

EGS6003W: DISSERTATION FOR THE DEGREE OF DOCTOR OF PHILOSOPHY

**THE CAPACITY OF THE CAPE FLATS AQUIFER AND ITS ROLE IN WATER
SENSITIVE URBAN DESIGN IN CAPE TOWN**



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PhD

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Abstract

There is growing concern that South Africa's urban centres are becoming increasingly vulnerable to water scarcity due to stressed surface water resources, rapid urbanisation, climate change and increasing demand for water. Furthermore, South Africa is a water-stressed country with much of its surface water resources already allocated to meet current demands. Therefore, in order to meet the future urban water supply requirements, countries like South Africa will need to consider alternative forms of water management that focus on moving towards sustainability in urban water management. WSUD is one such approach that aims to prioritise the value of all urban water resources through reuse and conservation strategies, and the diversification of supply sources.

This study investigates the capacity of the Cape Flats Aquifer (CFA), assessing the feasibility of implementing Managed Aquifer Recharge (MAR) as a strategy for flood prevention and supplementing urban water supply. The implementation of MAR on the CFA aims to facilitate the transition towards sustainable urban water management through the application Water Sensitive Urban Design (WSUD) principles. The fully-integrated MIKE SHE model was used to simulate the hydrological and hydrogeological processes of the CFA in Cape Town at a regional-scale. Using the results of the regional-scale model, four sites were selected for more detailed scenario modelling at a local-scale. Several MAR scenarios were simulated to evaluate the aquifer's response to artificial recharge and abstraction under MAR conditions. The first objective was to evaluate the feasibility of summer abstractions as a flood mitigation strategy at two sites on the Cape Flats prone to winter groundwater flooding, *viz.* Sweet Home and Graveyard Pond informal settlements. The second objective of the study was to assess the storage potential and feasibility of MAR at two sites in the south of the Cape Flats, at Philippi and Mitchells Plain. In addition, the migration of solute pollutants from the injected or infiltrated stormwater was simulated and climate change simulations were also undertaken to account for potential fluctuations in rainfall and temperature under climate change conditions.

The results indicated that flood mitigation on the Cape Flats was possible and was likely to be most feasible at the Graveyard Pond site. The flood mitigation scenarios did indicate a potential risk to local groundwater dependent ecosystems, particularly at the Sweet Home site. Yet, it was shown that a reduction in local groundwater levels may have ecological benefits as many of the naturally occurring wetlands on the Cape Flats are seasonal, where distinct saturated and unsaturated conditions are required. Furthermore, MAR was shown to improve the yield of wellfields at Philippi and Mitchells Plain through the artificial recharge of stormwater while also reducing the risk of seawater intrusion.

MAR was shown to provide a valuable means of increasing groundwater storage, improving the supply potential of the CFA for water supply while aiding the prevention or mitigation of the seasonal flooding that occurs on the Cape Flats. Furthermore, the case was made that MAR is an important strategy to assist the City of Cape Town in achieving its WSUD objectives. MAR and groundwater considerations, in general, are essential for the successful implementation of WSUD, without which, there is an increased risk of overlooking or degrading urban groundwater resources.

The findings of this study resulted in a number of recommendations to urban water resources managers, planners and policy makers. First, MAR is an important means for Cape Town to move towards becoming a truly water sensitive city. This study indicated that the CFA should be

incorporated as an additional source of water supply for Cape Town especially considering the recent drought conditions and due to its ability for the seasonal storage of water, this would improve the city's resilience to climate change. Furthermore, it was recommended that the application of MAR on the CFA could also be used to reduce groundwater related flooding on the Cape Flats. Second, it was emphasised that urban planning, using WSUD principles is essential for the protection of the resource potential of the CFA. Finally, for the implementation of WSUD and MAR to be successful, there needs to be appropriate policy development alongside the implementation of these strategies to ensure they are achieving their initial objectives and are not causing detriment to the aquifer.

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Glossary

The following terms and definitions are based on those outlined in Armitage et al. (2014) and Dillon et al. (2009).

Aquifer: is a porous, water-logged sub-surface geological formation. The description is generally restricted to media capable of yielding a substantial supply of water.

Aquitard: is a water-saturated sediment or rock whose permeability is so low (usually owing to a layer of clay, silt, or rock) that it cannot transmit any useful amount of water.

Attenuation: means the reduction of peak stormwater flow.

Bio-retention: refers to a depressed landscaped area that collects stormwater runoff and infiltrates it into the soil below through the root zone thus prompting pollutant removal.

Catchment: refers to the area contributing runoff to any specific point on a watercourse or wetland.

Channel: refers to any natural or artificial watercourse.

Climate change: is a continuous phenomenon and refers to the change in global climatic conditions, e.g. as a result of temperature increases due to anthropogenic emissions.

Confined Aquifer: is groundwater below an aquitard (solid rock or clay) is known as a confined aquifer. The rock or clay is called a confining layer. Typically, the groundwater within a confined aquifer is under pressure which usually exceeds atmospheric pressure.

Contamination: is the introduction of microorganisms, factory produced chemicals or wastewater in concentrations that render water unsuitable for most uses.

Detention pond: is a pond that is normally dry except following large storm events when it temporarily stores stormwater to attenuate flows. It may also allow infiltration of stormwater into the ground.

Drainage: may refer to: (1) the removal of excess groundwater or surface water by gravity or pumping; (2) the general flow of all liquids under the force of gravity; or (3) the component of the MIKE SHE model that allows for drainage of groundwater to occur at a certain depth below the ground surface.

Drainage system: refers to the network of channels, drains, hydraulic control structures, levees, and pumping mechanisms that drain land or protect it from potential flooding.

Evapotranspiration: means the evaporation from all water, soil, snow, ice, vegetation and other surfaces plus transpiration of moisture from the surface membranes of leaves and other plant surfaces.

Filter or Filtration: refers to the filtering out of stormwater runoff pollutants that are conveyed with sediment by trapping these constituents on vegetative species in the soil matrix or on geotextiles.

Flood: means a temporary rise in water level, including groundwater or overflow of water, onto land.

Hydrology: refers to the physical, chemical and physiological sciences of the water bodies of the earth including: occurrence, distribution, circulation, precipitation, surface runoff, streamflow, infiltration, storage and evaporation.

Impervious surface: denotes land surfaces which prevent the infiltration of water. Roads, parking lots, pavements and rooftops are typical examples of impervious surfaces in urban areas.

Infiltration: is the process of penetration of precipitation into the ground.

Infiltration device: is a typically a depression in the land used to aid the infiltration of surface water into the ground. Generally, this is performed in areas with rates of high infiltration.

Paleochannel: is ancient river bed buried overlain by newer geological deposits.

Permeability: describes the ability of water to flow through a porous media when fully saturated and subjected to an unbalanced pressure.

Peak discharge: is the maximum rate of flow of water passing a given point during or immediately after a rainfall event.

Precipitation: is the water received from atmospheric moisture as rainfall, hail, snow or sleet, normally measured in millimetres depth.

Receiving waters: are natural or man-made aquatic systems which receive stormwater runoff or treated wastewater, e.g. watercourses, wetlands, canals, estuaries, groundwater and coastal areas.

Runoff: is the overland flow of surface water resulting from rainfall or irrigation exceeding the infiltration capacity of the soil.

Sedimentation: is the deposition of soil particles, previously suspended in moving water, as a consequence of a decrease in the velocity of flow below the minimum transportation velocity.

Source controls: are non-structural or structural best management practices to minimise the generation of excessive stormwater runoff and/or pollution of stormwater at or near the source.

Stormwater: is rainfall that runs off all urban surfaces such as roofs, pavements, car parks, roads, gardens and vegetated open space.

Stormwater system: is constituted by both constructed and natural facilities including: stormwater pipes, canals, culverts, 'vleis', wetlands, dams, lakes, and other watercourses, whether over or under public or privately owned land, used or required for the management, collection, conveyance, temporary storage, control, monitoring, treatment, use and disposal of stormwater.

SuDS: is the abbreviation for sustainable drainage systems or sustainable urban drainage systems, which are a sequence of management practices and/or control structures or technologies designed to drain surface water in a more sustainable manner than conventional techniques.

Swale: is a shallow vegetated channel designed to convey stormwater, but may also permit infiltration. The vegetation assists in filtering particulate matter.

Treatment train: is a combination of different methods implemented in sequence or concurrently to achieve best management of stormwater. These methods include both structural and non-structural measures.

Total dissolved solids (TDS): is a measurement of the total dissolved solids or salts in a solution. Major salts in recycled water typically include sodium, magnesium, calcium, carbonate, bicarbonate, potassium, sulphate and chloride. TDS is used as a measure of soil salinity with the units of $\mu\text{g}\cdot\text{m}^{-3}$ used in this study.

Unconfined aquifer: is an aquifer that is open to receive water from the surface.

Vadose zone: is the portion of the earth between the land surface and the groundwater table (otherwise known as the unsaturated zone). In this zone, pore spaces are filled with water and air.

Volatilisation: is the conversion of water (stormwater/groundwater) compounds to gas or vapour typically as a result of heat, chemical reaction, a reduction of pressure or a combination of these.

Water table: is the upper most level of the zone of saturation below the Earth's surface, except where this surface is formed by an impermeable body.

Watercourse: means any river, stream, channel, canal or other visible topographic feature, whether natural or constructed, in which water flows regularly or intermittently including any associated storage and/or stormwater attenuation dams, natural vleis or wetland areas.

Wetland: refers to any land translational between terrestrial and aquatic systems where the water table is usually at or near the surface, or is periodically covered with shallow water, and which in normal circumstances supports or would support vegetation typically adapted to life in saturated soil. This includes water bodies such as lakes, salt marshes, coastal lakes, estuaries, marshes, swamps, 'vleis', pools, ponds, pans and artificial impoundments.

Acronyms and abbreviations

AAHMS	ACRU Agrohydrological Modelling System
AD	Advection-Dispersion
AR	Artificial Recharge
ARC	Agricultural Research Commission
ASR	Aquifer, Storage and Recovery
ASTR	Aquifer, Storage, Transfer and Recovery
AWRMS	Atlantis Water Resource Management Scheme
BMPs	Best Management Practices
BRNS	Biogeochemical Reaction Network Simulator
CBD	Central Business District
CFA	Cape Flats Aquifer
CMIP5	Coupled Model Intercomparison Project Phase 5
CoCT	City of Cape Town
CSIR	Council for Scientific and Industrial Research
DEA	Department of Environmental Affairs
DEM	Digital Elevation Model
DEM	Digital Elevation Model
DRDLR	Department of Rural Development and Land Reform
DWA	Department of Water Affairs
DWAF	Department of Water Affairs and Forestry
DWS	Department of Water and Sanitation
FAO	Food and Agricultural Organisation
GCMs	Global Circulation Models
GHG	Greenhouse Gas
ICBMs	Integrated Component-based Models
IUDMs	Integrated Urban Drainage Models
IUWCMs	Integrated Urban Water Cycle Models
IUWM	Integrated Urban Water Management
IUWSMs	Integrated Urban Water System Models
LAI	Leaf Area Index
LID	Low Impact Development
LiDAR	Light Detection and Ranging
MAR	Managed Aquifer Recharge
MUSIC	Model for Urban Stormwater Improvement Conceptualisation
NEIMO	Network Exfiltration and Infiltration Model
NGA	National Groundwater Archive
NLC	National Land Cover
OGS	OpenGeoSys
PHA	Philippi Horticultural Area

PT	Particle Tracking
RCP	Representative Concentration Pathways
RMSE	Root Mean Square Error
SAWS	South Africa Weather Service
SCMs	Stormwater Control Measures
SLAMM	Source Loading And Management Model
SLeakI	Sewer Leak Index
SUDS	Sustainable Urban Drainage
SuDS	Sustainable Drainage Systems
SUWM	Sustainable Urban Water Management
SWMM	Storm Water Management Model
TDS	Total Dissolved Solids
TMG	Table Mountain Group
TWM	Total Water Management
US EPA	United States Environmental Protection Agency
UVQ	Urban Volume and Quality
WCS	Water Sensitive Cities
WCWSS	Western Cape Water Supply System
WEAP	Water Evaluation and Planning
WSUD	Water Sensitive Urban Design
ZAR	South African Rand

Symbols

ET_o	Reference evapotranspiration
ET	Actual Evapotranspiration
Δ	Slope of the saturated vapour pressure versus temperature curve
G	Soil heat flux
K_c	crop factor
K_{sat}	Saturated hydraulic conductivity
M	Manning's M coefficient
N	Manning's n coefficient
R^2	Coefficient of determination
R_n	Net radiation
T_a	Average Temperature
T_{max}	Maximum temperature
T_{min}	Minimum temperature
T_r	Temperature range
A	Priestley and Taylor Constant
Γ	The psychometric constant
θ_{fc}	Water content field capacity
θ_s	Water content at saturation
θ_{wp}	Water content wilting point

1. Introduction

1.1 Background

There is growing concern that South Africa's urban centres are becoming increasingly vulnerable to water scarcity due to pressure on the current water supply system, rapid urbanisation, climate change and increasing demand for water (DWAF, 2009). Moreover, South Africa is classified as a water-stressed country due to its relatively low mean annual rainfall of approximately 450 mm.a⁻¹, of which about 91% is lost to evaporation owing to the high potential evaporation rates, which can exceed 3000 mm in the most arid regions of the country (Schulze, 1997; DEA, 2012). South Africa's rainfall is also unequally distributed across the country with the east coast experiencing between 600 and 1200 mm of rainfall compared to the dryer west coast, which receive less than 100 mm in some areas. Furthermore, it is estimated that 98 % of South Africa's surface water resources already allocated to meet various socio-economic demands, with limited capacity for expansion (Turton, 2008).

In order to overcome these issues of water scarcity, climatic variability and the uneven spatial distribution of rainfall, South Africa has developed a complex surface water management system of dams and inter-catchment transfer schemes. In doing so, South Africa is able to capture and store runoff generated in high rainfall, mountainous catchment areas and transport the stored water to the nearest cities, ensuring that the country's water requirements are met. Presently, there are approximately 320 major dams in South Africa with a total storage capacity of 32 412 Mm³, which equates to approximately two-thirds of the country's mean annual runoff (DWAF, 2004). Additional storage infrastructure, including small dams, direct abstractions from rivers and water use through afforestation, mean that many catchments in South Africa are water-stressed making the further exploitation of surface water using dams, increasingly unfeasible (Van Rooyen and Versfeld, 2010). To meet water demands in key catchments, inter-basin transfers are used to move water from catchments with a surplus water supply to those with a water deficit. The most stressed catchments in South Africa generally contain South Africa's major urban areas such as Johannesburg, Tshwane (Pretoria), Cape Town and eThekweni (Durban). Currently, over 3000 Mm³ or approximately 10 % of the natural flow regime is transferred per year (Van Rooyen and Versfeld, 2010; DEA, 2012; Bourblanc and Blanchon, 2014). Examples of these major transfer schemes include the Lesotho Highlands Water Scheme and the Thukela-Vaal transfer scheme, which supply water to Gauteng with transfers from Lesotho and Kwa-Zulu Natal, respectively. The ability to store and transfer high volumes of surface water is limited, as many catchments in South Africa have constrained surface water resources which are over-allocated (DWA, 2013a). Therefore, given the strain on South African surface water resources and the limitations of conventional engineering interventions such as constructing dams and inter-basin transfers, there is growing emphasis on identifying alternative supply options to meet water demand including groundwater, water conservation and demand management, reuse and desalination (Van Rooyen and Versfeld, 2010).

Groundwater is an attractive alternative water supply, that can be more resilient to climate change and offer protection from pollution risks, and in many cases is cheaper and more easily assessable than surface water resources (DWA, 2010; Shah, 2014). Furthermore, groundwater is

seen as valuable resource in South Africa as it has the potential to substantially contribute towards meeting the urban water demand of South Africa in the future (DWA, 2010). It is estimated that over 280 towns and settlements in South Africa are largely dependent on groundwater (Sililo *et al.*, 2001). However, most major urban areas rely on surface water from large dams, which is then pumped tens to hundreds of kilometres to the cities to meet urban and industrial water demands (DWA, 2010). A number of cities already use local groundwater to supplement their water supply. The City of Tshwane has used groundwater in the past and currently parts of the CBD and the suburbs to the east and west of the city centre receive 57 Mℓ.day⁻¹ of water from the groundwater springs and boreholes south of the city, which accounts for approximately 7.5 % of the total water demand (CoT, 2011; Dippenaar, 2013). The City of Cape Town has access to a number of groundwater resources: Albion Springs, the Atlantis Aquifer, the Table Mountain Group (TMG) Aquifer, the Cape Flats Aquifer and the Newlands aquifer. Cape Town's groundwater, and water resources in general, are discussed in more detail in the next section (Section 1.2.1), while a more general review of the groundwater resources of South Africa is discussed in Appendix A.

Although groundwater is a valuable resource with the potential to further contribute towards water supply in Cape Town and South Africa at large, it is likely to remain a relatively small component of the total water supply system. Therefore, there is a shift in urban water management away from encouraging conventional urban water management approaches and interventions, towards sustainable urban water management (SUWM). One such approach is Water Sensitive Urban Design (WSUD), which seeks to place value on all forms of water in the urban context with emphasis on conservation and reuse strategies. In addition, this thesis seeks to assess the optimisation of the resource potential of groundwater and aquifers using Managed Aquifer Recharge (MAR). MAR is the process of intentionally enhancing the recharge of an aquifer (Dillon *et al.*, 2009). The use of MAR in aquifers offers valuable opportunities for the potential storage and reuse of stormwater or treated wastewater (Water by Design, 2009; Wong *et al.*, 2012). This thesis seeks to investigate the potential opportunities for MAR to contribute toward the optimisation of WSUD and its successful implementation in South Africa.

1.2 Study Site

1.2.1 Cape Town

Located in a Mediterranean climate, Cape Town is a highly urbanised and ecologically sensitive area that is prone to water scarcity (DWA, 2009; Rebelo *et al.*, 2011). The city experiences regular episodes of hydrological drought, including the periods from 1986 – 1988, 1991 – 1992, 2000 – 2001, 2004 – 2005 and 2015 – 2017 (Mukheibir and Ziervogel, 2007). Cape Town is an important city for South Africa's economy contributing 11.3 % towards South Africa's GDP, the second largest contribution behind the City of Johannesburg at 16.8 % (CoCT, 2014a). The economic sectors of manufacturing, commerce, agriculture and tourism have driven the city's economy over the last two decades, exceeding the national average economic growth rate throughout this period. This growth is anticipated to continue at between 4 – 6% in the short to medium term (CoCT, 2013a; WESGROW, 2013). The population of Cape Town has also increased rapidly over the last two decades showing an increase of approximately 46% between 1996 and 2011. Further population growth is anticipated, with areas to the North of the city being earmarked for future

housing developments to accommodate the expected population increases (CoCT, 2013a). These projected economic and population growth rates are set to place additional pressure on the city's already strained water supply system. The current water supply system relies largely on surface water resources and it is estimated that this system will need to be augmented by 2020 (CoCT, 2013a). Given Cape Town's water scarcity, increasing water demands and stressed water supply system, coupled with global awareness around the issues of sustainable development, environmental protection and climate change, alternative urban water management strategies are required to ensure urban water is managed sustainably.

The City of Cape Town receives about 98.5 % of its water supply from surface water resources. Most of the rainfall in the Western Cape falls during the winter months (May – September), thus the runoff generated during these periods needs to be stored to meet the water demands of the city for the whole year. In particular, enough water needs to be stored in order to meet the peak demands during the hot dry summer. A network of major and minor dams that form the Western Cape Water Supply System (WCWSS) are used to store winter runoff. Six major dams, located in the mountain catchments areas in the Boland and Hottentots-Holland mountain ranges, supply 99.6 % of the CoCT's raw water. The remaining 0.4 % is met through minor dams such as Hely-Hutchinson and Woodhead on Table Mountain; Victoria, Alexandra and De Villiers near Constantia Nek; and Kleinplaats and Lewis Gay near Simon's Town. The total storage capacity of the major dams is 898 Mm³. The WCWSS has an annual yield of 556 Mm³ of which 399 Mm³ is allocated to the CoCT, while the remaining yield is allocated to other urban areas and for agriculture. In total, 73.2 % of the water supplied to the CoCT is from dams owned by the Department of Water and Sanitation (DWS) and the remaining 26.8 % is from dams owned by the CoCT. It is also important to note that 87 % of the CoCT's water is 'imported', meaning that it is stored in catchments outside of the municipal boundaries.

The city's groundwater resources include Albion Springs, the Atlantis Aquifer, the Table Mountain Group (TMG) Aquifer, the Cape Flats Aquifer (CFA) and the Newlands Aquifer, all of which are largely underutilised. Presently, only the Atlantis Aquifer is used for bulk water supply making up 1.7 % of the total bulk water supply to the City of Cape Town, however, the groundwater from the Atlantis aquifer meets more than 50 % of the town of Atlantis' water supply (CoCT, 2014b). The City of Cape Town actively promoted the use of some of these groundwater resources for private applications like urban and residential irrigation, however, this usage is largely unregulated and the only requirement is that households using groundwater for private irrigation purposes must register this use, and fix a notice on the property to indicate that water is being sourced from a well-point or borehole (CoCT, 2013b). The benefit of using groundwater for household irrigation is that it reduces the demand on potable water supply, but due to the lack of regulation, there is little knowledge about the potential impacts of this private use. The role of the CFA for private groundwater use is discussed further in Appendix B.

The CFA in particular, has been the subject of much debate over the last few decades. Its potential utilisation has been deliberated for the last 40 years, however no formal abstraction or management plans have been commissioned for the aquifer. A number of feasibility studies have been conducted on the aquifer since the late 1970's and early 1980's, which aimed at assessing the potential for bulk water supply and the artificial recharge of the aquifer using treated wastewater and stormwater. All of these studies concluded that the CFA was a valuable resource (Henzen,

1973; Wessels & Greef, 1980; Tredoux, 1981; Vandoolaeghe, 1990; Wright & Conrad, 1995; Fraser & Weaver, 2000). There has been a renewed interest in the groundwater of the CFA in recent years from the Department of Water Affairs (DWAf), Water Research Commission (WRC), City of Cape Town, Council for Scientific and Industrial Research (CSIR) and other independent organisations. This interest is mainly focused on the CFA as a bulk water supply source to help Cape Town meet its future water requirements.

1.2.2 Cape Flats Aquifer

The Cape Flats is a coastal sand plain in the city of Cape Town, South Africa covering an area in excess of 400 km², between the coast of False Bay in the South and the Tygerberg Hills in the North (Hay, 1981; DWAf, 2008). The CFA is a primary aquifer situated in the Quaternary sands of the Cape Flats. These sand deposits include fluvial, marine and aeolian deposited sands, underlain by Malmesbury Group and Cape Granite basement rock (DWAf, 2008). The bedrock of the CFA shows a unique channel formation or ‘paleochannel’ extending in a North-South direction through the central parts of the Cape Flats. This geological feature is believed to be an important factor in governing the groundwater flow processes of the CFA. It has been suggested that the groundwater of the CFA flows in a north – south direction towards False Bay, as much of the flow is conducted through the paleochannel (DWAf, 2008).

The topography of Cape Town is unique due to the relatively low change in elevation ranging from 0 m above sea-level at the coast to 110 m above sea-level in the north-east. By contrast, the topography surrounding the Cape Flats is marked by significant changes in elevation around Table Mountain, reaching a height of 1071 m above sea-level (DWAf, 2008). Elevated mountainous areas in the case of the Table Mountain range receives considerably more rainfall (>1000 mm.a⁻¹) than the lower altitude Cape Flats (~600 mm.a⁻¹) (Schulze *et al.*, 2008). As a result, many of the rivers that flow over the Cape Flats originate in these mountainous areas. The drainage of the Cape Flats is categorised into three main Quaternary catchments: G22D and G22E flow south into False Bay, while G22C flows east into Table Bay (Figure 1.1). Due to its flat topography, the hydrology of the Cape Flats is associated with wetland or seasonal wetland conditions. Yet, with the rise in urban development on the Cape Flats, many of these wetlands or “kuils” were removed through canalisation so as to ensure the rapid removal of stormwater. The introduction of canals has resulted in artificial river conditions with channels that are uniform in geometry and have a low surface roughness. Furthermore, because of the increased drainage the flow in these canals is maintained through the dry season, whereas most of the rivers that flow over the Cape Flats under natural conditions are non-perennial (Rebello *et al.*, 2011).

The land use over the Cape Flats is predominantly urban (*viz.* informal and formal residential and general urban land use) (Figure 1.2), but there are small pockets of agriculture and natural vegetation in various protected areas (Figure 1.1). Much of the urban development over the Cape Flats has occurred since the late 1970’s and early 1980’s where apartheid planning, guided by the Group Areas Act (1950), forced ‘black’ and ‘coloured’ people to live in outlying areas away from the City of Cape Town’s central business district. Thus, a substantial portion of the urban areas on the Cape Flats consists of low to medium income residential or informal settlements. As a result, this may have negatively impacted the CFA as many of these areas lacked formal housing and sanitation for periods prior to 1994. The risk of pollution remains in some areas as the lack of

sanitation is still a problem in many informal settlements. In addition to this, other sources of pollution including industry, landfill sites, cemeteries and wastewater treatment works have also become more prevalent (DWAF, 2008).

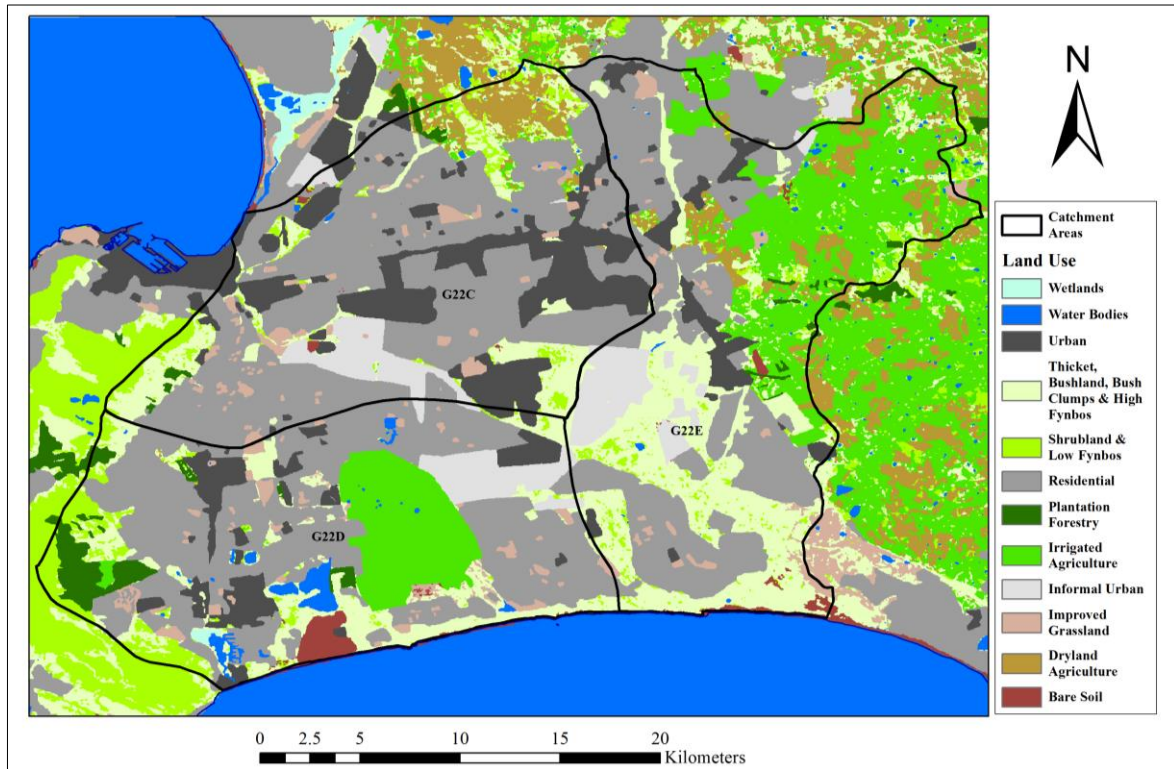


Figure 1.1: Land use classes within the Cape Flats area

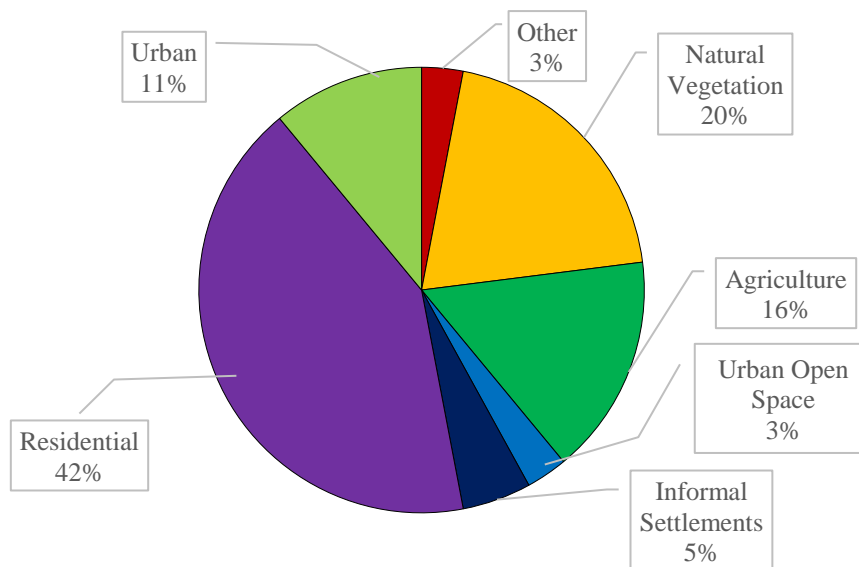


Figure 1.2: Land use in the Cape Flats

1.3 Problem Statement

1.3.1 Water Supply

As mentioned previously, the WCWSS relies heavily on surface water to meet the water demand for Cape Town. The WCWSS Reconciliation Strategy (DWA, 2013b) has forecast a number of possible scenarios for water demand and suggest possible options available to meet the future demands such as increasing the capacity of Voëlvlei dam, using the Table Mountain Group (TMG) aquifer, reuse strategies and desalination. The report estimates that in the event of a high annual growth in urban water demand (~3.38 %) due to an inability to satisfy water conservation and demand management objectives, the current water supply system would need to be augmented by the year 2022. However, if persistent dry conditions as a result of climate change lower the yield obtainable from dams, then the augmentation of the current water supply system would need to be achieved by 2021. Furthermore, projections of lower growth rates in future water demands suggest that the water supply system would need to be augmented by 2025 (DWA, 2013b). Therefore, based on these timeframes, it is evident that within the next decade additional sources of water supply will be required to meet the city's future water requirements. Based on the wide range of proposed sources of water, it is anticipated that there will be a shift away from large, high-yield water supply schemes, and rather moving towards a water supply system that consists of a blend or integration of a number of smaller, lower yielding water supply schemes. Many of the proposed water supply interventions such as desalination and proposals to increase in the capacity of Voëlvlei dam are likely to be expensive projects, while is desalination is known to have high running costs. Schemes such as the TMG aquifer have been considered for a number of years, but have struggled to gain momentum. The costs of the TMG scheme are also likely to be expensive due to the drilling required to access the high yielding TMG fractured sandstone aquifer. There are few details outlined with regard to the reuse strategies mentioned in the WCWSS Reconciliation Strategy and what these strategies might entail. Given the relatively short period of time before the augmentation of WCWSS is required, there is a need to identify water supply opportunities for Cape Town that optimise the available resource potential and reduce the per-unit cost of water supplied.

1.3.2 Flooding on the Cape Flats

In addition to the hot, dry summer conditions that result in limited access to water, Cape Town is prone to flooding during the wet winter months (Joubert and Martindale, 2013). The intensity and frequency of rainfall conditions are predicted to increase, worsening the impacts of flooding (Ziervogel and Smit, 2009). The flooding in Cape Town occurs mainly on the Cape Flats where there is poor drainage and is often associated with wetland conditions (Joubert and Martindale, 2013; Waddell and Ziervogel, 2014). Flooding on the Cape Flats mainly affects the poorest portions of the city as this was where many people were forced to live during the Apartheid era, pre-1994. Many of the 'Black' and 'coloured' communities in the Cape Town were moved away from the city centre and relocated to the Cape Flats. Additionally, migration has also contributed to the increased risk associated with flooding on the Cape Flats, as people who move to Cape Town seeking employment opportunities are often unable to afford accommodation. As a result, they are forced by their circumstances to live in cheaper informal areas that tend to be located on unsuitable land such as wetland or stormwater detention basins that are prone to flooding

(Olorunfemi, 2011). Thus, many of the areas that lack formal housing and infrastructure are at a greater risk of experiencing flooding.

There is little information on the processes that drive urban flooding on the Cape Flats, however it is generally linked to poor drainage, lack of or poor maintenance of local infrastructure or flooding as a result of a high groundwater table leading to ponding. A number of qualitative reports, based on stakeholder interviews, have shown there is a strong trend in literature towards groundwater related flooding often termed as ‘seepage’ flooding. Residents have noted that water rises up from underground to cause flooding in the home. It is important to note that this mainly occurs in areas associated with informal housing where dwellings are often constructed from scrap wood, plastic and corrugated iron without solid foundations. However, Pharoah (2013) in her thesis entitled *“Sometimes I Think the Shack was Better”: Examining Flood-Risk in Subsidised Housing Areas in Cape Town*, showed that merely supplying low-cost or subsidised housing does not overcome the issues of groundwater flooding on the Cape Flats. Pharoah (2013) noted issues such as rising damp and mould still remain a problem in formal dwellings. Therefore, it is clear that flooding as a result of elevated groundwater levels is a problem on the Cape Flats that impacts both formal and informal housing and posed the highest risk to the most vulnerable in Cape Town.

1.4 Research aim and objectives

This study examines the potential of groundwater and the application of WSUD for ensuring the sustainability of urban water management. The aim is to investigate the feasibility of implementing Managed Aquifer Recharge (MAR) —as a WSUD strategy—on the CFA, so as to (i) improve the water security of the City of Cape Town and (ii) to manage stormwater on the Cape Flats to prevent groundwater related flooding. Thus, the following research questions were posed:

1. *Is there sufficient storage capacity in the Cape Flats Aquifer for the recharge of winter stormwater from urban areas? Could this storage be enhanced through controlled summer abstractions for fit-for-purpose uses?*
2. *Can summer abstractions be used to mitigate groundwater related flooding on the Cape Flats?*

An integrated hydrological modelling approach was adopted to represent the hydrological and hydrogeological processes of the CFA. The physically-based MIKE SHE model was selected for the modelling of the CFA due to its ability to represent both surface and groundwater processes respectively. The first objective of this research was to prepare and calibrate the MIKE SHE model to identify the hydrological processes that influence groundwater recharge, storage and surface water interactions in the CFA—at a regional-scale —using available data, information from literature and past conceptual hydrogeological models for the CFA. This modelling approach is important for testing the established conceptual hydrogeological models, identifying the dominant hydrological and hydrogeological processes, and determining the groundwater resource potential of the CFA. The second objective was to test the applicability of WSUD techniques and technologies on the Cape Flats, specifically addressing the application of Managed Aquifer Recharge (MAR) for the management and storage of stormwater in Cape Town. This will provide insight into the application and management of MAR, helping answer the following questions:

- Is it feasible, in terms of the local hydrology and hydrogeology, to implement MAR for stormwater management on the Cape Flats?
- What are the rates and volume of possible recharge and storage for stormwater for short-term and long-term storage in the CFA?
- How much abstraction is required for the CFA during the summer months to prevent flooding or to optimise MAR storage?
- What are the risks of using the CFA for storage?
- What are the major hydrological and hydrogeological limitations for the implementation of MAR on the Cape Flats?

These research aims provide insight into the regional hydrological and hydrogeological processes that determine the resource potential of the CFA and its suitability for MAR. Therefore, it was hypothesised; (i) that the CFA is a valuable resource that has sufficient additional storage capacity for the artificial recharge of stormwater, thereby increasing the yield of the aquifer and; (ii) the seasonal groundwater flooding on the Cape Flats could be mitigated using localised groundwater abstractions. The research questions and the hypotheses have important implications for the future implementation of WSUD in the City of Cape Town.

This application of an integrated hydrological model is also valuable as it serves to increase the understanding of the CFA and the processes that govern its recharge, storage and flow. A valuable advantage of an integrated hydrological model like MIKE SHE is the physical-conceptual representation of surface water and groundwater interactions. In many of the past hydrogeological studies on the CFA, these interactions were assumed or estimated. Hence, the hydrological modelling in this study—which used a fully-integrated, physically-based modelling approach—allowed for the testing of different WSUD and MAR strategies on the Cape Flats so that the feasibility of each strategy could be evaluated.

1.5 Thesis outline

The structure of this thesis consists of six chapters followed by a list of references and appendices. A brief summary of the main chapters of the thesis is provided below.

Chapter 1 provides a brief overview of the context of water resources in Cape Town, South Africa. Addressing the problem statement and motivation for this study moving on to the aim and objectives of this research.

Chapter 2 conducts a detailed review of the literature pertaining to historical urban water management, alternative urban water management strategies such as Water Sensitive Urban Design (WSUD), urban groundwater, Managed Aquifer Recharge (MAR) and integrated hydrological modelling.

Chapter 3 details the research method used in this study, describing the data requirements for parameterisation, calibration and validation of the MIKE SHE model at a regional and local-scale that was used to evaluate the feasibility of MAR for stormwater management using the CFA.

Chapter 4 describes the results of the integrated hydrological modelling scenarios for determining the MAR potential, the potential for flood mitigation and the storage potential of the

CFA. This chapter then concludes with an assessment of the implications of contaminant transport and the threat of climate change.

Chapter 5 provides a discussion of the results in relation to literature and the research objectives. This chapter outlines the implications of the research and its applicability within the broad research fields of MAR, WSUD and urban water management in general.

Chapter 6 presents the key findings of this research and how these have contributed to knowledge. Furthermore, this chapter suggests management considerations for the CFA and groundwater in Cape Town, while discussing the limitations of this study and outlining recommendations for further research.

The **Appendices** provide supplementary documentation in support of the main thesis chapters, including, literature reviews relating to South African groundwater resources, the urban water cycle, the status of private groundwater use in Cape Town and the concept of Water Sensitive Settlements.

2. Literature Review

This chapter provides an assessment and insight into the literature relating to the evolution and current status of the relevant research areas including urban groundwater, Water Sensitive Urban Design (WSUD) and Managed Aquifer Recharge (MAR). The goal and objectives of each approach are compared, stressing the necessity for the research undertaken in this study. The chapter begins with a brief assessment of conventional urban water management practices, and the shift towards more sustainable urban water management strategies. The role of groundwater is also addressed, discussing the current perceptions of groundwater and groundwater management in urban areas. Thereafter, the concept and theory of WSUD are discussed as a strategy for achieving the conceptual ideals of the 'Water Sensitive City', aiming to ensure urban water is sustainably managed. The chapter then deals with the role of groundwater and groundwater management within the WSUD approach. The concept of MAR is discussed, explaining the related terminology, processes, risks and benefits, while assessing international and local case studies. Finally, the chapter concludes with a synthesis of the literature making note of the research gaps.

2.1 Urban water management

2.1.1 Conventional urban water management

The increasing growth of urban settlements has significantly altered the hydrological cycle when compared to the natural, pre-developed system (Niemczynowicz, 1999). The relationship between urban land use and hydrological response is outlined in Appendix C. In summary, Appendix C discusses the main impacts on the hydrological cycle which are, *inter alia*, increases in the amount and rate of urban runoff due to the increase in impervious surfaces like tar and concrete resulting in much greater stormwater generation in urban areas. Furthermore, the increases in urban runoff, together with decreases in evapotranspiration result in less groundwater recharge in urban areas.

Innovative and efficient management procedures are required to deal with the impacts of urban land use and imperviousness. Since water is essential for human survival, access to water is a prerequisite for human civilisation (Bergkamp *et al.*, 2015). The earliest civilisations focused attention on water supply. The Indus and Mesopotamian civilisations established complex water supply systems from as far back as *ca.* 2800-1100 BC (Mays *et al.*, 2007; Koutsoyiannis *et al.*, 2008). At a similar period, the Minoans on the Island of Crete had also developed a complex water supply system with the addition of a robust sewerage network (Burian and Edwards, 2002). Sanitation services in the form of sewage systems typically occurred after the basic requirements of water supply were met.

During the rise of the Roman Empire there were many significant advances in water management, observed in the complex sewerage networks that used excess water from aqueducts to ensure the rapid removal of sewage from the city. The Romans were also the first to drain stormwater from roads (Burian and Edwards, 2002). Yet, in Europe, during the Dark Ages, cities became decentralised coupled with a societal apathy towards sanitation services (Burian and Edwards, 2002). Thus, much of the urban drainage infrastructure established by the Romans fell into disrepair as it was considered an unnecessary service. In the later part of the Middle Ages, increasing urban populations in European cities like London and Paris resulted in increasing

quantities of human sewage that lead to health epidemics. It was during this time, between the 14th and 17th Centuries, that sewerage infrastructure was installed, accompanied by policies and laws that govern urban sanitation (Burian and Edwards, 2002). In the 19th and 20th Centuries, there were advances in all forms of urban water management in Europe and the United States, including; improved construction materials and methods, the identification of waterborne diseases, the introduction of wastewater treatment, advances in urban hydrology and increased environmental awareness (Burian and Edwards, 2002).

The advances in field of urban water management made during the 19th and 20th Centuries provide the foundations of many of the water systems currently installed in cities around the world. The use of these systems to manage urban water is known as conventional urban water management. Mitchell (2006) suggests that conventional urban water management is primarily concerned with the “*water system*”, that is, providing water supply, sanitation and drainage services. Delleur (2003 p. 568) argues that the two main objectives of current urban water management:

- i) “*To protect and maintain safety and health of communities (by the removal of flood waters without interference in the activities within the city, and the removal of human waste to maintain a sanitary environment)*”, and
- ii) “*To protect the natural environment (by maintaining environmental standards involving limits on the pollution of natural streams and the atmosphere).*”

While water supply, sanitation and drainage services are essential services for the survival and healthy functioning of urban areas, the conventional approach to urban water management does not address the broader issues of environmental, ecological and social-economic sustainability (Delleur, 2003; Mitchell, 2006; Brown *et al.*, 2008). Sustainability in urban water management involves planning and implementing urban water policies, procedures and management practises that are cognisant of their long-term and diffuse consequences (Delleur, 2003). Brown and Farrelly (2009) describes the traditional or conventional methods of urban water management as “linear” and “old-world”, going on to emphasise that sustainable urban water management approaches need to be more adaptive, participatory and integrated. It is difficult to find a consistent definition of sustainability in literature (Delleur, 2003). However, Loucks (1997 p. 518) propose the following definition: “*Sustainable water systems are designed and managed to fully contribute to the objective of the society, now and in the future, while maintaining their ecological, environmental and hydrological integrity.*”. According to this definition, the shift towards sustainability aims at achieving the objectives of conventional urban water management by providing essential water supply, sanitation and urban drainage services, but to do this in a way that ensures that the integrity of environmental, ecological and socio-economic elements of the city for future generations. This shift from conventional urban water management towards sustainable urban water management (SUWM) can be complex due a city’s specific history, geography, ecology, socio-political climate and the prevailing attitudes towards SUWM concepts (Brown *et al.*, 2008). As a result Brown *et al.* (2008) notes that the transitioning towards SUWM can be slow and advocates that there is a need for a framework to benchmark progress towards SUWM.

2.1.2 Water Sensitive Cities

Brown *et al.*, (2008) proposed the concept of a ‘Water Sensitive City’ (WSC) as a form of benchmarking for urban water management, envisioning a city where water is given due prominence in urban design, management and planning. This framework (Figure 2.1) uses ‘six

transition states' for benchmarking that are informed by the temporal, ideological and technological contexts of a city as it transitions towards a WSC (Brown *et al.*, 2008).

The first three transition states of the WSC: the 'Water Supply City', 'Sewered City' and 'Drained City', describe conventional or historical approaches to urban water management as outlined previously (Section 2.1.1). The remaining three transition states outline various conditions of future urban water management, which are designed to achieve sustainability in the cities of the future. The 'Waterways City' sees a progression from the service delivery role of conventional methods to one that seeks to preserve the ecosystem services of the waterways in urban areas and utilise these areas as open green spaces providing greater public amenity (Brown *et al.*, 2008). A 'Water Cycle City' recognises that water is a limited resource and that conventional means of urban water supply are likely to be unsustainable in the future. Strategies such as water conservation and identifying fit-for-purpose water supply sources are an important means of augmented supply in cities of the future. There are many sources of fit-for-purpose water such as rainwater, stormwater, treated wastewater and seawater, which all have varying quality and can be applied to a range of uses (irrigation, industrial and household). Wong and Brown (2008) also argue that the city itself should be viewed as a supply catchment, whereby a portion of rainfall and stormwater are reused instead of advocating expensive water supply schemes that are typically located outside the city's boundaries. The final transition state, the 'Water Sensitive City', is one that sets a vision of urban water management that is completely integrated, flexible and socially and environmentally sustainable. According to Brown *et al.* (2008, p. 9) the 'Water Sensitive City' incorporates "...environmental repair and protection, supply security, flood control, public health, amenity, liveability and economic sustainability, amongst others".

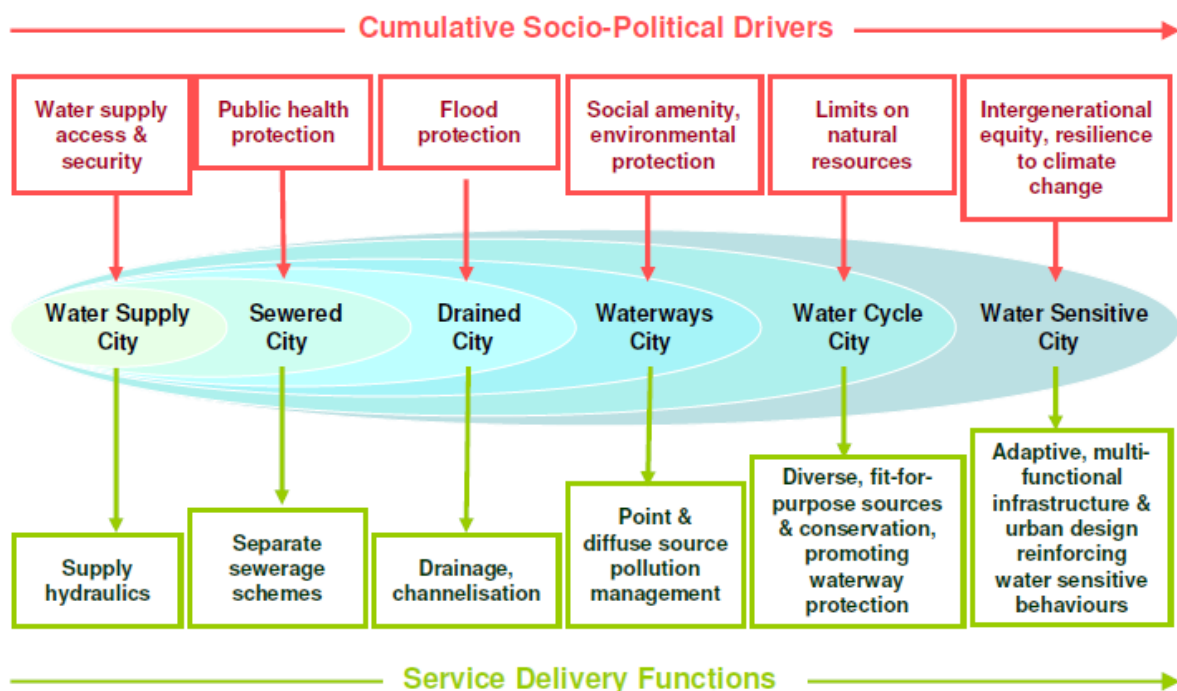


Figure 2.1: The transitions states outlined in the Water Sensitive Cities concept (Brown *et al.*, 2008)

Most South African cities are classified as ‘Drained Cities’ (Armitage *et al.*, 2014). However, due to the turbulent socio-political history of South Africa, Armitage *et al.* (2014) recognised that South African cities are complex, consisting of both well serviced formal areas and underserved informal areas. Thus, the WSC concept was modified to include the complex socio-political history that characterises the conditions in many South African cities and towns (discussed further in Appendix D).

The WSC framework is valuable for helping cities identify the past advancements, the current status, and the future objectives of urban water management. As a city progresses towards becoming a WSC, conventional urban water management strategies become insufficient and alternative urban water management strategies are required. The following section describes and discusses some of the alternative urban water management strategies that have been implemented globally and the progress to-date in South Africa.

2.1.3 Alternative urban water management

While conventional urban water management tends to concentrate on the basic water requirements for human settlement (Mitchell, 2006), there are a range of alternative water management strategies that aim to manage urban water in a more sustainable, integrated, adaptive and participatory manner (Brown and Farrelly, 2009). These include, *inter alia*, Green Infrastructure (GI) (Walmsley, 1995; US EPA, 2016), Total Water Management (TWM) (Rodrigo *et al.*, 2012), Integrated Urban Water Management (IUWM) (Bahri, 2012), Sustainable Urban Water Management (SUWM) (Brown and Farrelly, 2009), Sustainable Drainage Systems (SuDS) (Ellis *et al.*, 2004; Shepherd *et al.*, 2006; Armitage *et al.*, 2013; Fletcher *et al.*, 2014), Low Impact Development (LID) (Department of Environmental Resources, 1999; Dietz, 2007; Roy *et al.*, 2008; Fletcher *et al.*, 2014) and Water Sensitive Urban Design (WSUD) (Whelans *et al.*, 1994; Wong, 2006, 2007; Allen *et al.*, 2008) are terms commonly mentioned in literature.

It is clear that there are a diverse range of theoretical frameworks available, which can lead to confusion when describing the principles and practices of various alternative urban water management approaches (Fletcher *et al.*, 2014). Fletcher *et al.* (2014) tried to define the scope of a broad range of approaches and terminology that were used in the alternative urban water management literature. The study arranged some of the most commonly used terminology into a framework comparing the primary focus or range of application to the specificity of each term (Figure 2.2). The framework compares the main priorities of each management approach to the nature of its objectives, ranging from specific techniques to more conceptual objectives and then broader principles.

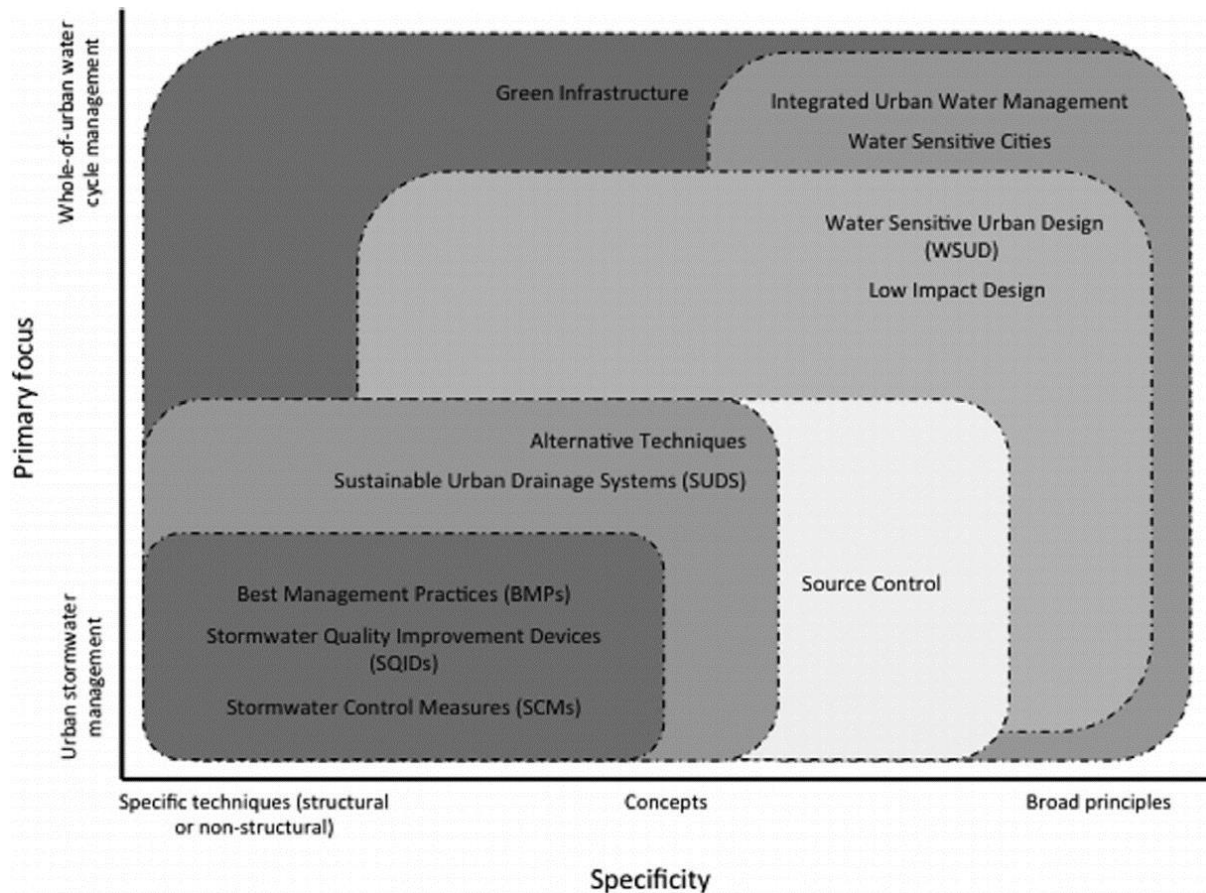


Figure 2.2: The classification of urban water management approaches and terminology based on their specific scope and objectives (Fletcher *et al.*, 2014)

The framework described in Figure 2.2 demonstrates how approaches such as Sustainable Urban Drainage Systems (SUDS), Sustainable Drainage Systems (SuDS) and Water Sensitive Urban Design (WSUD) better incorporate conceptual elements in their approach to urban water management, resulting in a more holistic view of water management within the complete urban water cycle. The broad focus on the complete water cycle of SuDS and WSUD, together with the combination of technical and conceptual interventions mean that these systems are well suited for the inclusion of other water management strategies such as Managed Aquifer Recharge.

SuDS is a term commonly used in the United Kingdom (UK) that became popular in the late 1980's and early 1990's (Fletcher *et al.*, 2014). SuDS is a relatively new concept in South Africa that is becoming increasingly popular, with a recent Water Research Commission (WRC) report published supporting the development and implementation of SuDS practices in South Africa (Armitage *et al.*, 2013). SuDS are commonly used to sustainably manage urban stormwater by reducing the amount and magnitude of peak stormflow discharges while reducing pollutant loads so as to protect the water quality and biodiversity of receiving waters (Water by Design, 2009). Water Sensitive Urban Design (WSUD) and Low Impact Development (LID) are concepts that maintain similar objectives to those of SuDS, but shift practice and thinking away from solely identifying specific techniques and technology, but to understand the conceptual challenges that inform many of the issues facing modern urban water management. LID is a term commonly used in the United States of America (USA) and in New Zealand (Armitage *et al.*, 2014; Fletcher *et al.*,

2014), while WSUD is commonly used in Australia (Wong, 2007). LID began with an emphasis on cities maintaining ‘natural hydrology’ in urban areas by designing and planning the urban landscape in a way that emulated the hydrological responses of the pre-developed system. Similarly, WSUD also aims to reduce the hydrological impact of urban developments; however, WSUD adopts a more philosophical approach through the integration of socio-economic factors which is noted in some of the earliest outlines of the WSUD objectives, for example, Whelans *et al.* (1994). The main framework for this study is WSUD which is discussed further in the following section.

2.2 Water Sensitive Urban Design (WSUD)

2.2.1 History and Background

The concept of Water Sensitive Urban Design (WSUD) became prevalent in Australia in the early 1990’s (Whelans *et al.*, 1994; Wong, 2007). Wong (2007, p. 1) notes that when WSUD was originally conceptualised in Perth, Australia, it “...*was as an alternative planning and design framework for urban development that attempts to break the dependency of urban environments on large water services infrastructure that is not integrated in a manner that manages all water streams as resources, promotes recycling, mitigates the impact of urban stormwater on the urban water environment through the promotion of at-source detention and retention of stormwater using landscaped features*”. It is apparent that from its inception, WSUD has aimed to address the problems of conventional urban water management through more sustainable urban planning and design. As a result, WSUD attempts to provide a more holistic perspective on urban water management that is focused on the sustainability of all urban water resources. WSUD is defined by Engineers Australia (2006 p. 4.1) as “...*an approach to urban planning and design that integrates land and water planning and management into urban design. WSUD is based on the premise that urban development and redevelopment must address the sustainability of water*”. Therefore, WSUD attempts to achieve sustainability in all areas of the urban hydrological cycle through multi-disciplinary engagement that ensures that the environmental, social and economic benefits of water are maximised (Water by Design, 2009). WSUD promotes a shift in emphasis from conventional, centralised stormwater drainage networks that are designed to rapidly remove stormwater, to diversified approach that includes both centralised and decentralised systems. WSUD encourages local or ‘source-control’ devices for stormwater management that involve managing rainfall where it falls by enhancing infiltration and evapotranspiration (Coombes *et al.*, 2000; Ellis, 2000; Wong and Brown, 2008; Wong *et al.*, 2012).

The objectives of WSUD have been outlined a number of times in literature by Wong (2006), the Queensland Department of Infrastructure and Planning (QDIP) (2009) and Whelans *et al.* (1994), however, it is evident that few of these publications mention groundwater, with the exception of Whelans *et al.* (1994). The WSUD objectives according to Whelans *et al.* (1994) provide a holistic approach to WSUD that attempts to consider the full urban water cycle, including groundwater. These key objectives according to Whelans *et al.* (1994) are stated as follows:

- i) To manage the urban water balance
 - a. Maintain appropriate aquifer levels, recharge and streamflow characteristics in accordance with assigned beneficial uses
 - b. Prevent flood damage in developed areas
 - c. Prevent excessive erosion of waterways, slopes and banks

- ii) To maintain and, where possible, enhance water quality
 - a. Minimise waterborne sediment loading
 - b. Protect existing riparian or fringing vegetation
 - c. Minimise the export of pollutants to surface or groundwater
 - d. Minimise the export and impact of pollutants from sewage

- iii) To encourage water conservation
 - a. Minimise the import and use of potable water supply
 - b. Promote the reuse of stormwater
 - c. Promote the reuse and recycling of wastewater
 - d. Reduce irrigation requirements
 - e. Promote regulated self-supply

- iv) To maintain water-related environmental values
- v) To maintain water-related recreational values

These objectives of WSUD advocate a broad scope for urban water resources management by considering the quantity (water balance), quality, and the conservation of water while preserving or enhancing the environmental and recreational value of urban water resources. Therefore, WSUD may provide an alternative means of addressing Cape Town's water insecurity as it attempts to assign a resource value to all forms of water in the urban context, *viz.* stormwater, wastewater, potable water and groundwater (Whelans *et al.*, 1994; Wong, 2006; BMT WBM Pty Ltd, 2009; QDIP, 2009; Water by Design, 2009). This concept proposes a shift away from the conventional view that potable water is the only valuable or usable water source available. A 'fit-for-purpose' water use, requires a lower quality water that can be used for purposes such as urban irrigation, sanitation and industry, thus relieving pressure on potable water supplies (Hancock, 2000).

While the stated objectives of WSUD according to Whelans *et al.* (1994) are arguably the most holistic, they are the only objectives that attempt to explicitly incorporate groundwater. The lack of inclusion of groundwater in the published WSUD literature and more importantly in publications that outline the aims and objectives of this particular approach to urban water

management, suggest that groundwater and the impacts to it, are unrecognised. As a result, urban groundwater is often neglected or given the least attention in the WSUD approach and urban water management in general (Mudd *et al.*, 2004). Foster *et al.* (1998 p. 52) suggest that this is because urban groundwater is “*out of public sight, and therefore out of political mind*”. Thus, because groundwater by nature is a ‘hidden’ resource that is sometimes difficult to conceptualise, the understanding and management of groundwater systems needs to improve through monitoring and evaluation, and enhancing groundwater policy. In the same way, there is a need for the improved representation of groundwater in WSUD that recognises its value as a resource and ensures its protection and conservation.

Furthermore, though most WSUD objectives outline a holistic approach to urban water management and planning, much of the early research in the field concentrated on stormwater management (Wong, 2007). Fletcher *et al.* (2014) further highlight this over-simplification of the WSUD objectives, noting that many practitioners would use the term in reference to a singular technology. However, Wong, (2007) suggests that WSUD has evolved from a narrow emphasis on stormwater management to the broader framework outlined in many of the earlier WSUD objectives. While the objectives of WSUD deal with a range of conceptual issues, the role of stormwater management remains an important component of WSUD.

2.2.2 Stormwater management in WSUD

Stormwater management in WSUD is concerned with two main issues (i) maintaining a flow regime that most closely represents that of the natural hydrology or specified environmental objectives, and (ii) managing the water quality of stormwater so as to protect receiving waters (Fletcher *et al.*, 2014). In order to achieve these objectives, a change from conventional urban drainage systems that utilise impervious surfaces and piped drainage networks to rapidly remove stormwater from urban areas, to drainage systems that attenuate the water and offer some level of treatment to remove contaminants is required.

The concept of attenuation in WSUD and SuDS is often referred to as ‘source control’ (Ellis, 2000; Allen *et al.*, 2008). Source control aims to manage stormwater at the point of rainfall by removing the connections between impervious surfaces that encourage rapid stormwater generation (Ellis, 2000). Devices like roof gardens or green roofs, permeable pavements, vegetated swales, bio-retention systems, constructed wetlands, detention ponds and infiltration basins all aim to reduce the quantity and rate of runoff from urban areas (Allen *et al.*, 2008; Hamel *et al.*, 2013). The attenuation of stormwater provides additional opportunities to enhance the urban environment. First, detention or retention provides an opportunity to collect stormwater or rainwater for reuse for potable or fit-for-purpose applications. This reduces the demand for potable water supply alleviating pressure on existing bulk water supply networks (Wong, 2000). Second, the use of source control devices can improve the ecosystem services of cities. Third, SuDS devices are able to add social and amenity value to an urban area through urban greening (Water by Design, 2009).

There is increasing pressure to protect the aquatic environments situated in urban areas. Wong (2000) notes that in Australia there had been significant improvements made to the quality of point-source pollutants resulting from sewage and industrial effluent. However, the poor quality of non-point source pollution from stormwater remained a contaminant risk. Therefore, SuDS or

source control measures that are implemented for water quality management, typically aim to reduce the contamination of receiving waters or to achieve a level of water quality that enables its use for fit-for-purpose applications (Wong, 2000; Lloyd *et al.*, 2002). Many of the devices used for stormwater attenuation offer varying improvements to stormwater quality by removing contaminants such as sediment, nutrients, heavy metals and litter (see further discussion on stormwater contaminants in section 2.5.3.1) (Water by Design, 2009). Some of the most commonly applied technologies are sand filters (Siriwardene *et al.*, 2007; Kandasamy *et al.*, 2008), bio-retention systems (Kazemi *et al.*, 2011; Lucke and Nichols, 2015) and constructed wetlands (Weiss *et al.*, 2006; Wong, 2006; Shutes *et al.*, 2010). Table 2.1 shows three categories (*viz.* primary, secondary and tertiary) that describe the application of stormwater management devices that are typically used in SuDS or WSUD and the role of each category in removing contaminants. The primary treatment of stormwater deals with pollutants – including litter and coarse sediment – that are largely removed by sedimentation. The secondary stage relies on filtration and sedimentation to remove fine particulates and absorbed contaminants. Vegetated swales and sand filters are commonly used to achieve secondary stormwater treatment. Tertiary stormwater treatment generally consists of an additional treatment process involving biological uptake. Constructed wetlands and bio-retention systems have been shown to effectively remove nutrients and heavy metals from stormwater (Davis *et al.*, 2003; Hunt *et al.*, 2006; Li and Davis, 2008).

Table 2.1: SuDS and WSUD stormwater treatment categories (after Dunstone and Graham, 2005; Water by Design, 2009)

Category	Definition	Pollutant Removal	WSUD Intervention
Primary	Physical screening or rapid sedimentation techniques	Gross pollutants and litter, coarse sediments, free oil and grease	Gross pollutant traps, sediment basins, oil and grit separators
Secondary	Finer particle sedimentation and filtration techniques	Fine particles and attached pollutants	Vegetated swales and Sand filters
Tertiary	Enhanced sedimentation and filtration, biological uptake, absorption onto sediments	Nutrients and heavy metals	Constructed wetlands and bio-retention systems, natural stream systems

Most of the WSUD or SuDS devices mentioned above rely on the process of infiltration to achieve the desired objectives for stormwater quantity and quality. Stormwater infiltration systems, such as bio-retention systems, constructed wetlands, detention ponds and infiltration basins all encourage infiltration to some degree. This can assist in enhancing groundwater recharge to pre-urban levels (Dierkes *et al.*, 2000). Furthermore, increases in infiltration can compensate for soil water deficits, increase baseflows, and reduce the volume and peak discharge of stormwater during storm events, and reduce the stress on current stormwater systems (Alfakih *et al.*, 1999; Water by Design, 2009). Therefore, infiltration incorporated into SuDS, WSUD and MAR (see section 2.5.1) as a means of attenuation and treatment, can be classified as secondary or tertiary interventions for contaminant removal. Yet, most devices including constructed wetlands and bio-retention

systems and infiltration basins, typically used in WSUD and MAR, fulfil a tertiary role in the contaminant removal process, as these devices are generally the final stage of the treatment train before stormwater is released to receiving water bodies or continues to recharge the aquifer.

2.3 Conventional urban groundwater management

Sustainable groundwater development is critical for urban planning and management (Collin and Melloul, 2001; Morris *et al.*, 2001), however, if groundwater management is to be successfully achieved, a strong institutional framework is required which includes the regulation of groundwater. Even if the appropriate institutional framework and legislation are in place, there still needs to be public and political will to manage and protect groundwater. In many instances, groundwater management is not politically attractive as it may only yield benefits in the long-term (Foster *et al.*, 1998). In general, groundwater lacks public, professional and governmental awareness (FAO, 2003). Knüppe (2011) asserts that, in South Africa, groundwater is particularly undervalued due to the absence of groundwater in bulk water supply and the lack of groundwater monitoring and regulation in urban areas. Knüppe (2011) propositions four reasons that explain why groundwater is undervalued. First, there is a stigma in South Africa that groundwater is a ‘poor man’s’ resource that should be limited for use in subsistence agriculture and rural communities. Second, there is a shortage of technical expertise and adequate data for groundwater resources. Third, groundwater governance is highly centralised and lacks the integration of national and local institutions, that may have better potential for managing the resource. Fourthly, there is a general disregard for the role of groundwater in providing ecosystem goods and services.

