ammonium in April 2014 (Figure 6.21) suggests, along with the K:Cl ratio data, that natural processes contribute to the majority of the high ammonium concentrations at SP1-1D.

The deeper groundwater adjacent to the wetland also show high ammonium concentrations (Figure 6.23) as observed by Shams (2000) and Woodward-Clyde (2000). As established above the deeper groundwater at this location has an evaporation signature associated with the through-flow wetland, thus the high ammonium is most likely derived from the wetland and not wastewater. Since wetlands are rich in natural organic matter (NOM) groundwater beneath is often low in oxygen (anoxic) due to the consumption of oxygen during microbial degradation of the NOM. Organic nitrogen present in the NOM is mineralised to ammonium which is transformed into nitrate in the presence of oxygen. However under anoxic conditions this does not occur. Anoxic conditions also favour the removal of nitrate through denitrification, thus resulting the in high ammonium concentrations observed.

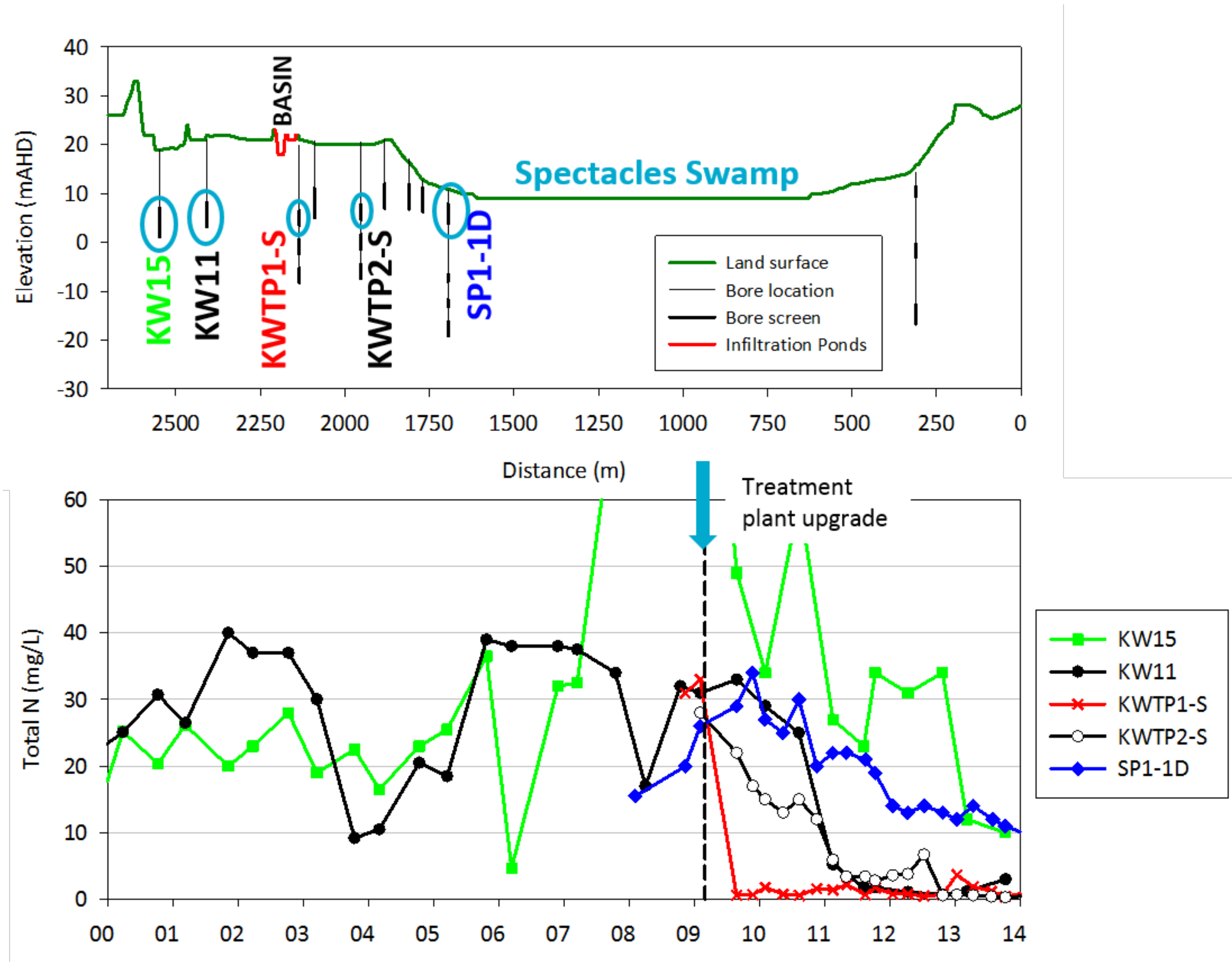

Figure 6.22 Temporal variation in groundwater total nitrogen concentration in the vicinity of the Kwinana WWTP. The upper panel shows the location of groundwater bore shown in the lower panel.

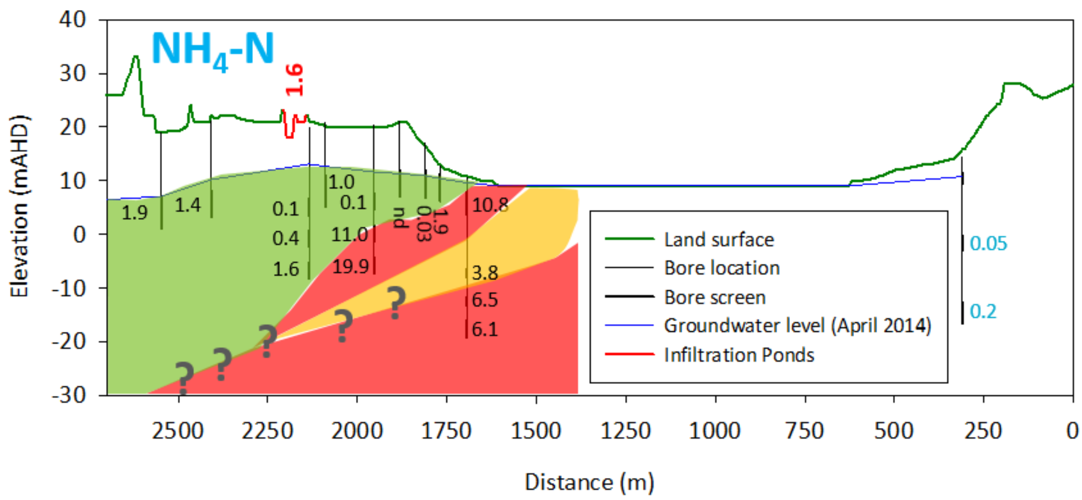

Figure 6.23 Spatial distribution of ammonium-N (NH_4-N, mg/L) in groundwater in the vicinity of the Kwinana WWTP and Spectacles wetland (April 2014). The position of the infiltration basins are shown in red along with the basin NH_4-N concentration.

Due to the small changes in TWW Total P concentration (Figure 6.15) following the WWTP upgrade there is no discernible change in the groundwater total P concentration close to the infiltration basins (Figure 6.24). Groundwater total P concentrations typically remain within the interquartile range of the TWW (2.6 to 6.1 mg/L) suggesting that the phosphors sorption capacity of the aquifer close to the basins has been exceeded. Considering the long-term application of phosphorus and the low P sorption capacity of the soils (Richie and Weaver, 1993; He et al., 1998) this is highly likely close to the basin. Shallow groundwater close to The Spectacles wetland (SP1-1D) shows an increasing trend in total P concentration between 2009 and 2010 (Figure 6.24). Given that there was little change in the TWW concentration or groundwater close to the infiltration basins then this may potentially be due to the breakthrough of phosphorus as the sorption capacity was exceeded. However as shown above the proportion of wastewater is at most one third (equivalent to ~1.4 mg/L TP), therefore other sources are likely to be contributing to the total P at this location. Total P concentrations in the three shallow bores to the west of SP1-1D generally show lower concentrations (Figure 6.25) supporting the notion that migration of wastewater total P is only a proportion of the observed concentrations at SP1-1D.

There is some evidence that there is elevated total P concentrations in the wetland itself with the limited number of measurements (26) ranging from 0.06 to 1.6 mg/L. Most total P readings from the wetland exceed the ANZECC (2000) trigger value for wetlands of 0.06 mg/L and could contribute to the total P observed at SP1-1D. The drying of the lake bed during summer (Figure 6.17) may also contribute to the increase P available for leaching through the microbial degradation of organic matter and subsequent release upon cell lysis as shown for North Lake (Qui and McComb, 1994; Qui and McComb, 1995). Another potential source is the associated with rainfall recharge on the dune system surrounding the wetland. Qui et al. (2004) found that P released from leaf litter from the upland areas around Thompson Lake had the potential to generate leachate high in P (2 to 5 mg/L).

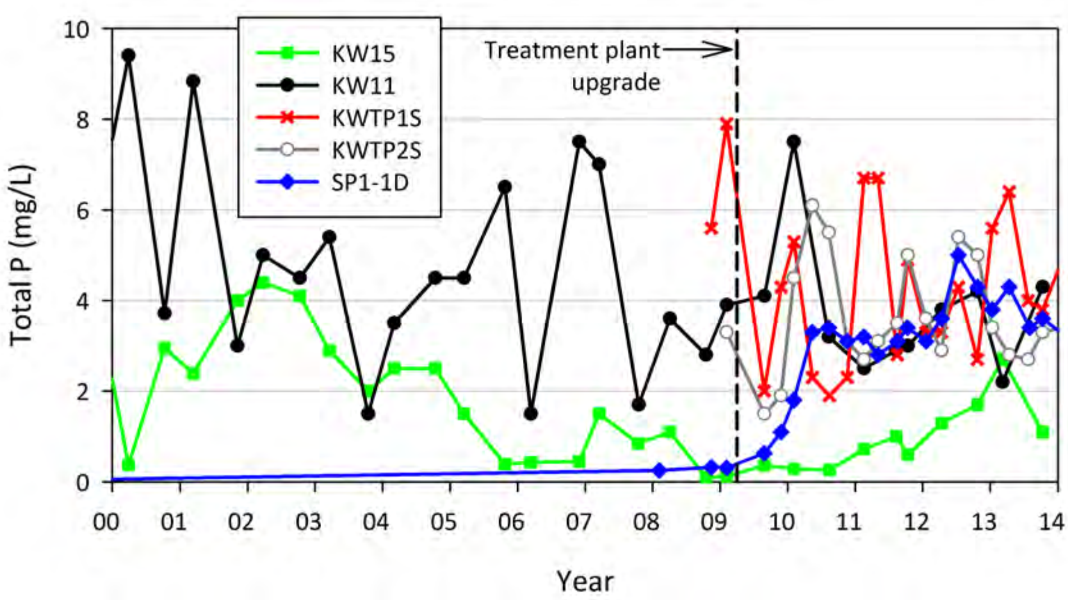

Figure 6.24 Temporal variation in groundwater total phosphorus concentration in the vicinity of the Kwinana WWTP. See Figure 6.22 for location of bores.

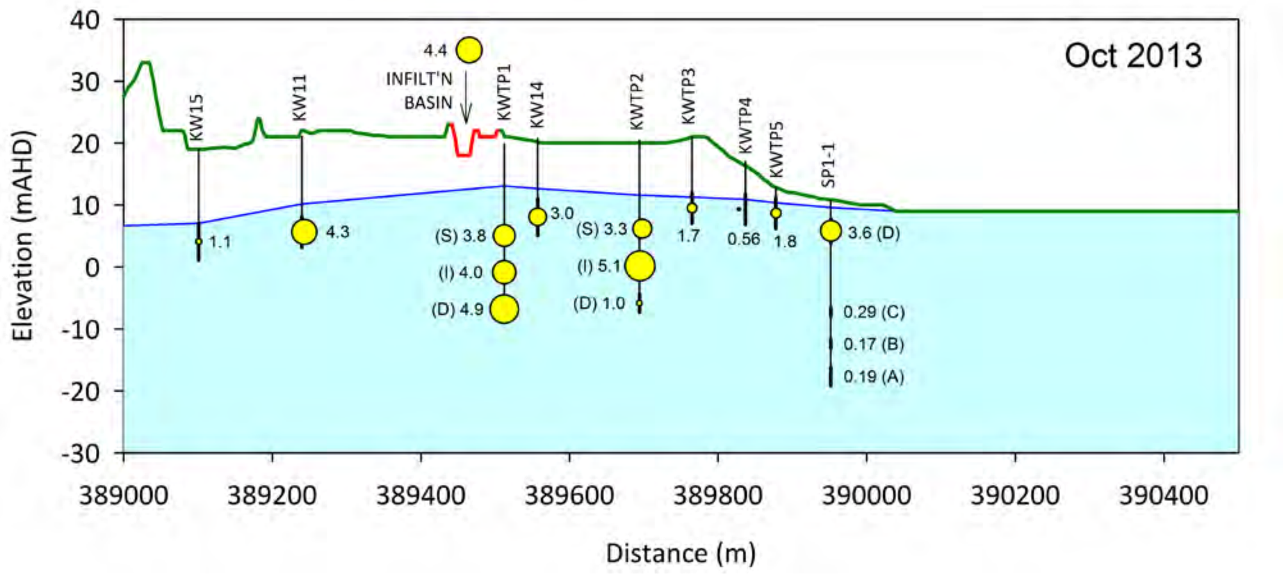

Figure 6.25 Spatial distribution of total phosphorus (mg/L) in groundwater in the vicinity of the Kwinana WWTP and The Spectacles wetland (October 2013, values given next to bubbles). The position of the infiltration basins are shown in red and indicated by the arrow.

The infiltration of TWW at the Kwinana WWTP was modelled as part of the current study with the results discussed in Section 10.4.5. The model indicated that while TWW did migrate towards The Spectacles, advective transport suggested that the TWW travelled deeper within the aquifer. While shallow flow paths are still possible especially as the density difference related to the aquifer salinity was not modelled they are less likely. Additional modelling to determine the source of shallow groundwater close to the wetland indicated that water originated from within the dunal system on the western edge of the wetland.

In summary, although groundwater gradients indicate that TWW may flow towards The Spectacles wetland from the infiltration basins, the groundwater chemistry indicates that:

- The Spectacles is a flow-through wetland and recharge of isotopically enriched, evaporated water from the wetland carries with it a natural ammonium signature
- a shallow zone of shallow groundwater of low salinity, low nutrient lies between the infiltration basins and the wetland
- High nutrient concentrations in the shallow groundwater immediately adjacent to the wetland (SP1-1D) may in part be derived from the TWW as indicated by the potassium tracer data, however the wetland and surrounding vegetated dune system also plays an important, and likely larger, role in the generation of the nutrients observed at this location.

Therefore it is unlikely that much TWW is flowing 500 m up-gradient into The Spectacles wetland.

6.5 Conclusions

The feasibility of MAR has been investigated at a regional scale across the Swan Coastal Plain and in local field trials. A review of various MAR studies conducted in the Perth-Peel region over the last three decades demonstrates increasing confidence in MAR for non-potable and potable purposes with some progression from primarily desktop modelling studies to field investigations and trials.

Over the last forty years, there has been a multi-pronged approach to investigating the requirements for MAR using infiltration in different units of the Superficial Aquifer:

- desktop aquifer suitability investigations that rely on hydraulic properties to either model or interpret the potential for excessive mounding of the watertable (and breaching at the land surface);

- groundwater flow modelling within increasing levels of sophistication and greater refinement of hydraulic properties to simulate the groundwater response to MAR, and the benefits to groundwater abstraction and environmental benefits from increased water levels in the Superficial Aquifer;
- analysis of the 'forensics' of forty years of wastewater infiltration at the Kwinana WWTP and the impacts on surrounding areas of wastewater mixed with ambient groundwater after a period of residence within the aquifer;
- review of the long-term hydraulic performance and maintenance requirements from discharge volumes of wastewater disposed of at coastal infiltration sites (predominantly in Tamala Limestone);
- field trials conducted in different units of the Superficial Aquifer to assess the water quality changes resulting from a period of residence in the aquifer and to assess clogging potential relative to infiltration rates and source water quality;
- sediment-filled column experiments testing the infiltration capacity and water quality changes resulting from trialling different combinations of aquifer material and source water treatments.

These varied and insightful methods have aided in identifying the criteria for successful infiltration of secondary treated wastewater for non-potable purposes from a technical perspective, barring other constraints (e.g. land use, economic, regulatory).

Nevertheless, there are relatively few well-documented, long-term field studies to provide guidance and confidence in infiltration gallery and basin techniques to recharge the Superficial Aquifer with wastewater at high rates of infiltration. As higher levels of wastewater treatment become more economically feasible and routine at WWTPs, there may be fewer obstacles to implementing MAR related to fewer concerns about clogging, nitrogen pollution of groundwater and long-term maintenance costs. Through greater experience and knowledge gained about the operation of infiltration galleries, there may be opportunity to explore multiple uses of the land overlying subsurface infiltration (thus making the best use of available urban land), and potential to reduce ocean outfall volumes to the marine environment, which is currently used to dispose of the majority of Perth's treated wastewater.

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7 Valuation of wetlands in the Perth area

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Key findings

- The hedonic property price analysis confirms that Thomsons Lake and Spectacles Lakes add value to surrounding property prices.
- Proximity to geomorphic wetlands, which are mostly dried out or drying out has a negative effect on sales price. Proximity to manmade wetlands on the other hand add value to surrounding properties.
- Only houses within 6km of Thomsons Lake derive benefit of being in close proximity to the lake in the form of higher property sales price. For the Spectacles Lakes, the distance is 10km.
- The estimated amenity value (e.g. aesthetic and recreation) of Thomsons Lakes and Spectacles Lakes that is capitalised in property prices is around $300 million.

7.1 Introduction

One of the objectives of this research project is to examine the additional benefits from recycling water via managed aquifer recharge (MAR). We *expect* that MAR will raise regional groundwater levels around throughflow lakes, and by doing so recharge wetlands that have dried or are in the process of drying out due to climate change and pumping. Groundwater in the study area supports a Ramsar-listed wetlands (Thomsons Lake), and two chains of wetlands that run north to south of the study area.

The manner in which MAR can recharge wetlands is as follows. The infiltration of MAR water will create a hydraulic barrier to sea water ingress while maintaining groundwater levels inland of the barrier that will make more groundwater available for use and maintain levels in throughflow wetlands. Groundwater modelling has indicated that infiltrating between 2.6 to 6.5 ML/day may influence levels up to 5 kilometres inland of the injection site (McFarlane et al., 2009) and the impact could be established within a year of adding the water (GHD, 2011).

Maintaining wetlands has significant ecological benefits. But wetlands provide social benefits as well. People who live near wetlands can enjoy the view of wetlands, as well as recreate around wetland areas. Studies have shown that wetlands can add value to nearby houses because of wetland view and proximity to wetlands (see e.g. Boyer and Polasky, 2004).

Since the cost benefit analysis framework suggest that all costs and benefits of a project should account for all the effects on the community and the economy, and that all the effects should be converted into dollar terms (Department of Prime Minister and Cabinet, 2014), the objective of this chapter is to present findings from the monetisation of wetland values in the study area.

7.2 Types of values

In the project study area, there are a number of geomorphic wetlands and man-made wetlands that may add a premium to surround house values. The objective of this wetland valuation analysis is to determine the amenity value of wetlands that are capitalized in property prices.

There are a number of ways wetland benefits can be valued. The most commonly used framework for monetising environmental assets is the Total Economic Value (TEV) framework. Pearce and Turner (1990) defined TEV as the economic benefits (implicitly derived by people) arising from the use (direct use, indirect use, and option values) and non-use value (quasi-option, bequest, and existence values) of natural resources. Use values are benefits derived from the actual use of the environment in which the use could be direct or indirect. Non-use values, on the other hand, are values that people assign to the environmental good regardless of whether they will use it or not.

The amenity value of wetlands is a type of direct use value that is non-extractive i.e. no reduction on the quantity or quality of the natural resource. Houses that are closer to wetlands (for easy access) or have a view of wetlands have been found to have a higher sales price than houses that are further away or do not have a view of wetlands (Boyer and Polasky, 2004; Brander et al., 2006; Brouwer et al., 1999).

7.3 Hedonic valuation of water bodies in Australia

Previous valuation studies of wetlands (Tapsuwan et al., 2009; Tapsuwan et al., 2012) have shown that house prices in Australia show a premium for being in close proximity to lakes and wetlands. For example, Tapsuwan et al. (2012) estimated that for an average property in the South Australian portion of the Murray-Darling Basin that is approximately 1 km away from the River Murray, moving half a kilometre closer will increase the property price by $245,000, holding every other variable constant at the mean.

A more closely related study is the valuation of wetlands in the northern suburbs of Perth Western Australia. Tapsuwan et al. (2009) estimated that

> "For a property that is 943 m away from the nearest wetland, which is the average distance to the wetland in this study, reducing the wetland distance by 1 m will increase the property price by AU$42.40. Similarly, the existence of an additional wetland within 1.5 km of the property will increase the sales price by AU$6976. For a randomly selected wetland, assuming a 20 ha isolated circular wetland surrounded by uniform density housing, the total sales premium to surrounding properties was estimated to be around AU$140 million (95% confidence interval of AU$40 million to AU$230 million)."

These two Australian studies are indicative of the fact that there may be wetland values that are capitalized in property prices in the Cockburn, Kwinana and Rockingham areas.

7.4 Methodology

Using the theoretical framework proposed by Rosen (1974) and later expanded by Freeman (1974), a property can be considered as a multi-attribute good that consists of structural attributes (e.g. bedrooms, bathrooms, land area), neighbourhood attributes (e.g. distance to town centre, shopping centres and transport hubs) and environmental attributes (e.g. distance to wetlands, nature conservation areas and neighbourhood parks). The price that buyers pay is in a sense reflective of how much they value each attribute of the house. This theory was then developed into a method called the Hedonic Property Price (HPP) approach.

Using this theoretical framework, we can estimate the values of environmental amenities that are capitalised in property prices. The general specification for the hedonic property price regression is

$$P_i = f(S_i, N_i, E_i, T_i) \qquad (1)$$

where

P_i is the purchase price of a property *i*

S_i is a vector of the structural attributes of property *i*

N_i is the neighbourhood attributes of property *i*

E_i is a vector of environmental attributes of property *i*

T_i is a dummy variable to express the sales month and year of property *i*

The marginal implicit price (MIP), an indicator of the willingness to pay of a buyer for an additional unit of a house characteristic, is estimated from taking the partial derivative of the HPP function with respect to the house characteristic.

7.4.1 STUDY SITE

The study area is relatively urbanised - with houses, commercial zones, and industrial zones built throughout the study area. However, there are some remaining geomorphic wetlands and nature conservations areas forming two belts running north to south on the west side and east side of the study site. Most of the geomorphic wetlands in the study area have dried out or are close to being permanently dry. The ones that have dried out are replaced by vegetation and no longer look like a wetland. In between the residential housing areas are manmade wetlands that act as substitutes to natural geomorphic wetlands that have dried out or are drying out. Unfortunately there is no information on how these manmade wetlands are maintained i.e. whether they are filled with groundwater or stormwater.

There are a number of significant wetlands of the Swan Coastal Plain within the project study area. In this analysis, the main wetlands of particular interest are Thomsons Lake and the Spectacles Lakes as they are both listed in the *Directory of Important Wetlands in Australia* (Environment Australia, 2001). Thomsons Lake in particular is a Ramsar wetland and is listed under the *List of Wetlands of International Importance under the Convention on Wetlands* (Conservation Commission of Western Australia, 2005). Additionally, Thomsons Lake is one of the few remaining refuge sites in Western Australia for a threatened Australian water species – the Australasian Bittern (Conservation Commission of Western Australia, 2005). Thomsons Lake is still filled with water but the amount of water varies dramatically between the wet and dry season. The Spectacles Lakes is generally filled with water all year round and has leafy vegetation growing in and around the wetlands creating a pleasant view around the area. Thomsons Lake is approximately 22km in a straight line distance away from Perth central business district (CBD). The Spectacles Lakes is approximately 30km away. Figure 7.1 shows a map of the study site.

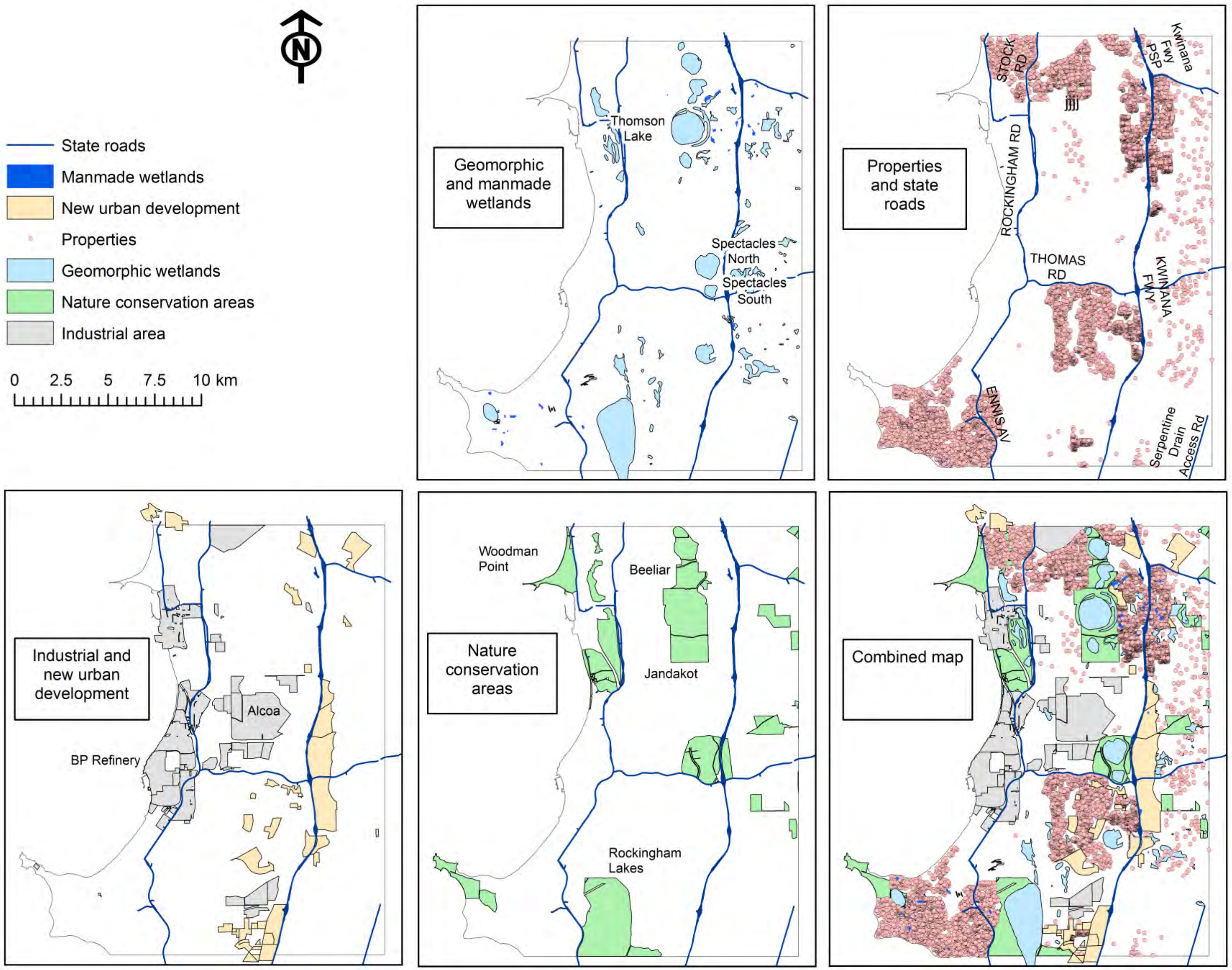

Figure 7.1 Landuses in the study area

7.4.2 DATA COLLECTION

Three main types of data were collected for this analysis: property sales data, geo-spatial data and demographic data. The geo-spatial data included centroids of properties and points of interests (e.g. shopping centres, hospitals, train stations etc) and polygons of local government areas, wetlands, nature conservation areas, industrial areas, and new urban development areas. A list of model variables and descriptive statistics is provided in Table 7.1.

Property sales data

The study site covers the residential market of three local government areas situated south of Perth city including City of Cockburn, City of Kwinana, and City Rockingham. Property sales data were acquired from Landgate Western Australia (formerly the Valuers General Office of Western Australia). The data set contained information about property sales from year 2009 until year 2014. The data set also contained geo-spatial reference (i.e. centroids) of each property.

Geospatial data

Values associated with environmental and neighbourhood attributes, such as distance to the nearest wetland or distance to the nearest airport were calculated in ArcGIS (ESRI, 2010). Geospatial data on environmental variables were acquired from Geoscience Australia, the Department of Water, and the Valuers General Office. Environmental amenities that were considered to add value to property prices in this study were wetlands, conservation parks, local parks and the beach. There were four types of wetlands in this analysis: geomorphic wetlands, man-made wetlands, Ramsar wetland (Thomsons Lake) and perennially wet wetlands (Spectacles Lakes). The HPP method measures the impact of environmental amenities on sale prices using proximity (e.g. distance in metres) from the centroid of the property to the centriod or edge of the amenity, the size (or area) of the amenity and the quality of the amenity (i.e. using Normalised Difference Vegetation Index (NDVI) to measure the greenness of parks). The variable used to represent the quality of the environmental amenity varies depending on the type of amenity and the data that was available.

Geospatial data on neighbourhood variables were acquired from Landgate. Neighbourhood amenities that were considered attributes contributing to sales price include shopping centres, transport hubs (i.e. airports, bus station, train station, freeway entrance), hospitals, schools (i.e. primary, secondary, high school, TAFE, university), golf courses, cinemas, industrial areas, police stations, cemeteries, and Perth CBD. In this analysis, the impact of neighbourhood amenities on sales price mostly used the proximity measure.

Demographic data

Two demographic data sets were included to capture the ‘quality’ of the neighbourhood. These include crime rates by suburb (Western Australia Police, 2014) and income by census area (Australian Burea of Statistics, 2013). Income by census area was matched with suburb data to generate an income by suburb data list.

Table 7.1 List of significant model variables and descriptive statistics

VARIABLES	MEAN	STD. DEV.	MIN	MAX	PER CENT
Sale price ($)	375,245	156,190	6,935	1,800,000	
No of bedrooms	3.19	0.66	1.00	6.00	
No of bathrooms	1.23	0.43	1.00	3.00	
House age (years)	36.90	11.51	1.00	92.00	
Land area (m^2)	777	272	209	4,725	
Distance to the nearest shopping centre (m)	2,120	1,136	34	10,304	
Distance to the nearest park (m)	1,913	1,102	14	9,691	
Distance to the nearest beach access point (m)	5,236	2,392	152	13,513	
Distance to nearest manmade wetland (m)	3,474	2,273	11	9,164	
Distance to nearest geomorphic wetland (m)	2,501	1,220	12	6,231	
Area of the nearest geomorphic wetland (ha)	24.20	31.31	0.15	103.97	
% of houses within 6km of Thomsons Lake					6
% of houses within 10km of Spectacles Lakes					21
Distance to the nearest nature conservation park (m)	3,175	1,353	33	6,412	
Distance to the nearest industrial area (m)	3,258	1,405	9	7,376	
% of house was sold in 2010					15
% of house was sold in 2011					18
% of house was sold in 2012					21
% of house was sold in 2013					25
% of house was sold in 2014					1
% of house is in Kwinana LGA					25
% of house is in Rockingham LGA					54

7.4.3 DATA ANALYSIS

The key objective of the HPP model is to estimate the amenity value of wetlands in the study area, using proximity as a proxy of amenity value, when controlling for all other affects on property price.

The property sales data set was cross-checked for improbable or missing values using market-average values and image checks via Google Maps. An example of an improbable value is a house that was sold for less than $1/m^2. This usually indicates a transaction that was not sold at the market price. Another example of an improbably value is a house that has no bedrooms or bathrooms despite appearing to be an average home on Google Maps. After removing observations with missing values and outliers, we obtained a dataset with 3,621 observations.

The statistical software package Stata13 (StataCorp, 2013) was used to analyse the data. A series of Box-Cox transformations suggested that a hedonic model with a natural log transformed dependent variable was an optimal functional form for the data. Regarding functional form of explanatory variables, the hedonic pricing model is often estimated in semi-log, where the natural log of price is the dependent variable with linear independent variables or in double log form where we take the log of both the dependent and independent variables (Tapsuwan et al., 2009).

7.5 Results

The final regression parameter estimates, standard errors and significance levels are shown in Table 7.2. All the variables were significant at the p<0.01 level, except for the dummy variable for houses sold in 2012. Other statistically insignificant variables were removed from the model for parsimony, such as distance to airport.

There are five types of wetland variables in this model: 1) distance to the nearest geomorphic wetland, 2) distance to the nearest man-made wetland, 3) number of houses within a 6km radius of Thomson Lake, 4) number of houses within a 10km radius of the Spectacles Lakes (both north and south grouped together and considered as a single wetland), and 5) size of the nearest geomorphic wetland.

Specific attention was given to Thomsons Lake and The Spectacles lakes because Thomsons Lake is a RAMSAR wetland, and Spectacles Lakes are near to the Kwinana wastewater treatment plant. The traditional linear specification, as well as other specifications including log, inverse and quadratic, did not produce significant parameter values for the variables capturing the marginal benefit of proximity to these two wetlands. Consequently, the distance of 6 and 10km radius for Thomsons Lake and Spectacles Lakes, respectively, were used instead. These distances were a result of a systemic incremental search (in increments of every 0.5km starting from 0.5km to 25km) for a 'premium' distance, for each of these wetlands that would yield the best model fit (using the Ramsey RESET F-test as the statistical indicator).

Table 7.2 Parameter estimates from the Hedonic Property Price model

VARIABLE	COEFFICIENT	(STD.ERR)
No of bedrooms	0.0315	(0.0079)***
No of bathrooms	0.1324	(0.0119)***
House age (years)	-0.1131	(0.0161)***
Land area (m^2)	0.3447	(0.0231)***
Distance to the nearest shopping centre (m)	1.06E-04	(6.60E-06)***
Distance to the nearest park (m)	-1.07E-04	(6.52E-06)***
Distance to the nearest beach access point (m)	-5.23E-05	(4.00E-06)***
Distance to nearest manmade wetland (m)	-5.83E-05	(5.02E-06)***
Distance to nearest geomorphic wetland (m)	2.40E-05	(5.94E-06)***
Area of the nearest geomorphic wetland (ha)	0.0013	(0.0007)*
Area of the nearest geomorphic wetland squared (ha)	-2.02E-05	(0.0000)**
House is within 6 km of Thomsons Lake	0.0874	(0.0256)***

House is within 10 km of Spectacles Lakes	0.0453	(0.0173)**
Distance to the nearest nature conservation park (m)	3.73E-05	(5.53E-06)***
Distance to the nearest industrial area (m)	2.27E-05	(5.72E-06)***
Dummy variable=1 if house was sold in 2010	0.0475	(0.0140)***
Dummy variable=1 if house was sold in 2011	-0.0313	(0.0134)**
Dummy variable=1 if house was sold in 2012	0.0080	(0.0129)
Dummy variable=1 if house was sold in 2013	0.1191	(0.0124)***
Dummy variable=1 if house was sold in 2014	0.1936	(0.0552)***
Dummy variable=1 if house is in Kwinana LGA	-0.5039	(0.0277)***
Dummy variable=1 if house is in Rockingham LGA	-0.7020	(0.0312)***
Constant	11.2786	(0.1472)***
Significance levels ***p<0.001, **p<0.05, *p<0.01		
N=3,621; F(22, 3598)=213.24; Adj R-squared=0.5633; Root MSE =0.24844		

Table 7.3 presents the MIP for all the significant variables in the model. The MIP is the premium a buyer is willing to pay for an additional unit of the attribute, for example, to have an extra bedroom, or to move 1km closer to the beach. A negative parameter value for distance variables indicate that if the amenity is one kilometre closer to the house, the sales price goes up, and vice versa for a positive parameter value. Based on the MIP values presented in Table 7.3, the interpretation for some of the variables is as follows.

For an average condition house that cost $375,000 with 3 bedrooms and 1 bathroom on a 700 m^2 block:

- an additional bedroom would add around $11,800 to the sales price,
- an additional square meter of land would add around $19,800 to sales price,
- an additional year of house age would lower the value of the house price by around $11,700
- moving 1km closer to the beach would add around $20,000 to sales price
- moving 1km to a geomorphic wetland would lower the value of the house price by around $9,000
- if the house was sold in 2010, as compared the baseline which is 2009, the house would be worth $18,200 more
- if the house was located in Kwinana LGA, as compared to the baseline which is Cockburn LGA, the house would be worth $148,000 less.

Table 7.3 Marginal Implicit Prices of significant variables

HOUSE ATTRIBUTE	MARGINAL IMPLICIT PRICE ($)
No of bedrooms	11,833
No of bathrooms	49,766
House age (years)	-11,776
Land area (m^2)	19,465

Distance to the nearest shopping centre (m)	40
Distance to the nearest park (m)	-40
Distance to the nearest beach access point (m)	-20
Distance to nearest manmade wetland (m)	-22
Distance to nearest geomorphic wetland (m)	9
Distance to the nearest nature conservation park (m)	14
Distance to the nearest industrial area (m)	9
Area of the nearest geomorphic wetland (ha)	139
House within 6km of Thomsons Lake	34,331
House within 10km of The Spectacles lakes	17,423
House sold in 2010	18,266
House sold in 2011	-11,594
House sold in 2013	47,555
House sold in 2014	80,261
House in Kwinana LGA	-148,764
House in Rockingham LGA	-189,575

Wetland premium

Based on the MIP of houses within 6km of Thomsons and 10km of Spectacles Lakes, it is possible to estimate the current premium of Thomsons Lake and The Spectacles lakes that is capitalised in the value of surrounding properties in the current time period and for the future. The number of houses in the current time period was based on data from the Valuer's General Office. Future housing was estimated using land planning data from the Western Australia Planning Commission (2012) and the rate of infill of new housing, which was estimated to be around 28 to 31%/year (Western Australia Planning Commission, 2014).

It was estimated that 5,303 houses are situated within the Thomsons Lake 6km premium zone. There are a number of houses that are situated within the premium zone of both wetlands. Regression analysis suggested that the premium value of being in both zones is additive. Other forms of model specification were tested, for example, a multiplicative premium of being in both zones was tested but regression results confirmed that an additive premium specification provides the best model fit. Figure 7.1 illustrates the premium zones for Thomsons Lake, The Spectacles lakes and the area where the premium of both lakes overlap.

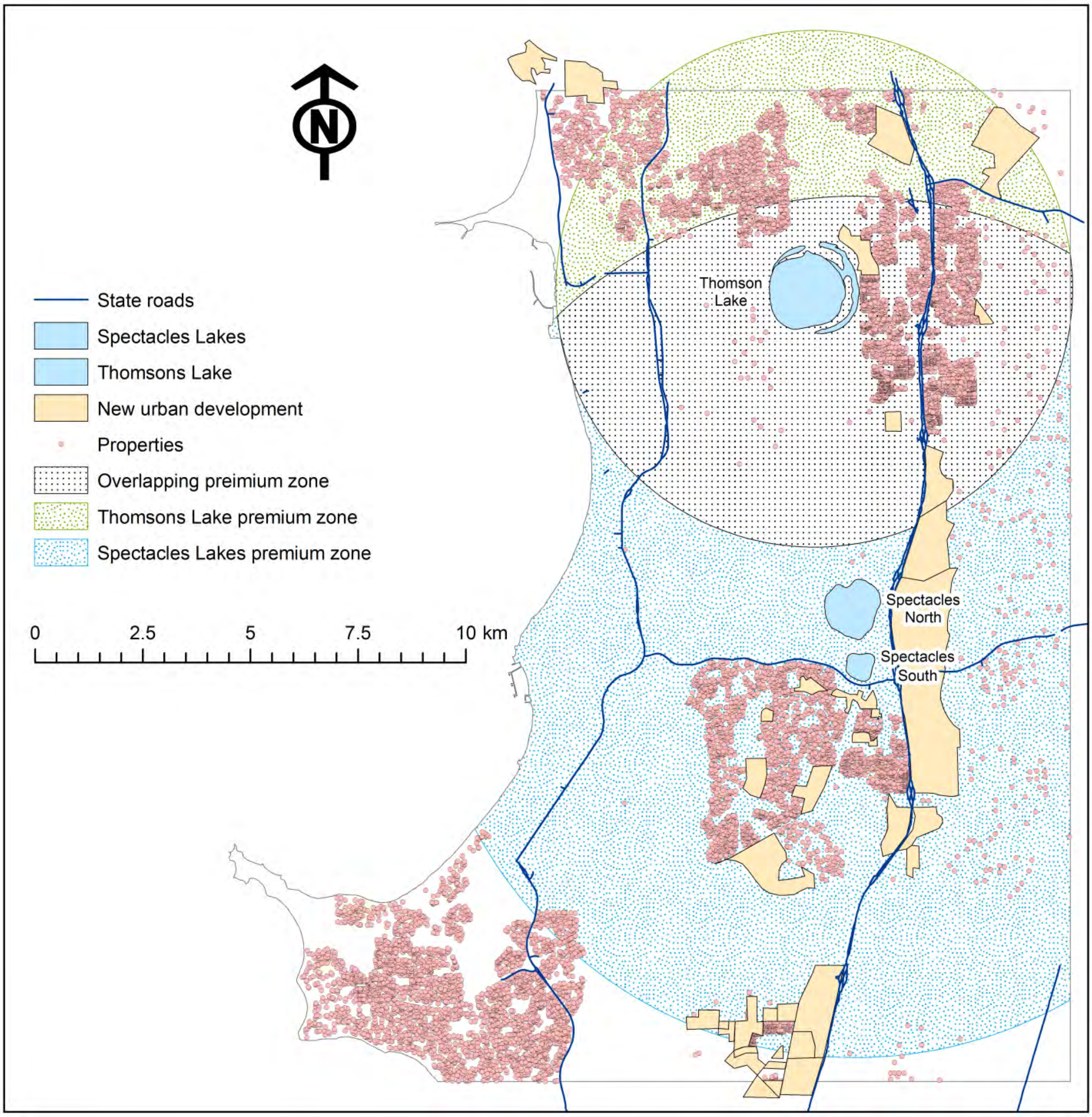

Figure 7.2 Map of study area indicating premium zones for Thomsons Lake and Spectacles Lakes

Geographical analysis in ArcGIS (ESRI, 2010) reveals that 3,507 houses lie within the premium zone of both Thomsons Lake and The Spectacles lakes, but, only 1,796 houses are within the premium zone of Thomsons Lake only. Table 7.4 provides a breakdown of the number of houses that are within each premium zone.

Table 7.4 Breakdown of the number of the current number of houses within each premium zone

CATEGORY	NO OF HOUSES (PERCENTAGE OF TOTAL)
Number of existing houses in the study area	8,589 (100%)
Number of houses in Thomsons Lake premium zone	1,796 (20.91%)

Number of houses in The Spectacles lakes premium zone	3,286 (38.26%)
Number of houses in overlapping premium zone	3,507 (40.83%)

Data on future land development (Western Australia Planning Commission, 2012) indicate that new area of urban development that is within the premium zone of Thomsons Lake is around 5.12 km^2. However, there is approximately 2.81 km^2 of that area that is situated between the premium zone of both lakes. Hence, the area of new urban development that is attributed to Thomsons Lake only is around 2.31 km^2. Based on a current housing density of 205 houses/km^2, it was assumed that the new housing development will have the same housing density[7]. As such, it was estimated that a total of 474 new houses will be built in the new urban development area. Table 7.5 provides a summary of the expected number of houses that will be built within each premium zone.

Table 7.5 Breakdown of the expected number of houses that will be built in new urban development areas

CATEGORY	NO OF HOUSES (% OF TOTAL)
Expected number of future houses in the study area	3,703 (100.0%)
Expected number of future houses in Thomsons Lake premium zone	474 (20.9%)
Expected number of future houses in Spectacles Lakes premium zone	2,653 (38.3%)
Expected number of future houses in overlapping premium zone	576 (40.8%)

According to the Western Australia Planning Commission (Western Australia Planning Commission, 2014), the rate of infill of new housing is around 28 to 32% per year. At this rate, the 3,703 new houses will be built within three years, at a growth rate of 3,703*((0.28+0.31)/2) = 1,092 houses/year.

Using information on the number of houses in each premium zone in the current time period and in the future, combined with the premium value indicated by the MIP of each wetland, one can estimate the premium of Thomsons Lakes and The Spectacles lakes for the current and future time periods.

Based on the MIP value indicated in Table 7.3, houses that are within the premium zone of Thomsons Lake are *on average* worth $34,300 more than houses that are outside this zone. Therefore, the total premium of Thomsons Lake in the current time period is estimated to be around $61,656,680 with a 95% confidence interval (CI) estimate of $24,752,472 to $98,560,888.

Assuming that the premium for Thomsons Lake stay the same for new houses, it was estimated that new urban development would overall add an additional $16,272,420 of premium to Thomsons Lake (95% CI of $6,532,668 to $26,012,172). Table 7.6 presents a summary of the estimation of Thomsons Lake premium.

[7] Average number of houses per km^2
From ArcGIS, 3 X 1km^2 grids reveal the no of houses to be
grid 1 – 256 houses (E of Thomson)
grid 2 – 132 houses (S of Spectacles)
grid 3 – 227 houses (N/W grid)
Average no of houses per km^2 is 205

Table 7.6 Thomsons Lake Premium calculation

ITEM	VALUE
Premium distance	<=6000m
No of houses within 6km of Thomson Lake	5,303
Minus no of houses within overlapping area	3,507
No of houses with Thomson Lake premium	1,796
Price premium (MIP) / house [95% CI in parenthesis]	$34,330 [$13,782 to $54,878]
Total premium (MIP X No of houses) [95% CI in parenthesis]	$61,656,680 [$24,752,472 to $98,560,888]
Total area of new urban development within 6 km of Thomson Lakes (km^2)	5.12
Minus area of new urban development within overlapping area	2.81
New development area with Thomson Lake premium	2.31
Average No of house per km^2	205
Estimated No of new houses with Thomson Lake Premium	474
Projected premium for new housing development [95% CI in parenthesis]	$16,272,420 [$6,532,668 to $26,012,172]

Following the same estimation technique, the estimated current premium for Spectacles Lakes is $57,248,692 [CI of $13,558,036 to $100,939,348], and the expected future premium for new urban development is $46,220,566 [CI of $10,946,278 to $81,494,854]. Table 7.7 presents a summary of the estimation of The Spectacles lakes premium. Note that these are also based on estimates of houses that are within the premium zone of the Spectacles Lakes only, and not houses that are within both premium zones.

Table 7.7 The Spectacles lakes premium calculation

ITEM	VALUE
Premium distance	<=10,000m
No of houses within 10 km of Spectacles	6,793
Minus no of houses within overlapping area	3,507
No of houses with Spectacles Lakes premium	3,286
Price premium (MIP) / house [95% CI in parenthesis]	$17,422 [$4,126 to $ 30,718]
Total premium (MIP X No of houses) [95% CI in parenthesis]	$57,248,692 [$13,558,036 to $100,939,348]
Total area of new urban development within 10 km of The Spectacles (km^2)	15.75

Minus area of new urban development within overlapping area	2.81
New development area with The Spectacles lakes premium	12.94
Average No of house per km^2	205
Estimated No of new houses with The Spectacles lakes premium	2,653
Projected premium for new housing development [95% CI in parenthesis]	$46,220,566 [$10,946,278 to $81,494,854]

The wetland premium for a house that is situated within both zones is around $51,000 ($34,000 for Thomsons Lakes premium and $17,000 for Spectacles Lakes premium). It was estimated that there are currently 3,501 houses that are situated within the overlapping area of the two premium zones. Therefore, estimated premium for current housing is $181,494,264 [CI of $62,803,356 to $300,185,172], and for future housing is $29,809,152 [CI of $10,315,008 to $49,303,296]. Table 7.8 presents a summary of the estimation of Thomsons Lakes combined with The Spectacles lakes premium.

Table 7.8 Overlapping premium area calculation

ITEM	VALUE
No of houses within overlapping area	3,507
Price premium (MIP) / house [95% CI in parenthesis]	$51,752 [$17,908 to $85,596]
Total premium (MIP X No of houses) [95% CI in parenthesis]	$181,494,264 [$62,803,356 to $300,185,172]
Total area of new urban development within overlapping area	2.81
Average No of house per km^2	205
Estimated No of new houses within new urban development area	576
Projected premium for new housing development [95% CI in parenthesis]	$29,809,152 [$10,315,008 to $49,303,296]

Taking all the premium values for the current time period, the estimated premium of Thomsons Lake and The Spectacles lakes is expected to be around $61,656,680 + $57,248,692 + $181,494,264 = $300,399,636 [95% CI of $101,113,864 to $499,685,408]. The total expected premium of the two wetlands from new housing development is $16,272,420 + $46,220,566 + $29,809,152 = $92,302,138 [CI of $27,793,954 to $156,810,322].

7.6 Discussion

Considering the size of Thomsons Lake, which is 341ha and Spectacles Lakes (north and south combined), which is 191ha, the estimated premium for these two wetlands of $300million is relatively conservative as compared to premium estimates by Tapsuwan et al. (2009). Tapsuwan and colleagues estimated that the value of a 20ha urban wetland in the northern suburbs of Perth was worth around $140 million. The differences in value between this study and the study by Tapsuwan et al. 2009 may stem from the fact that wetlands in the northern suburbs, such as Lake Monger and

Herdsman Lake, have higher amenity value because they are generally wet all year round - some of these northern suburb wetlands are naturally wet, while some are artificially maintained by pumping groundwater into these wetlands - and are visited by more people. As such, wetlands valued by Tapsuwan et al. (2009) are likely to have higher aesthetic appeal than Thomsons Lake, and more visitation than Spectacles Lakes, resulting in higher amenity value.

A separate study by (Marsden Jacob Associates, 2012) estimated that the value of urban and peri-urban wetlands on the Gnangara Mound was worth $4 billion. This study extrapolated the wetland value estimated by Tapsuwan et al. (2009) and assumed that all wetlands on the Gnangara Mound had the same value per hectare. Findings from this study revealed that not all wetlands in the study area add value to sales price. As such, it is unlikely that wetlands in the study area are worth as high as $4 billion.

The Spectacles Lakes may have high aesthetic value as compared to other geomorphic wetlands in the study area because historical images of wetlands (i.e. NDVI and Normalised Difference Wetness Index) suggest that the Spectacles lakes have always been wet through time. It is possible that the current infiltration of treated wastewater around the Kwinana wastewater treatment plant is having an effect on groundwater levels in the area. As a result, houses that are within a 10km radius of the Spectacles Lakes, on average have a higher sales price than houses that are further away, holding everything else constant at the mean.

With increasing housing density and urban expansion in Perth to cope with increasing population, it is expected that more houses will be built around wetlands. As previously mentioned, empirical evidence in Australia and the rest of the world confirm that proximity or view of wetlands add value to sales price. This means that future houses built around wetlands will have the amenity value of wetlands capitalised in their sales prices also. In other words, if wetlands remain wet into the future, they will be providing more direct use benefits to surrounding houses in the future. Additionally, if dried out wetlands are made wet once again, either through natural or engineering processes, there is potential for these wetlands to add value to surrounding homes as well. As a result, councils will benefit from being able to realise higher council rates from these new homes. On the other hand, the premium of being in close proximity to wetlands would be lost forever if the wetlands were allowed to go dry and terrestrialised by vegetation. It is unlikely that 'green space' of such nature i.e. wetlands becoming terrestrialised would add any premium value to surrounding households (see Tapsuwan et al., 2009).

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8 Groundwater model and simulations

Authors: Elise Bekele, Mike Donn, Irina Emelyanova and Don McFarlane

Key findings

- A local area groundwater model was developed based on data from previous modelling and local catchment-scale water level data to investigate groundwater response to Managed Aquifer Recharge (MAR) in the Cockburn Sound Catchment study area.
- A fairly good calibration of horizontal hydraulic conductivity values in the groundwater model was achieved based on the goodness of fit between simulated and observed water levels in the Superficial Aquifer.
- The groundwater model also satisfactorily reproduced a watertable mound produced by infiltration of treated wastewater at the Kwinana WWTP.
- Although there are limitations and uncertainties inherent in the model, the model can be used within these constraints to investigate the groundwater level response, flow rates and the impact on salt water intrusion interfaces to MAR for a range of climate and development scenarios.

8.1 Introduction

A local area groundwater model was developed to help evaluate the constraints and opportunities for managed aquifer recharge (MAR) in the 286 km^2 study area. It was used to simulate existing infiltration at the KWWTP for validation purposes and to investigate future groundwater response over a twenty year time frame to the implementation of MAR at different locations, using different volumes of wastewater recharge, and testing the sensitivity of the model results to different projections of future climate and groundwater abstraction. To meet these objectives, the model needed to have the flexibility to include spatially- and temporally varying recharge based on climate, land use and soil type. Moreover, it needed to have a relatively small model grid cell size which included grid refinements in the areas of interest (e.g. MAR sites; bore fields). The model contains publically-available groundwater level observations from DoW wells and data provided by Kwinana Industries Council (KIC) members and the most current groundwater abstraction data. The modelled period was from the beginning of 1990 to the end of 2012. Projected groundwater levels were from the end of 2012 until the end of 2032.

Groundwater models have been developed by previous modellers for sub-sections (e.g. Nield 1999; 2004) and overlapping portions of the study area (Marillier et al. 2012a) as discussed in Chapter 3. However, the best option was to use data files from the Perth Regional Aquifer Modelling System (PRAMS) and the modelling tool Visual MODFLOW (version 2010.1) to develop a local groundwater model specific to the needs of this study. Data files from PRAMS 3.2 were used as it was the most recent, fully-reviewed version available for general release by the DoW at the beginning of this project.

Table 8.1 provides details on the selection of PRAMS 3.2 input data files incorporated in the local area model.

Table 8.1 Summary of datasets extracted from PRAMS 3.2 and used in the local area model

DATA	DATA FILES ACQUIRED FOR PRAMS V3.2	MODIFICATIONS FOR THE LOCAL AREA MODEL FOR THE KWINANA STUDY
Horizontal grid resolution	500 m by 500 m grid cell sizes	Initially, the PRAMS grid cells were used to input data files. Grid refinement was made after selecting MAR sites in the catchment. Grid cells are mainly between 15 and 130 m in the horizontal plane.
Topographic elevation and layer elevations	Elevation of top of layers 1 to 8 (at 500 m by 500 m horizontal grid spacing)	LIDAR digital elevation data was used for the topography of layer 1 with a 30 m by 30 m resolution to more accurately estimate depths to groundwater in the unconfined aquifer. Elevations of layers 2-8 from PRAMS 3.2.
Hydraulic conductivity	Hydraulic conductivities for layers 4 to 8	Hydraulic conductivities for layers 1,2 and 3 were considered from PRAMS 3.2, and adjusted using values from Simon Nield (1999; 2004) and based on calibrations with more recently acquired groundwater data.
Other hydraulic properties	Specific yield, porosity, specific storage values from PRAMS did not vary significantly by layer. Average values for each parameter guided the assignment of values in the local area model.	These values were applied: Sy=0.2; porosity of 0.35; Ss = 1.0E-5 [1/m]
Groundwater abstraction	There was not enough detailed coverage of abstraction in the CSC in PRAMS 3.2 at the scale required for the local model.	See Section 8.2.6 for details.
Groundwater data for model calibration	The observed hydraulic head dataset from PRAMS was considered, but these same data were included in more recent datasets acquired from DoW.	A more comprehensive data set of observed head data was compiled based on data from the DoW and KIC members.
Time variant specified heads for the eastern boundary of the study area.	N/A as PRAMS covered a larger area with boundaries distant from the CSC.	Time variant specified heads along the eastern boundary of the Kwinana catchment were assigned using historical head data from observation wells along the east boundary.
Constant and initial head boundaries	N/A as PRAMS covered a different time period and a larger area. PRAMS assigned a head of 0.5 m to boundary cells along the Indian Ocean.	No flow boundaries were assigned along the North and South boundaries of the local area model; a constant head of 0 m was assigned to the Indian Ocean.
Recharge and Evapotranspiration	N/A; PRAMS uses the vertical flux model, which required additional resources that were not available for this study. A simplified approach to estimating recharge was used.	See Section 8.2.4 for details

8.2 Model specifications

The Visual MODFLOW software used in this study implemented the 2005 version of MODFLOW developed by the U.S. Geological Survey (Harbaugh 2005). It is a finite-difference groundwater model. Particle tracking using MODPATH was also conducted to predict advective transport directions and travel times.

8.2.1 SPATIAL DISCRETIZATION

Horizontal discretization

The study area is the onshore area bounded to the northwest at 375650m E, 6447000m N (GDA94 MGA Zone 50), and extending 23 km south and 20 km wide. The active model area is about 286 km^2. The finite-difference model grid is non-uniform with smaller grid cells surrounding sites selected for MAR infiltration basins and certain wells used to obtain modelled hydraulic heads in close proximity to the infiltration basins (Figure 8.1).

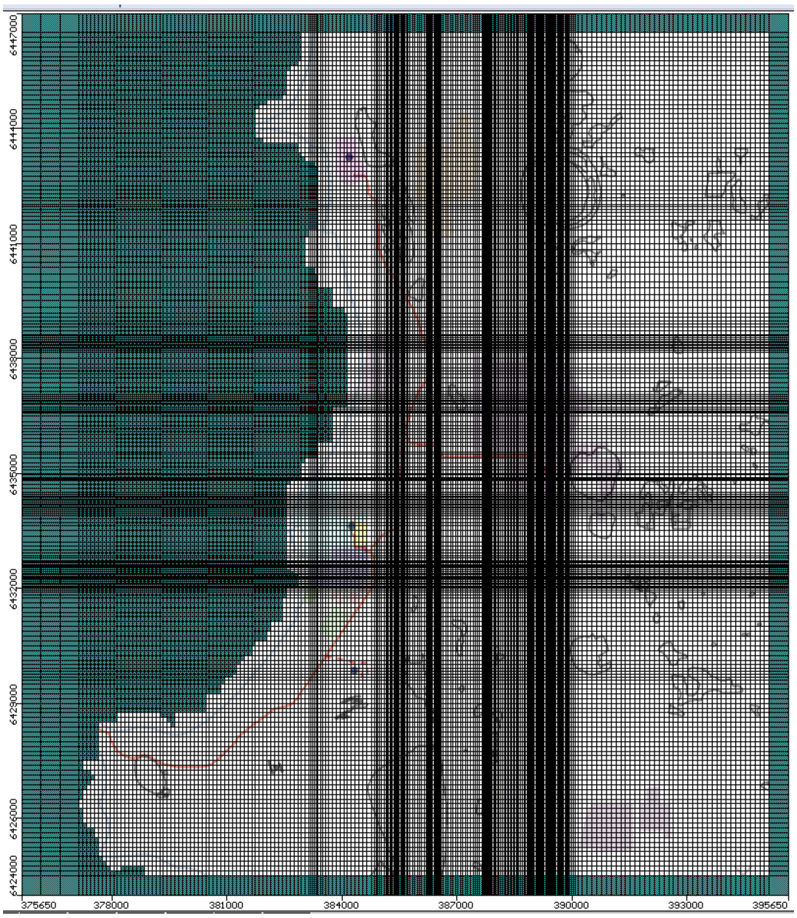

Figure 8.1 Map view of the study area showing the non-uniform grid resolution for the CSC

Vertical discretization

Figure 8.2 shows the conceptual hydrogeological model based on PRAMS (Davidson and Yu 2008; CyMod Systems 2009). The geological formations and corresponding layers in the model are shown in Table 8.2. Chapter 3 provides more details on the stratigraphy represented by the conceptual model.

Appendix A contains thickness maps for the model layers. The first three model layers represent the Superficial Aquifer. Layer 1 is not based on a geological formation. The top of layer 1 is the topographic surface, whereas the base of layer 1 is 10 m below the interpolated watertable of 1989 (CyMod Systems 2009). This layer thickness was based on predicted maximum observed changes in water levels over the area (CyMod Systems 2009).

Some geological formations do not fully extend over the entire model layer, in particular the Rockingham aquifer, Kardinya Shale, and Henley Sandstone and Pinjar Members of the Leederville aquifer. Isopach maps in Davidson (1995) provide more details on the areal distribution of the formations. The version of MODFLOW used in this study does not allow the pinching out or absence of a layer. As described in CyMod Systems (2009), the model layering is based on aquifers such that in areas where a formation is absent, the hydrogeological properties in these areas were assigned to values that represent the formation occupying the layer at that depth. Moreover, where there are layers that subcrop, layer thicknesses were adjusted to a minimum thickness of two meters and assigned the properties of the subcropping formation (CyMod Systems 2009).

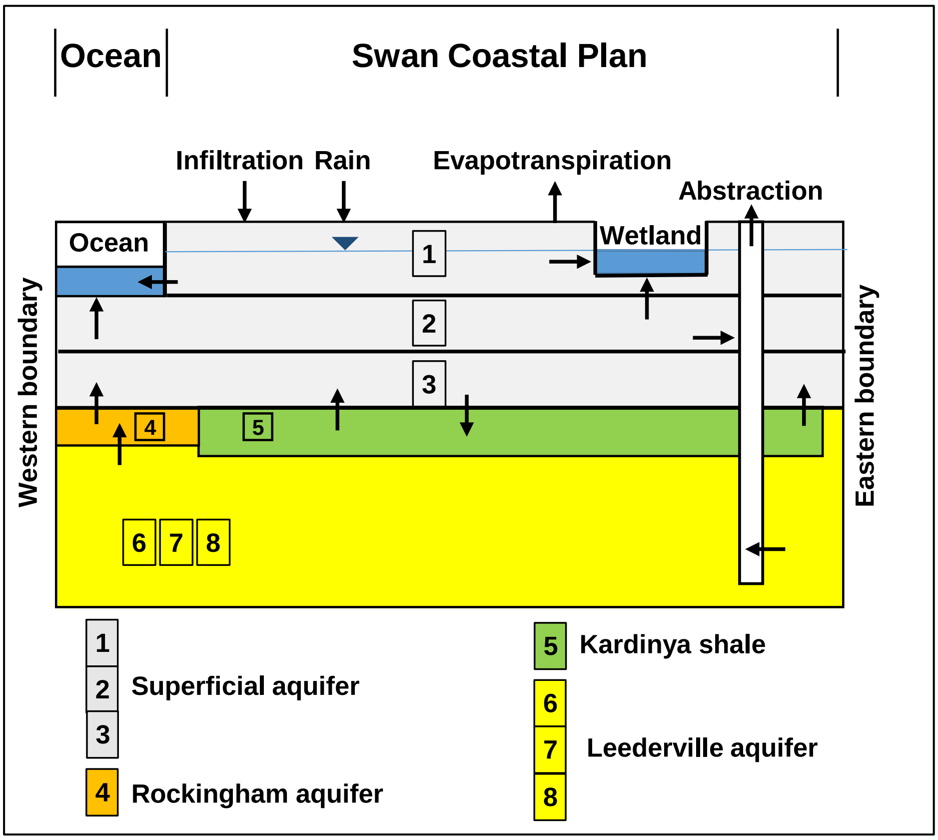

Figure 8.2 Conceptual hydrogeological model. Adapted from "Perth regional aquifer modelling system (PRAMS) model development: Hydrogeology and groundwater modelling," by W. Davidson and X. Yu (2008), Western Australia Department of Water, Hydrogeological record series HG 20. Copyright 2008 Government of Western Australia. Adapted with permission

Table 8.2 Summary of model layers (modified after CyMod Systems 2009)

FORMATION/AQUIFERS	MODEL LAYER	COMMENTS
Superficial Aquifer	1,2 and 3	Layer 1 is the top 10 m of saturated thickness (CyMod Systems 2009). The thickness of layer 2 is half the distance between the bottom of the Superficial aquifer and the water table of 1989 (De Silva et al. 2013).
Rockingham Aquifer	4	Where present
Kardinya Shale	5	Where present
Leederville Aquifer (Henley Sandstone and Pinjar Member)	6	Where present

Leederville Aquifer (Wanneroo Member)	7	Extends across the entire modelled area
Leederville Aquifer (Mariginiup Member)	8	Extends across the entire modelled area

Ground surface

A digital elevation model from LIDAR data area was obtained from the DoW to represent the ground surface topography in the CSC study. The data were input to Visual MODFLOW at a 30 m spatial resolution and interpolated by the software to the spacing of the model grid cells, using the Natural Neighbours interpolation scheme.

