The viability of transforming stormwater detention ponds into infiltration ponds on the Cape Flats, South Africa

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Thesis submitted in fulfilment of the requirements for the degree of Doctor of Philosophy (PhD)

Submission Date: 25 August 2023
Institution: University of Cape Town
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Declaration

I, **Craig Tinashe Tanyanyiwa,** acknowledge that plagiarism is wrong and declare that this thesis is my own work and I have appropriately refenced sections where other people's work – including my own – has been used. Over the course of the study, I worked with several BSc (Eng.) students whom I co-supervised and two MSc students, and their contributions to this thesis have been attributed to the respective individuals. Some findings from this thesis have been published in journals as a requirement for the provision of funding. Other than these journal articles I have not permitted anyone to copy this thesis with the intention of passing it off as their own work.

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Acknowledgements

I begin by expressing my gratitude to the divine presence that guides me – God, who accepts us as we are and remains a constant source of solace and strength.

I extend my profound appreciation to Professor Neil Armitage, my principal supervisor, whose unwavering academic guidance, and financial backing were pivotal in bringing this work to fruition. Through numerous meetings, drafts, discussions, and shared trips and experiences, you have shaped me into a more adept researcher and writer and ignited my curiosity. My new habit of questioning everything is a testament to your profound influence. Moreover, your generous support in funding professional development courses and conferences has enriched my understanding of general hydraulics and urban drainage. It is with immense pride that I acknowledge your role as my mentor and supervisor.

Dr. John Okedi, my co-supervisor, deserves special acknowledgement for laying the foundation for this research. Your expert guidance and assistance were instrumental in addressing the complex challenges presented in this thesis. The spontaneous discussions we had were helpful during times of uncertainty.

To Dr. Kirsty Carden, your emotional and financial support throughout my research journey was invaluable. Your relentless efforts to secure bursaries and scholarships and even lending your car for fieldwork made a significant difference.

This endeavour has been made possible by the gracious support of numerous scholarships and grants. My heartfelt thanks extend to the Future Water Research Institute for their funding under the 'Pathways to water-resilient South African Cities' (PaWS) project, grant number DFC-18-M05-KU, as well as The Donald McGregor Scholarship and the National Research Foundation (NRF) through the SARChI Chair in Governance and Economics for Ecological Infrastructure held by Professor Martine Visser, whose endorsement bolstered my application.

I am indebted to the South African Weather Services and the University of Cape Town (UCT) Climate Systems Analysis Group for providing vital climate data. I also thank the City of Cape Town and Umvoto Africa for the data and resources that enriched this study. Similarly, my thanks extend to Computational Hydraulics International, who provided a student grant for the PCSWMM licence. The UCT High-Performance Computing platform also deserves recognition.

Central to this work is the collaborative effort of the PaWS project jointly led by the UCT Future Water Research Institute and the University of Copenhagen. Support from fellow team members was valuable, and the insights and feedback were instrumental in shaping the direction of this work. I extend my gratitude to

Dr. Amber Abrams and Dr. Patience Mguni for their expertise in social aspects, which guided me in thinking and writing like a social scientist. The collective collaboration in research endeavours transformed my perspective and capabilities. I also want to thank Barry Lewis from UBU for the various contributions to the project and friendship over the last three years.

Rachelle Schneuwly's role in helping to establish the laboratory and field experiments was crucial in ensuring the investigations were conducted efficiently. The regular trips to the pond for sample collection, filled with problem-solving discussions, will be missed. Your competence, empathy and support have had lasting impacts.

Acknowledgement goes to the several B.Eng. students who contributed to this thesis through their final year projects, which I co-supervised: Michael Jones, Tashinga Dzurume, Firdaus Isaacs, Kirstin Naicker, and Bukasa Mukuna. Their efforts in initial site assessments and data analyses are greatly appreciated.

I would also like to express my heartfelt appreciation to my friends at the Future Water Institute. I wish to extend special recognition to Calvin van de Merwe, Jessica Fell, Tsepiso Lepota, Anabel Matalanga, and Dr. Sithabile Hlahla for their camaraderie and academic support. I especially want to highlight Teboho Mofokeng, my thesis 'buddy', whose assistance played a pivotal role in refining my thesis contribution. I am deeply grateful for your guidance and support. Your generous provision of Grammarly and Paperpal licences considerably enhanced the quality of my thesis. I look forward to reading your thesis soon.

Heartfelt appreciation also goes to my friends and family, whose support and encouragement sustained me. To my parents, Only Tanyanyiwa and Barbara Gwangwara, who invested in my future and motivated me to work hard, this achievement is a tribute to your love and sacrifices. I also thank Menfree and Philippa Tanyanyiwa for their encouragement and prayers. The 'boys' – Bakani, Michael, Terry, Tino, Tabeth, and Freeman – were pillars of emotional support. I also want to thank my Obs Med School life group friends for their support. A special mention to my friend and sister Sikhumbuzo Sithole for her constant presence and the many jokes we shared when I needed a laugh.

To Tafadzwa Mpofu, my partner, your steadfast love, patience, and encouragement were the bedrock of my journey from the start to the end. Your review of this thesis deserves special gratitude.

In closing, my heartfelt thanks extend to everyone who contributed to my research and its completion. Although I cannot name you all, your contributions have been immeasurable and deeply appreciated.

Abstract

The City of Cape Town (CoCT) is committed to implementing various measures to increase water security, including groundwater use with managed aquifer recharge (MAR). A previous simulation-based study found that stormwater harvesting (SWH) via MAR was viable in Cape Town and proposed implementing MAR in retrofitted detention ponds overlying the unconfined Cape Flats Aquifer (CFA).

This study investigated the viability of retrofitting detention ponds in Cape Town to facilitate urban MAR at the neighbourhood scale. The research methods used included desktop analysis, field investigations, unstructured participant observations, laboratory investigations, computational hydrological modelling, and financial analysis.

A field-scale case study in Mitchells Plain, Cape Town, demonstrated the importance of community engagement in successfully implementing detention pond retrofit projects. The developmental nature of retrofit projects requires collaboration between city officials and communities with continuous outreach efforts to address conflicting priorities and interests. This approach diverges from conventional engineering practices and requires a significant time investment. A computational hydraulic simulation using the PCSWMM Professional (version 7.5) software revealed that retrofitting a detention pond to improve infiltration led to a MAR increase of 290%. Laboratory studies showed that stormwater from the CoCT, which was of poor quality, could be treated by allowing it to infiltrate through the media from the CFA. This resulted in significant reductions in pollutants at varying depths in the vadose zone, with up to 99% reduction in contaminant concentrations. The resulting effluent met the established guidelines for six of the eight contaminants examined in this study. A financial viability assessment was conducted for two scenarios. In the shallow water table scenario, the benefit-cost ratio (BCR) was 0.55, indicating that the project was not financially feasible. However, in the alternate 'CFA groundwater use' scenario, the BCR was 1.14, demonstrating its financial viability. Furthermore, the unit reference value of the scenario was lower than that of desalination, suggesting that combining groundwater use with SWH via MAR is economically more attractive than seawater desalination.

Overall, this study provides evidence for the viability of retrofitting detention ponds for urban MAR in Cape Town from technical, social, and economic perspectives, considering the challenges and potential of the South African context. These outcomes were then used to develop a practical 'middle-out' approach for retrofitting detention ponds for urban MAR in Cape Town.

List of Abbreviations

AVFM	Area – Velocity Flow Meter
BCR	Benefit-Cost Ratio
CMIP	Coupled Model Inter-comparison Project
CoCT	City of Cape Town
CSAG	Climate Systems Analysis Group
DEM	Digital Elevation Model
DO	Dissolved Oxygen
DWA	Department of Water Affairs
DWAF	Department of Water Affairs and Forestry
DWS	Department of Water and Sanitation
E. Coli	*Escherichia coli*
EC	Electrical Conductivity
FV	Future value
GIS	Geographic Information System
GW	Groundwater
GWL	Groundwater Level
LID	Low Impact Development
LiDAR	Light Detection and Ranging
MAR	Managed Aquifer Recharge
MODFLOW	Modular Three-Dimensional Finite-Difference Ground-Water Flow
NPV	Net Present Value
PaWS	Pathways to Water Resilient South African Cities
PCSWMM	Personal Computer Stormwater Management Model
PV	Present Value
RCP	Representative Concentration Pathway
RTC	Real-Time Control
SAWS	South African Weather Services
StatsSA	Statistics South Africa
SuDS	Sustainable Drainage System
SWH	Stormwater Harvesting
SWMM	Stormwater Management Model
URV	Unit Reference Value
WSUD	Water Sensitive Urban Design

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1. Introduction

1.1 Contextual background

South African metropolitan areas, like many worldwide, have undergone rapid urbanisation due to rural-urban and regional migration (Hedden & Cilliers, 2014). Over the last 50 years, regions such as Mitchells Plain in the City of Cape Town (CoCT), previously covered by large stretches of dunes, have been transformed by urban development, resulting in an increase in impervious surfaces and a reduction in pervious areas that play an essential role in stormwater management (flood mitigation) by capturing and infiltrating stormwater runoff. Conventional engineering practices to mitigate flooding have led to the construction of various stormwater drainage structures, including stormwater pipes, canals, detention ponds, and retention ponds. Simultaneously, increasingly erratic rainfall patterns, possibly driven by climate change, have exacerbated the threat of physical blue water scarcity in Cape Town (Kummu *et al.*, 2016; Lake, 2017). Other major cities, such as San Francisco (USA), Sydney and Melbourne (Australia), and São Paulo (Brazil), have also experienced significant water scarcity in recent years (Tures, 2018). Consequently, various strategies have been proposed and implemented to mitigate water scarcity in major cities, including adopting alternative water supply sources and demand-management strategies (Armitage *et al.*, 2014).

The CoCT intends to expand its water supply by exploring alternative water sources, such as desalination, while increasing the use of existing sources, particularly groundwater, by tapping into the Atlantis Aquifer (AA), Cape Flats Aquifer (CFA), and Table Mountain Group Aquifer (TGA). The CoCT sourced 94% of its water supply from six primary surface reservoirs and 4% from groundwater in 2019 (CoCT, 2019). The CoCT seeks to increase its groundwater use with a groundwater expansion program that will increase groundwater use from 4% to 7% by 2040 (CoCT, 2019). However, increased groundwater abstraction could stress the resource through overuse, as natural groundwater recharge rates are typically lower than abstraction rates (Xu & Beekman, 2019; Cuthbert *et al.*, 2022). According to Parsons (2019), while aquifers in and around the CoCT do not presently face immediate threats or exhibit signs of overuse, they still require appropriate management, as they have finite capacities. Effective aquifer management entails controlling abstraction rates and managing and promoting aquifer recharge via managed aquifer recharge (MAR) (Dillon *et al.*, 2009). The CoCT recognises that MAR is critical for achieving sustainable groundwater use. Thus, its water strategy includes plans to increase groundwater recharge by expanding the use of treated effluent for infiltration into the Atlantis aquifer and borehole injection for the CFA (CoCT, 2019).

Research has indicated that stormwater harvesting can serve as a potential groundwater recharge source, both globally and in South Africa (Dillon *et al.*, 2009; Lim *et al.*, 2011; Payne *et al.*, 2015a; Fisher-Jeffes *et al.*, 2017). Stormwater harvesting (SWH) is defined by Fisher-Jeffes *et al.* (2017) as 'the collection and storage of runoff from an urban region and its subsequent use irrespective of location'. Okedi (2019) demonstrated the theoretical viability of SWH for the CoCT through a hydraulic model-based assessment of the Zeekoe catchment that indicated that MAR is the optimal storage mechanism for harvested stormwater. Okedi stated that some of the 230 + detention ponds of CoCT could theoretically be retrofitted to improve their infiltration capacity to facilitate the MAR.

Stormwater retrofitting is a well-established field in urban water management literature. However, the practical application of this technique to South African public infrastructure to facilitate urban MAR remains an underexplored research area. An improved understanding of this domain is essential for verifying the technical, social, and economic viability of retrofitting stormwater infrastructure in the CoCT to implement urban MAR and realise increased water security, as suggested by Okedi (2019).

1.2 Aim of the study

This study sought to develop a framework for retrofitting detention ponds in Cape Town to facilitate, *inter alia*, urban-managed aquifer recharge (MAR). To that end, the practicality and viability of stormwater retrofitting in the CoCT were investigated by establishing a pilot MAR scheme in an existing detention pond in Mitchells Plain on the Cape Flats.

The principal outcome of this study is a framework that provides a practical approach for retrofitting detention ponds in Cape Town to enable SWH through urban MAR. This study contributes to two Sustainable Development Goals (SDGs), specifically SDGs 6 and 11, which address clean water and sanitation and sustainable cities and communities, respectively.

1.3 Research Questions

The subsequent research enquiries were formulated to realise the aim of this study:

1. What are the principal requirements for facilitating stormwater harvesting through MAR?
2. What are the necessary technical and social aspects to consider when retrofitting detention ponds for urban MAR in Cape Town?
3. What are the water quality considerations for MAR, and how does Cape Town's stormwater quality compare with national guidelines and other countries?
4. How can the maximum MAR be achieved in retrofitted ponds in Cape Town, and what is the expected short- and long-term hydrological performance of such retrofitted ponds?
5. What range of water quality is expected in the saturated zone after stormwater infiltration and treatment in retrofitted ponds, and how does vadose zone depth affect treatment efficiency? What other benefits can be derived from retrofitted ponds in Cape Town?
6. What is the viability of retrofitting detention ponds for urban MAR in Cape Town?

1.4 Study limitations

- The study was limited to one case study located in one of the CoCT's 230+ detention ponds.
- The field data collected from the study detention pond does not necessarily represent the other detention ponds in the CoCT although the field conditions from this pond were similar to many other ponds in the vicinity suggesting that similar results are probable in other ponds
- A computational model was used to assess the extended performance of the pond and the effects of climate change. While the model was calibrated and validated, the results from this model only provide estimates of what might be expected.
- The economic viability of retrofitting detention ponds for MAR was evaluated based on the historical and market conditions prevailing at the time of the study and will require revision in the future.

1.5 Thesis outline

This thesis consists of twelve chapters, a list of references and various appendices. The remainder of this thesis is structured as follows:

Chapter 1 (this one) presents the contextual framework of the study, offering the broader context, research needs, and aims of the study. The main research questions addressed are also highlighted.

Chapter 2 provides a systematic literature review that examines the existing knowledge and research to establish a foundation for subsequent analyses.

Chapter 3 presents the research methods adopted to meet the study's aim.

Chapter 4 describes the study area, socio-economic context, and site selection method used to select the case study.

Chapter 5 evaluates Cape Town's historical stormwater quality data, examines its correlation with land-use patterns, and compares it with other countries.

Chapter 6 evaluates the technical and social viability of retrofitting a detention pond in Cape Town, using a field-scale case study.

Chapter 7 presents a modified community engagement tool used to assess the level of community participation in this study.

Chapter 8 presents the development and application of a hydrological model used to evaluate the theoretical recharge volumes and the short- and long-term effects of a retrofitted pond.

Chapter 9 presents the laboratory experiments conducted to evaluate the treatment performance of retrofitted detention ponds, and the results and implications thereof.

Chapter 10 assesses and highlights the additional benefits of retrofitted detention ponds within a sustainable drainage systems context.

Chapter 11 discusses the economic viability of retrofitted ponds based on a simple analysis of the benefits and costs of retrofitted detention ponds for MAR.

Chapter 12 is the closing chapter summarising the thesis' contribution to knowledge, conclusions, and recommendations for further research.

2. Literature Review

This section comprises a literature review focusing on Managed Aquifer Recharge (MAR) and its application in urban stormwater harvesting. The section begins by providing an examination of global, regional, and local literature on MAR, highlighting key concepts and challenges associated with its implementation. The concept and relevance of Sustainable Drainage Systems (SuDS) to MAR are explored, specifically considering their practical application within the South African context. In addition, the review examines stormwater retrofitting, investigating guidelines, and considerations for effective implementation. The treatment considerations for stormwater retrofitting for MAR are then discussed. Finally, an analysis of potential obstacles and opportunities within the field of MAR is presented, offering insights into the current state of knowledge, and highlighting the existing gaps in the literature.

2.1 Managed Aquifer Recharge overview

Managed Aquifer Recharge (MAR) is 'the purposeful recharge of water to aquifers for subsequent recovery or environmental benefit' (NRMMC-EPHC-AHMC, 2009). Aquifers are underground reservoirs or dynamic flowing systems of water contained in geological strata which are naturally replenished either by infiltration from water bodies or rainwater percolating through the subsurface into the aquifer (Dillon *et al.*, 2009, 2012; NRMMC-EPHC-AHMC, 2009).

The global volume of stored groundwater has been gradually declining on account of increased groundwater use spurred on by the development of improved drilling techniques, equipment and electric pumps (Jakeman *et al.*, 2016). Additionally, the global rate of groundwater depletion has been increasing (Konikow, 2011). Groundwater depletion has adverse effects such as higher pumping costs, deterioration of groundwater dependant ecosystems, deterioration of the groundwater quality, and depletion of surface water sources. It results in uncertainty in water supply for communities reliant on groundwater and reduced crop yields and income for groundwater reliant farmers. These threats make the management of groundwater crucial.

Dillon *et al.* (2012), proposed that a more appropriate way to manage groundwater is though flexible approaches to supply and demand management. MAR is one such approach to protect, extend, and augment groundwater supplies. MAR supports the active management of groundwater resources at a local and catchment-scale and involves using water sources such as rivers, stormwater, or recycled wastewater to recharge (refill) an aquifer under controlled conditions. This stored volume of water can then be utilised later (Dillon, 2009; Dillon *et al.*, 2009; NRMMC-EPHC-AHMC, 2009).

One of the benefits of MAR is that groundwater storage is often more efficient than surface storage as aquifer storage minimises evaporation loses – particularly in hot climates which have high evaporation rates. Aquifer storage also saves land, which can be used for other purposes, and the water is relatively safe from contamination (Bugan *et al.*, 2016). While the global MAR implementation rate has grown at a rate of 5% a year, the uptake falls short of matching the increasing groundwater extraction rates. As of 2019, the global MAR volume was estimated to be only 1% of the worldwide groundwater extraction (Dillon *et al.*, 2019).

2.1.1 MAR methods

Several MAR methods exist that can be used in different contexts. Some of the techniques developed were not explicitly designed to address the challenge of MAR such as injection wells, developed ~2000 years ago by the ancient Persian civilisation initially meant to deal with urban runoff (Burian & Edwards, 2002). Modern day MAR techniques have improved on these past methods (Dillon *et al.*, 2009; NRMMC-EPHC-AHMC, 2009). MAR techniques are case-specific and should only be implemented after careful consideration of the desired water quality, required infiltration rate, residence time, type of aquifer, land availability and cost of construction and abstraction (Dillon *et al.*, 2009; NRMMC-EPHC-AHMC, 2009).

MAR methods are broadly classified as: spreading methods – where water percolates naturally into the water table over a wide area; vadose zone dry-well injection methods – where water is discharged into dry wells which are in the vadose zone and filter into the aquifer; and direct aquifer recharge – where water is discharged directly into the aquifer. Figure 2-1 presents some examples of these methods as discussed by Dillon (2005) which are summarised as:

Aquifer storage and recovery (ASR) – encompasses the injection of water into a well for storage, followed by its retrieval from the same well.

Aquifer storage, transport, and recovery (ASTR) – involves introducing water into a well and subsequently recovering it from another well. This provides additional treatment by prolonging its residence time in the aquifer.

Roof runoff infiltration – involves directing roof runoff to a well, or waterproof compartment containing sand, facilitating percolation into the underlying water table. The stored water can be later retrieved by pumping from a well.

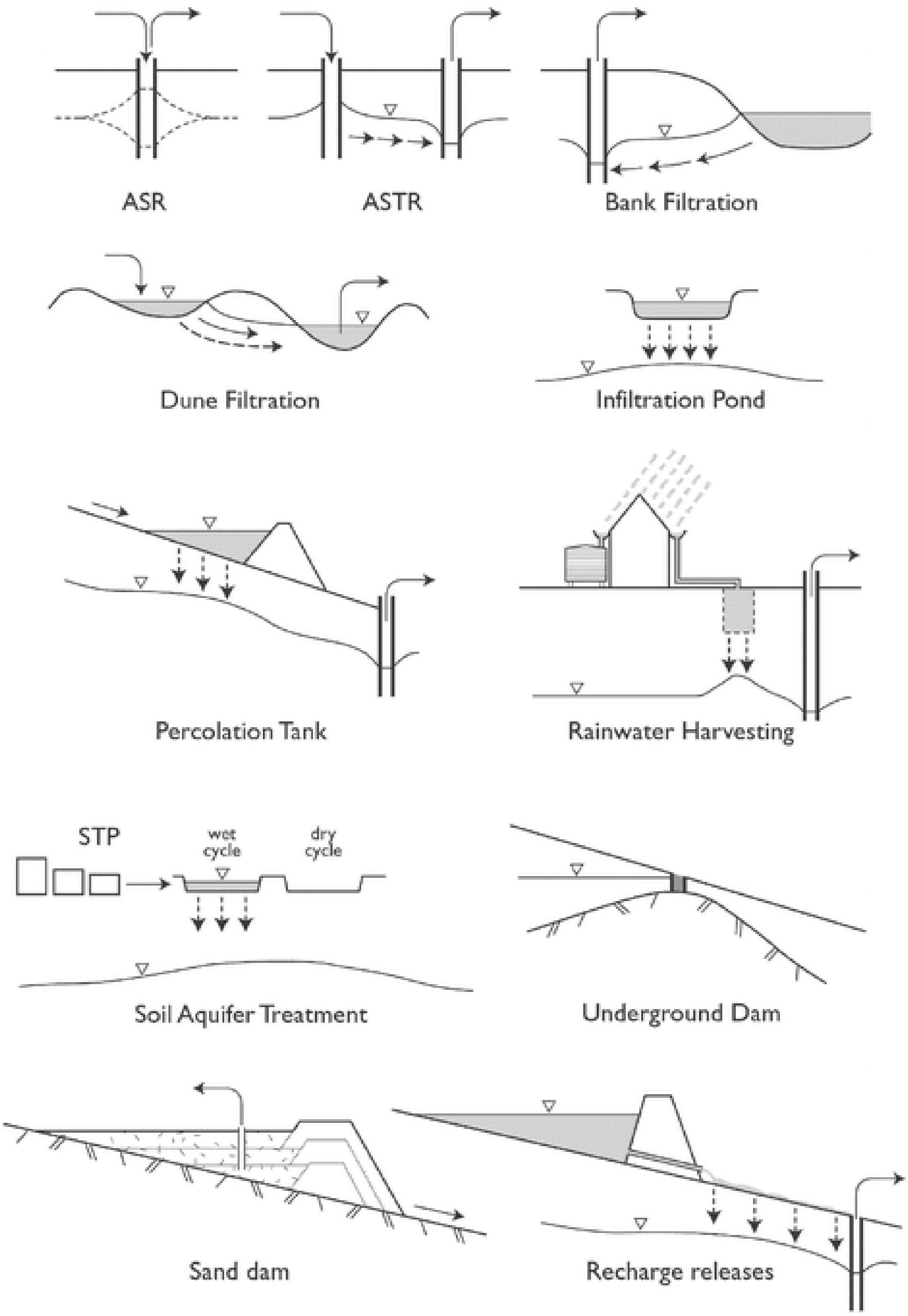

Figure 2-1: MAR methods (Dillon, 2005)

Dune filtration – water is infiltrated through ponds constructed in dunes. Water is subsequently drawn from lower-elevation wells or ponds to enhance quality and achieve equilibrium between supply and demand.

Infiltration ponds – are basins constructed over an unconfined aquifer. Surface water is diverted to the ponds and allowed to infiltrate to the underlying unconfined aquifer, generally through an unsaturated zone.

Soil Aquifer Treatment (SAT) – treated sewage effluent is periodically infiltrated via infiltration ponds, aiding in pathogen and nutrient removal in the vadose zone. Further residence time in the aquifer allows for further treatment, and the water is abstracted from recovery wells.

Various water sources can be used for MAR, such as potable water, treated sewage effluent, rainwater, river water, and stormwater. In an urban setting, these sources can often be found in multiple locations and incorporating them for MAR can help establish decentralized water systems, thereby contributing to a city's water resilience (Page *et al.*, 2018). However, the water quality may vary from poor to excellent due to human activities.

2.1.2 Establishing a MAR project

The successful establishment of a MAR project requires appropriate planning (Central Groundwater Board, 2007). This planning involves technical and regulatory considerations which can be condensed into five essential elements:

- Demand for the recovered water – the purposes of the recovered water need to be established, and a demand for the reclaimed water needs to be present to ensure the scheme addresses an existing problem.
- A source of water for recharge to meet the demand – there is a need to ascertain that there is enough water for recharge and that the scheme will not have a detrimental/adverse effect on other users of the recharge source and the aquifer.
- A suitable aquifer to store and recover water – the scheme must be in an aquifer with a satisfactory rate of recharge (usually guideline dependent) and must have enough storage to store the volume of water to be recovered.
- The availability of land to harvest and treat water – there should be adequate land to treat and recharge the water without causing flooding in the surrounding area.
- Capability to effectively manage a project – hydrological, geotechnical, and financial knowledge along with familiarity with water treatment, water

quality monitoring and management, water sensitive urban design and hydraulic modelling is required to manage a MAR project.

These requirements have been collated into a checklist by Dillon *et al.* (2009) (Figure 2-2).

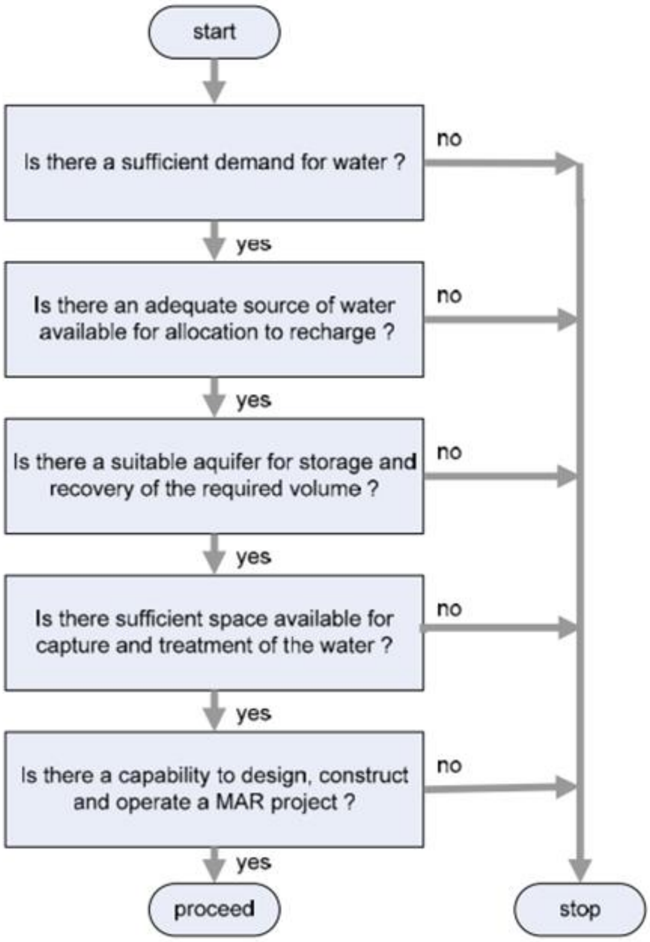

Figure 2-2: Issues to consider prior to undertaking a MAR project
(Dillon *et al.*, 2009)

2.1.2.1 Hydrogeology

Hydrogeology relates to the geology of an area in relation to the distribution and movement of groundwater. Several factors influence the occurrence, distribution, and movement of groundwater such as rainfall, soil type (which influences the porosity and permeability), the existing geology (lithology), topography, and the existence of surface water bodies. It is imperative to understand the underlying geology of an area as it influences the type of MAR method to be employed (Rahman *et al*., 2013; Parimalarenganayaki & Elango, 2015).

The Indian Central Groundwater Board (2007) provides a summary of the recommended hydrogeological studies to be conducted to establish the knowledge required to plan for a MAR project adequately (Table 2-1).

Table 2-1: Required hydrogeological studies for planning a MAR project (Central Groundwater Board, 2007)

Type of Study	Anticipated inputs
Remote Sensing Studies	• Variability in infiltration properties of different litho-units. • Drainage attributes and lineament density. • Delineation of distinct geomorphic units.
Hydrometeorological Studies	• Rainfall depth, duration, intensity on both daily and hourly scales, and the variability of precipitation.
Hydrological Studies	• Source water availability, infiltration characteristics of major soil types and various land use categories.
Geophysical Studies	• Depth of weathering in solid rock formations. • Attributes and dimensions of porous layers in sedimentary landscapes. • Configuration of aquifer layers, spatial fluctuations in hydraulic conductivity and vertical variations in hydraulic conductivity. • Geological disruptions like fault zones.
Hydrogeological Studies	• Regional hydrogeology and aquifer characteristics. • The behaviour of groundwater levels • Groundwater potential. • Groundwater flow pattern and hydraulic connection between groundwater and surface water bodies.
Hydro chemical Studies	• Water source quality considerations. • Spatial and temporal fluctuations in groundwater management.

MAR in unconfined aquifers presents more recharge opportunities compared to confined aquifers – where the groundwater is trapped in-between two impermeable strata (Rahman *et al*., 2013). This is because the water can be infiltrated into the underlying aquifer through spreading methods, vadose zone wells methods and direct injection wells, while in confined aquifers only direct aquifer recharge (such as injection wells) can be used. However, in unconfined aquifers, the soil type has a direct influence on the infiltration rate and storage potential of the aquifer. The groundwater level also plays a crucial part in the MAR, especially in unconfined aquifers as MAR is not feasible in places with shallow water tables.

Modelling studies have also shown that when storm runoff is utilised for MAR, projects may face operational challenges from seasonally fluctuating groundwater levels that affect recharge in aquifers with shallow water tables (Zheng *et al.*, 2018; Gxokwe *et al.*, 2020). This can be addressed by seasonal pumping where water is pumped out before the rainy season to lower the water table and create storage space for the water received during the rainy season (Gxokwe *et al.*, 2020).

2.1.2.2 Water source

The water source used for MAR is an essential consideration as it affects the quality, quantity, and the operational requirements of the project. The largest sources of water globally for MAR are natural streams and rivers – although the water quality from these sources is susceptible to anthropogenic influences. Aquifer recharge using natural water sources can be found in countries such as India and the USA (Rahman *et al.*, 2013; Parimalarenganayaki & Elango, 2015; Scanlon *et al.*, 2016)

Stormwater (urban and rural) has also been used in various MAR schemes that harvest and store storm runoff in aquifers. Urbanisation has resulted in increased stormwater owing to reduced infiltration capacity of urban landforms. The recharge method used depends on the quality of the runoff, which can contain several pollutants such as inorganic material (heavy metals, nutrients, and chemical waste etc.), litter, hydrocarbons, and bacteria (Aryal *et al.*, 2010; Schmidt *et al.*, 2011). The water is collected from the catchment area and conveyed to the recharge site which can be located within or outside the catchment area. Stormwater for MAR has been used countries like Australia, Brazil, China, Netherlands, the USA, and South Africa where it is infiltrated through different methods including spreading techniques – e.g., infiltration basins, soil aquifer treatment – and injection wells (Dillon, 2009; Sprenger *et al.*, 2017; da Costa *et al.*, 2019; Dillon *et al.*, 2019; Alam *et al.*, 2020; Shubo *et al.*, 2020).

A substantial percentage (60-80%) of the potable water supplied in urban areas is returned as wastewater. The wastewater is often treated, and the treated sewage effluent can then be used in MAR schemes. The treated effluent is sometimes of poor quality as extensive treatment is expensive. Consequently, the use of treated effluent for MAR is often limited to infiltration ponds as they allow for further treatment of the water (by trapping pollutants) as the water percolates to the water table. The Atlantis Water Resource Management Scheme in South Africa and the Orange County Water District in California, USA are examples of MAR schemes that use treated wastewater effluent (Tredoux *et al.*, 2009; Luxem, 2017). The aquifers provide ongoing passive treatment that can substantially improve water quality if the residence period is sufficiently long (NRMMC-EPHC-AHMC, 2009).

Several water scarce nations that experience high evaporation rates have embarked on large scale seawater desalination, storing the excess water via MAR. This practice is common in the Middle East with one such example being found in Abu Dhabi, UAE, which has a 50 Mm^3 groundwater reserve (Dillon *et al.*, 2019).

2.1.2.3 Health and environmental risks (water quality)

While MAR focuses on both the quantity and quality of the augmented groundwater, in some countries such as India, recharge schemes often ignore the water quality component. Such schemes should be termed artificial recharge and not MAR (Dillon *et al.*, 2019). Even when MAR is undertaken to protect groundwater resources from contamination due to seawater intrusion, it is often done so that the water can subsequently be reused (Luxem, 2017).

The guiding measure as to how the augmented groundwater is used is the quality of the water. The associated health risks from both chemical constituents and pathogenic microbes should be considered if the water is to be used for indirect potable use such as industrial and agricultural applications (Grobicki & Cohen, 1998). Stringent standards are required if the water or products produced will be ingested by humans. The specific acceptable water quality is dependent on the regulating authority, which is often guided by the World Health Organisation (WHO) regulations.

In South Africa, the mandate to provide the standard norms and water quality standards fall upon the Minister responsible for water and sanitation according to the Water Services Act of 1997 (Republic of South Africa, 1997). The agency that carries out the development and review of these standards is the Department of Water and Sanitation (DWS). The existing South African MAR guidelines do not explicitly mention the target water quality values for water used for MAR (Department of Water Affairs (DWA), 2010). MAR practitioners cannot rely on values specific to raw water quality as these values are made on the assumption that the water will undergo the typical water treatment processes. The absence of target water quality values for MAR is not exclusive to South Africa as only two pre-2022 guidelines have been developed that explicitly address the minimum quality of water to be used for aquifer recharge (Page *et al.*, 2016; Dillon *et al.*, 2019). These are the ANZECC-ARMCANZ (2000) and NRMMC-EPHC-AHMC (2009) water quality guidelines developed for Australia and New Zealand respectively. These risk-based guidelines recommend a residence time of more than six months to enable water quality improvement through natural geochemical processes. The guidelines also provide treatment requirements to address water quality hazards associated with MAR such as pathogens, nutrients, inorganic chemicals, organic chemicals, salinity, and turbidity (NRMMC-EPHC-AHMC, 2009).

The DWS does, however, require that the quality of water the aquifer receives should be of a higher quality than the aquifer's initial water quality (Department of Water Affairs, 2007). The requirement falls short of a specific target water quality and can encourage an accumulation of contaminants in the aquifer if the recharge is frequent and the water quality is poor. Therefore, in instances where non-potable water of poor quality is to be used for aquifer recharge, there should be a provision for pre-treatment before infiltration or injection.

2.1.2.4 MAR monitoring, operation, and maintenance

Existing guidelines (e.g., NRMMC-EPHC-AHMC, 2009; CoCT, 2018) require some form of water quality monitoring for alternative water systems such as MAR sites, as a measure of quality assurance. The need to monitor MAR sites is reflected in a study by Doza *et al*. (2020) which monitored the water quality for a MAR scheme in Bangladesh where faecal contamination was a challenge. Through their monitoring and assessments, they were able to show that, while the MAR scheme received water with high faecal counts in the monsoon season, the sand filter (the MAR method) was able to trap the bacteria and facilitate microbial decomposition which improved the water quality.

The hydraulic conductivity of the scheme should also be monitored to ensure that the scheme remains operational as MAR sites can become inoperative due to a lack of maintenance (Doza *et al*., 2020). Alcazar *et al*., (2008) describe a method to monitor and test the hydraulic conductivity of the soil media in an infiltration basin using the standard single-ring infiltration test (Figure 2-3) as per ASTM D5126 (ASTM, 2016). The single-ring infiltration test is a constant head test that uses a small metal or plastic ring that is driven 50 mm into the ground. Water is poured into the ring to a height of 50 or 150 mm, depending on the soil conditions. The head is kept constant during the whole experiment by pouring water at intervals when the volume of water decreases. The intervals typically vary from 30 seconds to five minutes depending on the soil type. The infiltration test is stopped when the water level in the ring remains constant for at least 30 minutes. A similar method was also used by Hutchinson *et al*. (2013), to monitor the infiltration rate of several MAR schemes in Orange County in the USA. Monitoring the hydraulic conductivity of the MAR scheme can also assist in the minimising the potential for clogging (Martin, 2013).

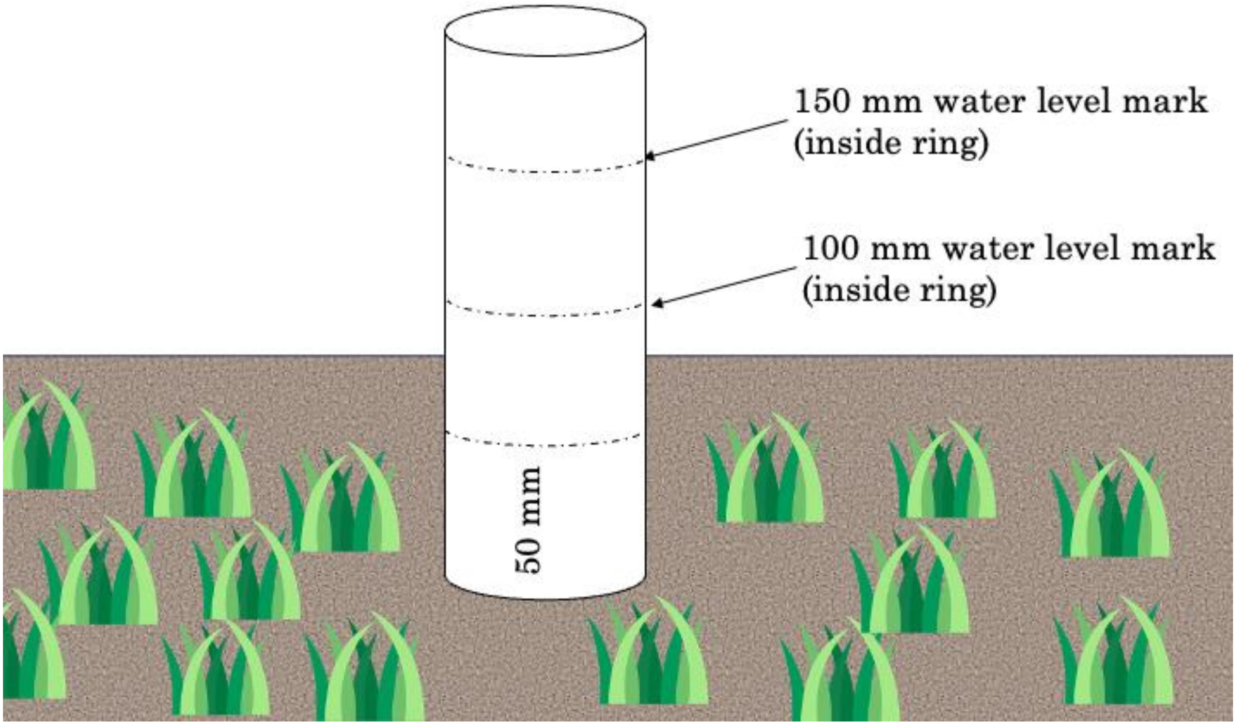

Figure 2-3: Single-ring infiltrometer
(Alcazar *et al.*, 2008)

Clogging is defined as the reduction of permeability in porous media. This reduction in permeability is a major technical challenge in MAR schemes as the objective is to infiltrate as much water as possible and clogging limits the volume groundwater infiltrated over a period and thus the efficiency of the project. Clogging occurs in both direct aquifer recharge and spreading methods.

Clogging may be categorised into four types: chemical (precipitation of metals such as aluminium or iron), physical (migration of fines, suspended solids), mechanical (gas binding/trapped air) and biological (growth of iron-reducing bacteria, algae growth) (Martin, 2013). A summary of these processes is presented in Table 2-2.

The hydraulic conductivity – an indicator of clogging – in infiltration basins decreases following construction owing to surface clogging and compaction. It tends to recover gradually in vegetated basins allowed to periodically dry out due to the creation of micropores by plant roots in the basin, with long-rooted vegetation performing better in this regard (Hatt *et al.*, 2007, 2008; Alcazar *et al.*, 2008; Benamar, 2013).

Table 2-2: Summary of clogging types and processes
(Martin, 2013)

Clogging type	Clogging Process
Chemical	• Geochemical reactions that result in the precipitation of minerals, e.g., iron, aluminium, or calcium carbonate growth. • Aquifer matrix dissolution (can also work to increase hydraulic conductivity). • Ion exchange and adsorption. • Oxygen reduction. • Formation of insoluble scales. • Formation dissolution.
Physical clogging	• Accumulation/injection of organic and inorganic suspended solids. • Velocity induced damage, e.g., migration of interstitial fines such as iolite or smectite. • Clay swelling (e.g., montmorillonite) • Clay deflocculating • Invasion of drilling fluids (emulsifiers) deep into the formation. • Temperature
Mechanical	• Entrained air/gas binding (includes nitrogen &/methane from the microbiological activity). • Hydraulic loading causing formation failure, aquitard failure or failure of the casing around joints or seals
Biological	• Algae growth and accumulation of biological flocs. • Microbiological production of polysaccharides. • Bacterial entrainment and growth.

Clogging in MAR is more comparable to clogging in soils rather than the granular media filters used in water treatment – e.g., sand filters – where the primary clogging mechanism is the mobilisation of suspended solids. Experiments have shown that in soils, clogging is associated with higher fluid velocities. As a result, direct recharge methods are more susceptible to clogging than spreading methods due to the fluid velocity being higher in the former (Martin, 2013).

Clogging when spreading methods have been used can be addressed by periodically scraping off the top layer of the infiltration basin as done at the Atlantis Water Resource Management Scheme (Dahlke *et al.*, 2018; Tredoux *et al.*, 2009). Pre-treatment devices such as forebays and check dams – small dams built across a detention pond, swale, or channel to minimise flow velocity (Figure 2-4) – are used to reduce sediment load and thus the clogging rate (Hutchinson *et al.*, 2013; Winston *et al.*, 2019). Extended drying periods have also been shown to result in increased infiltration capacity in biofilters (Hatt *et al.*, 2007).

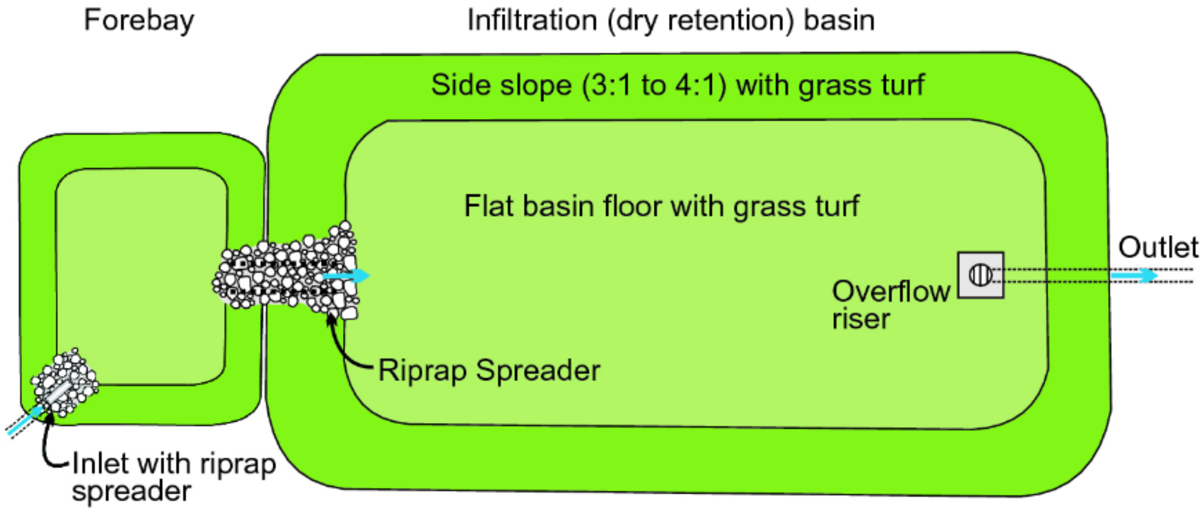

Figure 2-4: Top: Diagrammatic plan of an inflitration pond with forebay; Bottom: Rock check dams across a drainage ditch

(Winston *et al.*, 2019)

2.1.2.5 Economics of managed aquifer recharge

Investment for infrastructure projects must be justified and be viable from either a purely economic or socio-economic lens (Maliva, 2014). Megdal & Dillon (2015) curated a collection of papers that emphasise the importance of careful economic and financial analysis in MAR projects. These include case studies that address the methods of evaluating the economic benefits of MAR projects and demonstrate that, like all projects, a careful analysis of the costs and benefits of a MAR project is required for the successful implementation of a MAR project.

A Benefit-Cost Analysis (BCA) is often conducted to evaluate the economic viability of MAR projects. This involves quantifying the benefits and costs of the project in monetary terms. MAR projects should ideally cost less than the cost of alternative projects that provide the same benefit (Vanderzalm *et al.*, 2015). An example of such an evaluation is given by Dillon *et al.* (2009), where the cost of a desalination plant was compared to MAR by stormwater ASR. In that study, the MAR proved to have lower costs per kL – desalination is more energy intensive compared to MAR which drives up the cost of desalination. However, the yield of the MAR was significantly lower than the desalination plant yield.

The economic evaluation of MAR projects is often hampered by the difficulty of accurately quantifying the recharge volume and value of the water (Ross & Hasnain, 2018). Furthermore, in some countries, the data that relates to MAR is proprietary and hard to obtain, which makes economic evaluations difficult (Levy & Christian-Smith, 2011; Perrone & Rohde, 2016). Thus, alternative methods of evaluating and quantifying the value of water which are applicable to MAR projects are used. These methods include:

- Alternative costing – the cost of the next alternative
- Marginal value – the cost to obtain additional supplies.
- Markets pricing – the current price at which a good or service can be purchased or sold.
- Environmental value of in-situ groundwater – pertains to the advantages of having more groundwater present in an aquifer, as opposed to the benefits derived from withdrawing and using groundwater (Maliva, 2014; Vanderzalm *et al.*, 2015).

2.1.3 MAR via infiltration ponds: selected case studies

There are many examples of MAR case studies, e.g., France (Barraud *et al.*, 1999; Jakeman *et al.*, 2016), Australia (Dillon *et al.*, 2009; CRC, 2015; Payne *et al.*, 2015), South Africa (Tredoux *et al.*, 2009; Department of Water Affairs, 2010; Bugan *et al.*, 2016) and India (Central Groundwater Board, 2000). This review considers five relevant case studies that relate to MAR by spreading methods using stormwater.

2.1.3.1 Atlantis Water Resource Management Scheme, Cape Town, RSA

Recharge Type: Spreading methods (infiltration basins)
MAR Influent Source: Stormwater & treated wastewater

Water resource augmentation measures are not new concepts in Southern Africa as the region is prone to droughts and water scarcity. Drought prone Namibia's sand storage dams have acted as aquifers since the mid-20th century (Matengu *et al.*, 2019). However, the adoption of large-scale MAR schemes – particularly using spreading methods – in Southern Africa has been low with only four out of the 17 known MAR sites in South Africa using spreading methods (Ebrahim *et al.*, 2020) – the best-known being the Atlantis Water Resource Management Scheme (AWRMS).

Atlantis is a settlement situated approximately 40 km north of Cape Town's central business district. It is located along the semi-arid coast of South Africa, with a Mediterranean climate (Bugan *et al.*, 2016) and was developed as a growth point, with tax incentives for industry to their establishment there. Its water supply originally came from groundwater and springs that have been partly replenished by infiltrated storm runoff and treated wastewater since 1979 (Murray, 2016).

Atlantis' water was initially intended to be supplied from a nearby spring and several boreholes surrounding the spring (Tredoux *et al.*, 2009, 2011). However, the natural yield of the aquifer was found to be insufficient to sustain the water supply for the settlement (Department of Water Affairs, 2010). The long-term plan was to use surface water via a 70 km long pipeline; however, this exercise proved too costly in the early phases of the development and until the year 2000, the town wholly depended on the sustainable use of its groundwater. The yield of the aquifer was successfully increased through MAR.

The widespread practice in South Africa's coastal towns has been to discharge marine and urban stormwater runoff to the ocean (CoCT, 2018). However, for Atlantis, this practice was found to be expensive because of the prohibitive cost of the pipeline construction and long-term monitoring of the marine outfall (Bugan *et al.*, 2016). Infiltration into the underlying aquifer was then implemented as an alternative which also served as an aquifer recharge measure. The sandy surface of the area permits natural groundwater recharge of 15 to 30% of the mean annual precipitation, and this hydrogeology enabled MAR to be implemented around the town. The early practice of treating all the town's wastewater in a single treatment plant was discontinued in favour of separate treatment facilities for the industrial and domestic wastewater, due to water quality considerations. Currently (2023), the effluent from the residential areas is collected and treated via full secondary treatment processes and stored temporarily in maturation ponds. The DWS oversees the effluent quality. The storm runoff is collected via the twelve detention (1 to 4, 8 and 11) and retention basins (5, 6, 9 and 10). The storm runoff is then diverted for temporary storage in the Basin 6 where the low salinity stormwater is blended with the treated domestic wastewater effluent (Tredoux *et al.*, 2009) before being discharged into the main recharge Basins 7 and 12 (Figure 2 5).

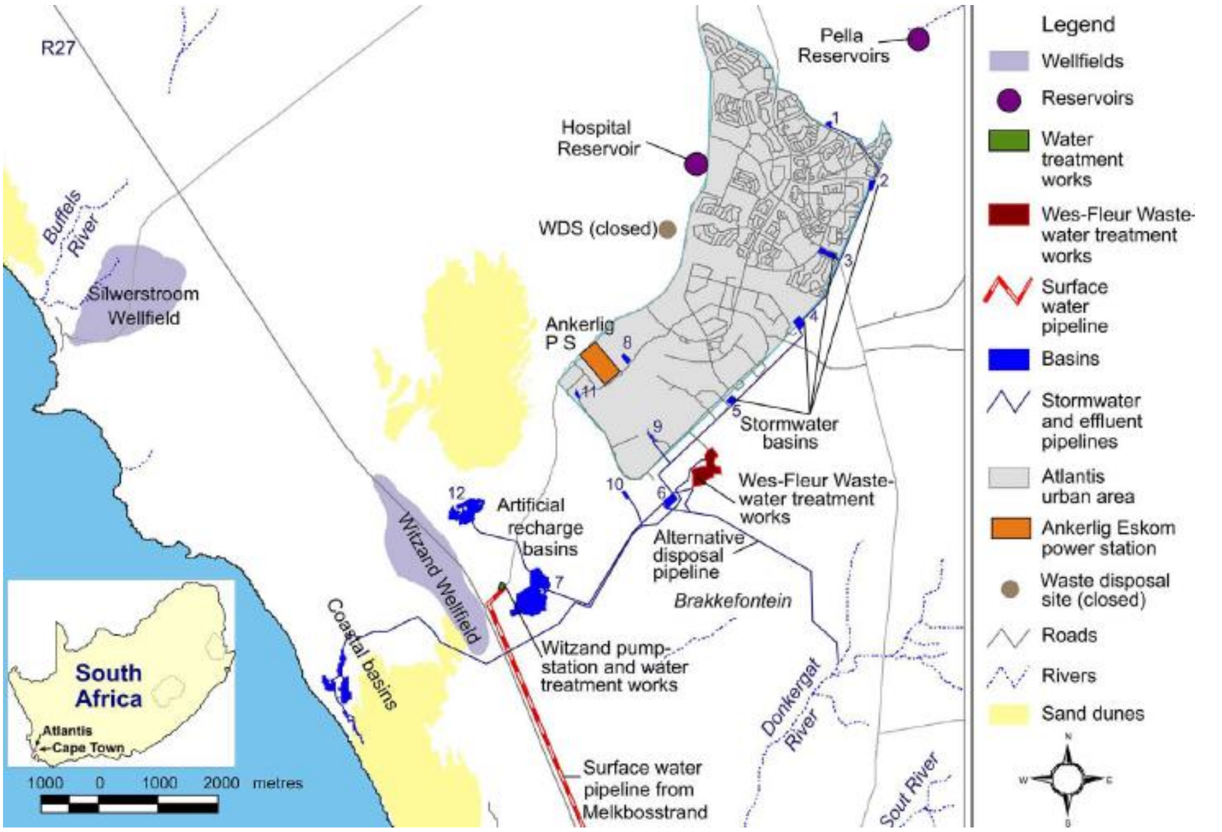

Figure 2-5: Location of Atlantis and its water supply system
(Bugan *et al.,* 2016)

Basin 12 is designed to receive peak storm runoff while Basin 7 is intended to collect the low flow blended mix of treated effluent and retained stormwater. A simple splitter box controls the flow into the two basins. It takes between 18 months and two years for the water to seep underground and reach the Witzand Wellfield (Department of Water Affairs, 2010). The industrial wastewater is treated at the industrial wastewater treatment plant, and the effluent is then discharged into coastal recharge basins where it seeps into the ground to serve as a mitigation measure against seawater intrusion by forming an underground hydraulic barrier by flattening the hydraulic grade line.

A simplified schematic of the treatment processes is presented in Figure 2-6. The numbered arrows represent the various water quality sampling points that were used to monitor the water quality at the scheme. MAR enables the AWRMS to augment the approximately 2.7 Mm^3 of groundwater harvested from Atlantis per year. The groundwater is abstracted via 58 wells at the Witzand and Silwerstroom wellfields (Tredoux *et al.*, 2009; Bugan *et al.*, 2016).

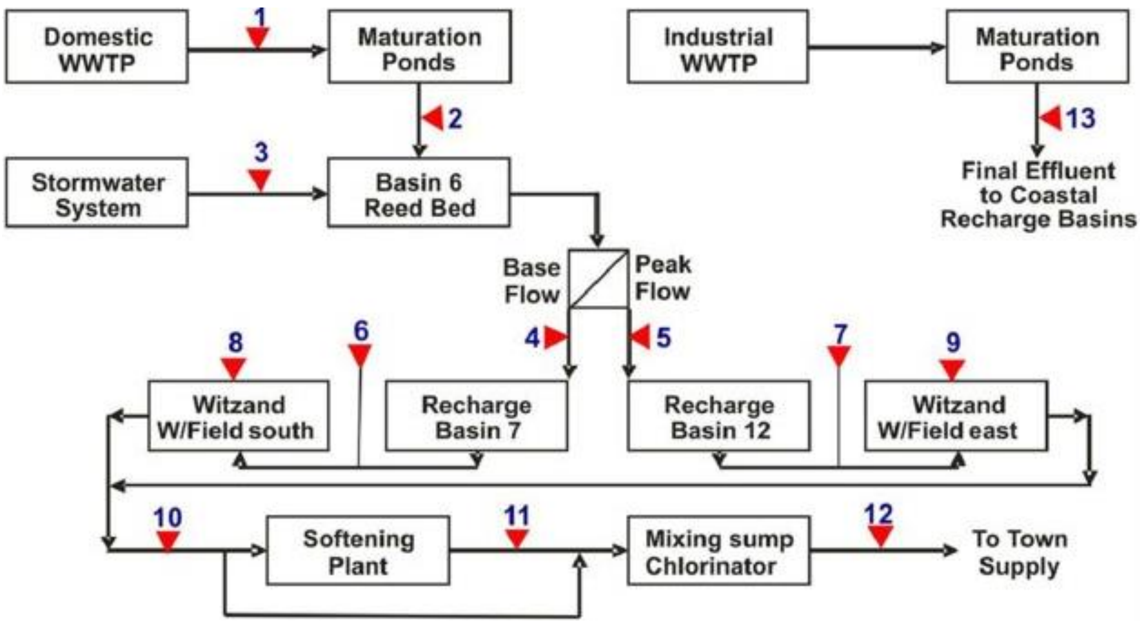

Figure 2-6: Schematic of the AWRMS showing the monitoring points (Bugan *et al.,* 2016)

Basin clogging is an operational challenge at the AWRMS as water is infiltrated into the aquifer via permanently flooded infiltration basins. The basin clogging is a result of the gradual build-up of fine sediments and organic material at the bottom of the basins which retards the infiltration rate. As a mitigation measure, one of the upstream basins (Basin 6) was converted to a wet basin with reeds which significantly reduced the volume of silt in the infiltration basin. Basin 6 thus acts as a forebay. The silt is removed annually by drying out the basin then manually and mechanically dredging the basin. Basin clogging is remediated by scraping off the top layer in the clogged basins. It is recommended that the scraping of the top layer in Basins 7 and 12 be done every fifteen years (Department of Water Affairs, 2010). The clogging in Basin 12 is further remediated by allowing it to regularly dry out, resulting in the cracking of deposited sediment which gets blown away. Litter build-up at the inlets and in the basins is another operational challenge. Screens have been installed to trap litter, but some litter is still found in the basins and often clogs the inlets and overflow outlets (Figure 2-7), which then require regular maintenance (Department of Water Affairs, 2010).

Figure 2-7: Litter build-up at one of the stormwater inlets in the AWRMS

(Department of Water Affairs, 2010)

The AWRMS is a cost-effective water supply option that demonstrates how MAR can be utilised in South Africa provided there is stringent management of the operations. It also demonstrates some of the challenges a MAR scheme could face in a South African context such as litter and clogging of the infiltration basin.

2.1.3.2 Harkins Slough MAR Project, California, USA

Recharge Type: Spreading methods (infiltration basins)

MAR Influent Source: Stormwater

The Harkins slough MAR (SH-MAR) project is in the Pajaro Valley in central coastal California (USA). The region receives an annual rainfall depth of ~500 mm and has perennial streams and wetlands that often flood during the rainy season. The project is operated by the Pajaro Valley Water Management Agency (PVWMA) which diverts surface runoff amounting to 2.5 Mm^3 annually from the surrounding catchment to the MAR site. Water is banked during winter – when the area receives rainfall – and recovered in summer (Levy & Christian-Smith, 2011). The surface runoff is passed through a sand filter which traps sediments and provides pre-treatment (Racz *et al.*, 2012). The water is then pumped up a 1.5 km pipeline to an infiltration pond (Figure 2-8). The soil profile beneath the pond (up to 2.5 m) mainly comprises of fine sand, silt, and clay. The maximum pond depth is ~6 m.

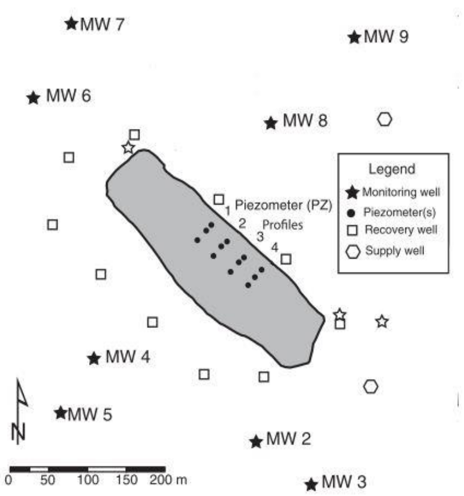

Figure 2-8: Harkins Slough Recharge Project – Left: Site Map; Right: Infiltration Pond

(Schmidt *et al.*, 2011; PVWMA, 2019)

The infiltration in the pond creates a local water table mound (rise in the water table due to groundwater recharge) and temporary storage in a perched aquifer that overlies a clay layer which is ~ 30 m below the pond (Schmidt *et al.*, 2011).

Runoff from the pond slopes is less than 1% of the inflow volume from the stormwater diversion. Groundwater from the shallow local aquifer is recovered via wells which are adjacent to the pond, blended with the regional groundwater, and distributed to agricultural lands (Schmidt *et al.*, 2011; Racz *et al.*, 2012).

Monitoring of the pond operation began in 2001 and entailed setting up soil sampling points, piezometers, hourly inflow flowrate measurement and infiltration tests within the pond (Racz *et al.*, 2012). The system's water quality (*viz.* nutrients, metals, biological and turbidity), point infiltration rates and infiltrated volumes (calculated by a mass balance) are monitored regularly. Groundwater samples from the underlying aquifer are collected from monitoring wells around the pond.

Through monitoring, it was found that the hydraulic conductivity of the ponds was declining with time because of clogging (biological & physical). The sedimentation is highest around the inlets, which was found to have sediments deposits thicker than 50 mm (Fisher *et al.*, 2010; Racz *et al.*, 2012). The clogging in the pond is remediated by first dying up the pond (through pumping out the water) and then mechanically scraping off the 'cake'. The dying off and cleaning of the pond is done every year at the end of the MAR season (Schmidt *et al.*, 2011). Scrapping off the top 100 mm results in an improved hydraulic conductivity (Fisher *et al.*, 2010).

2.1.3.3 Margaret River, Western Australia

Recharge Type: Spreading methods (biofilters)
MAR Influent Source: Stormwater

A large-scale multi-layer biofilter was constructed in 2008 in Margaret River, a town in the Southwest of Western Australia some 277 km south of Perth. The biofilter covers 1090 m^2 which is 2% of the catchment, as suggested by Alcazar *et al.* (2008) for satisfactory stormwater treatment performance. The biofilter receives and treats stormwater runoff from the Margaret River Central Business District watershed with the lower flows passed through a pollutant and sediment trap before entry into the biofilter whilst larger flows are allowed to bypass the trap. The stormwater then infiltrates down the biofilter profile and into the biofilter underdrains and is channelled towards the river via a bypass channel. The filtration process improves the runoff quality before it enters the river. The design of the Margaret River biofiltration system follows the Australian guidelines (CRC, 2015; Payne *et al.*, 2005). Biofilters are known to reduce the inflow of Total Suspended Solids and Total Phosphorus (TP) in stormwater by 80 – 90% (Hsieh & Davis, 2005). While the Margaret River biofilter is not a case of MAR, it introduces the concept of biofilters as a way of improving stormwater quality, which can then be infiltrated.

2.1.3.4 Mandurah, Western Australia

Recharge Type: Spreading methods (Infiltration basins)
MAR Influent Source: Reclaimed Wastewater

Mandurah is a coastal Western Australian city approximately 80 km south of the Perth Central Business District. It has a Mediterranean climate and receives a mean annual rainfall of 850 mm.

A public open park was to be established in the city precinct, which required irrigation (City of Mandurah, 2015). After evaluation of possible sources of irrigation water, the city officials decided to utilise groundwater alongside MAR to augment the groundwater supply in the area (City of Mandurah, 2014; Vanderzalm *et al.*, 2015). Groundwater augmentation was required as surveys in the area indicated that increased pumping without sufficient recharge would lead to seawater intrusion. The MAR scheme is located at the Caddadup Wastewater Treatment Plant, close to the public park, where treated wastewater is infiltrated (Figure 2-9).

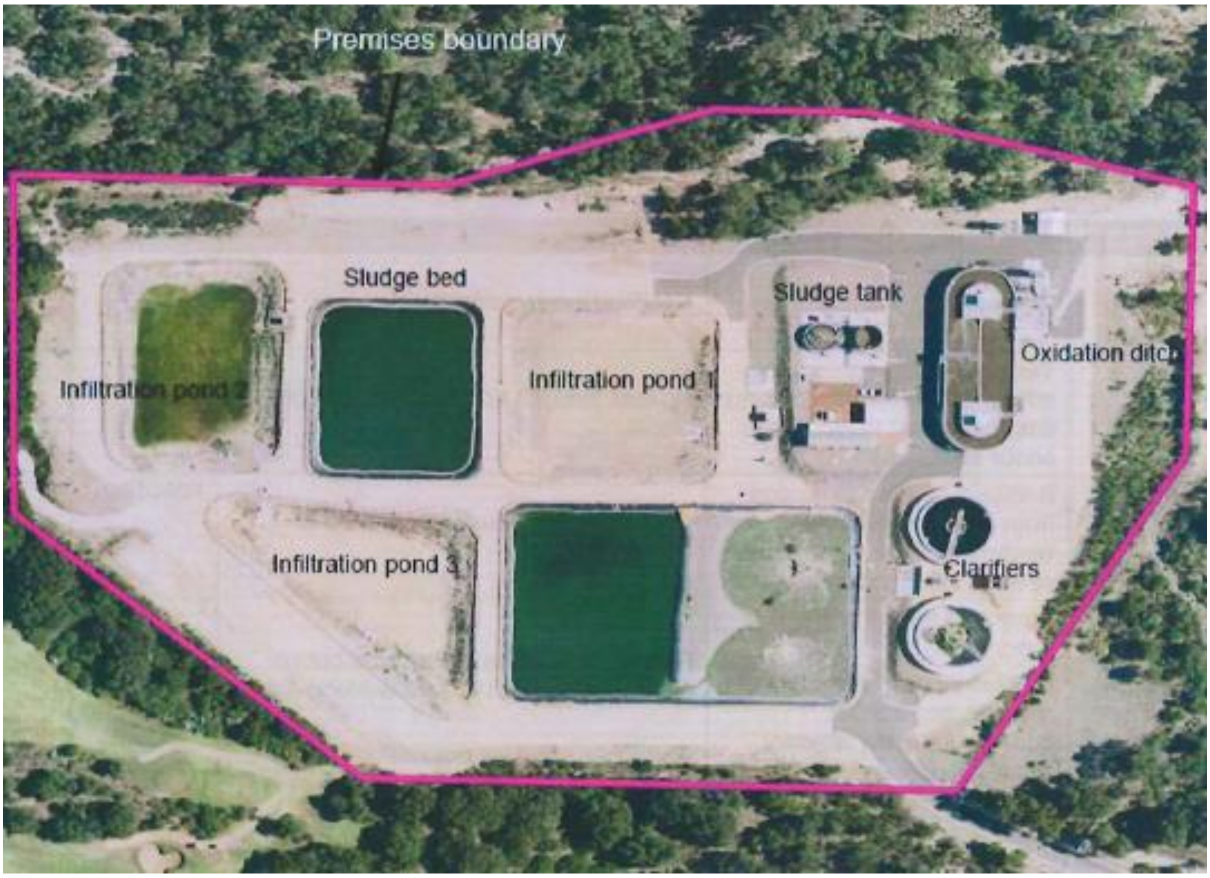

Figure 2-9: MAR Infiltration Ponds at the Caddadup WWTP
(City of Mandurah, 2015)

2.1.3.5 Menashe, Israel

Recharge Type: Spreading methods (infiltration basins)
MAR Influent Source: River water and Desalinated seawater

Israel derives some its water from desalination plants situated along the Israeli coast. During periods of low demand, the water produced at these plants needs to be stored cost-effectively as it is expensive to shut down the plants and restart them when the demand peaks. Therefore, in 1987, the Water Authority decided to store the excess water underground at the Menashe site (Figure 2-10) using MAR which was more cost-effective than constructing storage reservoirs.

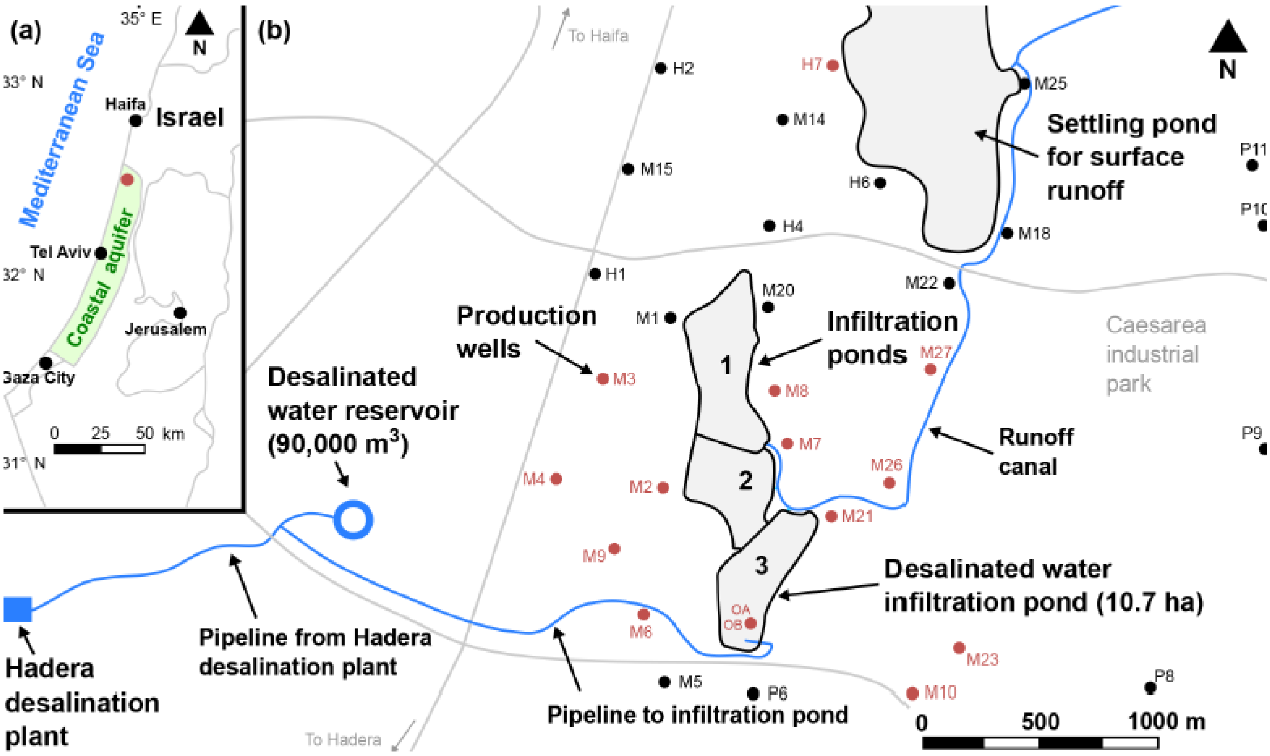

Figure 2-10: Menashe MAR Site Location
(Ganot *et al.*, 2018)

Two MAR methods are employed in this exercise, namely: injection wells and infiltration basins (Kurtzman *et al.*, 2015; Katz *et al.*, 2016). The infiltration ponds were originally only operational when there were large floods from a stream that conveys most of the runoff events in the catchment. The water was directed into a settling pond, which also acted as a forebay trapping suspended solids and silt, before being channelled into three infiltration ponds and later abstracted using the production wells which surround the infiltration ponds (Ganot *et al*., 2017).

The site overlies the Israeli coastal sandy unconfined aquifer that is ~ 80 m deep and was chosen as an infiltration site as it already had existing infiltration ponds – thus making the exercise cheaper. It is ~28 m above mean sea level. The regional groundwater level is ~3 m above sea level and seasonally fluctuates by ~2 m. The local climate is Mediterranean with a mean annual precipitation of 566 mm. The Menashe site is also used to infiltrate surplus desalinated water from a desalination plant 4 km to the west of the infiltration ponds (Katz *et al*., 2016).

A monitoring system that includes observation wells, soil sensors and temporary single ring infiltrometers was set up in 2014. Pressure sensors are used to monitor the ponding depth (Ganot *et al*., 2017). Disturbed samples obtained via hand auguring were collected to establish the underlying soil profile. The groundwater recharge at this site (after commissioning) was estimated through analytical and numerical modelling using observed data such as the pond levels, infiltration, and evaporation rates (Katz *et al*., 2016; Ganot *et al*., 2017). As of 2018, the monitoring studies have not yet revealed any evidence of physical and chemical clogging. The absence of clogging within the pond is attributed to the desalinated water having low turbidity values, although clogging may occur due to other processes such as clay swelling, dispersion, and deposition (Ganot *et al*., 2017).

2.2 Sustainable Drainage Systems (SuDS)

Sustainable Drainage Systems (SuDS) were originally termed Sustainable Urban Drainage Systems (SUDS). The phrase SUDS is understood to have been coined in 1997 to describe assorted stormwater technologies (Fletcher *et al*., 2015). However, some scholars and practitioners omit the term 'urban' to emphasise the inclusiveness of the technologies, which are also applicable in non-urban settings (Fletcher *et al*., 2015). This study will adopt the term 'SuDS'.