There is often a lack of foresight regarding the development of groundwater resources. The installation of boreholes, in the initial stages of groundwater development, is typically unplanned and unregulated. This has occurred in the City of Cape Town and in the Cape Flats, where the city has encouraged groundwater use for non-potable water uses such as watering gardens. This is a valuable for relieving pressure on the potable water supply, however, a lack of monitoring means that the impact of this water use is not known. The issue of private groundwater use in Cape Town is discussed further in Appendix B. In developing countries like South Africa there are additional problems. For example, the rapid increases in population in many developing countries often occurs before the required wastewater infrastructure is in place, leading to groundwater degradation. For instance, the use of on-site sanitation practices – which are commonly used in South Africa – can be detrimental to groundwater health (Foster *et al.*, 2010a).

In general, groundwater management in urban areas is concerned with improving or maintaining the appropriate quantity and quality of groundwater at the lowest cost, while preventing irreversible degradation (Todd and Mays, 2005). This can be enforced using specific regulatory codes or through planning and consultation (Foster *et al.*, 2010a, 2010b). The quantity of groundwater is dependent on the levels of recharge and abstraction which need to be carefully monitored (Foster *et al.*, 2010a). In terms of groundwater regulation, the licensing of groundwater users is the main method employed to manage urban groundwater resources. However, this is difficult in an urban context, particularly in a developing country, where most of the abstractions of groundwater are unregulated or illegal (Foster *et al.*, 2010a). The quality of groundwater also requires legislative and monitoring controls that prescribe the disposal of liquid effluents and solid waste, as well as other activities that could pollute groundwater.

2.4 Positioning Groundwater in the WSUD Framework

The academic literature on WSUD tends to focus on surface water themes including stormwater management (i.e. SuDS) and water conservation, while groundwater typically serves a secondary purpose as a means of storage or treatment. As a result, the role of groundwater management within WSUD appears more distant from the holistic and integrated urban water management vision outlined by Whelans *et al.* (1994). Based on the stated WSUD objectives groundwater should be viewed as a valuable resource that, if not managed correctly, could be over-exploited or overlooked completely (Hancock, 2000).

2.4.1 Groundwater in WSUD Best Management Practices (BMPs)

Water by Design (2009) outlines a number of Best Management Practices (BMPs) for WSUD techniques and technologies. BMPs are structural and non-structural elements of WSUD that outline the appropriate actions required for the implementation of WSUD, the limited or prohibited practices and maintenance procedures (Water by Design, 2009; Fletcher *et al.*, 2014). However, there are few BMPs that are focused on urban groundwater management that seeks to efficiently utilise groundwater for abstraction, storage and treatment, while ensuring this is done in a sustainable manner conserving the resource. Groundwater is indirectly considered in a number of the WSUD BMPs in terms of its storage potential or as a constraining physical characteristic (Water by Design, 2009). For example, a groundwater table at or near the ground surface is a constraining physical characteristic that may limit or preclude a number of WSUD practices. As a result, the Water by Design (2009) BMPs specify that an infiltration pond should be at least 1 meter above the seasonal high water table, as the saturated conditions associated can often increase the mobility of pollutants in the soil profile and reduce the efficiency of the infiltration device. Therefore, a number of WSUD techniques and technologies are constrained due to groundwater related risks.

In a review of the groundwater resources in the Australian city of Melbourne, Mudd *et al.* (2004) linked groundwater to WSUD through infrastructure, wetlands and aquifer storage and recovery (ASR). Although these groundwater links were specific to Melbourne, a few generalised roles of groundwater within the WSUD framework can be gleaned from this study *viz.* groundwater interacts with urban infrastructure, groundwater-surface water interactions are important for groundwater dependent ecosystems and groundwater or aquifers are a potential means of water storage.

2.4.2 Groundwater and infrastructure

Urban infrastructure, whether for stormwater, wastewater or water supply, interacts closely with groundwater and this interaction can alter the quantity and quality of groundwater (Morris *et al.*, 2003). Figure 2.3 highlights these interactions by showing the impact of the water supply system, urban engineering and waste disposal on urban groundwater leading to either over-abstraction, excessive recharge or deteriorating groundwater quality. The diagram further highlights the threat of negative feedbacks resulting in secondary problems for each of the components of the urban water system. For example, elevated groundwater levels as a result of leaking sewage infrastructure could infiltrate into the drinking water network and pose a contaminant risk (Figure 2.3).

Urban groundwater functions as both a source of water and a receptor of urban drainage (Hancock, 2000; Morris *et al.*, 2003). Over abstraction of groundwater can cause saline intrusions and land subsidence (Morris *et al.*, 2003), while excessive groundwater recharge can result in structural damage to buildings and flood underground basements and parking lots (Lerner, 2002). Groundwater recharge to urban aquifers has always been believed to be significantly reduced compared to pre-urban conditions. However, groundwater recharge is known to occur as a result of leaking water supply networks and from the over-watering of parks and recreational areas, leading to increases in groundwater levels (Lerner, 2000, 2002; Wolf *et al.*, 2006). Additionally, wastewater from on-site sanitation, leakages from sewerage networks or municipal and industrial wastewater can also cause increased water tables and can be detrimental to urban groundwater quality (Lerner, 2002; Morris *et al.*, 2003; Mudd *et al.*, 2004; Wolf *et al.*, 2006). Therefore, the implementation of WSUD requires a sound understanding of the relationship between groundwater and urban infrastructure to ensure the proper groundwater quantity and quality is achieved and maintained, as dictated by the WSUD objectives emphasised in section 2.2.1.

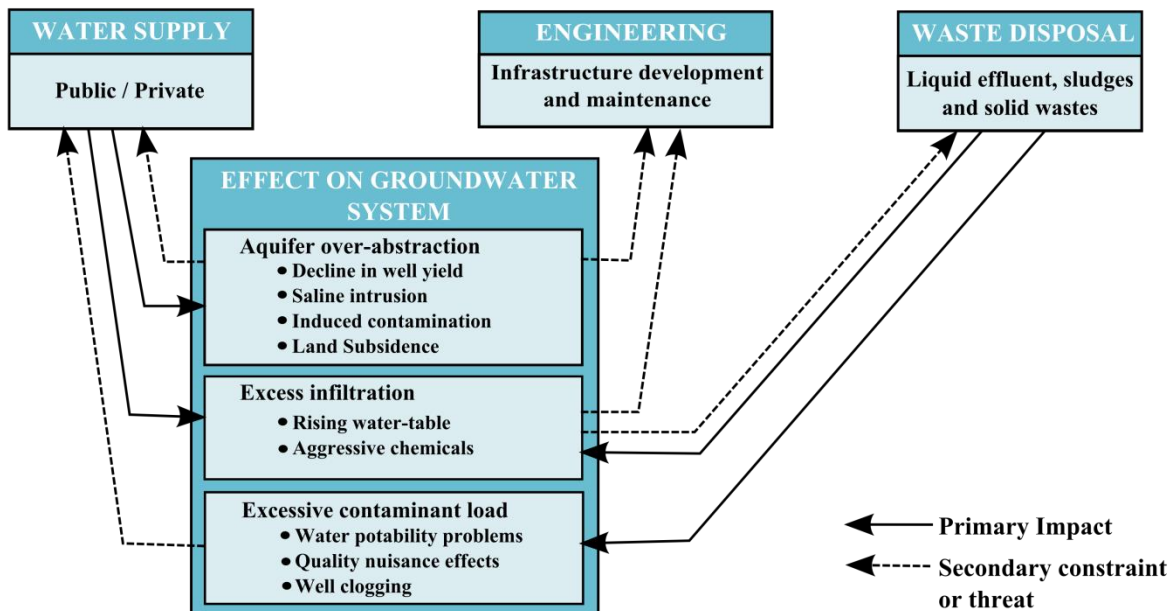


Figure 2.3: The impacts of urbanisation on groundwater degradation (Morris *et al.*, 2003)

2.4.3 Groundwater and surface water interactions

Groundwater can contribute significantly to streamflow in the form of baseflows, which describe the contribution of groundwater discharges to streamflow. Alternatively, streamflow can function as an essential source of groundwater recharge (Todd and Mays, 2005). This is particularly true in areas where the groundwater table is close to the surface, in topography associated with wetland or riverine areas (Winter *et al.*, 1998). Groundwater plays a crucial role in the health of ecosystems in rivers and wetlands, thereby offering valuable ecosystem goods and services, such as water supply, flow regulation, contaminant removal and food supply, along with recreational and aesthetic value (Weight, 2008). Mudd *et al.* (2004) suggest that there is a lack of understanding of the interactions between surface water and groundwater in a number of WSUD technologies. As

an illustration, Mudd *et al.* (2004) refer to an example of constructed wetlands in Melbourne, Australia. Constructed wetlands have become a popular method for attenuating and treating urban stormwater, however the authors noted: “...*there is a distinct lack of understanding of the hydrogeology of such systems*”. The study identified increasing nitrate concentrations in baseflows from a number of local urban wetlands, which were suspected to be a result of groundwater contamination from on-site sanitation or other sources. In Australia, there is no legal requirement to monitor groundwater in the proximity of constructed wetlands, however, Mudd *et al.* (2004) suggest that a minimum of three monitoring wells should be installed depending on the local hydrogeology to provide better hydrogeological information to improve the understanding of the impacts of WSUD strategies. Hence, groundwater is linked to these surface water systems and their interaction should be accounted for in the planning, implementation and operational phases of WSUD interventions to ensure that both groundwater and surface water are adequately managed.

2.4.4 Groundwater for storage

Probably the most established, role of groundwater within WSUD is for storage that is used to manage urban stormwater. The infiltration of stormwater into the ground is encouraged in WSUD (Coombes *et al.*, 2000; Dunphy *et al.*, 2007; Wong *et al.*, 2012) so as to attenuate peak stormflows from urban areas and augment groundwater recharge to its pre-urban level. Groundwater recharge increases the amount of water stored in an aquifer and the process of infiltration offers a means of treating recharged stormwater by filtering it as it passes through the unsaturated zone as it infiltrates or as the water moves through the saturated zone within the aquifer (Water by Design, 2009; Wong *et al.*, 2012).

Yet, the urban water cycle is complex and an attempt to return to ‘pre-urban levels’ can be misguided. Göbel *et al.* (2004) raised this concern in a study aiming to develop ‘near-natural’ stormwater infrastructure in the North Rhine-Westphalia region in Germany, suggesting that in areas where the soil surface is sealed by urban land use and infiltration is encouraged using SuDS devices, that the artificial recharge could exceed the natural groundwater recharge. This was due to limited evaporation losses that lead to above natural groundwater levels. The planning of sustainable stormwater infrastructure like that used in SuDS or WSUD needs to properly understand the local water balance and planning if infiltration devices need to extend beyond the infiltration component and examine the groundwater recharge elements (Göbel *et al.*, 2004).

Therefore, the role of aquifers for the storage and potential reuse of stormwater in WSUD is secondary to the primary objective of attenuating and treating urban stormflow. This can be seen in the WSUD literature, where there is strong emphasis on the stormwater management that often aims to enhance the natural processes of infiltration and evaporation of rainwater in urban areas, thereby minimising stormwater generation (Roy *et al.*, 2008; Water by Design, 2009; Wong *et al.*, 2012). It is the argument of this study that the secondary benefits of increased groundwater storage and potential reuse opportunities as a result of stormwater management using infiltration need to be better incorporated into WSUD so that both the stormwater management and storage aspects of WSUD are optimised. The role of storage in WSUD will be discussed further in section 2.5 looking at the implementation of Managed Aquifer Recharge (MAR).

2.4.5 Summary: groundwater in the WSUD

Armitage *et al.* (2014) summarises these WSUD considerations, outlining the role of groundwater within the WSUD framework (Figure 2.4). The schematic (Figure 2.4) shows the three main sources of urban water, *viz.* potable water (light blue), stormwater (dark blue) and wastewater (light brown), and how they interact with groundwater. Groundwater is a valuable means of water supply to urban areas; alternatively, aquifers can function as a means of storage for stormwater or treated wastewater, which can be reused at a later stage. The recharge of the aquifer can be achieved in two ways; by allowing the stormwater or treated wastewater to naturally infiltrate into the ground from infiltration devices, or by the subsurface injection of water into an aquifer for storage and later reuse (Figure 2.4). The quality of stormwater can vary and may contain contaminants, for instance, heavy metals, nutrients, salts and microorganisms. Before wastewater can be recharged via infiltration or subsurface injection, it must be treated at a wastewater treatment plant (WWTP) to the required standard. Infiltration devices are able to remove a number of potentially hazardous contaminants that could find their way into groundwater (as discussed further in section 2.5.3.1). Many urban ecosystems are dependent of groundwater contributions or may contribute to groundwater recharge like rivers or wetlands (green arrows). Leakages from urban infrastructure are a concern, as this can result in uncontrolled increases in water level and groundwater contamination. The areas that are at risk of contamination (marked with an 'X') include leakages from sewerage networks, the infiltration or subsurface injection of stormwater or wastewater, as well as the potential for polluted surface water that may contaminate groundwater, and *vice versa* (Figure 2.4).

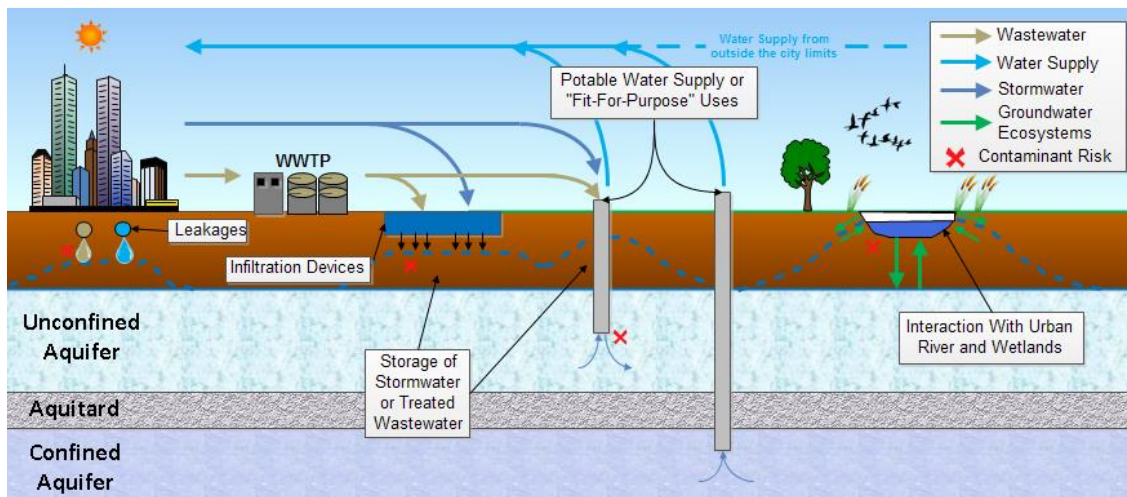


Figure 2.4: Summary of the considerations for groundwater in WSUD. The diagram highlights the main linkages of WSUD to groundwater, through infrastructure, ecosystems and storage (Armitage *et al.*, 2014)

Table 2.2 reiterates the interactions highlighted in Figure 2.4, showing the various WSUD-groundwater interactions, the risks associated with these interactions and the WSUD management responses required to address these risks. The WSUD responses outlined in Table 2.2 show that WSUD interventions extend beyond specific techniques (*i.e.* stormwater BMPs) but are also concerned with management and planning objectives. In each of the areas noted for interaction

with WSUD—infrastructure, groundwater dependent ecosystems and groundwater for storage—the potential contaminant risk at these sites can be mitigated through the WSUD policies that advocated pollution prevention, protection or rehabilitation of urban ecology and biodiversity. Furthermore, there is an important emphasis on continued maintenance to ensure that conventional and WSUD infrastructure meet minimum performance requirements. Monitoring is an important component that is often overlooked in WSUD, especially with regard to groundwater, as contamination can easily go undetected and the sources of contamination can be difficult to identify.

Table 2.2: The groundwater risks and WSUD groundwater management responses in terms of the various WSUD-Groundwater interactions (after Armitage *et al.*, 2014)

WSUD-Groundwater Interactions	Groundwater Risks	WSUD Groundwater Management Response
Infrastructure	Pipe Leakage – excessive recharge	<ul style="list-style-type: none"> • Preventing leakages from underground water pipelines (Potable, Stormwater and wastewater) • Urban land use planning • Installation of monitoring systems
	Pipe Leakage – groundwater contamination	
	Groundwater ingress into underground infrastructure	
	Contamination from urban land uses	
Groundwater dependent ecosystems	Polluted surface water ecosystems contaminating groundwater	<ul style="list-style-type: none"> • Preventing contamination of Groundwater and Surface water • Protect groundwater related ecosystem services • Groundwater rehabilitation • Monitoring
	Polluted groundwater Surface water ecosystems contaminating surface water ecosystems.	
	Loss of ecosystem goods and services	
Groundwater for Storage	Risk of contamination (Infiltration devices, ASR/ASTR)	<ul style="list-style-type: none"> • Careful planning, testing, design and assessment by suitably qualified personnel • On-going monitoring • Development of management plans
	Excessive increases in Groundwater levels	
	Compromised soil and aquifer structure	

2.5 Managed Aquifer Recharge

2.5.1 Definitions and Types of Managed Aquifer Recharge

Aquifers offer a valuable source of potential storage and reuse of stormwater and treated wastewater (Water by Design, 2009; Wong *et al.*, 2012). This process of intentionally enhancing the recharge of an aquifer is known as Management of Aquifer Recharge or Managed Aquifer Recharge (MAR) (Dillon *et al.*, 2009). Dillon (2005) describes MAR as the “...*intentional banking and treatment of water in aquifers*”. Recharged water can be stored in both confined and unconfined aquifers (Figure 2.5). In the case of a confined aquifer, generally, a borehole is required to penetrate the confining aquitard to directly inject and abstract water from the aquifer (Figure 2.5a). Unconfined aquifers are simpler as there is no impeding layer or aquitard so recharge can occur from the ground surface, thus infiltration and direct injection of water into the aquifer are available options (Figure 2.5b).

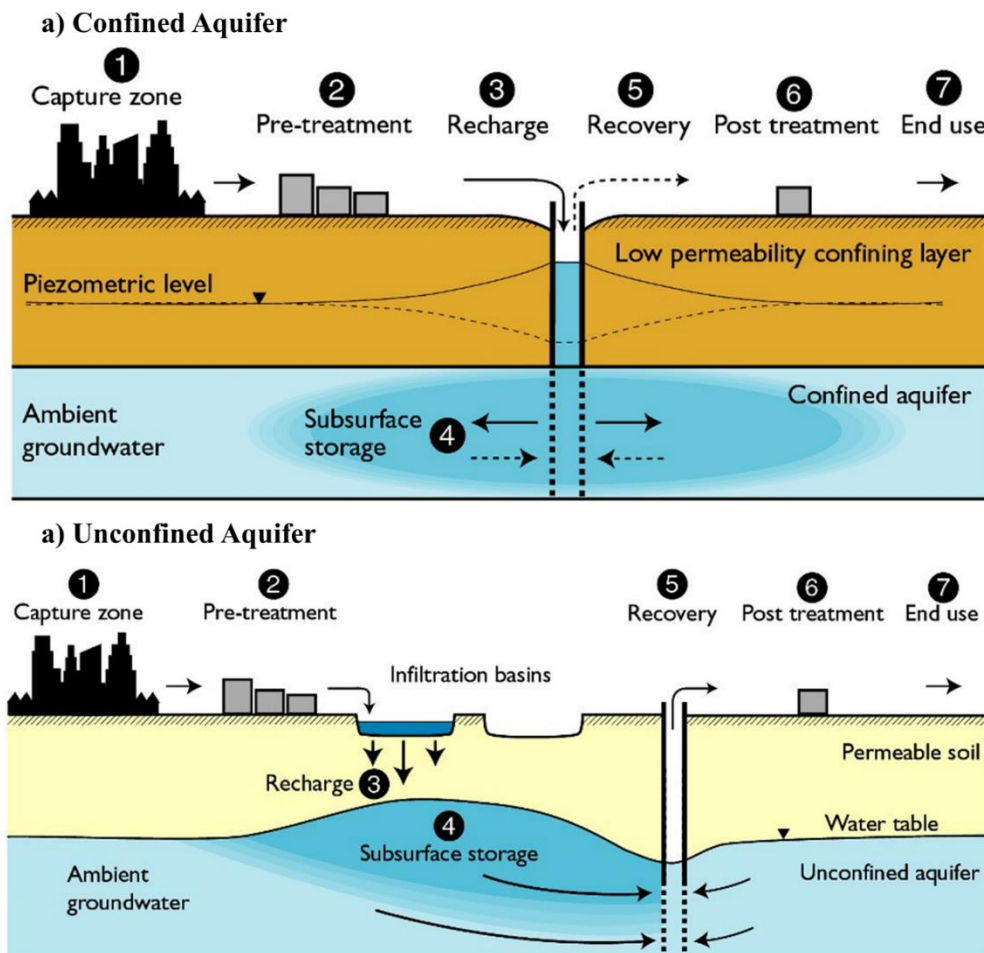


Figure 2.5: MAR applications in Confined (a) and Unconfined (b) Aquifers (after Dillon *et al.*, 2009)

There are a number of terms that are commonly referred to in literature with the same meaning as MAR. In South Africa a commonly used term is Artificial Recharge (AR) (Tredoux *et*

al., 1980, 2011a; Murray *et al.*, 2007), however the term has become unpopular due to the negative connotation attached to ‘artificial’ (Dillon, 2005). Aquifer Storage and Recovery (ASR) and Aquifer Storage Transfer and Recovery (ASTR) are also common terms that are used in literature (Melbourne Water, 2005; Wendelborn *et al.*, 2005; Wendelborn, 2008; Barry *et al.*, 2010; Steinel, 2012). However, Dillon (2005) acknowledges that these terms carry specific meanings and should be classified as part of a suite of options available under the broader term MAR (Figure 2.6). For example, ASR describes a situation where water is injected into an aquifer and abstracted out of the same well. ASTR is where water is injected into an aquifer and abstracted at another well, increasing the residence time of the groundwater and as a result provides additional treatment (Dillon, 2005). These techniques rely on the active injection of water through the borehole to recharge the aquifer, whereas other methods such as infiltration ponds, bank infiltration and underground dams all rely on infiltration to passively recharge water into the ground.

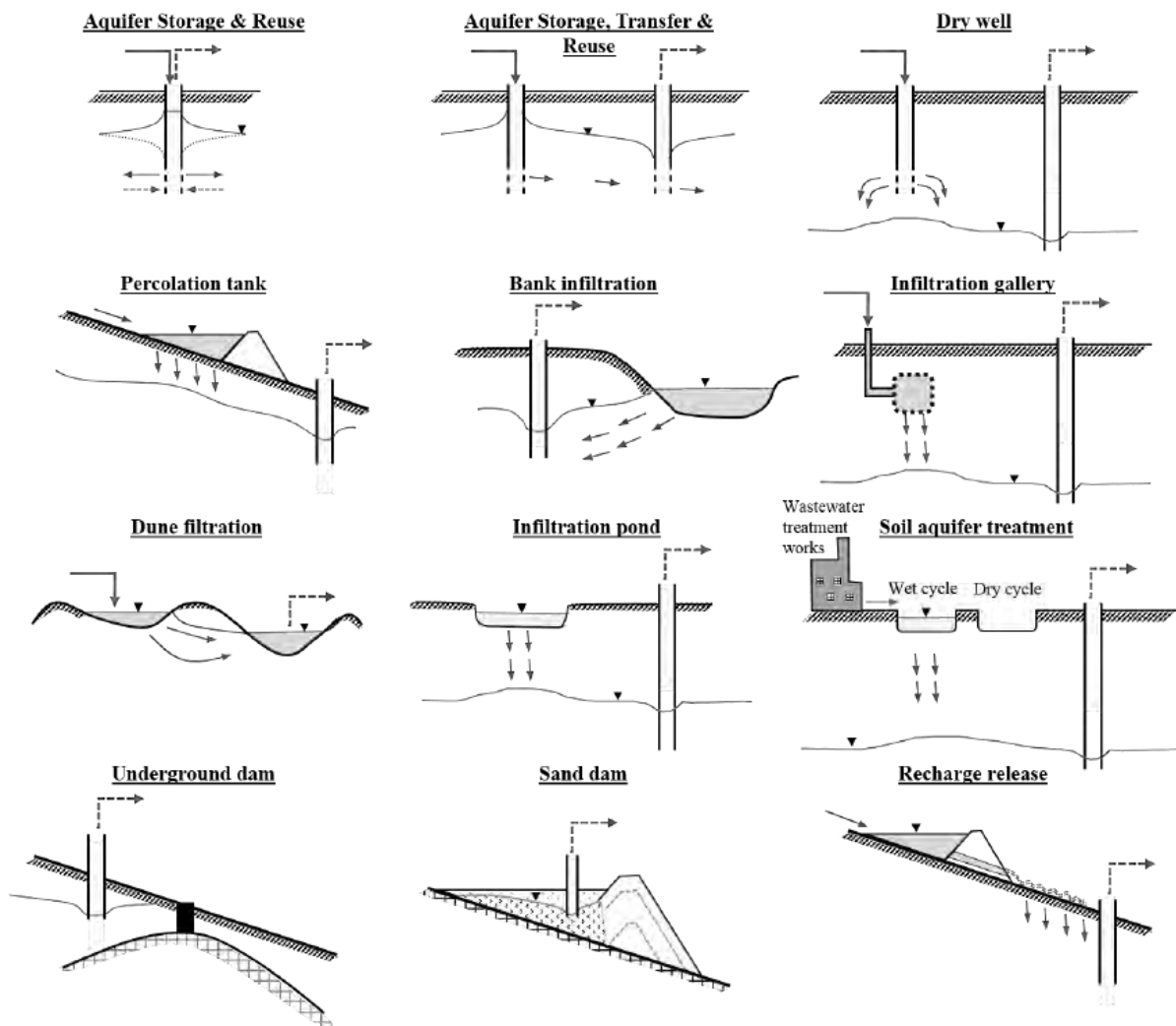


Figure 2.6: The various MAR technique available (Dillon, 2005; Fisher-Jeffes, 2015)

2.5.2 MAR Objectives

The objectives of MAR are site specific and are dependent on the desired outcomes of the MAR application (i.e. water supply, flood prevention etc.). The aquifer characteristics such as the geology, aquifer storativity and hydraulic conductivity are the most significant limiting factors, determining how much water can be stored in the aquifer and how easily it can be recharged and abstracted (Murray *et al.*, 2007). Moreover, there may be risks associated with MAR under certain geological conditions, for example, the formation of sinkholes in dolomitic aquifers or the risk of aquifer collapse after the aquifer has been dewatered following the recovery phase of MAR.

One of the most frequent reasons for the implementation of MAR schemes around the world is the ability to sustain or enhance water supply from an aquifer for urban and agricultural purposes (Figure 2.7). By supplementing aquifer recharge with water from additional water sources such as surface water from dams and rivers, groundwater from adjacent aquifers, recycled water from urban stormwater or treated wastewater, it is possible to enhance the assurance of supply obtainable from an aquifer. The application of MAR to sustain or enhance water supply from groundwater has proven successful in a number of countries such as, *inter alia*, the United States of America (Petrides *et al.*, 2015; Scanlon *et al.*, 2016), India (Shah, 2014), the Netherlands (Stuyfzand *et al.*, 2002) and Australia (Wendelborn *et al.*, 2005; Lennon *et al.*, 2014; Clark *et al.*, 2015). In addition to international experience, there are a number of good examples of the application of MAR in Southern Africa (Murray *et al.*, 2007). Most notable is the MAR scheme using the Atlantis aquifer in the Western Cape, which has been the primary water supply to the town of Atlantis for over 30 years (Murray *et al.*, 2007). In aquifers where the abstraction rate exceeds the natural recharge rate, MAR can assist in maintaining or restoring natural groundwater fluxes that are important for sustaining groundwater dependent ecosystems like wetlands and rivers. Recharged water can also be used to protect or improve the water quality of the groundwater resource (Figure 2.7).

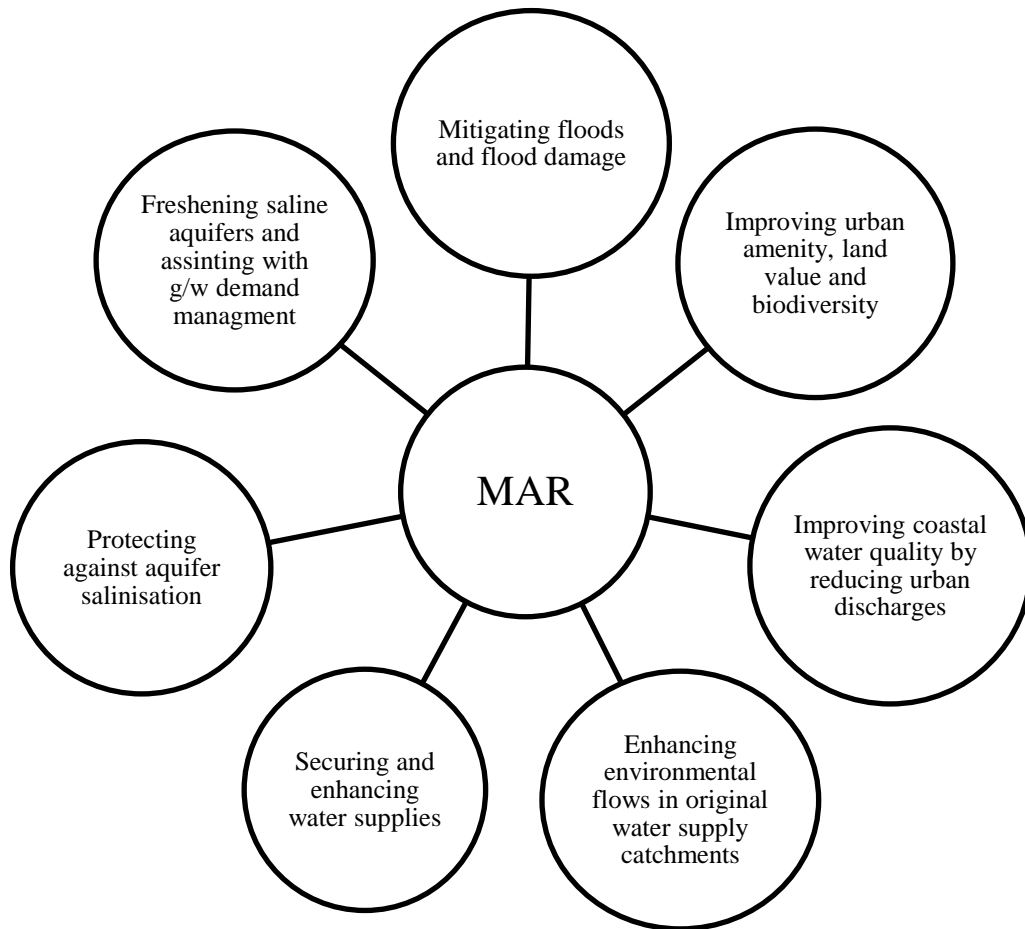


Figure 2.7: The main objectives of MAR (after Dillon *et al.*, 2009)

Dillon *et al.* (2009) list a number of benefits of MAR techniques (Figure 2.7), these include a range of themes such as optimising yield, improving water quality, resource protection and ensuring ecological and environmental sustainability. Furthermore, Dillon *et al.* (2009) suggest that MAR could improve local amenity, land value and biodiversity through MAR's focus on sustainability and resource protection. There is a close correlation that can be seen between the objective of MAR as outlined by Dillon *et al.* (2009) to those of WSUD as described by Whelans *et al.* (1994). Both MAR and WSUD share a common vision of sustainability that recognises the resource value of all urban water resources, but encourages the efficient utilization of these resources. There is a consistent emphasis on protection and conservation in both MAR and WSUD that recognises the innate value of urban water resources and their ability to provide ecosystem services, public amenity and sustain local biodiversity. Despite the common vision, groundwater and MAR remain secondary elements of WSUD with substantial emphasis on the management of surface water. Hence, the incorporation of MAR into WSUD could provide a means of addressing the current lack of inclusion of groundwater management in WSUD, while supplementing urban water supply.

2.5.3 MAR Risks

There are a number of risks associated with the subsurface injection or infiltration of stormwater. These include groundwater pollution (Alfakih *et al.*, 1999; Pitt *et al.*, 1999; Clark and Pitt, 2007; Weiss *et al.*, 2008), increasing groundwater temperature (Foulquier *et al.*, 2009), soil collapse or land subsidence, increases in local water table levels and the altering of natural groundwater flows (Ellis, 2000; Vázquez-Suñé *et al.*, 2004).

2.5.3.1 Groundwater contamination

The application of MAR using stormwater for groundwater recharge is common. Melbourne Water (2005) notes that increasingly stormwater is being used for direct injection schemes like ASR and ASTR. Furthermore, the use of infiltration devices for stormwater management are a commonly adopted method of groundwater recharge (Lloyd *et al.*, 2002; Dechesne *et al.*, 2004; Water by Design, 2009; Wong *et al.*, 2012). Stormwater can contain a wide variety of contaminants, including sediment, nutrients, toxic chemicals, micro-organisms and dissolved minerals. These contaminants come from a range of sources *viz.* vehicles, industry, lawn care products, organic waste and eroded sediments (Ellis, 2000; Clark and Pitt, 2007). However, stormwater infiltration systems can significantly reduce the risk of groundwater contamination (Burton and Pitt, 2002).

The presence of nutrients, such as phosphorous and nitrogen, in urban stormwater, are of concern because of their propensity to induce algal blooms and eutrophication in receiving water bodies (Weiss *et al.*, 2008). Sources of nutrients include motor oil, animal and plant material and fertilisers. The presence of nutrients in groundwater is not always as a result of anthropogenic impacts of stormwater or wastewater, but can be associated with natural soil and geological conditions (Pitt *et al.*, 1999). Nutrients can be removed during infiltration, through precipitation or chemical adsorption onto the surfaces of soil particles through chemical reactions with Iron, Calcium or Aluminium, however the effectiveness of nutrient removal is inconsistent (Weiss *et al.*, 2008). Nitrogen contamination of groundwater is more common than phosphorous, the potential for nitrate leaching from an infiltration device is high, due to the elevated rates of infiltration and the high solubility of nitrates, making it highly mobile through the soil profile (Pitt *et al.*, 1999; Burton and Pitt, 2002). Significant nitrate leaching is known to occur during cool, wet seasons, as denitrification and plant uptake are slowed due to the cool conditions. The risk of nitrate contamination of groundwater from stormwater is generally low to moderate due to the typically low concentrations of nitrate in urban stormwater runoff (Pitt *et al.*, 1999; Burton and Pitt, 2002). However, this is not the case in less developed countries. In many of the large cities of sub-Saharan Africa, poor sanitation infrastructure can result in high nutrient levels accelerating in the eutrophication of receiving waters (Nyenje *et al.*, 2009).

Heavy metals such as chromium, copper, lead nickel and zinc are often present in stormwater, resulting from vehicles and industrial processes (Barraud *et al.*, 1999; Pitt *et al.*, 1999; Dechesne *et al.*, 2004; Weiss *et al.*, 2008). Most of these metals are associated with sediment particulates and can be removed by the sedimentation of metal ions in the infiltrated water which is often adsorbed by soil particles in the vadose zone (Burton and Pitt, 2002; Weiss *et al.*, 2008). In sandy or loamy soil, copper, iron and zinc have been known to show higher mobility, and nickel and zinc have shown a high potential to contaminate groundwater if subsurface injection is used (Burton and Pitt, 2002). In general, the concentration of heavy metals decreases with depth in an

infiltration basin (Mikkelsen *et al.*, 1997; Barraud *et al.*, 1999; Datry *et al.*, 2004; Birch *et al.*, 2006). Dechesne *et al.* (2004) suggest that soil with alkaline pH conditions improves the retention of heavy metals, and Burton and Pitt (2002) suggests that infiltration devices should be kept moist as the drying of the soil allows the adsorption bonds between the sediment and the metals to be weakened, therefore allowing further percolation of heavy metals.

Organic compounds are also found in stormwater. Organic compounds may originate from natural sources, such as decaying animal and plant matter or from anthropogenic sources, such as petroleum hydrocarbons, tyre residue and the exhaust emissions from vehicles (Pitt *et al.*, 1999). Organic compounds, such as phthalate esters and phenolic compounds are most commonly found in groundwater but can include benzene, chloroform, methylene chloride, trichloroethylene, tetrachloroethylene, toluene and xylene. Polycyclic aromatic hydrocarbons (PAHs), such as benzo(a)anthracene, chrysene, anthracene and benzo(b)fluoroanthene have also been detected in groundwater near industrial areas (Burton and Pitt, 2002). Organic compounds can be removed from water in a soil profile in three ways: volatilisation, sorption or degradation. Volatilisation can reduce the concentrations of the most volatile compounds, but this becomes less effective with increasing soil moisture. Sorption onto soil organic matter can limit the mobility of less soluble compounds, however the sorption does not remove compounds and resolubilisation of organic compounds may occur during wet periods leading to groundwater contamination. Many organic compounds can be removed through microbial degradation; however, this is a function of temperature, pH, moisture content, and the ion-exchange capacity of the soil. The use of surface infiltration methods should reduce the concentrations of organic compounds; however, pre-treatment is recommended (Barraud *et al.*, 1999; Pitt *et al.*, 1999; Weiss *et al.*, 2008). Ellis (2000) suggests that stormwater containing high concentrations of organic compounds should not be directly injected into groundwater using ASR or ASTR due to their increased mobility under saturated conditions.

Pesticides that are used to control insects and plants can also be found in stormwater and can contaminate groundwater. The risk of pesticide contamination depends on the concentration of the pesticides in stormwater and the mobility of the pesticide. Most pesticides decompose in soil and water over time, but this can vary, taking between a few days to a few years to occur. Surface infiltration with pre-treatment could substantially reduce the risk of groundwater contamination; however, subsurface injection or infiltration of stormwater with high concentrations of pesticides should be avoided (Burton and Pitt, 2002; Weiss *et al.*, 2008).

Pathogens present in stormwater have the potential to contaminate groundwater. Viruses and bacteria are of concern as they can occur in high concentrations in urban stormwater and pass through the soil relatively easily (Weiss *et al.*, 2008). Pathogens can be removed from water through straining at the soil surface or adsorption to soil particles. However, viruses and bacteria can have lengthy survival times in soil of up to 5 years (Burton and Pitt, 2002). This means that they can be recollected by percolating water and transferred to groundwater. Stormwater with high concentrations of pathogens can result in the development of bacterial biofilms. Biofilms can cause clogging in boreholes and the aquifer surrounding a borehole, decreasing the storage potential and hydraulic conductivity of the aquifer, making recharge and abstraction more difficult (EPHC, 2009).

Salts are particularly problematic in stormwater, as soil offers little attenuation. Water that contains salt after it has passed through the vadose zone will contaminate groundwater. Sulphate and potassium concentrations have been known to decrease with depth; however, sodium, calcium, bicarbonate and chloride concentrations are all known to increase with depth. Salt is particularly problematic in the northern hemisphere where it is used to de-ice roads (Burton and Pitt, 2002).

A summary of the most common compounds present in stormwater is shown in Table 2.3. The table describes the potential of each of the compounds to contaminate groundwater, making reference to the mobility, abundance in stormwater and filtration potential. The table shows that the groundwater contamination potential for a number of contaminants is reduced when pre-treatment is used before surface infiltration. The table also indicates that the sub-surface injection of stormwater increases the risk of groundwater contamination. Of particular concern are enteroviruses and chloride contaminants that exhibit high potential for groundwater contamination for all of the recharge methods.

Table 2.3: The various pollutants found in stormwater and their potential to contaminate groundwater (after Pitt *et al.*, 1999)

Compounds		Mobility	Abundance in storm-water	Fraction filterable	Contamination potential: surface infiltration, no pre-treatment	Contamination potential: surface infiltration with sedimentation	Contamination potential: sub-surface injection
Nutrients	Nitrates	Mobile	Low/moderate	High	Low/moderate	Low/moderate	Low/moderate
Pesticides	2,4-D	Mobile	Low	Likely low	Low	Low	Low
	γ -BHC (lindane)	Intermediate	Moderate	Likely low	Moderate	Low	Moderate
	Malathion	Mobile	Low	Likely low	Low	Low	Low
	Atrazine	Mobile	Low	Likely low	Low	Low	Low
	Chlordane	Intermediate	Moderate	Very low	Moderate	Low	Moderate
	Diazinon	Mobile	Low	Likely low	Low	Low	Low
Other organics	VOCs	Mobile	Low	Very high	Low	Low	Low
	1,3-dichloro-benzene	Low	High	High	Low	Low	High
	Anthracene	Intermediate	Low	Moderate	Low	Low	Low
	Benzo(a)anthracene	Intermediate	Moderate	Very low	Moderate	Low	Moderate
	Bis (2-ethylhexyl) phthalate	Intermediate	Moderate	Likely low	Moderate	Low	Moderate
	Butyl benzyl phthalate	Low	Low/moderate	Moderate	Low	Low	Low/moderate
	Fluoranthene	Intermediate	High	High	Moderate	Moderate	High
	Fluorene	Intermediate	Low	Likely low	Low	Low	Low
	Naphthalene	Intermediate	Low	Moderate	Low	Low	Low
	Penta-chlorophenol	Intermediate	Moderate	Likely low	Moderate	Low	Moderate
	Phenanthrene	Intermediate	Moderate	Very low	Moderate	Low	Moderate
Pyrene	Intermediate	High	High	Moderate	Moderate	High	
Pathogens	Enteroviruses	Mobile	Likely present	High	High	High	High
	Shigella	Intermediate	Likely present	Moderate	Low/moderate	Low/moderate	High

	Pseudomonas aeruginosa	Intermediate	Very high	Moderate	Low/moderate	Low/moderate	High
	Protozoa	Intermediate	Likely present	Moderate	Low/moderate	Low/moderate	High
Heavy metals	Nickel	Low	High	Low	Low	Low	High
	Cadmium	Low	Low	Moderate	Low	Low	Low
	Chromium	Intermediate	Moderate	Very low	Low/moderate	Low	Moderate
	Lead	Very low	Moderate	Very low	Low	Low	Moderate
	Zinc	Low	High	High	Low	Low	High
Salts	Chloride	Mobile	Seasonally high	High	High	High	High

2.5.3.2 Additional MAR risks

The appropriate conditions need to be in place for MAR to be successfully implemented, such as the availability of aquifers and their storage capacity. Other areas of concern are the clogging of boreholes, the mobilisation and build-up of contaminants, and loss of permeability due to changes in biogeochemistry (Hancock, 2000; Mudd *et al.*, 2004). Therefore, before implementing an MAR scheme, it is imperative that all the potential impacts are thoroughly researched and understood (Mudd *et al.*, 2000).

An aquifer being considered for MAR needs to be protected to ensure the quality of water is maintained or improved (Dillon *et al.*, 2009). If the quality of recharged water is not appropriate this can lead to the long-term degradation of the resource potential of the aquifer. Poor water quality can also result in the clogging of the aquifer, negatively impacting the storage and hydraulic properties of the aquifer. Contaminants in poor water quality can also react with the rock or porous media of that aquifer which can lead to further degradation of groundwater. For example, Tredoux *et al.* (2009) suggested that the recharge of groundwater with a high dissolved oxygen content could alter the redox conditions of the quartzite Windhoek Aquifer resulting in chemical reaction with the rock leading to deteriorating water quality.

When implementing MAR, the appropriate structural or physical considerations of the local hydrogeology need to be observed. If the appropriate water levels or pressures are not maintained within the aquifer there can be negative impacts on the aquifer and surrounding ecosystems. If groundwater levels are not maintained and fall too low, there are higher risks of land subsidence resulting in the permanent loss of storage potential. At MAR sites located close to the sea, there are higher risks of seawater intrusion (Page *et al.*, 2012). In confined aquifers, the appropriate pressures are required to ensure the aquifer and aquitard structures remain intact. Reduction in pressure can also adversely affect local groundwater users (Water by Design, 2009; Page *et al.*, 2013).

2.5.4 MAR Costs

Aquifer storage and recovery (ASR) and aquifer storage, transfer and recovery (ASTR) are becoming increasingly attractive methods for the storage and treatment of water. The successful application of ASR and ASTR in Australia and the United States of America (USA) over the past 40 years has increased public acceptance of recycling stormwater through MAR related activities (Dillon *et al.*, 2009). Maliva (2014) notes that investment in MAR technology is often hindered by a lack of economic cost-benefit analysis. Maliva (2014 p. 1257) provides a detailed review of the appropriate procedures for evaluating the cost and benefits of MAR, by “...*direct and indirect measures of willingness to pay including market price, alternative cost, value marginal product, damage cost avoided, and contingent value methods.*” Maliva (2014) makes the point that the end water use for a MAR scheme should determine whether the costs are feasible or not. For example, a MAR scheme to supply water for a fit-for-purpose water use such as irrigation should not have high construction and operational costs. It should also be acknowledged that determining the value of water, especially water used for MAR is very challenging. Water has a priceless intrinsic value as it is essential for life, however due to its relative abundance in human civilisation the value of water is relatively low. Yet in times of shortage water can become a priceless commodity. In addition to a highly variable

monetary value, water used for MAR has an indirect or ‘hidden’ value. For example, when MAR is compared to other forms of water supply, MAR is a relatively flexible resource that can be made available on short notice for emergency demand. Therefore, MAR can play a valuable stabilising role for ensuring the assurance of water supply, potentially offsetting the economic costs of water storages. MAR adds the value of ground water supplies by increasing the storage potential of an aquifer, meaning more water can be obtained from the aquifer for longer. Increased aquifer storage can also delay the costs of required infrastructure upgrades, as a higher water table would negate the need for new or deeper boreholes to access groundwater. Additionally, there is an *in situ* value of merely having water present in the aquifer for environmental flows, avoidance of land subsidence and prevention of seawater intrusion. An additional benefit of MAR is the treatment process offered by the aquifer, removing contaminants as the water passes through the porous media (Maliva, 2014). Therefore, it is clear that there are no simple methods for evaluating the feasibility of MAR, as this often is a complex interaction of direct and indirect costs and benefits which are often site-specific.

The simplest means of determining the feasibility of MAR is to compare it to other water supply alternatives based on the unit cost of water from each source (Maliva, 2014). Australia has a number of ASR and ASTR applications that have been shown to be capable of producing water of drinking quality standards and is considered an economically attractive alternative to other treatment methods such as reverse osmosis (Dillon *et al.*, 2009). Dillon *et al.* (2009 p. 8) states that “*If 200 GL of the Water Services Association of Australia projected 800 GL shortfall in water in Australian cities by 2030 were met from stormwater ASR the cost savings in comparison with seawater desalination would be \$400 million per year in addition to significant environmental benefits*”. Dillon *et al.* (2009) compared the costs of various supply and demand management options for Adelaide, Perth, Newcastle and Sydney (Figure 2.8). Figure 2.8 outlines the supply options for all of these cities and their relative range of cost and those indicated by a star are those where MAR could be actively applied. MAR has the greatest cost saving potential if applied for reused strategies using stormwater and indirect potable water reuse showing costs ranges of \$0.10 – \$1.50 and \$1.68 – \$2.61, respectively. Non-potable water recycling is more expensive than stormwater reuse, thus is less economically attractive. However, this option may still be cheaper than surface water schemes that require long distance pipelines. It is also worth noting the groundwater development is already relatively inexpensive, thus the implementation of MAR may improve the yield and assurance of supply that could be obtained from such groundwater schemes.

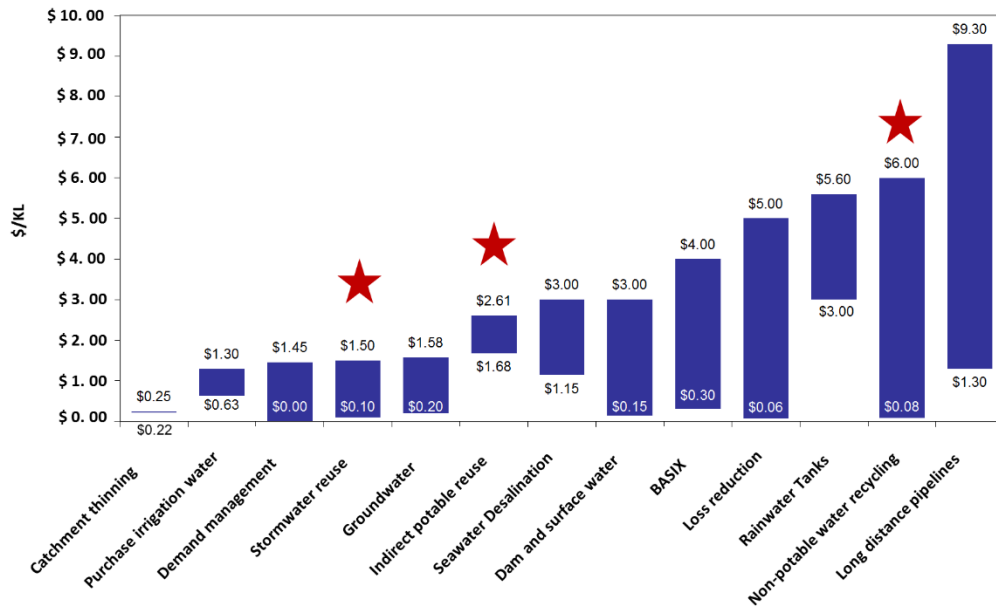


Figure 2.8: Comparison of cost of water supply enhancements and demand reduction measures (Dillon *et al.*, 2009)

2.5.5 MAR Applications

2.5.5.1 Australia

Australia has a long history of MAR applications, dating back to the mid-1960's. In the initial years, most of these MAR operations were located in rural areas and slowly started being applied in urban areas (Dillon *et al.*, 2009). Many of the existing MAR applications are located in the major cities of Perth, Adelaide and Melbourne, which have significant aquifer storage available for MAR. Perth has the highest available MAR capacity at approximately 250 Mm³ due to its unconsolidated sandy aquifer. Melbourne has approximately 100 Mm³ of MAR storage capacity in a fractured sandstone aquifer and in alluvial gravel deposits. There is a further MAR storage capacity in Adelaide of 20 – 80 Mm³ in confined limestone aquifers and fractured quartzite bedrock (Dillon *et al.*, 2009).

Burdekin Delta is Australia's oldest and largest MAR scheme and is mainly used for the irrigation of Sugar Cane and other crops. Up to 45 Mm³ of water is recharged into the coarse sandy aquifer using sand dams on the Burdekin River. Water is slowly released from upstream storages into the sand dams via natural and constructed channels. Turbidity is of concern in this scheme as clogging of the sand dams can become an issue, thus river water with the lowest turbidity is used for recharge to reduce the regularity with which the sand dams need to be cleaned. Typically, the sand dams need to be cleaned every two years by scraping away the top layer of the basin floor in order to restore the initial recharge rates.

Many of Australia's examples of urban MAR began in the late 1990's and early 2000's. A good example of this is in Salisbury near Adelaide, where ASR and ASTR systems have been developed making use of a formerly brackish aquifer for the recycling of stormwater. The stormwater, harvested from residential and industrial catchments, is treated in a reed bed wetland, after which the stormwater is injected into the underlying confined limestone aquifer, which is between 160 and 220 meters below ground (Dillon *et al.*, 2009; Page *et al.*, 2009; Miotliński *et al.*,

2014). The operation was designed to blend higher quality stormwater with brackish groundwater in the aquifer to levels that do not limit its productive use and to provide enough residence time in the aquifer to remove pathogens and organics to meet the Australian drinking water guidelines (Pavelic *et al.*, 2004). Around 0.4 – 0.6 Mm³ of stormwater is recharged to the aquifer through ASR near the Parafield wetlands, and ASTR to the South West of the wetlands. The ASR scheme is mainly used to regulate the pressure in the aquifer reducing excessively high water pressures that may result in the overflow of local wells or the damage of poorly constructed wells. Conversely, if the pressure is too low it may adversely affect the water availability of other local groundwater users. In terms of the ASTR operation, four outer injection wells (IW1, IW2, IW3 and IW4) are used to recharge the aquifer with stormwater and two inner abstraction wells (RW1 and RW2) are used to recover the recharged water. Currently, the recovered water is used as a fit-for-purpose water supply by blending the recovered water with reclaimed saline water from Mason Lakes (Figure 2.9). The blended product is suitable for non-potable uses such as garden irrigation and toilet flushing. Further investigation is underway to assess the feasibility of obtaining potable water from this ASR scheme.

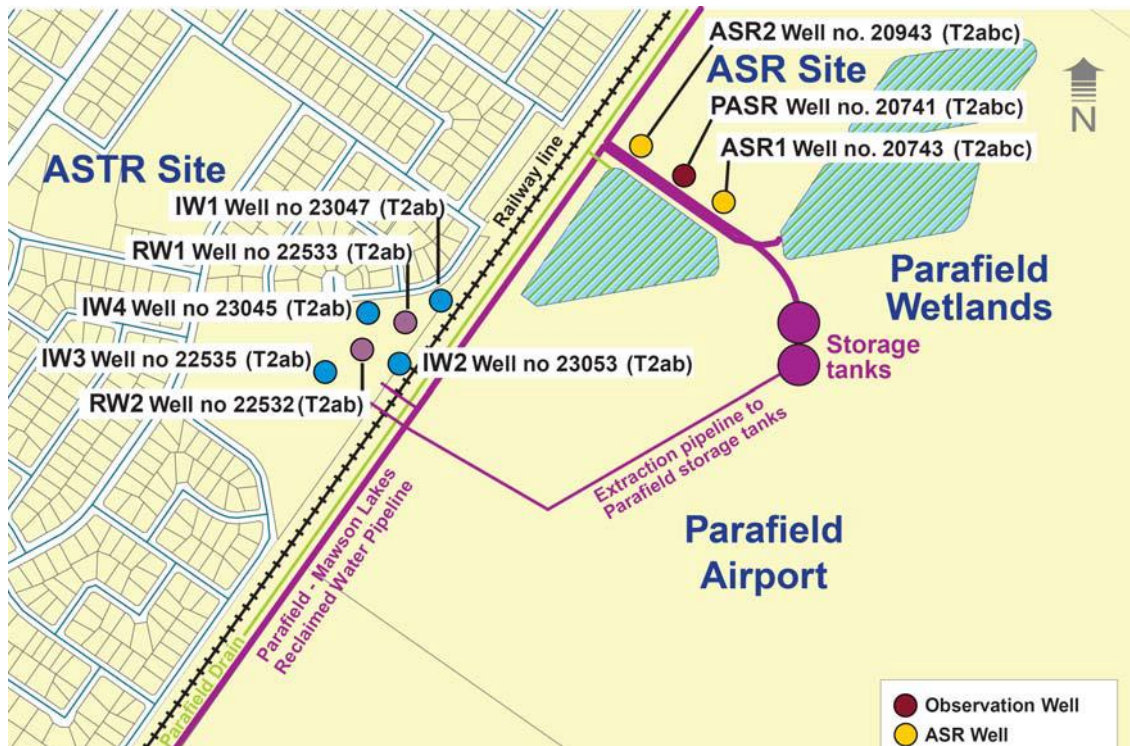


Figure 2.9: The Salisbury ASTR and ASR schemes (Dillon *et al.*, 2009)

2.5.5.2 The United States of America

Like Australia, the United States of America (USA) has a relatively long history of MAR, showing particular success with ASR schemes such as the Peace River ASR scheme in Florida and the Kerrville ASR scheme in Texas (Murray *et al.*, 2007). The Peace River ASR scheme is an example of a large-scale ASR scheme that diverts water from the Peace River into a surface reservoir. The water was treated to potable standards and injected into the underlying limestone aquifer. In 2005 a combination of 21 boreholes produced a recovery capacity of approximately 68 000 m³.day⁻¹.

The main aim of the Peace River system was to capture water from the peace river during peak flow and store the excess water in the aquifer, thereby reducing the required abstraction from the Peace River during low flow periods. This system has helped to defer and perhaps eliminate the need to develop further surface water infrastructure to meet the regional water demands, costing less than half that of the available alternatives (Murray *et al.*, 2007).

The Kerrville ASR scheme in Texas utilises a sandstone and conglomerate aquifer to store water for seasonal and long-term recovery. Like the Peace River scheme the water is injected at potable water quality into the aquifer. Currently, a recovery capacity of $9500 \text{ m}^3 \cdot \text{day}^{-1}$ is possible. The current storage volume is 5.7 Mm^3 to meet water supply requirements during dry periods and to meet the projected 2040 water demand (Murray *et al.*, 2007).

2.5.5.3 Windhoek, Namibia

Windhoek, the capital city of Namibia, is a semi-arid city. The water resources of Windhoek are limited and as a result, much of the water is sourced from dams many kilometres from the city. The main local water resource is the Windhoek aquifer found in the Auas Formation. This aquifer has a sustainable yield of approximately 20 Mm^3 per year, however the storage of the aquifer began to deplete after periods of drought. In response, the city has looked to MAR to restore the aquifer storage and enhanced the groundwater yield (Murray *et al.*, 2007; Tredoux *et al.*, 2009).

As of the year 2007, the volume of recharged water was approximately 2 Mm^3 per year and is planned to be expanded to 8 Mm^3 per year in the future (Murray *et al.*, 2007). The source for the artificial recharge is from the Von Bach Dam and from reclaimed treated wastewater. A number of pilot studies have been performed in the aquifer to test the MAR potential and have confirmed the aquifers storage and hydraulic properties. In addition, the pilot studies allowed for the suitable cost-benefit analysis to be carried out for the proposed MAR scheme. It was found that the MAR scheme outperformed alternatives such as long-range water transfers from groundwater and surface water sources further afield. The main purpose of this scheme is for the long-term storage of water for drought periods as an emergency water supply while preventing evaporative losses (Murray *et al.*, 2007; Tredoux *et al.*, 2009).

2.5.5.4 Atlantis, South Africa

The town of Atlantis in the Western Cape Province in South Africa is the location of one of the countries longest running and most successful MAR schemes, known as the Atlantis Water Resources Management Scheme (AWRMS). Atlantis is located on the west coast within the City of Cape Town metropolitan area, with a population of approximately 67 000 people. In 1976, Atlantis was declared a national growth point that would encourage industrial development on the outskirts of the city. This area has limited water resources as it is classified as semi-arid. The closest major water source was the berg river, located some 70 km away. However, in the initial stages of the development of the town, the Department of Water Affairs and Forestry started to develop the groundwater resources in the area. This was done to supplement water obtained from the small perennial Silwerstroom spring and to postpone the need for a major pipeline from the berg river, which was the expected permanent water source for the town (Tredoux *et al.*, 2011b). The AWRMS was first initiated as an alternative to the conventional marine discharge of wastewater. So in 1979 Atlantis started to recharge stormwater and treated wastewater into the sandy soils in the area.

Recognising that the long-term supply of the Atlantis aquifer was unsustainable, it was decided that there should be a shift away from wastewater disposal to water recycling. As a result, industrial wastewater was separated from domestic wastewater and stormwater so that the best quality water could be recycled using the aquifer (Tredoux and Cain, 2010).

The AWRMS is effectively an Aquifer Storage, Transfer and Recovery (ASTR) scheme. Treated domestic effluent and stormwater are diverted to two infiltration basins (Ponds 7 and 12) which recharge the aquifer up-gradient of the Witzand wellfield which is used for urban and industrial water supply (Figure 2.10). Wastewater and stormwater from the industrial 'noxious trade areas' was diverted to coastal recharge basins down-gradient of the Witzand wellfield, elevating the water table at the coast thereby protecting the aquifer from seawater intrusion (Figure 2.10). In addition to the artificial recharge elements of the AWRMS, the scheme is critical for stormwater management. Ponds 1 – 4, 8 and 11 were designed to be mostly dry and mainly serve to attenuated peak stormflow discharges.

As a result of the implementation of ASTR in Atlantis, it is estimated that approximately $7500 \text{ m}^3 \cdot \text{day}^{-1}$, equivalent to 2.8 Mm^3 per year is recharged to augment the water groundwater supply to Atlantis. A further 4000 m^3 of industrial wastewater was diverted to the coastal recharge ponds preventing significant losses to the ocean or the intrusion of seawater into the main production aquifer (Tredoux and Cain, 2010).

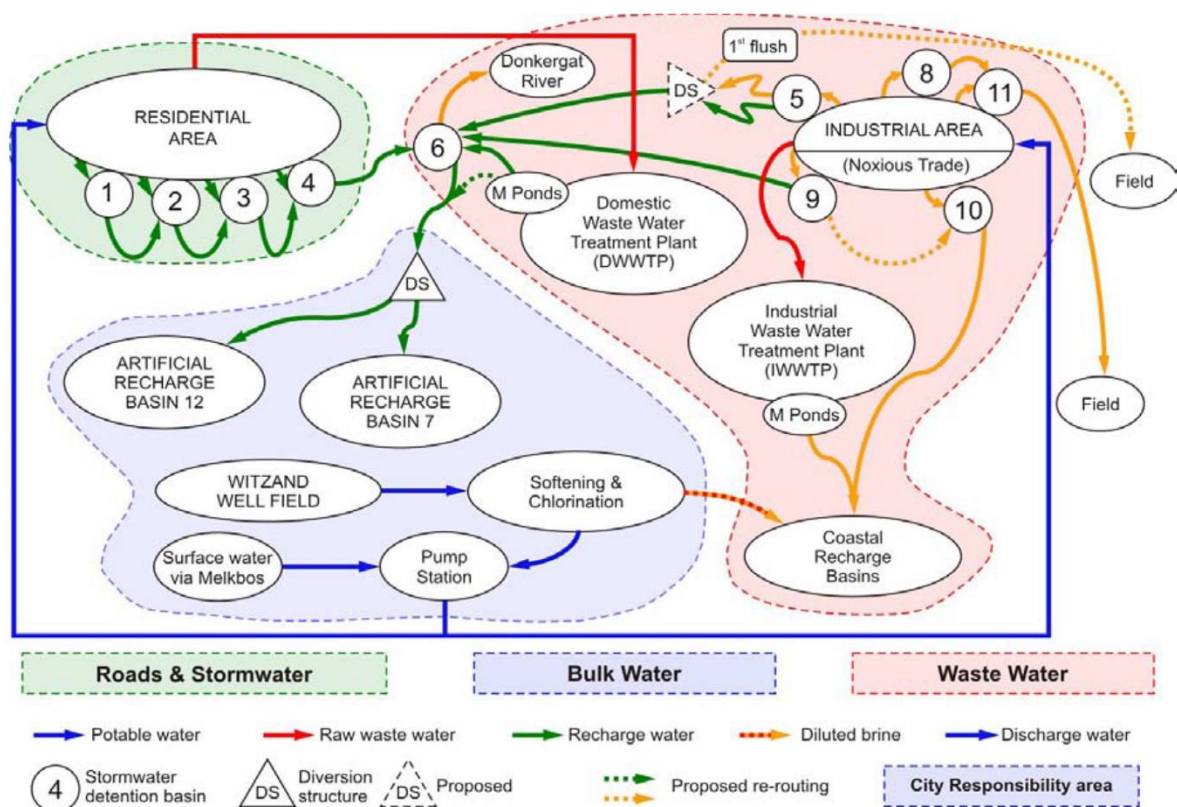


Figure 2.10: Schematic of the MAR elements of the Atlantis Water Resources Management Scheme (Tredoux and Cain, 2010)

2.6 Summary and research Gaps

WSUD is concerned with managing the quantity and quality of urban water and encouraging the protection of the environmental and aesthetic value of urban water resources. WSUD attempts to move away from the conventional attitude that wastewater and stormwater are ‘nuisance’ by-products of the urban environment that require rapid removal and disposal, but rather are valuable resources that could be reused. WSUD also encourages the reduction in water demand in a city through demand management strategies, while attempting to increase the supply capacity through the integration of alternative supply sources (i.e. groundwater or desalination) or reducing the stress on bulk water supply using ‘Fit-For-Purpose’ strategies. However, the review of the WSUD literature indicated that there are gaps relating to the inclusion of groundwater within the WSUD approach. The role of groundwater in WSUD literature is largely secondary to the primary emphasis on stormwater management. A limited number of studies have attempted to comprehensively incorporate groundwater into WSUD. Groundwater considerations in WSUD are generally limited to being considered purely as temporary storage for infiltrated stormwater and not as an integrated resource for water supply, storage and reuse that requires protection and monitoring. Hancock (2000) noted that in the Australian context, historically, urban groundwater has been over-exploited or, on the other hand, completely overlooked as a resource. Knüppe (2011) reiterated this sentiment for the South African context, asserting that groundwater is particularly undervalued in South Africa. Thus, groundwater management in general, and especially in urban areas in South Africa, is shown to be ineffective or poorly prioritised.

The MAR objectives, on the other hand, have a strong focus on sustainable groundwater utilisation, where groundwater and aquifers are of primary importance. Aligning the objectives of WSUD with those of MAR present a number of parallels. WSUD and MAR both seek to optimise the utilisation of urban water, while preserving its resource value. Both seek to increase the amenity value of urban water and maximise benefits such as flood prevention or mitigation and improving the yield of water supplies. Many of the WSUD stormwater management practices are directly or indirectly associated with various forms of MAR, fulfilling a number of WSUD objectives, such as stormwater management, stormwater or wastewater reuse and reducing demand for potable water by providing alternative sources of water for potable or ‘fit-for-purpose’ water supply. While MAR does attempt to facilitate groundwater management and protection there is a strong bias towards groundwater utilisation with an emphasis on storage and reuse for water supply. Therefore, based on the literature, the concepts of WSUD and MAR seem disconnected.

This disconnect highlights an opportunity for better integration of the concepts WSUD and MAR, where the broader scope of the WSUD aims and objectives can be better implemented through the specific techniques already established within MAR. The aim of encouraging the integration of WSUD and MAR, cannot solely focus on groundwater use but should facilitate adaptive thinking around groundwater and aquifers as a resource in urban areas. A resource that is not disconnected from the water cycle, but is intimately linked to ecological processes and may potentially have a number of benefits, such as meeting supply requirements or as a form of safe storage. This integrated approach calls on municipalities and other water institutions, to attach value to groundwater and to begin to monitor, evaluate and understand groundwater. This would assist in the realisation of the risks and opportunities associated with urban groundwater use and the costs of not protecting urban groundwater.