As the interpolated ground surface is an approximation, it is important to recognise that any variables that depend on this representation of the land surface will have errors of a similar magnitude. The elevation of the ground surface is used to calculate the depth to the watertable. This has particular relevance for evaluating the potential for inundation of the land surface by rising groundwater levels. The ground surface elevation is also used by MODFLOW in the calculation of the extinction depth and evapotranspiration rate as described in Section 8.2.4.

8.2.2 HYDRAULIC PARAMETERS

Estimates for the hydraulic properties for the aquifers and confining layers are given in Table 8.3 based on PRAMS (CyMod Systems 2009). These were the DoW's best estimates from reviewing available data for the formations on the Swan Coastal Plain and they aid in defining the upper and lower bounds for values that may be assigned during calibration. The actual values assigned in the model were obtained firstly using the data files from PRAMS and then making adjustments to these values during calibration to obtain the best fit to observed groundwater levels. The previous models by Nield (1999; 2004) and Marillier et al. (2012a) were also used as a guide. Figure 8.3 shows the hydraulic zones applied in the model developed by Nield (2004), which was particularly useful for calibrating near the Kwinana WWTP. Note the Tamala Limestone was assigned hydraulic conductivity values as high as 3000 m/day.

The vertical hydraulic conductivity was assigned a value one-tenth the horizontal hydraulic conductivity.

Section 8.3.6 provides the calibrated values used in the model and Appendix A contains the mapped distributions of hydraulic conductivity according to model layer.

Table 8.3 Estimated ranges for hydraulic properties of formations (CyMod Systems 2009; De Silva 2013)

FORMATION/AQUIFERS	HYDRAULIC CONDUCTIVITY (M/DAY)	SPECIFIC YIELD UNCONFINED UNITS; STORATIVITY (S) WHERE INDICATED (-)
Safety Bay Sand	10-15	0.2
Becher Sand	8	0.2
Tamala Limestone	100-1000	0.2-0.3
Bassendean Sand	10-50	0.2
Gnangara Sand	20	0.2
Guildford Clay	<1.0 to 10	0.05
Ascot Formation	8	0.2
Rockingham Sand	20	0.2

Kardinya Shale	$10^{-4} – 10^{-6}$	S: $10^{-3} – 10^{-4}$
Henley Sandstone	2-3	S: $10^{-3} – 10^{-4}$
Pinjar Member	1.0 to 2.0	S: $10^{-3} – 10^{-4}$
Wanneroo Member	1.0 to 10.0	S: $10^{-3} – 10^{-4}$
Mariginiup Member	<1.0	S: $10^{-3} – 10^{-4}$

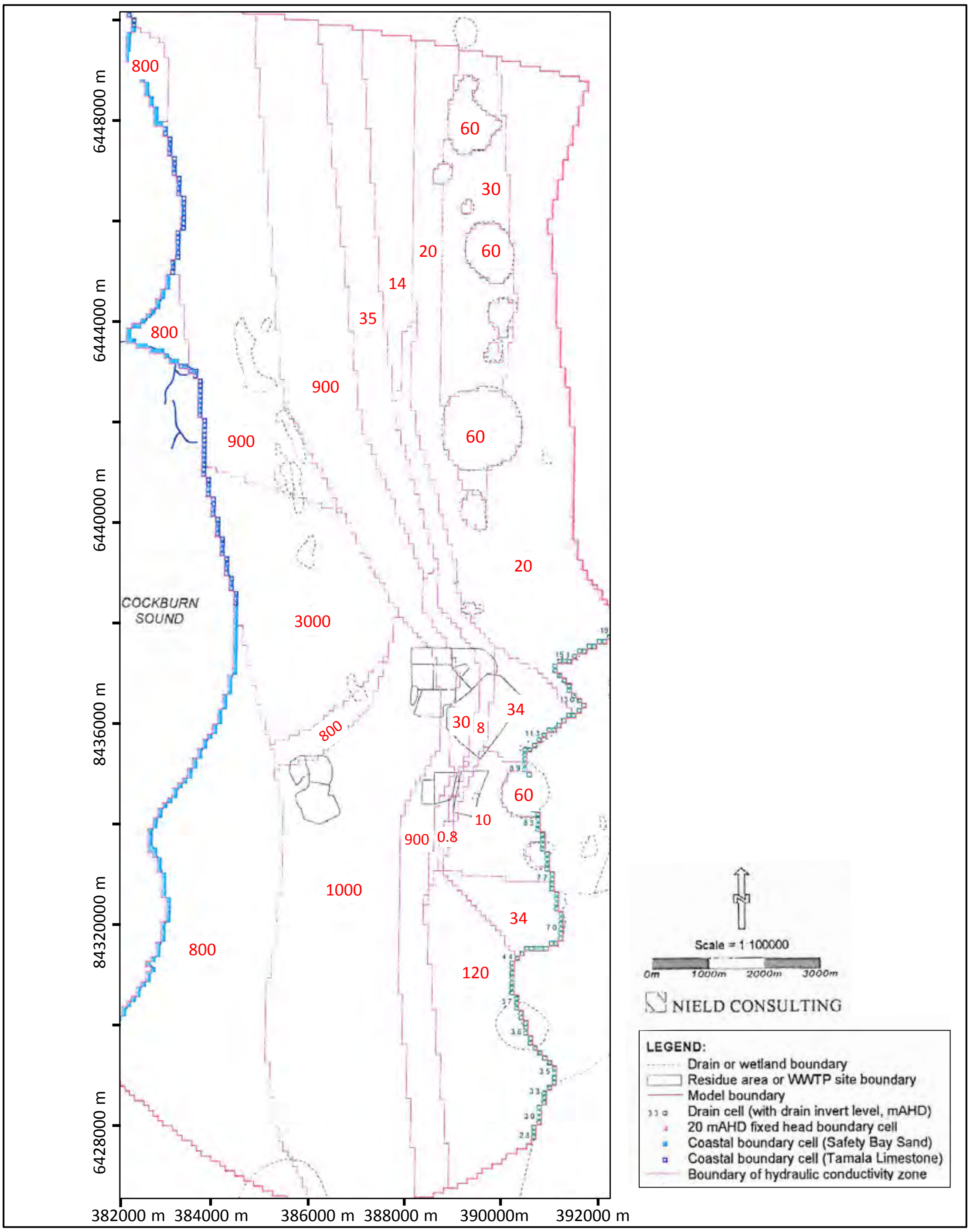

Figure 8.3 Zones of hydraulic conductivity (m/day labelled in red) for the study area modelled by Nield Consulting. Adapted from "Modelling of infiltration from the Kwinana Wastewater Treatment Plant," by S.

Nield (2004), Neild Consulting Pty Ltd Report for Water Corporation. Copyright 2004 by Water Corporation. Adapted with permission

8.2.3 HYDRAULIC HEAD BOUNDARY CONDITIONS

The boundary conditions for hydraulic head were assigned according to the conceptual hydrogeology of the catchment. This assumes that groundwater originates from rainfall recharge, which infiltrates into the Superficial Aquifer and flows vertically to greater depths in the absence of confining layers, and then migrates laterally through aquifers until it discharge into the Indian Ocean. The conceptual model assumes there can be discharge of groundwater offshore at a considerable distance from the coast, mainly in the layers below the Superficial Aquifer.

The northern and southern boundaries are modelled as no-flow boundaries in all layers. These boundaries lie on groundwater flow lines. The eastern boundary is a time-varying specified head boundary based on observed data from monitoring wells located near the boundary. Seasonal groundwater level data from monitoring wells in the Superficial Aquifer were interpolated to the easternmost grid cells in the model on a monthly basis for the simulation period of 1990 to 2012. A polynomial equation was used to interpolate the head data to the nodes that comprise the model grid. Polynomial coefficients in the equation were derived from fitting a curve to the measured data for each grid cell for every month. An example of one of the polynomial equations used to derive the head boundary conditions is given in Figure 8.4. The data for 2012 were the most current and complete data set available when the model was developed. There were no annual trends in the head data for the monitoring wells analysed along the eastern boundary; thus, there was no justification for using a trend to extrapolate time-varying heads for the projected period (2013-2032). Instead, the eastern boundary heads were assigned the dataset of time-varying heads from the final year (2012) for each year of the projected period.

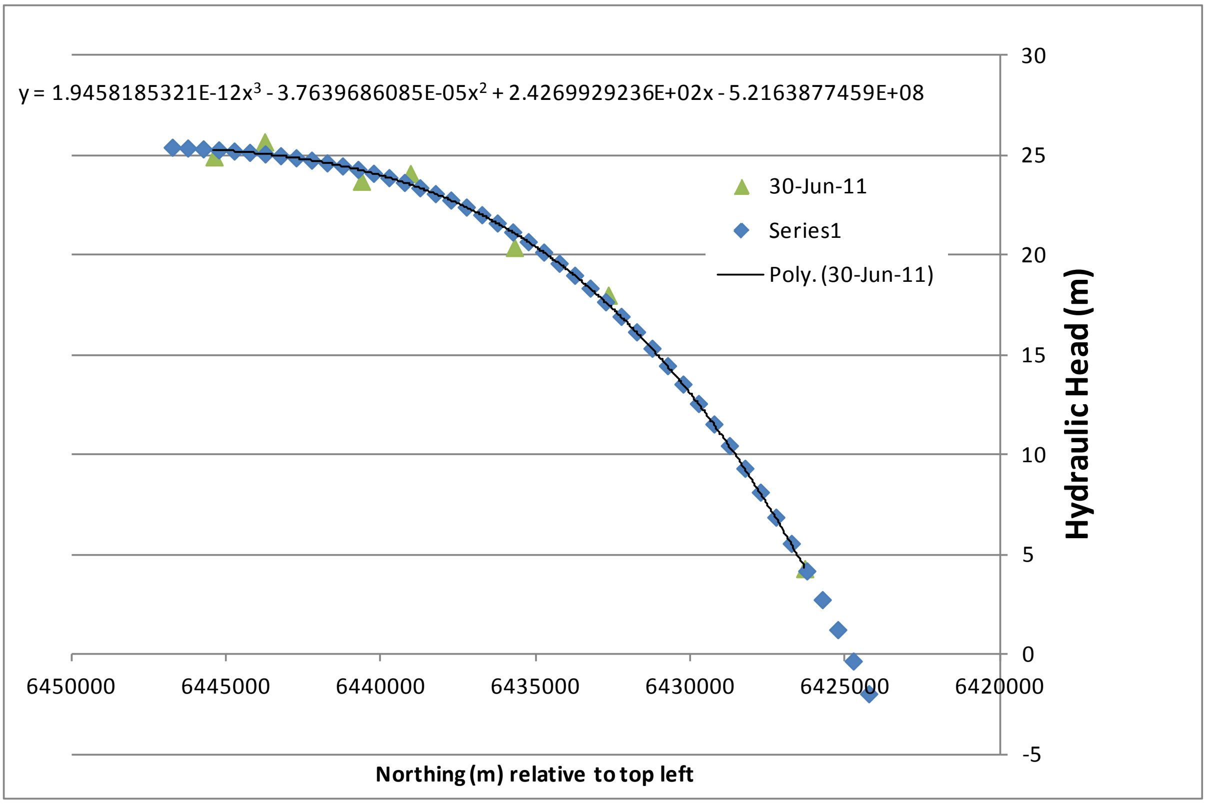

Figure 8.4 Polynomial equation used to approximate the heads along the eastern boundary for one of the modelled time periods (June 2011), based on measured head data from eight wells in the superficial aquifer near the boundary. The curve is a north-south transect or profile of the Jandakot Mound

In the Superficial Aquifer, the coastal boundary was assigned a constant head of 0 m. There are other methods for representing the coastal boundary, but this approach was used as it was simple and involved fewer assumptions.

Coastal boundary conditions used by other modelling studies were as follows: in the single-layer model of the Superficial Aquifer by Nield (1999; 2004), the coastal boundary was assigned constant fluxes according to the presence of Safety Bay Sand or Tamala Limestone outcropping at the surface. Higher flux values were assigned to the Tamala Limestone (2000 to 3200 m^2/day) than the Safety Bay Sand (200 to 600 m^2/day). In addition, a constant head of -0.05 m AHD was assigned to the coastal boundary as an approximation average sea level (Nield 2004). The flux across the coastal boundary was thus presumed to be proportional to the difference between the predicted head and the assigned value of head.

In the PRAMS model, a constant head of 0.5 m was assigned the coastal boundary cells in layer 1 based on the equivalent freshwater head for seawater and assuming an average aquifer thickness of 40 m at the coast (CyMod Systems 2009). This method was also used to impose a stationary saltwater interface at the coast and promotes upward flow from layer 2 into layer 1, consistent with a stationary saltwater interface (CyMod Systems 2009).

Discharge of groundwater offshore through the layers below the Superficial Aquifer was permitted by not imposing boundary conditions along the coast for these layers. The western-most cells of the rectangular grid were assigned as no-flow boundaries as these cells are 75 to 90 km west of the coastline in the study area.

The hydraulic heads for the beginning of the model for the Superficial Aquifer were initialised based on interpolated, measured groundwater level data. Below the Superficial Aquifer, PRAMS data for each of the underlying layers were used as initial head conditions.

8.2.4 RECHARGE AND EVAPOTRANSPIRATION

The conceptual model for natural rainfall recharge to the watertable assumes that rainfall less the amount of actual evaporation is the main driver, whilst land use and soil type are factors that diminish recharge relative to the amount which would occur under bare sandy soil.

Recharge estimation

This section describes the process used to estimate recharge in the catchment and its implementation in the model. This was the starting point; however, during calibration, recharge was adjusted as described in Section 8.3.6.

In the PRAMS model, a vertical flux model (VFM) is used to calculate the amount of recharge by simulating unsaturated zone processes and accounting for leaf area indices (LAI).

In this study, a simpler approach than the VFM was used to estimate recharge. The results from the VFM modelling by Dawes (2008) were applied, assuming they are applicable in the study area, which is fairly close to that of the Peel-Harvey groundwater model.

There were several steps involved in processing different data sets to obtain monthly estimates of net recharge to the watertable and of evapotranspiration (ET) required by MODFLOW as described below.

Historical climate data (daily rainfall and pan evaporation) were obtained from the SILO database, using 21 'data drill' locations that were evenly-spaced across the study area (Figure 8.5). These are synthetic data interpolated from point measurements for several climate stations. Rainfall data were interpolated to the spacing of the original model grid (500 m resolution of PRAMS).

According to the analysis of VFM results in Dawes (2008), maximum recharge in mm is a linear function of rainfall (in mm) given by the equation:

$$\text{Maximum Recharge} = 0.8 \times (\text{Rainfall} - 350)$$

This is essentially a correction factor that is applied before accounting for water loss by evapotranspiration (ET).

The next step involved calculating reduction factors for recharge under freely-draining bare soil using the matrix in Table 8.4. The matrix values are from Dawes (2008) for the Peel-Harvey region. An adjustment to recharge was required during calibration as described in Section 8.3.

The distribution of soil types was obtained from the DoW (SWSY dataset). The annual coverage of land use was obtained from Landsat TM data at a 25 m resolution. Landsat data were generated every other year from 1990 to 2002, and every year from 2002 to 2012. Table 8.5 summarises the predominant land use types that were identified in the study area.

MODFLOW requires recharge to be entered over spatial zones in layer 1 of the model. Each zone can have temporally varying recharge, but the size and location of the zones cannot vary over time. To overcome this limitation, a graphical analysis of annual net recharge was conducted to obtain averages for each zone for each year. Seasonal variations in net recharge were imposed using monthly rainfall.

Evapotranspiration

For evaporation, there was no significant spatial trend across the study area. Therefore average monthly values of pan evaporation were calculated based on the 21 data drill locations. Based on the dominant leaf area index in the catchment, pan evaporation was reduced by 50% to account for interception by vegetation (Warrick Dawes, personal communication). An algorithm in MODFLOW calculates the ET rate. It assumes the rate of ET is greatest at the land surface and diminishes linearly with depth to zero at a given 'extinction depth'. An extinction depth of 2 m was applied in the model.

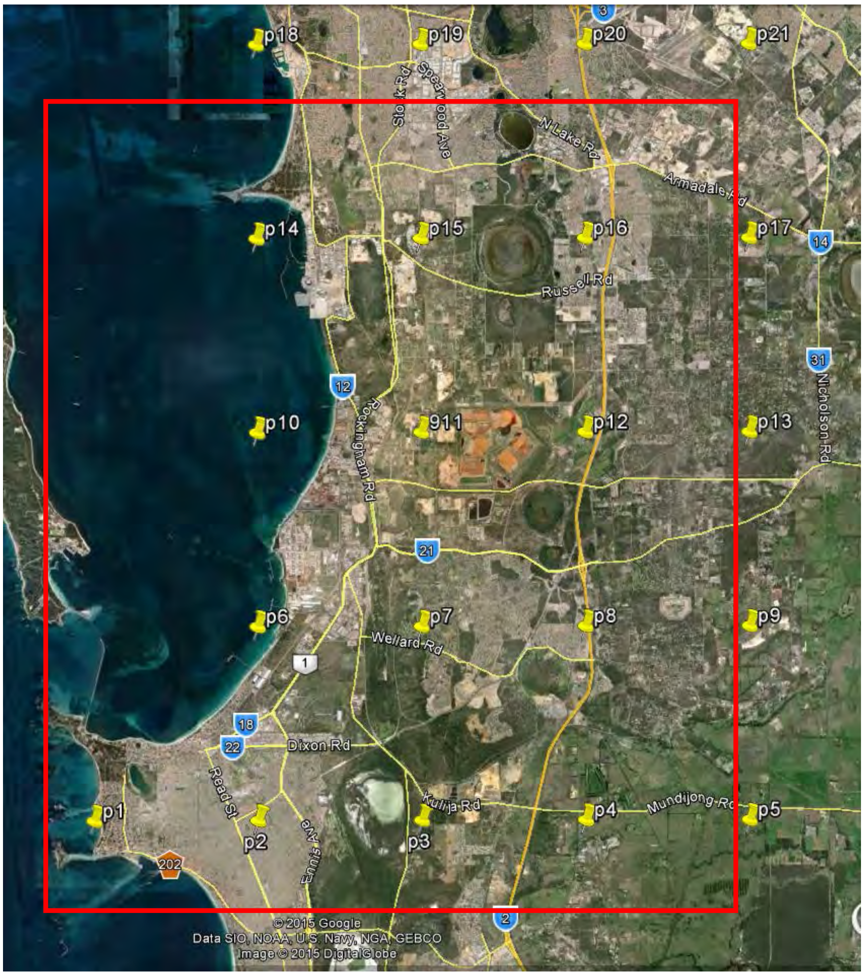

Figure 8.5 Location of 21 data drill sites for SILO data (yellow pins). Image adapted from Google Earth (2015)

Table 8.4 Recharge reduction factors according to land use and soil type

FACTOR	SAND DUNE	DUNE SWALE	FOOTHILLS
1 Bare / Urban (%)	100	75	50
2 Irrigated (%)	80	60	40
3 Cropping / Grazed (%)	60	45	30
4 Native Trees (%)	20	10	5
5 Plantation (%)	10	5	1

Table 8.5 Percentage of land use categories identified in the study area

YEAR	PERCENT OF GRID CELLS IN STUDY AREA (25 M BY 25 M RESOLUTION) FOR EACH LAND USE CATEGORY					
	WATER	MIX OF NATIVE TREES/PLANTATION	IRRIGATION	NATIVE TREES	BARE/URBAN	DRYLAND AGRICULTURE
1990	3	15	1	26	8	47
1992	3	12	1	27	12	45
1994	3	9	1	27	15	45
1996	3	11	1	23	22	40
1998	3	7	1	26	23	40
2000	3	21	2	13	23	38
2002	3	9	2	25	24	37
2003	3	5	1	29	25	37
2004	3	7	1	26	26	37
2005	3	5	2	26	27	37
2006	3	9	2	21	28	37
2007	3	4	1	26	28	38
2008	3	6	2	22	29	38
2009	3	8	2	21	29	37
2010	3	6	2	22	30	37
2011	3	4	1	24	31	37
2012	3	6	2	21	31	37

8.2.5 MANAGED AQUIFER RECHARGE

MAR is represented in the model by assigning a value for the recharge rate to grid cells in Layer 1. The MODFLOW model applies recharge to the uppermost groundwater layer of the model (i.e. the watertable) for each vertical column of grid cells.

The Kwinana WWTP has disposed of treated wastewater via infiltration to the Superficial Aquifer, which is akin to MAR. The site began operation in 1975 with the operation of a single basin for infiltration until a second basin was added in 2001 to allow the plant to operate using a system of alternating (east and west) basins. The spatial coverage of each basin during operation was estimated using historical aerial photos of the plant. The basin area is about 4781 m^2. For modelling purposes, a gridded area of this size was centred on the infiltration site and assigned an average linear rate of infiltration over time. These rates were calculated by dividing daily discharge volume of wastewater (less daily evaporation) by the approximate area of infiltration. Daily discharge volumes since 2000 were provided by the Water Corporation. It was assumed that discharge rates increased linearly from zero in 1975 to 2900 kL/day in 2000.

In the model runs for the 1990-2012 period, infiltration rates were assigned at a rate based on the current disposal of treated wastewater at the Kwinana WWTP, which is approximately 1 m/d and a doubling of this rate was used to test the sensitivity of infiltration to additional infiltration. These rates were also applied in scenario models for the projected period (2013-2032) where the SDOOL was selected as the source of the treated wastewater. In several of the scenarios, the Kwinana WWTP was selected as the source of the treated wastewater for infiltration. To determine the volume available for recharge (and the infiltration rates) in these scenarios, projected volumes of wastewater inflow to the Kwinana WWTP were obtained from the Water Corporation. As the disposal limit of wastewater via the infiltration basins at Kwinana WWTP is currently 4.7 ML/day, the excess wastewater that currently is disposed via the SDOOL was applied as recharge in scenarios of MAR on an annual basis (see Figure 5.2). Two different sites were used to simulate MAR using this wastewater that would otherwise be disposed of via the SDOOL. Infiltration for these scenarios was added at an increasing rate over the 20 year projected period in relation to the projected volumes of wastewater inflow to the plant.

8.2.6 GROUNDWATER ABSTRACTION

Groundwater abstraction from licensed bores in the Superficial and Leederville aquifers was modelled. Due to the lack of reliable metered data for the majority of bores, a dataset for groundwater abstraction was synthesized from available information. Previous models for the study area have also experienced issues with estimating extraction and used synthesized datasets of abstraction in a similar manner as discussed below.

The DoW provided licensed abstraction in terms of the maximum volume of water that may be extracted annually from specified wells. The database contained information about the aquifer for licensed abstraction, bore depths and locations. The private licensed abstraction database includes data for Water Corporation bores. For some licenses, the location for abstraction and a listing of the number of licensed bores at the site was provided, rather than the spatial coordinates for individual bores. For these sites, the unknown bore locations were approximated by evenly distributing them across the site. As the database did not contain the screened intervals for individual bores, it was assumed that abstraction occurred over the entire thickness of the aquifer at each location.

Groundwater use was assumed to be 80% of the allocation limits given in the license allocation database. This percentage was used in the models by Nield (1999) and Marillier et al. (2012a) as well as other modelling studies on the Swan Coastal Plain, which refer to original estimates between 1985 and 1995 from Davidson (1995). This approximation was not used in PRAMS 3.0; however, 20% of abstracted water was assumed to return to the Superficial Aquifer (CyMod Systems 2009), which in essence would achieve the same effect.

The licensed allocation database did not contain seasonal patterns of water use. According to a review of water use for licensed bores for the PRAMS model, the majority of licensed bores provide irrigation water, which has a strong season pattern (CyMod Systems 2009). This assessment was for the large area covered by PRAMS and may not be entirely applicable to the study area. However, in the model developed by Marillier et al. (2012b) monthly scaling factors from Sun (2005) were used based on estimates from different industries. Annual allocation was converted to a monthly abstraction rate using the scaling factors given in Table 8.6. This approach was adopted for this model as this was used for the Lower Serpentine model by Marillier et al. (2012 b), which overlaps with the catchment.

Table 8.6 Monthly scaling factors applied to annual allocation in Marillier et al. (2012b) that were applied in the CSC model

MONTH	JAN	FEB	MAR	APR	MAR	JUN	JUL	AUG	SEP	OCT	NOV	DEC
Scaling factor	15%	13%	12%	8%	4%	2%	2%	3%	4%	9%	13%	15%

The synthesized dataset for abstraction from 1990 to 2012 was anomalously low in certain years and the overall pattern of variations in annual abstraction did not correspond well to other estimates (Carey Johnston, Department of Water, pers. comm.). There were likely omissions in some of the files of licensed abstraction that were compiled to create the synthesized dataset. The likely underestimation of abstraction in the synthesized dataset was confirmed by comparing it with the abstraction data file for PRAMS 3.5, which showed total annual abstraction of approximately 40 GL for the study area from 2006 to 2012. Over the entire modelled time period (1990-2012), the dataset synthesized for the study contained about 4200 abstraction bores operating at different times, whereas PRAMS 3.5 models abstraction from about 790 bores. PRAMS 3.0 models abstraction from about 300 bores in the study area. The PRAMS model also locates abstraction bores at the spacing of the 500 m grid resolution rather than the geographic coordinates of individual bores.

As the synthesized dataset for total abstraction in 2001 was 39 GL/yr, it was assumed that this was likely correct for that year. In that year, 1550 bores were licensed to abstract. Given the unreliability of the synthesized dataset of abstraction for the majority of the model years, it was decided to apply the abstraction estimates for 2001 to all of the years in the model. There was no assumed growth rate and the abstraction was modelled as constant in space. This approach is not an ideal representation as bore locations have changed over time. To provide context for this estimate, it was compared with the sum of licensed entitlement volumes for the Kogalup, Thompsons, Valley and Wellard subareas that comprise the Cockburn Groundwater Area . The total of licensed entitlements for recent years, declined from 29.6 GL in 2009 to 28 GL in 2014 based on data from the Department of Water (Carey Johnston, Department of Water, pers. comm.). It should be noted that Kogalup extends 5 km north of the study area, and the data provided does not entirely cover the study area. It does not include the area that extends from Wellard to the south boundary of the study area (Rockingham GA) and the area between the four subareas and the east boundary of the study area.

Different approaches have been used in other models to hind-cast groundwater abstraction. In PRAMS 3.0, the year 1997 was identified as the time after which bore allocation records were most reliable. The abstraction for years prior to 1997 was modelled as constant in space with a 3% growth rate in time (CyMod Systems 2009). For the Lower Serpentine model, Marillier et al. (2012a) used the DoW's allocation records to synthesize a representative, estimated history of abstraction and used licenced abstraction locations from 2011, which assumes abstraction locations did not vary through time.

Previous models of the area have either neglected or estimated unlicensed groundwater abstraction. Unlicensed abstraction is permitted by the DoW from bores that abstract less than 1500 kL/yr. These are mostly private garden bores and are taken into account in a general manner in the DoW's database when allocation limits and estimates are made. The groundwater model in Nield (1999) commissioned by the DoW (then, the Water and Rivers Commission) for their review of groundwater allocation in the Cockburn Groundwater Area did not include unlicensed groundwater abstraction. Marillier et al. (2012a) included unlicensed garden bores by assuming 30% of residential properties used 800 KL/yr. In the model for the Lower Serpentine (Marillier et al. 2012a), garden bores within each 200 m grid cell were lumped as a single drawpoint; however, licensed allocations of less than 1500 kL/yr were excluded from the model because the volume abstracted was considered negligible. In PRAMS, estimates of abstraction by garden bores are included. Estimates were obtained from the DoW of the number of garden bores in each groundwater subarea. Abstraction from these bores was implemented in PRAMS using recharge flux values calculated from the number of bores estimated in each subarea, and the average bore usage divided by the size of the area. Recharge flux is then scaled by irrigation coefficients (PRAMS 3.5 documentation from CyMod Systems, Neil Milligan, 2015).

The model developed for this study did not included unlicensed bore use as estimates were not readily available. Figure 8.6 shows the spatial coverage of land cover. In 2012, the proportion categorised as urban residential was about 23% of the study area. Most private bores are likely to be located in the Rockingham area to the south because of the shallowness to the watertable and the large block sizes. Newly urban areas in the north and east, which MAR is more likely to affect, have small lot sizes and large houses making gardens very small and private bores of little value. The few bores that do occur in these areas may be offset by increased recharge from roofs and roads. Therefore their absence from the model is unlikely to cause a problem with estimating levels.

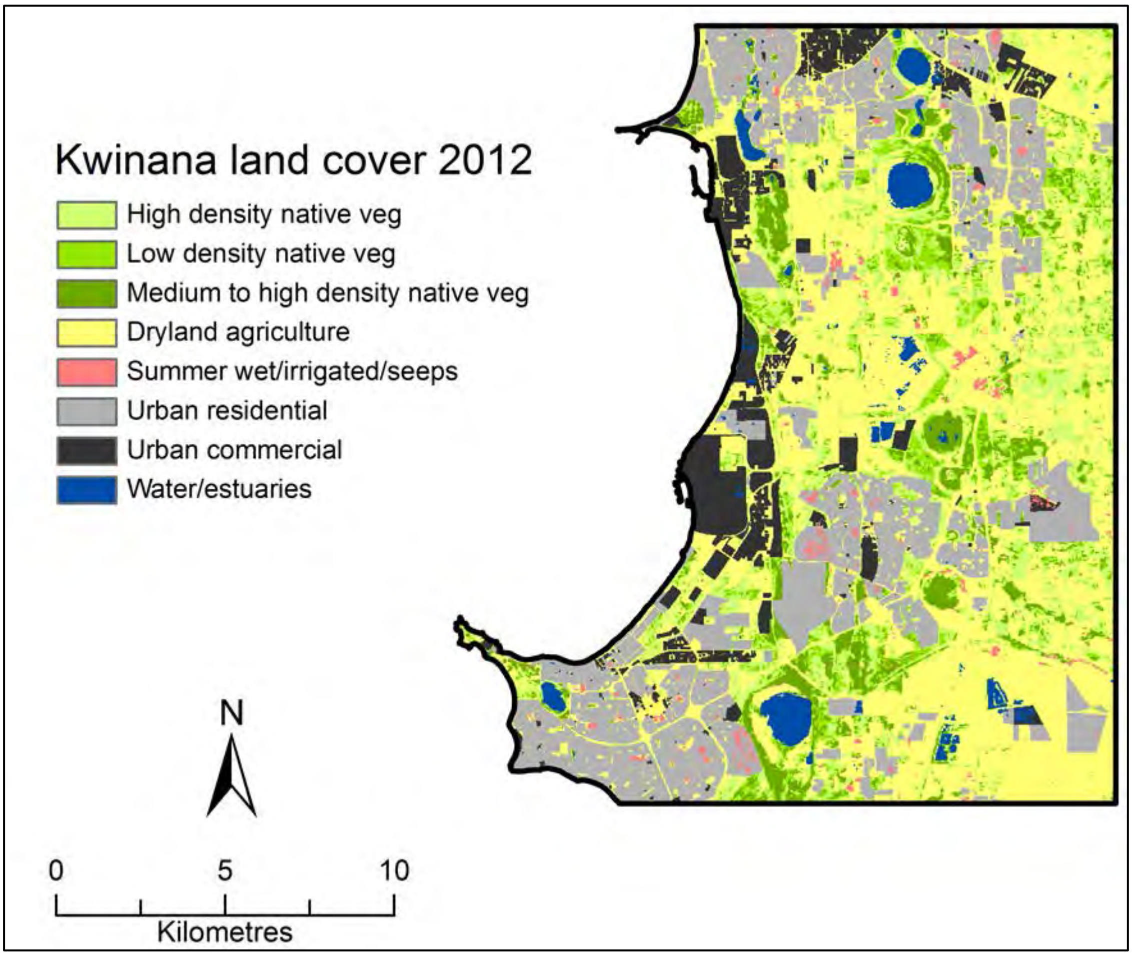

Figure 8.6 Estimates of land cover from Landsat images for 2012. About 23% of the study area was categorised as urban residential

8.2.7 CHOICE OF FUTURE CLIMATE SCENARIO

Groundwater levels in large parts of the Swan Coastal Plain have declined as a result of reduced rainfall and increased in the south-west of Western Australia since about 1975 (CSIRO 2009). Modelled groundwater levels have declined by between 0 and 3m over most of the study area between 1995 and 2012 with several local exceptions; an area near the Kwinana WWTP where treated wastewater has been added since 1975; and two areas in the north where pumping has ceased (Figure 8.7).

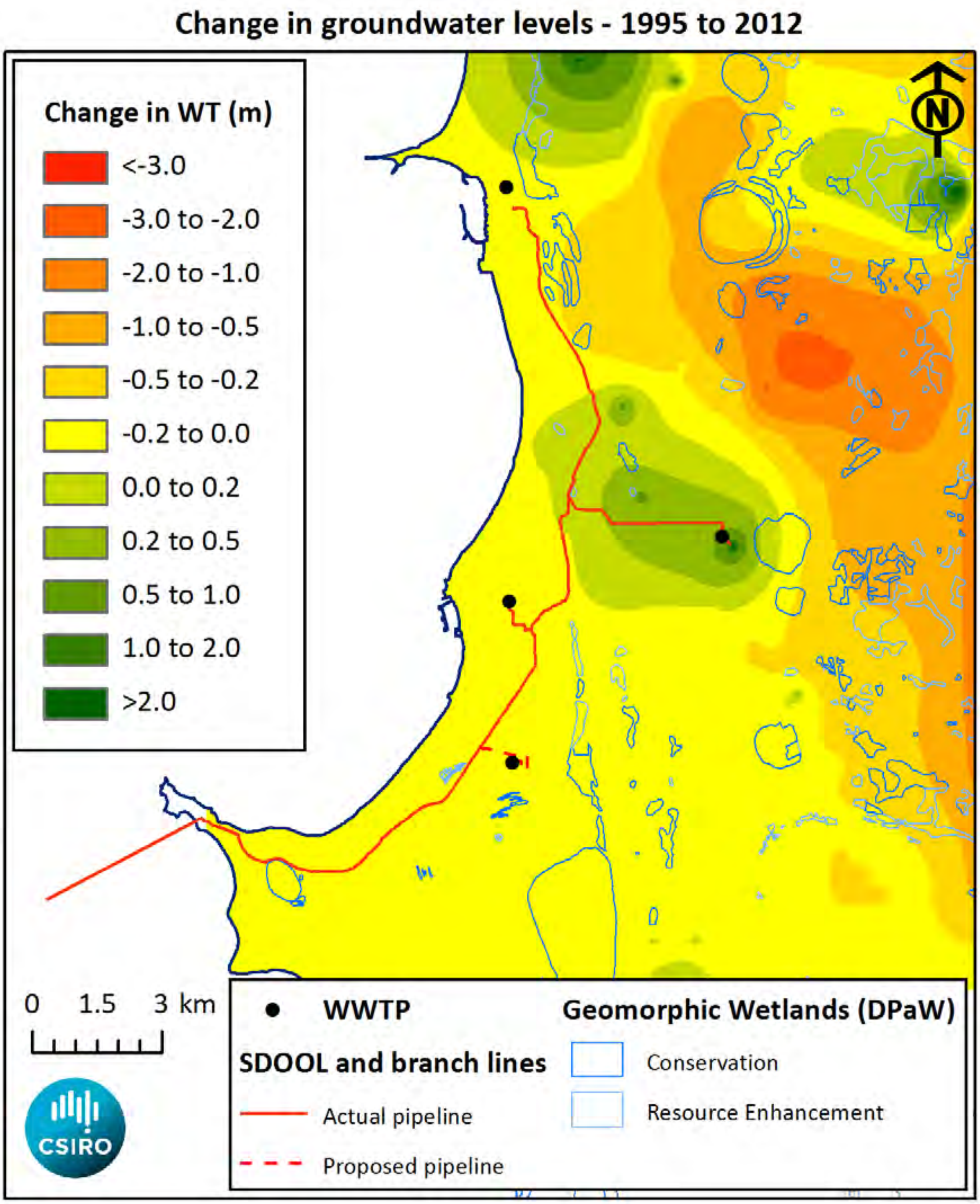

Figure 8.7 Changes in groundwater levels between 1995 and 2012 estimated using the calibrated groundwater model developed for this project

The projection of future groundwater levels after adding treated wastewater through managed aquifer recharge needs to take account of this long-term trend. The method used future climate projections used by the Department of Water. This uses a 1961 to 1990 baseline period, CMIP3 global climate models and a synthetic monthly rainfall amount that is based on the long-term average. From this, dry (90% rainfall exceedance probability), median (50% probability) and wet (10%) scenarios for 2030 were chosen to provide an estimate of the possible range of future climates.

Groundwater model runs were tested for these three scenarios and it was found that the trend in falling groundwater levels was best simulated using the 2030 dry scenario. Table 8.7 and Figure 8.8 show mean rainfall for the study area for the 5 year period 2008 to 2012, the 10 year period 2003 to 2012 and the 23 year period 1990 to 2012 compared with the DoW's median and dry scenario rainfalls. Using the median rainfall resulted in sharp increases in groundwater levels as soon as the simulation commenced in 2013, something that is not in the current record. It was decided that the dry scenario of 715 mm per annum would be used in all future simulations given that it closest to the trend line that has been apparent in recent decades. While it could be criticised for being conservatively dry because it is 30 mm lower than the last 23 year average, it is wetter than both the last 5 and 10 year periods.

Table 8.7. Mean rainfall in the study area for the last 5, 10 and 23 years compared with the DoW median and dry scenarios

PERIOD	MEAN RAINFALL (MM)
2008 to 2012 (5 yrs)	692.5
2003 to 2012 (10 yrs)	712.1
DoW Dry Scenario for 2030	715.3
1990 to 2012 (23 yrs)	746.6
DoW Median scenario for 2030	757.3

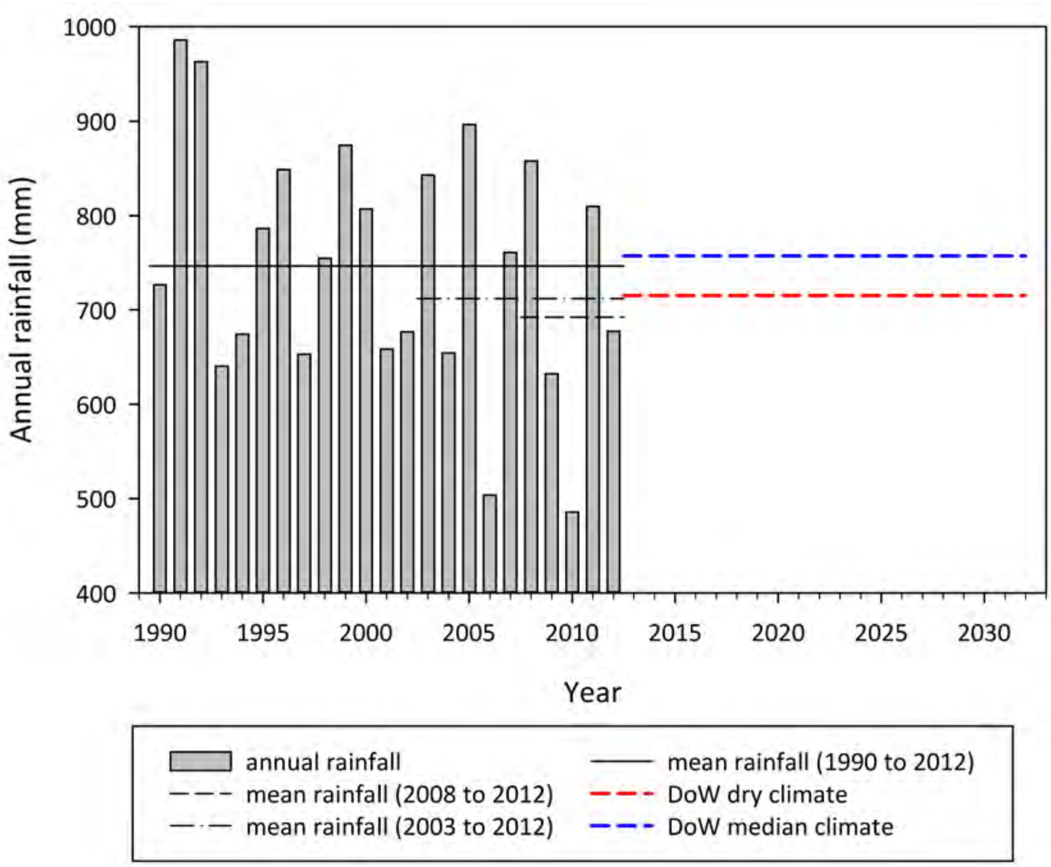

Figure 8.8 Historical record of annual rainfall (mm) from SILO data and 2030 climate projections from the DoW

8.3 Model calibration

8.3.1 METHOD

The typical procedure for calibrating MODFLOW simulations described in Waterloo Hydrogeologic (2005) was conducted. This involved adjusting hydraulic conductivities to minimise differences between simulated and measured groundwater levels. The calibration base for bore hydrographs was for the period from 1 January 1990 to 31 December 2012. A separate objective was to reproduce the general shape and height of the groundwater mound below the Kwinana WWTP. As the modelling analysis by Nield (2004) aimed to reproduce a contour map of measured groundwater levels for April 2004, a similar objective was used. In addition, more recent bore hydrograph data from the plant and The Spectacles were used in the calibration.

For the calibration procedure using bore hydrograph data (1990-2012) across the study area, the majority of the data were from the Superficial Aquifer. The hydraulic conductivities were primarily based on the values assigned in PRAMS 3.2, but these were modified as needed in certain areas to obtain better fits to the measured groundwater level data.

The criteria used to evaluate model calibration included:

- Modelled versus measured groundwater hydrographs for selected wells in the Superficial Aquifer;

- Calibration statistics versus time; and
- Scatterplots of modelled versus measured heads.

The main calibration statistic that was used was the normalised root mean squared (NRMS), which is a measure of the fit between calculated and measured data. The NRMS is the Root Mean Squared (RMS) divided by the maximum difference in the observed head values. The NRMS is considered a more representative measure of fit than the standard RMS because it accounts for the scale of the potential range of data values (Waterloo Hydrogeologic 2005). The RMS is based on the difference between the calculated results (X_{cal}) and the observed results (X_{obs}) at selected data points ($i \rightarrow n$) as indicated in the equations below:

$$RMS = \sqrt{\frac{1}{n}\sum_{i=1}^{n}(X_{cal} - X_{obs})_i^2}$$

$$Normalised_{RMS} = \frac{RMS}{(X_{obs})_{max} - (X_{obs})_{min}},$$

8.3.2 RUN PARAMETERS

The model developed for the study area is a transient flow model with 575 stress periods, beginning in 1990 and running until the end of 2012. A stress period is the time period in which all the stresses (e.g. boundary conditions, pumping rates) are constant. The maximum length of any stress period in the model was a month. The model calculated the number of time steps to use for each stress period and the time step multiplier used to increment the time step size within each stress period was set to a value of 2.

The Geometric Multigrid Solver in MODFLOW 2005 was used. The total number of iterations was limited to 1 outer iteration and 100 inner iterations. The maximum change in the solution at every cell after every outer iteration was limited to 0.01 (HCLOSE). The residual convergence criterion for every inner iteration was limited to 0.01 (RCLOSE).

A review of model results showed that the flow mass balance (total flow IN minus total flow OUT) expressed as a percentage of the total flow was 0.0 for the entire model run. This indicates that the simulation was successful for the selected run parameters and numerical solver.

8.3.3 CALIBRATION WELLS

Hydrographs from 174 wells were selected for model calibration. There were 28 DoW wells, 20 Water Corporation wells, and 126 KIC members' wells, which covered an important area under consideration for MAR and seawater intrusion. The calibration wells were selected based on the amount of data and coverage of the modelled period. The quality of the hydrographs was also considered in the selection of calibration wells; for example, the presence of many outliers was not desirable. Wells with high quality data for the most recent decade were also preferred in the selection. Most of the hydrographs were for wells in the Superficial Aquifer. The location of the calibration wells is shown in Figure 8.9 and Appendix A contains the calibration well data and a synopsis of water level data from industry voluntarily provided by several members of the KIC.

The distribution of hydrograph data is clustered in some areas, but sparse in between. As indicated in Figure 8.9, there is a tight clustering of wells near several industries that contributed data near the coastline, another tight clustering near Kwinana WWTP, a spread of wells around the Jandakot Mound area, but fairly sparse coverage elsewhere. In particular, the areas east of the Jandakot

Mound and near Rockingham have virtually no calibration data; thus, the model relies on the hydraulic properties from the PRAMS model in these areas.

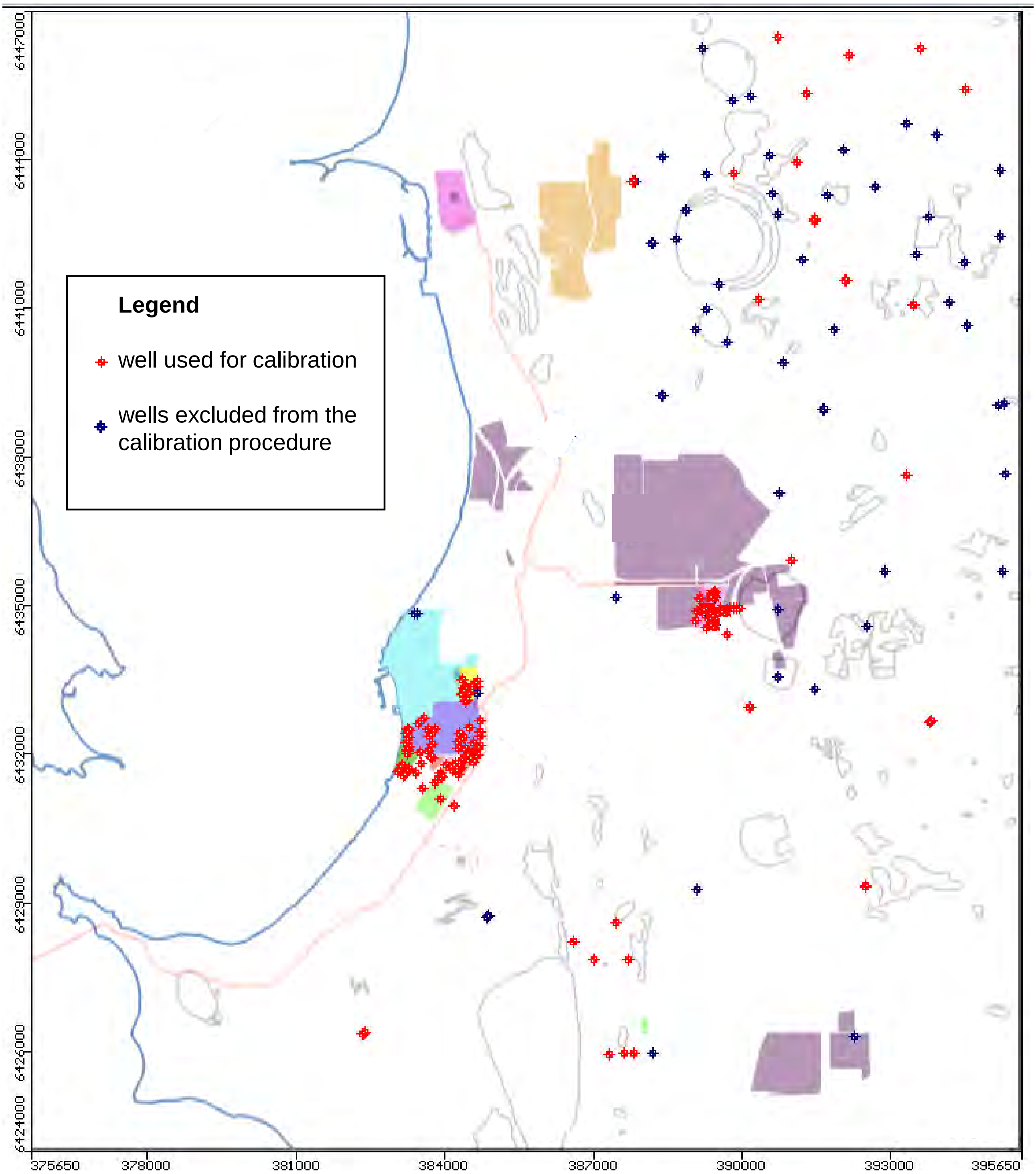

Figure 8.9 Location of calibration wells (red symbols). Wells that were excluded from the calibration procedure (blue symbols) had insufficient data or had water level trends influenced by local activities that were not included in the model

8.3.4 KWINANA WWTP GROUNDWATER MOUND CALIBRATION OBJECTIVE

To demonstrate the ability of the model to represent groundwater response to MAR, output from simulated infiltration at the Kwinana WWTP between 1990 and 2012 were compared with observed data and previous modelling by Nield (2004).

Hydraulic conductivities for the Superficial Aquifer near the Kwinana WWTP were adjusted in line with previous estimates from Nield (2004), which was a calibration to groundwater level that existed at that time (Figure 8.3). As there were more recent data from monitoring wells in the area, these data increased the dataset for calibration. This follows from the advice given in Nield (2004), which noted that the parameter values of the calibrated model developed by Nield Consulting are not unique and that further adjustment of parameter values may be required in the future as more information becomes available. The final set of hydraulic conductivity values in the Superficial Aquifer are in Appendix A.

Within about 500 m of the Kwinana WWTP, six monitoring wells were selected to compare modelled and observed hydraulic heads (Figure 8.10). The majority of the predicted hydrographs for these wells match favourably with the observed hydrograph. There is a stronger seasonality component evident in the simulated hydrographs that is not observed, but in general, the data overlap and the trends are reproduced. Well KW14 is one exception that shows a much larger rate of increase in water levels than predicted by the model. Also, the predicted elevation of the water table for KW14 is lower than observed. KWTP2(I) shows a similar underestimation of the observed water table elevation.

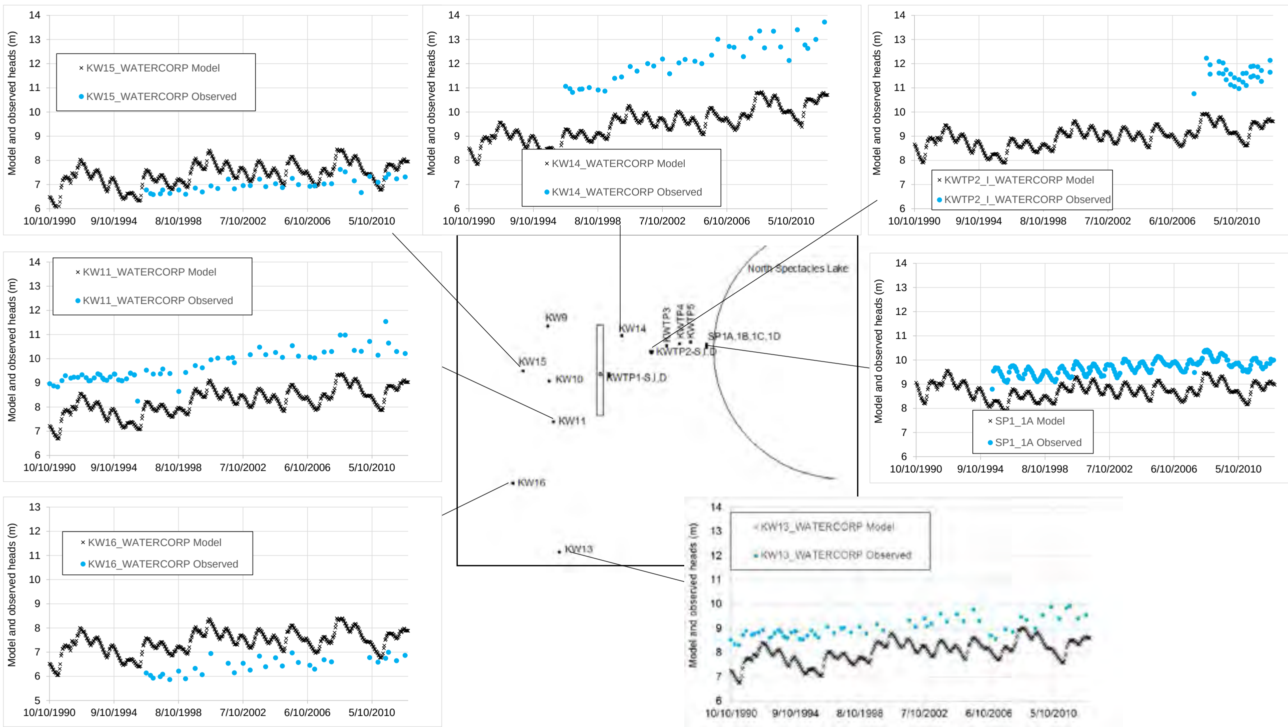

Figure 8.10 Calibration wells located within about a 500 m radius of the Kwinana WWTP infiltration site

To produce a contour map of the watertable, one requires a fairly large number of data points that are well spaced. Nield (2004) produced a contour map for April 2004 taking into account water level data outside the map shown in Figure 8.11 and estimates from other interpretations.

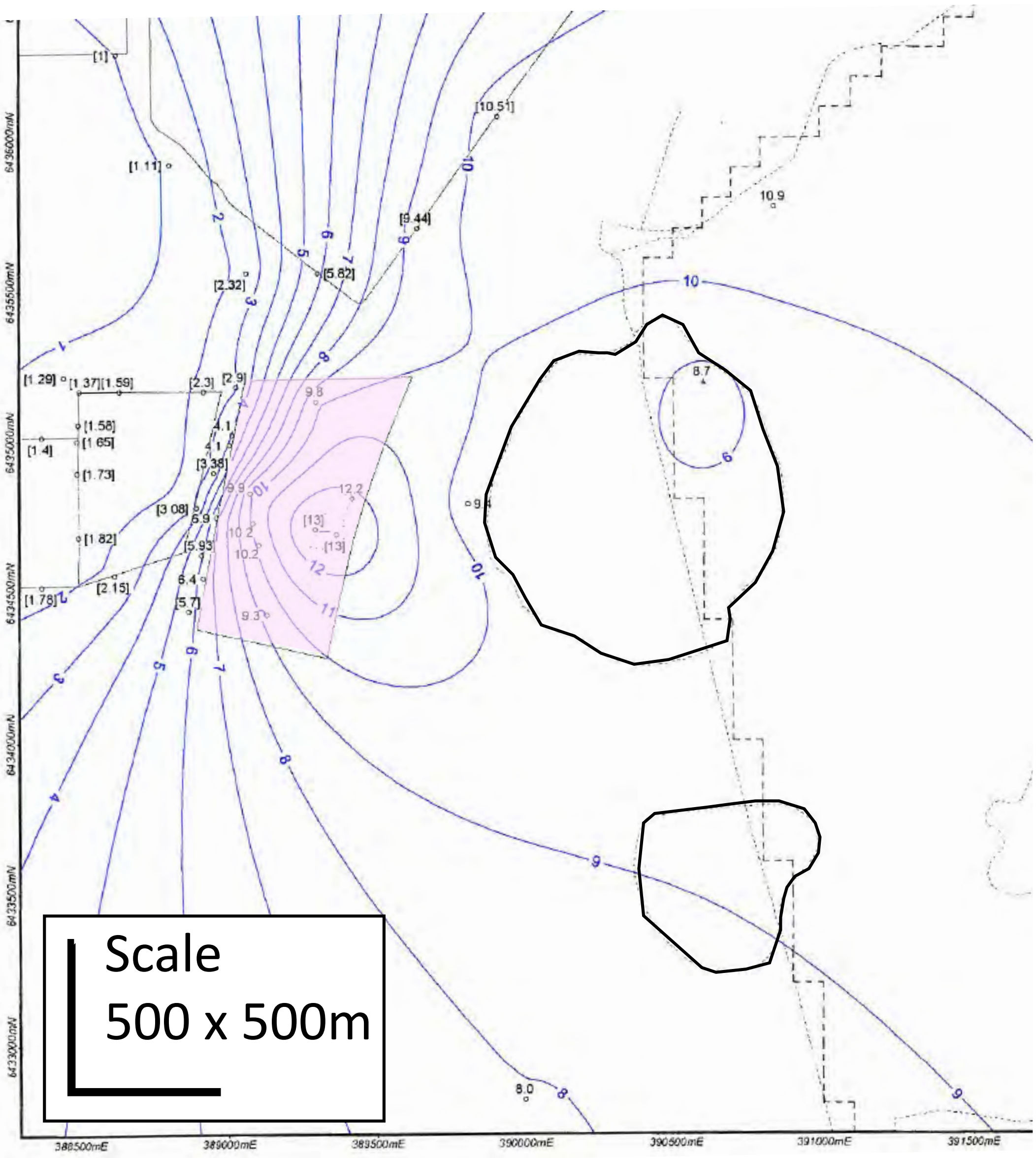

Figure 8.11 Watertable elevation map for April 2004. Adapted from "Modelling of infiltration from the Kwinana Wastewater Treatment Plant," by S. Nield (2004), Neild Consulting Pty Ltd Report for Water Corporation. Copyright 2004 by Water Corporation. Adapted with permission

A groundwater mound formed below the site with a maximum elevation of about 13 m at that time, but the groundwater model predicted an elevation of about 11 m. Figure 8.12 shows hydraulic heads and groundwater velocity vectors computed by the groundwater model. The size and areal extent of the predicted mound depends to some extent on the conceptualisation of the infiltration basin. The model assumes a representative rectangle for the infiltration site centred within the perimeter of the east and west basins that, in reality, are alternately wet and dry. As such, the model is less accurate at reproducing the exact spatial extent and mound elevation. It does appear to be performing suitably well for simulating the regional effects on the watertable of MAR at the site. A comparison of Figure 8.11 and Figure 8.12 shows similar contours of watertable elevation.

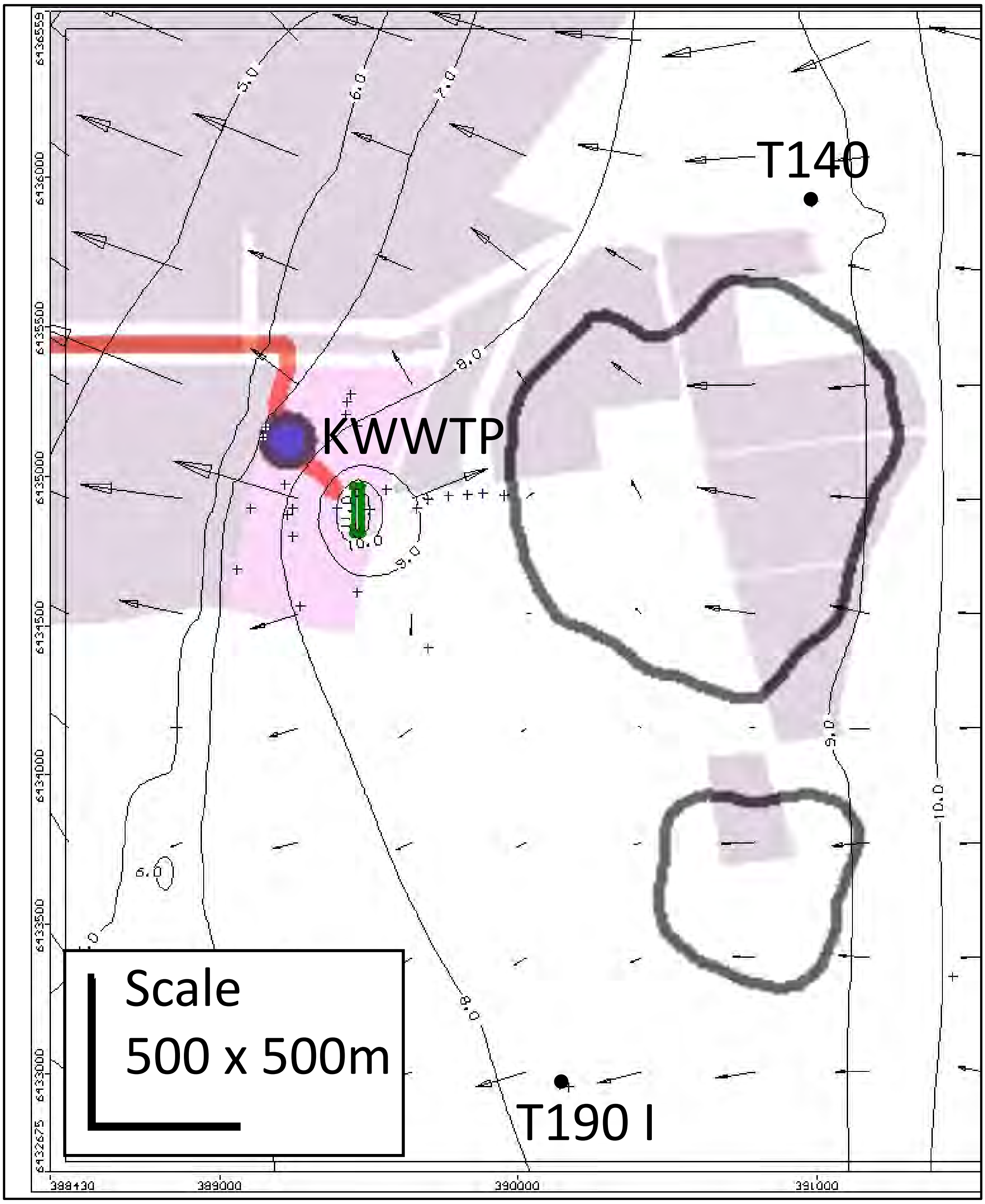

Figure 8.12 Location of monitoring wells T140 and T190 I relative to the Kwinana WWTP and model predicted hydraulic heads for April 2004 to compare with data and model in Nield (2004)

Wells T140 and T190I, which are located 1.8 km northeast and 2.0 km southeast of the Kwinana WWTP were the most distant wells used for calibration of the site (Figure 8.12). As shown in Figure 8.13, the predicted hydrograph for T190 matches the observed fairly well. It is reassuring to have the model perform well here as it is near an area of potential inundation due to the lower surface elevation.

The hydrograph for T140 is poorly matched by the model for the area to the northeast. As shown in Figure 8.13, the predicted hydrograph for T140 is lower overall compared with the observed data. It is not clear why this occurs, but further calibration could be considered in this area if necessary. The predicted hydrograph for T140 does reproduce the generally declining watertable trend in this area.

The discrepancy with the observed hydrograph could be because the simulated amount of groundwater abstraction from pumping bores to the north is too large or that site-specific measurements of hydraulic conductivities from this area are needed to constrain the model.