SuDS are 'drainage systems considered environmentally beneficial, causing minimal or no long-term detrimental impact' (Woods-Ballard *et al*., 2015). SuDS are referred to as Low Impact Development (LID) in Canada, the United States and China and Low Impact Urban Design and Development (LIUDD) in New Zealand (Randall *et al*., 2019; du Toit & Wagner, 2022). They offer a holistic means of managing stormwater by mimicking the natural hydrological cycle to preserve and recreate pre-development ecosystems while providing the most significant benefits (Cheng *et al*., 2001; Armitage *et al*., 2013).

SuDS ideally deliver high-quality runoff and drainage (S watuk *et al.*, 2021) with increased infiltration in developed areas and can potentially enhance aquifer recharge and be used for MAR (Woods-Ballard *et al.*, 2015). SuDS may also improve the quality of life in urban spaces by making them visually attractive through the provision and support of green infrastructure hence their association with Nature-based Solutions (NbS) (Department of Environment and Swan River Trust, 2006; Woods-Ballard *et al.*, 2015; Ferreira *et al.*, 2020). SuDS can achieve four main benefits, namely: improved water quality – to prevent pollution, controlling the quantity of runoff – for the management of flood risk and protecting water supply *vis* MAR, provision of amenities for communities, and promoting and sustaining biodiversity (Figure 2-11).

Figure 2-11: The four SuDS benefits

(After Woods-Ballard *et al.*, 2015)

SuDS options include: source controls – green roofs, permeable interlocking pavements, and rainwater tanks; local controls – filter strips, bioretention systems (e.g., rain gardens), swales, infiltration trenches and sand filters; and regional controls – constructed wetlands, detention, and retention ponds (Armitage *et al.*, 2013).

2.3 SuDS in the South African context

Post-Apartheid South Africa is still struggling with the social, economic, and environmental legacy of the unequal distribution of resources, infrastructure, and services. Further, the country is lagging in providing adequate water, sanitation, and ecosystem services, especially in the rapidly expanding informal settlements (slums) (Brown-Luthango *et al.*, 2017; Weimann & Oni, 2019). The government places a higher priority on achieving social equity through the provision of housing, water, and sanitation services than on environmental sustainability (Sutherland *et al.*, 2014). However, it would be erroneous to overlook the environmental sustainability aspect of water and sanitation services and associated deficits. This requires alternative water provision and management approaches that account for water quality and quantity constraints, such as Water Sensitive Design (WSD).

Fisher-Jeffes *et al.* (2017) suggest that any attempt toward WSCs in South Africa should consider the development of both the developed (formal) and presently informal settlements (Figure 2-12). Formal areas are envisioned to transition to water sensitivity by retrofitting and developing water-sensitive infrastructure (services) (Brown *et al.*, 2009). In contrast, informal areas should be redeveloped in a water-sensitive manner, learning from formal retrofits – and utilising SuDS – with an emphasis on maintenance of the installed systems (Fisher-Jeffes *et al.*, 2017).

Municipalities are charged with undertaking these transition pathways at city-scale i.e., in public spaces and through enabling regulation, while homeowners are responsible for implementing SuDS in private properties (Gauteng Provincial Government, 2020). Fisher-Jeffes *et al.* (2017), deemed it unreasonable to expect all municipalities to undertake these initiatives and implement technologies like SuDS due to limited funding and high-priority needs such as providing basic sanitation. Further, an analysis of the Upgrading of Informal Settlement Programme (UISP) reflects that municipalities are expected first to meet urgent needs such as housing and sanitation services, and SuDS are not considered urgent (Malulu, 2016). However, Fisher-Jeffes *et al.* (2017) advocate that municipalities attempt to retrofit WSD infrastructure (such as SuDS) if and where possible. While South African focused retrofit guidelines do not currently exist, guidelines and implementation manuals on greenfield SuDS have been developed for South Africa that municipalities can refer to as a starting point (Armitage *et al.*, 2013; Gauteng Provincial Government, 2020). In addition, the municipalities can partner with external organisations such as environmental non-profit organisations, universities and research institutes that have the funding to experiment with and implement experimental water-sensitive technologies in the localities (Fitchett, 2017).

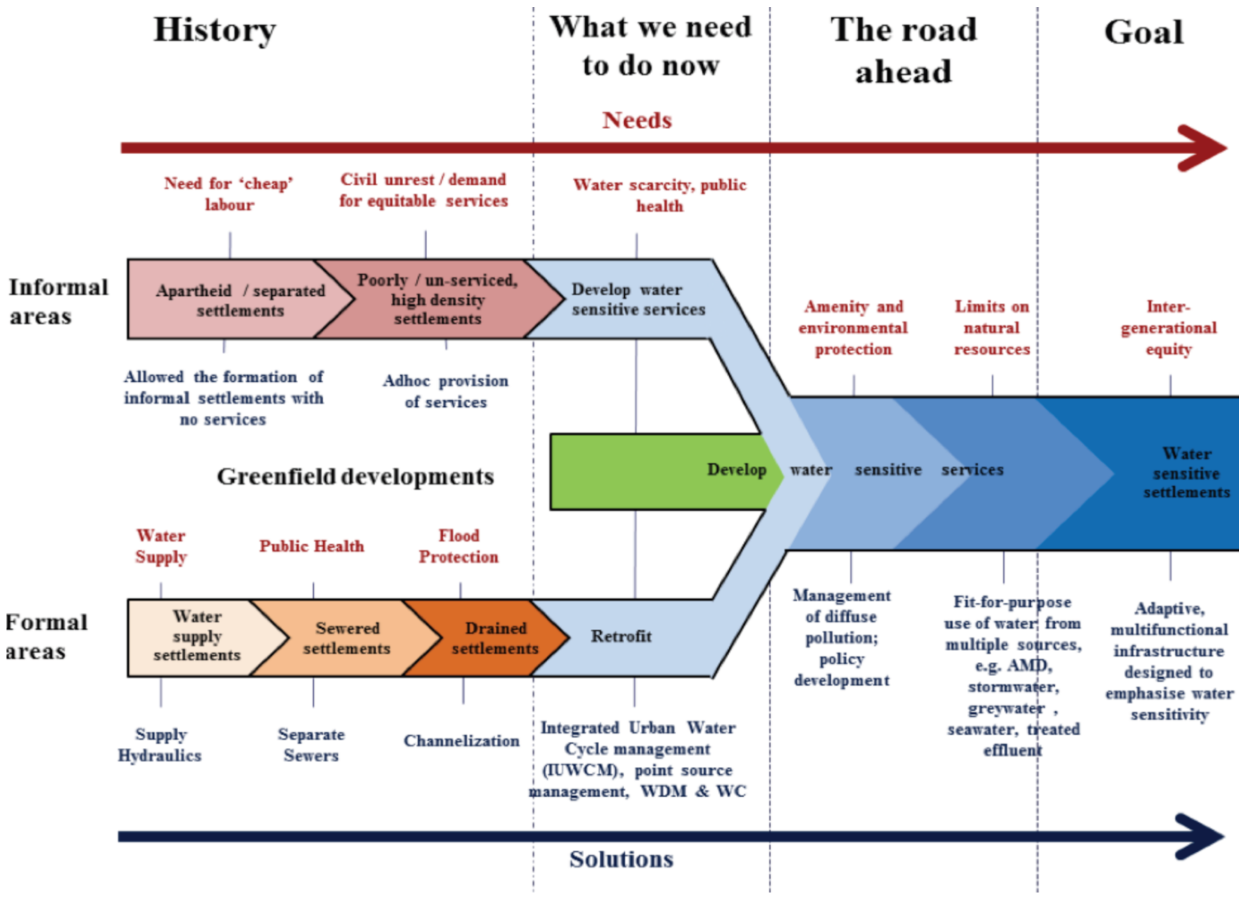

Figure 2-12: South Africa's transition to water sensitivity: 'Two histories, one future'
(Fisher-Jeffes *et al.*, 2017)

2.4 Stormwater retrofitting: From conventional systems to SuDS

Stormwater retrofitting is the installation of stormwater management systems where none existed or improving existing systems (NJDEP, 2004; Digman *et al.*, 2012). Stormwater retrofits are considered as a set of alternative stormwater management systems or devices such as SuDS, which are designed to provide or improve flood mitigation, pollutant reduction, and promote biodiversity and water resource augmentation (Claytor, 2004; Department of Environment and Swan River Trust, 2006; Digman *et al.*, 2010). SuDS can be cheaper than traditional drainage systems and often provide additional benefits apart from flood control (Armitage *et al.*, 2013). However, Schueler *et al.* (2007) assert that retrofit design, permitting, and construction are more complex, expensive, and time-consuming than new practices. Digman *et al.* (2010) counter Schueler *et al.*'s assertion and state that retrofitting SuDS can be more cost-effective – although not necessarily cheaper – than increasing the drainage capacity of conventional drainage systems.

Stormwater retrofitting can be implemented to achieve a multitude of goals which may include, *inter alia*: modifying the existing design or functional deficiencies, flood mitigation, addressing pollutants, and improving recharge and infiltration performance (US EPA, 2011). Retrofitting can be done at three levels; local/lot scale (Fitchett, 2017; Winston *et al*., 2019), neighbourhood scale (Bhaskar & Hogan, 2017; Khan *et al*., 2020; Pérez Rubi & Hack, 2021) and catchment scale where SuDS are installed across an entire catchment (MoFALD/DoLIDAR, 2013; Palla & Gnecco, 2015; Zheng *et al*., 2018; Chen *et al*., 2021).

South African guidelines developed specifically for retrofitting stormwater infrastructure for MAR are currently unavailable, and the existing guidelines mainly originate from the USA, the UK, and Australia. The primary retrofit guidelines, technical notes, and manuals reviewed in this document are as follows:

- The Pennsylvania Basin Retrofit Guide – Pennsylvania Environmental Council (2012)
- Stormwater Retrofit Techniques for Restoring Urban Drainages in Massachusetts and New Hampshire – US Environmental Protection Agency US EPA (2011)
- Stormwater Retrofits: Tools for Watershed Enhancement – Claytor (2004)
- New Jersey Stormwater Best Management Practices Manual: Chapter 8 – New Jersey Department of Environmental Protection NJDEP (2004)
- Center for Watershed Protection Manual 3: Urban Stormwater Retrofit Practices – Schueler *et al*. (2007)
- Stormwater Management Manual for Western Australia: Retrofitting – Department of Environment and Swan River Trust (2006)
- Retrofitting to Manage Surface Water – Digman *et al*. (2012)

The Department of Environment and Swan River Trust (2006), Schueler *et al*. (2007), US Environmental Protection Agency US EPA (2011), Pennsylvania Environmental Council (2012), and Digman *et al*. (2012) provide guidance for implementing stormwater infrastructure retrofits. These steps are summarised in Table 2-3.

An analysis of the reviewed documents reveals that no one document contains all the required steps to complete a successfully retrofit which complicates the retrofit process. For example, while the US Environmental Protection Agency US EPA (2011) and Pennsylvania Environmental Council (2012) contain retrofit steps, they lack technical retrofit design guidelines. However, they reference other guidelines, such as Schueler *et al.* (2007), who provide technical design considerations and examples for various stormwater retrofits. Consequently, in the absence of an all-encompassing guideline for stormwater retrofitting, a combination of several guidance documents on the planning, design, installation, and maintenance of SuDS must be used. These documents include Coffman (1999), Connecticut Department of Environmental Protection (2004), Armitage *et al.* (2013); US EPA (2013); Public Utilities Board (2018). In addition to technical considerations, the implementation of retrofits requires addressing other factors, particularly public and homeowner concerns. For example, safety concerns for children should be addressed if a retrofit results in an unfenced retention pond near a residential area as there may be a risk of drowning. Stagnant water resulting from retrofitting can create mosquito breeding conditions and lead to health issues. The landscaping of retrofitted ponds can introduce allergy-causing plants, and the retrofit project itself may become visually unattractive, affecting property values and community perceptions. However, public engagement is often not prioritised when initiating retrofit projects (Brink *et al.*, 2016; Vasconcelos *et al.*, 2022). For example, NJDEP (2004) only considers public engagement after preliminary design, topographical surveys, and retrofit cost estimates, while Claytor (2004) does not mention community engagement or participation at all. Further, in instances where community engagement is mentioned there is often no guidance on how to conduct the engagement although Digman *et al.* (2012) refers to external documents that offer engagement methods developed for the UK.

In summary, the absence of specific South African stormwater retrofitting guidelines necessitates the use of a combination of existing international documents. Moreover, while MAR is not explicitly mentioned as a retrofit objective in the aforementioned reviewed guidelines, certain retrofits can facilitate infiltration (NCDEQ, 2017). Nonetheless, the applicability of these international guidelines and documents in the South African context remains unknown and requires further investigation, particularly regarding the role and impact of community participation in stormwater retrofits.

Table 2-3: Stormwater retrofit steps

Steps	Reference
Develop a strategy: • Identify potential sites. • Establish the goal/aim of retrofit. • Establish project cost estimates. • Consider existing resources. • Create project schedule.	• Claytor (2004) • Department of Environment and Swan River Trust (2006) • Schueler *et al*. (2007) • US EPA (2011) • Pennsylvania Environmental Council (2012)
Create an inventory and prioritise (site selection): • Stakeholder mapping • Homeowners and community outreach addressing issues like visual aesthetics, health and safety concerns and site use/access.	• Department of Environment and Swan River Trust (2006) • Pennsylvania Environmental Council (2012) • Digman *et al*. (2012)
Design and construction: • Determine permit and regulations. • Conduct site assessment (soil tests and survey) • Appoint design consultant (depending on retrofit complexity) • Develop model for suggested retrofits. • Design and construct – include preconstruction meetings with identified stakeholders and appointed contractors. Continued outreach with the community to address concerns about safety, schedule, appearance, and overall purpose of the project.	• Department of Environment and Swan River Trust (2006) • Schueler *et al*. (2007) • US EPA (2011) • Digman *et al*. (2012)
Monitoring and maintenance: • Develop a monitoring plan and budget. • Develop a detailed maintenance plan with performance indicators.	• NJDEP (2004) • Department of Environment and Swan River Trust (2006) • Schueler *et al*. (2007) • Pennsylvania Environmental Council (2012)

2.5 The role of community participation in retrofits

2.5.1 Engagement vs outreach

Community participation is an activity that seeks communities' active and genuine involvement in projects that offer to solve problems that directly or indirectly affect them (McEvoy *et al.*, 2019). Community participation can be achieved – to different degrees – through community engagement or community outreach. The terms 'community engagement' and 'community outreach' are often mistakenly assumed to be synonymous but exist on opposite ends of a spectrum, though with some overlapping elements. On one end of the spectrum is community outreach that is defined as 'a one-way communication that informs the community about an issue, problem, opportunity, or decision' (Donovan, 2014). Community outreach often involves community leaders [or researchers] informing communities of a plan/decision and inviting their input and involvement (Joly *et al.*, 2012; Donovan, 2014). However, Babler *et al.* (2014) highlight that community outreach can inadvertently lead to the creation of self-defined communities (affinity groups). On the other hand, community engagement entails collaborative work with people affiliated by geographic proximity, special interest, or similar situations (Centres for Disease Control and Prevention, 2011). According to the Centres for Disease Control and Prevention (2011), community engagement is depicted as a continuum of community involvement, spanning from community outreach to shared leadership in projects and an enhanced collaborative partnership (Figure 2-13).

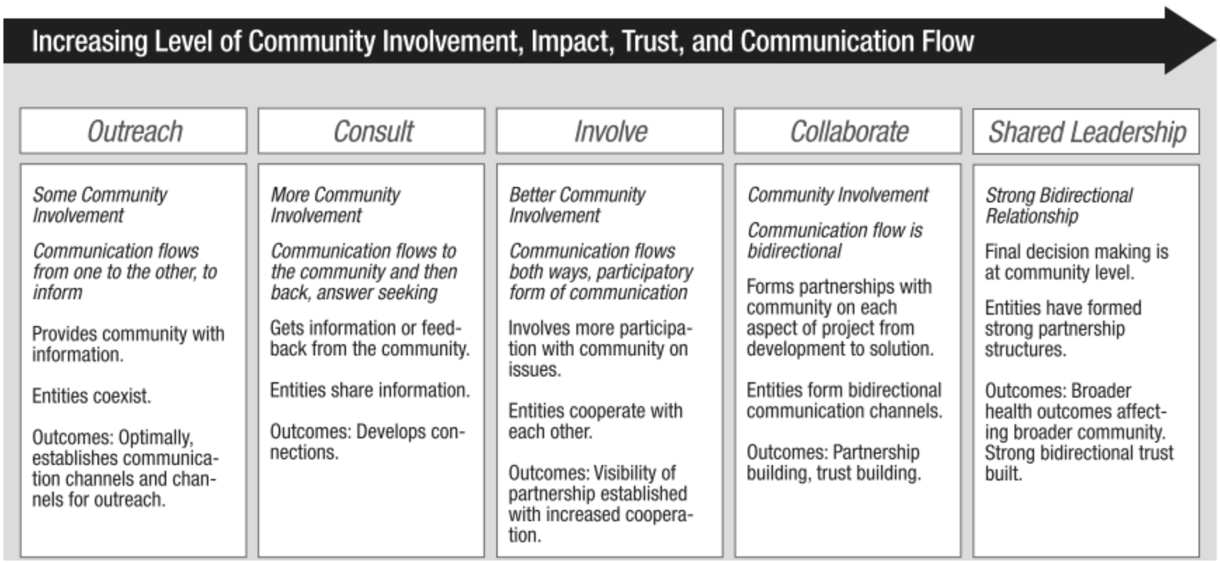

Figure 2-13: The continuum of community engagement
(Centres for Disease Control and Prevention, 2011)

Community engagement and outreach are critical because communities play an essential role in public projects (Leisher *et al.*, 2012). Rathenam & Dabup (2017), using a Relative Importance Index found that South African communities influence the success [completion] of public infrastructure projects more than other stakeholders such as the Project Manager or Contractor. The finding by Rathenam & Dabup (2017) contrasted with a study by Malkat & Kang (2012), who used Dubai as a case study and found that the role of project managers ranked higher than community influence in ensuring successful and timely public infrastructure projects (Malkat & Kang, 2012).

McCloskey *et al.* (2013) suggest that culture and community organisation play a substantial role in enhancing community engagement as a community's beliefs and practices can inform the ease of trust and willingness to engage with 'outsiders'. Further, knowledge and acknowledgement of the community organisational structure can aid researchers in identifying individuals or groups that have the political capital to sway opinions. However, some of these structures and community politics are not initially apparent, and when identified, the researchers need to navigate them with care to avoid confrontations that may lead to withdrawn consent.

While practitioners generally agree that community participation generally results in positive outcomes, Lizarralde *et al.* (2008) suggest that community participation can have negative outcomes, although they concede that their findings could result from the poor application of community engagement approaches. Further, Aule *et al.* (2019) assessed 54 cases – in planning areas such as housing, tourism, transportation, and urbanism (e.g., urban sanitation, waste management and pollution) – and established that while 70% of the projects had positive outcomes, 30% of the projects resulted in adverse consequences because of community participation.

2.5.2 Community participation in SuDS interventions

Literature on SuDS and the retrofitting of SuDS in residential areas predominantly focuses on the Global North (Vasconcelos *et al.*, 2022). Further, the role and impact of stakeholder/community participation in the SuDS arena, particularly SuDS retrofits, in both the Global North and South are sparsely discussed (Brink *et al.*, 2016; Vasconcelos *et al.*, 2022). The terminologies 'Global North' and 'Global South' are frequently used to describe the distinction between developed and developing nations, as defined by the International Monetary Fund (Nielsen, 2011, 2013). The Global North generally encompasses countries in North America, Europe, and East Asia, whereas the South comprises countries in Africa, Latin America, and certain parts of Asia. These terms may hold different connotations depending on their usage and are not always consistently applied (Khokhar & Serajuddin, 2015). In this study, the terms are employed to describe the economic disparity between developed and developing nations – which can impact social contexts.

This section of the review explores literature on community participation in the SuDS arena, including blue-green infrastructure and nature-based solutions, specifically in the Global South focussing on South Africa, East Africa, and Latin America.

2.5.2.1 Community Participation in South Africa

Winter *et al.* (2008a, 2008b) described their attempts to implement a collaborative, participatory action research approach to address greywater management in informal settlements in Cape Town, South Africa. These settlements are characterised by unplanned and densely populated residential areas that lack basic infrastructure and formal legal recognition. The study aimed to focus on community-centred retrofits for greywater management but achieved limited success. Initially, the research team faced challenges in establishing a local institutional structure to conduct workshops and build trust with residents, resulting in no significant demonstration effect. The researchers found that, in their context, most installations in previous studies were unsuccessful experiments carried out by outsiders, such as researchers and non-governmental organisations, who sought the involvement of local agents, including local representative social structures such as streets or section committees. They also noted that inadequate communication and trust between residents and local authority structures affected the development of social capital and hindered projects. Winter *et al.* (2008a, 2008b) suggest that implementing a sustainable participatory action research approach for managing grey water in informal settlements requires building trust and establishing a local institutional structure. These studies by Winter *et al.* (2008a, 2008b) highlight the importance of community engagement and collaboration in addressing greywater challenges in South African informal settlements.

Adegun (2015) studied the outcomes of two stormwater management interventions in Johannesburg, South Africa, focusing on two informal settlements. The research revealed that the lack of community involvement in state-led public infrastructure projects led to project failures, a pattern observed in other state-led initiatives. The state-led projects followed a top-down approach, with decisions made by government decision-makers and then implemented through a hierarchical structure with limited input from the community (Međugorac & Schuitema, 2023). Adegun (2015) also identified opportunities for meaningful community involvement in stormwater management initiatives. This alternative approach adopts a bottom-up perspective, emphasising community participation at the micro- or meso-level (Međugorac & Schuitema, 2023). A comparative analysis of the two cases demonstrates the limitations and potential of both top-down and bottom-up approaches and suggests the need to integrate the two approaches for sustainable stormwater drainage in informal settlements. Adegun (2015) found that community involvement is crucial for the success of stormwater management interventions in informal settlements and advocated for co-designing and producing solutions suitable for informal communities to bridge the gap between informal and formal infrastructural provision.

Fitchett (2014, 2017) investigated the application of sustainable drainage systems (SuDS) in an informal settlement for grey and stormwater management in Johannesburg, South Africa. Fitchett (2014, 2017) reveal that successfully implementing SuDS in informal settlements requires a comprehensive and multi-disciplinary approach considering social, environmental, and economic dimensions. Engaging with community members within the settlement is crucial to ensure success, as they have different priorities and attitudes towards the interventions. The studies highlight the importance of participatory and inclusive approaches in implementing SuDS in informal settlements rather than top-down approaches that do not address community needs or build trust. The researcher found that involving the community and promoting knowledge exchange can enable residents to drive the introduction and integration of SuDS as complementary to existing conventional drainage systems. However, the studies also reveal that even with extensive engagement, the absence of financial incentives for labour intensive retrofits can hinder community participation in informal settlements. Much like Winter *et al.* (2008a, 2008b) and Adegun (2015), Fitchett (2014, 2017) illustrates that implementing sustainable drainage systems in informal settlements requires a comprehensive and multi-disciplinary approach considering social, environmental, and economic dimensions. Finally, involving residents in the process is more effective than top-down approaches, but a lack of financial incentives can hinder community participation.

2.5.2.2 Community Participation in East Africa

Diep *et al.* (2022) conducted a study on retrofitting nature-based solutions (NbS) to manage soil erosion, greywater, and stormwater in informal settlements in Nairobi, Kenya, and Dar es Salaam, Tanzania. Four NbS sites were assessed in both countries. They found that community participation in the project's implementation phase positively influenced residents' perceptions and valuations of NbS. They also found that the aesthetic appeal of NbS fosters community engagement. Their study also demonstrated the potential of NbS to address multiple risks in informal settlements, providing benefits such as well-being, safety, and environmental awareness. However, the authors acknowledge that maintaining NbS projects within the context of low-income and low-land ownership remains challenging, and participatory approaches are necessary for community involvement and project sustainability. The authors argue that shared management and maintenance responsibilities between municipal authorities and residents are essential to ensure the long-term effectiveness of NbS and social justice. In addition, new NbS projects should integrate pre-existing systems managed by local organisations. Furthermore, intermediary organisations should create niche experiments that can eventually become advocacy tools for change. The case studies from Diep *et al*. (2022) demonstrate that early community involvement is crucial to stimulate stewardship and that continued monitoring, maintenance, and evaluation are necessary to confirm long-term outcomes.

2.5.2.3 Community Participation in Latin America

The Sustainable Water Management Improves Tomorrow's Cities' Health (SWITCH) research project described by Knauer *et al*. (2010) aimed to explore integrated urban water management in Brazil, specifically in Belo Horizonte. To that end, the study investigated technical, social, and governmental pathways for sustainable water management and addressed challenges related to urban drainage at the local, sub-catchment, and city scales. Knauer *et al*. (2010) highlight that the discernible growth in public participation within municipalities increased after the end of the military dictatorship in 1985. Initially, stakeholder participation was not an integral component of the SWITCH project, as it was assumed that existing public participation mechanisms would suffice. However, after its inception, researchers recognised the need to actively seek input from previously overlooked stakeholders and duly incorporate their perspectives.

The participatory structure pertaining to community involvement in Belo Horizonte was democratically inclined, encompassing the submission of project proposals by citizens and subsequent voting to determine project prioritisation. The involvement of local alliances, comprising community institutions, individual community members, municipal officials, and technicians, served as a driving force behind community participation. Demonstration activities and various events centred on training and raising awareness were employed to foster mutual learning and capacity-building endeavours. Emphasis was placed on the engagement of committees involved in the municipal participatory budgeting process to advocate for more sustainable drainage within the ambit of the participatory budget. The researchers identified five local demonstration sites featuring various initiatives, such as rainwater harvesting facilities at educational institutions, local infiltration trenches within parks, and wetlands situated near schools. These demonstrations were accompanied by community participation, incorporating seminars, and outreach activities for information dissemination. At the sub-catchment level, three sites were selected for research purposes, focusing on infiltration and wetlands, albeit devoid of community participation activities.

The demonstrations primarily revolved around SuDS features, necessitating both retrofitting of SuDS technologies into existing infrastructure and the construction of new facilities. These demonstrations augmented the public visibility of the SWITCH project. The rainwater harvesting demonstration facilitated a policy amendment within the municipality, promoting widespread adoption of this practice.

A major lesson derived from this research is the paramount significance of repeated engagements with relevant organisations. Knauer *et al.* (2010) reported that considerable time and effort were invested in engaging interested parties within the utility sector, achieved through workshops designed to communicate shared benefits, most notably pollution reduction. However, some challenges were encountered during this research process. For example, inadequate social sciences and ethnographic skills hindered the comprehensive analysis of engagement processes. Furthermore, political interference posed challenges at one of the project sites. Additionally, the language barrier – most of the local communities only spoke Portuguese while some SWITCH team members only spoke English and other non-Portuguese languages – restricted the participation of certain team members in training sessions and interactions with international network members. Overall, the findings from the SWITCH project underscore the pivotal role of community participation, municipal policies, demonstration activities, and organisational engagements in fostering sustainable water management practices in urban contexts in the Global South.

Diep *et al.* (2019) conducted a contextual analysis of the applicability of green infrastructure, including SuDS, to informal settlements in São Paulo, Brazil. They note that green infrastructure faces challenges in these settlements due to their high urban density, informality, and land tenure issues. However, green infrastructure also presents an opportunity to protect ecological zones while providing necessary services to urban residents. In many cases, informal settlements are situated in ecologically frail areas, leading to conflicts when these areas need protection, resulting in the removal of settled families. The authors suggest that the dominant policy direction in urban planning, which separates technoscience from social systems, is affecting the sustainability of green infrastructure implementation. Therefore, they propose that a combined understanding of planning regimes, governance systems, and residents' attitudes is necessary for successful implementation. From Diep *et al.* (2019), it is evident that green infrastructure can help protect ecological zones, but there is often a conflict between the need to protect ecologically fragile areas and the need to house settled families.

Another Latin America-based study was conducted by Pérez Rubi & Hack (2021) on co-designing and implementing nature-based solutions (NbS) to manage grey water in San Jose, Costa Rica. They found that an adaptive approach that considers contextual information and different knowledge sources is necessary to implement NbS. Further, engaging local dwellers is crucial in the retrofit process, resulting in solutions that are more likely to be successful. However, uncertainties around NbS make it challenging to engage local dwellers, and the authors suggest promoting NbS prototypes that include post-implementation monitoring and assessments to generate knowledge on NbS. Pérez Rubi & Hack (2021) suggest that co-design processes could help identify context-specific aspects relevant to prototype planning and design. There is a need for opportunities for regional experience exchange and communication to improve knowledge dissemination for the implementation of NBS.

2.5.2.4 Summary of the case studies

The studies analysed in this section offer valuable insights into the challenges and opportunities associated with sustainable water management in informal and low-income settlements in the Global South. These studies highlight the significance of community involvement and collaboration in addressing environmental challenges in informal settlements. Top-down approaches that fail to address community needs and build trust have been found to be inadequate, and participatory and inclusive approaches that engage with the community, promote knowledge exchange, and address social, environmental, and economic dimensions are deemed necessary. The studies also identify challenges such as urban density, informality, and land tenure issues that necessitate a combined understanding of planning regimes, governance systems, and residents' attitudes for successful implementation. The reviewed studies offer steps towards a framework for sustainable water management in informal settlements that considers community needs and aspirations, fosters trust and collaboration, and promotes social, environmental, and economic sustainability. Prominently, the findings of the reviewed studies indicate a need for more comparative studies across different countries and contexts to identify best practices for introducing SuDS and similar interventions. However, additional research is necessary to evaluate the economic sustainability of sustainable water management interventions in informal and low-income settlements. The findings from these studies also suggest the need for more research on the role of community engagement and participation in shaping and sustaining sustainable water management interventions in informal and low-income settlements within a Global South context.

2.6 Assessing the hydraulic performance of SuDS

The increasing importance placed on stormwater management and reuse (Cousins, 2018) has led to the increased use of SuDS which can be installed in green fields or retrofitted to existing systems. However, the stormwater design requirements for these systems differ, with some emphasising water quality while others focus on the potential stormwater quantity. Some local authorities/municipalities require all developments to include on-site sustainable stormwater management facilities (CoCT, 2002; Fletcher *et al.*, 2015). This requirement has amplified the necessity of hydrological modelling as it is often difficult to forecast the short and long-term impacts of the various forms of SuDS interventions in developments using traditional empirical methods (Burszta-Adamiak & Mrowiec, 2013).

Hydraulic models are used for extrapolation, fill temporal and spatial data gaps, and optimise networks. They are helpful for the planning, designing, and operation of water-related systems. Hydrological models have been developed to aid flood routing, forecasting and stormwater design, analysis, and optimisation (James, 2005). Models can also evaluate stormwater infrastructure like pipes, drains, and ponds. Their ability to predict the impact of various interventions and hydrological conditions makes them an essential and cost-effective component of water engineering, particularly stormwater design (USEPA, 2015). However, it is crucial to report the uncertainty when interpreting a model.

Various software has been developed that can be used to simulate single or long-term hydrological events (Lockie, 2010; Armitage *et al.*, 2013). Arguably the more popular one is the Stormwater Management Model (SWMM), developed by the United States Environmental Protection Agency (USEPA) and CDM Inc (Lockie, 2010). SWMM was developed in1971 and is an open-source, window-based desktop program that runs on the SWMM5 computational engine (USEPA, 2015; Rossman & Wayne Huber C., 2017). SWMM is a largely deterministic model – that is, it uses limited, known scientific knowledge with limited observed data. However, the

SWMM engine includes subroutines that rely on statistical models, including regression and stochastic models (James, 2005).

The freeware component of the SWMM5 engine has been further developed to create enhanced models, such as HydroSWMM and PCSWMM, which include graphical user interfaces and geographical information system capabilities and have gained popularity in South Africa. However, these software packages require the purchase of licenses, although both offer grants to students and researchers.

PCSWMM was first released in 1984 (James *et al*., 2010) and like most SWMM based software as some limited subsurface modelling capabilities. As a result, its use in modelling some LIDs, like infiltration ponds, is limited. The software performs reasonably well in areas with low water tables where the infiltration rate is minimally affected by the water table but can be limited in high water tables because the water table depth influences the infiltration rates. However, the software's functionality for shallow aquifers (high water tables) can be enhanced by subsurface calibration, application of lateral groundwater equations or modifying SWMM functions (Zhang *et al*., 2018). Subsurface calibration on PCSWMM is superior to the base SWMM engine because PCSWMM includes inbuilt calibration and sensitivity analysis functionality that improves modelling efficiency.

While SWMM engine-driven software, like PCSWMM, do not always optimally simulate groundwater interaction, they are still the most popular software for modelling LIDs (Palla & Gnecco, 2015; Zhang *et al*., 2018; Randall *et al*., 2019; Aparicio Uribe *et al*., 2022; Le Floch *et al*., 2022). Coupled models – linked surface and groundwater models – are better suited to modelling flow in areas with shallow water tables, but the coupling process is complex and often requires advanced coding experience to link the backends of the software and share pre- and post-processed data. Further, coupled models are only applicable and helpful if both models can be calibrated and validated (Rivard *et al*., 2004; Marchildon & Kassenaar, 2013; Wu *et al*., 2018; Babaie *et al*., 2021).

2.7 Treatment processes in infiltration basins

Stormwater contains, among other things, contaminants of emerging concern (CEC), heavy metals, hydrocarbons, microorganisms, nutrients, and suspended solids (Pitt *et al*., 2004a; Hamlyn-Harris *et al*., 2019; Robertson *et al*., 2019). These all have potential health risks when they exceed certain concentrations, and thus it is essential to ensure that stormwater is adequately treated before it is reused. The level of treatment in stormwater harvesting for MAR is tailored to the specific purpose of reuse, resulting in varying guidelines for acceptable contaminant concentrations (Hamlyn-Harris *et al.,* 2019). A widely accepted principle in MAR is to ensure that infiltrated or injected water quality does not deteriorate the existing groundwater quality (EPA SA, 2004; Murray *et al.*, 2007; Murray, 2009). However, there are some exceptions that may require individual investigation and approval (Murray *et al.*, 2007; Murray, 2009). Further, this principle assumes that the groundwater quality is initially satisfactory – but groundwater is often highly polluted (Teta & Hikwa, 2017; Ebrahim *et al.*, 2020; Banks *et al.*, 2021; Lapworth *et al.*, 2022), necessitating the consideration of stringent guidelines if the water is abstracted for direct domestic use.

One approach to treating urban stormwater before recharge is using SuDS, with infiltration basins/ponds being an example. Infiltration basins may incorporate treatment trains that facilitate partial stormwater treatment at various stages (Armitage *et al.*, 2013; Koch *et al.*, 2014). These typically begin by reducing suspended solids and litter in the forebay area by slowing the flow of water to a velocity slower or equal to the settling velocity of the suspended solids (typically 0.3 m/s). This is achieved by constructing forebays or check dams across the stormwater's flow path. A forebay is a small basin constructed after the inlet and located upstream of the main infiltration component. The forebay can be unvegetated or contain some vegetation to manage sediment (Armitage *et al.*, 2013; Woods-Ballard *et al.*, 2015).

In their study that investigated the hydrological impacts of retrofitting check dams in a bioswale (grass-lined channels used for the conveyance, storage, infiltration, and treatment of stormwater), Winston *et al.* (2019) observed that the introduction of check dams within the swale resulted in an increased detention time and a 45% reduction in suspended solids. These findings are reinforced by similar studies documented by Yu *et al.* (2001) and Butler *et al.* (2018). Check dams provide both: (1) a litter and suspended solids control through the creation of forebays; and (2) a detention structure that encourages infiltration (MAR). The presence of vegetation in the bioswale also aids in the trapping of suspended solids.

The processes in the forebay typically result in a reduction of the Total Suspended Solids (TSS) load by at least 60% (Bratieres *et al.*, 2008; Hatt *et al.*, 2009; Goonetilleke & Lampard, 2018) and are essential for removing heavy metals, which are often bound to the suspended solids (Maniquiz-Redillas & Kim, 2016).

Some infiltration basins include infiltration swales. Infiltration swales are shallow, flat-bottomed, vegetated open channels designed to convey, treat, and often attenuate stormwater (Woods-Ballard *et al.*, 2015). The vegetated top layer of a swale initially removes the coarser fraction of particulate solids in stormwater runoff before infiltration.

The final stage in the treatment train is the biotransformation and sorption of pathogens and dissolved minerals by the soil media as the water infiltrates into the ground. Biotransformation and sorption treatment mechanisms, such as fine filtration, microbe catabolism, denitrification, sorption, precipitation, plant uptake, and soil accretion, can occur leading to a further reduction in the concentration of contaminants.

Depending on the SuDS feature, the treated water is then discharged into the storm network, waterways, or stored in aquifers. However, for a comprehensive understanding of SuDS, such as infiltration ponds, it is imperative to gain insights into the origins and fate of contaminants in stormwater and the various treatment mechanisms contributing to pollutant removal. Moreover, it is essential to understand the main factors influencing these processes to inform the experimental design and interpret any laboratory outcomes. The subsequent section thus briefly discusses the origins and fate of common contaminants found in stormwater and their treatment processes in SuDS.

2.8 The fate of contaminants in SuDS

2.8.1 The fate of TN, TP, and TOC

Nitrogen is a common contaminant in stormwater emanating from various anthropogenic activities such as agriculture, gardening, and sewer pipe bursts. In this study, the term "contaminant" refers to any substance or material found in stormwater runoff that can potentially cause adverse effects on human health, aquatic life, or the environment when present at certain concentrations or exposure levels. The determination of harmful concentration levels relies on established guidelines and standards. For instance, certain chemicals such as heavy metals (e.g., lead) or toxic organic compounds can exhibit harmful effects even at low concentrations due to their inherent toxicity. Conversely, substances like sediment or nutrients may not possess inherent toxicity but can still have detrimental impacts when present in excessive quantities. Contaminants do not automatically imply a health risk associated with the water (Pitt *et al*., 1999; Ander *et al*., 2013).

The main forms of inorganic nitrogen are nitrite (NO_{2-}), nitrate (NO_{3-}), and ammonium (NH_{4-}), and these together with organic nitrogen make up a major part of measurable total nitrogen (TN) (Zinger *et al*., 2013). While nitrogen is desirable for plant growth, its presence in urban water in high concentrations has negative consequences, such as the eutrophication of rivers and coastal waters (Aryal *et al*., 2010; Goonetilleke & Lampard, 2018). Further, high nitrate concentrations in humans can result in methemoglobinemia, which is particularly hazardous in infants under three months of age (DWAF, 1996a; Lu *et al.*, 2009). Thus, the reduction of TN concentrations above allowable limits in water is imperative. TN concentration can be reduced by plant uptake and soil adsorption, as shown in a study by Bratieres *et al.* (2008) and Milandri *et al.* (2012), where vegetated columns removed greater amounts of TN than non-vegetated columns. Some reduction of TN may be achieved using slow sand filtration, but the method is unreliable as nitrates can later leach out. TN can also be removed by microbe catabolism and denitrification by denitrifying bacteria (e.g., *Lactobacillus* and *Pseudomonas*) (Feng *et al.*, 2012; Milandri *et al.*, 2012; Jacklin *et al.*, 2021; Geronimo *et al.*, 2022). Denitrification is an anaerobic process in which nitrates are used as an electron acceptor for anaerobic respiration – producing nitrogen gas, N_2 – by bacteria (NO_{3^-} → NO_{2^-} → NO → N_2O → N_2). The balanced redox equation is given in Equation 2-1.

$$2NO_3{}^- + 10e^- + 12H^+ \rightarrow N_2 + 6H_2O \qquad \textbf{Equation 2-1}$$

The presence of a carbon source has been reported to improve TN removal by enhancing denitrification (Blecken *et al.*, 2009; Rahman *et al.*, 2019; Tao *et al.*, 2022). Furthermore, laboratory and field studies have shown that creating a saturated (anaerobic) zone in filtration basins or columns improves TN removal efficiency by increasing denitrification (Li & Davis, 2014; Wu *et al.*, 2017). On the other hand, denitrification has been observed in systems with measurable dissolved oxygen concentrations in their waters (Lu *et al.*, 2009). The rate and magnitude of denitrification are therefore influenced by oxygen availability, temperature, the concentration of nitrate, organic carbon supply and the presence of denitrifying bacteria (Osman *et al.*, 2019). Typical SuDS have been reported to remove 40-69% of TN in stormwater (Jia *et al.*, 2015; Osman *et al.*, 2019; Lopez-Ponnada *et al.*, 2020). However, it is worth noting that other studies have reported both TN leaching as well as complete TN removal (Koch *et al.*, 2014).

Phosphorus, like TN, is also a common contaminant in stormwater derived from atmospheric deposition, fertilisers, organic materials (leaves, dead plants, animal excrement) and sewer leaks (Pitt *et al.*, 2012; Søberg *et al.*, 2020). In addition, excess total phosphorus (orthophosphate, total inorganic phosphate, or total dissolved phosphorus) builds up in catchments and is washed off into waterways, contributing to eutrophication (DWAF, 1996b; Yang & Toor, 2018). Laboratory studies have shown that phosphorus can be removed in SuDS, and the removal efficiency correlates to media depth in full-scale facilities and column studies. The retained phosphorus in the media is then removed by plant uptake in vegetated SuDS (Hatt *et al.*, 2007).

Sorption is the main process that leads to a reduction in phosphorus concentration via biofilter/swale media. Sorption is a process in which reactive elements, complexes or compounds are attached to surfaces, sometimes of solids or other elements and compounds (McGechan & Lewis, 2002). In this treatment process, phosphorus is bound to particulate matter – such as soil or phosphorus-sorbing matter like mineral surfaces such as iron or manganese oxides and clay (Qin *et al.*, 2018). The chemistry and process that govern the sorption process are complicated, and numerous mathematical models have been developed that seek to understand the process (McGechan & Lewis, 2002; Qin *et al.*, 2018). Nevertheless, there is a general agreement that some of the key factors that influence the rate of sorption are; (1) clay fraction content, (2) surface area of sorbents, (3) pH, (4) media sorption capacity, (5) soil organic matter, (6) sorptive concentration – see Hsieh *et al.* (2007), Fink *et al.* (2016), Ngatia *et al.* (2017); Abboud *et al.* (2018), Qin *et al.* (2018) and Yang *et al.*, (2019).

Total Organic Carbon (TOC) is the sum of dissolved organic carbon (DOC) and particulate organic carbon (POC) and is also commonly found in stormwater from natural and anthropogenic activities (Yuan *et al.*, 2019; Day *et al.*, 2020). TOC concentrations > 5 mg/L can cause some health risks (DWAF, 1996b). Low molecular weight organic carbon can be released into the atmosphere, but heavy molecular OC is suspended and accumulates in the soil where fungi and microbes can break into small labile forms and, ultimately, to carbon dioxide (DWAF, 1996a; Kalev & Toor, 2020). Therefore, TOC in stormwater is reduced through filtration in SuDS, where it accumulates and can be broken down by microbes and denitrifying bacteria.

2.8.2 The fate of heavy metals

Heavy metals, e.g., Chromium (Cr), Copper (Cu), Lead (Pb), Nickel (Ni), Iron (Fe) and Zinc (Zn), are often found in stormwater runoff at toxic levels that pose significant health and aquatic risks if unmitigated (ANZECC, 2000; Goonetilleke & Lampard, 2018; Sakson *et al.*, 2018; Brusseau & Chorover, 2019). Heavy metals are particularly prevalent in urban catchments where heavy metals arise from, *inter alia*, exhaust fumes, brake and tyre wear, de-icing agents, exposed soils, metal roofs, mines and other industries, and atmospheric deposition (Pitt *et al.*, 2004; Hwang *et al.*, 2016; Sakson *et al.*, 2018; Robertson *et al.*, 2019). Heavy metals in stormwater can be found either as free dissolved soluble cations or are particulate bound, which depends on the solubility of the metals (Maniquiz-Redillas & Kim, 2016) – Cr, Cu and Ni have been found to have higher solubility than most heavy metals, while Pb and Zn are more likely to occur as particulate-bound contaminants (Huber *et al.*, 2016).

Heavy metals pose potential risks to human, animal, and aquatic life, necessitating their removal from stormwater before discharge into open bodies or infiltration to aquifers. Several SuDS can be used for this, albeit the selection of the appropriate system depends on the form of the dominant heavy metal in the water. For example, filtration can remove particulate-bound metals, while sorption processes more readily remove soluble metals (soptives), which sorb to solid phase materials (sorbents). The main type of sorbents in soils are layer silicate clays, metal-oxyhydroxides, and soil organic matter (Thompson & Goyne, 2012). However, metals are not broken down in filtration and sorption but are 'held in suspension' in the filter material and can potentially leach out of the material and be reintroduced into the system (Koch *et al.*, 2014). Further removal of removal of heavy metals from SuDS can be achieved by removing contaminated filter media and soil washing (Wang *et al.*, 2020; Zhang *et al.*, 2022).

Soil washing is a remediation technique that aims to extract heavy metals from the soil using various extractants and reagents. The process involves excavating the polluted soil and thoroughly mixing it with a suitable solution for a specified period – determined by the target metal and concentration. Physical processes separate the most highly contaminated soil particles, followed by chemical extraction of the heavy metals. This extraction step utilises specific washing reagents to transfer the heavy metals from the soil matrix into the solution (Wang *et al.*, 2020; Zhang *et al.*, 2022). Another method used for heavy metal remediation in SuDS is phytoremediation, where plants uptake the heavy metals through their roots (Teta & Hikwa, 2017; Shehata *et al.*, 2019; Bhat *et al.*, 2022; Lamine & Saunders, 2022). Dissolved metals are transported to the plant roots via cation exchange, mass flow, diffusion, and osmosis and subsequently absorbed by the roots within the filter media (Sharma *et al.*, 2021). The plants are then uprooted and appropriately disposed after a pre-determined time or when heavy metal concentrations are found to have sufficiently decreased (Gong *et al.*, 2018; Liu & Tran, 2021).

2.9 Chapter summary

MAR is a well-established technique for enhancing groundwater resources. The choice of water sources for MAR can significantly affect the health, cost, and performance of MAR schemes with various implications. The literature suggests that MAR technologies can effectively manage water resources, provided careful consideration and appropriate management of the key components and potential challenges are implemented. The concept and relevance of water-sensitive cities and SuDS in the context of South Africa was also investigated. An assessment of stormwater retrofitting was then conducted. It was found that implementing stormwater retrofitting guidelines, mainly from the US, UK, and Australia, requires adaptation to the South African context. An assessment of the role and importance of community engagement was conducted. It was found that traditional top-down engagement approaches that failed to address community needs or build trust were inadequate. The literature suggests participatory and inclusive approaches that engage with the community, promote knowledge exchange, and address the social, environmental, and economic dimensions. Furthermore, the reviewed studies suggest the necessity for further research on the role of community engagement in sustaining SuDS interventions in the Global South context. The monitoring and modelling of retrofits were then investigated, revealing the availability of software programs that can simulate and evaluate the functionality of SuDS. Finally, water quality and treatment considerations for stormwater harvesting via MAR were discussed.

3. Research Method

3.1 Overview

As mentioned in Chapter 1, Okedi (2019) proposed that stormwater harvesting (SWH) in Cape Town's existing detention ponds could theoretically be achieved through MAR using a spreading method. This approach would reduce the infrastructure costs associated with SWH. However, the viability of converting existing detention ponds in Cape Town for MAR has not yet been established. Therefore, this study used a literature-guided mixed-methods research philosophy that combined desktop studies, laboratory work, computational hydraulic simulations, field experiments, and unstructured participant observations to investigate the viability of retrofitting Cape Town's existing detention ponds for MAR. Figure 3-1 shows a flowchart of the methods used in this study.

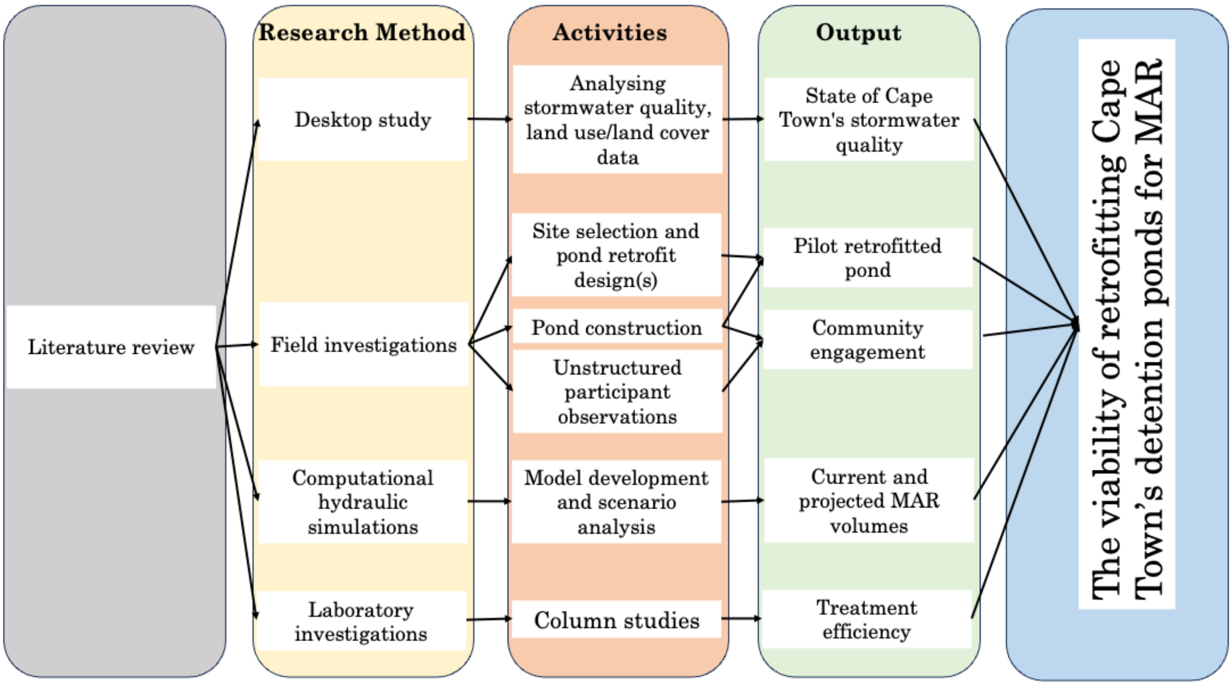

Figure 3-1: Research method flowchart

3.2 Desktop Study

A desktop study – presented in Chapter 5 – was conducted to determine the typical stormwater quality characteristics in Cape Town. The primary objective of this investigation was to compare stormwater quality in Cape Town's residential areas with the established water quality guidelines and stormwater quality observed in other developing and developed countries.

Representative water quality characteristics from four catchments in Cape Town over the period 2015 to 2020 were determined by conducting statistical analyses on the stormwater quality data from the catchment, combined with geospatial assessments of land use and land cover (LULC). The stormwater dataset encompassed monthly grab samples obtained from various sampling points in four catchments within the city. It was examined for six parameters: Ammonia, Dissolved oxygen, *E. coli*, TN, TP, and TSS. The outcomes of this assessment served as a basis for a comparative analysis between stormwater quality in CoCT residential areas and existing guidelines, as well as the quality observed in other countries – both developing and developed.

3.3 Field investigations

The principal requirements for SWH through MAR were determined by transforming a detention pond in a relatively deprived area of Cape Town. A site selection criterion was established based on insights from the literature review and considering the South African context. This criterion was used to identify potential sites that could be used to assess the feasibility of and requirements for retrofitting detention ponds in the CoCT for MAR. The detention pond in a residential area in Cape Town was then selected. A co-design process involving consultations with the local community was also conducted.

Retrofit designs were developed, and the pond was retrofitted to enhance its infiltration potential. The community participation exercise, pond design, and construction processes are discussed in Chapter 6. The stormwater and groundwater quality in the ponds were also evaluated. Supplementary data, such as stormwater inflow rates, groundwater levels, infiltration rates, and soil samples, were also collected from the pond and used in the hydraulic simulations and laboratory studies. The operation and maintenance requirements of the pond retrofit were also developed and evaluated. Unstructured participant observations were used to investigate the impact of the various interventions on the surrounding communities and to assess the role of communities in pond retrofits.

3.4 Hydraulic simulations

The nature and location of the study – described in Chapter 4 – meant that the geohydrological performance of the pond retrofits could be best appraised using hydraulic simulations. To that end, a calibrated and validated geohydrological model was developed using the PCSWMM software – Chapter 8. The model was used to run computational simulations to determine the retrofitted pond's short- and long-term performance and establish potential infiltration volumes from a range of rainfall events and durations. The model was also used to assess the factors influencing MAR volumes through sensitivity analysis and the impact of climate change under different climate change models. The influence of projected groundwater abstraction on MAR in a retrofitted pond was then investigated.

3.5 Laboratory investigations

The literature review highlighted a crucial concern related to the use of stormwater for MAR, namely, ensuring that the quality of the infiltrating water is not inferior to the existing groundwater within the aquifer. Employing stormwater for MAR introduces the potential risk of contamination. Therefore, evaluating the effectiveness of water quality treatment, specifically in terms of pollutant removal efficiency, is imperative when utilising in-situ materials (original soil) in retrofits instead of engineered materials (modified infiltration media optimised for enhanced stormwater treatment).

This study used laboratory investigations to understand the prospective treatment performance of retrofitted detention ponds within the study area. Controlled laboratory studies were conducted using three sets of 2 m tall PVC columns (each set consisting of two columns). Two sets were filled with soil from the Cape Flats Aquifer, whereas the other set was filled with clean silica sand as a control. Synthetic stormwater was applied to the columns to evaluate the performance of the retrofitted basins under varying contaminant loads and vadose-zone depths. The potential treatment performance of the retrofitted pond was inferred from the findings of these studies.

4. Description of the study area

4.1 Overview

The study was conducted in the City of Cape Town (CoCT) in the Western Cape province of South Africa (Figure 4-1). The CoCT is the second most populous city in South Africa with a population of approximately 4 600 000 in 2020 comprising more than 80% of the entire population of the province (Western Cape Government, 2020). It is also South Africa's legislative capital (CoCT, 2020). It is administered by the City of Cape Town Municipality ('the City') which is responsible for the provision and maintenance of water and sanitation services, including stormwater management.

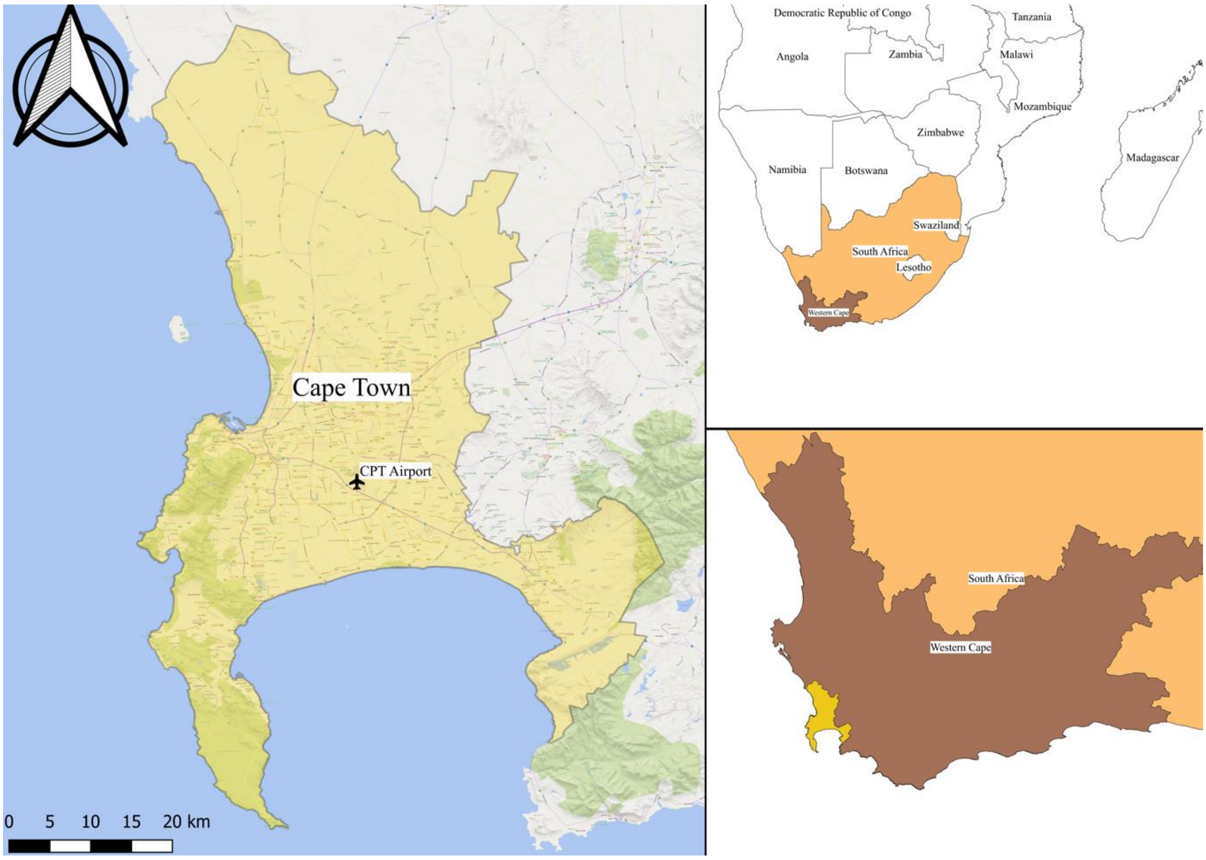

Figure 4-1: The City of Cape Town's location
(Image source: Google, 2022)

Cape Town is internationally renowned for the Table Mountain range and scenic beaches; however, it is also characterised by vast inequality – with a Gini coefficient of 0.62 (Western Cape Government, 2020). An example of this inequality is displayed in Figure 4-2 where the more affluent suburb of Penzance Estate and an informal settlement (Imizamo Yethu) are separated by a strip of land less than 120 m wide (yellow line in Figure 4-2).

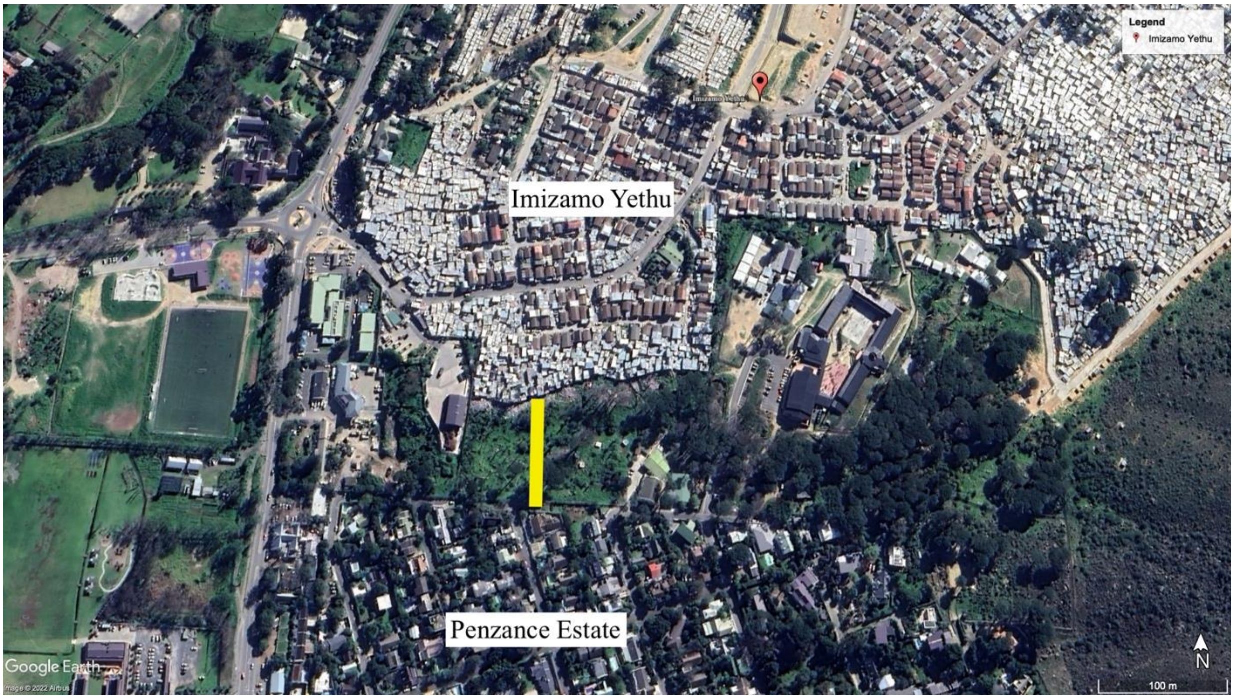

Figure 4-2: Imizamo Yethu informal settlement and Penzance Estate
(Image source: Google, 2022)

4.2 Cape Town's climate

The CoCT has a Mediterranean climate with warm dry summers (October-January) and cold wet winters (May-August) and a mean annual precipitation (MAP) of 619 mm with a range of 229 mm – 1037 mm (Adelana *et al.,* 2010; Ziervogel *et al.*, 2016; LaVanchy *et al.*, 2019; Olivier & Xu, 2019). The pronounced rainfall variability is a result of the topographical variations in the city with the mountainous regions enveloping the relatively level centre of the city called the Cape Flats (Figure 4-3) (Enqvist & Ziervogel, 2019). The CoCT – like much of South Africa – is a water scarce city whose scarcity is projected to increase due to climate change (Donnenfeld *et al.*, 2018).

Figure 4-3: Extent of the Cape Flats (area bounded by yellow line)
(Base image courtesy SRTM Team NASA/JPL/NIMA (2004))

4.3 Cape Towns' water supply and demand

Water use in South Africa is regulated by the Department of Water and Sanitation (DWS) (Enqvist & Ziervogel, 2019). Six major surface reservoirs provide the bulk of CoCT's water supply. The national government owns three of them – including the largest, Theewaterskloof – while the City owns the rest (CoCT, 2019a). The water from the dams is allocated to the Western Cape's agricultural sector, small towns and the CoCT – which receives around 69% of the total volume (Taing *et al.*, 2019). In 2019 the City derived 96% of its water from open water sources and 4% from groundwater (CoCT, 2019a). Increasingly erratic rainfall patterns and an increase in population growth rate mean the water supply from the existing water sources will likely decrease while the overall water demand will increase (Hedden & Cilliers, 2014). This mismatch will exacerbate the current levels of water scarcity in the City.

In July 2020, the City's target water demand was 650 ML/day, while the actual demand fluctuated because of varying consumer behaviour (CoCT, 2020a). The water demand is considered low for a metropolitan tourist city and is partly due to measures taken during the 2004-2005 and 2015-2018 droughts. These measures resulted in various water restrictions in the City and a paradigm shift regarding water demand and supply (Carden *et al.*, 2009; Parks & Mclaren, 2019; Taing *et al.*, 2019). However, these measures seem to have lost efficacy as the CoCT's water demand increased to 945 ML/day in March 2023, with a new target demand of 850 ML/day although the increase can also be attributed to population growth (CoCT, 2023).

The previous droughts, particularly the 2015 – 2018 drought, determined as a 1-in-590-year drought (CoCT, 2019), have resulted in the City pivoting towards being a more sustainable and water-sensitive city. This move has encouraged the exploration of various alternative water sources (CoCT, 2019). The City now plans to increase its level of water assurance from 98% to 99.5%. This means the City aims to theoretically have only one year in 200 where demand will not be fully met, an improvement from the current 1 in the 50-year level of assurance. To that end, the CoCT intends to: augment and protect its existing water supply by constructing more dams, increasing its use and regulation of groundwater, increasing water reuse, increasing the quantity of desalinated water, and implementing stormwater harvesting (Gosling, 2018; CoCT, 2019).

The CoCT and the DWS recognise that the increased groundwater use will require management and, to that end, the DWS has placed licensing restrictions on its use. The CoCT intends, *inter alia*, to augment the resource by injecting treated wastewater effluent into the CFA for later abstraction, treatment, and use (Umvoto, 2018; CoCT, 2019).

4.4 Alternative water sources for the City of Cape Town

The CoCT is committed to transitioning into a water-resilient city that recognises water scarcity is the 'new normal'. The CoCT thus supports the use of alternative water systems and has provided guidelines for such systems, such as greywater reuse, groundwater, treated effluent and rainwater (CoCT, 2010, 2018). These identified alternatives are regulated because they pose potential health and environmental risks (Johnstone & Manie, 2017).

The end-use of water from alternative sources is determined by the water quality (CoCT, 2018a). However, guidelines relating to the use of stormwater and seawater have yet to be developed (Rushmere & Mackay, 2017; Gosling, 2018). The feasibility of using stormwater and groundwater in the CoCT will be briefly discussed below as they relate to this study.

4.4.1 Stormwater

The CoCT receives an estimated mean annual storm runoff volume of 1200 GL, translating to ~2000 ML/day in a dry year (Armitage, 2020), far exceeding the reported target water demand of 850 ML/day (CoCT, 2023). A growing proportion of the rainwater the CoCT receives ends up as roof runoff intercepted into rainwater tanks (rainwater harvesting), while some is lost as evapotranspiration. However, most storm runoff is not captured (SWH) and is presently conveyed to the sea, meaning the resource is underutilised (Gosling, 2018).

While stormwater is already utilised in the City at the AWRMS (Section 2.1.3.1), this scheme remains the only documented and operational stormwater harvesting scheme. Fisher-Jeffes's (2015) study on the Liesbeek River catchment within the CoCT investigated the viability of stormwater harvesting in a South African urban context. This was done by modelling several scenarios using the Stormwater Management model developed for the study. Fisher-Jeffes (2015) found, amongst other things, that rainwater harvesting (RWH) and stormwater harvesting (SWH) can potentially reduce the stormwater runoff peak by 20 to 26% while reducing the potable water demand by up to 20%. However, this would only be viable if there were a high-level adoption of stormwater for residents' non-potable uses. SWH also offer additional benefits to the surrounding community such as: amenities, improved water quality, and biodiversity.

4.4.2 Groundwater

The DWS regulates groundwater use through the National Water Act (1998), while the CoCT regulates it through the 2018 amendment of the Water By-Law (CoCT, 2018c). The CoCT is utilising and exploring further groundwater use from the Atlantis, Table Mountain Group and Cape Flats aquifers (Olivier & Xu, 2019). The last two are augmentation schemes that involve drilling several boreholes and pumping the water to central distribution centres (Umvoto, 2018). The City has seen increased public and private groundwater use due to the 2015-2018 water crisis (Luker & Harris, 2019). However, the groundwater resource is finite and faces the risk of overexploitation and contamination via seawater intrusion. Parsons (2019) analysed historical borehole data from the DWS and concluded that the aquifers in and around Cape Town do not face an immediate threat or show signs of overuse. Nevertheless, aquifers still need to be appropriately managed. Aquifer management involves regulating abstraction rates and managing and encouraging the aquifer recharge. The CoCT's aquifer management plan involves increasing groundwater recharge in the City using treated effluent for infiltration and borehole injection, i.e., MAR (CoCT, 2019) particularly in the Atlantis aquifer and the CFA (CoCT, 2019).

4.5 The Cape Flats Aquifer location and geohydrology

The CFA is an unconfined, sandy aquifer that extends over ~ 630 km^2 – 29% of the Cape Town area (Adelana, 2010) with a varied terrain that ranges between 0 – 110 m above mean sea level (Department of Water Affairs, 2008). Adelana *et al.* (2010) demarcated the general extent of the CFA as the region bounded by three railway lines (Cape Town to Muizenberg, Cape Town-Bellville-Kraaifontein, Bellville-Eerste-River-Strand) and the False Bay coast. This boundary has since been accepted as the general extent of the CFA found in studies such as Hay *et al.* (2015); Seyler *et al.* (2016); Mauck (2017); and Gintamo *et al.* (2021) (Figure 4-4).

Figure 4-4: The CFA boundary

The CFA geology comprises of quaternary sands overlying the weathered Malmesbury formation, Table Mountain sandstone and Cape granite basement (Schalke, 1973; Adelana, 2010). The quaternary deposits consist of five formations: Elandsfontein, Langebaan, Springfontyn, Velddrift, Varswater and Witzand (Hay *et al.*, 2015). Some parts of the aquifer contain unconsolidated and semi consolidated sands, clay, calcareous layers, and embedded peat, resulting in semiconfined aquifers and perched water tables in those areas (Adelana, 2010; Gxokwe *et al.*, 2020).

The CFA has been long recognised as a vital groundwater resource for the CoCT and has a capacity of ~ 600 Mm^3 (Mauck, 2017; CoCT, 2019b; Gintamo *et al.*, 2021). The aquifer is a comparatively shallow and variable aquifer system 40 – 55 m thick (Hay *et al.*, 2015). The mean water level across the aquifer has a seasonal variation of 1 – 1.5 m (Seyler *et al.*, 2016). The CFA's drainage area includes five major urban river systems, which all discharge into the ocean. Three of the five rivers (Kuils, Elsieskraal and the Lotus) are canalised and lined with concrete (Department of Water Affairs, 2008).

The groundwater flow is primarily determined by the underlying bedrock and topology that influence groundwater elevation and, ultimately, the flow direction (Adelana *et al.*, 2010; Mauck, 2017; Gxokwe *et al.*, 2020). Groundwater generally tends to flow from the Belville region towards False Bay in the south (Hay *et al.*, 2015).

The aquifer is a primary source of irrigation water for the Philippi Horticulture Area (PHA) and has dispersed private use with borehole records maintained by the Department of Water and Sanitation and the CoCT (Rust, 1991; Department of Water Affairs, 2008). The groundwater quality is often assumed to be of poor quality due to various informal settlements spread across the aquifer's area with poor to no access to formal sanitation services. The unconfined aquifer is thus susceptible to point contamination due to the anthropogenic activities within its boundary.

4.6 The potential for SWH in Cape Town

This section undertakes an evaluation of the potential for stormwater harvesting in the CoCT based on two primary considerations: (1) the present need for alternative and sustainable water supply sources in the CoCT, as outlined in the City's 20-year water strategy (CoCT, 2019); and (2) the potential viability of SWH via MAR as a means of resource augmentation in the CoCT, as indicated by several studies (Fisher-Jeffes *et al.*, 2017; Mauck, 2017; Okedi, 2019; Gxokwe *et al.*, 2020).

In Section 4.3, it was highlighted that groundwater storage would likely decrease due to increased demand and abstraction rates. Furthermore, groundwater recharge is also likely to decrease without intervention due to climate change and the growth in urban settlements, resulting in a simultaneous decrease in rainfall and infiltration area, respectively. On the other hand, urbanisation tends to increase stormwater runoff (Jakeman *et al*., 2016), which can be harvested and utilised for other uses. The CoCT considers SWH an opportunity for the City (CoCT, 2019).

One major challenge in SWH is providing long-term storage, as storms are intermittent, and large volumes of stormwater must be stored for SWH to be viable. This challenge is appreciable in already built-up urban areas where land is both scarce and expensive. Open storage and MAR have been proposed as possible stormwater harvesting and storage techniques in the CoCT (Armitage *et al*., 2014; Fisher-Jeffes, 2015; Okedi, 2019). Rohrer & Armitage (2017) demonstrated that in the South African context, SWH in Cape Town through real-time control (RTC) of the existing stormwater system is viable.