This study aims to demonstrate the concept of improving the integration of WSUD and MAR using the CFA in Cape Town. This study explores the resource value of the CFA and options for its utilisation, while identifying how MAR can improve the implementation of WSUD in Cape Town. Furthermore, based on the similar vision and philosophy of MAR to that of WSUD, the increased incorporation of MAR into WSUD to improve the consideration of groundwater in urban water management is essential for assisting South African cities progress towards becoming Water Sensitive Cities.

3. Research Methods

The following chapter outlines the research methods of this study, in particular, the application of the MIKE SHE and MIKE 11 models at a regional-scale and MIKE SHE at a local-scale, including the sourcing of input data, model parameterisation and the development of a conceptual hydrogeological model. This chapter explains the calibration and validation procedures that were used to ensure that the conceptual assumptions pertaining to the hydrology and hydrogeology were reasonable. The site selection process is outlined, in which the geographical areas of the CFA that were suitable for the local-scale testing of MAR strategies using MIKE SHE, were identified. The scenarios used to test various MAR strategies on the CFA are then discussed. The chapter concludes with the methods for simulating the transport of groundwater contaminants and the potential impacts of climate change.

3.1 Hydrological Modelling

A model is a simplified representation of a real-world system (van Waveren *et al.*, 1999; Wainwright and Mulligan, 2013; Devi *et al.*, 2015). The aim of this simplification is to represent the processes of the real-world system in the simplest manner possible while attempting to attain the closest agreement between the outputs of the simulation and real-world observations. Therefore, modelling complex hydrological systems is valuable for the purpose of simplifying the system in order to test its sensitivity and investigate its response to changes in the environment or management (Schulze, 2004). In this study, the term '*model*' refers to a computer program or software package, many of which are commercially available. The following section outlines some of the main modelling concepts, procedures and models available.

3.1.1 Modelling concepts

The calibration and validation of hydrological or hydrogeological models is an important part of the modelling process. Calibration involves incrementally adjusting the modelling parameters so as to produce the "best-fit" between simulated and observed measurements (Barnett *et al.*, 2012). The purpose of the model validation is to test the calibrated model under different input conditions to ensure the conceptual model is a suitable representation of the actual system (Hassan, 2004). The validation is performed by comparing the simulation to observed data, not included in the calibration, to test and improve the model's performance (Friedel, 2006).

Models are used for a number of reasons that include the planning, design, operation, management or protection of a physical system (Schulze, 1995a). Developing models improves the understanding and prediction of current and future behaviour of a particular system. The information and understanding obtained from the creation of these models can then be applied to test different hypotheses and 'cause and effect' scenarios (Schulze, 2004). In the areas of hydrology, hydrological models provide an inexpensive and practical means of determining the sensitivity of a catchment's hydrological responses to different management and design conditions. Models also play a crucial role in decision support, allowing a number of management and planning decisions to be tested before implementation. Branson *et al.* (1981) suggest that models provide a means of transferring knowledge from areas where measurements or information is available to areas where information is needed so that appropriate decisions can be made. This has significant benefits in

terms of environment and social protection, reducing costs and improving the success of projects (Schulze, 2004). Models are also used for obtaining useful information from poor quality data or in some cases can be used to fill in or replace missing records in a dataset. The application of hydrological models can help to identify future fields of study or be used to identify areas where future monitoring programmes should be prioritised (Schulze, 1995a). Although there are an array of benefits of the use of models, Schulze (1995a) issues a few cautions for the application of hydrological models. First, models cannot substitute for a lack of knowledge, as they cannot create new information or data. Models can only use the information and data available to them to improve the understanding of the system and its behaviour. Second, models are imperfect representations of real world systems, hence Schulze (1995a p. 1.12) notes that models are “...*a means to an end, not an end in themselves*”. This means that models are dynamic, they can be adapted for different purposes, tasks and objectives and they can be made more complex or more simple depending on the aims and objectives of the modeller. Therefore, the application of a model is only as good as the assumptions and interpretation of the modeller.

3.1.2 Modelling urban water

There are a plethora of models available to simulate a wide range of urban hydrological and hydraulic processes. Armitage *et al.* (2014), after a comprehensive literature review, identified 98 IUWM/WSUD models of which 63 were still in use. An earlier study by Mitchell *et al.* (2007) identified 65 models within the IUWM field. Previous studies by Zoppou (1999) and Elliott and Trowsdale (2007) provided comprehensive reviews of the stormwater models available that are suitable to assess the feasibility of SuDS and WSUD. These studies reviewed a number of model characteristics, for instance, the model’s structure, capabilities, limitations and cost.

Bach *et al.* (2014) provide a critical review of a broader spectrum of models used in urban management. The authors suggest that urban water management has become increasingly more integrated, focussing on the combined management of all the components of urban water management (i.e. water treatment, distribution, sewerage and stormwater drainage). This shift towards integration has also been noted in the modelling of urban water, where it has become necessary to integrate numerous detailed models of isolated components of the urban water system in order to understand the whole system (Bach *et al.*, 2014). Bach *et al.* (2014) proceeds to develop a framework for the classification of the different levels of integration of urban water models (Figure 3.1) based on the model’s “...*(i) physical and institutional system delineation, (ii) model complexity, and (iii) differences between their development philosophies...*”. There are four main classification groups within this framework as outlined by Bach *et al.* (2014) are:

1. Integrated Component-based Models (ICBMs) is the simplest form of integration where a few components of an urban subsystem are integrated. (i.e. linking the models to represent the wastewater treatment train);
2. Integrated Urban Drainage Models (IUDMs) or Integrated Water Supply Models (IWSMs) are concerned with the integration of subsystems of urban drainage and water supply, particularly addressing treatment and conveyance processes;
3. Integrated Urban Water Cycle Models (IUWCMs) that links IUDMs and IWSMs into a common framework incorporating the complete urban water cycle;

4. Integrated Urban Water System Models (IUWSMs) is an extension of IUWCMs to include the integration of different urban infrastructure (physical and institutional) and disciplines (i.e. climate change and economics) (Bach *et al.*, 2014).

This framework not only helps to categorise the available urban water models but provides a valuable theoretical framework for guiding the selection and implementation of urban water models that adopt a more holistic view of urban water management and its aims and objectives (Bach *et al.*, 2014). The models that are most regularly used in the urban areas would be classified as IUDMs, IWSMs or IUWCMs, many of which have been extensively reviewed in the literature (Zoppou, 1999; Elliott and Trowsdale, 2007; Armitage *et al.*, 2014; Bach *et al.*, 2014). Some of the most common models in WSUD literature include: SWMM (Delleur, 2003; Fletcher *et al.*, 2013; Fisher-Jeffes, 2015), MUSIC (Wong *et al.*, 2002; Burns *et al.*, 2012; Imteaz *et al.*, 2013), MOUSE (Zoppou, 1999; Roldin *et al.*, 2012), MIKE URBAN (Siekmann *et al.*, 2010; Roldin *et al.*, 2012) and SLAMM (Burton and Pitt, 2002; Mitchell *et al.*, 2007), which are all stormwater models. Other models such as UVQ (Mitchell and Diaper, 2005), Urban Developer (eWater, 2016) and MUSIC are common urban water cycle models or IUWCMs. These models represent a significant advancement in their ability to represent the hydrological and hydraulic processes of urban water (Armitage *et al.*, 2014). Bach *et al.* (2014) go on to suggest that there are a few examples of IUWSMs in literature and that integration at this level is difficult to achieve.

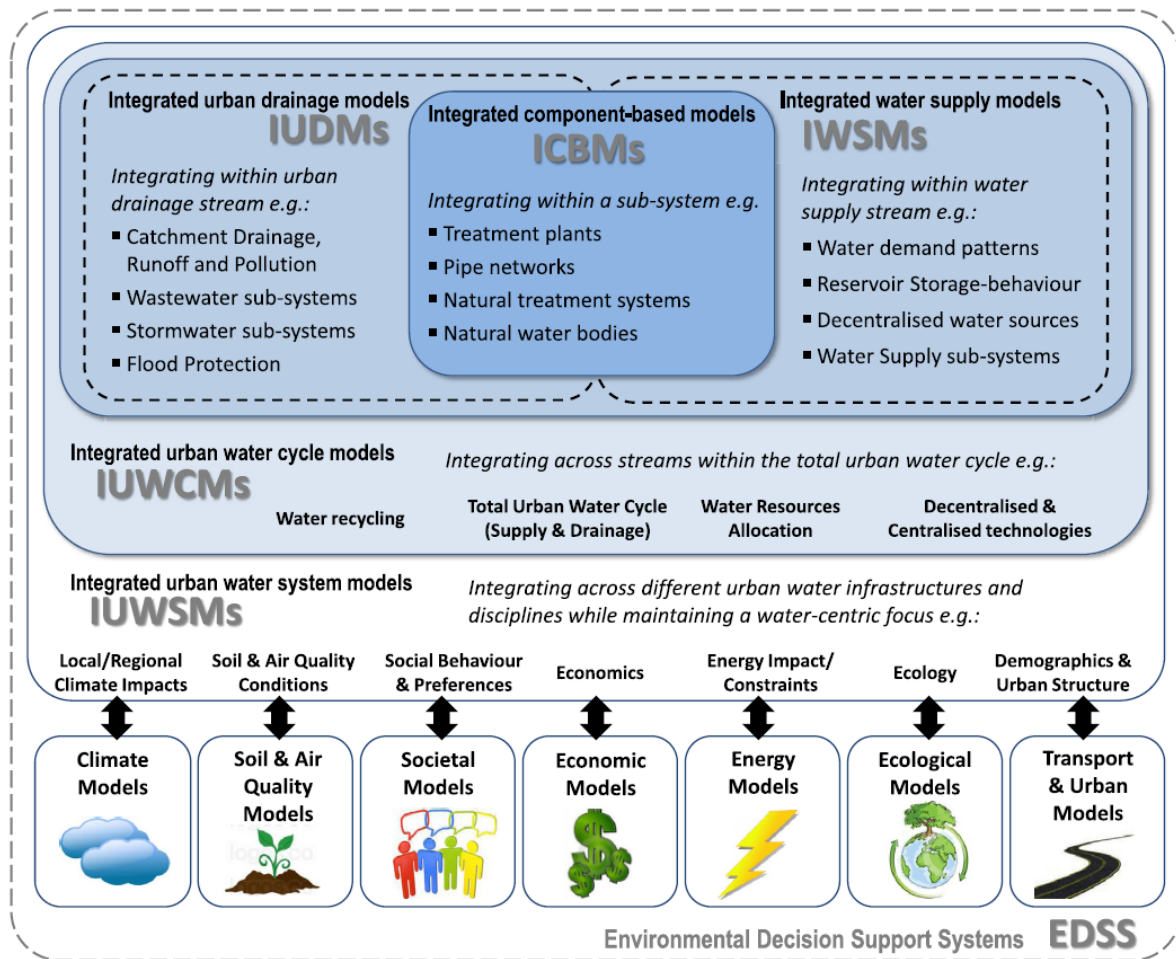


Figure 3.1: A framework for the classification of urban water models, within the context of the environmental design making (Bach *et al.*, 2014)

Despite the wide range of established urban water models, many lack the ability to represent the groundwater processes of urban areas, that are required for this study. The urban hydrological cycle, including groundwater, is influenced by a number of factors such as variations in land use, soil and geology. Additionally, urban areas have complex water supply systems, that typically import water supplies from outside the city's catchment area, combined with complex networks of stormwater and wastewater drainage networks for the removal and disposal of—what is considered to be—waste products. Thus, in order to model urban groundwater, both the surface water processes which control groundwater recharge and the groundwater processes that in turn influence surface water need to be understood. Thus, a modelling procedure that represents both surface and subsurface hydrological processes is required.

3.1.3 Modelling urban groundwater systems

In order to gain meaningful insight into groundwater processes, the collection of groundwater information is essential. Information such as groundwater levels and the characterisation of soil, geological and aquifer properties using pump tests are important for gaining an understanding of groundwater behaviour. This information is however, often limited to point measurements and

transect profiles and does not account for the variability between points of measurement. Thus, physically-based models are able to utilise the available data and information to inform the representation of the aquifer system within a numerical model able to simulate groundwater processes. Vázquez-Suñé *et al.* (2006) submit that groundwater modelling is necessary for the following reasons:

- Combining old and new information for aquifer characterisation and status,
- Improving the understanding of the groundwater mass balance,
- Assessing groundwater related environmental impacts,
- Testing groundwater-related plans and policies, and
- Assisting in the design and development of groundwater rehabilitation measures.

Hydrological modelling in urban areas is complex, as it is dependent on both surface and subsurface hydrological processes. For example, surface water, urban infrastructure, soil characteristics and pipe leakages all determine the amount and rate of groundwater recharge, flow and storage. A number of previous studies carry out the simulation of urban groundwater by linking standalone models that represent the individual components – surface and subsurface – of the urban hydrological cycle. This method has been used to develop decision support systems (DSSs) as shown in studies by Wolf *et al.* (2006), Droubi *et al.* (2008) and Kalbacher *et al.* (2012). Wolf *et al.*, (2006), as part of a project entitled “Assessing and Improving Sustainability of Urban Water Resource Systems” (AISUWRS), linked a number of models to account for the fluxes of water and contaminants between the surface and subsurface. The AISUWRS DSS incorporates an urban water balance model, Urban Volume and Quality (UVQ) and subsurface models, such as the **N**etwork **E**xfiltration and **I**nfiltration **M**odel (NEIMO), Sewer Leak Index (SLeakI) and Public Open Space Index (POSI). In the final stage of the AISUWRS DSS, groundwater flow modelling is performed using either MODFLOW or FEFLOW. Droubi *et al.* (2008) dynamically coupled the water evaluation and planning (WEAP) software and the groundwater flow model MODFLOW. Kalbacher *et al.*, (2012) used a number of different model combinations, such as the mesoscale hydrologic model (mHM), the EPA Storm Water Management Model (SWMM), aRoot and the Biogeochemical Reaction Network Simulator (BRNS), and linked them to the unsaturated and saturated subsurface model, OpenGeoSys (OGS). This DSS is known as the IWAS-Toolbox. Additionally, simpler applications of SWMM have been demonstrated by Rowan (2001) and Yergeau (2010). Rowan (2001) linked SWMM and MODFLOW using a ‘multiple model broker’, which allows for the exchange or feedback of information between the two models at each time-step during the modelling routine and Yergeau (2010) coupled the SWMM and MODFLOW models in a study of an urban wetland. Göbel *et al.* (2004) used the GwNeu Model to calculate the natural recharge for the North Rhine-Westphalia in Germany and the artificial recharge from infiltration devices was modelled using HYDRUS 2D/3D. The calculated recharge from both these models could then be used as input for the numerical groundwater model, SPRING. Many of these DSSs have not been extensively applied and only the AISUWRS DSS has been specifically designed for urban groundwater applications over a long period of time.

Most of the popular groundwater flow models such as MODFLOW and FEFLOW are three-dimensional, finite difference or finite element groundwater models, that solve a combination of Darcy’s law for water flow in saturated media and a mass balance equation for

various points in the study area. The study area in these models are represented as a matrix of cells, and within each cell is a 'node'. The combination of equations is solved in each of these nodes so that the cell-to-cell flow can be derived. MODFLOW is a good example of a three-dimensional, finite difference groundwater flow model, as it is one of the most well applied, tested and supported groundwater models available and is recognised as an industry standard for groundwater modelling (Yan and Smith, 1994; Camp Dresser and McKee Inc, 2001; Rowan, 2001; Kumar, 2002; Droubi *et al.*, 2008; Yergeau, 2010; Boskidis *et al.*, 2012). As described before, MODFLOW has been applied in a number of other applications in urban areas through the coupling of urban stormwater models and MODFLOW has a number of surface water – groundwater packages that can be applied to measure the interactions between surface water and groundwater. However, Brunner *et al.* (2009) suggest that MODFLOW has a number of limitations when dealing with surface-groundwater interactions. The first limitation is that the unsaturated zone is not considered in the flow interactions between rivers and groundwater. Yet, this is known not to be the case as water does interact with the unsaturated zone. Second, there is often a mismatch between the river width and cell size as the river is assigned to a particular cell which is often much wider than the river. Third, because the river is tied to a particular cell, it cannot be discretised horizontally. Fourth, because vertical discretisation is used to prevent drying out of the cells; this can lead to errors in water table simulations.

Given that current urban water management models (e.g. SWMM, MOUSE, P8 or MUSIC) have limited groundwater modelling capabilities as outlined by Elliot and Trowsdale (2007) and the limitations experienced through the application of the loosely coupled surface and groundwater models, there is a need for a fully-integrated surface water and groundwater modelling (Barron *et al.*, 2013). There are a number of surface water and groundwater models available, that are fully-integrated and spatially distributed, that may be more suitable for modelling surface-groundwater interactions, such as InHM (VanderKwaak and Loague, 2001), MODHMS (HydroGeoLogic, 2006), HydroGeoSphere (HGS) (Therrien *et al.*, 2009), MIKE SHE (Freeze and Harlan, 1969), Wash123D (Cheng *et al.*, 2005) and ParFlow (Kollet and Maxwell, 2006). A recent application of MODHMS by Barron *et al.* (2013) was used to identify the impact of urbanisation on shallow groundwater in Western Australia. It highlighted the potential for the further application and testing of fully-integrated, spatially distributed hydrological models in urban areas. MIKE SHE is another integrated hydrological model that is available that is able to represent land use, including urban land use and infrastructure (Graham and Butts, 2005). The MIKE SHE model was selected for this study and is discussed further in section 3.1.5.

3.1.4 Integrated hydrological modelling

The modelling of urban groundwater and surface-groundwater interactions is complex as there are a number of models available to achieve a wide range of modelling objectives. Therefore, the selection of modelling software needs to carefully consider the modelling objectives, available data, data requirements, model usability, past applications of the model and the cost of software. The loose coupling of surface or stormwater models and groundwater models, which have been commonly applied in urban areas, is limited by the lack of communication or feedback between the models. Specialised groundwater flow models are limited in their ability to deal with surface water and groundwater interaction. Therefore, models that are fully-integrated in terms of their representation of surface water and groundwater processes are desired, due to their holistic

representation of the hydrological cycle. The term ‘integration’ in this context differs from its meaning in section 3.1.2. In this context, ‘integration’ refers to the dynamic coupling of surface and subsurface processes in a hydrological model, processes that are typically modelled in isolation. Whereas, ‘integration’ in the context of section 3.1.2. refers to a shift in urban water modelling towards a more holistic representation the urban water cycle in the urban water modelling framework.

The examples of MODHMS and HGS highlight the improved capabilities of these models. MODHMS has been applied by Barron *et al.* (2013) in an urban area to assess groundwater impacts while also testing the impact of different water management approaches. HGS has been applied by Partington *et al.* (2011) in a study modelling source components in streamflow through the hydraulics of surface and groundwater interactions. The application of these models in studies dealing with urban land use and surface-groundwater interactions supports the selection of an integrated modelling approach in this study. However, there are a number of issues that limit these models for selection. HGS is limited as it does not have a graphic user interface (GUI) to assist with the pre-processing of data and inputs in the model. HGS is also noted as being inflexible with regard to representing hydrological process at different levels of complexity and is also unable to incorporate hydraulic structures like dams or weirs, which are important for urban applications (AquaResource Inc., 2011). MODHMS is also limited in that is unable to represent hydrological processes at different levels of complexity. Another option is MIKE SHE, which will be discussed in the following section.

3.1.5 MIKE SHE and MIKE 11

MIKE SHE was developed by Freeze and Harlan (1969) who described the physical processes of the hydrological cycle by their governing partial differential equations (Graham and Butts, 2005; Zhao, 2012). From 1977, further development of the work by Freeze and Harlan (1969) was done by three European organisations¹ which resulted in Système Hydrologique Européen (SHE) and finally MIKE SHE. Since the mid-1980’s the DHI Water and Environment has continued the development of the MIKE SHE model (Graham and Butts, 2005). MIKE SHE is a physically-based, fully-integrated surface and groundwater hydrological model. MIKE SHE represents the space as a grid and can function at a range of spatial scales, from a single soil profile to a large catchment system (>80 000 km²). MIKE SHE is a flexible modelling system that allows the user to develop an appropriate numerical model within MIKE SHE that is consistent with the user’s conceptual model, available data, desired outcomes and the required detail and complexity (Graham and Butts, 2005). However, there are a number of limitations of physically-based, fully-integrated models such as MIKE SHE, *viz.*:

- The high cost and difficulty of data acquisition,
- The higher computer processing requirements and simulation run time,
- The over-parameterisation of relatively simple or unimportant model processes may lead to an unnecessarily complex model, and

¹ The Institute of Hydrology (United Kingdom), SOGREAH (France) and The Danish Hydraulic Institute (Denmark)

- The disparity between the experimental scale and the scale of the grid used by MIKE SHE.

MIKE SHE is a fully-integrated surface and groundwater model this means the model is able to represent a number of surface water processes including evaporation, overland flow and can be linked to MIKE 11 for channel flow modelling, while also being able to simulate subsurface hydrological processes such as unsaturated flow and saturated groundwater flow (Figure 3.2).

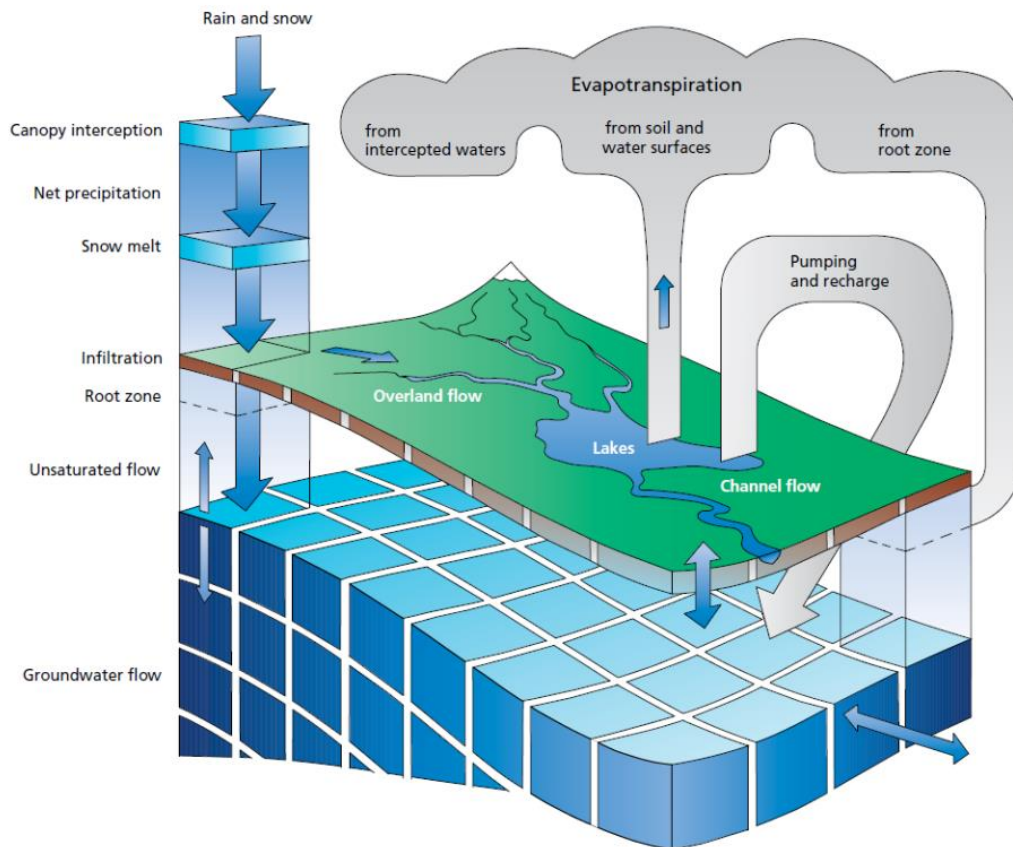


Figure 3.2: A schematic diagram of the MIKE SHE model (DHI, 2014a)

Additionally, another important component for the hydrological modelling of urban areas is the ability to represent the complete urban water system. The MIKE SHE model has the capability of coupling with the urban stormwater model, MIKE URBAN. While MIKE URBAN is not utilised in this study it is important to note that MIKE SHE is able to account for the hydrological components that are specific to the urban context, for instance, the flow of water in sewerage and stormwater pipes.

3.2 Input data requirements and model parameterisation

Two models were established for the Cape Flats: a regional-scale model, and a local-scale model (Figure 3.3). In the regional-scale model, the combination of MIKE SHE coupled to MIKE 11 was used to simulate the surface and groundwater processes of the Cape Flats region for the period from 1980 to 1984. Integrated hydrological modelling at a regional-scale gives insight into the

storage and flow processes of both the surface and groundwater system. The representation of surface water and groundwater processes also means that model calibration can be conducted using a combination of surface water and groundwater elements, *viz.* streamflow, groundwater levels, and items of the water balance. The selection of the calibration period from 1980 to 1984 was based on the availability of sufficient surface and groundwater observations.

The local-scale MIKE SHE model focused on a more detailed representation of the hydrogeological processes of the CFA, so as to simulate the proposed flood mitigation and MAR scenarios. The local-scale model was used to validate the conceptual model for the period from 2000 to 2015. This model was parameterised to focus on groundwater simulations with a more simplified representation of the surface water system by excluding river flow using MIKE 11. Since there are two models that are used in this methodology, the following sub-sections describe the input data and model parameterisation that were used to address the different input data requirements for both the regional- and local-scale models.

The parameterisation of MIKE SHE was based on the principle of parsimony as outlined by Pinder (2002), where the development of a model should progress from a relatively simple model and adding more complexity as and when additional information is required. This serves to ensure that the model is not unnecessarily complex. Overly complex conceptual representation within a hydrogeological model can have negative effects on the performance of the model and may have limited returns in terms of improving the model calibration.

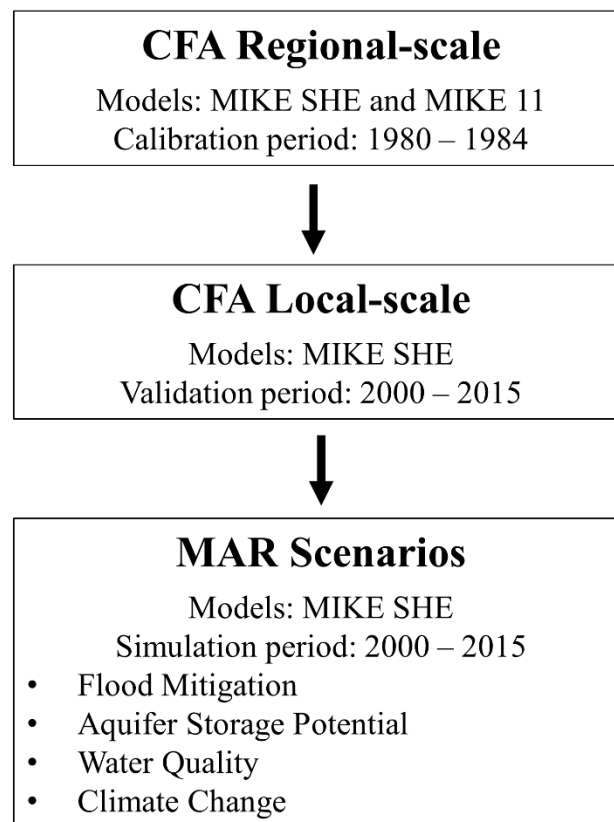


Figure 3.3: A schematic outlining the modelling methodology applied in this study

3.2.1 Rainfall

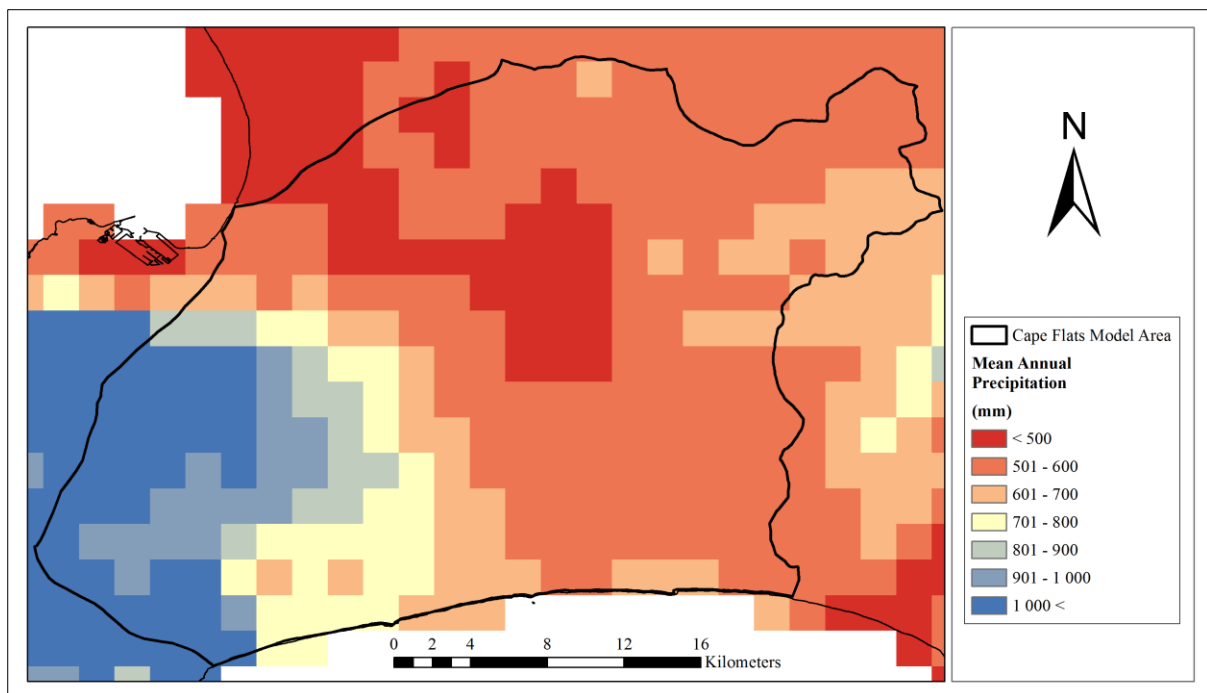
Rainfall is the driver of the hydrological cycle and is therefore an important input data requirement for integrated hydrological models such as MIKE SHE. Rainfall data were observed at six locations over the Cape Flats for the period from 1980 to 1984 (Table 3.1). The rainfall data for the six available rainfall stations were sourced from the data extraction utility within the ACRU Agrohydrological Modelling System (AAHMS)—which has daily rainfall records available through its built-in Quaternary Catchment database (QCD)—and the South African Weather Service (SAWS). The selection of these rainfall stations was based on their proximity to the study area, record length, and data coverage over the calibration period (1980 – 1984). There is significant variation in Mean Annual Precipitation (MAP) over the Cape Flats and the surrounding areas (Figure 3.4). Table Mountain has a high MAP in excess of 1000 mm per year, whereas the Cape Flats is relatively dry between 400 – 700 mm per year. Therefore, in addition to proximity, the rainfall gauges need to be representative of the main types and patterns of rainfall that are experienced on the Cape Flats.

For the calibration period the rainfall areas around Table Mountain were represented by the rainfall station at Kirstenbosch Gardens, which had a high MAP of approximately 1335 mm. The other rainfall stations were used to represent the other areas of the CFA that generally displayed lower MAP values of between 580 and 685 mm. The areas of the model assigned to a particular rainfall station, were based on the proximity of the station and the altitude. Therefore, the Durbanville rainfall station was used to represent the northern, and higher altitude, parts of the model area near the hills of Durbanville. The Airport and Kuilsriver rainfall stations were used to represent the central and eastern parts of the model area respectively. The Observatory rainfall station was used to represent the rainfall at the base of Table Mountain, while Rondevlei represented rainfall along the slightly higher rainfall regions along the False Bay coastline.

Only one reliable rainfall station could be sourced for the validation period from 2000 – 2015. The rainfall station at the Cape Town Airport was selected as it was the only station that was available with data spanning the validation period and that was representative of the typically lower MAP experienced on the Cape Flats. The more confined nature of the local-scale model boundary used in the validation phase means that much of the study site lies within the lower rainfall areas of the Cape Flats. Therefore, due to the relative uniformity of rainfall in the local-scale model, it was assumed that the use of just one rainfall station was sufficient to represent the rainfall of the entire model area.

Table 3.1: Available rainfall stations for the input into MIKE SHE

Raingauge No.	Source	Location	Record Length (Years)	MAP (mm)	Period	Model Setup
0020779 0020780	CSAG	Kirstenbosch Gardens	98	1335	1914 – 2012	Calibration
0020866	CSAG	Observatory	162	607	1850 – 2012	Calibration
0021178 0021179 0021178A3	CSAG	Cape Town Airport	66	580	1950 – 2015	Calibration and Validation
0021260	CCWR	Durbanville	44	597	1950 – 1994	Calibration
0021325	CCWR	Kuilsriver	44	628	1950 – 1994	Calibration
0004874	CCWR	Rondevlei	44	685	1950 – 1994	Calibration

**Figure 3.4: Mean Annual Precipitation (MAP) over the Cape Flats Model area (Schulze *et al.*, 2008)**

3.2.2 Reference evapotranspiration

Reference evapotranspiration (ET_0) is a climatic parameter that describes the magnitude of the evaporative demand of the atmosphere, also known as the potential evaporation (Allen *et al.*, 2006). ET_0 is the rate of evapotranspiration from a reference surface (hypothetical grass) with an

unlimited amount of water (DHI, 2014b). The multiplication of the ET_o by a crop factor (K_c) provides an estimate of the actual evaporation that could be expected from a particular vegetation or land use type. ET_o is an important variable required for the MIKE SHE model so that the Actual Evapotranspiration (ET) can be calculated internally within the models' water balance algorithms. ET is calculated in MIKE SHE using one of two methods, the Kristensen and Jensen (1975) method or the Yan and Smith (1994) method. The Kristensen and Jensen (1975) method is used if the unsaturated zone is modelled using the Richards equation or the Gravity method and Yan and Smith (1994) method is used when the Two-layer water balance method for representing the unsaturated zone is selected (DHI, 2014a).

There are a number of methods for calculating ET_o such as the Penman-Monteith method which is regarded as the most comprehensive means of estimating ET_o , however the Penman-Monteith method is data intensive, requiring a number of inputs, such as wind speed and relative humidity, which are not readily available over most of the Cape Flats particularly during the calibration period. An alternative is the Priestley and Taylor method (Priestley and Taylor, 1972) which requires fewer input data than the Penman-Monteith method. The equation for the Priestley and Taylor method is given as:

$$ET_o = \alpha \left[\frac{\Delta(R_n - G)}{\Delta + \gamma} \right] \quad (1)$$

where α is an empirical constant with an average value of 1.26 determined by Priestley and Taylor (1972), R_n is the net radiation, G is the soil heat flux, Δ is the slope of the saturated vapour pressure versus temperature curve and γ is the psychrometric constant. Solar radiation data was not available for the Cape Flats during the calibration period. Thus, a modified version of the Priestley and Taylor method – developed by Crespo *et al.*, (2011) – was utilised for this study, where the maximum temperature (T_{max}) and minimum temperature (T_{min}), latitude and longitude, altitude and the proximity to the coast, were used to empirically derive R_n . The empirical constant α has also been modified to account for arid climatic regions (Crespo *et al.*, 2011), which makes the modified method more applicable to the study area than the original Priestley and Taylor (1972) method.

The ET_o was calculated for the meteorological station at the Cape Town Airport as this was the only site that had complete T_{max} and T_{min} datasets available during the calibration period (1980 – 1984). The T_{max} and T_{min} records for the Cape Town Airport were obtained from the South African Weather Service (SAWS). The model validation from 2000 – 2015 also used data from the Cape Town Airport meteorological station due to the extensive length and quality of the data record. Further climate data was available for the validation period such as relative humidity, mean daily temperature, mean dew point temperature, and the number of sunshine hours. The FAO ET_o calculator was used to calculate ET_o for the validation period using the additional data available (Raes, 2012). Where no data are available for a specific piece of climate information, the ET_o calculator can estimate these values based on related empirical relationships similar to those describes earlier for the Priestley and Taylor method (Raes, 2012).

3.2.3 Topography

Topography is an important data requirement for MIKE SHE, giving the model geographical context and allowing the model to determine slope, which governs the overland flow and drainage within the model. Therefore, it is important that the topography is as accurate as possible as any inaccuracy may result in errors in the model. Of particular concern is the presence of artificial depressions or “sinks” within the topographic layer which causes the ‘ponding’ of water within the model. This is particularly problematic in MIKE SHE as the overland component of the model is computationally intensive, especially under ponding conditions. A 2 m resolution Digital Elevation Model (DEM) – developed from LiDAR data – of the Cape Flats model area was sourced from the City of Cape Town (Figure 3.5). The resolution and accuracy are sufficient as this resolution is simplified in the model to regional- (500 m) and local-scales (60 m). In order to remove artificial depressions, the DEM was pre-processed using the ‘Fill’ function in ArcGIS version 10.2. The Fill function is part of the hydrology suite of tools available in ArcGIS that removes ‘sinks’ or ‘depressions’ which are considered inaccuracies in the topography.

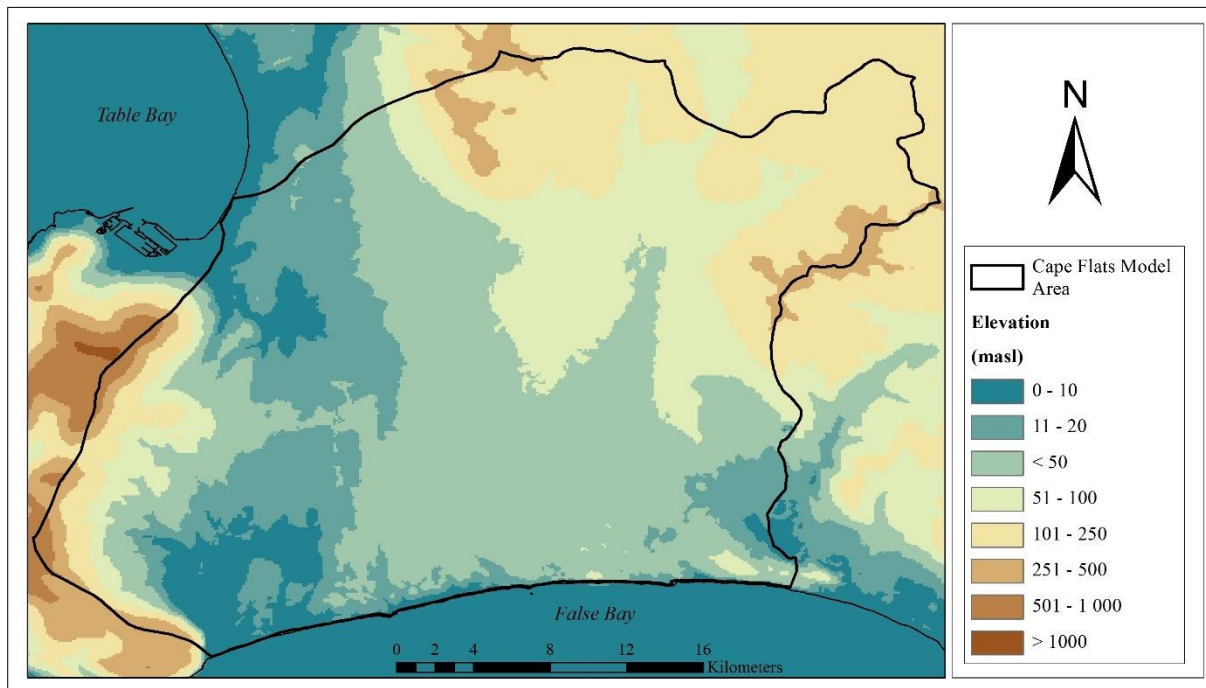


Figure 3.5: Topography of the Cape Flats

3.2.4 Vegetation and urban land use

The primary function of land use and vegetation information in the model is to determine how rainfall is partitioned into different hydrological responses such as infiltration, runoff and evapotranspiration (DHI, 2014b). The land use and vegetation layer for the calibration period used the earliest available digital land cover map, the National Land Cover for the year 2000 (NLC 2000). This layer was edited to remove urban areas that had developed between 1983 and the year 2000, based on aerial photography from 1983 (Figure 3.6). The removed urban land use was

assumed to be the same as the adjacent land uses, which was typically natural vegetation or agriculture. The 31 NLC 2000 vegetation classes were reclassified into 11 broad vegetation classes (Table 3.2).

For each vegetation class, the Leaf Area Index (LAI) and the rooting depth were required as inputs of MIKE SHE. The LAI and rooting depth for water bodies, which have no vegetation, were assigned a zero value, respectively. The values for LAI and rooting depth for natural vegetation were based on a study by Jovanovic *et al.* (2013) who specifies the vegetation properties of fynbos in the Atlantis area, approximately 50 km north of the Cape Flats. This vegetation was similar to that of the Cape Flats as it grows in comparable soil and climatic conditions on the Cape Flats, thus these values are assumed to be an acceptable representation of the natural vegetation. There are limited values for LAI and rooting depth for wetlands in South Africa, however a study by Clulow *et al.* (2012) suggests a LAI of approximately 2 $\text{m}^2\cdot\text{m}^{-2}$ for a wetland in the Northern KwaZulu-Natal. Values for the rooting depth of wetlands could not be sourced and thus were assumed to be the equivalent of those specified in the natural vegetation category. LAI and rooting depth values for plantation forestry were taken from studies by Canadell *et al.* (1996) and Bulcock and Jewitt (2010). The main species of trees in the areas is pine (*Pinus radiata* or *Pinus pinaster*), which has a high LAI of approximately 4 $\text{m}^2\cdot\text{m}^{-2}$ and rooting depths of up to 4 m. The vegetation in open spaces is assumed to consist of grasses such as Kikuyu (*Pennisetum clandestinum*), which typically the preferred turf grass for open and recreational areas; it was also listed as an invasive species so any fallow land was likely to be invaded by Kikuyu (Macdonald *et al.*, 2003). The values for Kikuyu were estimated from two studies Brauman *et al.* (2012) and FAO (2016). Both these studies gave very high values for LAI ($>3 \text{ m}^2\cdot\text{m}^{-2}$) and deep roots (5.5 m), however more conservative estimates were selected as most of these areas are likely to be mown regularly so it was assumed that the LAI would be greatly reduced and therefore a LAI of 1 $\text{m}^2\cdot\text{m}^{-2}$ was selected. The FAO (2016) suggest that 90% of the roots of the Kikuyu grass occur in the first 60 cm, thus a more conservative estimate of root depth was selected at 1 m. Information relating to the typical LAI and rooting depth values used for urban applications of MIKE SHE was scarce. Thus, these values were estimated but were based on a study by Alonzo *et al.* (2015) who used satellite imagery to estimate the LAI for of an urban area in Santa Barbara, California in the United States of America. Much of the urban impervious areas had LAI values of approximately 0.5 $\text{m}^2\cdot\text{m}^{-2}$ and most of these areas were relatively well vegetated residential areas. The rooting depth was assumed to be equivalent to that used for Kikuyu as lawns that were a typical occurrence in residential areas. The “Urban” land used includes areas that were associated with a high level of imperviousness such as commercial and industrial land uses. Therefore, the LAI and rooting depths were reduced from the typical values for residential areas, to 0.2 $\text{m}^2\cdot\text{m}^{-2}$ and 500 mm to account for the reduced vegetation in these areas.

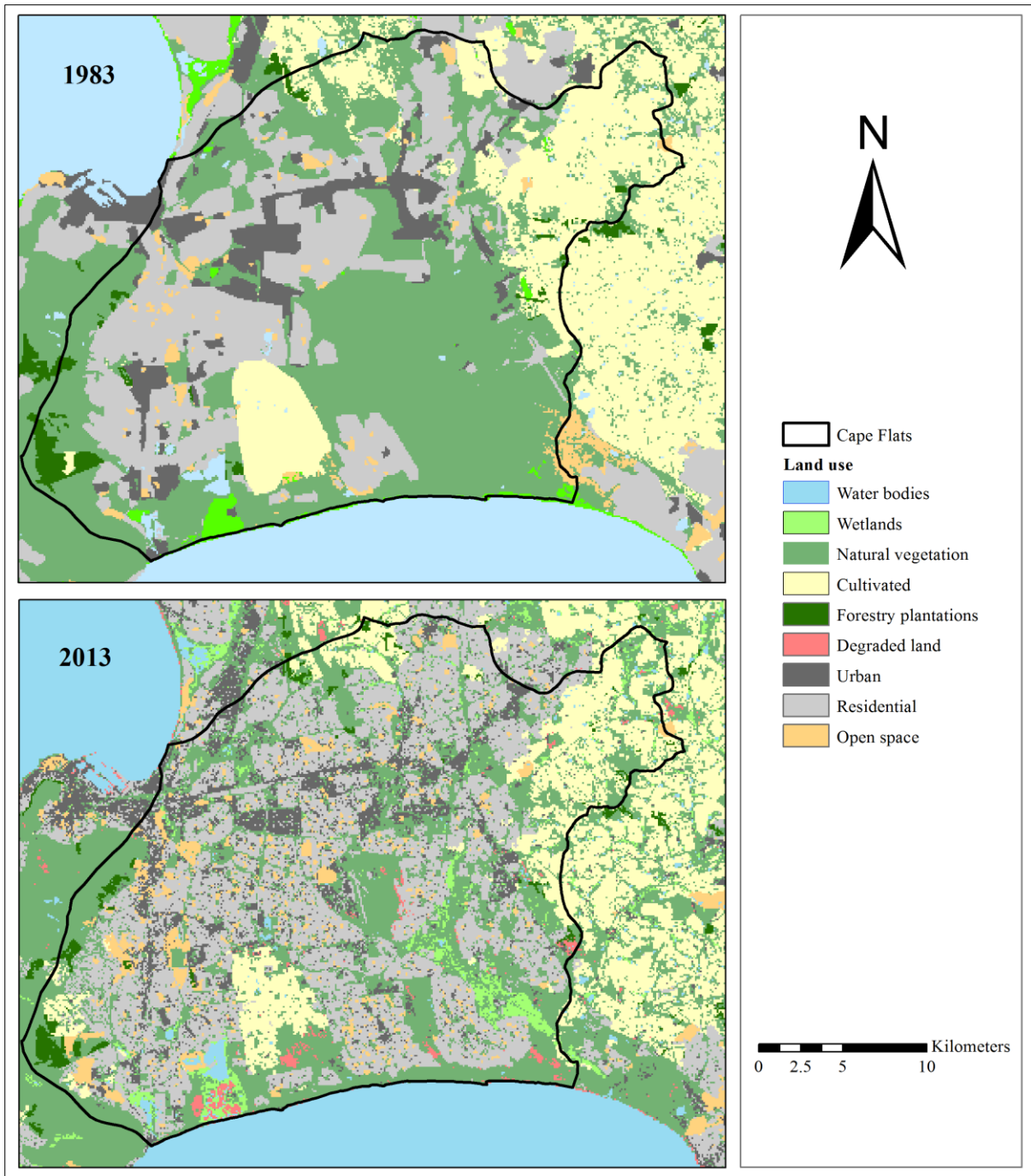


Figure 3.6: A comparison of the land use used for the regional-scale model in 1983 and the land use used in the local-scale model used on 2013

Table 3.2: The simplified land use types specified in the CFA model area showing the Leaf Area Index (LAI) and Rooting Depth

Land use type	Leaf Area Index (LAI)	Rooting Depth (mm)
Water bodies	0	0
Wetlands	2	1500
Natural vegetation	1.3	1500
Cultivated land	2	1500
Forestry plantations	4	4000
Degraded land	0.1	500
Urban	0.2	500
Residential	0.5	1000
Open space	1	1000

Additional information was required for the representation of urban land use and paved areas in MIKE SHE. The imperviousness of urban land use was represented as ‘paved areas’. The Paved Runoff Coefficient specifies the fraction of overland flow or ponded water that is routed directly to drainage. The paved areas were specified for two land use types: ‘urban’ and ‘residential’, and the Paved Runoff Coefficients were determined for these areas through calibration (see section 3.3.2).

Furthermore, MIKE SHE required inputs for surface roughness using the Manning’s M value, with the unit’s $\text{m}^{-1/3} \cdot \text{s}^{-1}$, to determine overland flow. Manning’s M is equivalent to the Stickler Coefficient, which is the inverse of the more commonly used Manning’s n . The typical range for Manning’s n is 0.01 for smooth channels to 0.10 for thickly vegetated channels. Therefore, these values correspond to the equivalent range for Manning’s M of 100 and 10, respectively (DHI, 2014b). The Manning’s M coefficient was important as it determined the rate at which water was removed from the surface into the drainage system. The Manning’s M values were only applied to the urban and residential areas, while it was not included for the other vegetation classes as it was assumed that the overland flow would be negligible given the dense vegetation and highly permeable sandy soils. The typical values for Manning’s M for urban areas range from 90 – 70 or 0.011 – 0.014 in the Manning’s n equivalent (Chow *et al.*, 1988; Trinh and Chui, 2013; Akhter and Hossen, 2014). The typical values Manning’s M for residential areas range from 60 – 40 or 0.016 – 0.025 in the Manning’s n equivalent (Chow *et al.*, 1988; Goodrow, 2009). Therefore, the Manning’s M values for this study were set to 80 for urban areas and 50 for residential areas.

3.2.5 River network

MIKE 11, when linked to MIKE SHE, functions as a collection system for overland flow and drainage including the drainage of groundwater. Inversely, MIKE SHE can also receive recharge from water bodies in the event that the hydraulic gradient between the river and the groundwater is reversed (DHI, 2015). The MIKE 11 model developed for this study was relatively simple, as the main purpose of the model was to remove runoff and drained water from MIKE SHE without detailed routing calculations.

The MIKE 11 network was relatively coarse, only considering the main rivers and excluding minor rivers, local artificial drainage and stormwater infrastructure (Figure 3.7). The river network was digitised using an ArcGIS shapefile of South African national rivers obtained from the Department of Rural Development and Land Reform (DRDLR). Cross-sectional data was not readily available for these rivers and collecting cross-sectional data manually would be expensive and time consuming given the number of cross-sections required. Thus, Google Earth was utilised to measure the channel width, while the channel dimensions were estimated using Google Street View and verified with field visits where possible. The elevation of the cross-sections was determined using the DEM developed for the Cape Flats and surrounding areas.

In order to simulate the flow of water in a river, the model must be able to calculate how a flood wave or hydrograph moves down the river reach. Typically, this is achieved through the solution of St. Venant equations for the river reach; however, this requires detailed cross-sectional information, which is frequently unavailable. Routing is the simplified means of calculating the progression of a hydrograph by determining the transformation of an upstream hydrograph as it progresses downstream, which is a function of the hydrodynamic properties of the river channel. There are two types of routing options available in MIKE SHE: 'Routing' and 'Kinematic Routing'. Kinematic Routing is used when the main objective is to simply route water down the main river system. As a result, Kinematic Routing cannot incorporate additional hydraulic structures, such as weirs or account for backwater effects (DHI, 2015). This routing method is ideal for this model application where the main purpose of the river network is to provide a means of drainage for the MIKE SHE model. The main parameters required for Kinematic Routing are the Manning's M value for characterising the bed resistance and the leakage factor, which determines the rate at which water is gained or lost by the riverbed. Both these values were used as calibration parameters when calibrating to observed streamflow (see section 3.3.2).

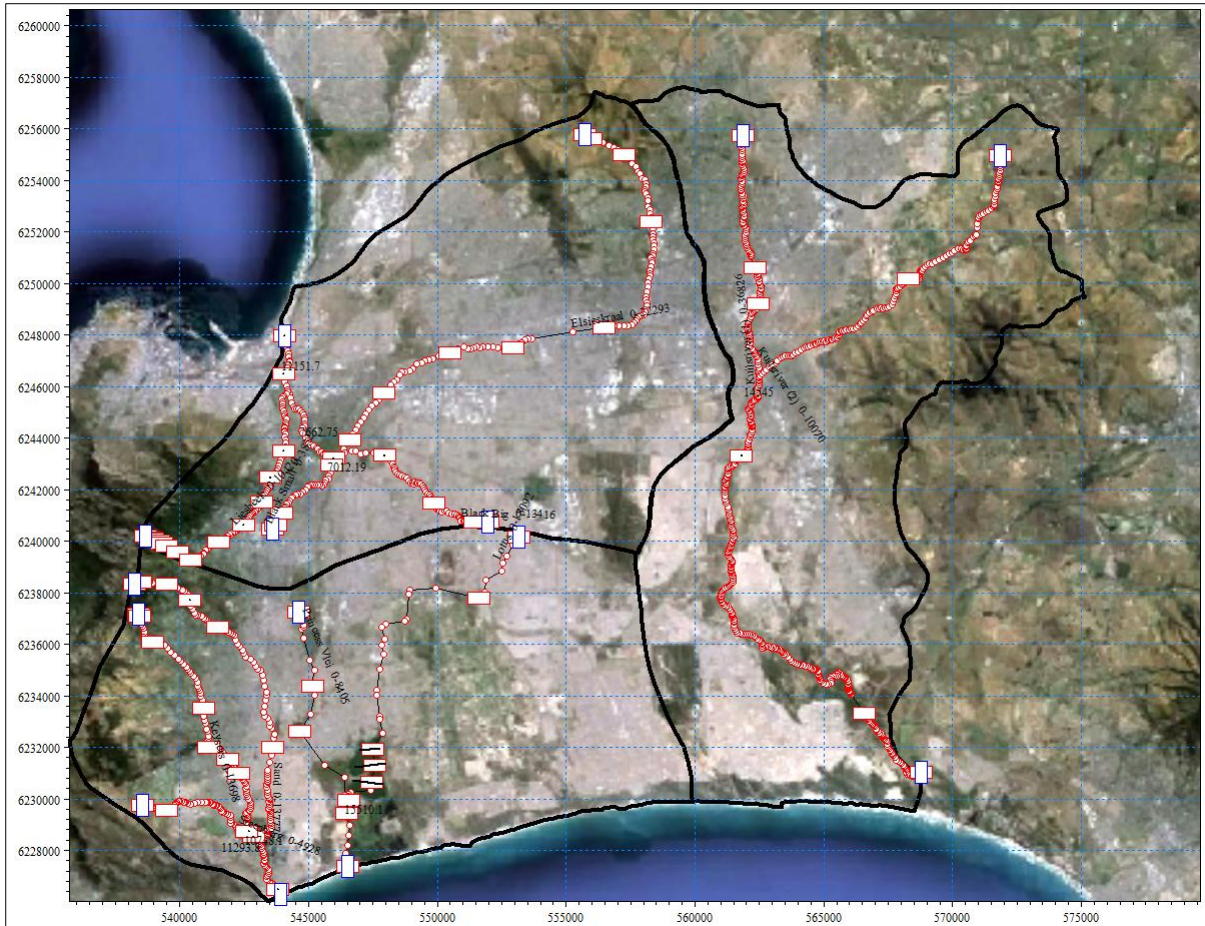


Figure 3.7: Screen-shot of the MIKE 11 river network that was coupled to MIKE SHE

3.2.6 Soil

Three methods for representing the hydrological processes of the unsaturated zone are available in MIKE SHE *viz.* the Richards equation, the Gravity method and the Two-layer water balance method. The method selection was dependent on the soil information available, the model's size and resolution, the required detail of the soil layer, and the computational capacity available and the objectives of the simulation. The Richards Equation generally requires detailed descriptions of the soil's moisture retention and effective hydraulic conductivity relationships. Much of this information was not available for the Cape Flats sands and developing this information would be costly and time-consuming, and would have limited value due to the variation in soil conditions at a regional-scale. The Gravity method is a simplification of the Richards equation but ignores capillary effects. While the Two-layer water balance is a further simplification, representing the average conditions of the soil profile as two layers. The Richards equation is the most computationally intensive method and can greatly increase the run-time of the model. The Gravity method does reduce the run-time somewhat due to simplifications in the Richards equation, however the Two-layer water balance method results in the shortest model run-time due to the simplified representation of the unsaturated zone. Furthermore, the simple Two-layer water balance method accounts for limited data availability, only requiring data that is readily available

such as the soil water content at saturation, field capacity and wilting point and hydraulic conductivity. This is important on the Cape Flats as there are no known, detailed soil studies and therefore all of the soil parameters used in the model setup were derived from a range of literature values. Therefore, the Two-layer water balance method was selected for the study. Furthermore, the simple water balance concept for the representation of unsaturated flow is commonly used for water resources modelling applications in South Africa such as ACRU—developed in South Africa—which employs a similar method of representing the unsaturated zone (Schulze, 1995b). ACRU has been successfully applied for the purpose of identifying hydrological processes that are dependent on the unsaturated zone such as groundwater recharge, thus demonstrating the effectiveness of the soil water balance method (Lorentz *et al.*, 2003).

The primary sources for unsaturated zone data were from the SCS-SA soil maps (Schmidt and Schulze, 1987) and the ARC Land-type maps (ARC, 2003). These maps were used to construct a soil distribution map required by MIKE SHE. The map was based on the SCS-SA soil map (see Figure 3.8) however some simplifications have been made, namely the ‘Deep Lamotte formation’ and the ‘Regic sands and other soil’ were merged, leaving two dominant layers covering the majority of the model area. The main reason for the simplification of the soils over the CFA was to reduce complexity for the model calibration, thus the calibration process only used soil zones 5 and 6. These two sand types represent most of the sand of the Cape Flats overlying the CFA. The 1:50 000 geological map series guided the simplification process of the soils zones as it indicates two predominant type of Quaternary sands: the quartzose sands in the northern parts of the Cape Flats (Soil code 5) and the calcareous sands in the Southern parts of the Cape Flats (Soil code 6).

The SCS-SA soil map provided the water content at saturation (θ_s), field capacity (θ_{fc}) and wilting point (θ_{wp}) (Table 3.3). The typical values for θ_s , θ_{fc} , and θ_{wp} range from 0.420 – 0.456, 0.181 – 0.210 and 0.066 – 0.120. There is limited information on saturated hydraulic conductivity for each soil zones, thus estimates of these values were required. The values were estimated from a study by Schulze (1995), however in order to make this estimation the soil texture class is required, which were sourced from the ARC Land-type maps (ARC, 2003). The upper and lower limits that are required for the calibration of the unsaturated zone for soil zones 5 (quartzose sand) and 6 (calcareous sand) have been specified in Table 3.3. In order to limit the selection of inappropriate or unrealistic parameters for the calibration, a range of appropriate or realistic properties for sandy soil must be specified.

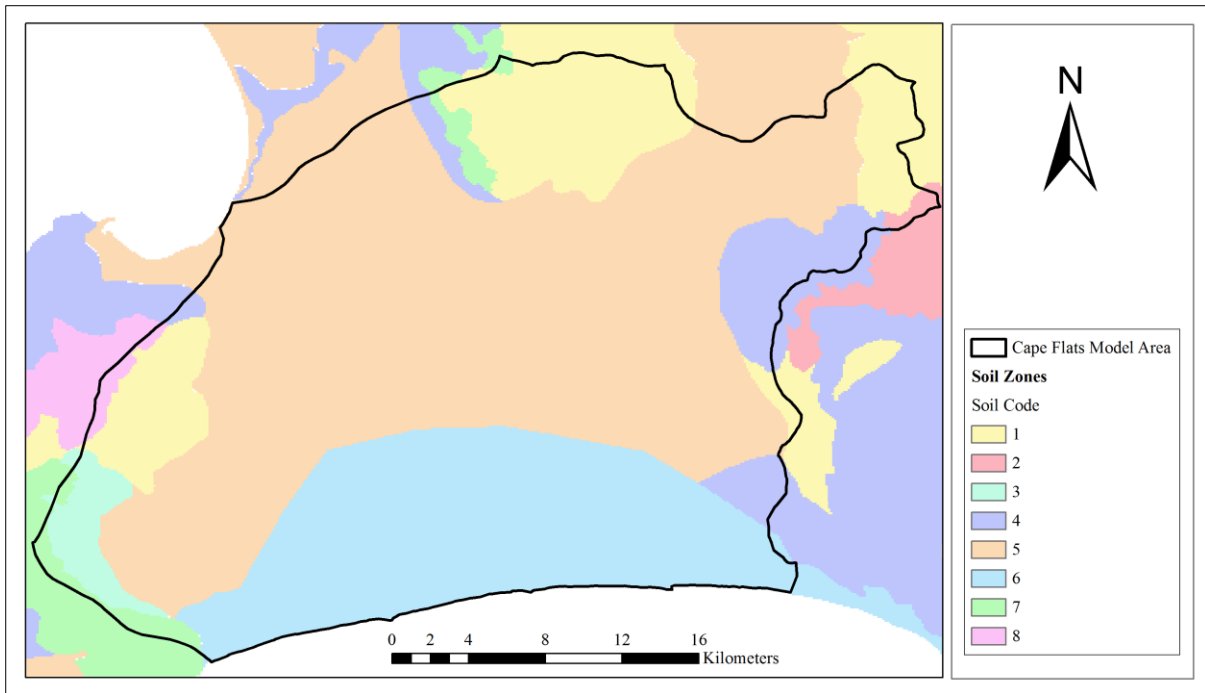


Figure 3.8: The distribution of soil over the Cape Flats model area.

Table 3.3: The soil properties for the soils of the Cape Flats and surrounding areas.

Soil Code	Calibration	Water Content (θ)			*Saturated Hydraulic Conductivity ($m.s^{-1}$)
		*Saturation (θ_s)	*Field Capacity (θ_c)	Wilting Point (θ_{wp})	
1	No	0.456	0.188	0.097	4.17×10^{-7}
2	No	0.430	0.209	0.120	4.17×10^{-7}
3	No	0.446	0.192	0.100	4.17×10^{-7}
4	No	0.439	0.200	0.110	7.22×10^{-6}
5	Yes	0.3 – 0.45	0.15 – 0.22	0.066	$1 \times 10^{-6} - 0.0001$
6	Yes	0.3 – 0.45	0.15 – 0.22	0.086	$1 \times 10^{-6} - 0.0001$
7	No	0.450	0.181	0.091	5.83×10^{-5}
8	No	0.42	0.21	0.1	4.17×10^{-7}

* Values with ranges for water content and saturated hydraulic conductivity were the ranges used for specifying the upper and lower limits for the model calibration.

3.2.7 Geology and aquifer properties

The Cape Flats Aquifer (CFA) is a primary aquifer situated in the Quaternary sands of the Cape Flats. These sand deposits include fluvial, marine and aeolian deposited sands, underlain by Malmesbury Group and Cape Granite basement rock (Figure 3.9). The Malmesbury Group is a metamorphosed Greywacke, consisting of phyllite, shale and quartz. The Cape Granite Suite heavily intrudes into the Malmesbury Group. Overlying the basement layers of the Malmesbury and Cape Granite is the topographically dominant Peninsula Formation, which is a major unit of the sedimentary sequence of the Cape Supergroup. The Cape Supergroup is absent over the Cape Flats due to erosion, resulting in the Quaternary sands that dominate the Cape Flats (DWAF, 2008). A number of studies have described the stratigraphy of the Quaternary sands of the Cape Flats through the analysis of borehole logs (Henzen, 1973; Wessels and Greef, 1980; Hay, 1981) (Figure 3.9). These studies indicate that the Quaternary sands reach a maximum thickness of ~55 m in the South West of the Cape Flats. Henzen (1973), Wessels and Greeff (1980), Hay (1981), and Vandoolaeghe (1989) have interpreted the stratigraphy of the Quaternary sands as highly heterogeneous, comprising of discrete, continuous layers of gravel, clean sand and calcareous sands with discontinuous, interbedded lenses of calcrete, clay and peat. DWAF (2008) provided a detailed analysis of the stratigraphy described in all of the local studies and attempted to summarize these in order to describe the 'typical' stratigraphy for the Cape Flats for analysis at a regional-scale. The study concluded that at a regional-scale the unconsolidated deposits could be reasonably classified into two layers, with coarse sediments below sea-level and finer peaty sediments above sea-level. However, after testing this interpretation using a numerical model, the best results were achieved using a three-layer conceptual model.

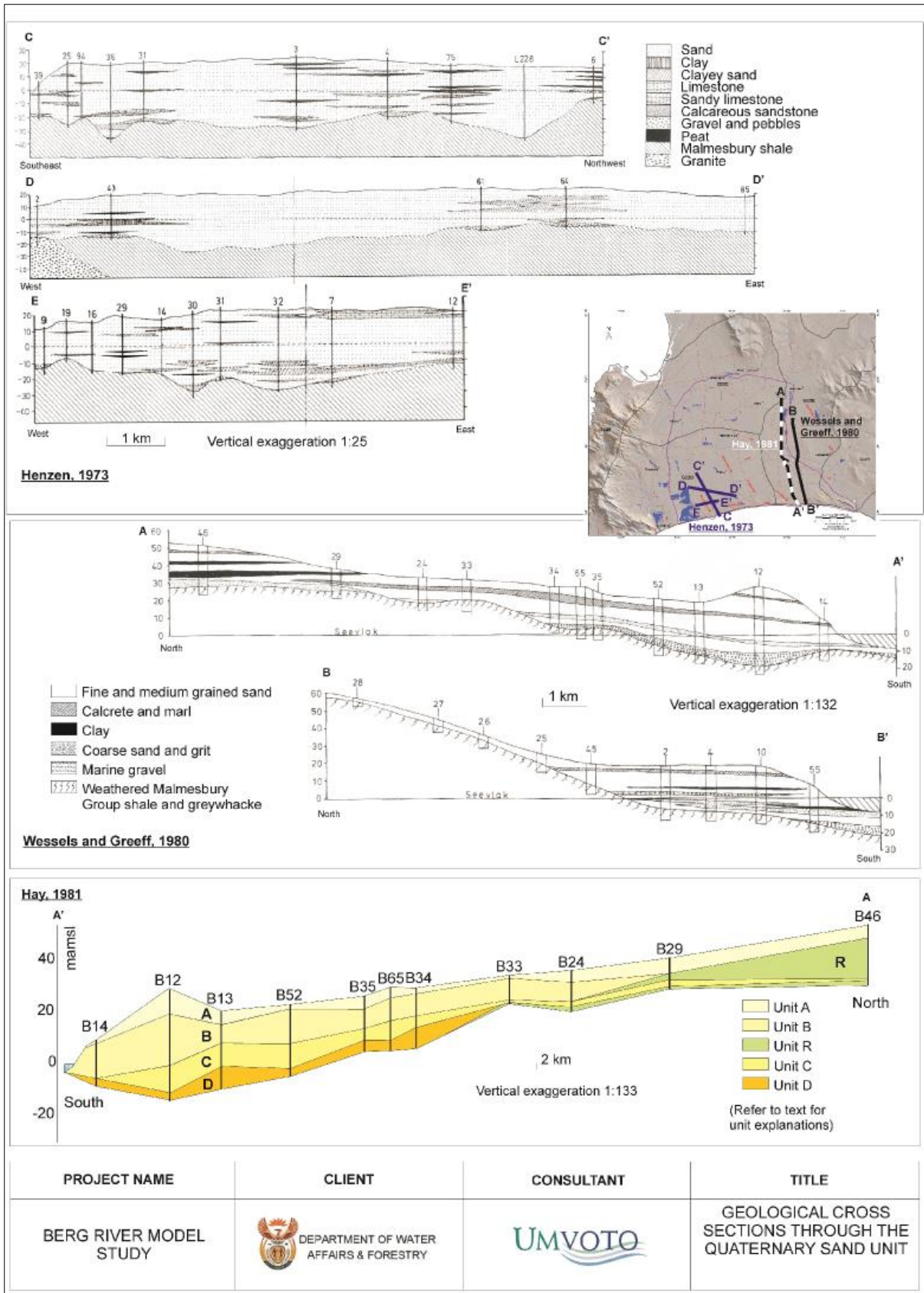


Figure 3.9: The stratigraphy of the Cape Flats Quaternary Sands (DWAF, 2008)

The conceptual hydrogeological model for the CFA for this study was developed from the information collected in studies by Henzen (1973), Wessels and Greef (1980) and Hay (1981), the 1:50 000 geological map series and the conceptual model outlined in the DWAF (2008) modelling report. The DWAF (2008) conceptual model was developed based on a detailed review of all the past hydrogeological information on the Cape Flats, thus this study was assumed to be a good starting point for further model development. The main outcomes of the DWAF (2008) modelling report, in terms of the conceptual hydrogeological model, was that the CFA was assumed to be regionally unconfined and therefore can be reasonably modelled using a relatively simple 4 layer model. It is important to note that the conceptual model developed by the DWAF (2008) study was focused on the saturated zone and used the FEFLOW groundwater model. Based on a summary of the geology and stratigraphy, the study specified the following conceptual model. The first layer of the model consisted of the first 1 meter below the topography. This layer was required to represent the model's interaction with surface water features, such as rivers and wetlands. The second layer extended from the bottom of Layer 1 to half way between Layer 1 and the mean sea-level. Layer 3 extended from the bottom of Layer 2 to mean sea-level and Layer 4 extended from Layer 3 to the basement topography or bedrock. These layers were then calibrated to observed data.