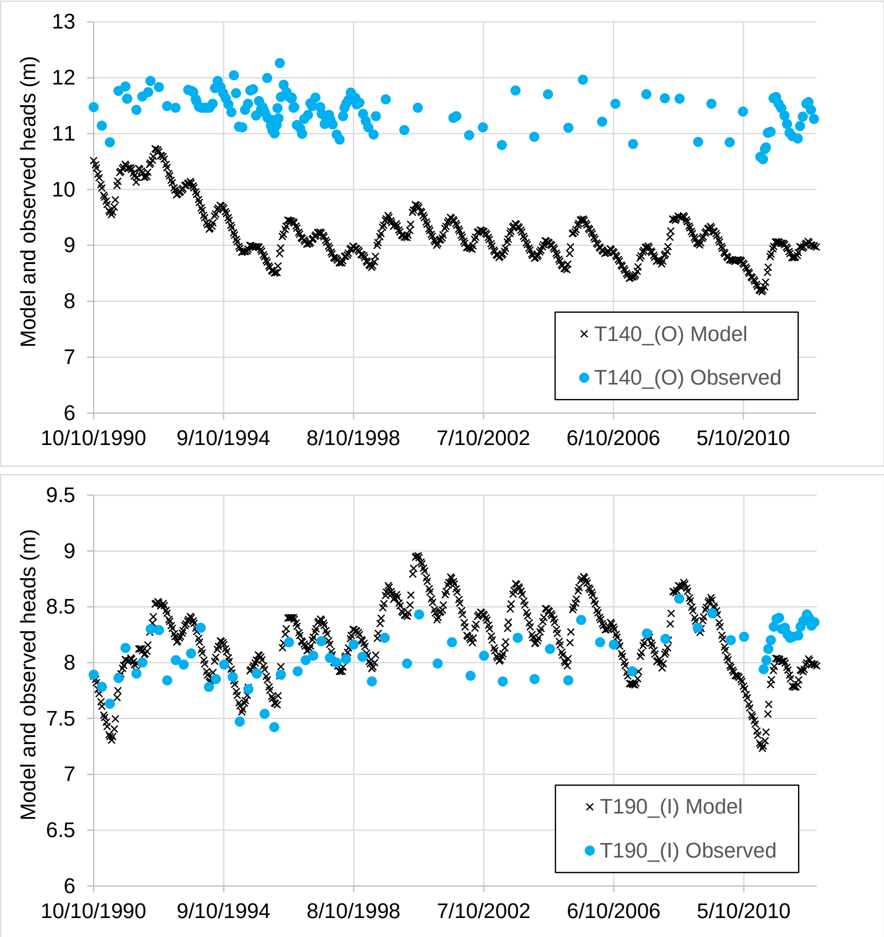

Figure 8.13 Modelled and observed hydraulic heads for wells T140 and T190

8.3.5 CALIBRATION STATISTICS

The average NRMS was 4.4% over the entire modelled period. This was based on water level from 174 wells (total of 9168 measured data). The correlation coefficient for the calculated versus observed head plot (Figure 8.14) is 0.99. The statistics versus time plot (Figure 8.15) shows varying NRMS over the modelled period with a slight reduction toward the end of the simulated period.

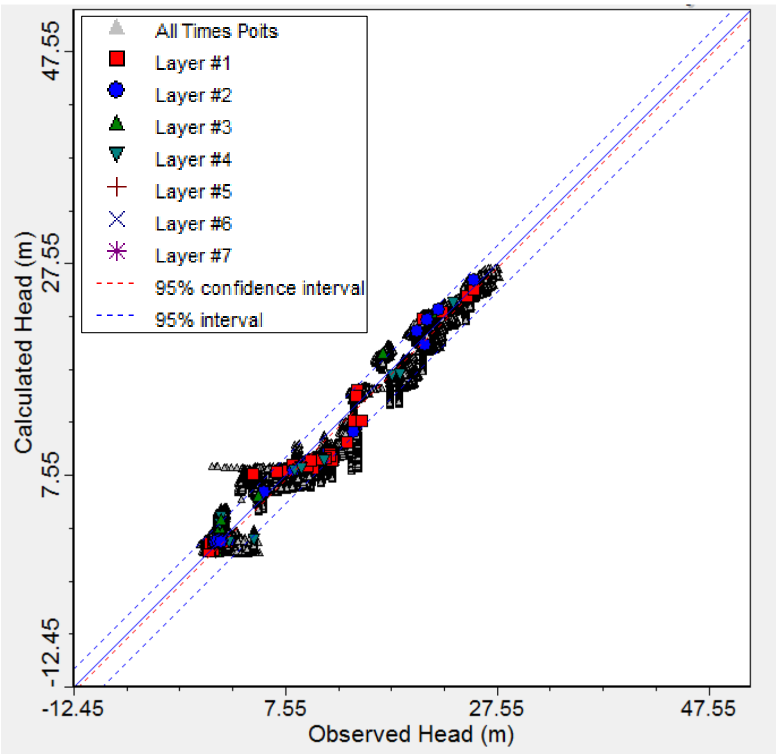

Figure 8.14 Calculated versus observed head plot. The coloured symbols refer to the data for end of the simulated period (end of 2012). Calibration was for the Superficial Aquifer represented by Layer 1, 2 and 3 in the model

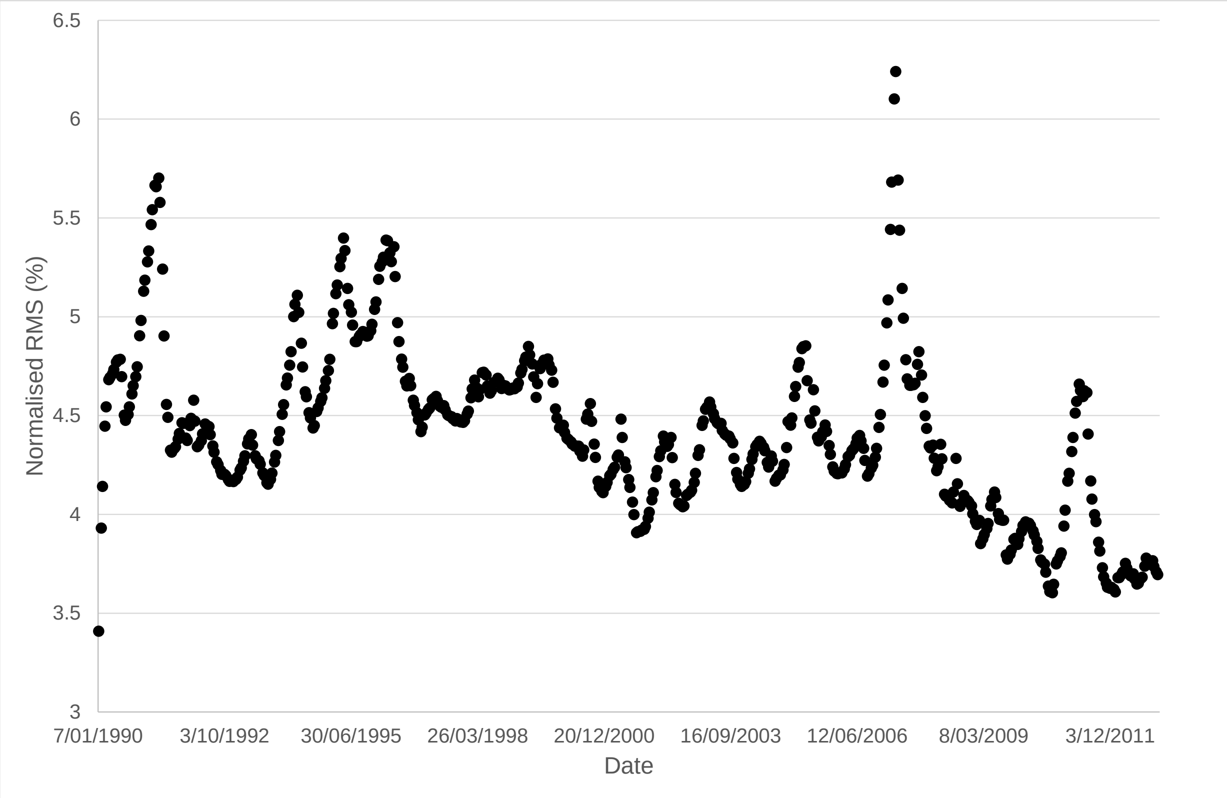

Figure 8.15 Normalised root mean squared results from calibration. The average NRMS was 4.4%

8.3.6 CALIBRATED MODEL PARAMETERS

Horizontal hydraulic conductivity values in the Superficial Aquifer (Layers 1, 2 and 3) and Rockingham Aquifer (Layer 4) were the only hydraulic property adjusted during calibration. The vertical hydraulic conductivity was assigned a value one-tenth the horizontal hydraulic conductivity, storage and specific yield were based on estimates from PRAMS (Table 8.3).

There was no adjustment of the hydraulic head boundary conditions during calibration, but there were adjustments to recharge. It was not valid to assume that recharge estimates derived from the Peel-Harvey VFM were applicable. This may be due to the need for a more localised relationship between rainfall and maximum recharge (i.e. the linear relationship derived in Dawes (2009) derived specifically for the study area; see Section 8.2.4). Similarly, this may be needed for the recharge reduction factors that were used (Table 8.4). The calibration of recharge involved incrementally adjusting a multiplier of the recharge factors to minimise differences between simulated and measured groundwater levels. A multiplier of 1.5 produced the best calibrated model.

Maps showing the zones of calibrated hydraulic conductivity are given in Appendix A. The ranges of the calibrated values are consistent with estimates from previous modelling investigations and based on measured data in this area (Table 8.3; Figure 8.3).

8.4 Model limitations and uncertainties

The hydraulic conductivity of the Superficial Aquifer is highly uncertain as indicated by other models of the coastal area in the study area: horizontal hydraulic conductivity for the Superficial Aquifer is 85 m/day (PRAMS3.2), 20 to 100 m/day (Marillier et al. 2012a), and 1660 to 3000 m/day (Nield

1999; 2004). Groundwater data voluntarily provided by industry was used for calibration, but there remain large gaps between clusters of wells used for calibration. The spatial extent of highly conductive areas remains uncertain without more data.

The model does not account for fine-scale aquifer heterogeneity due to the approach of grouping the stratigraphy into model layers. An area that would particularly benefit from refinement in the stratigraphy is the coastal area of seawater intrusion. The basal silty layer in the Safety Bay Sand is thought to influence the geometry of the salt wedge along the coast. This would require geological mapping and resources beyond the scope of this project.

Since historical records of groundwater abstraction are uncertain, the model relies on a synthesized dataset of abstraction based on licensed allocation. As discussed in Section 8.2.6, this has been a common problem encountered by others and different methods have been used by other modellers to hind-cast groundwater abstraction for the Swan Coastal Plain.

Moreover, the amount of unlicensed abstraction was highly uncertain and this component was neglected in the model. This approach should be reasonably accurate for regional flow modelling. There may be less accuracy in localised areas that have a high density of residential properties (e.g. areas southwest of The Spectacles and near East Rockingham based on land use from remote sensing; Figure 8.6), particularly if there is heavy abstraction by private garden bores; however, increasingly, smaller residential blocks are being developed with less green space.

Calibration for Layers 5 to 8 (Leederville aquifer and overlying confining layer) was not conducted as it was presumed that the calibrated hydraulic conductivities from PRAMS 3.2 were suitable. As the development of the model for this study did not focus on these deeper layers, it would not be suitable for simulating MAR injection scenarios for the Leederville aquifer.

The model assumes freshwater density for groundwater and wastewater. This may underestimate the watertable elevation in coastal areas with seawater intrusion due to buoyancy effects.

For the simulation of groundwater response to MAR in the future, the model has several limitations. It does not included future projected rates of abstraction and it assumes a dry 2030 climate which is very similar to that of the past decade. The model could be run with other climate projections; however, it was felt that a dry climate for the future is most likely. Also, the dry climate scenario provides insight into the potential groundwater response to an increase in groundwater supply. A median or wet climate scenario would predict a rising watertable to elevations that are closer to the ground surface than the dry scenario, and lead to more opportunities for surface expression of the watertable.

While the model can simulate surface water in terms of when the watertable intersects the ground surface, it is not suitable for investigating complex groundwater-surface water interactions or surface drains such as those which connect The Spectacles with Bollard Bulrush Swamp. This limits the utility of the model for simulating changes in areal extent and depth of water in wetlands.

8.5 Submarine groundwater discharge of water and nutrients

Output from the groundwater flow model was used to estimate submarine groundwater discharge and nutrient discharge to the Indian Ocean, and the location of the seawater interface. This section describes the methods used in these applications, but the results are presented in Chapter 10.

To estimate submarine groundwater discharge along the coast, an analysis using the subregional water budget package, Zone Budget in MODFLOW was conducted (Harbaugh 1990). A zone is a subregion of the model for which Zone Budget calculates a water budget. The grid cells adjacent to the coastline in Layers 1, 2 and 3 (Superficial Aquifer) were selected individually as separate zones, whereas grid cells west of these in the Indian Ocean were grouped into a single zone (Ocean zone).

This zonation was used because water budget data along the coastline were needed at a grid scale level for estimating nutrient discharge along sections of the coast as described below. Inflows and outflows from the zones were analysed to determine the total volume of discharge based on the sum of outflows from coastline grid cells to the Ocean zone. These outputs were collated for different times during the simulation period and the projected period as discussed in Chapter 10.

To estimate the loads of total nitrogen (TN) and total phosphorus (TP) to Cockburn Sound due to SGD a spatially distributed groundwater concentration was multiplied by the SGD derived from the model. Smith et al. (2003) reported spatially distributed groundwater concentrations based on submarine porewater analyses. These point data were linearly interpolated between sampling locations to assign concentration values to each cell in the model grid. The concentrations were then multiplied by the corresponding SGD in each cell to provide a TN and TP load and aggregated to determine the total loads to Cockburn Sound.

Another application of the groundwater model results was to estimate the location of the salt water interface using the Ghyben-Herzberg approximation. The Ghyben-Herzberg method predicts the position of the SWI based on the density difference between freshwater and seawater and the elevation of the watertable. It assumes a sharp interface exists between freshwater and saltwater, hydrostatic conditions within the aquifer and that the thickness of the freshwater zone is zero at the shore where the watertable has a zero elevation (Figure 8.16). This approximation typically over-estimates the inland extend of the SWI position because it assumes no flow through the aquifer (Aitchison et al. 2003). If a hydraulic gradient discharging groundwater to sea exists, then the SWI will be displaced toward the sea (Smith et al. 2005). In a freshwater aquifer it predicts that the depth (z) to an abrupt seawater-freshwater interface below sea level will be approximately forty times the height of the groundwater elevation above mean sea level (h). This assumes a density of freshwater of 1.0 g/cm^3 and a density of seawater of 1.025 g/cm^3. The mathematical formula for the Ghyben-Herzberg approximation is

$$z = \frac{\rho_f}{(\rho_s - \rho_f)} h = 40h$$

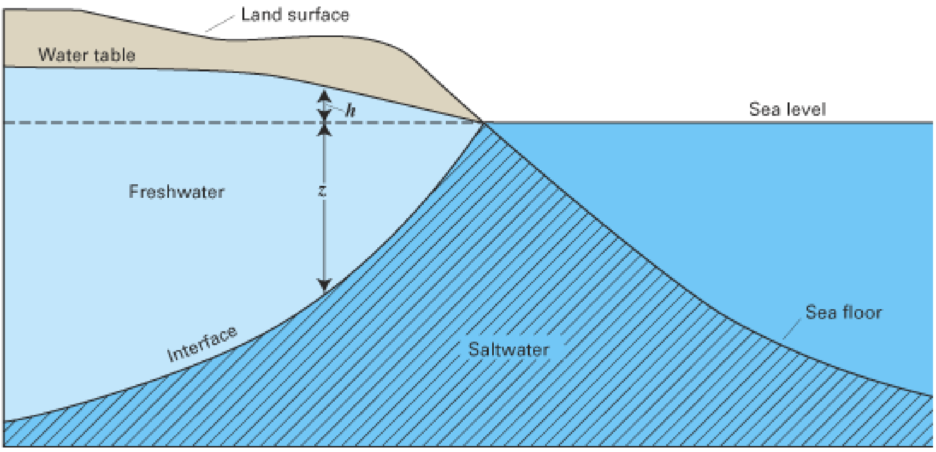

Figure 8.16 The freshwater-saltwater interface along a coastline. Reprinted from "Ground water in freshwater-saltwater environments of the Atlantic Coast," by P. Barlow (2003), U.S. Geological Survey Circular 1262. Public domain image

The location of the 'toe' of the salt water wedge (i.e. the point where the salt water interface intersects the base of the aquifer) was calculated using the elevation of the base of Layer 4.

8.6 Discussion

This chapter describes the development of a groundwater flow model based on a conceptual model for the hydrogeology that applies boundary conditions and approximations for recharge and groundwater abstraction based on a number of different datasets. Groundwater models have been developed by previous modellers for sub-sections and overlapping portions of the study area; however, the best option was to use data files from PRAMS, supplemented with additional data from previous modelling and local catchment-scale water level data for calibration. The distribution of hydrograph data is clustered in some areas, but sparse in between. As indicated in Figure 8.9, there is a tight clustering of wells near several industries that contributed data near the coastline, another tight clustering near Kwinana WWTP, a spread of wells around the Jandakot Mound area, but fairly sparse coverage elsewhere. In particular, the areas east of the Jandakot Mound and near Rockingham have virtually no calibration data; thus, the model relies on the hydraulic properties from the PRAMS model in these areas. The results from scenario modelling in these areas should be regarded in this context. More site-specific hydrogeological characterisation studies, additional data from industries in these areas, and pilot studies of MAR will greatly add to our knowledge of these systems.

There are limitations and uncertainties inherent in the model. Thus, the model should be used in view of these constraints when considering the results of scenario testing and analysis of groundwater response to MAR in the Cockburn Sound Catchment study area.

A decision was made to use a future projection of climate based on the assumption of continual drying as it resembles the trend in recent decades. While this assumption could be criticised for being conservatively dry, this approach provides insight into the potential groundwater response to an increase in groundwater supply. A median or wet climate scenario would predict a rising watertable to elevations that are closer to the ground surface than the dry scenario, and lead to more opportunities for surface expression of the watertable. The model could be run with other climate projections; however, it was felt that a dry climate for the future is most likely.

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9 Constraints and opportunities for managed aquifer recharge in the Cockburn Sound catchment

Authors: Mike Donn, Sorada Tapsuwan and Don McFarlane

Key findings

- The main constraints to infiltrating treated wastewater to the Superficial Aquifer are increasing the risk of eutrophication to wetlands and Cockburn Sound (where past nitrogen additions have affected seagrass); mobilising contaminated sites; the cost of accessing and possibly further treating the treated wastewater; and the cost of transporting it across land that contains transport, pipe and communications infrastructure
- There are also opportunities to recover drying wetlands to raise environmental, social and economic benefits; providing a low cost non-potable water supply which would benefit industry, employment and providing more options to manage contaminated sites and seawater intrusion
- Successfully overcoming the constraints in the Cockburn Sound catchment could result in a wider application of managed aquifer recharge along the western coast of Australia where there are increasing problems associated with groundwater level decline, the loss of throughflow wetlands and seawater intrusion.

9.1 Introduction

The development of managed aquifer recharge (MAR) schemes in the Cockburn Sound catchment has the potential to provide an additional water source for industry and other groundwater users, support wetlands in a drying climate and the management of seawater intrusion and possibly contaminated sites. However the infiltration of treated wastewater (TWW) into the Superficial Aquifer is not without its risks and constraints which include the impacts of additional nutrients on water quality in the groundwater, wetlands and Cockburn Sound; engineering constraints associated with the transport of TWW and the siting and operation of MAR; impacts on groundwater users including mobilising contaminated sites and the potential impact to human health associated with TWW.

These constraint and opportunities for MAR are discussed in general for the Cockburn Sound catchment in this chapter. More detailed discussion for individual MAR sites tested is provided in Chapter 10.

9.2 Environmental

9.2.1 WETLANDS

There are a large number of wetlands classified as conservation category wetlands having high ecological values under the geomorphic wetland of the Swan Coastal Plain classification (Department of Parks and Wildlife, 2014) in the study area (Figure 2.6). Addition of TWW through MAR has the potential to impact the water quality and hence ecological function within these wetlands. The salinity of the TWW is unlikely to constrain the infiltration in the vicinity of wetlands within the study area since it falls within the ANZECC trigger values for wetlands in South-west Australia (Chapter 5). In addition the salinity of wetlands in the study area is generally higher than observed in the TWW (Figure 9.1), suggesting that adverse impacts related to increased salinization of wetlands within the study area are unlikely. Wetlands of the western chain closer to the coast, such as Lake Coogee, Brownman Swamp, Lake Mt Brown and Long Swamp have higher salinities than the eastern chain of wetlands (Arnold, 1990; Davis et al., 1993). Inputs of fresher water into these wetlands from MAR may potentially impact the ecological assemblages that have developed as a result of the higher salinity which (Davis et al., 1993) indicated was related to formation from near shore oceanic lagoons at times of higher sea level. Infiltrating water of too different a salinity runs a risk of the water types not mixing with native groundwater creating salinity stratification between the surface and base of the aquifer. Evaporation of water from throughflow wetlands already results in a slightly denser plume descending to the base of the down-gradient aquifer e.g. The Spectacles (Section 6.4) and Thomson Lake (Turner and Townley, 2006). Future potable water reuse plans at the Woodman Point WWTP could increase the salinity of TWW in the SDOOL (Roman Harasymow pers. comm. 2015).

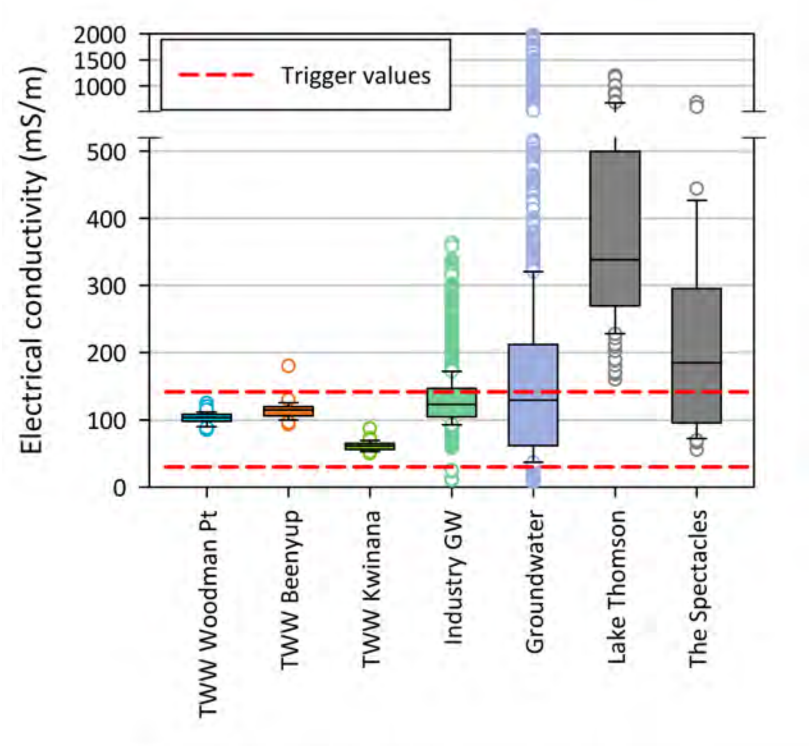


Figure 9.1 Comparison of electrical conductivity between treated wastewater (TWW), groundwater (industrial abstraction and general) and two major wetlands. Box plots show 25th, 50th and 75th percentiles (box and line), 10th and 90th percentiles (error bars) and outliers (<10th and >90th percentiles).

The major threat to the health of the wetlands is the high nutrient concentrations in the TWW and the potential alteration of the wetland nutrient balance should nutrients derived from wastewater enter the wetlands. Therefore this constrains the placement of MAR sites relative to the wetlands with the greatest impact likely when TWW is infiltrated immediately up-gradient of a wetland. Generally, due to the east to west flow of the regional groundwater, MAR sites are most appropriately located on the western side of the wetlands. However due to the local mounding of groundwater a suitable buffer should also be determined between the MAR site and the wetland of interest to prevent eastward migration of the TWW. Chapter 6 examined the risk of infiltrating 1.8 GL/y of treated wastewater about 400 m down gradient of The Spectacles lakes. In this case the infiltrated water is unlikely to have entered the lake and an upgrade to the treatment plant has resulted in wastewater having a lower total nitrogen concentration (median 4.5 mg/L) than exists in the lake because of mineralising organic matter (6 to 10 mg/L).

The location of the wetlands within the inter-dunal swales in the study area may limit the location of MAR sites if the stance was taken that a MAR site could not be located up-gradient of a wetland. However other factors such as the natural nutrient removal processes (e.g. denitrification (Gerritse et al., 1990; Salama et al., 2001), phosphorus sorption (Bekele et al., 2011)) and mixing of the TWW with the ambient groundwater would mitigate the risk to wetlands given sufficient residence time prior to encountering a wetland.

While groundwater resources in general are enhanced by MAR, the localised mounding of the watertable beneath MAR sites alters the flow of the regional groundwater in the vicinity. In a static system the groundwater flow created from enhanced infiltration will result in a roughly symmetrical flow in all directions. However when superimposed on a regional groundwater flow system, groundwater up-gradient is forced to flow around the localised area of high watertable (groundwater mound). Due to the increased tortuosity of the groundwater flow path (greater travel time) groundwater levels up-gradient of the infiltration zone rise disproportionally to those down-gradient. As is shown later, the area of groundwater rise can be many times larger than the area that receives infiltrated water. Therefore if a wetland were located up-gradient of a MAR site there is the potential for surface water levels within the wetland to be increased relative to the case when there is no additional recharge. While actual water levels in the wetland may not rise the degree of decline may be reduced.

Using a groundwater model, McFarlane et al. (2009) showed that this could be achieved for Perry Lakes at an infiltration rate of 4 m/d. While the infiltrating area and rates would differ in different hydrogeological settings the concept would still apply in the Cockburn Sound catchment. Modelled rises in groundwater level as a result of infiltration of up to 1.8 GL/yr at the Kwinana WWTP were shown in Chapter 8.

In a drying climate falling wetland and groundwater levels may potentially expose pyritic materials to oxygen thus resulting the formation of acid sulfate soils (Appleyard and Cook, 2009). A secondary benefit of maintaining or raising water levels in wetlands is in the prevention of exposing potentially acid sulfate soils with the release of acidity and potentially arsenic, aluminium and iron as happened in the City of Stirling (Appleyard et al. 2004).

9.2.2 NITROGEN LOADS INTO COCKBURN SOUND

The infiltration of TWW will increase the load of nutrients being delivered to the groundwater. The nutrient load associated with 4.8 ML/d TWW infiltration are shown in Table 9.1 for the three WWTPs, Woodman Point (current SDOOL quality), Beenyup (representing future upgrade of Woodman Point) and Kwinana. This load for the Kwinana TWW is approximately what is currently infiltrated at the WWTP. These loads represent the worst case scenario should mitigation and/or attenuation process fail to reduce the nutrient concentrations during migration through the aquifer to Cockburn Sound regardless of the residence time.

Nutrient reduction may take place through a number of mitigation or attenuation processes such as;

1. Interception of the groundwater plume

Due to the large amount of groundwater abstraction in the catchment there is the potential for TWW to be intercepted by existing abstraction bores, thus reducing the nutrient load that could potentially discharge to the Sound. This is similar to pump-and-treat techniques commonly employed in groundwater contaminant remediation.

2. Attenuation of nutrients along the groundwater flow path

As shown in Chapter 5 the nitrogen and phosphorus are present as different species in the TWW. These nutrient species behave differently under differing conditions.

Phosphorus, largely present as phosphate, is removed by adsorption and precipitation in calcareous sands and soils (Gerritse, 1993; Whelan, 1988; Whelan and Barrow, 1984). However the P retention capacity of the vadose zone and aquifer is finite, and as such saturation of the retention capacity and P breakthrough may occur. This was observed at the Floreat Infiltration Gallery where Bekele et al. (2011) showed that P was present in the groundwater 2.3 m down-gradient of the gallery, although 31% of the P was removed.

Attenuation of nitrogen is more complex than phosphorus, it depends upon the species on nitrogen present, the aquifer redox conditions and the availability of organic carbon. The available groundwater data (WIR database, DoW) suggest that generally the groundwater is anoxic, median dissolved oxygen 0.33 mg/L (interquartile range 0.10 to 1.32 mg/L) and contains relatively high dissolved organic carbon (median 19 mg/L; interquartile range 8.8 to 36 mg/L). These conditions are likely to result in denitrification (removal of nitrate-N) occurring while ammonium-N in the TWW is unlikely to be attenuated unless it is converted to nitrate before entering the groundwater. This is reflected in the persistence of ammonium plumes in the industrial areas (Trefry et al., 2006). However due to the high nitrogen inputs in the horticultural areas to the east of Lake Coogee the potential for aquifers to remove nitrate through denitrification is exceeded resulting in high nitrate concentrations (Department of Water, 2010).

The attenuation of both phosphorus and nitrogen before discharging to the Sound also increases with the residence time within the aquifer. Therefore MAR sites further inland pose less of a threat to water quality in Cockburn Sound.

Table 9.1 Treated wastewater nutrient loads to groundwater were 4.8 ML/d to be infiltrated and the concentration corresponding to the 50th and 95th percentiles for three wastewater treatment plants

NUTRIENT LOAD		WOODMAN POINT		BEENYUP		KWINANA	
		50th percentile	95th percentile[1]	50th percentile	95th percentile	50th percentile	95th percentile
Total P	t/yr	7.7	18.7	14.5	17.5	8.1	17.1
Total N	t/yr	25.7	50.2	30.3	37.4	9.7	16.0
NO_x-N	t/yr	13.0	21.9	23.8	31.5	2.1	4.5
NH_3-N	t/yr	7.2	21.3	0.9	3.7	1.9	8.8

[1] The 95th percentile is used to eliminate outliers and to indicate the worst case

9.2.3 MOBILISING AND TREATING POLLUTANTS

The infiltration of TWW will alter the groundwater flow field around MAR locations through the mounding of the watertable. This in turn has the potential to alter the geometry of existing contaminant plumes depending on the location relative to the MAR plume.

The two major constraints associated with exiting contaminant plumes are altering the plume flow direction such that existing groundwater abstraction bores are impacted and a general increase in the groundwater discharge to Cockburn Sound increasing pollutant loads. A knowledge of plume geometries relative to existing groundwater users and the strategic placement of MAR sites could mitigate the former. However due to the greater spatial impact of MAR on groundwater levels increased groundwater discharge to Cockburn Sound is expected over a larger area with associated increase in pollutant load.

Conversely through the appropriate placement of MAR sites may benefit groundwater users through the stabilisation of contaminant plumes by creating a barrier to plume movement and a reduction of impacts relating to abstraction induced movement of the plumes. Additionally, where pump-and-treat schemes are limited by seawater intrusion, increasing recharge up-gradient would result in increased abstraction rates and hence increasing remediation of contaminants. At least one member of the KIC sees MAR as a means of accelerating the remediation of a contaminated site through providing additional water, if not carbon to accelerate *in-situ* microbial breakdown.

9.3 Engineering

The engineering constraints and opportunities outlined below were informed by two reports produced by GHD on *Industrial Water Supply Options Assessment* and *MAR Infrastructure Concepts and Cost Estimates* as part of the project (GHD, 2015a; 2015b). Discussion and conclusions from these reports are included in this summary of engineering constraints and opportunities. For further information please refer to the reports.

9.3.1 ACCESSING THE TREATED WASTEWATER

The sizes of typical MAR schemes investigated as part of this project were 1.75 GL/yr and 3.50 GL/yr which is equivalent to, and double, the Kwinana WWTP current TWW disposal rates, respectively. If MAR was to solely meet the projected water demand for heavy industry of an increase by around 15 GL/yr by 2031 (Figure 4.1) and assuming 100% recovery, then 9 and 5 such MAR schemes, respectively would be required. Due to the high volumes of TWW disposed of through the Sepia Depression Ocean Outlet Landline (SDOOL), currently 58.4 GL/yr and predicted increases; TWW sourced from the SDOOL is unlikely to limit the implementation of MAR. However, if the Kwinana WWTP were the source the additional TWW volumes are much less, increasing from 0.2 GL/yr in 2013 to 2.5 GL/yr in 2032. This may limit the viability of MAR sites which rely solely on the Kwinana WWTP for water supply.

Access to TWW from the SDOOL is limited to existing section valves which are between 1.6 and 3.1 km apart (Figure 9.1). In order to hydraulically separate the MAR water supply from the SDOOL as required by the Water Corporation a 'break tank' is required (GHD, 2015b). This requires construction of a wet well near to the section valve and separate pumping infrastructure to move TWW to the MAR site.

For TWW sourced from the Kwinana WWTP, the engineering assessment assumes that all pipework for MAR will originate from the WWTP itself. While connection to the SDOOL and the constraints associated with accessing the TWW are less, depending on the MAR site location pipe networks required to convey the TWW may be long.

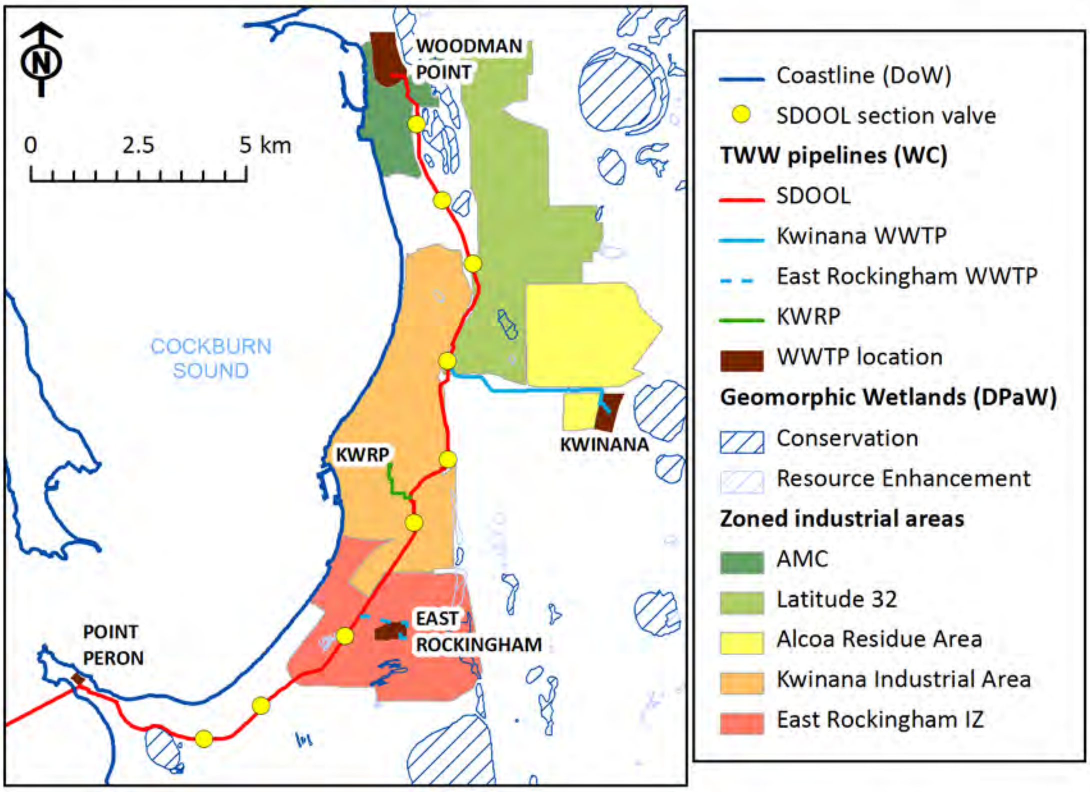


Figure 9.2 Section valve locations along the Sepia Depression Ocean Outlet Landline (SDOOL) indicating the limited locations where TWW could be accessed

9.3.2 PRE-TREATMENT OF TREATED WASTEWATER

The requirement for pre-treatment of the TWW depends on the quality of the source water (Woodman Point WWTP or Kwinana WWTP) as discussed in Chapter 5. Pre-treatment is not required to adjust the salinity and pH of the TWW for either the purposes of protecting the environmental values (data falls within the ANZECC trigger values) or abstraction for industrial use (within the groundwater salinity and pH range currently used by industry).

Suspended solids in the MAR source water potentially limits the infiltration capacity due to physical clogging that occurs when the suspended solids are filtered out in the base of the basin or gallery used for infiltration. To achieve low to moderate rates of clogging during aquifer storage and recovery (deep well injection), the MAR guidelines recommend a total suspended solid (TSS) concentration of <10 mg/L (NRMMC-EPHC-NHMRC, 2009). Further to this the median TSS design target of <6 mg/L was adopted during the conceptual infiltration gallery design of the Perry Lakes managed aquifer recharge project (GHD, 2011). The TWW from the Kwinana WWTP meets these targets with a median TSS concentration of 5.6 mg/L (2010 to 2013). However current TSS in the TWW from the Woodman Point WWTP (median 8.0 mg/L) and indicative future TSS based on Beenyup WWTP (median 15 mg/L) indicate that additional filtration is required before the water from the SDOOL would be suitable for infiltration unless infiltration is done via a recharge basin which can be easily cleaned as currently occurs in WWTPs on the Perth Basin which don't have access to an ocean outfall. Infiltration galleries will require solid reduction however.

Given the need for filtration of TWW from the SDOOL provision for disposal of the backwash water is also required at gallery installations. This is a potential constraint for the development of a MAR site and associated infrastructure where the SDOOL is utilised as the source water. Project costs and land take are likely to be impacted depending on the method of disposal, e.g. pipeline to the nearest sewer connection.

Depending upon the degree of natural attenuation provided by the aquifer, or interception by pumping bores, nitrogen concentrations may be required to be reduced prior to infiltration. Wastewater from the Kwinana and East Rockingham WWTPs uses an oxidation ditch method which reduces Total N concentrations to a median of 4.5 mg/L and these waters may not pose a problem, depending on ambient values in the wetland, aquifer or Sound. A Moving Bed Biofilm Reactor nitrogen removal process proposed by GHD (2015b) is proven mature technology utilised as a tertiary treatment option in WWTPs. Thus it is unlikely to prove to be a constraint to the operation of a MAR scheme, though it represents an additional cost and requires supplementary carbon dosing for the biological denitrification process.

While general observation can be made based on regional groundwater quality regarding the potential of the aquifer to remove nitrogen and phosphorus, further evaluation of the removal capacity should be undertaken at the pre-feasibility stage of MAR scheme development.

9.3.3 EASEMENTS, PIPES AND GALLERIES/BASINS

For TWW to be delivered to a MAR site a pipeline is required to connect it to the access point (SDOOL section valve or the Kwinana WWTP). Section valves allow a simplified offtake from bypass pipework and alleviates the need to hot tap the SDOOL itself. The locations where a MAR pipeline could be run may be limited by the availability of suitable reserves/easements and the existing infrastructure that may need to be crossed. Existing and proposed infrastructure include:

- Roads (current and proposed) and associated drainage infrastructure
- Railway tracks and associated infrastructure
- Overhead and underground powerlines
- Oil and high pressure gas pipelines
- Sewer and water mains
- ALCOA pipelines and infrastructure connecting the refinery to the residue area

MAR sites located further from the TWW access point will incur greater costs not only based on the length of the pipe network and associated pumping costs, but also potentially from increased need to cross existing infrastructure. However this may be off-set by the environmental benefits that may be provided if for example TWW from the SDOOL is piped further east this enables greater residence time before reaching Cockburn Sound and results in a greater spatial extent rise in groundwater level potentially influencing wetland water levels and continued/greater access for groundwater users, e.g. industry and local governments.

Two options were considered for infiltrating the TWW; underground galleries and open basins. They both have their advantages and disadvantages (Table 9.2). In areas that are remote from residential areas then recharge basins are more likely to be used since the negatives associated with aesthetics and potentially public health risks outlined in Table 9.2 become less of an issue. While land take is similar in terms of the footprint of both options, buried infiltration galleries may be offset by other passive uses such as recreation since access would only be limited during period of remediation due to clogging.

Table 9.2 A comparison of the relative benefits of recharge basins verses buried infiltration galleries compiled by GHD (Table 3, 2015b). ✓ - Preferred, ✗ - Non-preferred

CONSIDERATION	OPEN RECHARGE BASINS	BURIED GALLERIES
Footprint	- (dependent on design infiltration rate, potentially similar)	- (dependent on design infiltration rate, potentially similar)
CAPEX	✓ (lower cost option, particularly if use of recharge basins obviates need for tertiary filtration facilities)	✗
OPEX	✓ (potentially lower OPEX due to ease of access to the infiltration surface but dependent on the frequency of solids removal)	✗ (potentially higher, depending on frequency at which galleries need to be "re-built" to remove clogged layer at/below infiltration surface)
Aesthetics	✗ (recharge basins are unsightly and would need to be fenced, and there is potential for nuisance odours if algal blooms occur)	✓
Public health risk	✗ (risk of human contact greater by virtue of open water body, plus potential for nuisance mosquito breeding in the basins)	✓
Failure risk	✓ (proven system, long term performance is well understood)	✗ (there is a risk that the infiltration rate could over time reduce to an unacceptable level, requiring the galleries to be "re-built")
Potential for pre-infiltration WQ deterioration	✗ (potential for growth of algae in the open water body, and subsequent clogging of the infiltration surface with algal solids)	✓ (no sunlight therefore no potential for algal growth, however this does not prevent bacterial biofilm growth which also induces clogging)

9.3.4 HOW MAR WATER COMPARES WITH ALTERNATIVES

GHD (2015a) examined MAR in comparison with alternative water sources for industry in their report entitled *Kwinana Managed Aquifer Recharge Study: Industrial Water Supply Options Assessment*. GHD made an assessment of the current and future water demand in the Western Trade Coast area which contain the major industrial areas shown in Figure 2.1 and discussed in Chapter 4. The following conclusions were made regarding water demand and potential for different sources to meet the future water requirements for industrial water uses.

- By 2031 the industrial water demand is expected to be approximately 45 GL/yr, which is approximately 17 GL/yr higher than the current demand.
- At present, industry abstracts approximately 17 GL/yr of groundwater from local aquifers to meet their water demands. The balance of the current water demand (approximately 11

GL/yr) is met with low TDS recycled water supplied from the Kwinana Water Reclamation Plant (KWRP), scheme water and a range of other sources including on-site stormwater runoff.
- Based on supply-demand considerations, cost and/or environmental considerations; neither scheme water, seawater nor stormwater are likely to be viable sources that could be used to meet significant new industrial water demands.
- Two potentially viable sources able to meet significant new industrial water demands are TWW sourced from the SDOOL, or a combination of TWW sourced from the SDOOL and groundwater sourced from the Superficial Aquifer. In the longer term groundwater is not considered an option in its own right as there is not sufficient groundwater available to meet industrial water demand increases in the longer term.
- Noting the limitations that apply to the preliminary order of magnitude cost estimates documented by GHD (2015a), of the five water supply options assessed, the calculated comparative unit water costs indicate that the least cost supply option is i) abstraction of groundwater, followed by ii) indirect reuse of TWW following infiltration by a recharge basin type MAR scheme, iii) indirect reuse of TWW following infiltration by a infiltration gallery type MAR scheme and iv) decentralised direct reuse of TWW). The highest cost option by a significant margin was the option pertaining to centralised treatment and distribution of high quality recycled water.

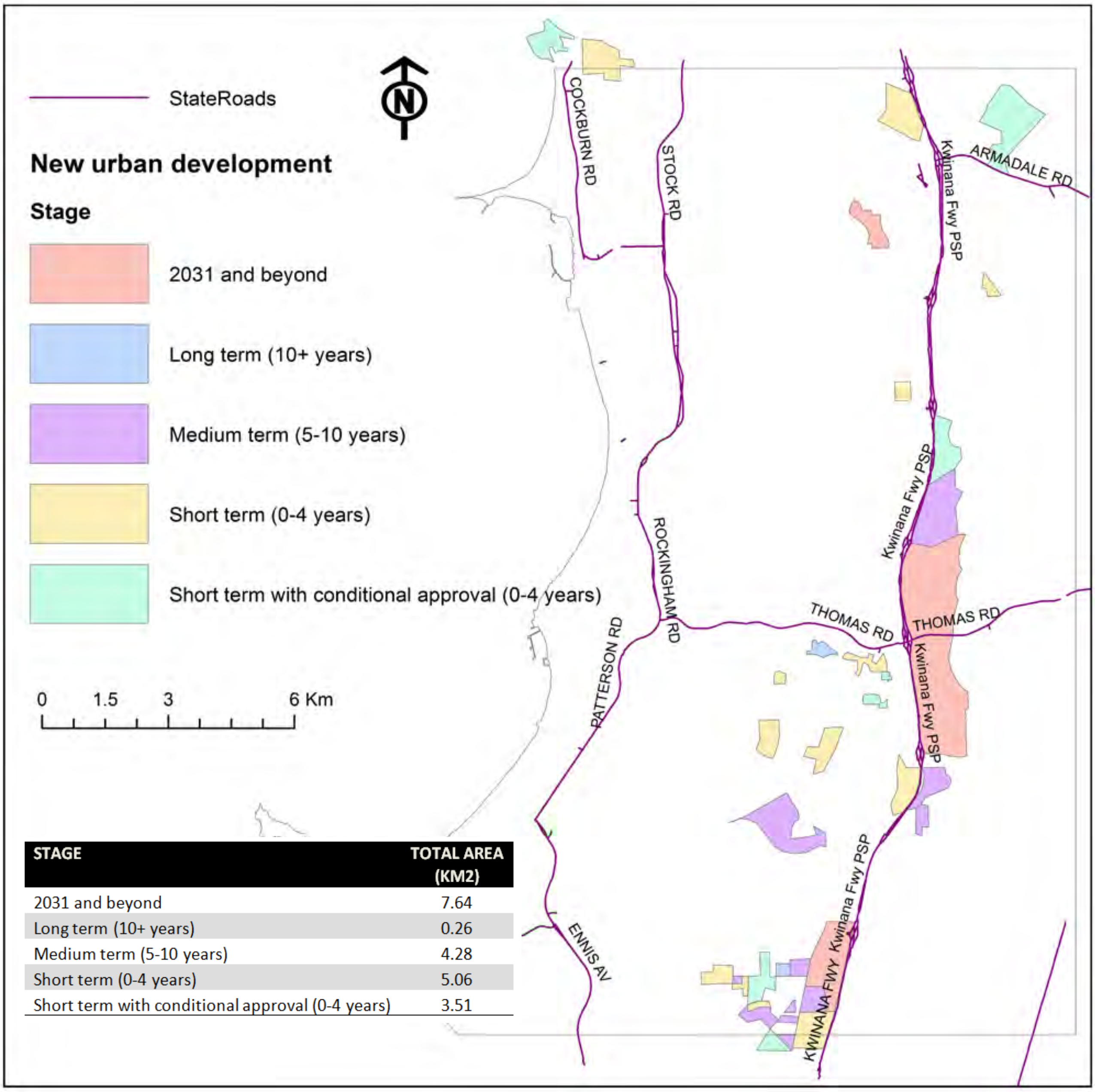
Thus given the constraints on the availability of groundwater as indicated in Chapter 4, the assessment of engineering costs to supply water to industry through indirect reuse of TWW following MAR is a viable option.

9.4 Economics

9.4.1 NON-POTABLE WATER SUPPLIES OF INDUSTRIES, LGAS, HORTICULTURALISTS AND RESIDENTS

Water supply reliability is a key factor that promotes the continuous growth of industrial activities in the study area. However, confined aquifers in this area are either too saline or fully allocated, and as mentioned in Chapter 4 and above there is very limited amount of water is available from the Superficial Aquifer for abstraction (Department of Water, 2013).

In addition to the limited groundwater supply situation, there is growing demand for water through the natural increase in population growth. More new suburbs are being built in the study area (Figure 9.2). Each suburb has green public open space areas that require water for irrigation. Since these areas cater for many individuals, the social value of public open space is quite high as illustrated by (see e.g. for studies estimating the value of green space, Hatton MacDonald et al., 2010; Mahmoudi et al., 2013). It is likely that the 'value' of water used for irrigating public open space could be higher than the value of water to industrial areas, rendering industries less competitive if water is going to be allocated to the highest value user in the future. However, most of the new urban development planned in the study area may not be within the same groundwater catchment as the industrial area, therefore, competition for water with urban users may not be an issue.



STAGE	TOTAL AREA (KM2)
2031 and beyond	7.64
Long term (10+ years)	0.26
Medium term (5-10 years)	4.28
Short term (0-4 years)	5.06
Short term with conditional approval (0-4 years)	3.51

Figure 9.3 Map of various stages of planned new urban development

There are also benefits of using groundwater to maintain wetlands and groundwater dependent ecosystems. Wetlands have been shown to have high aesthetic value to surrounding households (see e.g. Tapsuwan et al., 2009; Tapsuwan et al., 2012). To retain these values water levels may need to be protected from further reduction. In order to ensure that wetlands are protected or maintained, large amounts of groundwater abstraction need to be monitored or controlled to prevent potential environmental effects, including terrestrialisation (i.e. wetlands taken over by vegetation) of wetlands due to prolonged dropping of groundwater levels. Some of these restrictions are already currently in place in the study area, for example, shallow groundwater abstraction for industrial use is prohibitive in the Rockingham Industrial Zone due to environmental approval conditions EPBC 2010/5337 (Hyd2o, 2013). Chapter 7 of this report presents the value of wetlands in the study area in more detail.

Another long term user of groundwater in the study area is agriculture. However, growing demand for urban and industrial space is pushing agricultural activities out of the study area. There is a proposal to replace agricultural with industrial activities in Latitude 32, which will free up 2GL of water currently allocated for irrigation. This water may become available for industrial supply in the future (Department of Water, 2013).

When resources, such as water, become limited, the choice of whom to allocate water to becomes a critical social challenge. Even though industries employ people in their businesses, thus offering a form of social benefit, the extent of public benefit from using water to irrigate public open space and maintain wetlands may also be high and benefit a large amount of people. As such, industries could face these environmental users as a competing user for water in the future. The choice to allocate water could be driven by market forces i.e. water allocation is given to the highest value user first, or regulatory instruments could be used to ensure equitable distribution of the much needed water supply.

9.5 Seawater intrusion

For groundwater users that require a source of freshwater, seawater intrusion poses a constraint on groundwater use close to the coast. As discussed in Section 3.2 the seawater interface (SWI) extends up to 2 km inland from the coast in the Superficial Aquifer. Groundwater abstraction is also likely to influence seawater intrusion locally as upconing may be induced (Department of Water, 2007; Ivkovic et al., 2012). Climate-driven decreases in groundwater levels and falling hydraulic gradients (Ivkovic et al., 2012) and the impacts of abstraction along the Cockburn Sound coast have the potential to extend the seawater interface landward, thus impacting existing bores and resulting in abandonment and limitations on new abstraction within the groundwater allocation limits.

The rise in groundwater level associated with MAR has the potential to influence the location of the SWI by increasing the hydraulic gradients near the coast provided that the increase groundwater abstraction is less than the increase in recharge. Thus MAR may prevent the landward movement of the SWI and potentially reverse it. Similar schemes using well injection are used to manage SWI along the Californian coast (Los Angeles County Public Works Department, 2015). However, in the case of the Cockburn Sound catchment a balance would be required between controlling the position of the SWI and managing nutrient export to Cockburn Sound. SWI is currently limiting the rate that contaminated sites can be pumped and treated so MAR may assist the cleanup of the aquifer.

9.6 Human health

The main risks to human health associated with the infiltration of TWW in a MAR scheme arise from microbial hazards (NRMMC-EPHC-NHMRC, 2006). In Western Australia, the Operational policy on MAR states that “MAR proposals may require Department of Health approval under the *Health Act 1991*, if the proposed end use of the recharged water has the potential to affect human health” (Department of Water, 2011). The main contact pathways are associated with contact with the TWW during infiltration (e.g. in recharge basins), or through surface expression or recovery of the infiltrated TWW. While it is assumed that controls on access to infiltration sites will limit that exposure route, greater care is required to assess the potential risks of surface inundation associated with groundwater level rise, the risk to existing groundwater users and the installation of new recovery bores.

Investigation of the groundwater use close to proposed MAR sites can indicate whether current abstraction bores are likely to be impacted by MAR and hence whether the potential health risks need to be assessed. Groundwater modelling may indicate both the expected travel path of the infiltrated TWW as well as the potential for inundation to occur as a result of groundwater level rise. However to fully assess the microbial risks to human health local scale investigations are required to,

- Assess the microbial hazards associated with the source water as well as those present in background groundwater, and
- The ability of the aquifer to act as a treatment barrier to remove microbial hazards.

This is emphasised by investigations undertaken at the Floreat Infiltration Galleries (FIG), a MAR study site. At this site it was observed that microbial indicators were present in the background groundwater (Bekele et al., 2009), which may potentially influence the assessment of microbial removal. A microbial pathogen risk assessment conducted for the FIG site also found that the recovered water did not meet the Australian Guidelines for Recycled Water when used for some private green space irrigation scenarios (Toze et al., 2010). The risk assessment was influenced by the local hydrogeology and residence time of the TWW in the aquifer. It was found that the most important factor influencing the risk was the pathogen numbers in the TWW. This emphasises the need for validation and verification of the aquifer residence time and the subsequent potential of the aquifer to remove microbial pathogens at new MAR sites. There is the potential to use existing TWW disposal sites, such as the Kwinana WWTP to provide further evidence to support microbial risk assessments. For industrial uses greater control on exposure and risk is possible due to the section of low exposure uses, training of staff in risk management with these uses, and excluding or limiting public access to open space uses of TWW.

9.7 Proof-of-concept for other areas

As reviewed in Chapter 4, the possibility of adding treated wastewater to address falling groundwater levels on the Swan Coastal Plain has been around since the Perth Urban Water Balance study (Cargeeg et al. 1987). Cumulative Difference from the Mean (CDFM) plots using Perth rainfall has indicated that groundwater levels in the Gnangara and Jandakot Mounds may have peaked in the late 1960s (Yesertener 2008) meaning that groundwater levels have been falling in some urban and peri-urban areas for over 45 years. The loss of valuable wetlands has been widespread since this time.

Seawater intrusion emerged as a problem in the late 1970s and increasing seasonal groundwater salinity has been an ongoing problem in suburbs in the Mosman Peninsula ever since. Seawater intrusion is an increasing problem in coastal and estuarine areas between Broome and Busselton so any success in reversing it using treated wastewater to create a barrier as is done in the USA and Israel amongst other parts of the world may have wide Australian applicability.

The Kwinana area has a number of advantages for testing MAR using treated wastewater:

1. There is almost 60 GL per annum of treated wastewater going to the Sepia Depression Ocean outfall and groundwater modelling in this study shows that this volume is more than enough to greatly increase groundwater levels if added to the aquifer;
2. There is a pipeline that takes this water past the area most in need of MAR
3. The area has a heavy industry zoning so that infiltration could occur away from residential areas
4. No groundwater is extracted for human consumption over the catchment apart from the up-gradient Jandakot Mound and there is little to no extraction for residential garden irrigation
5. The area has an emerging shortage of non-potable water for industrial use
6. The area has a well monitored set of groundwater bores and several local groundwater models used to manage contaminated sites in the area
7. Treated wastewater has been safely added to the Superficial Aquifer since 1975 at the Kwinana WWTP so there is a site that can be used as a surrogate for other additions. This

site is especially propitious because it has taken place close to a Conservation Category Wetland which appears to have been prevented from drying as have other wetlands not located near a disposal site

8. Scientific evaluations of MAR have taken place in similar hydrogeological settings (especially at the Floreat Infiltration Galleries and Halls Head)
9. The area has well-documented saltwater intrusion problem in the Superficial Aquifer since the 1970s

Therefore the benefits that may accrue from the trial of MAR in this catchment could be far wider and these need to be considered when assessing the cost benefit analysis of a specific MAR project in the study area.

9.8 Discussion

Given that the project is a regional assessment of the potential for MAR covering an area of 307 km^2, appropriate site selection plays an important role in the mitigation of the various constraints described above. However there are some trade-offs between the different types of constraints when assessing the suitability of sites, for example costs accrued as a result of connections between the TWW source and the MAR site may be unavoidable in order to protect environmental assets such wetlands and the Cockburn Sound from unacceptable increases in nutrient loads. The availability of TWW for MAR may be limited if sourced solely from the Kwinana WWTP which has the best water quality, while pre-treatment of the TWW from the SDOOL may be required prior to MAR to manage the higher suspended solid loads and clogging potential depending on the infiltration method that is used.

Further discussion of the constraints and opportunities specific to individual sites selected for groundwater modelling and economic analysis is included in Chapter 10.

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10 Managed Aquifer Recharge scenarios

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Key findings

- A consultative process was used to select locations for testing of managed aquifer recharge (MAR) scenarios utilising several iterations of groundwater modelling and assessment of the environmental and engineering constraints. A total of six locations with ten scenarios were run in the final phase of modelling.
- Due to the regional nature of the study the risks of introducing MAR in the study area are only expressed semi-quantitatively. Site selection has a strong impact on the final outcome of the risk analysis and was incorporated in the initial site selection process.
- Generally inland sites (E1, E2 and E3) and northern site (N) have lower risks especially related to increasing nutrient loads entering Cockburn Sound due to groundwater interception and smaller changes to submarine groundwater recharge rates.
- Site selection largely mitigates impacts on wetlands however impacts are likely to be greater for Site E3 which shows likely impacts on Long Swamp, and potentially Site S2 although the wetlands impacted are not Conservation Category and may be impacted by the golf course
- At the sites and infiltration rates chosen the surface expression of treated wastewater does not appear to be likely
- Engineering constraints are largely accounted for in the benefit-cost analyses allowing for costs of piping treated wastewater and provision for different levels of pre-treatment
- As a water source all MAR sites and options are economically favourable over alternative water sources (KWRP and scheme water) within the constraints of the modelling
- All MAR scenarios had a benefit cost ratio that exceeded 1 except where it was assumed that 30 or 40% of the added water would be unavailable for use at one site where added volumes were small and piping costs were high.
- Economically-favourable options do not necessarily have the lowest risks

10.1Introduction

Previous chapters have detailed the characteristics of the study area, and the opportunities and constraints that face managed aquifer recharge to provide non-potable water for heavy industry (and potential other users) and reverse seawater intrusion. This chapter consolidates this material so as to identify promising sites for further investigation.

A three-stage process involving three workshops with Steering Committee members and key experts was used to short-list the most prospective sites:

1. An evaluation of how groundwater levels throughout the study area may respond to additions of treated wastewater was carried out to assess aquifer responsiveness. The impact of adding MAR volumes equivalent to that already occurring at the Kwinana WWTP (i.e. 1.7 GL/yr or 1 m/d through the ponds) and of twice that amount was modelled at

eleven sites. The responses to the addition of MAR were compared to the future projection of the 'Business as Usual' (BAU) case. The BAU case reflects the current (2012) abstraction rates and the disposal of 4.7 ML/d of treated wastewater (TWW) at the Kwinana WWTP site. The same future climate scenario that was applied to the MAR scenarios was applied to the BAU scenario. This assessment also looked at the path TWW may take away from these indicative sites (particle tracking) to see whether the added water would enter downstream wetlands or enter Cockburn Sound after only a short period. The sites were located along two north-south lines; one close to the SDOOL and therefore the Kwinana Industries Area, and another inland at the same distance from the Sound as the Kwinana WWTP (Figure 10.1 left panel). As explained in the next section, this first workshop prioritised six sites for a more detailed assessment – 2, 3, 4, 5, 7 and 9.

2. More detailed groundwater modelling, including evaluating the potential impact of MAR on saltwater intrusion and site constraints were considered in a second workshop to reduce the six potential sites to locations around three sites; North (near site 2), South (between sites 4 and 5) and East (site 9). When these were examined in more detail, and discussed with nearby industries, these three sites were expanded to include six sub-options outlined below (Figure 10.1 right panel):

    a. North (N) = just north of Site 2 to test impact of MAR on known area of saltwater intrusion;

    b. South 1 (S1) = near the railway reserve between Sites 4 and 5. Moved to vacant land, close to industry abstractors but short travel time to Cockburn Sound;

    c. South 2 (S2) = on the Kwinana Golf Course, to test the effects of shifting MAR inland on travel time to Cockburn Sound and the potential for broader coverage of industry abstractors;

    d. East 1 (E1) = Kwinana WWTP, increased infiltration rates at current location;

    e. East 2 (E2) = excess water from the KWWTP being infiltrated onto the Medina Agricultural Research Station, in case East 1 was not viable; and

    f. East 3 (E3) = excess water from the KWWTP being infiltrated on ALCOA land near their tailings facilities, in case East 1 was not viable.

    Note that the Kwinana WWTP is operating as for the BAU scenario in addition to the 'new' MAR infiltration for all the above scenarios except for East 1, where infiltration rates increase according to TWW availability.

3. A third workshop examined detailed modelling and costings of MAR options for these six locations, the details being provided towards the end of this chapter.

Because the process may have application in other MAR siting investigations, or could be improved in some way, a summary of each stage is provided below.

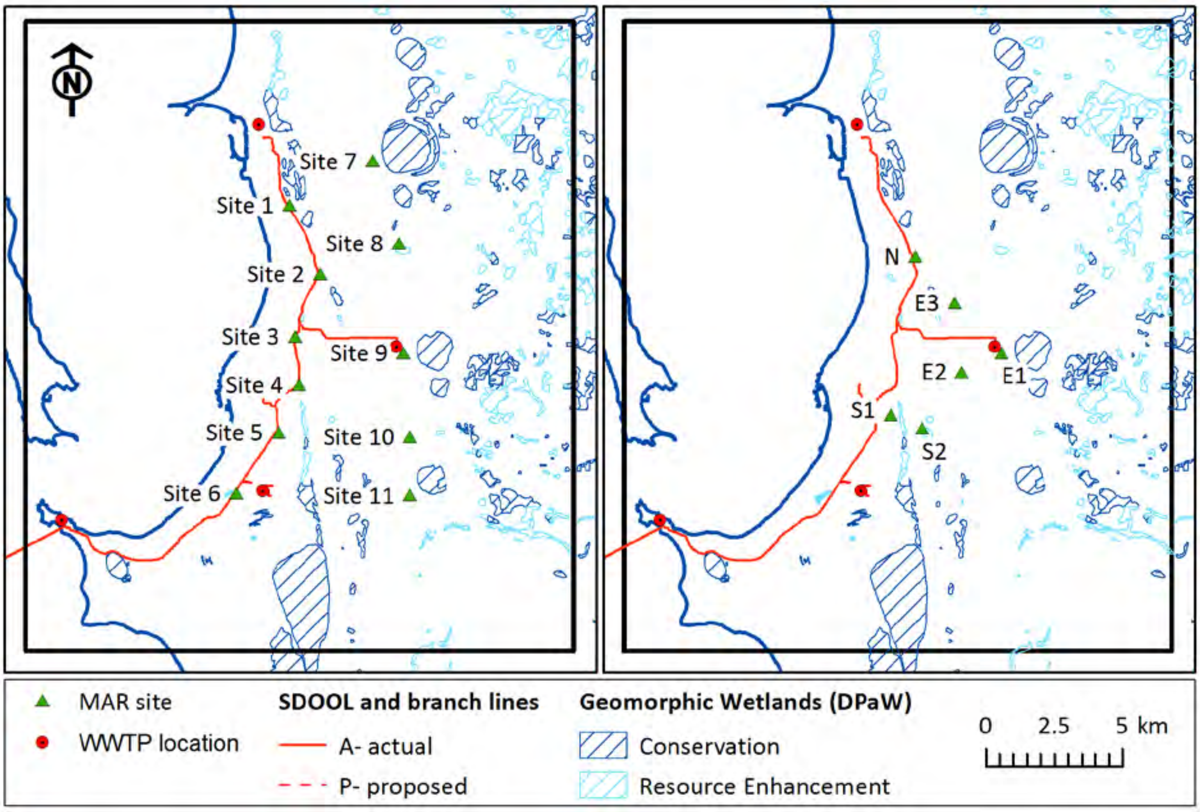

Figure 10.1 Location of eleven initial sites (left) and six final locations (right) used for assessing the most prospective areas for managed aquifer recharge

10.2 MAR site selection process

10.2.1 REDUCING THE POTENTIAL MAR SITES FROM ELEVEN TO SIX

The eleven potential sites for MAR were selected to provide a reasonable spatial coverage of the study area with which to test the impacts of MAR on groundwater levels and provide an initial indication of TWW plume extent/travel direction. No consideration was given specifically to land use, environmental assets or other factors that may impact on site selection at this stage. Six sites were within Tamala Limestone and Safety Bay Sand aquifers close to the SDOOL and five were 4 to 6 km inland in less transmissive sandy aquifers.