Mauck (2017) established that there was a capacity for MAR in the CFA, which spreads over 630 km^2 (29% of the Cape Town area) using a MIKE model. He evaluated how MAR could be incorporated into Water Sensitive Design (WSD) for the CoCT whilst simultaneously seeking to address the flooding issues that are common in mainly informal areas found within the aquifer boundary. The best scenario involved recharging the aquifer in the winter months and using recharged water in the summer months, which allowed for greater storage. The recommended abstraction rates ranged from 3 L/s to 5 L/s per borehole, which would substantially lower the water table and reduce the probability of flooding exceedance. He evaluated that the MAR and abstraction exercise would result in a sustainable yield of 18 Mm^3/year as just two sites located in Philippi and Mitchells Plain. The CFA has a capacity of 600 Mm^3 (Mauck, 2017; CoCT, 2018), suggesting that a higher yield from the CFA can potentially be realised with more MAR schemes over the aquifer. Mauck (2017) recommended using injection wells over infiltration ponds for MAR as it would result in higher yields; however, it would also require the pre-treatment of the stormwater, thereby increasing the operation cost. He also claimed that there was insufficient land for infiltration basins but overlooked the readily available detention ponds noted by Rohrer & Armitage (2017). Of interest to this study is Mauck's evaluation of the water table in the Mitchells Plain area where he noted that MAR over this area would likely have lower yields due to a reduced aquifer thickness that results in higher water table levels. This finding was confirmed by Gxokwe *et al*. (2020), who suggested that any practical MAR schemes over this region of the aquifer would require that water be pumped during the summer months, as simulated in the Mauck study.

In a subsequent study, Okedi (2019) established, through a hydraulic model of the Zeekoe Catchment covering 89 km^2 of the city's 2460 km^2 area, that SWH was a volumetrically and economically viable way to augment the water supply in the CoCT. He evaluated both RTC and MAR as stormwater storage techniques in the catchment. The catchment used in the study is situated over a deep, unconfined, sandy aquifer (CFA). Okedi (2019) found that the best method to store harvested stormwater in this instance is MAR, which could result in a 30% increase (9 – 12 Mm^3/year) in the groundwater resource in the catchment. He postulated that the substantial MAR could occur using some of the City's 800+ ponds, particularly the detention ponds.

The review conducted in Section 2.1.2 indicated that one of the necessary components for a successful MAR project is an appreciation of the water quality to be used to mitigate health and environmental risks (Jiang *et al*., 2015). South Africa's stormwater is known to be of poor quality as it contains high concentrations of pollutants (Knight, 2017; Okedi, 2019). A study on the runoff characteristics from a Cape Town freeway (R300) found that all but one of the parameters assessed exceeded the South African effluent quality guidelines (Robertson *et al*., 2019). These pollutants can be attributed to various point and non-point sources due to varying anthropogenic activities, *viz.,* urban farming, residential activities, industrial waste disposal and car tyre wear. The prevalence of informal settlements that often have poor or no access to formal sanitation systems also results in highly polluted storm runoff (Fell, 2017).

Studies have shown that the most common pollutants in urban stormwater are litter, suspended solids that transport heavy metals, hydrocarbons and volatile organic compounds from tyre wear and car oil leaks, nutrients, and microbial pathogens (Boogaard *et al*., 2014; Sauvé & Desrosiers, 2014; Osman *et al*., 2019; Feng *et al*., 2022; He *et al*., 2022). Contaminants of emerging concern (previously undetected and unregulated chemicals) and steroids have also been found in urban storm runoff (Ellis, 2000; Murphy *et al*., 2014; Hwang *et al*., 2016; Knight, 2017). Due to the potentially poor quality of South Africa's stormwater, MAR projects may require case-specific designs different from the typical approaches in more developed nations, which assume relatively better stormwater quality.

4.7 The South African socioeconomic context

The exclusionary pre-1994 policies are still socially and economically affecting post-Apartheid South Africa. The economic struggles are exacerbated by the nation's status as one of the most financially unequal countries globally, with a Gini coefficient of 0.67 – most nations outside southern Africa have a Gini coefficient below 0.5 (StatsSA, 2019; Díaz Pabón *et al.*, 2021). South Africa's inequality, coupled with economic instability and a slow recovery from the 2007/08 global financial crash, has led to an increase in unemployment which further increased from 29.1% in 2019 to 32.6% in the first quarter of 2021 due to the Covid-19 pandemic (StatsSA, 2019, 2021). Furthermore, campaign promises of transformation and subsidised housing made during the transition to a post-Apartheid South Africa are yet to be realised, with jobs, housing, and basic sanitation still inaccessible to a significant portion of the population (Bradlow *et al.*, 2011; Parikh *et al.*, 2020). Consequently, the Gini coefficient has not significantly improved post-Apartheid – from 0.68 in 1994 to 0.67 in 2020 (Chatterjee *et al.*, 2020). This backdrop creates a hotbed for social disgruntlement, including distrust directed at the government and a high crime rate.

In the CoCT, this inequality is observed as soon one approaches the Cape Town International Airport and commutes from the airport towards the leafy green southern suburbs or the central business district. The city has a high crime rate crime and is South Africa's 'murder capital' – ranked the 10th most violent city globally in 2020 (Citizen council for public safety and criminal justice (CCPSCJ), 2020). Most violent crimes are concentrated in the gang strongholds, primarily in the Cape Flats area, which overlie the CFA (StatsSA, 2020).

The socio-economic challenges suggest that retrofitting existing ponds is not as straight forward as the retrofit steps presented in Table 2-3 as security becomes an important consideration in site selection. For instance, while 32.1% of Cape Town ponds are located within the CFA boundary (319 retention and detention ponds), 122 (38.1%) of these are also located in and around crime hotspots, as per the South African Police Service (2021) quarterly report on national crime statistics (Figure 4-5). The crime hotspots are defined in terms of the policing boundaries that are in the Western Cape's top 20 of the 151 precincts in five crime categories, namely: community reported serious crime, carjacking, murder, common robbery, and malicious property damage.

Working in these hotspots becomes precarious for both researchers and CoCT officials as there is a high likelihood of theft, carjacking and loss of monitoring equipment that would otherwise be secured and is often used in more developed nations.

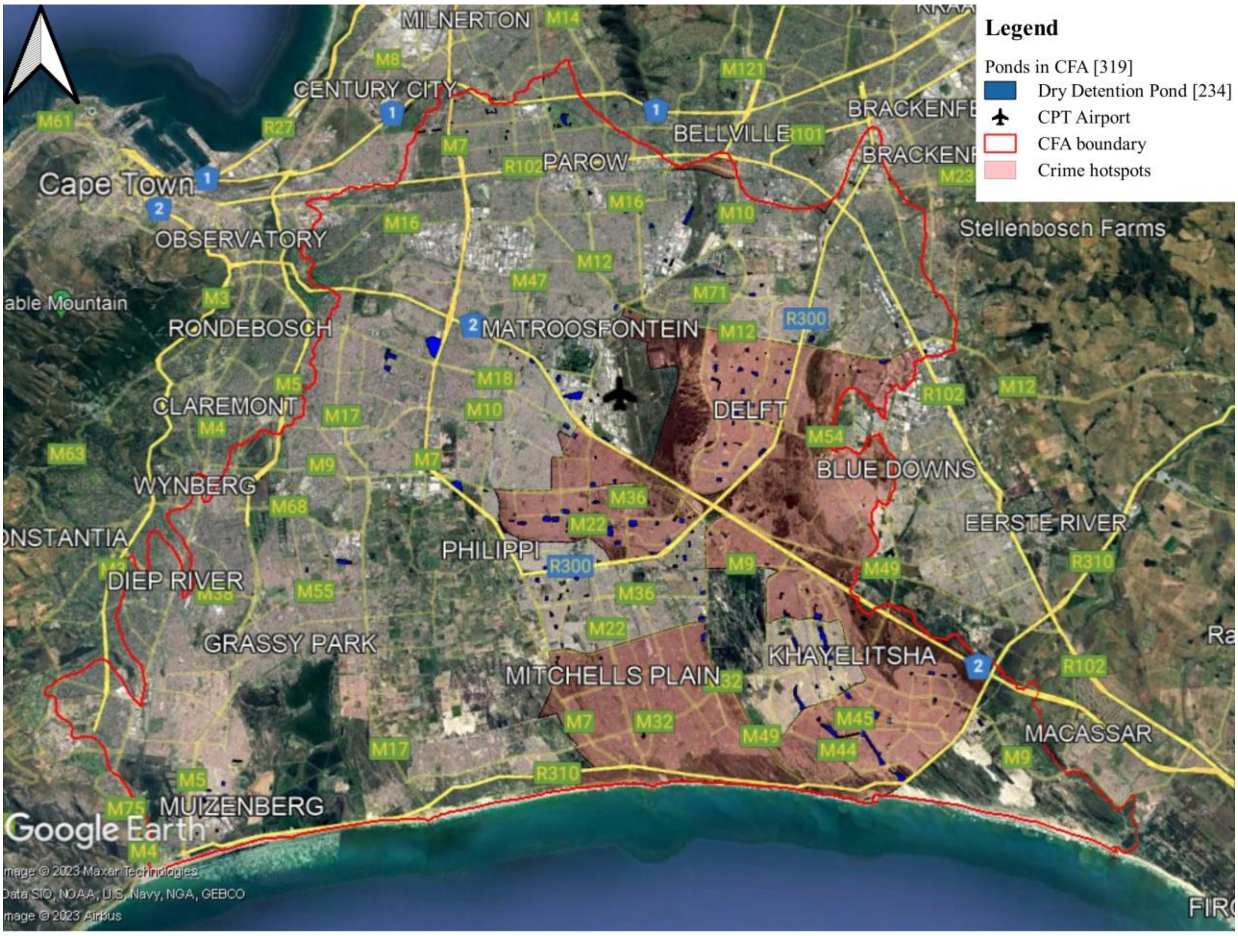

Figure 4-5: Crime hotspots within the CFA
(Map data: ©2021 Google, SAPS, 2021)

Another challenge is obtaining a community's informed consent to work in their area, often due to local competing interests driven by increasing competition for jobs or political ambitions, as highlighted by Adato *et al.* (2005) and Emmett (2010). These socio-economic challenges are, however, not unique to Cape Town and are reported in other South African cities like Johannesburg (Mathee *et al.*, 2010; Sindall *et al.*, 2021).

4.8 Study site selection

This research aimed to explore the potential for urban MAR in a retrofitted stormwater detention pond. There are 319 ponds located over the CFA with 234 potentially retrofittable detention ponds (Figure 4-6). One of the 319 ponds in the CFA was selected as the case study for retrofitting detention ponds to facilitate MAR in this study. Conducting research in multiple ponds was not feasible due to the high costs associated with travel, chemical analysis, and construction, which exceeded the available budget. Therefore, focusing on a single pond provided a more feasible approach for the study. The pond was selected following a desktop study and site visits conducted with the CoCT officials and their consultants in 2019 to identify suitable detention ponds that could be retrofitted for infiltration purposes (Jones, 2019 – unpublished BSc (Eng.) report).

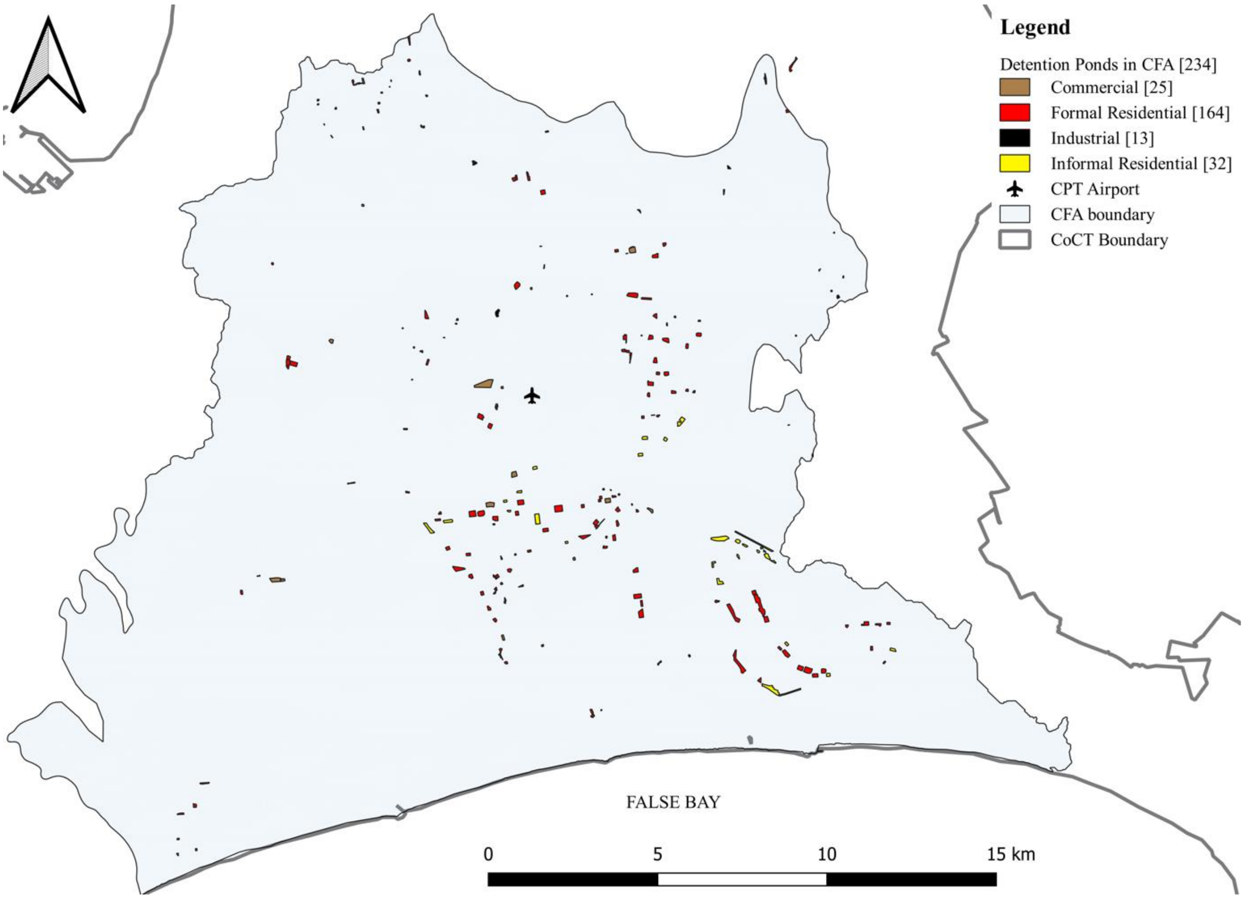

Figure 4-6: Detention ponds overlying the CFA
(CFA ponds shapefile courtesy of Jessica Fell, 2023)

The criteria used in the pond selection exercise included: location above a suitable unconfined aquifer; water table – high water tables would negatively impact infiltration; removed from existing groundwater extraction points (in case of contamination); and safety – many parts of the city have reported high levels of violent crime. The short-list of potential study ponds were then physically inspected, after which they were reduced to four (Figure 4-7).

A multi-criteria selection tool was then developed to identify which ponds would be best suited for MAR, building upon the requirements for a successful MAR project recommended by Dillon *et al.* (2009) but factoring in the South African socio-economic and political context. Eight elements were identified as essential for a successful stormwater-based urban MAR installation in South Africa: i) site geology and hydrogeology, ii) pond inlet and outlet positions, iii) access, and site security, iv) land-use, v) potential amenity and burden/risk to the surrounding community, vi) biodiversity, vii) land use and the viii) anticipated cost of the MAR retrofit. Each element was assigned a score out of 5.

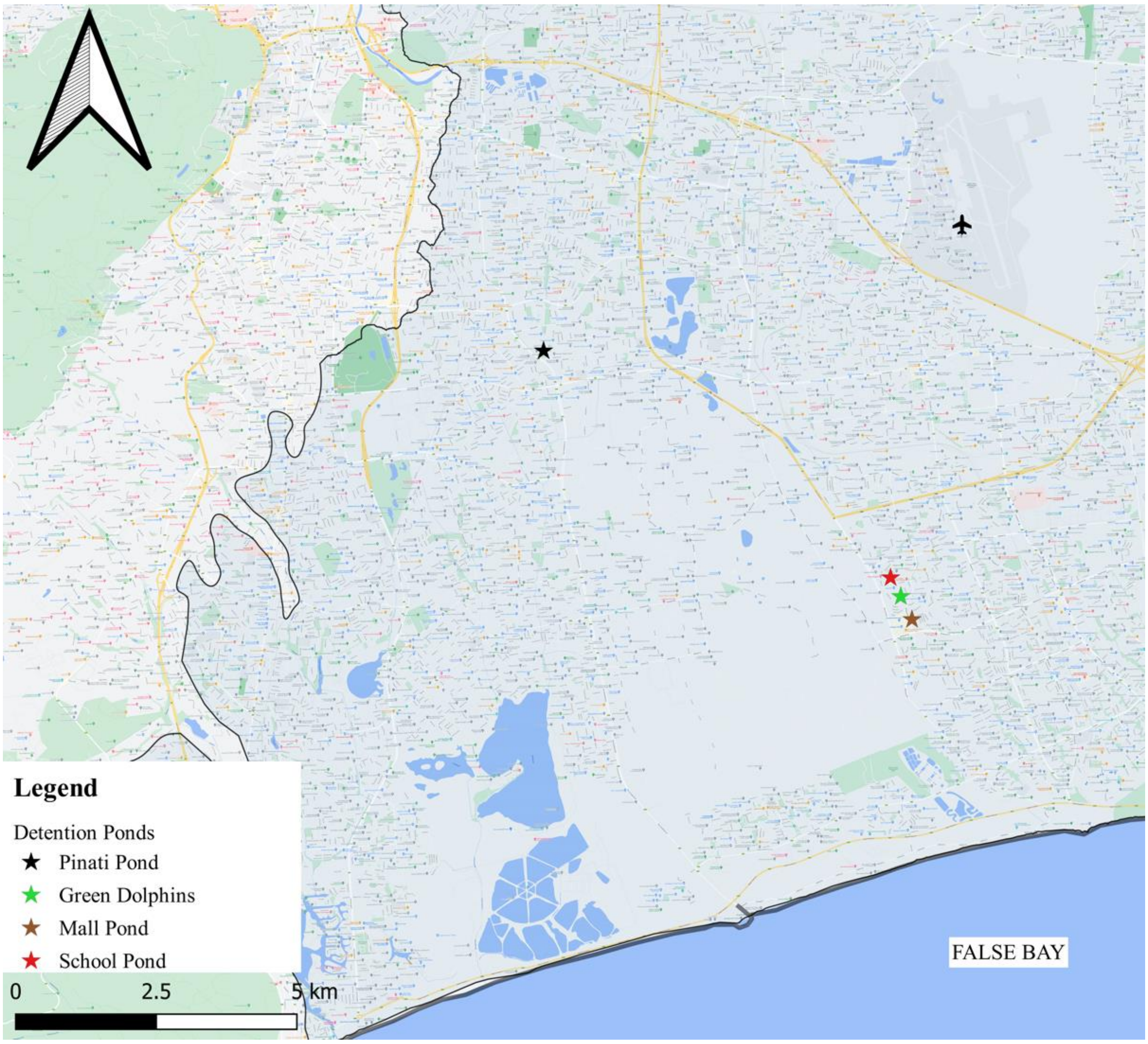

Figure 4-7: Ponds considered in study
(Image source: Google, 2023)

The evaluation of the candidate ponds included: site visits, the conducting of infiltration tests, and a desktop study of the hydrogeology data provided by the CoCT's groundwater consultants. A radar chart system was then used to identify the best suitable candidates for MAR (Figure 4-8). The Mall and Green Dolphins ponds performed adequately in most assessment categories; however, their inlets and outlets' positioning presented complexities for retrofit design. Hence, these ponds were considered unsuitable for MAR retrofitting. Likewise, the Pinati pond demonstrated favourable characteristics in several assessment criteria, but the presence of a very high-water table, located less than one metre below ground level, posed risks of groundwater contamination. Therefore, the Pinati pond was also deemed unsuitable for MAR. In contrast, the detention pond adjacent to a school, hence named the 'School Pond' showed the most favourable features and was thus the selected as the most suitable candidate for retrofitting.

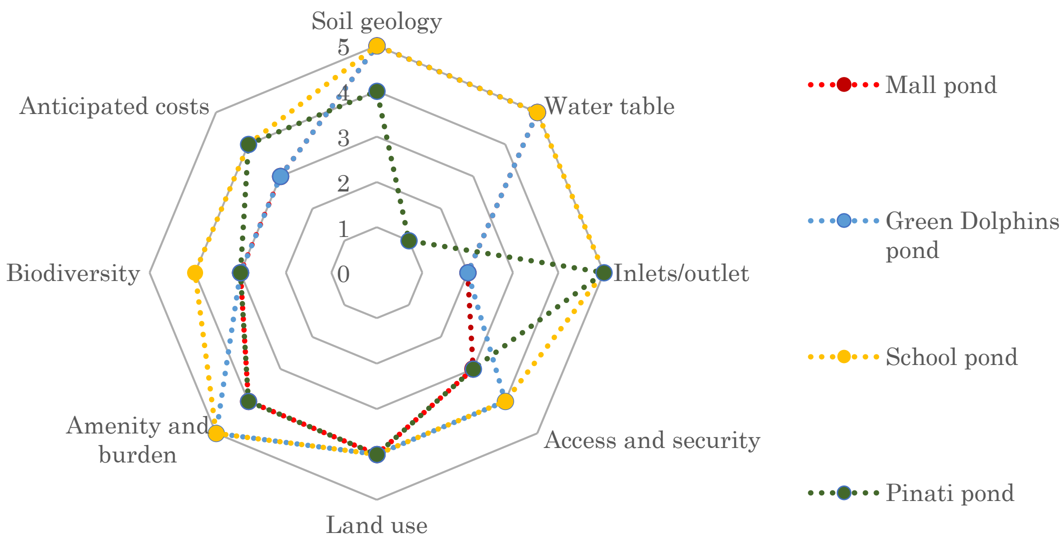

Figure 4-8: Radar chart for pond selection
(Jones, 2019 – unpublished BSc (Eng.) report)

4.9 The School Pond

The School Pond, situated in Rondevlei Park, Mitchells Plain, is a dry detention pond under the ownership and maintenance of the CoCT. It falls under the Mitchells Plain West hydrological catchment and overlays the CFA (Figure 4-9).

Figure 4-9: School Pond locality map
(Image source: Google, 2023)

The School Pond is on Fulham Road adjacent to The Leadership College, a privately run primary school. The pond was originally designed as a flood attenuation structure holding back runoff from storm events from the surrounding 170 600 m^2 catchment (Figure 4-9) and gradually releasing it into downstream storm conduits. The pond has a surface area of 9950 m^2, a maximum depth of ~2 m, two inlets and one outlet with trash regularly observed inside and around the pond.

The Mitchells Plain West catchment receives a mean annual rainfall of 530 mm – as measured from the Wolfgat Weather Station located 3.5 km away from the pond – mostly during the winter months of May to August (Figure 4-10).

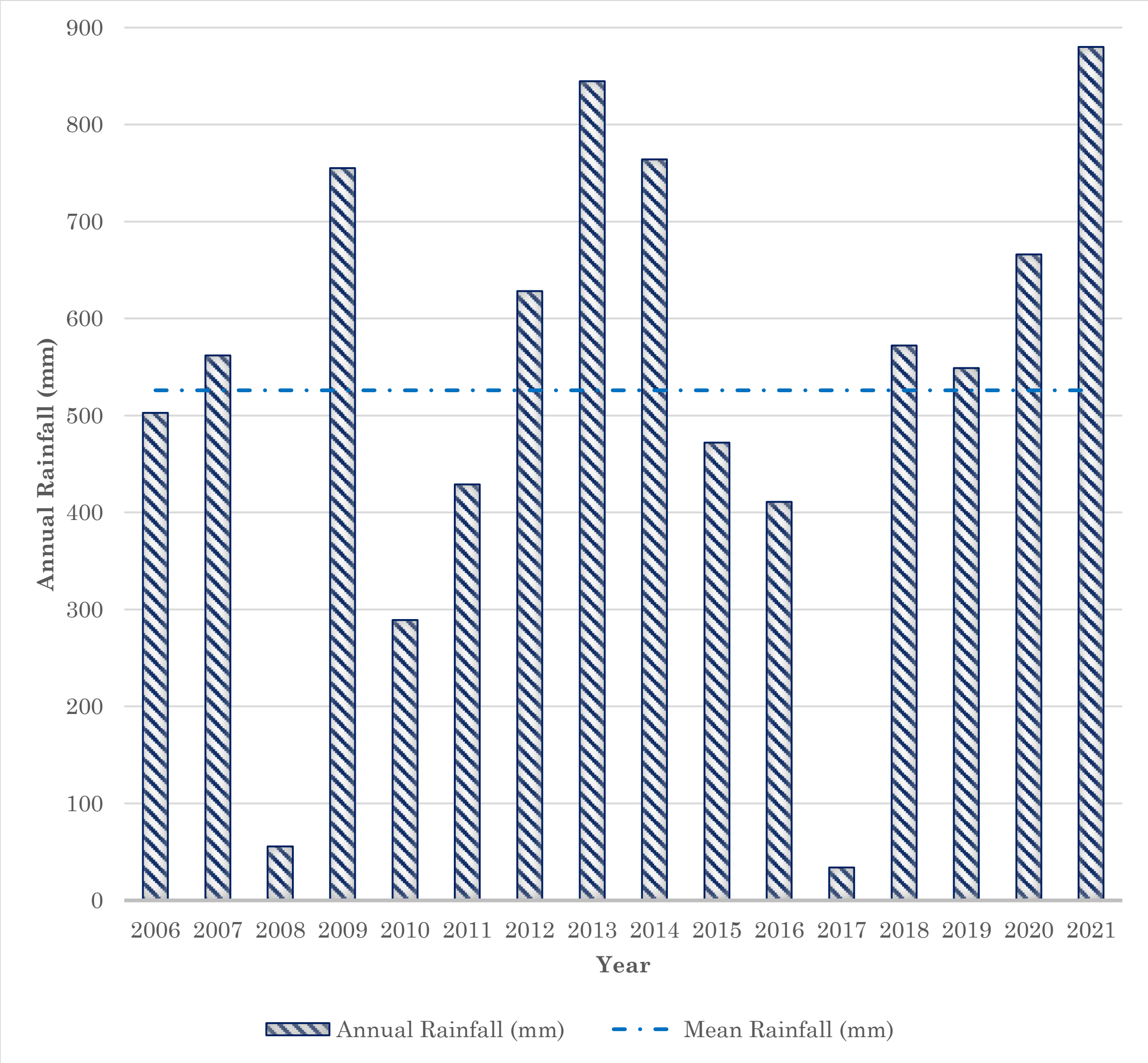

Figure 4-10: Historical rainfall data from Mitchells Plain West catchment

4.10 Subsurface information

One of the key considerations for MAR is the water table level – a high water table reduces the thickness of the unsaturated (vadose) zone, thus limiting the amount of water that can infiltrate. Furthermore, the vadose zone thickness influences the treatment capabilities of the infiltration basin, with a thicker zone producing improved treatment. The recommended minimum vadose thickness for infiltration basins using stormwater is 0.6 – 1.2 m (Conn *et al.*, 2010; NRWM, 2014; Hubner *et al.*, 2016). The vadose zone thickness should be considered during the highest rainfall period naturally recharge the aquifer and increase groundwater levels. This gives a 'worst-case' scenario.

Preliminary investigations by Jones (2019 – unpublished BSc (Eng.) report). suggested the water table below the School Pond was 3 - 4 m below the lowest point in the pond (elevation - 26.9 m above mean sea level (m a.m.s.l). The water table depth was derived by extrapolating the water table depth from a nearby monitoring well, MON-71 ~400 m from the School Pond (Figure 4-11). The topographical height of the water table was obtained by subtracting the depth to the water table from the topographical height of the top of MON-71. The resulting water table topographical height was then compared with the topographical height of the School Pond's lowest point from Google Earth data.

Figure 4-11: Location of CoCT monitoring well 71 (MON-71)
(Image source: Google, 2022)

The extrapolated groundwater elevation beneath the School Pond assumed that the School Pond has similar subsurface characteristics (soil type and soil profile) with MON-71 site. It was seen from the borehole logs from MON-71 that the subsurface around the area consists of medium sand to a depth of 16 m below ground level (mbgl) (elevation – 12.2 mamsl) and then breaks into medium to shells coarse sand with shells from 16 – 30 mbgl (Umvoto, 2020).

Three-hand augured observation wells were dug to establish the subsurface below the School Pond (Figure 4-12) in different locations on the 18th of February 2021. The trial wells produced varying soil profiles at similar intervals suggesting the subsurface was not homogeneous.

Figure 4-12: a) Hand augured observation well b) Soil profile from augured well

The water table in two trial wells was ~1 mbgl, while the water table at the lowest point in the pond was 0.5 mbgl. The water table depth variation within a 400 m distance between the trial wells led to a hypothesis that the pond might be lying over a calcrete layer that then formed a perched water table. A topographical survey of the School Pond site and MON-71 was then conducted. The topographical study and additional groundwater data from the CoCT's groundwater consultant revealed that the Jones (2019) study had erroneously calculated the water table level beneath the pond. The error resulted from a water level elevation calculation based on a different wells' (MON-72) water level depth but using MON-71's ground elevation – which was also initially wrong but corrected after the topographical survey. This meant that the water table in MON-71 at the time of observation was 25.7 mamsl and not 23.5 mamsl reported by Jones (2019). The water level in the SP was then 1.2 mbgl. This value agreed with the results from the site investigations conducted on the 18th of February 2021. Additionally, it was found that the water table in the School Pond was at least 0.6 m higher than the recorded level in MON-71 on similar dates. A further investigation of the pond's subsurface was then conducted on 18/06/2021 to better understand the sub-terrain. The investigation was carried out during the installation of monitoring wells in the pond. The wells were installed using mechanical and hand auguring with depths ranging from 1.5 m – 3.5 m. The wells are shallow wells that comprise three nested wells (each group has a shallow, medium, and deep well) and one individual 3.5 m deep well. Figure 4-13 shows the position of the 4 'deep' wells.

Figure 4-13: School Pond monitoring well positions
(Image source: Google, 2022)

The disturbed samples from the auguring were analysed at varying depths, resulting in a recreated soil profile (Table 4-1). The results from this hydrogeological study also showed a variable sub-terrain. However, the profile was limited to the monitoring well depth (max. 3.5 mbgl), and thus the presence of a perched water table could not be established.

Table 4-1: Recreated subsurface profile from 'deep' monitoring wells

	GWMW1D	GWMW2D	GWMW3D	GWMW4D
0 – 0.8 m	Medium sand	Medium sand	Clayey sand	Medium sand
0.8 – 1.5 m	Medium sand	Medium to fine sand	Sandy clay	Silty sand
1.5 – 2.5 m	Silty sand	Medium to fine sand	Clay sand with shell fragments up to 2.8 m	Medium sand
2.5 – 3.5 m	Medium sand with shell fragments	Medium sand	–	Medium sand

An Electrical Resistivity Survey (ERS) was then done by Cape Geophysics to investigate the subterrain beyond 3.5 m at the School Pond on the 9th of March 2022. An ERS produces a subsurface profile up to 24 m below the ground and reveals subsurface strata changes – such as an impervious formation typical of a perched water table. Three sections/traverses across the pond were chosen (Figure 4-14).

Figure 4-14: School Pond ERS traverses
(Image source: Google, 2022)

Traverse 1 had 400 mm long electrodes inserted in the ground at 4 m electrode spacing yielding an investigation depth of ~ 24 m. Traverses 2 and 3 were conducted with 2 m electrode spacing resulting in a maximum investigation depth of ~ 12 m. Measurement of the resistivity of the ground was carried out by transmitting a controlled current between two electrodes inserted in the ground while measuring the potential difference (voltage) between two other electrodes.

Figure 4-15 shows the survey equipment and an electrode connection. Electrical resistivity varies across distinct geological materials, predominantly due to fluctuations in groundwater content and dissolved ions. Consequently, resistivity surveys were used to discern areas with contrasting electrical characteristics that were subsequently denoted as distinct geological strata. Resistivity, also referred to as specific resistance, is the reciprocal of conductivity or specific conductance. Common minerals forming soils and rocks typically exhibit high resistivity when dry; thus, the resistivity of these materials is generally dependent on the quantity and water quality within the pores and fractures, along with the extent of tropical weathering in the formation. This variability often remains confined to specific geological regions and shifts in resistivity within soil or rock types represent fluctuations in their physical properties. Appendix B contains an in-depth description of the ERS procedure.

Figure 4-15: a) Abem SAS 1000 Terrameter and ES 10-64 switching unit on Traverse 1 b) An electrode connected to the main traverse line

The resistivity values were used to develop a subsurface model, and the logarithmic resistivity values were used to illustrate the resistivities (Figure 4-16).

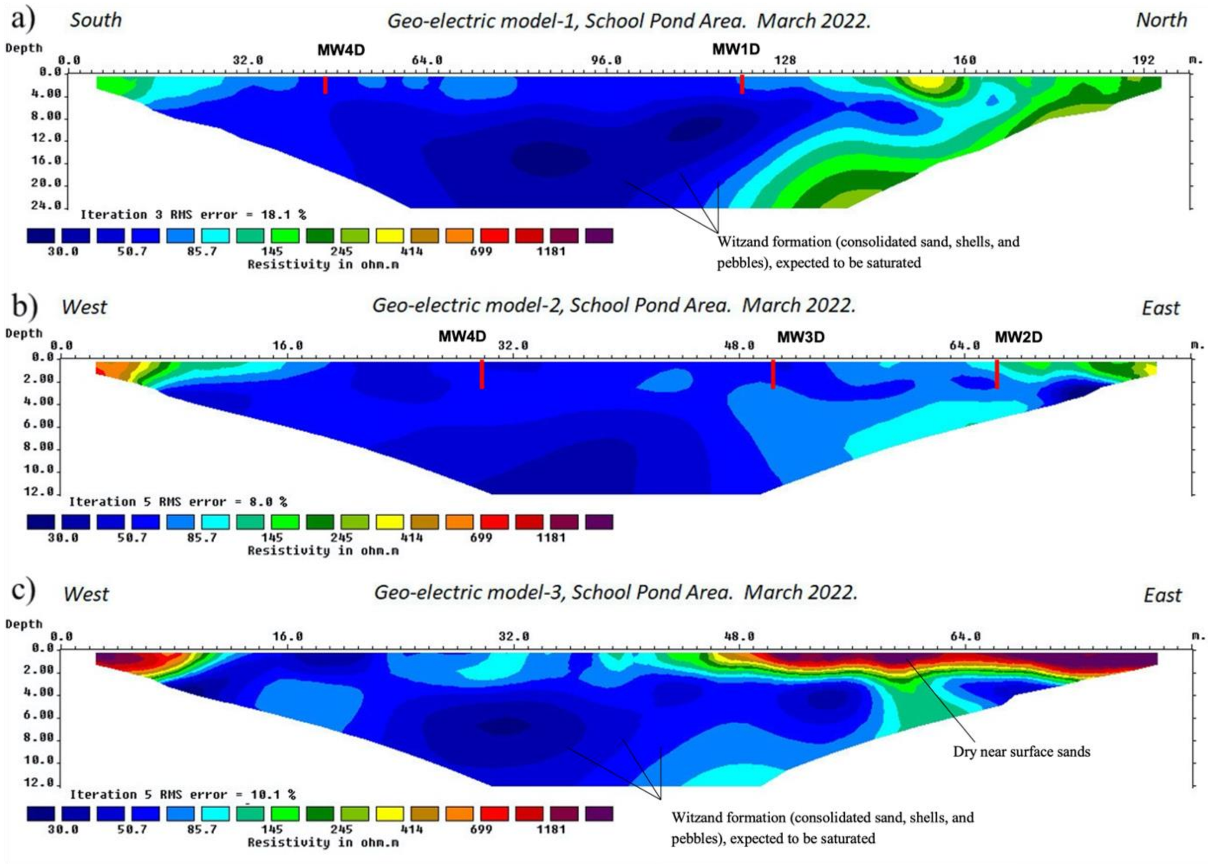

Figure 4-16: Geo-electric models of the School Pond subsurface; a) Traverse 1 b) Traverse 2 c) Traverse 3

The results from the subterrain model indicated that the strata below the pond form part of the Witzand Formation (a geological unit that forms part of the CFA – Section 4.5). There is no clear impervious stratum to suggest that a perched water table existed – for the depth investigated. The resistivity values associated with the edges of the pond were due to dry sand, which, compared to the resistivity of the wet vadose zone, has high resistivity. The magnitude of the resistivity values from 30 Ohm.m to 145 Ohm.m were so small that the strata were not easily distinguishable without the aid of borehole logs.

Incorporating the ERS with previous geohydrological investigations (Table 4-1) suggested that the water table variation, especially in the swampy area of the pond (green area in Figure 4-17), is due to the higher clay content in that area (in relation to the rest of the pond). The clay retains water better than sand, leading to higher water levels in that area which results in a swampy, waterlogged area. The swampy area was only discovered in later investigations in 2021 after the site selection process and had not been observed beforehand owing to the drought from 2015 to 2019.

Figure 4-17: Monitoring well locations in relation to the ERS traverses
(Image source: Google, 2022)

The presence of clay also results in high conductivity and lower resistivity compared to dry sand (clay better retains water which increases the conductivity and results in less resistance as explained by Ohm's Law), and this was seen in both the ERS, and the conductivity measurements recorded during groundwater sampling from the wells. The higher resistivity in the area below and between monitoring wells GWMW2D and GWMW3D was likely due to infiltrated stormwater – with lower conductivity than groundwater – which accumulates before spreading out.

Groundwater mounding was also considered as a possible reason for the observed water table level variations between the pond and MON-71 data, but this is unlikely as MON-71 is also next to a detention basin which would be expected to also have mounding (possibly to a larger extent due to it overlying medium sand up to 3 mbgl which is expected to better infiltrate stormwater the sand at the School Pond).

In conclusion, the differences in observed groundwater levels in the pond and MON-71 data are now understood to be a consequence of the high clay content under some parts of the S.P (particularly the swampy area). The clay retains water better than the mixed sand under the MON-71 location, resulting in a higher water table but not necessarily a perched water table in the 'traditional' sense. This finding also explains the presence of a swampy area within the pond throughout the year, except during very dry years. While the presence of a swampy area was suboptimum, it was found that the allowable vadose zone depths could still be attained around the elevated sections of the pond, specifically the area to the east of GWMW2D in Figure 4-17. Additionally, the pond's proximity to a school made it more appealing for this study, as the School served as a convenient entry point for the community engagement.

4.11 Chapter summary

The CoCT has experienced two significant droughts in the past two decades, underscoring the impact of escalating regional water scarcity. The CoCT has proposed water augmentation programs to secure its water supply, including increased groundwater abstraction. However, sustainable groundwater use necessitates the management of abstraction rates and replenishing aquifers through recharge methods. SWH at the catchment level has been suggested as a viable approach to augment the CoCT's water supply to address this challenge. Multiple studies have proposed MAR as a potential storage method for harvested stormwater. Okedi (2019) suggested that MAR could be implemented in existing detention ponds through a spreading method, potentially reducing the infrastructure costs of SWH. However, no specific approach to retrofitting the CoCT's detention ponds for MAR has been developed and evaluated. Therefore, this study proposed a pilot study to develop and evaluate a practical approach for SWH via MAR in existing detention ponds in the CoCT. A detention pond located in Rondevlei Park, Mitchells Plain, Cape Town, South Africa, was selected as the case study site. A minimum water table depth of 0.6 to 1.2 m is recommended to implement infiltration basins, which can be achieved in specific sections of the pond.

5. The state of Cape Town's stormwater quality

5.1 Overview

The typical characteristics of Cape Town's stormwater quality were established using historical stormwater data grab sampled monthly over five years (2015-2020) by the CoCT. Four catchments covering 45% of the CoCT total area (Table 5-1) were investigated, namely the Diep River, Sand River, Salt River and Zeekoe catchments (Figure 5-1). The catchments have varying numbers of sampling points spread over each catchment.

Table 5-1: Summary of analysed catchments

Catchment	Area (km^2)	No. of sampling points
Diep River	726	14
Salt River	214	18
Sand River	92	23
Zeekoe	89	14

Stormwater characteristics of the four catchments were assessed using the same method. This chapter presents findings from the Salt River and Zeekoe catchments, which overlie the CFA and include 102 dry detention ponds (out of 243 in the CFA), which is of greater importance to this study. The findings from the other catchments are presented in Appendix C.

Coordinates of the sampling points were obtained from the CoCT database uploaded to a geographic information system, in this case (QGIS). Vector data in the form of land-use and land-cover shapefiles for the catchments were obtained from the South African National Land Cover Database (Thompson, 2019). The catchments' bare earth 10 m digital elevation models (DEM) were also obtained from the University of Cape Town's GIS department. The DEMs were used to delineate the sub-catchments and identify the areas that contributed to each sub-catchment stormwater flow and the associated sampling point. The correlation between land use and stormwater quality parameters was analysed in Microsoft Excel using descriptive statistics to assess stormwater quality and correlation analysis between the stormwater quality parameters and land use. The association between land use and stormwater quality was established by connecting data from sampling points to their corresponding land use categories. Pearson's correlation coefficients were used to assess the strength of the relationships (R-values) considering a significance level of $p < 0.05$.

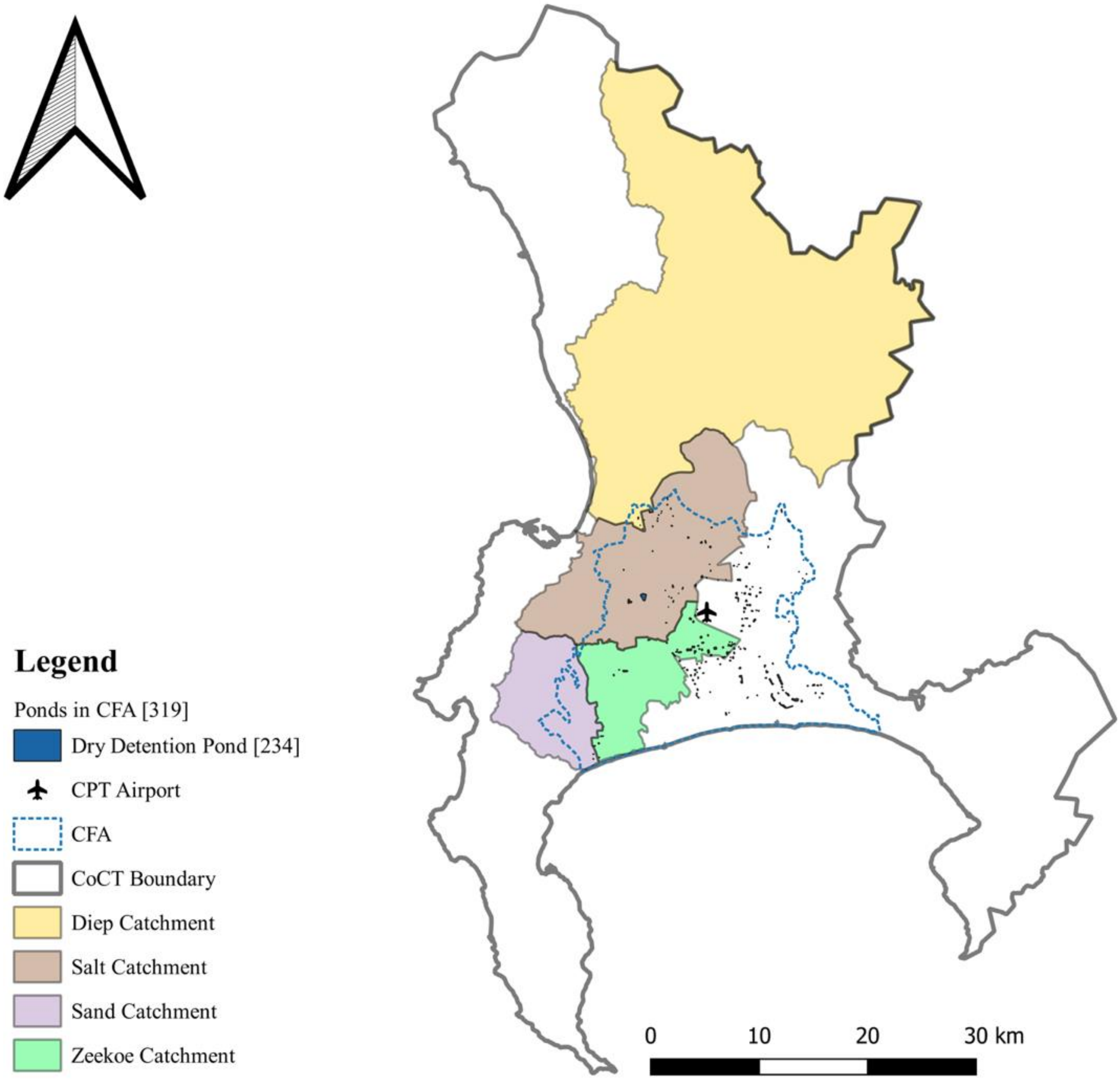

Figure 5-1: Investigated catchments

The nine stormwater parameters' mean, median, and 95th percentile values from all sampling points within the catchment were calculated. The stormwater quality was then compared with the existing national guidelines for recreational use and data from a South African study (DWAF, 1996c; Nel *et al.*, 2013) (Table 5-2).

Table 5-2: South African guidelines for recreational use

Parameter	SA Quality Guideline (DWAF, 1996c; Nel *et al.*, 2013)				
	Natural	Good/ Target	Fair	Poor/ Risk	Unacceptable
Ammonia (mg/L)	< 0.015	0.015 - 0.058	0.058 - 0.1	0.1 - 0.2	> 0.2
DO (mg/L)	> 8	8 - 6	6 - 4	4 - 2	< 2
E. Coli (/100mL)	–	0 - 130	131 - 200	201 - 400	> 400
pH	8 - 6.5	9 - 8	10	> 10	
Temperature (°C)	–	5 - 30	–	–	–
TN (mg/L)	< 0.25	0.25 - 1	1 - 4	4 - 10	> 10
TP (mg/L)	< 0.005	0.005 - 0.025	0.025 - 0.125	0.125 - 0.25	> 0.25

Stormwater quality data from other countries were also compared to provide a broader context. The mean stormwater parameters from residential areas in India, Singapore, Malaysia, the USA, and the Netherlands were examined in this general analysis (Table 5-3). The decision to employ mean values for comparison instead of the median or the 95th percentile was driven by the availability of publicly accessible data sets from the developing nations under consideration (India, Singapore, and Malaysia), which were predominantly presented in terms of means or ranges. Despite mean values being generally regarded as suboptimal for statistical inference in assessing stormwater parameters due to their inherent variability and non-normal distribution, this compromise was made due to data limitations to enable some level of comparison.

Table 5-3: Mean stormwater characteristics for residential land use in selected countries

Parameters	**India** [a]	**Singapore** [b]	**Malaysia** [c]	**USA** [d]	**The Netherlands** [e]
Ammonia (mg/L)	–	–	0.14	0.3	–
E. coli (CFU/100 mL)	4.44E+06	–	–	7.00E+02	1.90E+04
TN (mg/L)	9.2	1.2	–	1.4	1.9
TP (mg/L)	0.7	0.1	0.73	–	0.4
TSS (mg/L)	27.8	31.9	37	49.0	17

– no data

[a] Arora & Reddy (2013), [b] Song *et al.* (2019), [c] Chow *et al.* (2013), [d] Pitt *et al.* (2004), [e] Boogaard *et al.* (2014

5.2 The Salt River and Zeekoe catchments

Located in central Cape Town, the Salt River catchment extends over ~ 214 km^2 of Cape Town's area from the Tygerberg Hills, through several northern and central suburbs of Cape Town, to the eastern slopes of the Table Mountain. Runoff from the catchment discharges through the Salt River Canal into Table Bay. The main tributaries are the Elsieskraal, Vygekraal, Kromboom, Black, and Liesbeek Rivers (Figure 5-2).

The Salt River catchment has 34 sampling points distributed over the catchment (Figure 5-2). The catchment's land use comprises: residential – formal and informal – (32%), industrial (29%), commercial (14%), natural vegetation (19%), wetlands (2%) and barren land (2%). Other land uses, such as water bodies, agricultural mining, and parks, cover 3% of the catchment area. The Salt River catchment area contains approximately 46 dry detention ponds.

The Zeekoee Catchment covers 89 km^2 and is situated in the south-central part of Cape Town, adjacent to the Salt River catchment to the east and the Sand River catchment to the north (Figure 5-1). Within the catchment is Zeekoevlei, a lake fed by three streams: the outlet from Rondevlei (a small shallow lake), Little Lotus River, and Great Lotus River. The Big Lotus River is an 18 km canal that was constructed to drain Cape Town International Airport and flows through the townships of Crossroads, Nyanga, and the Philippi horticultural area, Ottery, and Grassy Park before discharging into Zeekoevlei (Figure 5-3).

The Zeekoe catchment has 14 sampling points (Figure 5-3). The catchment's land-use composition is ~33% formal residential, 33% natural vegetated areas, 16% agricultural and 6% informal residential, while commercial areas, industrial areas, barren land, and parks comprise the rest of the catchment. There are approximately 56 dry detention ponds in the catchment.

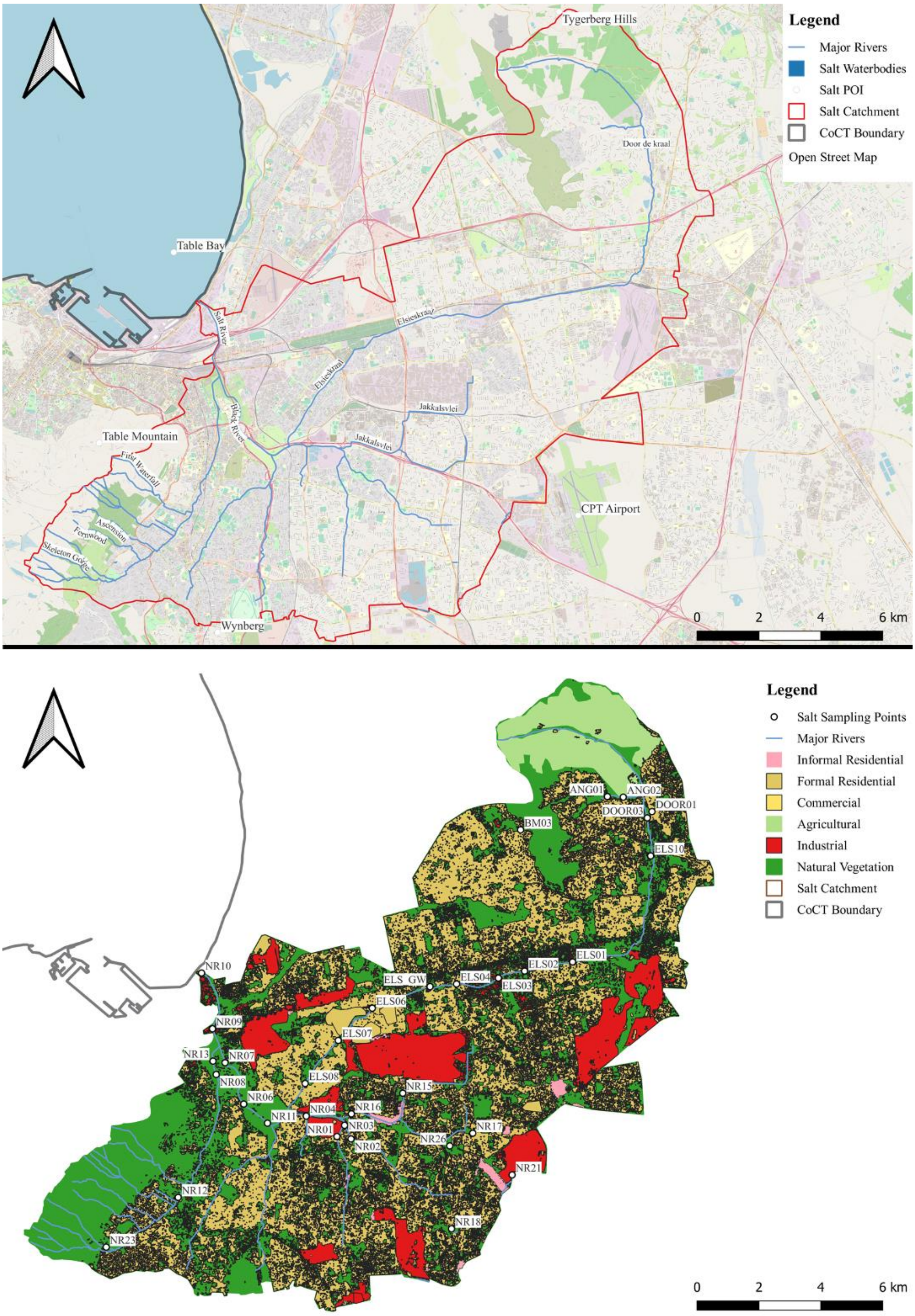

Figure 5-2: Salt River catchment locality plan (top); LULC composition and sampling points (bottom)

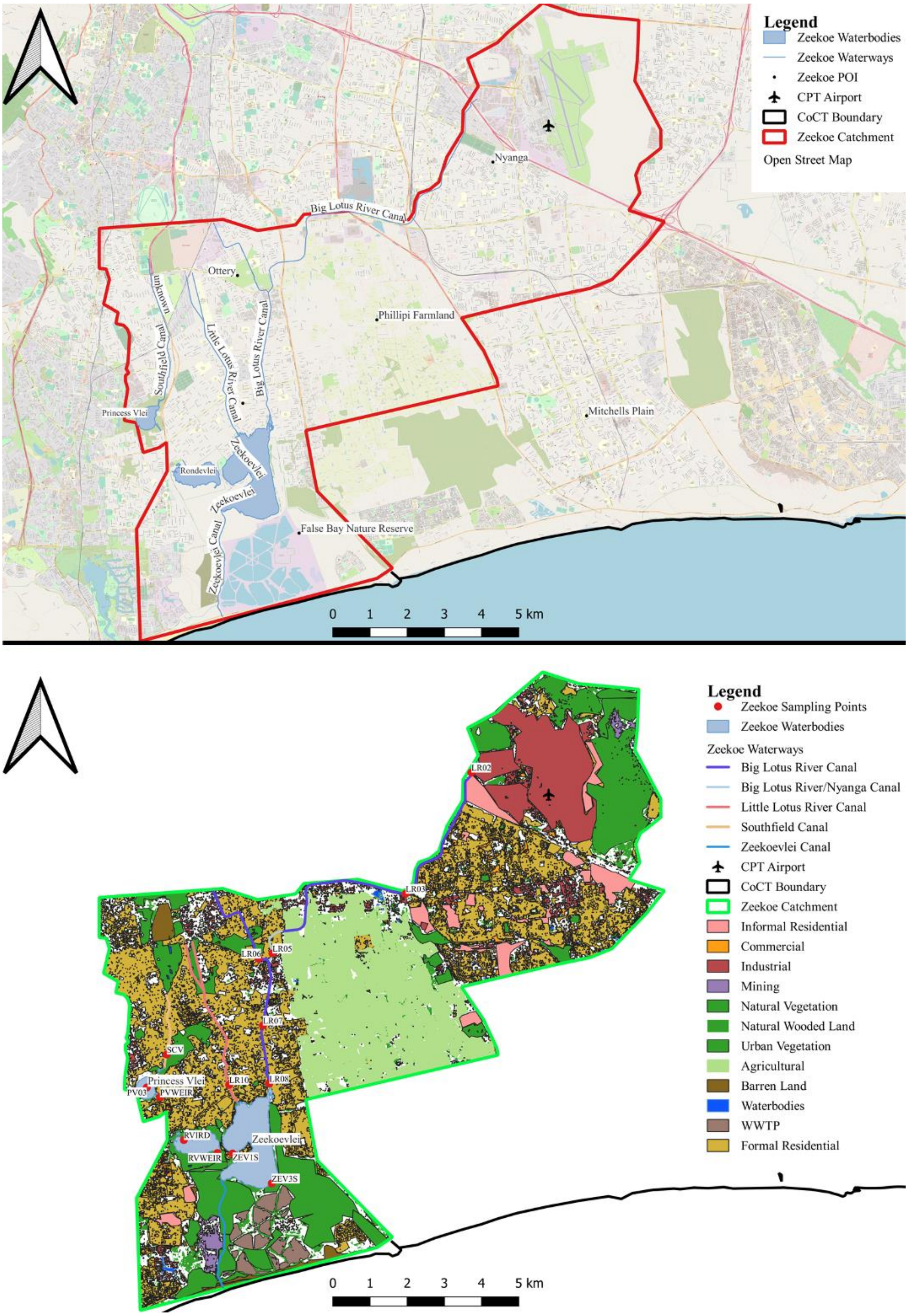

Figure 5-3: Zeekoe catchment locality plan (top); LULC composition and sampling points (bottom)

5.3 Residential stormwater characteristics

5.3.1 Salt River catchment

The stormwater quality characteristics of the residential land use in the catchment and the evaluation thereof are presented in Table 5-4. The median water quality parameter values from the residential areas in the Salt catchment indicate that the stormwater quality is generally poor, with Ammonia, *E. coli* and TP median values classified as unacceptable based on the South African recreational-use standards. Further, the mean and 95th percentile values for Ammonia, *E. coli*, TN and TP are unacceptable – except for the mean value of TN, which is classified as poor. This finding suggests there might be contamination from fertiliser use in agricultural activities, contamination from poorly serviced informal settlements and possibly sewer pipe leaks/bursts resulting in higher mean and 95th percentile concentrations.

Table 5-4: Stormwater quality characteristics for residential land use in the Salt River catchment

Parameters	Salt River (n = 745)			Evaluation of the median value
	Mean	Median	95th percentile	
Ammonia (mg/L)	3.91	0.8895	18.9	Unacceptable
DO (mg/L)	8.55	6.4	16.2	Target
E. coli (CFU/100 mL)	7.46E+05	3.40E+04	3.62E+06	Unacceptable
EC (mS/m)	100.4	102	150	–
pH	7.86	7.80	9.00	Natural
Temperature (°C)	20.2	19.0	25.6	Target
TN (mg/L)	7.9	3.4	28.0	Fair
TP (mg/L)	1.48	0.48	5.56	Unacceptable
TSS (mg/L)	37.3	13.0	113.3	–

The mean stormwater characteristics of the Salt River residential areas in relation to the five selected countries revealed notable distinctions in several parameters (Table 5-5). The Salt River demonstrated higher mean concentrations of Ammonia, *E. coli*, and TP than all the selected countries. Additionally, the Salt River residential areas exhibited the second highest mean TN concentration, surpassed only by India, and the second highest mean TSS concentration, after the USA. Based on these results, it can be inferred that the Salt River is subject to higher levels of pollution than most of the nations included in this study.

Table 5-5: Comparing Salt River residential stormwater with selected countries

	Ammonia (mg/L)	*E. coli* (CFU/100 mL)	TN (mg/L)	TP (mg/L)	TSS (mg/L)
Salt River	3.91	7.46E+05	7.9	1.48	37.3
India	–	4.44E+06	9.2	0.7	27.8
Singapore	–	–	1.2	0.1	31.9
Malaysia	0.14	–	–	0.73	37
USA	0.3	7.00E+02	1.4	–	49
The Netherlands	–	1.90E+04	1.9	0.4	17

5.3.2 Zeekoe Catchment

The stormwater characteristics for the residential areas in the Zeekoe are then presented in Table 5-6. The residential areas analysed for the Zeekoe catchment comprise formal and informal land uses. Both land uses were included in the analysis because of the shared stormwater sampling points.

Table 5-6: Stormwater quality characteristics for residential land use in the Zeekoe catchment

Parameters	Zeekoe (n = 636)			Evaluation of the median value
	Mean	Median	95th percentile	
Ammonia (mg/L)	3.60	0.27	20.92	Unacceptable
DO (mg/L)	7.20	7.86	13.02	Target
E. coli (CFU/100 mL)	2.5E+06	3.6E+03	5.4E+06	Unacceptable
EC (mS/m)	81.4	77.0	138.0	–
pH	8.01	7.80	9.20	Natural
Temperature (°C)	18.2	18.3	25.2	Target
TN (mg/L)	5.65	2.34	23.9	Fair
TP (mg/L)	0.60	0.31	1.95	Unacceptable
TSS (mg/L)	29.3	15.0	92.4	–

The median values for Ammonia, *E. coli* and TP from the residential areas in the Zeekoe catchment are classified as unacceptable as per the South African recreational-use standards (Table 5-2). Moreover, the mean and 95th percentile values for these same parameters were also found to be unacceptable apart from the mean value of TN, which is classified as poor. The mean and median characteristics suggest that the stormwater from the Zeekoe catchment is contaminated, possibly from poorly serviced informal settlements, agricultural runoff – fertiliser wash-off, and leaky sewers – validated by the high 95th percentile concentrations.

The comparative analysis of Zeekoe residential stormwater with India, Singapore, Malaysia, the USA, and the Netherlands revealed significant variations in mean concentrations (Table 5-7).

Table 5-7: Comparing Zeekoe residential stormwater with selected countries

	Ammonia (mg/L)	*E. coli* (CFU/100 mL)	TN (mg/L)	TP (mg/L)	TSS (mg/L)
Zeekoe	3.6	2.50E+06	5.65	0.6	29.3
India	–	4.44E+06	9.2	0.7	27.8
Singapore	–	–	1.2	0.1	31.9
Malaysia	0.14	–	–	0.73	37
USA	0.3	7.00E+02	1.4	–	49
The Netherlands	–	1.90E+04	1.9	0.4	17

The residential areas in Zeekoe catchment had higher mean Ammonia concentrations than in Singapore and the USA, while data was unavailable for India and the Netherlands. The *E. coli* count in the Zeekoe catchment was lower than in India and USA datasets but higher than that in the Netherlands dataset. The Zeekoe catchment had lower mean concentrations of TN than India, but higher concentrations than Malaysia, Singapore, and the Netherlands, with no TN data available for the USA. TP concentrations in the Zeekoe were lower than in India and Malaysia but higher than in Singapore and the Netherlands. Furthermore, the Zeekoe catchment had lower TSS concentrations than Malaysia, Singapore, and the USA but higher TSS concentrations than India and the Netherlands. These findings suggest that the stormwater quality in the Zeekoe is not significantly different from that in other developing countries (India). However, the stormwater from the Zeekoe catchment has higher mean TN and TP concentrations than most developed countries.

5.4 Chapter summary

In this section the stormwater quality in Cape Town was examined, focusing on four catchments that collectively cover 45% of the Cape Town's total area. Historical stormwater parameters (2015 – 2020) from sampling points in residential areas (including formal and informal settlements) were compared against national guidelines and previous South African research (DWAF, 1996c; Nel *et al.*, 2013). Building upon Day *et al.* (2020), this study incorporated further statistical analyses and established correlations between stormwater quality and land-use patterns.

The study's findings revealed that the median stormwater quality in residential areas is generally poor, failing to meet South African recreational-use water quality standards (Table 5-2). Therefore, pre-treatment is necessary before direct mixing with groundwater. Overall, the mean stormwater characteristics from residential areas in the assessed catchments were more polluted than those of most developed countries investigated (Malaysia, Singapore, the Netherlands, and the USA). Organic and biological pollutants (TN, TP, and *E. coli*) showed notably high levels, likely stemming from poor sanitation access in informal areas, sewer bursts, wastewater treatment plant overspills, and fertiliser runoff from agricultural areas and golf courses (Day *et al.*, 2020; Ghoor, 2023; Matthews, 2023). However, the stormwater quality in two of the four analysed catchments was generally better than that of a comparable developing country (India). These variations among the assessed countries and analysed catchments highlight the significance of site-specific factors in the stormwater characteristics. Furthermore, socioeconomic status does not appear to solely determine stormwater quality, reinforcing the need for tailored approaches to address the unique characteristics of each location.

6. Retrofitting a detention pond in Cape Town

6.1 Overview

Okedi (2019) found that MAR was the most suitable method for storing harvested stormwater in the CFA and could increase groundwater resources in the catchment under consideration by up to 30%. He proposed implementing MAR in retrofitted detention ponds in the CoCT. However, the practical implementation of retrofitting stormwater detention ponds to promote urban MAR is not well researched. This knowledge is crucial for validating the technical, social, and economic viability of retrofitting the CoCT's existing stormwater infrastructure to facilitate MAR and realise the benefits suggested by Okedi (2019). Therefore, this research investigated the technical and social viability of retrofitting the City of Cape Town's stormwater detention ponds to enable MAR. A mixed-methods approach was used, and this chapter presents the case study located in Mitchells Plain, Cape Town that was used to evaluate the technical and social viability of retrofitting the CoCT's stormwater detention ponds to enable MAR.

The retrofitting of detention ponds is not a new concept, as displayed by the wide range of stormwater management retrofit guidelines from the United States, United Kingdom, and Australia – discussed in the Section 2.4. The retrofit process involves four key steps: (1) developing a retrofit strategy that includes costing and funding, (2) creating an inventory of potential retrofit sites and prioritising them, (3) designing, constructing, maintaining, and monitoring the retrofit to ensure its effectiveness, and (4) ongoing evaluation of its performance. Some guidelines suggest the need for community outreach if the retrofit site is highly visible to the public (Department of Environment and Swan River Trust, 2006; Digman *et al.*, 2012; Pennsylvania Environmental Council, 2012). However, while these guidelines may be generally applicable, particularly from a technical perspective, they are centred on Global North contexts and make certain socio-economic assumptions that are inapplicable in Global South countries such as South Africa. Contextualisation and adaptation of stormwater management retrofit guidelines is thus necessary to ensure their effectiveness in different settings.

6.2 Designing a retrofitted stormwater detention pond

The South African context presented in Sections 4.7 and 4.8, suggests that three key aspects require a modified strategy for pond retrofitting: (1) choosing ponds for retrofitting, (2) engaging with the community, and (3) selecting construction materials and construction techniques for retrofitting ponds. These factors were considered in the selection of a stormwater detention pond in Mitchells Plain that was used as the study site for SWH via MAR.

The pond's design and construction necessitated the integration of technical design principles typically found in stormwater design manuals with participatory principles. This approach incorporated inputs from landscape experts, anthropologists (social scientists), and water scientists, with extensive community participation, to enable the successful implementation of a retrofit that could be replicated in other areas. The subsequent sections provide a description of the retrofitting process including both the design choices, and context-specific engagement activities implemented in this study.

6.2.1 The retrofit co-design process

Co-design has been proposed as a means of encouraging engaged participation which is vital in South Africa's context, where projects tend to fail because of community dissatisfaction that often manifests in protest action and vandalism (Sambo *et al.*, 2018; Sindall *et al.*, 2021; Mugumbate & Kruger, 2022).

In this study, the co-design process was initiated by a team of transdisciplinary researchers from the University of Cape Town's Future Water research institute in collaboration with the University of Copenhagen. The multidisciplinary team included various disciplines such as anthropology, environmental sciences, civil engineering, and landscape architecture.

The Principal of the Hyde Park Leadership College, next to the School Pond was contacted and supported the research team's interest in undertaking a project in the pond. This was an important step as it established an entry point into the Rondevlei community. The principal then helped arrange a meeting at the school and invited the local Ward Councillor, who was vital in liaising with the residents. The ward councillor suggested holding an open meeting where the broader residents could be informed and engaged. The engagement efforts sought to establish and understand the resident's perceptions of the pond and their needs, wants and fears concerning retrofitting the pond.

The researchers obtained the contact details of the chairperson of the Ratepayers Association – a private group comprising homeowners in the Rondevlei area, some parents, and a few teachers from the local school. The school staff also offered to relay information to other community members via existing community WhatsApp groups. Flyers informing the community of the proposed meeting were also distributed.

Käppeli (2020) led the initial engagement process as part of their MSc research. Käppeli (2020) formulated research questions with support of the project team and gathered input from the residents attending community meetings and individual interviews. The residents present at the first community meeting on the 4th of May 2020 resolved to create a 'Pond Committee' comprising interested community members. A second community meeting was then held on the 10th of March 2020 (Figure 6-1), where some residents, *inter alia*, expressed a desire to turn the pond into a park where they could take walks and socialise. One of the residents suggested that the pond could also be used as a community food garden. The various visions expressed by the community member were noted down (Figure 6-1).

At the meeting on the 10th of March 2020, it became evident that this was not a monolithic community, and several competing interests had to be well managed. The primary concern was the selection of the School Pond over other ponds in the area. The technical site selection process was explained to the residents to address any suspicions of favouritism, which they accepted.

With the onset of the Covid-19 pandemic and the subsequent national lockdown (initiated on the 27th of March 2020), face-to-face meetings became impossible, and the engagement process had to shift to virtual platforms such as Zoom and WhatsApp messaging.

A third community meeting was convened following the relaxation of lockdown restrictions. At this meeting, the researchers noted the emergence of new stakeholders, along with the expressed desire of previously absent residents to participate in the pond committee, prompting a re-selection of the committee. The new community members emphasised the research team's pre-existing intent to source labour for any work on the pond from within the community. Furthermore, members of the community underscored the need for job opportunities in the area, given the high unemployment rate that had been further compounded by the Covid-19 Pandemic.

The approach of involving community members in the retrofitting process faced a hurdle when some residents expressed a desire to assume the project contract for constructing the retrofits. The research team emphasised that the study was primarily a research project with limited funding. An account of the available funds was provided to reassure them that funding was limited. This transparent approach fostered increased trust and collaboration among stakeholders at this point of the retrofitting process.

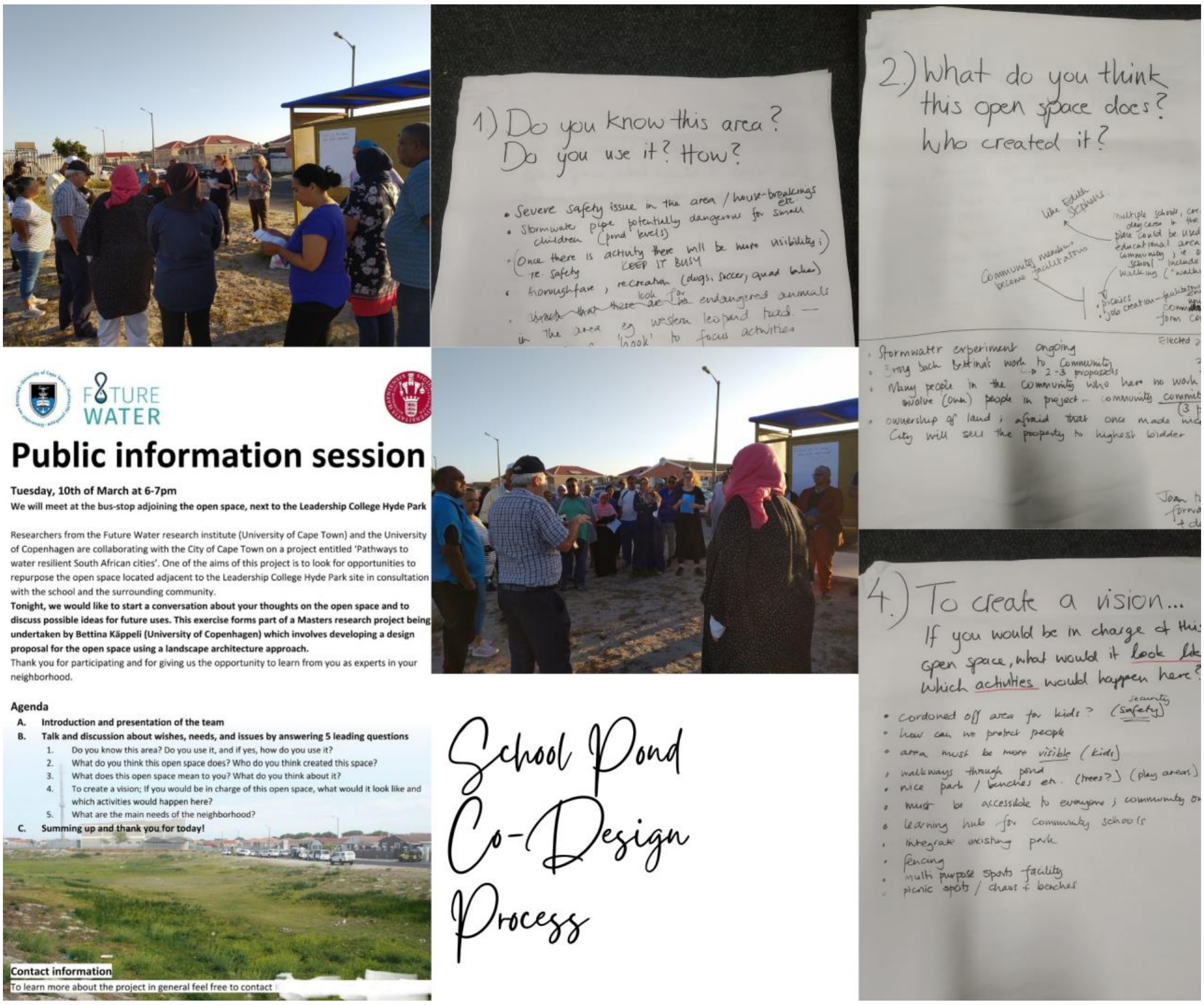

Figure 6-1: The School Pond co-design process in pictures

Upon the suggestion of the Ward Councillor, the selection of workers for the retrofit work was proposed to be done via the South African government's Expanded Public Works Programme (EPWP) jobs portal. The EPWP is a government initiative with the aim of generating employment opportunities for the unemployed or underemployed, with a particular focus on infrastructure and social development projects (Maphanga & Mazenda, 2019). However, the community members present at the meetings rejected this suggestion, citing concerns that individuals who are not part of the Rondevlei Park community might be recruited through the EPWP selection process. Instead, interested residents were requested to submit their details to a WhatsApp group, created and managed by two community members who had assumed unofficial representative roles. This community-led process insulated the research team from any potential conflict that could arise from the recruitment process while allowing for greater participation and ownership among residents.