Table 3.4: The conceptual model developed for the CFA by DWAF (2008)

Layer	Top	Bottom	Thickness	Comment
1	Topography	Topography – 1 m	1m	Required to represent rivers
2	Topography – 1 m	Half way between (topography – 1) and 0 mamsl, or 0 mamsl + 0.1 m	0.45 – 28.0 m	Numerical model layers do not represent geological boundaries exactly.
3	Half way between (topography – 1) and 0 mamsl, or 0 mamsl + 0.1 m	0 mamsl. 0 mamsl + 0.1 m where basement > 0 mamsl	0.45 – 28.0 m	Numerical model layers do not represent geological boundaries exactly.
4	0 mamsl. 0 mamsl + 0.1m where basement > 0 mamsl	Basement topography	0.10 – 29.8 m	Represents higher proportion of higher hydraulic conductivity below 0 mamsl, including basal gavels

The conceptual model developed for this study was similar to those described above with a few exceptions. This conceptual hydrogeological model consists of three layers, namely the Layers 2, 3 and 4 from those used in the DWAF (2008) study. Layer 1 was excluded due to MIKE SHE's ability to represent the unsaturated zone, which controls the partition of rainfall into recharge, evaporation and overland flow. Additionally, MIKE SHE is linked to MIKE 11, which accounts for the interaction of groundwater and the river network. Therefore, the layering structure is equivalent to that of the DWAF (2008) study, except the upper most layer which extends from the topography to half way between the topography and sea-level (Figure 3.10).

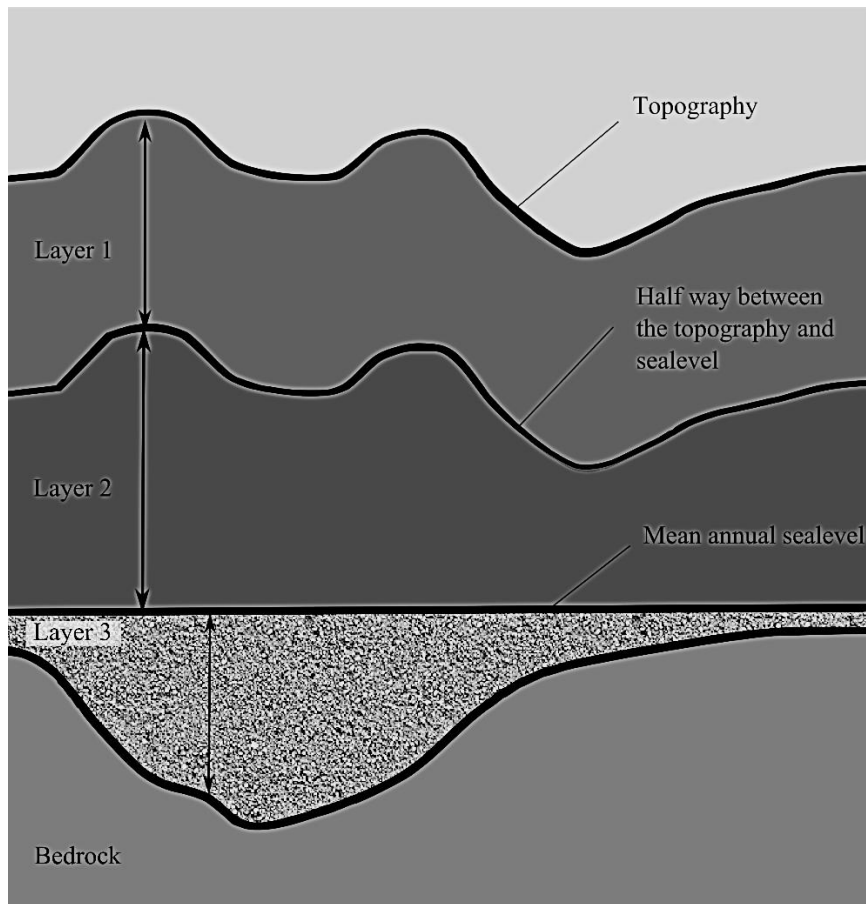


Figure 3.10: The conceptual hydrogeological model of the CFA developed for the MIKE SHE model

The geology of the layers used in the parameterisation of the MIKE SHE model were conceptualised as ‘geological units within layers’, which allows for variation of geology within a particular layer. There are three main types of Quaternary sediments that were used to specify the geology of the layers used in the MIKE SHE model. Namely, calcareous sand, quartzose sand and basal gravel. The first layer was assumed to have a higher conductivity than the second layer, while the third layer was assumed to have a very high hydraulic conductivity due to its high porosity (DWAF, 2008). In the first layer the geology is distributed, based on information from the 1:50 000 geological map series. The geological maps show distinct types of Quaternary sands, *viz.* calcareous sands in the south and central parts of the model area and quartzose sand in the north (Figure 3.11). The geology of Layer 2 was assumed to be uniform quartzose sand. The geological composition was distributed in Layer 3, with basal gravels dominating those areas that are typically below sea-level, running along the coast and inland along the postulated paleochannel. The adjacent geology was assumed to be quartzose sand. It is also important to note that in many of the borehole logs for those areas where basal gravels were found, clay or clayey sand were identified overlying the basal gravel. Thus, a clay lens was incorporated with a spatial extent equivalent to that of the basal gravel at a thickness of approximately 5 m.

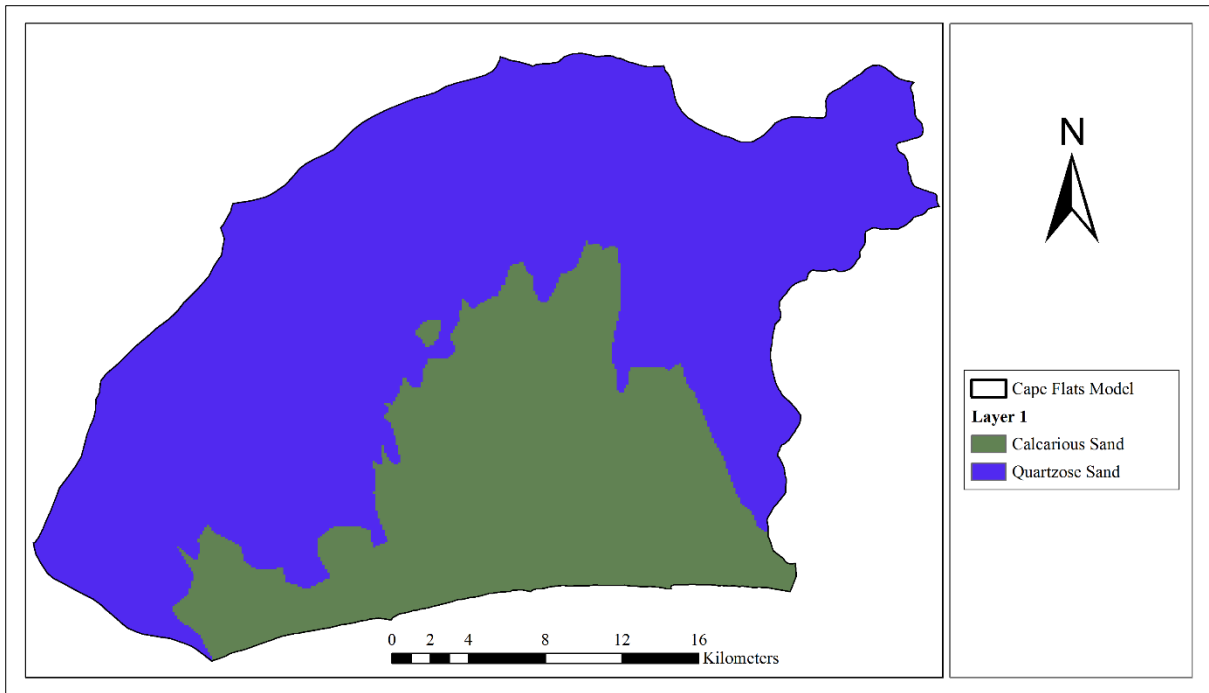


Figure 3.11: The spatial distribution of quartzose and calcareous Quaternary sands within Layer 1

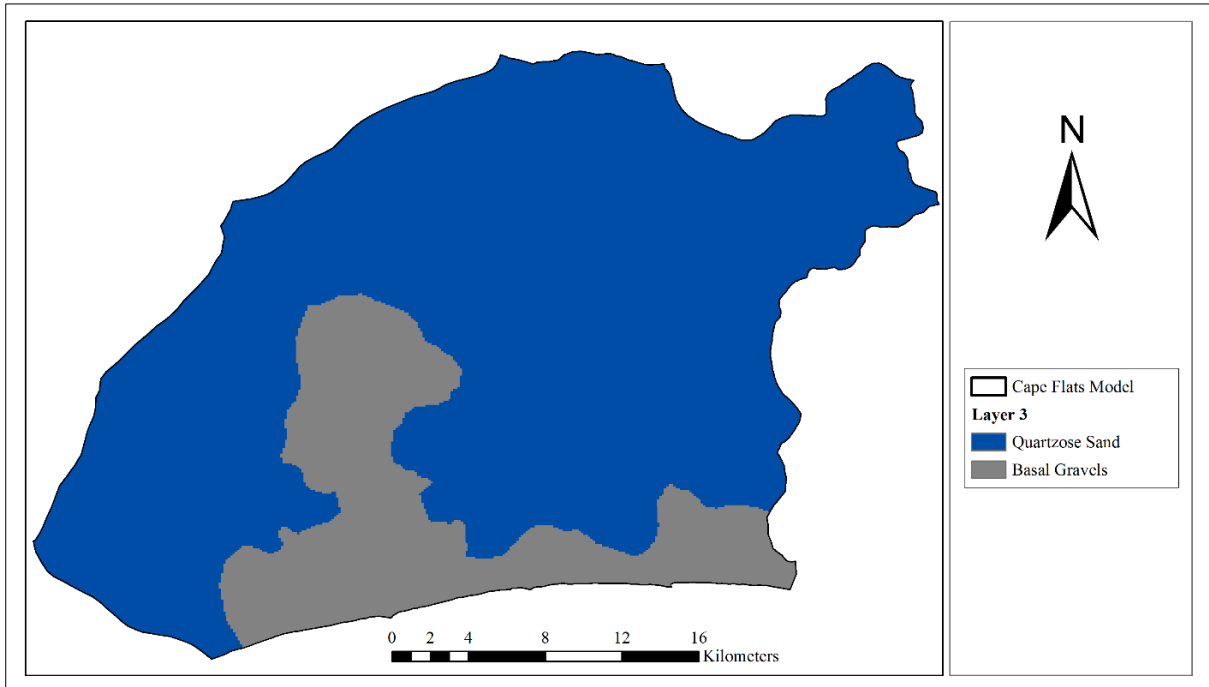


Figure 3.12: The spatial distribution of quartzose Quaternary sands and basal gravels within Layer 3

3.3 Model calibration and validation

It is necessary that an integrated hydrological model, used for investigating surface and groundwater processes, be calibrated to both surface and groundwater information to ensure that the complete hydrological cycle is represented realistically. Therefore, a range of information—quantitative and qualitative—was used to calibrate this MIKE SHE model including visual and statistical comparisons of the simulated and observed groundwater levels, head elevation, and streamflow discharge, as well as information from past studies and modelling applications. The calibration of MIKE SHE was subject to a number of limitations and uncertainties. Firstly, due to the fact that the majority of the groundwater monitoring performed on the Cape Flats was during the late 1970's early 1980's, the window for the calibration period was restricted to the relatively short period from 1980 to 1984. Second, the data required for this period for model calibration was further limited as only four boreholes had sufficient and complete record lengths and minimal interference from external influences such as pumping. In addition, only one streamflow gauging station could be obtained for this period. Thus, given these data limitations, in particular, the short record length, it was concluded that the available data would be most effectively used to calibrate the model; however, there was insufficient data to conduct a comprehensive validation of both surface and groundwater processes.

3.3.1 Groundwater calibration

The selection of model parameters for the calibration of the saturated zone in MIKE SHE was guided by literature recommendations and manual testing within the model. MIKE SHE's auto-calibration function, AUTOCAL, was used to calibrate the selected parameters. AUTOCAL performs multiple model simulations, iterating the calibration values within the specified lower and upper bounds until the best statistical outcome is achieved (Table 3.5). In this study, MIKE SHE was calibrated to borehole level measurements and the root mean square error (RMSE) evaluated for each calibration point. The statistical performance of each borehole was then aggregated into an objective function that describes the best overall performance. The main parameters used to calibrate groundwater levels were soil (unsaturated zone) and aquifer (saturated zone) properties. The conceptual model for the saturated zone consisted of three layers of unconsolidated deposits (i.e. Calcareous Sand, Quartzous Sand and Gravel). The horizontal hydraulic conductivity (K) of these layers was calibrated and the best results were obtained with the upper calcareous sand with a relatively high K value of 2.3×10^{-4} , underlain by quartzous sand with lower K values of 5.6×10^{-5} and finally the basement gravel layers were associated with high K values of 6×10^{-4} . The soils of the Cape Flats were classified into two main soil types, calcareous sand in the South and quartzous sand to the North. The unsaturated zone parameters determine the amount and rate of recharge as governed by the saturated hydraulic conductivity (K_{sat}), water content at saturation (θ_{sat}) and field capacity (θ_{fc}), and the bypass coefficient. The K_{sat} , θ_{sat} and θ_{fc} are typical soil water variables. The bypass coefficient describes the fraction of a rainfall event that bypasses the soil matrix and contributes directly to groundwater recharge. The calibrated bypass coefficient was higher in calcareous sands (0.34) than the quartzous sands (0.19), however the K_{sat} was lower in the calcareous sands (1.6×10^{-5}) than the quartzous sands (3.2×10^{-5}). These calibrated values suggest that the calcareous sands have a higher initial recharge response from water bypassing the soil matrix, perhaps due to macro-pore flow, with a slower recharge response from the soil matrix. The quartzous sands show a lower bypass coefficient and higher K_{sat} values

promoting a greater flow of water through the soil matrix. The quartzous sand shows lower θ_{sat} values at 0.37 when compared to the calcareous sand at 0.45, however the calcareous sand has a lower calibrated θ_{fc} value at 0.15 compared to the quartzous soil at 0.19. The difference in the calibrated θ_{sat} and θ_{fc} values indicated that the calcareous soil has a higher specific yield than the quartzous sands, as a result the groundwater recharge is higher in the calcareous sands.

Simulated groundwater levels were calibrated to observed groundwater levels for four available monitoring boreholes, *viz.* the Cape Town Airport, Gugulethu, Skaapkraal, and Mitchells Plain, for the period 1980 to 1984 (Figure 3.13). A visual assessment of the simulated groundwater levels indicated that the model captures seasonal fluctuations reasonably well and the simulations were relatively consistent over all four boreholes. However, it was clear that the model performed better in the boreholes with deeper groundwater levels, *viz.* Mitchells Plain and Skaapkraal. The groundwater levels at these points show a good correlation with observed data, displaying a coefficient of determination (R^2) values of 0.80 and 0.76, and RMSE of 0.21 and 0.42 m respectively. The Airport and Gugulethu both have average groundwater depths of less than 3 m and the model was unable to replicate the magnitude of fluctuations seen in the observed groundwater levels. The airport showed a good representation of the mean groundwater level, however the minimum and maximum observations were not well simulated. In the case of Gugulethu, the maximum values were reasonably well represented, however the minimum observations were poorly simulated. The poorer performance of the boreholes located in areas with shallow groundwater could be due to the simplistic representation of the unsaturated zone. More detailed, physically-based unsaturated zone methods, such as the Gravity or Richards methods—with improved soil data—will improve the description of the soil water retention and hydraulic conductivity characteristics and evapotranspiration from the unsaturated zone within the model, thereby enabling the model to better capture the seasonal variations in areas where the average groundwater level is high. Refinement of the conceptual geological model might aid the representation of the hydrogeological processes of the Cape Flats. Detailed studies have been conducted in many areas of the Cape Flats, but few studies have assessed the stratigraphy of the areas to the north east where the model shows poorer performance.

Table 3.5: The selected calibration parameters with the Calibrated value, Initial, Lower and Upper bounds.

Profile	Parameter	Unit	Geology	Calibrated Value	Initial Value	Lower Bound	Upper Bound
Saturated Zone	Hydraulic Conductivity	ms ⁻¹	Calcareous Sand	2.3×10^{-4}	1×10^{-4}	5×10^{-5}	4×10^{-4}
	Horizontal Hydraulic Conductivity	ms ⁻¹	Quartzous Sand	5.6×10^{-5}	1×10^{-4}	5×10^{-5}	4×10^{-4}
	Horizontal Hydraulic Conductivity	ms ⁻¹	Gravel	6×10^{-4}	9×10^{-4}	6×10^{-4}	1×10^{-3}
Unsaturated Zone	Water Content at Saturation	Ratio	Calcareous Sand	0.45	0.25	0.15	0.40
	Water Content at Field Capacity	Ratio		0.15	0.45	0.30	0.60
	Bypass Coefficient	Fraction		0.34	0.20	0.10	0.35
	Saturated Hydraulic Conductivity	ms ⁻¹		1.6×10^{-5}	5×10^{-5}	1×10^{-4}	4×10^{-3}
	Water Content at Saturation	Ratio	Quartzous Sand	0.37	0.25	0.15	0.40
	Water Content at Field Capacity	Ratio		0.19	0.45	0.30	0.60
	Bypass Coefficient	Fraction		0.19	0.20	0.10	0.35
	Saturated Hydraulic Conductivity	ms ⁻¹		3.2×10^{-5}	5×10^{-5}	1×10^{-5}	1×10^{-4}

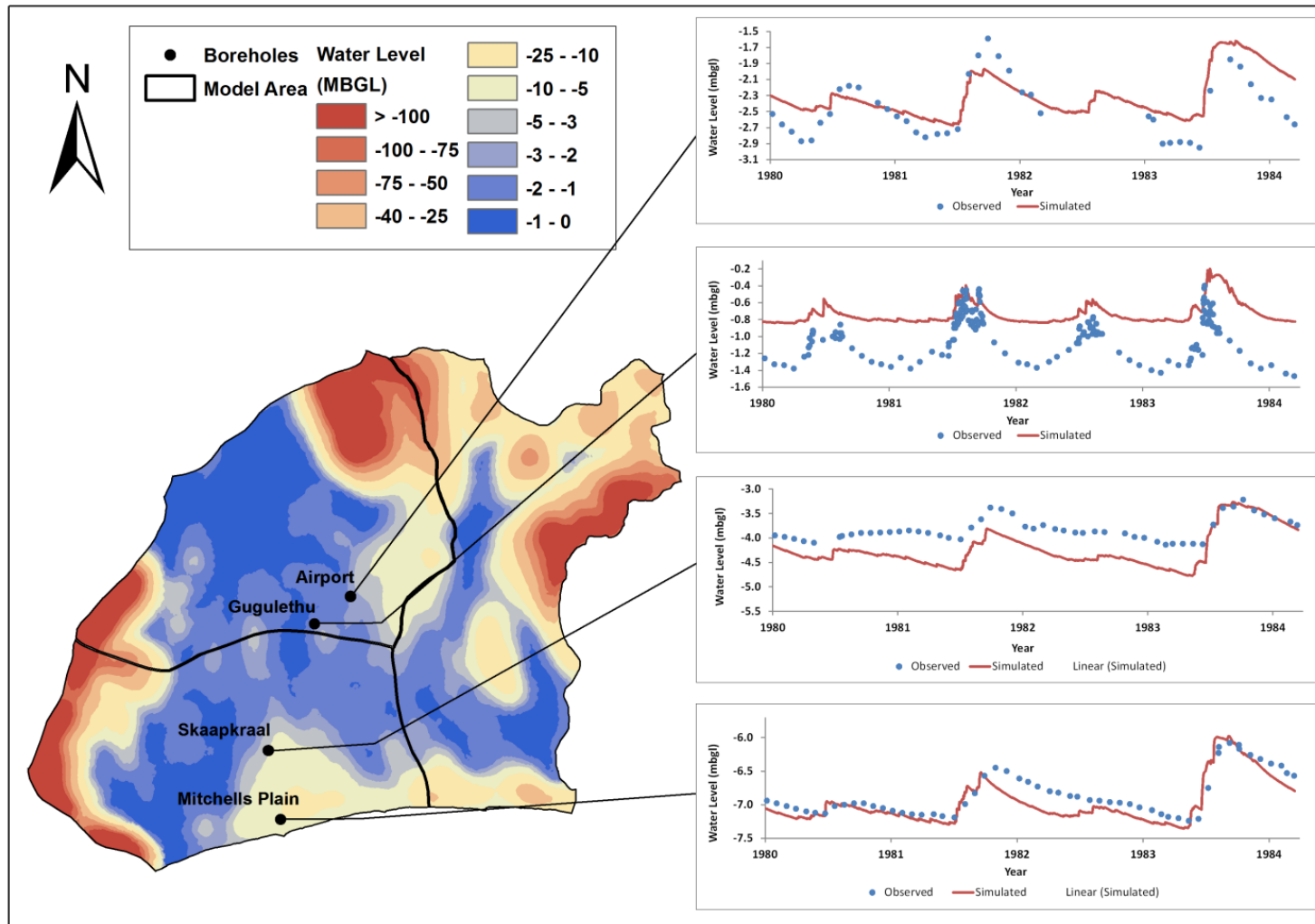


Figure 3.13: Simulated groundwater levels (mbgl) for the Cape Flats and the simulated vs. observed groundwater levels for the Cape Town Airport, Gugulethu, Skaapkraal, and Mitchells Plain from 1980 - 1984

The simulated mean groundwater head elevations were compared to mean observed groundwater head elevations over the Cape Flats (Figure 3.14). The observed groundwater elevation dataset was based on data that were used in a previous study by DWAF (2008). The dataset was estimated from point measurements and averaged groundwater elevations from monitoring boreholes extracted from the National Groundwater Archive (NGA). The majority of the data points were located in the most productive areas of the Cape Flats, in the central and southern parts of the modelling domain. Thus, only these areas were used in the model calibration, as it was assumed that they are the most accurate due to the higher density of data points and that further interpolation beyond this region would be inaccurate.

The simulated and observed mean groundwater elevations are visually similar (Figure 3.14). The groundwater elevation in both maps follows the topographic trends showing lower groundwater elevation in the low-lying areas in the north west, south west and south eastern parts of the calibration area, while the groundwater elevation increased from the central region to the north eastern parts of the calibration area. The low-lying areas were typically associated with wetland or coastal conditions while the areas with higher head elevations are associated with increasing topographical elevation towards the north east. Two elevation profiles (Transect 1 and Transect 2) were extracted to further illustrate these trends and compared to the simulated and observed groundwater elevations (Figure 3.15). Transect 1 showed the change in head elevation from the north east to the south west and Transect 2 showed the elevation change from the north to the south. The simulated groundwater elevation corresponds to the observed groundwater elevation for both Transect 1 and 2, with mean error values of -1.6 m and -0.4 m respectively. Comparing the spatially distributed mean simulated and observed groundwater elevation is useful for assessing the model's ability to capture the spatial trends.

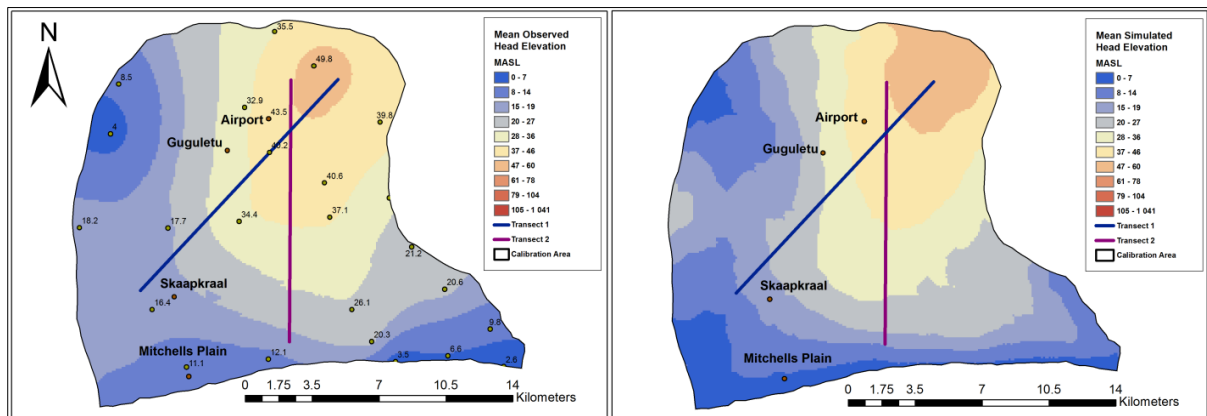


Figure 3.14: The mean observed groundwater head elevation (left) compared to the mean simulated groundwater head elevation (right) for the CFA

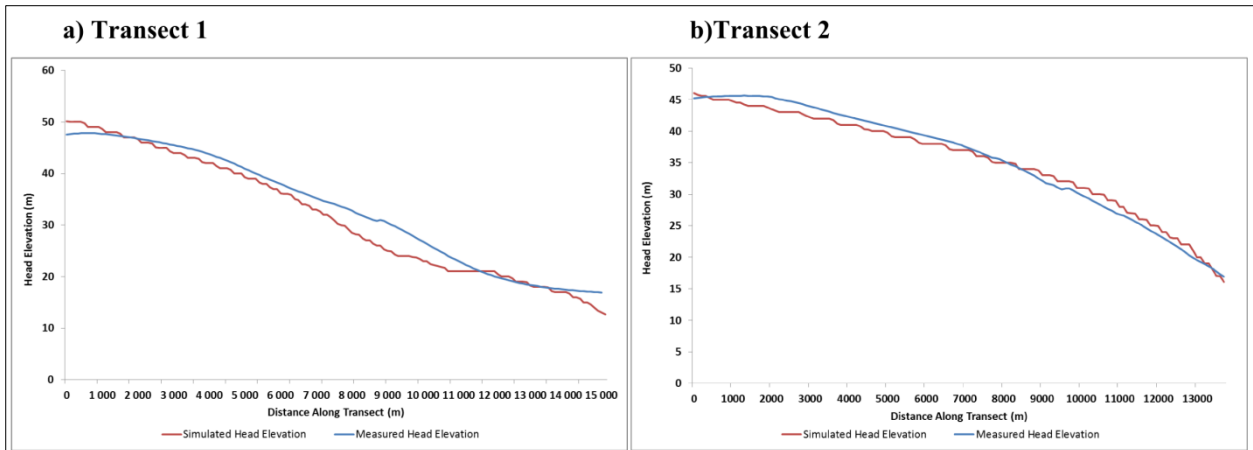


Figure 3.15: The simulated groundwater head elevation plotted against observed groundwater head elevation for a) Transect 1 and b) Transect 2

3.3.2 Surface water calibration

The calibration of surface water processes such as streamflow were important for the integrated hydrological modelling to ensure that the complete hydrological cycle was simulated reasonably, that water was partitioned realistically within the model and that surface and groundwater assumptions were consistent during the simulation. However, streamflow data for the Cape Flats during the calibration period was limited with only one active gauging station in the upper reaches of the Kuils river. Furthermore, the quality of this data was questionable, as the data does not correlate well to the rainfall data due to incorrect dating resulting in a lag of two days between the rainfall event and peak discharge. The peak flows are also inconsistent over the calibration period. During 1983 and 1984 there were a number of rainfall events in excess of $40 \text{ mm}\cdot\text{day}^{-1}$ resulting in mean daily discharge rates of $>8 \text{ m}^3\cdot\text{s}^{-1}$. However, prior to 1983 the mean daily discharge seldom exceeded $4 \text{ m}^3\cdot\text{s}^{-1}$ for rainfall events $>40 \text{ mm}\cdot\text{day}^{-1}$ (Figure 3.16). Despite the limitations of the streamflow data it was useful in the calibration process.

Streamflow was sensitive to a number of model parameters that were used for calibration *viz.* the paved area coefficient, channel roughness (Manning's M) and the groundwater leakage coefficient. The paved area coefficient was the most important variable used to represent urban land use in the MIKE SHE model, thus streamflow was sensitive to changes in the paved area coefficient. There are two main types of urban land use that are used in this model setup, Urban and Residential. The best calibration was obtained using paved area coefficients of 0.6 for Urban and 0.3 for Residential. The Manning's n value showed little impact on discharge and thus was set to 0.014, which is a typical value for a concrete canal. Discharge, particularly during periods of low flow, was sensitive to the groundwater leakage coefficient. The groundwater leakage coefficient was calibrated to 3×10^{-5} . The calibration of streamflow discharge for 1980 – 1984 displayed a poor R^2 of 0.55, but the R^2 improved, at 0.58, for the years 1983 – 1984 and again at 0.65 when only the year 1984 was considered.

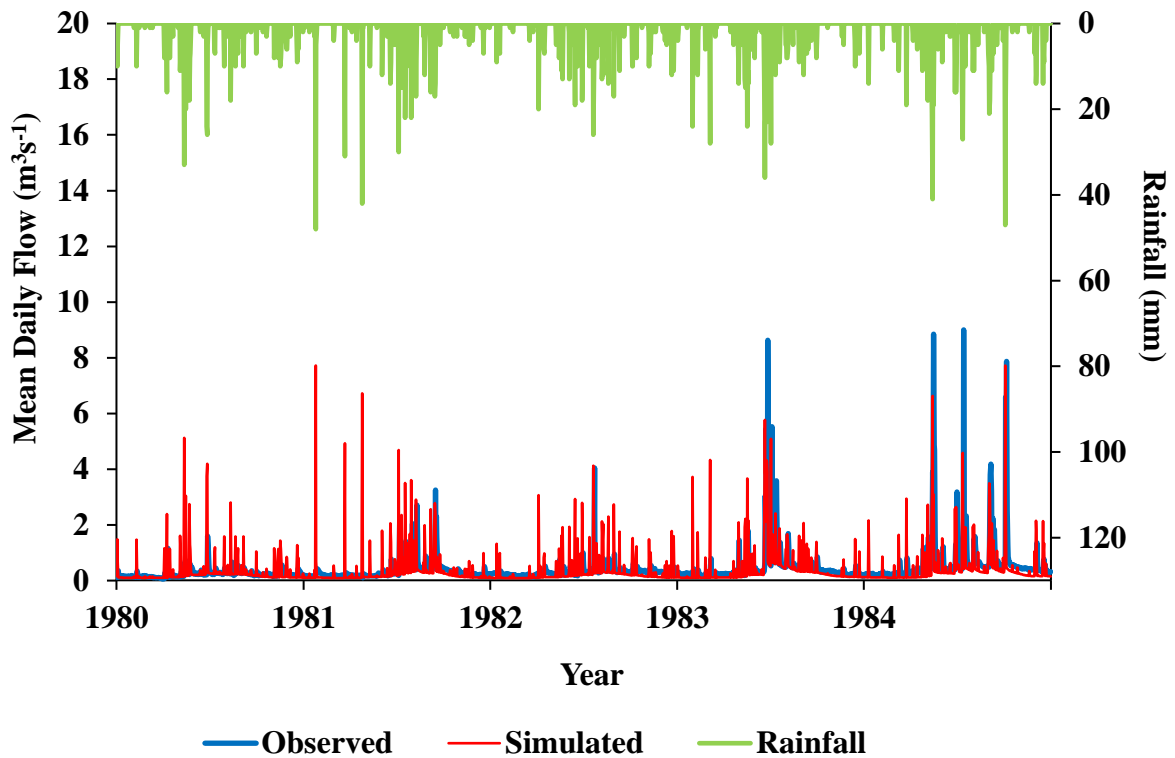


Figure 3.16. Simulated and observed mean daily streamflow discharge for the upper reaches of the Kuils river

3.3.3 Water balance

The water balance for the calibration period was used to evaluate the model's performance (Figure 3.17). The water balance gives an indication of how water is partitioned in the model and this can be evaluated based on local knowledge, literature values and previous modelling results. The total rainfall for the calibration period was 3112 mm of which 1748 mm (56%) was lost to evapotranspiration, 981 mm (32%) was recharged to groundwater, a further 368 mm (12%) was discharged by rivers. The simulated recharge was in line with estimates by Vandoolaeghe (1989). Vandoolaeghe (1989) estimated that the range of recharge for the Cape Flats varied between 15 – 37% of the annual rainfall and that the average recharge was 33% of annual rainfall. Furthermore, the total streamflow discharge was also within the ranges estimated by the WR90 report for the G22C (14%), G22D (15%) and G22E (11.5%) (Midgley *et al.*, 1994). The close correspondence of these water balance components indicates that the model's performance is consistent with past estimates of groundwater recharge and streamflow discharges and suggested that model's representation of surface and groundwater processes were reasonable.

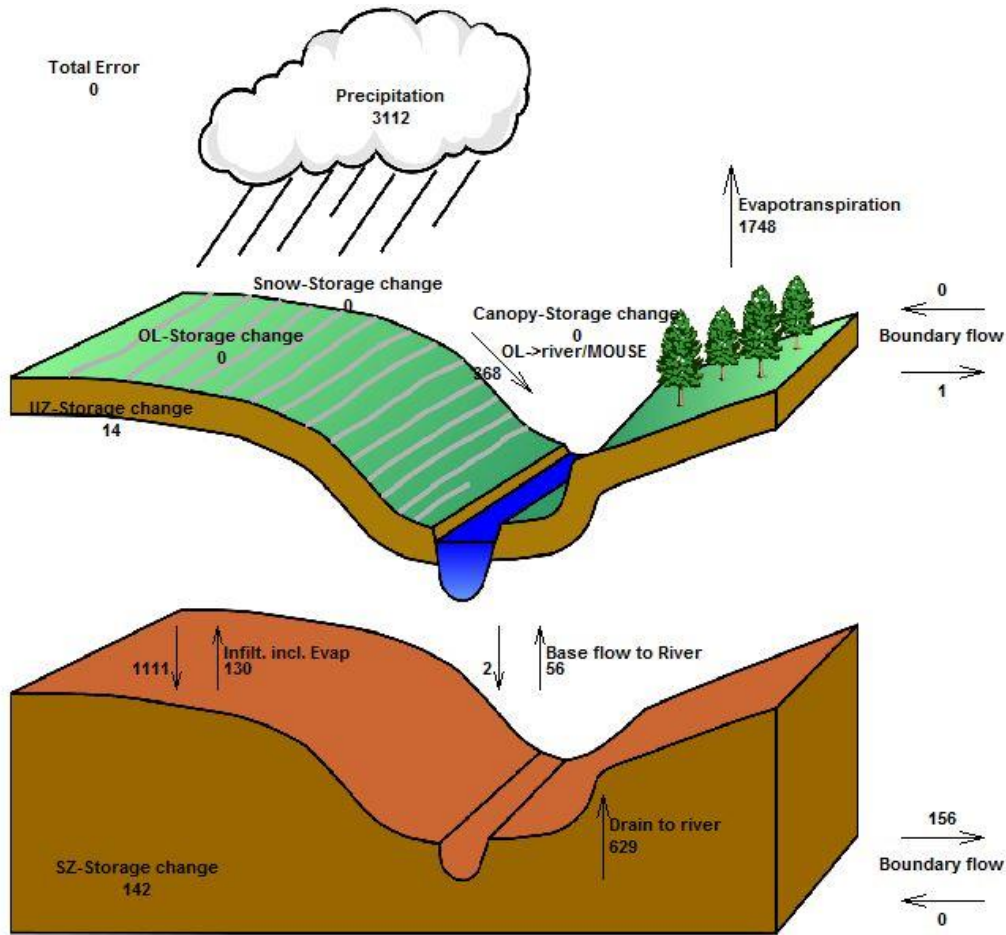


Figure 3.17: MIKE SHE water balance output (mm) for the calibration period: 1980 to 1984

3.4 Model validation

MIKE SHE was validated to a different dataset at a later period between the 1st of January 2000 and the 31st of July 2015. It is important to note that due to the different period used in the model validation, different rainfall and potential evapotranspiration series were used. Other changes to the model for validation included land use and paved areas, as these were updated for land use information from the year 2013. Irrigation was also incorporated for the agricultural areas overlying the PHA. No changes were made to neither the conceptual model informing the structure of the unsaturated and saturated zones within the numerical model nor to the model parameters describing groundwater and surface water processes.

Due to a lack of accurate streamflow data, the surface water component of the MIKE SHE model could not be included in the validation. Groundwater level data was available for 22 boreholes on the Cape Flats, which were collected by the DWS at monthly intervals. Though, a majority of these boreholes show intense variations in groundwater level due to the proximity of local pumping. Thus, these boreholes cannot give a reasonable representation of the natural fluctuations of groundwater level, as data could not be incorporated into MIKE SHE that account for the fluctuations in groundwater levels due to pumping. The available data on groundwater abstractions are limited to licencing information from the DWS in the form of the WARMS

database, which effectively gives an indication of the maximum yearly water volume available to a farm or industry, offering little information regarding the actual pumping rates of farms or industry in proximity to the observation borehole. Thus, the calibrated MIKE SHE model was validated using groundwater level data obtained from two boreholes: G32961 and G32973. The lengths of these datasets were more extensive than those that were available during the calibration period. BH G32961 extends almost 12 years, collected from 2004 to 2015, while BH G32973 extends for almost 9 years from 2007 to 2015. Both boreholes show ‘natural’ variations in groundwater levels showing seasonal variation by remaining within an expected 1 m range of variation over the entire extent of the dataset; a similar trend was seen during the calibration period where the anthropogenic influences on the hydrogeology were greatly reduced.

A visual comparison of the simulated and observed groundwater levels for BH G32961 (Figure 3.18) and BH G32973 (Figure 3.19) showed a good fit. The simulation showed similar seasonal responses to recharge and rates of decline as the groundwater level drops during dryer periods. Assessing the coefficient of determination (R^2) of the simulation compared to the observations supports the visual assessment. For BH G3296, an R^2 value of 0.73 indicated that the simulated results showed a good fit with the observed data. BH G32973 showed an excellent fit to the observed data with an R^2 value of 0.89. The RMSE for the correlation between simulated results and observed data for BH G32961 is 0.19 m and 0.17 m for BH G32973. This indicates that mean error between each simulated and observed data point is 17 % of the maximum and minimum groundwater levels for BH G32961 and 15 % for that of BH G32973. These values indicate a good representation of the model over a relatively extensive period incorporating a good seasonal contrast between wet and dry years. These results are also good given the relatively large model domains used for the calibration and validation of MIKE SHE.

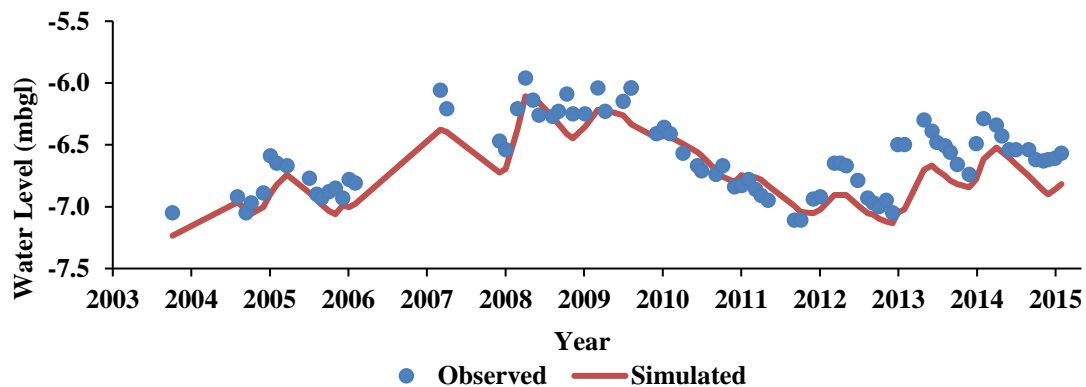


Figure 3.18: Simulated groundwater levels (meters below ground level) for the Cape Flats and the simulated vs. observed groundwater levels for BH G32961 from 2004 – 2015

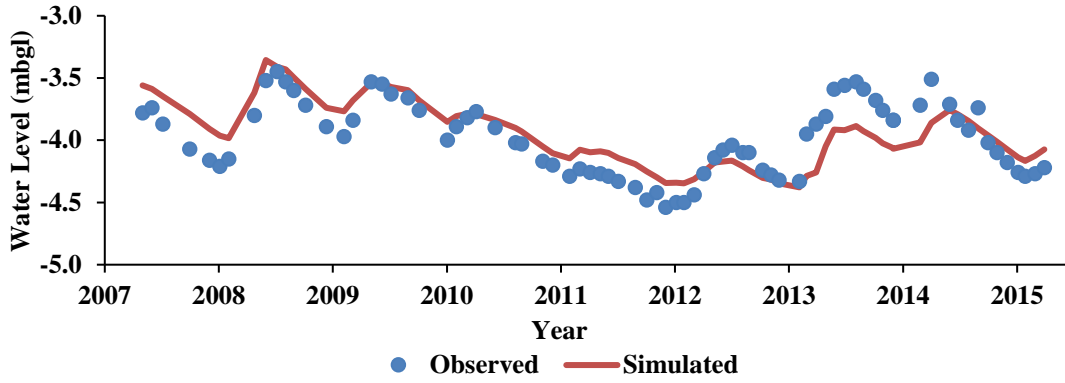


Figure 3.19: Simulated groundwater levels (meters below ground level) for the Cape Flats and the simulated vs. observed groundwater levels for BH G32973 from 2007 – 2015

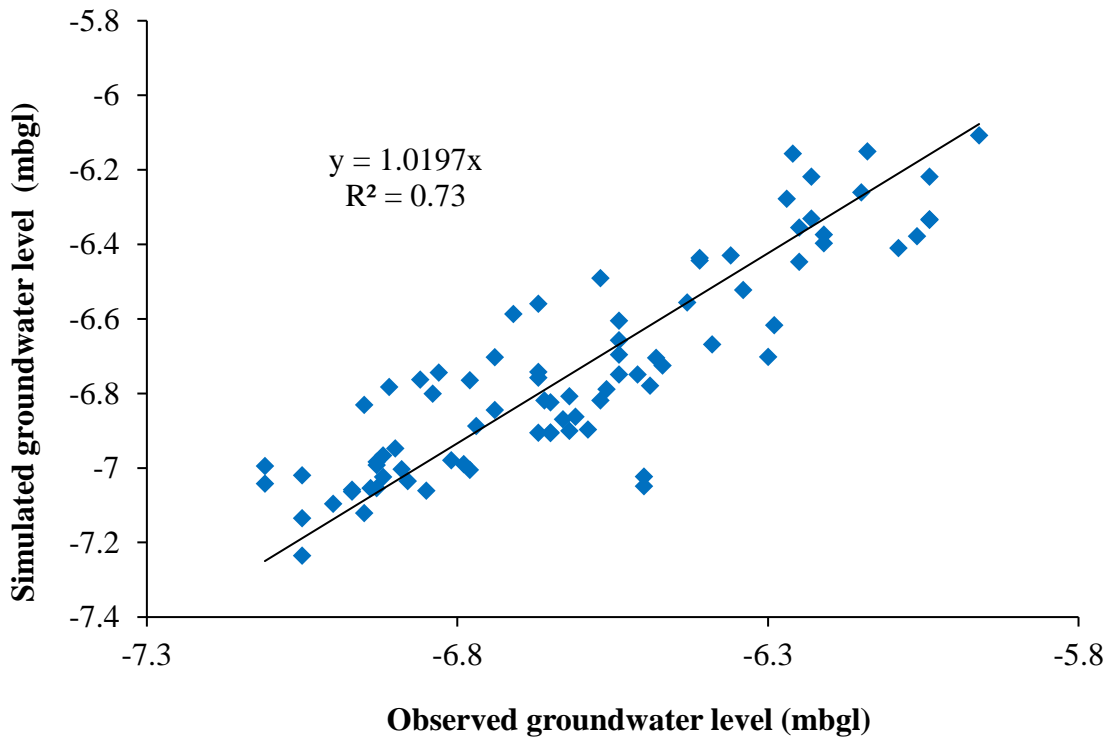


Figure 3.20: Simulated vs observed groundwater levels for BH G32961

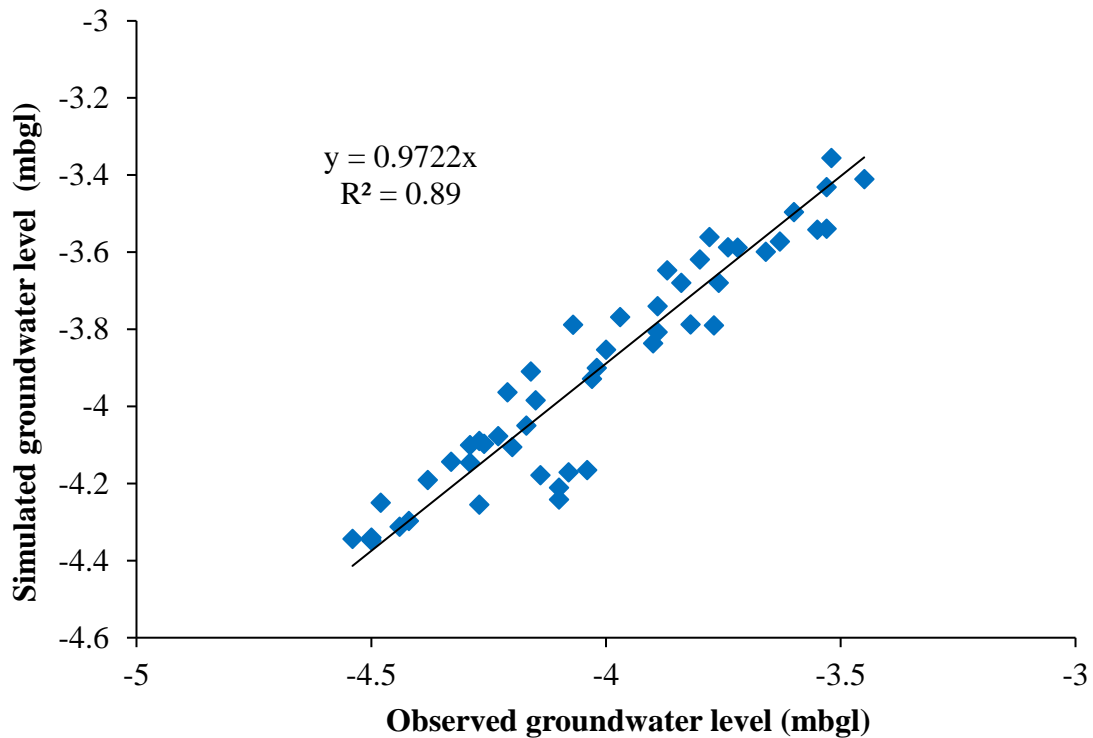


Figure 3.21: Simulated vs observed groundwater levels for BH G32973

3.5 Overall model performance

The water balance, together with the calibrated groundwater levels, head elevations and streamflow discharge, demonstrated that the model was able to reasonably represent the groundwater and surface water processes of the Cape Flats at a regional-scale during the calibration period. The results of the model validation indicated that the calibration parameters and conceptual model used to develop the MIKE SHE model were valid using different input data and during a later period. The model performance demonstrated the value of the integrated hydrological modelling approach for urban applications, by showing sensitivity to urban land use and its ability to simulate surface and groundwater processes and their interaction. The integrated hydrological modelling approach was particularly valuable as recharge was represented in detail and was simulated at a daily time step and in a spatially distributed manner. This was particularly important on the Cape Flats as groundwater recharge and discharge vary dramatically, with some areas associated with both recharge and discharge depending on the time of year. Urban areas also determine the amount and spatial distribution of recharge, therefore the spatial representation of urban land use in an integrated hydrological model such as MIKE SHE is essential for simulating recharge in urban areas. MIKE SHE was able to produce calibrated and validated results that were captured from limited available data. Based on the calibration and validation results, the MIKE SHE model setup for regional- and local-scales was deemed sufficient to investigate that application of MAR scenarios on the CFA.

3.6 MAR site selection for local-scale modelling

The benefit of regional-scale modelling using an integrated hydrological model is that both groundwater and surface water processes can be simulated. This is particularly important as groundwater level and groundwater head elevation are important factors for determining the feasibility of MAR. Additionally, due to the representation of surface water within the model, further useful information can be obtained from the model such as groundwater recharge. Groundwater recharge is important as it gives an indication of natural or current recharge for a particular site and where a MAR intervention would be most effective.

3.6.1 Selection process using regional-scale model outputs

The site selection process for identifying areas suitable for testing MAR at a local-scale on the CFA required information from a number of sources, including aquifer information from hydrogeological surveys and regional hydrogeological modelling results. Since this study aims to help mitigate flooding it was also important to consider where flooding occurs on the Cape Flats. The results from the regional-scale MIKE SHE model provided the main basis for the selection of local-scale sites for testing MAR on the CFA. The most important model results were the mapped groundwater levels, groundwater head elevation and recharge. The most significant limiting factor to MAR was finding locations where there was significant storage capacity within the aquifer for infiltrated or injected stormwater. By identifying locations where the groundwater level was at its greatest depth below the ground surface, it was assumed these areas are likely to have sufficient capacity for additional recharge. Figure 3.22 shows those areas with the low groundwater levels as yellow and orange, whereas those areas with high groundwater levels are indicated in blue. The lowest levels on the Cape Flats occur in the North East and in the South near Philippi and Mitchells Plain. In terms of flooding on the Cape Flats, there are a number of low-income areas that are at risk of seasonal winter flooding, *viz.* Sweet Home, Kosovo, Phola Park and Kanana are listed by the City of Cape Town as areas that are affected most by seasonal winter flooding (Pharoah, 2013). All these suburbs fall within areas that are associated with high mean groundwater levels indicated by the blue areas in Figure 3.22.

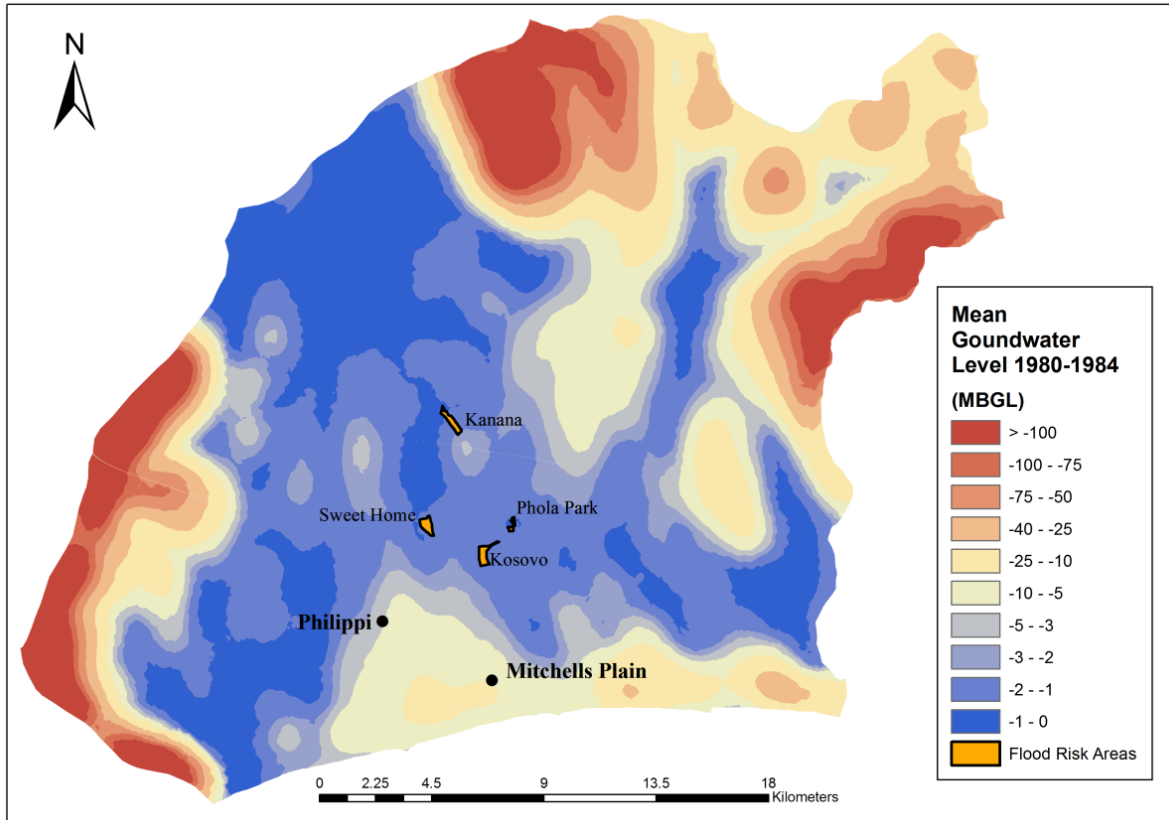


Figure 3.22: Simulated mean groundwater levels (mbgl) for the Cape Flats from 1980-1984

The hydraulic gradient must also be considered for MAR interventions as the hydraulic gradient determines the flow of groundwater. The hydraulic gradient of the CFA (Figure 3.23) showed a decreasing trend from north to south and east to west, with the highest groundwater elevation found in the north east. This means that the aquifer to the north east of those suburbs affected by flooding were not ideal for stormwater MAR as they are ‘down gradient’ of the areas where storage could be maximised. However, the area to the South of the flood risk areas between Philippi and Mitchells Plain hold more potential as they are ‘down gradient’ and also have the required storage capacity.

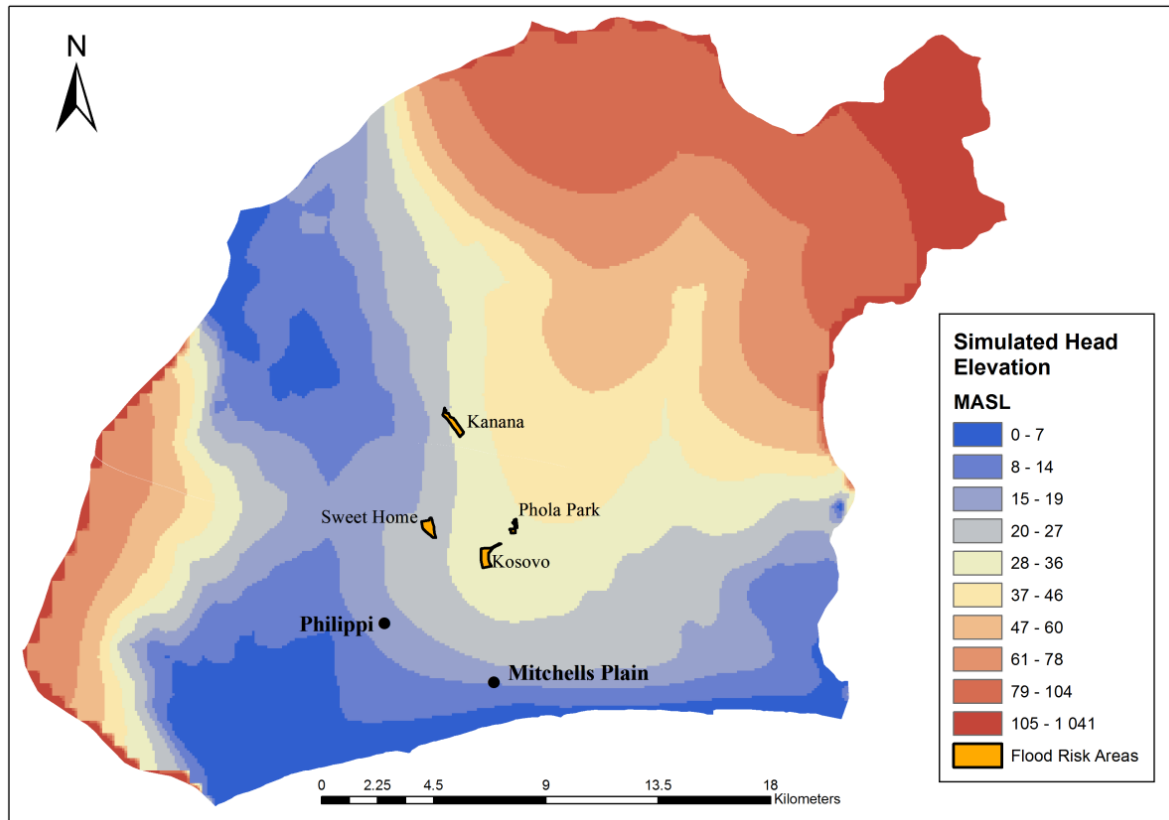


Figure 3.23: Simulated mean groundwater head elevation (meters below ground level) for the Cape Flats from 1980-1984

3.6.2 Mapping MAR potential

The regional-scale model output described previously, allowed for the MAR potential of the CFA to be determined using the simulated mean head elevation for the period from 2000 – 2015. The method demonstrated by Murray *et al.* (2007) was used to calculate the MAR potential of the CFA (Figure 3.24). This method assumes that MAR is most suitable in an aquifer where there is an additional storage volume due to lack of natural recharge. The MAR Potential was calculated as half the difference between the “top of the aquifer” and five meters below the mean groundwater level. The top of the aquifer was the theoretical maximum groundwater level after which additional recharge was lost as drainage. The “top of the aquifer” was derived by simulating the groundwater levels using an artificially high rainfall dataset. This dataset involved doubling the daily rainfall for 2007, which was the wettest year between 2000 and 2015 and running the model with that data for 15 years. The mean groundwater levels were calculated using the 15-year MIKE SHE simulation of groundwater levels from 2000 to 2015 under baseline conditions.

The simulated MAR potential was highest in those areas that have the highest topographic elevation, such as Table Mountain in the west and the hills of Tyger-valley and Kuilsriver in the north and northeast. However, these areas are associated with aquifer conditions beyond the scope of this study, as they are fractured aquifers and the MIKE SHE model has not been optimised for these conditions. Thus, the main interest for MAR mapping is within the primary aquifer of the Cape Flats Quaternary sands, the aquifer unit for which this MIKE SHE model was calibrated and validated. The results of this mapping indicate that the highest MAR potential exists in the

Southern parts of the CFA, showing between $75\,000\text{ m}^3\cdot\text{ha}^{-1}$ per year to over $150\,000\text{ m}^3\cdot\text{ha}^{-1}$ per year. These are areas where the aquifer was at its thickest due to the deeper sections of the bedrock topography and of higher topographic elevation. The Philippi MAR site exhibits a particularly large storage volume available for MAR due to the presence of a paleochannel that runs in a north – south direction below much of the central parts of the CFA. This depression in the bedrock topography together with the increasing elevation of surface topography towards the False Bay coastline marks the thickest parts of the CFA (approximately 55 m). The areas in red have a MAR potential of less than $75\,000\text{ m}^3\cdot\text{ha}^{-1}$ per year.

It is important to note that the method used by Murray *et al.* (2007) was developed to assess MAR potential on a national scale and thus is a conservative estimate of MAR Potential. It is anticipated that the upper recharge limit – half the difference between the “top of the aquifer” and five meters below the mean groundwater level – and lower drawdown limit of 5 m can be safely extended on the CFA. Thus, the following section examines the areas of the CFA with the highest MAR potential by assessing the upper recharge limits and the potential safe drawdown limits.

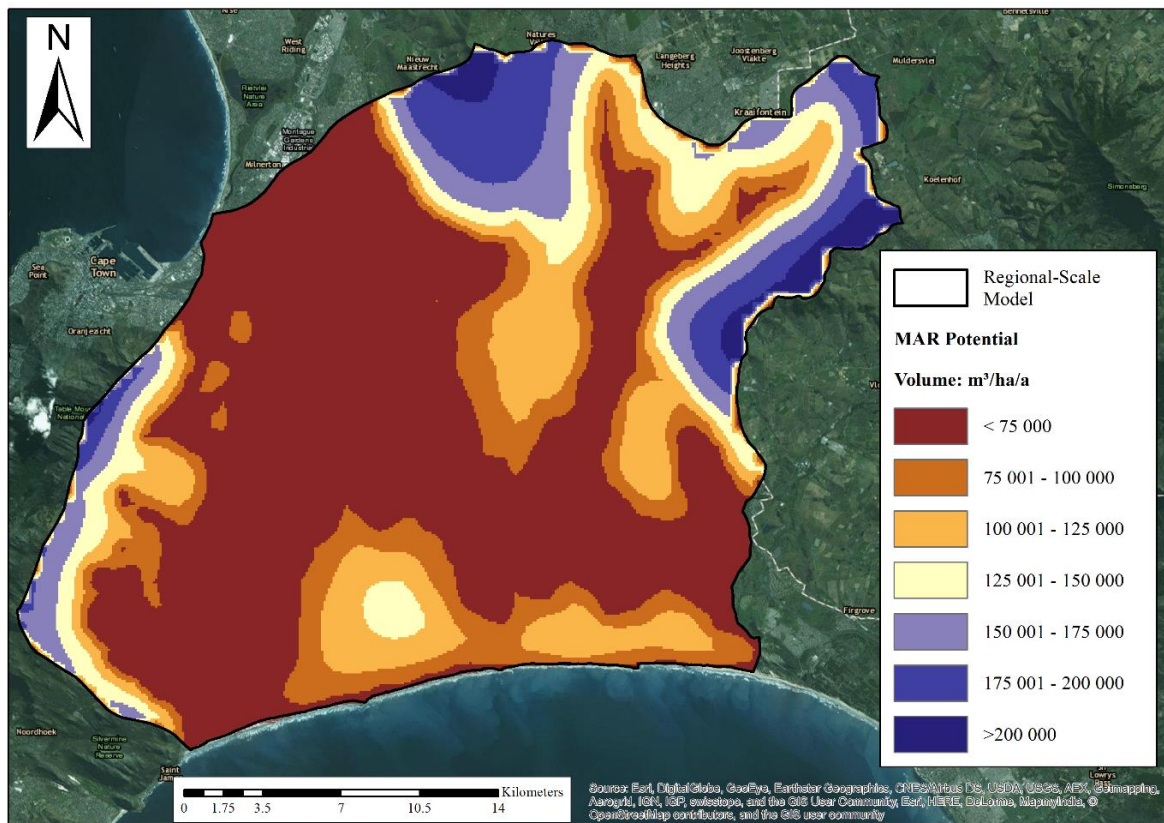


Figure 3.24: MAR potential for the Cape Flats Aquifer

3.6.3 Recharge potential

Further support for the selection of a site near Philippi and Mitchells Plain was the fact that the recharge in this area is high ($0.5 - 1 \text{ mm.day}^{-1}$) or very high ($>1 \text{ mm.day}^{-1}$) (Figure 3.25). This means that there was sufficient available land for recharge to take place and that this location has the required soils for infiltration of stormwater to recharge the aquifer. Additionally, the area around Philippi and Mitchells Plain has been studied extensively in the past and much of the hydrogeological investigations performed in the late 1970's and early 1980's were concentrated around Mitchells Plain and Philippi (Henzen, 1973; Gerber, 1980; Tredoux, 1980). The CFA around Mitchells Plain and Philippi is known to produce relatively high borehole yields ($> 5 \text{ l.s}^{-1}$) due to the high hydraulic conductivity and transmissivity values (Gerber, 1980). Furthermore, Tredoux *et al.* (1980) tested the concept of wastewater recycling on the CFA applying MAR principles. The study showed many promising aspects in terms of the infiltration and reclamation of treated wastewater in the CFA. However, further research on the CFA lost momentum as attention was shifted onto the Atlantis aquifer, a MAR project initiated at a similar time to that of the CFA.