Groundwater flow model results were presented for each of these sites operating individually at either 4.8 or 9.6 ML per day. In addition, results from combinations of sites were presented, namely pairing of sites 1 and 7; sites 3 and 9; and sites 5 and 11; all 6 coastal sites along the SDOOL, all 5 inland sites, and all of the sites operating simultaneously.

Table 10.1 shows the wastewater volumes that were applied in these model runs. To provide a frame of reference, the simulated MAR volumes were compared with the volume of wastewater discharged via the SDOOL as this would potentially be the source water for MAR. For example, operation of all 11 sites at 9.6 ML/d (2 m/d) of infiltration would recharge the aquifer with 76% of the wastewater currently discharged via the SDOOL and nearly half of the projected wastewater discharge for 2032. In contrast, the operation of only one MAR site would use only 3.4% of the wastewater currently discharged via the SDOOL and 2.1% of the projected wastewater discharge for 2032.

Table 10.1 Volumes of wastewater modelled in simulations of MAR in the CSC presented at the 21 August 2014 workshop. These are shown relative to the SDOOL discharge volumes as a percent.

SCENARIO	1 m/day (4.8 ML/day)			2 m/day (9.6 ML/day)		
	VOLUME (GL/yr)	PERCENT 2013	PERCENT 2032	VOLUME (GL/yr)	PERCENT 2013	PERCENT 2032
1 site (individual)	1.7	3.4	2.1	3.5	6.9	4.3
2 sites (paired)	3.5	6.9	4.3	7.0	13.7	8.6
5 sites (inland)	8.7	17.1	10.7	17.5	34.3	21.4
6 sites (coastal)	10.5	20.6	12.9	20.9	41.2	25.7
11 sites (all sites)	19.2	37.7	23.6	38.4	75.5	47.1

The model results showed changes in groundwater levels relative to a ‘Business As Usual’ (BAU) case which may include a gradual fall in regional levels as the climate becomes slightly drier. The BAU assumes the current operation of the KWWTP infiltrating at 4.8 ML/d. An example of the effect of adding 4.8 and 9.6ML/d (1.7 and 3.5 GL/yr) at site 2 on levels is shown in Figure 10.2 which is just east of the ALCOA Refinery. The rise in groundwater levels is asymmetric with the greatest rises being up-gradient (to the east of the infiltration point) because groundwater flow is restricted from discharging to Cockburn Sound. This is because the mounds acts like an underground dam with inflow water backing up behind the mound ‘wall’. At 9.6 ML/d the impact could extend as far east as Thomson Lake, a Ramsar-listed wetland, and The Spectacle lakes. MAR closer to these lakes (e.g. sites 7 and 9 respectively) would have much greater impacts of course. Details of all simulations can be found in Appendix A.

Figure 10.3 shows predicted spatial extents for the difference in hydraulic head (greater than 15 cm) between the MAR scenario and the BAU scenario in 2032. These simulations were run separately (i.e. only on site at a time) but have been superimposed here for comparison purposes.

The largest impacts of MAR are predicted for four of the inland sites, and in the north for the coastal sites, which reflects the relative transmissivities of the aquifer (it being much higher in the south west). Large transmissivities prevent a groundwater mound from forming due to the greater dissipation of water. These early screening runs used a version of the model that was later recalibrated so that impacts in the southwest are now projected to be slightly greater than in this version. However the relativities remained the same so the selection of sites was not affected by this later refinement.

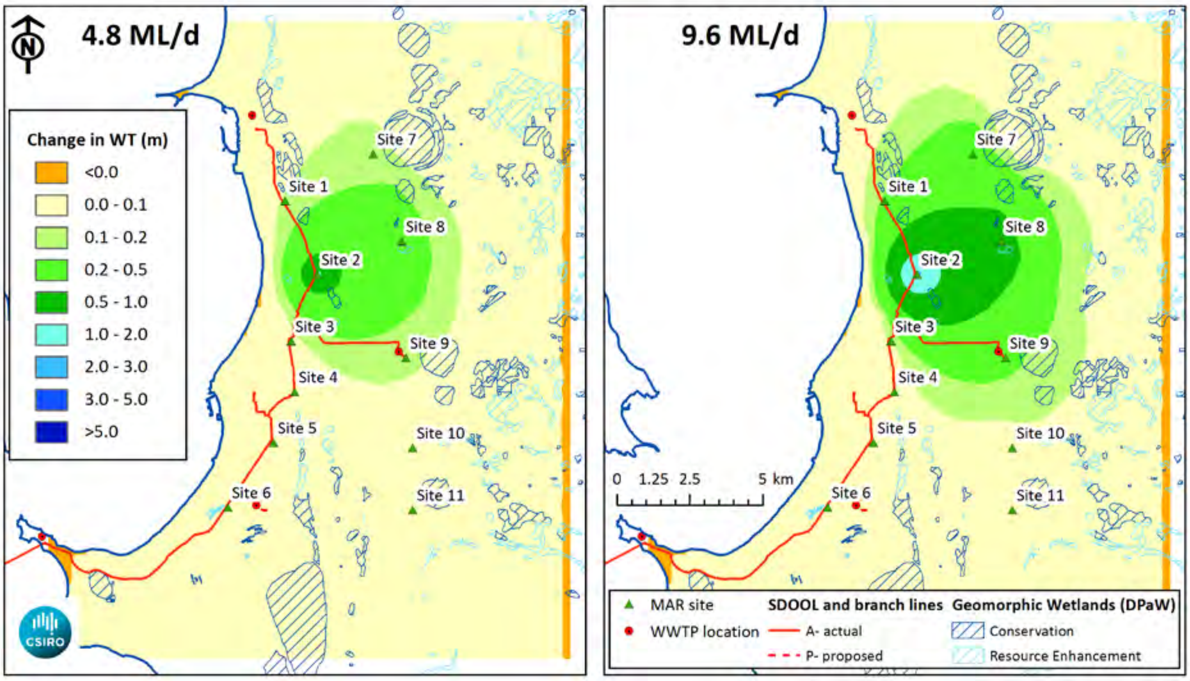

Figure 10.2 Predicted changes in the watertable (WT in metres) at site 2 computed based on the difference in hydraulic heads for 2032 less those for the BAU scenario for 2032.

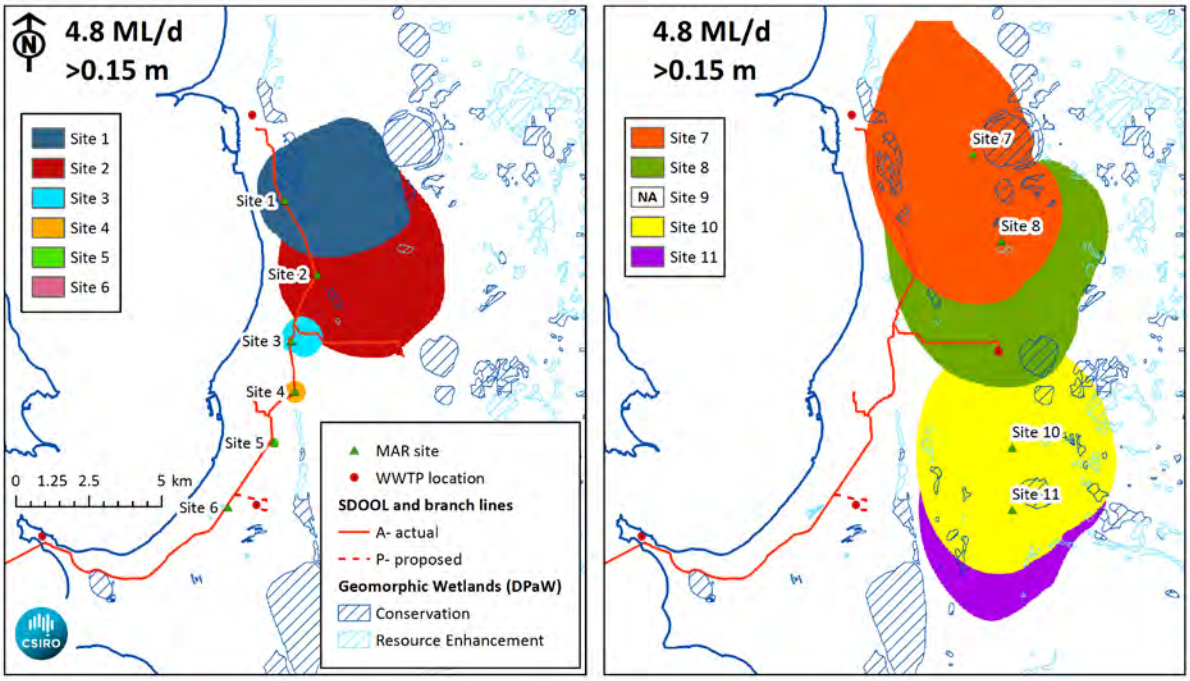

Figure 10.3 Predicted spatial extent for 2032 for the difference in hydraulic head greater than 15 cm between the MAR scenario and the BAU scenario. For clarity, model results are shown in two separate images. The sites shown in the images were modelled separated, but are plotted together here for comparison. There is no change for site 9 as it is BAU.

MAR at multiple sites leads to cumulative impacts and much greater mounding across a larger area as is shown by the combination of all inland sites at 4.8 ML/d (understanding that this is BAU for the central site 9, thus the limited groundwater response) and 9.6 ML/d (Figure 10.4). The 4.8 ML/d would utilise about 17% of current available wastewater or 11% of the water available in 2032 (Table 10.1). It would raise groundwater levels in the inland chain of Beeliar lakes by more than 1m compared with the BAU case. Such a scenario would however require wastewater to be pumped inland from the SDOOL (and/or East Rockingham WWTP) because there is insufficient water at the Kwinana WWTP.

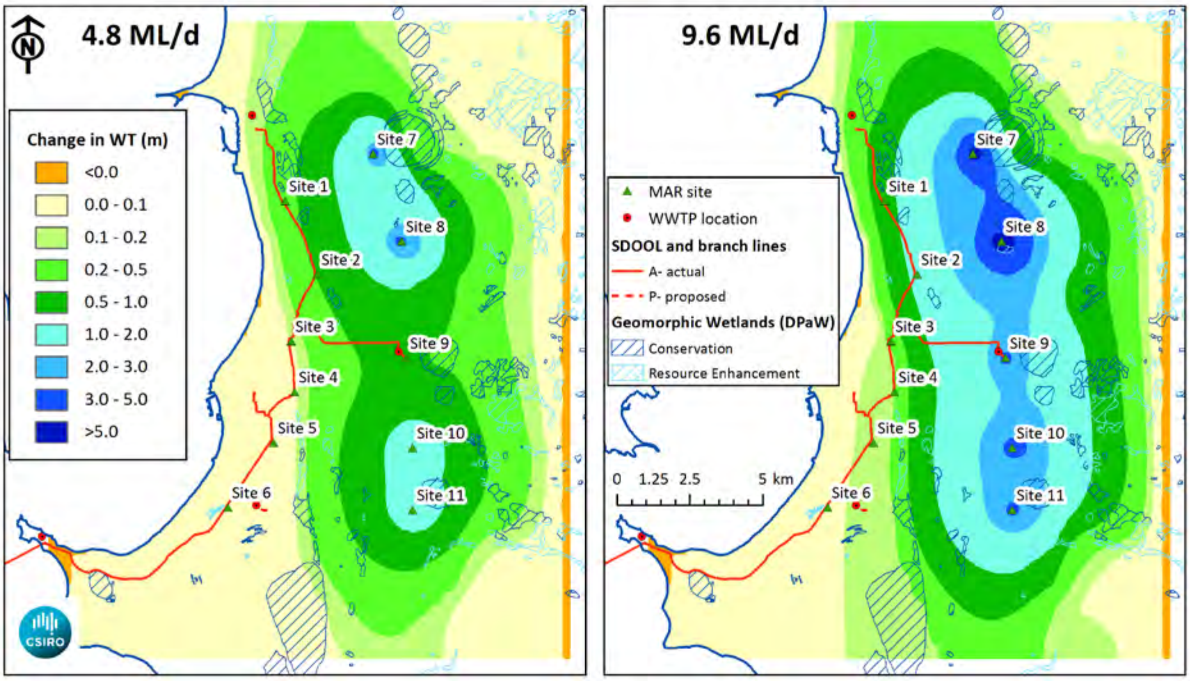

Figure 10.4 Impact on groundwater levels of adding 4.8 ML/day in the four inland site (left) and at 9.6 ML/day (right) which increases MAR at the Kwinana WWTP from 4.7 to 9.6 ML/day.

An analysis of particle tracking pathways was conducted (Figure 10.5). The early version of the model predicts that constituent particles transported by advection in groundwater should reach the coast for the 6 sites coastal situated near the SDOOL by 2032 but none of the inland sites would reach Cockburn Sound. Later analyses using the improved groundwater model gave similar results but also showed that particles can be intercepted by large pumping bores and may therefore not travel far from the site of infiltration. The maximum aerial extent of particle pathways or spread in map view is influenced by variations in hydraulic conductivity. In areas of high hydraulic conductivity, a narrow spread of pathways develops. The model predicts limited up-gradient advective transport and travel distances (e.g. distances are usually less than a few hundred metres).

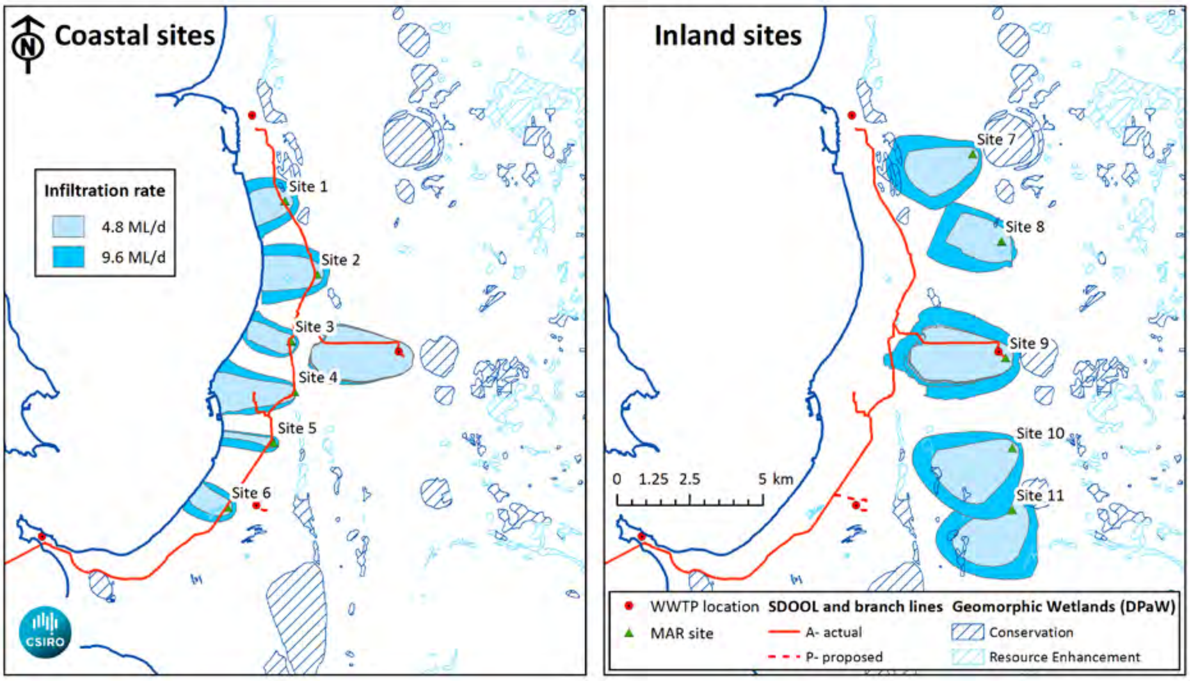

Figure 10.5 The maximum areal extent of particle tracking pathways computed by MODPATH over the 20 year period, 2013 to 2032. Coastal sites have a shorter travel time due to as particles are transported to Cockburn Sound. MAR scenarios for each of the 11 sites were simulated separately, but are presented here on combined images for comparison.

An analyses of the strengths and weaknesses of MAR in the eleven sites arising from the first workshop concluded:

1. **Site 1** is too far north for industry to benefit and may cause problems for mobilising nitrogen plumes into Jervois Bay. It is close to the Beeliar Nature Reserve but adding 9.6 ML/d could marginally raise water levels in Thomson Lake
2. **Site 2** is located where the salt water intrusion in most evident and would benefit the northern part of the industrial area (near ALCOA). Travel times to the Sound range between 8 and 11 years depending on the addition rate. Projected groundwater level rises are highest at this site compared with the other coastal sites, potentially resulting in movement of the salt water wedge towards the coast.
3. **Site 3** would benefit industry around the Kwinana Power Station and possibly reduce salt water intrusion to the north. Travel times are between 2 and 4 years. Projected groundwater rises are much lower at this site compared with sites 1 and 2.
4. **Site 4** would benefit industry; particle tracking shows it would extend under the BP refinery site. Travel times range between 5 and 7 years. Projected groundwater levels are lower again compared with Site 3.
5. **Site 5** would benefit industry in the CSBP area although groundwater rises would be very modest given the high transmissivity of the aquifer in this area. Travel times range between 5 and 7 years.
6. **Site 6** is too far south to benefit industry. Groundwater level rises are negligible and travel times range between 3 and 5 years. Water could be sourced from the East Rockingham WWTP which will use oxidative ditch methods which produce low nitrogen levels. Being a conservation reserve there is a requirement that Water Corporation not infiltrate treated

wastewater on site. This has resulted in disposal to the ocean via the SDOOL. Volumes in this WWTP will increase substantially in future.

7. **Site 7** has the potential to raise groundwater levels in Lake Thomson and the related Beeliar chain of lakes. Over time it could also add groundwater to the Henderson / Jervois area. The nearest industry is Cockburn Cement which may benefit depending on their licenses. The water may travel 2.4 to 3 km from the site by 2032.
8. **Site 8** would provide indirect benefit to the chains of lakes to its north and south, and delayed impact to industry in the northern KIC area. The water may travel 1.9 to 2.6 km from the site. This area had the greatest reduction in groundwater levels between 1990 and 2012 (possible because of horticulture).
9. **Site 9** is the Kwinana WWTP site and it was assumed that 4.8 ML/d would continue until 2032. It is about 4.5 km from the coast and infiltrated water is calculated to travel between 3.2 and 4.2 km between 2012 and 2032. Past infiltrated water has therefore almost certainly reached the KIA and Cockburn Sound having started in about 1975 and travel times not being very sensitive to volumes. Doubling infiltration at this site (if water was diverted from the SDOOL) would raise levels in The Spectacles by 0.5 to 1 m. It would also raise levels in a wooded depression immediately east of the DAFWA Medina Research Station. Its effect on site 3, if jointly operated, is to increase water for industry in the central KIA area.
10. **Site 10** is near the Kwinana townsite and therefore close to residential and council bores. It is not close to industry or important wetlands. Travel distances range between 2.8 and 3.4 km over the 20 year period.
11. **Site 11** is near Bollard Bullrush Swamp. Added water is likely to move south west towards Lake Cooloongup which is naturally saline.

It was therefore decided to concentrate most effort on sites 2, 3, 4, 5, 7 and 9.

10.2.2 REDUCING THE POTENTIAL MAR SITES FROM SIX TO THREE

Specific engineering and environmental constraints for the remaining six sites were discussed at a workshop along with more detailed groundwater analyses, especially of seawater intrusion.

Site 7 was eliminated because of potential cost of bringing treated wastewater to this inland site (although there is a rarely used stormwater pipe that passes nearby this site and the Woodman Point WWTP), and water may enter Jervoise Bay which already has a eutrophication problem. Advantages could accrue to Thomson Lake and a large cement manufacturer but it is too far north for other industries.

Site 3 was eliminated because there are few current groundwater users in this area (although new industries may require water) and the impact (positive and negative) on contaminated sites was unclear.

Sites 4 and **5** were combined and included for further work on a site between them because of large groundwater users in this area although it is accepted that any MAR would need to take account of contaminated sites in the area.

Site 2 was also included for further study because of the need to address seawater intrusion caused by extraction to treat contaminated sites amongst other things. Siting the infiltration to the east would benefit more industries because of the greater mound, increase flow time before any water reached the Sound and could improve water levels in Long Swamp.

Site 9 (Kwinana WWTP) was considered the easiest site to increase MAR if the additional water from the plant could be diverted to the existing ponds (through a change in licensed disposal conditions) or to areas adjacent to the plant.

10.2.3 PRIORITISING THE REMAINING THREE MAR SITES

Detailed engineering, cost-benefit analyses and site investigations were carried out on the remaining three sites. A new naming convention (described above) was utilised to distinguish these results from previous results in similar locations. In locating potential MAR infiltration basins, the East and South sites had alternative locations (Figure 10.1) which were separately evaluated for reasons outlined above. This brings the total number of scenarios run in this phase to ten including business as usual. A summary of the details that were examined is given in Table 10.2.

Table 10.2 Summary of details examined in final MAR site assessment

<table>
<tr><th>MAR LOCATION</th><th>SOURCE WATER</th><th>SCENARIO INFILTRATION RATES</th><th>KWINANA WWTP OPERATION</th><th>ENGINEERING ASPECTS (GHD, 2015A)</th><th>COST-BENEFIT ANALYSIS</th></tr>
<tr><td>Site N</td><td>SDOOL</td><td rowspan="3">Constant
4.8 ML/d and
9.6 ML/d</td><td rowspan="3">Continued operation at constant rate 4.7 ML/d</td><td rowspan="6">Access to TWW
Transport of TWW and pipelines necessary
Pre-treatment options for TWW (filtration and nitrogen removal)
Infiltration options (recharge basin verses infiltrate gallery)
Estimate of associated cost</td><td rowspan="6">Comparison of scenarios
Based on engineering and other cost estimates</td></tr>
<tr><td>Site S1</td><td>SDOOL</td></tr>
<tr><td>Site S2</td><td>SDOOL</td></tr>
<tr><td>Site E1</td><td>Kwinana WWTP</td><td>Increasing rate
4.9 ML/d up to
7.2 ML/d</td><td>As per scenario</td></tr>
<tr><td>Site E2</td><td>Kwinana WWTP</td><td rowspan="2">Increasing rate
(in excess from WWTP)
0.2 ML/d up to
2.5 ML/d</td><td rowspan="2">Continued operation at constant rate 4.7 ML/d</td></tr>
<tr><td>Site E3</td><td>Kwinana WWTP</td></tr>
</table>

10.3Groundwater model results

This section describes the groundwater modelling results for the projected influence of MAR infiltration over the 20 year period between 2013 and 2032. Additional information on model setup is found in Chapter 8, though it should be noted that while the climate was varied (Section 8.2.7) the groundwater abstraction was maintained at the same rate and in the same locations as utilised in the calibration period (1990 to 2012; Section 8.2.6). Similarly future changes to land cover were not modelled, thus changes to recharge resulting from land cover change could not be modelled. While access to and abstraction of 'additional' water as a result of MAR due to the regional nature of the current study it was not feasible to include these changes in the modelling. Further investigations would be required in future studies to assess the impact of additional abstraction at a more local level. Capture of the infiltrated TWW through targeted abstraction could be used to limit any potential environmental impacts.

10.3.1 GROUNDWATER MOUNDS

Each of the MAR scenarios generates a watertable mound below the site of infiltration. The major factors that influence the size and spatial extent of the mound for these scenarios are: the rate of water infiltrated to the aquifer, the hydraulic conductivity of the aquifer near the infiltration site, and the abstraction rate and proximity of pumping bores.

Figure 10.6 shows the relative change in predicted groundwater levels between each MAR scenario and the BAU (at the end of the 20 year simulation). These were computed by subtracting the computed hydraulic heads at the end of 2012 from those at the end of the 2032. In general, the larger the infiltration rate (e.g. 9.6 ML/d and 4.8 ML/d, equivalent to 1 m/d and 2 m/d), the wider the spatial extent of water table response to MAR as can be seen for the N, S1 and S2 sites that were each run with these two rates. As similarly noted in the theoretical modelling by Wu (2003) and discussed in Chapter 6, at low rates of infiltration, the mound spreads more than rises, and mound height increases under high infiltration rates. The relative mound heights predicted for N, S1 and S2 with 9.6 ML/d are about 20 to 50 cm higher than those predicted with 4.8 ML/d of infiltration (Figure 10.6).

Another factor to consider in interpreting the relative groundwater mound results is the distribution of hydraulic conductivity at the infiltration sites and surrounding these sites. The hydraulic conductivity in the Superficial Aquifer below all of the Sites was 85 m/d, except for S1 where the hydraulic conductivity was 300 m/d; however, within a short distance of some of these sites, the hydraulic conductivity is more variable. To the east and northeast of Site N beyond 1.5 km, the hydraulic conductivity of the aquifer is lower and there are zones of hydraulic conductivity oriented northwest-southeast. This causes water table mound to develop along a similar orientation. A highly transmissive aquifer will also not produce a very large groundwater mound as there is greater opportunity for lateral flow. At site S2, the same hydraulic conductivity (85 m/d) extends to the north, south, and east of this location, whereas to the west, the hydraulic conductivity is one order of magnitude larger (300 m/d). Consequently, the relative groundwater mound results show larger spread inland and to the north and south, than to the west. At site S1, the hydraulic conductivity is 300 m/d, but in close proximity (250 m to the east), the hydraulic conductivity decreases to 85 m/d. There is less mound development in the more transmissive section of the aquifer west of S1 compared with areas to the east of this site.

It should be noted that the volume of water in the groundwater mounds produced by MAR is not entirely composed of the infiltrated wastewater; it is a mixture of the recharge water and ambient groundwater. The infiltrated water has a damming effect on the ambient groundwater as it discharges generally in the direction of the Indian Ocean.

To provide insight into where the wastewater will migrate, the results from particle tracking are shown in Figure 10.6. Particles were released from the perimeter of the MAR site in each scenario, at the beginning of the projected period, and the resulting maximum areal extent (at 2032) of advection of these particles is shown. In essence, these images indicate the extent of the MAR plume within the aquifer in map view. The amount of lateral spread depends on the infiltration rate. Scenarios E2 and E3 involve gradually increasing infiltration rates based on the projected increase in the volumes of wastewater inflow to the Kwinana WWTP. The MAR for these scenarios is the excess beyond what is currently infiltrated at the Kwinana. The small amount of infiltration for these scenarios compared with the other scenarios does not generate groundwater mounds as large as the other scenarios. Moreover, the particle tracking pathways are quite focused toward abstracting bores to the northwest and west of E2 and E3, respectively, and do not spread as far laterally away from the infiltration sites as in the other scenarios. It should be noted, that the particle tracking pathways develop within a time-varying flow field. If particles were to be released at later times when higher infiltration rates were applied in E2 and E3, there would be greater lateral spread. As the particles were released at the start of 2013 when the applied infiltration rates were low, they initially migrated in response largely to the existing flow field and were not influenced by MAR. Further exploration of the advective particle travel distances at various times after release is provided in Section 10.4.

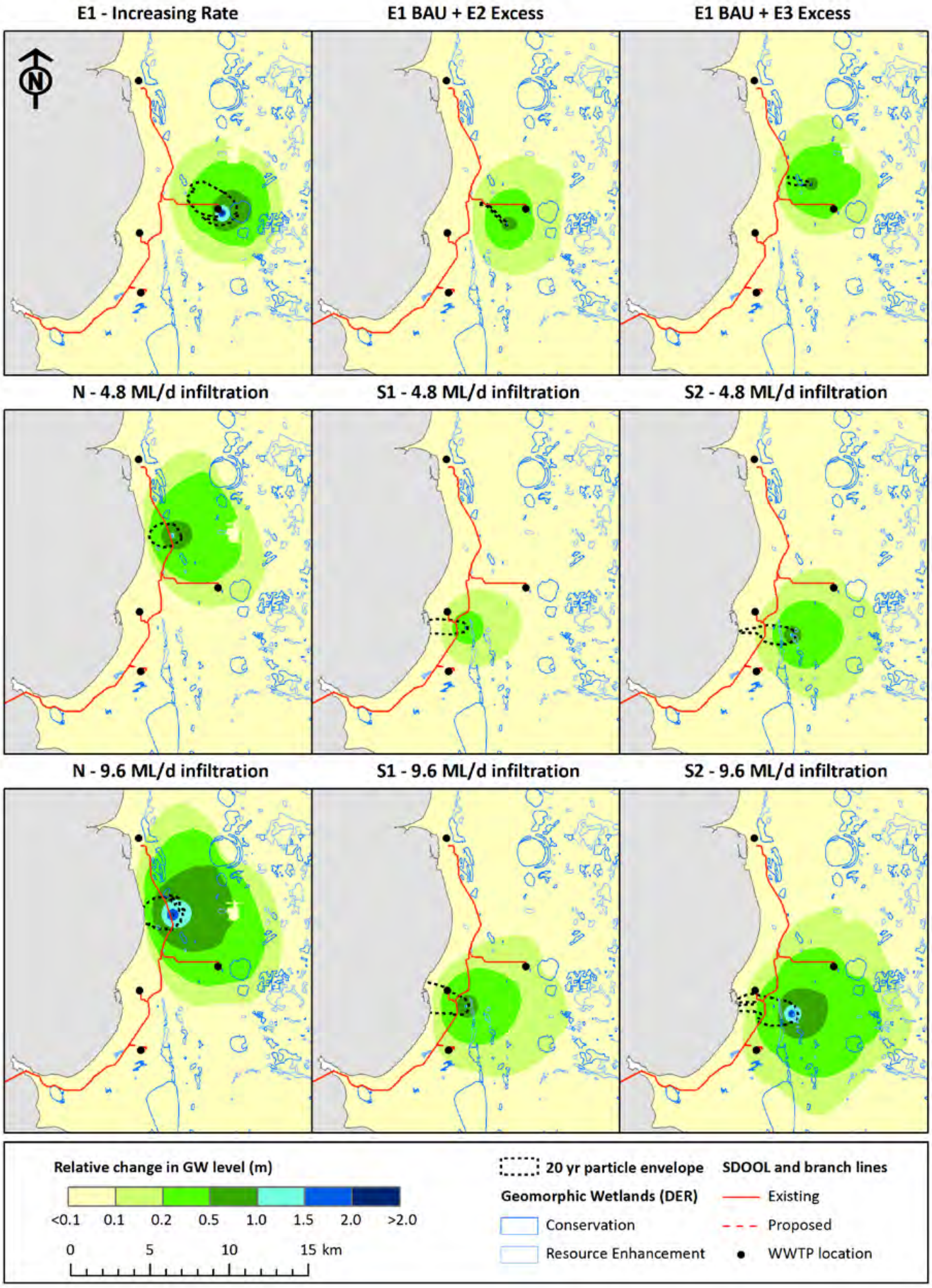

Figure 10.6 Relative change in predicted groundwater levels between each MAR scenario and the BAU (at the end of the 20 year simulation).The maximum areal extent of particle tracking pathways at the end of the 20 year simulation is superimposed on the maps.

10.3.2 SIMULATED HYDROGRAPHS NEAR MAR SITES

Figure 10.7 shows hydrographs for 'fictitious' observation wells that were used to extract data at a distance of 200 m East of the MAR site. Each of the MAR schemes were simulated separately, but the hydrographs from different models are shown together here for comparison. In particular, the BAU model hydrograph at a 'representative' observation well adjacent to each scheme is shown. The purpose of these hydrographs is to show the relative changes in groundwater level in response to the adjacent MAR scheme. The distance of the fictitious observation well relative to the MAR site was selected to be generally indicative or representative of the groundwater level response. Results from other fictitious observation wells around each site were plotted, but show the same general trends.

The predicted hydrographs for the BAU model at all of the sites shows the typical seasonal variation, but also a gradually declining trend in response to the projected dry climate scenario implemented in the model (Figure 10.7). The results for the projected period for Sites N, S1 and S2 clearly show that infiltrating a larger volume of water (i.e. 9.6 ML/d versus 4.8 ML/d, equivalent to 1 m/d and 2 m/d, respectively) results in higher groundwater levels in the Superficial Aquifer and certainly higher levels than predicted for the BAU model (Figure 10.7). The hydrographs for N, S1 and S2 (for both modelled infiltration rates), show fairly rapid equilibration within about one year. The effect of MAR at these sites appears to have a stabilizing influence on the water table, in contrast to the BAU hydrographs, which show gradual decline. The predicted hydrographs for simulated MAR at Sites E1, E2 and E3 show a delayed response to the respective infiltration rates applied in these scenarios. The hydrographs for these models track fairly closely with the hydrographs predicted for the BAU case for several years, and then deviate as larger volumes of water are infiltrated according to the scenarios (Table 10.2) based on the projected increase in TWW volumes from the Kwinana WWTP (Figure 5.2).

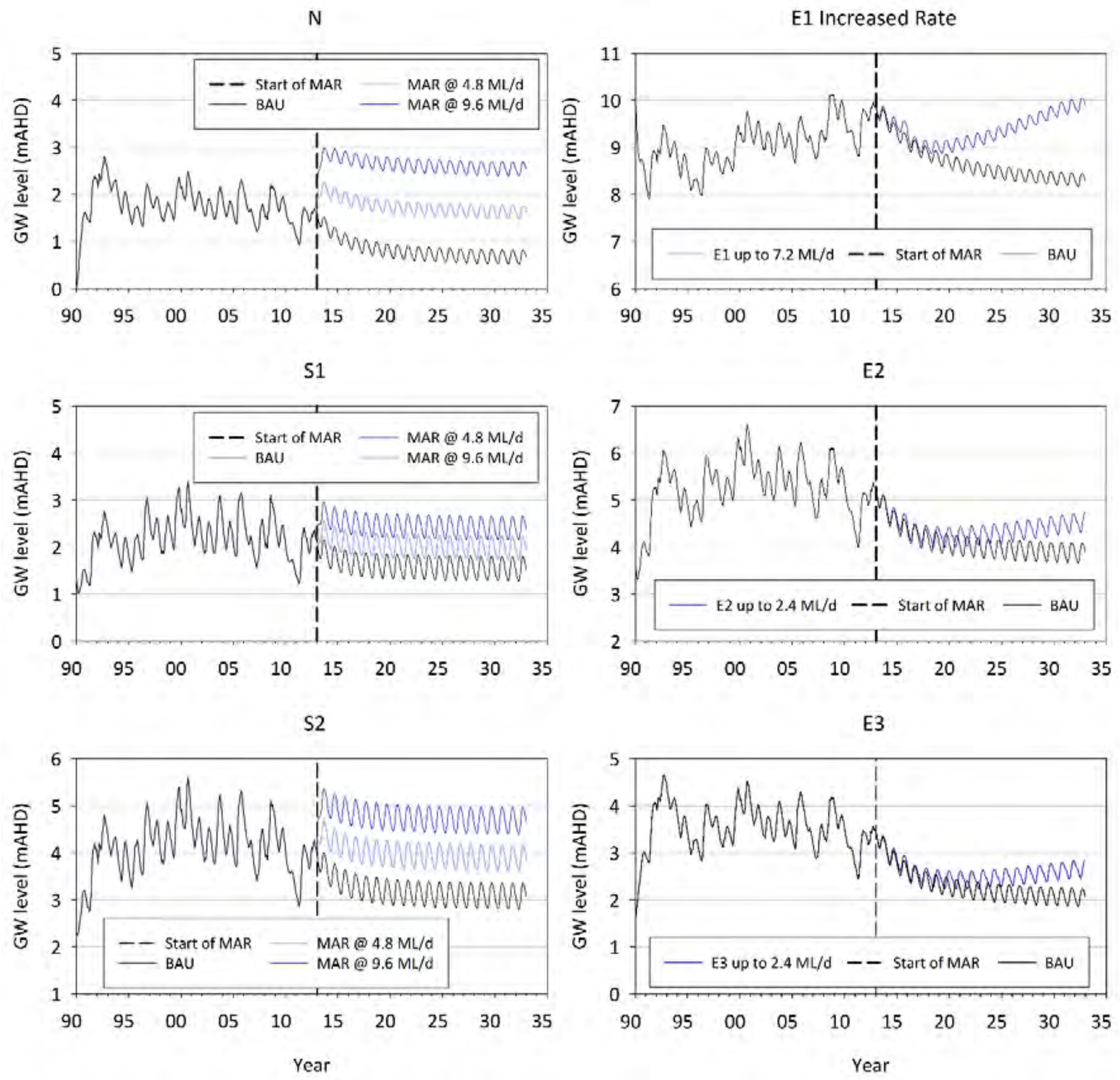

Figure 10.7 Simulated groundwater levels for a 'fictitious' observation well located adjacent to each of the six MAR sites relative to the BAU model. Several of the Sites (N, S1 and S2) were simulated at two infiltration rates and these results are plotted together with the BAU for comparison.

10.3.3 ESTIMATES OF SUBMARINE GROUNDWATER AND NUTRIENT DISCHARGE

For the calibration period the annual submarine groundwater discharge (SGD) rates predicted by the model vary quite widely (e.g. Figure 10.8). The annual SGD is strongly dependent on the rainfall, both for the calendar year of interest and the year preceding. Thus for the years show in Figure 10.8 the SGD is lowest in 2010 corresponding to the lowest rainfall (485 mm). The SGD was highest in 2000 and while the 2000 rainfall (807 mm) was lower than 2005 (897 mm), the preceding year, 1999 has 875 mm compared to 654 mm in 2004. This inter-annual variability is not captured for the projected period since climate parameters are constant between years.

Spatially the distribution of SGD (Figure 10.8) shows the inverse pattern of that modelled by Smith and Nield (2003) due to large spatial differences in hydraulic conductivity used in the respective models. The SGD distribution largely reflects the relatively lower hydraulic conductivity (85 m/d) for the Tamala Limestone in the north compared to 300 m/d in the area of Safety Bay Sand in the south. Smith and Nield (2003) used the inverse pattern of hydraulic conductivities, higher in the north (1660 to 3000 m/d) and compared to the south (400 to 800 m/d). The spatial distribution in SGD in

Figure 10.8 indicates that the majority of the flux to Cockburn Sound occurs from the area of Safety Bay Sand which will impact on nutrient inputs to the sound as it hosts a number of contaminant plumes, especially nitrogen (Figure 3.6).

The computed submarine groundwater discharge (SGD) for all of the scenarios show a reduction in SGD compared to BAU SGD of 32.8 GL/y in 2012 (Table 10.3) because it was assumed that a 2030 dry climate scenario would commence from January 2013. A decrease of approximately 3.5 GL/y in SGD was projected for 2013 compared to 2012 although annual rainfall increased from 677 mm to 713 mm. The differences in the rainfall in the preceding year may explain these differences with 810 mm in 2011 resulting in the higher SGD in 2012 despite the lower rainfall while the modest increase in rainfall in 2013 may also reflect the low 2012 rainfall. It is important to note that exceptionally dry years such as 2010 result in lower SGD with the model only producing 22.8 GL discharged to the Sound. In the long-term a constant annual rainfall of 713 mm produced a similar SGD for the BAU scenario in 2032 (22.9 GL).

The change in SGD for each scenario with respect to BAU scenario in the same year is shown in Figure 10.9. MAR at Sites E1 (increased rate), E2 and E3 doesn't change the flux of groundwater discharged to the ocean in the initial few years (2013 and 2014) since the MAR volumes are initially low and these sites are distant from the Sound. The addition of about 4.8 ML/d of MAR at S1 increases discharge by over 1 GL/yr or 5% compared with business as usual (BAU) over the entire projected period and by twice this at a rate of 9.6 ML/d (Table 10.3). The impact of MAR at S2 is slightly mitigated by its greater distance from the Sound and the likelihood that pumping bores may intercept the flows (see Section 10.4**)** MAR at the northern site has a similar impact to that at S1 by 2032, though the initial increases in SGD are slower. While the dry scenario may not eventuate, the relative fluxes between MAR scenarios will be similar whichever climate scenario is used.

Table 10.3 Computed submarine groundwater discharge fluxes in GL/yr. The flux for the business as usual (BAU) model for 2012 was 32.8 GL/yr

SCENARIO	YEAR: 2013	YEAR: 2014	YEAR:2032
BAU	29.3	26.3	22.9
E1 (increased rate, up to 7.2 ML/d)	29.3	26.3	23.3
E2 (up to 2.5 ML/d)	29.3	26.3	23.1
E3 (up to 2.5 ML/d)	29.3	26.3	23.4
S1 (4.8 ML/d)	30.2	27.4	24.2
S1 (9.6 ML/d)	30.8	28.6	25.5
S2 (4.8 ML/d)	29.7	27.0	23.9
S2 (9.6 ML/d)	30.0	27.8	25.0
N (4.8 ML/d)	29.8	27.2	24.1
N (9.6 ML/d)	30.2	28.1	25.4

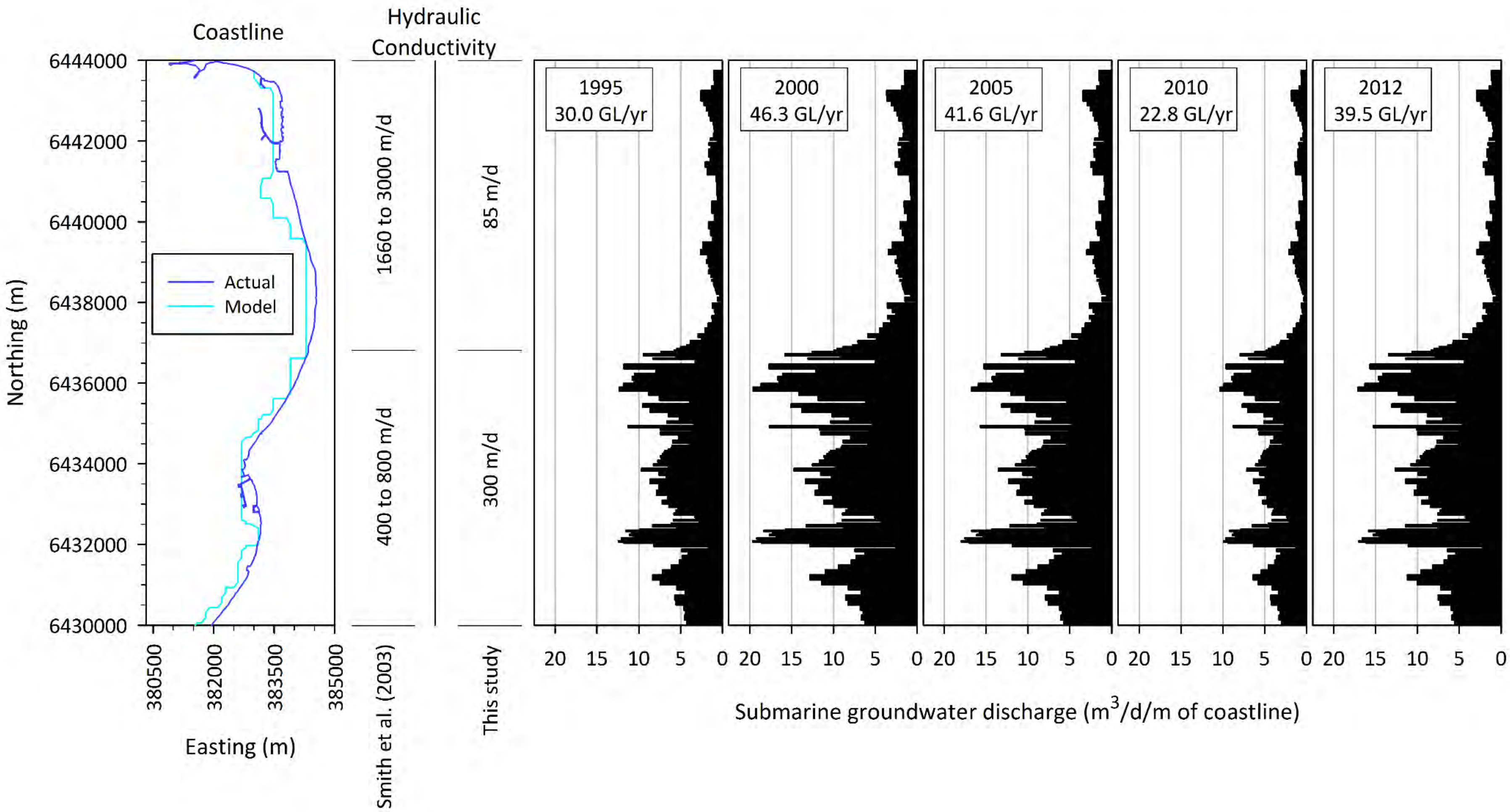

Figure 10.8 Spatial distribution of modelled submarine groundwater discharge (this study) for five selected years. Approximate distribution of model hydraulic conductivities provided for comparison and coastline for spatial reference.

There are two potential sources impacts to nutrient loads to Cockburn Sound resulting from MAR. First is the mobilisation of existing nutrients within the groundwater through the increase in SGD relative to BAU for each MAR scenario. Second is the addition of nutrients with the infiltrated TWW and subsequent transport to Cockburn Sound. To estimate the first case, the spatially distributed SGD computed from the model was multiplied by the spatially distributed nutrient concentrations determined from pore water sampling reported by Smith et al. (2003). This method indicates that the total nitrogen and total phosphorus loads to Cockburn Sound determined for BAU in 2012 were 655 t/yr and 181 t/yr, respectively. The existing nutrient discharge is computed based on separate multipliers applied to the SGD results for total nitrogen and total phosphorous loads. Generally for all scenarios the future projections of nutrient loads from existing sources decrease relative to the BAU in 2012 due to the decrease in SGD as a result of the drying climate scenario used (Figure 10.10). However relative to the BAU loads in 2032, MAR increases nutrient loads from existing sources (black bars, Figure 10.10).

Several different estimates were conducted to assess the impact of the addition of nutrients with the TWW. The loads for 2032 were calculated assuming SGD was either the proportion of TWW reaching the coast (based on particle tracking, see Section 10.4) or 100% reaching the coast, and concentrations based on the 50th and 95th percentile of nutrient concentrations for the Woodman Point WWTP (sites N, S1, S2) or Kwinana WWTP (E1 increased rate, E2, E3). Three estimates are indicated in Figure 10.10; moderate (proportional TWW volume and 50th percentile concentration), high (proportional TWW volume and 95th percentile concentration) and worst case (100% of TWW and 95th percentile concentration). Finally these estimates also assume that there is no attenuation of nutrients in the aquifer. For the eastern sites and N at 4.8 ML/d either the TWW is not predicted to make it to the coast or is intercepted by groundwater abstraction, thus for the moderate and high cases there is no increase in loads to the sound. As the proportion of TWW reaching the coast increases for S2 and S1 the loads also increase. For the worst case scenario the total loads are at least double that of the increase to due to mobilisation of existing sources.

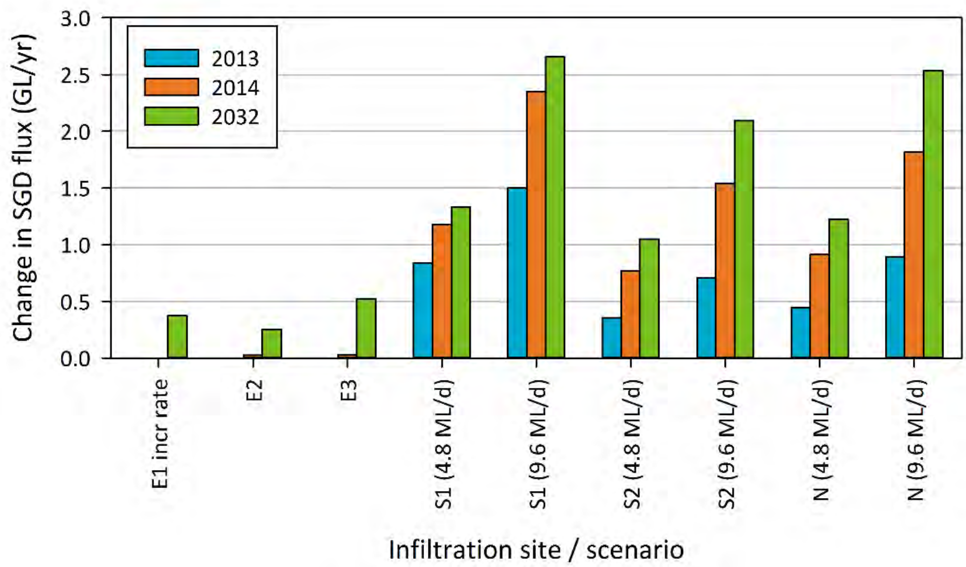

Figure 10.9 Change in submarine groundwater discharge (SGD) along the coastline relative to the business as usual scenario for the same year. Scenarios as outlined in Table 10.2.

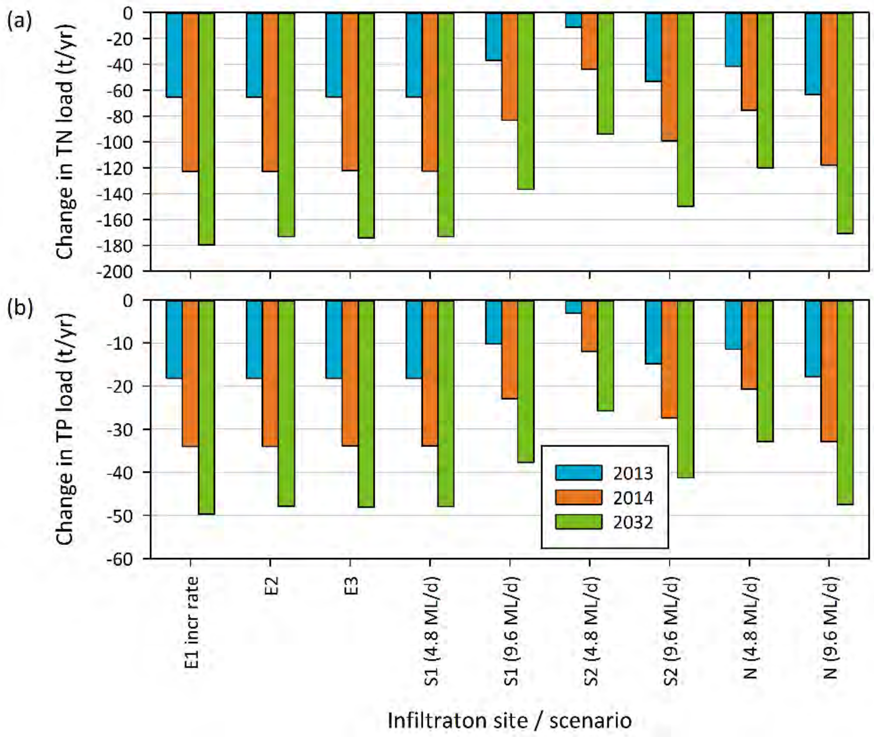

Figure 10.10 Change in total nitrogen (TN) and total phosphorus (TP) loads in submarine groundwater discharge along the coastline relative to the business as usual case in 2012. Scenarios are as outlined in Table 10.2.

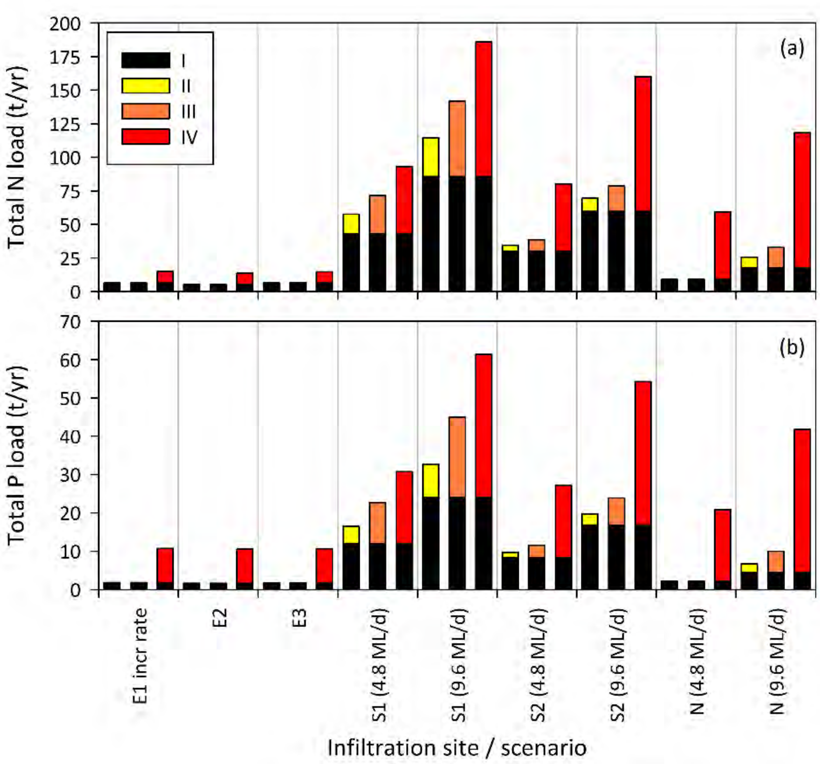

Figure 10.11 Increase in (a) nitrogen and (b) phosphorus load to Cockburn Sound relative to BAU in 2032. (I) mobilisation of existing nutrients, and mobilisation of nutrients introduced with MAR (II) moderate case, (III) high case, and (IV) worst case.

10.3.4 SEAWATER INTRUSION

The calculated locations of the 'toe' of the salt water wedge (i.e. the point where the sea water interface (SWI) intersects the base of the Superficial Aquifer) are shown in Figure 10.11 for several different models. In the BAU model for 2012 (close to present day), the predicted location varies along the coastline, but is generally less than 1 km from the coastline, excluding Woodman Point where the distance is as much as 2 km (Figure 10.11A).

Figure 10.11A shows a comparison of the simulated SWI for the BAU model at the end of 2012 and 20 years later. Due to the implementation of the dry climate projection in the model, groundwater levels in the Superficial Aquifer decline over the 20 year period, which causes the SWI to migrate inland by a maximum of about 920 m over that time period. While the change in the position of the SWI is only an estimate, the greatest impact is in the north of the study area, though uncertainty in this area is high due to the lack of calibration bores (Figure 8.9). The position of the SWI for the BAU model at 2032 indicates that Brownman Swamp and Lake Mt Brown may potentially be impacted, especially as these wetlands are close to sea level (Department of Conservation and Land Management, 2006). The degree to which these wetlands may be impacted however is not clear as they are already saline (Davis et al., 1993; Strehlow et al., 2005) which relates to the formation of the deposits which host the wetlands during previous periods of high sea levels (Davis et al., 1993).

Figure 10.11B shows a comparison of the simulated SWI at the end of the projected period for the BAU model and the Site N (9.6 ML/d infiltration) model. The results suggest that implementation of MAR at Site N at 9.6 ML/d could cause the SWI to be up to 1100 m farther to the west than under the conditions of the BAU model. The largest impact of MAR on the SWI is predicted down-gradient (west) of the N site.

Figure 10.11C shows a comparison of the simulated SWI at the end of the projected period for the BAU model and the Site S1 (9.6 ML/d infiltration) model. The results suggest that implementation of MAR at Site S1 at 9.6 ML/d could cause the SWI to be up to 300 m farther to the west than under the conditions of the BAU model. The largest impact of MAR on the SWI is predicted down-gradient (west) of the proposed MAR at Site S1.

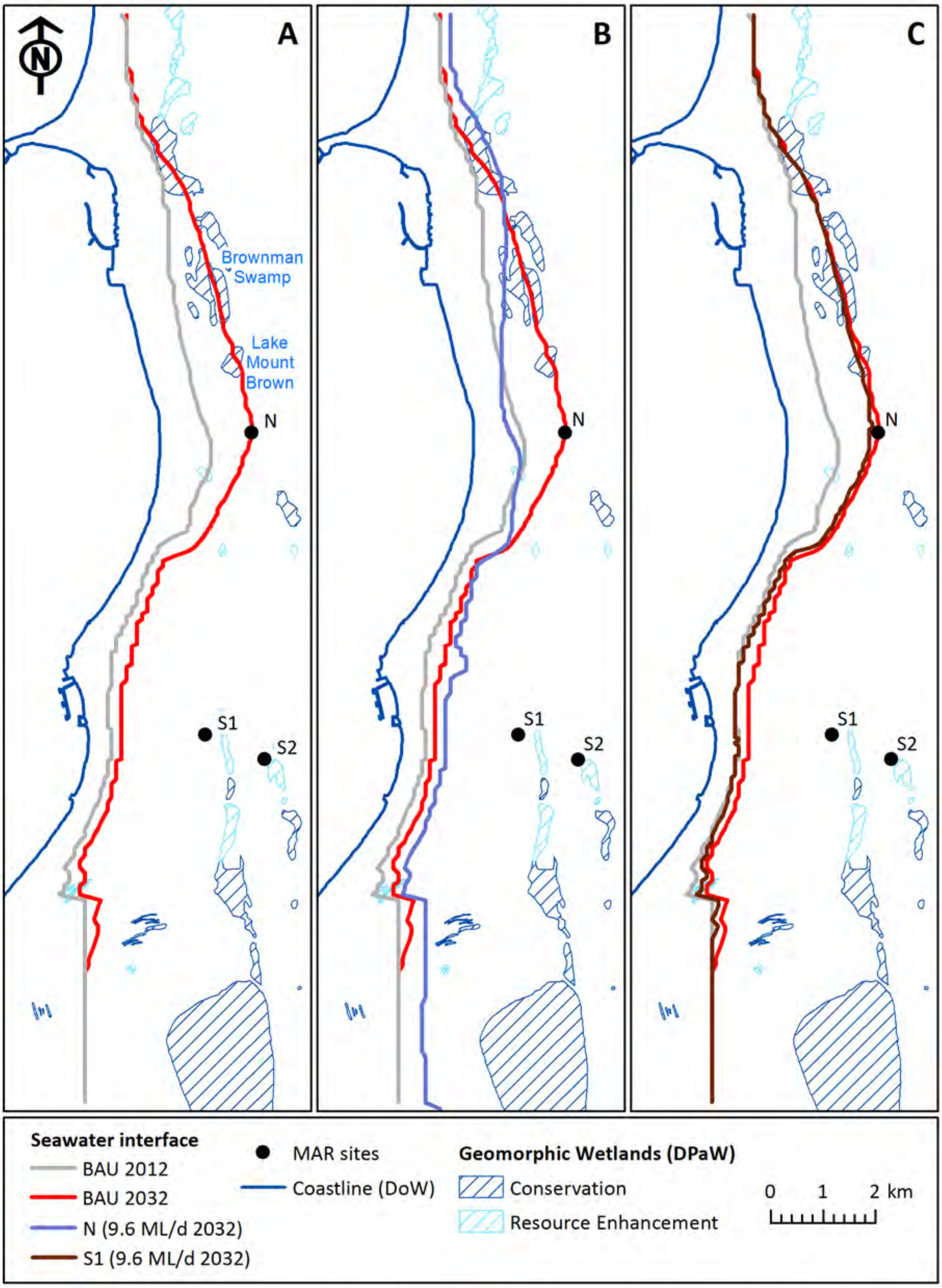

Figure 10.12 Calculated locations of the seawater interface (SWI) using the Ghyben-Herzberg approximation and simulated freshwater heads from the model. (A) SWI location in the business as usual (BAU) scenario between 2012 and 2032, (B) 2032 SWI location for 9.6 ML/d (2 m/d) MAR infiltration at site N and (C) 2032 SWI location for 9.6 ML/d (2 m/d) MAR infiltration at site S1

10.4 Constraints and opportunities

10.4.1 INTRODUCTION

In the following section the constraints for the six sites selected for detailed analysis (Figure 10.1) in the final stage of the project will be discussed. This discussion is mostly limited to the engineering and environmental constraints which relate to the connection of the MAR site to the source of TWW and the modelled transport of the infiltrated TWW.

The engineering constraints are largely derived from the analysis by (GHD, 2015a).

Environmental constraints are examined using outputs from the groundwater model, in particular the advective particle tracks and depth to groundwater. The theoretical particles added to the model at the beginning of the projected infiltration period (1 Jan 2013) will be used to show the modelled advective path of the TWW in the aquifer. Initially 120 particles were added at the perimeter of the infiltration basin and ran in forward mode. Particle tracking was also run in backward mode for 100 particles introduced to the model around the circumferences of The Spectacles northern lake to examine the origin of groundwater potentially discharging to the wetland. Further details on how the model was set up can be found in Chapter 8. The maps show the areal extent of these particles through time, however the particles also move vertically through the aquifer.

The other model output used in the interpretation of the constraints at each site is the depth to groundwater. This is determined from the difference between the land surface elevation (model input) and the groundwater table elevation (model result) at the end of the modelling period, 2032.

10.4.2 NORTHERN SITE (N)

Engineering

Of all the sites, Site N is located closest to the TWW source with a 150 m pipeline required to deliver the TWW to the site. As TWW is to be supplied from the SDOOL depending upon the method of infiltration, filtration of the TWW may be required prior to infiltration to limit clogging. As site N is located in the Latitude 32 industrial area there is the potential for infiltration to take place in recharge basins rather than infiltration galleries.

Perhaps the biggest engineering constraint is related to the proposed Fremantle to Rockingham Controlled access highway, within the footprint of which site N is located (Appendix C, GHD, 2015a). While the Beeliar Regional Park restricts the siting of MAR further to the west there is the possibility of it being moved to the east to avoid the highway. Further investigation would be required to determine the impact of MAR on groundwater abstraction for the management of the groundwater plume associated with Kwinana power plant fly ash disposal.

Environmental

Following 20 years of infiltration the particle tracks (Figure 10.12) indicate that the infiltrated water does not by advection reach Cockburn Sound at a rate of 4.8 ML/d, however 15% of the particles are expect to reach Cockburn Sound at an infiltration rate of 9.6 ML/d within 20 years. A minimum advective travel time to Cockburn Sound of 18 years was determined for the 9.6 ML/d scenario. Thus nutrient impacts to Cockburn Sound are likely to be relatively low due to the high residence time within the aquifer assuming that natural attenuation (denitrification and phosphorus sorption) occurs.

Similarly the model predicts that the areal extent of the TWW only impacts wetlands at the 9.6 ML/d infiltration rate (Figure 10.12). Two wetlands are potentially impacted, Lake Mt Brown and an unnamed wetland SW of the infiltration site, with minimum travel times of 14 years and 8 years, respectively. Though due to the complexity of the geology in this area (northerly extent of Safety Bay Sand/Becher Sand, Chapter 3) the impact on the wetlands would need to be confirmed locally taking the geology into account.

A number of large abstractors are present in the vicinity of the northern site forming two main clusters associated with Alcoa (western cluster) and abstraction to control a plume emanating from fly ash disposal (eastern cluster, abstracted groundwater used in power station operations). The larger abstractors perturb the particle flow paths and along with several other minor users intercept TWW (particles) as it moves away from the site of infiltration. This results in 29% and 36% of the particles remaining in the model domain following 20 years of infiltration for the 4.8 ML/d and 9.6 ML/d, respectively. This implies that there would be a commensurate reduction in the nutrient load that may eventually enter Cockburn Sound even if the nutrients behave conservatively and there is no attenuation within the aquifer. As expected the travel times to the bores are faster at the higher infiltration rate, 1-2 years to the closest bore, 3-4 years to the eastern cluster, and 4-5 years to the western cluster.

A depression in the landscape extends from site N north towards Lake Mt Brown and south towards Long Swamp associated with the inter-dunal swale. Infiltration at the site N has the greatest impact on depth to groundwater along this line (expansion of area green zone, Figure 10.13). There was a small area (~0.7 ha) immediately north of the infiltration zone groundwater rise to within 1 m of the ground surface at the high infiltration rate.