During community meetings and interviews, residents expressed a desire for amenities, such as benches, a mural, a pathway, and a community food garden around the pond. Käppeli (2020) incorporated these requests into conceptual landscape proposals, including a park within the pond, stepped terraces, and walkways (Figure 6-2a). Käppeli's design also aimed to maximise the pond's functionality, such as using the terraces as infiltration sites during rainfall and as benches on dry days (Figure 6-2b). The proposed infiltration trenches ran along the length of the terraces and were to be constructed using exported media (Figure 6-2c). The use of alternative and potentially low-cost construction materials, such as compacted soil, rocks, or wood, to support the terraces was also considered (Figure 6-2c).

The developed design was deemed impractical and financially unfeasible despite its visual appeal. Thus, to balance practical and financial constraints, it was decided to retain only feasible features within the project budget. Benches and a mural were among the selected amenities, while the community food garden was earmarked for a follow-up project. The community members were informed of these changes and that their suggestions had been considered. Thus, a revised design was required to meet the research and community needs and desires while ensuring practicality and financial feasibility.

The revised design aimed to keep the technical interventions as simple as possible to ensure they were feasible for local companies/individuals to take responsibility for after a short training program. This approach would help to make the endeavour financially and technically viable while building the capacity of local companies/individuals, ensuring long-term sustainability. Further community engagement was achieved through group and individual interviews, public meetings, field trips, co-design workshops, and interactions with the research team.

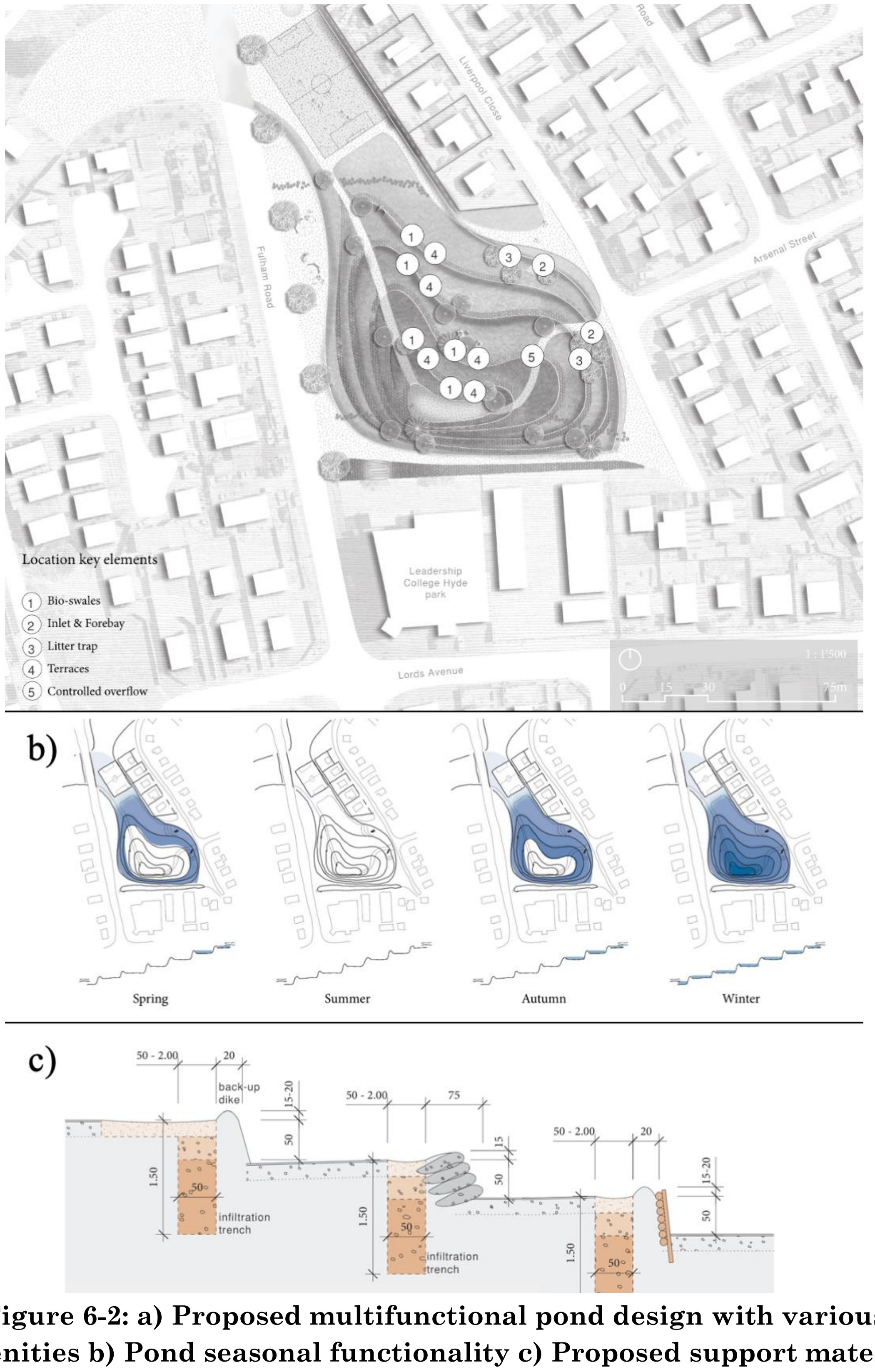

Figure 6-2: a) Proposed multifunctional pond design with various amenities b) Pond seasonal functionality c) Proposed support material for stepped terraces

(Käppeli, 2020)

6.2.2 Technical design and construction process

The School Pond – the study site – is a dry detention pond originally designed to attenuate flooding by storing water from the local stormwater network during storms and slowly releasing it into downstream storm conduits. It is an excavated open area connected to a stormwater system with two inlets and an outlet. Detention ponds are designed to typically retain the maximum amount of water possible for a maximum of 24 to 72 hours (GSMM, 2001; CoCT, 2002; Armitage *et al.*, 2013) while slowly releasing it into the downstream stormwater pipelines, ultimately drying out the pond. Stormwater detention mitigates flooding by confining large volumes of stormwater in a 'safe' area and then slowly discharging it out of the pond, thus abating pipe and culvert surcharges, and urban flooding. Therefore, the design of the pond retrofit had to ensure that the pond would be able to fulfil its initial intended purpose, that is flood attenuation, while enhancing infiltration and providing amenities for the local community. The guiding principles behind the retrofit design were thus:

- A berm should be installed in the pond to retain stormwater and promote infiltration in the ensuing swale. In this study, sandbags were selected as the most suitable material for berm construction due to their practicality, cost-effectiveness, and low value making them less vulnerable to theft. The flexibility and ease of construction of sandbags make them a preferred option over earth berms, which were suggested as an alternative by Käppeli (2020) (Figure 6-2c). The modular construction method with sandbags also made it easy to dismantle the berm if the CoCT or residents wish. The sandbags used in this study were made from recycled bottles.
- The retained water should not be higher than 300 mm to prevent human, and vegetation drowning (Maliva, 2020), and the overflow weirs should be as remote from the inlets as reasonably possible to ensure maximum spread of the trapped water and thus opportunity for infiltration.
- The bottom of the infiltration swale created by installing the berm should be at a level at least 0.6 m above the winter water table (New Jersey Department of Environmental Protection, 2021).
- The detention time within the swale should not be longer than 48 hrs to counter mosquito breeding – 50% of the design storm should drain in 24 hours (Department of Human Settlements, 2019)
- The sand for the sandbags should be obtained from the pond to avoid transporting sand from other areas – ensuring zero net cut and fill.

6.2.2.1 Preliminary topographical and geotechnical surveys

A topographical survey was conducted in parallel with the community engagement efforts described in Section 6.2.1. The survey was aimed at determining the precise locations, elevations, and contours of the pond and its surrounding features. A grid was created over the pond area, and selected points within and around the pond were surveyed using a Garmin GPSMAP 60CSx device to obtain their x- and y-coordinates. The z-elevations were determined using a Dumpy level and referenced to CoCT survey benchmarks around the pond, considering the elevation above the mean sea level for each point. The survey data, including coordinates and elevations, were logged in an Excel spreadsheet, and then imported into the QGIS software and saved as a shapefile to visualise the spot heights.

The Surfer software was used to generate contour lines for the pond, resulting in 0.2 m interval contours. The contour lines were stored as shapefiles and imported into QGIS, where a Google Map overlay was used to check the precision of the coordinates and how they matched up with the Google Earth image. Shapefiles outlining the pond and its components were generated and stored for further analysis. The QGIS shapefiles containing pond outlines, contour lines, and spot heights were exported to AutoCAD. The files were georeferenced, scaled, and saved in .dwg format using AutoCAD software for future use in pond design.

Geotechnical surveys were also conducted, as elaborated in Section 4.10, using hand augurs to assess the underlying soil profile and soil type, and to determine the depth of the water table. The surveys provided insights into the pond surface and sub-surface characteristics and aided in the development of the retrofit designs.

6.2.2.2 Designing and constructing the hydraulic elements

Fast flowing storm runoff contains considerable kinetic energy, which can erode loose and uncovered surfaces. Stilling basins comprising a stone riprap – ground cover made up of large, loose angular stones laid over a geotextile – can be used to dissipate the stormwater's energy. Riprap placed at the inlets were included in the School Pond's retrofit design where they served two primary purposes: 1) to slow down the stormwater flow and thus prevent erosion, and 2) to allow sediment deposition due to the lower stormwater velocity. The design of the riprap thus factored in the required sediment deposition velocity of ~0.3 m/s (Federal Highway Administration (FHWA), 2006) and the expected maximum velocity of the incoming stormwater. The procedure and equations used to determine the School Pond riprap design are found in Appendix H. 150 mm Class 2 riprap was chosen from the FHWA guideline. The apron length and depth were 4D and $3.3D_{50}$, respectively – from FHWA (2006). Figure 6-3 illustrates the layout of the riprap.

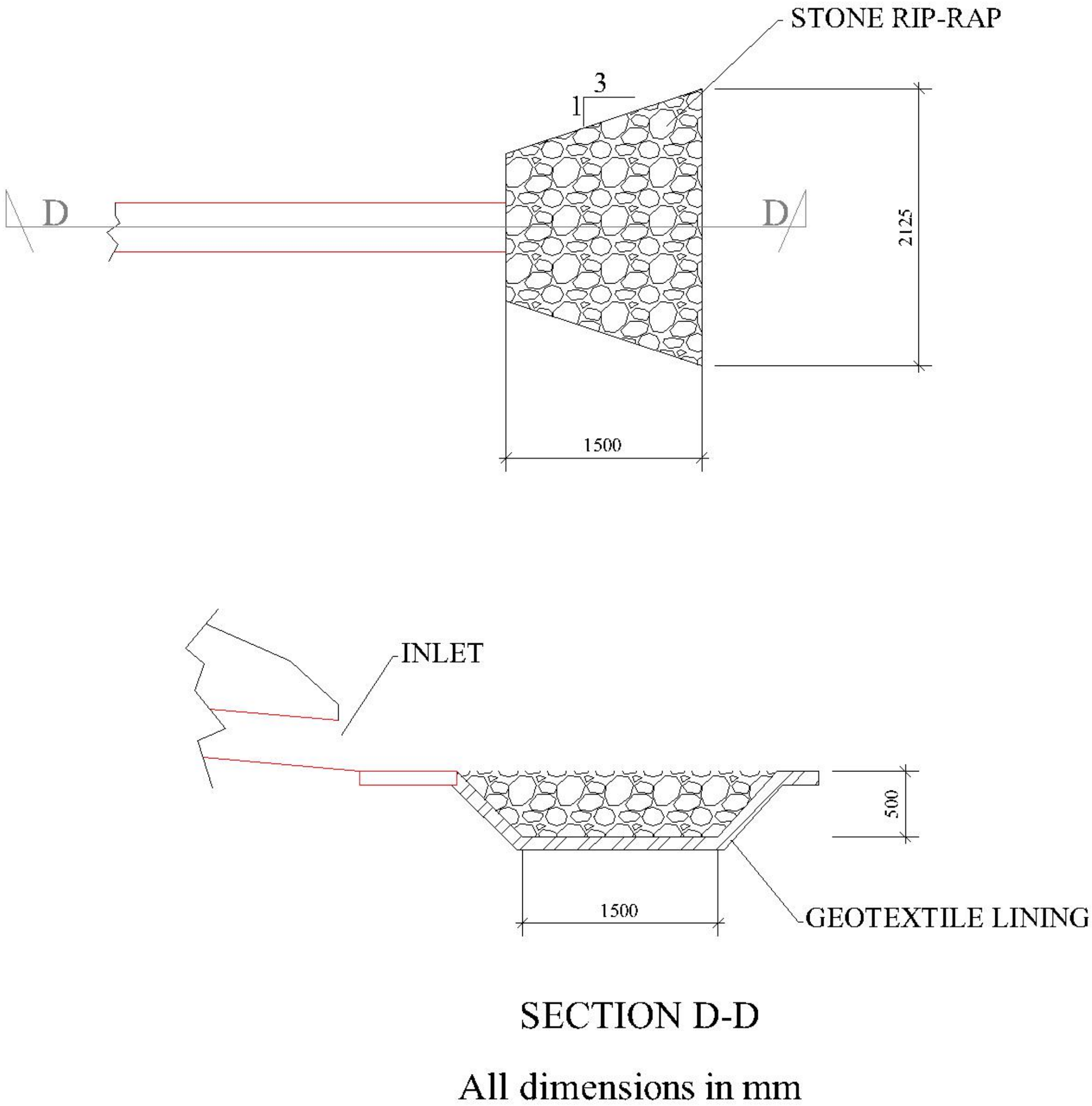

Figure 6-3: School Pond riprap detail

The riprap were then installed at the two inlets with slight modifications to the original riprap design: the lengthening of the riprap section and the installation of rock check dams within the riprap to further retard the flow and trap sediment and litter (Figure 6-4).

Figure 6-4: Inlet dissipator detail

The location of the berm line that defined where the berm would be constructed was identified in conjunction with riprap installation. Trial pits were excavated (Figure 6-5a) to determine the depth of the water table, and the topographical heights of regions in the pond with a depth to water of more than 0.6 m were used to define the elevation of the berm line. This evaluation was conducted in 2021, following a wet season that received 880 mm of rainfall exceeding the historical mean annual precipitation as measured from a weather station located 3.6 km away from School Pond. Consequently, the water table was higher than that observed in previous years (2019 and 2020). Nevertheless, this provided useful insights, as it allowed for the consideration of a worst-case scenario with minimal vadose zone (unsaturated depth).

Figure 6-5: a) Trial pit b) setting out the berm line

The determined berm position had an elevation of ~ 600 mm above the winter water table at the conclusion of the 2021 rainy season. A dumpy level was then used to set out the associated contour (Figure 6-5b) and the elevation and spot coordinates along the contour were transferred to QGIS (Figure 6-6). A retrofit design was then developed encompassing the details and coordinates of the necessary features. The materials chosen for the construction were considered low-value and easily replaceable. The sandbags for the berm wall were filled with sand from the site: the outlet weir used to measure the flow rate was fabricated from plywood and Perspex: while the litter traps were in the form of rock check dams.

Figure 6-6: School Pond layout
(Image source: Google, 2022)

The sandbag berm design is presented in Figure 6-7. The construction was carried out using local labour and hand tools (picks and shovels) as follows:

- The grassed ground cover in the vicinity of the berm line was carefully stripped into sods that were placed aside for later use as cover over the sandbag berm.
- A level terrace on the up-slope side of the berm line was then excavated into the bank to a maximum depth of around 0.5 m to form the infiltration swale. Its width varied with the contours (Figure 6-6). The newly created bank was trimmed to a maximum slope of 1:4 to ensure its stability. The excavated material was stockpiled for later use in the sandbags.
- A trench measuring 300 mm in width and 300 mm in depth was excavated along the predetermined sandbag wall line to house a 1000-micron thick by 1.5 m wide PVC water-proof barrier placed vertically in the trench with the upper section exposed (Figure 6-8).

- The trench was then backfilled on either side of the barrier using the previously excavated material compacted in layers.
- The exposed top of the barrier was then interwoven into the sandbags that were placed around it to a height of approximately 450 mm and filled using the excavated material.
- The grass sods were laid over the berm and secured in place using a woven hessian covering fastened to the sandbags with wooden stakes.

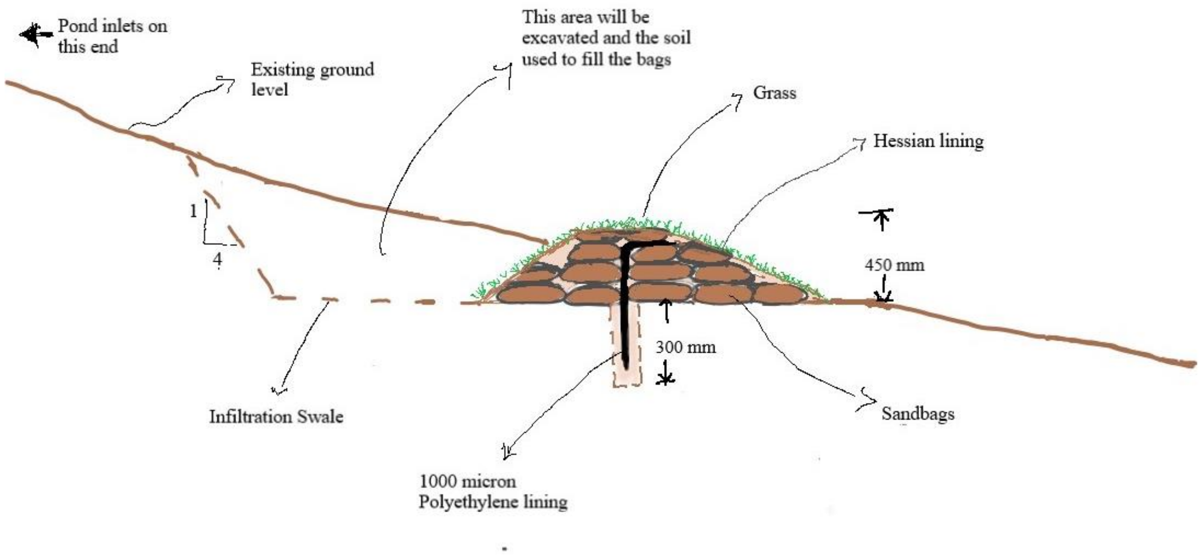

Figure 6-7: Sketch showing the sandbag berm detail

Figure 6-8: a) Residents positioning the PVC sheet b) Assembling the sandbag berm

Two 2 m wide weirs were provided on the far ends of the berm wall for surplus inflow that was unable to infiltrate along the trench. The weir openings were lined with a PVC sheet, which was then covered by sandbags to limit the PVC sheet's ultraviolet (UV) light exposure (Figure 6-9). The weirs were set 300 mm above the trench level with a maximum weir height of 150 mm prior to overtopping of the berm. These dimensions ensure that when the stormwater depth in the swale surpasses 300 mm, the water will cascade over the weir into the larger infiltration area – the area between the berm and the pond outlet (Figure 6-6). During the wall construction, the level on the top of the berm was frequently checked with the dumpy level to ensure it was reasonably constant level, while the terrace floor level was checked to ensure it was relatively flat to ensure the water spread out with minimal localised ponding. To facilitate this, the heights on the top of 'sand towers' constructed of sandbags stacked to the desired wall height and positioned at roughly 5 m intervals along the length of the wall were determined with the dumpy level and adjusted as necessary with a tolerance of ±25 mm. The sandbag wall was then built between the towers, with the top of the wall flush with the sandbag towers. The levels along the wall were then checked using the dumpy level (Figure 6-9).

Figure 6-9: Weir set 150 mm below the wall height

The level control exercise also provided the opportunity for skills transfer whereby two residents were trained to set up and operate a dumpy level and reduce the ground levels using known datums. The practical training involved explaining the concept of land surveys and the importance of level control in construction. The residents expressed an appreciation of these concepts and quickly understood how to set up and record levels. The trained residents were then able to carry out the level control when the researcher was not on site.

Litter traps made from loose rock check dams were installed across the infiltration swales to trap litter transported into the pond via the inlets. Trapping the litter within confined areas in the swale makes collecting the litter easier and limits litter migration into the general pond area. The original plan had been to remove the trapped litter periodically – initially by the researcher – and later by the City or residents acting on behalf of the City. However, some community members conveyed interest in conducting regular pond clean-ups which they did sporadically. The teachers at the school next to the pond also expressed interest in helping with litter management through their environmental club.

The construction process took 25 working days with minimal deviation from the proposed design and the completed retrofit elements are shown in Figures 6-10 and 6-11. The system functions as follows:

- The stone riprap at the stormwater inlets retards the velocity of the stormwater entering the pond reducing local erosion and encouraging the coarser sediment transported from the catchment area to settle out.
- Rock check dams on either side of the riprap create a forebay enclosure by impeding the flow allowing sediment to settle out while simultaneously trapping litter. This both helps keep the remaining infiltration swale free from excessive sediment build-up as well as facilitating easier pond clean-up as the trash is concentrated in the two forebays as the deposited silt can be shovelled out when necessary.
- The stormwater is then channelled through the infiltration swale, which serves as the primary infiltration site with the sandbag berm and weirs restricting the maximum depth of around 300 mm. Any excess water then flows over the two end weirs.
- The overflow cascades to the central part of the pond – denoted as a 'swampy area' in Figure 6-10. This area is situated in the lower part of the pond and becomes saturated with groundwater during periods of high rainfall. The topography of the swampy area, featuring depressions, forms natural storage that support diverse fauna and flora.
- Any additional overflow results in water flowing from the pond over an overflow weir at the outlet and into the existing stormwater network.

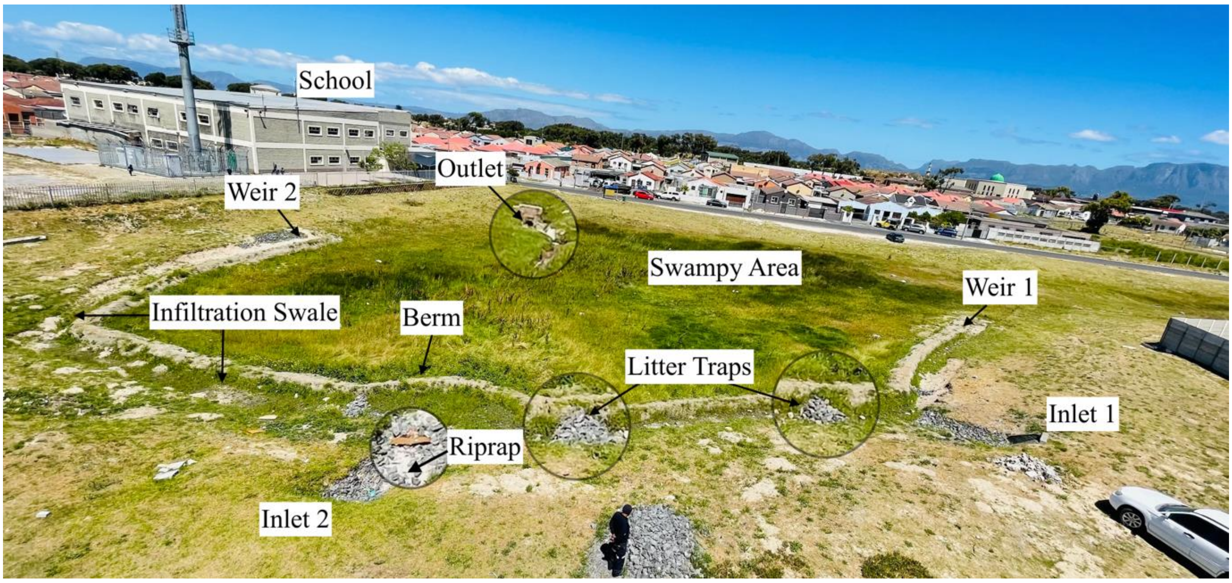

Figure 6-10: Completed pond retrofit

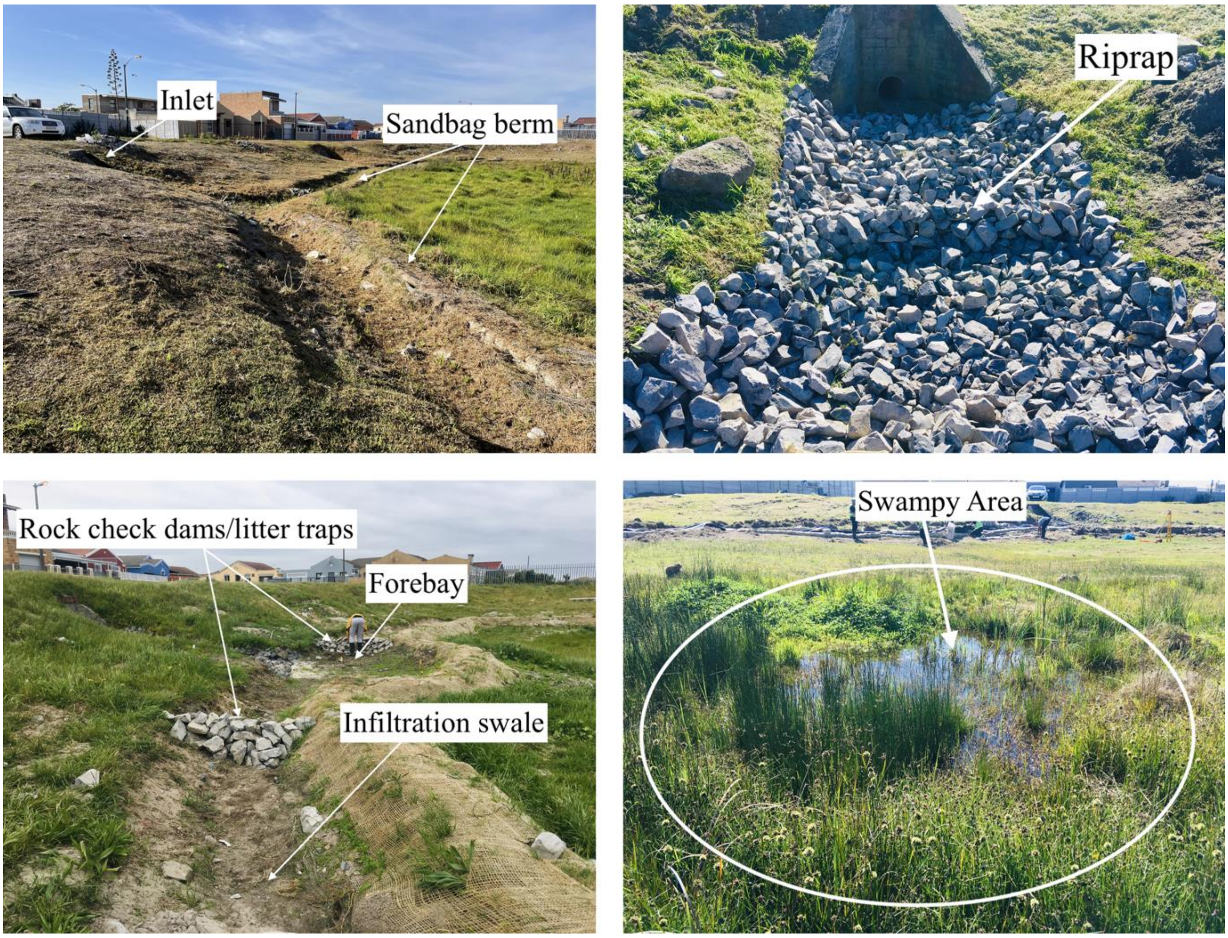

Figure 6-11: Retrofitted pond elements

6.3 Functionality and maintenance of the retrofitted pond

6.3.1 Berm and infiltration swale functionality

The sods of grass on the berm (Figures 6-12a and 6-12b), rapidly took root and eventually resulted in a vegetated berm (Figure 6-12d). The grass growth was aided by a prolonged period of rainfall shortly after construction. The installed sandbag berm was sturdy enough to bear the weight of people walking and children playing (Figure 6-12d).

Figure 6-12: a & b) Grass placed on the berm during construction c) Hessian fibre staked to the ground d) Resident walking along the berm

40.8 mm rainfall recorded at the School Pond weather station on the 26th of August 2021 showed that the swale ground levels were acceptable along the trench. The ponded water in the infiltration swale infiltrated in less than 24 hours (Figure 6-13).

Figure 6-13: a) Ponded infiltration swale after a storm, 26 August 2021 (16:51) b) Dry infiltration swale, 27 August 2021 (10:35)

Another rainfall event was recorded on the 17th of November 2021. The event resulted in 5.4 mm of rainfall in ~ 6 hours. Water in the trench adjacent to inlet 2 – Figure 6-10 – ponded to a depth of ~100 mm. However, Cape Dune Mole-Rats (*Bathyergus Suillus*) had dug a tunnel beginning in the infiltration swale – between the litter traps shown in Figure 6-10 – underneath the berm to the general pond area (between the berm and the swampy area – Figure 6-10) and the water escaped the infiltration swale through this hole (Figure 6-14). Water which did not escape via the mole tunnel infiltrated within 24 hours.

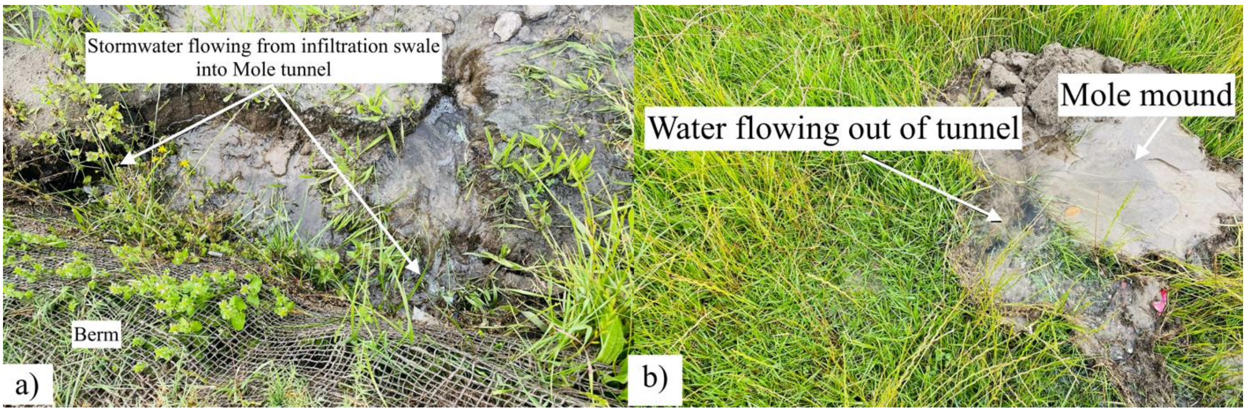

Figure 6-14: Stormwater loss through a mole tunnel

A third rainfall event was observed on the 6th of May 2022, resulting in a rainfall depth of 10.4 mm in 7 hours – the total rainfall depth for the day was 12.2 mm. The storm coincided with a maintenance exercise – during which an attempt was made to plug the holes created by the moles in the infiltration trench (a further six holes had been observed since the 17th of November 2021 event) using a combination of stones and sand to plug the mole holes, but this proved ineffective as the stormwater continued to leak through the holes as before (Figure 6-14). The leaks limited the infiltration in the swale (Figure 6-15).

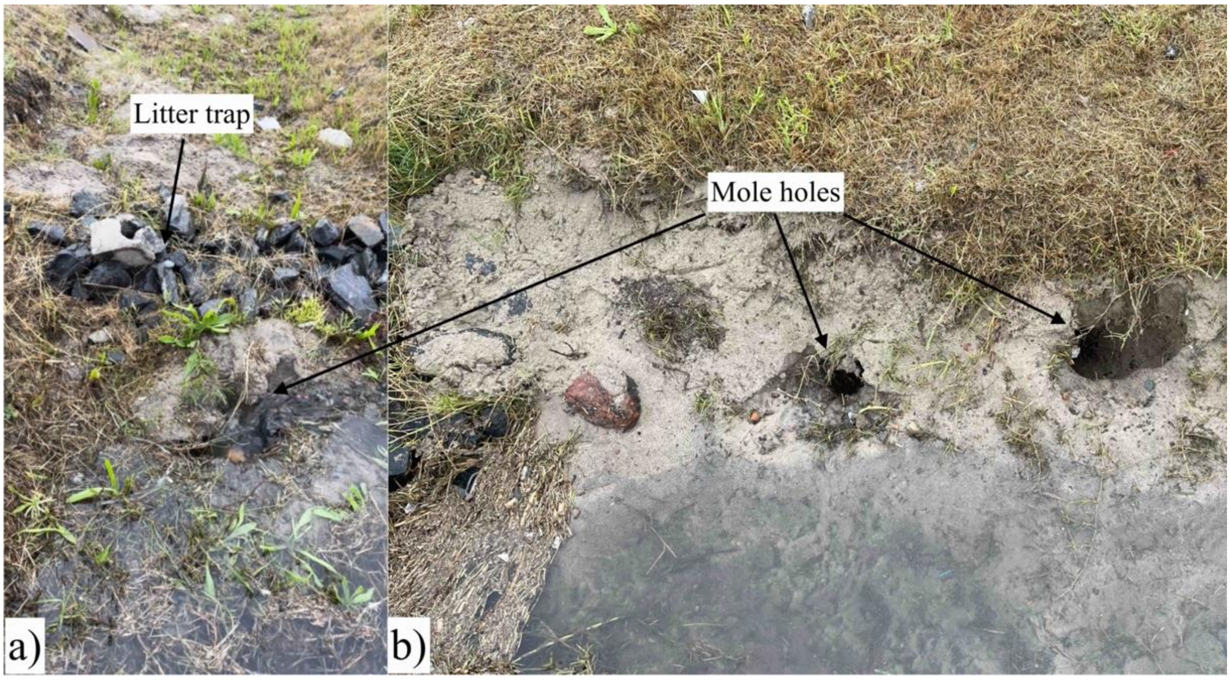

Figure 6-15: a & b) Mole holes in the infiltration trench

An attempt was then made to plug the mole holes with bentonite (Figure 6-16b). Bentonite was selected because it would not be toxic to the moles and expands when in contact with water to fill the voids left by the rocks and sand. While clay has a much lower porosity than sand, it was speculated that this would limit the short-circuiting.

The mole holes were first plugged with rocks after which bentonite mixed with sand was poured into the hole and compacted using a spade until the hole was sealed. However, the seal did not hold, and the stormwater scoured the plug and the hydrostatic pressure resulted in water flowing out through the hole as before (Figure 6-16c)

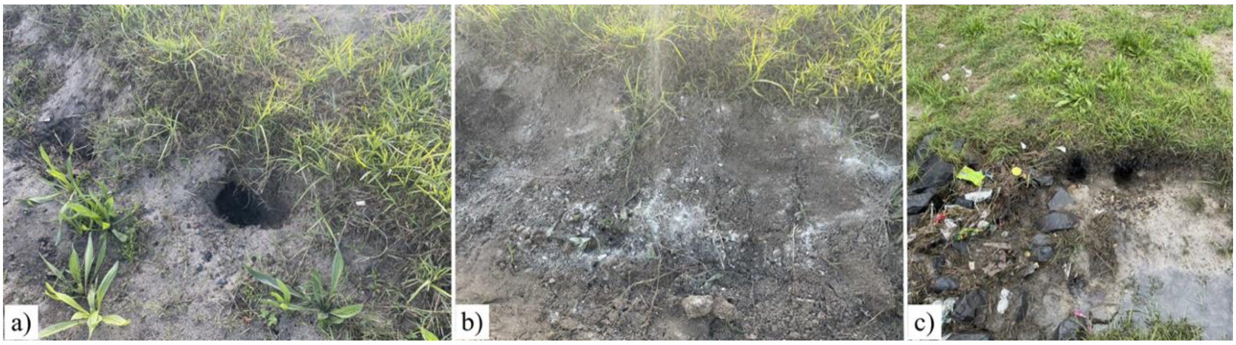

Figure 6-16: a) Mole hole b) Plugged mole hole c) Failed mole hole plug

Another storm was recorded on the 13th of June 2022 during which the pond received 7.8 mm in 3.5 hours. The riprap retarded the fast-flowing flow, while the berm retained the water in the swale (Figure 6-17). However, most of the water was lost via the mole holes as the plugs failed.

Figure 6-17: a) Full swale during a storm

More storm events were observed at the School Pond, yielding similar results with respect to the berm and swale functionality. However, the presence and impact of moles in the pond posed an unforeseen challenge, necessitating new considerations in design and construction operations. After unsuccessful attempts to plug the mole holes using rocks and bentonite, a long-term solution was devised to address the mole issue. The approach adopted involved revising the retrofit design by installing a mole barrier employing a geosynthetic material with environmentally safe properties developed by a local firm.

The mole barrier installation was completed in April 2023, with the same local contractor engaged in the initial pond retrofit. Community members were also enlisted using the same labour selection methodology, ensuring a collaborative approach that maintained stakeholder engagement and promoted a sense of ownership and environmental stewardship throughout the project. The mole barrier installation process was as follows:

- A trench, 0.5 m wide and 1.5 m deep was dug along the downstream side of the existing berm (Figures 6-18 and 6-19).
- Grass sods from the excavated trench were preserved.
- A mole barrier made of two 1 m wide flexible high-density polyethene (HDPE) sheets was placed in the trench, with an approximate 0.5 m overlap and leaving 0.5 m exposed (Figure 6-19).
- The mole barrier was held in place by replacing the excavated sand on either side in compacted layers of ~ 300 mm (Figure 6-20)
- The exposed top of the mole barrier was then pushed against the back face of the existing berm wall, and sand was placed over it and compacted using shovels to prevent movement (Figure 6-19). This barrier section was placed to prevent moles from burrowing through the sandbag berm.
- A portion of the sand from the trench excavation was used to fill the sandbags. The sandbags were used to repair sections of the berm that had been disturbed during the installation of the mole barrier.
- The preserved sods of grass were then placed on the berm to integrate the structure with its ecological surroundings, as before the repair.
- A Hessian fabric cover was positioned over the berm and secured using wooden pegs to hold the sods of grass in place until they were integrated into the berm.

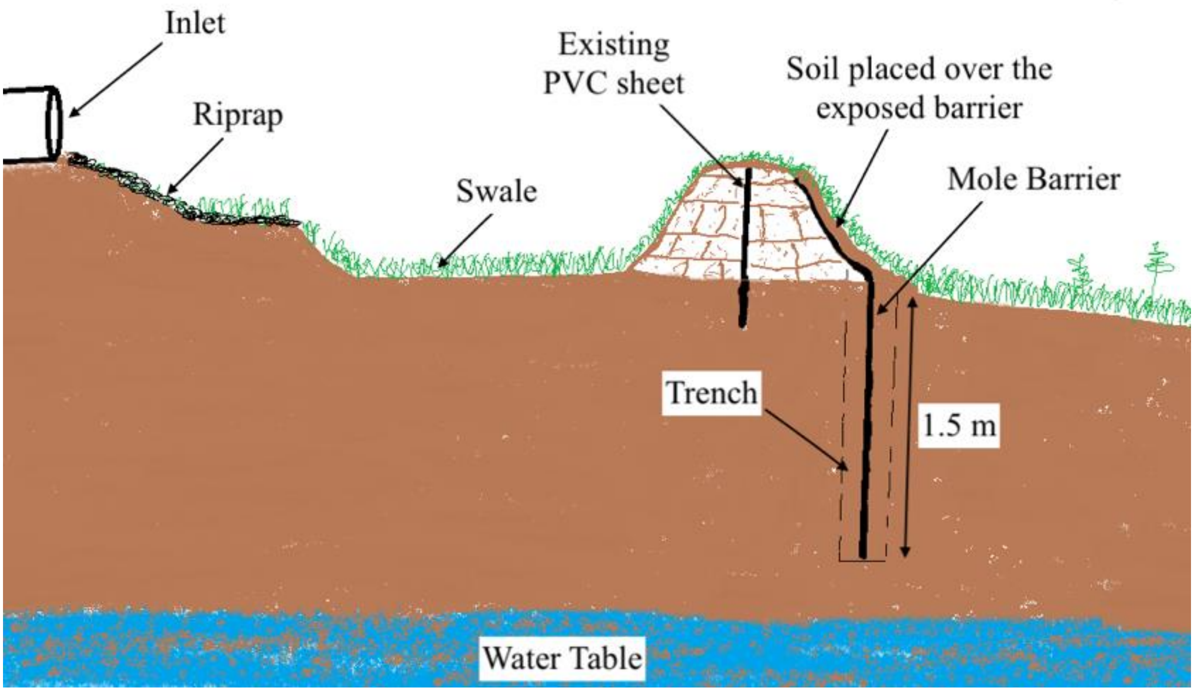

Figure 6-18: Sketch of the mole barrier installation

Figure 6-19: Excavating the trench for the mole barrier

Figure 6-20: a) placing the mole barrier b) compacting the trench

6.3.2 Riprap and litter trap performance

The performance of the energy dissipator (the stone riprap) was also observed during various storms. The use of riprap and rock check dams proved effective as litter traps and energy dissipators and substantial quantities of silt quickly accumulated in the forebay. However, the accumulation of silt in the forebay raised concerns due to the potential acceleration of clogging within the infiltration swale, necessitating frequent maintenance.

The riprap was observed to retard the flow of stormwater, transforming the flow from a fast and turbulent motion to a calm and steady flow evenly dispersed across the infiltration swale area (Figures 6-17 and 6-21a). The grass that grew in the infiltration swale can aid in retarding the clogging process, and the plant roots facilitate and improve infiltration rates (Hua *et al.*, 2014; Carrasco-Acosta *et al.*, 2019; Jacklin *et al.*, 2021).

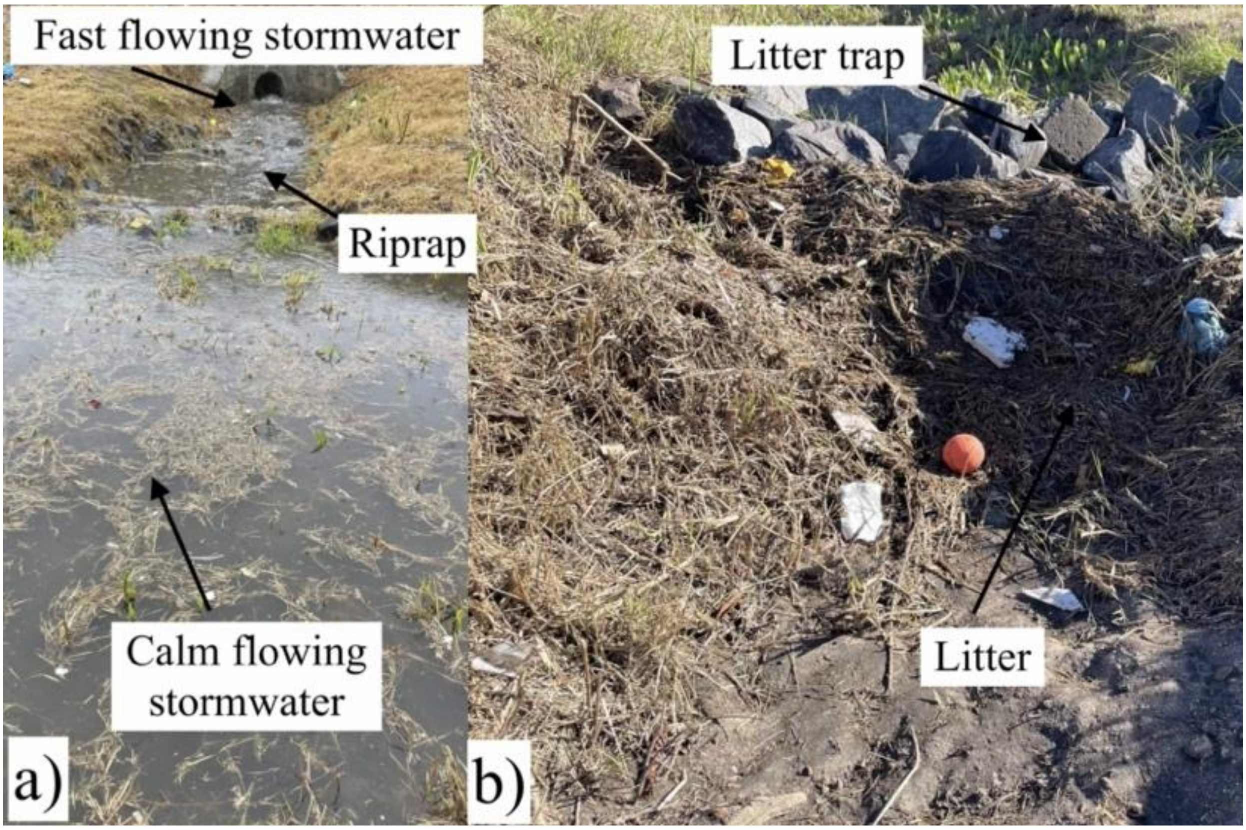

Figure 6-21: Riprap and litter functionality

Suspended solids, such as silt, which are transported in urban stormwater, typically contain high levels of heavy metals that adhere to the silt particles (Hwang *et al.*, 2016; Sakson *et al.*, 2018). Consequently, the deposition of silt potentially reduces the heavy metal load, while the vegetation in the area downstream of the riprap can aid in the phytoremediation of the deposited heavy metals *(Shehata et al.*, 2019; Lamine & Saunders, 2022). During maintenance operations, the deposited silt and the vegetation containing accumulated heavy metals and other pollutants can be removed as necessary.

During a storm, a motor oil spill was observed, and further investigation revealed that the oil originated from a nearby mechanic's workshop. It was determined that an individual from the shop was responsible for the incident, having disposed of motor oil by dumping it into a catch pit (Figure 6-22a). As the stormwater flowed through the network, carrying the oil, the strategically placed check dams effectively trapped and confined a portion of the oil within the forebay area (Figure 6-22b).

Figure 6-22: a) Motor oil dumped in a storm catch pit b) Trapped oil and litter

6.3.3 Maintenance of retrofits

Maintenance of storm infrastructure like detention basins is essential for continued and optimal functionality of the infrastructure and the overall stormwater system.

Typical stormwater maintenance procedures include clearing litter and debris in stormwater pipes, catching pits, and trimming grass and vegetation in detention ponds (Digman *et al.*, 2012). Woods-Ballard *et al.* (2015) provide a guide on the maintenance requirements of stormwater infrastructure. They recommend the monthly removal of litter, debris, and trash and the cutting of grass in these systems to manage vegetation and weeds. In addition, other maintenance actions can be periodically taken, such as the realignment of the riprap, rehabilitation of inlets, outlets, and overflows, and scrapping of the topsoil in the infiltration trench/basin if the infiltration performance deteriorates.

A maintenance plan was thus required for the School Pond to ensure its optimum performance. This plan entailed the removal of debris, trimming of vegetation in and around the pond and on the berm, clearing of inlet and outlet channels, as well as the upkeep of the infiltration trench including corrective actions, frequency, and accountable parties and is presented in Appendix D.

The management of grass cutting and litter collection in detention ponds, such as the School Pond, has historically been the responsibility of the CoCT. As of 2023, both the Parks and Recreation Department, as well as the Roads and Stormwater

Department, have been actively involved in undertaking these maintenance tasks at the School Pond.

The grass within the School Pond is typically cut twice a year, with additional monthly cuts available upon special request to the CoCT. Litter collection takes place once a month. However, following the retrofitting of the pond, the school leadership expressed a willingness to contribute by managing the litter during the school term. They committed to collecting litter from the pond every Wednesday as part of their environmental club activities. Site visits were conducted at least once a month, on Wednesdays, where the school was observed carrying out this task on two occasions. Furthermore, certain community members involved in the pond retrofit marked out in stones the area known as the 'swampy' section, where some residents believed that the endangered Western Cape Leopard toad species breed, (Figure 6-23a). They requested that the CoCT maintenance team refrain from cutting the grass in that specific area. As a result, the swampy section became densely covered with vegetation (Figure 6-23b). However, the growth of tall vegetation raises potential safety concerns, as it could offer concealment for potential intruders or burglars. The grass in the surrounding areas, including the berm and pond slopes, were trimmed by the Recreation and Parks Department.

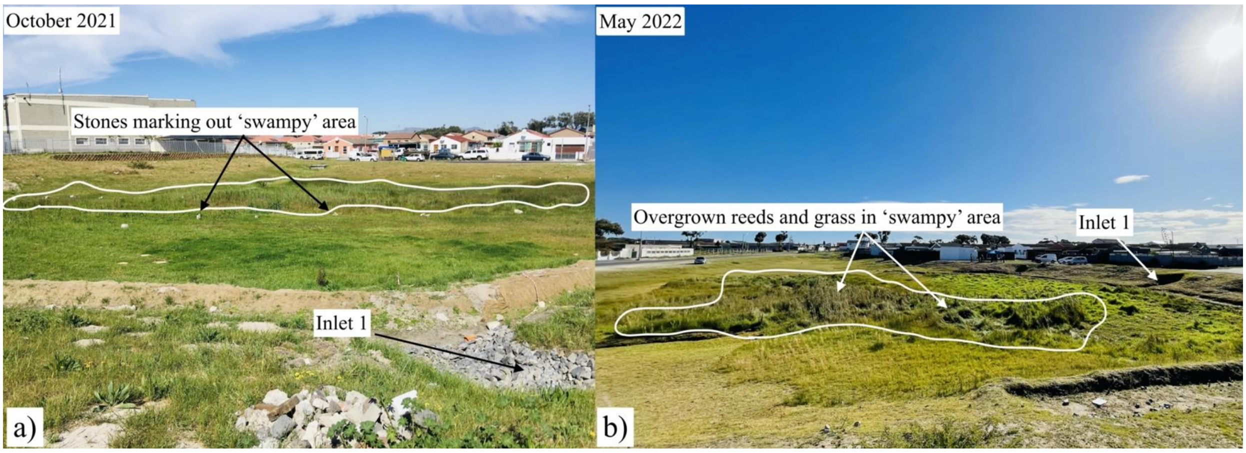

Figure 6-23: a) Marked out swampy area b) Overgrown reeds and grass after eight months without trimming

Throughout the research period, both the CoCT and the research team provided resources and labour to maintain the pond. However, to ensure sustainable management, resident involvement and responsibility for maintenance while holding municipal officials accountable are crucial. To facilitate this shift, the operation and maintenance plan, developed as a component of this study, was shared with the School Pond committee. Their input was integrated into initial drafts, making the plan collaborative. This maintenance strategy functions as a blueprint, enabling the gradual transfer of responsibilities to the community. This empowerment sought to encourage active engagement from residents in overseeing and maintaining the pond.

6.4 Chapter summary

In this chapter, the technical evaluation of retrofitting detention ponds to facilitate MAR was undertaken using a case study in Mitchells Plain, Cape Town – the School Pond. The design, construction, operation, and maintenance of a retrofitted detention pond were investigated. Existing stormwater retrofit guidelines were recontextualised and adapted to consider South African socioeconomic factors.

The design process involved engaging with residents living near the pond through a co-design approach, and labour for the pond retrofit was sourced from the local community, completing the initial construction process in 25 working days. The retrofit design incorporated riprap at the pond inlet to reduce stormwater velocity, mitigate local erosion, and promote sediment settling. A 90 m long, 450 mm high sandbag berm was built in front of the inlets to detain stormwater and facilitate infiltration. In addition, rock check dams were constructed to act as litter traps.

The materials chosen for construction were low-value and easily replaceable, with sandbags filled using on-site sand, while the outlet weir, utilised to measure flow rate, was fabricated from plywood and Perspex. Non-metallic elements were preferred as metal components can be removed and sold as scrap metal in South African neighbourhoods. The choice of materials was a clear departure from the northern guidelines reviewed (Section 2.4) but was an important consideration in the South African context.

Observations of several storm events at School Pond demonstrated the functionality of the retrofit, facilitating MAR and silt trapping. However, the presence of moles that dug tunnels under the swale hindered the infiltration potential of the pond. Nonetheless, this issue was resolved by installing a mole barrier. A maintenance plan was also developed for the pond.

7. Evaluating community participation in pond retrofits

This section presents an evaluation of the level of community participation achieved as a result of the School Pond retrofitting project. Community participation was evaluated using a tool initially developed by the Nexus Community Engagement Institute (NCEI) (2018), which was modified in this study. The tool was developed to, *inter alia*, aid individuals or organisations in assessing the level of community participation in their projects.

The NCEI (2018), refer to their tool as a 'community engagement assessment tool', emphasising that community engagement (CE) should be the goal of community-centred projects. The foundation of this assessment tool lies in recognising that community participation encompasses both 'community engagement' and 'community outreach' – terms often mistakenly assumed to be synonymous but exist on opposite ends of a spectrum, as discussed in Section 2.5.1.

The assessment tool comprises five main questions and fifteen sub-questions formulated and designed to guide user analysis (Appendix F). The questions are a set of continua in which users rank their project activities from 'doing primarily outreach' to 'doing community engagement'. The tool provides a self-assessment key described as follows:

- Uncertain what we are doing: This column is chosen if the user is uncertain whether their project activities can be classified as outreach, community engagement, or somewhere in between.
- Primarily engaged in outreach: This column is chosen if the user considers project activities primarily to involve outreach rather than community engagement.
- Beginning to talk about moving to CE: This column is chosen if the user considers that all or some of the project team members have started serious discussions about incorporating community engagement practices and principles, but no specific strategies have yet been implemented.
- Working toward CE: This column is chosen if the user considers members of the project team have begun implementing some community engagement practices, but engagement is not yet the foremost aspect of the project.
- Actively engaged in CE: This column is chosen if the user considers that most project activities are community engagement and systems and practices to

continually learn and adapt through community engagement have been created.

The original community engagement assessment tool was entirely qualitative, meaning that the overall assessment of a project relied on interpreting non-numerical data (responses to questions) to gain insight into the nature and characteristics of the project. However, qualitative interpretations are often subjective and difficult to replicate. The assessment tool was thus modified to incorporate qualitative and quantitative aspects to address this limitation. This hybrid approach seeks to enhance the objectivity and replicability of the assessment process. The assessment tool was modified and implemented as follows:

- During each assessment period, the designated assessor (in this case, the researcher) responded to the fifteen questions presented in the assessment tool. As explained previously and depicted in Figure 7-1, each question provided five continua options, offering a spectrum of community participation levels.
- The statistical mode indicating the continuum most frequently chosen by the assessor for each question was recorded. For instance, if the assessor selected the 'Working toward CE continuum' for five of the fifteen questions, the mode value for that continuum was five.
- The modes for the five continua were tabulated and presented using a bar chart. The mode of the continua was then used to evaluate the level of community participation at each assessment period.

By adopting this method, the assessment process attained a standardised and replicable approach, aiming to mitigate the potential subjectivity that might have been present in the original tool.

Q: WHAT KIND OF RELATIONSHIP DO YOU HAVE WITH COMMUNITY MEMBERS?

OUTREACH	UNSURE WHICH WE ARE DOING	DOING PRIMARILY OUTREACH	BEGINNING TO TALK ABOUT MOVING TO CE	WORKING TOWARD CE	DOING CE	COMMUNITY ENGAGEMENT
• Relationships are primarily TRANSACTIONAL, for the purpose of completing a project.				✓		• Relationships are FOUNDATIONAL, continually built between and among people and groups. Staff/institutions continually build the relationships they need to know their community.
• Relationships are often NOT INCLUSIVE of all racial or cultural groups in the community.					✓	• Relationships reflect the DIVERSITY within the community.
• Relationships can be LIMITED to a few community members, often giving influence to those with the loudest voices.					✓	• Relationships are built not just with current leaders, but also with people with an interest and/or POTENTIAL TO BE LEADERS.
• Relationships are SHORT-TERM, so staff have to rebuild them as other projects or issues come up.					✓	• Relationships are transformational and LONG-TERM, so community leaders/members can engage in projects and issues as they come up.

Figure 7-1: Screenshot of assessment form showing responses to Question 1 and its sub-questions

The modified tool was used to assess the level of engagement at four distinct stages of the retrofit process in this study. The four stages occurred during the pre-construction and during-construction phases (Assessment Period 1) spanning from January 2020 to October 2021, followed by post-construction assessments in Periods 2, 3, and 4, encompassing November 2021 to May 2022, June 2022 to January 2023, and February 2023 to June 2023, respectively (Figure 7-2).

An overall evaluation of the project's progress until July 2023 was then conducted, providing an overview of community engagement throughout the project's observed lifecycle. The responses to the fifteen questions from all assessment stages were categorised into two groups: 'outreach leaning' and 'engagement leaning' responses, to facilitate this evaluation. The responses were aggregated to indicate the level of community participation achieved during the School Pond retrofit project. The participant observations conducted in these evaluations were covered by an ethics clearance that was issued for the PaWS project (Appendix J2).

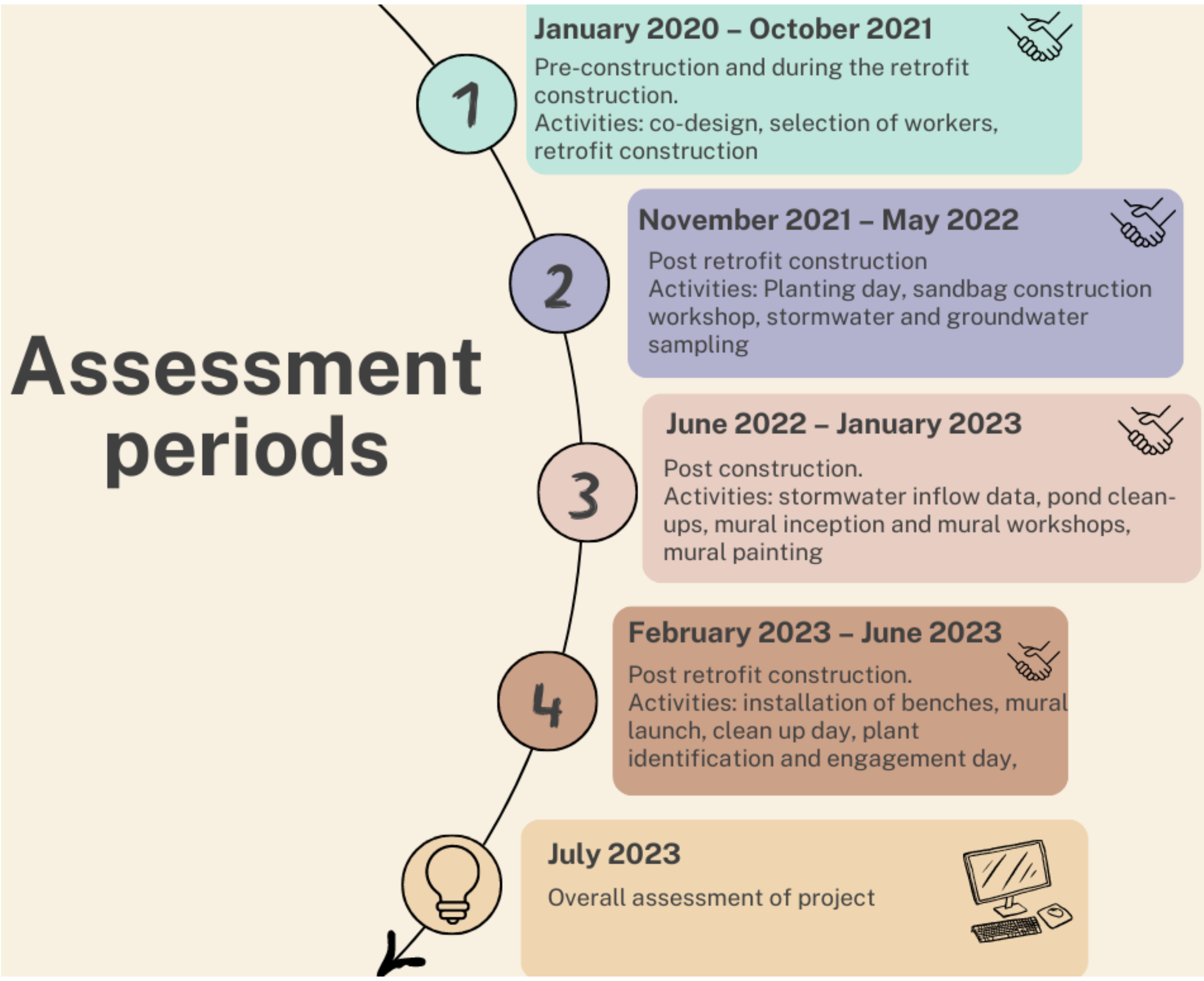

Figure 7-2: Timeline of assessment periods

7.1 Evaluating community participation at the school pond

The subsequent sections present the outcomes derived from applying the modified assessment tool at each assessment period, providing insights into the varying levels of community engagement throughout the project's progression.

7.1.1 Assessment Period 1 – pre-and during construction

Figure 7-3 depicts the modes for each continuum based on the responses to the fifteen questions during the first assessment period, leading up to the conclusion of the construction phase in October 2021.

The findings from Assessment Period 1 (Figure 7-3) indicate that the social activities taking place at the School Pond predominantly fell under the category of community outreach. Nonetheless, certain activities such as the co-design exercise and the selection of workers to construct the retrofits were indicative of a transition towards community engagement. This observation was consistent with the study's context, given that this assessment was conducted during the construction phase where interactions with the community had recently been initiated and relationships were still being formed. The level of community participation was thus expected to evolve with time as more activities were conducted at the pond.

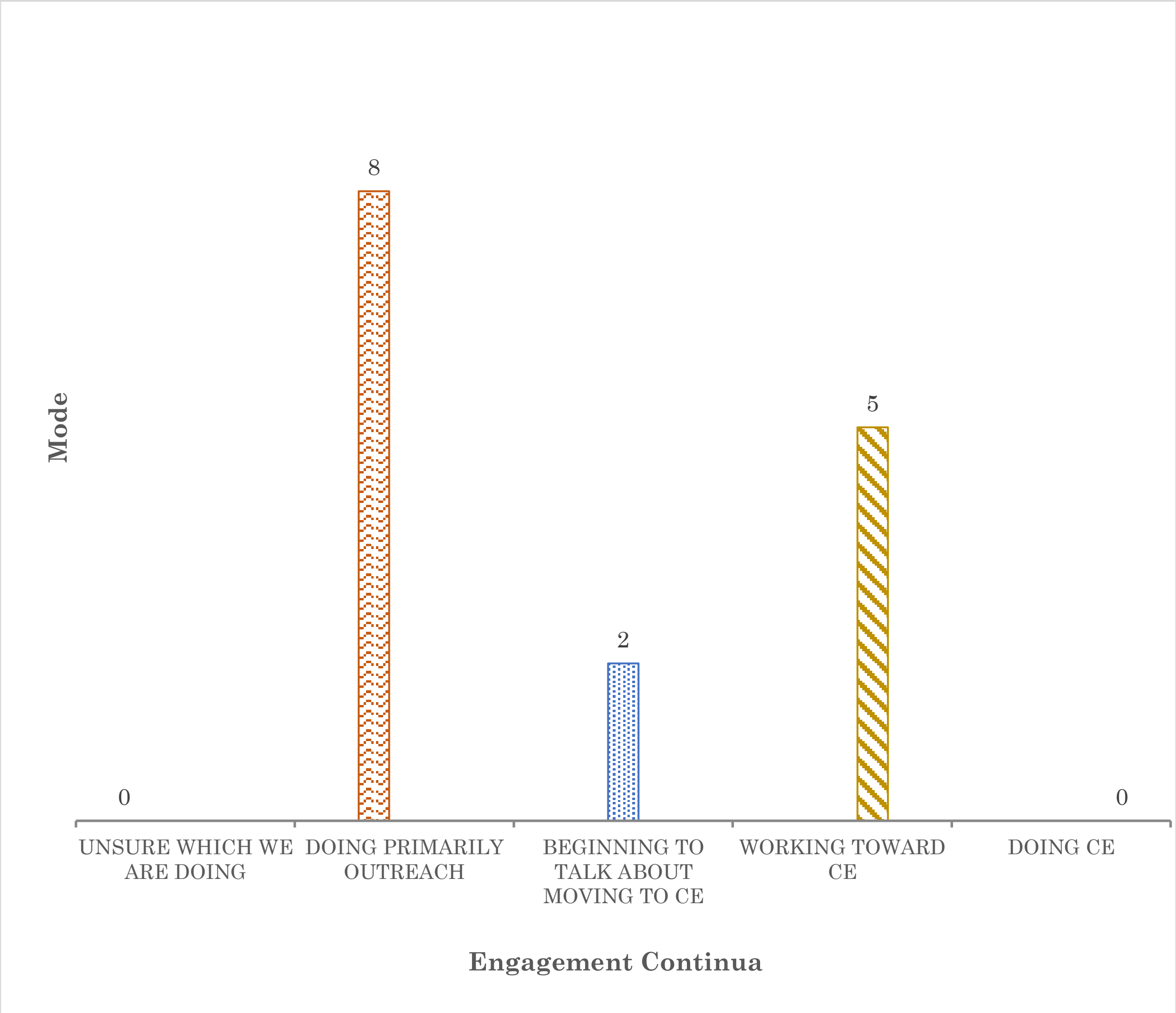

Figure 7-3: Modes of continua (Assessment Period 1)

7.1.2 Assessment Period 2 – post construction

Various activities were carried out at the School Pond during the second assessment period, which extended from November 2021 to May 2022. These activities included a planting and engagement day that involved pupils from two nearby schools (Figure 7-4a), a sandbag construction workshop (Figure 7-4b), and several stormwater and groundwater quality sampling events (Figures 7-4c and 7-4d).

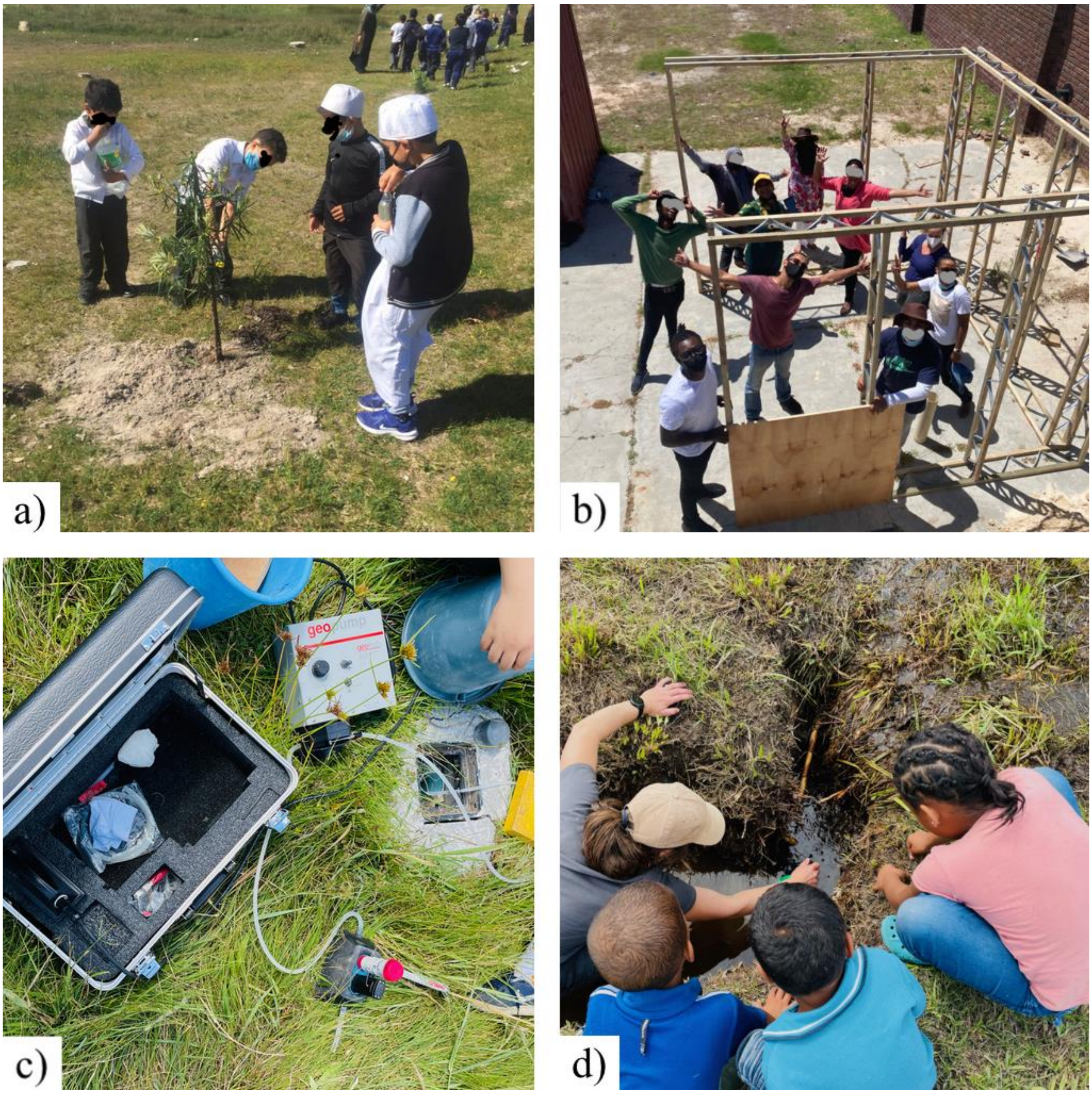

Figure 7-4: Images from Assessment Period 2

These activities provided opportunities for meaningful interactions between the community members and researchers. The engagement/outreach assessment for this period was conducted in May 2022.

The modes for each continuum, based on the responses to the fifteen questions, were captured and tabulated to evaluate the engagement rating effectively (Figure 7-5). The overall assessment of the activities indicated that researchers were still primarily conducting outreach work at this stage. While some activities were considered community outreach, there was an increase in activities that signalled a shift towards community engagement, with one response alluding to some level of community engagement.