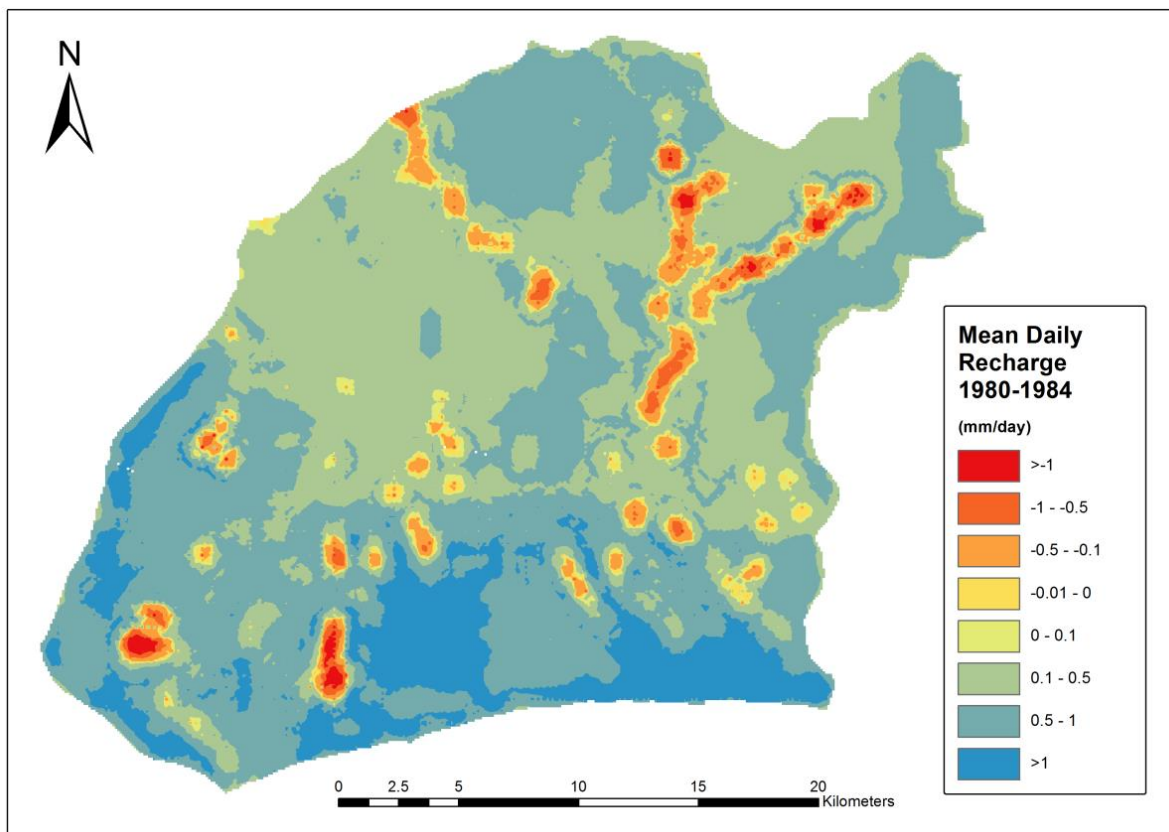


Figure 3.25: Simulated mean recharge/discharge (or net water flux) for the Cape Flats from 1980-1984

3.6.4 Site Selection: Summary

Based on the information from the regional-scale modelling using MIKE SHE, additional information from literature and hydrogeological surveys of the CFA and the proximity to areas that have a high risk of urban flooding, the region around Philippi and Mitchells Plain was evaluated as the most suitable for local-scale flood mitigation and MAR simulations. At the most northern boundary, the local-scale model area focuses on the informal residential suburbs of Sweet Home and Graveyard Pond, as these areas are prone to winter flooding linked to elevated groundwater tables. The areas in the south of the study site are the parts of the CFA that have been evaluated as suitable for MAR based on the aquifer's storage capacity and high hydraulic conductivity.

3.7 Water quality

Groundwater is under increasing threat of contamination as many of the contaminants released into nature ultimately percolate through the unsaturated zone to contaminate groundwater and continue to migrate along its flow paths (Patil and Chore, 2014). According to the Environmental Protection Agency (2007 p. 14.3), the primary purpose of modelling the quality of groundwater is to “...simulate the movement and chemical alteration of contaminants as they move through the subsurface”. The models used to simulate groundwater quality are also called contaminant fate or transport models. A calibrated flow model is required for the simulation of contaminant transport, as the magnitude and direction of the flow vectors determine the rate and direction of contaminant transport. Thus, groundwater flow models typically have an optional water quality module for simulating the fate or transport of solutes (EPA, 2007). The EPA (2007) suggest that most water quality models aim to simulate the following:

- The movement of contaminants (advection and dispersion),
- The sorption or desorption of contaminants to rock and soil particles in the subsurface, and
- Changes to the chemical composition of the contaminant due to biological or physical processes, or by chemical reactions.

Advection is the movement of a dissolved solute through the porous media of an aquifer that is influenced by the flow of the groundwater. Thus, advection is governed by Darcy's law which states that the flow rate of water through a porous media from one point to another is proportional to the head loss and inversely proportional to the length of the flow path. This means that if advection was the only consideration, the flow of dissolved solutes would be equivalent to the groundwater flow. However, in a real-world situation the transport of solutes is less than the extent of the groundwater flow due to dispersion and retardation. Dispersion is the dilution of the dissolved solute, due to diffusion and mechanical mixing. Diffusion involves the movement of solutes from a higher concentration to a lower concentration thereby lowering the concentration of the contaminant plume. While mechanical mixing is the simpler process where the concentration of the contaminant plume is reduced as it moves through and mixes with the surrounding uncontaminated water (Patil and Chore, 2014). Sorption describes the process of ion and molecule exchange between the solid and liquid phase. Adsorption is the process where the molecules and ions of the solute adhere to the particles of the porous media in the solid phase

thereby lowering the concentration. This process is also known as retardation. The reverse of this process is desorption where the molecules and ions in the solid phase transfer back to the liquid increasing the concentration (Patil and Chore, 2014).

3.7.1 Modelling Water Quality

MIKE SHE has two methods for the simulation of the fate and transport of solutes, the Advection-Dispersion (AD) method and Particle Tracking (PT). The AD method is the more traditional approach using the groundwater flow model to describe the exchange of solutes from sources and sinks based on fluxes as a result of advection and retardation because of dispersion. PT is an alternative to the AD method that calculates the position of a particle at each time-step of the simulation based on the local flow vectors and dispersion coefficients. PT is mainly used to delineate capture zones for borehole or well-points or determining the age and solute transport time (DHI, 2014a, 2014b).

The basic water quality simulations conducted in this study were to assess the flux of potential contaminants from areas in proximity to poor quality stormwater such as the informal settlements of Sweet Home and Graveyard Pond, and at sites where stormwater is being considered for injection and subsurface storage. The concentration of Total Dissolved Solids (TDS) was used to evaluate the risk of pollution migration. The focus was on stormwater from informal settlements as these are the areas that are most affected by winter flooding and the people in these areas are the most vulnerable to the impacts of flooding. There is little contemporary information on the quality of stormwater from informal settlements on the Cape Flats. Thus, a study on the quality of stormwater performed in the early 1990's by Wright *et al.* (1993) was used to guide the estimation of daily TDS concentrations for the water quality simulation. This study found that the typical TDS concentration of stormflow from the Khayelitsha informal settlement was $1472 \text{ mg}\cdot\text{l}^{-1}$ or $1.48 \times 10^9 \text{ }\mu\text{g}\cdot\text{m}^{-3}$. In order to construct a daily TDS concentration dataset, the average monthly TDS concentration was estimated by assigning stormflow concentration of $1.48 \times 10^9 \text{ }\mu\text{g}\cdot\text{m}^{-3}$ for every day with rainfall in a given month. The average monthly TDS concentration was calculated for each month based on rainfall from 2000 – 2015 and divided by the number of days in the month to get the daily TDS concentration. The same time series dataset was used for a flood mitigation scenario and a MAR scenario. The flood mitigation scenario simulated the migration of a TDS plume from Sweet Home and Graveyard Pond until the year 2100. In this scenario, the stormwater with a high TDS concentrations was applied to the soil surface to represent pollutant sources that typically occur at the soil surface such as on-site sanitation and contaminated runoff. In the MAR scenario, the migration of a TDS plume was simulated from Philippi and Mitchells Plain until the year 2100. In this case, the contaminated stormwater was applied in the subsurface to represent the direct injection of poor quality stormwater.

3.8 Climate change

The global community is experiencing an increasing urgency to address the issues of climate change and the level of global CO₂ emissions. The evidence of which is noted in the signing of the Paris Agreement which is regarded as a landmark achievement aimed at keeping global warming to well below 2° C (Dimitrov, 2016; Rogelj *et al.*, 2016). There has been extensive research focused on understanding the impacts of climate change on issues such as food security (Gregory *et al.*, 2005), health (Comrie, 2007), infrastructure (Gill *et al.*, 1978; Hallegatte, 2009), ecosystem services and biodiversity (Grimm *et al.*, 2008). South Africa is particularly vulnerable to climate change because of the high rates of poverty and inequality (Ziervogel *et al.*, 2014). Thus there is a significant emphasis on adaptation to climate change in South African research (Ziervogel *et al.*, 2014; Stuart-Hill and Schulze, 2015; Vogel *et al.*, 2015). In South Africa the effects of climate change are anticipated to be more difficult to detect due to the variable climatic conditions. Furthermore, it is anticipated that climate change is likely to increase the variability of the climate in South Africa (Schulze, 2000).

In the field of hydrology, modelling is an essential tool for assessing the impacts of climate change on the hydrological cycle. In order to simulate the impacts of climate change, climate scenarios from Global Circulation Models (GCMs) are required. The latest climate scenarios are called Representative Concentration Pathways (RCP), which describe a future climate based on anticipated long-term energy demands and greenhouse gas (GHG) emissions, based on factors such as population size, economic activity, lifestyle, energy demand, technology and the adoption of climate policy (IPCC, 2014). There are four RCPs that represent atmospheric CO₂ concentrations that result in four levels of radiative forcing: 2.6, 4.5, 6.0 and 8.5 W.m⁻² by the year 2100 (Riahi *et al.*, 2011; IPCC, 2014). RCP 2.6 scenario represents the lowest climate change impact aimed at keeping the increase in global temperatures at less than 2° C. In order for this to be achieved international climate change policies and commitments need to be extensively adhered to. RCP 4.5 and 6.0 are both intermediate scenarios. While, the RCP 8.5 scenario is anticipated to be the ‘worst case’ scenario, based on the assumption that the proposed policies for the mitigation of GHG emissions are poorly adopted and implemented resulting in a considerably higher radiative forcing. Furthermore, the current CO₂ emissions remain above the RCP 8.5 projection indicating that if emissions are not mitigated through the adoption of global climate treaties, the RCP 8.5 is likely to be a realistic representation of this future (Sanford *et al.*, 2014). Figure 3.26 shows the projections of the four RCP scenarios simulated by a number of GCM as part of the Coupled Model Intercomparison Project Phase 5 (CMIP5). The blue line indicates the RCP 2.6 and the red line denotes the RCP 8.5, with the shaded areas representing the associated uncertainty. The graphs indicate that RCP 2.6 is likely to have greatly reduced temperatures and sea-level rises when compared to RCP 8.5 which predicts global mean temperature to increase by approximately 4° C by the year 2100 and sea-levels to rise by just less than 0.75 m (Figure 3.26).

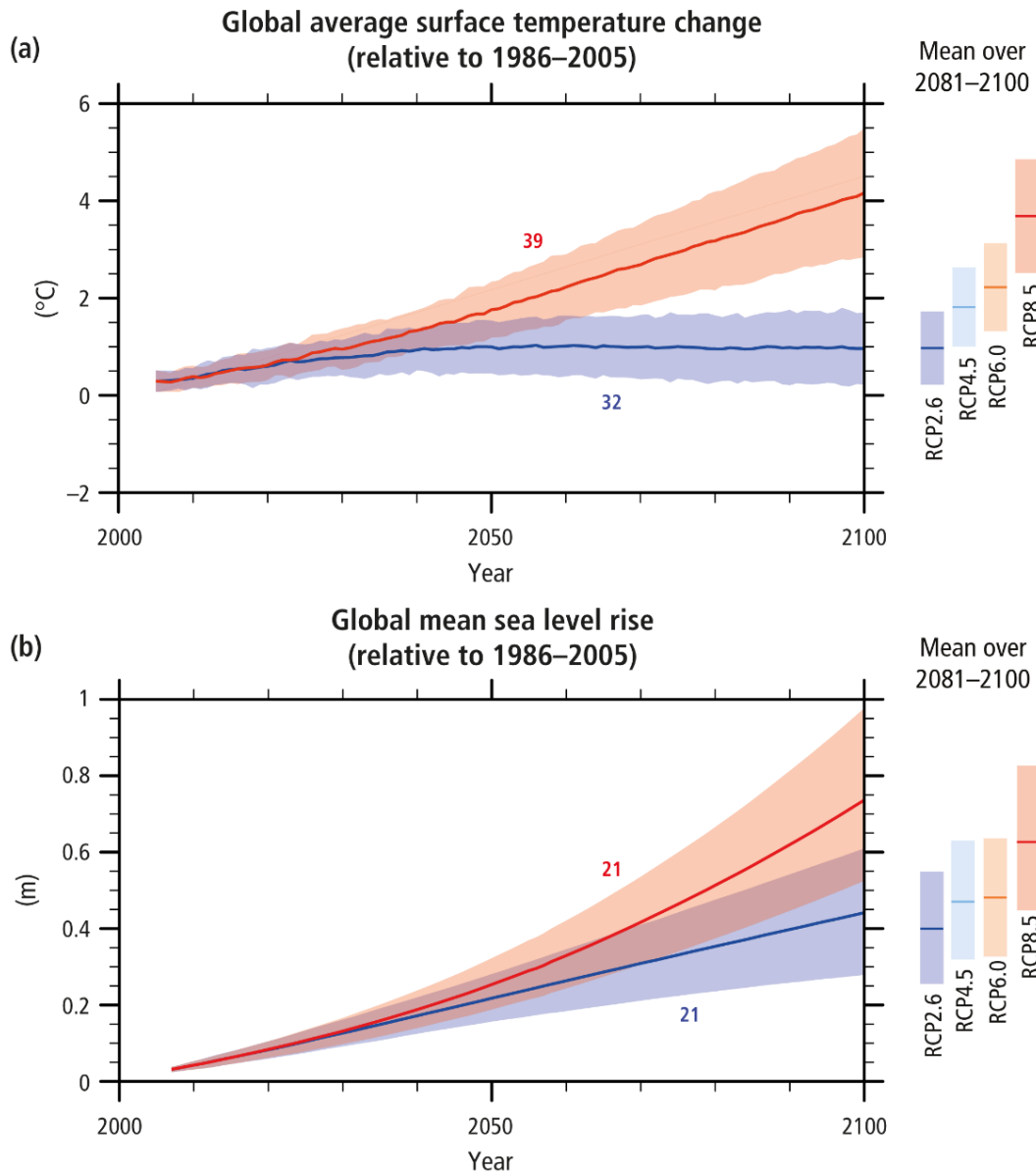


Figure 3.26: The a) global average surface temperature and b) global mean sea-level rise, projected using the Coupled Model Intercomparison Project Phase 5 (CMIP5) models used to calculate the multi-model mean is for RCP 2.6, 4.5, 6.0 and 8.5 (IPCC, 2014)

The information from the GCM models (i.e. CMIP5) can then be incorporated into the hydrological model through the delta change factor method, which indicates how much a certain climate variable will change over time when compared to a reference or baseline period (DHI, 2016). The delta change factor method is a relatively simple, linear method for calculating the future climate condition based on monthly data. However, this method is not well suited for datasets that do not have a normal distribution (i.e. daily rainfall) or for use with extreme values (Trzaska and Schnarr, 2014). For this reason, the climate change analysis in this study was not conducted at a daily time step, but assessed the impacts of climate change at a monthly time step for rainfall and ET_0 . The hydrological modelling of anticipated climate change is discussed in the following section.

3.8.1 Modelling climate change

Climate change modelling was performed to assess the impacts of projected future climate change on the water balance of the hydrological cycle. This climate change scenario analysis utilised the Representative Concentration pathways (RCP) 8.5, which describes a future climate that with high long-term energy demands and greenhouse gas emissions that result in a radiative forcing of 8.5 W.m⁻² by the year 2100 (Riahi *et al.*, 2011). Therefore, this climate scenario was selected as an anticipated ‘worst case’ scenario, where it is assumed that the proposed policies for the mitigation of greenhouse gas emissions are poorly adopted and implemented resulting in considerably higher global temperatures. Furthermore, the current CO₂ emissions remain above the RCP 8.5 projection indicating that if emissions are not mitigated through the adoption of global climate treaties, the RCP 8.5 is likely to be a realistic representation of this future (Sanford *et al.*, 2014). In total data from 16 Global Circulation Models (GCMs) for the City of Cape Town were sourced from the Climate Systems Analysis Group (CSAG) at the University of Cape Town. The hydrological and hydrogeological impacts of climate change were assessed using the MIKE SHE model by simulating projected changes in future ET_o and Rainfall. In order to simulate the impacts of climate change on water resources using MIKE SHE, the climate input data for the model, *viz.* rainfall and ET_o, need to be adjusted to reflect the anticipated mean state of the projected future climate. In order to achieve this the observed datasets were multiplied by a monthly adjustment factor or delta value. The monthly delta values were determined by calculating the average monthly ET_o and rainfall for a historic period (1980 – 2010) and comparing them to a future period (2040 – 2070).

3.8.1.1 Future reference potential evapotranspiration (ET_o)

Monthly ET_o was calculated using the monthly minimum and maximum temperatures from GCM's from 1960 – 2100. The Hargreaves and Samani (1985) method was used to calculate the ET_o as it only requires minimum and maximum temperatures which are available from GCM's. The Hargreaves and Samani (1985) method for calculating ET_o (mm.month⁻¹) is given as:

$$ET_o = 0.0023 \times R_a \times T_r^{0.5} \times (T_a + 17.8) \quad (2)$$

where, R_a is the extra-terrestrial radiation as a millimetre equivalent per day. T_r is the monthly temperature and T_a is the average monthly temperature. However, before the maximum and minimum temperatures of the 16 GCMs could be used to calculate ET_o, the temperatures needed to be corrected for any bias by comparing the GCM simulations to observed maximum and minimum temperatures during the historical period from 1980 – 2010. Monthly correction factors were calculated for each of the 16 GCMs and applied to both the historical and future maximum and minimum temperatures, providing corrected datasets for the calculation of ET_o. The average monthly ET_o for both the historical and future periods was then calculated. The delta values were then calculated by determining the percentage change between the monthly averages of the historical climate (1980 – 2010) and future climate period (2040 – 2070). The results of the calculation of ET_o using temperature predictions from GCMs indicate that most of the models predict greater increases in ET_o during winter than in summer, with a median increase of up to

12.4 % for the typically wet winter months from May to August (Figure 3.27). This suggests that the stronger increases in temperature during winter under future climate conditions is likely to result in greater increases in the actual evapotranspiration during the winter months.

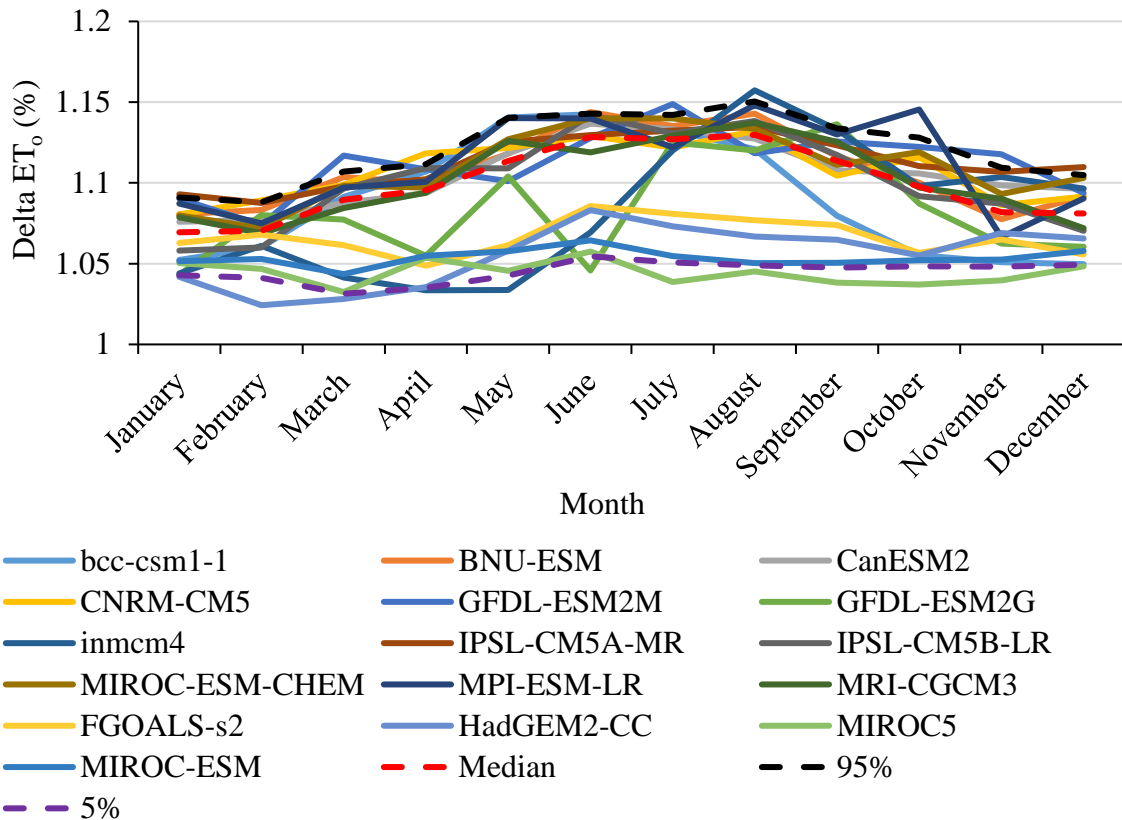


Figure 3.27: The monthly delta values describing the mean change in potential evapotranspiration (ET_0) between an historical climate (1980 – 2010) and future climate period (2040 – 2070)

3.8.1.2 Future rainfall

A similar procedure was adopted to calculate the monthly delta values required to estimate a theoretical rainfall record representing rainfall under the effects of future climate change. First, the average monthly rainfall was determined for the historical climate (1980 – 2010) and future climate period (2040 – 2070) for each of the 16 GCMs. The percentage change was then calculated to determine the mean monthly delta factors for rainfall. The mean monthly delta values are shown in Figure 3.28 for all 16 GCMs. Most of the models show decreases in rainfall for all the months of the year, with a median rainfall of approximately 0.9 % over all 12 months. An assessment of the median monthly delta values for rainfall indicates that most of the models indicate that the greatest declines in rainfall are anticipated to occur during late summer and early autumn months from February to April.

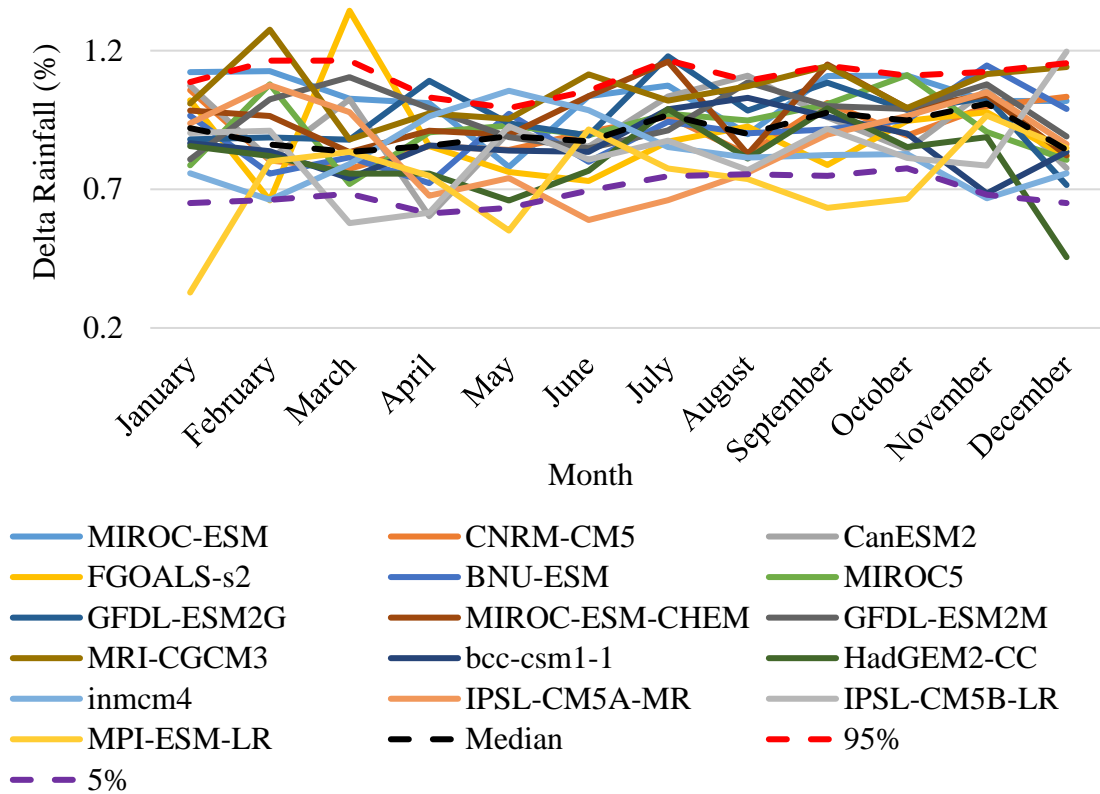


Figure 3.28: The monthly delta values describing the mean change in rainfall between a historical climate (1980 – 2010) and future climate period (2040 – 2070)

3.8.2 Selection of climate model for MIKE SHE simulations

As mentioned previously, MIKE SHE requires daily rainfall and ET_0 data for the simulation period (2000 – 2015). These datasets were constructed by applying the monthly delta values to the baseline rainfall and ET_0 to produce hypothetical rainfall and ET_0 datasets representative of a future climate. Although the current “best practice” in the climate change assessment process is to use the entire available ensemble of climate projections, due to time constraints and for the sake of transparency of the results, it was decided to perform analyses for selected projections only. The GCMs to be used for hydrological modelling were selected based on their specific future climate characteristics, so that the selected projections reflected the range of the entire 16-member ensemble. Individual GCMs were used rather than using the monthly statistical composition of all 16 GCMs so as to ensure the physical assumptions of each GCM are conserved. The selection of appropriate GCMs was based on the plotting of the mean delta values for the winter months from May to August for both rainfall and ET_0 , as this is when the impacts of climate change are likely to be most significant (Figure 3.29). Figure 3.29 shows the plotted delta values for rainfall on the x-axis, and ET_0 delta values plotted on the y-axis for the 16 GCMs as indicated by the blue markers, while the red markers show the 5th, 95th and median values of the 16 GCMs. In terms of ET_0 , the higher the delta value, the higher the future ET_0 indicating a hotter climate. The same is true of rainfall, where an increasing delta rainfall value corresponds to a wetter future climate. It is clear that for all the future climate scenarios, increases in future temperatures are anticipated. Many of the climate

scenarios predict a drier future with less rainfall, however some models show slight increases in rainfall.

Four GCMs were selected for hydrological modelling with MIKE SHE, *viz.* IPSL-CM5A-MR, MRI-CGCM3, MIROC5 and GFDL-ESM2M (Table 3.6). These models represent the following future climate conditions: “much hotter and much drier”, “much hotter and slightly wetter”, “slightly hotter and slightly drier” and the “median” conditions, respectively. IPSL-CM5A-MR is “much hotter and much drier” as it has both a high delta ET_0 and a low delta rainfall. MRI-CGCM3 has a similar delta ET_0 to IPSL-CM5A-MR. However, this model has a slightly higher delta rainfall, hence this simulation is “much hotter and slightly wetter”. MIROC5 was selected as it indicates the scenario with the least change in ET_0 in the future and only slightly reduced rainfall. Conversely, GFDL-ESM2M indicates a much higher change in ET_0 , but exhibits a similar predicted change in future rainfall as that of MIROC5.

Table 3.6: The four GCMs selected for MIKE SHE climate change simulations with MIKE SHE

GCM	Climate Change Description
IPSL-CM5A-MR	much hotter and much drier
MRI-CGCM3	much hotter and slightly wetter
MIROC5	slightly hotter and slightly drier
GFDL-ESM2M	median

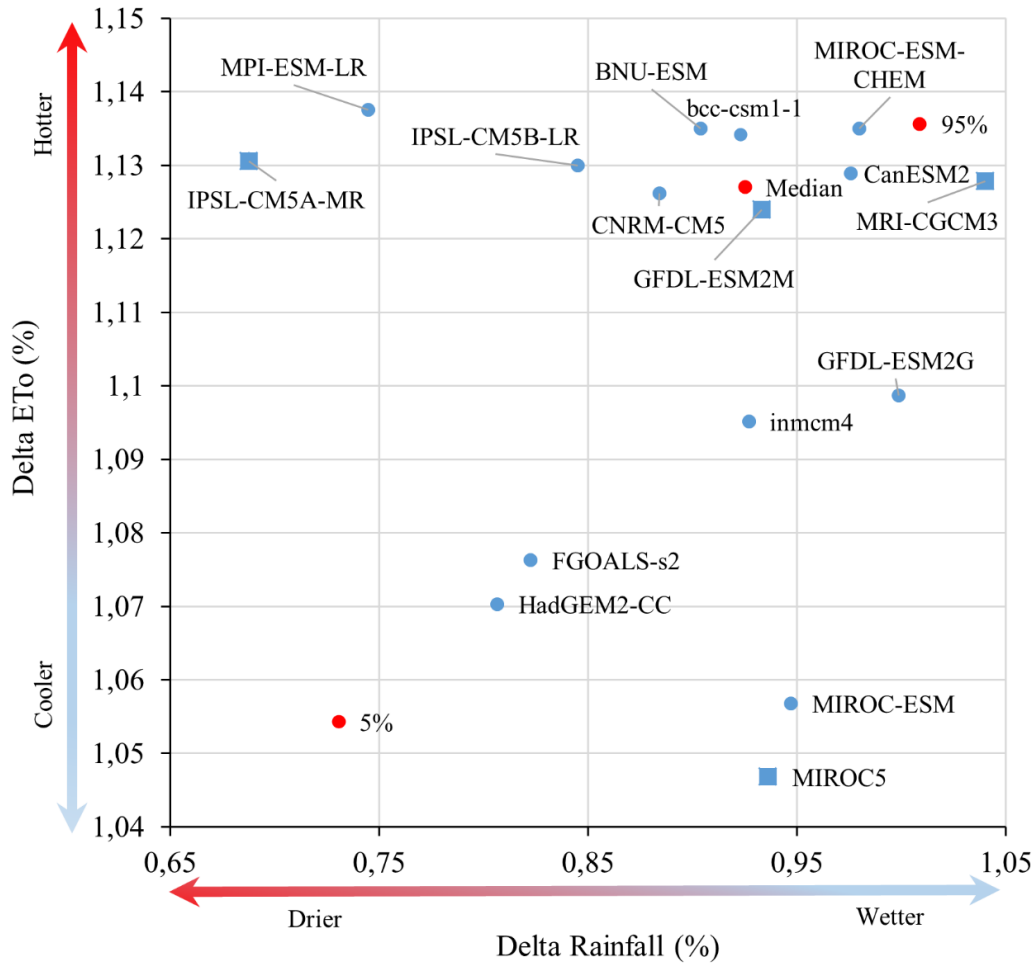


Figure 3.29: The plotted delta values for rainfall (x-axis) and ET_o (y-axis) for the 16 GCMs as indicated in blue, while the red markers show the 5th, 95th and median values of the 16 GCMs. The selected GCMs are indicated by the square data points

3.9 Summary of research method

This chapter has presented the research method that was used to address the research questions and objectives of this thesis, outlining the methods of this study, in particular, the application of the MIKE SHE and MIKE 11 models at a regional-scale and MIKE SHE at a local-scale, including the sourcing of input data, model parameterisation and the development of a conceptual hydrogeological model. The most important element of Chapter 3 was the calibration and validation of MIKE SHE for the CFA, demonstrating that the conceptual hydrogeological model for the CFA was reasonable and that the model could be appropriately used for scenario analysis.

Chapter 3 further outlined the site selection process, identifying the areas of the CFA that were suitable for the local-scale testing of MAR strategies using MIKE SHE. The scenarios used to test various MAR strategies on the CFA were then discussed. The chapter concluded with the methods for simulating the flow of groundwater contaminants and the potential impacts of climate change. The following chapter (Chapter 4) presents the results of the hydrological modelling with MIKE SHE.

4. Results

The following chapter describes the results of the MIKE SHE modelling. The first section presents the results of the regional-scale MIKE SHE model, describing the regional groundwater processes of the Cape Flats and discussing groundwater flow and recharge. Of particular importance is the spatial and temporal variability of groundwater recharge which has not been previously determined for the Cape Flats using a physically-based modelling approach such as MIKE SHE. This regional-scale MIKE SHE model was used to determine the boundary conditions of the local-scale MIKE SHE model application. The second and third sections examine the results of a scenario analysis of flood mitigation and MAR on the Cape Flats. Flood mitigation and MAR scenario simulations were conducted at four sites on the Cape Flats. Flood mitigation scenarios were conducted on two sites prone to winter flooding, and an assessment was made of the efficacy of utilising summer abstractions in order to mitigate seasonal winter flooding. A further two sites were selected to test the feasibility of MAR on the Cape Flats to determine where there is sufficient storage and aquifer yield within the CFA. The last two sections deal with the dispersion processes that govern the transport of contaminants from possible stormwater pollution and the potential threat of climate change.

4.1 Regional-scale simulation

The regional-scale MIKE SHE model was used to determine the current status of the CFA in terms of groundwater elevation, flow rate and direction, aquifer storage capacity and the hydrological and hydrogeological processes that influence groundwater recharge and discharge. All these model outputs were mapped and used to analyse the groundwater resources of the CFA.

4.1.1 Groundwater flow

The groundwater flow in the CFA is largely determined by the groundwater elevation which is influenced by the topography and underlying bedrock. The elevation of the water table decreases from the northeast of the model area towards the False Bay coastline in the south (Figure 4.1) with decreasing topographic and bedrock elevation towards the False Bay coastline. In addition, there are higher groundwater elevations in the western parts of the model, that are closer to Table Mountain. Thus, the regions with the highest groundwater elevation are generally associated with higher rates of groundwater flow. Groundwater in the north-east tends to flow in a southerly direction towards False Bay, flowing either along the eastern Kuils river valley or along the central Lotus river channel. The groundwater in the south-eastern model boundary flows towards False Bay, whereas the groundwater near Table Mountain in the west, flows north into Table Bay. The same is true for the groundwater in the north and north-western parts of the CFA. The groundwater flow in the central parts of the CFA along the Lotus River valley is higher than the surrounding areas due to the higher hydraulic conductivity of the coarse gravel within the paleochannel which dominates the central and southern parts of the CFA.

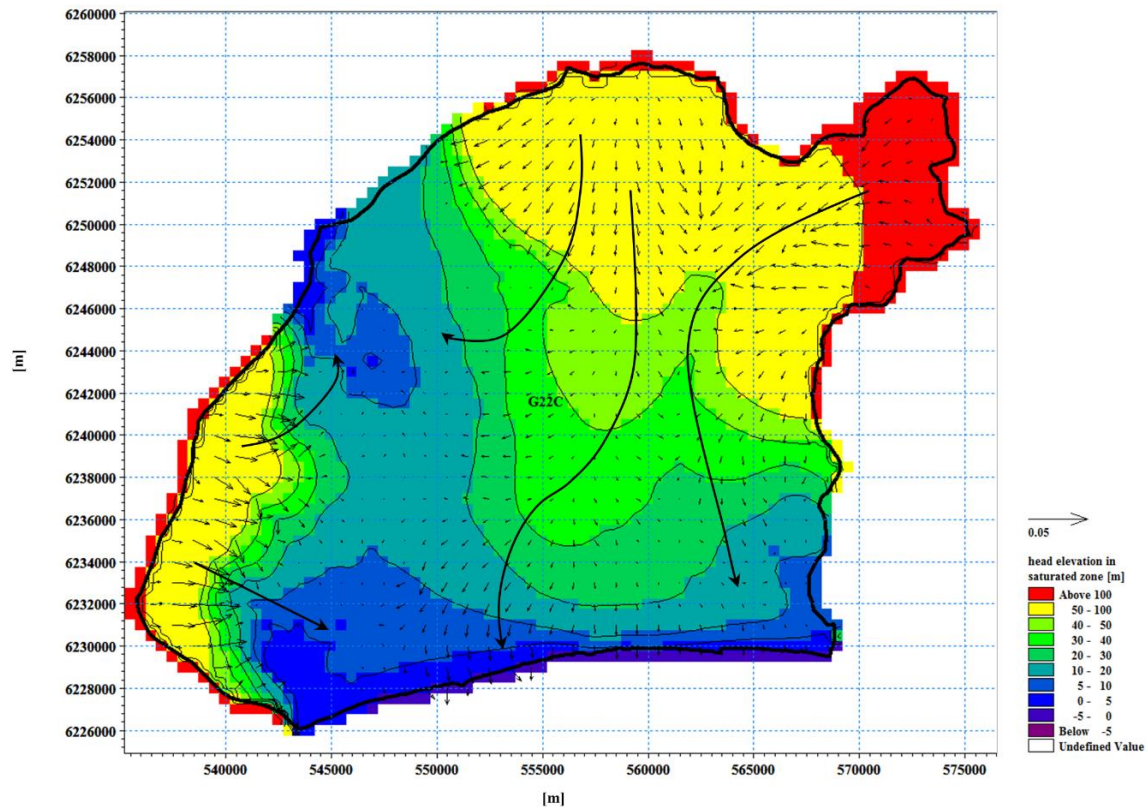


Figure 4.1: Simulated head elevation and generalised flow direction based on flow vectors for 31st of July 2015

4.1.2 Groundwater Recharge

The volume of groundwater recharge to the CFA was important to identify, because it is necessary to determine the available storage in the aquifer and volume available for sustainable abstraction. Past estimates of groundwater recharge have been based on mean annual precipitation and do not explicitly represent the influences of land use and evapotranspiration (Vandoolaeghe, 1989; DWAF, 2008). However, utilising a physically-based model like MIKE SHE, it is possible to determine the spatial and temporal variability of groundwater recharge by accounting for the spatial distribution of land use, soil and aquifer characteristics which determine groundwater recharge at a daily interval.

Figure 4.2 compares the simulated groundwater recharge, experienced under altered land use conditions between 1983 and 2013. In order to isolate the impact of changing land use, or more specifically the increase in urban land use, the same period from 2000 to 2015 was used to simulate both the 1983 and 2013 land use coverages. Simulating the impacts of urban land use over the same period means that interference from varying climatic conditions can be excluded. The land use for 1983 represents the recharge under ‘historical’ conditions and the land use for 2013 represents recharge under ‘current’ conditions, assuming the same climate, soil and aquifer conditions. The regions in blue denote recharge to the saturated zone. Whereas, the regions in red or pink identify areas where there is a negative recharge or an effective loss from the saturated zone, attributed to surface water discharges, drainage, losses to the unsaturated zone and evapotranspiration. A reduction in groundwater recharge is observed over much of the model area

for the current period when compared to the recharge observed under historical land use conditions. The most notable reductions in recharge occur in the central and northern regions of the Cape Flats which experienced between 0.2 and 0.3 mm.day⁻¹ under historical conditions. However, under current conditions, the recharge is reduced to between 0.1 and 0.2 mm.day⁻¹ for much of this region. And in some areas the recharge has become negative indicating net losses from the saturated zone either, to drainage or evaporative loss. In the southern parts of the Cape Flats, there is a reduction in recharge of between 0.1 and 0.2 mm.day⁻¹, with values of > 0.4 mm.day⁻¹ under historical land use, which has reduced to between 0.3 – 0.4 mm.day⁻¹ under current conditions. These changes in groundwater recharge are a result of the increase in urban land use that has occurred on the Cape Flats since the 1980's.

Noticeably, in many areas, there was an increase in negative recharge values under current conditions. Under the historical land use conditions this typically occurred in areas that were associated with surface water discharge from the saturated zone including surface water bodies, wetlands and rivers. However, under current conditions the regions that indicated a negative groundwater recharge had expanded, with the majority of these indicating small negative recharge values (-0.4 – 0 mm.day⁻¹). This is a result of urban land use expanding over previously natural or agricultural areas, thus recharge from infiltration is reduced, but the groundwater elevation in these areas mean that evaporation from plants and drainage result in a slight net negative recharge.

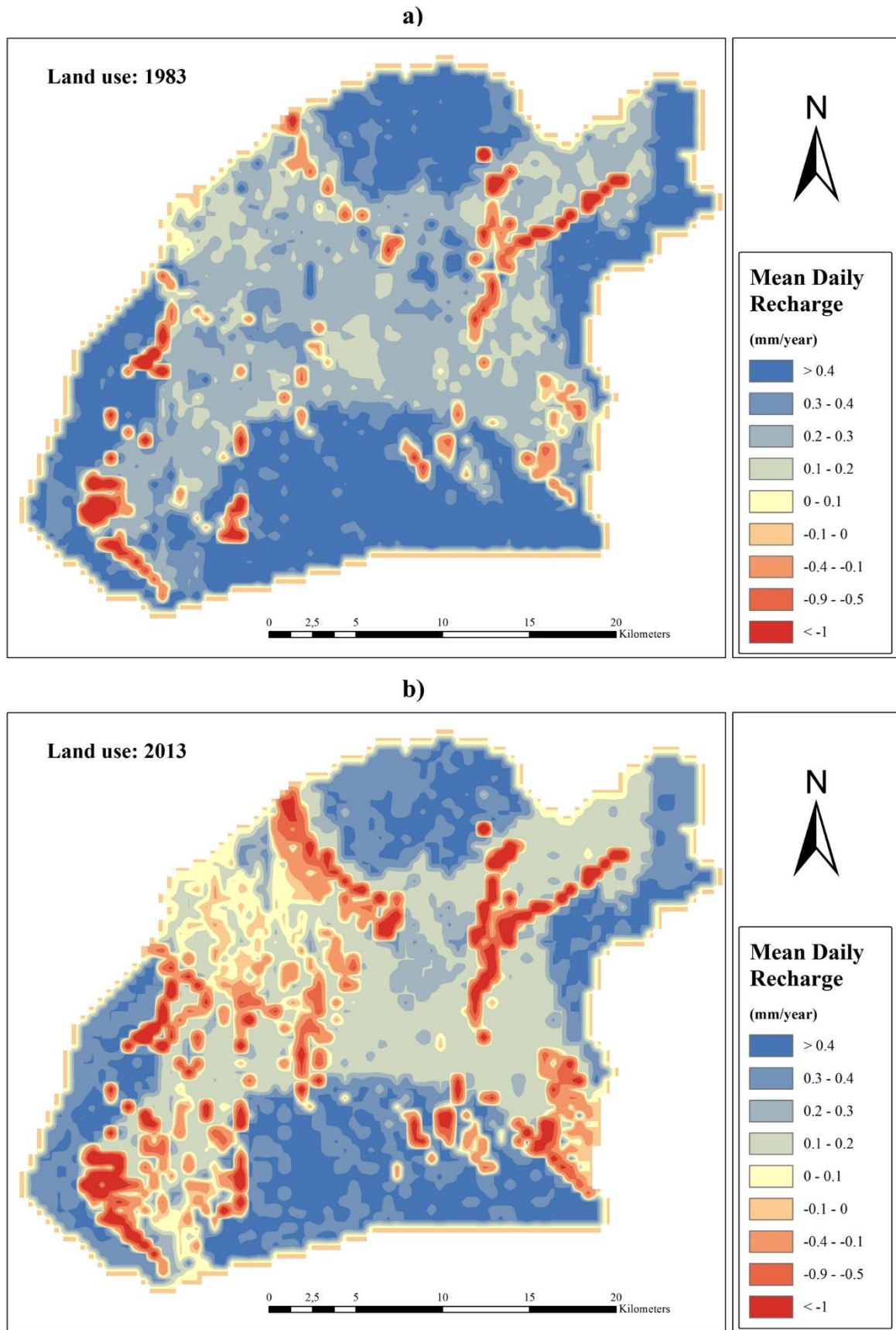


Figure 4.2: The impact of land use change on the CFA, comparing the simulated mean groundwater recharge for the land use of 1983 (a) and 2013 (b)

4.2 Flood mitigation

The areas that are most affected by flooding were identified from records found in the literature and obtained from the City of Cape Town. The informal settlements of Sweet Home, Never-Never, Phola Park and Graveyard Pond experience flooding annually (Figure 4.3). A number of these flood risk areas are located on or in proximity to locations where the simulated groundwater levels from 2000 to 2015 exceed -1.5 m above 80 % of the time, as indicated by the regions highlighted in red. Thus, it was assumed that flooding in these areas could be a result of groundwater seepage or poor drainage conditions from elevated groundwater levels.

Two sites in particular, Sweet Home and Graveyard Pond have received increasing public attention and are regularly reported as flood zones on the Cape Flats. Observed flooding in these areas was assumed to be a result of high groundwater levels, sometimes referred to as ‘seepage’ (Musungu and Drivdal, 2011; Pharoah, 2013). Sweet Home is located on land that was originally designated for agriculture and subsequently used for dumping building rubble (Pharoah, 2013). The areas around the site are known for the presence of seasonal wetlands, which support the assumption that an elevated groundwater table in the area is associated with seepage flooding or poor drainage conditions that increase runoff. Graveyard Pond is a site where a number of informal dwellers have moved into an area designated as a stormwater detention pond. Stormwater ponds are typically dry in summer but fill with stormwater during the wet winter months. In a study of Graveyard Pond, Musungu and Drivdal (2011) noted that 70 % of the residents claimed to have experienced flooding as a result of ‘underground water’.

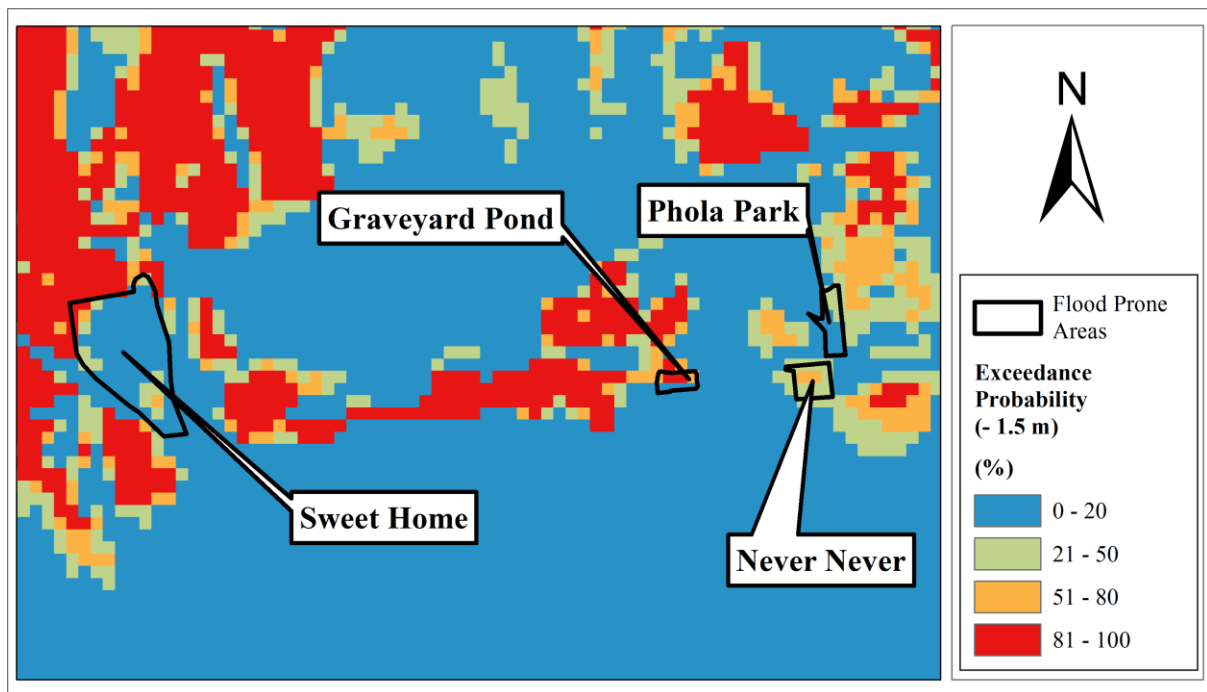


Figure 4.3: Probability of simulated groundwater levels exceeding a -1.5 m level and corresponded to flood-prone settlements on the Cape Flats (see Figure 3.22Figure 3.23 for context)

The flood mitigation scenarios for Sweet Home and Graveyard Pond are based on the assumption that flooding in these informal settlements is, in part, a result of elevated groundwater levels that prevent the infiltration of surface water, or a water table that exceeds the ground level such that ponding results. This model application does not explicitly represent flooding due to the required level of detail within the unsaturated zone and overland flow components of the model, as the required data was not readily available. Furthermore, a detailed representation of the unsaturated zone would have negative effects on the computational run time of the model. Alternatively, a threshold value for flooding of 1.5 meters below ground level (mbgl) was specified based on the proximity of the flood risk settlements to areas where the groundwater levels exceed -1.5 m more than 80 % of the time (Figure 4.3). Thus, it is assumed that groundwater levels that exceed the -1.5 m threshold are at risk of flooding. Conversely, if the groundwater levels are reduced to lower than the 1.5 m threshold through groundwater abstraction, it is assumed that potential risk of flooding will be mitigated.

The abstractions for all these scenarios is limited to half the year (184 days) during the summer months from the 1st of November until the 30th of April. The abstraction of groundwater for flood mitigation alone was assumed unlikely, due to the costs of pumping and infrastructure. Thus, an economic incentive for abstraction is required. Utilising the abstracted groundwater as a fit-for-purpose water supply to meet current water demands, including agricultural and residential irrigation, would create an economic incentive for the abstracted water and relieve the demand for potable water supply.

4.2.1 Sweet Home

Six abstraction scenarios were conducted at the Sweet Home site, consisting of three borehole arrangements of 9, 18 and 27 boreholes, 9BH, 18BH and 27BH respectively (Figure 4.4). The boreholes were positioned on available land not used for urban or agricultural purposes and spaced as uniformly as possible given the area of available land and number of boreholes required. Borehole yields in this area are typically between 3 $\ell.s^{-1}$ and 5 $\ell.s^{-1}$ (DWAF, 2000; GCS (PTY) LTD, 2015) therefore the three borehole arrangements with pumping rates of 3 $\ell.s^{-1}$ and 5 $\ell.s^{-1}$ were simulated in MIKE SHE (Table 4.1).

Table 4.1: Summary of the scenarios for flood mitigation at Sweet Home informal settlement

Summary of scenario descriptions	
Scenario 1	9 Boreholes pumping at a rate of 3 $\ell.s^{-1}$ during the summer months
Scenario 2	9 Boreholes pumping at a rate of 5 $\ell.s^{-1}$ during the summer months
Scenario 3	18 Boreholes pumping at a rate of 3 $\ell.s^{-1}$ during the summer months
Scenario 4	18 Boreholes pumping at a rate of 5 $\ell.s^{-1}$ during the summer months
Scenario 5	27 Boreholes pumping at a rate of 3 $\ell.s^{-1}$ during the summer months
Scenario 6	27 Boreholes pumping at a rate of 5 $\ell.s^{-1}$ during the summer months
Scenario 7	27 Boreholes pumping at a rate of 5 $\ell.s^{-1}$ during the summer months. Wellfield rearranged, moving six boreholes away from Edith Stephens Wetland.

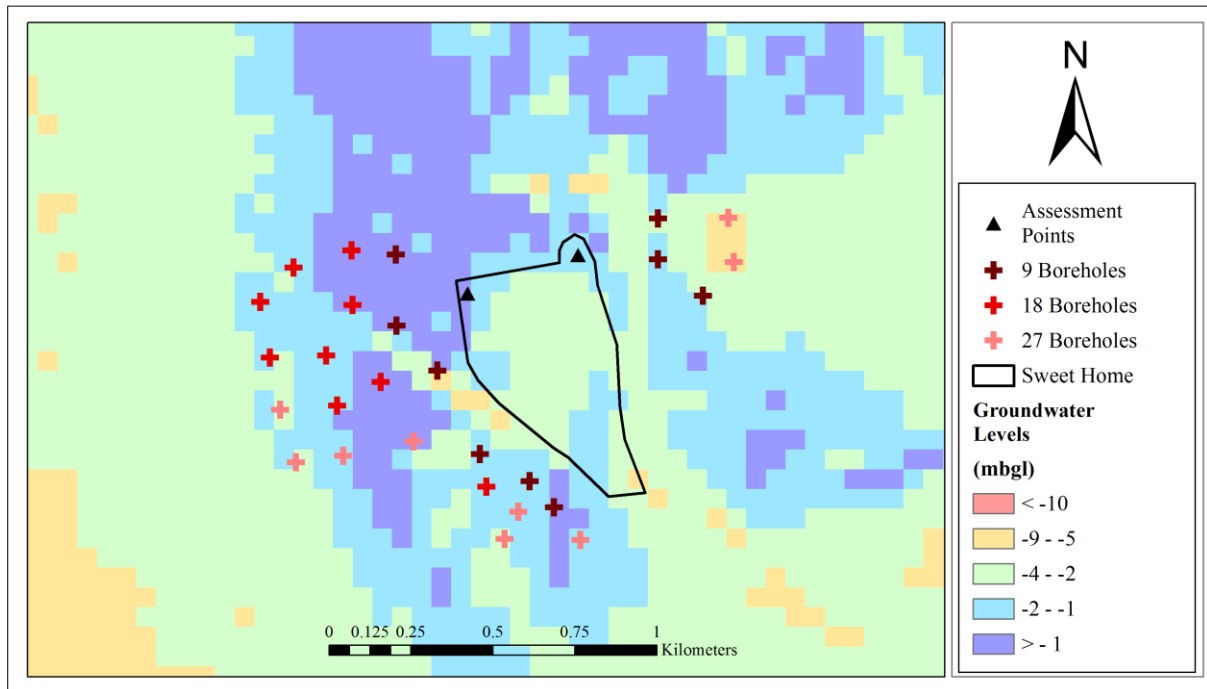


Figure 4.4: Borehole arrangement and assessment points for the Sweet Home flood mitigation scenario

The groundwater levels for the west and east assessment points showed that the baseline simulated groundwater levels were above a depth of -1 m for much of the simulation period, receding towards the -1 m level during summer and rising to near the ground surface in winter, particularly in the wet years of 2007 and 2008 (Figure 4.5 and Figure 4.6). The application of the six pumping scenarios showed an increase in drawdown at the Sweet Home site with increasing numbers of boreholes and abstraction rates. An abstraction rate of 3 l.s^{-1} showed limited impacts on the drawdown at both sites for all the borehole arrangements, with Scenarios 1 and 3 exceeding the -1.5 m threshold for much of the simulation period. Scenario 5 did reduce the groundwater level to below the threshold on a number of occasions after a steady-state was achieved between 2005 and 2015. At the west assessment point, 5 of the 10 wet seasons between 2005 – 2015 were below the flood threshold, while at the eastern site, only 2 sites were below the flood threshold. The analysis of the exceedance probabilities of groundwater levels in excess of -1.5 m for each site support the interpretation of the flood mitigation potential using groundwater levels (Figure 4.7 and Figure 4.8). The probability of exceedance was calculated once the groundwater levels had reached a steady-state for each pumping application, approximately 5 years into the simulation from 2005 – 2015. The western site shows a greater response to the abstraction, due to the larger wellfield on the western side of Sweet Home. The 9BH arrangement (Scenario 1) exceeded the threshold 93 % of the time, while the 18BH (Scenario 3) and 27BH (Scenario 5) arrangements reduced the probability of exceedance to 49 % and 17 % respectively. While, at the eastern site, the 9BH, 18BH and 27BH borehole arrangements exceeded the threshold 82 %, 62 % and 34 % of the time.

The 9BH borehole arrangement at 5 l.s^{-1} (Scenario 2) showed similar drawdown and seasonal variations to that of 18BH at 3 l.s^{-1} (Scenario 3) at the western site, exceeding the threshold 49 % of the time. However, at the eastern site the 9BH borehole arrangement at 5 l.s^{-1} (Scenario 2) showed similar drawdown and seasonal variations to that of 27BH at 3 l.s^{-1} (Scenario 5), exceeding the threshold approximately 35 % of the time. The 18BH and 27 BH pumping at 5 l.s^{-1} (Scenarios 4 and 6) were the only scenarios that resulted in simulated groundwater drawdown to levels that were regularly below the specified -1.5 m threshold. After an initial period of 5 years, where the groundwater levels decline to reach a steady-state, the groundwater levels were assessed. The simulated groundwater levels only exceed the threshold value at the eastern site 5 % of the time for the 18BH borehole arrangement pumping at 5 l.s^{-1} . Whereas, the 27BH borehole arrangement utilising an abstraction rate of 5 l.s^{-1} results in simulated groundwater levels that remain permanently below the threshold value.

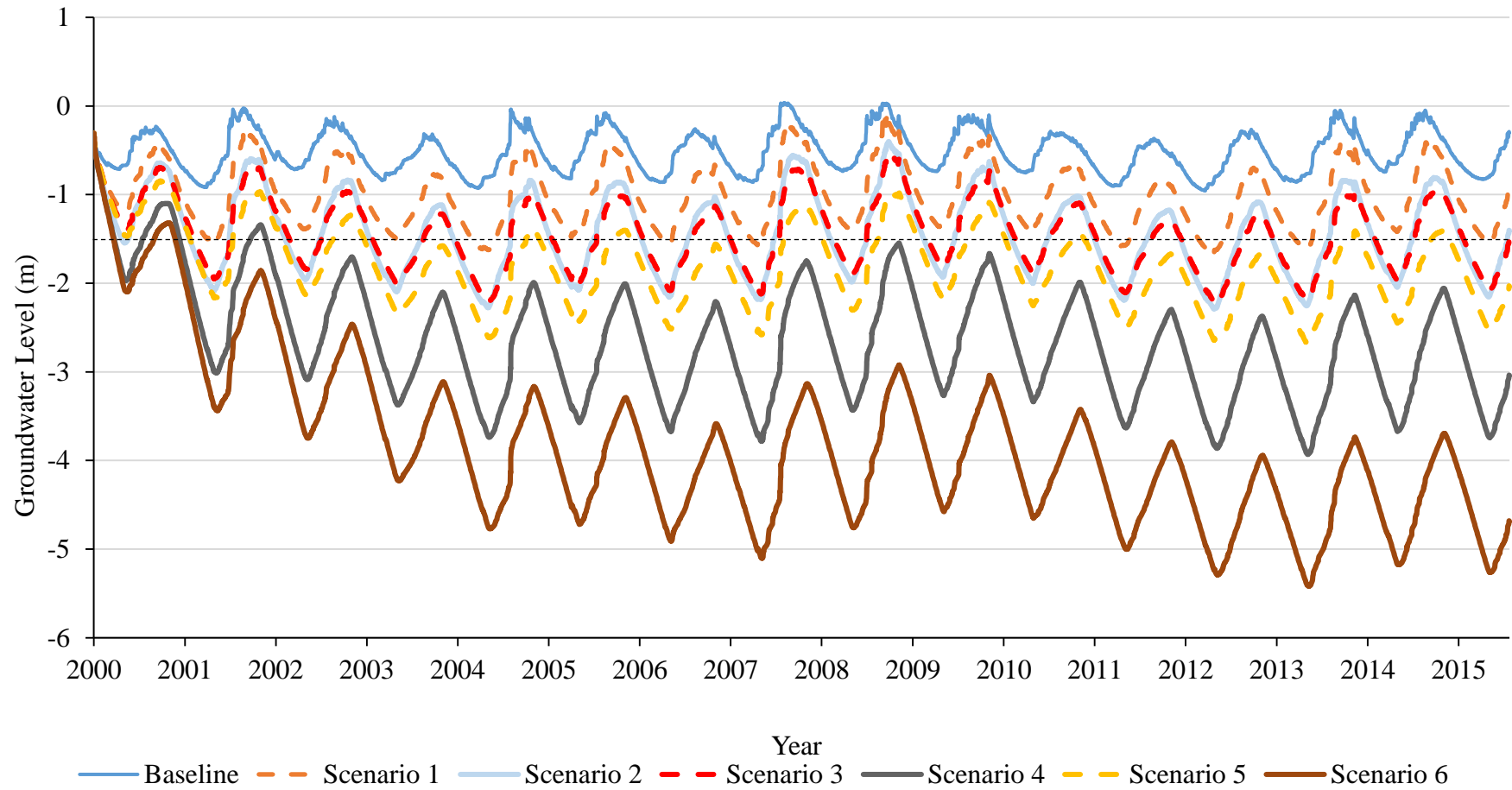


Figure 4.5: Simulated groundwater levels in the west of Sweet Home comparing three different borehole arrangements (9BH, 18BH and 27BH) at two different pumping rates (3 l.s^{-1} and 5 l.s^{-1})

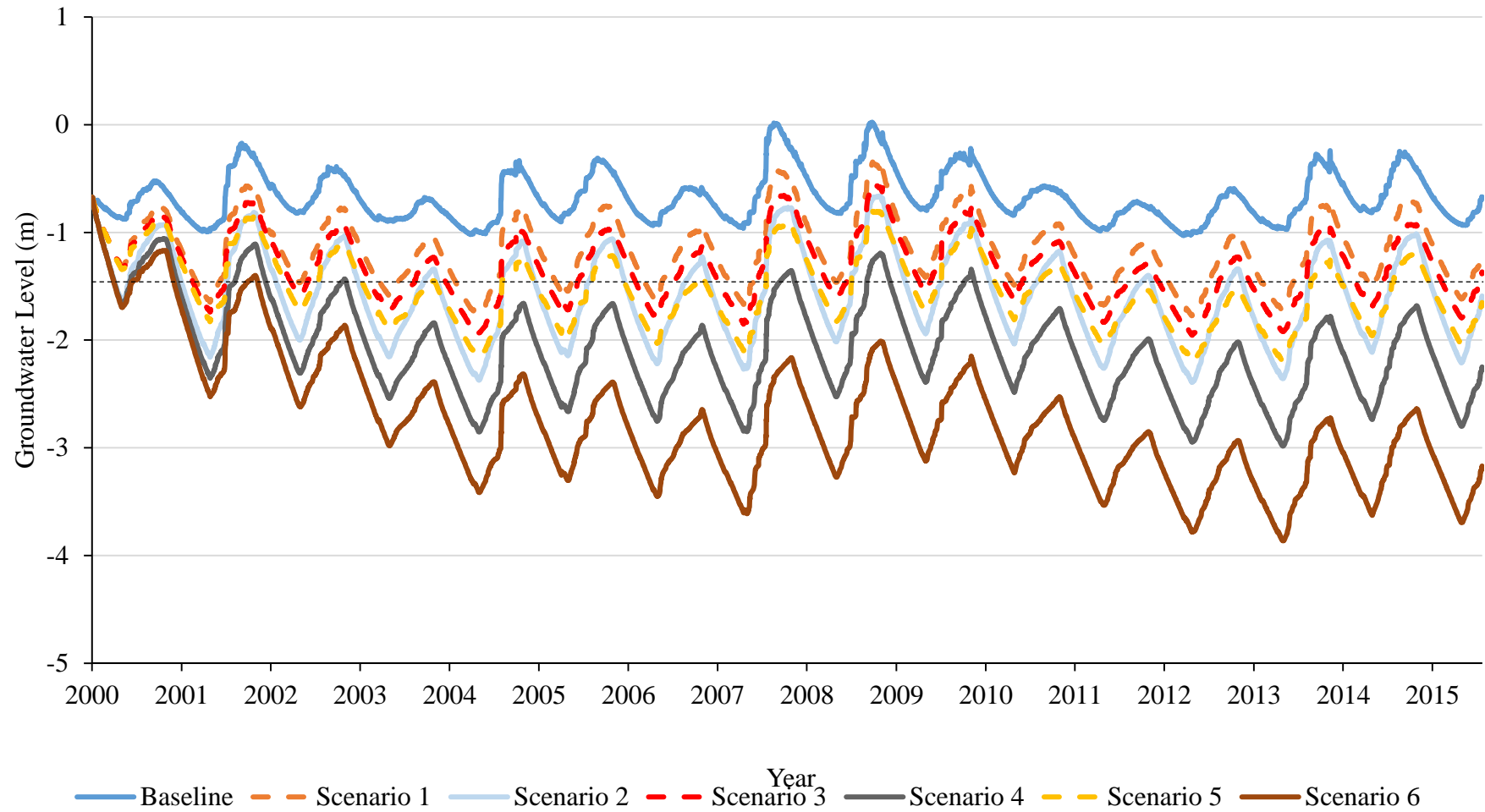


Figure 4.6: Simulated groundwater levels in the east of Sweet Home comparing three different borehole arrangements (9BH, 18BH and 27BH) at two different pumping rates (3 l.s^{-1} and 5 l.s^{-1})

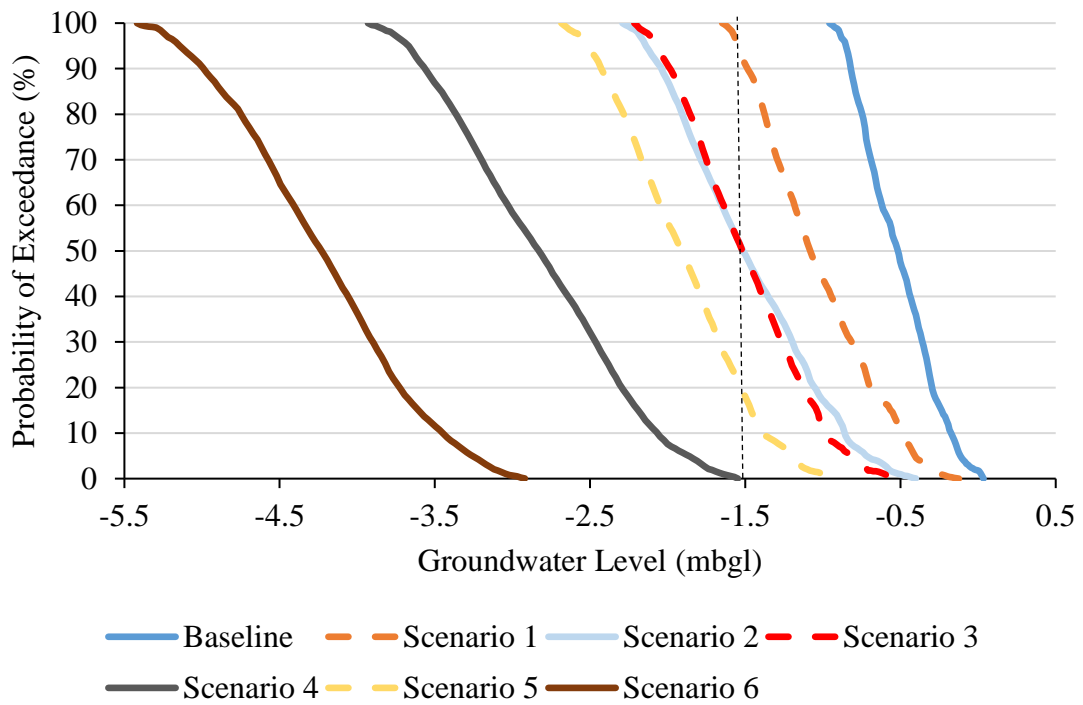


Figure 4.7: Probability of exceedance for the seven pumping scenarios in the west of Sweet Home

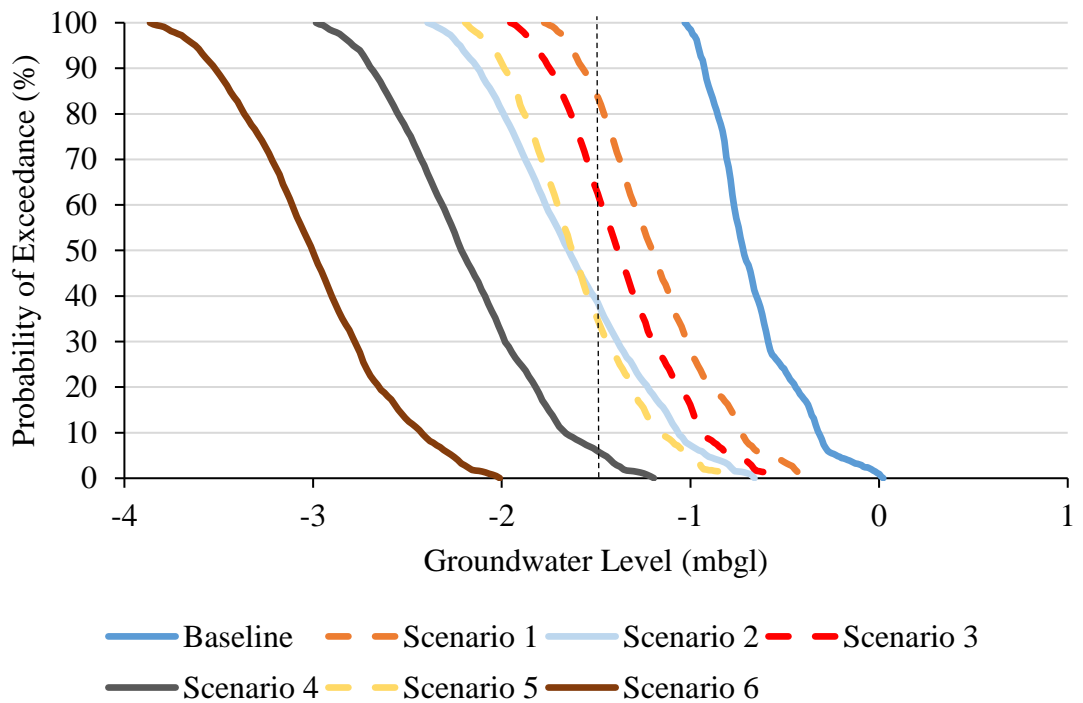


Figure 4.8: Probability of exceedance for the seven pumping scenarios in the east of Sweet Home

The cone of depression for each scenario gives an indication of the extent of the effects of the abstractions from each scenario, helping to inform how effective each scenario is for the specified objective of drawing down the groundwater level as a flood prevention measure. Furthermore, it is possible to gain insight into the unintended impacts of the scenarios through the potential impact to local water bodies, such as wetlands and rivers. It is important to note that the drawdown is relative to the baseline groundwater level and not the ground surface level, therefore the threshold value and the cone of depression cannot be compared directly. The extent and magnitude of the cone of depression for each of the scenarios were mapped comparing the wellfield arrangement and the abstraction rate (Figure 4.9). The results indicated that Scenarios 1 and 2, which represent the 9 borehole arrangements only showed a drawdown of less than 1 mbgl with a maximum radius of 0.72 km for Scenario 2. Scenario 3 showed similar results to that of Scenario 2 in terms of the magnitude of the drawdown, however the extent of the cone is greater primarily due to the increase in the number of boreholes from 9 to 18. Figure 4.5 indicates that only Scenarios 4, 5 and 6 that begin to reach drawdown magnitudes that are below the specified threshold value of -1.5 m. These scenarios show more extensive depression areas with radii of 1.6, 1.5 and 2.2 km for Scenarios 4, 5 and 6 respectively.