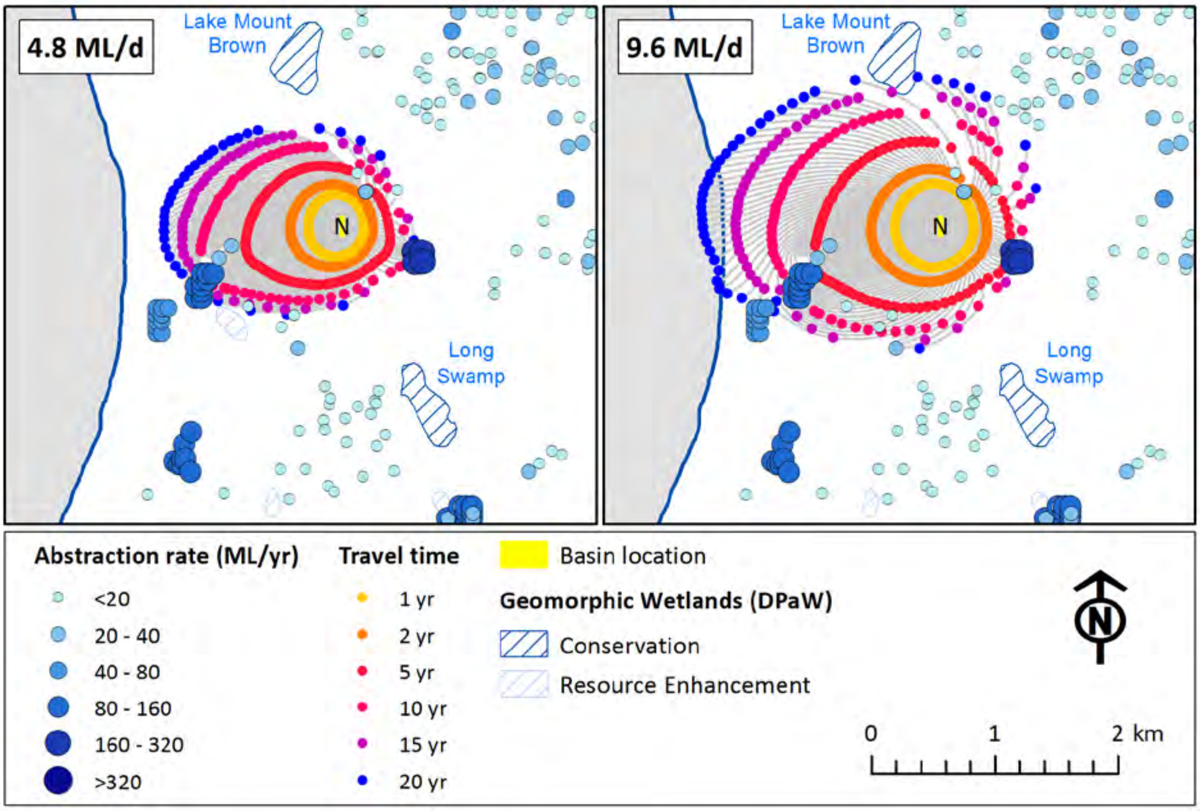

Figure 10.13 Areal extent of 20 years of treated wastewater infiltration at Site N for 4.8 ML/d (left panel) and 9.6 ML/d (right panel) based on model particles released at the beginning of MAR period (1 Jan 2013). The spatial extent of groundwater abstraction bores is also shown

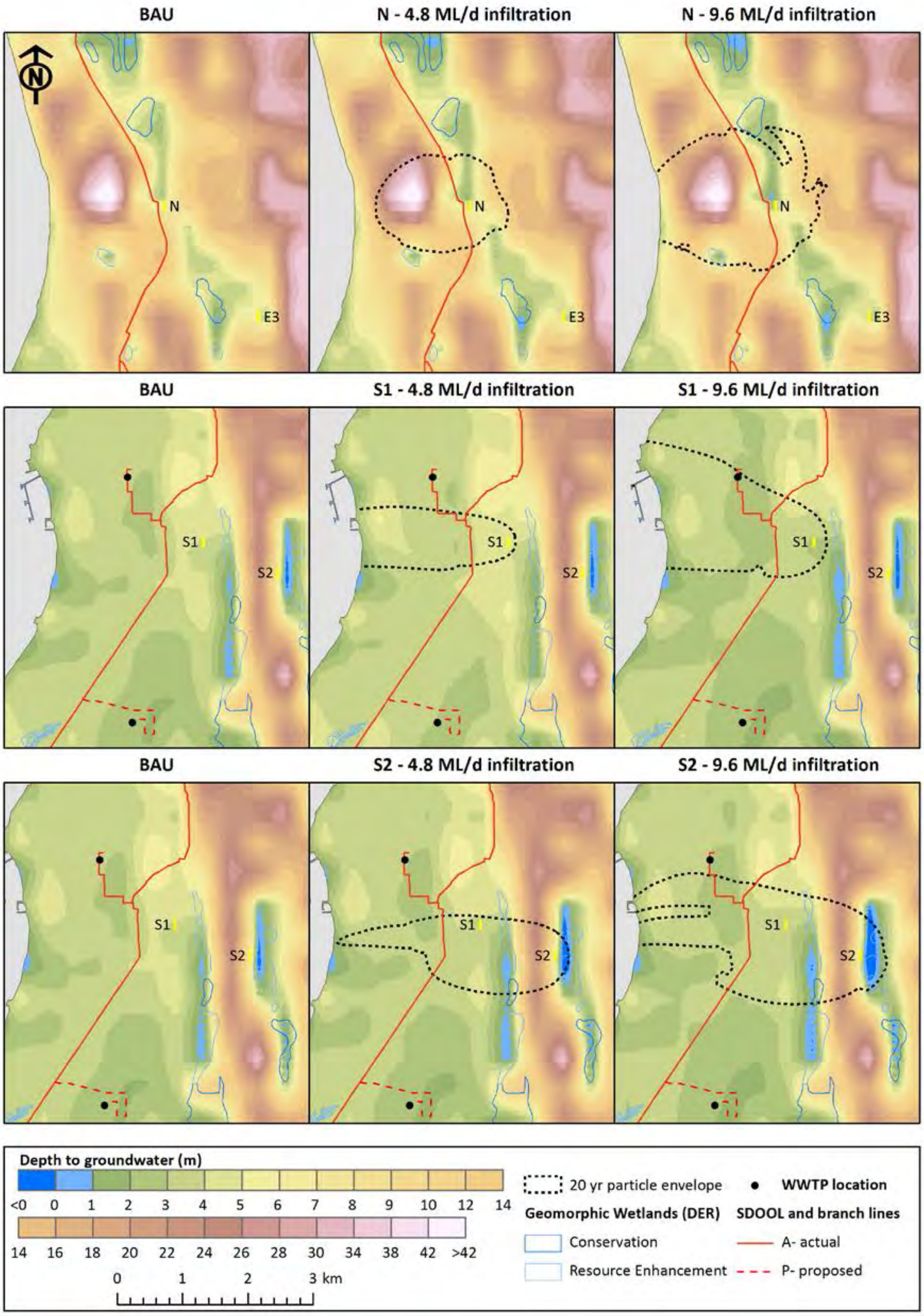

Figure 10.14 Minimum depth to groundwater (October 2032) at sites N, S1 and S2 under business as usual (BAU, left panel), 4.8 ML/d infiltration (central panel) and 9.6 ML/d infiltration (right panel). The 20 year particle envelope is overlaid for comparison

10.4.3 SOUTHERN SITE 1 (S1)

Engineering

Site S1 is located relatively close to a SDOOL section value with approximately 1000 m of pipeline required to deliver the TWW to the site. As TWW is to be supplied from the SDOOL depending upon the method of infiltration, filtration of the TWW may be required prior to infiltration to limit clogging. As Site S1 is located in industrial land there may be the potential for infiltration to take place in recharge basins rather than infiltration galleries. Other engineering challenges include the crossing of several minor roads and a natural gas pipeline (Appendix C, GHD, 2015a).

Environmental

During the 20 year projected period it is expected that Site S1 would have the greatest influence on loads to Cockburn Sound. The particle paths predict that the minimum travel time for TWW to reach the coastline by advection is 9-10 years at 4.8 ML/d infiltration rate and 7-8 years at 9.6 ML/d (Figure 10.14). This is the lowest aquifer residence time relative to the Sound for all sites.

There are a number of groundwater users in the area with CSBP being the major user according to the groundwater allocation dataset. Groundwater users capture approximately 44% of particles introduced into the model indicating that a substantial proportion of the nutrients added may be removed by groundwater pumping. Since the actual locations of the CSBP production bores (Figure 10.14) are more spatially distributed there may be greater interception of TWW in reality for MAR at Site S1. However the licence conditions imposed on CSBP and other industries that dispose of wastewater to the SDOOL that limit N loads may prove a constraint especially as the travel time to the bores are of the order of 1-2 years.

The siting of MAR at S1 is also likely to impact on existing pollutant plumes the approximate locations of which are indicated in Figure 10.14 based on those published by (Trefry et al., 2006). This may have a two-fold effect. Firstly plumes down-gradient of the MAR site will be flushed into Cockburn Sound at a faster rate than previously. Secondly the flow direction of those plumes on the periphery of the TWW plume will be altered. More information on the plume geometry is required to be able to predict what will occur if a MAR site is established at Site S1.

No direct impact from the TWW entering nearby wetlands is expected from MAR at Site S1. However there may be a beneficial benefit of the additional water which results in enhanced availability of water in two chains of wetlands (unnamed) to the east of Site S1 (Figure 10.14) where areas of depth to groundwater <1 m, including inundated area increase with increasing infiltration. This occurs with no appreciable changes to the depth to groundwater in the industrial area adjacent to Site S1.

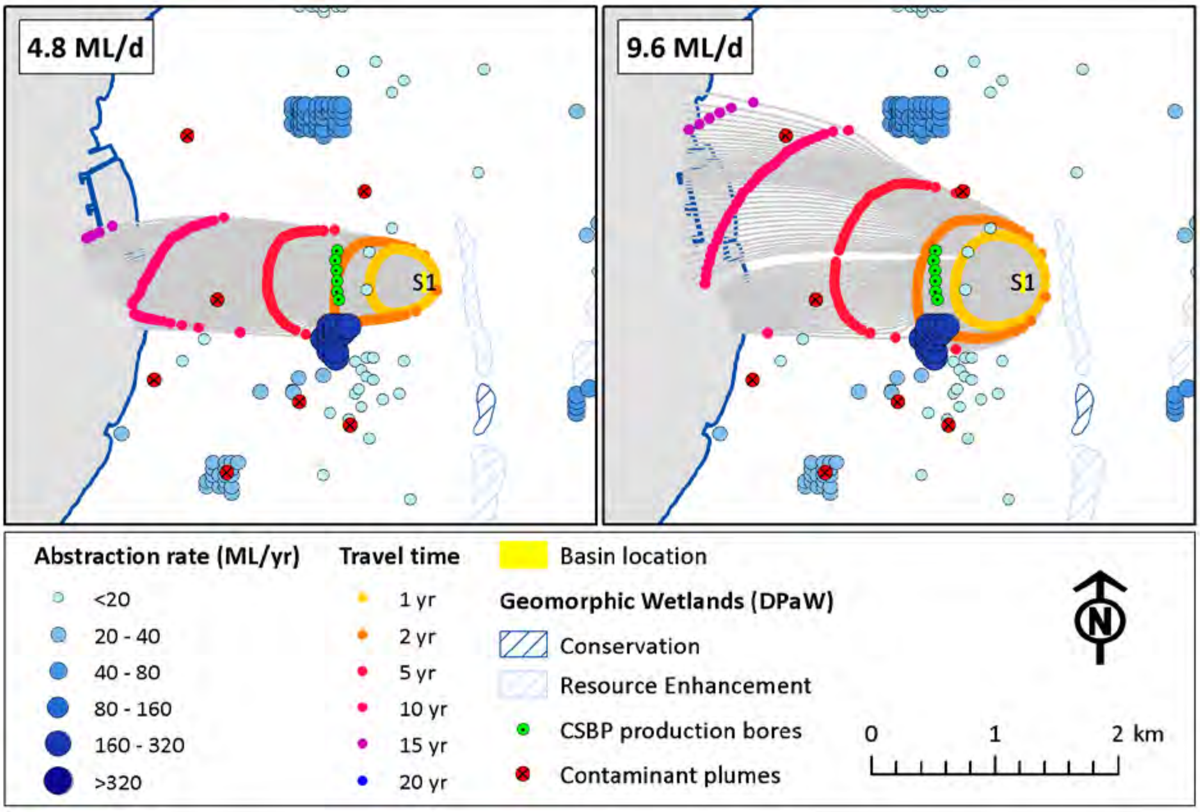

Figure 10.15 Areal extent of 20 years of treated wastewater infiltration at Site S1 for 4.8 ML/d (left panel) and 9.6 ML/d (right panel) based on model particles released at the beginning of MAR period (1 Jan 2013). The spatial extent of groundwater abstraction bores is also shown along with the location of CSBP production bores and approximate locations of existing contaminant plumes

10.4.4 SOUTHERN SITE 2 (S2)

Engineering

Site S2 is the most distant of the three sites to take TWW from the SDOOL, with 1800 m of pipeline required to deliver the TWW to the site. Due to the close proximity to the Kwinana Golf Club it is unlikely that recharge basins could be used at this site indicating that filtration would be required prior to infiltration to limit clogging. The site is also in an elevated location relative to the SDOOL thus requiring greater pumping infrastructure to transport the TWW.

The requirement of the pipeline to cross a railway reserve substantially impacts the costs to the pipeline construction along with the crossing of several roads and a natural gas pipeline.

Environmental

Due to the greater distance from the coast in comparison to Site S1, the TWW travel time by advection to reach the coast from Site S2 is longer, greater than 20 years for 4.8 ML/d and a minimum travel time of 15-16 year for 9.6 ML/d (Figure 10.15). Those particles which reached the coast between 15 and 20 years represent 13% of the particles released. A substantial proportion of the particles introduced into the model were removed through groundwater abstraction, with only 17% remaining within the aquifer at 20 years for the 4.8 ML/d rate and 25% either remaining within the aquifer or making it to the coast at 9.6 ML/d.

Of greater concern is the proximity of Site S2 to wetlands both up-gradient and down-gradient. Extremely short travel times (<1 year) to the two up-gradient (resource enhancement) wetlands located within the Kwinana Golf Club and bushland to the north (Figure 10.15). There is a high likelihood that infiltrated TWW may enter these wetlands. While the wetland within the golf club is likely to be already impacted by fertiliser use the health risks are potentially greater due to the recreational use and irrigation using groundwater within the golf club. There are slightly longer (2 to 4 years) aquifer residence times before the infiltration TWW reaches the down-gradient wetlands hence potentially less impact. Further assessment of the ecological function of the wetlands surrounding Site S2 would need to be conducted before MAR could proceed at Site S2.

The model indicates that inundation occurs within the wetlands up-gradient of Site S2 even under the BAU scenario (Figure 10.13). This combined with the close proximity of the MAR site to the wetlands results in increases to the area where the groundwater is above the land surface. This again may impact on the health risks. Additional benefits are observed in wetlands to the southeast and southwest of the MAR site with groundwater levels rising in these areas. Similar to S1 there is no substantial change in depth to groundwater in the industrial areas to the west.

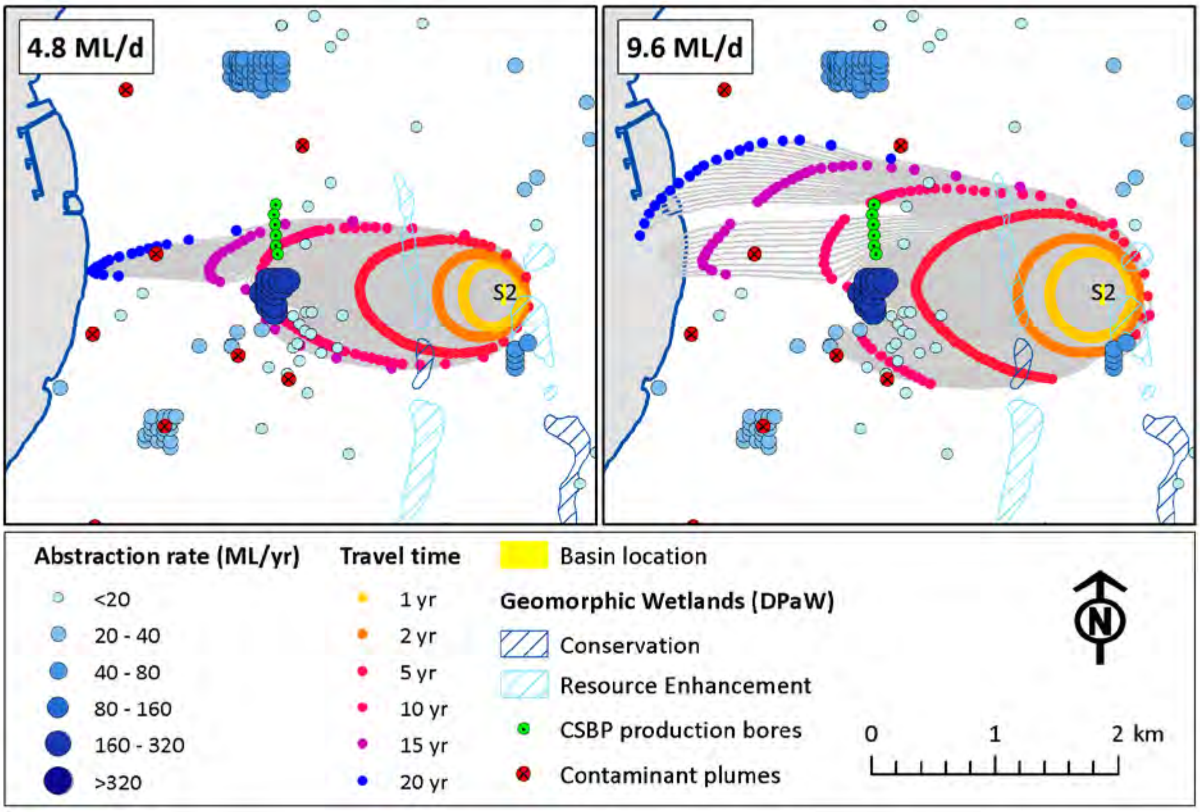

Figure 10.16 Areal extent of 20 years of treated wastewater infiltration at Site S2 for 4.8 ML/d (left panel) and 9.6 ML/d (right panel) based on model particles released at the beginning of MAR period (1 Jan 2013). The spatial extent of groundwater abstraction bores is also shown along with the location of CSBP production bores and approximate locations of existing contaminant plumes

10.4.5 EASTERN SITE 1 WITH ADDITIONAL INFILTRATION (E1 INCREASED RATE)

Engineering

Currently the disposal of TWW to the recharge basins at the Kwinana WWTP (Site E1) is limited by the licensed disposal rate of 4.7 ML/d. Outflow from the WWTP is currently in excess of the disposal limit and is expected to increase into the future (Chapter 5). The additional TWW applied in this scenario increased from 0.2 ML/d in 2013 to 2.5 ML/d in 2032.

Due to the existing disposal of TWW to recharge basins located at the Kwinana WWTP the engineering requirements are thought to be minimal. However, further investigation should be undertaken to investigate whether groundwater level rise associated with increased infiltration rates and subsequent connection between the ponded water and groundwater beneath the basins will limit the infiltration capacity. If this proves to be the case then additional basins may be required to meet the extra capacity.

Environment

Unlike other sites the particles travel in a north-westerly direction away from the Site E1 (Figure 10.16) with the flow direction consistent that observed for the calibration period and previous modelling (Nield, 2004). Abstraction of groundwater to the north and northwest strongly influences the flow direction since previous model versions with lower abstraction resulted in a westerly flow direction (e.g. Figure 10.5). The abstraction has resulted in a stable plume extent with little change

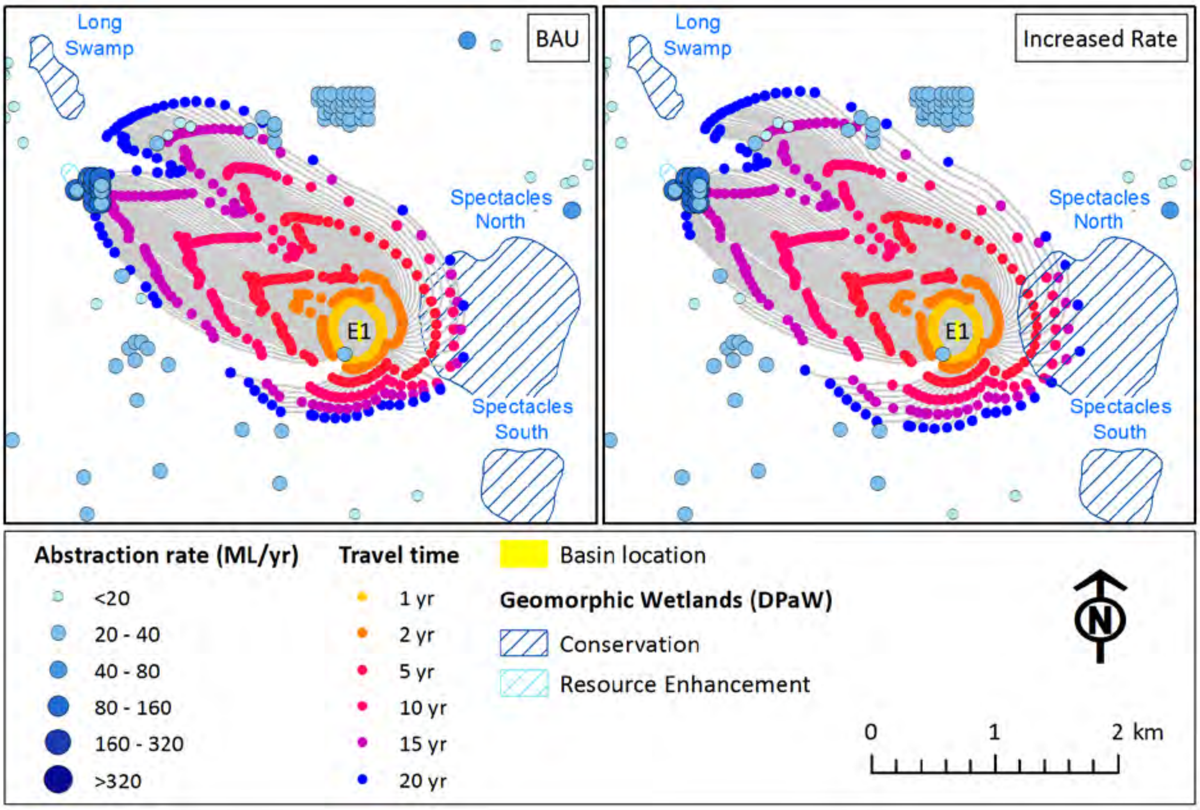

Figure 10.17 Areal extent of 20 years of treated wastewater infiltration at Site E1 for business as usual (BAU, 4.7 ML/d, left panel) and increased infiltration rate (up to 7.2 ML/d in 2032, right panel) based on model particles released at the beginning of the projected MAR period (1 Jan 2013). The spatial extent of groundwater abstraction bores is also shown

during the whole model period (particles released from 1995) with a 14 year advective travel time to the cluster of bores to the western extent of the particle track. The earlier particle release times suggest that the outer edge of the TWW plume suggest that it advectively takes approximately 30 years to reach Long Swamp. This is also reflected in the minimal differences between the BAU and increased rate scenarios with only a slight spreading of the plume with increased infiltration (Figure 10.16).

In plan-view particles travel east from the recharge basins towards The Spectacles wetland with both BAU and increased rate scenarios taking similar times to travel (by advection) to the western edge of the wetland; 3 to 4 years. Subsequent particle release dates when infiltration rates were higher suggests that travel times may be shorter; 2 to 3 years. Within the model particles are also tracked in the vertical direction. For those particles that migrated towards The Spectacles they were transported deep within the aquifer instead of migrating towards the watertable surface. While the model was not setup specifically to investigate the vertical distribution, for example not modelling density changes in the Superficial Aquifer, the particle distribution within the aquifer provides additional evidence to that presented in Section 6.4 to which indicated that TWW reaching the wetland is minimal.

To further investigate the potential for TWW to discharge into The Spectacles (north) additional particle tracking was done in backwards mode to determine the origin of particles placed around the perimeter of the wetland (Figure 10.17). The particles were introduced into the model at the beginning of 2013 and tracked backwards to the beginning of the model (1990). It is important to note that not all particles remained in the model in 1990 as removal by sinks, which in this case represented evapotranspiration from the water table, occurred for many particles. Particles on the eastern shoreline originated almost due east and with a few exceptions show paths lengths equivalent to 22 years. This would represent the main capture zone of the wetland. Particle paths to the northeast and south originate close to the 2013 positions, while the model predicts that the major release zone of the wetland is likely to be in the northwest.

Of most interest to the movement of TWW are the particles on the western margin of the wetland (Insert A, Figure 10.17). These particles originate from west of the wetland corroborating the direction of travel for the forward particle tracking from the MAR basin. However these particle tracks are relatively short (~100 m) and terminate in evapotranspiration sinks at the water table. Thus it is reasonable to suggest that local rainfall recharge in the dune system on the western margin of the wetland is the likely source of groundwater discharge to the wetland. This further suggests that while there is a general easterly flow towards the wetland the infiltrated TWW is unlikely to contribute much if at all to nutrient discharge to the wetland.

While it appears from the modelling and other data present in Section 6.4 suggest that TWW is not reaching the Spectacles, due to the complex chemistry in the aquifer below the Spectacles and the Kwinana WWTP basins a local scale solute transport model would be required to better understand the distribution of TWW in the aquifer.

Due to the relatively deep groundwater table at Site E1 there is only a minor increase in potential inundation in Spectacles North (Figure 10.18). This is likely due to the existing mound and relatively small increases to the infiltration rate relatively to the sites that source TWW from the SDOOL.

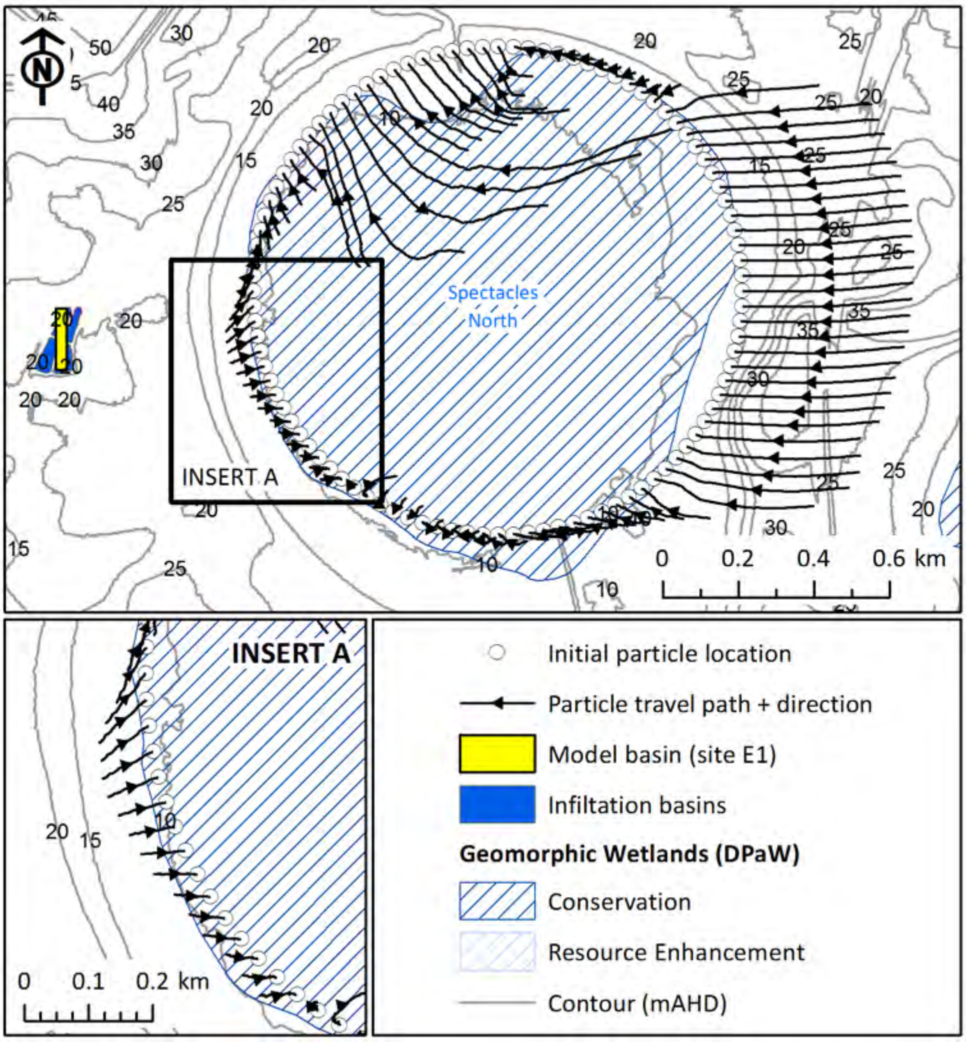

Figure 10.18 Flow direction and source of groundwater around the Spectacles North wetland as revealed by backward advective particle tracking (2013 particle release)

10.4.6 EASTERN SITE 2 (E2)

Engineering

Site E2 was included as an alternative to the addition of TWW in excess of the current license disposal rate of 4.7 ML/d at the Kwinana WWTP. As such is constrained by the projected increase in TWW volumes in excess of 4.7 ML/d, increasing from 0.2 ML/d in 2013 to 2.5 ML/d in 2032. The pipeline to Site E2 connects with the TWW supply at the Kwinana WWTP with the only major piece of infrastructure encounter being the sewer main leading to the WWTP from Kwinana townsite (Appendix C, GHD, 2015a). As Site E2 is located on the Medina Agricultural Research station and in a future light industrial area recharge basins are the most likely form of infiltration.

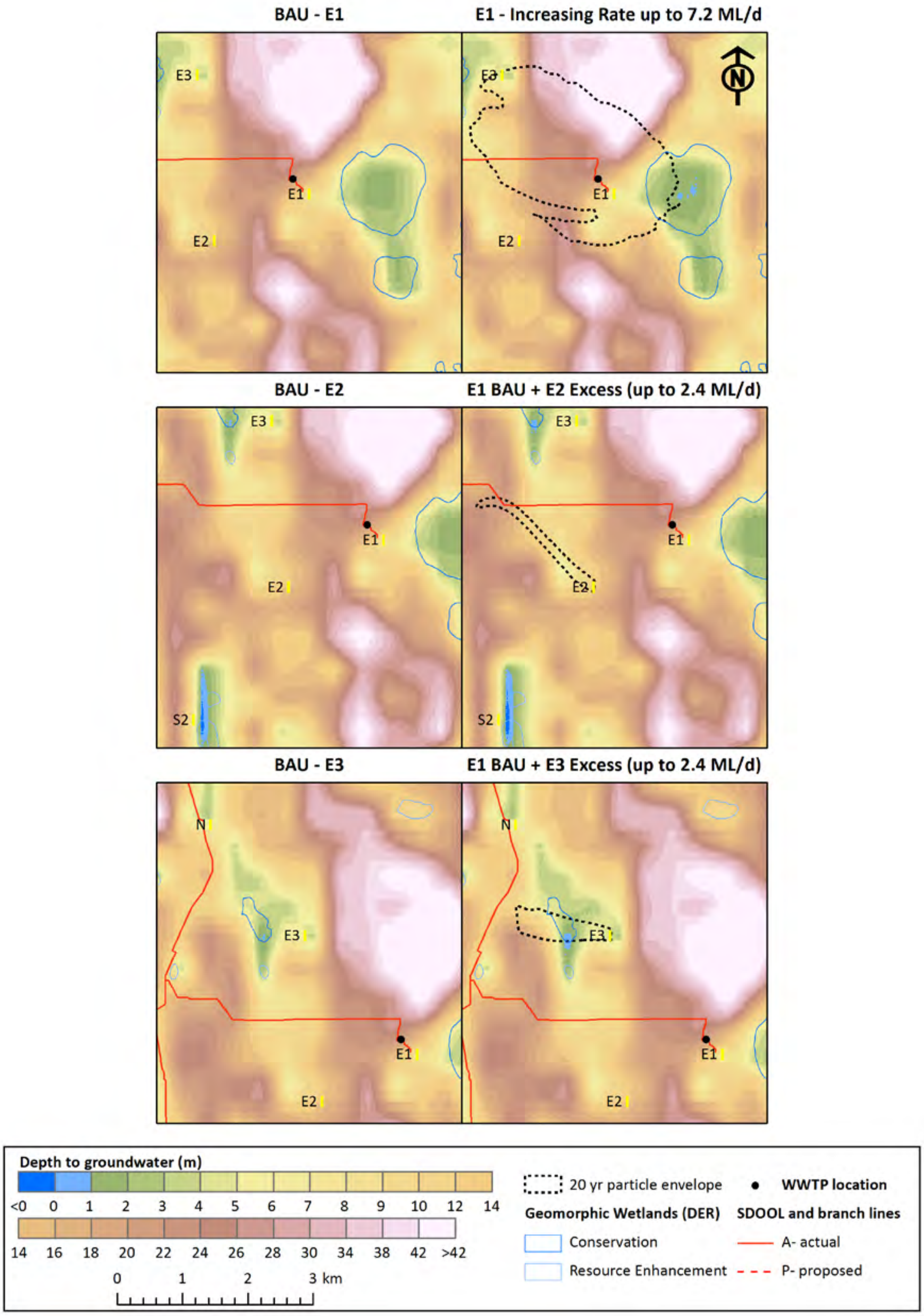

Figure 10.19 Minimum depth to groundwater (October 2032) at Sites E1, E2 and E3 under business as usual (BAU, left panel) and increasing infiltration rate in accordance with Kwinana WWTP outflow in excess of 4.7 ML/d. At Site E1 the additional TWW is applied along with the 4.7 ML/d. The 20 year particle envelope is overlaid for comparison

Environment

The environmental constraints are low at Site E2. The major impact is on existing groundwater users. Due to the initial low volumes of TWW infiltrated at this site the spread of the TWW plume indicated by the 2013 particle envelope (Figure 10.19) is relatively narrow compared to those sites with higher infiltration rates (N, S1, S2 and E1). The TWW flows northwest towards a cluster of abstraction bores which altered the flow direction 15 years after infiltration. This groundwater flow direction avoids the ALCOA ABC residue area passing to the north. A number of smaller groundwater users are impacted especially once higher infiltration rates are applied (e.g. 2027 particle envelope, Figure 10.19). However this interception is not likely until 4-5 years following infiltration. The same general flow direction is also followed at the higher infiltration rates.

The depth to groundwater at Site E2 is relatively unchanged however there is a minor increase in potential inundation within Long Swamp to the north (Figure 10.18).

10.4.7 EASTERN SITE 3 (E3)

Engineering

Similar to Site E2 infiltration at Site E3 was included as an alternative to the addition of TWW in excess of the current license disposal rate of 4.7 ML/d at the Kwinana WWTP and to see if there was any impact on groundwater beneath the ALCOA residue area to the east. It is constrained by the projected increase in TWW volumes in excess of 4.7 ML/d, increasing from 0.2 ML/d in 2013 to 2.5 ML/d in 2032. As Site E2 is located next to ALCOA's residue area recharge basins are the most likely form of infiltration.

To deliver the TWW to Site E3 the pipeline has to run parallel with ALCOA's infrastructure that runs between the refinery on the coast and the residue area along Anketell Road. Hence it easement access may be constrained along this corridor and also by the requirement to cross Anketell Road, a major freight route between the Kwinana Freeway and the KIA industrial area.

Environment

Site E3 is located approximately 500 m up-gradient of Long Swamp, thus may potentially impact the ecological health of the wetland. Initially due to the low infiltration rates a fairly narrow TWW plume forms with particles taking 5-6 years to reach the wetland (Figure 10.19). The aquifer residence time decreases with increasing infiltration rate such that for particles released in 2027 (1.9 ML/d) arrive at Long Swamp after 3-4 years. As the depth to groundwater is not altered substantially (Figure 10.18) it is not clear whether TWW would be expressed in the wetland, though should infiltration increase to rates similar to those modelled for Sites N, S1 and S2 (4.8 and 9.6 ML/d) the impact on Long Swamp is likely to be considerable. There a number of small groundwater users (<20 ML/yr) to the west of Long Swamp that may be impacted by the TWW after 15 years of infiltration, though no major uses appear to be impacted at least at the low infiltration rate modelled.

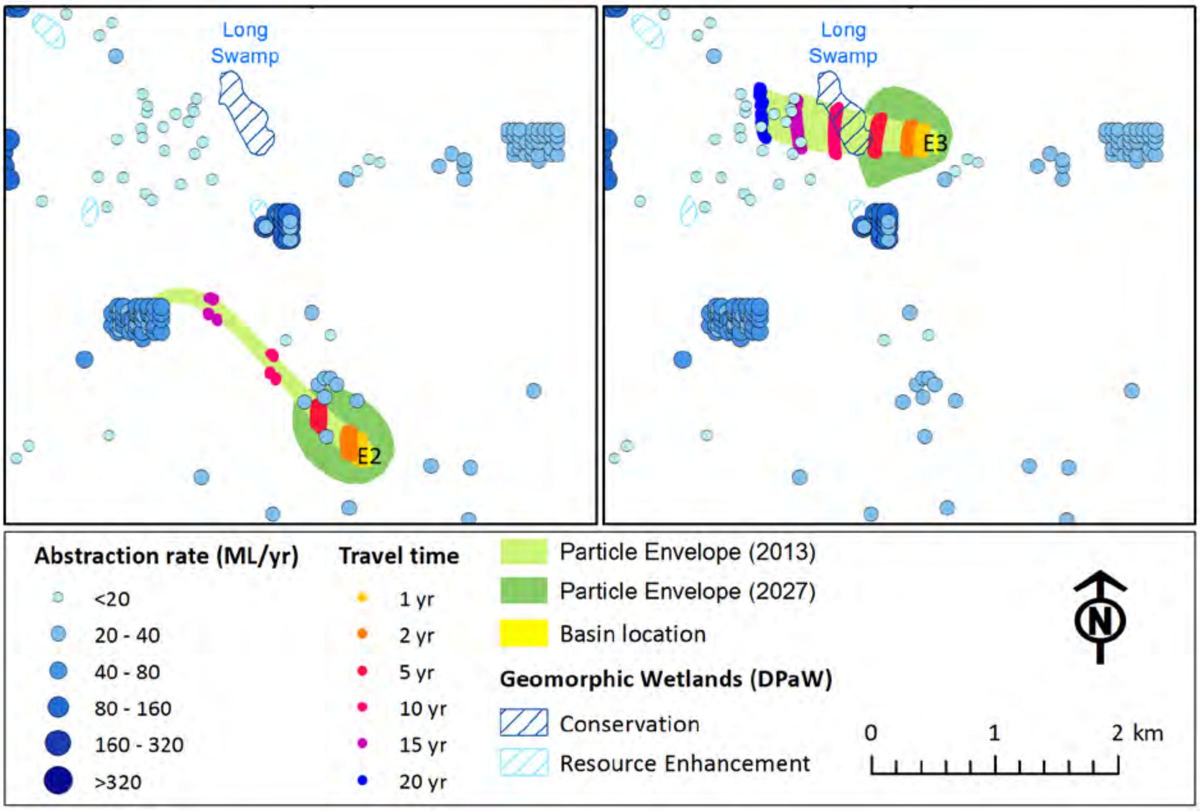

Figure 10.20 Areal extent of 20 years (2013 particle release) and 5 years (2027 particle release) of treated wastewater infiltration at Site E2 (left panel) and Site E3 (right panel). The spatial extent of groundwater abstraction bores is also shown.

10.5 Economic assessment of the MAR locations

The industrial demand for water is expected to increase in coming years. Presently, around 60% of the current demand for heavy industrial use is being met by groundwater (GHD, 2015). However, groundwater is fully allocated in this area and industries are looking for efficiencies or alternative sources of water to meet their water demands. This economic assessment evaluates the cost effectiveness of heavy industry using treated wastewater via MAR. The method applied follows a cost benefit analysis (CBA) framework, the preferred method of analysis for State and Commonwealth agencies. CBA is widely applied for evaluating both private and public investments because it allows ranking of multiple options with disparate timing and scale of costs and benefits with a single summary metric (Kahn, 1998).

As part of the CBA, the non-economic benefit of wetland maintenance from recharge was monetised in order to have as complete a picture of the benefits and costs of the project as possible. However, because wetlands are only a metre or so deep and their values vary enormously between being wet or dry, groundwater and wetland levels in the study area would need to be very accurately modelled, something that is beyond the model that was developed. It was therefore not possible to accurately estimate the costs of wetland loss (from doing nothing) or the benefits of wetland retention or recovery as a result of MAR, even though these values are very high.

Without the inclusion of non-economic costs and benefits, the CBA becomes a relatively simple commercial viability analysis of heavy industry accessing treated recycled water via MAR, where the heavy industry stakeholders are the focus of the commercial viability analysis. The heavy industries include heavy industries in the Kwinana Industrial Area and in the Rockingham Industrial Zone for water demand projection analysis.

The analysis in this report follows the Economic Assessment Tool developed for the Australian Water Recycling Centre of Excellences (AWRCoE) by Marsden Jacob Associates (2013). The Tool utilises the principles of discounting where the costs and benefits are reduced to a single net present value (NPV). A benefit-cost ratio (BCR) is then used as an indicator to indentify the most cost-effective MAR solution to address emerging water shortage problems faced by heavy industries in the study area.

10.5.1 ASSUMPTIONS

General

- The evaluation period was from 2015 to 2031. Year 2031 was chosen as the end period of the analysis because demand projections for heavy industrial use estimated by the Department of Water (2013) were projected up to this year.
- A widely used discount rate, recommended by the New South Wales Treasury (2007), of 7% was applied to the future costs and benefits, with sensitivity testing undertaken at 4% and 10%.
- The cost and benefit of MAR water per kilolitre is based on Marsden Jacob Associates (2013), which is the levelised cost or benefit of water, expressed as follows:

$$Levelised\ cost\ or\ benefit = \frac{NPV(cost\ or\ benefit)}{NPV(water\ conserved\ or\ supplied)}$$

Supply and Demand

- The base case demand scenario follows the High demand scenario. Water demand projections for the Medium and Low demand scenarios are presented in Table 10.4.
- Demand projections for heavy industrial use by 2031 were estimated to be 42.91 GL/yr for the high demand scenario, 40.88 GL/yr for the medium demand scenario, and 37.13 GL/yr for the low demand scenario.
- It was assumed that the existing demand, as of 2014, of 26.78 GL/yr is being met by existing supply sources and is not considered in the analysis. The economic analysis is therefore only concerned with meeting future demands that are above 26.78 GL/yr. For example, the projected high demand scenario forecasts demand to be 42.91 GL/yr by 2031. The costs and benefits will be calculated based only on meeting the surplus demand of (42.91 - 26.78 =) 16.13 GL.

Table 10.4 Demand projections and average annual growth rates for heavy industries 2014-2031 (Department of Water, 2013)

DEMAND SCENARIOS	WATER DEMAND PROJECTIONS (GL/yr)				AVERAGE ANNUAL GROWTH (%)		
	2014	2016	2021	2031	2014-2016	2016-2021	2021-2031
High	26.78	28.57	32.76	42.91	3.29	2.78	2.74
Medium	26.78	28.17	31.50	40.88	2.57	2.26	2.64
Low	26.78	27.58	30.42	37.13	1.50	1.98	2.01

Source: Department of Water (2013)

- It was assumed that any future water demand that is above and beyond the current level of water use of 26.78 GL/yr will be met by groundwater abstraction of MAR water first, followed by Kwinana Water Recycling Plant (KWRP) water until the surplus supply of KWRP runs out, and lastly by scheme water for the remaining amount of water demanded.
- It was also assumed that there will be no future limitations around scheme water infrastructure to the industrial area. Hence, any level of increase in demand can be met by scheme water in the future.
- Construction of all MAR infrastructures is completed within the first year i.e. by 2015/2016. Infiltration is assumed to commence immediately the year after.
- It was assumed that KWRP can be expanded from 6 GL/yr to 9.6 GL/yr without any additional capital cost.
- Infiltration volumes for site N, S1 and S2 are 5 ML/d (approximately 1.825 GL/yr) and 10 ML/d (approximately 3.65 GL/yr), whereas the infiltration volume at sites E1, E2 and E3 will vary with the amount of treated wastewater coming from the Kwinana WWTP that is in excess of the current infiltrated volume of 4.7 ML/d. It is expected that the amount of TWW coming from the Kwinana WWTP will increase gradually every year, up to around 900 ML/yr.
- The base case assumes that there will be no loss to the aquifer from infiltration; i.e. the groundwater available to industries is equal to the amount of treated wastewater infiltrated. A sensitivity analysis for ratios 0.8:1; 0.7:1 and 0.6:1 was also assessed.

Costs

All capital and operating costs estimates for each MAR project were provided by GHD and are articulated in GHD (2015). According to GHD (2015), the cost estimates are preliminary probable costs using information available to GHD staff and are made on assumptions and judgements by GHD. The main assumptions are that:

- Pipelines that will have to be built to sites N, S1, S2, E2 and E3 are 150 m, 1,000 m, 1,800 m, 2,000 m and 4,000 m in length respectively. Site E1 uses the existing infrastructure therefore no new pipeline is required to be built to that site,
- Treated wastewater for sites N, S1 and S2 are sourced from the SDOOL and treated wastewater for E1, E2 and E3 are sourced from the Kwinana WWTP,
- Treated wastewater quality at the Kwinana and Woodman Point WWTPs is assumed to be unchanged for the duration of the study time period,
- Land acquisition costs, monitoring and water quality analysis costs, capital and operating costs associated with end users recycled storage and delivery infrastructure, and capital and operating costs of end user cooling tower pre-treatment facilities and blowdown disposal facilities are not included in the cost estimates,
- Cost of electricity is assumed to be 35 c/kWhr,
- Maintenance costs of civil infrastructure is assumed to be 0.5% of capital expenditure and maintenance cost of mechanical and electrical infrastructure is assumed to 1.5% of capital expenditure,
- The analysis focuses only on two infiltration methods – recharge basins or galleries,
- Sites E1, E2 and E3 do not require any treatment or removal of suspended solids,
- Site E1 will use the existing recharge basin at the site,
- The process of nitrogen removal includes suspended solids reduction as well.

- The cost of industries pumping and treating groundwater is \$0.28/kL (pers. comm. Chris Oughton).

Benefits

The benefit of heavy industry accessing treated recycled water via MAR is the avoided cost of having to meet future water demand with KWRP or scheme water.

- It is assumed that the DoW would allow industries to pump more groundwater (i.e. in addition to their current allocations) immediately after infiltration commences, which is by year 2016/2017. However, they cannot collectively pump more than the amount infiltrated each year.
- Water price
  - The price of treated wastewater (i.e. the cost of buying SDOOL water or treated wastewater from the Kwinana WWTP was assumed to be between \$0/kL to \$0.5/kL. By assuming that industries are contributing towards the State's target of using 30% recycled water by 2030 (Water Corporation 2015), industries may be able to obtain treated wastewater at zero cost. The \$0.5/kL price is based on the market price of groundwater being traded in the WTC (Department of Water, 2013), and is assumed to be the maximum amount industries were willing to pay for treated wastewater. A sensitivity analysis was undertaken for \$0.25/kL also.
  - The price of KWRP water is assumed to be \$2/kL (Department of Water, 2013).
  - The price of scheme water to industries is \$2.093/kL (Water Corporation 2015).

10.5.2 SCENARIOS

Business as usual (BAU) scenario

The BAU scenario is described as the situation where projected demand is expected to increase up to 37.13, 40.88 and 42.91 GL/yr by 2031 for the low, medium and high demand scenarios, respectively. Under BAU, industries will still be extracting groundwater to meet the existing demand of 26.78 GL/yr, but will not have any more groundwater allocation to meet demands that are greater than 26.78 GL/yr because the MAR scheme is not implemented.

In order to continue with business as usual into the future, heavy industries will source additional KWRP and scheme water to meet their demand requirements.

Figure 10.20 shows that in the BAU scenario, the increased demand (above 26.78 GL/yr) can only be met by either KWRP or scheme water. MAR is not available under the BAU scenario. Industries will try to use KWRP first because KWRP costs less than scheme. However, there is only 3.6 GL/yr of KWRP that could be expanded (Department of Water, 2013), so the rest of the growing demand will have to be met by scheme water.

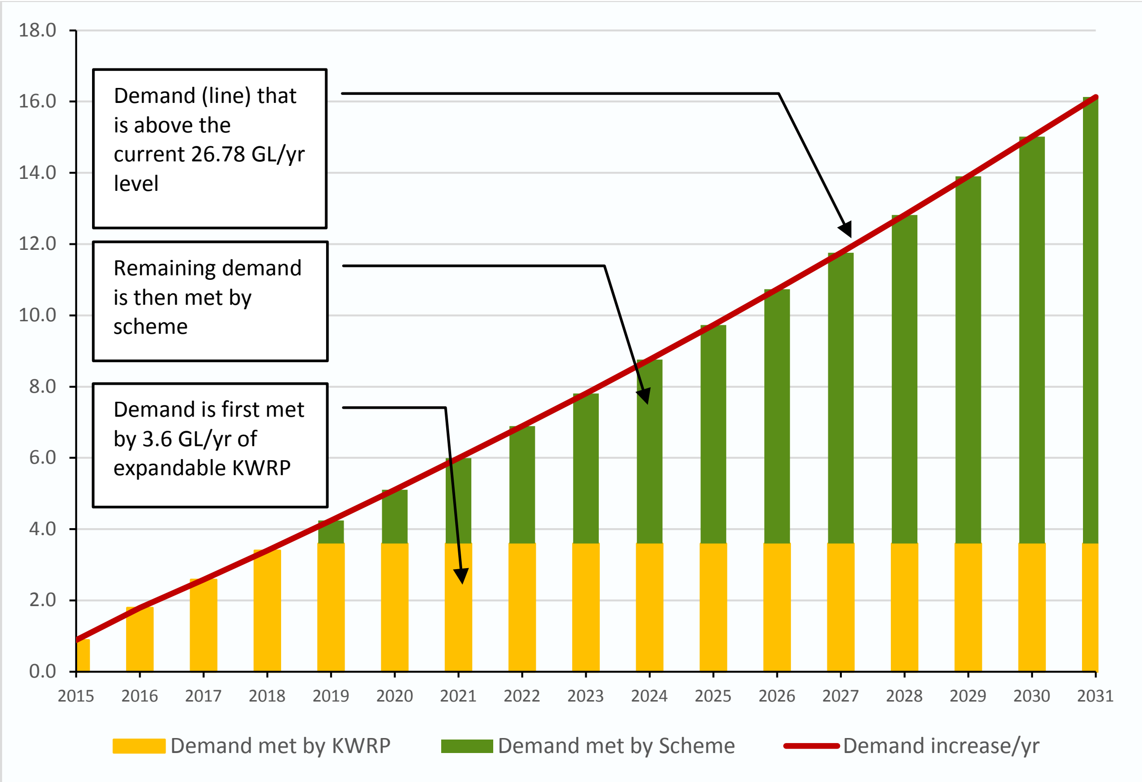

Figure 10.21 Predicted demand (in GL/yr) up to 2031 and supply source for heavy industry under the BAU high demand scenario

Based on these assumptions, the cost of buying KWRP and scheme water up to 2031 is estimated to be $85M, $115M, and $139M for the low, medium, and high demand scenario, respectively (in NPV$). The benefit of alternative scenarios is that heavy industries can avoid paying for the cost of KWRP and scheme water.

Alternative scenarios

In contrast to the BAU scenario, heavy industry would have access to more groundwater in the future through the various MAR scenarios. Therefore, to meet additional water demand in the future, industries will first try to meet that excess demand using groundwater (i.e. the additional groundwater that comes from the MAR project), followed by KWRP water and then by scheme water. Figure 10.21 shows the amount of water from each source that industries will use to meet the growing demand each year that is above 26.78 GL/yr. In this first year (2015/2016) it is assumed that the MAR facilities are under construction and industries will use KWRP water to meet their demand in that year. Once MAR becomes available in the following year, industries will switch to MAR water to meet their growing demand first.

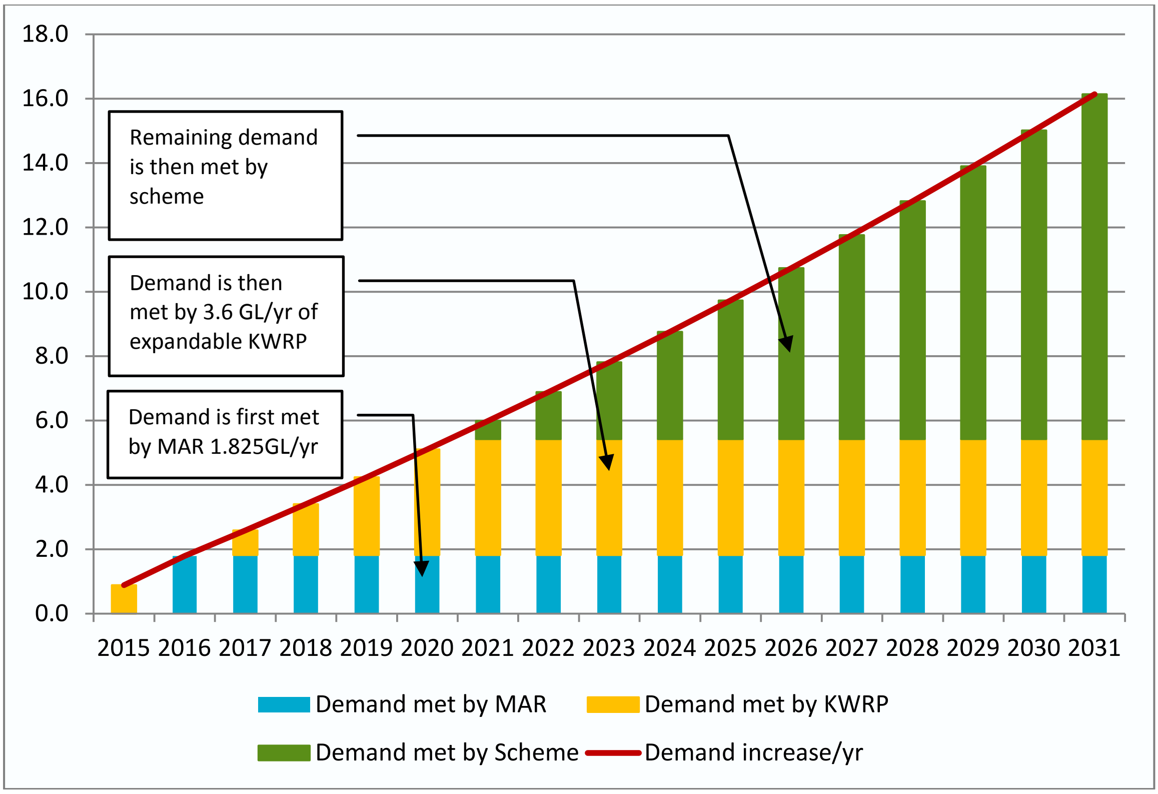

Figure 10.22 Predicted demand (in GL/yr) for heavy industry from each supply source under the alternative scenario, based on infiltration of MAR water of 5 ML/day for the high demand scenario

The demand assumption used in this analysis followed a constant annual growth rate for the periods of 2015 to 2021 and 2021 to 2031. In practice, growth in demand is likely to be in step changes as businesses decide to expand operations or enter the industrial area.

The alternative scenarios under consideration are based on the location at which infiltration through MAR will occur (sites N, E1, E2, E3, S1, S2), the amount of water infiltrated per day (5 ML/d, 10 ML/d or >4.7 ML/d), the method of infiltration (recharge basin, infiltration galleries), and the level of treatment before infiltration (solids removal only, solid removals and nitrogen reduction).

Table 10.5 Alternative water supply scenarios

SCENARIO	SITE	SUPPLY RATE (ML/d)	INFILTRATION METHOD	TREATMENT METHOD	
				NITROGEN REDUCTION	SOLIDS REDUCTION
1	N	5	Recharge basins	No	No
2	N	5	Recharge basins	Yes	Yes
3	N	5	Galleries	No	Yes
4	N	5	Galleries	Yes	Yes
5	N	10	Recharge basins	No	No
6	N	10	Recharge basins	Yes	Yes
7	N	10	Galleries	No	Yes
8	N	10	Galleries	Yes	Yes
9	E1	<5	Recharge basins	No	No
10	E2	<5	Recharge basins	No	No
11	E2	<5	Galleries	No	No
12	E3	<5	Recharge basins	No	No
13	E3	<5	Galleries	No	No
14	S1	5	Recharge basins	No	No
15	S1	5	Recharge basins	Yes	Yes
16	S1	5	Galleries	No	Yes
17	S1	5	Galleries	Yes	Yes
18	S1	10	Recharge basins	No	No
19	S1	10	Recharge basins	Yes	Yes
20	S1	10	Galleries	No	Yes
21	S1	10	Galleries	Yes	Yes
22	S2	5	Recharge basins	No	No
23	S2	5	Recharge basins	Yes	Yes
24	S2	5	Galleries	No	Yes
25	S2	5	Galleries	Yes	Yes
26	S2	10	Recharge basins	No	No
27	S2	10	Recharge basins	Yes	Yes
28	S2	10	Galleries	No	Yes
29	S2	10	Galleries	Yes	Yes

10.5.3 SENSITIVITY ANALYSIS

Sensitivity analysis was undertaken on the following parameters:

- Demand scenarios – high (base case), medium and low.
- Discount rates of 4% and 10% (7% for base case).
- Price of MAR water – heavy industry can buy treated wastewater from the Water Corporation at $0/kL, $0.25/kL or $0.50/kL ($0/kL is base case). The price of MAR water is not the cost of treating wastewater to the quality suitable for MAR but a 'joint scheme contribution' where the Water Corporation requires a user to contribute towards the cost of operating the wastewater system.
- Loss to aquifer – under MAR, there is a chance that not all of the water infiltrated can be recovered for industrial use due to the natural loss to the aquifer (e.g. some will be

evaporated and some will discharge to Cockburn Sound). There is also the possibility that the regulator may require a portion of groundwater to be retained for environmental purposes in a drying climate. More detailed analysis is required to determine how much water will be lost exactly. For this analysis four levels of loss to the aquifer were considered – 0% (base case), 20%, 30% and 40%. Some loss of groundwater allocations may take place even without MAR because of declining levels, the loss of wetlands and/or seawater intrusion.

10.5.4 RESULTS

Table 10.6 presents a summary of the benefit cost analysis of the base case scenario (i.e. high demand, 7% interest rate, zero loss to aquifer, treated wastewater price of $0/kL). The values in the table are the benefit cost ratio of each scenario. A value greater than 1 indicates that the benefits exceed the costs, a desirable outcome. The higher the benefit cost ratio, the more desirable the outcome. The following bullet points summarises findings from this table:

- The benefits outweigh the costs in all cases.
- Despite recharge basins having higher operating costs (GHD 2015), they were the most cost effective method in this analysis when assessed on sites N, S1, S2 and E1.
- As expected the more treatment that is required to remove suspended solids and nitrogen, the higher the cost becomes. High treatment levels reduce benefit to cost ratios but they still exceed 1.
- For sites N, S1 and S2, the benefit to cost ratio is higher where it was possible to infiltrate 10 ML/d compared with 5 ML/d. However, there are still advantages of infiltrating smaller volumes at sites E1, E2 and E3 which are limited by the availability of TWW.
- For each type of infiltration and treatment method, the most cost effective location is site N, followed by sites S2 and S2.
- Worst site – The least cost effective location is site E3, regardless of whether the infiltration method is via a recharge basin or galleries. This is most likely driven by the cost of the pipes as site E3 is quite far away from the WWTP and the available volumes are small.

Table 10.6 Benefit cost ratio for each of the MAR scenarios for the base case (ratios are NPV of benefits over NPV of costs)

SITE	INFILTRATION AND TREATMENT METHOD	5ML/DAY	10ML/DAY	>4.7ML/DAY
N	Option 1: Recharge basin	4.48	4.73	
	Option 2: Recharge basin (nitrogen & solid reduction)	1.81	2.02	
	Option 3: Galleries (nitrogen reduction)	2.48	2.65	
	Option 4: Galleries (nitrogen & solid reduction)	1.62	1.79	
S1	Option 1: Recharge basin	4.11	4.41	
	Option 2: Recharge basin (nitrogen & solid reduction)	1.75	1.95	
	Option 3: Galleries (nitrogen reduction)	2.34	2.55	
	Option 4: Galleries (nitrogen & solid reduction)	1.56	1.74	
S2	Option 1: Recharge basin	3.32	3.61	
	Option 2: Recharge basin (nitrogen & solid reduction)	1.58	1.78	
	Option 3: Galleries (nitrogen reduction)	2.06	2.26	
	Option 4: Galleries (nitrogen & solid reduction)	1.43	1.60	
E1	Option 1: Recharge basin			2.65
E2	Option 1: Recharge basin			1.66
	Option 2: Galleries			1.47
E3	Option 1: Recharge basin			1.36
	Option 2: Galleries			1.24

Interest rates may have effect on the cost of E3 if the price of TWW is not $0/kL. At an interest rate of 10% infiltrating at site E3 cost exceed benefits if the cost of TWW is $0.25/kL (Table 10.15).

Under a 7% interest rate assumption, MAR is a cost effective option even if heavy industries have to pay up $0.5/kL for treated wastewater. MAR becomes even more cost effective as the price of treated wastewater approaches $0/kL. Table 10.7 provides a summary of the cost range for each infiltration and treatment method for each site under the three different treated wastewater price assumptions. Table 10.9, Table 10.12 and Table 10.13 provides a more detailed comparison of MAR cost by site, infiltration volume and treatment method.

Table 10.7 Levelised cost of MAR water under different treated wastewater price assumptions

INFILTRATION AND TREATMENT METHOD	TREATED WASTEWATER PRICE		
	@$0.5/kL	@$0.25/kL	@$0/kL
	($/kL)	($/kL)	($/kL)
Recharge basin (no treatment)			
N	0.90-0.96	0.65-0.71	0.40-0.46
S1	0.93-1.00	0.68-0.75	0.43-0.50
S2	1.02-1.12	0.77-0.87	0.52-0.62
E1	1.29	1.04	0.79
E2	1.76	1.51	1.26
E3	2.03	1.78	1.53
Recharge basin (nitrogen & solid reduction)			
N	1.44-1.64	1.19-1.39	0.94-1.14
S1	1.47-1.68	1.22-1.43	0.97-1.18
S2	1.56-1.80	1.31-1.55	1.06-1.30
Galleries (no treatment)			
E2	1.92	1.67	1.42
E3	2.19	1.94	1.69
Galleries (nitrogen reduction)			
N	1.21-1.33	0.96-1.08	0.71-0.83
S1	1.24-1.38	0.99-1.13	0.74-0.88
S2	1.34-1.50	1.09-1.25	0.84-1.00
Galleries (nitrogen & solid reduction)			
N	1.56-1.77	1.31-1.52	1.06-1.27
S1	1.59-1.82	1.34-1.57	1.09-1.32
S2	1.68-1.94	1.43-1.69	1.18-1.44

Note:
1. All other assumptions are at the base case values
2. The lower and upper bound values correspond to the 10ML/day and 5ML/day injection rates, respectively

Losses to the aquifer of 20-40% have significant effects on the cost effectiveness of a number of alternative scenarios (Table 10.8; Table 10.14). As expected, the higher the amount lost to the aquifer, the less cost effective a scenario becomes. E3 is not cost effective under 20 and 30% losses and high pre-treatment assumptions but all other scenarios remain cost effective.