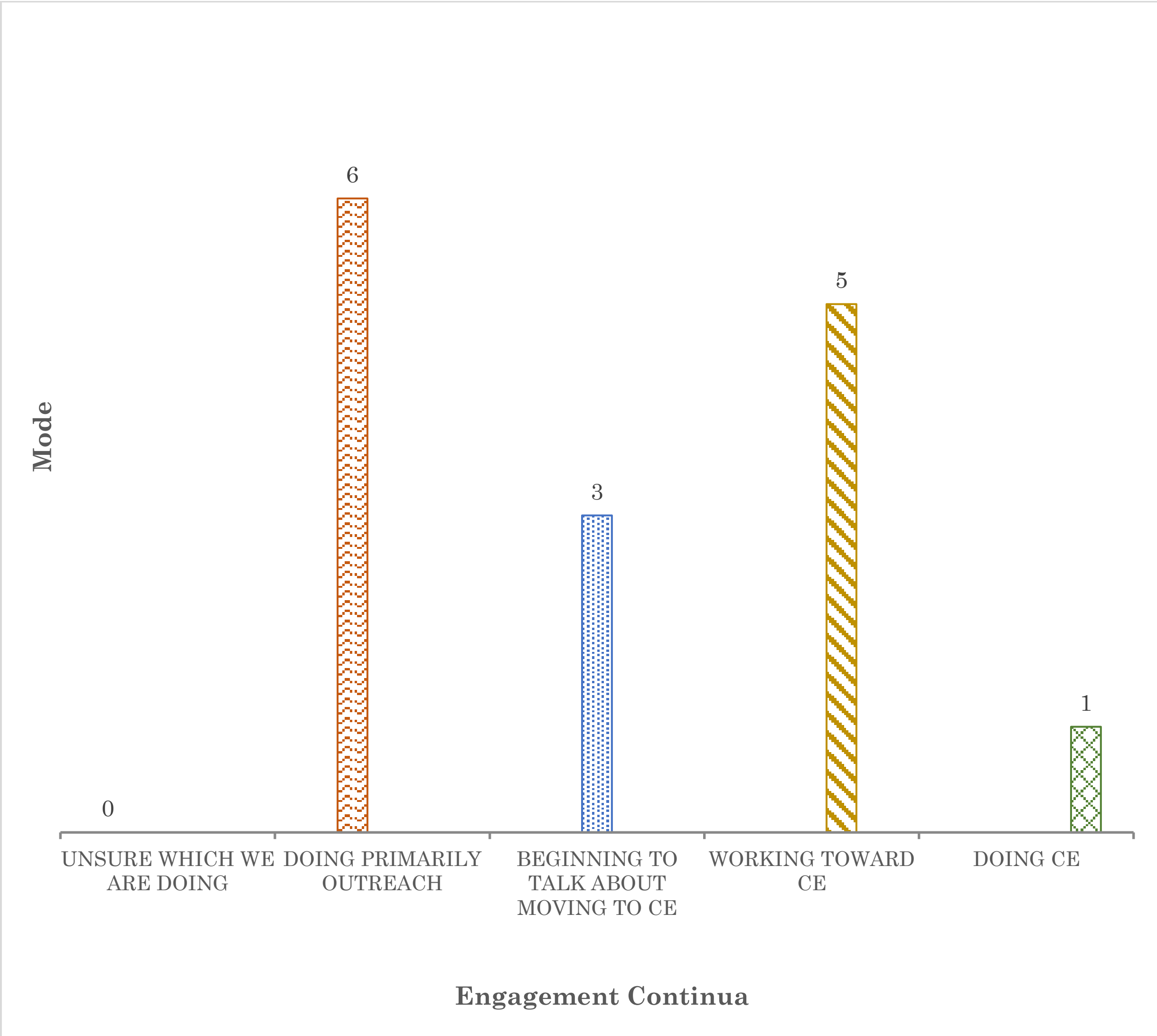

Figure 7-5: Modes of continua (Assessment Period 2)

7.1.3 Assessment Period 3 – post construction

The third assessment period, spanning from June 2022 to January 2023, followed pond retrofit construction completion. Stormwater inflow data, additional stormwater grab samples, and groundwater samples were collected during this phase. Concurrently, community interactions and activities were conducted, including pond clean-up days and several events centred around painting a mural at the School Pond. The process leading up to the mural painting involved engaging the community through discussions about the mural's content, workshops to gather ideas and suggestions for the mural, and the actual painting of the mural.

The engagement-outreach rating for this monitoring period was also evaluated and presented as a bar chart (Figure 7-6). The findings from this exercise indicated that the observed activities hinted at a shift towards more community engagement, and the overall assessment of the activities at the pond indicated that they were mainly community engagement.

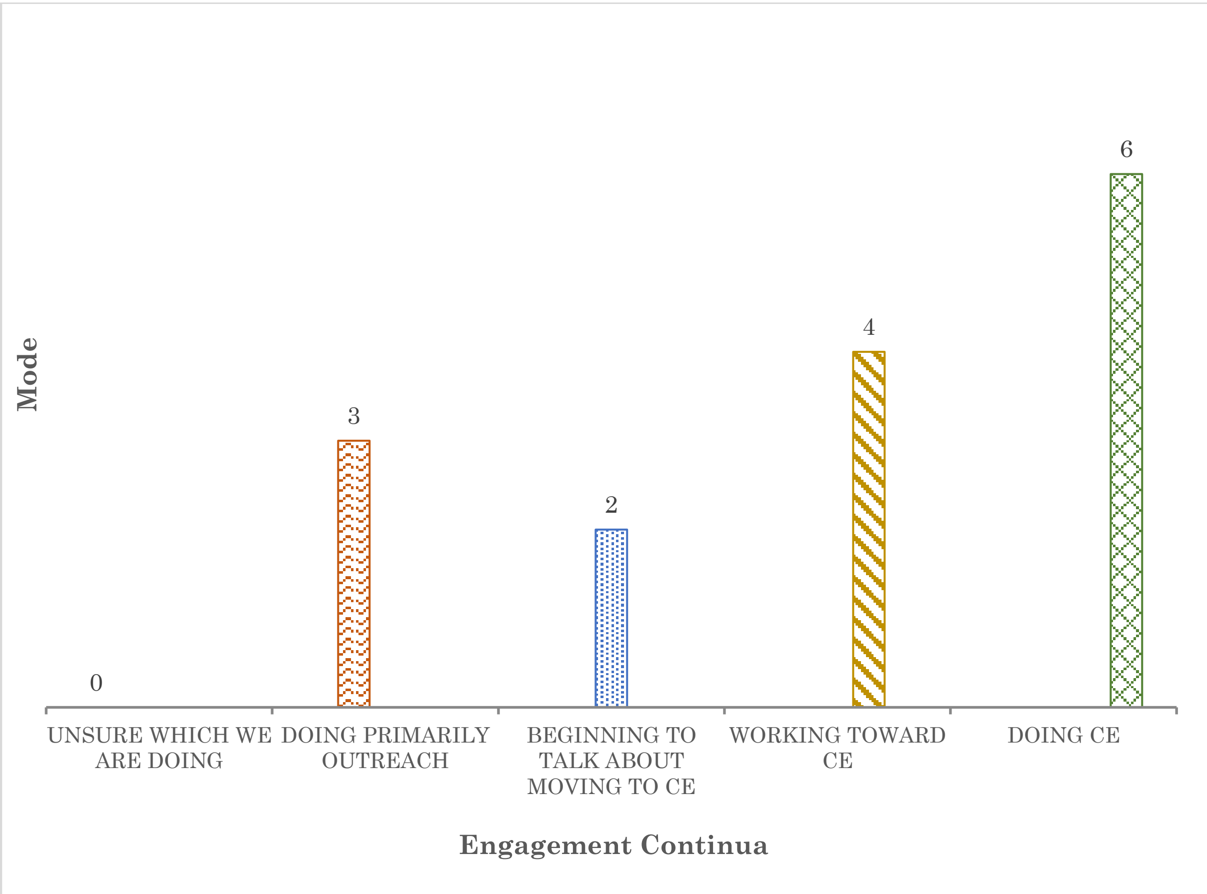

Figure 7-6: Modes of continua (Assessment Period 3)

The transition from an outreach project to an engagement project was evident in more activities during the second monitoring period. An example of this was seen during a mural painting activity at the pond where the community was engaged – via workshops – to determine the artwork and message they wanted on the mural (Mclachlan *et al.*, 2023). These workshops involved a broad mix of members that reflected the diversity (gender and age) of the local community (Figures 7-7a and 7-7b). All the workshop suggestions were incorporated into the mural (Figure 7-7c), with community members assisting with the painting of the mural.

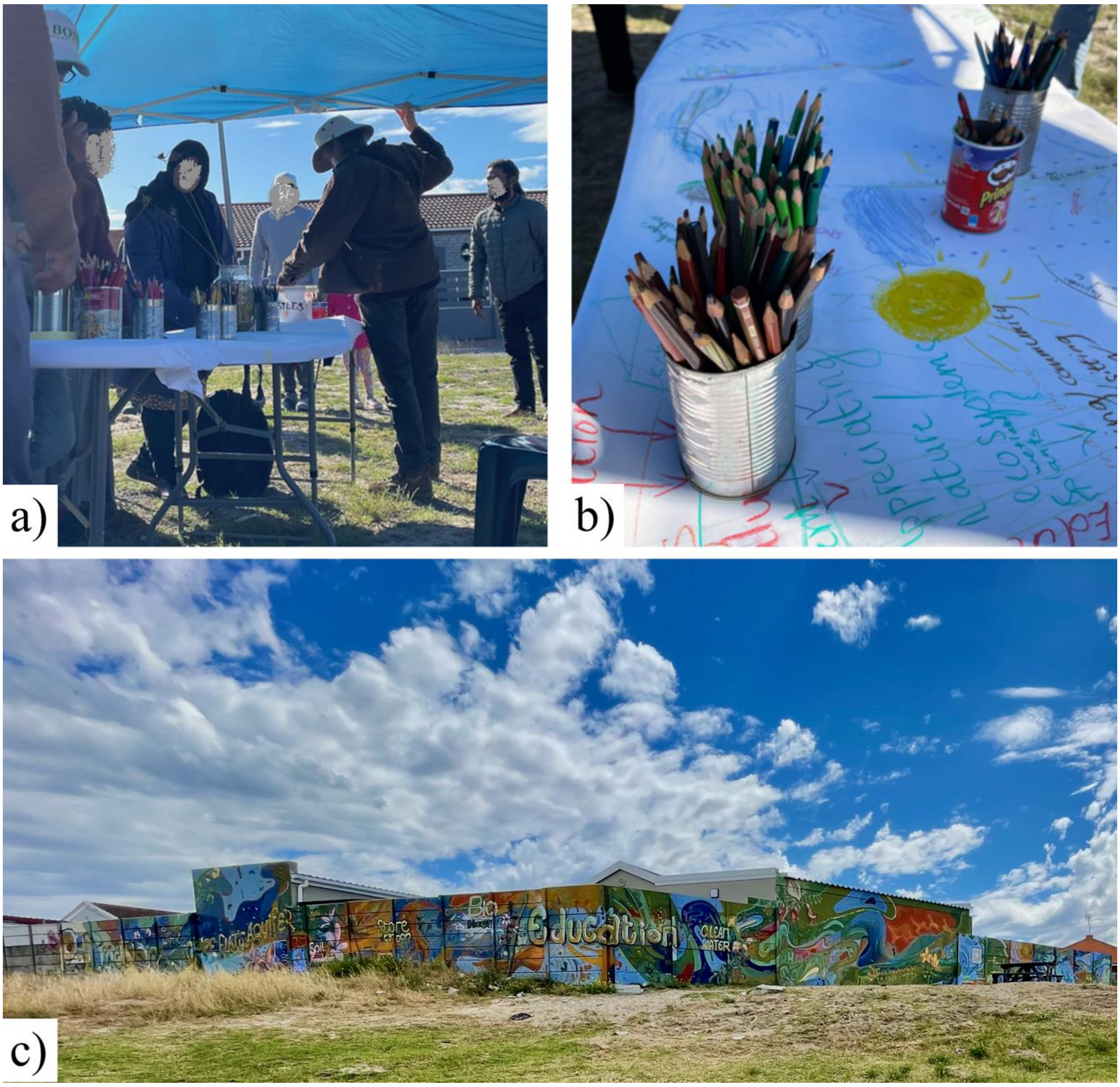

Figure 7-7: a & b) Mural workshop c) School Pond mural

The community was predominantly Muslim but demonstrated inclusivity by creating spaces that welcomed individuals from different cultural backgrounds. For instance, the artist collective involved in the mural project consisted of members of Khoi descent who practiced their ancestral beliefs, such as burning incense at School Pond during community meetings (Figure 7-8). This collaborative effort showcased the openness of some members of the community in embracing different cultural practices and fostering a sense of unity. Moreover, the researchers fostered meaningful and cooperative relationships with willing members of the community. These relationships were built on trust and rapport, leading to informal interactions where community members often visited and engaged in casual conversations during the sampling period. This enhanced level of participation from community members exemplifies the significance of the connections formed, further enriching overall engagement in the project.

Figure 7-8: A Khoi welcome ceremony at the School Pond b) Burning incense

7.1.4 Assessment Period 4 – post construction

An evaluation of the project's progress up to July 2023 was then carried out. The results portrayed in Figure 7-9 illustrate that the project underwent a transition from predominantly focusing on outreach to primarily emphasising community engagement over the duration of the project (January 2020 – June 2023). This shift is supported by the increasing number of engagement-leaning responses (from 7 to 12), while the number of outreach-leaning responses declined (from 8 to 3). These findings also indicate that with adequate time and intentional effort, a project initially centred on outreach can evolve into one that prioritises community engagement. Furthermore, these findings align with the growing body of literature advocating for a 'slow science' approach (Stengers, 2018; Frith, 2020), emphasising allowing projects to unfold gradually. Prolonged project durations enable increased stakeholder interaction, facilitating improved understanding between groups and yielding more favourable outcomes (Bahrami *et al.*, 2012).

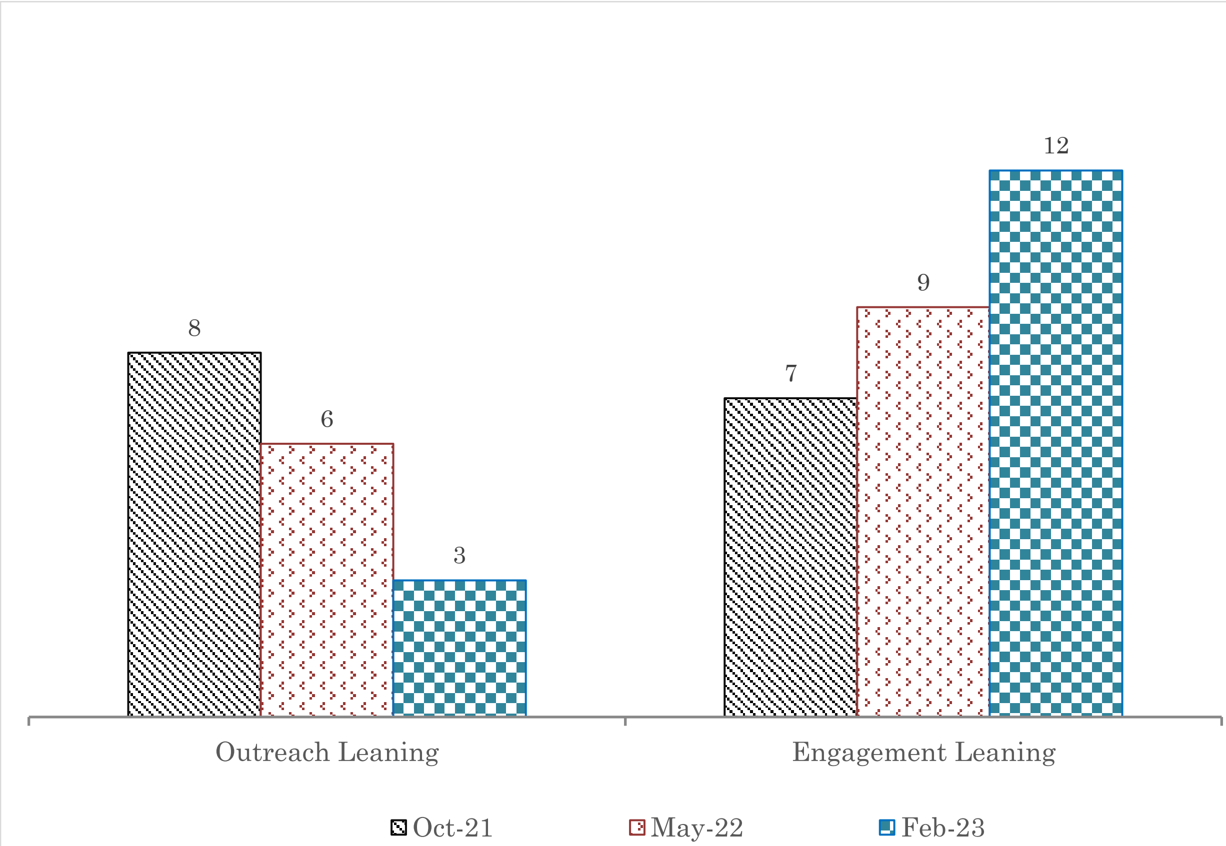

Figure 7-9: Aggregate of responses for all assessment periods

From these assessments and observations, it is argued that pond retrofits in a South African context rely heavily on engaged communities, which suggests that retrofits are developmental projects – commonly found in the humanitarian and WASH sectors (Sorensen & Snel, 2020; Shields *et al.*, 2022). Furthermore, the technical engineering needs of such projects deviate from conventional engineering practices as they require social engagement from the project manager (Tanyanyiwa *et al.*, 2023).

7.2 Chapter summary

In this chapter, a modified community engagement tool was used to evaluate the extent of community participation (outreach vs. engagement) achieved during the School Pond retrofit. Community participation was assessed over four periods of the retrofitting process. An overall evaluation of the project's progress was then conducted.

This evaluation found that the project was initially outreach-centred but transitioned to an engagement-centred project. It was also seen that achieving community engagement required continuous outreach efforts, considering the diverse interests, and competing priorities of community actors. Furthermore, it was found that the engagement process often requires specialised skills beyond the purview of typical water managers or engineers and requires substantial investment in time. Overall, the study highlights that pond retrofits in a South African context rely heavily on engaged communities, and the level of engagement should be continuously evaluated at each stage to ensure that project objectives are met. The modified engagement tool presented in this study can be used for such an assessment.

8. Evaluating the hydrological performance of a retrofitted detention pond

8.1 Overview

This section presents the development and application of a PCSWMM-based hydrological model with surface and subsurface (groundwater) interactions. The model was developed using field- and literature-derived data and was calibrated and validated. Twelve scenarios were developed to evaluate the pond's short- and long-term hydrological performances under different climatic and operational conditions. The PCSWMM software was used because of its extensive technical and research support, including a student grant for the use of the software. The software also explicitly simulates the 'design and placement of SuDS features on surface water and groundwater hydrology' (Mooers *et al.*, 2019).

8.2 School Pond model conceptualisation

Model conceptualisation is abstracting real-world conditions to be used in a simulated environment (Robinson, 2015). It enables the modeller to choose what to model. This process is a major step in understanding the model's complexity, as there is a risk of either overcomplicating or simplifying the problem (James, 2005). Therefore, the first step in model conceptualisation is defining the problem and required outcomes.

In this study, a PCSWM model was developed to evaluate the hydraulic performance of a retrofitted detention pond. The principal purpose of the retrofit was to encourage SWH via MAR by enhancing infiltration in the pond. It was thus required to quantify the infiltrated stormwater volumes to assess the retrofit's geo-hydraulic performance. The infiltrated stormwater volume could have been estimated from a water balance that included inflow, outflow, and evapotranspiration using continuous (real-time) flow measurements in and out of the pond. However, real-time monitoring could not be implemented at the pond due to the risk of instrumentation theft. Consequently, a model was required to estimate the unknown parameters.

The retrofit is idealised in the model as two distinct components: the surface component, which simulates surface interactions such as precipitation, evapotranspiration, and runoff volumes, and the subsurface component, which accounts for surface-subsurface interactions, including infiltration and aquifer storage. These components are interconnected and interact within the PCSWMM environment. Hence, the model required input parameters for surface and subsurface models. The modelling approach employed to simulate the retrofit is presented in the subsequent section as well as the underlying assumptions made during the analysis.

8.2.1 The School Pond's surface model

Precipitation falls onto the pond's catchment area – divided into sub-catchments in the model – with varied topography, land-use/land cover, and infiltration parameters. The precipitation is first stored in any depressions within the sub-catchments and either infiltrates the ground if the sub-catchment is pervious and/or evaporates (James *et al.*, 2010). Once the depression storage (within the sub-catchment) is filled, any additional precipitation becomes runoff which flows along topographical defined flow paths towards the road network. Once in the road, the runoff flows down the road until it meets a catchpit whereupon it is directed to the stormwater network (Figure 8-1). The pond receives stormwater from the network via the two inlets into the pond, in addition the pond receives precipitation. Any stormwater that does not infiltrate or evaporate within the pond exits via an outlet pipe and into the trunk stormwater network.

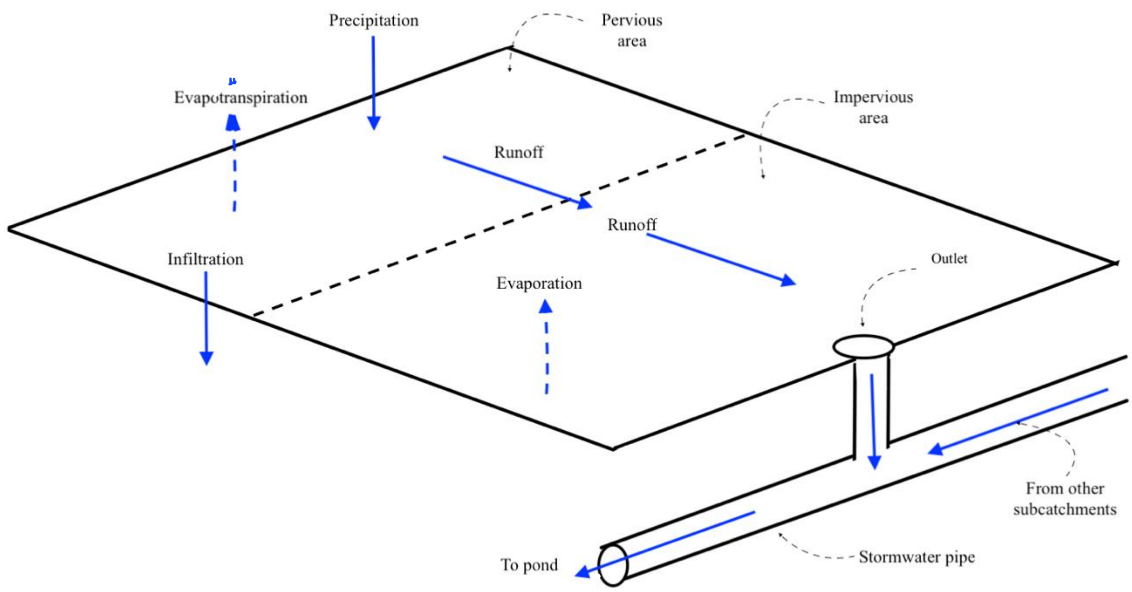

Figure 8-1: Conceptual surface model

The simulation of these processes required consideration of several key model inputsnamely:

- Temporal precipitation data: This parameter was crucial for determining the rate, depth, and intensity of precipitation, which directly influence water input into the system.
- Catchment characteristics: Factors such as land use/cover and slope had a significant role in determining the response time of the catchment to rainfall events and the velocity at which runoff occurred.
- Stormwater network: Parameters related to the stormwater network, including pipe diameters, invert levels, pipe slopes, catch pit locations and geometries, were essential for determining the network's response. These parameters influence flow rates, potential flooding, and overall drainage efficiency.
- Temporal temperature data: Temperature variations influence the evapotranspiration rate, which impacted the system's water loss.

By considering these input parameters, the model can effectively capture and analyse the complex interactions within the system, providing valuable insights into its behaviour and performance.

8.2.2 The School Pond's sub-surface model

Part of the precipitation received in a sub-catchment stored as depression storage which may infiltrate into the ground for pervious surfaces. The rate of infiltration, and hence the volume, is a function of time, percentage of the pervious sub-catchment, soil cover and soil properties. These parameters must be defined in a model but are often impossible to determine accurately. They can be estimated from literature, GIS tools and field tests and adjusted during the model calibration and validation process. In the modelling process, the modeller can incorporate a connection between sub-catchments and an aquifer, enabling the storage of infiltrated volumes within the aquifer. Alternatively, they may opt to neglect this aquifer property, resulting in a model where the infiltration capacity recovers more rapidly compared to real-world conditions. This accelerated infiltration recovery is particularly pronounced when the groundwater table is elevated relative to the ground level – a reduced vadose zone. However, the inclusion of an aquifer introduces greater model complexity, albeit yielding a more realistic representation of the system.

In this study, the subsurface processes were modelled with a particular focus on the processes occurring within the School Pond. The pond has been retrofitted to enhance infiltration and a model was employed to estimate the inflow and volume of stormwater infiltrated into the pond. The retrofitted pond was thus modelled as a SuDS/LID feature. The retrofit process and final design are discussed in Chapter 6.

The PCWSMM software has a LID component which allows the user to model several LIDs, such as infiltration trenches, bio-retention cells, rain gardens, permeable pavements, rain barrels, and rooftop disconnections (gutters) and vegetative swales (James *et al*., 2010). Figure 8-2 illustrates the various layers that an LID can possess and the processes that occur within and between the layers.

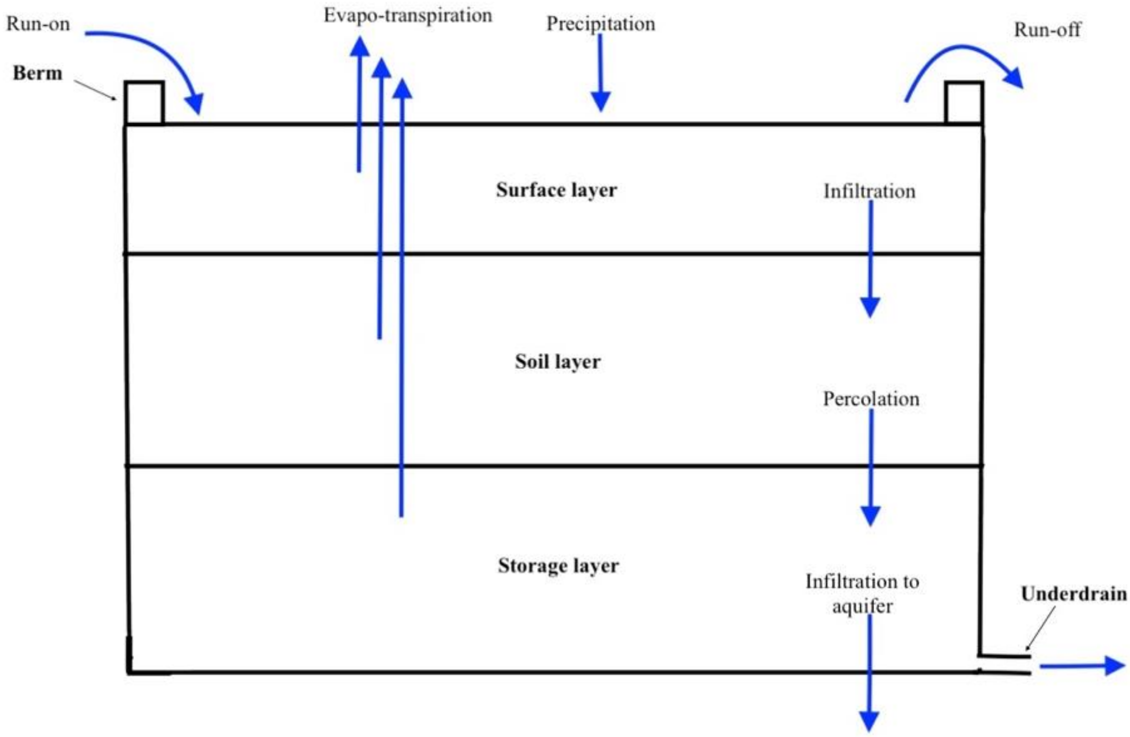

Figure 8-2: Typical LID layers and processes in PCSWMM

The available LID layers in PCSWMM may be described as follows:

- The surface layer corresponds to the ground surface receiving direct rainfall and run-on from upstream areas, storing surplus inflow in depressional storage, and producing surface outflow, entering either the drainage system or flow into adjacent lower catchments.

- The soil layer is the engineered soil mixture used in bio-retention cells and rain gardens to support vegetation growth. It can percolate water into the underlying storage zone.
- The storage layer is a bed of crushed rock or gravel that provides storage for the infiltrated water in bio-retention cells, permeable pavements, and infiltration trench systems.
- The underdrain conveys water from the gravel storage layer into a standard outlet pipe or chamber.

PCSWMM performs a moisture balance calculation during a simulation for each layer and keeps track of water movement and storage. Table 8-1 indicates the combination of layers that apply to each LID type.

Table 8-1: LID types and associated layers
(after James *et al.*, 2010)

LID Type	Surface	Pavement	Soil	Storage	Drain	Drainage Mat
Bioretention Cell	x		x	x	o	
Green Roof	x		x			x
Infiltration Trench	x			x	o	
Permeable Pavement	x	x	o	x	o	
Rain Barrel				x	x	
Rain Garden	x		x	o		
Rooftop Disconnection	x				x	
Vegetative swale	x					

'x' denotes required layers, and 'o' means optional.

In this study, the 'School Pond' infiltration swale was modelled using a PCSWMM component that simulates the infiltration swale, the retaining berm, and the groundwater exchange. The rain garden LID with no storage was used as the 'School Pond' retrofit was not intended to store water within its layers but transfer it to the underlying aquifer. The conceptualised infiltration swale consisted of two zones: the topsoil (surface layer), and the initially unsaturated zone (soil layer), whose saturation level varied because the water table fluctuates.

The surface layer, which receives direct rainfall and runoff from the catchment, was allocated the infiltration and land cover parameters. However, the soil layer used in the 'School Pond' was not an engineered soil mixture and the infiltration swale overlies the existing soil. Water infiltrates the soil layer from the surface layer and percolates down into the aquifer below the pond (Figure 8-3). To represent the soil layer, parameters were assigned based on site and laboratory studies, as well as relevant literature when on-site or laboratory measurements were not feasible.

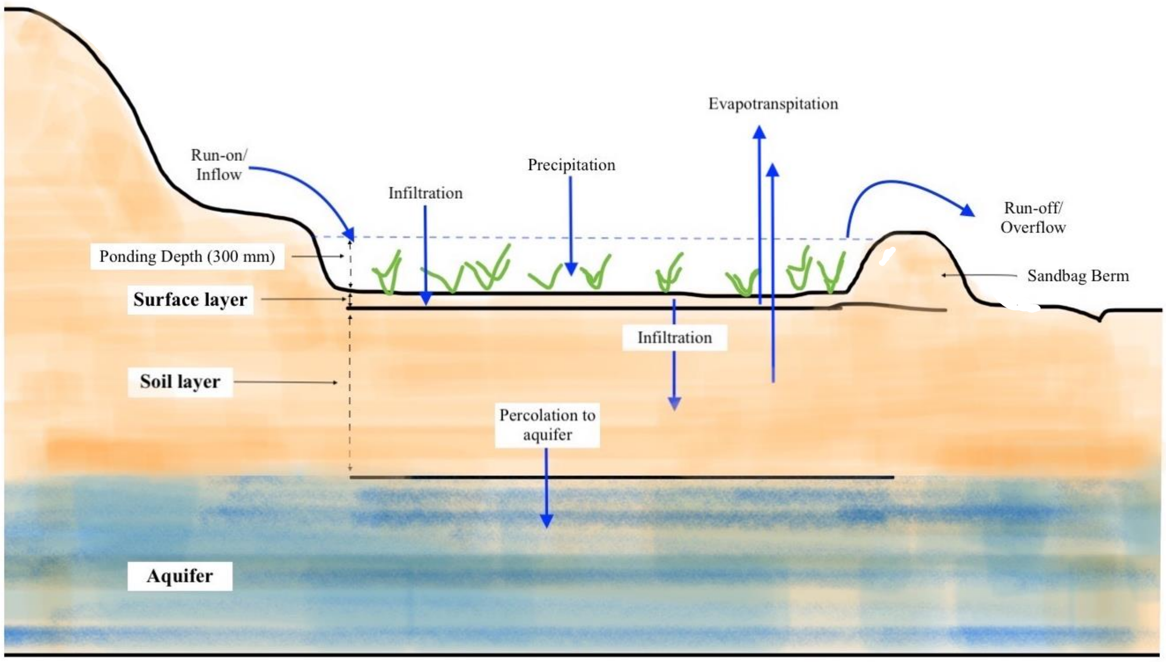

Figure 8-3: Conceptualised 'School Pond' surface-subsurface processes

8.3 Model development

The School Pond PCSWM model was developed using data from the CoCT, the UCT-GIS laboratory, literature, and field surveys. Local five and 15-minute interval rainfall, wind speed, and temperature data were obtained from an automated weather station installed at the school. Extended climatic data from 2005 – 2022 was obtained from two weather stations managed by the South African weather services (SAWS). The water table levels, soil data, and infiltration characteristics were obtained from a site-level hydrological survey.

A high-resolution 0.5 m bare earth digital elevation model (DEM) obtained from a LiDAR survey was loaded onto the PCSWMM Professional (ver. 7.5) interface as a raster file. A Google map background was also added to the model project to aid in visualising the location of the ponds. The CoCT stormwater infrastructure shapefile was then loaded on the interface, and the Hartebeesthoek94_lo19_(E-N) ESRI:102562 projection system – the WGS 1984 ellipsoid was implemented for all shapefile layers and utilised as the models' default projection system.

The watershed and sub-catchment delineation tool – which analyses a DEM and creates sub-catchments and flow paths where each sub-catchment drains into a node – was then applied to the DEM to delineate the study area's sub-catchments with a discretisation (level of detail of sub-catchments) of 0.25 hectares. A sub-catchment is an area of land usually containing a mix of pervious and impervious surfaces whose runoff drains to a set node or outlet. The flow paths and areas that contributed to the pond inflow and outflows were then defined. Non-contributing parts of the DEM were excluded from the delineation output if they did not contribute toward flow into the pond. The remaining area, comprised of contributing sub-catchments, was considered the pond's catchment. The resulting sub-catchments were modified to depict separate land uses. (Figure 8-4)

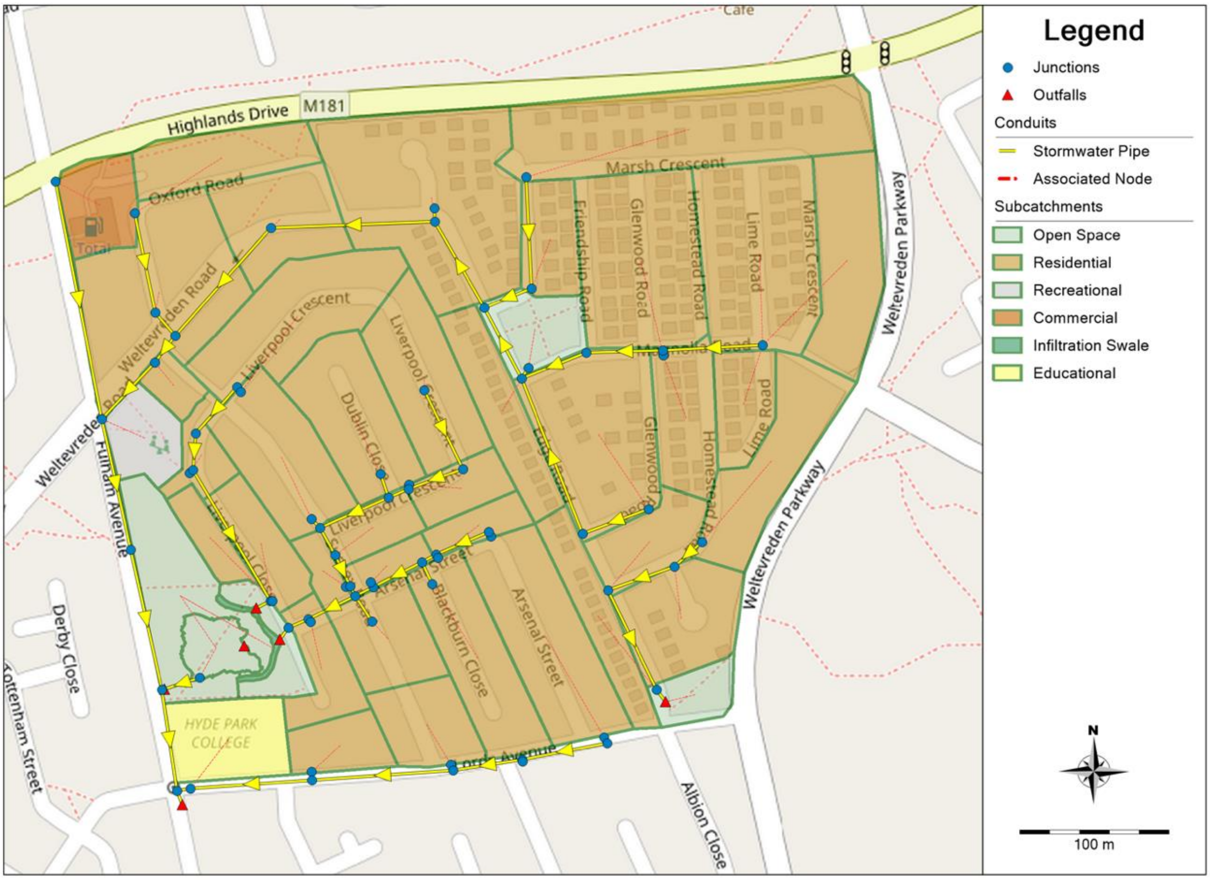

Figure 8-4: 'School Pond' PCSWM Model

(Image source: Google, 2023)

All sub-catchment geometric properties, such as contributing area and slope, were automatically assigned and stored on the software. Other parameters, such as land cover and soil properties, were assigned from literature and field data. The model was calibrated and validated using the observed experimental data (rainfall depths, temperature, and inflow rates). A LID was then allocated to the pond to simulate the retrofitted swale.

The calibrated model with the infiltration swale was then used to assess the hydraulic performance of the retrofitted infiltration swale under different scenarios. The model development process is summarised in Figure 8-5 while the process of obtaining, setting up and validating the model input parameters is described in the subsequent sections.

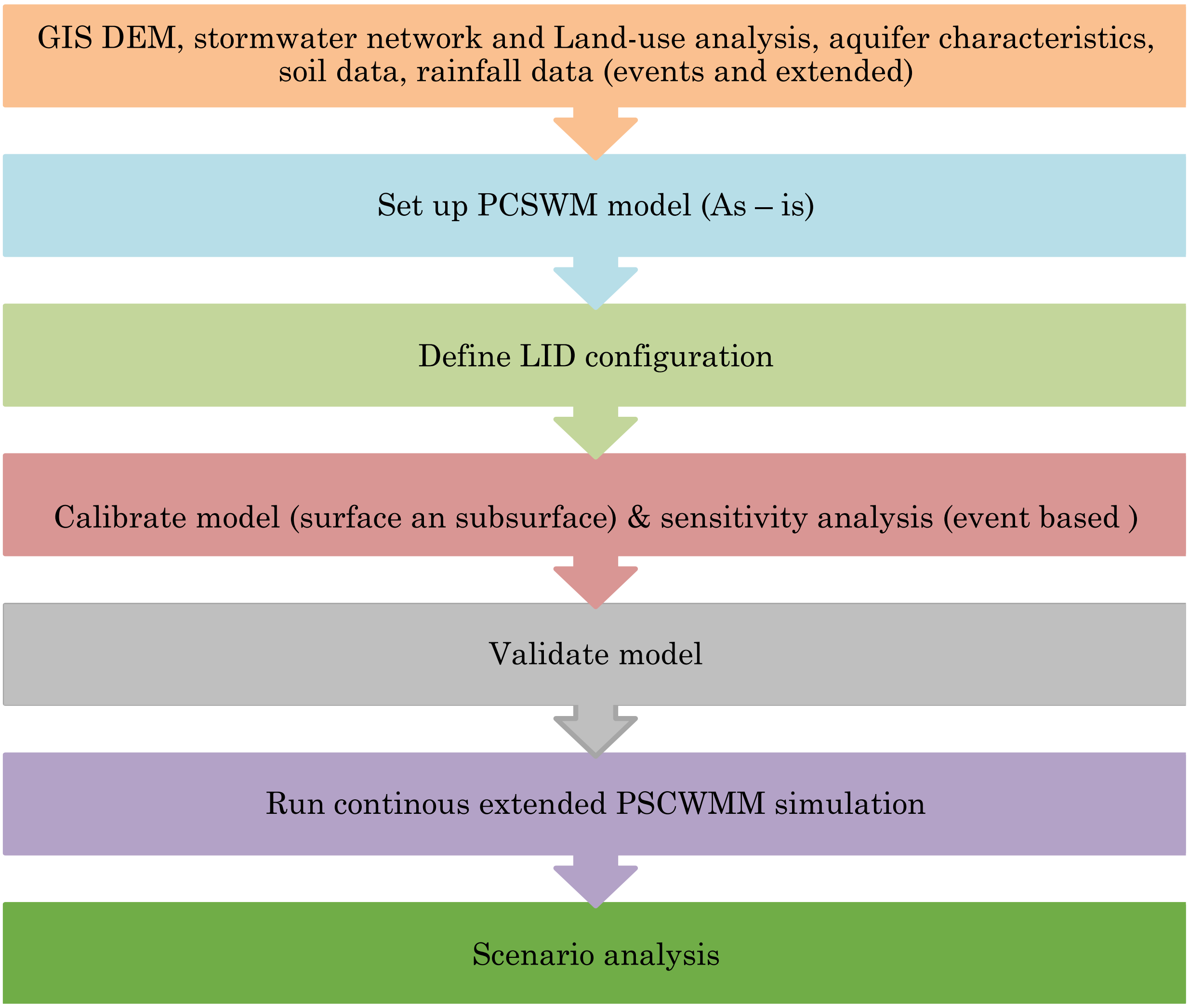

Figure 8-5: Model development flowchart

8.3.1 Model input parameters

This section covers the input parameters utilised in the model after the various stages of data collection.

8.3.1.1 Rainfall data

The model developed in this study relied on two distinct rainfall datasets. The first dataset was utilised for model calibration and validation, specifically using defined continuous rainfall events. Obtained from an internet-connected weather station installed at the School in September 2021, this dataset ensured accurate and localised rainfall information. The second rainfall dataset was utilised for the continuous extended simulation. This dataset was sourced from a rain gauge located 3.6 km away from the pond, known as the Wolfgat Weather Station (Figure 8-6). Owned and maintained by the South African Weather Services (SAWS), this rain gauge provided reliable rainfall data for the extended simulation period.

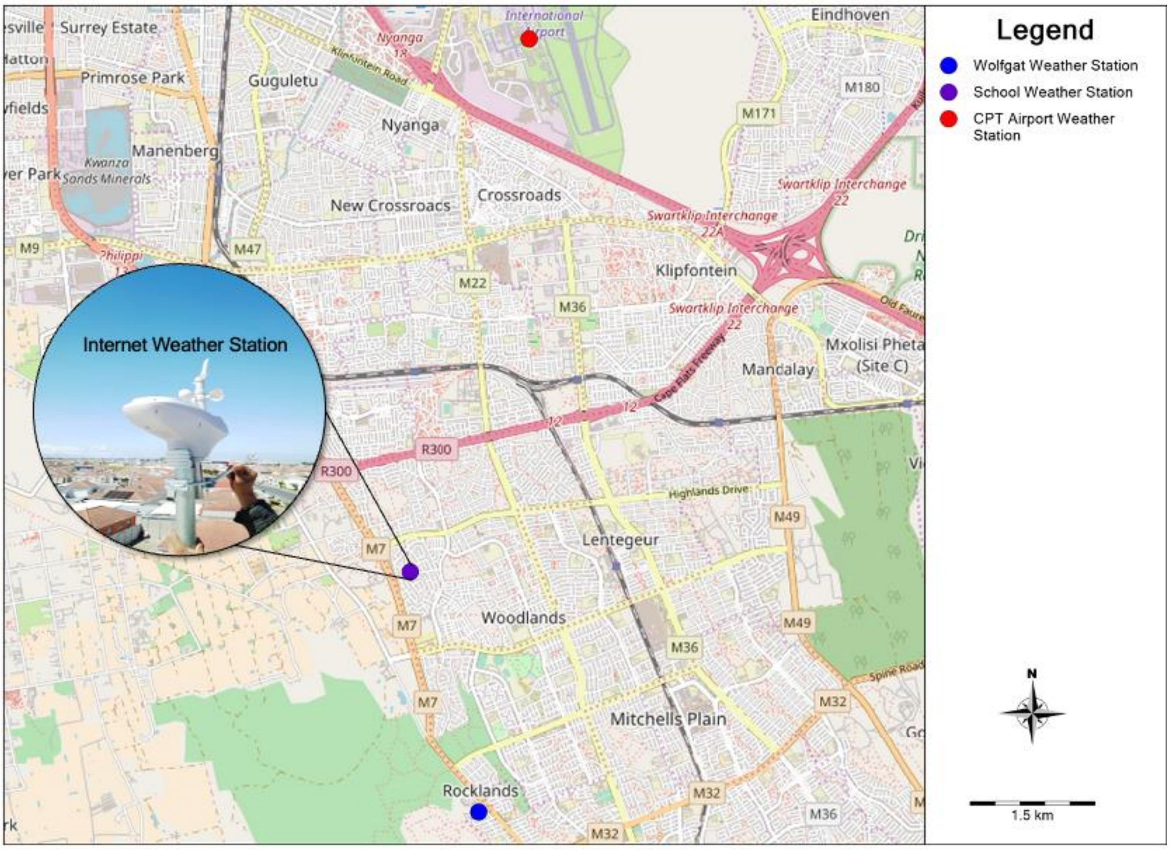

Figure 8-6: Location of weather stations used in the study
(Image source: Google, 2023)

A comparative analysis was conducted to establish the consistency and similarity of the two rainfall datasets. Rainfall depths from both stations were compared using data from October 2021 to February 2022, a period characterised by uninterrupted data collection without power disruptions. Figure 8-7 illustrates the rainfall data for the mentioned periods.

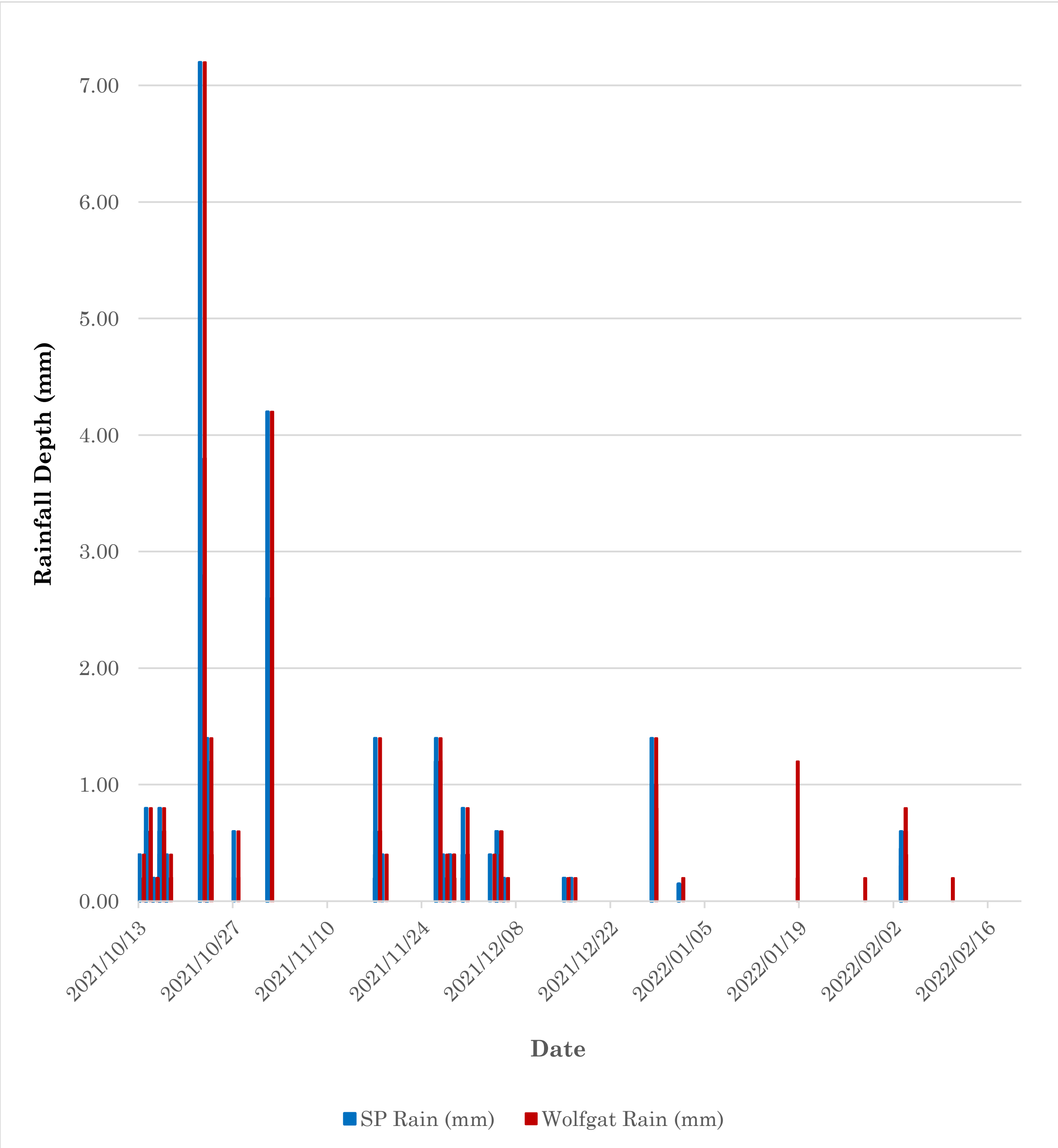

Figure 8-7: Comparing the School Pond and Wolfgat rain gauges

T-tests, detailed in Table 8-2, were carried out to evaluate the equality of means between the two datasets. The null hypothesis assumes that the means are equal. The p-value from this analysis was > 0.05 indicating an inability to reject the null hypothesis. Thus, the mean rainfall from the two datasets is statically similar and comparable. Furthermore, the regression analysis revealed a strong positive correlation between the datasets, with a correlation coefficient of 0.97. This correlation further validated the consistency and reliability of rainfall measurements.

Table 8-2: Paired t-Test for the School Pond and Wolfgat datasets

Parameter	School Pond Rain (mm)	Wolfgat Rain (mm)
Mean	0.00356	0.00362
Variance	0.00469	0.00475
Observations	37728	37728
Sum (mm)	134.4	136.5
Hypothesized Mean Difference	0	
Degrees of freedom	75451	
t stat	-0.117	
P(T<=t) one-tail	0.453	
t critical one-tail	1.64	
P(T<=t) two-tail	0.907	
t critical two-tail	1.969	

The SAWS provided the unprocessed data from the Wolfgat station in Excel formats. The rainfall data from the Wolfgat and School Pond weather stations were processed on Microsoft excel, creating PCSWMM-supported data files (.dat), which were interpreted as time series files. The continuous rainfall events recorded by the School Weather station were used to calibrate and validate the model. The rainfall data from the Wolfgat station (Figure 8-8) was used for the extended continuous runs. The time series files produced on PCSWMM were then used to create rain gauges within the software (Figure 8-8).

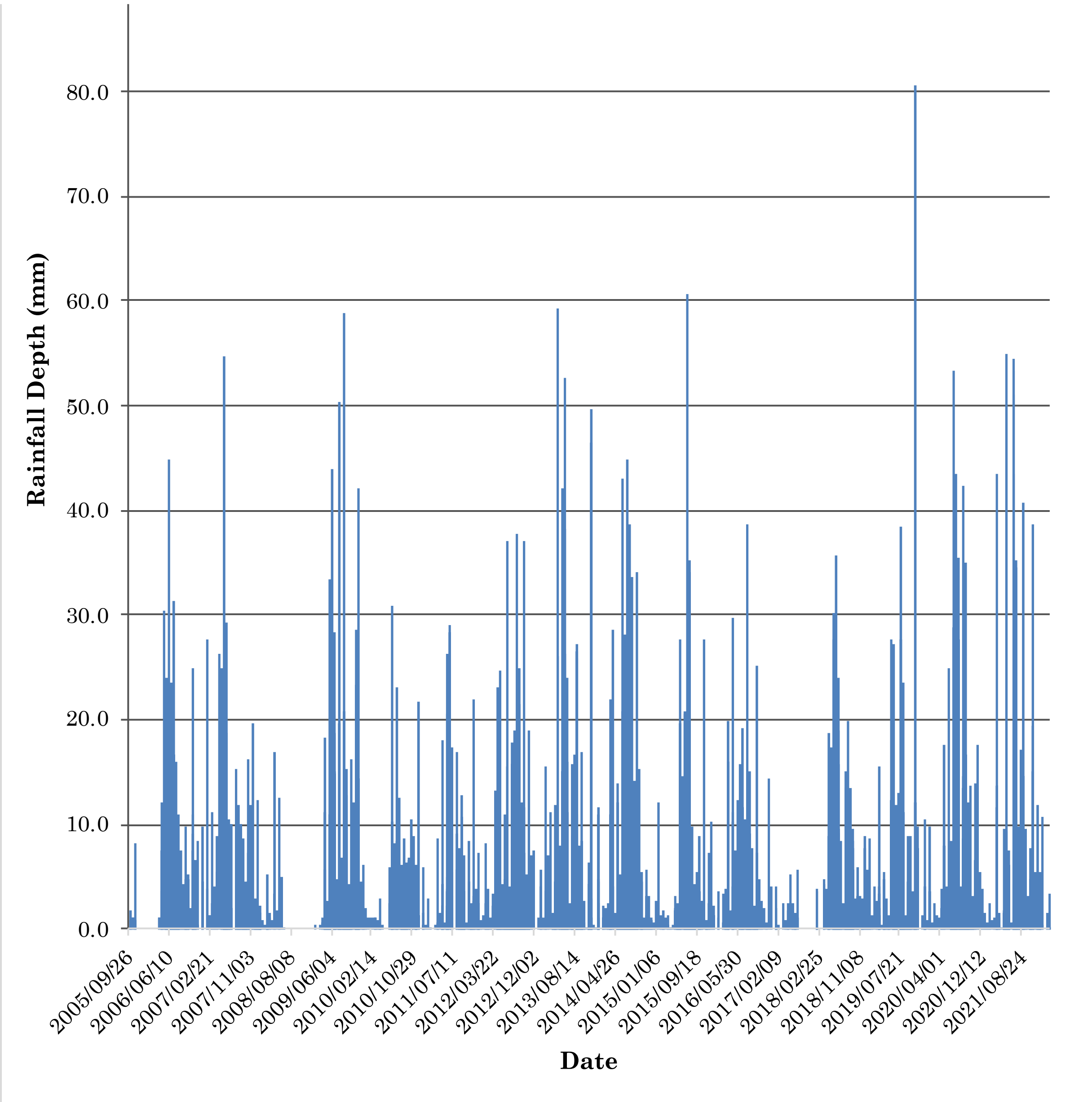

Figure 8-8: Extended continuous daily rainfall data from the Wolfgat station (2005 - 2022)

8.3.1.2 Stormwater network

The residential area where the School Pond is located (Rondevlei Park) was developed in 2000 in two phases, with Phase 2 completed in 2008. The As-built drawing for the residential area, including the road and stormwater network, was obtained from the CoCT's catchment, stormwater & river management department. The As-built drawings included the pipe diameters, slopes, invert levels, catch pit locations, and the inlet and outlet positions in the detention pond (Figure 8-9). They were digitised on PCSWMM, while the pipe parameters were captured on an Excel sheet and then transferred to the software (Figure 8-4).

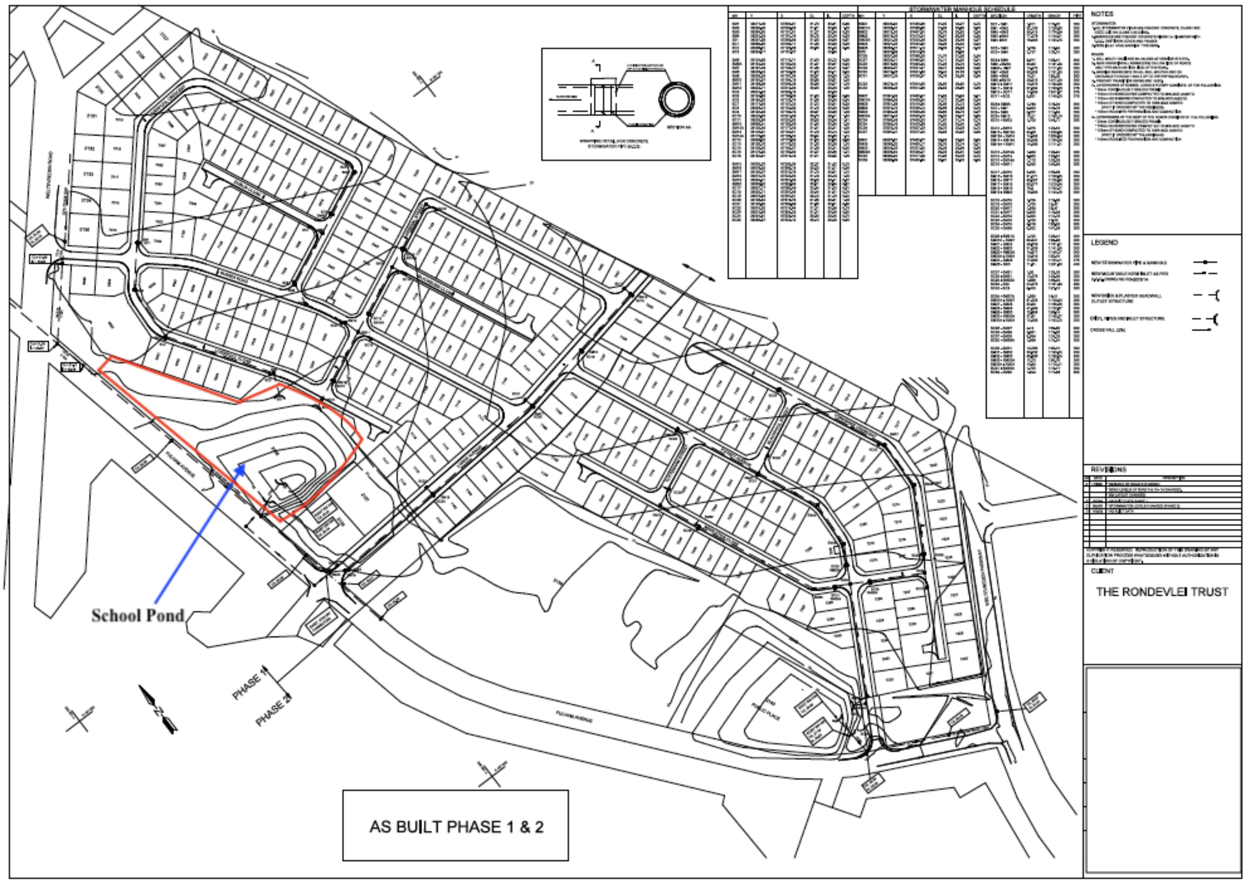

Figure 8-9: An extract from the Rondevlei Park as-built stormwater network drawings

8.3.1.3 Sub-catchment characteristics

The SWMM5 computational engine conceptualises each sub-catchment as a non-linear 'reservoir'. The sub-catchments experience sheet flow – runoff flowing as a thin continuous layer of water over the surface, also called overland flow (Davie, 2010). In the non-linear reservoir conceptualisation, the inflow (run-on, precipitation) is routed and captured in a 'reservoir' with a set depth d and a storage depth d_p (Rossman & Huber, 2016). The mass flow balance then comprises run-on, precipitation, infiltration, evaporation, and sub-catchment flow Q, defined by Manning's equation (Figure 8-10). The net change in depth of water, d, per unit time is updated with each timestep according to the numerical solution of the mass balance (Equation 8-1).

$$\frac{\partial d}{\partial t} = i - e - f - q$$ **Equation 8-1**

Where i is precipitation including run-on (m/s), e is the surface evaporation rate (m/s), f is the infiltration rate (m/s), and q is the runoff rate (m/s).

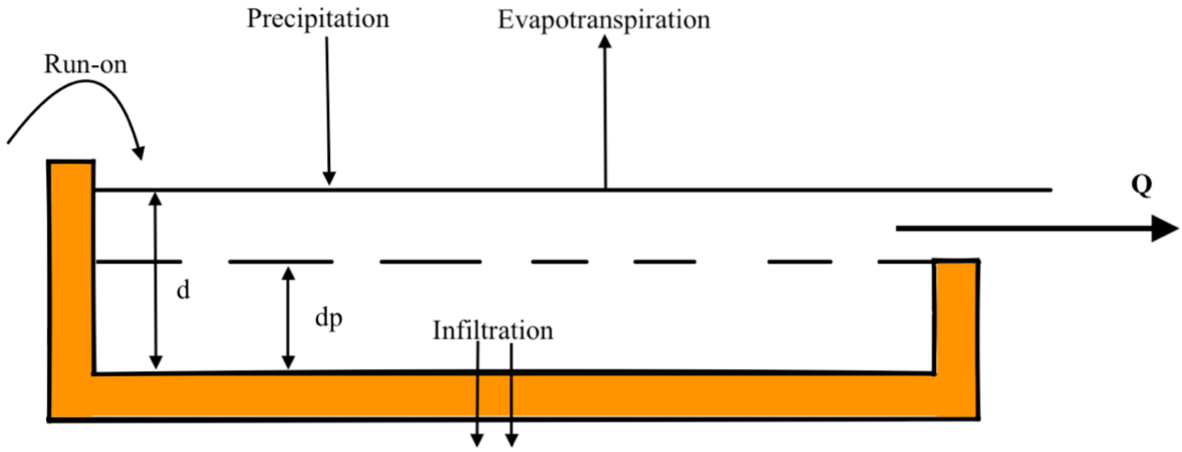

Figure 8-10: Conceptual view of surface runoff
(After James *et al.*, 2010)

In this study, the mass balance in the SWMM5 engine was calculated using user-declared parameters that influenced sub-catchment runoff. First, the sub-catchment use was categorised as residential, educational, commercial, and open land. These land-use categories were then allocated properties based on literature and desktop study results. The main sub-catchment runoff properties utilised in the model were impervious percentage, Manning's coefficient (n) and depth of depression storage for the impervious and pervious surfaces.

The imperviousness percentage – the percentage of the impervious sub-catchment area – was initially evaluated using a supervised classification GIS tool. The Semi-Automatic classification plugin in the QGIS software was used to classify the land use in the study area. Satellite data (imagery) from the Sentinel-2A open dataset was downloaded, pre-processed and post-processed using the Semi-Automatic classification tool. The satellite data contains raster files with different spectral bands used to evaluate parameters such as vegetation cover, water, built-up and bare soil (Congedo, 2021). However, the raster data obtained from the

Sentinel-2A satellite had a pixel density of 10 m – each pixel in the raster represents a 10 x 10 m area on the ground – making it impossible to clearly distinguish between objects in the same pixel. Consequently, training the tool to classify pixels as vegetation, roads, pervious and impervious was not fully successful (Figure 8-11a) but produced a better land-use classification from the study area compared with the publicly available South Africa National land cover classification (SANLC) (Thompson, 2019) (Figure 8-11b).

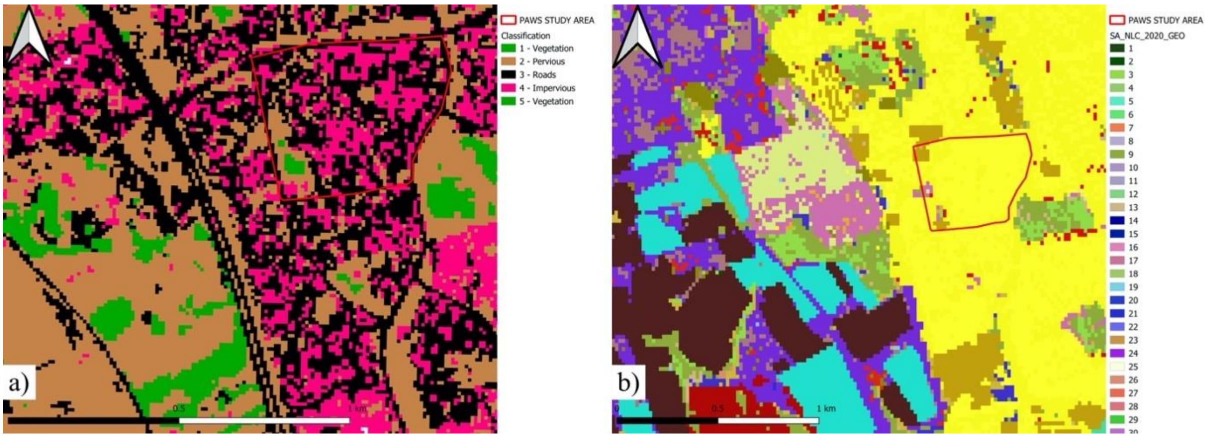

Figure 8-11: a) land-use classification (Semi-automatic classification tool) b) land-use classification (SANLC dataset)

High-resolution satellite imagery (not raster) was acquired through Google Earth Pro to evaluate the imperviousness percentage. The high-resolution satellite image of the study area was used as a backdrop, and for each sub-catchment, the visible pervious regions (bare earth and grass) marked up, and their areas recorded (Figure 8-12). The pervious areas for each sub-catchment were summed, and the impervious fraction of each sub-catchment was obtained. The impervious percentage for each sub-catchment was then obtained.

Manning's roughness coefficients (n) for the impervious and pervious areas in the sub-catchments were derived from literature and applied to the model. The depth of depression storage (D_{store}, denoted as d_p in Figure 8-10) for impervious and pervious areas was also derived from the literature. Finally, the percentage of impervious areas with no depression storage (zero impervious %) was estimated from reference manuals.

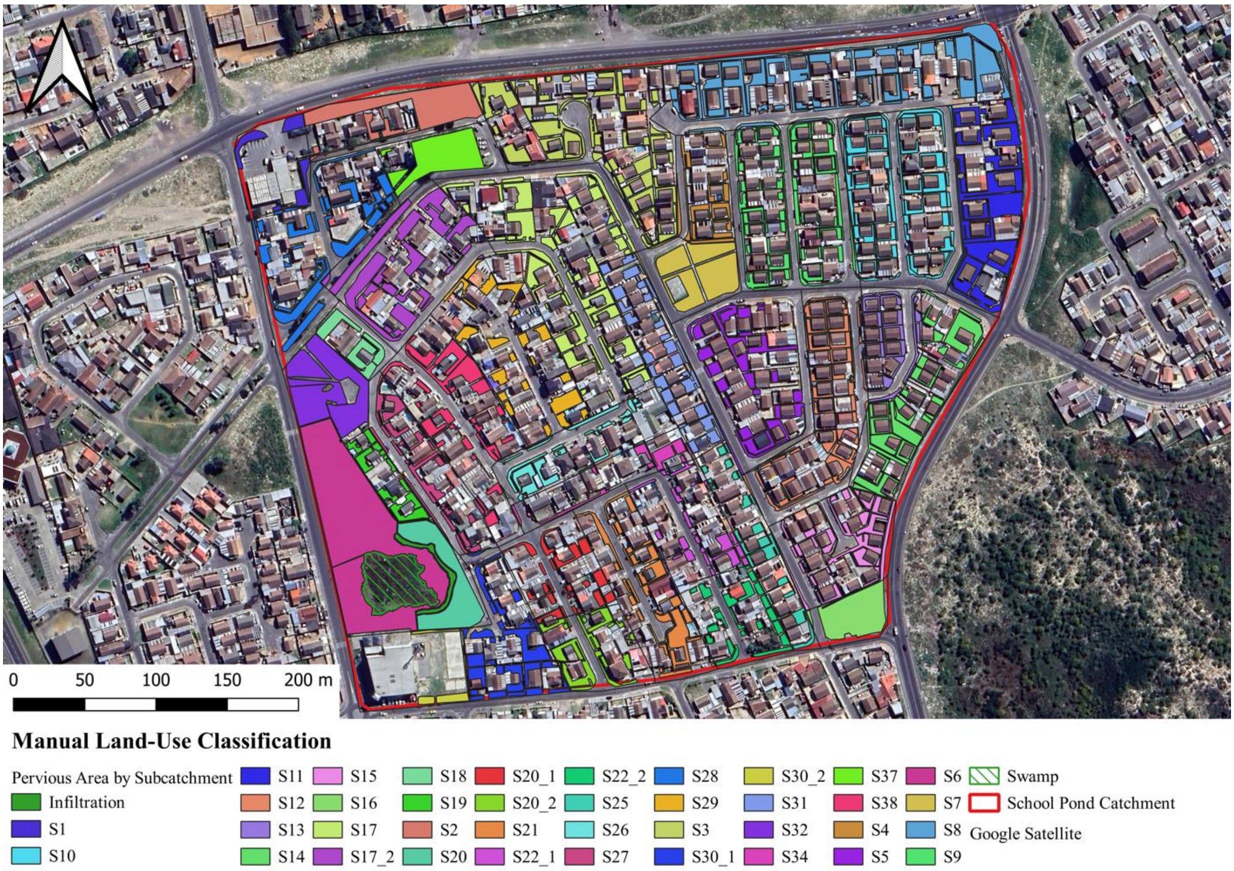

Figure 8-12: Manually estimated pervious areas per sub-catchment
(Image source: Google, 2022)

These parameters and their respective references are summarised in Table 8-3. A set of properties was assigned to each land use and replicated for all sub-catchments within the same land use class.

Table 8-3: Sub-catchment runoff properties for the 'School Pond'

Parameter	**Reference/Source**
Imperviousness (%)	GIS Catchment analysis – QGIS and Google Earth Pro
N Impervious	Chow, 1959
N Pervious	Chow, 1959
Dstore Impervious (mm)	Rossman & Huber, 2016
Dstore Pervious (mm)	Rossman & Huber, 2016
Zero Impervious (%)	Rossman & Huber, 2016

8.3.1.4 Infiltration

The infiltration parameters considerably influence the hydrograph and must be carefully considered in hydraulic modelling. In this study, the infiltration parameters were defined by empirical and experimental methods. The Soil Conservation Service (SCS) hydrologic method (Hubner *et al.*, 2016) was used for the built-up areas, while the infiltration in the pond area was derived from earlier field experiments where infiltration in the pond was measured at ten different locations at the surface and 300 mm below ground level.

The Soil Conservation Service (SCS) hydrologic method estimates peak flows and hydrographs for all applications, but its application is limited to catchments smaller than 2000 acres (Hubner *et al.*, 2016). The method required the user to know the runoff curve number (CN) determined by soil group type, land cover type and hydrologic condition. A previous hydrogeological survey of the area and texture analysis based on Ollis *et al.* (2013) categorised the soil as sand. Therefore, the Rondevlei hydrologic soil group is classified as Group A under the SCS method. A curve number of 77 was obtained from Table 2.1.5-1 in GSMM (2001) using the mean residential lot size in the catchment (200 m^2) for Group A soils.

The infiltration parameters of the undeveloped areas, including the pond area, were derived using the Horton infiltration model defined by Horton's equation:

$$f_p = f_c + (f_0 - f_c)e^{-kt} \qquad \textbf{Equation 8-2}$$

Where f_p is the infiltration capacity at time t, f_c is the final steady state infiltration capacity f approaches asymptotically, f_0 is the soil's initial infiltration capacity at the start of the infiltration event, and k is Horton's decay constant.