An area of concern when abstracting water for flood mitigation was the risk to local groundwater dependent ecosystems. In the case of Sweet Home, a local wetland called Edith Stephens Wetland is located 1.3 km away from Sweet Home. As suggested, the most appropriate scenario for achieving the flood mitigation objective is Scenario 6. However, the effects of the cone of depression from Scenarios 6 on Edith Stephens was assessed based on maps of the extent of the cone of depression, together with the transect profiles of Scenarios 1– 6 (A to B) (Figure 4.10). The results indicate that all the scenarios have some effect on the Edith Stephens Wetland, however, Scenarios 4, 5 and 6, in particular, are likely to have a significant impact on Edith Stephens, showing a decline in local groundwater levels in this area of between 1.5 to 2 m, which would potentially have a negative impact on the wetland. In order to try and mitigate the impacts on the Edith Stephens Wetland, Scenario 6 was modified into an additional scenario, Scenario 7, where 6 boreholes were relocated away from the wetland to the north-west of the Sweet Home site. Scenario 7 results in similar drawdown at the Sweet Home site, meeting the flood threshold for the entire simulation period for both assessment points, but reducing the drawdown at Edith Stephens's Wetland by almost one meter (Figure 4.10).

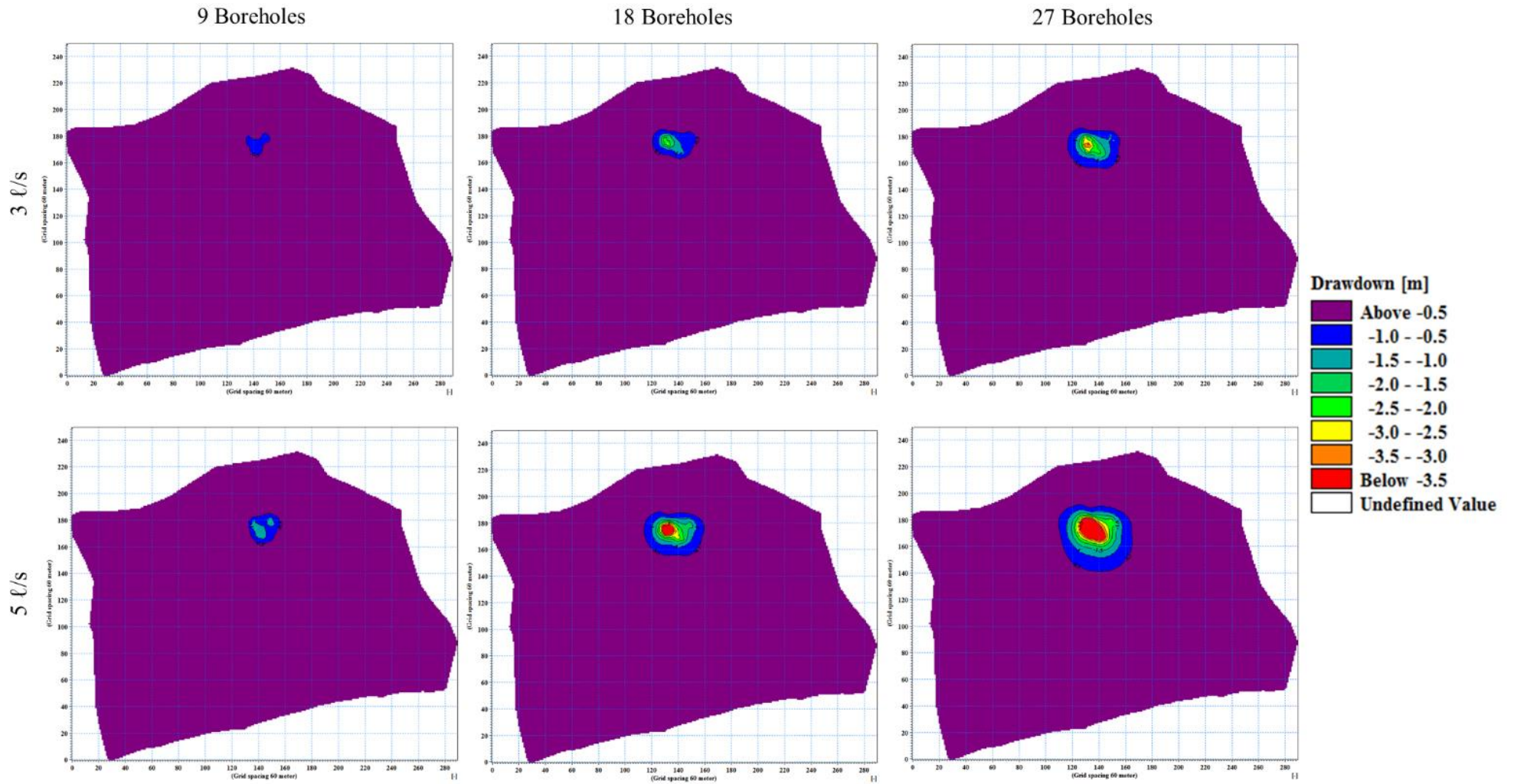


Figure 4.9: The cone of depression for the six scenarios comparing the three borehole arrangements and two pumping rates

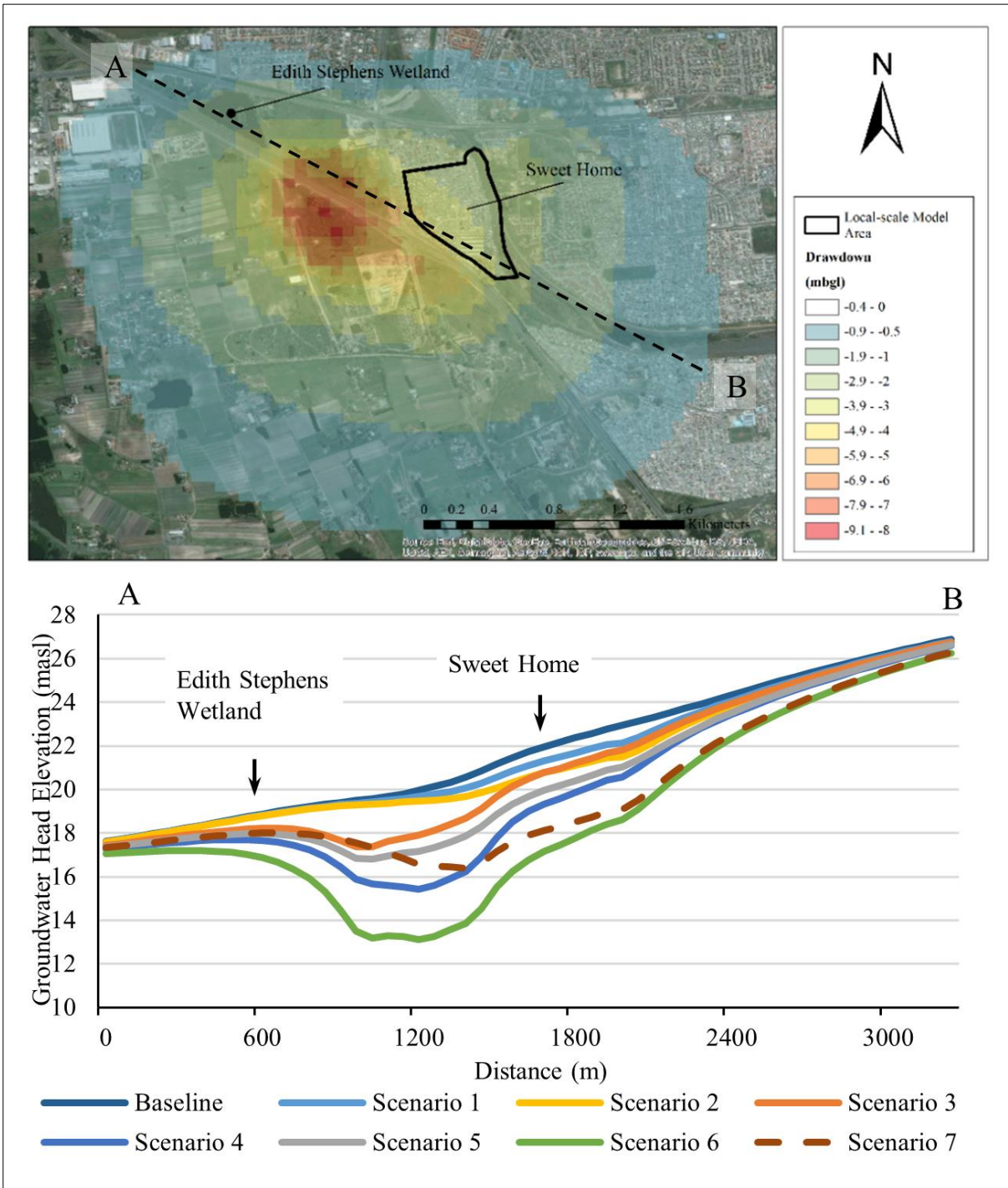


Figure 4.10: The extent of the cone of depression under Scenario 6, showing the transect of groundwater head elevation for Scenarios 1 to 7

4.2.2 Graveyard Pond

Graveyard Pond is an informal settlement to the east of Sweet Home, that was selected to further test the viability of summer abstractions as a flood mitigation strategy. A number of studies have suggested that the site experiences flooding as a result of an elevated groundwater table. Six

abstraction scenarios were simulated at the Graveyard Pond site, consisting of three borehole arrangements of 4, 8 and 12 boreholes, *viz.* 4BH, 8BH and 12BH (Figure 4.11). Similar to the Sweet Home scenarios, the abstraction boreholes were positioned on available land free from any specific land use. Available land in proximity to Graveyard Pond was limited, with the exception of three local stormwater ponds and vacant land near Phola Park. The same abstraction rates (3 l.s^{-1} and 5 l.s^{-1}) that were applied for the Sweet Home scenarios were used to assess the viability of utilising groundwater abstraction for flood mitigation at Graveyard Pond (Table 4.2). The same threshold value of -1.5 meters was specified for this flood mitigation scenario, assuming that reductions in groundwater levels below the threshold would reduce groundwater flooding and improve drainage.

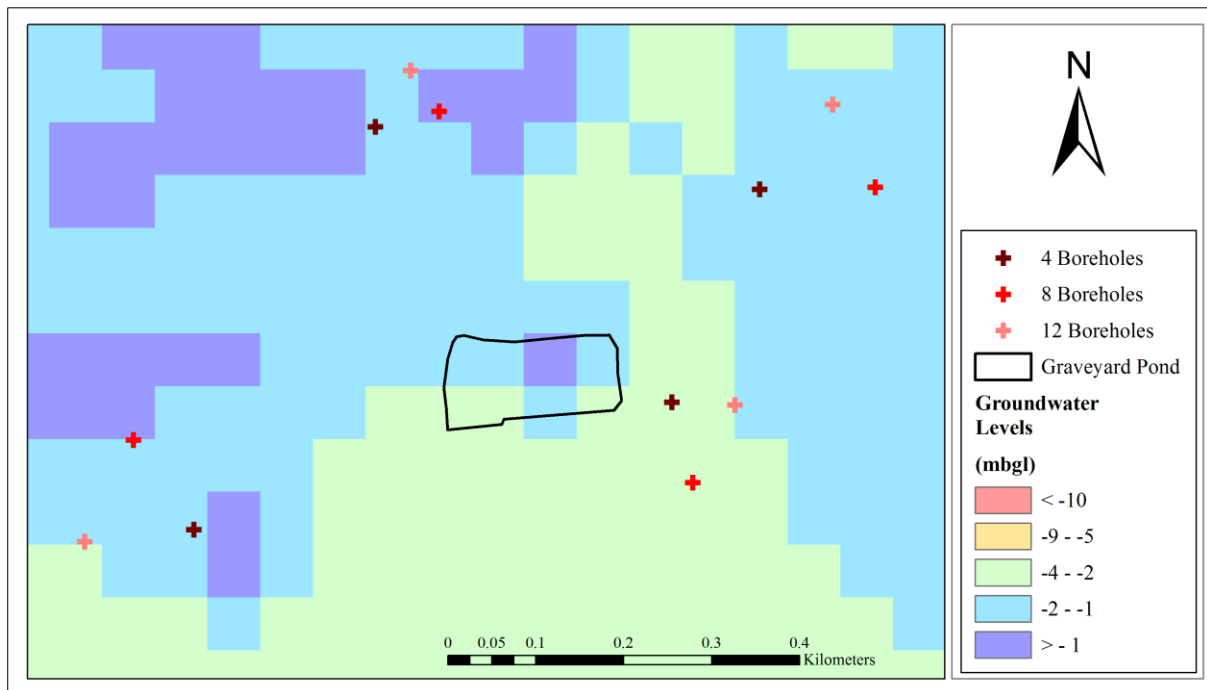


Figure 4.11: Borehole arrangement and assessment points for the Graveyard Pond flood mitigation scenario

Table 4.2: Summary of the scenario descriptions for flood mitigation at Graveyard Pond informal settlement

Summary of scenario descriptions	
Scenario 8	4 Boreholes pumping at a rate of 3 l.s^{-1} during the summer months
Scenario 9	4 Boreholes pumping at a rate of 5 l.s^{-1} during the summer months
Scenario 10	8 Boreholes pumping at a rate of 3 l.s^{-1} during the summer months
Scenario 11	8 Boreholes pumping at a rate of 5 l.s^{-1} during the summer months
Scenario 12	12 Boreholes pumping at a rate of 3 l.s^{-1} during the summer months
Scenario 13	12 Boreholes pumping at a rate of 5 l.s^{-1} during the summer months

The groundwater levels under baseline conditions at the Graveyard Pond site exceeded the threshold of -1.5 m for the entire simulation period from 2000 – 2015 and exceeded -1 m for 11 wet seasons (Figure 4.12). The six abstraction scenarios showed increasing drawdown of groundwater levels at Graveyard Pond with increasing numbers of boreholes and abstraction rates. Abstraction at a rate of 3 l.s^{-1} showed limited impact on the drawdown of groundwater levels for the 4 and 8 borehole arrangements (Scenarios 8 and 10), both showing probabilities of exceeding the -1.5 m threshold at 72 % and 32 % respectively (Figure 4.12 and Figure 4.13). However, the 12BH arrangement (Scenario 12) achieved some success in reducing the groundwater level to below the threshold for approximately 11 % of the time, only exceeding the threshold on 5 of the 15 wet seasons within the simulation period (Figure 4.13). The 5 l.s^{-1} abstraction rate for the 4BH arrangement (Scenario 9) exceeded the threshold value for 44 % of the simulation period. However, Scenario 11 reduced the groundwater levels to below the threshold for 94 % of the simulation period, only exceeding the threshold during the notably wetter period from 2007 to 2010. Similar to Scenario 12, Scenario 11 did reduce the amount of time that the groundwater level exceeds the threshold, with the peak groundwater levels only exceeding the threshold in 5 of the 15 wet seasons during the simulation period. Only Scenario 13 was able to maintain reduced groundwater levels for the entire simulation period, only exceeding the threshold on one occasion, which occurred in the initial stages of the simulation prior to the groundwater level stabilising after 2005. After the groundwater reaches a steady-state in 2005, Scenario 13 was able to maintain the prescribed threshold for the rest of the simulation period, even during the particularly wet period from 2007 to 2015 following a period of high rainfall (Figure 4.12). Thus, the results suggest that Scenario 13 is the most viable as a flood mitigation scenario. However, Scenarios 11 and 12 both showed substantial reductions in the time that the groundwater levels exceeded the flood threshold, only exceeding the threshold in very wet years.

The risk to local wetlands or surface water bodies from local groundwater abstractions is minimal. Figure 4.14 shows the simulated drawdown and the extent of the mean cone of depression obtained from each scenario. The simulated groundwater levels at Graveyard Pond (Figure 4.12) are similar to those at Sweet Home as Scenarios 8, 9 and 10 all indicate groundwater levels that are above the threshold value of -1.5 m and therefore do not fulfil the flood mitigation objectives as the mean drawdown is above the threshold. Scenarios 11, 12 and 13 all reduced the groundwater level to below the threshold for more than 88 % of the time, resulting in cones of depression extending 1.0, 0.9 and 1.5 km respectively. The extent of these cones of depression are only marginally less extensive than Sweet Home, especially given that there are fewer boreholes at the Graveyard Pond site. This is due to the shallower bedrock at Graveyard Pond meaning that there is less water available due to the thinner aquifer layer, thus the extent of the cone of depression is relatively more extensive as water from further away migrates towards the source of abstraction.



Figure 4.12: Simulated groundwater levels in Graveyard Pond comparing two different borehole arrangements (4BH and 8BH) at two different pumping rates (3 l.s^{-1} and 5 l.s^{-1})

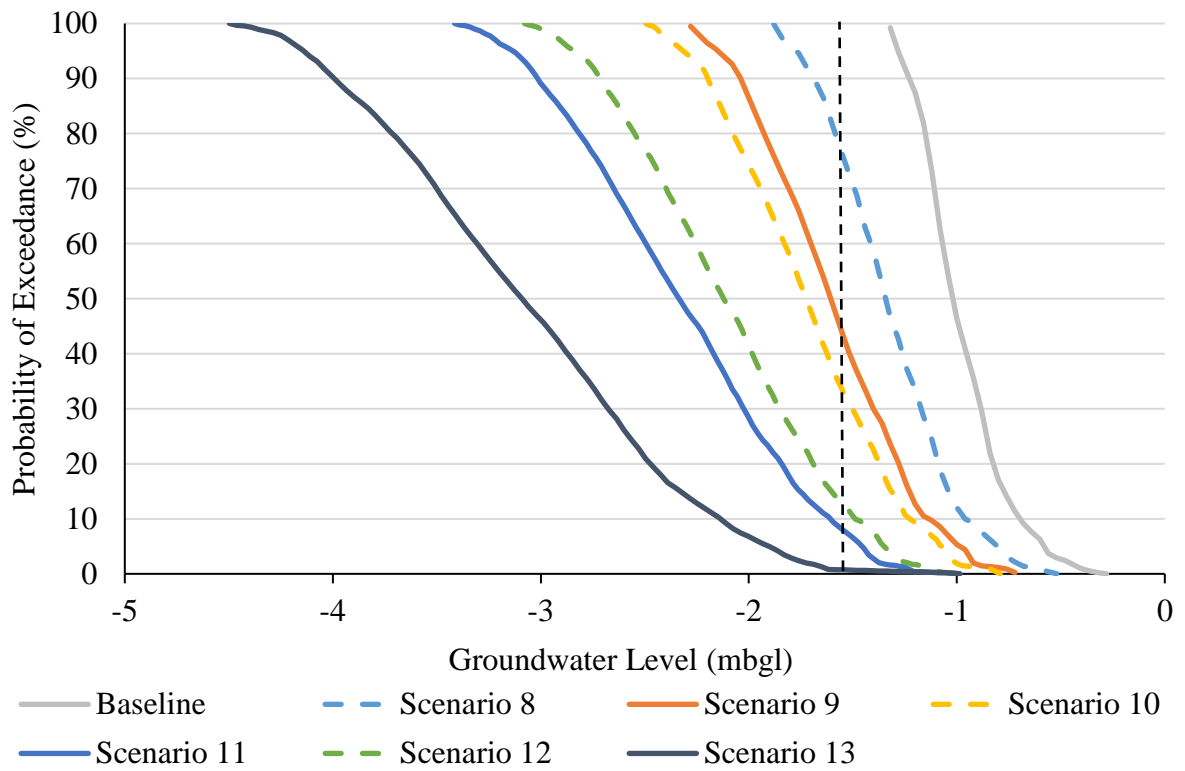


Figure 4.13: Probability of exceedance for the seven pumping scenarios in the east of Sweet Home

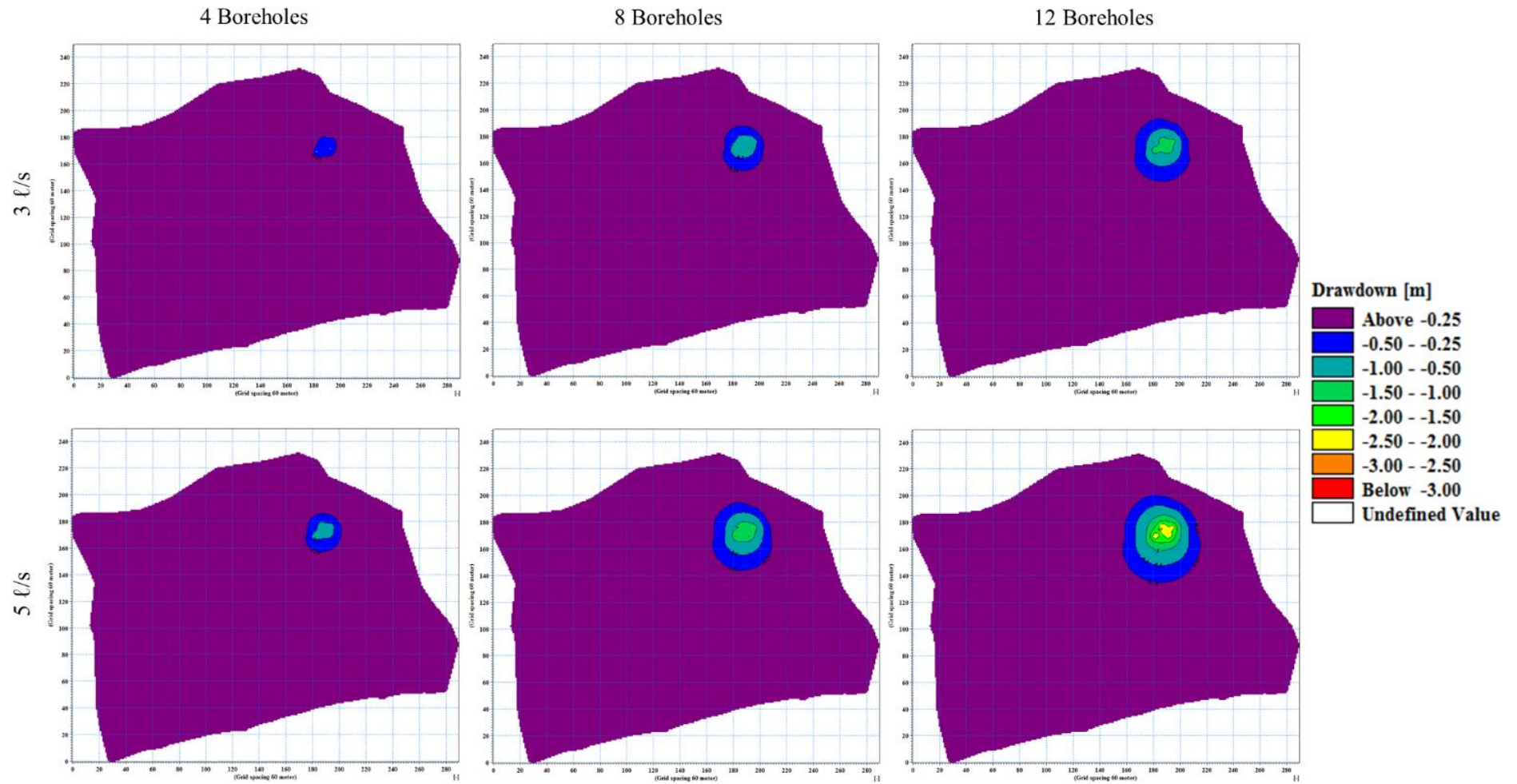


Figure 4.14: The mean cone of depression at Graveyard Pond for the six abstraction scenarios comparing the three borehole arrangements and two pumping rates

4.2.3 Philippi and Mitchells Plain MAR sites

The Philippi MAR site was selected due to the available storage within this area of the CFA based on MAR potential maps, which indicated that the site may provide a valuable form of inter-seasonal storage and short-term storage. The aquifer at the Philippi site has the greatest depths of unconsolidated material and therefore has the highest MAR potential. Based on the available storage capacity, it was assumed that the greatest opportunity for inter-annual or long-term storage is at Philippi. Thus, more detailed scenario analysis was conducted at the Philippi site to simulate the storage processes of the CFA. Therefore, as described in Table 4.3 the first scenario (Scenario 14) describes the aquifer recharge-storage relationship in order to identify the feasibility of long-term stormwater storage in the CFA at Philippi. Scenarios 15 to 17 test various MAR scenarios at Philippi as a form of inter-seasonal storage for winter stormwater. Scenarios 18 and 19 assess the best performing scenarios at the Philippi site and evaluate their feasibility under different aquifer conditions at Mitchells Plain.

Table 4.3: Summary of the scenario descriptions for MAR at Philippi and Mitchells Plain

Summary of scenario descriptions	
Scenario 14	The artificial recharge of 'stormwater', during the winter months, through infiltration and injection in increasing volumes from 2 Mm ³ to 10 Mm ³ .
Scenario 15	Equivalent summer abstraction and winter recharge rates using a conservative rate of approximately 15 ℓ.s ⁻¹ .
Scenario 16	Summer only abstraction at maximum pumping rates at 32 ℓ.s ⁻¹ with no artificial recharge at the Philippi site.
Scenario 17	Double summer abstraction at approximately 32 ℓ.s ⁻¹ and a conservative winter recharge rate of approximately 15 ℓ.s ⁻¹ at the Philippi site.
Scenario 18	Summer only abstraction at maximum pumping rates at 32 ℓ.s ⁻¹ with no artificial recharge at the Mitchells Plain site.
Scenario 19	Double summer abstraction at approximately 32 ℓ.s ⁻¹ and a conservative winter recharge rate of approximately 15 ℓ.s ⁻¹ at the Mitchells Plain site.

4.2.3.1 Philippi: CFA recharge-storage potential

In Scenario 14, recharge was simulated as infiltration through the ground surface and as subsurface injection to determine the upper limits of the storage depth available within the CFA in the Philippi region. This was achieved by incrementally increasing the volume of water recharged into the CFA via injection and infiltration to assess the recharge-storage relationship, indicating the efficiency of the CFA as a form of storage. In order to represent surface infiltration, a depth of rainfall equivalent to the daily volume of recharged water was applied to the soil surface for infiltration (Table 4.4). In terms of the infiltration scenario, an area that is roughly a similar size to Pond 7

(280 000 m³) in the Atlantis MAR operation north of Cape Town was used to simulate artificial recharge into the CFA at the Philippi site.

Table 4.4: Infiltrated water recharge volume and equivalent rainfall depth

Total Recharge Volume (Mm ³ .year ⁻¹)	Daily Volume (m ³)	Relative Rainfall Depth (mm)
2	10869.57	37.74
3	16304.35	56.61
4	21739.13	75.48
5	27173.91	94.35
6	32608.70	113.22
7	38043.48	132.10
8	43478.26	150.97
10	54347.83	188.71

The alternative to infiltration is to utilise a borehole wellfield for both the abstraction and injection of water. The size of the wellfield is a function of the amount of water intended to be recharged, the allowable borehole yield or abstraction rate and the storage capacity of the aquifer. Since this scenario is investigating the feasibility of winter storage, the time for groundwater recharge was limited to 184 days during the winter months, placing further constraints on the wellfield design. The maximum borehole yield in the southern parts of the CFA in the Mitchells Plain and Philippi areas is approximately 32 l.s⁻¹ with an estimated sustainable yield of approximately 17 l.s⁻¹ (Tredoux *et al.*, 1980; Vandoolaeghe, 1989). Thus, in the design of the wellfield at Mitchell's Plain, 20 boreholes were used to maximise the recharge volume given the limited injection and abstraction rates. Furthermore, the same limits that were applied to the injection rates, were applied to the abstraction rates.

The results for Scenario 14 indicate that initially both the infiltrated and injected recharge scenarios show proportional increases in storage depth with increasing volumes of recharge (Figure 4.15). The injected MAR scenario showed greater storage with increasing recharge, as it is applied over a larger area than the infiltrated recharge, meaning that the same volume is distributed more evenly throughout the available storage in the aquifer. This results in a smoother hydraulic gradient around the recharge site improving the storage efficiency. The proportional relationship continues until the recharge volume of approximately 6 Mm³ per year, after which the storage efficiency of the recharge begins to decline due to drainage losses. Drainage losses affect the infiltrated scenario to a greater extent, where the storage efficiency is reduced earlier at a recharge volume of 4 Mm³ per year. Thus, the optimum storage volume for the injected MAR and infiltrated storage MAR is 237 mm and 149 mm respectively. Indicating that the injection scenario is likely to be more efficient than the infiltration MAR strategy.

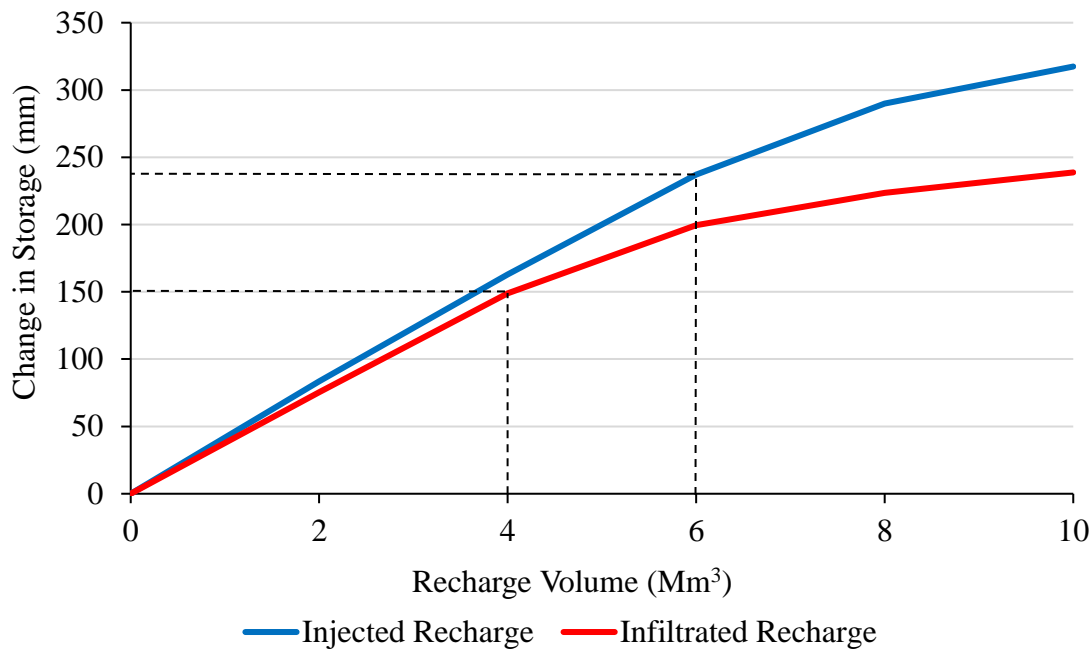


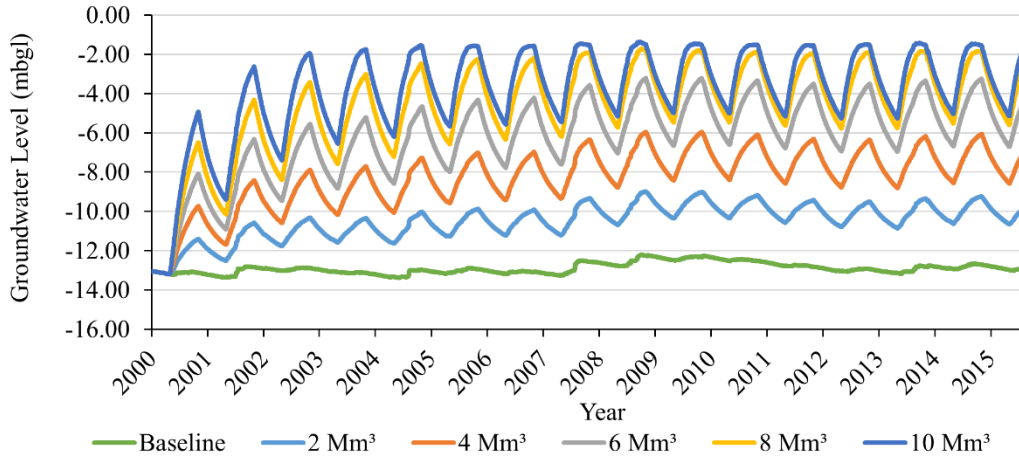
Figure 4.15: Recharge-storage relationship for the Philippi MAR site using infiltration and direct injection

In order to assess the risk to urban infrastructure of recharging water into the CFA at the Philippi site, the groundwater levels for each of the recharge scenarios were compared for the recharge site and two suburbs in close proximity to the recharge site (Figure 4.16). The baseline groundwater levels at the MAR recharge site fluctuated between -12 and -14 m. All the scenarios showed increased groundwater levels in response to increased recharge. All the scenarios indicated increasing groundwater levels from the year 2000 to 2009. After 2009, all the scenarios reached a new equilibrium under the new recharge conditions, indicating that a minimum of nine years is required for the MAR to reach an equilibrium state or maximum storage capacity. In order to assess the risk to urban infrastructure, the same threshold value used in the flood mitigation scenarios of -1.5 m was applied in this scenario. The results indicated that all recharge volumes are below -1.5 m, suggesting a limited risk of urban flooding due to MAR activities.

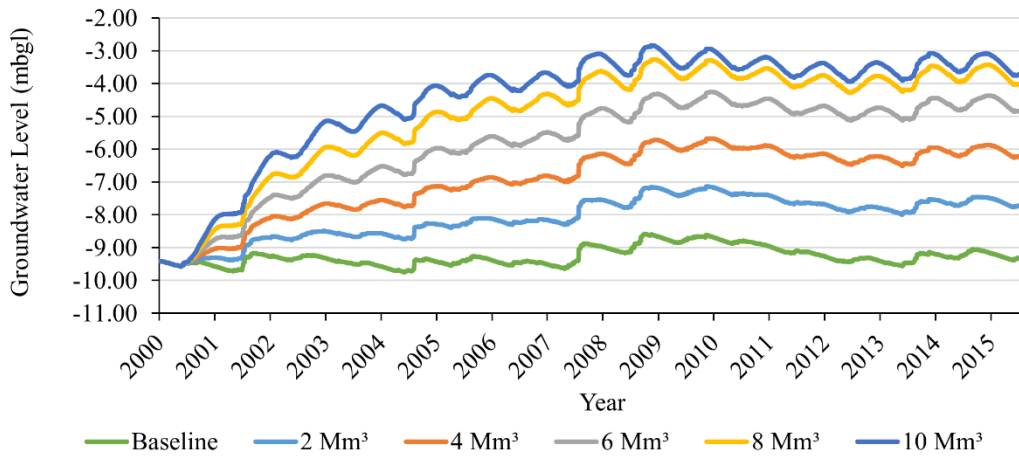
Mitchells Plain is an urban area to the east and up-gradient of the Philippi MAR site (Figure 4.16). The baseline groundwater levels at Mitchells Plain ranged from -8.5 and -10 m. All the scenarios showed increased groundwater levels in response to increased recharge, however the fluctuation in response to recharge was less exaggerated than that of the MAR site. The graph for this site more clearly illustrates increasing groundwater levels from the year 2000 to 2009, after which all the scenarios reach an equilibrium state under the new recharge conditions. All the scenarios were below the threshold depth indicating that there is a limited threat to urban infrastructure and a low risk of flooding at Mitchells Plain even under the highest recharge conditions. The second urban area, Strandfontein, is located south and downgradient of the Philippi MAR site. All the sites showed increases in groundwater levels in response to recharge, however Strandfontein showed a greater variation in water table response due to the hydraulic gradient increasing the rate of flow of groundwater in a south-westerly direction. Despite variation

in hydrogeological response, the groundwater levels at the Strandfontein site are still below the threshold and can be assumed to have no negative impacts on local infrastructure and minimal risk of urban flooding. Thus, it is concluded based on the simulated groundwater levels the risk of urban flooding or damage to urban infrastructure is unlikely with the maximum groundwater levels only reaching -3 m and -4 m in Mitchells Plain and Strandfontein respectively.

Philippi MAR Site



Mitchells Plain



Strandfontein

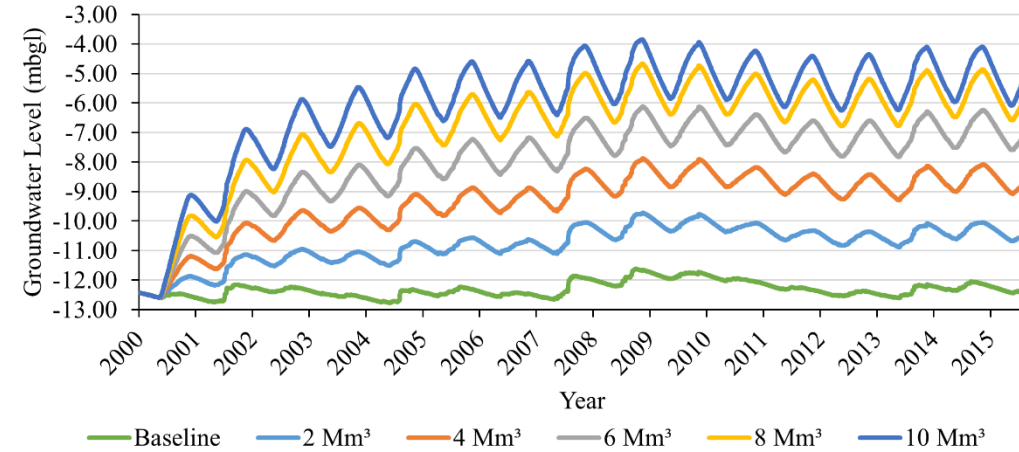


Figure 4.16: Simulated groundwater levels for the five recharge scenarios compared to the baseline from 2000 – 2015

4.2.3.2 Long-term storage potential

The long-term storage potential of the CFA at the Philippi site was evaluated using two additional scenarios that aimed to recharge stormwater into the CFA for long-term storage, also known as water “banking”. The scenario was based on a conservative recharge rate of 5 Mm^3 per year or 15.72 l.s^{-1} during the winter months. Three recharge periods were selected: 1) for the full simulation from 2000 to 2015, 2) from 2000 to 2006 and 3) from 2000 to 2009 (Figure 4.17). The first case was to assess the impact of continuous seasonal recharge for the duration of the simulation period, mainly for comparison purposes. In this case, the increase in storage with winter recharge reached a new equilibrium or steady-state after approximately nine years in 2009, indicating the upper limit to the amount of storage that is possible given this area and its recharge rate. In cases 2 and 3, after six and nine years respectively, summer abstractions were initiated to simulate the demand for stored water after the recharge period. For example, this might occur in a season of prolonged drought. The availability of the stored water was evaluated based on the length of time between the initiation of the summer abstractions until the point where the storage of the CFA declined until it intersected the simulated baseline storage (Figure 4.17). In case two, after six years of recharge, abstraction occurs for three years and one month, dropping below the baseline level in February 2009. However, in case three after nine years of recharge, abstraction occurs for three years and ten months, dropping below the baseline level in November 2009. These results indicated that the amount of retrievable stored water is proportional to the amount of recharge, where an increase of three years’ worth of recharge, from 6 to 9 years, only results in 10 months or one additional season of abstraction.

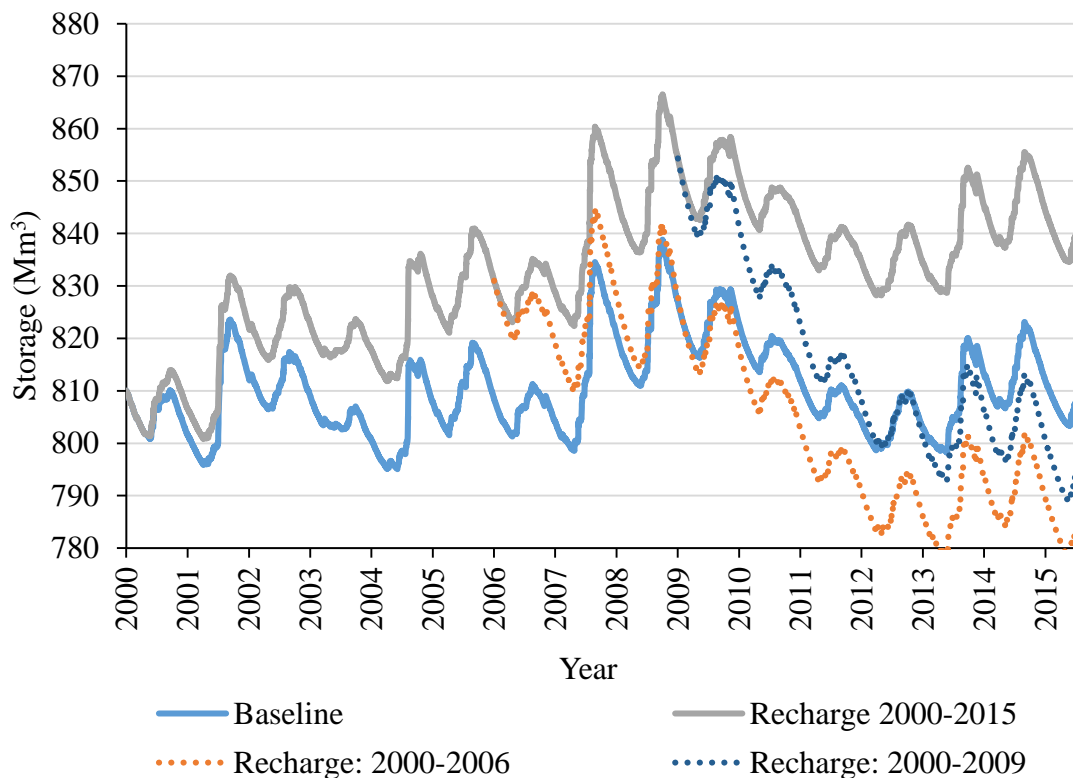


Figure 4.17: The long-term storage of the CFA in at the Philippi site comparing recharge at 5 Mm^3 per year for 6 and 9 years respectively

4.2.3.3 Philippi and Mitchells Plain wellfield yield

The objective of this scenario analysis of MAR is to assess the feasibility of the CFA for its potential to store stormwater on a shorter term basis, where winter recharge is made available for reuse in the following summer months. Thus, the following section examines five recharge and abstraction scenarios at two sites suggested for MAR by this study. Scenarios 15, 16 and 17 were conducted at the Philippi site, while Scenarios 18 and 19 were simulated at the Mitchells Plain site.

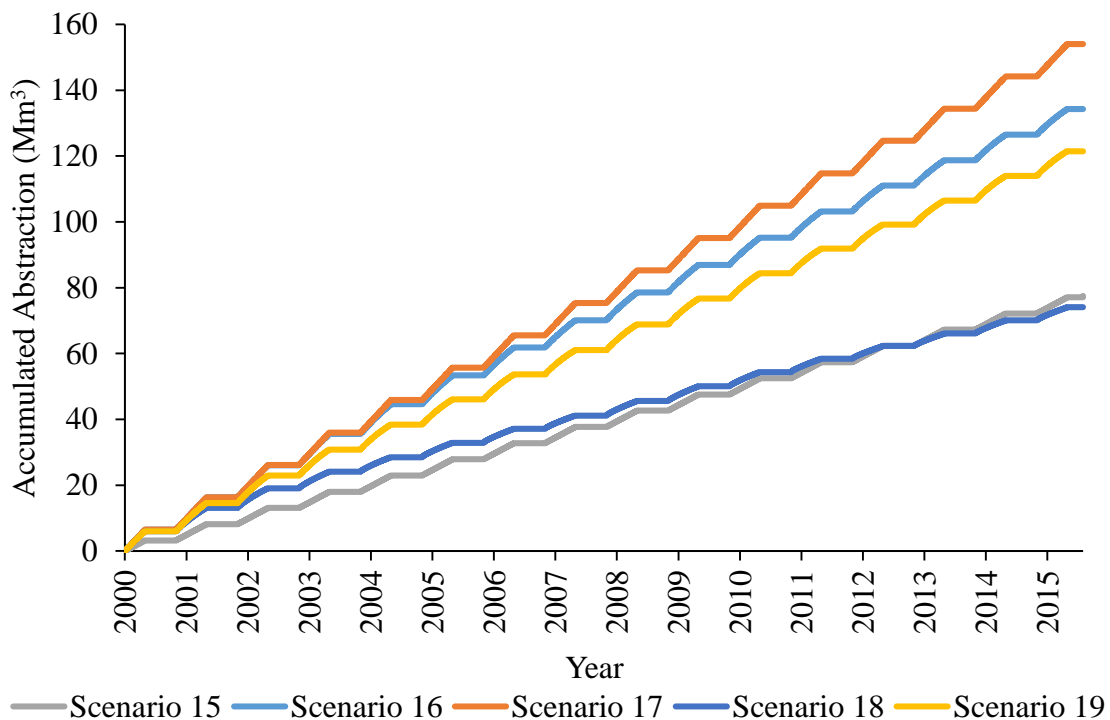
Scenario 15 is a relatively conservative MAR scenario using a 15 l.s^{-1} winter recharge rate and 15 l.s^{-1} summer abstraction rate. In this scenario, the recharged winter stormwater was completely depleted by the summer abstractions cancelling out the recharge gained during the winter months. A recharge and abstraction rate, equivalent to an annual recharge volume of 5 Mm^3 per year, was determined based on the storage threshold for preventing drainage losses as specified previously in Figure 4.15. Additionally, this rate, equivalent to 15 l.s^{-1} , is just less than the suggested sustainable pumping rate of 17 l.s^{-1} based on the work of Tredoux *et al.* (1980) and Vandoolaege (1989). Scenario 16 was an abstraction scenario and did not include recharge. The abstractions were conducted at a maximum abstraction rate of approximately 32 l.s^{-1} during summer. Scenario 17 assessed the impact of maximum abstractions at a rate of 32 l.s^{-1} during summer and incorporating winter recharge at 15 l.s^{-1} . The yield of each of the scenarios was assessed based on the accumulated abstractions. Although a given abstraction rate was specified in a particular simulation, due to the wellfield arrangement, aquifer characteristics and abstraction schedule, some of the boreholes within the wellfield may run dry. In this event, abstraction ceases until the groundwater levels recover to the point where pumping can resume. The expected yield of the wellfield for the simulation period was approximately 77 Mm^3 for Scenario 15, while the expected wellfield yield for Scenarios 16 and 17 is approximately 154 Mm^3 due to the same pumping rate used in both scenarios. The actual wellfield yield of Scenario 15 is equal to the expected yield indicating that the boreholes in this scenario did not run dry and therefore this scenario has a 100% abstraction efficiency. However, Scenario 16 has a reduced efficiency, only abstracting 87 % of the intended volume as the boreholes ran dry on occasion (Table 4.5). However, the recharge of the CFA during the winter months increases the groundwater levels prior to the summer abstraction period, thereby preventing boreholes in the wellfield from running dry and ensuring the expected wellfield yield is achieved.

Given the potential increases to wellfield yield using MAR as demonstrated by Scenario 17, the next step was to see if these results could be transferred to another site, Mitchells Plain. Scenarios 18 and 19 were simulated at Mitchells Plain and consisted of the same simulation procedure as those conducted for Scenarios 16 and 17 at the Philippi site. The same expected yield for both scenarios was 154 Mm^3 , however both scenarios show greatly reduced yields when compared to the corresponding scenarios at Philippi, with Scenario 18 showing a 48 % reduction compared to scenario 16 and Scenario 19 showing a 21 % reduction compared to Scenario 17. This is due to the reduced aquifer thickness at Mitchells Plain resulting in less storage volume around the wellfield to maintain borehole yields. The implementation of MAR did improve the wellfield yield by 39 %, however considering the recharged volume was 50 % of the abstracted volume, suggests that the returns on the recharged water are not observed.

Table 4.5: The wellfield yields for MAR scenarios at Philippi and Mitchell Plain

Scenario	MAR Site	Expected Wellfield Yield (Mm ³)	Actual Wellfield Yield (Mm ³)	Efficiency (%)
Scenario 15	Philippi	77	77	100
Scenario 16	Philippi	154	134	87
Scenario 17	Philippi	154	154	100
Scenario 18	Mitchells Plain	154	74	48
Scenario 19	Mitchells Plain	154	121	79

These wellfield yield results are further supported when assessing the accumulated abstractions for Scenarios 15, 16 and 17 at Philippi (Figure 4.18). Scenario 15 showed consistent pumping and rates until the total of 77 Mm³ was achieved with no incidents of borehole drying. Scenarios 16 and 17 showed similar increases in the initial stages of the model simulation up until the year 2004. After 2004, Scenario 16 began to reduce the level of the groundwater table in the aquifer to the extent that the boreholes began to dry out showing increasing declines in yield from the wellfield. Alternatively, Scenario 17 maintained the specified abstraction rates and did not experience declines in yield because of the drying boreholes. At Mitchells Plain, the accumulated abstractions were significantly lower than the equivalent pumping rate at Philippi (Scenario 16) and after 15 years the wellfield yield was even less than the conservative MAR scenario. The addition of 5 Mm³ of stormwater recharge to improve the wellfield yield did show increases in the accumulated yield but remains well below the yields experienced at Philippi.

**Figure 4.18: Accumulated abstracted water volume for Scenarios 15, 16, 17 18 and 19**

4.2.3.4 Mitchells Plain MAR flood risk

Unlike Philippi, the land use of the Mitchells Plain site consists of predominantly urban and residential land use, meaning that there are risks to urban infrastructure if recharged stormwater raises the groundwater levels excessively in the area. Additionally, due to the urban land use, there is limited space for the infiltration of stormwater. Thus, for the simulation of MAR at this site, it was assumed that the most appropriate means of recharging stormwater was the use of injection wells. Twenty injection wells were located approximately 1 km inland from the False Bay coast. This scenario (Scenario 19) is identical to that of Scenario 17, with a winter recharge of approximately 15 l.s^{-1} , equivalent to 5 Mm^3 per year and an abstraction rate of approximately 32 l.s^{-1} , equivalent to 10 Mm^3 per year.

The mean groundwater levels were mapped (Figure 4.19) to assess the potential impact of Scenario 19 on the overlying urban infrastructure. Under baseline conditions, the mean groundwater levels at Mitchells Plain are relatively high exceeding -5 m over much of the area, increasing to between $-0.9 \text{ m} - 0 \text{ m}$ in the north east of Mitchells Plain. These elevated mean groundwater levels are not conducive to recharge, due to limited storage capacity. Additional recharge could increase the mean groundwater levels to a point that could result in the damage of overlying infrastructure. Scenario 19 shows a net reduction in mean groundwater levels at Mitchells Plain with much of the area below -5 m and the areas that previously had mean groundwater levels in excess of -2.9 m have been significantly reduced.

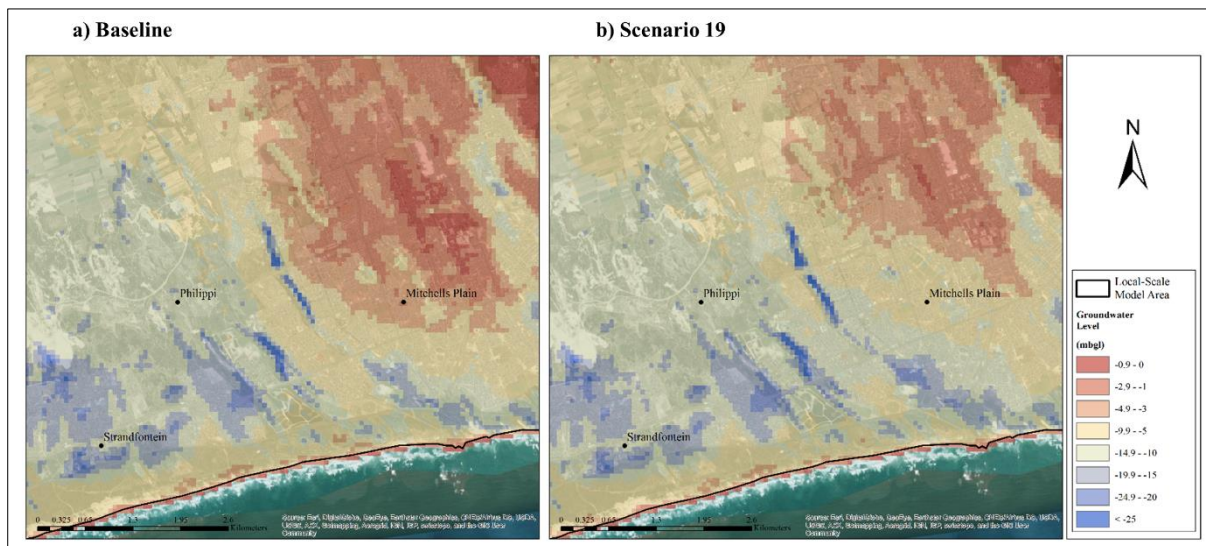


Figure 4.19: Groundwater levels for the baseline scenario and Scenario 19 for the Mitchells Plain MAR site

4.2.4 Seawater intrusion

Due to the close proximity of the Philippi and Mitchells Plain MAR sites to the False Bay coast, it was important to ensure that the risk of seawater intrusion was mitigated. The change in simulated hydraulic gradient and the extent and magnitude of the cone of depression were evaluated to assess the risk of seawater intrusion. Figure 4.20 shows the minimum groundwater head elevation for the baseline conditions and Scenarios 15 and 16 at Philippi. The minimum head elevation was selected to assess the absolute lowest groundwater levels obtained for the simulation period. The baseline simulation indicated that there was a strong hydraulic gradient with groundwater levels above 20 meters above mean sea-level (masl) in the north eastern parts of the model area hydraulically up-gradient of the MAR site, while the MAR site itself has a hydraulic head elevation of between 10 and 20 masl. The hydraulic gradient is consistent along the False Bay coastline, going from sea-level and reaches 5 masl at approximately 1 km inland. The hydraulic head elevation of Scenario 15 indicates that minimal drawdown at the MAR site was observed. The hydraulic gradient along the False Bay coastline remains largely unaffected by the recharge and abstraction at the MAR site. Furthermore, there appears to be an increase in groundwater head elevation at the MAR site and increases in groundwater head elevation near Zeekoeivlei, suggesting potential discharge of groundwater into Zeekoeivlei. Scenario 16 represents the upper limits of the possible drawdown using the 20 borehole arrangement and by pumping at 32 l.s^{-1} during the summer months. Scenario 16 shows significant drawdown at the MAR site to well below -5 m reaching a maximum drawdown of around -16 m. The hydraulic gradient along the coast is reduced in Scenario 16, where the area between the MAR site and the False Bay coastline is reduced to a maximum of 1 m compared to a maximum of 5 m under the baseline conditions. The hydraulic gradient, while reduced, still maintains a north-south inclination.

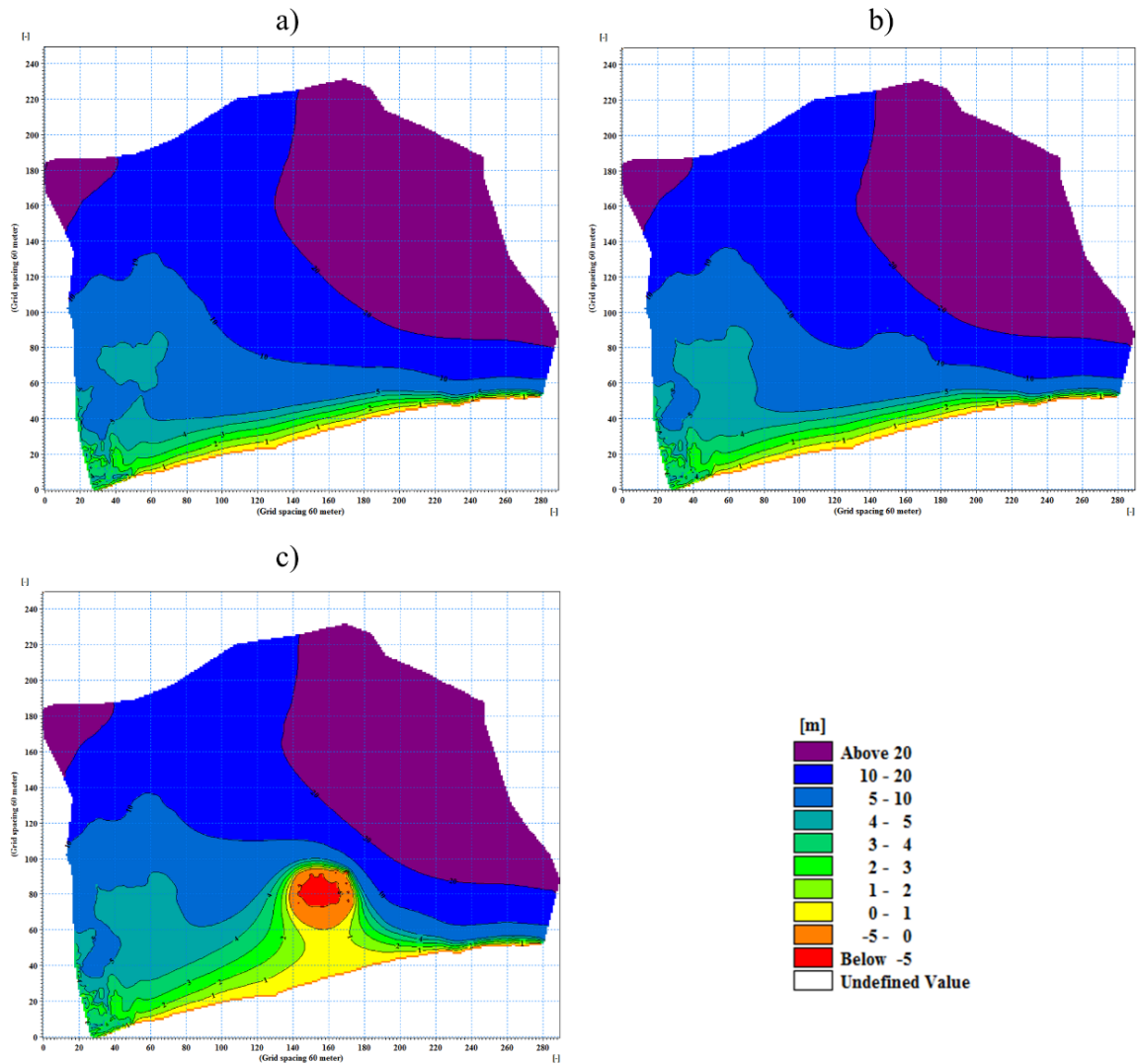


Figure 4.20: Groundwater minimum head elevation (masl) at Philippi for a) the baseline conditions, b) Scenario 15 and c) Scenarios 16

The mean cone of depression for Scenario 16 further highlights the impacts of the maximum drawdown conditions (Figure 4.21). As indicated by the graphed groundwater elevation profile for Scenario 16 the minimum groundwater elevation was approximately 5 m below the mean sea-level, indicating that despite the fact that the north-south hydraulic gradient was maintained in scenario 16, groundwater levels below sea-level may be at risk of saltwater upconing. In order to minimise the risk of seawater intrusion and to maximised the amount of water available for abstraction, an additional scenario (Scenario 17) was simulated that consisted of summer abstractions at the theoretical maximum rate of $32 \ell.s^{-1}$ while the injected recharge was simulated at a rate of $15 \ell.s^{-1}$. It was assumed that a lower recharge rate would lower the cost of recharging water while still maintaining a sustainable yield of approximately 10 Mm^3 at the maximum abstraction rate.

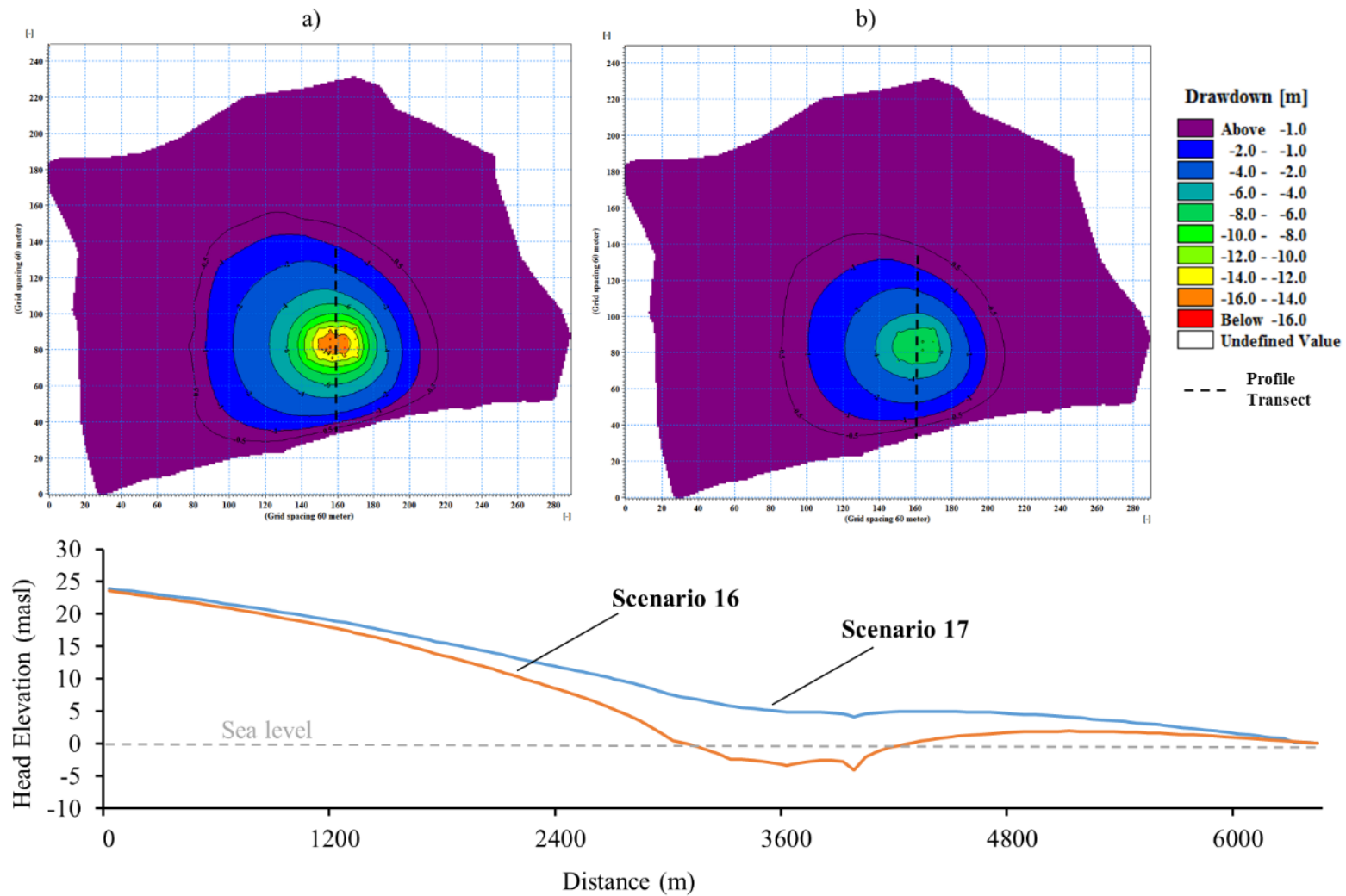


Figure 4.21: Simulated mean cone of depression and the groundwater elevation profiles at Philippi for a) Scenario 16 and b) Scenario 17

Based on the cone of depression and groundwater elevation profile for Scenario 17 (Figure 4.21) the recharge considerably offsets the drawdown effects for abstractions at 32 l.s^{-1} . The maximum drawdown for Scenario 17 was half that of Scenario 16 at approximately -8 m with an average of between -6 and -8 m for much of the MAR area. This was supported by the groundwater elevation profile indicating that the magnitude of the drawdown was reduced. Furthermore, the reduction in drawdown ensured the cone of depression did not extend below sea-level, thereby reducing the risk of seawater intrusion or upconing of seawater.