Table 10.8 Sites that become cost ineffective (i.e. costs outweigh benefits) at various loss to aquifer ratios

SITE	INFILTRATION AND TREATMENT METHOD	LOSS TO AQUIFER			
		0%	20%	30%	40%
N	Recharge basin				
	Recharge basin (nitrogen & solid reduction)				
	Galleries (nitrogen reduction)				
	Galleries (nitrogen & solid reduction)				a
S1	Recharge basin				
	Recharge basin (nitrogen & solid reduction)				
	Galleries (nitrogen reduction)				
	Galleries (nitrogen & solid reduction)				a
S2	Recharge basin				
	Recharge basin (nitrogen & solid reduction)				a
	Galleries (nitrogen reduction)				
	Galleries (nitrogen & solid reduction)				a
E1	Recharge basin				
E2	Recharge basin				c
	Galleries				c
E3	Recharge basin			c	c
	Galleries		c	c	c

a = 5 ML/d infiltration; b = 10 ML/d infiltration; c => 4.7 ML/d infiltration

10.5.5 DISCUSSION

The analysis shows that infiltration of treated wastewater via MAR gives industries another source of water that is cheaper than KWRP and scheme water. Industries will choose to use as much groundwater as possible first in order to save costs. Assuming that the DoW allows for a 1:1 injection to abstraction rate (base case), the change in the consumption of each water source, as compared to the BAU scenario, is shown in Figure 10.22.

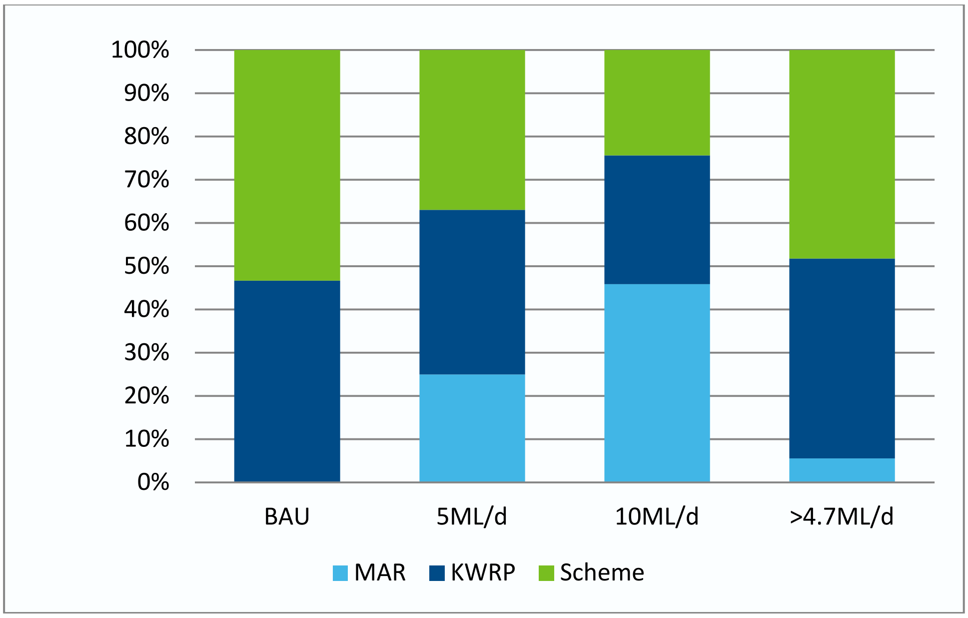

Figure 10.23 Percentage of additional demand met by different sources of water under the BAU and alternative MAR scenarios for the high demand projection

By extracting more MAR water from existing bores industries could significantly reduce the amount of scheme water they would have to buy to meet their demands, particularly if 10 ML/d of water is infiltrated. Under infiltration of 10 ML/d, industries would only have to supplement 24% of their demand requirements (above 26.78 GL/yr) with scheme water, rather than 53% per year under the BAU scenario. This assumes that large volumes of scheme water can be delivered to the KIC area.

The cost effectiveness of each site is affected by the distance of the pipeline from the treated wastewater access point to each site. Site N is the clearly the most cost effective site because it is only 150 m (in pipe length) away from the wastewater access point. Having sites built further away from the wastewater source requires significant benefits to outweigh the costs associated with infrastructure costs. The cost effectiveness of each treatment method may also vary depending on the quality of the water coming from the two wastewater treatment plants. Lower water quality not only will increase the cost of treatment prior injection, but will also increase the cost of maintenance and possibly monitoring.

Although the benefit cost ratios (Table 10.6) clearly show that it is more cost effective to infiltrate 10 ML/d over 5 ML/d, the benefit of infiltrating 10 ML/d could be further maximised. If we examine only the benefit of infiltrating 10 ML/d versus 5 ML/d (Figure 10.22) the benefit of infiltrating 10 ML/d is lower than 5 ML/d in the first few years. This results from 10 ML/d being injected but only a fraction is used to supplement industry demand increase (i.e. all volumes greater than current demand of 26.78 GL/yr). Hence, there is recharge in excess of demand. Consequently, the full benefit of 10 ML/d is not realised in these first few years. The benefit for the 10 ML/d infiltration rate remains lower than the 5 ML/d infiltration rate until 2019 (Figure 10.23). If, however, there is an expansion of industrial activities in the study area in the near future, and projected water demand in the first 4 years is significantly higher than the predictions used in this analysis, more water will be used by industries, which means more benefits will be accrued from the 10 ML/d infiltration scenario. Alternatively, one could stage the development of MAR to best match the expected industry demand level over time. In this instance, the benefit of MAR could be maximised.

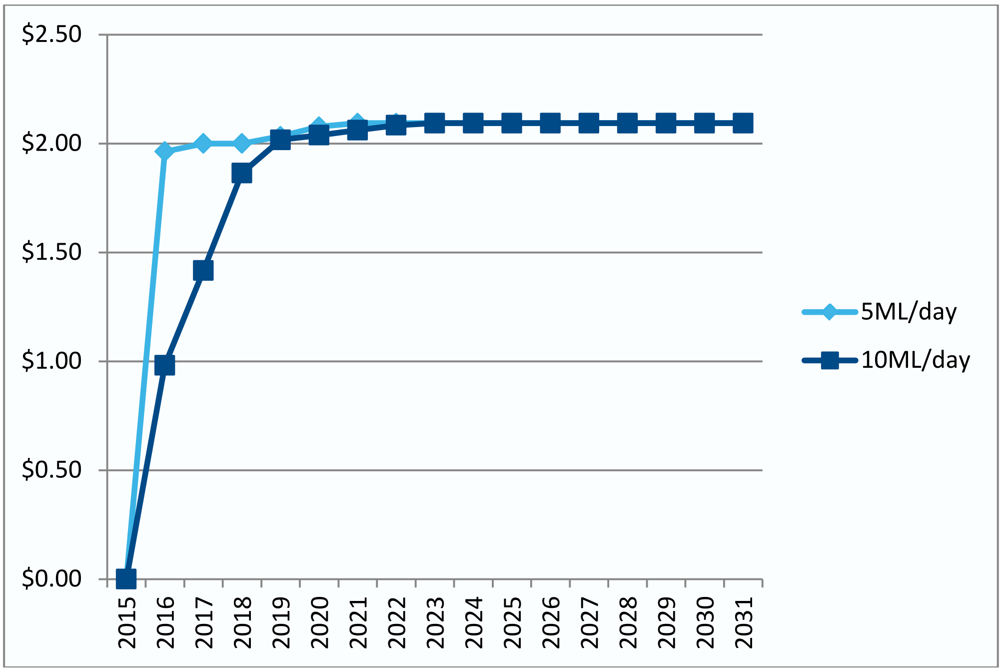

Figure 10.24 Discounted benefits per kilolitre of water infiltrated for 5ML/day and 10ML/day infiltration rates

10.5.6 LIMITATIONS AND CAVEATS

In this analysis, it was assumed that KWRP could be expanded from 6.0 GL/yr to 9.6 GL/yr without any additional capital cost. In reality, KWRP will require capital investment, and without this investment water availability from KWRP will be restricted to 6 GL/yr. Therefore, the price of KWRP water of $2/kL is considered a lower bound estimate of what KWRP water would cost to buy. It is more likely that the price of KWRP would be closer to scheme water if KWRP were to be expanded to 9.6 GL/yr.

It should be noted that unlike the cost of KWRP and scheme water, the cost of MAR estimated by GHD (2015) does not include the cost of land acquisition, monitoring and water quality analysis. As such, the cost of MAR used in this commercial viability analysis is considered a conservative cost estimate. Nonetheless, for most of the MAR options presented here, the benefits outweigh the costs quite significantly.

The drying climate limits the sources of water that industries could use in the future e.g. stormwater is expected to be limited due to reduced rainfall. As such, industries are likely to depend more on climate independent sources, such as desalinated sea water or recycled wastewater to meet their demands in the future. However, it is likely that the drying climate will force the prices of water supply up due to scarcity. Hence, industries may have to pay more for scheme or KWRP water in the future i.e. more than the assumption made of $2.03/kL for scheme water and $2/kL for KWRP water. The outcome of this is that industries would be even more likely to want to use MAR water because the cost of other sources would become higher in the future.

The environmental benefits stemming from groundwater replenishment not covered in this analysis include the maintenance and enhancement of wetlands and groundwater dependent ecosystems, prevention of soil acidification from drying wetlands, prevention of saltwater intrusion, and the increased security of water for industry. These benefits could be monetised using appropriate non-market valuation methods when reliable models as a response to groundwater change become available. On the other hand, any negative environmental impacts arising from MAR that could have been monitised, such as the risk of eutrophication to wetlands and Cockburn Sound, were not included in this analysis either. However, if groundwater abstraction by industries were to increase to equal the volume recharged via MAR and effectively captures the infiltrated TWW, then it is unlikely that there would be significant nutrient discharges to Cockburn Sound.

Other social benefits not included in this analysis include the benefits to local government of being able to access more groundwater in the future for irrigation of public open space. However, this is subject to local government being granted a license and the groundwater quality being suitable and safe for public open space irrigation.

The main ecological benefits stemming from groundwater replenishment is the maintenance and enhancement of freshwater and marine ecosystems, which lead to biological diversity, and abundance of ecological processes (Huynh, Martin, & Moscovis, 2013)

Wetlands have been shown to have high amenity value to surrounding households (see e.g Tapsuwan et al., 2009; Tapsuwan et al., 2012). Wetlands can be considered to be a high value user of water. Some pumping restrictions are currently in place in the study area to protect the environment, for example, shallow groundwater abstraction for industrial use is prohibitive in the Rockingham Industrial Zone due to environmental approval conditions EPBC 2010/5337 (Hyd2o, 2013).

One of the benefits not accounted for in this analysis is the increase in the number of jobs in the study area. If heavy industries are not constrained by the supply of water and are able to grow their businesses as projected by the Department of Water (2013), then more jobs will be created as a result.

In the future, the increase of KWRP water supply to the industrial area will require an expansion of the existing pipe network to consumers, which will significantly increase the cost of this option as a source of water supply. Due to the lack of information around the costs associated with expanding KWRP, the additional costs have not been covered in the analysis. The increase of scheme water supply to the industrial area, on the other hand, can still be supplied through the existing pipe network. Therefore, the cost of increasing supply of scheme water to the industrial area will only increase marginally from amplification costs.

Future CBA for MAR in the study area should include the costs and benefits of MAR for light industries in the study area when projected demand for light industries becomes available.

The cost and benefits estimates presented in this study are sensitive to the changes in the demand projection figures. The timing and magnitude of the demand projection itself will depend on local and global market demands for heavy industry goods, the availability of gas as a source of energy for production, and the value of the Australian dollar. The supply of groundwater to existing industries may also not be assured and the DoW is conducting a review of groundwater allocation limits in 2015.

Table 10.9 High demand, 7% interest rate, $0/kL TWW cost, 0% loss to aquifer (all values are expressed in NPVs)

SCENARIOS		5ML/DAY		10ML/DAY		>4.7ML/DAY	
	Total injection volume	17,240,134		34,480,267		3,834,678	
	Total benefit	$35,469,486	$2.06/kL	$65,081,808	$1.89/kL	$8,000,708	$2.09/kL
SITE	**INFILTRATION & TREATMENT METHOD**	**TOTAL COST**	**COST/KL**	**TOTAL COST**	**COST/KL**	**TOTAL COST**	**COST/KL**
N	Option 1: Recharge basin	$7,909,023	$0.46	$13,746,812	$0.40	-	-
	Option 2: Recharge basin (nitrogen & solid reduction)	$19,607,877	$1.14	$32,286,937	$0.94	-	-
	Option 3: Galleries (nitrogen reduction)	$14,329,350	$0.83	$24,535,945	$0.71	-	-
	Option 4: Galleries (nitrogen & solid reduction)	$21,902,266	$1.27	$36,437,152	$1.06	-	-
S1	Option 1: Recharge basin	$8,622,409	$0.50	$14,752,211	$0.43	-	-
	Option 2: Recharge basin (nitrogen & solid reduction)	$20,321,263	$1.18	$33,292,336	$0.97	-	-
	Option 3: Galleries (nitrogen reduction)	$15,142,736	$0.88	$25,541,344	$0.74	-	-
	Option 4: Galleries (nitrogen & solid reduction)	$22,715,652	$1.32	$37,442,551	$1.09	-	-
S2	Option 1: Recharge basin	$10,682,114	$0.62	$18,035,961	$0.52	-	-
	Option 2: Recharge basin (nitrogen & solid reduction)	$22,380,969	$1.30	$36,576,085	$1.06	-	-
	Option 3: Galleries (nitrogen reduction)	$17,202,441	$1.00	$28,825,093	$0.84	-	-
	Option 4: Galleries (nitrogen & solid reduction)	$24,775,357	$1.44	$40,726,301	$1.18	-	-
E1	Option 1: Recharge basin	-	-	-	-	$3,021,509	$0.79
E2	Option 1: Recharge basin	-	-	-	-	$4,828,762	$1.26
	Option 2: Galleries	-	-	-	-	$5,428,762	$1.42
E3	Option 1: Recharge basin	-	-	-	-	$5,863,096	$1.53
	Option 2: Galleries	-	-	-	-	$6,463,096	$1.69

Table 10.10 High demand, 4% interest rate, $0/kL TWW cost, 0% loss to aquifer (all values are expressed in NPVs)

SCENARIOS		5ML/DAY		10ML/DAY		>4.7ML/DAY	
	Total injection volume	21,265,439		42,530,879		5,127,227	
	Total benefit	$43,853,651	$2.06/kL	$81,568,764	$1.92/kL	$10,704,273	$2.09/kL
SITE	INFILTRATION & TREATMENT METHOD	TOTAL COST	COST/kL	TOTAL COST	COST/kL	TOTAL COST	COST/kL
N	Option 1: Recharge basin	$9,335,386	$0.44	$16,372,770	$0.38	-	-
	Option 2: Recharge basin (nitrogen & solid reduction)	$22,341,487	$1.05	$37,093,670	$0.87	-	-
	Option 3: Galleries (nitrogen reduction)	$16,320,822	$0.77	$28,116,652	$0.66	-	-
	Option 4: Galleries (nitrogen & solid reduction)	$24,868,050	$1.17	$41,675,882	$0.98	-	-
S1	Option 1: Recharge basin	$10,075,246	$0.47	$17,426,127	$0.41	-	-
	Option 2: Recharge basin (nitrogen & solid reduction)	$23,081,346	$1.09	$38,147,026	$0.90	-	-
	Option 3: Galleries (nitrogen reduction)	$17,160,682	$0.81	$29,170,009	$0.69	-	-
	Option 4: Galleries (nitrogen & solid reduction)	$25,707,910	$1.21	$42,729,238	$1.00	-	-
S2	Option 1: Recharge basin	$12,359,027	$0.58	$21,149,702	$0.50	-	-
	Option 2: Recharge basin (nitrogen & solid reduction)	$25,365,128	$1.19	$41,870,602	$0.98	-	-
	Option 3: Galleries (nitrogen reduction)	$19,444,464	$0.91	$32,893,584	$0.77	-	-
	Option 4: Galleries (nitrogen & solid reduction)	$27,991,691	$1.32	$46,452,814	$1.09	-	-
E1	Option 1: Recharge basin	-	-	-	-	$3,604,649	$0.70
E2	Option 1: Recharge basin	-	-	-	-	$5,494,653	$1.07
	Option 2: Galleries	-	-	-	-	$6,094,653	$1.19
E3	Option 1: Recharge basin	-	-	-	-	$6,535,716	$1.27
	Option 2: Galleries	-	-	-	-	$7,135,716	$1.39

Table 10.11 High demand, 10% interest rate, $0/kL TWW cost, 0% loss to aquifer (all values are expressed in NPVs)

SCENARIOS		5ML/DAY		10ML/DAY		>4.7ML/DAY	
	Total injection volume	14,278,268		28,556,537		2,928,080	
	Total benefit	$29,306,913	$2.05/kL	$53,013,558	$1.86/kL	$6,104,773	$2.08/kL
SITE	INFILTRATION & TREATMENT METHOD	TOTAL COST	COST/kL	TOTAL COST	COST/kL	TOTAL COST	COST/kL
N	Option 1: Recharge basin	$6,859,489	$0.48	$11,814,603	$0.41	-	-
	Option 2: Recharge basin (nitrogen & solid reduction)	$17,596,457	$1.23	$28,750,089	$1.01	-	-
	Option 3: Galleries (nitrogen reduction)	$12,864,003	$0.90	$21,901,220	$0.77	-	-
	Option 4: Galleries (nitrogen & solid reduction)	$19,720,009	$1.38	$32,582,436	$1.14	-	-
S1	Option 1: Recharge basin	$7,553,395	$0.53	$12,784,715	$0.45	-	-
	Option 2: Recharge basin (nitrogen & solid reduction)	$18,290,363	$1.28	$29,720,200	$1.04	-	-
	Option 3: Galleries (nitrogen reduction)	$13,657,909	$0.96	$22,871,332	$0.80	-	-
	Option 4: Galleries (nitrogen & solid reduction)	$20,513,915	$1.44	$33,552,547	$1.17	-	-
S2	Option 1: Recharge basin	$9,448,223	$0.66	$15,744,835	$0.55	-	-
	Option 2: Recharge basin (nitrogen & solid reduction)	$20,185,190	$1.41	$32,680,320	$1.14	-	-
	Option 3: Galleries (nitrogen reduction)	$15,552,736	$1.09	$25,831,452	$0.90	-	-
	Option 4: Galleries (nitrogen & solid reduction)	$22,408,743	$1.57	$36,512,667	$1.28	-	-
E1	Option 1: Recharge basin	-	-	-	-	$2,604,880	$0.89
E2	Option 1: Recharge basin	-	-	-	-	$4,352,470	$1.49
	Option 2: Galleries	-	-	-	-	$4,952,470	$1.69
E3	Option 1: Recharge basin	-	-	-	-	$5,381,708	$1.84
	Option 2: Galleries	-	-	-	-	$5,981,708	$2.04

Table 10.12 High demand, 7% interest rate, $0.25/kL TWW cost, 0% loss to aquifer (all values are expressed in NPVs)

SCENARIOS		5ML/DAY		10ML/DAY		>4.7ML/DAY	
	Total injection volume	17,240,134		34,480,267		3,834,678	
	Total benefit	$35,469,486	$2.06/kL	$65,081,808	$1.89/kL	$8,000,708	$2.09/kL
SITE	INFILTRATION & TREATMENT METHOD	TOTAL COST	COST/kL	TOTAL COST	COST/kL	TOTAL COST	COST/kL
N	Option 1: Recharge basin	$12,219,056	$0.71	$22,366,879	$0.65	-	-
	Option 2: Recharge basin (nitrogen & solid reduction)	$23,917,911	$1.39	$40,907,004	$1.19	-	-
	Option 3: Galleries (nitrogen reduction)	$18,639,384	$1.08	$33,156,012	$0.96	-	-
	Option 4: Galleries (nitrogen & solid reduction)	$26,212,300	$1.52	$45,057,219	$1.31	-	-
S1	Option 1: Recharge basin	$12,932,442	$0.75	$23,372,278	$0.68	-	-
	Option 2: Recharge basin (nitrogen & solid reduction)	$24,631,297	$1.43	$41,912,403	$1.22	-	-
	Option 3: Galleries (nitrogen reduction)	$19,452,770	$1.13	$34,161,411	$0.99	-	-
	Option 4: Galleries (nitrogen & solid reduction)	$27,025,686	$1.57	$46,062,618	$1.34	-	-
S2	Option 1: Recharge basin	$14,992,148	$0.87	$26,656,028	$0.77	-	-
	Option 2: Recharge basin (nitrogen & solid reduction)	$26,691,002	$1.55	$45,196,152	$1.31	-	-
	Option 3: Galleries (nitrogen reduction)	$21,512,475	$1.25	$37,445,160	$1.09	-	-
	Option 4: Galleries (nitrogen & solid reduction)	$29,085,391	$1.69	$49,346,367	$1.43	-	-
E1	Option 1: Recharge basin	-	-	-	-	$3,980,178	$1.04
E2	Option 1: Recharge basin	-	-	-	-	$5,787,431	$1.51
	Option 2: Galleries	-	-	-	-	$6,387,431	$1.67
E3	Option 1: Recharge basin	-	-	-	-	$6,821,765	$1.78
	Option 2: Galleries	-	-	-	-	$7,421,765	$1.94

Table 10.13 High demand, 7% interest rate, $0.50/kL TWW cost, 0% loss to aquifer (all values are expressed in NPVs)

SCENARIOS		5ML/DAY		10ML/DAY		>4.7ML/DAY	
	Total injection volume	17,240,134		34,480,267		3,834,678	
	Total benefit	$35,469,486	$2.06/kL	$65,081,808	$1.89/kL	$8,000,708	$2.09/kL
SITE	INFILTRATION & TREATMENT METHOD	TOTAL COST	COST/kL	TOTAL COST	COST/kL	TOTAL COST	COST/kL
N	Option 1: Recharge basin	$16,529,090	$0.96	$30,986,946	$0.90	-	-
	Option 2: Recharge basin (nitrogen & solid reduction)	$28,227,944	$1.64	$49,527,070	$1.44	-	-
	Option 3: Galleries (nitrogen reduction)	$22,949,417	$1.33	$41,776,079	$1.21	-	-
	Option 4: Galleries (nitrogen & solid reduction)	$30,522,333	$1.77	$53,677,286	$1.56	-	-
S1	Option 1: Recharge basin	$17,242,476	$1.00	$31,992,345	$0.93	-	-
	Option 2: Recharge basin (nitrogen & solid reduction)	$28,941,330	$1.68	$50,532,469	$1.47	-	-
	Option 3: Galleries (nitrogen reduction)	$23,762,803	$1.38	$42,781,478	$1.24	-	-
	Option 4: Galleries (nitrogen & solid reduction)	$31,335,719	$1.82	$54,682,685	$1.59	-	-
S2	Option 1: Recharge basin	$19,302,181	$1.12	$35,276,094	$1.02	-	-
	Option 2: Recharge basin (nitrogen & solid reduction)	$31,001,035	$1.80	$53,816,219	$1.56	-	-
	Option 3: Galleries (nitrogen reduction)	$25,822,508	$1.50	$46,065,227	$1.34	-	-
	Option 4: Galleries (nitrogen & solid reduction)	$33,395,424	$1.94	$57,966,434	$1.68	-	-
E1	Option 1: Recharge basin	-	-	-	-	$4,938,847	$1.29
E2	Option 1: Recharge basin	-	-	-	-	$6,746,101	$1.76
	Option 2: Galleries	-	-	-	-	$7,346,101	$1.92
E3	Option 1: Recharge basin	-	-	-	-	$7,780,434	$2.03
	Option 2: Galleries	-	-	-	-	$8,380,434	$2.19

Table 10.14 High demand, 7% interest rate, $0/kL TWW cost, 20% loss to aquifer (all values are expressed in NPVs)

SCENARIOS		5ML/DAY		10ML/DAY		>4.7ML/DAY	
	Total injection volume	17,240,134		34,480,267		3,834,678	
	Total benefit	$28,452,428	$1.65/kL	$54,037,974	$1.57/kL	$6,400,566	$1.67/kL
SITE	INFILTRATION & TREATMENT METHOD	TOTAL COST	COST/kL	TOTAL COST	COST/kL	TOTAL COST	COST/kL
N	Option 1: Recharge basin	$7,909,023	$0.46	$13,746,812	$0.40	-	-
	Option 2: Recharge basin (nitrogen & solid reduction)	$19,607,877	$1.14	$32,286,937	$0.94	-	-
	Option 3: Galleries (nitrogen reduction)	$14,329,350	$0.83	$24,535,945	$0.71	-	-
	Option 4: Galleries (nitrogen & solid reduction)	$21,902,266	$1.27	$36,437,152	$1.06	-	-
S1	Option 1: Recharge basin	$8,622,409	$0.50	$14,752,211	$0.43	-	-
	Option 2: Recharge basin (nitrogen & solid reduction)	$20,321,263	$1.18	$33,292,336	$0.97	-	-
	Option 3: Galleries (nitrogen reduction)	$15,142,736	$0.88	$25,541,344	$0.74	-	-
	Option 4: Galleries (nitrogen & solid reduction)	$22,715,652	$1.32	$37,442,551	$1.09	-	-
S2	Option 1: Recharge basin	$10,682,114	$0.62	$18,035,961	$0.52	-	-
	Option 2: Recharge basin (nitrogen & solid reduction)	$22,380,969	$1.30	$36,576,085	$1.06	-	-
	Option 3: Galleries (nitrogen reduction)	$17,202,441	$1.00	$28,825,093	$0.84	-	-
	Option 4: Galleries (nitrogen & solid reduction)	$24,775,357	$1.44	$40,726,301	$1.18	-	-
E1	Option 1: Recharge basin	-	-	-	-	$3,021,509	$0.79
E2	Option 1: Recharge basin	-	-	-	-	$4,828,762	$1.26
	Option 2: Galleries	-	-	-	-	$5,428,762	$1.42
E3	Option 1: Recharge basin	-	-	-	-	$5,863,096	$1.53
	Option 2: Galleries	-	-	-	-	$6,463,096	$1.69

Table 10.15 High demand, 10% interest rate, $0.25/kL TWW cost, 0% loss to aquifer (all values are expressed in NPVs)

SCENARIOS		5ML/DAY		10ML/DAY		>4.7ML/DAY	
	Total injection volume	14,278,268		28,556,537		2,928,080	
	Total benefit	$29,306,913	$2.05/kL	$53,013,558	$1.86/kL	$6,104,773	$2.08/kL
SITE	INFILTRATION & TREATMENT METHOD	TOTAL COST	COST/kL	TOTAL COST	COST/kL	TOTAL COST	COST/kL
N	Option 1: Recharge basin	$10,429,056	$0.73	$18,953,737	$0.66	-	-
	Option 2: Recharge basin (nitrogen & solid reduction)	$21,166,024	$1.48	$35,889,223	$1.26	-	-
	Option 3: Galleries (nitrogen reduction)	$16,433,570	$1.15	$29,040,354	$1.02	-	-
	Option 4: Galleries (nitrogen & solid reduction)	$23,289,576	$1.63	$39,721,570	$1.39	-	-
S1	Option 1: Recharge basin	$11,122,962	$0.78	$19,923,849	$0.70	-	-
	Option 2: Recharge basin (nitrogen & solid reduction)	$21,859,930	$1.53	$36,859,334	$1.29	-	-
	Option 3: Galleries (nitrogen reduction)	$17,227,476	$1.21	$30,010,466	$1.05	-	-
	Option 4: Galleries (nitrogen & solid reduction)	$24,083,482	$1.69	$40,691,681	$1.42	-	-
S2	Option 1: Recharge basin	$13,017,790	$0.91	$22,883,969	$0.80	-	-
	Option 2: Recharge basin (nitrogen & solid reduction)	$23,754,758	$1.66	$39,819,454	$1.39	-	-
	Option 3: Galleries (nitrogen reduction)	$19,122,303	$1.34	$32,970,586	$1.15	-	-
	Option 4: Galleries (nitrogen & solid reduction)	$25,978,310	$1.82	$43,651,801	$1.53	-	-
E1	Option 1: Recharge basin	-	-	-	-	$3,336,900	$1.14
E2	Option 1: Recharge basin	-	-	-	-	$5,084,490	$1.74
	Option 2: Galleries	-	-	-	-	$5,684,490	$1.94
E3	Option 1: Recharge basin	-	-	-	-	$6,113,728	$2.09
	Option 2: Galleries	-	-	-	-	$6,713,728	$2.29

10.6 General discussion

Several MAR options have been considered in this analysis, and their risks and economic benefits have been separately identified. In this section, we present the outcome of a combined risk and economic assessment of each MAR option.

Based on the baseline economic assumption where the interest rate is at 7%, the cost of TWW is $0/kL and the loss to aquifer is 0%, the benefit-cost ratio (Table 10.6) is presented in graphical format against the risk of MAR, which is represented by the rate of total nitrogen (TN) mobilisation for each MAR site at various injection volume levels (Figure 10.10). The nitrogen mobilisation was used to represent the risk as (i) the economic analysis assumed that all MAR water was abstracted, and (ii) it was the only risk parameter quantifiable due to uncertainties and large number of sites. While the other risk parameters were not quantified they were addressed partially by the economic analysis (e.g. the piping distance/difficulty was accounted in the costing) and site selection was such that impacts on wetlands were minimised.

All MAR options have a benefit-cost ratio greater than one (Figure 10.24) indicating they are cost-effective methods of meeting industrial water requirements. The x-axis portrays the risk of nitrogen mobilisation of each option, where the higher the TN load/risk the higher the risk. The best MAR options are recharge basins at Site N, where the benefit-cost ratio is high and risk is low. The two least favourable scenarios are 10 ML/d infiltration into galleries at Site S1 with solids only and solids plus nitrogen removal, because of low benefit-cost ratio and higher risk level.

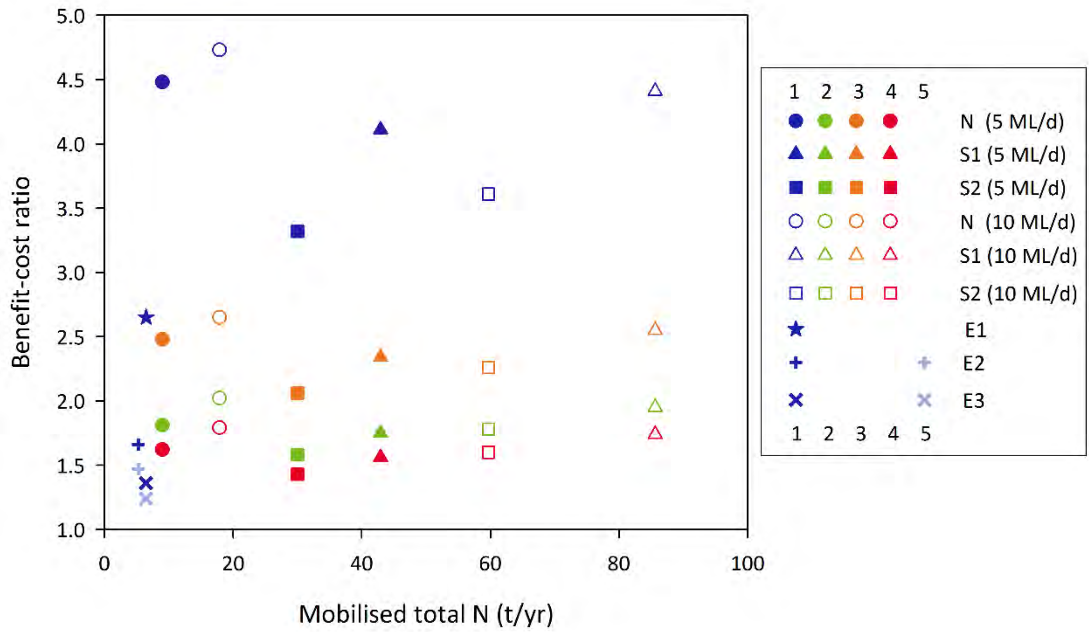

Figure 10.25 Comparison of benefit-cost ratio and environmental risk (potential mobilisation of current groundwater total nitrogen) for various managed aquifer recharge scenarios. 1 = Recharge basin; 2 = Recharge basin with nitrogen and solid reduction; 3 = Galleries with solid reduction); 4 = Galleries with nitrogen and solid reduction; 5=Galleries (no treatment)

Although the level of risk tends to increase with infiltration volume, at Site N, despite an infiltration rate of 10 ML/day, the risk level is lower than for Sites S1 and S2 where injection volume is only 5 ML/day. This difference arises because Site N is further from areas of high groundwater TN concentration than Sites S1 and S2. Thus the impact of the addition of MAR water at Site N has less effect on submarine groundwater discharge (SGD) in the high groundwater TN area relative to Site S1 and S2 which are located within these areas. A similar explanation exists for Sites S1 and S2 although both these sites are situated inland of the area of high groundwater TN. Site S2 has a lower risk than S1 due to the increased distance from the coast which results in the greater spread of the groundwater mound produced as a result of MAR, a greater likelihood of uptake by a bore and a lower SGD and TN mobilisation.

Worth noting are sites E1 to E3 where the benefit-cost ratio may not be as high, but the risk levels are very low compared to the other options. There are two reasons why risks are low at these sites: (i) the injection volume is low relative to sites N, S1 and S2 based on the predicted increase in TWW from the Kwinana WWTP; and (ii) sites E1-E3 are further inland than other sites resulting in a higher likelihood of uptake and minimal changes to SGD volumes.

From this analysis, we conclude that the following key factors influence on the viability of each MAR option:

1. Distance to the coast which impacts on the changes in groundwater levels at the coast and hence SGD.
2. Travelling time to the coast. The greater the distance from the coast the longer the residence time and hence the greater the potential for removal of nutrients within the aquifer by natural processes such as denitrification and for interception by extraction bores
3. The amount MAR water being intercepted. If less than 100% then it is likely that there would be increased nutrient loads entering to Cockburn Sound and wetlands down-gradient of MAR sites.
4. Industrial water quality impacts. As the end use is mainly for industrial non-potable uses the risks are low. However the closer to the MAR site the higher the risk of intercepting low quality water.
5. Distance to the wastewater treatment plant. Long distances to the wastewater access point has a significant effect on the infrastructure cost of each MAR option. In this analysis, we attempted to locate each MAR site where the project could most feasibly be constructed i.e. not on private property, not interfering with existing infrastructure, etc.
6. Injection volume. High injection volumes reduce the average cost per kilolitre of the MAR project, as well as increase water supply reliability to end-users. However, the economic benefits need to be balanced with the risk of increased N and P loads entering Cockburn Sound.
7. Treatment and infiltration method. The economic analysis showed that treatment of treated wastewater increases the cost of MAR water. It would be ideal if the treated wastewater coming from the wastewater treatment plant is treated to the point where no additional treatment is required before MAR. The oxidation ditch process at the Kwinana WWTP most closely approximates this because nutrient concentrations are about an order of magnitude lower than less advanced WWTPs.

Caveats and recommendations

It was estimated that from 2015 to 2019, not all infiltrated MAR will be intercepted by industries because the amount of MAR water infiltrated exceeds the projected industry demand. Under these circumstances, the risk of nitrogen going into the Sound will increase. However, these risks were not monetised and internalised in the benefit-cost analysis due to the lack of information around quantifying the outcomes. Under site-specific conditions, MAR volumes could be adjusted up and down to meet industry needs and bores could be located to intercept all MAR water. It should also be noted that the groundwater modelling did not incorporate any increase in abstraction in response to MAR, therefore groundwater responses may be less and travel times greater if current allocations are increased or new abstraction bores installed. This will have a flow-on effect on the risks associated with MAR as only bores in the travel path will intercept nutrients thus reducing the associated risks of SGD, although the zone of groundwater rise is much larger.

Timing of groundwater abstraction is also another aspect of the study which could be studied in more detail. It was assumed in the economic analysis that industries would be permitted to abstract the same amount of groundwater that was injected via MAR the same year the MAR project commences. In reality, groundwater travelling time may prevent industries from accessing MAR water straight away. The longer the travelling time from the point of injection to the point of abstraction (or pressure response) the longer the benefit of MAR is delayed, which would have implication on the benefit-cost scenario. Benefits of MAR may also be realised outside the envelope in which TWW itself is expected to impact the aquifer. This results from the increases in groundwater levels (pressure response) which occurs over a much larger area (often 4 to 10 times) than the zone impacted by the actual TWW. Abstraction outside the added TWW zone may therefore have little influence on mitigating the risk of nutrient additions that may be achieved through increased groundwater abstraction.

Future analysis should consider quantifying the risk of MAR for each injection site in order to have a better understanding of them. By focusing on each site at a time, quantification of the risks and mitigation methods will be more precise. Better information of environmental and health risks will also allow for better monetisation of their impact, which can then be internalised into the benefit-cost analysis. It is possible that the ranking of the benefit-cost ratio may change once risks are better estimated.

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11 Discussion

11.1 Questions that the project has helped to clarify

Topics are covered under a series of questions to make discussion areas clear.

How do our results compare with previous analyses?

Previous regional work (e.g. Smith and Pollock 2010) have indicated that there are large parts of the study area that are hydrogeologically suited to MAR and both the groundwater modelling and analysis of discharge from the Kwinana WWTP supports this assessment. Modelling of MAR at Perry Lakes by both CSIRO (McFarlane et al. 2009) and GHD (2011) showed similar responses of groundwater levels to the addition of 1 to 2 GL/yr of treated wastewater to similar soil and aquifer types. Groundwater mounds under historic treated wastewater disposal sites also show similar responses (Smith et al. 2011) to that measured and modelled around the Kwinana WWTP. Particle tracking estimates of where groundwater flows after infiltrating is also close to the measured changes in groundwater nutrient levels after significant upgrading of the Kwinana WWTP in 2009.

There can therefore be some confidence that MAR impacts have been reasonably well modelled in this catchment given the 40 year history of wastewater disposal at sites such as Kwinana and Gordon Road in Mandurah. The advective transport of Kwinana WWTP water has travelled to the northwest and largely captured by groundwater abstraction in the Alcoa residue disposal area with indications that in the long term (30 years travel time) the plume may enter Long Swamp. Based on the modelling there is little evidence that the infiltrated wastewater has made it to Cockburn Sound. The early wastewater treatment processes resulted in total nitrogen concentrations exceeding 40 mg/L but the current concentrations from this site are now an order of magnitude lower; therefore, fewer risks should be present over time. However only a 32% reduction in total phosphorus was observed upon plant upgrade may pose an issue as it appears as though the capacity of the aquifer to remove phosphorus has been exceeded.

How cost effective is Managed Aquifer Recharge (MAR) compared with its alternatives?

The analyses have shown MAR to be a cost effective source of water supply for heavy industries assuming that their current main source of non-potable water (shallow groundwater) becomes less secure and industrial demand continues to grow.

The analysis shows that the cost of MAR ranges between $0.40 and 1.69/kL with the lower costs being for no pre-treatment and infiltration by recharge basins close to the SDOOL and the higher costs being the converse. These cost estimates are based on the assumption of 7% interest rate, $0/kL for accessing treated wastewater and 0% loss to aquifer (i.e. all added water can be recovered for use). Pumping distance and the need to remove nitrogen increase costs the most. When compared to the price of KWRP water ($2.00/kL) or the price of scheme water ($2.093/kL), the cost of MAR is more competitive.

A caveat to these costs is that it is based on the assumption that industries can fully extract the amount of water that has been infiltrated. While there is a chance that a certain amount of infiltrated water will need to be retained for environmental reasons, existing groundwater allocations may have to be foregone to the environment in a drying climate. The cost estimates for MAR also do not include land acquisition costs, scheme monitoring and compliance costs or costs for end user (infrastructure and groundwater pumping costs) where these apply to a scheme.

Technical, logistical, environmental and risk management constraints will be important when considering the cost effectiveness of MAR. Hence, the choice of whether to use MAR over other sources of water will require careful site-specific analyses and possibly a pilot that can be progressively expanded if it proves successful. The need to learn from MAR experience so that it can be adopted in more complex situations may require government support in the form of regulatory support if not financial assistance as the first few cases will almost certainly require additional study and monitoring and therefore be more expensive. However, it may also be important not to set a support precedent that is extended to subsequent MAR cases that no longer need to solve these problems.

How much extra treatment is required to make MAR safe?

The need to remove suspended solids and/or nitrogen to meet operational constraints can increase its cost by between \$0.30 and 0.40/kL. Studies in social psychology and economics have shown that the degree of safety is a matter of perception, and that for some people no level of treatment would be enough for them to use the water (Leviston et al., 2006; Gibson and Burton, 2014). Most of these studies, however, look at household use of MAR water, rather than industrial use. Therefore, findings around acceptance from these studies may not directly apply to industrial users. Nonetheless, the encouraging message from these studies is that acceptance of treated wastewater use increases as proximity to human contact decreases (Po et al., 2005), and as such, there should not be any social resistance from heavy industry using MAR water for their production. There are also encouraging signs of social acceptance of MAR water from the Beenyup wastewater treatment plant project where MAR is being used to augment public drinking water supply.

Apart from concerns to human health, there are also concerns of the impact of MAR on the environment, particular on wetlands, groundwater dependent ecosystems, and seagrasses. The addition of 1.7 GL/yr of treated wastewater within 400m of The Spectacles wetlands did not result in large volumes of treated wastewater entering the lakes, and remote sensing and groundwater measurement (and modelling) have indicated that water levels in the wetlands have benefited because they are the only ones in the eastern Beeliar chain to avoid drying out in recent decades. Bollard Bullrush Swamp has also been relatively resilient and this may be because it is connected by a drain to The Spectacles or it is located lower in the landscape and close to Bassendean palusplain areas. More measurements and surface water-groundwater modelling would need to be carried out to properly answer this question.

Can the community benefit from MAR aimed mainly at assisting industry?

Although the focus of this research project was to examine the benefits of MAR for heavy industry, there are other stakeholders that could potentially benefit from its uptake, especially if carried out in inland and northern areas where watertables respond more and there are several inter-dunal wetlands and higher demands for irrigation water. The project examined the value of wetlands to nearby properties using house price as a hedonic measure i.e. measure of amenity value of wetlands that is capitalised in property sales price. Depending on the number and closeness of the houses, and whether the wetland has recreational and visual appeal (e.g. open water qualities), the impact can be substantial (i.e. hundreds of millions of dollars on property sales price). However, the potential value of MAR to maintaining or increasing the social value of wetlands has not been examined in this report and requires further investigation.

Higher groundwater levels can also translate into greater availability and security of non-potable water for local government to irrigate public open space. One site (Kwinana Golf Course) could benefit were MAR to occur at the S2 location because irrigation water and wetlands could be more secure. Capturing these benefits by a third party MAR proponent may however be difficult.

Would MAR using treated wastewater add significantly to groundwater discharge and nitrogen loads to Cockburn Sound?

The groundwater model developed in this project has further refined past estimates of submarine groundwater discharge and nutrient loads entering Cockburn Sound. Submarine groundwater discharge may increase by 5 to 10% if 1.7 to 3.5 GL/yr of treated wastewater were added to sites adjacent to the Sepia Depression Ocean Outfall Line and therefore adjacent to industry. This assumes that there is no additional abstraction of groundwater by industry to intercept the added water. Total discharge estimates vary widely according to the assumptions that are made about the future climate. If the 2030 climate is even drier than the past decade then discharge volumes could decrease by 25 to 40% making any MAR additions minor. If extra treatment was required to remove nitrogen from the treated wastewater then this would reduce MAR's cost-effectiveness but it would remain profitable under the assumptions that have been made in this analysis. Adding high-quality treated wastewater from the oxidation-ditch Kwinana and East Rockingham WWTPs would reduce the risk of nitrogen loads entering Cockburn Sound compared with SDOOL water from the Woodman Point WWTP. Recharge operations could be scaled back if the projected increase in industrial water use did not materialise. This could be factored into the decision into whether and what type of additional treatment would be required prior to infiltration. Furthermore tailoring of recharge operating scale to water use could also be of benefit in making infiltration more cost effective for industry.

Would additional nitrogen from MAR result in environment impacts to seagrasses in Cockburn Sound?

This is a critically important question for any proponent wanting to undertake MAR in the Cockburn Sound catchment unless the added water was designed to be intercepted by existing or additional industrial use. This was not the subject of work in this project but the question has triggered a study of nitrogen trends, what influences nitrogen concentrations in the water column in the Sound, and what role submarine groundwater discharge plays compared with other nitrogen sources including reworking of bottom sediments. Annual reports on the state of the Sound by the Cockburn Sound Management Council have indicated nitrogen concentrations have reduced and light penetration and nuisance algal growth to have improved in recent years. However seagrasses have continued to decline in both Cockburn and in Warnbro Sounds. This has resulted in a separate study to determine what factors may be affecting seagrass health, including whether a reduction in submarine groundwater discharge may be affecting the supply of critical nutrients. This work will influence whether MAR is seen as a threat or possibly even a benefit to the marine environment.

Can MAR help manage salt water intrusion?

The modelling showed that it is possible to influence the position of the salt water wedge by several hundred metres which may be important where production- or contaminant remediation-bores are located within a few kilometres of the coastline and where the base of the aquifer is deep. The future climate will also affect the location of the wedge, with some scenarios indicating that it could be more influential than MAR. However the groundwater model would need to be improved before a definitive conclusion could be reached. If MAR was operated such that all of the added water was taken by industry its impacts on salt water intrusion may be limited.

Is the Kwinana Industrial Area a safe area to test MAR?

The location of the project has advantages in that the area is zoned for heavy and light industry away from residential and public drinking water users of groundwater and in an area with heavy demand for groundwater which is becoming increasingly unreliable because of a drying climate and projected use. The close proximity of wastewater from the SDOOL and smaller treatment plants at Kwinana and the soon-to-be-commissioned East Rockingham WWTP also makes it very suited. The main drawback is the potential impact of additional nitrogen loads entering Cockburn Sound given its history of seagrass loss because of past nitrogen discharges. However, the current situation of nitrogen and environmental health is unclear, apart from Jervoise Bay in the north where a long-term problem exists of nutrients entering and resulting in eutrophication in a restricted marine environment. Therefore, until the current studies being supported by the Cockburn Sound Management Council and KIC on current nitrogen sources and impact on seagrass are finished this area remains unclear.

Contaminated sites add significant complexity with MAR in the KIA. Currently industry adjusts their pumping rates to avoid contamination entering in their production bores as their only control method. Were they able to add treated wastewater at defined locations and volumes they may be able to extract more production water, and also increase the rates at which they extract and treat contaminated sites without exacerbating seawater intrusion. The chemistry of the contaminated sites and of the treated wastewater would need to be well understood as this could make the contamination harder or easier to remediate. The presence of nutrients and carbon in MAR water may allow some contaminated sites to be remediated in situ.

Acid sulphate soils are believed to be common in wetland areas in the catchment. The drying of peaty soils has been known to result in acidification and the release of heavy metals such as aluminium, arsenic and iron. MAR may be a method of preventing acid sulphate soil formation while maintaining or recovering the environmental, social and economic values of wetlands. If acidified groundwater flows through the Tamala Limestone then the acidity will be neutralised. The drying of wetlands appears also to be releasing nitrogen and other nutrients into the groundwater at similar if not greater rates when wastewater that has undergone oxidation-ditch treatment.

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12 Conclusions and recommendations

12.1 Conclusions

A number of general conclusions can be drawn from the project. Specific findings are detailed in previous chapters and are not repeated here.

- Managed aquifer recharge with treated wastewater is a cost-effective source of non-potable water for future heavy industry in the Kwinana Industrial Area (KIA), under the assumptions used in this investigation. The issues that pose the greatest challenge to advancing MAR in this area related to (1) managing contaminated sites which may be impacted (positively or negatively) by MAR additions, (2) estimating and monitoring interception of added water by existing and new abstractors, including those who may benefit but are outside the treated wastewater plume, and (3) determining required levels of additional water treatment prior to MAR to reduce nitrogen loads entering Cockburn Sound. Further work would be required on specific sites to properly quantify MAR options.
- The KIA is uniquely positioned to benefit from trialling MAR due to the absence of other users, the ability to infiltrate water through low cost open pits, and the proximity of wastewater to areas of demand. Moreover, the KIA has a major impetus for trialling MAR due to emerging water security concerns because of the drying climate and growing industrial water demands. Given the existence of the SDOOL outlet, which will continue to dispose of increasing volumes of treated wastewater via ocean outfall, MAR trials can be conducted and terminated without jeopardizing the continuity of wastewater disposal. A robust system of monitoring of impacts would be required to learn as much as possible from MAR, to obtain early warning of emerging risks and to provide assurance to regulators and the general community that the practice was operating as intended.
- Significant knowledge of MAR has been gained from the 40-year history of adding treated wastewater to the Superficial Aquifer at the Kwinana WWTP in the heart of the catchment. This activity appears to have allowed nearby wetlands that are in hydraulic connection with the underlying aquifer to continue to exist by virtue of increasing wastewater discharge to the aquifer whilst climate has continued to dry. An upgrade of the treatment system at the plant has resulted in greatly improved water quality such that Total Nitrogen levels are now lower than that generated by mineralising peat in the wetland. The previous, poorer-quality wastewater has migrated through the aquifer below the industrial zone and has been partly if not completely removed by pumping bores before reaching Cockburn Sound. The quality of seawater in Cockburn Sound does not appear to have been significantly impacted by any influx of treated wastewater, given its improvement in quality and interception over this period.
- Risks and management costs are associated with the current site-specific factors:
    - The degree to which treated wastewater needs to have solids removed (if added via galleries) or nitrogen (if loads pose a risk to downstream wetlands and Cockburn Sound).
    - Long distances to the wastewater access points will have an effect on the cost of each MAR option.
    - Whether the added water mobilises or interacts in a negative way with contaminated sites in the KIA. If done intelligently, however, MAR could assist in contaminant site management as well as reducing the risk of acid sulphate soil formation caused by the drying of wetlands and associated acid sulphate soils.
    - The fate of added water in terms of its uptake by down-gradient bores, degradation or transformation of nutrients, its mixing with ambient groundwater, and whether it is

expressed at the surface, especially in wetlands. The presence of a well-monitored wastewater infiltration site near The Spectacles wetlands would enable the fate of added water at site E1 to be monitored and ceased if risks became unacceptable. This site had a high benefit-to-cost ratio (2.65).

  - Whether all added treated wastewater can be used by industry or a proportion needs to be used to replace environmental losses due to the drying climate. If industry intercepts all added water from within the treated wastewater plume then it reduces nutrient load risks to Cockburn Sound. However, if the proportion is reduced or abstracted in areas of groundwater level increase outside the treated wastewater plume, the cost effectiveness of MAR is also lowered. In the analysis no account was made for benefits to the environment through improvements to water regimes in wetlands, nor to non-industrial use of groundwater.
  - Distance and travelling time to the coast will have an effect on the potential for removal of nutrients by natural process that occurs within the aquifer, or by interception as has been happening to the treated wastewater added at the Kwinana WWTP since about 1975.
  - Higher injection volumes of treated wastewater lead to economies of scale, which reduces the cost of MAR option. However, the cost savings need to be balanced with the possible risk of increased N and P entering Cockburn Sound.
  - The extent to which MAR can halt or abate the effects of seawater intrusion, especially the siting of MAR locations and volumes of infiltration to gain control of this threat to groundwater quality. Again this must be balanced against the risk of increased nutrient loads to Cockburn Sound.
- Workshops have been held in association with this project on the policies associated with MAR, governance and regulatory practices (in light of the Water Resource Management Bill) as well as conditions for getting third party access to provide water services that are currently not being offered to industry. Determining the most appropriate proponent of a MAR scheme in the KIA was beyond the scope of this investigation, but this issue is very important if regulatory approval were to be sought for a scheme.

12.2 Recommendations

The project has identified the following investigation / research needs:

- Methods to better detect the presence of treated wastewater in aquifers and to estimate the relative proportions of infiltrated wastewater to ambient groundwater over time;
- A fully coupled, density-dependent, solute transport model of MAR options in the KIA to better predict and manage seawater intrusion;
- A detailed investigation of the water budget of The Spectacles wetlands and Bollard Bulrush Swamp to assess how these areas have withstood the drying climate and how this may change under a future dry climate and additional MAR at the Kwinana WWTP site;
- A detailed study of MAR interactions with currently known contaminated sites, and predictive modelling of MAR benefits to the management and/or clean-up of specific contaminated sites
- The effects of nitrogen added to groundwater from both batch and oxidation ditch wastewater treatment plants
- Quantifying the volumetric influx of nitrogen from submarine groundwater discharge (SGD) relative to other components of the nutrient balance in Cockburn Sound; better knowledge of the potential risk to seagrass health from MAR activity and whether SGD adds nutrients (such as silica and potassium) to seagrasses. Some of this work has already commenced, but it is possible that additional follow-up research will be needed.

- A more comprehensive coupled groundwater-surface water model is required to assess wetland responses (e.g. Thomsons Lake) to groundwater recharge, including changes to wetlands water levels and changes to flora and fauna species.
- Better estimations of non-market values associated with changes to wetlands and groundwater dependent ecosystems as a result of MAR. The cost benefit analyses in this report don't include these benefits because of difficulties determining them accurately. The sensitivity of values to changes in groundwater levels are outside the resolution of the groundwater model. A more detailed model would overcome this deficiency and allow these potentially very large benefits to be included in future analyses.
- Determining how the costs and risks identified in this study may be shared, if needed, between industries, government and water service providers in trials of MAR in the KIA area.

It is also recommended that, based on KIC needs for additional water, a trial MAR scheme be established using the options explored in this study as a guide to its location. The recharge rates should be linked to abstraction requirements and/or the need to push back the salt water interface. Several overlapping MAR projects could be implemented over a suitable time scale that is commensurate with increased demand for groundwater and the need to replenish the aquifer in the drying climate. Such a trial could have wide ranging implications for MAR in similar locations along the West Australian coastline. Additional regulatory and research support could enable the KIC area to trial MAR as a demonstration of best practice under DoW's MAR Operational Policy 1.01.

Appendix A

1. Calibrated maps of hydraulic conductivity

The calibrated maps for upper layers in the model (Layers 1 to 4) are given below. The maps of hydraulic conductivity for Layers 5 through Layer 8 are the distributions from PRAMS 3.2 (extracted for this study area) and were not re-calibrated for the purposes of this study.