Alternative infiltration models, such as the Green-Apt and the Philip two-term infiltration models, can also be used. However, studies have demonstrated that when these models are utilised within calibrated models, their performances are comparable (Michaud & Sorooshian, 1994; Chahinian *et al.*, 2005; Wang *et al.*, 2017; Vand *et al.*, 2018). The parameters for Horton's infiltration used in this study are detailed in Table 8-4. They are derived from data obtained from double-ring infiltration tests performed before the retrofit. The double-ring infiltration test is described in many texts, such as Davie (2010). This study divided the pond into four sub-catchments (Table 8-4), and for each sub-catchment, Horton parameters were calculated from field data. In addition, the SCS curve numbers were used in the other sub-catchments in the study and were important calibration parameters.

Table 8-4: Horton's Infiltration parameters
(Jones, 2019 – unpublished BSc (Eng.) report)

Sub-catchment	F_o (mm/hr)	F_c (mm/hr)	k (/hr)
Infiltration_swale	862.3	265.7	44.5
Swampy_area	389.0	204.4	37.3
General_pond_area	862.9	163.7	43.1
Slope_to_pond	1308.4	360.0	43.9

8.3.1.5 Evapotranspiration

Evapotranspiration (ET) is the process of water transfer from the soil surface through evaporation and from plant stomata through transpiration (Stanhill, 2005). This parameter is significant in surface modelling as evapotranspiration can result in a substantial amount of water volume lost to the atmosphere and overlooking it results in an overestimate of the infiltrated water volumes and groundwater levels.

Evapotranspiration can be calculated using recorded data, Class A pan evaporation rates and pan coefficients or empirical evapotranspiration models. Several empirical evapotranspiration models have been developed to calculate evapotranspiration, such as the Penman-Monteith, Priestly Taylor, Blaney-Criddle and Hargreaves methods (Penman, 1948; Blaney, 1952; Priestley & Taylor, 1972; Hargreaves & Allen, 2003). While each method (model) has its advantages, the Hargreaves equation is often used in data-scarce regions as it requires the least data to estimate ET_0 – only the extra-terrestrial radiation and maximum and minimum temperature are required. It is an empirical model that has evolved since its inception in 1975. Updated and larger datasets have been used to improve their reliability, and modifications have been implemented to account for variables such as cloud cover. The latest and most popular model was derived in 1985. When evaluated against other evapotranspiration models and experimental data, the Hargreaves method produces similar values (Temesgen *et al.*, 1999; Hargreaves & Allen, 2003; Okedi, 2019). The 1985 Hargreaves model is defined by Equation 8-3.

$$\mathrm{ET} = 0.0023\mathrm{R_a}\,(\mathrm{TC} + 17.8)\,\mathrm{TR}^{0.50} \qquad \textbf{Equation 8-3}$$

Where ET – Evapotranspiration (mm day^{-1}); R_a – Extra-terrestrial radiation (MJ m^{-2} day^{-1}); TC – mean temperature (^oC); TR – daily temperature range (^oC) (i.e., T_{max} - T_{min}, where T_{max} and T_{min} are the mean daily maximum and minimum temperature respectively).

In this study, the Hargreaves method (Equation 8-3) was used to derive the hourly evapotranspiration values using the best publicly available extra-terrestrial radiation dataset collated and published on an online repository by Allen *et al.* (1998). The mean extra-terrestrial radiation data for the 'School Pond' (-34 S) obtained from Allen *et al.* (1998) is shown in Table 8-5.

Table 8-5: Daily mean extra-terrestrial radiation for 34^{o}S
(Allen *et al.*, 1998)

Month	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Rs (MJ m^{-2} day^{-1})	43.4	39.2	33	25.3	19.2	16.2	17.4	22.3	29.6	36.7	42	44.1

In addition, historical temperature data were obtained from the closest weather station to the study area that had, *inter alia*, historical daily mean, maximum and minimum temperature data, which was the Cape Town Airport station located ~ 6.5 km from the 'School Pond' (Figure 8-6). The then mean daily temperature and extra-terrestrial radiation were subsequently utilised to compute the daily evapotranspiration, which was then incorporated into the PSWMM software to simulate evapotranspiration.

8.3.1.6 LID control parameters

The retrofitted infiltration swale was modelled as a LID (SuDS) feature on the PCSWMM software. The swale LID was modelled as an infiltration swale comprising of surface and soil layers. Table 8-6 contains the surface layer data used in PCSWMM's LID control editor, while Table 8-7 contains the soil layer parameters.

Table 8-6: Surface layer parameters

Parameter	Values	Source
Berm height (mm)	300	Constructed berm height
Vegetation volume (fraction)	0.8	Google Earth imagery
Surface roughness (Manning's)	0.03	Chow (1959)
Surface slope (%)	3.3	Constructed infiltration swale slope

Table 8-7: Soil layer parameters

Parameter	Values	Source
Thickness (mm)	500	Swale surface to lowest recorded water table depth
Porosity	0.29	Laboratory experiments
Field capacity	0.19	James *et al.* (2010)
Wilting point	0.085	James *et al.* (2010)
Conductivity (mm/hr)	110	Mavundla (2022)
Conductivity slope	9.8	Mavundla (2022)
Suction head (mm)	49.5	James *et al.* (2010)

The parameters used to model the LID were obtained from literature, field studies and laboratory experiments. The soil layer thickness was derived from borehole logs recorded during the installation of monitoring wells.

The soil from the monitoring well installation was collected, mixed, dried, and sieved. The representative porosity of the soil layer was estimate using two 0.5 m test columns of 200 mm diameter (Figure 8-13).

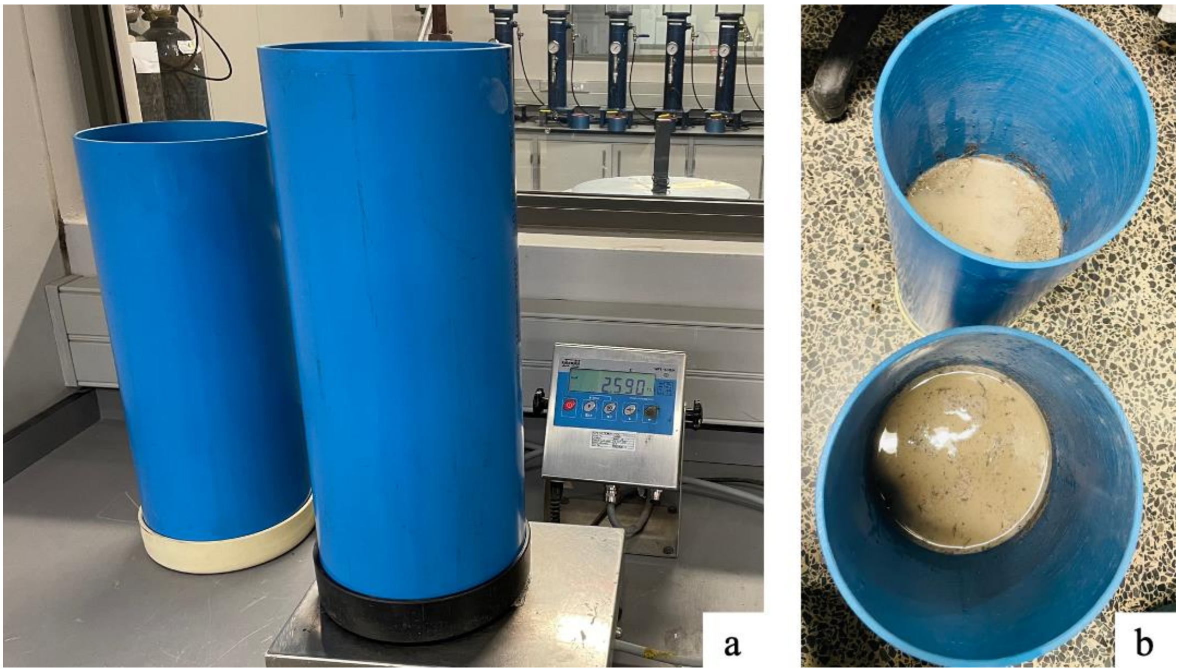

Figure 8-13: a) Weighing the columns, b) Saturated columns

Each column was fitted with an end cap and weighed before packing the soil. A full 10 L bucket of soil was added to each test column, and a needle vibrator – ordinarily used in concrete works – was used to vibrate and compact the soil in the column for

90 seconds. The column was again weighed, and the bulk density of the soil was calculated. Next, water was added from a measuring cylinder until the column was completely saturated, but no free water was present above the media. The column was weighed once more, and the porosity of the packed media was calculated by determining the void volume ratio (volume of infiltrated water) and the total volume (volume of compacted soil and saturated water). All calculations used the standard soil property equations found in soil mechanics and geology texts such as Craig (2004).

8.3.1.7 Aquifer attributes

The SWMM computational engine can simulate subsurface features such as bank storage, groundwater-surface interaction and establish the threshold saturated water zone above which saturated outflow can occur. A simple two-zone groundwater routine is incorporated into the SWMM5 engine to provide subsurface features and processes. Each sub-catchment groundwater flow is independently routed and analysed with the subsurface represented by two zones: an unsaturated upper zone (UZ) that overlies a lower saturated zone (LZ) (Rossman & Huber, 2016).

The water table height fluctuates depending on the inflow and outflow of the water in the upper zone (UZ) and lower saturated zone (Figure 8-14). Although the UZ receives vertical inflow from the impervious sub-catchment areas, the UZ also losses moisture through evapotranspiration. If the UZ becomes saturated due to the water table rising, infiltration will cease, and runoff will be produced from the excess precipitation, i.e., the pervious area becomes impervious. Flow from the unsaturated upper zone to the lower saturated zone; f_u is controlled by the percolation equation that depends on the declared soil parameters. Losses and outflow from the lower zone consist of deep percolation, saturated zone evapotranspiration, and lateral groundwater flow. The depths of the two zones and the water content of the upper zone are determined by the volumetric fluxes (Figure 8-14). Where: f_I = infiltration from the sub-catchment surface; f_{EU} = evapotranspiration from the upper zone, a fixed fraction of the unused surface evaporation; f_U = percolation from the upper to lower zone, which depends on the upper zone moisture content θ and upper zone depth d_U; f_{EL} = evapotranspiration from the lower zone, which is a function of the depth of the upper zone d_U; f_L = percolation from the lower zone to deep groundwater, depending on the lower zone depth d_L; f_G = lateral groundwater seepage to the conveyance network, which depends on the lower zone depth d_L and the water surface elevation in the receiving node

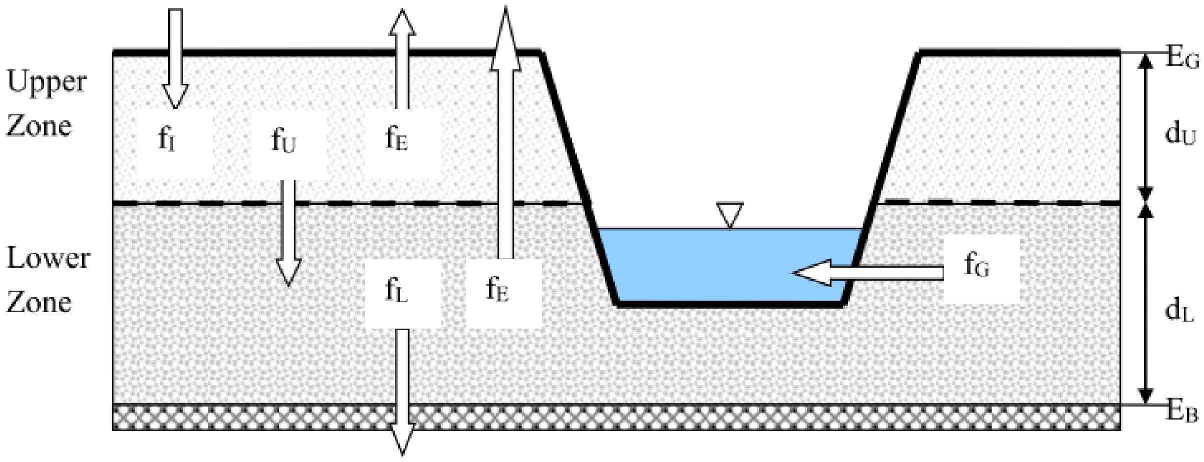

Figure 8-14: The two-zone groundwater model
(Rossman *et al.*, 2016)

The lateral groundwater flow process that can be modelled in PSWMM are illustrated in Figure 8-15. In the figure, Q_{GW} is the groundwater flow (m^3 per hectare), H_{GW} is the height of the saturated zone above the bottom of the aquifer (m), H_{SW} is the height of surface water at receiving node above the aquifer bottom (m) and H^* is threshold groundwater height – the groundwater elevation that must be reached before any lateral flow can occur.

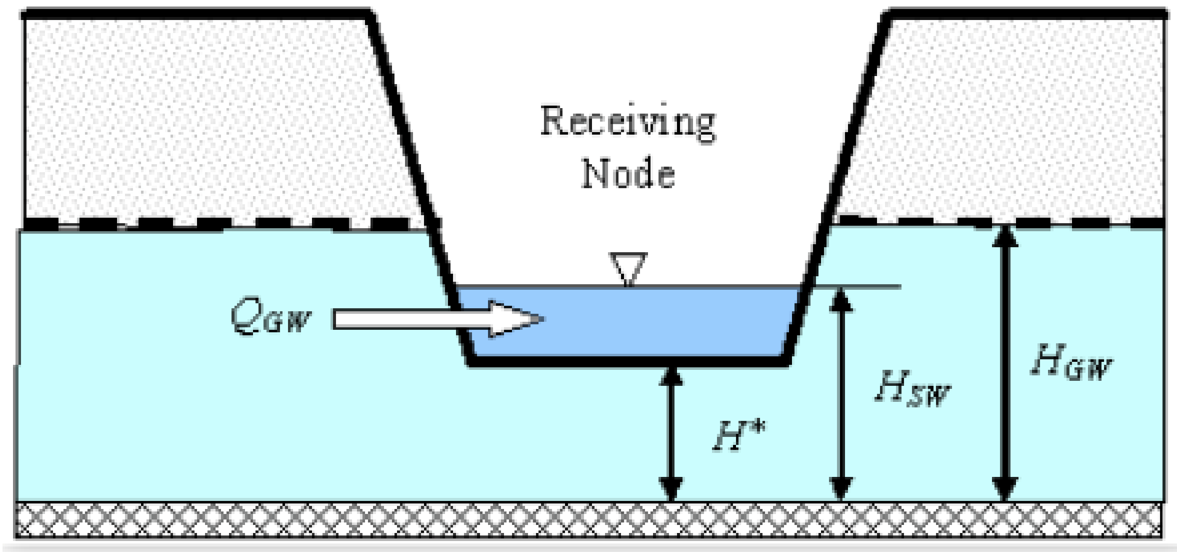

Figure 8-15: Lateral groundwater flow in PCSWMM
(Rossman *et al.*, 2016)

The lateral groundwater seepage, f_G, can be defined by three equations: the Linear reservoir equation, the Dupuit-Forchheimer equation, and the Hooghoudt equation (Equation 8-4 to Equation 8-6).

$$Q_{GW} = A1(H_{GW} - H^*)$$ **Equation 8-4**

$$Q_{GW} = A1(H_{GW} - H^*)^{B1} + A3(H_{GW} * H_{SW})$$ **Equation 8-5**

$$Q_{GW} = A1(H_{GW} - H^*)^{B1} - A2 + A3(H_{GW})$$ **Equation 8-6**

A1, A2, A3, B1 and B2 are the lateral flow equation coefficients initially set based on the model objective, and their typical values can be found in PCSWMM reference manuals and literature (Rossman, 2010; Charbonneau & Bradford, 2016; Rossman & Wayne Huber C., 2017). The coefficients are fine-tuned in the calibration exercise.

The SWMM two-zone groundwater model has several limitations:

- It is a simplistic representation comprising one saturated and unsaturated zone and does not allow for multi-layered aquifer systems modelling.
- Water is assumed uniformly spread across the entire catchment area (the water table responds uniformly under pervious and impervious areas), which means groundwater mounding under infiltration areas cannot be simulated.
- The capillary fringe of the saturated zone cannot move upwards by either diffusion or capillary action.
- Lateral groundwater flows between aquifer systems that underlie several sub-catchments cannot be simulated.

Alternatively, it is possible to direct all inflow or infiltration from the sub-catchments to an aquifer, treating it as a storage tank. The PCWMM software offers the Dupuit-Forchheimer equation for the modelling of the lateral groundwater flow to a channel or node, considering both the surface and groundwater head (Rossman & Huber, 2016). A realistic representation of long-term groundwater response can be achieved by calibrating the Dupuit-Forchheimer parameters that influence the aquifer's response (Charbonneau & Bradford, 2016). This calibration process enhances the accuracy and reliability of the model's groundwater component. This approach effectively controls the acceleration of groundwater response time within the sub-catchments.

The surface-groundwater interactions are modelled in the PCSWMM software, and the aquifer of interest can be defined using the groundwater editor. For the specific case of the CFA, the relevant attributes were obtained from literature sources, observed data from groundwater monitoring wells in the 'School Pond' area as well as monitoring well owned by the City of Cape Town (MON-71). Detailed information on these attributes is presented in Table 8-8.

Table 8-8: CFA attributes

Parameter	Values	Source
Porosity	0.30	Seyler *et al.* (2016)
Wilting point	0.085	James *et al.* (2010)
Field capacity	0.190	James *et al.* (2010)
Conductivity (mm/hr)	110	Mavundla (2022)
Conductivity slope	9.80	Mavundla (2022)
Tension slope	0.00	Rossman & Huber (2016)
Upper evaporation fraction	0.35	James *et al.* (2010)
Lower evaporation depth (m)	2.00	Max. depth into a lower saturated zone over which evapotranspiration can occur (initially assumed)
Lower GW loss (mm/hr)	0.002	Rate of seepage to deep groundwater when the aquifer is wholly saturated from Rossman & Huber (2016)
Bottom elevation (m)	-16.7	Aquifer basement is 45m below MON-71, whose elevation is 28.2 mamsl
Water table elevation (m)	27.5	Initial water table elevation observed in Jan 2021
Unsaturated zone moisture	0.30	James *et al.* (2010)

8.4 Model calibration and validation

Model calibration involves the adjustment and optimisation of model parameters to achieve a statistical and realistic fit (i.e., produce values within an expected range) between the simulated values and observed values. Models typically have three categories of uncertainty:

1. input uncertainties – due to the input files such as precipitation.
2. parameter uncertainties – due to the user declare parameters such as infiltration; and
3. model structure uncertainties – due to the complex relationship of and between processes (James, 2005).

These uncertainties can result in model errors which can be reasonably minimised by calibrating or restructuring the model.

The adjustment of modelled parameters is guided by the modeller-declared confidence zones/levels and depends on the data source and quality. The quality of the data required for calibration must be as accurate as is reasonably possible, while data for design or inference need only be credible. Further, only the 'As-is' Scenario – the current or existing state of a system without any modifications or interventions – can be calibrated. After model calibration, model verification is typically conducted to assess how well the calibrated model matches the observed data. Model verification will provide an indication of the model's accuracy within the specified confidence intervals (Titterington *et al*., 2017).

Calibration can be performed using either short- or long-term (time-series) data. James (2003) suggests that short-term time series are acceptable for a credible model calibration, provided the data is of good quality and has been accurately measured. Long-term data, such as precipitation, flow data, and groundwater levels, can then be used for model validation and inference. Calibrating using short-term data involves parameter estimation and optimisation against short-term, accurate, observed input functions, for example, flow rate, depth, and water level.

Input parameter estimation and optimisation are achieved by statistically analysing the observed and modelled data and minimising errors using either the maximum likelihood function or the sum of squares methods. The two methods aim to find the parameters' values so that the simulated variable values become the most likely possible. An in-depth description of these and other statistical methods can be found in texts like Casella & Berger (2002) and Johnson & Wichern (2007).

PCSWMM calibrates any of the following parameters: sub-catchment runoff, sub-catchment pollutant wash-off, groundwater flow, groundwater elevation, node depth, lateral node inflow, node flooding, node water quality, and link flow rate (Rossman & Huber, 2016). For this, the observed data must be uploaded into the model where a sensitivity analysis is conducted. A sensitivity analysis is used to identify which parameters will be most effective in fitting the simulated to the observed data and aims to minimise the number of calibration parameters. It is performed by changing the value of one relative parameter at a time, and the resulting percentage change in the simulated flow/level is computed relative to the results obtained from a single run of the pre-calibrated model (Krebs *et al*., 2013).

The PCSWMM software, by default, runs four sensitivity points for each identified calibration-sensitive parameter using the assigned uncertainty percentages. Uncertainty measures the user's confidence in the value chosen for a parameter or attribute. For example, if the sub-catchment slope parameter was assigned an uncertainty value of 50% with four (4) sensitivity points, four models will be created where the slope parameter is changed by ± 25% and ± 50%. A sensitivity analysis is then done for the model, where it is run many times with one parameter adjusted in accordance with the declared uncertainty for each run. For instance, if four parameters are evaluated and four sensitivity points are specified, the model runs 16 times. The results are stored, and the parameters can be manually adjusted using the Sensitivity-based Radio Tuning Calibration (SRTC) tool until the simulated data matches the observed data to a satisfactory degree.

The accuracy of the fitted model is evaluated using pre-set coefficients (objective function – OF), namely, Integral Square Error (ISE), Nash–Sutcliff Efficiency (NSE), Coefficient of Determination (R^2), Standard Error Estimation (SEE), Least Square Error (LSE) and the Root Mean Squared Error (RMSE). The equations for each objective function are found in Appendix E. Table 8-9 displays the ranges and interpretation of the two commonly used model objective functions – the ISE and NSE.

Table 8-9: ISE and NSE ranges for model calibration and validation

(After, Sarma *et al.*, 1973; Moriasi *et al.*, 2007)

OF	Poor	Fair	Good	V. Good	Excellent
ISE	≥ 25	10 ≤ ISE <25	6 ≤ ISE <10	3 ≤ ISE < 6	0 ≤ ISE < 3
NSE	0.3 ≤	0.3 ≤ NSE < 0.5	0.5 ≤ NSE < 0.65	0.65 ≤ NSE < 0.75	0.75 ≤ NSE < 1

The application of a model guides the acceptable value of the calibrated models OF, i.e.:

- OFs with 'poor' ratings should only be used for screening.
- models with 'fair' ratings can be used for screening and planning.
- models with a 'good' rating can be used for planning and preliminary design.
- models with 'very good' and 'excellent' ratings can be used for final design.

The adjusted parameters represent a calibrated model (Randall *et al.*, 2019). Finally, the model is re-run using the adjusted parameters, and the resulting output is a verified model output which can also be evaluated using the OF described above. Generally, a model is deemed sufficiently calibrated when a 25% accuracy is reached, although some requirements are more stringent and demand a 10% accuracy. Nevertheless, the ISE and NSE values mentioned above are still suitable measures, and acceptable values are often guided by the model's intended use (James, 2005; Titterington *et al.*, 2017). The recommended level of accuracy for wet weather is ±10% of the volume and ± 15% of the peak (James, 2005). It is recommended that at least three significant events are used for calibration. The model can then be evaluated on the following criteria:

- A model volume of flow between -10% to +20%
- A model peak of flow between -15% to +25%
- The general shape of the hydrographs should be similar.

8.5 Calibrating and validating the School Pond model

8.5.1 Calibrating the surface model

The School Pond model was calibrated using observed flow into the pond from one of the inlets. The flow was recorded using a portable area-velocity flow meter (AVFM) (Figure 8-16b). A portable AVFM was used because a permanent meter with a logger could not be installed due to the risk of theft. The AVFM was used in conjunction with weather alerts which incoming storms and were obtained from a weather site (windy.com) (Figure 8-16a). The ACFM was deployed at the pond's inlets before rainfall events to measure the inflow rates (Figures 8-16c and 8-16d).

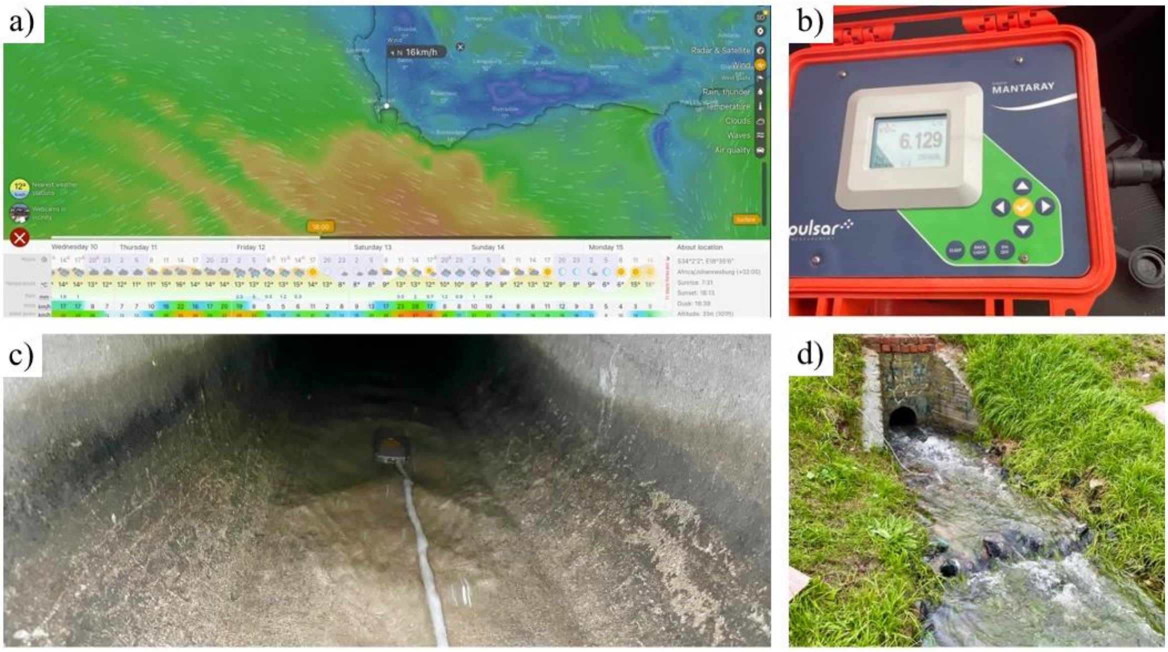

Figure 8-16: a) Stormwater inflow data collection process

The portable AVFM can accurately measure flows with velocities ranging from 0.1 to 6.2 m/s and an accuracy of ± 0.25% of the reading, and operational for all flow regimes. The measured flow data is recorded and stored in a dedicated data logger for further analysis.

A portable meter was preferred to mitigate the risk of theft, although the possibility of on-site robbery or hijacking remained. The risk of robbery was mitigated by the presence of a neighbourhood watch which conducted regular patrols during flow logging events. The AVFM supplier calibrated the meter, and its accuracy was verified at the hydraulics laboratory of the University of Cape Town through a comparison of the AVFM-derived flow rates with those obtained from a calibrated magnetic flow meter.

Seven storm events were recorded, of which only four were usable due to a logger error attributed to the logger software, which the manufacturer later rectified. The storm events had a minimum duration of three hours. Three storm events were used for calibration, and the fourth was used for validation (Table 8-10).

Table 8-10: Selected storms for calibration and validation

Event Date	Model Process	Rainfall/Duration	Comment
13 Jun 2022	Calibration	7.8 mm/3.6 hours	Meter logging error: stopped logging after 3.6 hours. Pre-error data was used.
14 Jun 2022	Calibration	6.0 mm/3.5 hours	The logger missed some flow rate values and two peaks, and these missed events were skipped.
23 Jun 2022	Calibration	8.6 mm/3 hours	The school weather station did not record the rainfall event due to load-shedding. Used Wolfgat data
17 Aug 2022	Validation	4.2 mm/5.5 hours	Data collected with no problems

The flow data from the AVFM was converted to a PCSWMM-compatible data file. The associated rainfall and temperature data (which was used to calculate evapotranspiration) were obtained from the weather station. The data was then used to run the model.

First, a sensitivity analysis using the SRTC was conducted, where the sensitivity of twenty-one parameters that contributed to the inflow at Inlet 1 (Junction C13 in the model) was analysed. The analysis used eight sensitivity points and resulted in 168 model runs. Twelve parameters were then identified as the most sensitive parameters to optimise in the model (Figure 8-17).

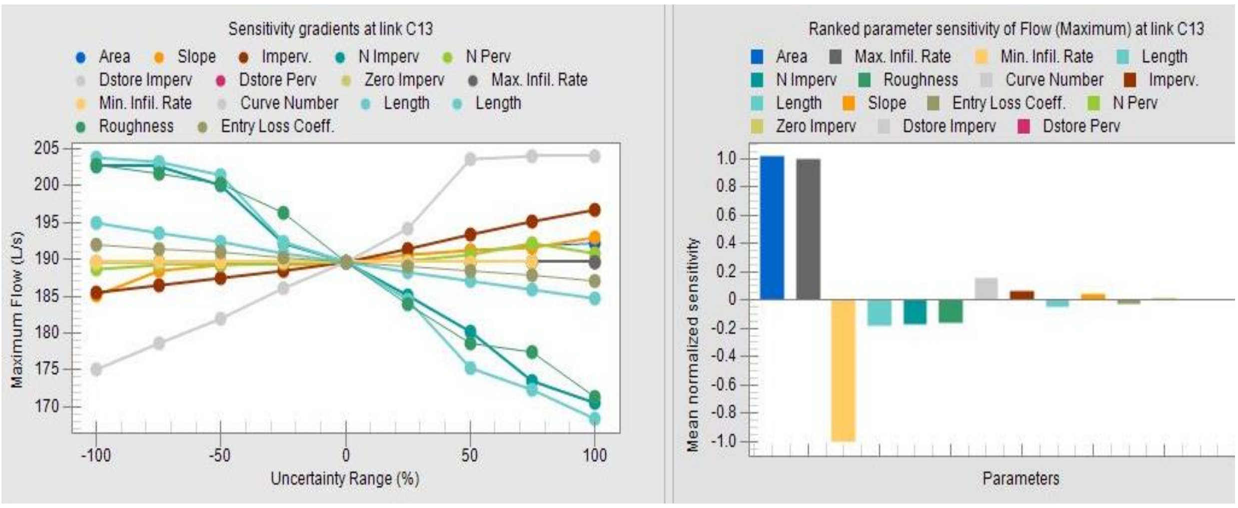

Figure 8-17: Sensitivity gradients for sub-catchment and pipe parameters that influence the inflow rate at inlet 2

The SRTC tool was then used to optimise the model parameters by comparing the simulated flow rate (referred to as the current response) from the pond's inlet to the AVFM-derived flow rate. Finally, the model parameters were calibrated manually by finding the best fit for the shape of the simulated hydrograph, peak flow rates and the time of peak (Figure 8-18). The calibration process employed the ISE and NSE as the objective functions, with the aim of achieving a balanced trade-off between the two. The model parameters were iteratively adjusted until satisfactory values of NSE and ISE were obtained, indicating a reasonable calibration of the model (Figure 8-18).

The model over-predicted the observed peaks and runoff volumes, likely due to overestimated percentages of pervious area and contributing sub-catchment areas compared to reality. However, the shape of the simulated hydrograph closely matched the measured one. The peaks and volume offsets were minimised during calibration to maximise the NSE coefficient and minimise the ISE. The calibrated response (green curve in Figure 8-18) from the SRTC tool is generated by linearly interpolating the results between the appropriate parameter value based on the number of sensitivity points.

The calibrated response is an estimated resultant response and must be verified by re-running the model to determine the actual model results when parameters are adjusted – the re-run results in a verified response (grey curve in Figure 8-18). Thus, the School Pond model was re-run with the optimised parameters producing a verified hydrograph (Figure 8-18).

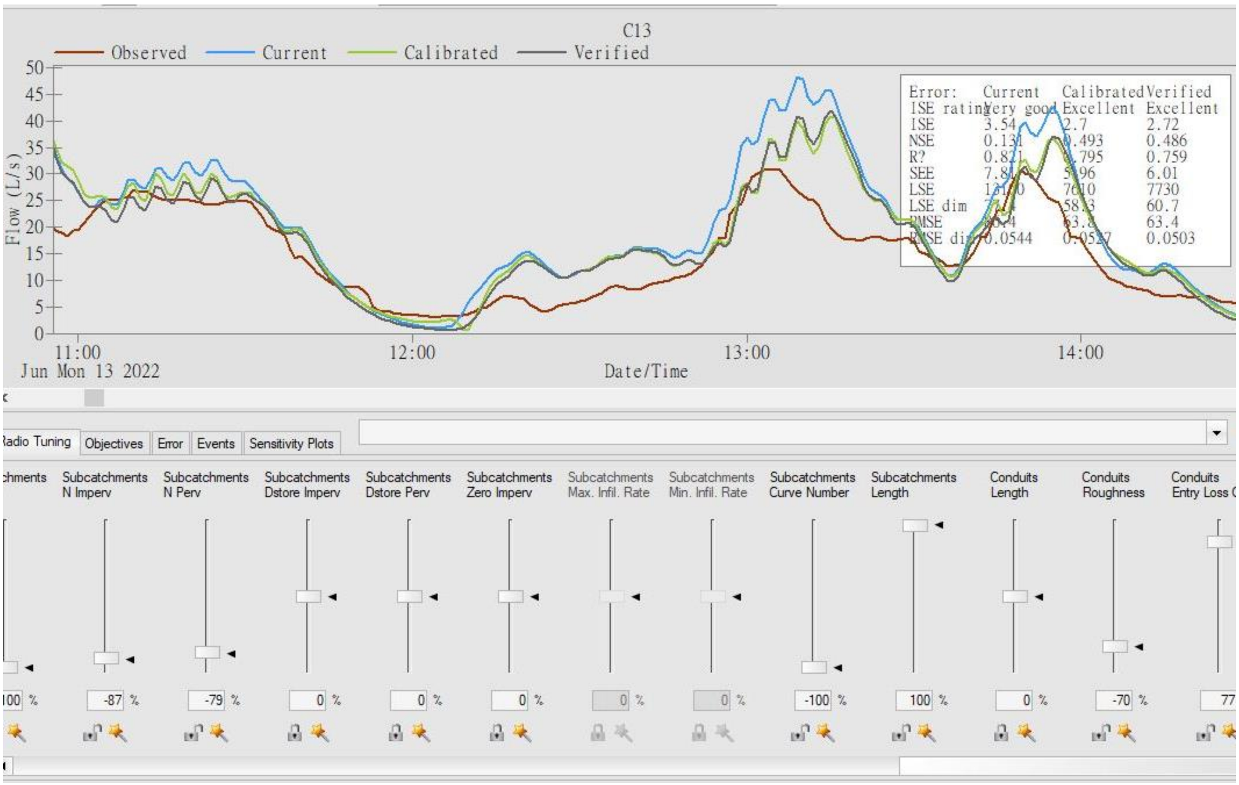

Figure 8-18: Inflow-based 'School Pond' calibration and verification

The pre-calibrated time to peak of the simulated hydrograph and the observed pond inflow hydrograph was offset by ~5 minutes. The offset implied the model had a comparatively longer time of concentration – initially attributed to the sub-catchment width and area, the depression storages, percentage pervious area and pipe slope. However, during a data collection event on 18 Aug 2022, a waste management company sub-contracted by the CoCT was observed unblocking a partially blocked stormwater pipe that contributed to the pond inlet.

The company used a high-pressure jet to clear debris in the pipe and associated culvert (Figure 8-19). This observation helped explain the peak offset, as the partial blockage meant the flow to the pipe inlet was delayed. The blockage did not significantly influence the observed inflow volume as the data collection was initiated before the start of a storm and finalised after no flow into the pond was observed and thus the total inflow volume was recorded.

Table 8-11 then summarises the values and interpretation of the objective functions.

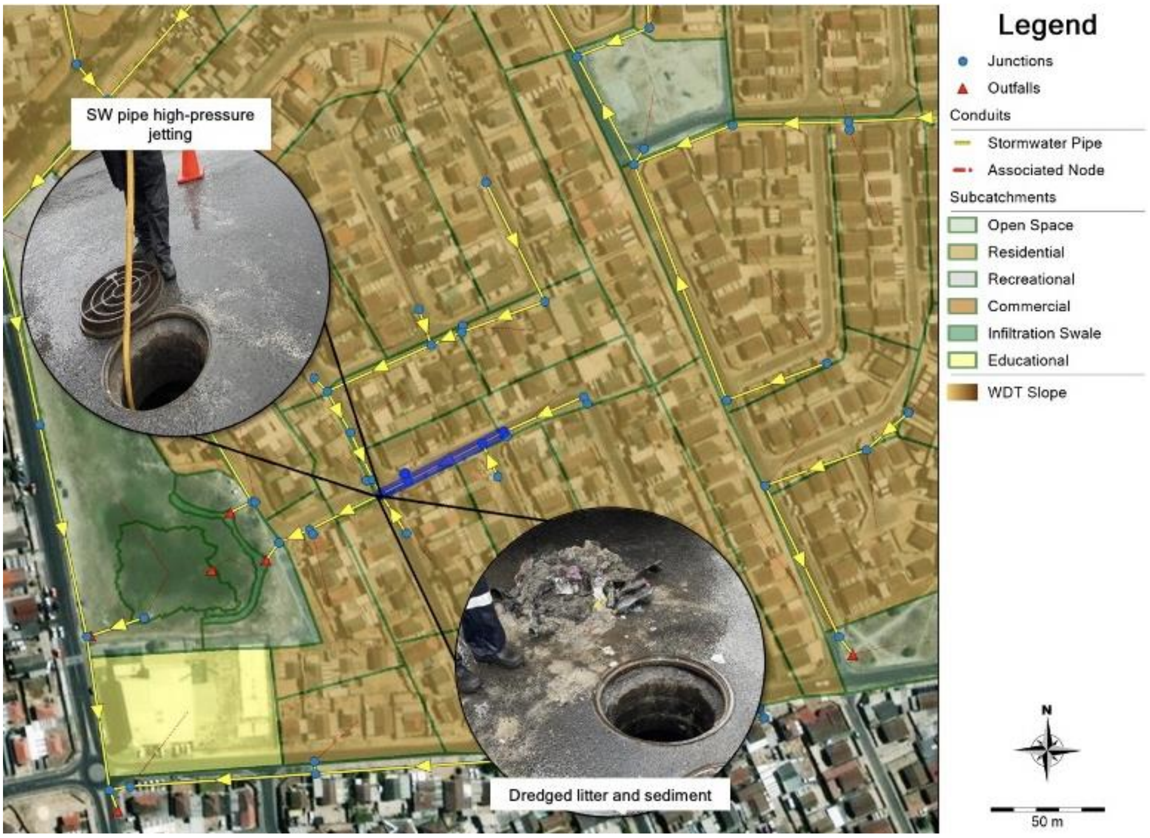

Figure 8-19: Service providers clearing a blocked pipe

Table 8-11: Summary of the surface model calibration results

Objective function	Value			Comment
	Simulated	Calibrated	Verified	
ISE	3.54	2.7	2.72	The ISE value is 'Excellent' (from Table 8-9).
NSE	0.131	0.493	0.486	The NSE value 'Fair' (also from Table 8-9).

Event 1 was primarily used for the calibration and verification exercise upon realising that using the combined but discontinuous set of events as presented in this study (global calibration) would misrepresent the system. It was found that the NSE performs poorly if discrete and widely spaced events are used as the NSE optimised for continuous data like river gauges because the errors are more consistent in continuous logging (Hossain *et al.*, 2019). This is evident when Events 1 and 3 are jointly analysed using the same verified parameters as Event 1. The resulting NSE is 0.321, rated as 'Fair' but less than Event 1's NSE value of 0.49 (Figure 8-20). The ISE value, however, improves and is rated as 'Excellent'.

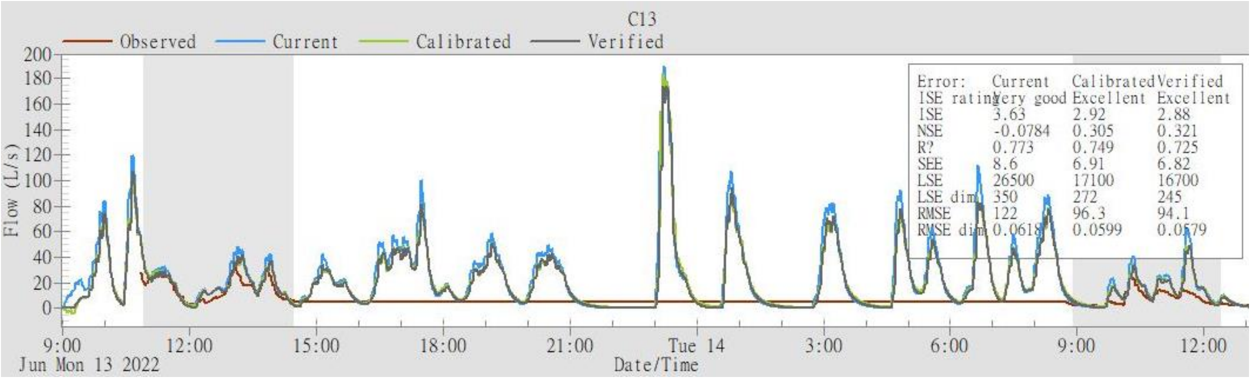

Figure 8-20: Global calibration (all events)

Figure 8-21 presents the validation results. The shape of the calibrated verified hydrograph matched well with the measured one, and the OFs are given in Table 8-12.

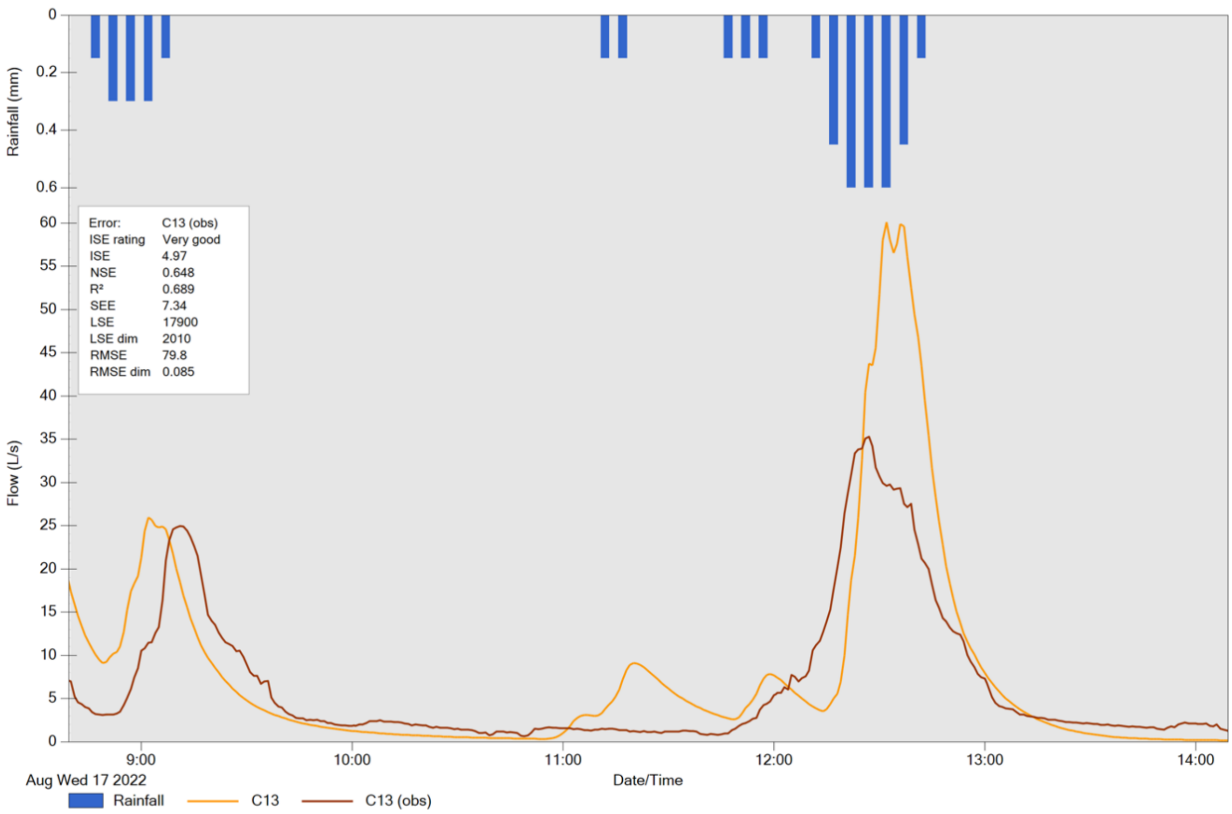

Figure 8-21: 17 Aug 2022 event used for model validation

The validated model shows good performance for the NSE and ISE objective functions. The observed and simulated event volumes difference of +18.4% is within the recommended model volume, which should be between -10% to +20%. However, the simulated peak is 41% higher than the observed peak, which is beyond the recommended range, although not uncommon, as exemplified by Randall *et al.* (2019). Since the model was only used to estimate volumes and not peak flows, this discrepancy was overlooked in this study.

Table 8-12: Summary of the validation results

Objective function	Value		Comment
	Verified	Validated	
ISE	2.72	4.97	The ISE value is rated 'Very Good' (from Table 8-9).
NSE	0.486	0.648	The NSE value is considered 'Very Good' (also from Table 8-9).
Event Volume (L)	159900	130400	+18.4%. Within recommended range
Peak Flow (L/s)	60.1	35.3	+41% difference. Beyond the recommended range

8.5.2 Calibrating the subsurface model

The groundwater response in the pond influences the infiltration rate and the volume of stormwater than can be harvested in the retrofitted pond. Thus, this parameter had to be calibrated in the model. The calibrated surface model was used in the groundwater calibration. The aquifer properties were applied only to the pond area because of the PCSWMM limitation – lateral groundwater flows within an aquifer system that underlies several sub-catchments cannot be simulated as all the sub-catchment groundwater flows are routed to the aquifer 'tank'.

To illustrate this point, consider the 'School Pond' catchment area (Figure 8-4) and assume that the groundwater flows from the northern to the southern boundary. If one were to assign an aquifer that extends across the entire catchment to a random sub-catchment found along the northern boundary, the infiltrated water from the sub-catchment would immediately influence the groundwater response in a sub-catchment that is along the southern boundary. This is because the 'tank' routing in PCSWMM means the water from that sub-catchment immediately increases the volume of water in the tank and, by extension, the height of the water in the tank – which is the water table depth. This scenario is unrealistic in practice as the infiltration from the northern sub-catchment would take a few hours or years to reach the southern boundary – the travel time depends on the aquifer's hydraulic gradient, conductivity, and transmissivity. To counteract this unrealistic 'tank' scenario the aquifer was assigned to only the sub-catchments within the pond. The groundwater transfer from the aquifer below the pond to the 'regional' aquifer is facilitated by incorporating an outflow node to 'transfer' the groundwater using the Dupuit-Forchheimer approximation, much like the scenarios investigated by Charbonneau & Bradford (2016). The rate and characteristics of the aquifer transfer were calibrated using observed data to best represent the system.

The calibration period was limited to periods between installing the infiltration swale and discovery of the mole holes that short-circuited the infiltration swale (October 2021 – April 2022). This period was chosen as it would best reflect the increase in groundwater levels due to the infiltration swale and calibrating for this would aid in the extended simulation of the swale functionality. The calibration process for the groundwater response followed a similar process as described in Section 8.5.1. However, in this instance, the simulated groundwater levels were compared with those derived from the installed monitoring wells (Figure 8-22).

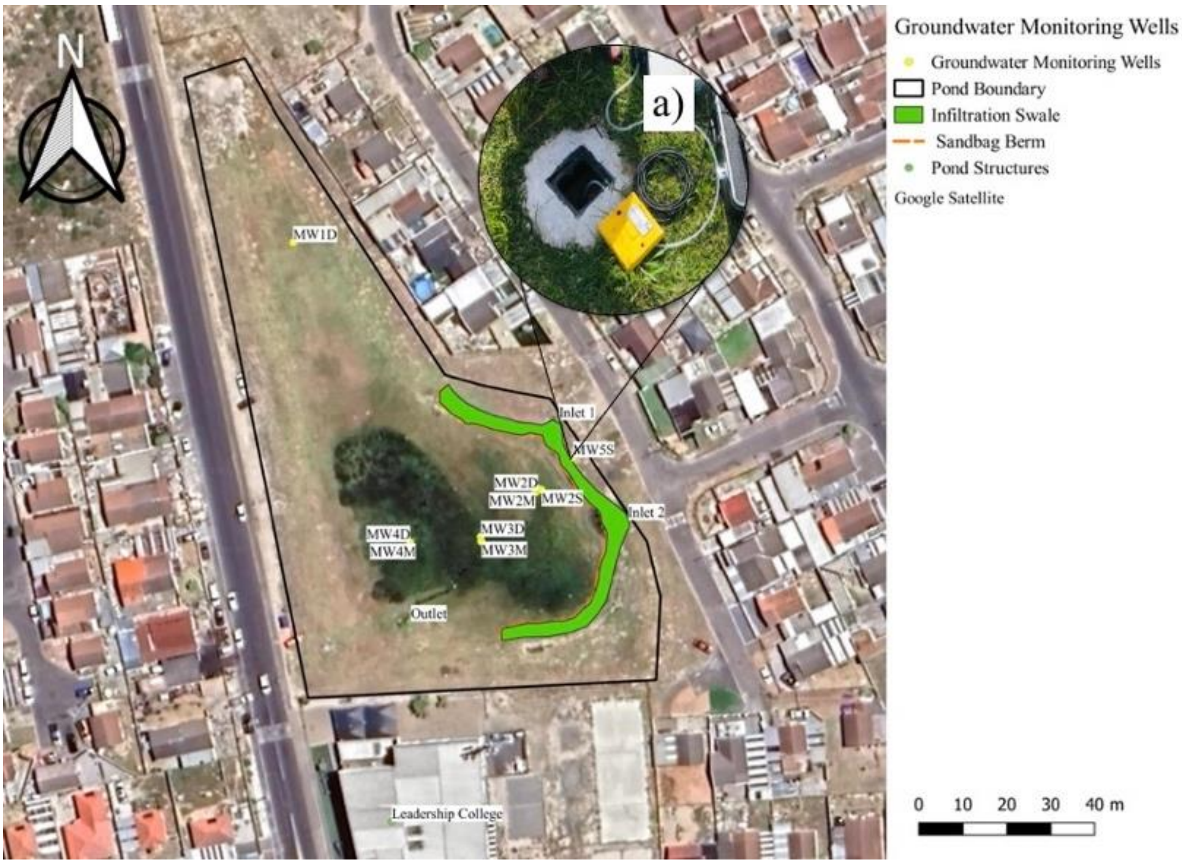

Figure 8-22: Positions of installed monitoring wells MW5S

Groundwater elevations were collected monthly during summer and fortnightly during winter. The water elevation data from the shallow monitoring well adjacent to the swale – MW5S, which is 2 m deep – was used to calibrate the model. The top of MW5S is at an elevation of 28.8 mamsl, while the swale elevation is at 28.2 mamsl (Figure 8-23). The groundwater data from 13 Oct 2021 to 04 May 2022 (203 days) was converted to a PCSWMM-compatible file and used as the observed data file.

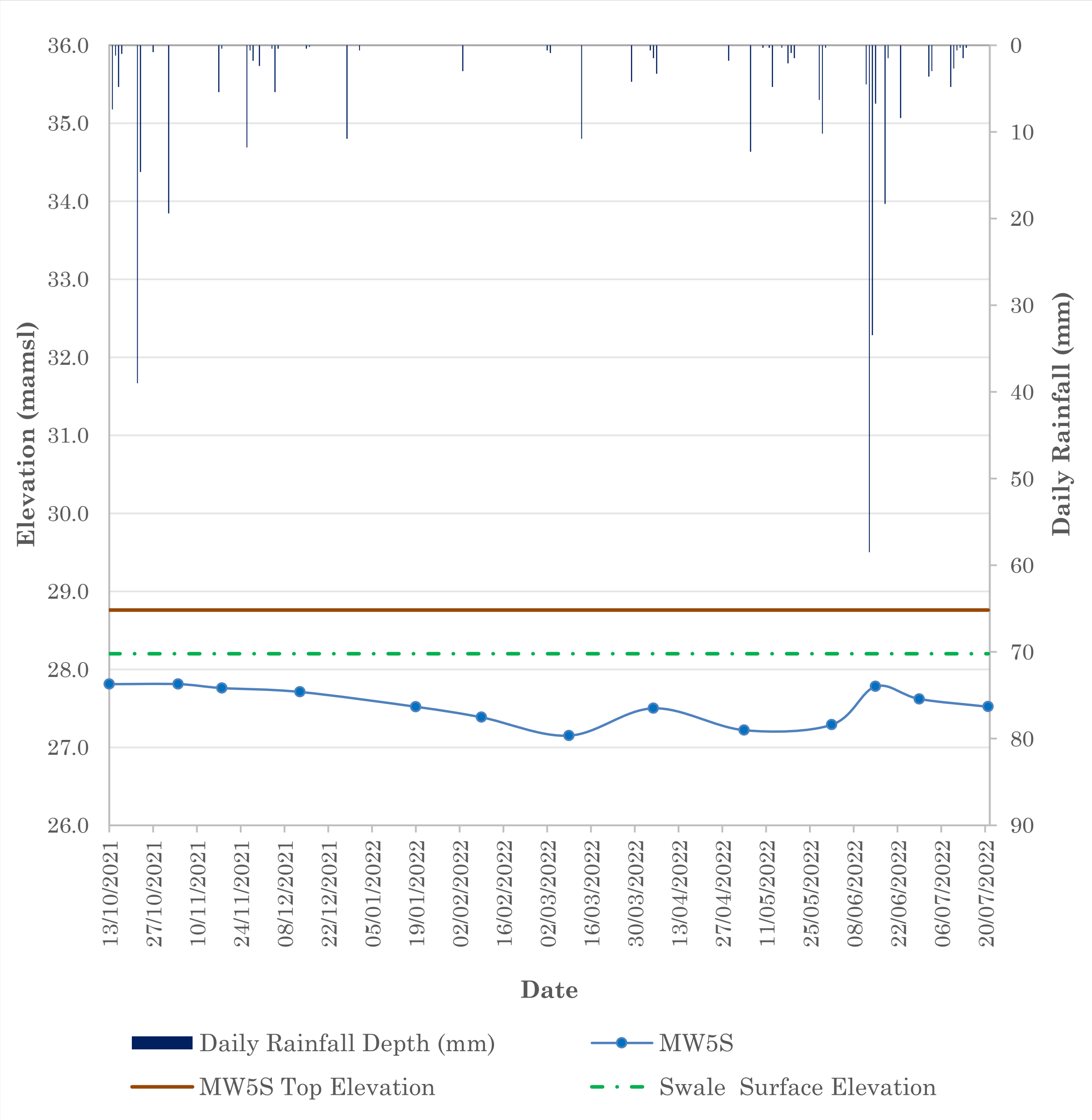

Figure 8-23: Water table fluctuation beneath the infiltration swale

The aquifer parameter's uncertainty ranges were defined and inputted into the software, and a sensitivity analysis was conducted using the SRTC tool. This analysis encompassed 34 parameters and employed four sensitivity points, resulting in 136 model runs (Figure 8-24). The parameter influencing the mean groundwater level the most was the lower evaporation depth – the maximum depth into the lower saturated zone over which evapotranspiration can occur (James *et al.*, 2010). The SRTC was used to optimise the parameters first using the NSE as the objective function.

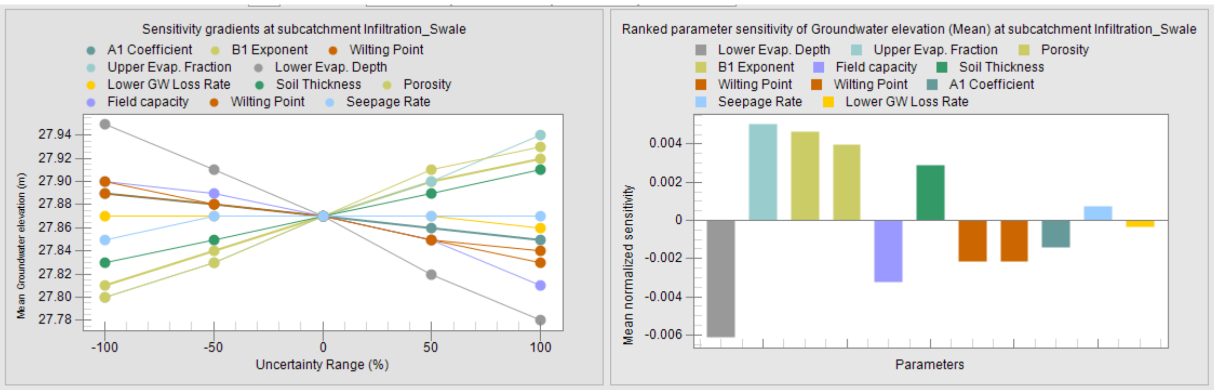

Figure 8-24: Sensitivity gradients for sub-catchment and aquifer parameters that influence the water table beneath the pond

While the groundwater level response was sensitive to eleven parameters, optimising some of the parameters resulted in an unrealistic groundwater profile – in some instances, the resultant water table elevation was higher than 28.2 mamsl, which is the swale surface elevation. Further, the parameters that gave optimum ISE and NSE values (0.309 and 0.216, respectively) were found to be unrealistic with parameters such as the field capacity, which is the faction of water remaining in a soil after it has been thoroughly saturated and allowed to drain freely – being higher than the soil porosity. Some parameters such as the B1 coefficient – which influenced the rate of lateral groundwater flow – could have been adjusted to produce a more desirable positive NSE value, but the 'optimal' values were higher than literature-suggested values and were, in this case, ignored which is recommended according to established SWMM users and developers (Klaver, 2019). This process is regarded as responsible modelling (James, 2005). However, some researchers, such as Charbonneau & Bradford (2016), use an adjusted B1 value of 2.07, which is higher than the accepted value of 2 (from Dupuit-Forchheimer approximation for groundwater flow). This study adhered to a dimensionally homogenous Dupuit-Forchheimer approximation for lateral groundwater flow; thus, the B1 value was kept as 2.

The parameters were iteratively fine-tuned, making trade-offs between the profile shape, realistic aquifer parameter values, and NSE and ISE values. Finally, the adjusted parameters were accepted, run, and verified (Figure 8-25). The blue curve termed 'current' in PCSWMM in the Figure 8-25 represents the SWMM5 simulated groundwater profile, while the red curve represents the observed groundwater elevations. The green curve corresponds to the calibration process in PCSWMM, where certain parameters were optimised, but some of them may not reflect realistic conditions as they are derived from a linear estimation process. The grey curve depicts the verified curve which represents the best fit between observed and simulated values, using realistic and optimised parameters.

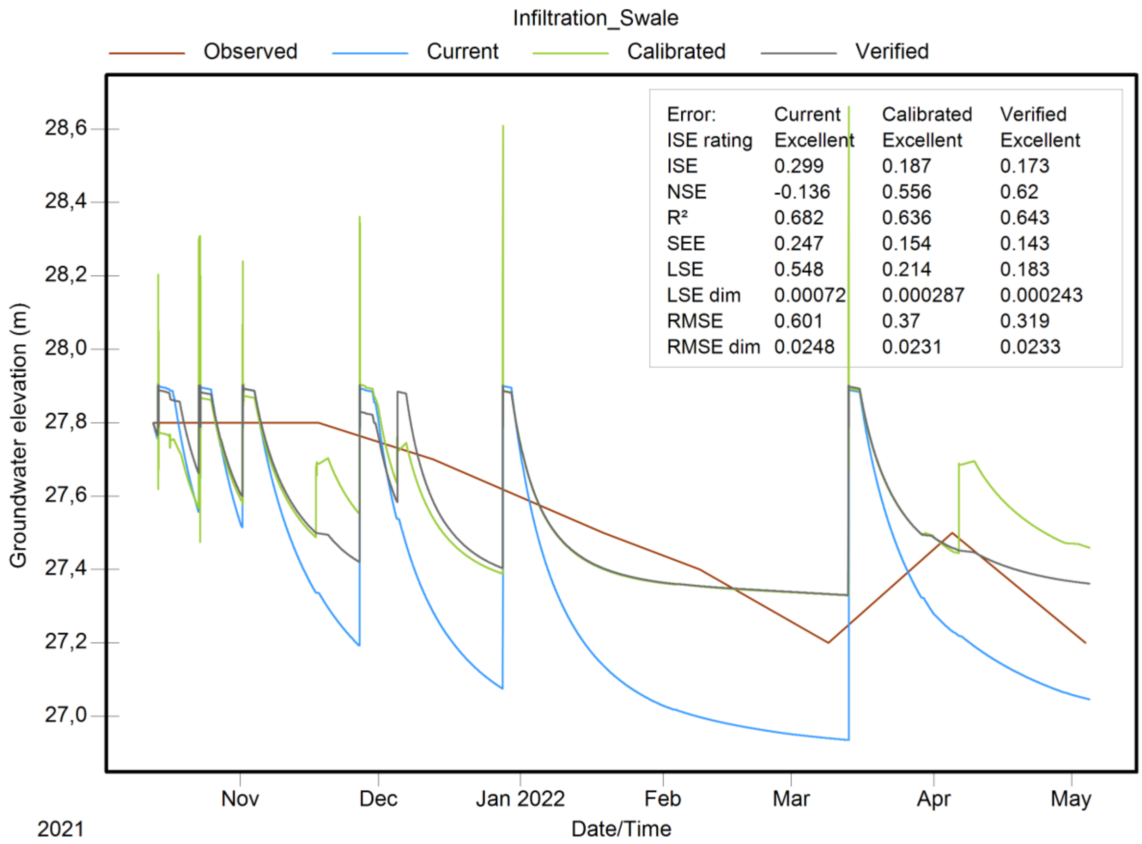

Figure 8-25: Groundwater calibration and verification

The values of the objective functions and the evaluation thereof are given in Table 8-13. The verified parameters were assigned to the sub-catchments, LID, and aquifer properties.

Table 8-13: Summary of the groundwater model calibration results

Objective function	Value			Comment
	Simulated	Calibrated	Verified	
ISE	0.299	0.187	0.173	The ISE value is rated 'Excellent' (from Table 8-9).
NSE	-0.136	0.556	0.62	The NSE value is considered 'Good' (also from Table 8-9).

The validation of models necessitates using previously unused data to evaluate the correspondence between the verified response and observed data. In this study, the groundwater model could not be validated using observed data due to a lack of usable information caused by the Cape Moles Rats creating tunnels beneath the infiltration swale. These disturbances disrupted the infiltration process, rendering the water level in the monitoring well unrepresentative. However, the calibration and verification results show that the groundwater model had an excellent ISE rating (0.173) and a good NSE value (0.63). In addition, the ISE and NSE values indicate that the model is a good long-term predictor of water elevation.

The PCSWMM engine computes NSE and ISE values by comparing instantaneous data points with their corresponding observed data points. Although this approach is generally adequate for continuous data comparisons, challenges may arise when the observed data is collected at longer intervals than the simulated data. In such scenarios, where the simulated data is continuous while the observed data is not, moving averages (means) become more appropriate for comparison within the PCSWMM software. The simulated groundwater levels, recorded at 1-minute intervals over 203 simulation days, were exported to Excel, and the moving average of the simulated groundwater level was calculated to address this limitation. A 25-day interval was used, which coincided with the mean groundwater monitoring interval at the School Pond – once a month. The data points from dates that matched when the groundwater elevations were recorded at the pond there then extracted from the moving average data. The resultant groundwater profile better matched the observed data (green and red curve in Figure 8-26) than the raw simulated data, while the recalculated NSE and ISE values were better than the PCSWMM-derived values (Table 8-14). These findings indicate that, despite differences in the visual representation of the groundwater profile from the response dynamics of the LID in PCSWMM, the data over the extended simulation period is consistent with what was observed.

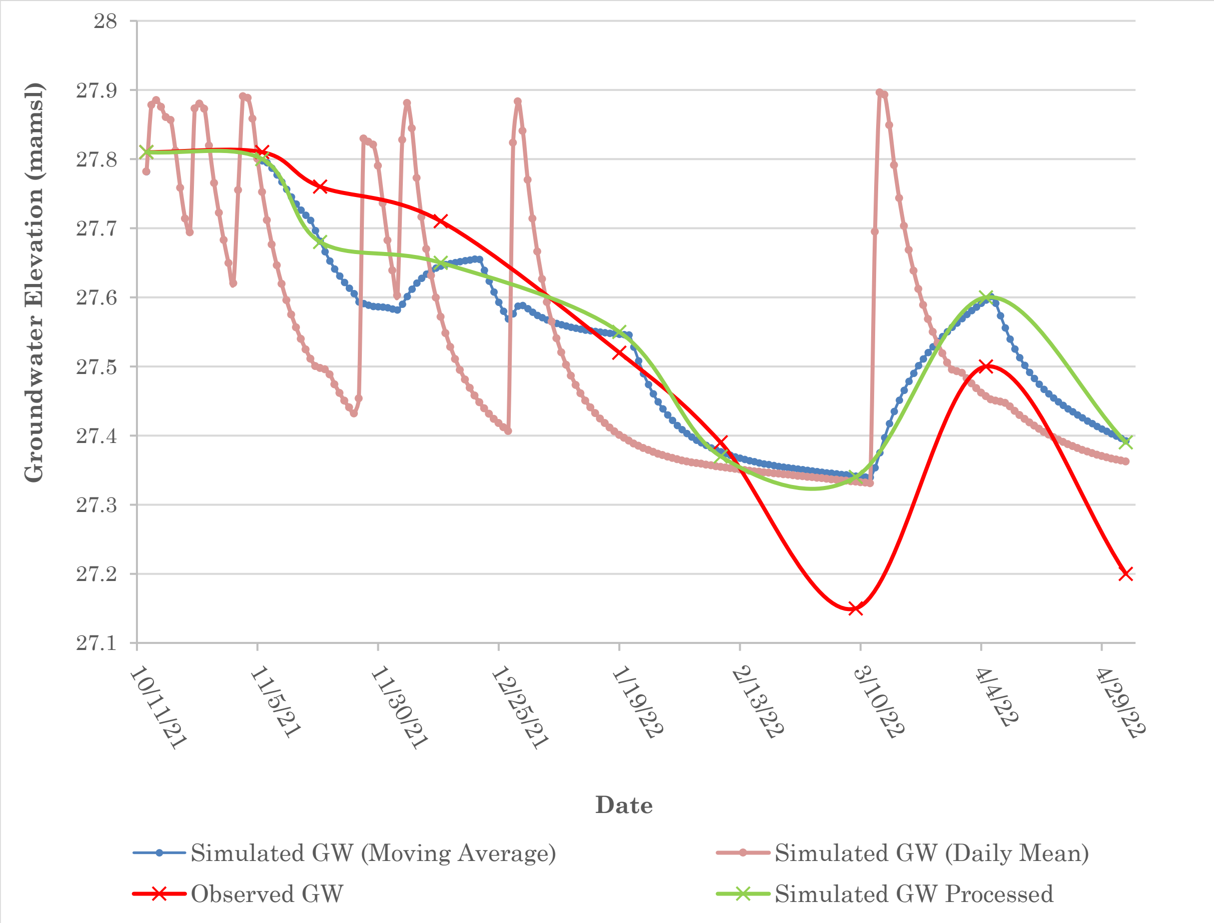

Figure 8-26: Groundwater elevation curves with processed observed data

Table 8-14: Groundwater calibration using processed data

Objective function	Value		Comment
	Verified	Cleaned	
ISE	0.173	0.0012	The ISE value is rated 'Excellent' (from Table 8-9).
NSE	0.62	0.82	The NSE value from the processed data is rated 'Excellent' (also from Table 8-9).

In summary, the observed groundwater elevation graphs exhibit a high degree of similarity, while the ISE is excellent and the NSE is rated as 'Good' or 'Excellent' depending on the data processing method used. Moreover, the mean difference between the observed and simulated water tables is only 0.04 m. These findings suggest that the model can be confidently used for long-term groundwater simulations. Consequently, the calibrated surface and subsurface models were utilised to simulate various model scenarios.

8.6 Model scenarios

The PCSWMM model of the School Pond was used to evaluate the geo-hydraulic short- and long-term performance of the retrofitted infiltration swale under different scenarios. This study evaluated the volume of infiltrated stormwater (MAR) from twelve scenarios. The scenarios are summarised in Table 8-15 and described in the subsequent sections.

Table 8-15: Summary of modelled scenarios

Scenario No.	Simulation Year(s)	Description
1	2005 - 2022	Pre-retrofit harvested volume using extended rainfall data
2	2005 - 2022	Post-retrofit harvested volume using extended rainfall data
3	2005 - 2022	Post-retrofit harvested volume using extended rainfall data and with dropped water table due to CoCT groundwater abstraction and MAR
4	2005 - 2022	Post-retrofit harvested volume using extended rainfall data, a dropped water table, and an extended LID area.
5	2010	Harvested volume for a year with low rainfall – Pre-retrofit
6	2010	Harvested volume for a year with low rainfall – Post-retrofit
7	2019	Harvested volume for a year with low rainfall – Pre-retrofit
8	2019	Harvested volume for year with low rainfall – Post-retrofit
9	2021	Harvested volume for year with highest rainfall – Pre-retrofit
10	2021	Harvested volume for year with highest rainfall – Post-retrofit
11	2084 - 2100	Post-retrofit harvested volume using extended climate change adjusted data (intermediate emissions scenario)
12	2084 - 2100	Post-retrofit harvested volume using extended climate change adjusted data (high emissions scenarios)

For all scenarios, the infiltrated volume in the detention pond – considered the volume contributing to MAR – was calculated using the water balance presented in Equation 8-7.

$$INF_{gw} = I_v + R_v - O_v - Evap_{surface} \quad \textbf{Equation 8-7}$$

Where, INF_{gw} is the infiltration to groundwater, I_v is the inflow volume into the pond from the two inlets, R_v is the runoff volume resulting from rainfall falling over the pond area, O_v is the resulting outflow that drains to the existing stormwater network, and $Evap_{surface}$ is the evaporation from the pond surface mainly evaporation from depression storage and ponded water. The stored groundwater volume (MAR) is then derived from Equation 8-8.

$$\text{MAR} = \text{INF}_{\text{gw}} - \text{Evap}_{\text{gw}} - \text{GW}_{\text{seepage}} \qquad \textbf{Equation 8-8}$$

Where, $Evap_{gw}$ is evaporation from the upper and lower saturated aquifer zones via evapotranspiration, and $GW_{seepage}$ is groundwater seepage defined as percolation into the deep aquifer (Rossman & Huber, 2016).

8.6.1 Scenario description

The twelve scenarios in Table 8-15 are categorised into five segments and further described below.

8.6.1.1 Pre- and post-retrofit extended scenarios using the extended rainfall data (Scenarios 1 & 2)

These scenarios were used to establish the potential recharge volumes and evaluate the long-term performance of the infiltration swale by running the model for 16.4 years of available rainfall data. There were two drought periods within the simulation period 2007 – 2009 and 2015 – 2018. The droughts meant less precipitation than usual and more time for the aquifer to recover. These two conditions mean that the results for the scenarios with an initially high-water table may be an overestimate.

The volume of infiltrated stormwater in the swale was established by calculating the infiltrated volume over the simulation period using Equation 8-7. The initial water table was determined from mean of the observed water elevation (27.5 mamsl). The calculated infiltration volume was then divided by the total precipitation received during the simulation. Scenarios 1 and 2 were used to assess the recharge volumes before and after the retrofit, respectively, using the same climatic data.

8.6.1.2 Post retrofit extended scenarios with water table adjustment (Scenarios 3 & 4)

The CoCT intends to abstract water from the CFA as part of its New Water Program (CoCT, 2019b). This planed abstraction, from sixty-nine installed production wells (CoCT, 2021), will be coupled with MAR by injecting high-quality treated wastewater effluent into sixty-one recharge wells.