The drawdown impacts of Scenarios 18 and 19 at the Mitchells Plain site were also examined. Scenario 18 represents the mean drawdown under maximum summer pumping (32 l.s^{-1}) with no winter recharge. The mean drawdown under this scenario is between -12 and -10 m (Figure 4.22a), which was less than the same abstraction rate in Scenario 16 (Figure 4.21a) at the Philippi site. The reason for the reduced drawdown impact is the shallower aquifer depth at Mitchells plain, meaning that boreholes dried out quicker due to the weaker hydraulic gradient in the vicinity of the Mitchells Plain wellfield. Similar impacts were seen in Scenario 19, however due to the additional winter recharge, the drying out of boreholes was less frequent. When compared to Scenario 17, Scenario 19 showed a drawdown of between -4 and -6 m, which was only 1 – 2 m higher than that of Scenario 17. Additionally, the profiles of mean groundwater elevation for both Scenarios 18 and 19 showed that they were well above sea-level, indicating a limited risk of seawater intrusion and up-coning of seawater into the boreholes. This is primarily due to the strong north to south hydraulic gradient and the elevated bedrock in the area preventing the flow of seawater in a northerly direction into the CFA.

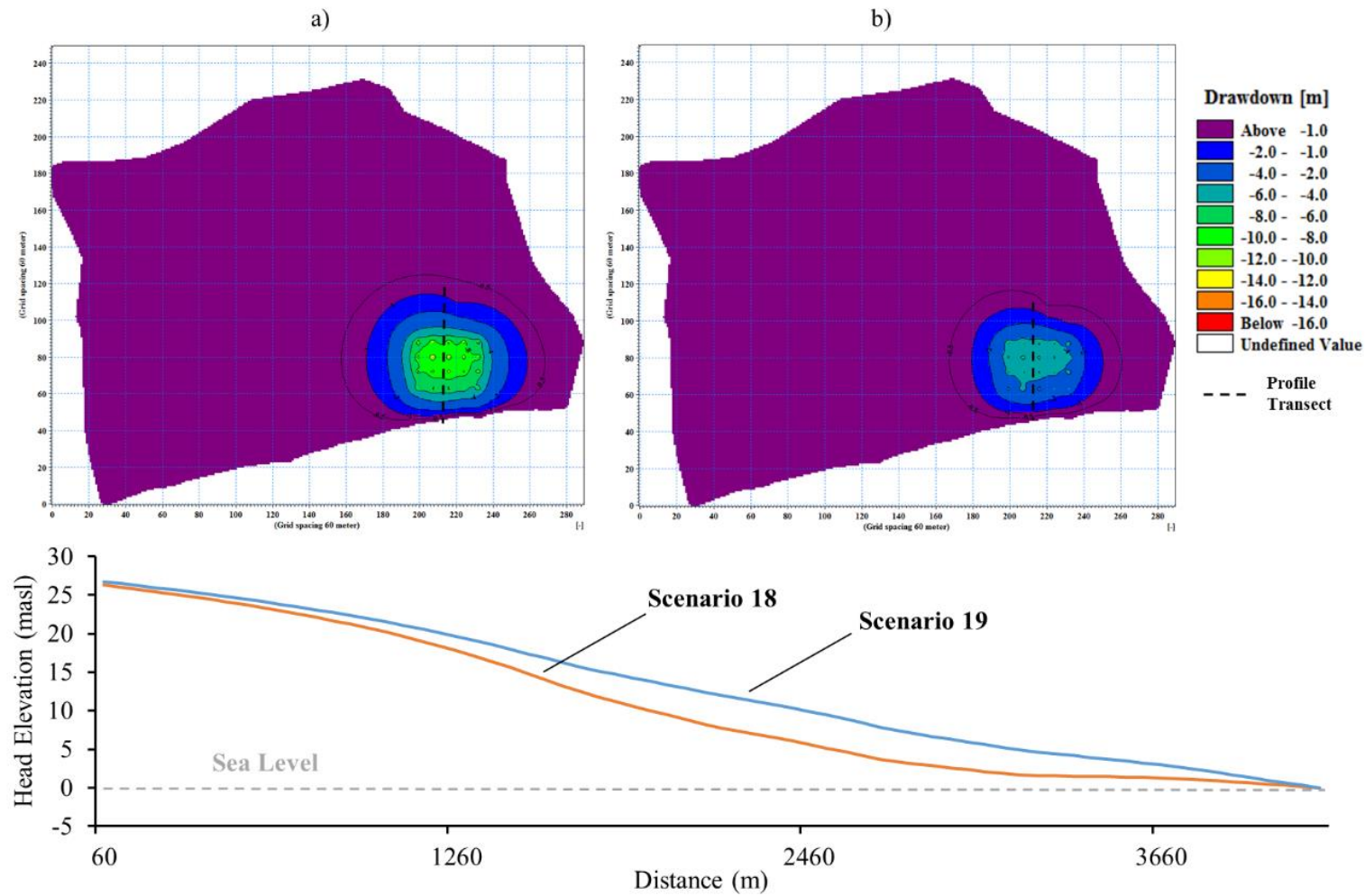


Figure 4.22: Simulated mean cone of depression and the groundwater elevation profiles at Mitchells Plain for a) Scenario 16 and b) Scenario 17

4.3 Water quality

Some basic water quality simulations were performed for the flood mitigation sites (Figure 4.23) and the MAR sites (Figure 4.24). These simulations were conducted to assess the flux of potential contaminants from areas in close proximity to poor quality stormwater such as the informal settlements of Sweet Home and Graveyard Pond, or at sites where stormwater is being considered for injection and subsurface storage. The water quality modelling at the flood sites compared the migration of a TDS pollutant plume under baseline conditions and flood mitigation scenarios. The model was run from the year 2000 to 2100 with the extent of the plumes for each scenario assessed after 25, 50 and 100 years (Figure 4.23). Under baseline conditions, the TDS plume moves through the saturated zone relatively unimpeded. The contaminant plume from Graveyard Pond moves in a westerly direction, merging with the contaminant plume from Sweet Home by 2025. The contaminant plume from Sweet Home was shown to move in two directions, where plumes with weak concentrations started to migrate in northerly and southerly directions by 2025. In 2050, the plume extending in a southerly direction became more extensive than the one to the north of Sweet Home, extending well into the central parts of the PHA. Furthermore, the TDS concentrations at Sweet Home have increased by the year 2050 to above $1 \times 10^7 \mu\text{g.m}^{-3}$. By 2100 these concentrations are shown to have continued to increase, with a large area surrounding Sweet Home exceeding $1 \times 10^6 \mu\text{g.m}^{-3}$ and localised areas that exceed $1 \times 10^8 \mu\text{g.m}^{-3}$. The weaker contaminant plume extending in a southerly direction has continued to extend in 2100. The plume has followed the general flow patterns of groundwater in the area, flowing into central parts of the model area, through the vicinity of Zeekoeivlei and exiting along the False Bay coast. Therefore, the flow path of the contaminant plumes resulting from the stormwater at Sweet Home and Graveyard Pond, under baseline conditions, is likely to have negative impacts on local wetlands such as Zeekoeivlei. The addition of groundwater abstractions at the two flood mitigation sites sees a significant reduction in the extent of the contaminant plumes for all the future dates. In particular, the weaker contaminant plumes prevalent in the baseline scenario were prevented. However, elevated contaminant levels continue to persist in the vicinity of each site.

The same simulation procedure was applied for the MAR sites, Philippi and Mitchells Plain, assessing the migration of a TDS contaminant plume under baseline conditions and abstraction and recharge conditions. The contaminant plume showed similar concentrations and extent at 2025, 2050 and 2100, due to the strong hydraulic gradient in the southern parts of the CFA diluting the TDS concentration by rapidly moving water towards False Bay and preventing a build-up of TDS in any location. With the addition of abstraction and recharge, there is a similar trend where the concentration and extent of the TDS plume remains consistent throughout the simulation period. However, the concentration of TDS is much higher in the vicinity of both MAR sites, with the central parts of the wellfields exhibiting concentrations greater than $1 \times 10^8 \mu\text{g.m}^{-3}$, while the outer sections of the wellfields generally show lower concentrations of between $1 \times 10^6 \mu\text{g.m}^{-3}$ and $1 \times 10^7 \mu\text{g.m}^{-3}$. The extent of the plumes were reduced from those exhibited in the baseline scenario. Philippi showed almost no lateral flux of TDS concentrations, while at Mitchells Plain the plume extending towards the False Bay coast was slightly reduced. However, the concentrations of the Mitchells Plain plume exceeded those of the baseline, with a large extents displaying concentrations of greater than $5 \times 10^7 \mu\text{g.m}^{-3}$ and extend all the way to the False Bay coastline. The steeper

hydraulic gradient and greater aquifer depth at Philippi means that seasonal abstractions and recharge prevent the migration of TDS. However, at the Mitchells Plains site, the aquifer is shallower and the bedrock is above sea-level meaning that the hydraulic gradient at Mitchells Plain was still strong enough under MAR conditions to carry the TDS plume south towards False Bay.

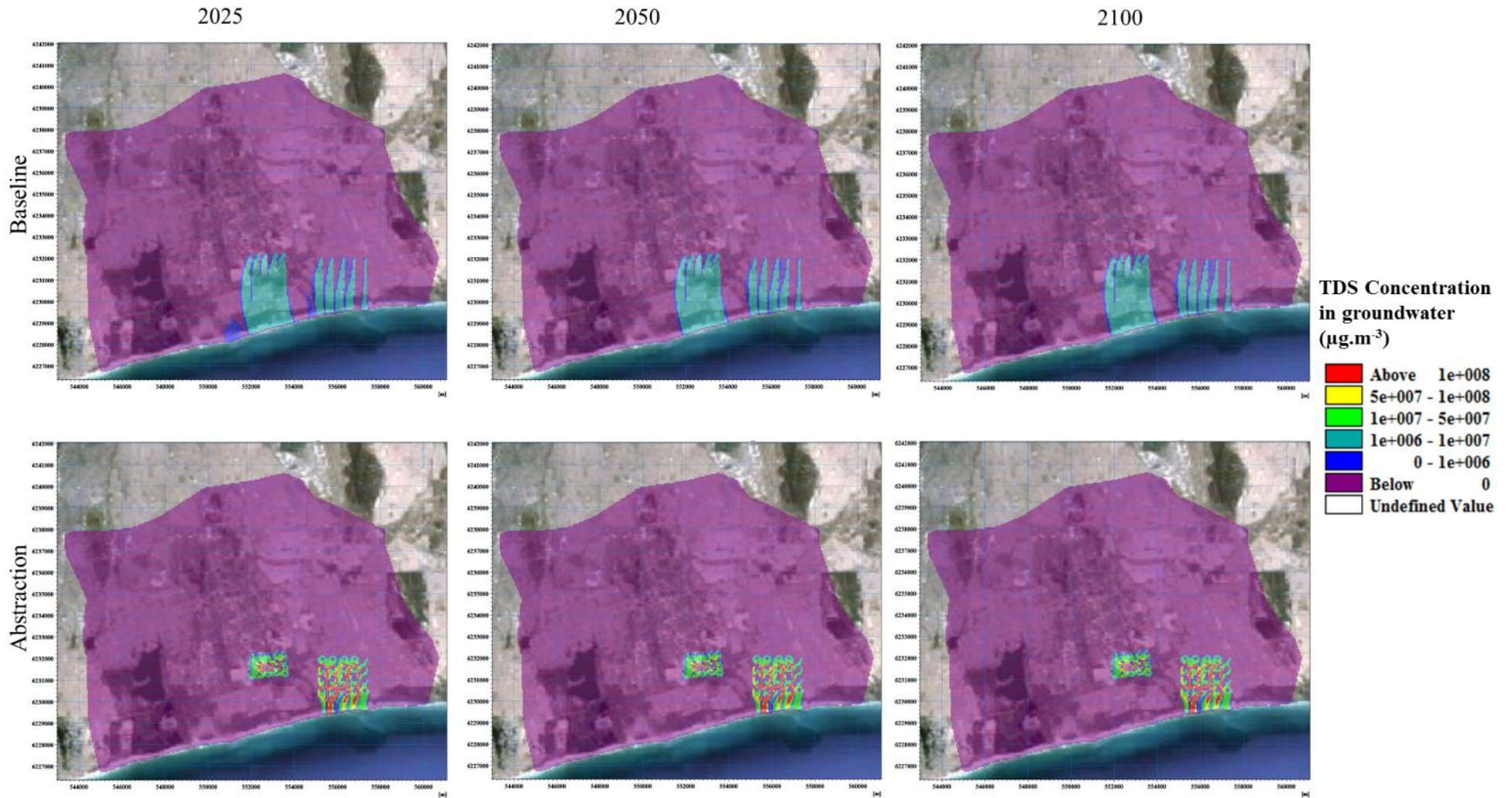


Figure 4.24: Total Dissolved Solids (TDS) concentration in groundwater for the MAR sites of Philippi and Mitchells Plain, comparing the transport of contaminants under baseline and MAR conditions after 25, 50 and 100 years

4.4 Climate change

The results of the climate change simulations with MIKE SHE were focused on the water balance and changes in the saturated zone. The climate change scenarios considered the changes in temperature and rainfall from a historical period (1980 – 2010) to a future period (2040 – 2070). Four GCMs were selected for hydrological modelling with MIKE SHE, *viz.* IPSL-CM5A-MR, MRI-CGCM3, MIROC5 and GFDL-ESM2M. These models represent the following future climate conditions: “much hotter and much drier”, “much hotter and slightly wetter”, “slightly hotter and slightly drier” and the “median” conditions respectively.

The changes to the water balance of the hydrological cycle (Figure 4.25) can be used to evaluate the sensitivity of the model to changes in future temperature and rainfall. All the scenarios except for MRI-CGCM3 predicted decreases in rainfall. MIROC5 and GFDL-ESM2M showed similar reductions in precipitation at -6 and -4 % respectively. However, GFDL-ESM2M was much hotter than MIROC5, yet the effects on the water balance were similar for both GCM's. This suggests that elements of the hydrological cycle such as groundwater recharge and discharge are less sensitive to change in temperature than rainfall. This notion was confirmed when comparing IPSL-CM5A-MR and MRI-CGCM3, both models predict high future temperatures, however MRI-CGCM3 predicts slightly higher rainfall. MRI-CGCM3 showed minimal changes to the water balance of the hydrological cycle except for discharge which showed a slight reduction. The reduction in drainage suggests that changes in temperature have a greater impact on drainage than on the other components of the water balance such as recharge. IPSL-CM5A-MR shows a 24 % reduction in rainfall which resulted in a large reduction in recharge of around 66 % and drainage was reduced by 64 %. This suggests that the hydrological cycle is highly sensitive to changes in rainfall, where an increase in rainfall from 4 % in GFDL-ESM2M to 24 % in IPSL-CM5A-MR results in a more than doubling of recharge and discharge.

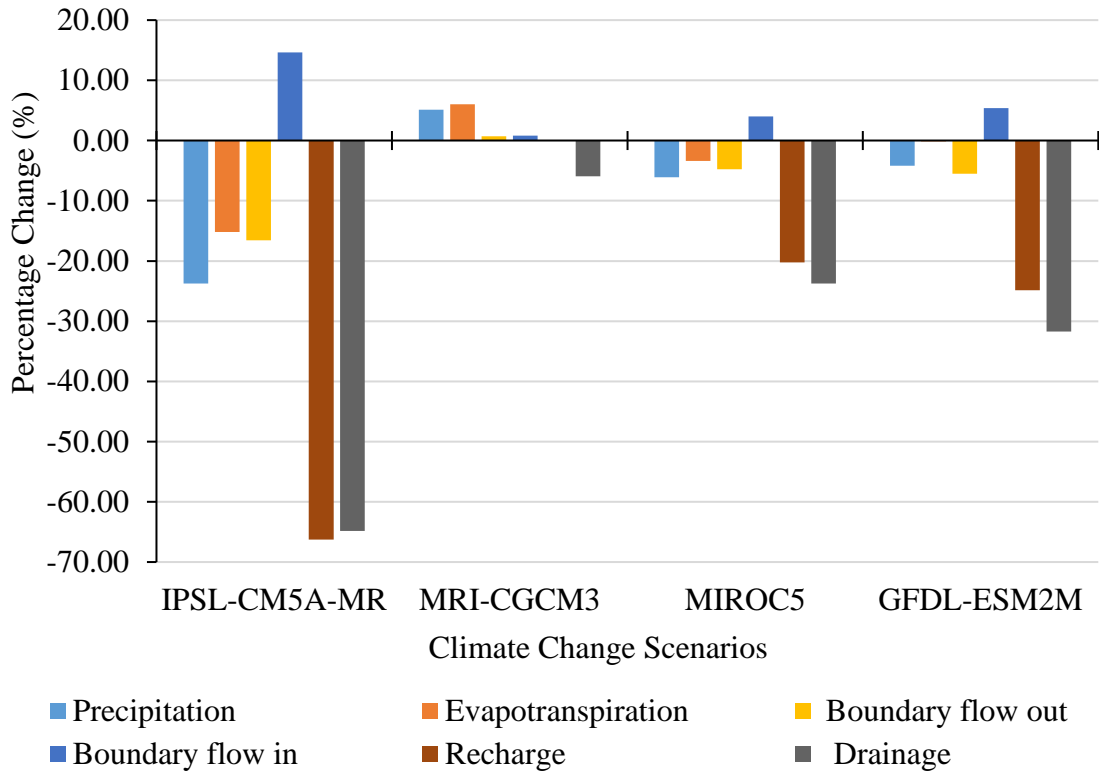


Figure 4.25: The response of the hydrological cycle to scenarios of future climate change (2040-2070)

Figure 4.26 shows the change in groundwater storage for the southern section of the CFA. The results indicated that MRI-CGCM3 showed similar seasonal trends to that of the baseline simulation, with only a marginally greater storage due to the slightly greater rainfall projected by MRI-CGCM3 and despite the higher projected temperature. This further supports the notion that higher future temperatures have less of an impact on the groundwater response than rainfall. MIROC5 and GFDL-ESM2M showed declining groundwater storage trajectories ending the simulation 71 mm and 81 mm below the baseline storage. Both these simulations have similar reductions in rainfall of 6 % and 4 % respectively, while GFDL-ESM2M exhibits a higher increase in ET_o . Thus, the greater declines in groundwater storage can be attributed to the elevated ET_o , given that the greater reduction in rainfall experienced in MIROC5 did not result in greater groundwater storage losses. The greatest declines are seen by IPSL-CM5A-MR where the combination of significant reductions in rainfall and increases in ET_o , have resulted in a 256 mm or 5 % reduction in groundwater storage at the end of the simulation.

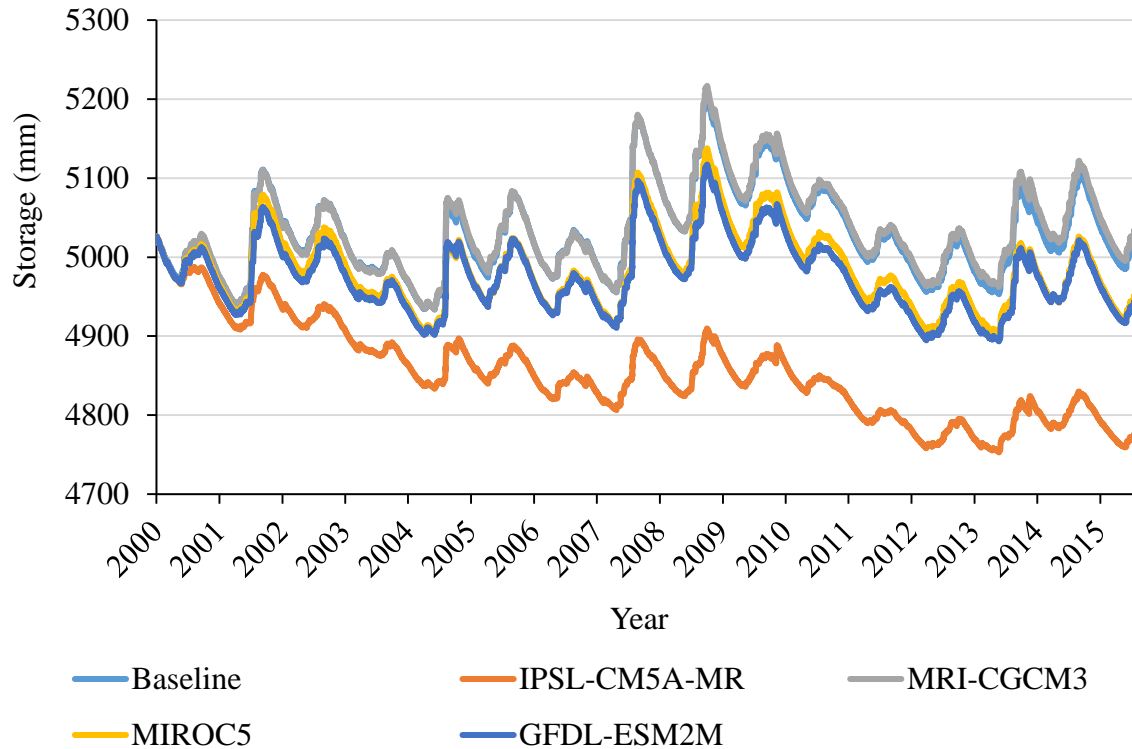


Figure 4.26: The impacts of climate change of groundwater storage from 2000 – 2015

4.5 Summary

The results of the scenarios simulated with MIKE SHE identified the MAR potential for the CFA, while also developing the understanding of the regional flow and hydraulic potential of the CFA. Furthermore, the application of an integrated hydrological model enabled these groundwater processes to be examined within the complete water cycle, while accounting for other influences such as land use and land use change. Most notable was the ability to simulate daily recharge, which has not been performed on the CFA previously.

A local-scale model was established to assess the impacts of the MAR scenarios evaluating (i) opportunities to mitigate groundwater related flooding, and (ii) to assess the storage capacity and recharge potential of the areas of the CFA with the highest MAR potential. The local-scale modelling was also utilised to assess the impacts of additional factors such as contaminant transport and the potential impacts of climate change. The following section discusses these modelling results in relation to the broader fields of MAR and WSUD specifically with regard to the aim and objectives of this research.

5. Discussion

This chapter discusses the results of the hydrological modelling to determine the feasibility of MAR on the CFA. In addition to the description of the regional hydrological and hydrogeological processes and mapping of the MAR potential of the CFA, the MIKE SHE simulations aimed to firstly, determine the feasibility of seasonal groundwater abstractions for the purpose of flood mitigation, and secondly determine the storage potential of the CFA for stormwater recharge. These two modelling objectives are evaluated in relation to the risks of aquifer contamination and climate change. Finally, this chapter discusses the role of MAR in the application and implementation of WSUD.

5.1 Regional Hydrogeology and Recharge

The groundwater flow in the CFA was shown to be largely determined by the regional groundwater elevation which is influenced by the local topography and underlying bedrock. The elevation of the water table of the CFA decreases from the northeast towards the False Bay coastline in the south as a result of decreasing topographic and bedrock elevations. Consequently, the regions with the highest groundwater elevation are generally associated with higher rates of groundwater flow. Groundwater in the north-east tends to flow in a southerly direction to False Bay, whereas the groundwater near Table Mountain in the west, flows north into Table Bay. The same is true for the groundwater in the north and north-western parts of the CFA. The groundwater flow in the central parts of the CFA along the Lotus River basin is higher than the surrounding areas due to the higher hydraulic conductivity of the coarse gravel within the paleochannel, which dominates the central and southern parts of the CFA. These findings agree with the conceptual models and results of past studies of the CFA, most notably those of DWAF (2008).

The simulated groundwater recharge, experienced under altered land use conditions between 1983 and 2013 were compared. The land use for 1983 represented the recharge under 'historical' conditions and the land use for 2013 represented recharge under 'current' conditions, assuming the same climate, soil and aquifer conditions so as to isolate the influence of land use change. Simulating the impacts of urban land use over the same period (2000 – 2015) meant that interference from varying climatic conditions could be excluded. It was shown that reductions in groundwater recharge extend over much of the model area under current land use conditions when compared to the historical land use conditions. The reductions in recharge occurred in the central and northern regions of the Cape Flats which experienced between 0.2 and 0.3 mm.day⁻¹ under historical conditions, however under current conditions the recharge has been reduced by 0.1 and 0.2 mm.day⁻¹. Further declines in recharge were noted in the southern parts of the Cape Flats, there is a reduction of between 0.1 and 0.2 mm.day⁻¹, with values of > 0.4 mm.day⁻¹ under historical land use, which was reduced to between 0.3 – 0.4 mm.day⁻¹ under current conditions. The reduction in recharge can be attributed to the increase in urban land use that has occurred on the Cape Flats since the 1980's. Much of the urban development occurring over the Cape Flats has replaced natural and agricultural land uses which has reduced groundwater recharge. This suggests that further urban development of the Cape Flats is likely to have negative implications for the natural recharge of the aquifer. Thus, artificial recharge of the CFA using MAR may become increasingly important in the future to sustain the aquifer's resource potential.

The results of the regional-scale model and its correlation with past studies serves to support the conceptual hydrological model utilised in this study. Regional flow and groundwater head elevation were essential for the calibration and validation of the model which could also incorporate surface water processes at a regional-scale. Groundwater elevation was of particular importance as these results were used to develop the MAR potential maps that were essential for confirming the available storage of the CFA and selecting sites that were appropriate for MAR scenario analysis. Furthermore, incorporating the complete water cycle for calibration and validation provided additional support for the theorised conceptual hydrogeological model, as this allowed the daily calculation of groundwater recharge which is influenced by land use and soil properties that were not fully represented in past modelling applications on the CFA. The decline in groundwater recharge on the CFA over the last 30 years suggests that there an opportunity for MAR to supplement natural recharge with artificial recharge using stormwater or treated wastewater.

5.2 MAR: Flood mitigation

The study assessed the feasibility of actively lowering the water table in parts of the CFA in order to mitigate flooding. Flooding is a serious problem on the Cape Flats, particularly in the informal settlements of Sweet Home and Graveyard Pond, which have been linked to high groundwater levels that exceed the ground level in a number of past studies (Drivdal, 2011; Musungu and Drivdal, 2011; Pharoah, 2013), often referred to as ‘seepage’ or ‘rising flooding’ (Pharoah, 2014). High levels of poverty and unemployment in these areas also mean that people are more vulnerable to the impacts of flooding.

A threshold approach was used to evaluate the flood risk of these areas, where the relative potential for flooding based on the exceedance levels of the groundwater table was assessed, rather than the actual flooding or ponding of groundwater. This was due to the relatively coarse structure of the MIKE SHE model that was required to simulate flooding and aquifer processes at a large scale. In addition, due to limitations of cost and the scope of this study, it was not possible to develop and calibrate a very high resolution (i.e. <10 m) model for each flood site as currently available data do not support such models, and this would have required extensive investment in groundwater monitoring and geological investigation at a local-scale. The use of a physically-based hydrological model meant that it was possible to meaningfully represent the general storage and hydraulic properties of the CFA for the purpose of determining the feasibility of the MAR using the CFA as demonstrated in the calibration and validation stages (Chapter 3.3). The physically-based hydrological model was conducted at a regional-scale (500 m resolution) and local-scale (60 m resolution). The utilisation of hydrological models that have a more detailed, physically-based representation of the hydrological cycle have been shown to be valuable in ungauged or data scarce applications in South Africa, which are vital for understanding hydrological processes and decision making (Schulze, 1995a, 2004; Warburton *et al.*, 2010; Smithers *et al.*, 2013).

The flood mitigation threshold was based on the proximity of the flood-prone areas to areas with elevated simulated groundwater levels that exceed -1.5 m more than 80 % of the time, in the period from 2000 to 2015. Therefore, the assumption was that flooding in these areas is known to be a result of groundwater seepage, and areas with groundwater levels that regularly exceed the assigned -1.5 m threshold have a higher likelihood of experiencing groundwater related flooding.

5.2.1 Sweet Home

Three summer abstraction scenarios (9BH, 18BH and 27BH), at rates of 3 l.s^{-1} and 5 l.s^{-1} , were simulated to represent a potential fit-for-purpose water supply using groundwater in the vicinity of Sweet Home. The results of the Sweet Home flood mitigation scenarios showed limited success in reducing groundwater levels to below the threshold level with an abstraction rate of 3 l.s^{-1} , only Scenario 5, with 27 boreholes, showed some success. The only Scenarios that were consistently below the threshold value were observed in the scenarios with 18 and 27 boreholes at abstraction rates of 5 l.s^{-1} . The groundwater mapping and profile transects at Sweet Home were used to determine if the drawdown was sufficient to meet the requirements of the flood mitigation threshold and determine the extent of the influence of the groundwater abstractions. These scenarios indicate that it is possible to reduce groundwater levels to below the flood threshold based on the specified abstraction rates and borehole arrangement. However, in order to achieve this, a substantial number of boreholes (18 – 27 boreholes) are required to attain the specified flood mitigation objectives. The relatively large number of required boreholes has obvious cost implications. Colvin and Saayman (2007), based on survey data, estimated that the cost of installing a cased borehole in the Philippi area was approximately 20 000 ZAR, making the total costs of such a scenario approximately 360 000 – 540 000 ZAR for the installation of these boreholes, excluding pumps and operation and maintenance costs. Therefore, due to the likely costs flood mitigation at Sweet Home the fit-for-purpose water supplied from this scheme would be associated with a higher cost of water. As a result, the use of the water becomes more important as the water user will need to be able to afford the cost of the scheme so that the flood mitigation potential can be realised.

The area influenced by the drawdown of groundwater levels suggested that this could impact local artificial wetlands such as the Edith Stephens Wetland (Day, 2016). The reduction in mean groundwater levels at the Edith Stephens Wetland was simulated to be around 1.5 – 2 m. The reduction in groundwater levels at the Edith Stephens wetland could have a number of negative impacts including the loss of biodiversity (natural or artificial) and a reduction in the effectiveness of these abstraction scenarios for flood mitigation. In an effort to minimise the impact of the scenario when the total yield is highest (Scenario 6), an additional scenario (Scenario 7) was tested to assess possible outcomes if the wellfield design could reduce the impacts to the wetland. By shifting 6 boreholes a further 1 km away, from the wetland from the western side of Sweet Home to its eastern boundary, it was possible to reduce the drawdown of approximately 1m. Of significant concern on the Cape Flats is that seasonal wetlands rely on periods of saturation during winter and drying out completely during summer. These seasonal wetlands, where they still exist, are important for endemic species (Day, 2016). However, flooding on the Cape Flats and the detention of stormwater in ponds and artificial wetlands can result in inundation all year round, which has a negative impact on species that require saturated-unsaturated cycles (Day, 2016). This means that the drawdown in the Sweet Home area could have a secondary benefit of ensuring the seasonal drying out of local wetlands during summer, while the diversion of stormwater to these wetlands in winter would ensure periods of inundation.

5.2.2 Graveyard Pond

The fit-for-purpose abstraction scenarios that were simulated for Graveyard Pond, with three summer abstraction scenarios (4BH, 8BH and 12BH), at rates of 3 l.s^{-1} and 5 l.s^{-1} , showed that the 8 and 12 BH arrangements both significantly reduced the groundwater levels to below the flood threshold. The 8BH scenario at 5 l.s^{-1} (Scenario 11) and the 12BH scenarios at 3 l.s^{-1} (Scenario 12) showed similar groundwater fluctuations, with both only exceeding the threshold on three occasions during the wet period from 2007 to 2010. Scenario 13, with 12 boreholes at the highest abstraction rate, reduced the local groundwater levels to below the flood threshold for the entire simulation period including the wet years from 2007 to 2010. Scenarios 11, 12 and 13 all reduced groundwater levels to below the flood threshold more than 88 % of the time.

The results of scenarios 11, 12 and 13 indicated that summer abstractions at Graveyard Pond provide a reasonable means of mitigating groundwater levels that could potentially cause flooding by reducing groundwater levels to below the flood mitigation threshold. The best performance was achieved in Scenario 13, with a 12 borehole arrangement pumping at 5 l.s^{-1} . Scenario 13 would be a more expensive option given the higher abstraction rate and greater number of boreholes required. However, based on Scenarios 11 and 12, these costs could be reduced with only marginal increases in the time that the local groundwater levels exceed the threshold. Based on the simulation of these scenarios, the flood mitigation threshold was only exceeded during three wet seasons between 2007 and 2010. Therefore, by implementing Scenarios 11 and 12 it would be possible to reduce the frequency and severity of flooding during the wet seasons, while reducing the total cost of the intervention.

The cone of depression for the Graveyard pond scenarios are comparative to those at Sweet Home, despite significantly lower total volumes of water abstracted. This was attributed to the thinner aquifer layer at Graveyard Pond, meaning that there is less water volume available for abstraction within the immediate vicinity of the wellfield. Thus, the extent of the cone of depression is relatively more extensive as water from further afield migrates towards the source of abstraction to replace the abstracted water. Despite the comparatively similar cone of depression to that at Sweet Home, there are no likely impacts to local groundwater dependent ecosystems, as there are no wetlands or rivers within the area of influence of the abstractions. Furthermore, any available land in proximity to Graveyard Pond serve as stormwater detention ponds and therefore have significantly altered the natural hydrology and biodiversity at these locations. As discussed previously, the use of controlled summer abstractions could contribute to ensuring that these stormwater ponds function more like the naturally occurring seasonal wetlands by ensuring defined dry periods during summer and saturated conditions during winter.

5.3 MAR: water supply

5.3.1 Recharge-storage relationship

The CFA in Philippi was shown to have the greatest MAR potential. Thus, it is important to determine the upper limits of the recharge potential of the CFA based on the recharge-storage relationship at the Philippi site, and to compare the two methods of recharge, *viz.* injection and infiltration. This process involved incrementally increasing the amount of winter stormwater recharge, applied as infiltration or injection, and simulating the change in the aquifer storage (Scenario 14). By simulating incremental increases in recharge it is possible to isolate the point at which the proportional relationship between recharge and storage ceases, where increasing volumes of artificially recharged water result in diminishing increases in aquifer storage. The declining storage volumes are a result of the rising water levels intersecting with local topographic drainage or the increase in hydraulic gradient increasing discharge at local water bodies such as Zeekoeivlei.

The results of Scenario 14 indicated that the direct injection method of recharge was more efficient than the infiltration method, showing greater storage for each volume of water recharged. This was attributed to the greater volume of aquifer storage utilised in the injection method when compared to the infiltration method. The available storage volume in the aquifer was reduced using the infiltration method, due to the sharper ‘mounding’ of the water table as a result of the slower infiltration process, meaning the water takes longer to reach the aquifer (Figure 5.1). Whereas in the case of the injected water, the aquifer is recharged almost instantaneously and the increase in pressure more rapidly facilitates the dissemination of water throughout the aquifer resulting in a higher storage potential (Figure 5.1). The greater level of ‘mounding’ meant that the drainage losses for infiltrated recharge occurred at lower recharge volumes than in the injected recharge as the higher groundwater levels began to intersect with local topography above 4 Mm³. While, the injection method of recharge was found to only start incurring drainage losses after 6 Mm³, which suggested that injection would be a more efficient method for stormwater recharge for MAR on the CFA. Figure 5.1 demonstrates this by comparing the two MAR methods of recharge, injection and infiltration, using the maximum winter recharge rate for the Philippi site of 10 Mm³ per year. The maximum volume more clearly illustrated the effects of the steeper hydraulic gradient in the infiltration scenario that results in the ‘mounding’ effect which leads to increased drainage losses. At this recharge volume, the injection method also incurs drainage losses however this is mitigated as more of the aquifer is used for the storage of recharge water from the smoother hydraulic gradient and less pronounced mounding effects.

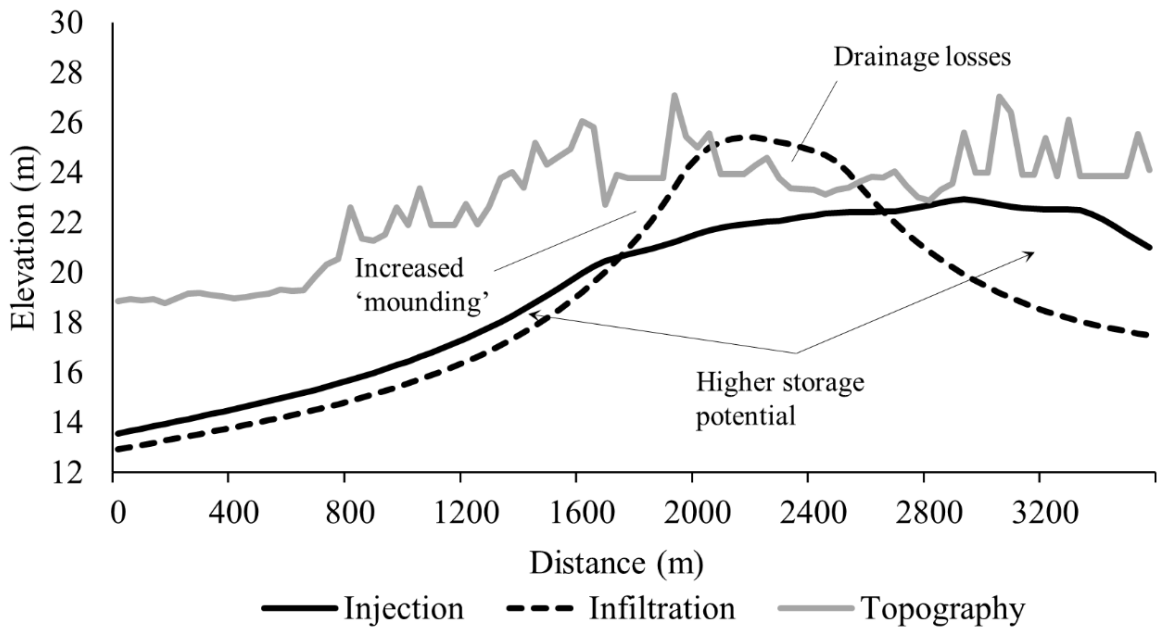


Figure 5.1: An elevation profile through the Philippi MAR site comparing injection and infiltration of 10 Mm³ per year in relation to the topographic elevation

The groundwater levels at the Philippi MAR site and two adjacent urban areas, Mitchells Plain and Strandfontein, were evaluated to assess the risk of local groundwater flooding as a result of the injection of winter stormwater. The injection scenario was used as it was the most efficient scenario. The simulated groundwater levels for the 15 years from 2000 – 2015 showed that none of the injection scenarios exceed the same flood threshold of -1.5 m used in the flood mitigation scenarios. All the scenarios reach an equilibrium within the first nine years, yet none of these are high enough to cause groundwater flooding. The closest was the 10 Mm³ scenario which exceeded -2 m but remained below the flood threshold. Therefore, it is apparent that the optimum storage scenario, consisting of the direct injection of 6 Mm³ winter stormwater into the CFA, has no anticipated risks in terms of groundwater flooding.

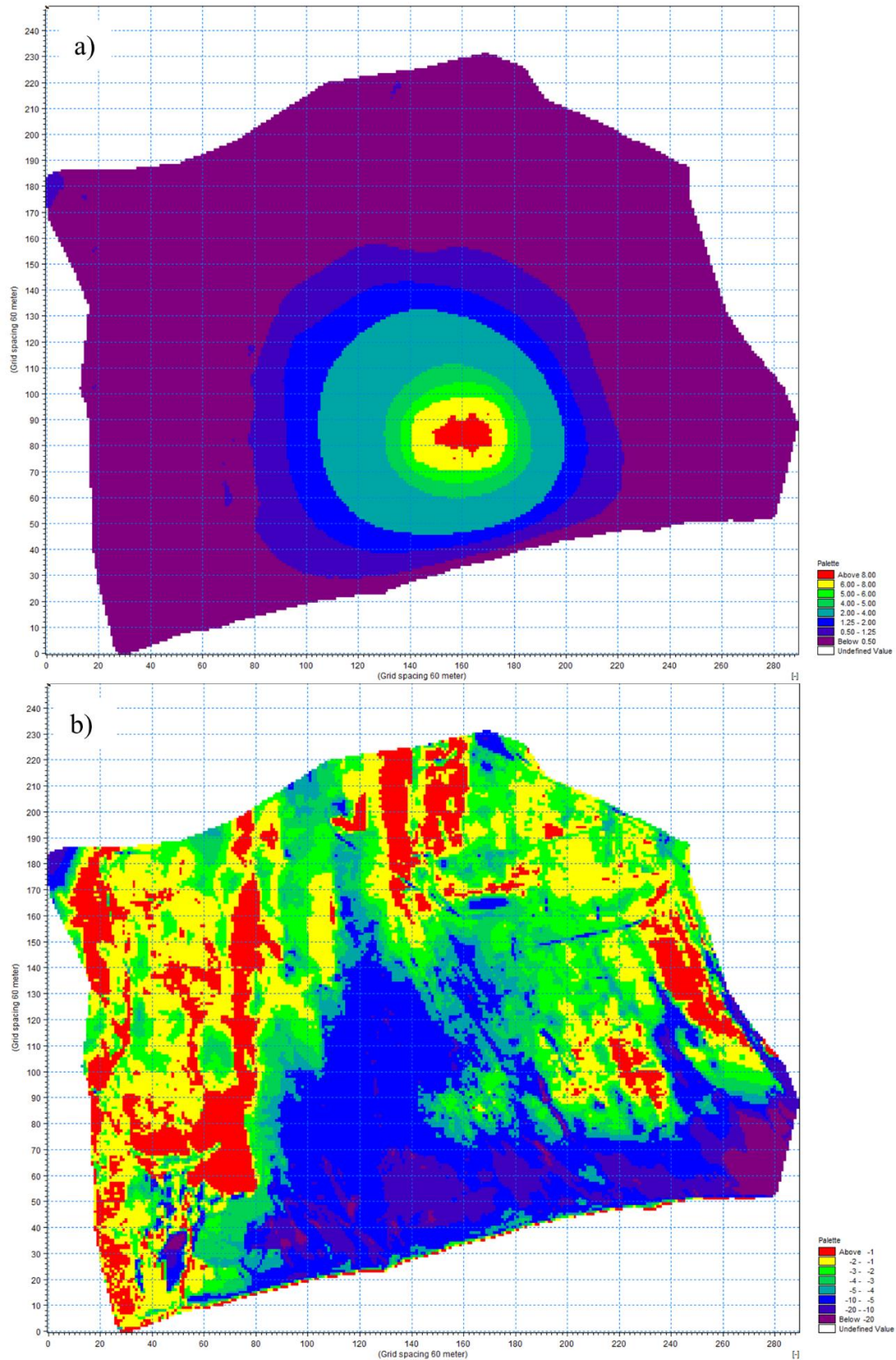


Figure 5.2: MIKE SHE results mapping the a) the extent of 'mounding' effect and the b) groundwater levels (mbgl) of the 6 Mm³ injected winter recharge scenario

5.3.2 Long-term storage

After determining the static storage potential without any abstractions, the dynamics of long-term storage were investigated. The long-term storage scenarios considered the storage of stormwater from wet winter seasons for a number of consecutive years for use in times of drought. In this way, water is 'banked' in the aquifer much like a dam for the storage of surface water. Thus, two things are important to understand the long-term storage potential of the CFA: the length of time until the recharge of the aquifer yields no increase in storage to reach an equilibrium, and how long that recharged water is available.

In order to test this, a conservative volume of water was recharged at a rate of 5 Mm³ per year or 15.72 ℓ.s⁻¹ during the winter months and abstracted during the summer months at the same rate. The length of time that the water is available is evaluated based on the period between the end of the recharge phase until the recharge scenario storage intersects the baseline storage (Figure 4.17). By comparing the long-term storage scenarios to the baseline scenario, it is possible to evaluate the increase in storage for longer recharge periods. Two recharge lengths for six and nine years were simulated and abstractions were initiated thereafter. It was shown that by increasing the recharge period by three years, the benefit for storage yield was only ten months or one additional season of abstraction. In other words, the additional recharge only returned an increase in the available storage volume of approximately 33 %. Therefore, while there is value in the long-term storage in the CFA, for periods longer than 6 years there is a decreasing yield in available storage. This suggests that the storage volume is too small for an extended storage term and would be optimally utilised for seasonal storage or for the storage of a few wet seasons.

5.3.3 Short-term or seasonal MAR

Following the assessment of the long-term storage potential of the CFA, an assessment of the application of MAR in regard to short-term or seasonal storage on the CFA was evaluated. Short-term or seasonal MAR, in this application, is concerned with the collection and storage of stormwater, which occurs predominantly during the winter months, within the CFA. This stored stormwater is then available to be used during the subsequent dry summer months. As with the long-term storage scenario, recharge was simulated for half the year during the winter months, and abstractions simulated for half the year during the summer. Due to the shorter cycles between recharge and abstraction, it is not necessary to have the greatest volume of storage as was required for long-term storage at the Philippi site. Therefore, the assessment of short-term storage water was carried out at Philippi (Scenarios 15, 16, and 17) and Mitchells Plain (Scenarios 18 and 19). The objectives of this MAR modelling were to maximise the available storage at Philippi and Mitchells Plain to obtain the highest yield from the CFA at the lowest cost and risk to the environment or the aquifer itself.

5.3.3.1 Philippi MAR

At the Philippi site, three scenarios were tested to assess the yield and the impacts of drawdown due to abstractions such as seawater intrusion. First, Scenario 15, was simulated as a relatively conservative application of MAR based on the recharge-storage relationship outlined in Scenario 14, involving the recharge of 5 Mm³ per year of winter stormwater. Thereafter, the same volume was abstracted over the summer months. At this rate of injection and abstraction, there were no

interruptions due to boreholes drying out, and this scenario was able to support the anticipated yield of approximately 77 Mm³. Furthermore, this scenario was shown to have a negligible impact on the mean annual water table, indicating seawater intrusion is unlikely.

Due to the conservative nature of Scenario 15, it was necessary to identify the point at which drawdown results in the drying of boreholes, increasing the risk of seawater intrusion. Therefore, Scenario 16 was simulated as an abstraction only scenario, with an abstraction rate of 10 Mm³ per year during the summer months. This abstraction only scenario resulted in the drying out of the boreholes due to excessive drawdown and as a result, the anticipated yield was reduced by 13 %. In an attempt to offset these losses, the 10 Mm³ per year summer abstraction rate was maintained, but 5 Mm³ per year of stormwater was injected into the CFA during the winter so as to mitigate the excessive summer drawdown (Scenario 17). As a result, it was possible to preserve groundwater levels to maintain a consistent abstraction rate and maximise the wellfield yield. Furthermore, Scenario 17 was shown to reduce the risk of seawater intrusion by reducing the drawdown by approximately 50 %, maintaining a stronger north-south hydraulic gradient towards the False Bay coastline.

5.3.3.2 Mitchells Plain MAR

MAR strategies could also be transferred to the adjacent Mitchells Plain area, which was also highlighted as a potential MAR site, despite the potentially shallower aquifer. Scenario 18, equivalent to Scenario 16, considered only summer abstraction at the maximum rate of 10 Mm³ per year. However, at this rate, the wellfield was not able to maintain the abstraction rate due to the boreholes drying out. As a result, the efficiency of this wellfield was approximately 48 % of the anticipated yield. Applying approximately 5 Mm³ per year of stormwater recharge to offset these losses in Scenario 19 did improve the efficiency of the wellfield to 79 %, but only with a 39 % improvement from the abstraction only scenario (Scenario 18). Thus, this indicates that the infiltration of this quantity of water was inefficient as only 61 % of the infiltrated water is available during the summer months for abstraction. This was likely due to the steep bedrock topography creating a high hydraulic gradient in the area, rapidly transporting injected water towards the False Bay coastline. This means that reductions in abstraction rates may improve the reliability of the wellfield yield at Mitchells Plain thereby preventing the boreholes from drying out. In order to address these issues, an option would be to move the injection borehole further north or up-gradient of the abstraction site so as to prevent losses of injected water. Additionally, the steep nature of the bedrock and strong hydraulic gradient of the CFA at Mitchells Plain makes seawater intrusion into the aquifer unlikely. Moreover, the bedrock underlying the aquifer is higher than sea-level, thus for seawater intrusion to occur a strong reversal of the hydraulic gradient would be required. This is unlikely, where even under maximum abstraction the hydraulic gradient was only weakened, but not reversed.

5.4 Water Quality

This section discusses the results of the water quality or contaminant transport simulations for the flood mitigation and MAR storage scenarios on the CFA. The migration of TDS concentrations was used as an indicator of how contaminants from sources such as stormwater can migrate through the aquifer. There is little information on the quality of stormwater from informal

settlements on the Cape Flats. So this study used information from a previous study by Wright *et al.* (1993) to estimate the quality of stormwater based on measured TDS concentrations from informal settlements. This study found that the typical TDS concentration of stormflow from the Khayelitsha informal settlement was $1472 \text{ mg}\cdot\ell^{-1}$ or $1.48\times 10^9 \mu\text{g}\cdot\text{m}^{-3}$. In order to construct a daily TDS concentration dataset, the average monthly TDS concentration was estimated by assigning a stormflow concentration of $1.48\times 10^9 \mu\text{g}\cdot\text{m}^{-3}$ for every day with rainfall in a given month. The average monthly TDS concentration was calculated for each month based on rainfall from 2000 – 2015 and divided by the number of days in the month to get the daily TDS concentration. This dataset was then used for both the flood mitigation scenarios and MAR storage scenarios.

It is clear that under the baseline conditions the migration of contaminants is likely to result in pollution downstream. In this case, Zeekoeivlei and ultimately the False Bay coastline are projected to experience contamination between 2050 and 2100 based on the mapping of contaminant plumes (Chapter 4.3). Strategic abstractions at Sweet Home and Graveyard Pond were shown to prevent the excessive migration of TDS concentrations. These abstractions will help to prevent the degradation of the water quality at Zeekoeivlei and at the False Bay coastline, which in turn is likely to have positive impacts on biodiversity and ecosystem services in these areas. An additional benefit would include the protection of potential MAR schemes closer to the coast near Philippi and Mitchells Plain. However, the removal of contaminants via abstraction does pose risks for the water users and would need to have some form of treatment before it is used as a fit-for-purpose water supply.

Following the flood mitigation scenarios, the contaminant transport of the MAR at Philippi and Mitchells Plain were simulated. The similarity of TDS contaminant plumes in terms of concentrations and extent under baseline scenarios for each time step from 2025 – 2100, demonstrated that the strong hydraulic gradient in the southern parts of the CFA dilutes the TDS concentration by rapidly moving water towards False Bay and preventing the accumulation of TDS. With the application of MAR, the concentration of TDS is much higher in the vicinity of both MAR sites, with the central parts of the wellfields exhibiting higher TDS concentrations when compared to the outer sections of the wellfields. The extent of the contaminant plumes was reduced when compared to the baseline scenario. The Philippi site shows almost no migration of TDS meaning that any potential contamination is contained within the vicinity of the Philippi site. At Mitchells Plain, the contaminant plume present in the baseline scenario remains present and was only slightly reduced in its extent, still extending to the False Bay coastline. However, the concentrations of Mitchells Plain plume exceeded those of the baseline scenario by 2 orders of magnitude in some areas. The areas in green denote TDS concentrations exceeding $10\,000 \text{ mg}\cdot\ell$ which is classified as saline water which is limited to industrial and other limited uses. The areas in red are in excess of $35\,000 \text{ mg}\cdot\ell$ denoting TDS concentrations that were classified as brine. Brine can only be used for industrial and mining processes. Therefore, at the Mitchells Plains site, the aquifer is shallower and the bedrock is above sea level meaning that the hydraulic gradient at Mitchells Plain is still strong enough under MAR conditions to carry the TDS plume south towards False Bay. The migration of such high contaminant load could jeopardise the economic value of recharged water for fit-for-purpose applications in Cape Town. Furthermore, such high TDS concentrations could pose a risk for the False Bay coastline where excessively high concentrations are likely to impact coastal ecosystems. If TDS was assumed as an indicator for other potential contaminants such as pathogens and nutrients, the result would see an intensification of algal

blooms and general poor coastal water quality, which are ongoing problems on the False Bay coastline.

5.5 Climate change

Dedicated simulations with MIKE SHE allowed for the evaluation of the impacts of climate change based on projected temperature and rainfall data from four GCMs, *viz.* IPSL-CM5A-MR, MRI-CGCM3, MIROC5 and GFDL-ESM2M. These models represent the following future climate conditions: “much hotter and much drier”, “much hotter and slightly wetter”, “slightly hotter and slightly drier” and the “median” conditions respectively. Firstly, the impacts of climate change were evaluated using the simulated water balance to determine the sensitivity of the hydrological cycle to changes in future temperature and rainfall. Most of the GCMs showed a projected decrease in rainfall. In particular, two GCMs (MIROC5 and GFDL-ESM2M) were noted for showing similar reductions in precipitation. However, GFDL-ESM2M is associated with a much hotter climate than MIROC5, yet the effects on the water balance are similar for both GCMs. GCMs with high future temperatures (IPSL-CM5A-MR and MRI-CGCM3) were compared, however MRI-CGCM3 predicts slightly higher rainfall than IPSL-CM5A-MR, which is a much drier model. MRI-CGCM3 showed minimal changes to the water balance of the hydrological cycle, whereas the drier model, IPSL-CM5A-MR, showed substantial reductions in recharge and drainage. Demonstrating that the recharge and storage of the CFA, are less sensitive to changes in temperature than rainfall. These results indicate that understanding the sensitivity of elements of the hydrological cycle is pertinent for determining the impacts of environmental and climate changes on groundwater and the feasibility of MAR.

The greater impact of rainfall on the hydrological cycle, and groundwater, was supported in the assessment of the groundwater storage of the CFA. MRI-CGCM3 showed a marginally increased groundwater storage when compared to baseline conditions due to slightly higher projected rainfall in this scenario and despite the much higher projected temperature. Higher future temperatures have less impact on the groundwater response than rainfall. The other three GCMs all showed declining groundwater storage trajectories. These reductions are mainly attributed to a reduction in rainfall, however evaporative losses are also partly responsible for reductions in storage. This is evident in the hottest and driest model (IPSL-CM5A-MR), where the combination of significant reductions in rainfall and increases in evapotranspiration, resulted in the most significant reduction in groundwater storage.

In summary, most of the climate change models suggest a hotter drier future climate for Cape Town. Due to the sensitivity of the CFA to rainfall, it is likely that under a future climate natural aquifer recharge will decrease, resulting in decreasing aquifer storage. A declining water table in the CFA could have a significant impact on local groundwater dependent ecosystems like rivers and permanent or seasonal wetlands. This is particularly true for seasonal wetlands that rely on a high groundwater table for seasonal inundation.

There are also potential socio-economic implications for a declining water table in the CFA. Many of the farmers in the PHA depend on the groundwater of the CFA to irrigate their crops. If the water table drops below the level of their current groundwater infrastructure, they may need to invest in deeper boreholes and more powerful pumps to abstract water. This would affect the poorest farmers the most, as they would have limited access to the capital or credit required to

adapt to this impact of climate change. It has also been noted by a number of farmers in the PHA that the winter rainfall provides a fresh layer of water on the top of the aquifer and that as the fresh layer depletes through summer, the water quality of the CFA deteriorates, becoming increasingly saline. Therefore, under drier climatic conditions the amount of winter recharge is likely to be reduced, resulting in a more rapid decline in water quality during the summer months. High salinity will affect all the farmers in the area, both commercial and subsistence, having negative impacts on food security and the economic stability of many of the people who depend on the PHA for their livelihood.

Therefore, the application of MAR on the CFA could be used to offset some of these negative impacts of climate change through the artificial recharge of the aquifer using stormwater. However, it may be necessary to consider other sources of water for recharge than stormwater, which may be less abundant if there is less rainfall. Treated wastewater might be a viable option on the CFA for increasing water levels for ecosystem services, while also being a reuse option for irrigation and industrial purposes. Further treatment using reverse osmosis could be used to treat the wastewater to a potable water standard and then used to recharge the CFA. This form of reuse makes the CFA a plausible potable water source improving the adaptability of Cape Town to the threat of climate change, by protecting the high value treated wastewater from evaporation through subsurface storage. The same could be achieved with water from reservoirs that form part of the current water supply system. The raw water, which is generally of a reasonable quality, would be protected from evaporative losses. Evaporative losses are likely to be higher in hotter conditions reducing the yield reservoir, while also exacerbating issues such as eutrophication (Helfer *et al.*, 2012; Paerl and Paul, 2012). Therefore, MAR is a potential alternative to surface water storage that can prevent losses while sustaining current ecosystem and socio-economic needs.

5.6 The feasibility of MAR

5.6.1 MAR for flood mitigation

Graveyard Pond is shown to be the most feasible option of the two flood mitigation sites. Reductions in local groundwater levels to below the flood mitigation threshold were more easily achieved at Graveyard Pond than at Sweet Home because fewer boreholes were required to achieve the same objective. The installation of 12 boreholes instead of 27, is likely to be more than half the cost required to achieve the same objectives at Sweet Home. Furthermore, the lower abstracted volumes obtained from the Graveyard Pond site would be easier to attach to local fit-for-purpose water demands at a lower per-unit cost of water supplied.

The migration of TDS concentrations was used as an indicator of how contaminants from sources such as stormwater can migrate through the aquifer. It is clear that under the baseline conditions, the migration of contaminants is likely to result in pollution downstream. In this case Zeekoeivlei and ultimately the False Bay coastline are projected to experience contamination between 2050 and 2100 based on the mapping of contaminant plumes (Chapter 4.3). Strategic abstractions at Sweet Home and Graveyard Pond are shown to prevent the excessive migration of TDS concentrations. Abstractions for flood mitigation were shown to prevent the migration of contaminant plumes that could be detrimental to local ecosystems and biodiversity, including Zeekoeivlei and the False Bay coastline. The mitigation of the contaminant plumes emanating

from Sweet Home and Graveyard Pond also hold the potential to protect the MAR schemes at Philippi and Mitchells Plain from contamination. Nevertheless, the removal of potentially contaminated water does pose a health risk and the appropriate treatment measures should be applied before it is used as a fit-for-purpose water supply.

5.6.2 The CFA as a water supply option

The results of the MAR scenarios (Scenarios 14 – 19) at Philippi and Mitchells Plain provide strong evidence to suggest that there is sufficient storage capacity within the CFA for the application of reuse strategies using stormwater. The use of MAR was shown to improve the wellfield yield at Philippi allowing the maximum abstraction rate to be sustainably achieved, resulting in a sustainable yield of approximately 10 Mm³ per year without risking seawater intrusion. The improvements of MAR to the yield of Mitchells Plain were less efficient than those observed at Philippi, only yielding a maximum of 7.8 Mm³ per year. Therefore, the total sustainable yield of these schemes was approximately 18 Mm³ per year.

Given the stressed water supply of the City of Cape Town, this potential water supply from a relatively small part of the CFA is important for contributing towards the city's water security. The estimated 18 Mm³ per year of water supply is relatively small in comparison to the current water demands which are just over 300 Mm³ per year. However, projections of the city's water demand into the future are estimated to grow to just under 900 Mm³ per year, by the year 2035, assuming a high growth rate of 3.38 %, or 750 Mm³ assuming a lower increase in water demand of 2.3 % (DWA, 2013b). The Western Cape Water Supply System (WCWSS) Reconciliation Strategy has acknowledged that the current supply system—relying largely on surface water—is under pressure with few additional surface water options available. As a result, the WCWSS Reconciliation Strategy has opted for an integrated or blended approach to available water supply options to meet the future water demands (Table 5.1). Therefore, although small, the 18 Mm³ potential contributions from Philippi and Mitchells Plain within this range of supply options could prove to be an invaluable sustainable resource offering further adaptability to the WCWSS Reconciliation Strategy.

The use of MAR on the CFA offers additional adaptability for water supply as groundwater storage is less susceptible to fluctuations in weather or long-term climate change. Furthermore, by utilising MAR as a reuse strategy, with water that would otherwise be considered a useless by-product of the urban water supply system (i.e. stormwater or treated wastewater), means that water managers have greater control of the assurance of supply of these systems. Thereby, the overall uncertainty of the water supply system is reduced. Maliva (2014) argues that MAR serves a valuable stabilising role for urban water supply, suggesting that surface water resources should be used first when it is available and that fresh groundwater resources should be strategically reserved for dealing with water scarcity. Maliva (2014) asserts that MAR could be used as a dynamic groundwater resource, able to fill the gap in water supply when surface water is limited and before the use of high-quality groundwater, or where MAR can be used to optimise the available groundwater supplied. A good example of this is the Table Mountain Group (TMG) aquifer is a substantial, high-quality water resource ($\sim 70 \text{ Mm}^3 \cdot \text{a}^{-1}$) for the City of Cape Town. Utilising the CFA could help to postpone the need for water from the TMG, by using MAR on the CFA to meet non-potable water requirements. If the high quality, hence higher value water from the TMG

is reserved for emergency periods such as drought conditions, the higher the likelihood that that water will be assigned to higher value end uses such as drinking water and other indoor uses. The reason for this is that during drought conditions the implementation of water restrictions will encourage only essential water use while discouraging non-essential use such as residential garden irrigation. Meaning that the TMG would be reserved for essential potable water demand, rather than wasting high-quality water on end uses that do not require a high level of water quality.

MAR could reasonably be incorporated into the WCWSS either as a part of the ‘Generic Reuse’ category or as a supplementary water supply source (Table 5.1). The WCWSS Reconciliation Strategy addresses reuse mainly as a fit-for-purpose strategy using treated wastewater for irrigation purposes. However, the use of the CFA for the storage and/or treatment of stormwater or treated wastewater could contribute significantly to the Generic Reuse scheme scheduled for the year 2025 (Table 5.1). This would potentially be a less expensive option when compared to reuse strategies that are more technologically intensive. A good example of technologically intensive water reuse strategies is NEWater in Singapore, in which wastewater is treated for indirect potable water reuse (Wintgens *et al.*, 2008). Although NEWater is less expensive than desalination, MAR is likely to be an even cheaper option when compared to technologically intensive solutions such as NEWater, due to the lower infrastructure and energy costs. It is also important to note that if the City of Cape Town were to consider large-scale reuse programs—like NEWater—for indirect potable reuse, it is likely that the CFA would be considered for the storage of treated wastewater. This is due to that fact that most of the reservoirs are located well outside the city’s boundaries, making the transport of treated wastewater expensive. Furthermore, most of the wastewater treatment plants are located on the coast. Therefore, the aquifers of the Cape Flats sands (CFA and Atlantis) offer the best option for storage when considering large-scale indirect reuse strategies.

Table 5.1: Projected water supply option for the City of Cape Town (after DWA, 2013)

Proposed WCWSS Intervention	Year	Yield (Mm ³ /a)	Potential MAR intervention using the CFA
Berg River – Voëlvlei	2022	23	None – increasing the volume of the Voëlvlei dam will have short term gains that cannot be met by the CFA in the short-term.
TMG Scheme – 1	2024	20	Potential to postpone the utilisation of this high-value resource.
Generic Reuse – 1	2025	40	Potential opportunity for MAR with higher value end water uses, such as an indirect potable water reuse scheme.
TMG Scheme – 2	2028	50	MAR on the CFA would help reserve the TMG for emergency uses.
Generic Reuse – 2	2030	40	Potential opportunity for MAR with lower value end water uses, such as an urban irrigation and industrial use.
Desalination	2032	80	MAR on the CFA would help postpone the adoption of desalination, having a number of potential economic benefits.

The intended use of the water is another factor that will influence the feasibility of MAR on the CFA. The surrounding urban areas of Mitchells Plain, Philippi and Strandfontein provide areas suitable for the collection of stormwater for recharge at Philippi or Mitchells Plain. The quality of

stormwater from these areas may be poor, due to some areas having insufficient stormwater infrastructure and pollution problems. Therefore, it is important to make sure that the collection of stormwater is conducted from precincts with the appropriate stormwater infrastructure. The incorporation of WSUD or SuDS technology would potentially offer a pre-treatment stage before the stormwater is injected. Common WSUD or SuDS technology, including artificial wetlands, sand filters and microfiltration and Granular Activated carbon filtration (GAC) are used for the pre-treatment of rainwater or stormwater prior to aquifer storage (Dillon *et al.*, 2009). However, it is the end use of the recharged water that is an important factor in determining if further treatment is required in the recovery phase. For example, Dillon *et al.* (2009) suggest that fit-for-purposed water, for uses like industry, irrigation, toilet flushing and sustain ecosystem services, require little to no post-treatment. Whereas, if the end use is for drinking water supply, disinfection would be required.

The quality of stormwater is an important factor to consider when selecting the recharge method for MAR. Recharge via infiltration offers the potential additional stage of the treatment for stormwater by removing an array of contaminants. Whereas, poor quality stormwater may preclude the direct injection of water into the CFA as this method offers no further treatment risking the aquifer health. Furthermore, the saturated conditions of an aquifer can facilitate the rapid transport of contaminants throughout the aquifer, which could have negative impacts on the local surface water bodies such as Zeekoeivlei.