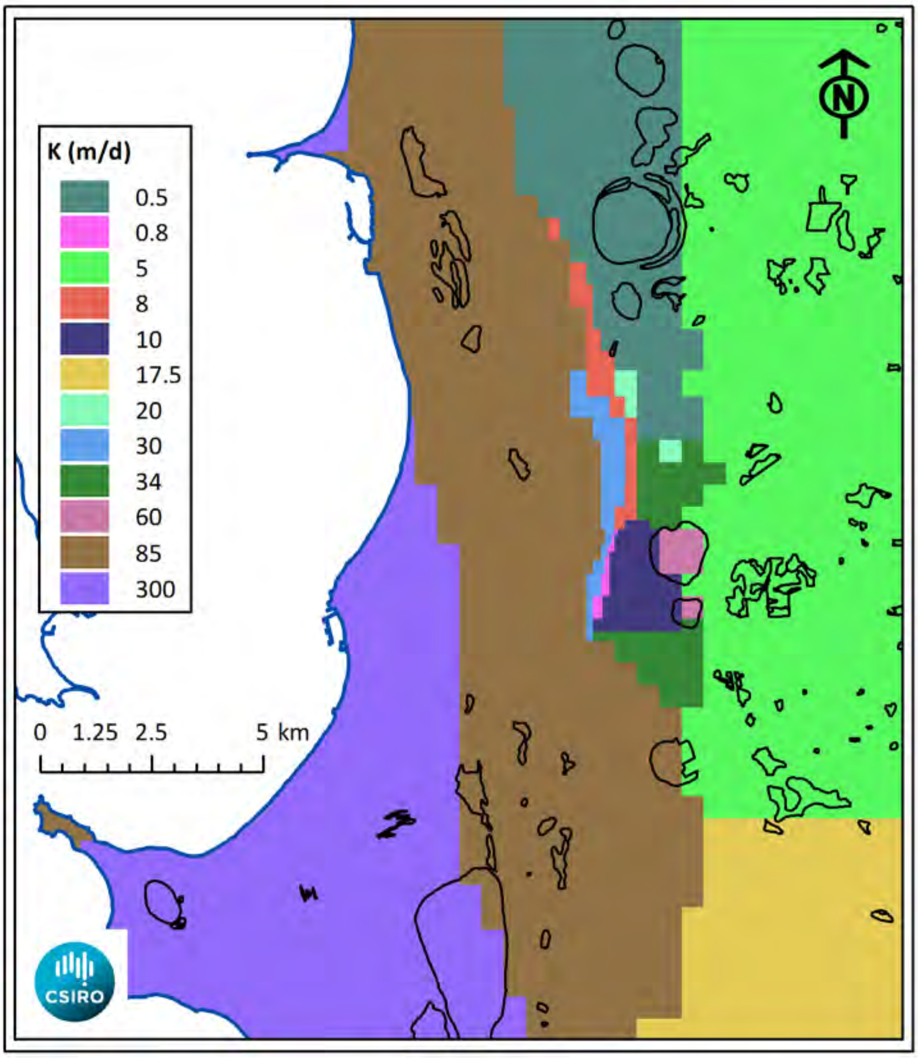


Figure 26 Zones of hydraulic conductivity for Layers 1 and 2

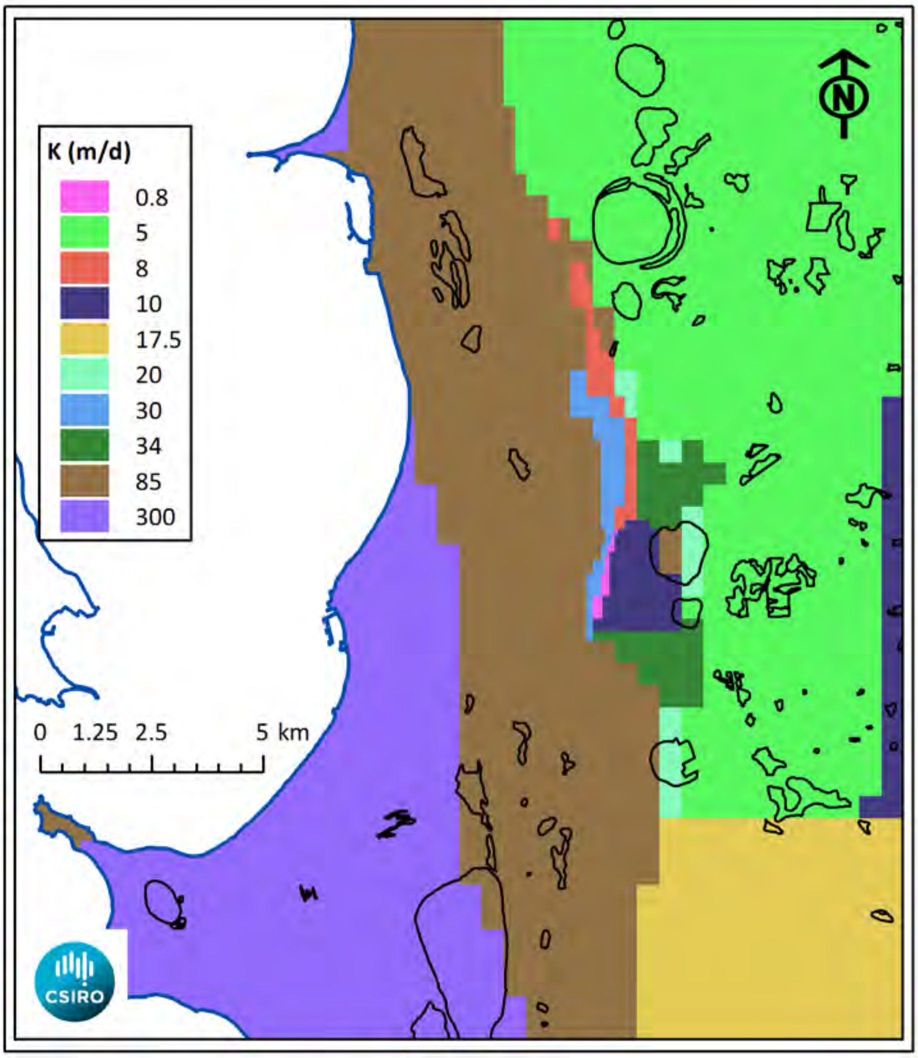


Figure 27 Zones of hydraulic conductivity for Layer 3

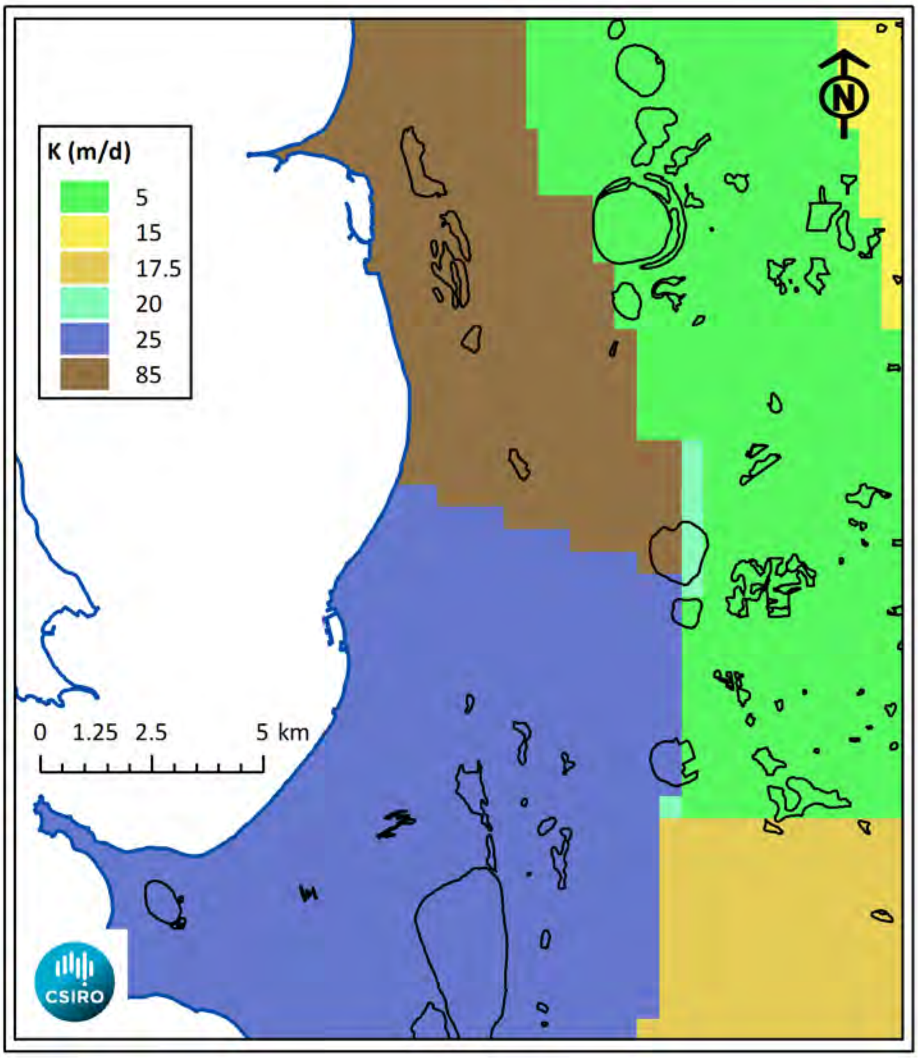


Figure 28 Zones of hydraulic conductivity for Layer 4

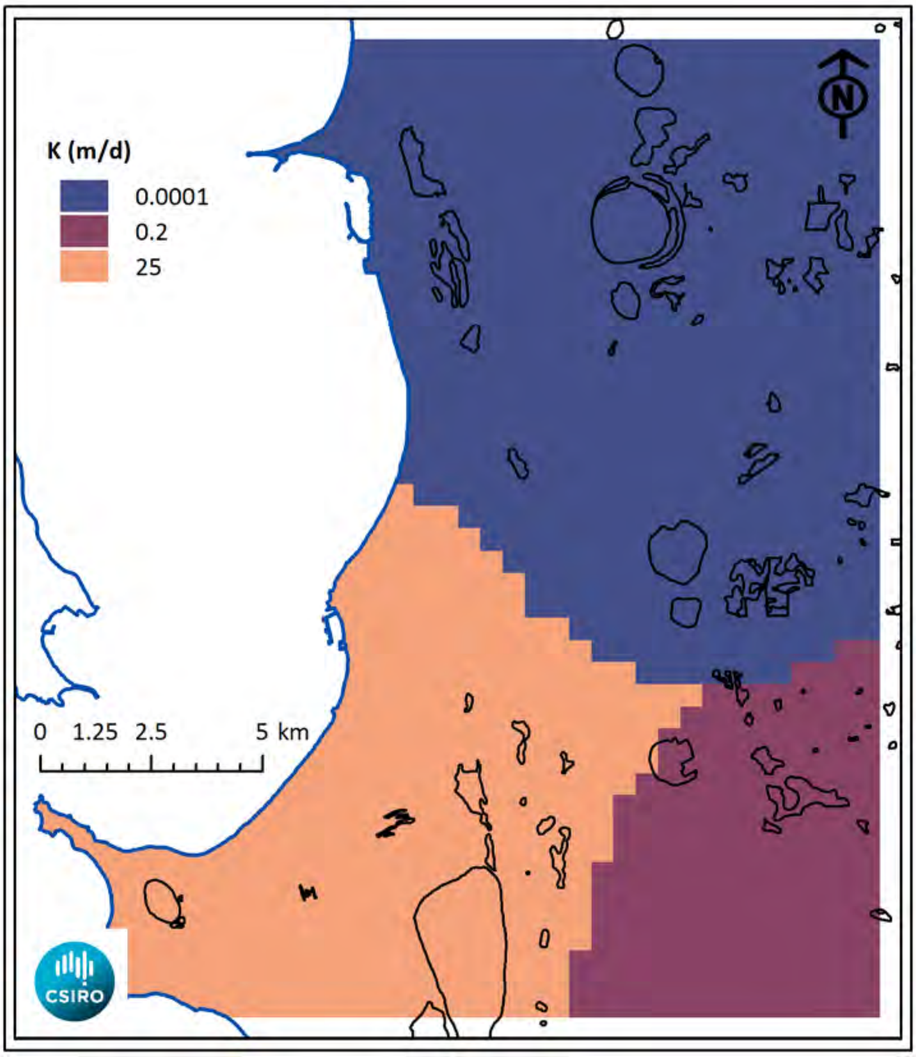


Figure 29 Zones of hydraulic conductivity for Layer 5

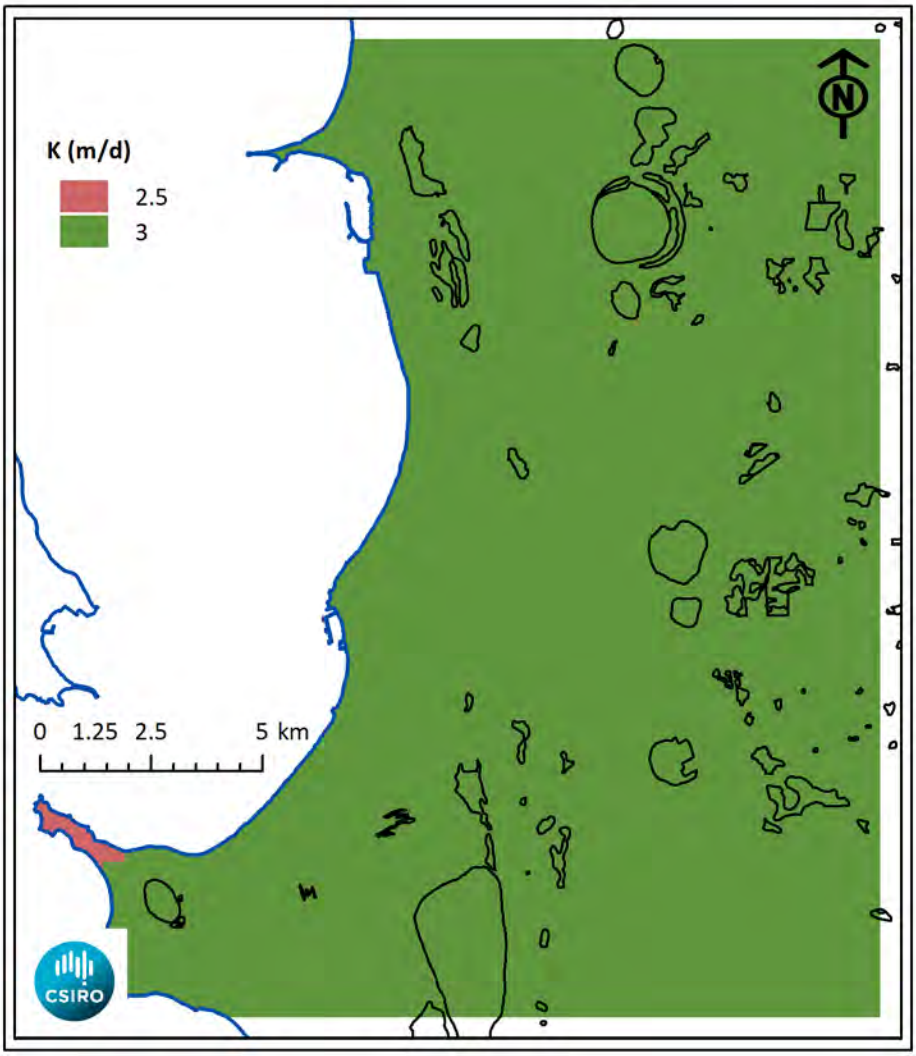


Figure 30 Zones of hydraulic conductivity for Layer 6

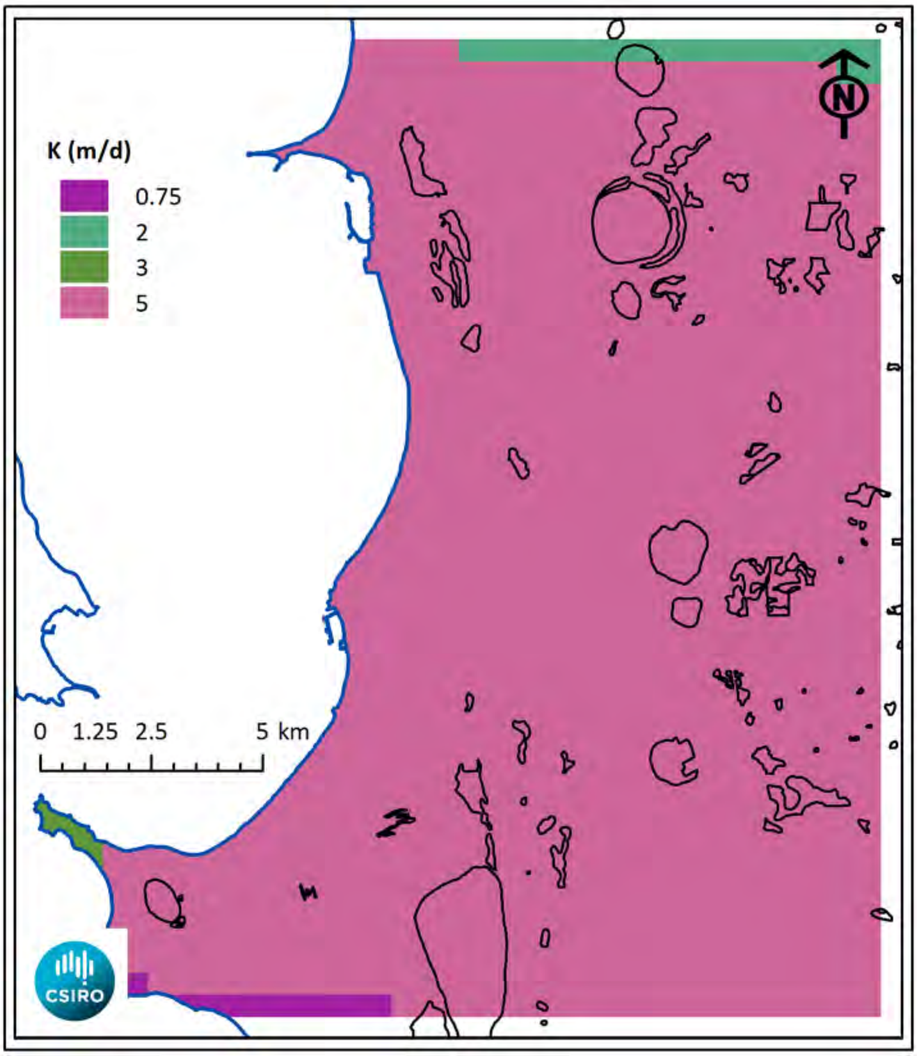


Figure 31 Zones of hydraulic conductivity for Layer 7

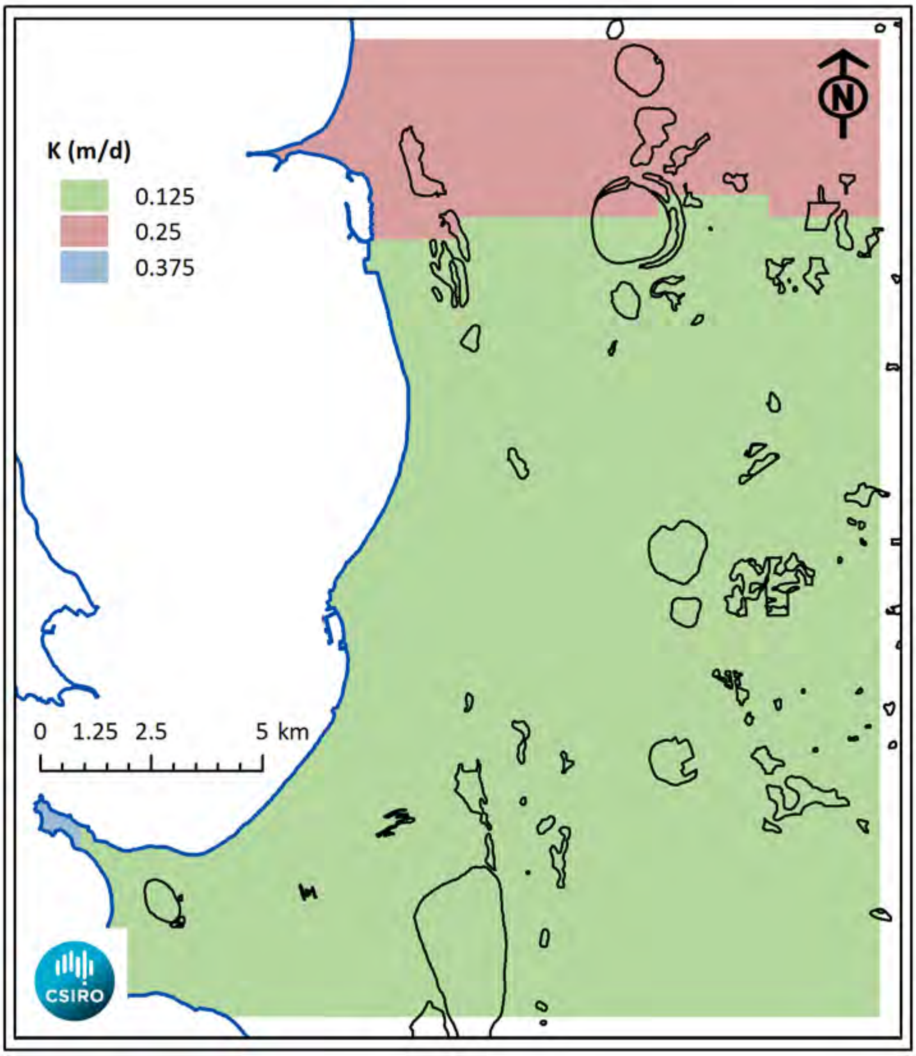


Figure 32 Zones of hydraulic conductivity for Layer 8

2. Layer thickness maps

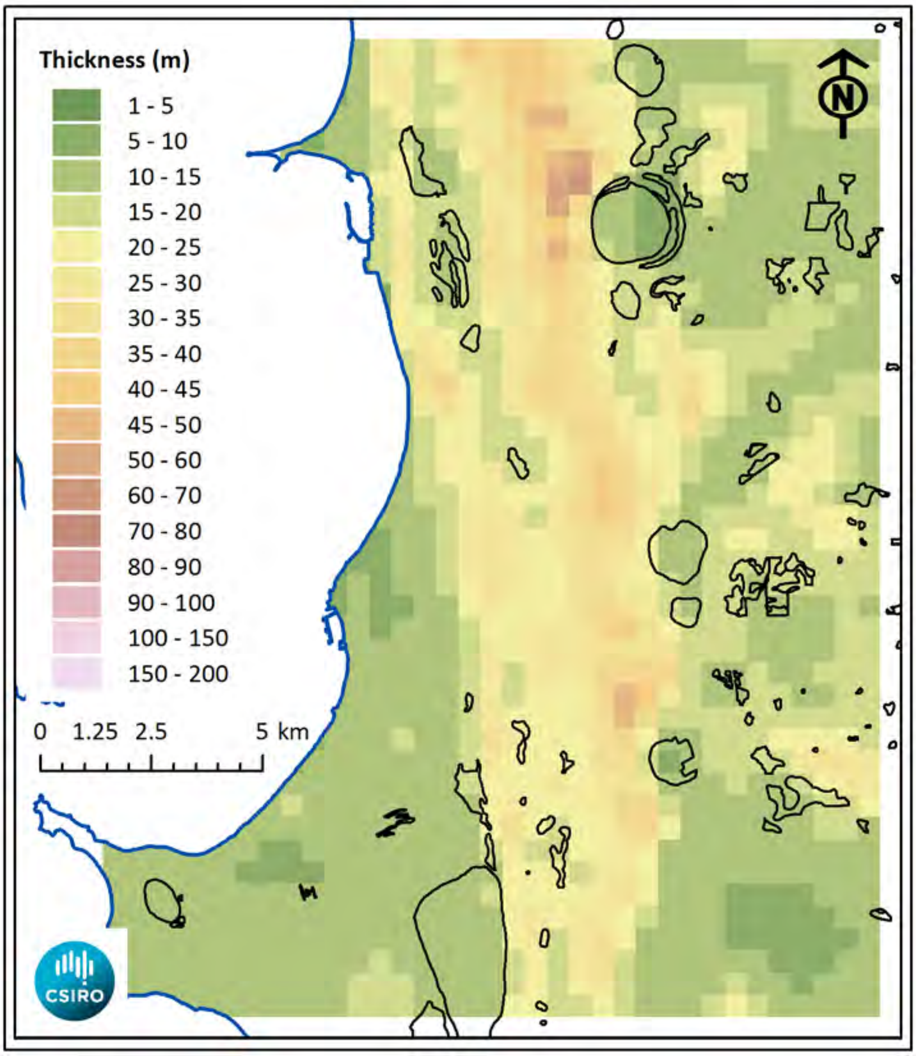


Figure 33 Thickness of Layer 1.

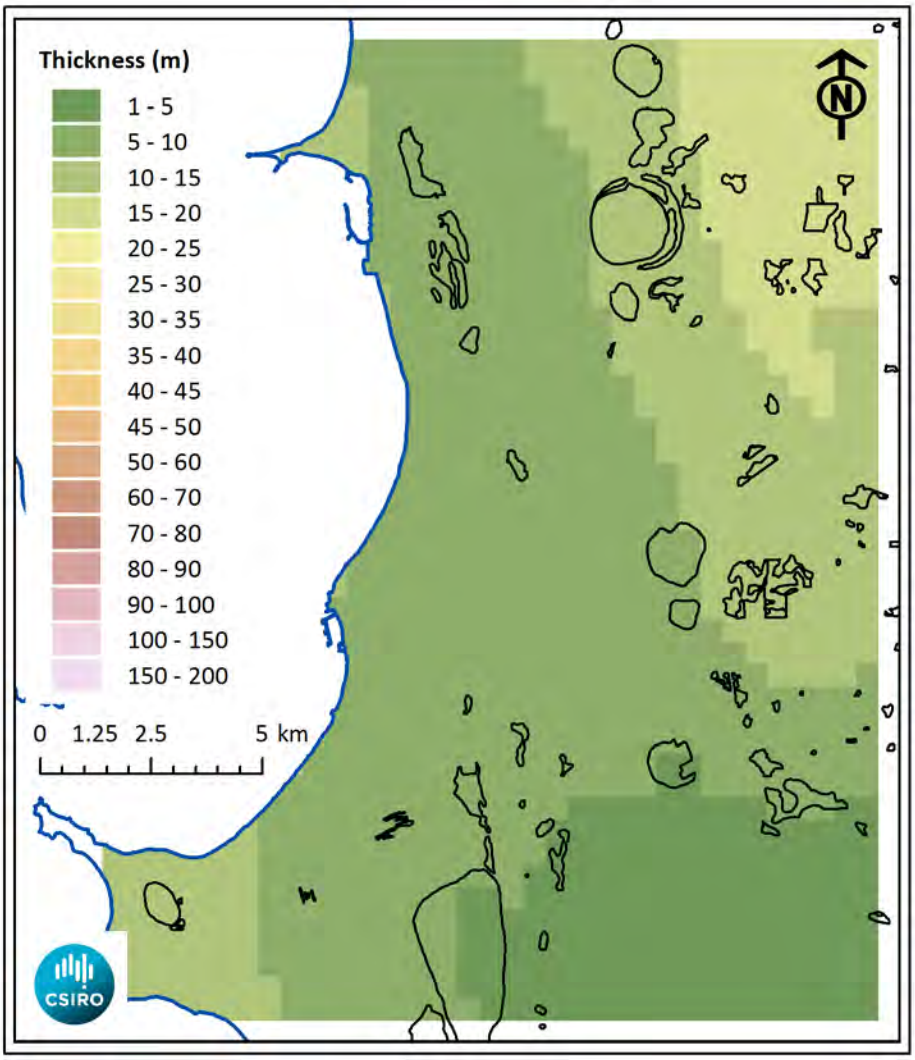


Figure 34 Thickness of Layer 2.

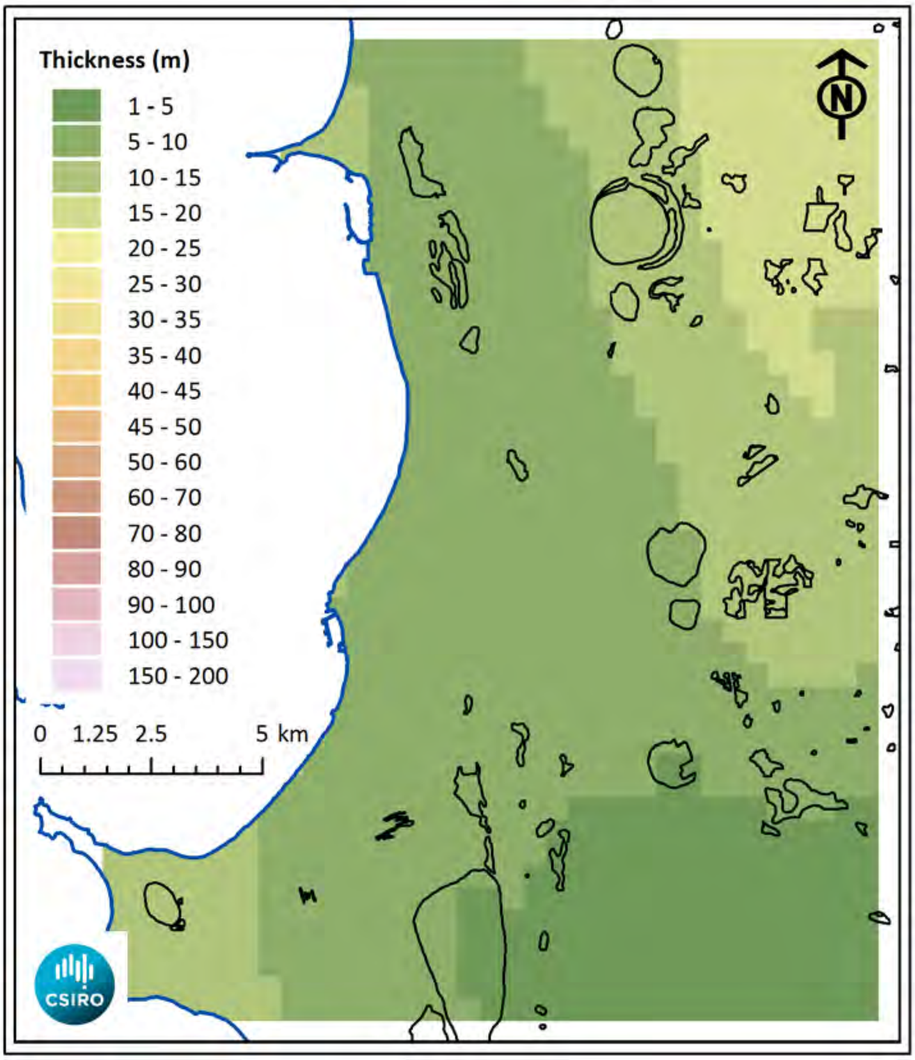


Figure 35 Thickness of Layer 3.

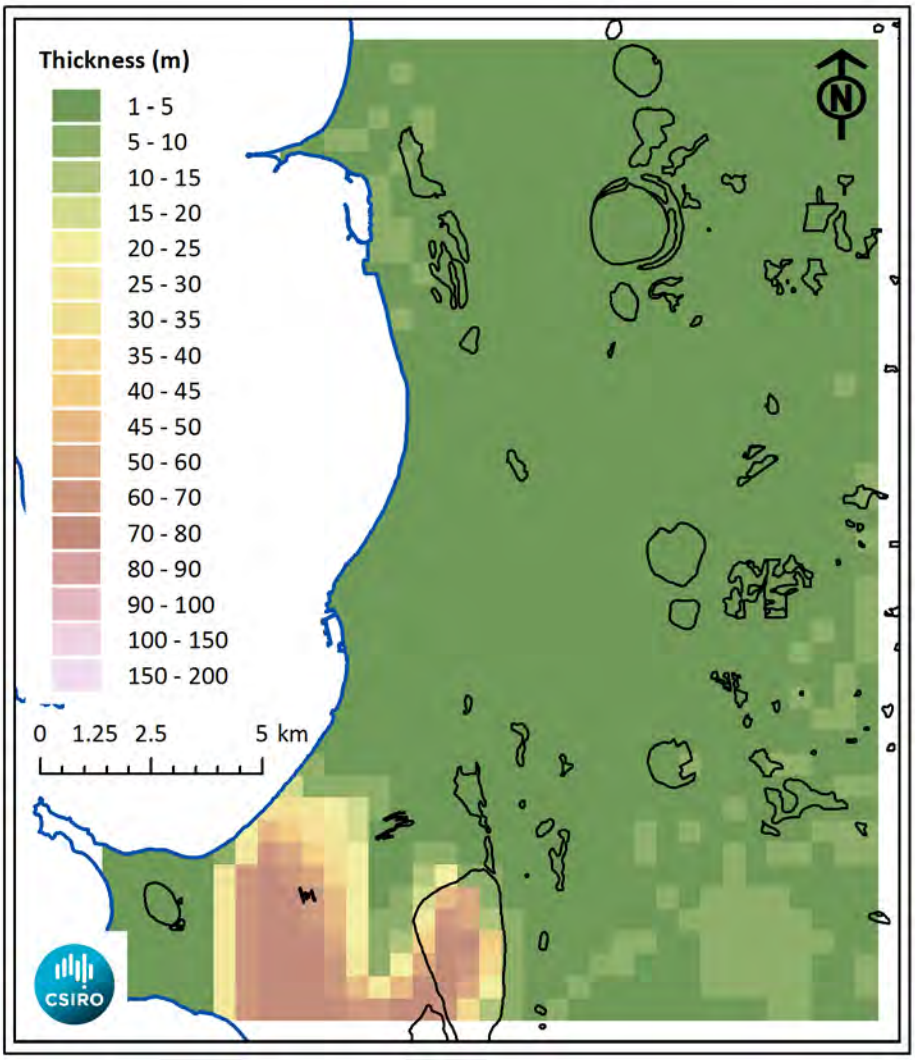


Figure 36 Thickness of Layer 4.

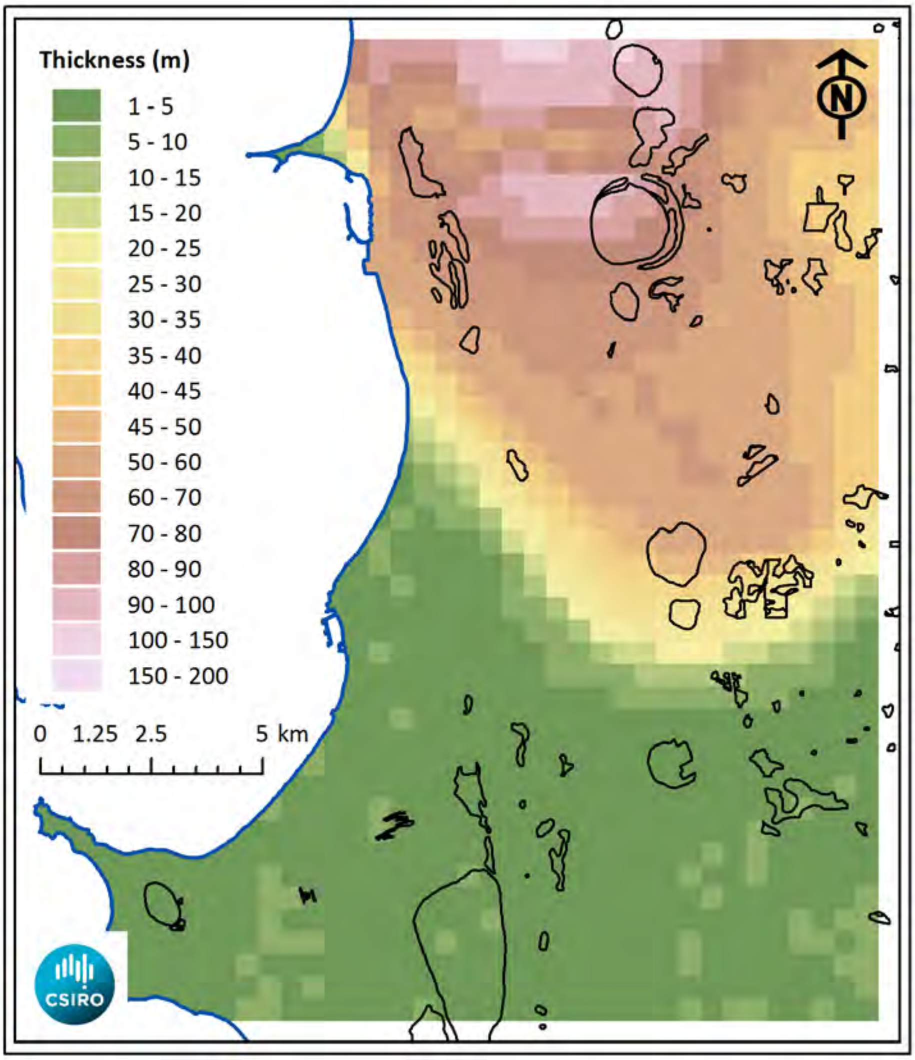


Figure 37 Thickness of Layer 5.

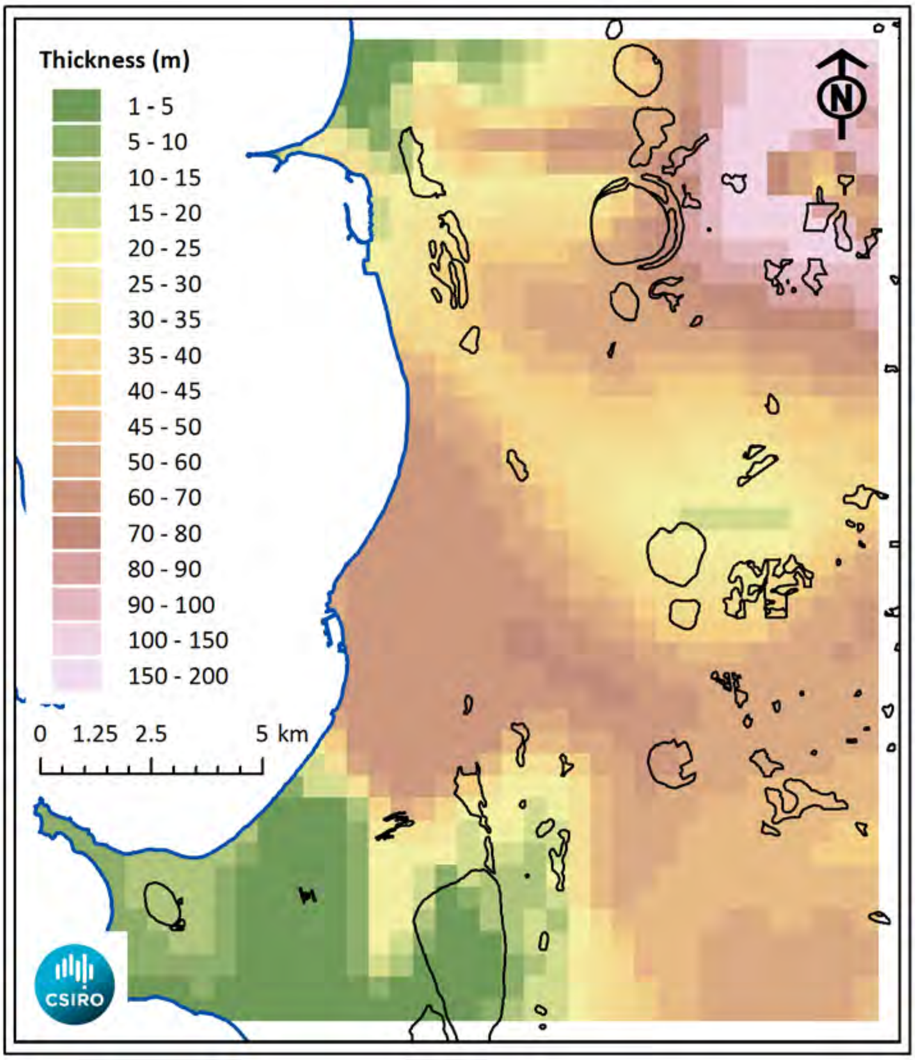


Figure 38 Thickness of Layer 6.

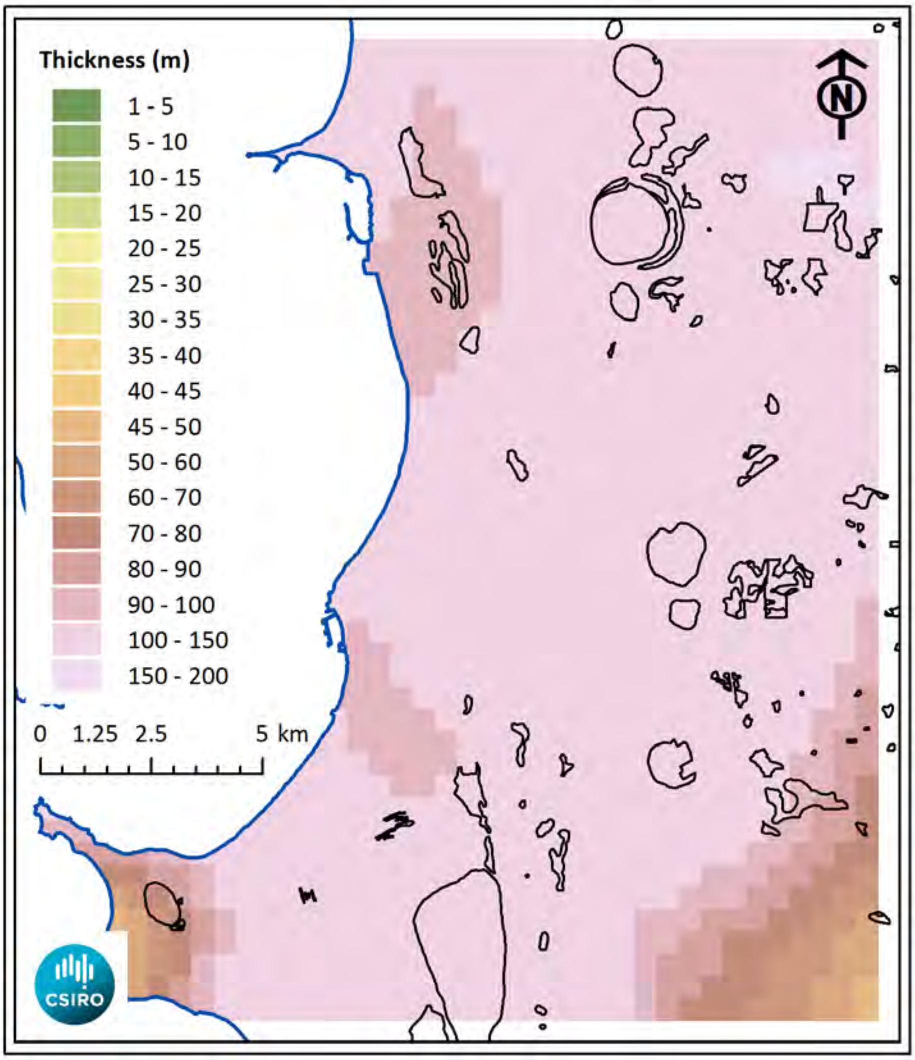


Figure 39 Thickness of Layer 7.

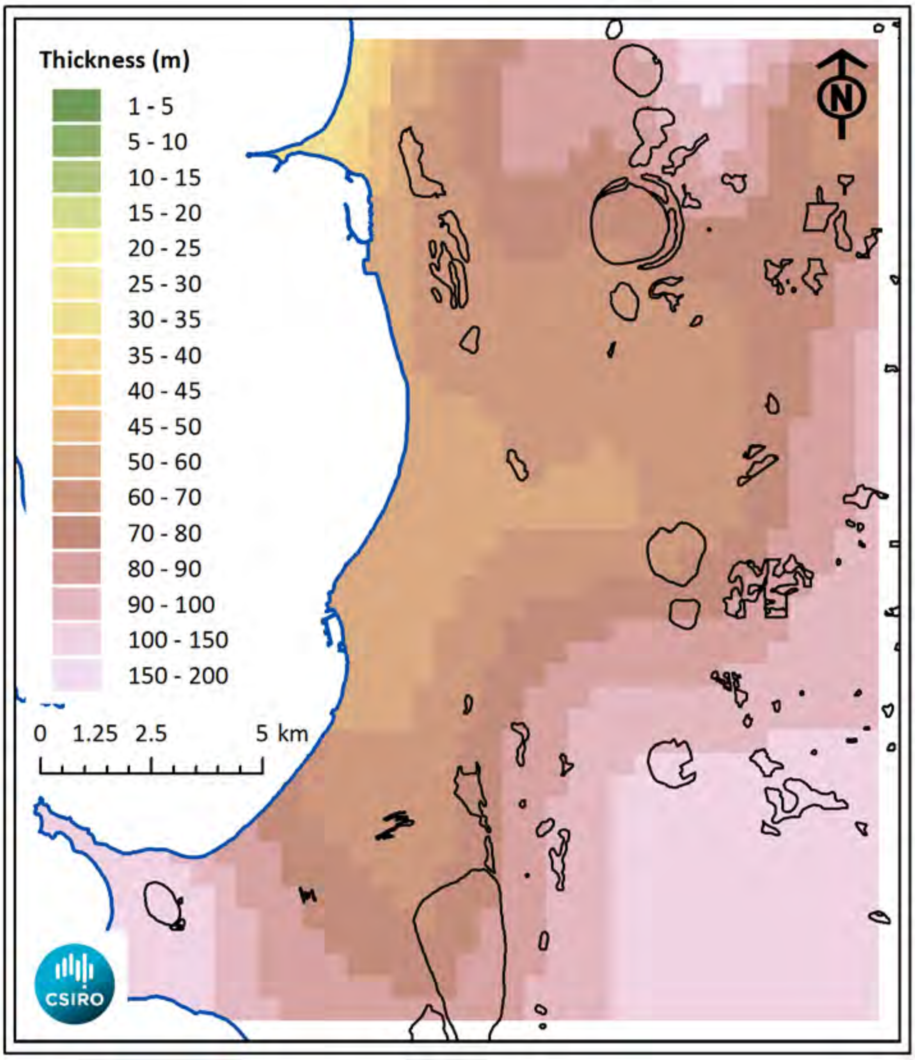
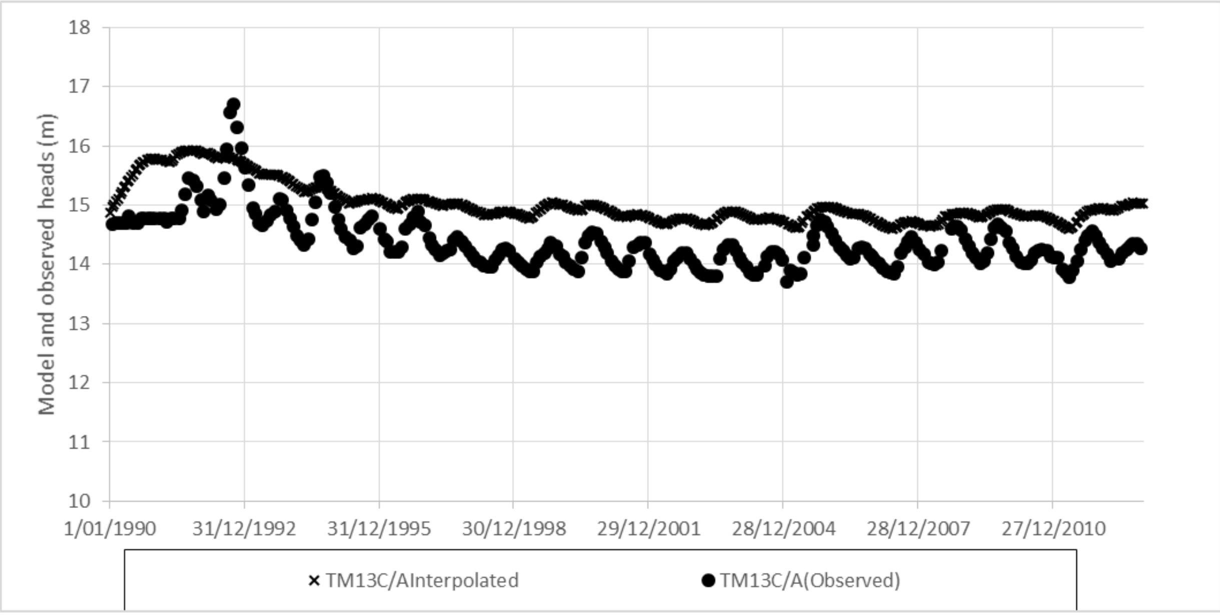


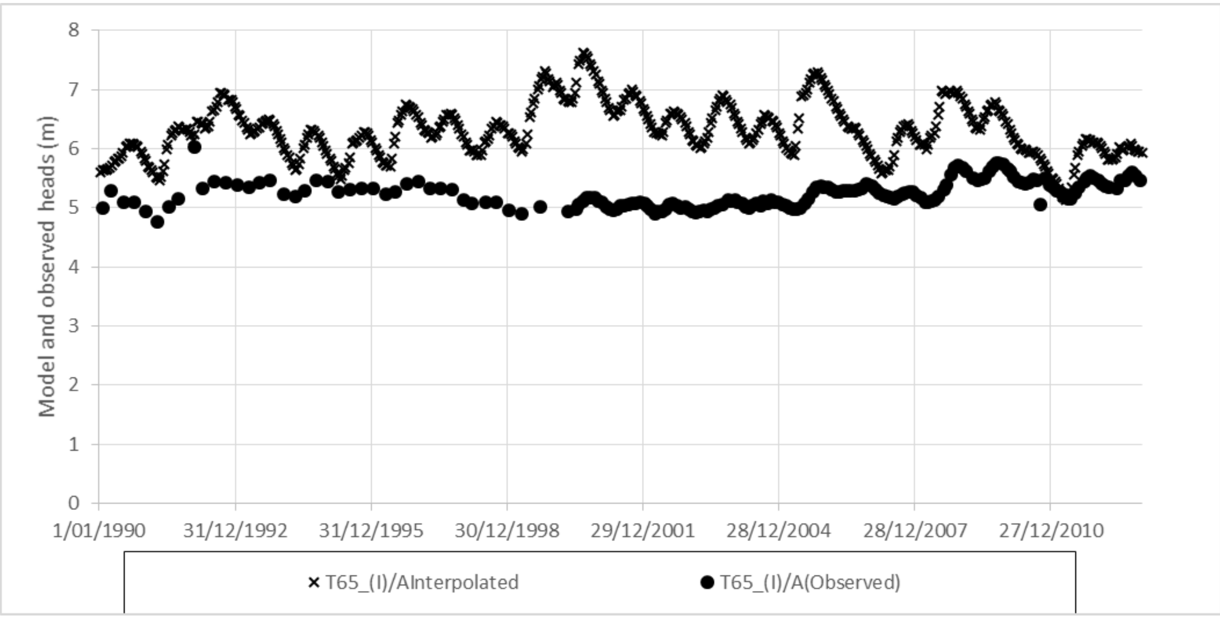
Figure 40 Thickness of Layer 8.

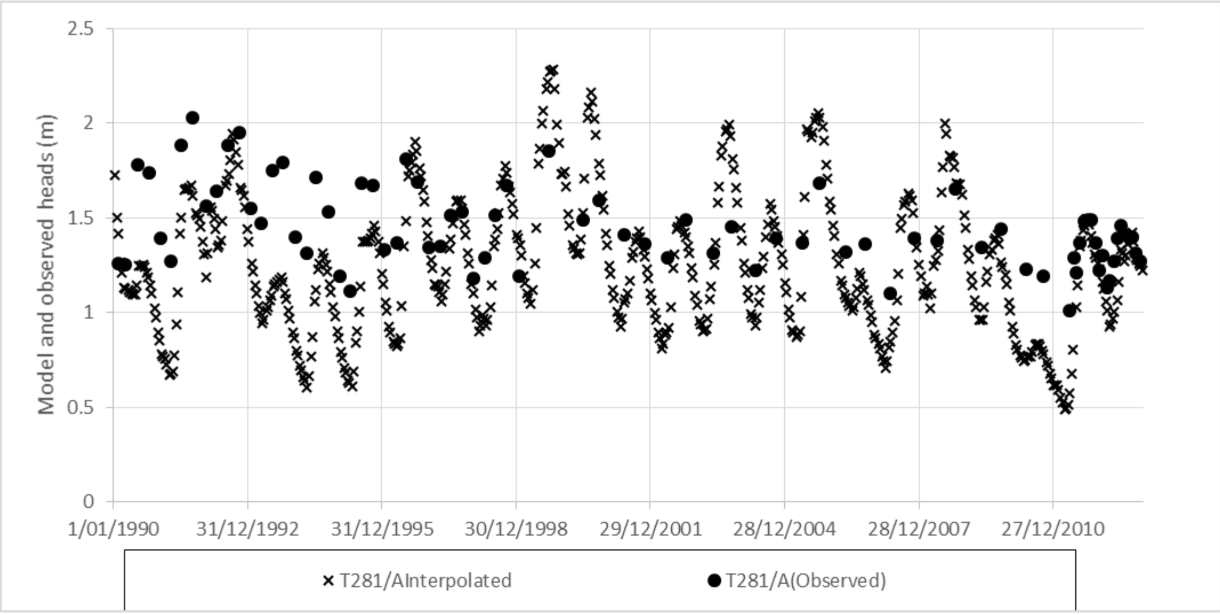
3. Calibration well hydrographs

OBSERVATION WELL DATA FROM THE DEPARTMENT OF WATER

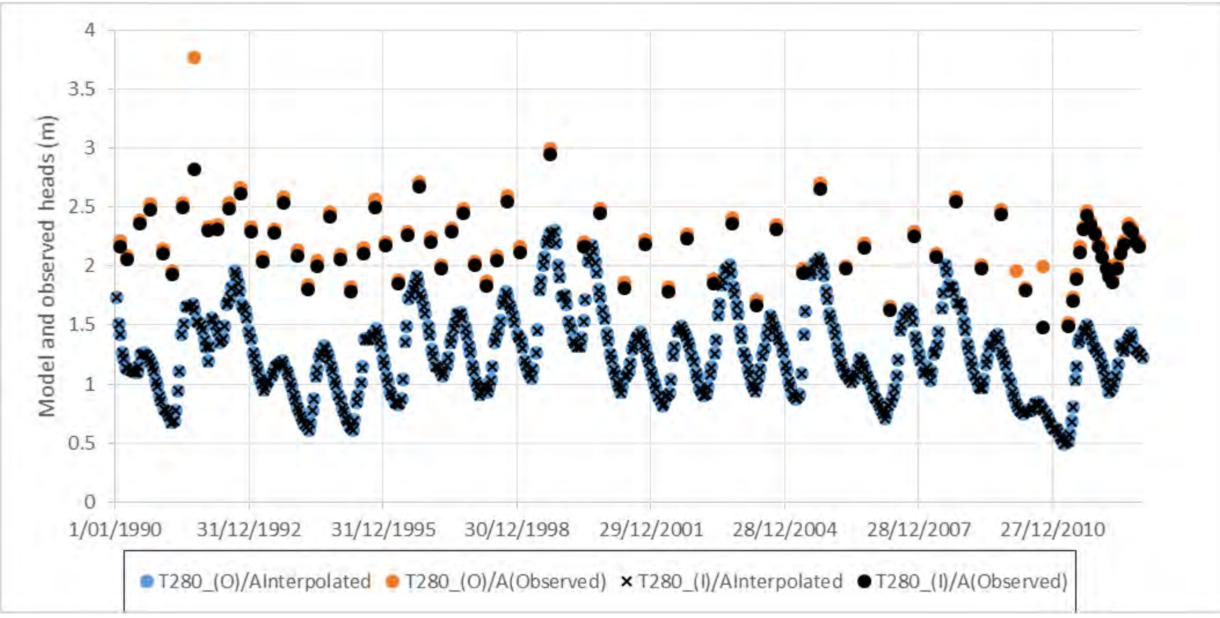
In all of the plots shown below, the model results are labelled as 'Interpolated' data.



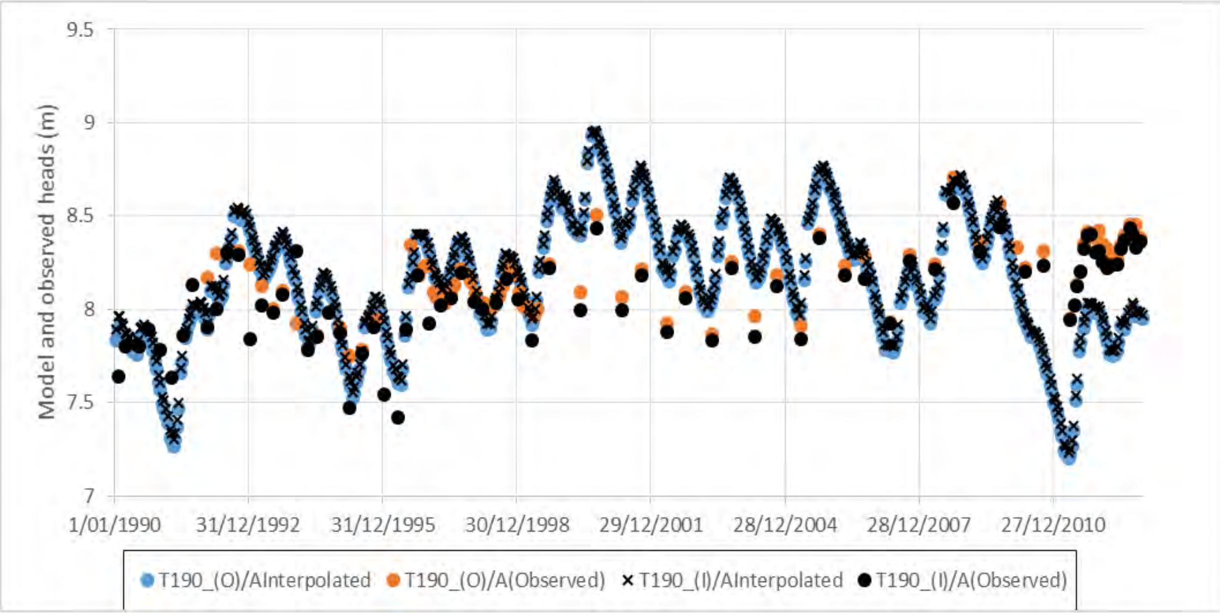




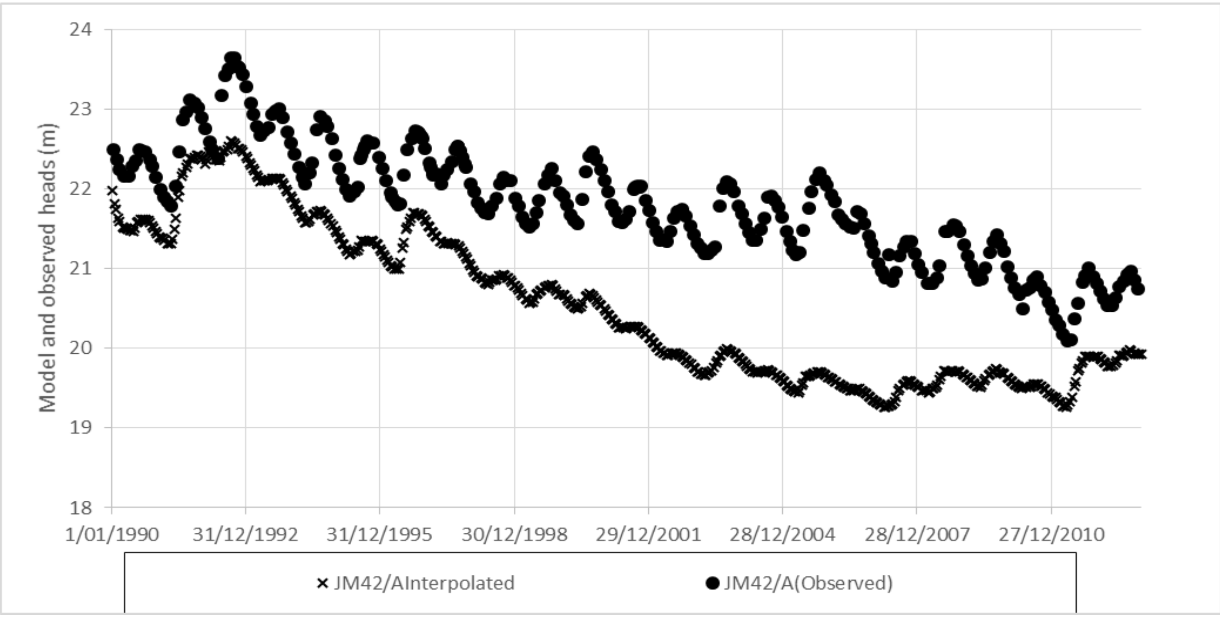
Model and observed heads (m)
2.5
2
1.5
1
0.5
0
1/01/1990
31/12/1992
31/12/1995
30/12/1998
29/12/2001
28/12/2004
28/12/2007
27/12/2010
T281/AInterpolated
T281/A(Observed)



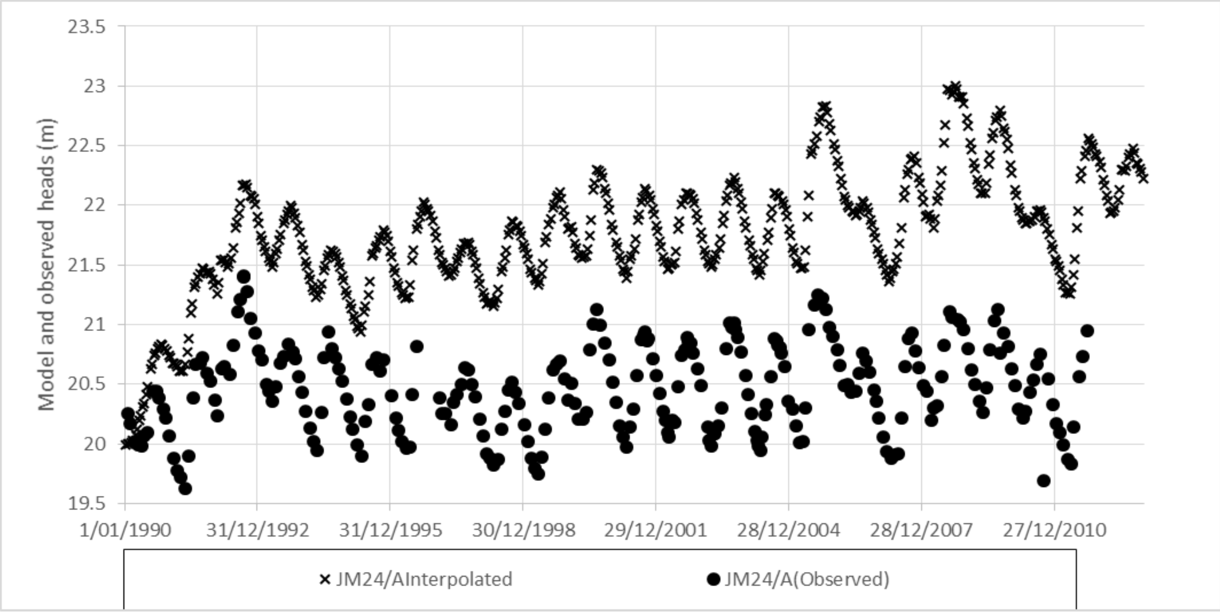
Model and observed heads (m)
4
3.5
3
2.5
2
1.5
1
0.5
0
1/01/1990
31/12/1992
31/12/1995
30/12/1998
29/12/2001
28/12/2004
28/12/2007
27/12/2010
T280_(O)/AInterpolated
T280_(O)/A(Observed)
T280_(I)/AInterpolated
T280_(I)/A(Observed)

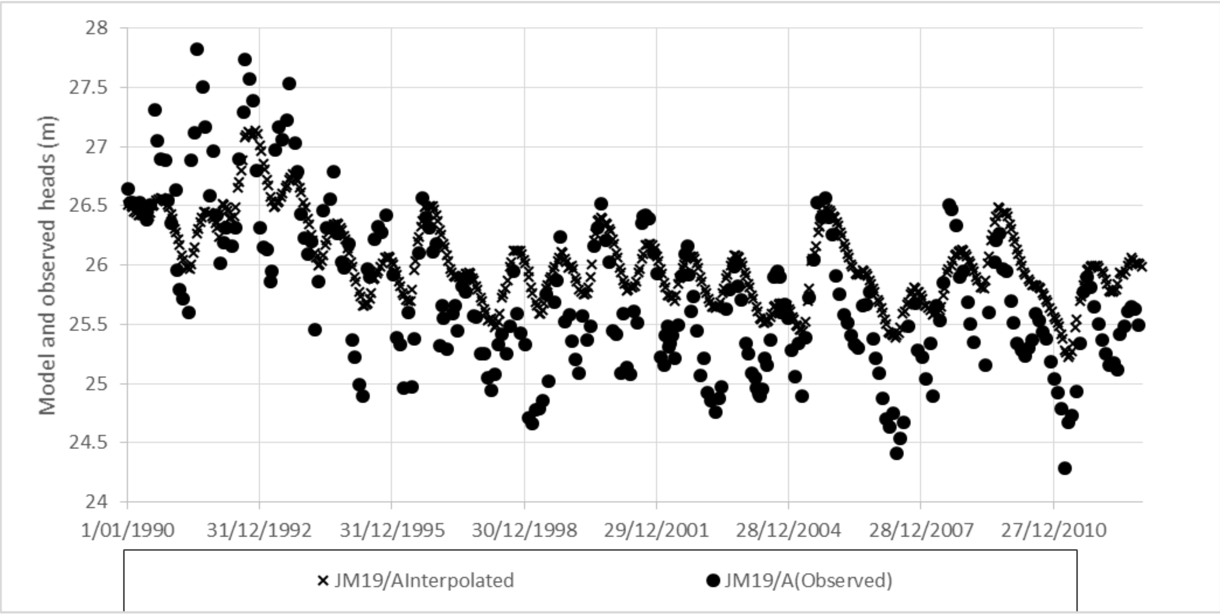


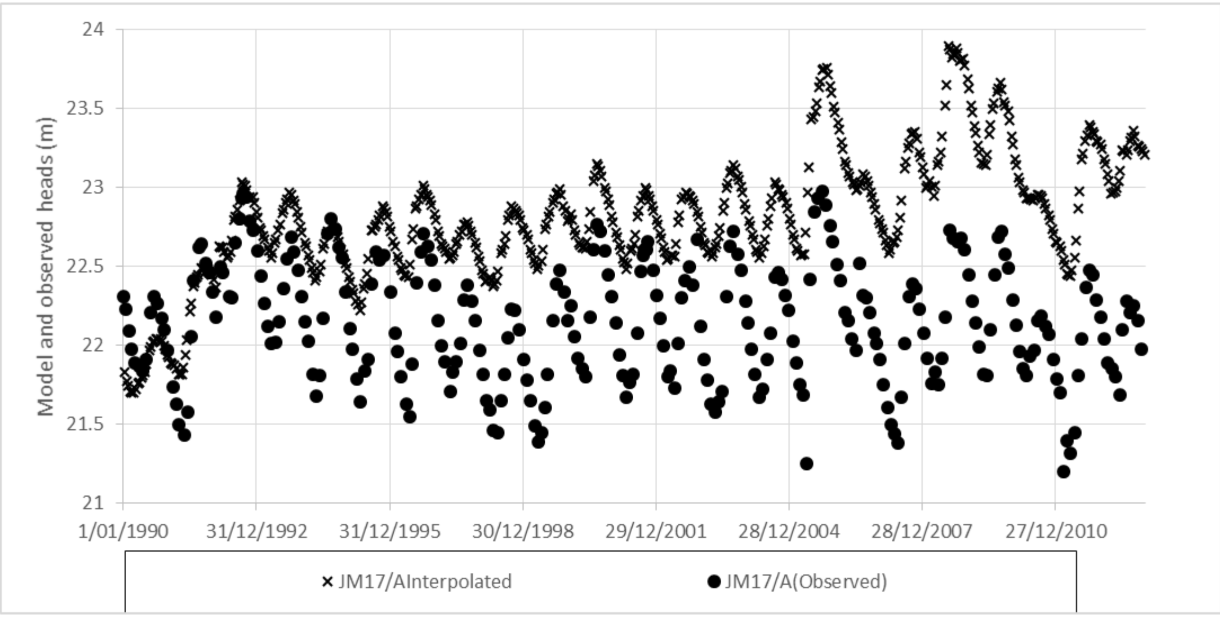
Model and observed heads (m)
9.5
9
8.5
8
7.5
7
1/01/1990
31/12/1992
31/12/1995
30/12/1998
29/12/2001
28/12/2004
28/12/2007
27/12/2010
T190_(O)/AInterpolated
T190_(O)/A(Observed)
T190_(I)/AInterpolated
T190_(I)/A(Observed)

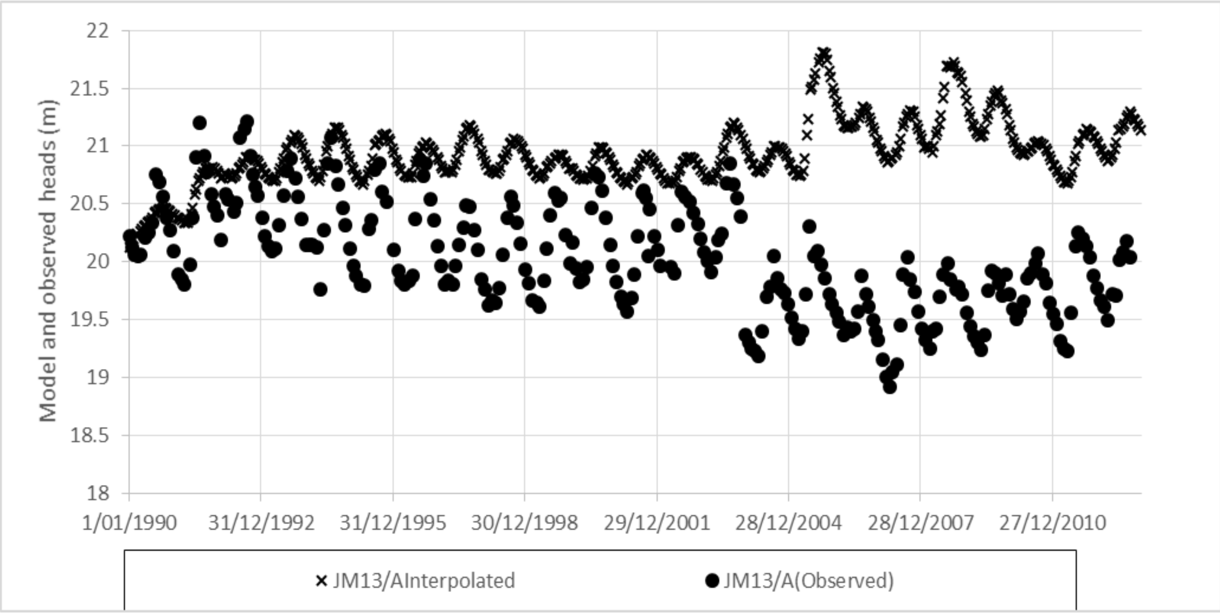


Model and observed heads (m)
24
23
22
21
20
19
18
1/01/1990
31/12/1992
31/12/1995
30/12/1998
29/12/2001
28/12/2004
28/12/2007
27/12/2010
JM42/AInterpolated
JM42/A(Observed)

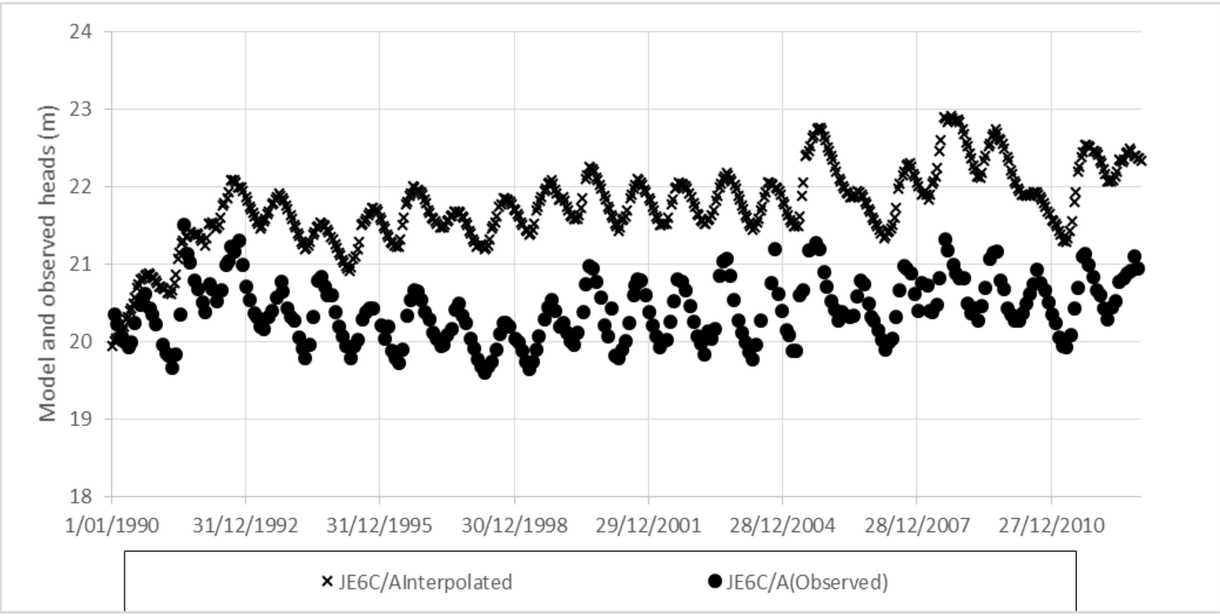

23.5
23
22.5
22
21.5
21
20.5
20
19.5
Model and observed heads (m)
1/01/1990
31/12/1992
31/12/1995
30/12/1998
29/12/2001
28/12/2004
28/12/2007
27/12/2010
JM24/AInterpolated
JM24/A(Observed)