The abstraction and MAR processes in the CFA were simulated using MODFLOW, a groundwater modelling programme that simulates groundwater hydraulics. The CFA model was developed by the CoCT's groundwater consultants.

Figure 8-27 illustrates the simulated groundwater table profile obtained from the groundwater model.

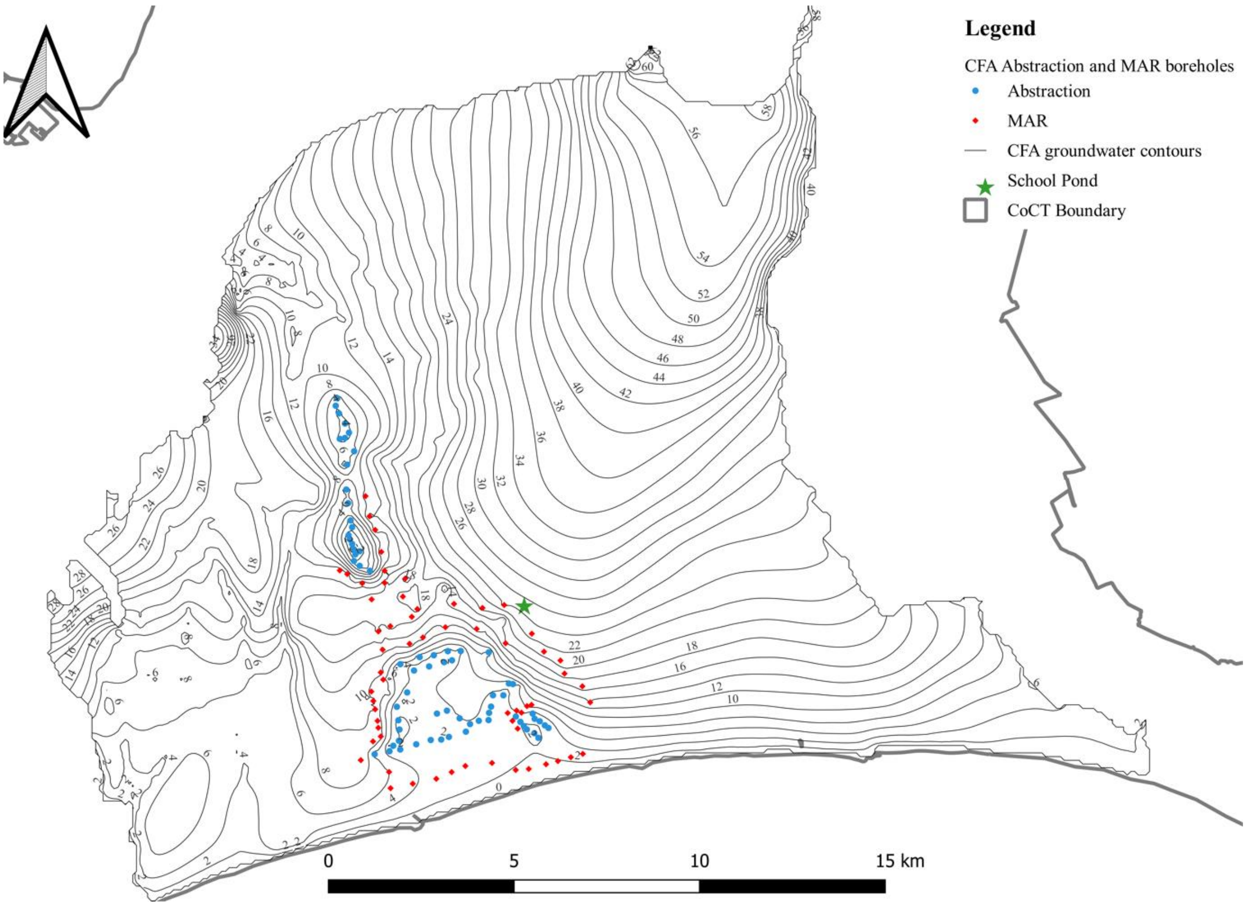

Figure 8-27: Mean CFA water table profile from a simulated 50 ML/D abstraction and 40 ML/D MAR scenario using MODFLOW
(MAR data obtained from Umvoto Africa, October 2022)

The modelled mean water table from the groundwater model results show that if the CoCT abstracted 50 ML/day and injected 40 ML/day as planned, with the difference being made up by natural recharge, the mean annual water table beneath the School Pond would be 25 mamsl. The resultant water table level would thus be 2.5 m below the present observed mean water table. The extended unsaturated (vadose) depth presents potentially more infiltration volume but also possibly more evapotranspiration.

Scenario 3 was thus used to evaluate the retrofitted pond's potential MAR if the CoCT implemented its planned abstraction and MAR and dropped the water table to a mean of 25 mamsl at the pond location.

Earlier designs of the School Pond retrofit had envisioned a larger infiltration area (henceforth referred to as Extended_LID). This was not realised due to the high-water table beneath the pond. However, the CoCT MAR scenario presented an opportunity to model and evaluate the potential infiltration volume from the Extended_LID design. The model structure for this scenario (Scenario 4) is described in Section 8.6.2.3.

8.6.1.3 Pre- and post-retrofit scenarios using different climatic conditions (Scenarios 5 – 10)

The recharge volume in this study was influenced by stormwater inflow, evaporation, evapotranspiration, and outflow (Equation 8-7 and Equation 8-8). Therefore, scenarios were developed to investigate how three climatic conditions – described as dry (year with low rainfall), mean (year with mean rainfall), and wet (year with the highest rainfall) as evaluated from the Wolfgat station's dataset (Figure 8-28) – would influence the recharge volume. This assessment aided in understanding the processes that influence aquifer recharge.

The performance of the pre- and post-retrofit scenarios were assessed in Scenarios 5 to 10 using the three climatic conditions that all spanned one full year with the initial water table set at the mean January water table as determined from monitoring well data. 2017 and 2008 were the driest years which occurred during drought periods such that they received 34 mm and 55.8 mm of rainfall, respectively. Therefore, these precipitation depths are considered outliers with the preliminary model runs indicating no MAR, as the water was lost as evaporation and evapotranspiration. The 2010 data – the third driest year in the dataset – was then used as the representative dry year where some MAR could be observed.

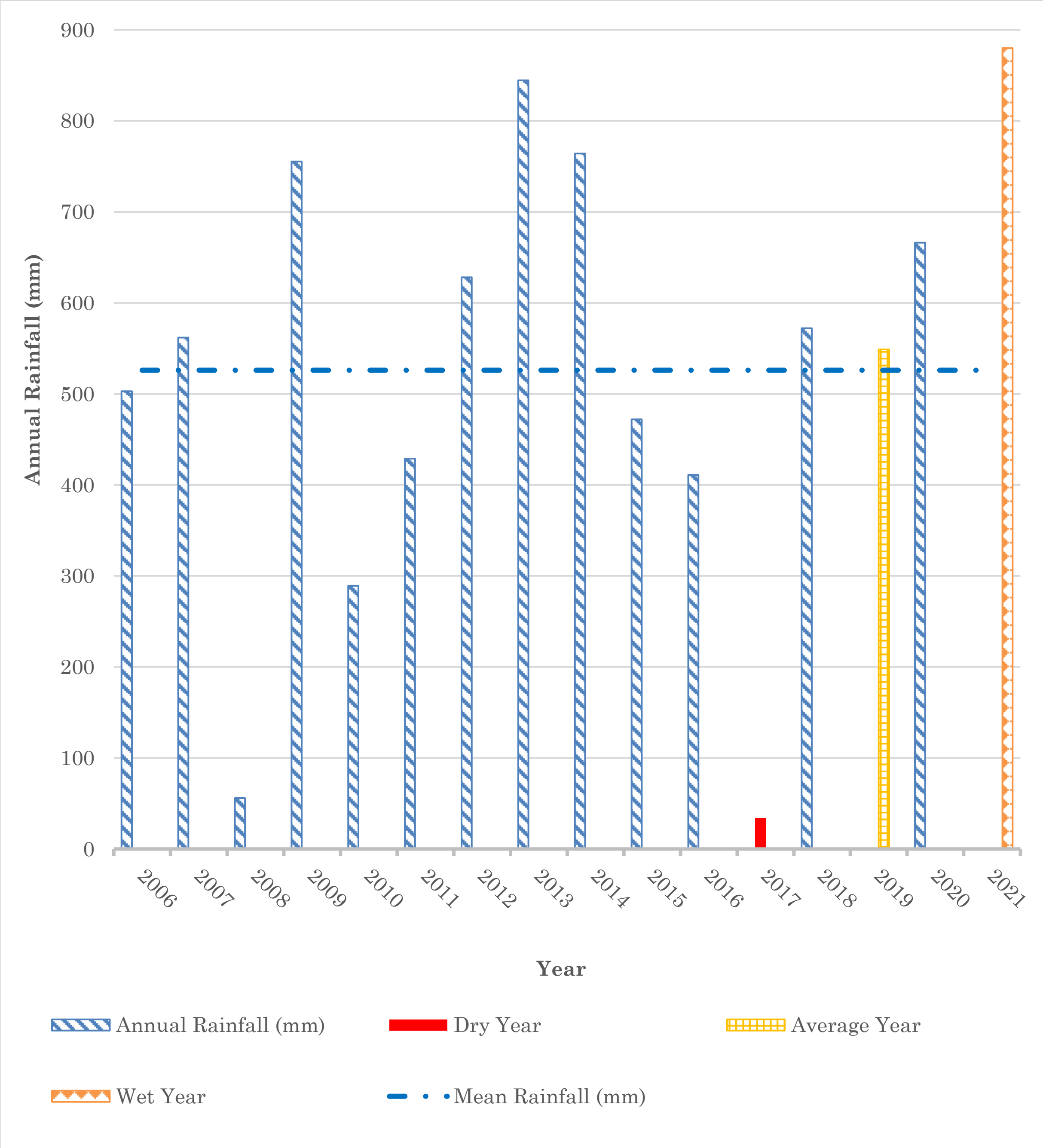

Figure 8-28: Periods used to simulate observed climatic variations

8.6.1.4 Post retrofit extended scenarios using climate change data (Scenarios 11 & 12)

Climate change is predicted to significantly impact the Global South and could exacerbate water scarcity in South Africa (Donnenfeld *et al.*, 2018; Dosio *et al.*, 2021; Muyambo *et al.*, 2022; Ziervogel *et al.*, 2022). Consequently, research into the potential impact of climate change on meteorological conditions has been conducted

(Kitoh & Endo, 2016; Pohl *et al*., 2017). The University of Cape Town (UCT) Climate Systems Analysis Group (CSAG) has developed 26 regional climate change scenarios from the Coupled Model Intercomparison Project Phase 5 (CMIP5) global projection (Taylor *et al*., 2012). These models were downscaled for several regions in the country using existing weather stations, including the Cape Town International Airport weather station, located 6 km from the School Pond (Figure 8-6).

The University of Cape Town (CSAG) analysed the rainfall and temperature datasets for the Cape Town International Airport weather station derived from the 26 climate change models and found that 25 of the 26 climate models predict that the CoCT is likely to face drier and hotter years, with only one model predicting a wetter than average year. Further, studies suggest that while South Africa will experience a decreased volume of annual precipitation, it will see an increase in the volume of rainfall – heavier than the historical mean – during particular events (Pohl *et al*., 2017; Jury, 2018; Muyambo *et al*., 2022). These observations have important implications for SWH when considering the retrofit designs proposed in this study. In scenarios with fewer rainfall events but higher intensity, retrofitting ponds can effectively capture and store the excess water that would otherwise be lost from the system, helping replenish groundwater resources.

This study used two climate change models for the evaluation of climate change impacts on MAR in the retrofitted pond. The climate models were derived from two representative concentration pathway (RCP) scenarios, namely the intermediary mitigation scenario (RCP 4.5) and the high emission scenario (RCP 8.5) (Riahi *et al*., 2011; Thomson *et al*., 2011).

The chosen climate change models represented the wettest and driest scenarios from the available 26 models, making them suitable for simulating the effects of climate change on MAR in the pond. The two selected models are the Hadley Global Environment Model 2 climate change RCP 8.5 (HadGEM2-CC-rcp85) and the Institute of Numerical Climate Model version 4 RCP4.5 (inmcm4-rcp45) models.

Daily rainfall and temperature data from selected climate change models were obtained from the UCT-CSAG group. The evapotranspiration was calculated using the Blaney-Criddle Method. The decision to exclude the Hargreaves Method which had been used previously, was based on its inclusion of the extra-terrestrial radiation component, which has the potential to vary across different emission scenarios. The Blaney-Criddle Method was deemed more appropriate because atmospheric conditions, which influence extra-terrestrial radiation, are expected to change under these scenarios. Furthermore, the Blaney-Criddle Method considers the mean daily sunlight hours, which are projected to undergo minimal changes in the considered climate change scenarios (Hargreaves & Allen, 2003).

The daily rainfall data was disaggregated into 15-minute intervals using paired stochastic rainfall disaggregation methods based on Socolofsky *et al.* (2001). The stochastic method uses observed shorter interval rainfall from the same or nearby rain gauge and stochastically disaggregates the longer duration rainfall using the observed rainfall distribution to desired shorter intervals. The paired disaggregation can be done using manual calculations or software like NetSTORM.

The NetSTORM software was used to disaggregate the HadGEM2-CC-rcp85 and inmcm4-rcp45 daily rainfall data into hourly and 15-minute intervals (Figure 8-29). The disaggregated data was then converted to time series files on the PCSWMM software.

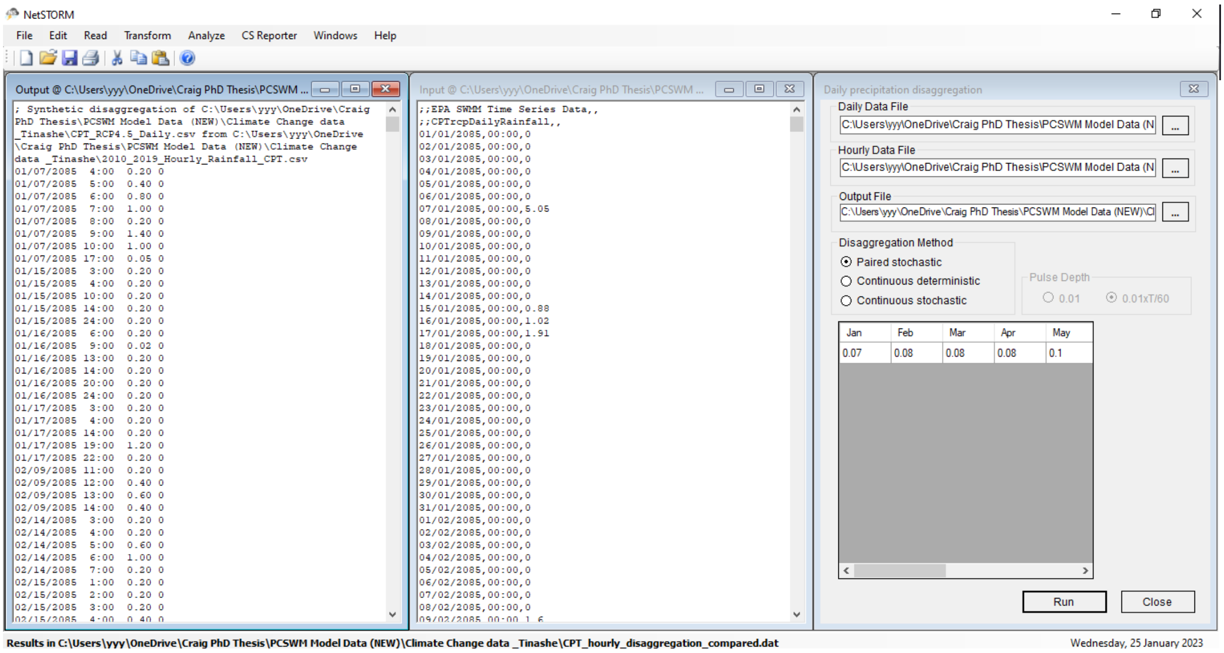

Figure 8-29: Paired stochastic disaggregation of daily interval rainfall

The simulation duration for this study spanned from 2084 to 2100, maintaining the same model duration of 16.4 years. Figure 8-30 presents the projected changes in rainfall from the climate change models, comparing them with the historical data.

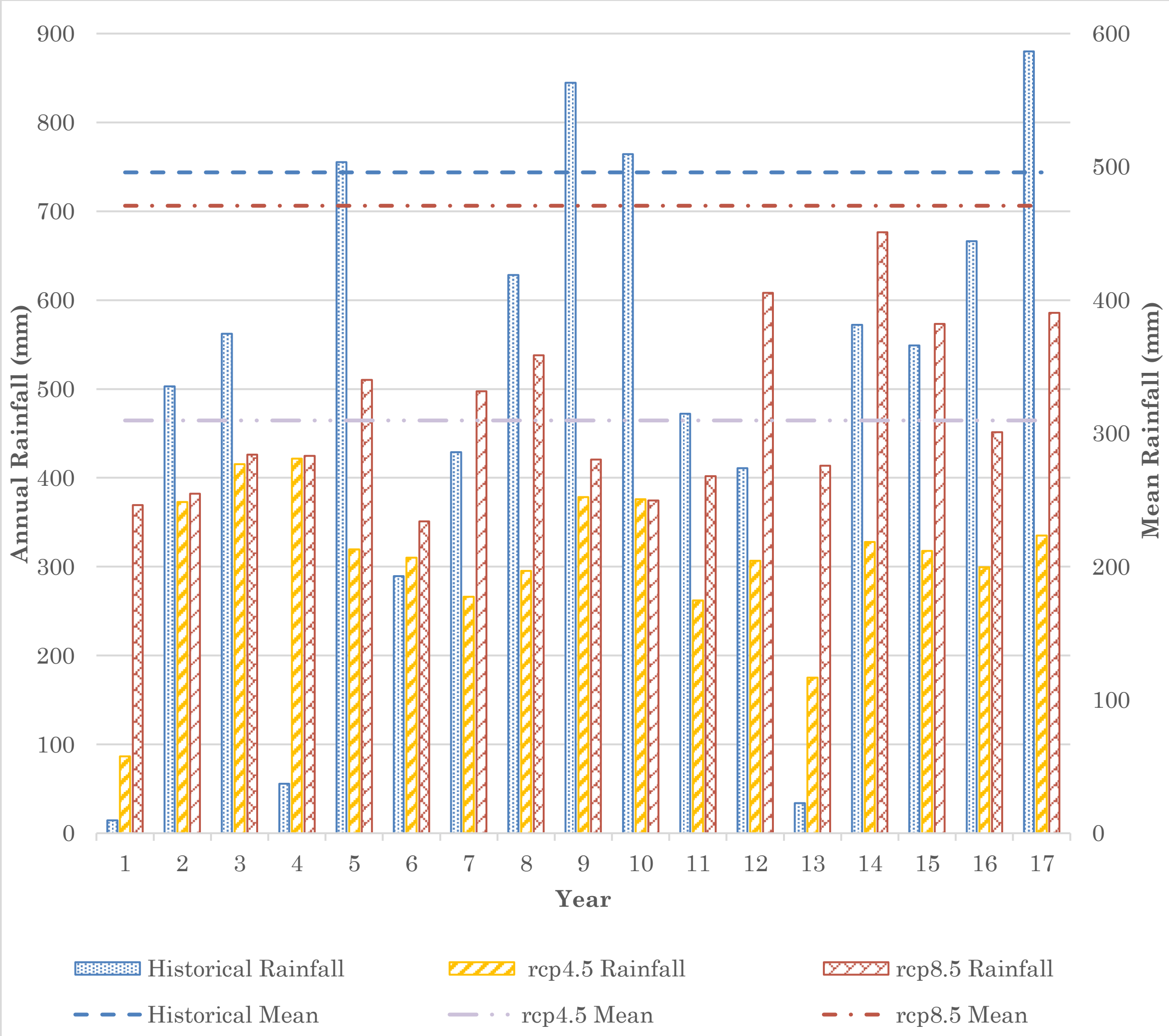

Figure 8-30: Variation of annual rainfall depths

Despite experiencing a drought between 2015 and 2018, the historical data used in this study shows a higher mean rainfall of 496 mm over the observed 16.4 years compared to the climate change scenarios. Further, even with the intermediary scenario of RCP 4.5, there is a possibility of drier years than the historical mean with a mean rainfall of 310 mm over 16.4 years. In contrast, the high-emissions scenario of RCP 8.5 predicts wetter years than the intermediary scenario with a mean rainfall of 471 mm for the 16.4 years assessed. The high-emissions scenario assumes continuously high levels of greenhouse gas emissions throughout the 21st century, resulting in increased carbon dioxide (CO_2) and other greenhouse gas concentrations in the atmosphere (Riahi *et al.*, 2011; Thomson *et al.*, 2011). Warmer temperatures resulting from the increased greenhouse gas concentrations can alter the hydrological cycle, influencing precipitation patterns.

Figure 8-31 displays the mean daily evapotranspiration for the historical and climate change scenarios examined. The intermediate scenario (RCP 4.5) indicates higher temperatures and, consequently greater evapotranspiration compared to the historical means recorded at the Cape Town Airport weather station (22% higher evapotranspiration). However, the high emissions scenario (RCP 8.5) exhibits the most substantial annual increase in daily temperatures among the three scenarios, leading to the highest annual evapotranspiration across all scenarios considered (25% higher evapotranspiration than the historical data).

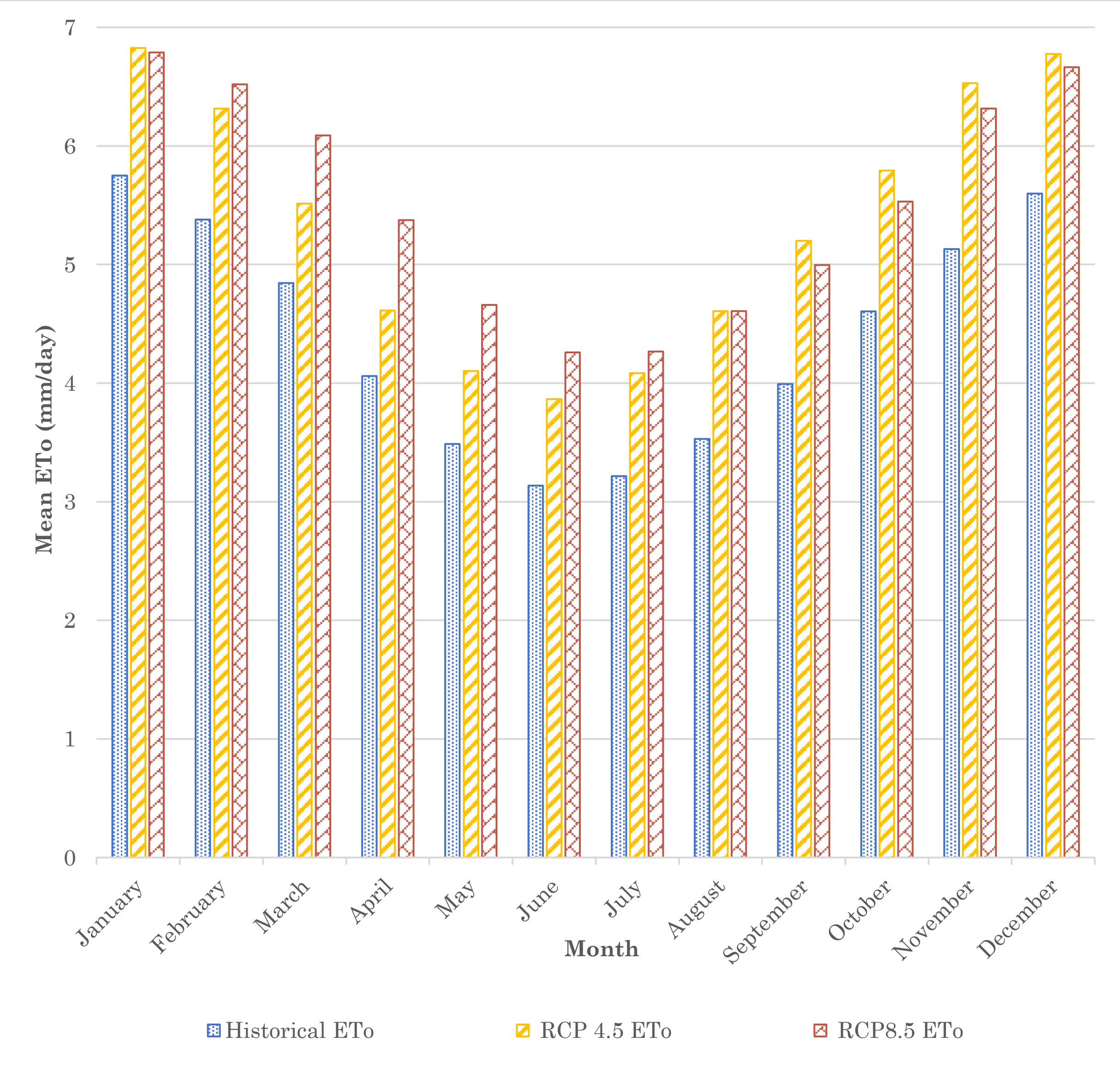

Figure 8-31: Impact of climate change on evapotranspiration

8.6.2 Description of model structures for all scenarios

8.6.2.1 Pre-retrofit scenarios

The School Pond was divided into sub-catchments in this section: the general_pond_area and the swampy_area. These sub-catchments were assigned the same slope, infiltration, and pervious area properties as the pond before implementing the infiltration swale retrofit. This configuration was used to simulate Scenarios 1, 3, 5, and 9 (Figure 8-32).

Figure 8-32: Pre-retrofit model structure

In the model, run-on and runoff from the 'general_pond_area' – that did not infiltrate – were routed to the 'swampy_area' sub-catchment and contributed to its inflow. The 'swampy_area' inflow and run-on – less infiltration and evapotranspiration – flowed out of the pond into the CoCT's trunk stormwater network if the depression storage was exceeded. The water table response was then simulated. If the water level rose to the surface, some water evaporated, while the rest was routed out of the pond via the pond outlet and into the trunk stormwater network.

The pond was assigned the parameters detailed in Section 8.3.1.4. The initial groundwater level was set as the lowest observed January 2022 water table level (27.5 mamsl) – as observed from the monitoring wells in Figure 8-22 for the 'As-is' scenarios. The pond sub-catchments and the CFA aquifer were all assigned the calibrated parameters described in Sections 8.3.1.7 and 8.5.2.

8.6.2.2 Post–retrofit scenarios

The School Pond was represented as four sub-catchments, namely, slope_to_pond, infiltration_swale, general_pond_area, and the swampy_area (Figure 8-33) for the post-retrofit scenario.

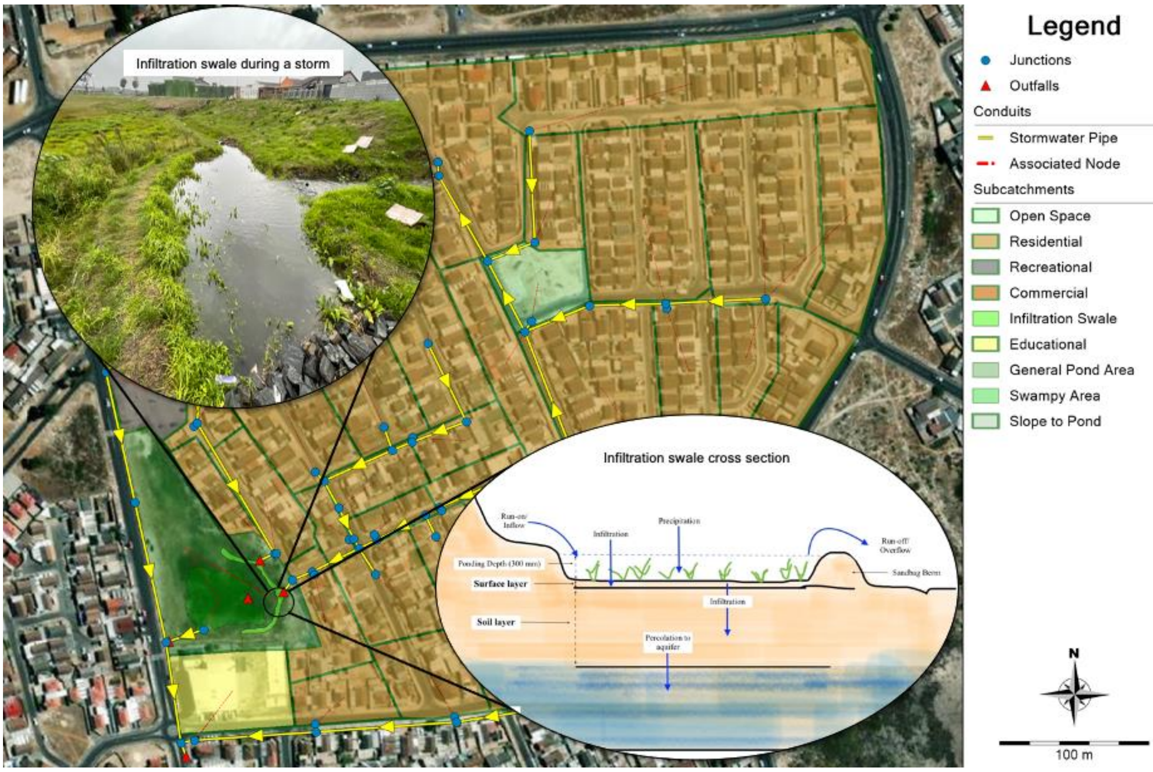

Figure 8-33: Post-retrofit model structure

These sub-catchments were also assigned the infiltration parameters described in Section 8.3.1.4 and calibrated and adjusted in Section 8.5.2. In addition, the 'Infiltration_swale' sub-catchment was assigned the LID component described in Section 8.3.1.6 and the overflow was directed over the berm and into the swampy_area. Any resultant groundwater outflow – if the water table rose to the surface – was simulated to flow out of the pond via the pond outlet and into the trunk stormwater network. This scenario was used to evaluate the pond's performance for Scenarios 2, 4 and 6 and the climate change scenarios.

8.6.2.3 Post–retrofit Extended_LID scenario

This scenario aimed to assess the recharge potential of a retrofitted stormwater system in the event of a water table decline caused by the CoCT's groundwater abstraction from the CFA. The primary objective was to enhance the capacity of the system to detain and infiltrate the stormwater. By adopting this design, an unsaturated zone of approximately 2.5 meters was established at the lowest point of the pond, allowing for improved water infiltration, storage, and the potential to retrofit a more extensive section of the pond.

Adopting a larger area involved implementing the designs discussed by Käppeli (2020) and presented in Section 6.2.1, which featured terraces covering a significant portion of the pond. The Extended_LID scenario (Scenario 4) thus incorporated five terraced infiltration swales (Figure 8-34) positioned at elevations below the lowest inlet invert level – 29.2 mamsl.

Figure 8-34: Extended LID area model structure

The stormwater inflow into the pond was directed into the first infiltration swale and ponded to a depth of 300 mm berm, where some water infiltrated, evaporated, and any additional inflow overflowed. The overflow then cascaded to the next infiltration swale, located 300 mm below the previous one, with a berm. This sequence continued until any additional inflow exited the pond via the outlet and into the stormwater trunk. The combined area of infiltration swales was 4857 m^2, covering 49% of the entire pond area. The terraced infiltration swales were represented by five LID units that covered the extended infiltration swale area in the model. All other hydrological processes were the same as those described in Section 8.6.2.2. The Extended_LID scenario was modelled using extended climatic data for 2005 – 2022, with the initial water table set to the mean water table from the CoCT MAR and abstraction scenarios.

8.7 Scenario results

The following section presents the outcomes obtained from the modelled scenarios.

8.7.1 Scenarios 1 - 4

A 16.4-year (2005 – 2022) continuous simulation was conducted across four scenarios to assess the infiltration swale performance under various geohydrological conditions. The following scenarios and their associated rainfall data (in brackets) are presented:

1. Pre-retrofit harvested volume (2005 – 2022 rainfall data).
2. Post-retrofit harvested volume (2005 – 2022 rainfall data).
3. Post-retrofit harvested volume considering the impact of a lowered water table resulting from CoCT groundwater abstraction and MAR (2005 – 2022 rainfall data).
4. Post-retrofit harvested volume when the water table is lowered, and the infiltration swale area is extended (2005 – 2022 rainfall data).

A summary of simulation results is presented in Figure 8-35. The total rainfall received amounted to 8435 mm. The pond received runoff from its contributing sub-catchments through Inlets 1 and 2 and the precipitation that fell on the pond. Thus, the total stormwater in the pond was taken as the sum of the pond inflow and surface runoff. In these four scenarios, the total volume of stormwater passing through the pond over the 16.4 years totalled 256 000 m^3 (256 ML).

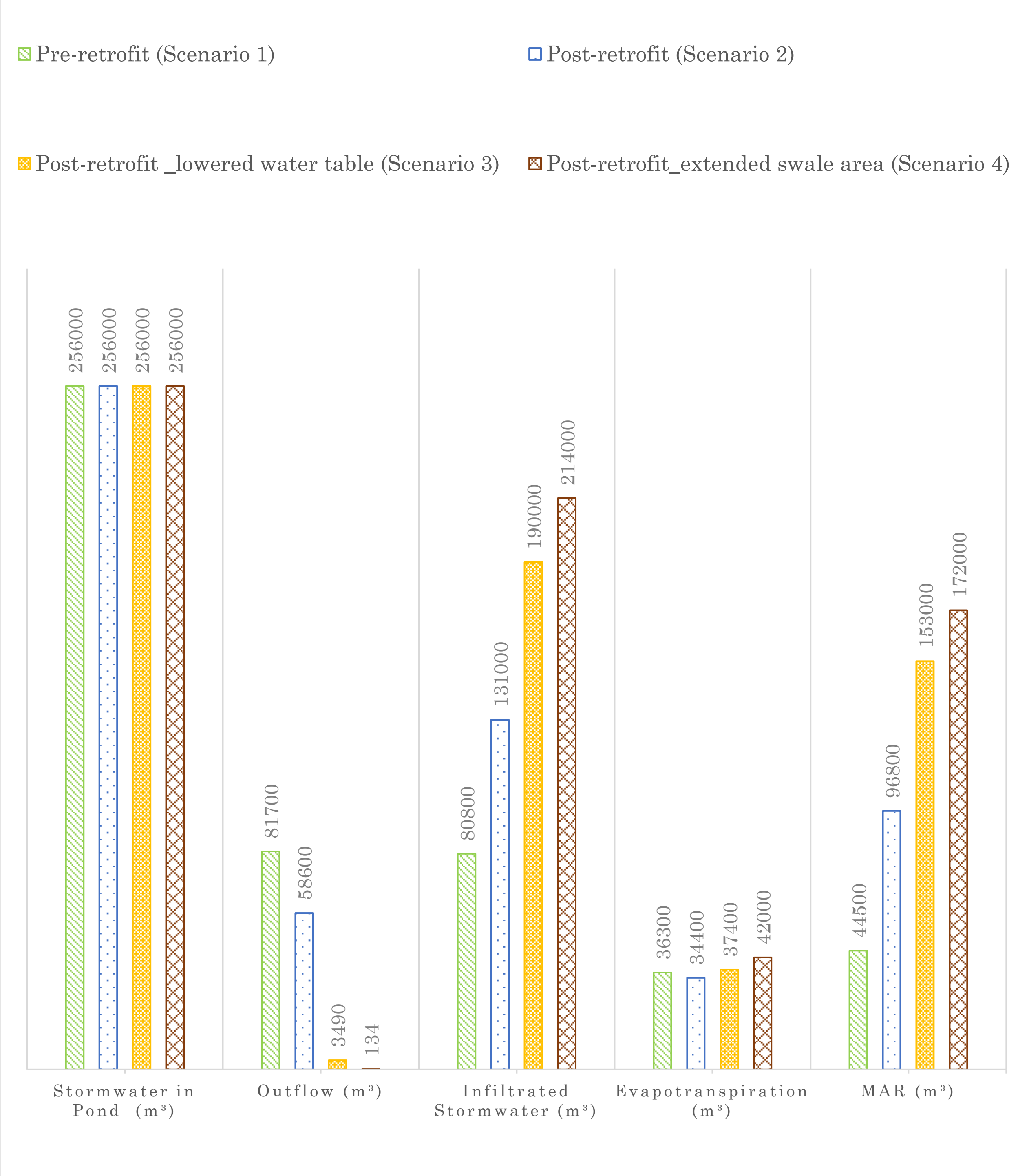

Figure 8-35: Water balance for Scenarios 1 – 4

The School Pond had some infiltration potential even before the retrofit exercise – infiltrating 17% of the stormwater volume in the pond (Figure 8-35). The infiltration volume presented here is the water that infiltrates less surface evaporation. It can be seen, however, that ~45% of the water that infiltrates in Scenario 1 is lost as evapotranspiration – a combination of evaporation from groundwater and plant roots. The evapotranspiration volume is similar for all scenarios, although it is seen to increase in scenarios with a reduced water table (Scenario 3 & 4) as more water is available for evapotranspiration in the vadose zone due to increase infiltration. The reduced evapotranspiration in Scenario 2 compared to Scenario 1 is a result of the pre-retrofit scenario having a larger area of low permeability (clayey soil) which means more water is available for evapotranspiration. The infiltration swale is more permeable than the general_pond_area and the swampy_area; thus, more water is directly stored in the CFA, and less is available for evapotranspiration.

The total MAR volumes for the four scenarios are compared in Table 8-16 and Figure 8-36. Over the 16.4 years, the total MAR was predicted to increase by 52 300 m^3 (118%) if the retrofit was implemented. Over 16.4 years, the MAR volume contributes 17% of the CoCT's peak daily demand in the summer of 2022 (Western Cape Government, 2022). Lowering the water table by 2.5 m results in an increase in MAR by 244% and a total MAR volume increase of 108 500 m^3. The MAR volume further increases if the infiltration swale size (area) is extended in the lowered water table scenario; the resultant MAR increase is 127 500 m^3 – a 286% increase in MAR compared to the pre-retrofit scenario.

Table 8-16: Results for Scenarios 1 – 4 (2005 - 2022)

	Pre-retrofit (Scenario 1)	**Post-retrofit (Scenario 2)**	**Retrofitted with lowered water table (Scenario 3)**	**Retrofitted extended infiltration swale (Scenario 4)**
Infiltrated Stormwater (m^3)	80800	131000	190000	214000
Evapotranspiration (m^3)	36300	34400	37400	42000
MAR (m^3)	44500	96800	153000	172000
SW infiltrated to GW (%)	17	38	60	67
MAR (m^3/mm)	5.3	11.5	18.1	20.4
Percentage change (%)	–	118	244	286

The model outcomes indicate an increase in MAR when the retrofit is introduced is a consequence of an increase of the infiltration by detaining and infiltrating water that would otherwise be lost to the stormwater network.

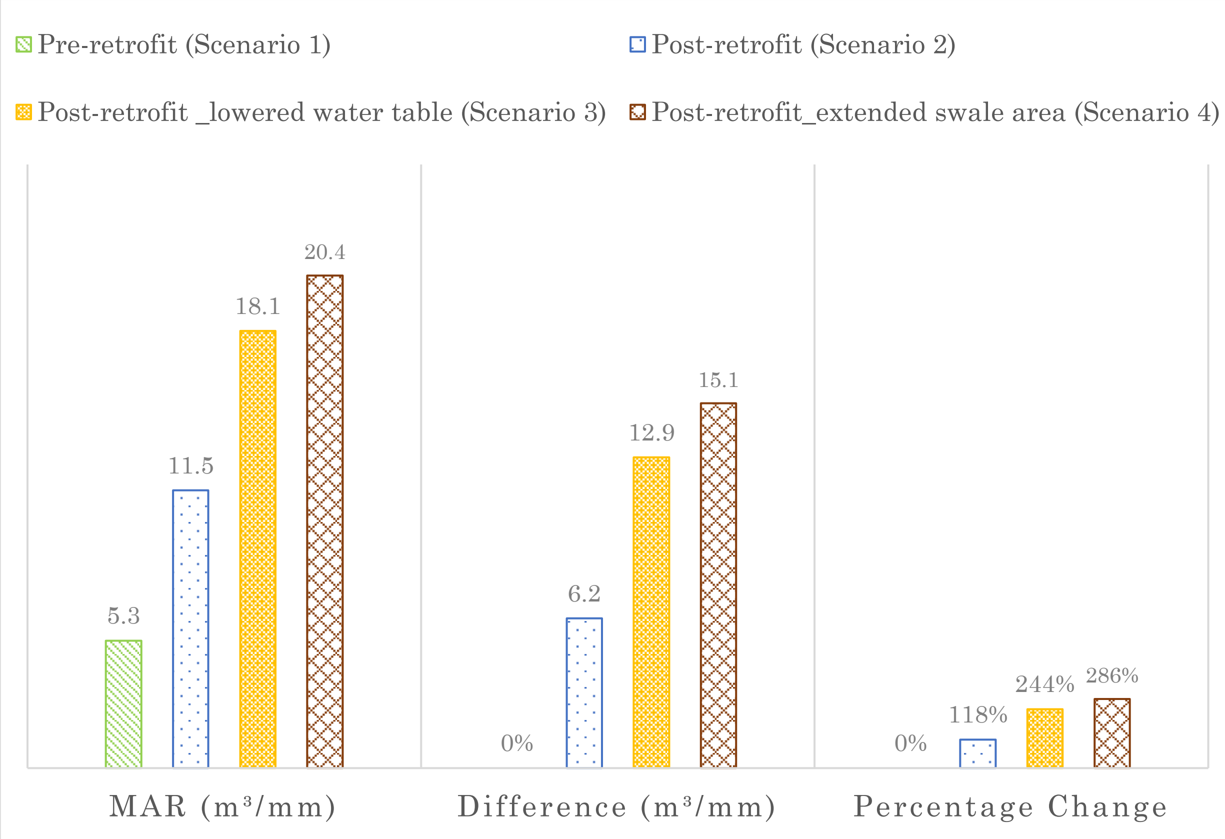

Figure 8-36: Impact of retrofitting the School Pond on MAR under different geohydrological conditions

8.7.2 Scenario 5 – 10

The preceding section outlined the outcomes of long-term simulations examining the volume of harvested stormwater under different geohydrological conditions. Although these results provide insights into the long-term performance of the pond retrofit, they may not fully capture its behaviour under extreme climatic conditions. Consequently, considering both climatic extremes and a 'mean year', six additional scenarios were evaluated.

The six scenarios investigated were as follows:

5. Harvested volume in a year with low rainfall – Pre-retrofit
6. Harvested volume in a year with low rainfall – Post-retrofit
7. Harvested volume for a year with mean rainfall – Pre-retrofit
8. Harvested volume for a year with mean rainfall – Post-retrofit
9. Harvested volume in a year with the highest rainfall – Pre-retrofit

10. Harvested volume in a year with the highest rainfall – Post-retrofit

The rainfall data was sourced from the Wolfgat weather station and the rainfall patterns for the three years were comparable (Figure 8-37). However, the annual total rainfall amounts varied, with recorded depths of 290, 550, and 880 mm for 2010, 2019, and 2021, respectively.

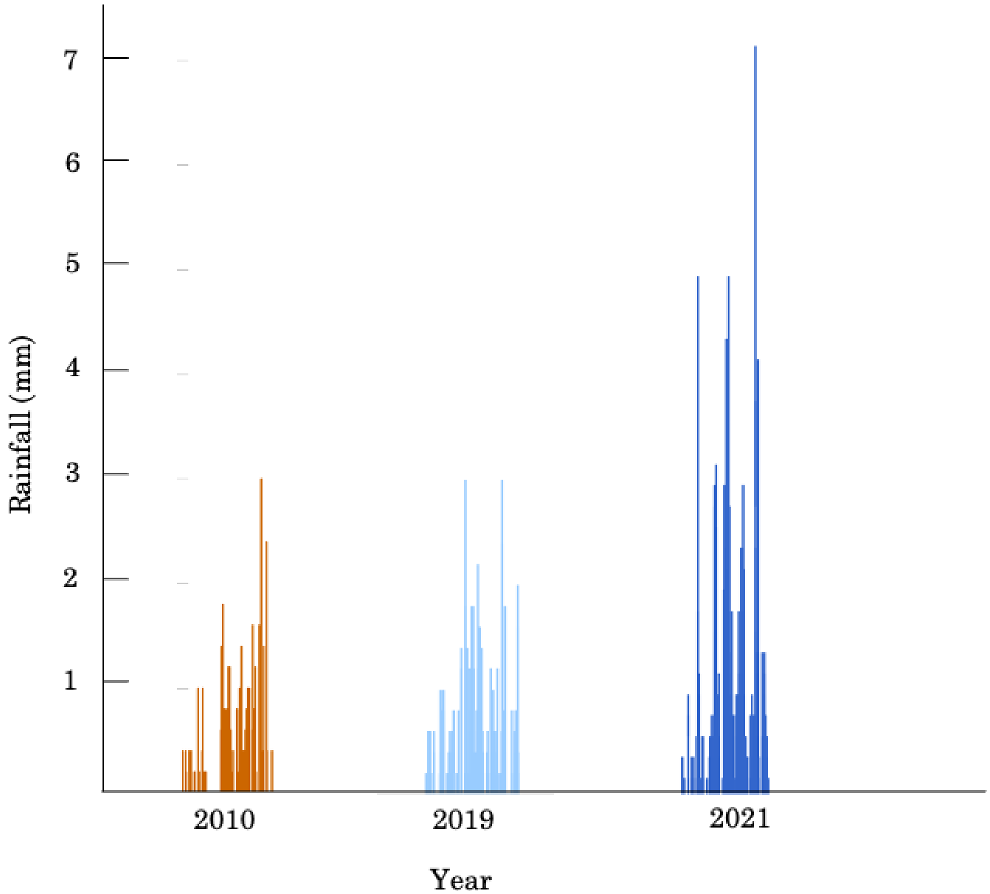

Figure 8-37: Rainfall depths for the three climatic conditions

The results from the three scenarios indicate that the retrofit increases the MAR in the pond to varying extents (Figure 8-38). The percentage MAR increase is highest in the 2010 scenario due to the retrofit reducing the outflow by 74% (the 2019 and 2021 scenarios have 43% and 32% outflow reduction, respectively). The total evapotranspiration from the three years shows an incremental trend.

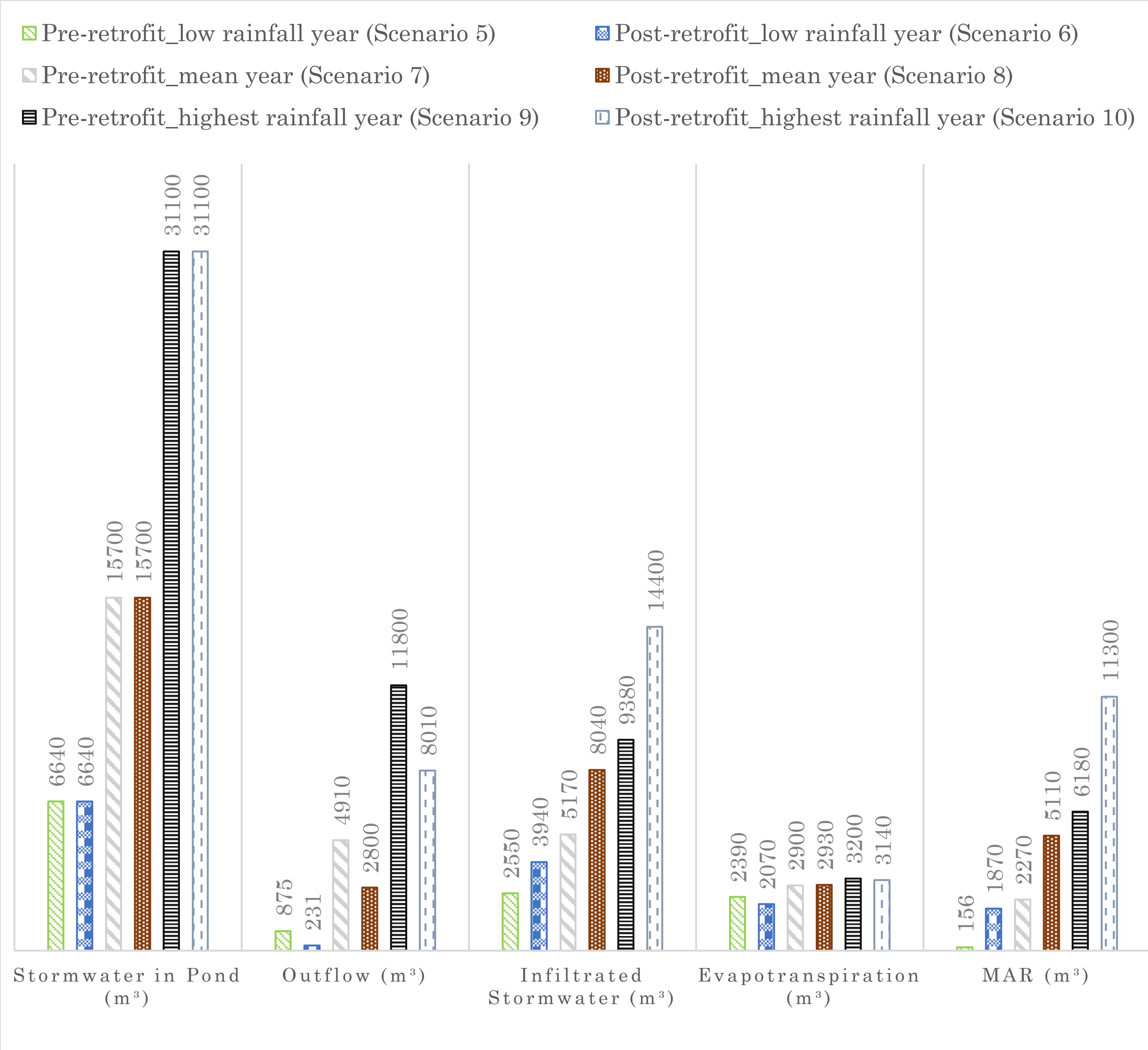

Figure 8-38: Water balance for Scenarios 5 – 10

Figure 8-39 shows a higher MAR increase (~1100%) in the drier year. The increase is a result of the lower volume of precipitation and more antecedent dry days, so the aquifer is not fully saturated for extended periods, thus allowing storage recovery. The aquifer recovery then means that when the next storm occurs, the resultant infiltrated stormwater volume can be stored before the soil is fully saturated and the water overflows out of the pond. This effect is magnified in the post-retrofit scenario as the infiltrated volume increases due to the extended stormwater detention. In short, the aquifer provided more storage in the drier year, and the infiltration swale provides increased infiltration.

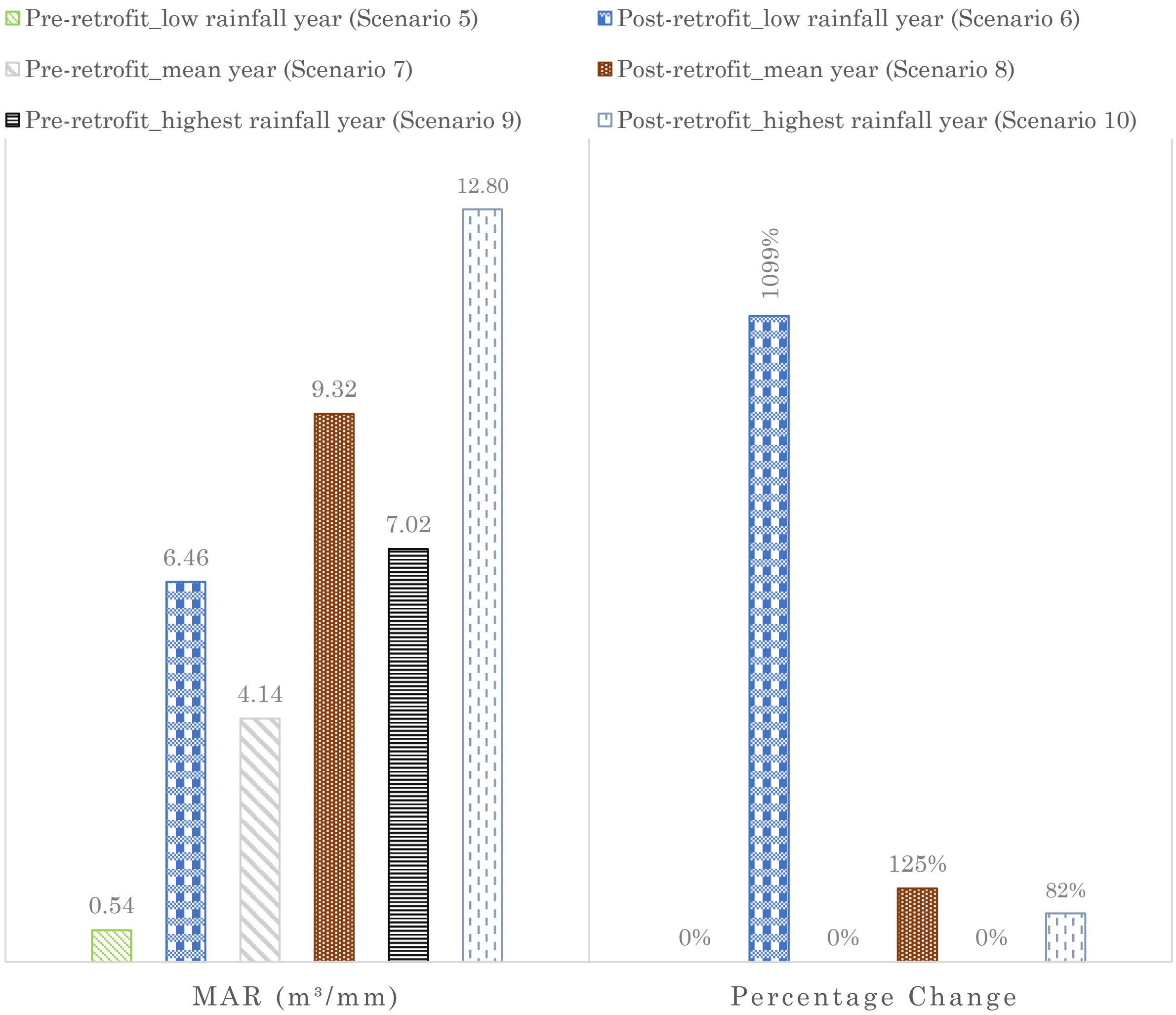

Figure 8-39: Impact of the pond retrofit on MAR under three hydrological conditions

It was initially assumed that antecedent dry days play a more significant role in groundwater table recovery – more dry days result in more storage, and the inverse was true. This phenomenon is illustrated Figure 8-40 where the 2010 post-retrofit scenario had more antecedent dry days and experienced a more considerable elevation drop. However, the precipitation depth received was found to have more influence on MAR, possibly due to 1) the rate of groundwater transfer being the constant for all scenarios considered, and 2) the sandy CFA quickly transmitting water over the aquifer such that there is room receive and store precipitation before the aquifer is saturated. The 2021 scenario had more frequent (and wetter) precipitation events and saturated the available storage quickly, but the aquifer also recovered quickly if the rainfall events were spread out (Figure 8-40).

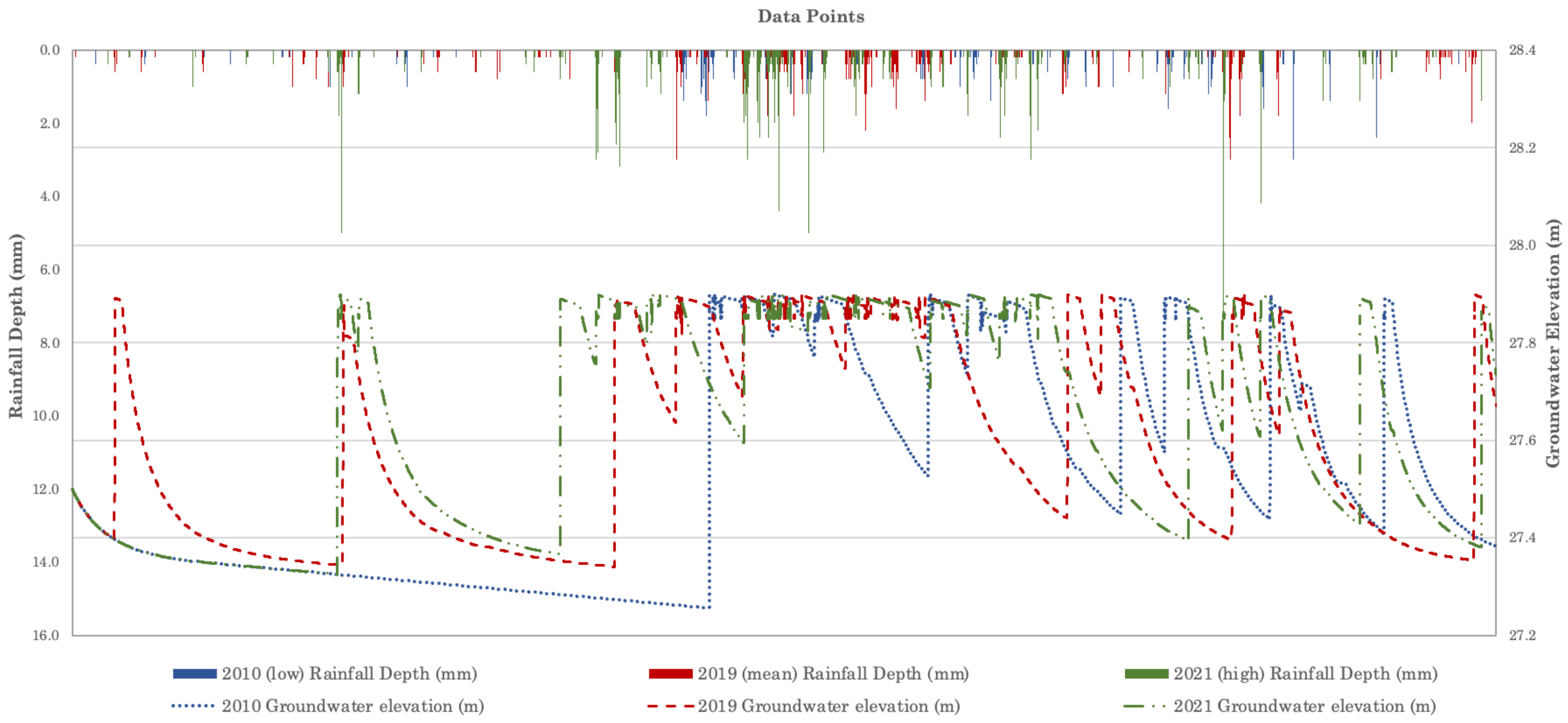

Figure 8-40: Groundwater elevation and rainfall depths for the three post-retrofit scenarios

8.7.3 Scenario 11 – 12

Two scenarios using continuous rainfall data were simulated to investigate the performance of the infiltration swale under two climate change scenarios – intermediate and high emissions pathways. These scenarios are then compared with Scenario 2 – obtained from historical data from 2005-2022. Section 8.6.1.4 already established that the three scenarios have different rainfall patterns and mean rainfall, as shown in Figure 8-30. The total projected rainfall for the Cape Town station from 2084-2100 amounted to 5285 mm and 8025 mm for the RCP 4.5 and RCP 8.5 scenarios, respectively.

A summary of the simulation results is shown in Figure 8-41. The total volume of stormwater in the pond over in three scenarios, was 257 000 m³, 108 000 m³ and 181 000 m³ for Scenarios 2, 11 and 12, respectively.

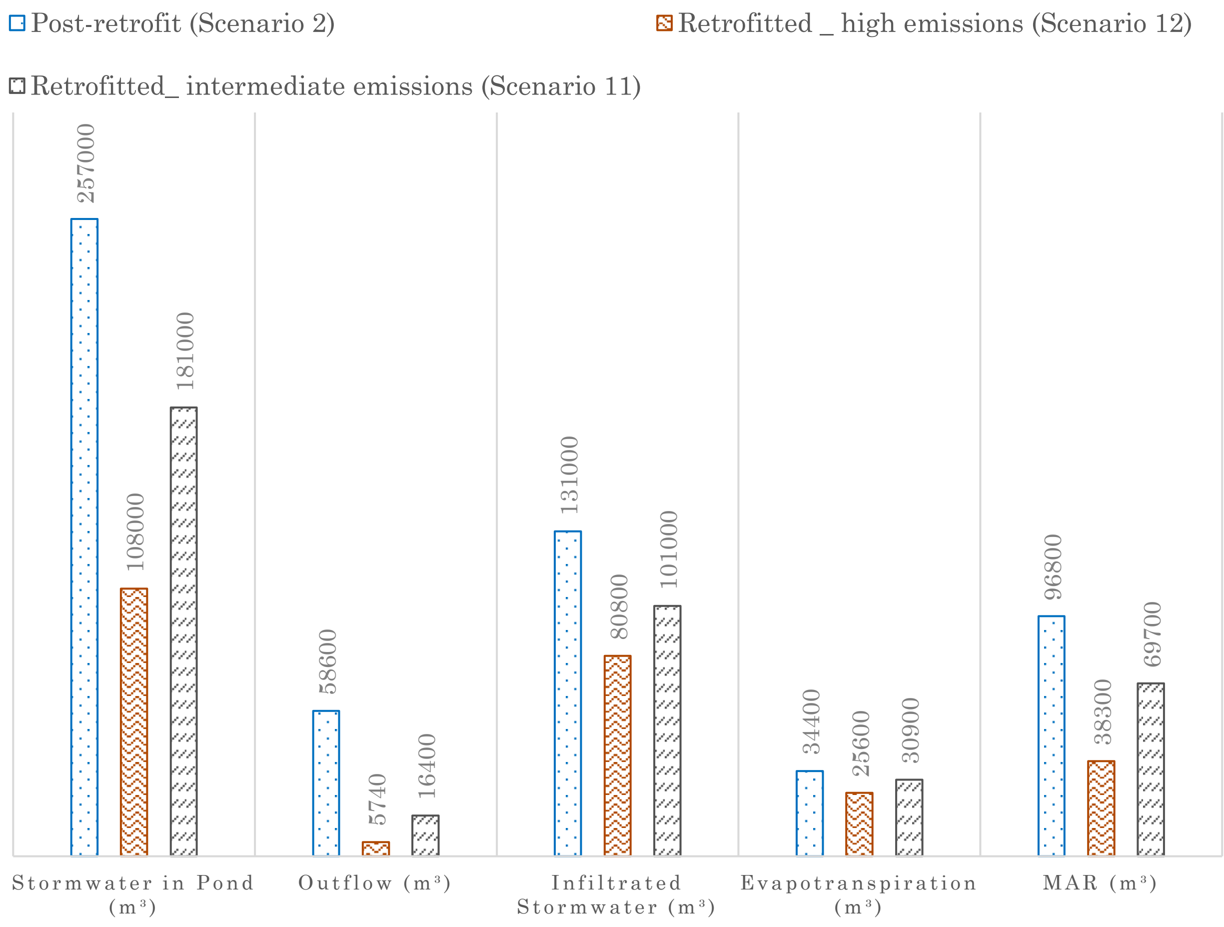

Figure 8-41: Water balance for Scenarios 2, 11 and 12

The outcomes of the three scenarios demonstrate a correlation between MAR in the pond and the amount of rainfall received. A decrease in rainfall directly corresponds to a reduction in MAR. Table 8-17 provides insights into the percentage of stormwater infiltrating the ground in each scenario. Scenario 11 exhibits the highest infiltration rate, with 74% of stormwater effectively infiltrating into the ground. Conversely, Scenarios 12 and 2 display lower infiltration percentages of 56% and 51%, respectively. These disparities stem from the frequency and intensity of rainfall events. During drier years, the retrofitted basin captures a greater proportion of water, resulting in reduced outflow. In this instance, Scenario 2 experiences a loss of 23% of its stormwater as outflow, whereas Scenarios 11 and 12 show losses of only 5% and 9%, respectively. Furthermore, drier spells allow for more substantial water table recovery, facilitating enhanced infiltration. However, dry spells also result in increased infiltrated stormwater loss through evapotranspiration. Scenario 11 had the highest stormwater loss rate at 31%, followed by Scenarios 12 and 2 at 31% and 26%, respectively.

Table 8-17: Results for Scenarios 2, 11 and 12

	Post-retrofit (Scenario 2)	**Retrofitted_ intermediate emissions (Scenario 11)**	**Retrofitted _ high emissions (Scenario 12)**
Infiltrated Stormwater (m^3)	131000	80800	101000
Evapotranspiration (m^3)	34400	25600	30900
MAR (m^3)	96800	38300	69700
SW infiltrated to GW (%)	38	35	39
Mean Annual MAR (m^3)	5900	2340	4250
MAR (m^3/mm)	11.5	7.3	8.7
Percentage Change	–	-37%	-24%

The decrease in MAR in Scenario 11 results from the high evapotranspiration also present in Scenario 12 (Table 8-17); however, in the latter, it is counter-balanced by a greater volume of infiltrated stormwater. The MAR volume for Scenarios 11 and 12 – over 16.4 years – would contributes 4% and 8% of the CoCT's peak daily demand in the summer of 2022, respectively (Western Cape Government, 2022).

8.8 Discussion

This chapter investigated the long-term impacts of the retrofitted pond under twelve different scenarios with varied results, which are thematically discussed below.

8.8.1 Impact of retrofitting detention ponds on MAR

As outlined in Section 8.7.1, the model indicates an increase in MAR following the implementation of the retrofit – a 120% increase compared to the pre-retrofit scenario (Scenario 1 vs Scenario 2). This outcome underscores the potential of retrofitting Cape Town's detention ponds, as described in Section 6, to enhance the MAR. Retrofit measures achieve this by facilitating increased infiltration, enabling the detention and absorption of water that would otherwise be discharged into the stormwater system. Nonetheless, consistent with the studies conducted by Seyler *et al*. (2016) and Mauck (2017), this research affirms that the high-water table constrains the MAR in the CFA during winter. Nevertheless, there are encouraging prospects for augmenting MAR volumes and replenishing the aquifer through groundwater abstraction efforts undertaken by the CoCT in the CFA that would also expand the infiltration swale area, leading to a 290% increase in MAR volumes compared to the pre-retrofit scenarios.

The modelling shows the considerable potential of retrofitting methods across various realistic scenarios, even though the contribution of a single pond to the CoCT's daily water demand is minimal. The anticipated benefits of retrofitting are expected to extend to other ponds overlying the CFA. This assertion is validated by the observed dominant influence of CFA characteristics compared to other modelled parameters, except for LID soil thickness (the vadose zone). The dominance of the CFA characteristics was established through the sensitivity analysis, which identified that ten of the eleven parameters influencing the groundwater response were associated with the modelled CFA parameters. Thus, incorporating infiltration swales into any pond situated in the CFA is projected to lead to similar improvements in MAR, as observed in this study.

Consequently, it can be concluded that retrofitted ponds overlying the CFA, are highly likely to experience increased MAR, even when applied at the upper limits of the allowable vadose depths (0.6 m).

8.8.2 Impact of extreme historical climatic conditions on MAR

The results from the different climatic conditions (wet, mean, and dry years) indicate that while MAR is a function of rainfall depth, it is the rainfall frequency and evapotranspiration that play a significant role in determining the final volume of stormwater stored under MAR. High frequency and intensity rainfall result in increased pond outflow, while high frequency and low intensity rainfall results in increased infiltration primarily from the stormwater volume retained by the berm. However, in the case with limited vadose zone (high water table), high-frequency rainfall can result in less infiltration due to the aquifer filling up and having fewer antecedent days and less time to recover. Therefore, optimal MAR is linked to rainfall patterns facilitating optimum aquifer storage, transfer, and subsequent recovery.

8.8.3 Impact of climate change on MAR

This section of the study established that climate change will significantly impact the amount of rainfall and, consequently, the volume of stormwater received at the pond. The investigated scenarios showed a decrease in the magnitude of MAR in both climate change scenarios, although the high emissions scenario had more rainfall and consequently MAR than the intermediary emissions scenario. Additionally, as described by the two representative concentration pathways, climate change will also lead to an increase in evapotranspiration (Figure 8-31) as the summers (and winters) get hotter and drier. The higher evapotranspiration rates from the retrofitted pond could positively contribute to urban cooling and mitigate the heat island effects though evaporative cooling – a process where evaporated water absorbs heat from surrounding air and surfaces (Coutts *et al.*, 2013; Qiu *et al.*, 2013; Moss *et al.*, 2019; Bakhshoodeh *et al.*, 2022).

8.8.4 Model limitations

The geohydrological model's development and implementation were constrained by several factors, which are essential for interpreting the results. Despite attempts to mitigate these constraints, it is important to acknowledge the limitations of the findings and conclusions.

Constraints arise from the nature of the calibration data. This study relied on storm events rather than continuous flow data because of the impracticality of installing a permanent flow meter at the study site. Only four of the seven recorded storm events were usable because of a logger error that rendered the other events unreliable. Although Titterington *et al.* (2017) acknowledge the appropriateness of the calibration events used in this study, it is essential to recognise that such a model may not capture system responses under extreme conditions. Therefore, a potential overestimation of the MAR values should be noted.

The use of PCSWMM for simulating the Cape Flats aquifer's characteristics was predicated on a constant aquifer transfer rate. To account for this limitation, it is recommended to employ a coupled model, such as SWMM-MODFLOW, which allows for variable aquifer transfer rates. Despite efforts to incorporate this approach in the present study, the necessary data to calibrate and validate the flux-dependent aquifer boundary conditions in MODFLOW was unavailable, leading to the conclusion that the coupled model would not provide superior results compared to the stand-alone model that had been calibrated and validated.

A lack of appropriate instrumentation hindered the recording of pond outflow volumes. Despite the initial plans to install an instrumented weir, prohibitive costs and theft risks made this unfeasible. Consequently, it was not possible to validate the pond water balance. It is crucial to recognise that the significance of the outflow parameter may be somewhat diminished, as water exiting the system did not contribute to the MAR. Infiltration, on the other hand, was evaluated using the calibrated groundwater module in PCSWMM. Nevertheless, uncertainty remains regarding the accuracy of the overall water balance predictions.

Beyond the technical constraints, there were also practical challenges that played a role. Specifically, the model did not account for factors that could affect MAR, such as the presence of moles. This omission was based on the simulation relying on historical and future data, assuming no leaks in the swale via the mole holes, which was validated by installing a mole barrier. However, the long-term effectiveness of this barrier remains unknown. Finally, the exclusion of mole holes from the model is unlikely to have made a substantial impact on the computed MAR volumes in the pond.

8.9 Chapter summary

The hydrological implications of retrofitted detention ponds were examined in this chapter. The assessment was conducted using a calibrated and validated surface-subsurface PCSWM model. A modelling approach was adopted as a substitute to continuous monitoring of the pond because of security concerns surrounding the monitoring equipment installed at the School Pond and to determine the implications of the CoCT implementing groundwater abstraction from the CFA resulting in a lowered water table and the impact of climate change scenarios. The results presented in this section highlight the potential benefits of ponds retrofitted for MAR, as they contribute to water augmentation while fulfilling their primary function as detention ponds.

9. Evaluating the treatment potential of retrofitted detention ponds

This chapter presents findings from laboratory experiments investigating the treatment performance of retrofitted detention ponds overlying the CFA under varying pollutant loads and vadose zones. The primary objectives were to determine the stormwater quality range before and after treatment and to assess the treatment performance at different vadose zone depths, focusing on the implications for MAR under varying vadose zone conditions.