It is also important to account for the land use and economic impacts of these different recharge techniques. At Philippi, much of the proposed site is currently being used for sand mining. However, there are plans to develop the land for housing. Therefore, direct injection might be an appealing option as injection wells can be strategically sited to accommodate the layout of a housing development. Additionally, it is also likely that a housing development will have a significant socio-economic benefit, creating jobs and revenue through the construction and sale of the housing units. The site could be used for infiltration recharge, which would lower unit cost of water produced due to the lower costs associated with a more passive, less technologically intensive form of MAR. At Mitchells Plain, much of the most suitable land overlying the aquifer is dominated by residential areas which prevent the installation of large-scale infiltration devices. One possible option is to install a number of smaller, on-site, infiltration devices using existing stormwater ponds or the development of precinct-scale infiltration measures including, *inter alia*, infiltration basins, pervious pavements and rain gardens. However, this decentralised system would be difficult to maintain and manage to ensure the system is performing at an optimum level. A decentralised system also means there is more uncertainty in relation to the volume of recharge that could be achieved, meaning that it is likely that there will be greater uncertainty with regard to the anticipated yield of the aquifer. Moreover, a decentralised MAR system would mean difficulty in monitoring and managing the quality of stormwater that is being recharged, and this would be problematic if the end use of the water is intended for potable use. It is also important to note that the objectives of MAR aim to ensure the health of the aquifer is either maintained or improved, further motivating the careful consideration of the quality of recharged stormwater. Therefore, the use of direct injection is recommended as it can be incorporated into the current urban layout while allowing greater control over the quality and quantity of stormwater for recharge. Additionally, based on the fact that 20 boreholes are required to achieve the design yield of 7.8 Mm³ per year at the Mitchells Plain wellfield, the existing infrastructure would be available for

direct injection. An additional economic consideration for MAR in the CFA, is the inherent value of a unit of storage space within the aquifer. Obviously, the parts of the aquifer with the highest storage capacity will have the highest value, due to the ability to store more water in a smaller space. Maliva (2014) addresses this concept, suggesting that the value of a unit of aquifer storage will increase as more of the aquifer's storage capacity is utilised. This thesis has identified Philippi and Mitchells Plain as areas well suited for MAR given their high storage and hydraulic conductivity, therefore these areas have the highest relative value. Consequently, planning and management of the CFA needs to account for this value to ensuring that these parts of the aquifer are utilised appropriately. In doing so it would be possible to guarantee that the resource value of the CFA is preserved and that the appropriate schemes are implemented, supplying water for the highest value end uses.

5.7 The integration of MAR and WSUD

There are a wide range of anticipated benefits from MAR as described by Dillon *et al.* (2009) who outlines specific MAR objectives. The flood mitigation scenarios were able to achieve a number of these objectives, such as flood prevention and mitigation, enhancing the ecological and biodiversity as well as the management of groundwater quality. Many of these objectives are also consistent with the WSUD objectives outlined by Whelans *et al.* (1994). Most specifically, the flood mitigation scenarios help to maintain and regulate the urban water balance by “*maintaining appropriate aquifer levels, recharge and streamflow characteristics in accordance with assigned beneficial uses*” and “*preventing flood damage in developed areas*”. A further contribution is the role of this MAR scenario in maintaining or enhancing the water quality of the CFA. In particular, the flood mitigation scenarios demonstrated that MAR is able to “*protect existing riparian or fringing vegetation*” such as the Zeekoeivlei and the False Bay Coastline. In addition, MAR was also shown to be able to “*minimize the export of pollutants to surface or groundwater*”. Furthermore, the fit-for-purpose water use of abstracted water also contributed towards the water conservation aspects of WSUD, by helping to “*minimize the import and use of potable water supply*” and “*promote regulated self-supply*”.

Based on the results of this thesis and the preceding discussion, it is apparent that groundwater considerations are essential for the application of WSUD practices and cannot be carried out in isolation. Managing the characteristics of an aquifer can also be utilised to serve both the WSUD and MAR objectives. The results of this scenario analysis also demonstrated that urban groundwater systems are complex, yet integrated, physically-based hydrological models such as MIKE SHE; allow a number of management scenarios, potential risks or anticipated climate or land use changes to be evaluated with minimal cost. This section of the discussion chapter deals in more general terms with how the concept of MAR contributes towards the application and progress of WSUD in South Africa and internationally.

5.7.1 Water Supply

A common motive for the implementation of MAR is the ability to sustain or enhance water supply from an aquifer for urban and agricultural purposes (Dillon *et al.*, 2009). The results of the MAR modelling on the CFA showed that by supplementing natural aquifer recharge using urban stormwater or treated wastewater, it is possible to enhance the assurance of supply obtainable from an aquifer. The scenario analysis of MAR on the CFA showed that it is important to identify the

upper and lower limits of recharge and storage. Understanding the limits of the physical properties of the resources are essential for its sustainable and optimised utilisation. The aquifer should not be ‘filled’ to the point that could result in local flooding or damage to subsurface infrastructure. On the other hand, uncontrolled abstraction can cause excessively low groundwater tables that could change the local hydraulic gradients, altering the natural flow regime of an aquifer or resulting in seawater intrusion. These are important issues for consideration in a number of WSUD devices that incorporate groundwater for storage and treatment, such as infiltration basins, stormwater ponds, rain gardens and permeable pavements. The WSUD literature often overlooks the potential hydrogeological impacts and feedbacks of WSUD interventions. This is particularly evident in the promotion of infiltration as a means of stormwater management. However, the implications of recharge can have significant impacts on local infrastructure and surface water bodies. The use of an integrated hydrological model provides a means of identifying these impacts and their severity so that appropriate design and planning measures can be taken.

The MAR water supply scenarios conducted at Philippi and Mitchells Plain demonstrated that MAR can substantially contribute towards the urban water supply objectives for the City of Cape Town. Based on this research, MAR is an important storage resource that will be vital for the implementation of any water reuse strategies for fit-for-purpose or potable water supply to the City of Cape Town. Furthermore, the application of MAR on the CFA will also encourage the use of a previously underutilised resource as the CFA has remained largely unused over the last 40 years. Therefore, MAR may well provide a more concrete link between groundwater and WSUD and encourage its inclusion as a water supply option where it is available. Even in areas where the water quality in the aquifer is relatively poor, having a high salinity, it is possible to use MAR to improve the water quality of the aquifer and hence its resource potential (Dillon *et al.*, 2009; Steinel, 2012; Miotliński *et al.*, 2014). This would make groundwater resources that were previously overlooked for WSUD, more attractive. This is the case on the CFA where some parts of the aquifer suffer from high salinity thought to be linked to local industrial and agricultural processes (Aza-gnandji *et al.*, 2013). By recharging the aquifer with water of an improved water quality, a further improvement to the general aquifer health is likely. Therefore, with an approach to urban water management that is guided by WSUD principles that advocate alternative, sustainable and reusable water resources, groundwater in aquifers like the CFA become increasingly more difficult to ignore—even more so with the implementation of MAR that aims to increase the efficiency and value of the resource. As a result, MAR can provide greater incentives for the inclusion of groundwater in the WSUD. While WSUD can position MAR and its objectives within an urban water strategy with the vision of creating water sensitive cities.

5.7.2 Flood prevention and mitigation

In terms of flood prevention and stormwater management, there are two considerations that this thesis makes. The first deals with groundwater related flooding resulting from a combination of elevated groundwater levels and low-lying land. In these areas, the application of the flood mitigation simulations (Chapter 4.2) were conducted to evaluate groundwater abstraction as a direct flood mitigation intervention. The second element is a more indirect method of stormwater management by the quantifying and assessing of the storage potential of the CFA for stormwater recharge. In this indirect approach, the areas of the Cape Flats with the highest MAR potential do not have the same risk of urban flooding, as the water table is lower and the relatively elevated

topography improves drainage, preventing excessive ponding. Flooding in these areas does occur, but not as regularly as the low-lying land that is known for wetland conditions. Currently, stormwater in these areas is managed using an array of small stormwater detention ponds. Within these temporary storage devices, stormwater is readily available for treatment and recharge into the CFA. Therefore, the evaluation and confirmation of the storage and hydraulic properties of the CFA provide a realistic seasonal storage option for recharged stormwater which, in turn, will improve the capacity of stormwater ponds.

While the flood mitigation strategy of controlled summer abstractions from flood mitigation is not MAR in the strict sense of its definition, outlined by Dillon (2005 p. 313) as the “...*intentional banking and treatment of water in aquifers*” as it does not have an ‘artificial recharge’ component. The main purpose of these scenarios is to counteract current or natural recharge in order to prevent groundwater flooding. However, this strategy is aimed at controlling the storage and groundwater levels in the aquifer to achieve specific MAR and WSUD objectives, *viz.* preventing flooding and enhancing urban water supply. This thesis argues that the main thrust of MAR is focused on the resource potential of the aquifer. Therefore, groundwater abstraction or artificial recharge are secondary components of MAR that serve the primary purpose which is to manage and control the resource potential of the aquifer. The storage potential at Sweet Home and Graveyard Pond is low, and to create additional storage volume through summer abstractions is unfeasible due to the steep hydraulic gradients and relatively high hydraulic conductivity. Therefore, the flood mitigation scenarios are not conventional MAR, but rather a form of controlled aquifer management. Nevertheless, this strategy does fulfil a number of MAR and WSUD objectives.

5.7.3 Ecosystem services, biodiversity and amenity value

WSUD and MAR both advocate for environmental sustainability in their application, attempting to incorporate ecosystem goods and services while improving the social amenity value (Whelans *et al.*, 1994; Dillon *et al.*, 2009; Water by Design, 2009). This section examines some of the MAR scenarios, showing the potential environmental and ecological contributions of MAR towards the environmental and ecological goals of WSUD on the CFA.

The flood mitigation scenarios were also shown to mitigate the transport of contaminants throughout the CFA. As a result, this intervention protects the aquifer and the groundwater dependent ecosystems downstream of the informal settlements against further contamination. Mitigating the transport of contaminants also helps to sustain natural biodiversity by reducing the pressure on species that are sensitive to pollution. This is particularly true for many of the wetland and rivers that interact with the CFA such as: the Lotus river; the Zeekoeivlei and the Princess Vlei wetlands. The status of these ecosystems is considered to be poor to unacceptable, with severe pollution from high levels of phosphorus and nitrates. The poor water quality results in declining populations of natural, and in some cases endemic, flora and fauna while causing the proliferation of invasive species (CSIR, 2005). Nel *et al.* (2013) suggest these ecosystems are a high priority that should receive the most urgent attention and implementation of remedial interventions.

Furthermore, abstracting contaminated water could be used to isolate and treat contaminated WSUD sites. WSUD sites are not invulnerable to contaminant risk as the quality of stormwater or treated wastewater are difficult to control and accidental contamination is inevitable. MAR and its ability to control hydraulic gradient and groundwater flow direction could prove

instrumental in reducing the rate of contamination in an aquifer, acting as a barrier to further contamination.

The conventional MAR scenarios demonstrated that recharged water could also be used to protect or improve the water quality of the groundwater resource. This could benefit a number of local groundwater users, especially the farmers of the PHA who could benefit from longer periods of fresh water before it becomes more saline towards the end of summer due to abstractions. The use of stormwater will require some level of pre-treatment and this is where WSUD and MAR are able to complement one another most significantly. MAR is able to function as a storage and treatment component in the WSUD treatment train. WSUD has an established set of Best Management Practices (BMPs) dealing with the treatment and management of stormwater. Therefore, the installation of WSUD devices such as bio-retention systems and artificial wetlands offer a relatively inexpensive and passive form of pre-treatment of stormwater. Furthermore, the inclusion of these devices in amongst the urban environment will enhance the social amenity through the creation of open public space that encourages community engagement with water. This engagement can have strong educational benefits by improving the awareness of the close interaction between the urban environment and the groundwater below. Further educating communities and encouraging them to take ownership of the resource and to reduce pollution where possible, could improve the quality of stormwater so that it does not exceed the capacity of the pre-treatment devices and does not put the health of the aquifer and themselves at risk.

5.7.4 Transitioning to a Water Sensitive City

As discussed in Chapter 2 there has been a change in attitude towards conventional urban water management approaches and a shift to strategies that are more sustainable, resilient and adaptable (Brown and Farrelly, 2009). The rise of alternative management approaches such as WSUD and SuDS in recent decades have attempted to provide a framework for the management of urban water that is wholistic in its approach (Water by Design, 2009; Whelans *et al.*, 1994). This research has demonstrated several contributions within the field of WSUD as outlined in the theoretical framework by Fletcher *et al.* (2014) (Figure 2.2). This theoretical framework illustrates that much of the literature on WSUD tends to focus on the conceptual elements and broader theories. However, this study has proposed several strategies that seek to improve the technical aspects of WSUD, by stressing the value of groundwater in WSUD and opportunities for the implementation of MAR to assist in achieving specific WSUD objectives. The demonstration of these technical aspects serves to strengthen WSUD as an increasingly wholistic urban water management approach that is not merely a theoretical framework, but one that is able to provide practical strategies for the management of urban water resources. To this end, this research has shown that MAR can be used to manage the storage of an aquifer to maximise its potential for benefit to society in terms of stormwater management and improving urban water supply, while improving the aquifer's health and the associated ecosystem services.

Yet, this study has also sought to further develop the theoretical concepts and principals that provide the basis for these technical strategies. As a result, this research has highlighted the lack of groundwater consideration within WSUD and demonstrated the value of incorporating groundwater through the implementation of MAR. Implementing MAR as part of a city-scale WSUD strategy, will also have the added benefit of improving groundwater management in

general. The CFA is evidence of this, where its utilisation and protection have gone unregulated and managed for decades. Thus, it is important that cities begin to adopt urban water management approaches and frameworks that maintain some level benchmarking for their urban water management. The argument can be made that if Cape Town is genuinely striving to become a truly water sensitive city, one where all water is given due prominence in urban design, management and planning (Brown *et al.*, 2008). Then then the poor utilisation or mismanagement of any resources such as the CFA, will be detrimental to this cause. The study demonstrates that there is scope for the MAR to be a viable means of assisting Cape Town by providing means of flood prevention and increasing urban water supply.

Although this strategy is shown to be feasible for the city of Cape Town, the transferability of these findings is limited, as the properties of an aquifer are site specific. Furthermore, there are climatic and socio-political conditions that are unique to Cape Town, that need to be considered when assessing the transferability of the findings of this study. There is scope to apply the same rationale to similar issues in other South African cities and around the world. For example, the application of MAR in Johannesburg may be feasible, however the available aquifers in this area mainly dolomitic with difference hydraulic properties and different storage and yield dynamics. There are also number of different risks in the Johannesburg relating to the feasibility of MAR such as Acid Mine Drainage and geological instability as a result of artificial recharge. Therefore, while these results are not fully transferable, this study has shown that all water resources can assist a city as it progresses toward becoming truly water sensitive.

6. Conclusions

6.1 Outline of the study

Cape Town is becoming increasingly vulnerable to water scarcity due to stressed surface water resources, rapid urbanisation and the threat of climate change (DWAF, 2009; Rebelo *et al.*, 2011). Water-stress is an inherent problem in Cape Town, given its relatively hot, dry climatic conditions. Surface water resources, which are the primary source of water supply, are under pressure (Schulze, 1997; Turton, 2008; DEA, 2012). As a result, there is growing pressure to find alternative supply options to meet water demand. In addition to the water supply problems in Cape Town, the city is also prone to regular, seasonal flooding on the Cape Flats—a sandy, low altitude, flat plain within the city. This flooding has long been associated with elevated groundwater or ‘seepage’ flooding, which mainly affects the poorest settlements in Cape Town (Joubert and Martindale, 2013; Pharoah, 2013; Waddell and Ziervogel, 2014).

This thesis has aimed to contribute to the theory and application of WSUD by building on the knowledge of MAR and arguing for its incorporation in WSUD. The CFA in Cape Town is used to demonstrate this by assessing the potential role of the CFA for ensuring the sustainability of urban water management in the City of Cape Town. The main strategy to achieve this vision for the CFA has been to investigate the feasibility of implementing Managed Aquifer Recharge (MAR) on the CFA as this would maximise the yield or supply potential of the aquifer, while providing a means of reusing or recycling stormwater on the Cape Flats. The aim of this research was to evaluate if MAR could: (i) improve the water security of the City of Cape Town and (ii) to manage stormwater on the Cape Flats to prevent groundwater related flooding. In summation, the aim of this research was to examine the following questions:

1. *Is there sufficient storage capacity in the Cape Flats Aquifer for the recharge of winter stormwater from urban areas? Could this storage be enhanced through controlled summer abstractions for fit-for-purpose uses?*
2. *Can summer abstractions be used to mitigate groundwater related flooding on the Cape Flats?*

The specific objectives of this research involve establishing an integrated hydrological model, MIKE SHE, to identify the hydrological processes that influence groundwater recharge, aquifer storage and groundwater-surface water interactions in the CFA, on a regional-scale. The study makes use of available data, literature and past conceptual hydrogeological models of the CFA. The regional-scale model was used to develop a local-scale application of MIKE SHE, allowing for finer spatial resolution to determine the applicability of MAR for stormwater management on the Cape Flats.

6.2 Summary of the main findings

6.2.1 MAR Potential

The first contribution of this study has been to determine the storage potential or MAR potential of the CFA. The mapping of MAR potential has enabled the various intended MAR scenarios to be planned and sited. The MAR potential map has specified that many of the areas that are prone

to flooding on the Cape Flats are not ideal for artificial recharge as there is not enough additional aquifer storage in these areas. It was concluded that the scenarios aiming to achieve the objectives of flood prevention, specifically flooding as a result of groundwater flooding, should mainly focus on abstraction as additional infiltration would pose a flood risk. These abstraction scenarios have confirmed this assumption, as relatively extensive abstractions were able to achieve the specified flood mitigation objectives of reducing the local groundwater levels to below the flood mitigation threshold of -1.5 m. However, further abstraction aiming to create increased capacity for artificial recharge are not anticipated to be feasible due to the steep hydraulic gradient between the reduced groundwater levels at the flood sites and high groundwater elevation of the surrounding groundwater table. This means that any local storage created through summer abstractions near Sweet Home and Graveyard Pond recover significantly with natural recharge alone. In order to increase the aquifer storage at the flood mitigation sites, significantly more groundwater would need to be abstracted. Because the abstraction rates for the wellfields at Sweet Home and Graveyard Pond are restricted to maximum abstraction rate of 5 l.s^{-1} , additional boreholes would be needed to generate the required abstracted volume. The additional boreholes would increase the cost of the proposed flood mitigation scheme due to the need to install additional boreholes and pumps as well as increasing the operational cost due to the increased energy requirements of a larger wellfield. Increasing the capacity of the wellfields is unnecessary, given that other regions of the CFA (*viz.* Philippi and Mitchel's Plain) are better suited for seasonal and long-term storage. An alternative is to increase the length of the abstraction period, which would mean abstracting water when there is a low demand. The areas with the highest MAR potential have been presumed ideal for MAR, given the available aquifer storage, allowing for seasonal and long-term storage and reuse simulations.

Therefore, the mapping of MAR potential addresses the first question that has been posed by this thesis: "*Is there sufficient storage capacity in the Cape Flats Aquifer for winter stormwater?*". The answer to this question is complex. In terms of total storage potential in the CFA, there is additional storage capacity in the CFA for stormwater. However, an issue arises when the location for stormwater recharge is considered. Winter flooding on the Cape Flats often occurs in the areas where there is a low storage potential or where the groundwater table is already high and close to the ground surface. Therefore, storage opportunities in the CFA for stormwater where flooding regularly occurs is limited as these tend to be in low-lying areas. Thus, the areas with the highest storage potential are often located away from stormwater, especially in areas known for flooding. In short, there is sufficient storage in the CFA for stormwater recharge and storage, but it must be acknowledged that the stormwater will be required to be transported to the sites with the highest storage potential. This would require extensive infrastructure and energy to move storm water to where it can be recharged. As a result, this thesis has taken the view that groundwater abstractions are a valuable tool for flood mitigation and prevention along with more conventional strategies such as improving drainage and access to stormwater infrastructure. The stormwater in these areas should be considered for alternative treatment, reuse and fit-for-purpose strategies in line with WSUD objectives. Whereas, the MAR areas are better suited to deal with stormwater from the adjacent Philippi, Strandfontein and Mitchells Plain areas. In sequence, the following sections focus on the conclusions of the flood mitigation and MAR scenarios.

6.2.2 Flooding Mitigation

The simulations at Sweet Home and Graveyard Pond have shown that controlled summer abstractions are a feasible flood mitigation intervention to prevent groundwater related flooding. Hydrological modelling has demonstrated that the Graveyard Pond settlement is the most suitable for this type of MAR flood mitigation intervention, due to the relative ease of achieving the flood mitigation threshold when compared to Sweet Home. Thus, it can be anticipated that the costs of such an intervention will be significantly reduced to more than half of that at Sweet Home to achieve similar flood mitigation goals. The flood mitigation scenarios also demonstrate that some WSUD techniques and technologies cannot be assumed to be fully transferable. The success of WSUD and MAR, in particular, is dependent on a range of site-specific factors. The scenarios at Sweet Home and Graveyard Pond reiterate that the appropriate steps need to be considered before MAR is considered. The use of physically-based models are a valuable means of providing a low-cost method for assessing the risks, constraints and opportunities of a proposed WSUD or MAR intervention.

It was concluded that the flood mitigation strategy of controlled summer abstractions from flood mitigation is not conventional MAR. Yet, the main purpose of the flood mitigation scenarios is to offset current or natural recharge in order to avoid groundwater flooding. This strategy focuses on controlling the storage and groundwater levels in the aquifer to achieve specific MAR and WSUD objectives *viz.* preventing flooding and enhancing urban water supply. Therefore, the argument was made that the flood mitigation scenarios show substantial potential for MAR, due to its ability to control the water balance and properties of an aquifer that optimises its resource potential. Thereby, groundwater abstraction or artificial recharge were argued to be secondary components of MAR that serve the primary purpose of managing and regulating the resource potential of the aquifer. Therefore, this strategy does indirectly fulfil a number of MAR and WSUD objectives.

While the flood mitigation strategy at Graveyard Pond is likely to be the most cost effective and efficient, the implementation of controlled groundwater abstraction at Sweet Home settlement illustrates a number of indirect benefits to this strategy. The Sweet Home scenario highlighted the importance of wellfield design, where the use of scenario analysis helped to improve the placement of boreholes, thereby reducing localised impacts of the groundwater drawdown on the neighbouring, Edith Stephens wetland. Furthermore, localised abstractions are also connected to the indirect benefit of assisting protection of seasonal wetlands where they exist, helping to ensure these wetlands are sufficiently wet in winter and that drying does occur during summer. Therefore, MAR can be optimised to support and sustain local biodiversity and groundwater dependent ecosystems.

MAR in the form of controlled groundwater abstractions was shown, through simulation of contaminant transport, to be able to prevent or mitigate the migration of potential contaminants. The migration of pollution plumes from Sweet Home and Graveyard Pond have indicated that there is a potential long-term risk to downstream ecosystems, including the Zeekoeivlei wetland and the False Bay coastline. Therefore, strategic abstractions at the flood mitigation sites are likely to have positive impacts on biodiversity and ecosystem services in these areas. The removal of potential contaminants at these settlements is also anticipated to offer protection to potential MAR interventions downstream at Philippi and Mitchells Plain.

6.2.3 Water Supply

The MAR potential at Philippi and Mitchells Plain indicates that there is an opportunity to implement MAR as a stormwater reuse strategy. Storage analysis at the Philippi MAR site have shown that MAR is valuable for long-term or seasonal storage. However, after six years of recharge, the returns in yield start to decrease, suggesting that the system is more suitable for seasonal storage and reuse.

MAR was shown to improve the wellfield yield at Philippi allowing the maximum abstraction rate to be sustainably achieved, resulting in a sustainable yield of approximately 10 Mm³ per year. While, at Mitchells Plain, the wellfield yield using MAR could not achieve the same increases in wellfield yield when compared to Philippi at 7.8 Mm³ per year, making the total sustainable yield from both these MAR sites just under 18 Mm³ per year. Although this potential water supply yield seems relatively small when compared to the size of the current water supply system and urban water demands, there is still value in incorporating this yield with the integrated or blended approach to water supply options. In doing so, the use of MAR on the CFA will offer further flexibility for water supply as groundwater storage is less susceptible to fluctuations in weather or long-term climate change.

Due to restrictions on available land and the consequential cost implications for the establishment and running of a direct injection MAR scheme, it was concluded that injection wells are an appropriate option for MAR at Philippi and Mitchells Plain in order to achieve significant yields and to control the aquifer's storage appropriately. Selecting this option means that the unit cost of water is likely to be more expensive than the less energy intensive, passive infiltration methods. Due to the absence of filtration as a treatment stage in direct injection, the stormwater will require a level of pre-treatment, ideally within a SuDS or WSUD treatment train. By incorporating MAR as the main storage component within a larger WSUD stormwater treatment train, it is possible to achieve additional benefits for the urban environment including improved ecosystem health and biodiversity, improved social amenity and the utilisation of ecosystem services. Due to the costs and management requirements for such a system, it is recommended that MAR at Philippi and Mitchells Plain be utilised as an indirect potable water option. This is especially true if this stormwater reuse option is supplemented with the injection of recycled wastewater. It is essential to utilise the areas of the CFA with the highest storage potential for the highest value water use. This was based on the notion that the value of aquifer storage is greatest where there is the highest storage available with the greatest rates of injection/infiltration and abstraction. The areas of the CFA in the vicinity of the Philippi and Mitchells Plain are a relatively small fraction of the total area available. Thus, additional MAR sites could adopt a less expensive infiltration based MAR approach geared towards supplying fit-for-purpose water supply. These adjacent fit-for-purpose MAR systems could also assist in preventing groundwater contamination at the MAR schemes at Philippi and Mitchells Plain by assisting in maintaining positive hydraulic gradients.

Most notably, at Mitchells Plain, the extensive urban development covers much of the land overlying the aquifer, making the installation of a large centralised infiltration device—similar to that used in Atlantis—unfeasible. The use of many smaller, on-site, infiltration devices is impractical as it would be difficult to maintain and manage to ensure the system is preserving maximum storage and yields from the system. Therefore, the use of direct injection is suggested

as it can be more easily incorporated into the current urban layout while allowing greater control over the quality and quantity of stormwater for recharge.

6.3 Implications of research findings

The following section discusses the implications of the findings of this research and provides recommendations to urban water managers, planners and policy makers.

6.3.1 Recommendations for water managers

The results of this study have demonstrated that MAR implemented as part of a greater WSUD strategy holds significant value for the sustainable management of urban water. The findings of the flood mitigation scenarios have demonstrated that controlled summer abstraction from the CFA can reduce groundwater levels sufficiently to lower the risk of localised flooding in areas groundwater related flooding has been problematic. This study has shown that with, strategic planning and design, MAR can be utilised to mitigate the seasonal groundwater relating on the Cape Flats. Therefore, it is recommended that further work be carried out to understand seasonal groundwater related flooding in more detail to identify priority areas to implement MAR flood mitigation strategies. Future proposals for MAR focusing on urban water supply should be cognisant of the secondary benefits of potential flood mitigation on the CFA, which could be used as further justification for MAR.

In terms of MAR for water supply, the CFA presents a valuable opportunity for the implementation of MAR to contribute towards improving the water security of Cape Town. As demonstrated in the discussion chapter of this thesis, the implementation of MAR is well aligned with the WCWSS Reconciliation Strategy offering a means of achieving the effective reuse of water and providing seasonal storage that is resilient to climate change. Given the findings of this research it is recommended that pilot studies be implemented as soon as possible to identify any technical issues that may arise and so that site specific testing can be carried out. Given the current draught conditions (2016 - 2018) additional water resources are in high demand, therefore the CFA is a resource that can prove an invaluable means of relieving the risk of the current water crisis, while helping to build an improved resilience to future droughts. It must also be mentioned that in times of crises, there is the temptation to spend a large amount of capital on emergency water supplies such as desalination, that may not yield long term returns on the initial investment. In the authors opinion, MAR could be more naturally integrated with the current surface water supply system as it is less technically complex and has comparatively lower energy requirements.

6.3.2 Recommendations for urban planners

If MAR is to be implement in the CFA it is essential that the city overlying the aquifer is well planned and designed. This is where the integration of MAR with WSUD is essential to ensure that the aquifer is protected from pollution and degradation. The adoption of SuDS strategies will improve the quantity and quality of groundwater recharge, improving the health of the aquifer. The Cape Flats in particular needs improved planning and design. The current land use, has many potential contaminant risks including industry and informal settlements. Thus, to resolve these risks, land use planning is essential making sure area are protected or at least accounted for at the design stage. In terms of informal residential areas, the provision of appropriate housing and

sanitation services is essential, not only for aquifer health but as a socioeconomic imperative as housing and sanitation services are basic human rights. This study also demonstrated the value of modelling and its ability to predict the likely impacts to a resource. The improved modelling of urban land use and the hydrological cycle is a valuable tool that will assist urban planners and designers to ensure that our cities are progressively becoming more water sensitive.

6.3.3 Recommendations for policy makers

For MAR and WSUD strategies to be successful, there needs to be a strong policy framework to ensure that these strategies are properly developed and maintained. It is important that the implementation of MAR and WSUD follow international best management practice and standards. Furthermore, there needs to be improved monitoring of groundwater resources to ensure the devices and strategies that have been implemented are achieving their initial objectives and are not causing detriment to the aquifer. Groundwater monitoring will also help to regulate land use and to ensure the pollution is minimised or prevented. Improved policy relating to MAR and WSUD will also have the added benefit of assisting areas where there has previously been a lack of policy, such as groundwater resources.

6.4 Contribution to knowledge

This thesis contributes to knowledge in a number of ways. These include:

- The use of integrated hydrological modelling, able to represent both surface and groundwater processes, is demonstrated as a valuable means of understanding the complete urban hydrological cycle for aiding urban water management, such as WSUD.
- The integrated modelling approach allows for a physically-based determination of groundwater recharge for the CFA. Previous attempts for evaluating recharge for the CFA have relied largely on indirect empirical methods.
- The MAR potential is mapped for the CFA which is valuable for future MAR planning and design. With improved information on aquifer characteristics, water quality, land use, soil and geology, this mapping methodology is an essential tool for groundwater management and planning on the CFA.
- The use of MAR, through controlled groundwater abstractions, to artificially lower the water table demonstrates a feasible means of reducing groundwater related flooding on the Cape Flats, Cape Town.
- MAR is shown to be a viable water supply option for the City of Cape Town by facilitating the reuse or recycling of stormwater or treated wastewater, potentially contributing approximately 18 Mm³ per year towards the city's water supply.
- The evaluation of contaminant (TDS) transport simulations demonstrates the primary contaminant flow paths from the Flood and MAR sites. This shows downstream sites that are at risk of contamination, regardless of whether MAR is conducted or not.
- The integrated hydrological modelling approach is used to quantify the likely impacts of future climate change on the entire water balance. This is particularly essential from a

groundwater perspective as these impacts have not been evaluated for the CFA as yet and can easily go undetected due to the 'hidden nature' of the resource.

- Based on the results of the integrated hydrological modelling, the resource value of the CFA is reiterated. As a result, it was acknowledged that a pilot study is an essential step towards ensuring this resource is utilised in the near future.
- It was shown that MAR is a tool for the application of WSUD in Cape Town, offering value for water supply and stormwater management. Additionally, WSUD and its application in MAR offer valuable benefits including pre-treatment, enhanced public amenity and improved biodiversity.
- Therefore, given all of the above, this thesis provides a new theoretical dialogue for the improved integration and use of MAR strategies to aid the implementation of WSUD and the creation of Water Sensitive Cities in South Africa and internationally.

6.5 Limitations and further research

One noteworthy limitation of this study was a lack of data and monitoring of groundwater and surface water on the Cape Flats. An improvement to the amount and quality of data will assist the development of more realistic conceptual models and more accurate and reliable numerical models. In the City of Cape Town, there is a lack of consistent and accurate streamflow data. The current datasets, where they exist, are highly error prone and inconsistent with large periods without measurements. Groundwater monitoring could benefit from further investment. The current datasets for groundwater levels are monitored by the Department of Water and Sanitation (DWS) and are located in a relatively random array across the CFA with some boreholes clustered around particular points. As a result, it is difficult to get a representative evaluation of the nature of groundwater levels on a regional-scale. Furthermore, many of these boreholes are located in the central and southern parts of the CFA with little coverage in the peripheral areas to the west, east and north. Many of the current borehole records have a coarse temporal resolution and are only measured once a month. Some of the boreholes are clearly located near to sites that experience significant amounts of groundwater abstractions and without sufficient groundwater usage data, it is difficult to determine the natural groundwater levels or represent this process within a modelling framework.

An example of this data shortage was noted at the Philippi MAR site. The simulated groundwater levels from the calibrated model showed groundwater levels below -10 m below the ground level. This calibration was based on boreholes that were within 1 to 2 km range of the Philippi site. However, an environmental assessment report for the Oakland housing development that was published during the final stages of this research, suggested that the groundwater levels in the vicinity of the Philippi MAR site may be higher than the groundwater levels simulated with the model, suggesting the possibility of a perched water table (UDWC, 2016). A perched water table resulting from the presence of a clay or calcrete lens is near impossible to incorporate into the conceptual hydrogeological model without detailed mapping of the regional hydrogeology. This highlights the need for improved groundwater monitoring and mapping of aquifer characteristics. More specifically, there is a need for a pilot study to more fully test MAR in terms of its efficiency and costing. Furthermore, more detailed monitoring and the implementation of

pilot studies will enable a comprehensive sensitivity analyses to be undertaken to ensure the conceptual model is appropriate.

In addition to the need for observed data for calibration, this modelling application using MIKE SHE could also be improved with more detailed information describing the land use and soils of the Cape Flats. The measurement of LAI and rooting depth would improve the accuracy of simulated evaporation and groundwater recharge. Improving the representation of the unsaturated zone in MIKE SHE with more detailed soil information is also essential to improve the performance of the model. The measurement of hydraulic properties of the Cape Flats soils is important. This can be derived from field and lab experimentation. Of particular value in MIKE SHE is to derive the soil water retention and conductivity curves for each soil type which are required for the application of the more physically-based Richards equation or Gravity flow method.

With improved monitoring, there are greater opportunities for more detailed modelling applications at much higher spatial and temporal resolutions. A particular field of further study would be a more detailed simulation and measurement of the processes causing flooding on the Cape Flats. Such a study would require detailed measurement of surface water processes such as runoff and ponding to account for flooding as a result of poor drainage. Whereas a more detailed understanding of groundwater flooding would require more information on the unsaturated and saturated zones. In this case, soil moisture and accurate groundwater level data would be essential. Furthermore, the development of a hydrological model in conjunction with the establishment of a specific pilot and monitoring programs would improve the calibration and validation of models.

Based on the conclusions, limitations and recommended future research outlined in this thesis, the following management actions are recommended. Firstly, the emphasis of WSUD is needed to sustainably utilise all available resources to meet the required demand. The CFA is a resource that has long been considered for development but has been deferred in favour of larger surface water projects. Secondly, in order to break the cycle of inaction in the utilisation of the CFA, it is recommended that the scenarios from this thesis are used to design an appropriate pilot study that is able to describe the hydrogeological conditions in detail and allow the aquifer responses to be tested under field conditions. A pilot study also allows for experimentation and a level of 'fine tuning' of elements that arise under field conditions that would otherwise not be apparent when using a model. Furthermore, pilot studies are essential for deriving actual costing and yield output necessary for leveraging the support and investment of stakeholders. Finally, and in addition to a pilot study, further investment should be directed towards broader groundwater monitoring programs so that baseline conditions are better understood. This is particularly important for groundwater quality as this is necessary to distinguish between the current state of groundwater pollution and the influence of potential future pollution sources.

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Publications

This following section lists the contributions of this study towards other publications and conference proceedings.

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Appendix A: Groundwater in South Africa

Much of the information on the status of groundwater resources in South Africa was developed by the Department of Water Affairs and Forestry (DWAF), through the Groundwater Resources Assessment (GRA). The results of this project were incorporated into a Water Research Commission project entitled “Water Resources of South Africa, 2005 Study (WR2005)” (Middleton and Bailey, 2009). The WR2005 attempts to quantify the groundwater resources in South Africa, so as to estimate the Average Groundwater Resource Potential (AGRP). The AGRP is the maximum volume of groundwater that could be abstracted from an aquifer on a sustainable basis, on the assumption that boreholes are evenly distributed across the aquifer system; however this is both physically and economically impractical (Middleton and Bailey, 2009). Therefore, in order to account for the groundwater that could actually be utilised, exploitation and potability factors were introduced, which led to the concept of Utilisable Groundwater Exploitation Potential (UGEP) for South Africa (Figure A A.). The total UGEP for South Africa is said to be 10 350 million m³ per annum under normal rainfall conditions and 7 500 million m³ per annum under drought conditions (Middleton and Bailey, 2009). The current use of groundwater is between 2000 and 4000 million m³ per annum, indicating that there is potential to make use of the additional groundwater available. It is clear that over most of the country the values of UGEP are generally low due to poor borehole yields and water quality, however a number of the countries large metropolitan areas are located near areas with high UGEP such as Pretoria, Johannesburg and Durban (Figure A A.). The highest UGEP values (25 001 – >100 000 m³.km⁻².a⁻¹) are located in Limpopo, Mpumalanga, the North West, Eastern and Western Cape provinces. These are the areas that are able to provide potable water at a high yield.

Figure A B. shows groundwater occurrence or groundwater yield, which is the amount of water that can be extracted from aquifers in different parts of the country. The areas in green and blue are associated with the highest level of groundwater yield with volumes in excess of 200 m³.day⁻¹. These areas are typically associated with fractured rock aquifers such as the green areas demarcating the Table Mountain Group Aquifer (TMG) close to the City of Cape Town and the blue areas near Johannesburg which are highly productive dolomitic karst aquifers. Intergranular aquifers tend to dominate the rest of South Africa and are associated with lower groundwater yield.

The quality of groundwater is an important factor that determines the usability of the water as different applications require different water quality standards. Electrical Conductivity (EC) is a common indicator of the water quality of water resources, as it gives an indication of the presence of dissolved minerals in the water which can cause the water to become brackish or even saline (DWA, 2010). High EC values can be attributed to natural causes such as the geological composition of the aquifer and residence time, but can also be linked to anthropogenic factors such as agriculture and industry (DEA, 2012). The areas that show the worst EC are in the northern and western parts of the Northern Cape as well as the northern parts of the Western Cape (Figure A C.).

The depth of groundwater or water table in meters below ground level (mbgl) gives an indication of how accessible groundwater resources are (Figure A D.). The accessibility of groundwater has economic implications, because with increasing depth the costs of groundwater

infrastructure (pumps and piping) and installation (drilling) increase with depth. There are also environmental implications, where the risk of pollution of groundwater resources is higher in areas where the groundwater is closer to the surface.

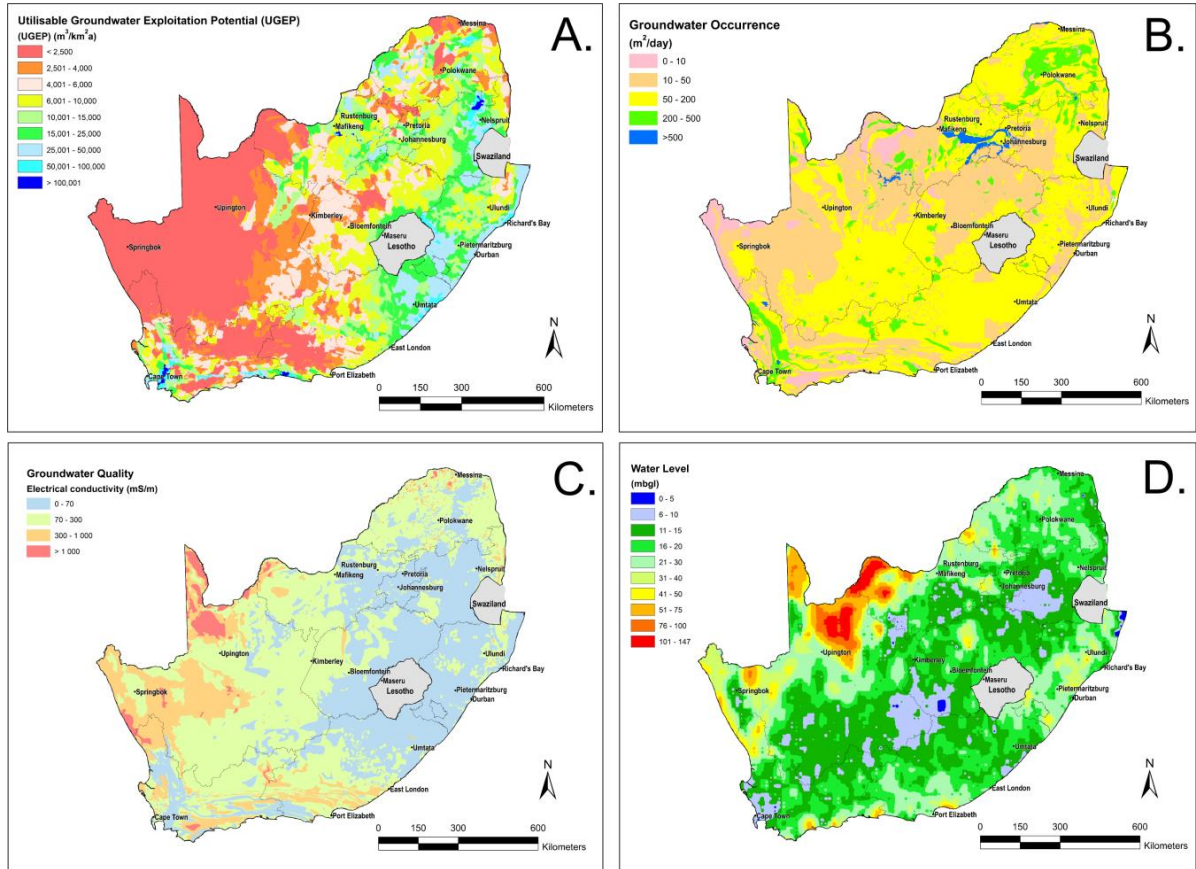


Figure A.1 The Utilisable Groundwater Exploitation Potential (UGEP) (A), the groundwater occurrence or yield (m^3/day) (B), Groundwater Quality (mS/m) (C) and the water level of groundwater (mbgl) (D) for South Africa (after Middleton and Bailey, 2009)

Appendix B: Private groundwater use in Cape Town

The private use of groundwater in Cape Town provides some insight into the value of groundwater as a fit-for-purpose supply. The City of Cape Town municipality actively encourages groundwater use for applications that do not require high water quality such as refilling swimming pools and watering gardens. The municipality also encourages the voluntary registration of private wells and boreholes for research and environmental monitoring purposes (Figure A.1) (CoCT, 2013c). Although the registrations may not be an accurate representation of the total number of wellpoints and boreholes since registrations are voluntary, but it does give an indication of who is using groundwater and the location of these registrations. From 2004 to 2012 the City of Cape Town has registered approximately 3035 groundwater users (Figure A.1). The largest number of registered users is found in the northern parts of the city in areas including Melkbosstrand, Milnerton, Bellville, Kuilsrivier, Brackenfell and Durbanville. The number of groundwater registrations is also high in the eastern parts of the city in Somerset West and Strand. The areas in north are drier compared to areas in the south where the number of groundwater registrations are lower, which is likely to be associated to the demand for groundwater for garden irrigation during the dry winter months. The availability of groundwater also plays a role in groundwater registrations, where a majority of the areas of high groundwater use are located on or near aquifers of high or moderate productivity. According to the National Groundwater Strategy (NGS) the areas of high productivity are likely to have borehole yields of greater than $5 \ell.s^{-1}$ and the areas of moderate productivity are likely to have borehole yields of between 2 and $5 \ell.s^{-1}$.

Another notable trend is the socio-economic restrictions that limit groundwater use. This is evident in areas such as Khayalitsha and Gugulethu and other low-income areas, have access to productive aquifers, but the resource is not being used due to costs associated with the installation of wellpoints and boreholes (Colvin and Saayman, 2007). Of the groundwater registrations for the year 2012, 61 % were wellpoints and 39 % were boreholes. The installation of a borehole, which is deeper than a well-point, is often significantly more expensive to install than a wellpoint (Colvin and Saayman, 2007). This limits who can access groundwater and how much water they are able to abstract as deeper boreholes will be less likely to run dry. Due to the lack of regulation and monitoring there is a risk that groundwater could be overused or potential contamination may go undetected, resulting in negative impacts on local ecosystems, on human health and the value of resource itself.

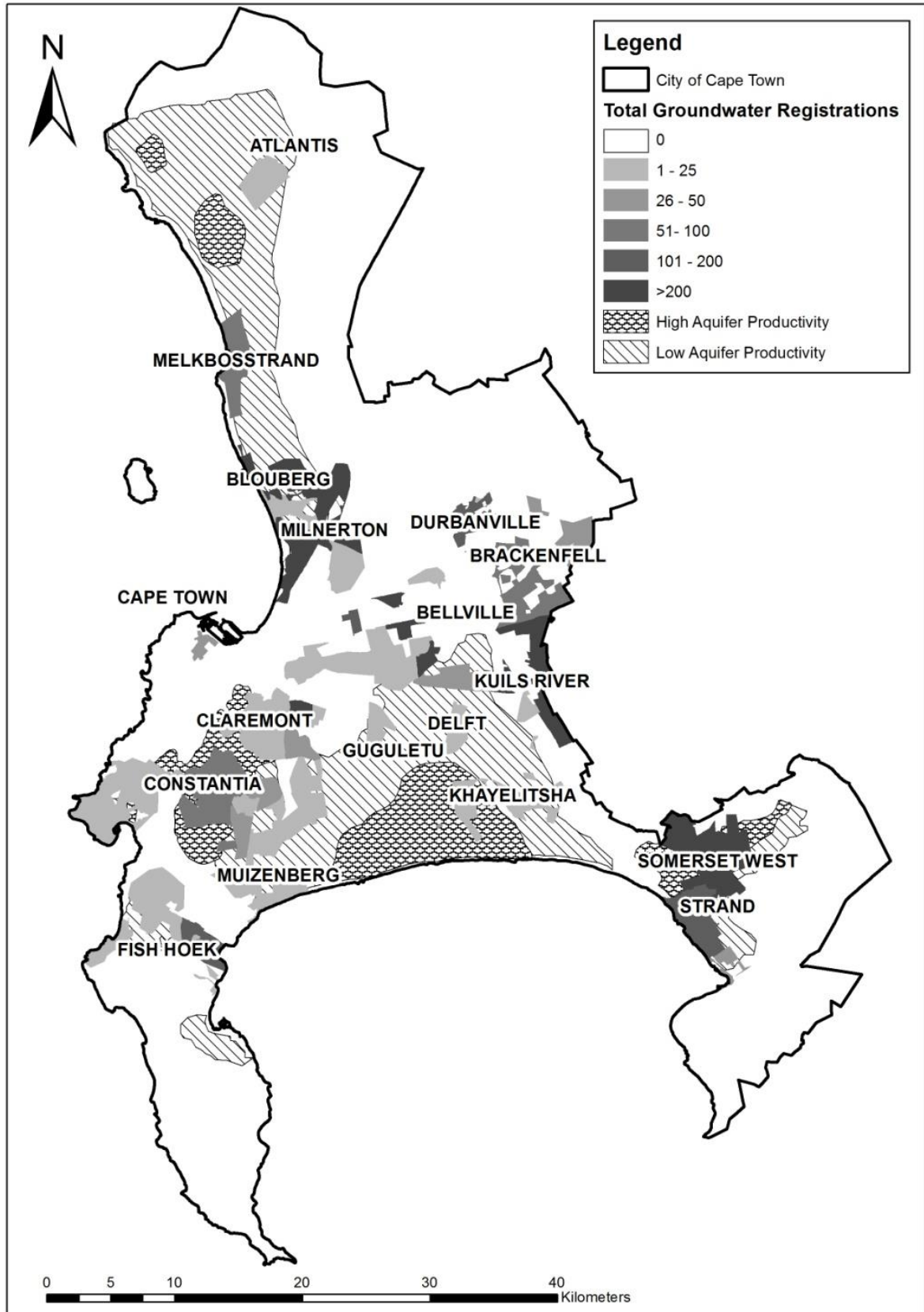


Figure A.1: The distribution of groundwater registrations and aquifers in the City of Cape Town from 2004 – 2012

Appendix C: The Urban Water Cycle

A.1 Land use and the water Cycle

Land use characteristics such as the vegetation and soil influence the hydrological response that can be expected from a catchment (Falkenmark *et al.*, 1999). Thus, a change in land use is associated with a change in hydrological response (Falkenmark *et al.*, 1999; DeFries and Eshleman, 2004; Warburton, 2012). Land use determines the hydrological response because it determines how water is partitioned in the hydrological cycle. Partitioning describes how rainfall is separated into the different hydrological components of the water cycle such as interception, runoff, infiltration and groundwater recharge. There are three main partition points in the hydrological cycle, *viz.* the vegetation, the soil surface and the root zone (Figure A.2) (Falkenmark *et al.*, 1999; Schulze, 2004; Mauck, 2012).

The first partitioning point is at the vegetation where rainfall is either, captured and held on the leaves or stem of the plant as interception or passes through the vegetation to the soil surface as throughfall or stemflow. Intercepted water that is captured on the vegetation eventually evaporates contributing to the total evaporation (Ward and Robinson, 2000; Mauck, 2012). Water that reaches the soil surface is partitioned into runoff or infiltration. The structure and texture of the soil determines whether infiltration or runoff will occur. Generally, runoff is generated when the intensity of a rainfall event exceeds the infiltration capacity of the soil (Ward and Robinson, 2000). Thus, soils with relatively low infiltration rates, such as clayey soils, have a larger runoff component than sandy soils, which have a relatively higher infiltration rate. The difference between these two soils is due to the difference in texture, where sandy soils generally have a higher porosity than clayey soils. Meaning that the ratio of voids to solids in the soil's composition is higher in the sandy soil than the clayey soil. Permeability is another factor that determines the rate of infiltration in a soil. Permeability is the capacity of the soil to allow water through it, it is also known as the hydraulic conductivity of the soil (Whitlow, 2001). Additionally, topographic elements also determine how much runoff or infiltration is experienced (Falkenmark *et al.*, 1999). Steep slopes for example, will increase the chances of lateral flow reducing the time available for infiltration to occur. The final partitioning point is located at the root zone. The roots in the root zone determine how much water will be abstracted from the soil profile for transpiration required by the plants. Any water that passes beyond the influence of the plant roots is assumed to contribute to groundwater recharge.

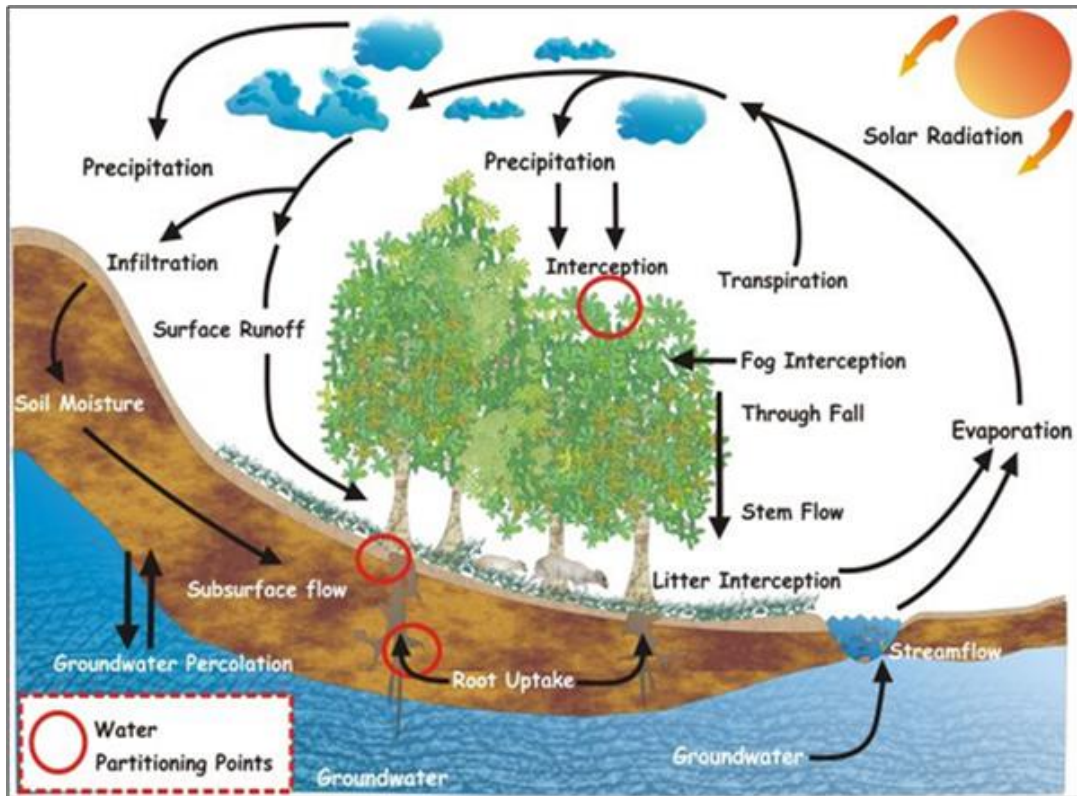


Figure A.3: The water cycle partitioned at the vegetation, soil surface, and the root zone (Jewitt, 2005)

A2 Urban impacts on the water cycle

The United Nations (2014) estimates that currently 54 % of the world's population live in urban areas and that this percentage is likely to increase to 66 % by the year 2050. The increase in population living in the world's urban centres will inevitably lead to the expansion or 'sprawling' of urban growth in cities. Generally, this sprawling urban growth occurs on natural or agricultural land on the periphery of cities (Braud *et al.*, 2013). Characteristically, urban land use consists of hardened, impervious surfaces such as concrete and tar, which consists of roads, pavements, parking lots and roofs. Therefore, as urban growth occurs it typically entails the replacement of natural or agricultural land uses, which have natural or slightly modified flow regimes, with hardened impervious surfaces. The hardened impervious surfaces are critical for drainage in cities, rapidly removing stormwater, which can be dangerous if allowed to pond. In general, rainfall in the natural water balance is partitioned into evapotranspiration and infiltration, with only a small amount of runoff generated (Figure A.2). Conversely, the urban water balance is significantly altered with the evaporation and infiltration components of the water cycle being reduced due to the increased runoff generated from impervious surfaces (Figure A.2).

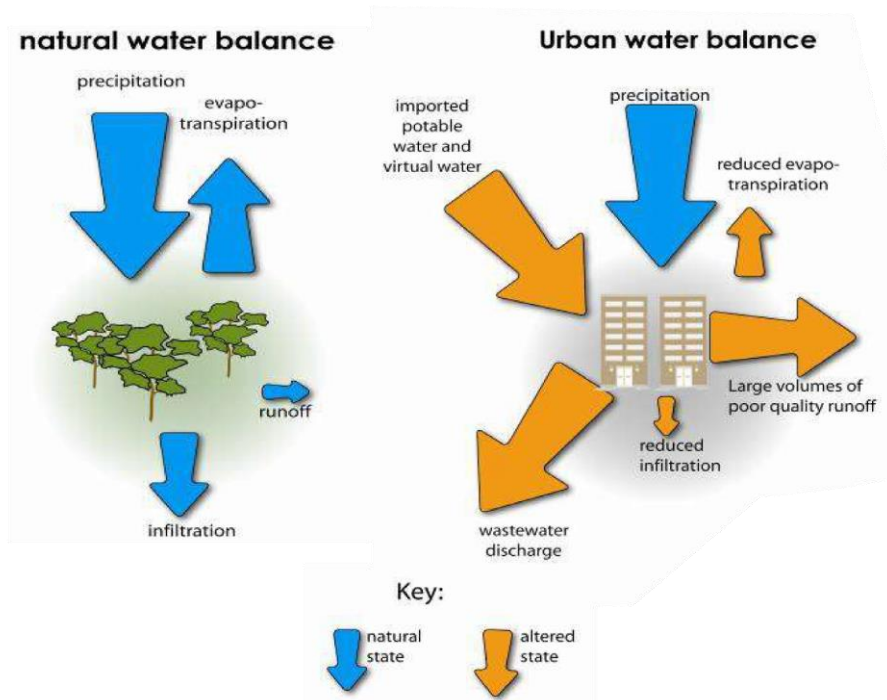


Figure A.2: Urban water cycle illustrating how urban land use alters the natural hydrological responses (Hoban and Wong, 2006)

The presence of impervious surfaces in urban areas are known to have a number of negative hydrological impacts and have been well documented (Leopold, 1968; Hollis, 1975; Hall, 1984; Ward and Robinson, 2000; Jenerette and Potere, 2010). A strong focus in past studies is the determination of downstream flooding risks due to increasing imperviousness in urban areas preventing infiltration and increasing the rate and volume with which water is drained away. Assessing the impacts of urban land use on the flow regime in a river can be done by assessing the changes to the stormflow hydrograph. Figure A.3 illustrates a typical pre-development or natural stormflow hydrograph compared to a typical urban stormflow hydrograph. Most notably the time-to-peak or lag time is more rapid in the urban hydrograph due to the rapid removal of stormwater from the urban areas via impervious surfaces and stormwater drains. The lag time describes the time difference between the peak of the runoff and the peak rainfall (Figure A.3). The volume of the peak discharge is also higher than the pre-development hydrograph. This means that the river will experience more ‘flashy’ flow conditions, where the rising and falling limbs of the urban stormflow hydrograph are steeper than those of the pre-development hydrograph.

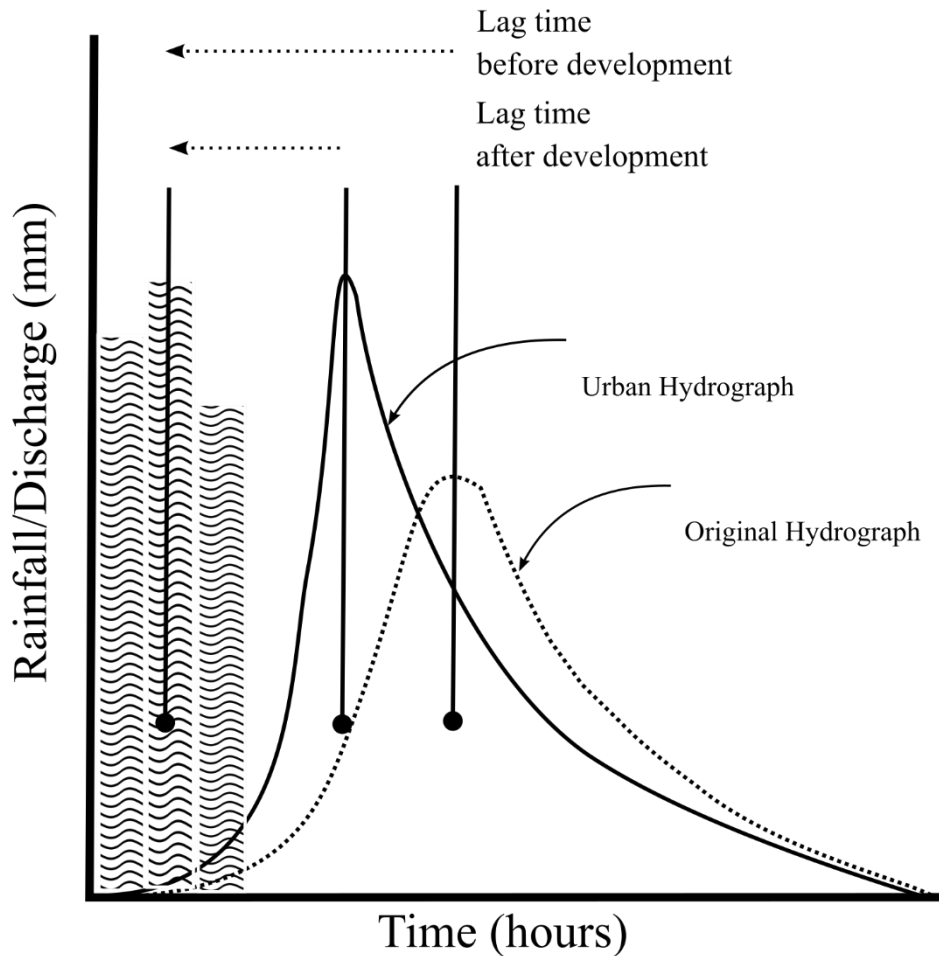


Figure A.3: Stormflow hydrographs for before and after urban development (after Leopold, 1968)

Groundwater is also impacted by urban land use, where increased imperviousness in urban areas reduces the ability of water to penetrate the soil surface and percolate down to the water table (Rose and Peters, 2001; Mauck, 2012). However, Lerner (2002) raises the issue that the reduced groundwater recharge in urban areas as a result of imperviousness can be offset by unintentional recharge from leaking sewerage and bulk water mains and the over-watering of parks and gardens. Recharge from leaking sewerage and bulk water mains are commonly experienced, especially in South Africa (Younger, 2007; Mckenzie *et al.*, 2012). Furthermore, many cities often exceed their local water supply capacity and require additional water supplies from adjacent catchments. Therefore, any water lost through leakages in sewerage and bulk water networks is water that would not naturally occur within that catchment. Therefore, groundwater recharge in urban areas can be a substantial part of the total water balance (Younger, 2007; Mauck, 2012).

Appendix D: Water Sensitive Settlements

This appendix addresses how the concept of a Water Sensitive City has been reconceived and adapted for the South African context. The original concept of the Water Sensitive City was developed by Brown *et al.* (2008). Brown *et al.* (2008 p. 2) suggest that the process of moving towards the goal of a Water Sensitive City is influenced to additional factors such as “...*specific histories, ecologies, geographies and socio-political dynamics*”. This is particularly true in the South African context where the Apartheid government pre-1994, created a socially and economically segregated urban landscape. After more than 20 years of democracy, the legacy of apartheid remains in most of South Africa’s major Cities. There is still a significant gap between the rich and poor and many households lack access to basic services, such as water and sanitation. Most affected by these conditions are those who live in informal settlements. Thus, Armitage *et al.* (2014) recognised that this benchmarking framework may have limited applicability given that informal settlements often lack basic services such as urban drainage. Therefore, the framework was adapted to improve the representation of South African conditions (Figure A.4). The new framework shows the concurrent development of two settlement types, formal and informal settlements. The aim of this development is to provide services to informal settlements in a sustainable manner, while retrofitting formal settlement with sustainable infrastructure.

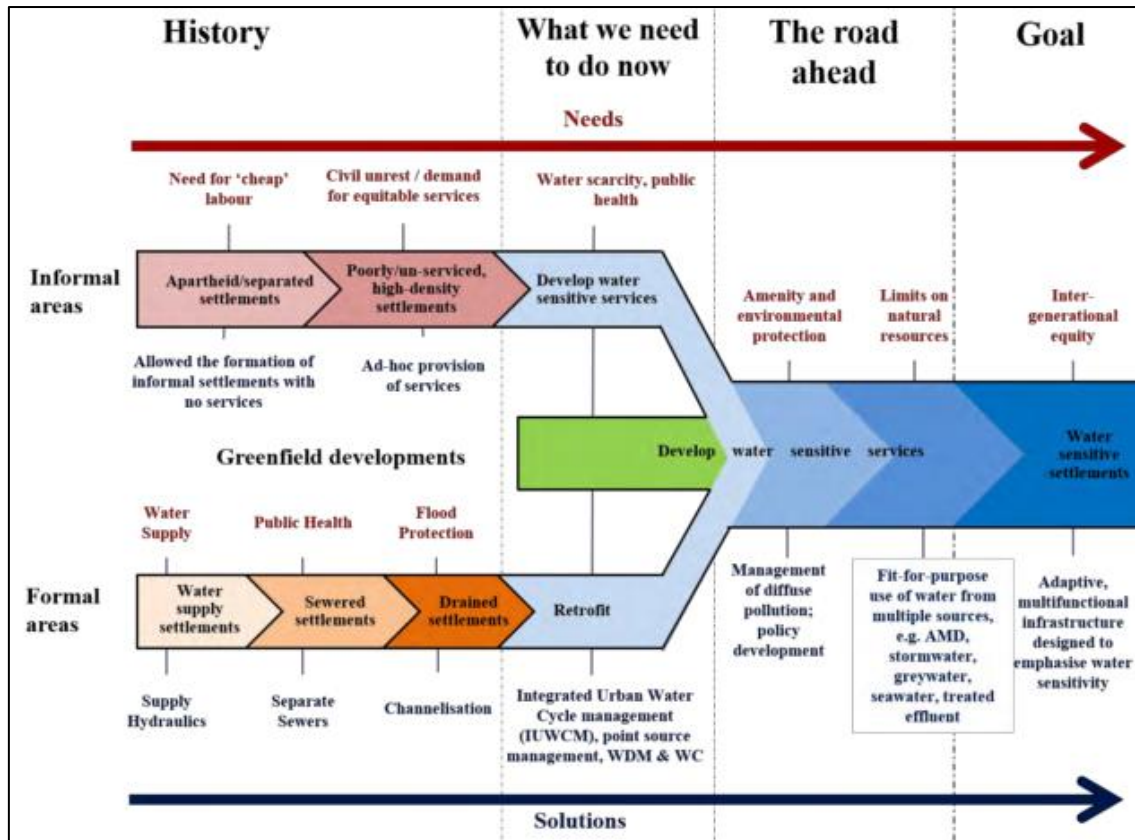


Figure A.4: The framework describing the transition states towards a ‘water sensitive settlement’, highlighting the different development process required for informal and formal areas (Armitage *et al.*, 2014)

Appendix E: Photographs of the Cape Flats

The following photographs serves to give some context to this study highlighting the soil conditions, land use and natural vegetation of the Cape Flats as referred to in this study.

Figure A.5 shows an area of the Philippi Horticultural Area (PHA) facing east towards Stellenbosch and Somerset West where the Helderberg Mountains can be seen in the distance. This photograph highlights the commercial farming of vegetables in the white sandy soils of the Cape Flats. The sandy soils in this area are well drained and are ideal for growing vegetables. The future use of this area is contentious as property developers have moved to develop the land as a housing development. Subsequently, they have been met with stern opposition from the farmers of the PHA.



Figure A.5: A view over the Philippi Horticultural Area (PHA) facing east towards the Helderberg Mountains near Stellenbosch and Somerset West. This area is utilised for the commercial farming of vegetables

Figure A.6 shows a view from the dunes of the Wolfgat Nature Reserve located just south of Mitchells Plain overlooking the False Bay coastline towards Somerset West and Gordons Bay. The vegetation in this area is one only of the few small pockets of natural fynbos vegetation known as Cape Flats Dune Strandveld (Rebelo *et al.*, 2011).



Figure A.6: A view from Mitchells Plain over the False Bay coastline towards Somerset West and Gordons Bay

Figure A.7 shows some soil testing performed by an environmental science honours student, performing infiltration tests using a single ring infiltrometer and mini disc portable infiltrometer. This picture gives a clearer indication of the height of the vegetation which is relatively short, approximately 20 – 40 cm tall. Larger, denser thickets can also be seen in the back ground which are between 1 – 1.5 m tall.



Figure A.7: Performing infiltration tests on the soils of the Cape Flats using a single ring infiltrometer and mini disc portable infiltrometer