28
27.5
27
26.5
26
25.5
25
24.5
24
Model and observed heads (m)
1/01/1990
31/12/1992
31/12/1995
30/12/1998
29/12/2001
28/12/2004
28/12/2007
27/12/2010
JM19/AInterpolated
JM19/A(Observed)

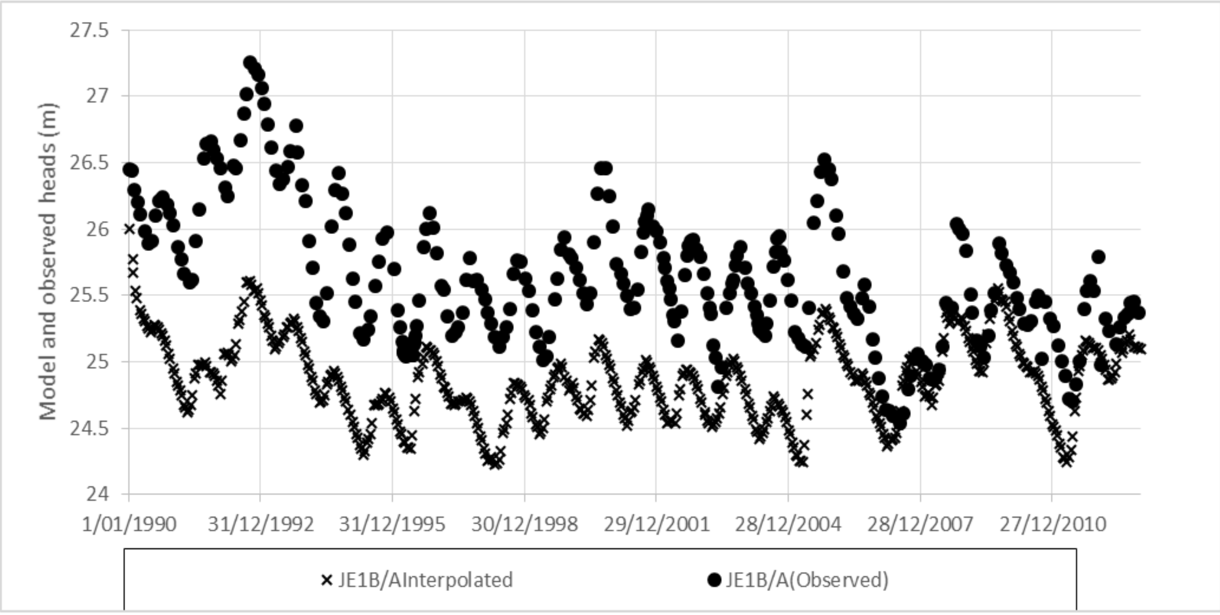

24
23.5
23
22.5
22
21.5
21
Model and observed heads (m)
1/01/1990
31/12/1992
31/12/1995
30/12/1998
29/12/2001
28/12/2004
28/12/2007
27/12/2010
JM17/AInterpolated
JM17/A(Observed)



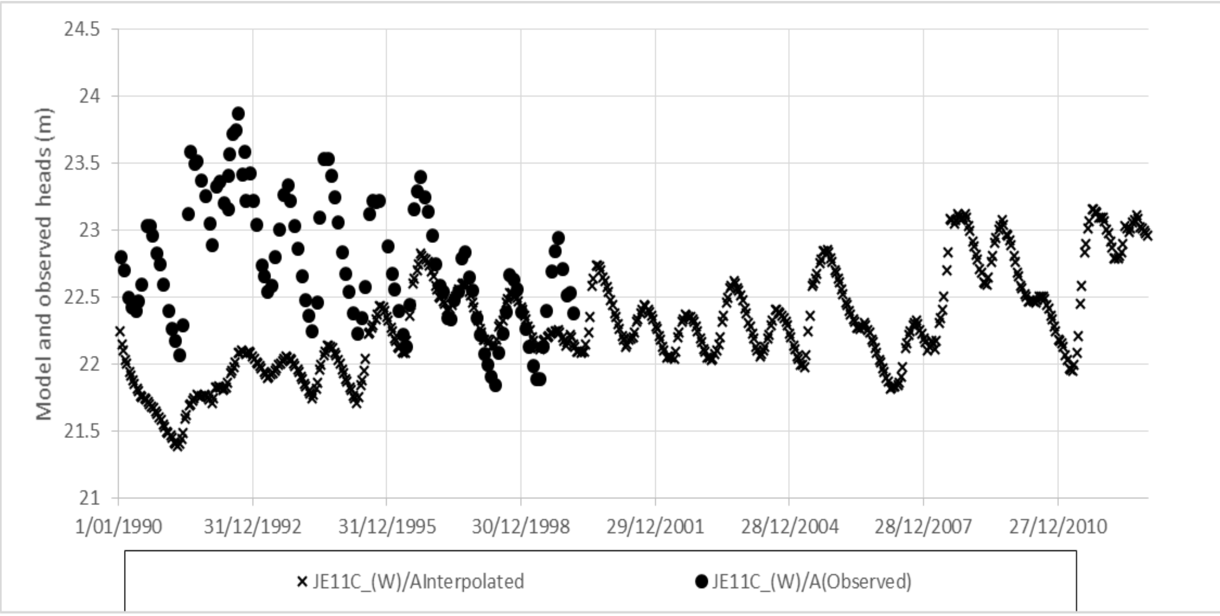
Model and observed heads (m)
22
21.5
21
20.5
20
19.5
19
18.5
18
1/01/1990
31/12/1992
31/12/1995
30/12/1998
29/12/2001
28/12/2004
28/12/2007
27/12/2010
JM13/AInterpolated
JM13/A(Observed)



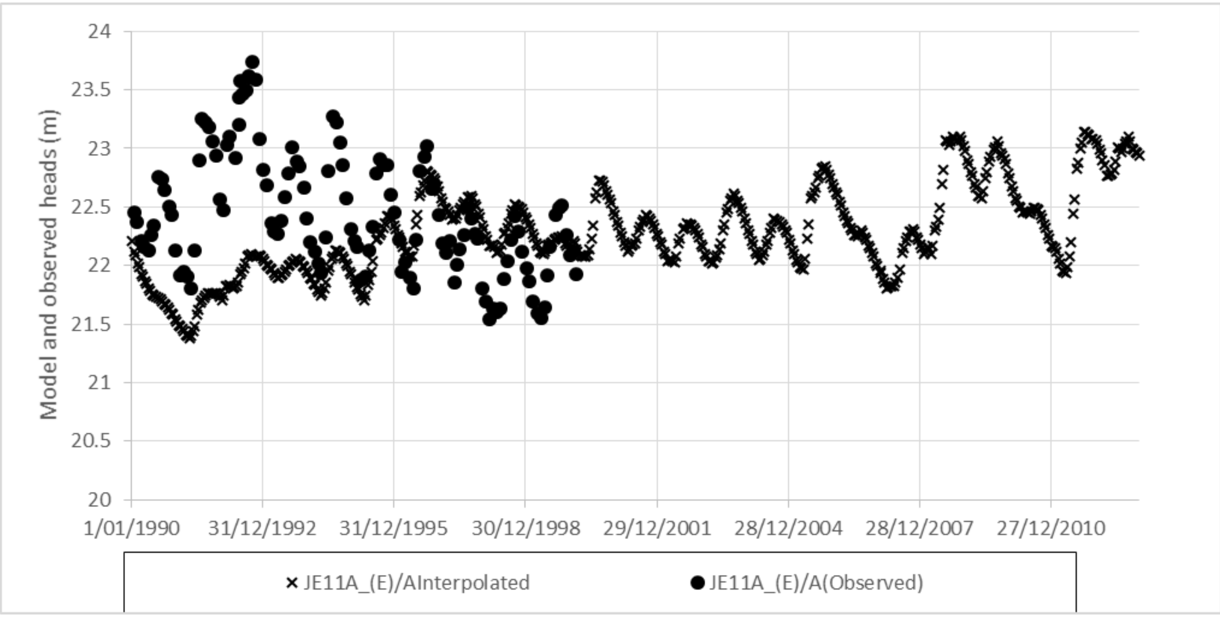
Model and observed heads (m)
24
23
22
21
20
19
18
1/01/1990
31/12/1992
31/12/1995
30/12/1998
29/12/2001
28/12/2004
28/12/2007
27/12/2010
JE6C/AInterpolated
JE6C/A(Observed)



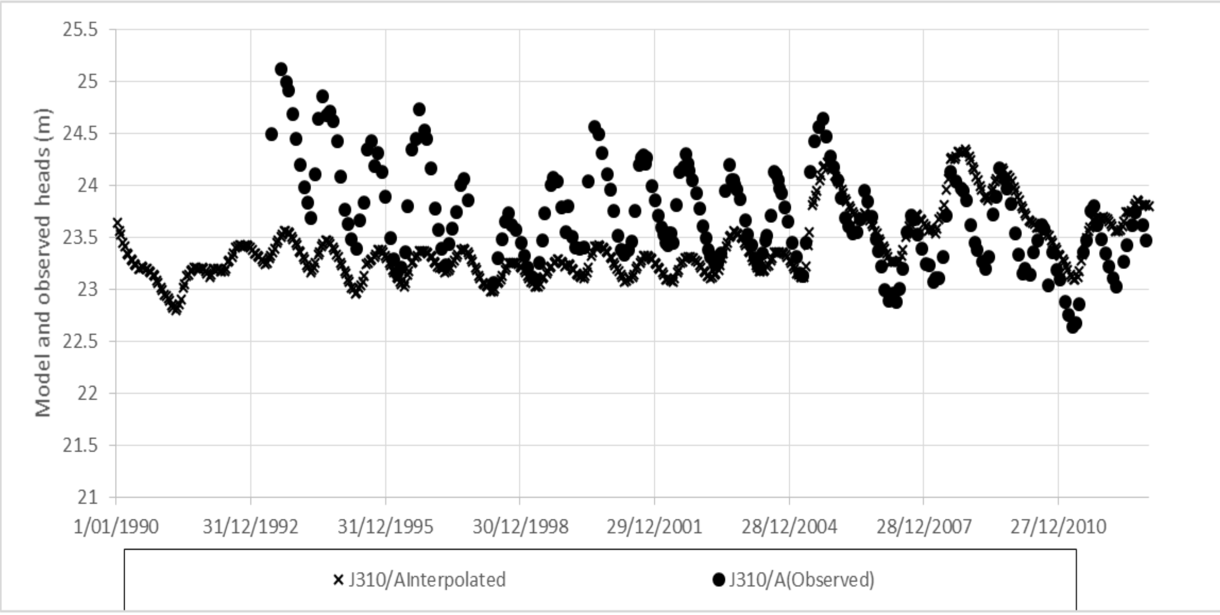
Model and observed heads (m)
27.5
27
26.5
26
25.5
25
24.5
24
1/01/1990
31/12/1992
31/12/1995
30/12/1998
29/12/2001
28/12/2004
28/12/2007
27/12/2010
JE1B/AInterpolated
JE1B/A(Observed)



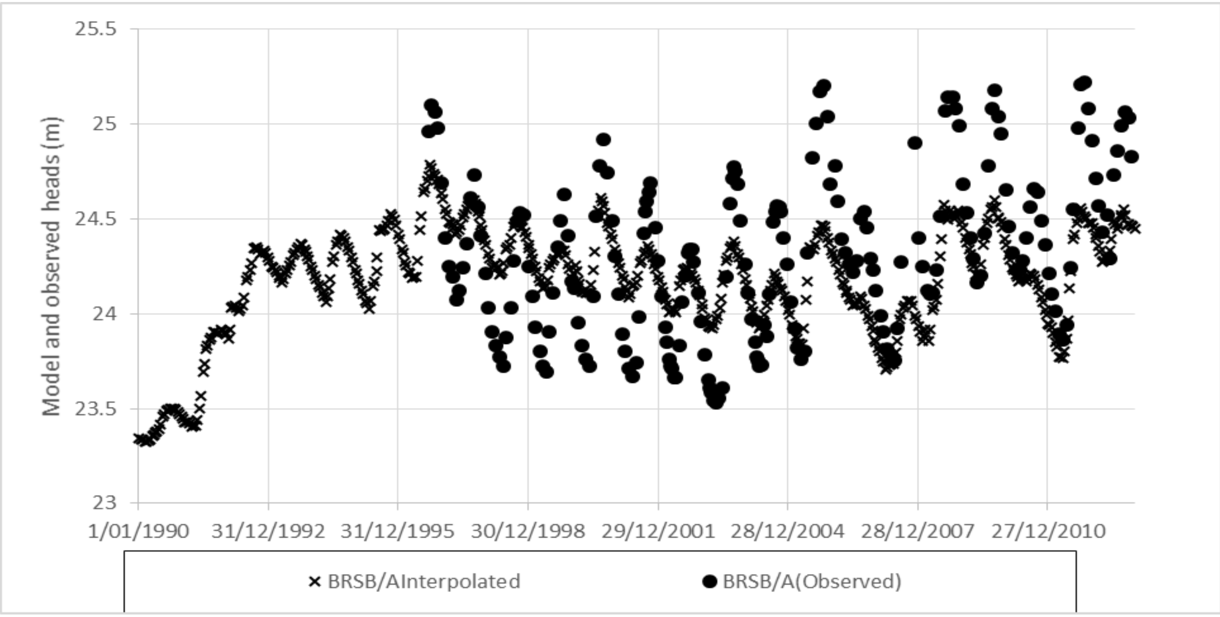
Model and observed heads (m)
24.5
24
23.5
23
22.5
22
21.5
21
1/01/1990
31/12/1992
31/12/1995
30/12/1998
29/12/2001
28/12/2004
28/12/2007
27/12/2010
JE11C_(W)/AInterpolated
JE11C_(W)/A(Observed)



Model and observed heads (m)
24
23.5
23
22.5
22
21.5
21
20.5
20
1/01/1990
31/12/1992
31/12/1995
30/12/1998
29/12/2001
28/12/2004
28/12/2007
27/12/2010
JE11A_(E)/AInterpolated
JE11A_(E)/A(Observed)



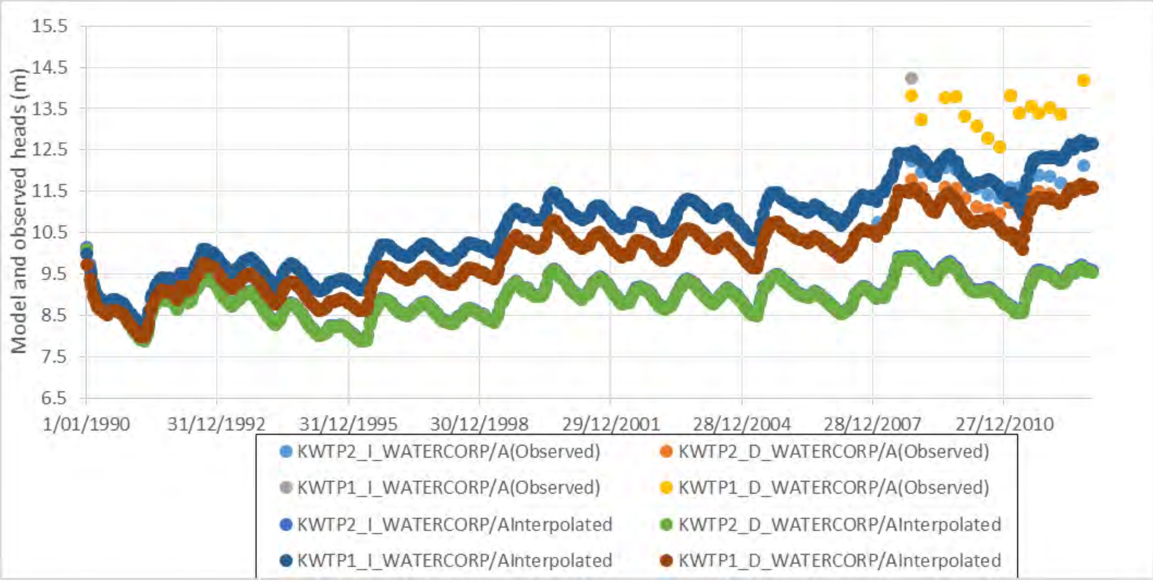
Model and observed heads (m)
25.5
25
24.5
24
23.5
23
22.5
22
21.5
21
1/01/1990
31/12/1992
31/12/1995
30/12/1998
29/12/2001
28/12/2004
28/12/2007
27/12/2010
J310/AInterpolated
J310/A(Observed)

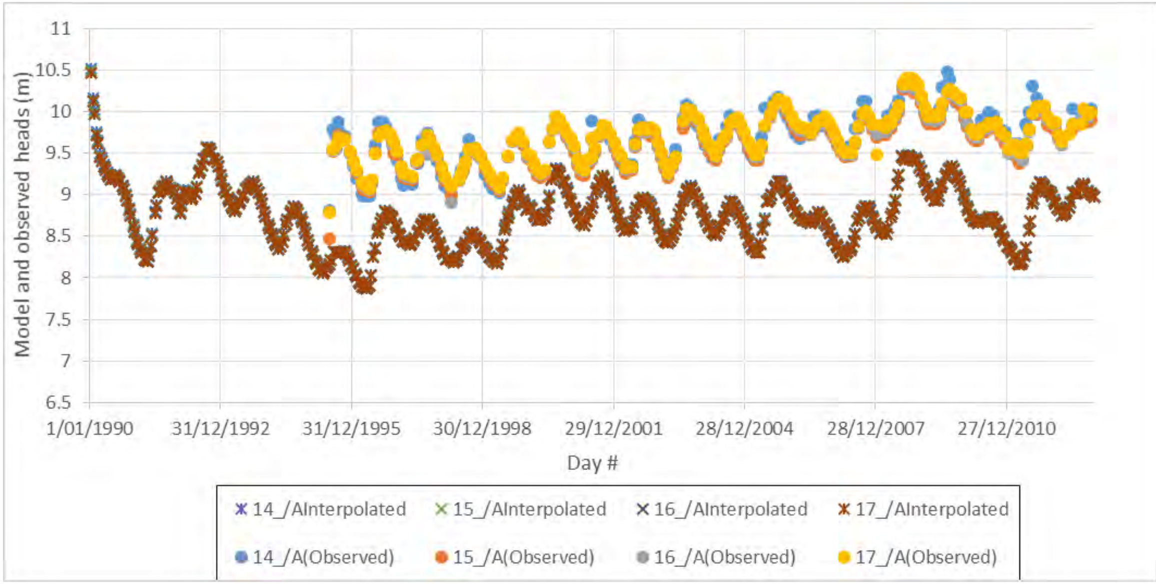


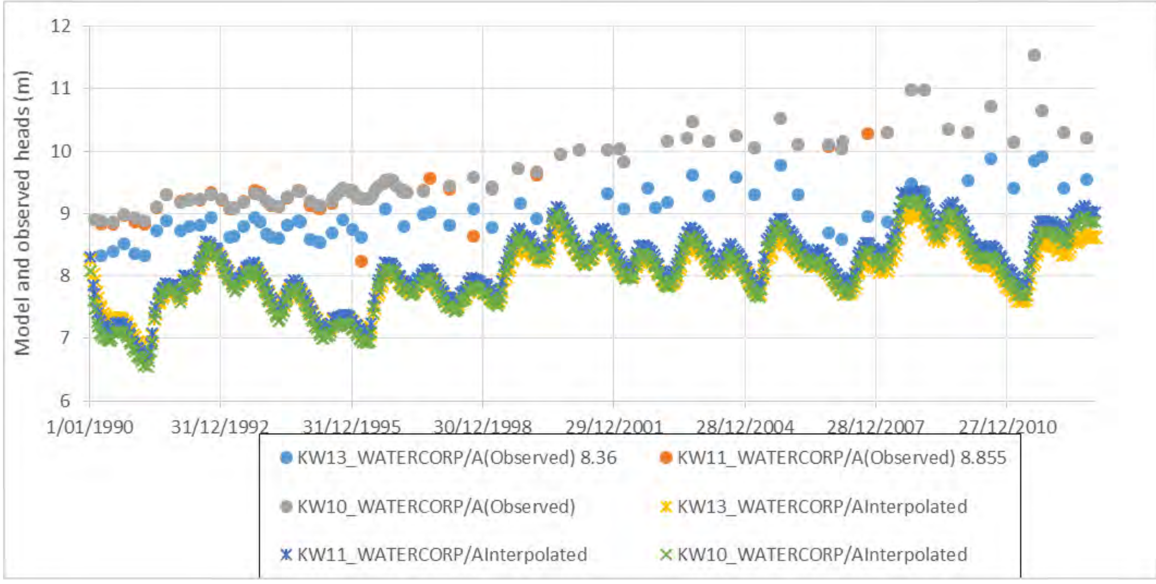
Model and observed heads (m)
25.5
25
24.5
24
23.5
23
1/01/1990
31/12/1992
31/12/1995
30/12/1998
29/12/2001
28/12/2004
28/12/2007
27/12/2010
BRSB/AInterpolated
BRSB/A(Observed)

OBSERVATION WELL DATA FROM THE WATER CORPORATION

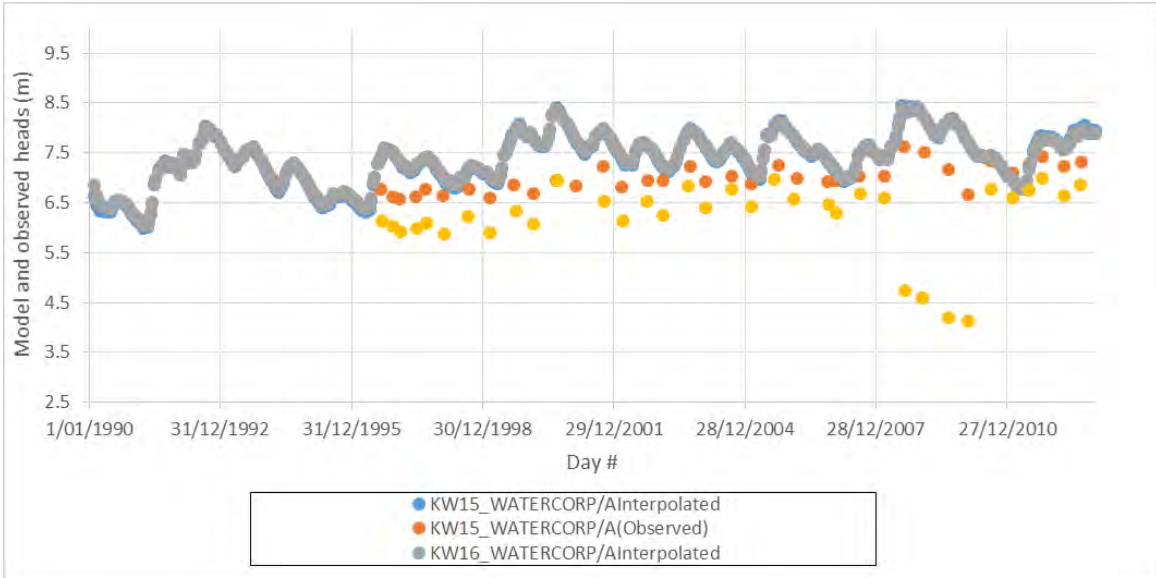
In all of the plots shown below, the model results are labelled as 'Interpolated' data







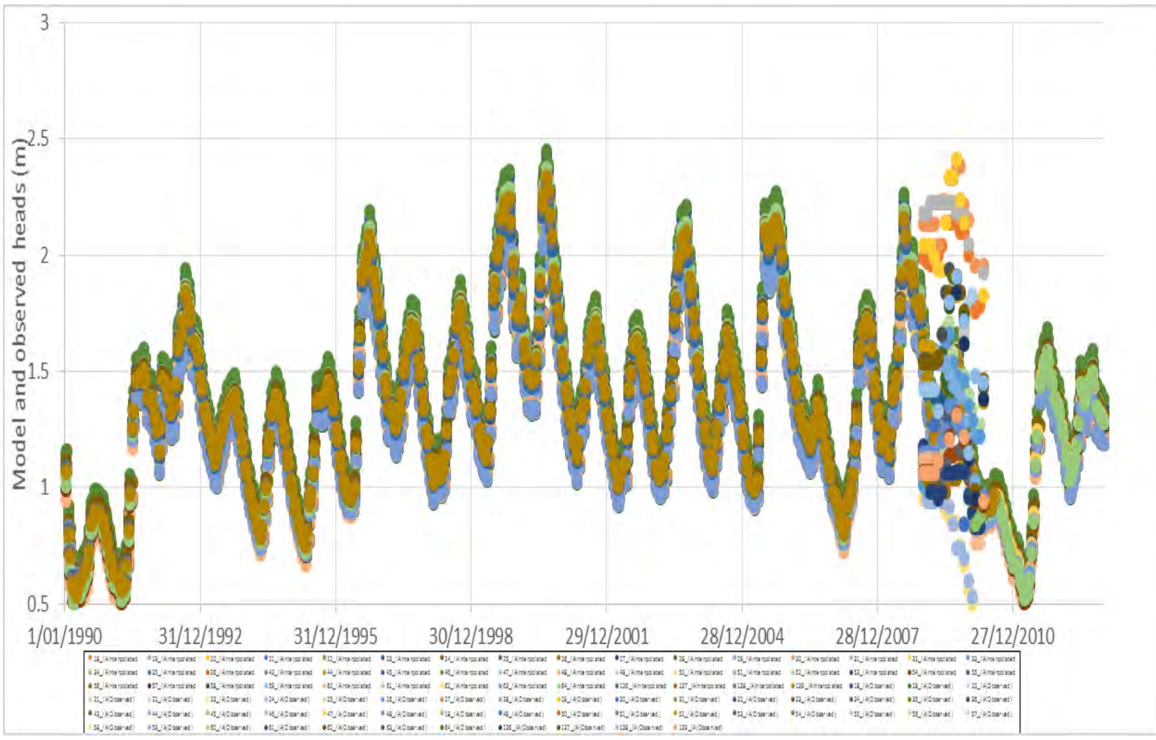
Model and observed heads (m)
12
11
10
9
8
7
6
1/01/1990
31/12/1992
31/12/1995
30/12/1998
29/12/2001
28/12/2004
28/12/2007
27/12/2010
KW13_WATERCORP/A(Observed) 8.36
KW11_WATERCORP/A(Observed) 8.855
KW10_WATERCORP/A(Observed)
KW13_WATERCORP/AInterpolated
KW11_WATERCORP/AInterpolated
KW10_WATERCORP/AInterpolated

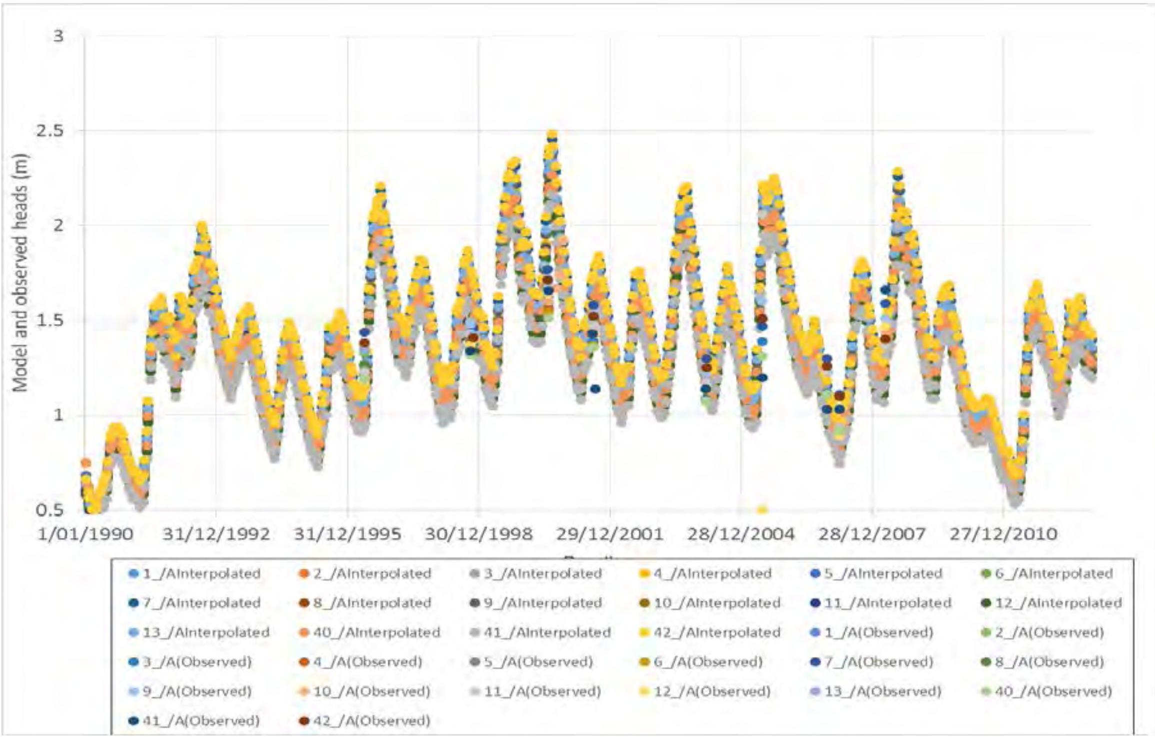


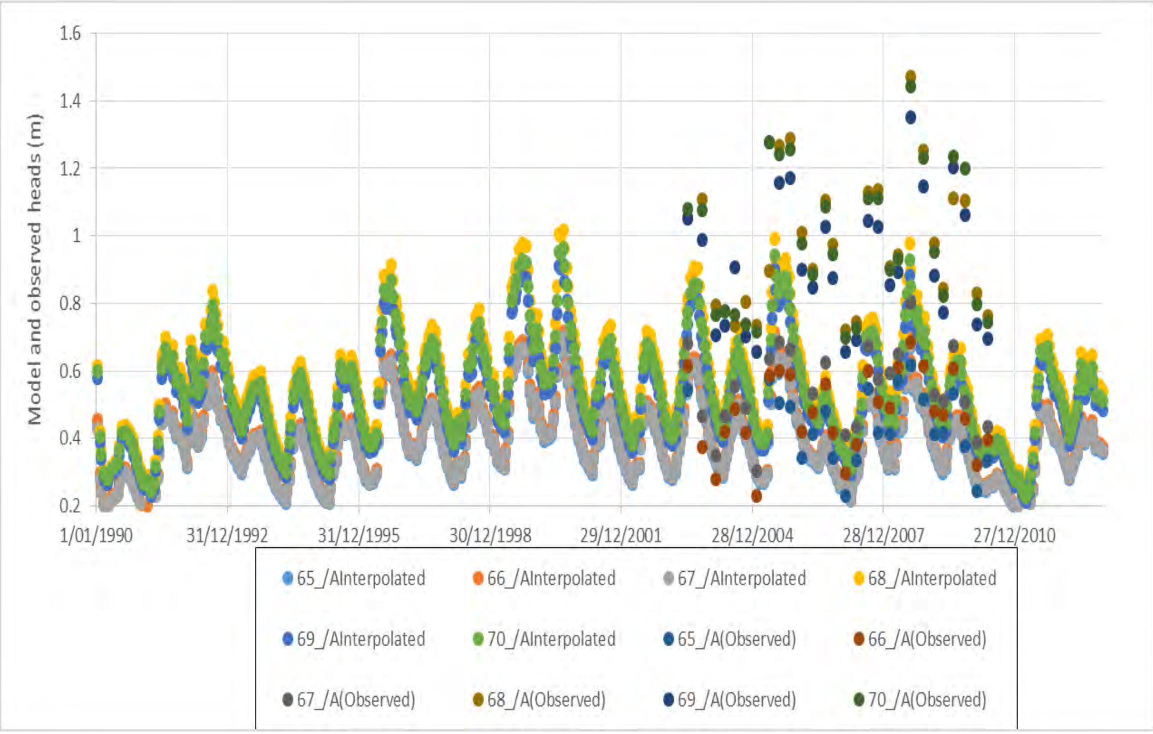
Model and observed heads (m)
9.5
8.5
7.5
6.5
5.5
4.5
3.5
2.5
1/01/1990
31/12/1992
31/12/1995
30/12/1998
29/12/2001
28/12/2004
28/12/2007
27/12/2010
Day #
KW15_WATERCORP/AInterpolated
KW15_WATERCORP/A(Observed)
KW16_WATERCORP/AInterpolated

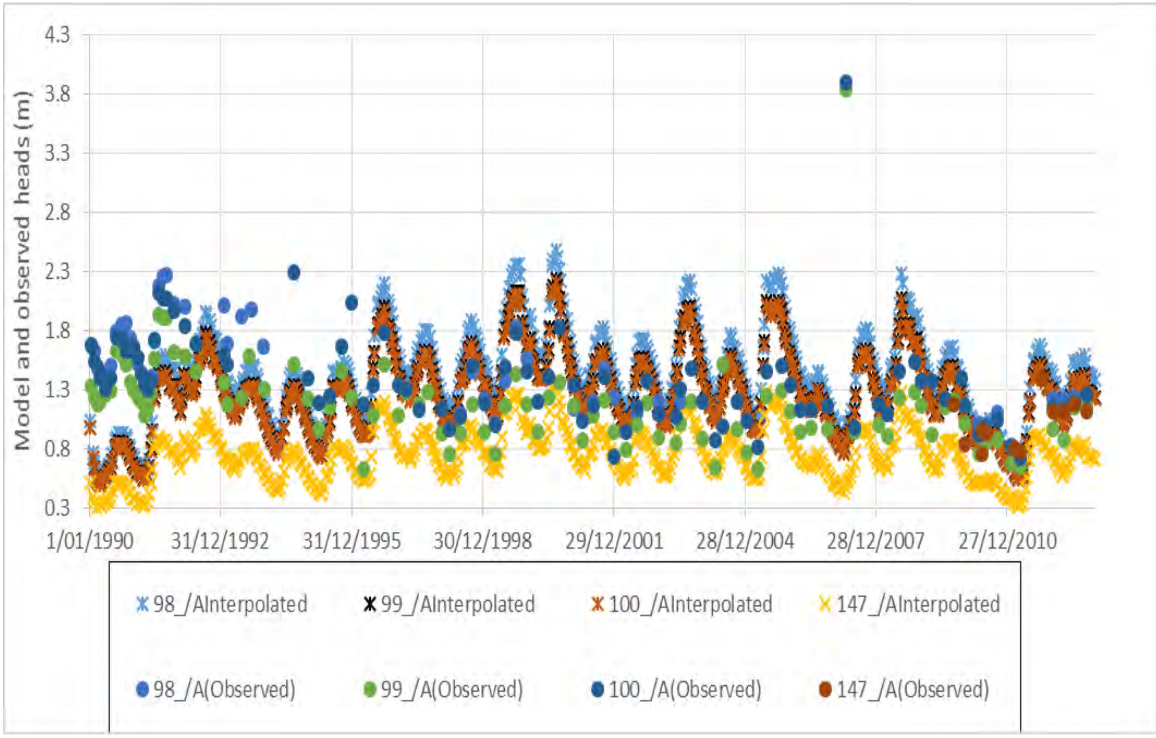
OBSERVATION WELL DATA FROM KIC MEMBERS

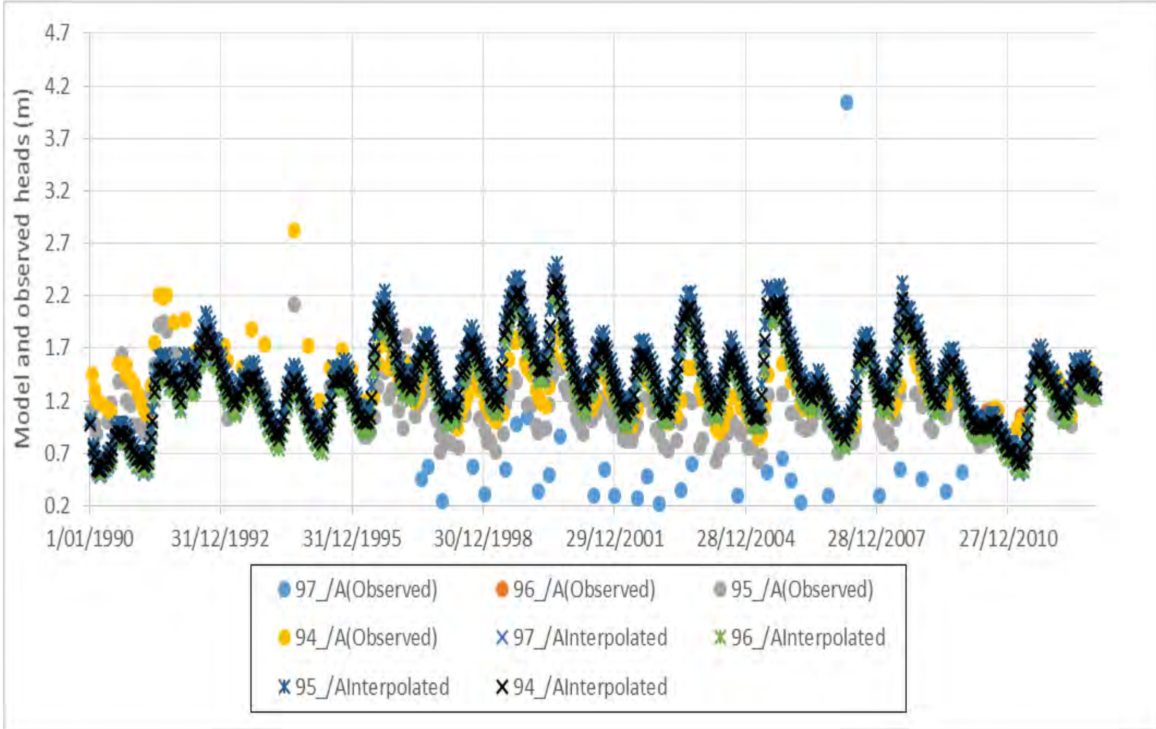
In all of the plots shown below, the model results are labelled as 'Interpolated' data. As the original data were provided under a confidentiality agreement, the well names have been modified.

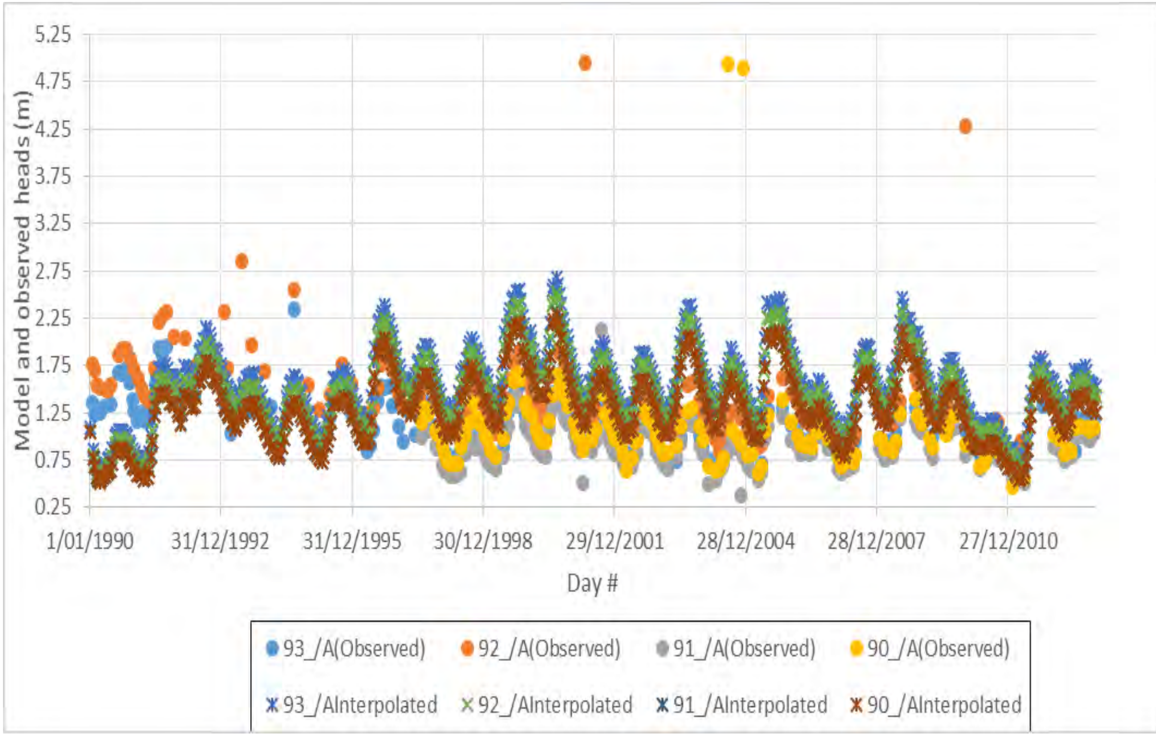


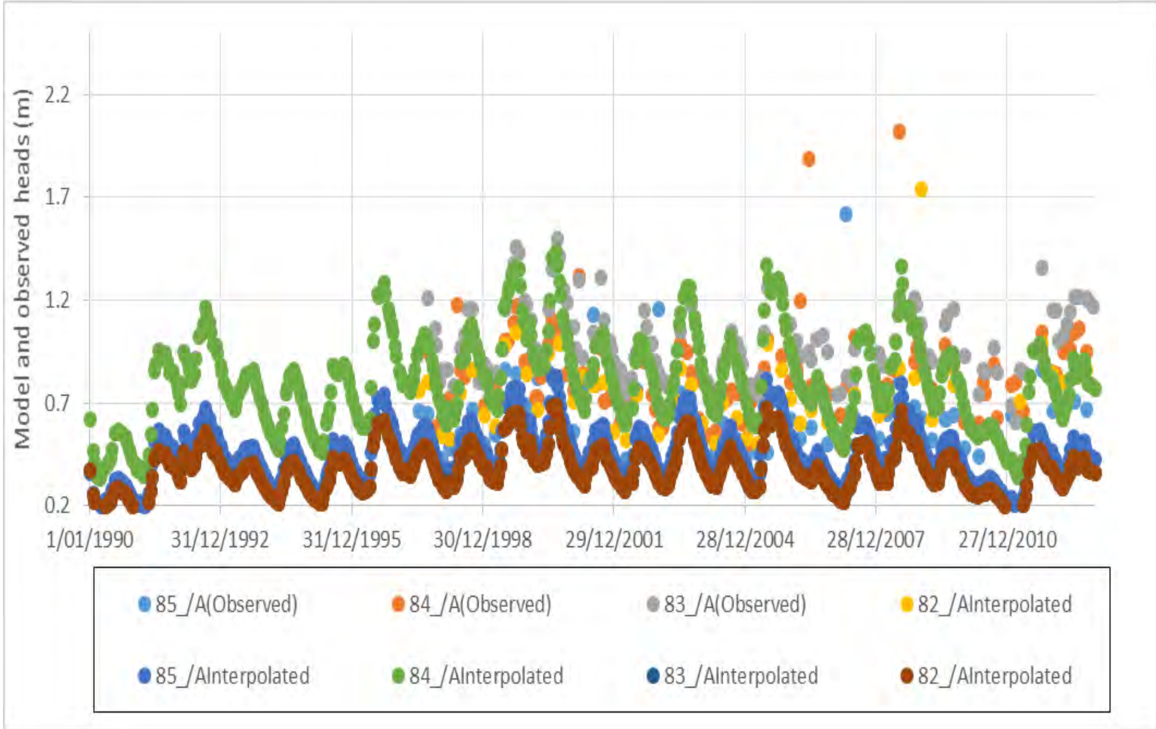


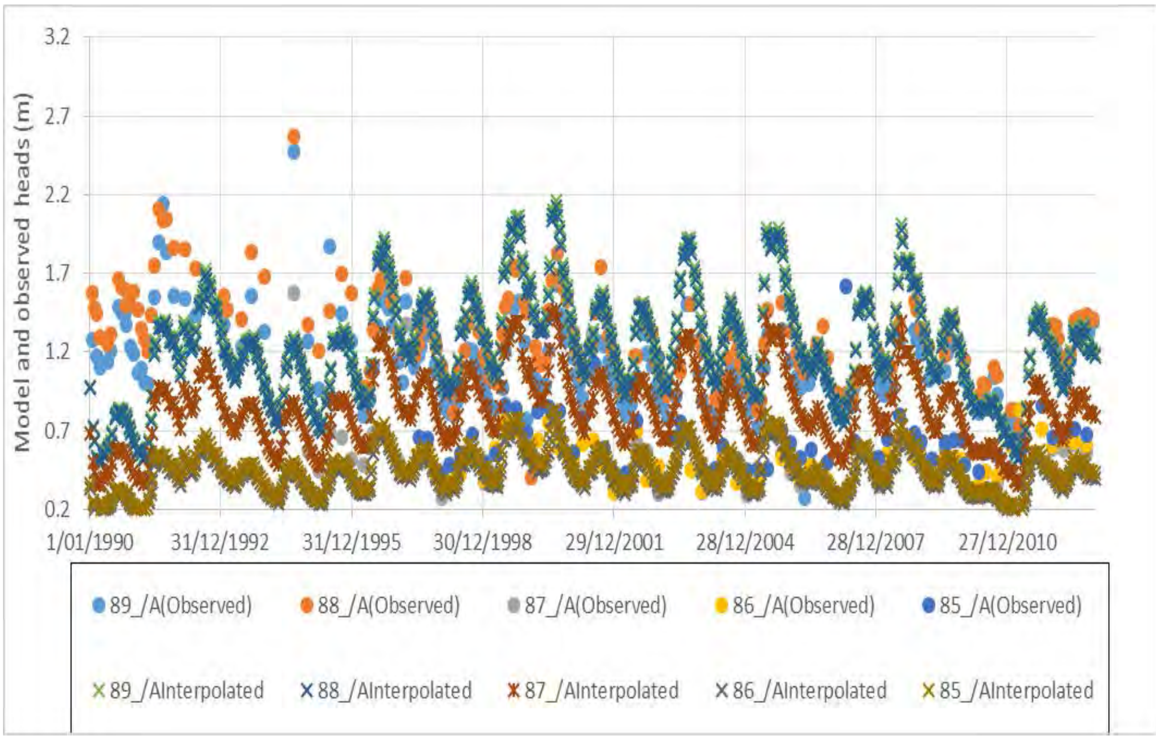

Model and observed heads (m)
1.6
1.4
1.2
1
0.8
0.6
0.4
0.2
1/01/1990
31/12/1992
31/12/1995
30/12/1998
29/12/2001
28/12/2004
28/12/2007
27/12/2010
65_/AInterpolated
66_/AInterpolated
67_/AInterpolated
68_/AInterpolated
69_/AInterpolated
70_/AInterpolated
65_/A(Observed)
66_/A(Observed)
67_/A(Observed)
68_/A(Observed)
69_/A(Observed)
70_/A(Observed)

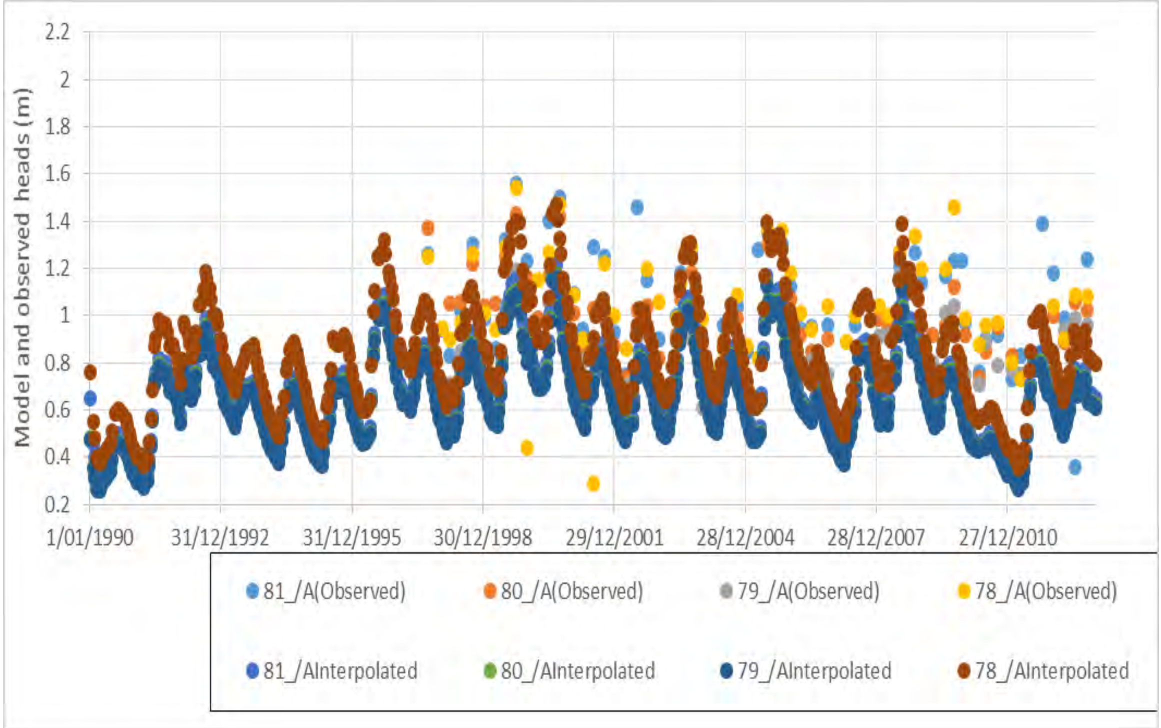

Model and observed heads (m)
4.3
3.8
3.3
2.8
2.3
1.8
1.3
0.8
0.3
1/01/1990
31/12/1992
31/12/1995
30/12/1998
29/12/2001
28/12/2004
28/12/2007
27/12/2010
98_/AInterpolated
99_/AInterpolated
100_/AInterpolated
147_/AInterpolated
98_/A(Observed)
99_/A(Observed)
100_/A(Observed)
147_/A(Observed)


4.7
4.2
3.7
3.2
2.7
2.2
1.7
1.2
0.7
0.2
Model and observed heads (m)
1/01/1990
31/12/1992
31/12/1995
30/12/1998
29/12/2001
28/12/2004
28/12/2007
27/12/2010
97_/A(Observed)
96_/A(Observed)
95_/A(Observed)
94_/A(Observed)
97_/AInterpolated
96_/AInterpolated
95_/AInterpolated
94_/AInterpolated

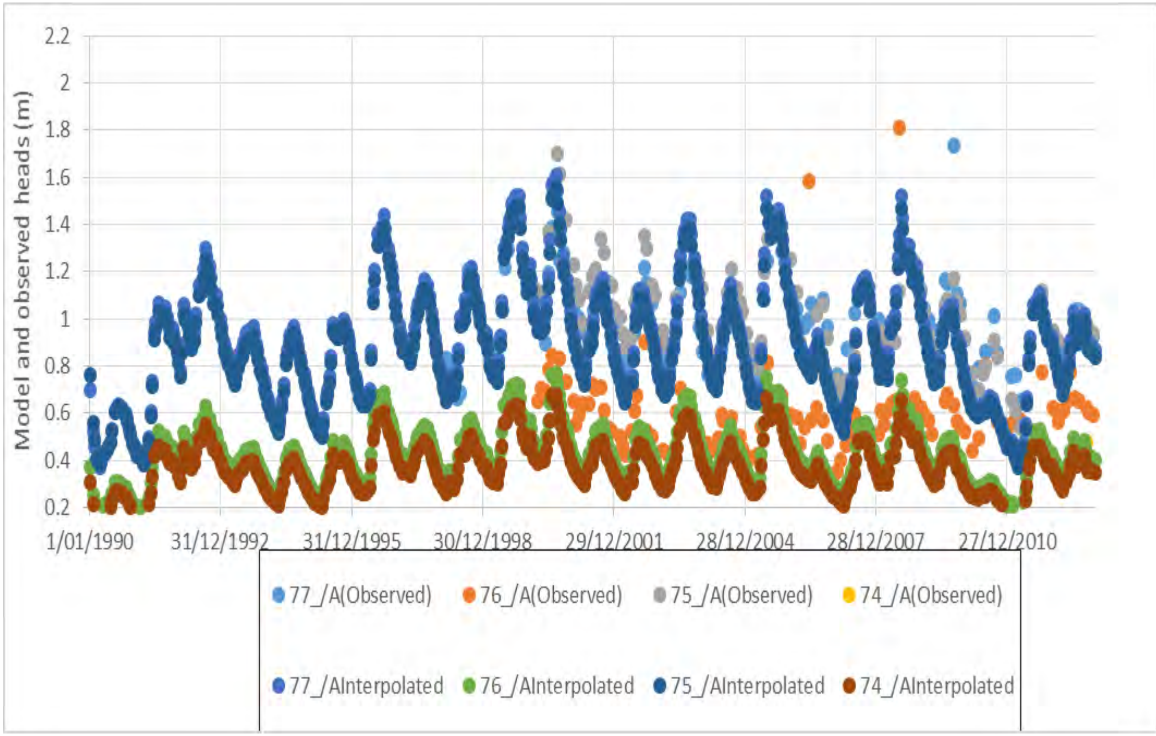

5.25
4.75
4.25
3.75
3.25
2.75
2.25
1.75
1.25
0.75
0.25
Model and observed heads (m)
1/01/1990
31/12/1992
31/12/1995
30/12/1998
29/12/2001
28/12/2004
28/12/2007
27/12/2010
Day #
93_/A(Observed)
92_/A(Observed)
91_/A(Observed)
90_/A(Observed)
93_/AInterpolated
92_/AInterpolated
91_/AInterpolated
90_/AInterpolated


Model and observed heads (m)
2.2
1.7
1.2
0.7
0.2
1/01/1990
31/12/1992
31/12/1995
30/12/1998
29/12/2001
28/12/2004
28/12/2007
27/12/2010
85_/A(Observed)
84_/A(Observed)
83_/A(Observed)
82_/AInterpolated
85_/AInterpolated
84_/AInterpolated
83_/AInterpolated
82_/AInterpolated

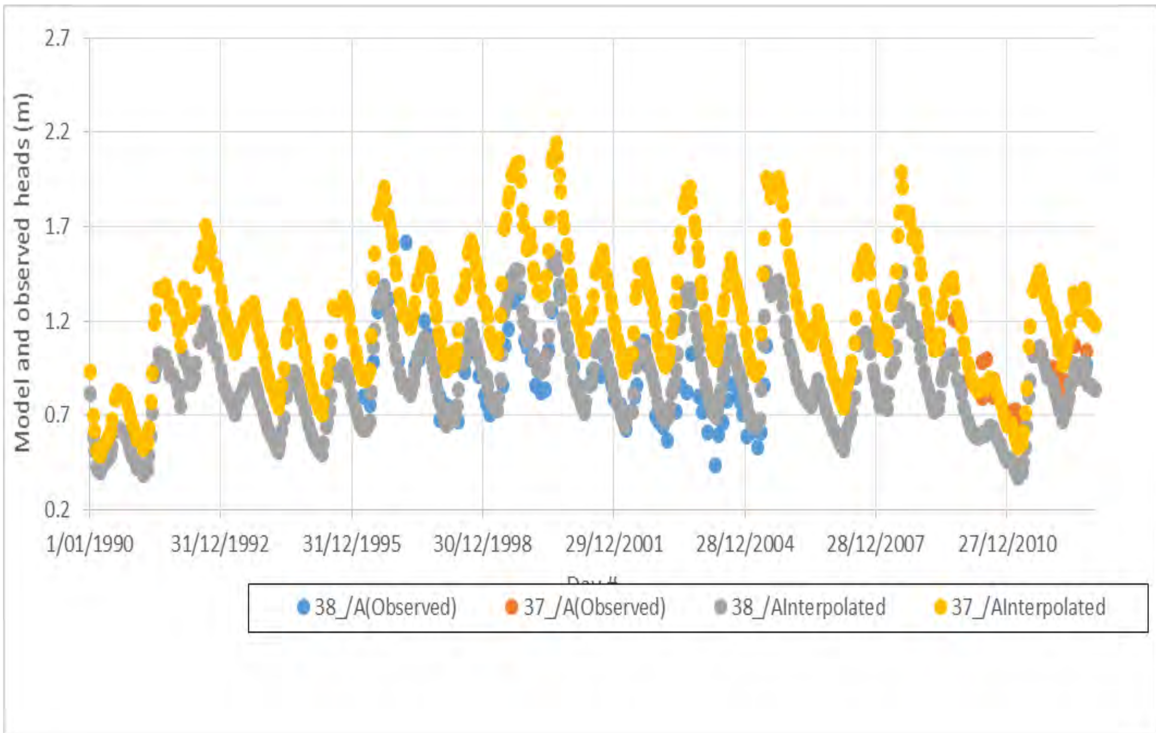

Model and observed heads (m)
3.2
2.7
2.2
1.7
1.2
0.7
0.2
1/01/1990
31/12/1992
31/12/1995
30/12/1998
29/12/2001
28/12/2004
28/12/2007
27/12/2010
89_/A(Observed)
88_/A(Observed)
87_/A(Observed)
86_/A(Observed)
85_/A(Observed)
89_/AInterpolated
88_/AInterpolated
87_/AInterpolated
86_/AInterpolated
85_/AInterpolated

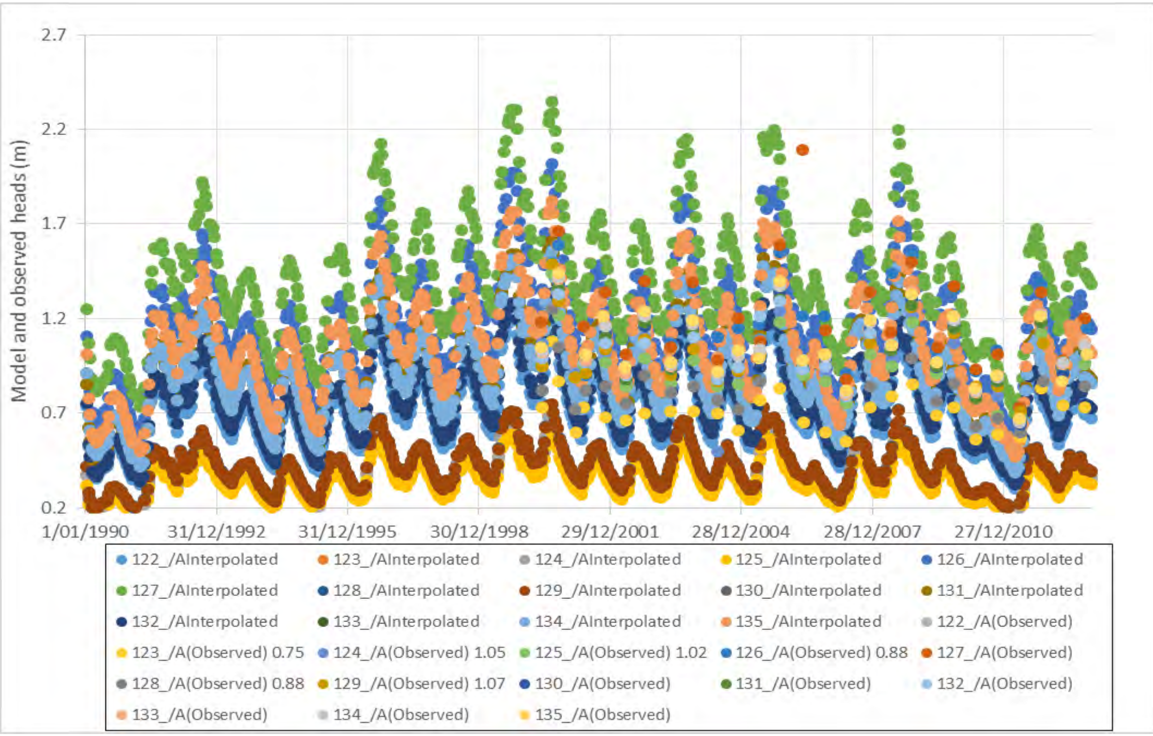


Model and observed heads (m)
2.2
2
1.8
1.6
1.4
1.2
1
0.8
0.6
0.4
0.2
1/01/1990
31/12/1992
31/12/1995
30/12/1998
29/12/2001
28/12/2004
28/12/2007
27/12/2010
81_/A(Observed)
80_/A(Observed)
79_/A(Observed)
78_/A(Observed)
81_/AInterpolated
80_/AInterpolated
79_/AInterpolated
78_/AInterpolated



Model and observed heads (m)
2.2
2
1.8
1.6
1.4
1.2
1
0.8
0.6
0.4
0.2
1/01/1990
31/12/1992
31/12/1995
30/12/1998
29/12/2001
28/12/2004
28/12/2007
27/12/2010
77_/A(Observed)
76_/A(Observed)
75_/A(Observed)
74_/A(Observed)
77_/AInterpolated
76_/AInterpolated
75_/AInterpolated
74_/AInterpolated


Model and observed heads (m)
2.7
2.2
1.7
1.2
0.7
0.2
1/01/1990
31/12/1992
31/12/1995
30/12/1998
29/12/2001
28/12/2004
28/12/2007
27/12/2010
38_/A(Observed)
37_/A(Observed)
38_/AInterpolated
37_/AInterpolated


Model and observed heads (m)
2.7
2.2
1.7
1.2
0.7
0.2
1/01/1990
31/12/1992
31/12/1995
30/12/1998
29/12/2001
28/12/2004
28/12/2007
27/12/2010
122_/AInterpolated
123_/AInterpolated
124_/AInterpolated
125_/AInterpolated
126_/AInterpolated
127_/AInterpolated
128_/AInterpolated
129_/AInterpolated
130_/AInterpolated
131_/AInterpolated
132_/AInterpolated
133_/AInterpolated
134_/AInterpolated
135_/AInterpolated
122_/A(Observed)
123_/A(Observed) 0.75
124_/A(Observed) 1.05
125_/A(Observed) 1.02
126_/A(Observed) 0.88
127_/A(Observed)
128_/A(Observed) 0.88
129_/A(Observed) 1.07
130_/A(Observed)
131_/A(Observed)
132_/A(Observed)
133_/A(Observed)
134_/A(Observed)
135_/A(Observed)

4. Summary of groundwater data from KIC members

As the original data were provided under a confidentiality agreement, the quantity of groundwater data are summarised here. Some water level data provided by industries could not be included in the calibration because there was insufficient information about the construction of the well (e.g. the slotted interval) or the geographic coordinates.

Table 16 Summary of groundwater level data from KIC members that were used in the calibration procedure.

KIC MEMBER	NUMBER OF WELLS WITH WATER LEVEL DATA UTILISED IN THE CALIBRATION PROCEDURE	TOTAL NUMBER OF WATER LEVEL MEASUREMENTS AVAILABLE FOR CALIBRATION	TIME RANGE (YEARS)
BHP	21	553	2000-2013
Coogee Chemical	45	404	2009-2013
CSBP	34	3251	1990-2013
Fremantle Ports	6	160	2003-2010
TRONOX	16	107	1996-2008
Water Corporation	24	1644	1990-2014