The laboratory experiments involved column studies employing eight uPVC pipes filled with media from the CFA and 'clean' silica sand. Synthetic stormwater (SSW) was used to maintain uniformity in the experiment by ensuring that stormwater had similar pollutant concentrations. Eight columns were dosed with SSW using dosing cycles representative of the historical rainfall pattern in Cape Town. The SSW effluent samples were collected after each dosing cycle, starting at the bottom sampling points. As the column saturated, the 'water table' rose, prompting a change in the sampling points (i.e., moving up the column). Sampling ports at 500 mm intervals were used to collect effluent samples tested in various laboratories. The pollutant removal efficiency of each column was determined by comparing influent and effluent pollutant concentrations. The effective treatment depth was identified by analysing pollutant outflow at different vadose zone depths.

This study used non-vegetated columns as previous studies have shown that vegetation improves treatment thus it was deemed more valuable to investigate the 'worst case' where only media is used (Bratieres *et al.*, 2008; Milandri *et al.*, 2012).

9.1 Experimental procedure

9.1.1 Column description

The column experiment comprised eight columns constructed from opaque uPVC pipes with a diameter of 200 mm and a length of 2 m. The bottom of the PVC pipe was sealed using flat uPVC caps. Opaque PVC pipes were used to mimic, as far as possible, the below-ground natural dark conditions in retrofitted ponds. The columns had three sampling ports – 25 mm drilled holes connected to 25 mm ball valves through a threaded adapter positioned 0, 0.5, and 1 m from the bottom of the column. Cheesecloth fabric was used to ensure sand (mud) did not flow out of the sample ports during the sampling. In addition, the inner pipe walls were sandpapered (Figure 9-1) to increase the surface roughness and prevent preferential flow paths down the sides of the column. A 2.0 x 2.5 m steel frame was constructed to secure the columns (Figure 9-2).

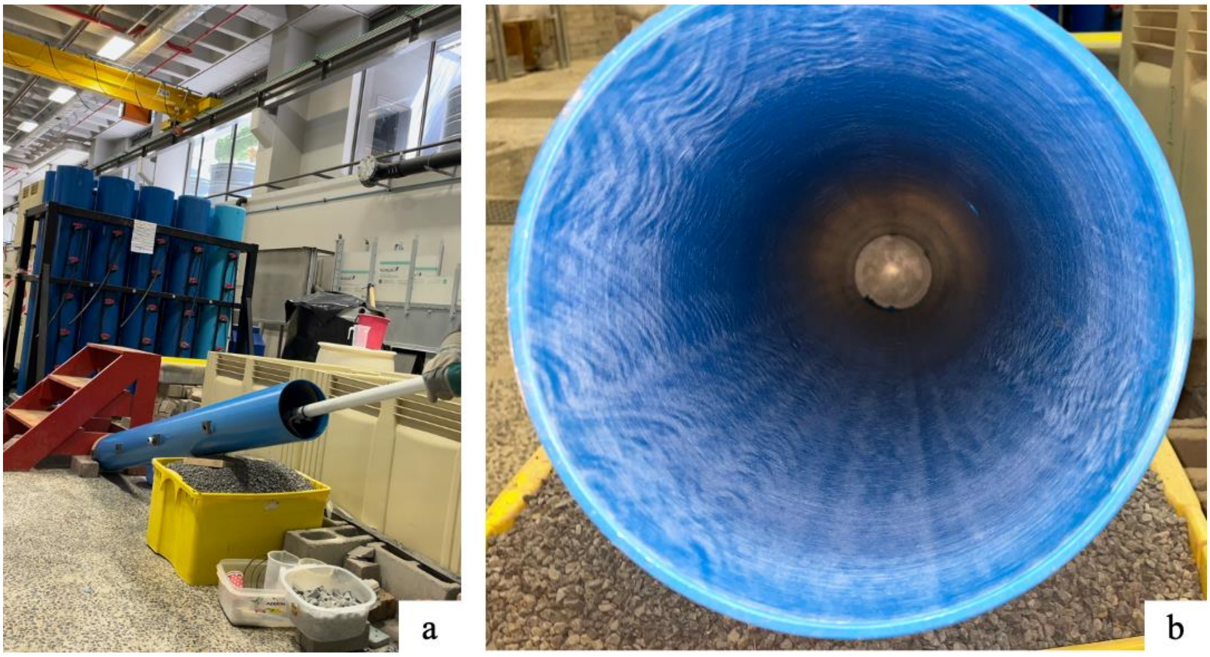

Figure 9-1: Abrading the column walls

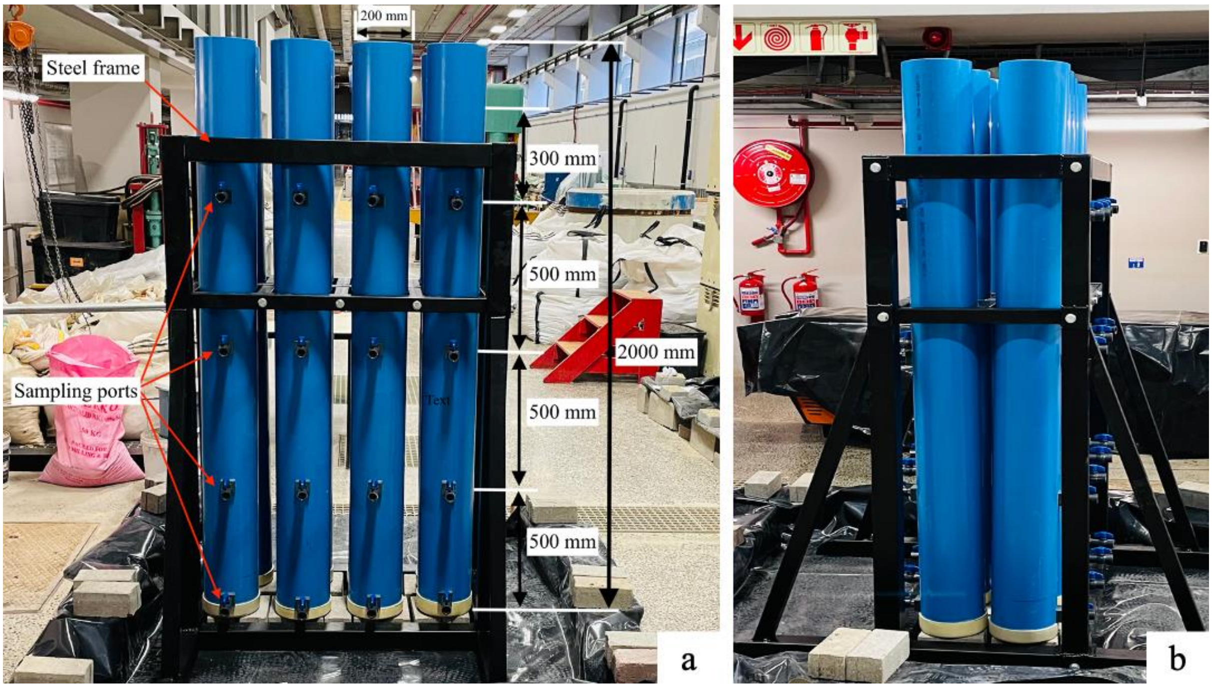

Figure 9-2: Experimental setup

9.1.2 Experimental setup

The columns were divided into four sets, each consisting of two columns. Three sets were filled with soil from the CFA, and the fourth was filled with clean silica sand, which served as the control medium. The soil from the School Pond was used to represent media over the CFA, while the clean silica sand represented a worst-case scenario which did not contain any soil constituents, such as clay and organic matter, which aid in water treatment. The experiment was set up as follows:

- Soil from the School Pond was collected during the monitoring well installation outing on the 18th of June 2021. Approximately 600 kg of soil was air-dried at the University of Cape Town Hydraulic Laboratory.
- The soil was then sieved using a 2.36 mm mesh sieve (No. 8) per the American Society for Testing and Materials (ASTM) D6913M-17 guidelines (ASTM, 2017). This procedure effectively eliminated all particles deemed larger than sand (Craig, 2004).
- The soil was mixed well in bags, and three samples were collected and sent to an external laboratory for analysis because of the unavailability of appropriate equipment and testing standards at the Civil Engineering Laboratory. The soil was then packed into six of the eight columns.

Soil analysis was performed to determine the soil's baseline concentrations of soluble and exchangeable nutrients and heavy metals. The properties analysed included five-fraction soil texture, composition, and total organic carbon. Tables 9-1 and 9-2 present the soil composition and extraction results, respectively. The heavy metal concentrations have a reported uncertainty of ± 5% of the derived value, while the phosphorus concentration values have an uncertainty of ±10%.

Table 9-1: Media properties and composition

Media	Classification	Sand (%)	Clay (%)	Silt (%)	CEC (cmol charge/kg)
Silica Sand	Coarse Sand	97.5 ± 3.1	0.60 ± 3.1	2.00 ± 2.0	1.60 ± 0.3
School Pond	Sand	92.2 ± 0.0	5.80 ± 0.0	2.00 ± 2.0	2.01 ± 0.2

**Cation exchange capacity (CEC) is a measure of the soil's ability to retain and exchange cations*

Table 9-2: Nutrient and heavy metal concentrations in soil extracts

Contaminant	Media	
	Silica Sand	School Pond
Total Nitrogen (mg/L)	0.20 ± 0.02	2.49 ± 0.3
Total Organic Carbon (mg/L)	0.94 ± 0.1	6.72 ± 0.3
Total Phosphorus (μg/L)	15 ± 1.0	13.67 ± 1.5
Chromium (μg/L)	< 0.2*	< 0.2*
Copper (μg/L)	<1*	< 1*
Lead (μg/L)	< 0.4*	< 0.4*
Nickel (μg/L)	0.43 ± 0.6	< 0.2*
Zinc (μg/L)	4.57 ± 0.8	2.3 ± 0.7

**Below detectable laboratory limits*

The soil from the School Pond was a disturbed sample that did not reflect the field conditions, requiring repacking to replicate field conditions in terms of porosity and bulk density. Consequently, an experimental procedure was devised to determine the most suitable approach to column packing. The objective was to attain bulk density and porosity values consistent with the established field-based values. The column was deemed adequately packed if a bulk density of 1200 – 1500 kg/m^3 was attained (Lewis & Sjöstrom, 2010; Fouché & Day, 2022). Various texts, such as Craig (2004), have elaborated on the procedure for determining bulk density and porosity.

The packing method was established using two trial short columns, 0.5 m long and 200 mm in diameter, fitted with an end cap and then weighed. The short columns were packed with sand from the pond and silica sand. The sand was poured using a 10 L bucket in increments and compacted using a concrete poker (Figure 9-3) that vibrated the sand for 90 s.

The columns were reweighed, and the sand bulk density, ρ_b, was calculated using Equation 9-1. The porosity was then determined by pouring water into the column until it was fully saturated, weighing the columns again, and then calculating porosity n using Equation 9-2.

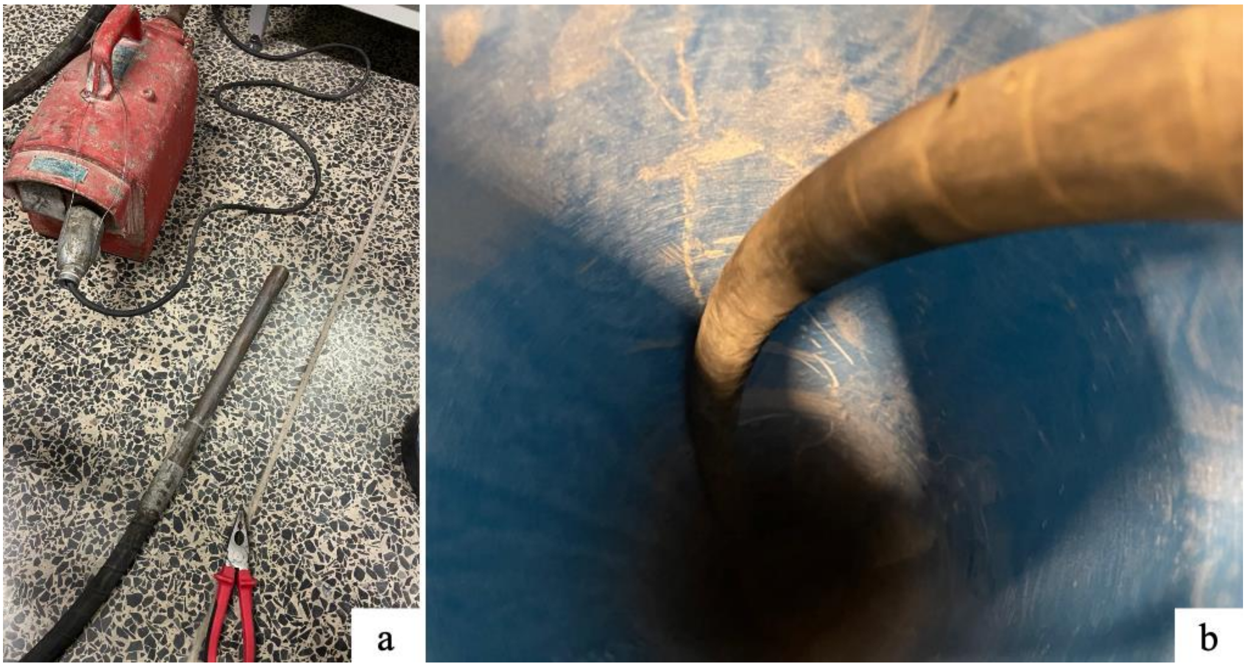

Figure 9-3: a) Concrete 'poker' vibrator, b) compacting the soil in the column

$$\rho_b = \frac{M_{solids}}{V_T} = \frac{m_{packed\ (dry)} - m_{empty}}{\pi r^2 h}$$ **Equation 9-1**

$$n = \frac{V_{voids}}{V_T} = \frac{(m_{saturated} - m_{packed\ (dry)}) * \rho_{water}}{\pi r^2 h}$$ **Equation 9-2**

Where m_{solids} is the dry mass of the solids, V_T is the volume of the solids in the packet column (including the volume of voids, V_v), $m_{packed\ (dry)}$ is the dry mass of the soil (including the mass of the column), m_{empty} is the mass of the empty column, $m_{saturated}$ is the mass of the saturated packed column, ρ_{water} is the density of water, r is the radius of the column, and h is the height of the soil in the column.

The eight experimental columns were then packed by pouring one bucket filled with sand and vibrating it for 90 s. The sand was then lightly scratched using a wire before the next load was applied to ensure its fusion with the next layer (Hatt *et al.*, 2008). The media was then packed incrementally up to 1.5 m above the bottom of the column.

9.1.3 Preparation of stormwater samples

The composition of the synthetic stormwater (SSW) used in this study was determined from datasets obtained from an analysis of stormwater in Cape Town (Chapter 5) and stormwater quality data found in the literature. The SSW was designed to represent a typical urban catchment in the CoCT (denoted as a low-concentration SSW in this study) and a more polluted catchment (denoted as a high-concentration SSW). A stormwater quality study determined the characteristics of a typical urban catchment (Chapter 5). The characteristics of a highly polluted catchment were determined by multiplying the typical values of a formal residential catchment in the CFA, determined in (Chapter 5), by five. This multiplier was calculated based on the magnitude difference between the mean concentrations and 95th percentile values of residential stormwater pollutant concentrations. The analysis revealed that the 95th percentile values were 3.4 times higher than the mean concentrations, leading to the decision to round off the multiplier to five to incorporate outliers and variance.

The SSW constituted of nutrients and heavy metal contaminants. The concentrations of organic constituents in SSW, namely Total Nitrogen (TN), Total Phosphorus (TP), and Total Organic Carbon (TOC), were obtained from the median concentrations in Cape Town's residential areas (Chapter 5).

The CoCT dataset did not include metal analysis therefore data from a study by Pitt *et al.* (2004), which examined stormwater quality from residential catchments in the USA, was used to establish 'typical' metal concentrations. The composition of the SSW was determined using the mean concentrations of several heavy metals, namely chromium (Cr), copper (Cu), lead (Pb), nickel (Ni), and zinc (Zn). Heavy metal concentrations are primarily attributed to exhaust fumes, fuels, oils, and tire degradation (Sakson *et al.*, 2018; Robertson *et al.*, 2019). The USA dataset has higher values than South African residential areas due to increased vehicle usage in the USA (World Bank, 2014). As a result, in this study the SSW data based on the USA dataset can be considered a worst-case scenario. Table 9-3 summarises the target concentrations for each contaminant in SSW.

Table 9-4 lists the column identifiers and SSW concentrations. As part of a parallel study investigating the impact of carbon on denitrification, some batches of SSW were supplemented with organic carbon (Table 9-4).

Table 9-3: Target concentrations for SSW constituents

	TOC (mg/L)	TN (mg/L)	TP (mg/L)	Cr (µg/L)	Cu (µg/L)	Pb (µg/L)	Ni (µg/L)	Zn (µg/L)
Typical Formal Residential	13.53	3.90	0.22	2.00	14.0	2.00	2.00	84.0
Highly Polluted (x5 formal residential)	67.65	19.5	1.10	10.0	70.0	10.0	10.0	420.0

*As and Fe were not added to the SSW but were detected in the samples.

Table 9-4: Description of the columns and concentration of SSW

Columns	A1	A2	B1	B2	C1	C2	D1	D2
SSW	Low_c	Low_c	$High_c$	$High_c$	High	High	$High_c$	$High_c$
Column Media	SP	SP	SP	SP	SP	SP	SS	SS

*Subscripts indicate the presence of carbon in the SSW where c means 'with carbon', and a blank entry means no added carbon

Key: Low = Formal residential concentration, High = 5x Formal residential concentration, SS = Silica Sand Media, SP = School Pond Media

9.1.4 Column loading, operation, and sampling

SSW was prepared using stock chemicals with the desired SSW constituents in 20 L batches for each loading cycle and column pair. The SSW was prepared and applied (loaded) as follows:

- The dry chemical compounds were weighed on a mass balance to prepare stock solutions – pre-mixed chemical solutions with constituents of known concentrations.
- The stock solutions were mixed with deionised water in a 100 mL volumetric flask (Figure 9-4a) and then added to a container of deionised water. The container was shaken to mix SSW.
- The concentration of the constituents in the SSW was verified by analysing the 500 mL sample drawn from the mixed SSW in the laboratory. Four batches of synthetic stormwater were prepared for each cycle to load eight columns.

- The SSW was carefully poured into the columns using a sprinkling can – to ensure the soil surface was not scoured (Figures 9-4b and 9-4c). 9 L of SSW were required to achieve a ponding depth of approximately 300 mm over the soil surface.

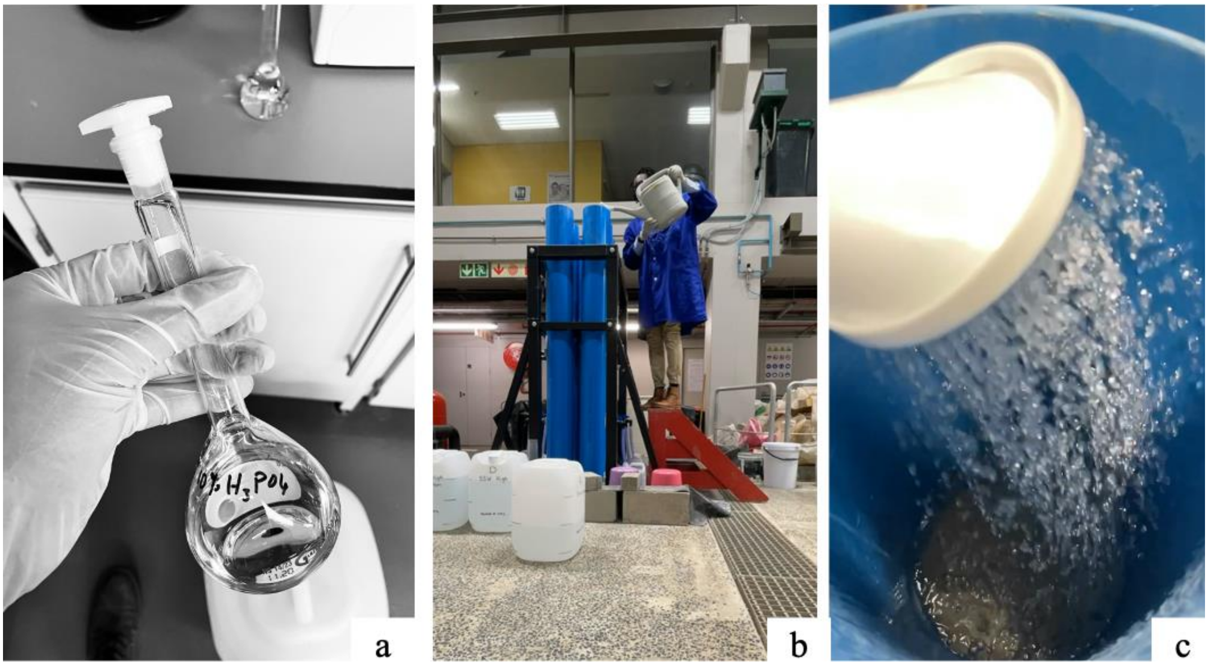

Figure 9-4: a) Preparing the SSW, b&c) loading the columns

The loading cycle for the column study was determined considering the following assumptions: (1) similar retrofits to the School Pond would be implemented in the CFA using infiltration trenches allowed to pond up to a depth of 300 mm above the swale surface; (2) most retrofitted ponds would be in residential areas with similar land use to the Rondevlei Park catchment where the School Pond is situated and thus have similar runoff characteristics, and (3) the infiltration swale would be allowed to dry out (dry detention) between rainfall events. Thus, the loading cycle for the column study was based on a scaled-down historical rainfall frequency derived from rainfall data collected at the Wolfgat station in Mitchells Plain. This frequency was calculated using rainfall data from the traditional Cape Town rainy season, which spans April to September (Mahlalela *et al.*, 2019; Roffe *et al.*, 2022; Cash *et al.*, 2023). Rainfall days were characterised as days with recorded rainfall exceeding 3 mm, the minimum depth required to generate runoff at the School Pond. Analysis of five rainfall seasons (2013, 2014, 2016-2018) revealed that out of the 615 days examined, 138 experienced rainfall exceeding 3 mm. This indicates three days in a 14-day cycle, with rainfall depths equal to or exceeding 3 mm. The loading sequence of the experiment was then defined as three consecutive rainfall days (loading days), followed by 11 dry days, culminating in a 14-day (two-week) cycle.

The experiment comprised nine cycles, spanning 18 weeks, with three consecutive 'wet' days (Days 1, 2, and 3) and 11 'drying' days in each cycle. During the 'wet' days, the columns were loaded with SSW to enable infiltration. Samples were collected from the sampling ports on days two, three, and four of each cycle.

The loading procedure aimed to simulate a rising water table, such as the fluctuations observed in the CFA, particularly the School Pond, which ranged from 0.6 to 1.2 m below the infiltration swale. This approach allows for a controlled representation of the natural conditions in the field. The loading configuration is shown in Figure 9-5.

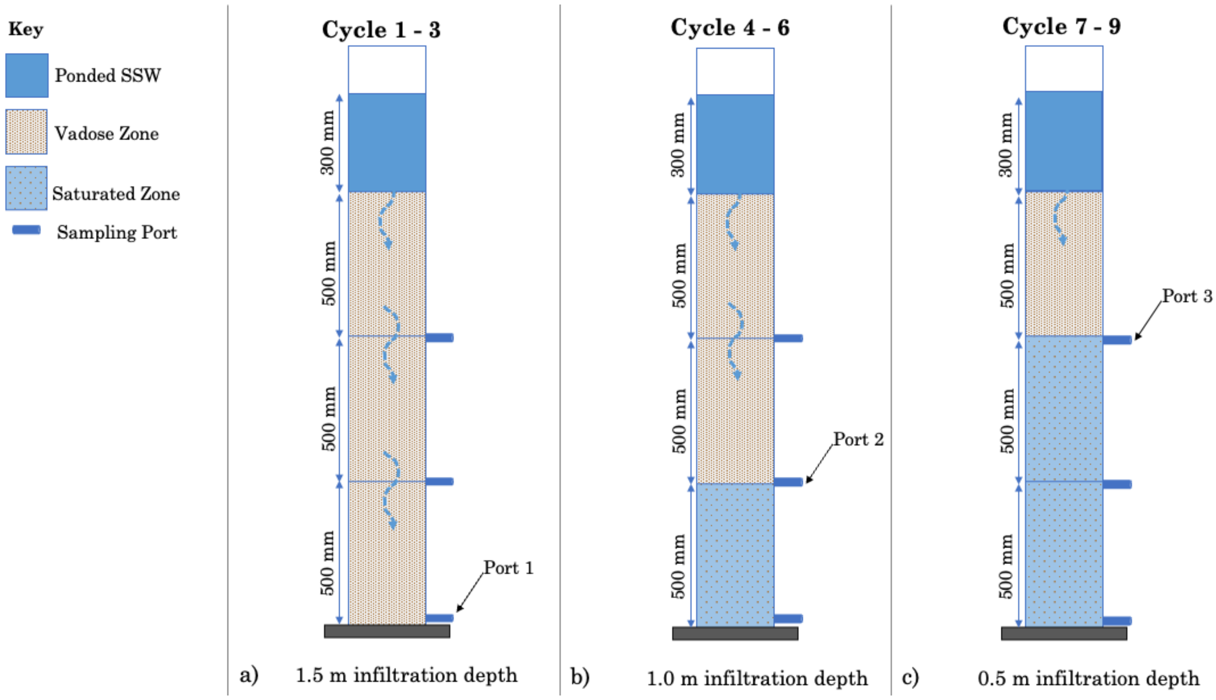

Figure 9-5: Column loading configuration

The loading and sampling procedures were as follows:

- In Cycles 1 – 3, SSW effluent was sampled from Port 3, which was positioned 1.5 m below the free surface (Figure 9-5a).
- This process was repeated for Cycles 4 – 6, with the SSW effluent sampled from Port 2 to 1 m below the free surface. During these cycles, a 0.5 m saturated zone developed below port 2 (Figure 9-5b). Stormwater effluent samples were also collected from the saturated zones of Port 1 on day 4 of each cycle and analysed.

- Finally, the columns were loaded in cycles 7 – 9, and the sampling location shifted to Port 1, which was situated 0.5 m below the free surface. A 1 m saturated zone developed under Port 1 (Figure 9-5c). The SSW effluent was sampled from the saturated zones of ports 2 and 3 on day 4 of each cycle.

9.1.5 Analysis of stormwater effluent

The infiltrated SSW effluent was sampled from the ports (Figure 9-6a) and first tested using handheld probes to analyse electrical conductivity (EC), pH, redox potential (ORP), and temperature.

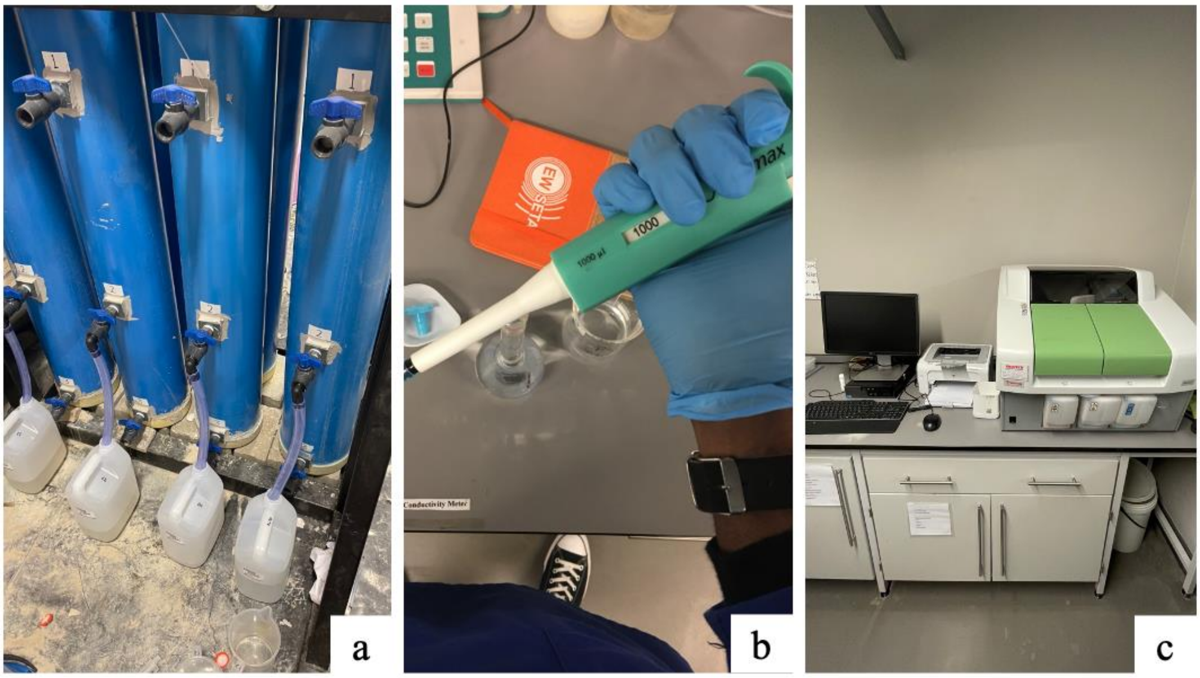

Figure 9-6: Stormwater analysis procedures and equipment

The effluent was then filtered using 0.45 µm syringe chromatography filters. The dissolved metals and TP in the filtered effluent were preserved in dilute nitric acid (HNO3) and stored overnight in a freezer. The samples were subsequently analysed at two external laboratories, the UCT Chemical Engineering Laboratory (CEBER) and Element Materials Technology, as the UCT Civil Engineering Water Quality Laboratory lacked the necessary equipment and standards for this analysis.

A Themo Fisher automated discrete analyser (Figure 9-6c) was used to analyse the nutrients (NH_4, NO_3, NO_2, and PO_4) at the UCT water quality laboratory. Total organic carbon (TOC) and TN were analysed at the Chemical Engineering

Department using an Analytikjena multi-N/C TOC analyser. The dissolved metals and TP were analysed at the external laboratory, where 64 samples per cycle were couriered to the external laboratory for dissolved metal analysis using mass spectrometry-based a modified USEPA 200.8/6020A and BS EN ISO 17294-2 2016 method. The results from the external laboratory were entered into an Excel sheet for further analysis.

9.2 Column treatment efficiency

The treatment performance of the columns and retrofitted ponds was evaluated by determining the removal efficiency of each column pair at different infiltration depths. The media in the columns facilitates the removal of contaminants through filtration, sorption, ionic adhesion, precipitation, and surface complexation – and in the case of nutrients bioconversion (Hatt *et al.*, 2008).

The removal efficiency was determined by considering the fraction of the pollutant load removed at the end of each loading cycle, that is, the difference between the inflow and outflow concentrations expressed as a percentage of the SSW influent concentration (Equation 9-3).

$$\mathrm{Removal}_{\mathrm{efficiency}} = \frac{C_{\mathrm{reduction}}}{C_{\mathrm{influent}}} = \frac{C_{\mathrm{inf}} - C_{\mathrm{eff}}}{C_{\mathrm{inf}}} * 100$$ **Equation 9-3**

C_{inf} and C_{eff} are the mean influent and effluent pollutant concentrations, respectively, for three consecutive cycles. The central assumption was that the media matrix was not saturated (adsorption capacity was not reached) and had the potential to facilitate the bioconversion of nutrients and sorb TP and metals.

9.3 Column study results

Column pairs were used to ensure the replicability of the results, which were verified through statistical tests that assessed whether the mean effluent concentrations from both columns were statistically similar. The effluent concentrations of the column pair were compared across the three vadose zone depths. When the effluent concentrations drawn from the columns were statistically similar, the respective effluent means were combined to determine the joint performance of the column pair. For instance, if the concentrations from columns A1 and A2 were statistically identical, the data for each vadose depth were combined to represent column pair A.

The treatment performance of each column pair was then compared with that of other column pairs for all contaminants. This comparison aimed to establish whether there were any statistically significant differences in treatment efficiency among column pairs A, B, C, and D and to determine the influence of soil media type on the treatment efficiency for all contaminants.

A single-factor analysis of variance (ANOVA) pairwise comparison was used to assess the relationship between pairs (or groups) of mean concentrations (Nordstokke & Stelnicki, 2014), with the p-value as the test statistic. The Null hypothesis (H_0) in the ANOVA tests for the column pairs was that there was no difference in the column means, signifying similar treatment efficiency between a pair of columns. If the p-value is > 0.05, the null hypothesis is not rejected, suggesting no statistically significant difference in the column means. Conversely, a p-value < 0.05 indicates a significant difference between the means of the column pairs and signifies that the column pairs had different treatment efficiencies and were not replicable.

The stormwater harvested from retrofitted ponds will be directly discharged into the CFA. Thus, the effluent must meet specific water quality requirements. However, as discussed earlier, South Africa lacks explicit water quality parameter concentrations for groundwater discharges. However, the Department of Water and Sanitation (DWS), formerly the Department of Water Affairs, stipulates that the water introduced into an aquifer should match or surpass the quality of the receiving aquifer (Department of Water Affairs (DWA), 2010).

Data from a CoCT report that detailed the groundwater quality from 140 groundwater wells within the CFA were examined to obtain the baseline groundwater quality in the CFA, specifically the mean characteristics of the groundwater in the aquifer (Table 9-5). This baseline was considered the threshold for satisfying DWS requirements (CoCT, 2021). Furthermore, the quality of the column effluent was evaluated based on the existing domestic use guidelines, aquatic ecosystem guidelines, and limits for wastewater discharge into water resources (DWAF, 1996d,a; Department of Water Affairs, 2013). Domestic water use guidelines were considered because of the potential risk of residents using groundwater from CFA for domestic purposes without pre-treatment. Additionally, the presence of rivers and streams interconnected with the CFA (Seyler *et al.*, 2016; Gxokwe, 2017) necessitated the consideration of aquatic ecosystem guidelines and wastewater discharge limits.

Table 9-5: Summary of target water quality limits

Contaminant	Domestic Water use	Aquatic Ecosystems	Wastewater discharge general limits	CFA Baseline
TN (mg/L)	6	0.5	6	1.5
TOC (mg/L)	5	–	–	20.9
TP (µg/L)	–	5	10000	10500
Cr (µg/L)	50	7	50	2.7
Cu (µg/L)	1000	1.5	10	3.0
Pb (µg/L)	10	1	10	0.5
Ni (µg/L)	70*	–	–	1.8
Zn (µg/L)	3000	3.6	100	5.6

9.3.1 Treatment performance for organics

ANOVA was performed for the organic contaminants assessed in this study (TN, TP, and TOC), and the pairwise p-values for all eight columns are presented in Table G-1 (Appendix G). All the calculated p-values were greater than 0.05, indicating that the means of the column pairs were not statistically different at the 95% confidence level. This result indicated that the column pairs produced statistically similar means at all vadose depths; thus, the data could be combined and analysed.

The following section discusses the treatment efficiencies of the columns for TN, TP, and TOC. Treatment efficiencies at 1.5, 1.0 and 0.5 m vadose depth are presented in Table 9-6. Appendix G.2 present the mean influent and effluent concentrations for all organics across all columns, vadose depths, and saturated cycles.

Table 9-6: Summary of treatment efficiencies for organic contaminants

Contaminant	Column Pair	Removal Efficiency (%)				
		1.5 m vadose zone	1.0 m vadose zone	0.5 m vadose zone	0.5 m saturated zone	1.0 m saturated zone
TN	A	-24%	-80%	16%	58%	45%
	B	48%	38%	23%	89%	86%
	C	17%	0%	-20%	88%	84%
	D	35%	27%	24%	21%	46%
TP	A	66%	91%	92%	93%	93%
	B	94%	91%	98%	99%	99%
	C	97%	81%	99%	98%	98%
	D	88%	62%	23%	78%	80%
TOC	A	-42%	79%	87%	32%	-25%
	B	62%	96%	85%	88%	85%
	D	85%	92%	78%	96%	93%

Key: A – CFA media dosed with typical SSW containing added carbon; B – CFA media dosed with highly contaminated SSW containing added carbon; C – CFA media dosed with highly contaminated SSW without added carbon; D – Silica sand dosed with highly contaminated SSW containing added carbon

Table 9-7 presents the treatment performance for the three organic contaminants analysed in this section and compares the mean effluent concentrations with the CFA baselines.

The columns show varying levels of treatment. Column A, which contained media from the CFA, exhibited leaching in the 1.5m vadose zone for TN and TOC. Column A also leached out TN at the 1.0 m vadose zone. The leaching is attributed to the soil's pre-existing TN and TOC loads from the CFA being released in a 'first wash' event. This first wash phenomenon was observed in all columns, but its effect is believed to be more significant in Column A – because of the lower influent concentration. The treatment efficiency is a function of influent and effluent concentrations (Equation 9-3), and the first wash is more influential when the effluent has a lower concentration.

Table 9-7: Treatment performance for organics in relation to CFA baseline

Contaminant	Column Pair	CFA Baseline	Mean SSW Effluent				
			1.5 m vadose zone	1.0 m vadose zone	0.5 m vadose zone	Saturated zone (cycle 4-6)	Saturated zone (cycle 7-9)
TN (mg/L)	A	1.50	6.59	4.61	4.09	2.15	2.65
	B		12.7	14.1	18.7	2.68	3.32
	C		21.1	25.5	30.9	3.03	4.21
	D		15.7	17.2	18.2	19.0	12.9
TP (µg/L)	A	10500	65.7	17.4	16.3	14.8	14.0
	B		61.9	97.7	19.1	16.3	16.8
	C		37.8	212	14.6	20.7	18.0
	D		128	440	898	248	230
TOC (mg/L)	A	20.9	19.1	2.54	1.62	8.74	15.2
	B		26.0	2.40	9.48	8.07	9.32
	C		12.5	2.62	1.54	5.55	4.92
	D		10.1	5.31	14.4	2.65	4.41

Key: A – CFA media dosed with typical SSW that contains added carbon; B – CFA media dosed with highly contaminated SSW that contains added carbon; C – CFA media dosed with highly contaminated SSW without carbon added; D – Silica sand dosed with highly contaminated SSW that contains added carbon

Red fill – Mean effluent value above CFA baseline.

Green fill – Mean effluent value below CFA baseline.

The treatment efficiency for TN ranged from -24% to 89% effluent from all columns; however, all columns had mean effluent concentrations that exceeded the CFA baseline (Table 9-7). Furthermore, the post-hoc ANOVA showed no statistically significant difference between the effluents from the CFA media and silica sand columns ($p = 0.942$). These findings indicated poor TN treatment at all depths and columns in the vadose zone. This presents a risk of groundwater contamination if poor-quality stormwater is introduced into the aquifers. However, TN is degraded through nitrification-denitrification processes carried out by nitrifying and heterotrophic denitrifying bacteria (Saeed & Sun, 2012). Denitrification occurs under conditions that promote optimal microbial activity, such as anaerobic zones, DO concentration, presence of organic carbon, hydraulic retention time, and pH (Vymazal, 2007; Liou & Madsen, 2008; Li *et al.*, 2017). The effect of denitrification was observed as a reduction in TN in the saturated zone, which was attributed to the more favourable conditions for denitrification (Wu *et al.*, 2017; Rahman *et al.*, 2019). Column pairs A, B, and C reduced TN to concentrations below the SA drinking water guidelines and SANS-241-1:2015 limits. Column pair D, which contained silica sand, had the worst treatment performance, attributed to the 'clean' Silica sand containing less organic matter required for denitrification. Regardless, Column D reduces the TN concentration to a limited extent.

There was improved treatment efficiency for TP compared to TN, ranging from 24% to 99%. The silica sand column had a statistically different and lower treatment efficiency for SSW with high concentrations of TP compared to the School Pond soil columns with the same SSW ($p = 0.003$). Column pair D had a treatment efficiency of at least 20% less than the other high-concentration SSW columns with media from the School Pond. The CFA baseline for TP (10500 µg/L) was higher than the aquatic ecosystem limit of 5 µg/L, suggesting TP contamination in the CFA. The mean effluent concentrations for all columns and vadose depths were below the CFA baseline (Table 9-7), although the effluent concentrations also exceeded the aquatic guidelines. This finding implies that in the absence of groundwater guidelines with explicit permissible values, the effluent derived from the infiltration of typical and highly polluted stormwater is permissible, as it is below the CFA baseline. There was an overall improvement in the treatment efficiency of the columns in the saturated zone; however, this improvement was not statistically significant, with p-values of 0.103, 0.277, 0.212, and 0.062 for columns A, B, C, and D, respectively.

The columns displayed varied treatment performance for TOC, with treatment efficiencies ranging from -42% (leaching) to 96%. The CFA baseline of 20.9 mg/L exceeds the South African domestic water-use target limit of 5 mg/L and is considered to pose health risks and significant aesthetic effects for taste, odour, and colour (DWAF, 1996a). This indicates that CFA groundwater is unsuitable for direct domestic use without pre-treatment. However, the column experiments in this study revealed that the effluent was below the CFA baseline for all columns except for Column B, which contained media from the CFA and was dosed with highly contaminated SSW as the 1.5 m vadose zone. However, Silica sand reduced the effluent concentration to levels below the CFA baseline in the 1.5 m vadose zone, indicating that the Column B result might have been caused by TOC leaching out due to the first flash effect in Cycles 1-3. The mean effluent concentrations for the saturated cycles (Cycles 4-9) were inconclusive, with Columns pairs A and B showing an increase in the effluent concentrations, while Column pair D showed a decrease in effluent concentrations and an increase in treatment efficiency. The observed increases in A and B might result from microbes dying off or increased organic carbon leaching, whereas column D initially had minimal soil organic carbon.

9.3.2 Treatment performance for heavy metals

As with organic contaminants, ANOVA tests were performed, and the pairwise p-values are presented in Table G-2 (Appendix G). All the computed p-values were > 0.05, indicating that the column pairs produced statistically similar means at all vadose depths. The data were analysed to assess the overall treatment efficiency at 1.5, 1.0 and 0.5 m vadose depths (Table 9-6).

The following section discusses the treatment performance of column pairs in reducing heavy metal contaminant concentrations. Appendices G.2 and G.3 present the mean influent and effluent concentrations of all heavy metals assessed across all columns, vadose depths, and saturated cycles.

Chromium (Cr) treatment efficiency ranged between 38% and 99%, indicating a reduction in Cr concentration across all four column pairs at all vadose depths. The column pairs (B: CFA media, C: CFA media), and D: silica sand), which have higher concentrations of SSW, have higher treatment efficiencies. This finding is attributed to sorptive concentration playing a significant role in the accumulation of sorbates in soils – considered as treatment efficiency in this study (Thompson & Goyne, 2012; Brusseau & Chorover, 2019). The relationship is linear at low sorptive concentrations (by Le Chatelier's principle of chemical equilibrium) but breaks down at higher sorptive concentrations once the sorbent (soil) capacity is reached (Huang *et al.*, 2003). Furthermore, the low concentrations of Cr could be close to the irreducible pollutant concentration limit, as discussed by Schueler (1996). There was no statistically significant difference in the treatment efficiencies of the columns with silica sand and those with CFA media ($p = 3.89E\text{-}08$). Similar efficiencies were unforeseen, as it was anticipated that silica sand would have lower treatment efficiencies due to a lower percentage of humic and layer silicate clays (Table 9-1), which play a significant role in the sorption processes. All four column pairs had mean effluent concentrations below the CFA baseline of 2.7 µg/L (Table 9-9), lower than the wastewater discharge limits, South African domestic water-use limits, and stringent SA aquatic ecosystem limits (Table 9-5). This finding indicates that, according to current requirements, the effluent from the columns can be discharged into the CFA without further treatment.

Table 9-8: Summary of treatment efficiencies for heavy metals

Contaminant	Column Pair	Removal Efficiency (%)				
		1.5 m vadose zone	1.0 m vadose zone	0.5 m vadose zone	0.5 m saturated zone	1.0 m saturated zone
Cr	A	38%	85%	79%	91%	87%
	B	87%	98%	97%	98%	99%
	C	89%	94%	95%	98%	98%
	D	88%	94%	90%	94%	97%
Cu	A	91%	93%	92%	96%	89%
	B	98%	98%	98%	99%	98%
	C	99%	91%	98%	99%	98%
	D	59%	79%	63%	91%	92%
Pb	A	54%	90%	87%	92%	90%
	B	93%	97%	94%	97%	96%
	C	93%	97%	97%	98%	97%
	D	97%	96%	94%	97%	96%
Ni	A	25%	83%	93%	83%	86%
	B	81%	94%	96%	96%	96%
	C	87%	93%	98%	96%	96%
	D	86%	88%	45%	86%	84%
Zi	A	91%	87%	76%	91%	75%
	B	98%	98%	96%	98%	95%
	C	98%	97%	97%	97%	91%
	D	95%	96%	76%	93%	86%

Key: A – CFA media dosed with typical SSW that contains added carbon; B – CFA media dosed with highly contaminated SSW that contains added carbon; C – CFA media dosed with highly contaminated SSW without carbon added; D – Silica sand dosed with highly contaminated SSW that contains added carbon.

Table 9-9: Treatment performance for heavy metals in relation to CFA baseline

Contaminant	Column Pair	CFA Baseline	Mean SSW Effluent				
			1.5 m vadose zone	1.0 m vadose zone	0.5 m vadose zone	Saturated zone (cycle 4-6)	Saturated zone (cycle 7-9)
Cr (µg/L)	A	2.70	1.11	0.25	0.37	0.15	0.22
	B		1.02	0.17	0.23	0.13	0.12
	C		0.83	0.46	0.35	0.15	0.15
	D		0.92	0.49	0.82	0.45	0.22
Cu (µg/L)	A	3.00	1.3	0.97	1.06	0.50	1.50
	B		1.1	1.25	1.03	0.50	1.17
	C		0.6	5.72	0.94	0.50	1.00
	D		27	15.4	23.9	6.00	5.83
Pb (µg/L)	A	0.50	1.17	0.20	0.25	0.20	0.20
	B		0.49	0.20	0.23	0.20	0.20
	C		0.61	0.20	0.20	0.20	0.20
	D		0.26	0.20	0.2	0.20	0.20
Ni (µg/L)	A	1.80	1.57	0.33	0.13	0.35	0.25
	B		1.78	0.52	0.35	0.37	0.33
	C		1.21	0.66	0.14	0.37	0.37
	D		1.32	1.07	5.13	1.25	1.43
Zn (µg/L)	A	5.6	8.27	12.0	24.27	8.50	22.0
	B		8.67	10.5	21.58	7.95	23.45
	C		7.84	12.8	15.71	12.1	44.28
	D		20.9	18.4	112.7	34.0	65.97

Key: A – CFA media dosed with typical SSW that contains added carbon; B – CFA media dosed with highly contaminated SSW that contains added carbon; C – CFA media dosed with highly contaminated SSW without carbon added; D – Silica sand dosed with highly contaminated SSW that contains added carbon. Red fill – Mean effluent value above CFA baseline; Green fill – Mean effluent value below CFA baseline

The copper (Cu) removal rate ranged from 59% to 99%. There was a general improvement in the treatment efficiency in the saturated zones for all the column pairs. The mean effluent concentrations for Columns A, B, and C from all vadose zone depths were below the CFA baseline of 3 µg/L (Table 9-9), which is lower than the wastewater discharge limits and the South African domestic water-use limit but higher than the stringent SA aquatic ecosystem limits (Table 9-5). However, Column D, which contained silica sand, had mean effluent concentrations above the CFA baseline, possibly due to its soil composition containing fewer silicate clays and soil organic matter. This indicates that for all vadose zone depths, clean silica sand failed to produce effluent that could be directly discharged into the CFA. However, the effluent concentration was well within the domestic water use limit of 1000 µg/L.

54–98% treatment efficiencies were observed for the lead (Pb) contaminant. The CFA baseline for Pb (0.5 µg/L) is lower than the wastewater discharge limits, South African domestic water-use limits, and stringent SA aquatic ecosystem limits (Table 9-5). The effluent concentrations from the column pairs were mainly below the CFA baseline (Table 9-9). The exceptions were Column A, containing CFA media and dosed with typical SSW with low Pb concentrations in the 1.5 m vadose zone. Column A had significantly less treatment than the other columns ($p = 0.0417$), possibly due to leaching or the effect of the low sorptive concentration on the accumulation of sorbates in the soil, as with Cr. The column pairs packed with silica sand (D) had similar treatment to those packed with media from the CFA at all vadose zone depths ($p = 0.0655$). These findings also indicate that, according to the current requirements, the effluent from the columns can be discharged into the CFA without further treatment of highly contaminated SSW.

The treatment efficiencies for nickel (Ni) ranged from 25% to 98%. Column A at 1.5 m vadose zone had the lowest treatment efficiency, possibly due to the low concentration SSW being close to the irreducible pollutant concentration limit, as Schueler (1996) discussed. The mean effluent concentrations from all but one column were below the 1.8 µg/L CFA baseline. The exception was the column pair that contained clean silica sand at a 1.5 m vadose depth. Furthermore, the column pairs packed with silica sand (Column D) provided less treatment than those packed with media from the CFA at all vadose zone depths (p-value = 4.95E-10), and the effluent concentration was at least 14 times higher than that in columns A, B, and C. These results indicate that columns with silica sand alone perform poorly at reducing contaminants at 1) shallow vadose depths and 2) in comparison to columns with CFA media. However, the results from this assessment also indicate that, like Pb, effluent from the columns with CFA media can be discharged into CFA without further treatment, according to the current requirements.

The zinc (Zn) removal efficiency ranged from 75% to 99%. The treatment efficiencies of column pairs A, B, and C decreased slightly at a vadose depth of 0.5 m, but the change in the mean effluent concentrations at the 0.5 m vadose zone depths was not statistically significant. The effluent concentrations of all the column pairs at all depths were above the CFA baseline of 5.6 µg/L (Table 9-9), which was below wastewater discharge limits, the South African domestic water-use limits but higher than stringent SA aquatic ecosystems limits (Table 9-5). The Zn results indicated poor Zn treatment in the vadose zone at all depths and columns. This presents a risk of groundwater contamination if poor-quality stormwater is introduced into the CFA.

9.4 Discussion

The columns generally improved the SSW quality, as indicated by the reduced effluent concentrations of most contaminants. The column pairs exhibited statistically similar results for all contaminants and at all vadose zone depths, with one exception (zinc treatment efficiency in Column D at 0.5 m vadose zone depth). Dissolved carbon in the SSW influent was only significant in reducing TN concentration in the effluent.

The columns produced effluents with mean concentrations below the CFA baselines for two organics assessed (TP and TOC). While the columns produced largely reduced effluent concentrations from TN with some observed leaching, the overall treatment efficiency of the columns at all depths and for all produced effluents was above the CFA baseline. However, there was a significant improvement in the treatment efficiency of all contaminants in the saturated zone. Practically, this indicates the possibility of further reducing contaminant concentrations within the aquifer. The efficiency of silica sand is encouraging, as media in most of the ponds in the CFA are expected to have more fine sediments, inorganic colloids, and humic and particulate materials (see Fouché & Day (2022) and Mavundla (2022)), which would improve the treatment efficiency of the ponds in the infiltration zone (even at 0.5 m – as observed from the findings Columns A, B and C). However, the mean effluent concentration of TN remained above the CFA baseline. This result indicates a risk when using stormwater for MAR in CFA, as it may introduce poor-quality water. Nevertheless, the columns produced effluent below the domestic water use limits for TN.

The results of the heavy metal analysis revealed that the columns produced effluent concentrations below the CFA baseline for all metals except Zn (all columns and vadose depths) and Cu (for the silica sand column pair at all vadose zone depths). The treatment efficiencies of the heavy metals in the vadose zones (Cr, Cu, Pb, Ni, and Zn) ranged from 33% to 98%. Furthermore, the mean effluent concentrations of Zn and Cu were below domestic water-use standards. These findings indicate that the effluent produced from infiltrating typical and highly polluted stormwater using CFA media satisfies the available recommendations (effluent should not be more contaminated than the aquifer baseline) for four of the five contaminants analysed.

Most contaminants investigated in this section were removed by filtration and sorption processes, and their efficiencies were correlated to media characteristics. However, silica sand has characteristics that are not favourable for these processes, such as fewer inorganic colloids, organic compounds such as humic and particulate materials, more porosity, and potentially fewer microbes.

A correlation between the vadose zone depth and treatment efficiency was observed. Thus, the worst-case scenario analysis can be conducted by considering the treatment efficiency of silica sand in the 0.5 m vadose zone with stormwater, which is five times more polluted than stormwater from a typical residential area in Cape Town. For the 0.5 m vadose zone depth, the silica sand columns reduced the concentrations of all contaminants; however, for TN, Cu, and Zn, the silica sand columns could not treat the SSW to concentrations below the CFA baseline. Poor treatment implies that infiltrating poor-quality stormwater might contaminate shallow aquifers, a concern also raised by Gxokwe (2017) and Mauck (2017) for the CFA.

9.5 Chapter summary

The primary objective of this chapter was to evaluate the treatment capabilities of media, specifically those derived from the CFA and 'clean' silica sand. This study aimed to determine the efficacy of these media in reducing pollutant loads in infiltrated stormwater before mixing with groundwater. Furthermore, this study investigated the treatment performance in varying vadose zone depths within retrofitted detention ponds overlying the CFA. The findings indicate that media from CFA and silica sand can reduce the contaminant concentration of the stormwater to varying extents, ranging from -24% to 99%, and produce acceptable effluent for six of the eight contaminants assessed.

10. Additional benefits of retrofitted detention ponds

The previous chapters assessed the practical implementation of retrofits in Cape Town (Chapter 6), potential MAR volumes under different scenarios (Chapter 8), and the treatment capabilities of retrofitted ponds (Chapter 9). This chapter examines the added advantages that retrofitted basins could bring to the residents of Cape Town.

10.1 Contextual background

The 2015 to 2018 drought showed the susceptibility of Cape Town and numerous other South African cities to water scarcity. Consequently, there is now a revived consensus within the City of Cape Town leadership ('the City') that conventional water management practices are no longer sustainable. This has propelled the exploration and implementation of a range of alternative and diversified water sources (CoCT, 2018). One way of ensuring improved water management is through the Water Sensitive Cities (WSCs) concept that calls for cities to, *inter alia*, look within to provide and protect their local water resources instead of the current conventional model that often sees cities importing water from other catchments. The City is now committed to becoming a WSC by 2040 (City of Cape Town, 2019; Sanya & Shefer, 2021). Wong & Brown (2009), proposed three pillars of practice that characterise WSCs:

1. Water supply catchments that provide diverse centralised, decentralised water sources.
2. Providers of ecosystem services for built and natural environments in urban centres.
3. Hubs of water-sensitive communities that promote water-sensitive decision-making and behaviours within and around their communities.

The WSC approach can be achieved by applying various blue-green interventions such as SuDS, a key component of WSCs. SuDS align with key WS design practices (Oral *et al.*, 2020; Ramírez-Agudelo *et al.*, 2020). SuDS are 'drainage systems considered environmentally beneficial, causing minimal or no long-term detrimental impact' (Woods-Ballard *et al.*, 2015a). SuDS can achieve four main benefits, namely: improved water quality – to prevent pollution, controlling the quantity of runoff – for the management of flood risk and protecting water supply, provision of amenities for communities, and promoting and sustaining biodiversity.

In this study, the pillars of practice (PoP) for water-sensitive cities (WSCs), as proposed by Wong & Brown (2009), were recontextualised by superimposing them with the benefits of SuDS presented in Woods-Ballard *et al*. (2015).

The first PoP views WSCs as catchments that provide centralised and decentralised water sources. This pillar encompasses two SuDS benefits, namely improved water quality and flood risk management. The rephrased theme proposes that 'WSCs should be viewed as catchments that use SuDS to improve water quality and supplement water supply'. The second PoP proposes that WSCs should provide ecosystem services for built and natural environments in urban centres, which aligns with the SuDS benefits of providing community amenities and sustaining biodiversity. This PoP is rephrased as 'WSCs should provide ecosystem services that promote and sustain biodiversity and community amenities in urban centres.' Lastly, the third PoP emphasises the need to create and sustain water-sensitive communities within a WSC. The second and third rephrased pillars are merged into one theme that proposes that 'WSCs should provide urban ecosystems for biodiversity and restorative ecology that promote and support water-sensitive communities.'

The recontextualised PoP are consolidated into two narratives which consider a WSC as:

1. A catchment which utilises blue-green infrastructure – such as SuDS – to improve water quality and supplement water supply. Shortened to 'The city as a catchment.'
2. A provider of urban services for biodiversity and restorative ecology that promote and support water-sensitive communities. Shortened to 'The city as a provider of urban services.'

The recontextualized PoPs were then used to assess the additional benefits from the retrofitted dry detention pond described in Section 6.2 within a WSC framework.

10.2 Lessons from Rondevlei, Mitchells Plain

10.2.1 The city as a catchment

Several countries, such as Singapore and Australia, have taken significant strides in implementing the 'City as a catchment' concept. This concept involves the development of diverse, integrated, resilient, and decentralised water sources within cities' boundaries (Wong & Brown, 2009; Wong *et al.*, 2020). For example, Singapore has succeeded in securing nearly 40% of its water supply from within its borders. By 2018, Singapore captured stormwater from two-thirds of its metropolitan area, storing it for later use. (Lim *et al.*, 2011; Irvine *et al.*, 2014; Song *et al.*, 2019). This initiative was necessitated by Singapore's need to be self-sufficient in water, thereby reducing reliance on Malaysia's water system, and was facilitated by an effective institutional and technological environment (Tortajada, 2006).

The CoCT can also leverage stormwater as an alternative source through SWH (Fisher-Jeffes, 2015; Seyler *et al.*, 2016; Rohrer & Armitage, 2017; Okedi, 2019; Gxokwe *et al.*, 2020). Retrofitting existing infrastructure, e.g., dry detention ponds, is beneficial as cities can avoid allocating more land for new stormwater harvesting sites and aids in SWH (Schueler *et al.*, 2007; Pennsylvania Environmental Council, 2012; Lamond *et al.*, 2015). Retrofitting of existing infrastructure in urban areas, might be deterred by the social and financial implications. Nevertheless, should the projected water scarcity levels materialise, the opportunity cost of not retrofitting could prove to be even more significant. The School Pond retrofit explored in this study demonstrated the potential stormwater harvesting capabilities of low-cost community-involved infiltration basins. The pond will support the use of stormwater as a water resource through MAR into the CFA.

The inflow volumes were obtained from a hydraulic model (Chapter 8) and the mean annual MAR from the pond retrofit was found to be 5900 m^3 – about 0.002% of Cape Town’s annual demand for 2022 (Western Cape Government, 2022). The harvested volume from one pond is minimal but the School Pond retrofit resulted in a 120% increase in MAR. Thus, retrofitting can be considered worthwhile considering that the CoCT has 230 potentially retrofittable ponds overlying the CFA that can likewise increase MAR in the city. In addition, the retrofitted ponds have the potential to mitigate the urban heat island effect through evaporative cooling as they increase evapotranspiration which cools the surrounding areas. Further, the pond will continue to support flood risk management through the provision of detention storage for large storms.

In addition to augmenting a catchment's water quantity through MAR the retrofitted pond enhances the quality of stormwater originating from the pond catchment area. This enhancement is accomplished through a treatment train – a series of SuDS facilities that emulate a natural hydrological response to a storm event, effectively treating polluted stormwater using various method (Bastien *et al.*, 2010). Silt and pollutants are conveyed to the pond through the stormwater network. The riprap plays a vital role in reducing the stormwater flowrate, capturing silt, often carrying heavy metal pollutants. Slowing the flow of stormwater allows for sediment to settle and reduces the silt load at the inlet point. (Figure 10-1a). The riprap also traps some transported litter (Figure 10-1b), while additional litter is trapped in front of the rock check dams. The bioswale then further improves the water quality by trapping the finer sediments in the soil matrix through filtration, where pollutants bind to the soil particles and can be degraded. Further treatment through denitrification and adsorption was shown through the laboratory experiments described in Chapter 9 where it was seen that soil from the CFA can reduce pollutants like heavy metals by up to 98%.

Figure 10-1: a) Deposited silt in riprap b) Riprap during after a storm

10.2.2 The city as a provider of urban ecosystems

Urban ecosystems are ecological systems located within cities or other densely settled areas that consist of blue and green spaces and the ecology that exists therein (Haase *et al.*, 2014). Urban blue/green spaces such as parks and community gardens can be private or public and serve an essential function in urban settings (WHO, 2016). Green spaces can also be viewed as centres of community interaction that provide many benefits, such as relaxation and social bonding (Gibson, 2018; Enssle & Kabisch, 2020; Grima *et al.*, 2020). Also, green spaces are increasingly important, particularly in the context of expanding 'concrete jungles' as they can help mitigate the urban heat island effect (Li *et al.*, 2004; Hyder & Haque, 2022). Green urban spaces are often characterised by diverse fauna and flora (Wang *et al.*, 2019), forming sensitive ecosystems that require conservation and protection. While this protection is often provided by the local authorities (and environmental agencies), some scholars argue that involving communities in the management of urban ecosystems promotes effective parks programmes and can strengthen users' attachment to their local ecosystems (Kobori & Primack, 2003; Huang, 2010; Shan, 2012).

In this section, it is emphasised – using observations from the School Pond – that pond retrofits around the CoCT can provide an arena for engagements that promote community involvement in the upkeep and guardianship of urban green spaces.

The School Pond investigated in this study had some fauna and flora prior the retrofit, however, some residents who assisted in the retrofit construction communicated having little to no appreciation of the pond ecology. However, earlier consultations with the community revealed that residents would like to have more plants and flowers in the pond (Section 6.2.1). Consequently, one of the retrofit project partners invited and introduced members of the community to an ecologist who specialises in ecosystem restoration in the Cape Flats area. The ecologist brought plants – particularly the Cape Flats Sand Fynbos, which are indigenous to the Cape Flats (Rebelo *et al.*, 2011) – which were planted by the primary school pupils from the two neighbouring schools to mark Spring Day on the 1st of September 2021. They engaged the students and members of the community in the concept of restorative ecology – which is the process of rehabilitating ecosystems that have been damaged or degraded (Aradottir & Hagen, 2013; Maunder, 2013; Tapia *et al.*, 2021).

The ecologists addressed the members of the community and then has a discussed with the community on how restorative ecology aligns with restorative justice – a concept of justice that relies on interaction with the victim and addressing past injustices (Wenzel *et al.*, 2008; Davis, 2019; Murhula & Tolla, 2021) – in a post-Apartheid South Africa. Both concepts seek to encounter, repair, and transform ecosystems and communities, respectively. Restoration of fynbos vegetation, which was historically cleared for settlement and replaced with alien/invasive species (Stirton, 1978), represents a full-circle process as displaced people of colour, who were also deprived of their homes and moved to the Cape Flats, gradually receive compensation for their land. Thus, the restoration of fynbos creates a symbolic reconnection with their ancestral territories. This multidimensional process not only encompasses ecological rehabilitation but also addresses historical injustices by recognising the rights of marginalised communities. The restoration efforts serve as a testament to the role of restoration in retrofitted ponds in fostering socio-ecological resilience.

The activities at the pond increased the residents' awareness of the biodiversity in the pond, sparking a newfound enthusiasm and appreciation among both the visiting residents and school pupils. The primary school students living near the pond gained knowledge about its ecosystem and showed a sense of responsibility in preserving its ecological balance.

The retrofit project brought about notable revelations to both community members and primary school pupils, particularly regarding the presence of the endangered Western Leopard Toad (*Sclerophrys pantherina*) within the pond. The Western Leopard Toad is classified as an endangered species (IUCN SSC Amphibian Specialist Group and South African Frog Re-assessment Group (SA-FRoG), 2016; Measey *et al*., 2017). Moreover, a resident residing near the pond opined that despite a small permanent pool of water, the absence of mosquitoes was attributed to the frogs actively preying on mosquito larvae. Thus, the project stimulated interest in the pond's diverse fauna and flora, as evidenced by the active participation of community members in capturing and sharing images of the pond's biodiversity through the WhatsApp group (Figure 10-2). Some of the fauna and flora that were captured and shared were identified by the ecologist. In the first row of Figure 10-2 from left to right, are the Cape Dune Mole-Rat (*Bathyergus Suillus*), Western Leopard Toad (*Sclerophrys pantherina*), Heady Maiden Moth (*Amata Cebera*). The second row shows the Western Leopard Toad, Wetland/swamp within the pond, and Blue Emperor Dragonfly (Anax imperator). The third row shows Sand Babooncabage (*Othona coronopifolia*), Cape Skink (*Trachylepis capensis*), and Common Rush (*Juncus effusus*).

The active engagement of the community in vegetation restoration and retrofit construction increased their interest in the biodiversity of the pond, particularly among school pupils. The pupils actively participated in environmental club activities such as regular litter collection (Figure 10-3).

The residents demonstrated a sense of ownership and appreciation for the pond by voluntarily picking up litter within and around the pond. However, interest in the pond was inconsistent, and the school club occasionally failed to visit as committed. Furthermore, the community's involvement was mainly during the construction period and when researchers organised clean-up efforts (Figure 10-4).

Figure 10-2: School Pond ecosystem diversity
Photo Credit: Abe Human and Ganief Nell

Figure 10-3: Trash collected from the pond

Figure 10-4: Pond clean-up during the mural launch day

The importance of green spaces as centres of community interaction that offer various amenities, including relaxation and social bonding, has been highlighted in previous discussions. In the context of SuDS, amenity goes beyond recreational areas and aesthetics and includes education and awareness, health and safety, and environmental risk assessment and management, as noted by Armitage *et al.* (2013). Through an initial conversation with the community, it became clear that amenity was a critical consideration for some members of the community who envisioned the School Pond as a safe space for individuals and families to socialise, relax, and take walks after work. Various landscaping designs were proposed during the community interactions, but the final design presented in this thesis was the compromise aimed at keeping retrofit costs low while providing the necessary elements for multifunctional infiltration basins. This decision was necessary as the CoCT has limited resources, and the use of high-value materials in the City's ponds may be risky due to theft. While one suggestion from the community involved fencing off the pond to address concerns regarding high levels of crime in the area, this would limit the pond's functionality as a green space and create a negative perception about the safety of the area. Torres *et al.* (2018) assert that fencing signals that something or someone undesirable needs to be kept out or kept in. The School Pond is located in the Lentegeur policing precinct, which has relatively high crime rates, ranking 5th, 13th, and 19th among 151 precincts in the Western Cape province for common robbery, carjacking, and malicious property damage, respectively (South African Police Service, 2021). However, the Rondevlei community benefits from an active neighbourhood watch team – security service that patrols the neighbourhood – that had, at the time of writing (2023), kept the pond area relatively safe. Given the presence and apparent effectiveness of this neighbourhood watch there appears to be no immediate need to fence off the School Pond.

The School Pond also exhibited potential to provide amenities to the Rondevlei Park community. Several individuals and groups were observed sitting in and around the pond (Figure 10-5), suggesting that the community is beginning to use the space for social activities.

Figure 10-5: a) A family sitting by the pond b) The environmental club picking trash c) Pupils assisting with the planting

Active community involvement, along with additional features like the mural and benches (Figure 10-6), will continue to play a vital role in encouraging the use of the pond for walks and other social activities.

Figure 10-6: Mural and installed benches next to the School Pond

10.3 Chapter summary

This chapter demonstrated the additional benefits of retrofitted ponds for the residents of Cape Town, as demonstrated through a case study of a retrofitted pond in Rondevlei, Mitchells Plain, Cape Town. This chapter examined two narratives using recontextualised characteristics of water-sensitive cities (WSC):

- Cape Town as a catchment which uses blue-green infrastructure such as SuDS to improve water quality and increase water supply.
- Cape Town as a provider of urban services that foster biodiversity and restorative ecology, thus aiding water-sensitive communities.

11. Economic analysis of pond retrofits

In this chapter the economic viability of the retrofitted pond is assessed by considering the direct economic cost and benefits from the potential water volume that can be harvested in the School Pond. The scope of the assessment is limited to a simple economic model that considers the direct financial implications associated with the volumes harvested from the pond and does not include the financial implications of the broader befits benefits, such as water quality improvement, amenities provision, and biodiversity enhancement, discussed in Chapter 10.

11.1 Evaluating the School Pond benefit cost ratio

A benefit-cost analysis was conducted to evaluate the project's feasibility, requiring an assessment of the pond's benefits and associated costs over an extended period. A 20-year project lifespan was adopted, aligning with the predetermined operational period for water schemes set by the CoCT (CoCT, 2018d).

The benefit-cost ratio (BCR) defined is as ‘the ratio of the discounted total benefits of the program to the discounted total costs over some specified period’ (Schultz *et al.*, 2010). This ratio represents the present value of all associated project benefits to the total costs. If the BCR is greater than 1, the program is financially beneficial.

A discount equation is used to assess the present value of the project’s cost (including the capital, maintenance, and operational costs) and cash inflows (benefits) as defined by Equation 11-1.

$$\mathrm{PV} = \frac{\mathrm{FV}}{(1+\mathrm{r})^{\mathrm{N}}}$$ **Equation 11-1**

PV is the present value, FV is the future value, *r* is the discount rate, and N is the operational period.

The PV is defined as ‘the current value of a future amount, calculated by discounting the FV back at a known discount or interest rate for a specified period’, while the FV is defined as ‘the value that a current amount will grow to at a given interest rate over a given period’ (Besley & Brigham, 2014; Dahlquist *et al.*, 2022). The discount rate is ‘the annual rate of reduction on a future value and is the inverse of the growth rate' (Dahlquist *et al.*, 2022). It is the interest rate used to determine the present value of future cash inflows.

The BCR was derived by initially evaluating the PV of maintenance and cash inflow spanning 20 years – aligned with the operational duration suggested by the CoCT (CoCT, 2020b) – which were compounded to project their future worth. Subsequently, these compounded costs were aggregated and discounted back to their PV. The capital cost was already expressed in PV terms and was combined with the maintenance costs to ascertain the overall expenditure. This approach assumes that the maintenance cost and cash flows only increase because of the discount rate, whereas in practice, other variables might influence the rate of cost variation. However, the BCR should not be the only decision rule, and other factors, such as net present value (NPV) and Unit Reference Value (URV), should be considered (Conningarth Economists, 2014). The NPV is 'the present value of the cash inflows of a project minus the present value of the cash outflows of the project' (Dahlquist *et al.*, 2022). The NPV assesses the value the project will add. The URV (Equation 11-2) is commonly used in water resource planning in South Africa to compare the financial efficiency of water schemes (Bester *et al.*, 2020). It is defined as a 'cost-effectiveness measure that computes the unitary cost of supplying a cubic metre of water at the required assurance of supply, over the portion of the water management or augmentation project's lifespan during which it produces economic benefits for society' (Bester *et al.*, 2020).

$$\mathrm{URV} = \frac{\mathrm{PV\ life\ cycle\ cost}}{\mathrm{PV\ additional\ yield}} \qquad \textbf{Equation 11-2}$$

The discount rate used in this evaluation was the social discount rate, used for projects with social benefits, such as water augmentation schemes (e.g., MAR). A social discount rate of 8% is recommended for South African capital expenditure calculations (van Niekerk & du Plessis, 2013). This value was adopted in the current analysis, complemented by a sensitivity analysis encompassing 6%, the value cited in the WCWS report (Department of Water and Sanitation, 2021), and 10%, as suggested by the South African National Treasury (Department of National Treasury, 2021).

11.1.1 Evaluating the capital cost

The capital cost is the once-off expenditure to acquire, construct or set up a new project. Calculating the retrofit's total cost involved examining various expenses, such as stakeholder engagement, retrofit design, supervision during construction, labour wages, and material procurement. Professional fees for design and supervision were determined using time-based fees as stipulated by the Engineering Council of South Africa's (ECSA) Guideline for Professional Fees (ECSA, 2021).

The required technical staff roles, such as a senior stormwater engineer, resident engineer, community development officer, and maintenance manager, were categorised based on ECSA's criteria which classify technical staff at an institution such as the CoCT as category C. The hourly rates for the technical staff were calculated as follows:

- The Cost of Employment (COE) for these roles was determined by reviewing job advertisements on the CoCT job portal and Indeed.com. A higher value was selected when multiple COE values existed for the same role.
- The ECSA guidelines indicate that the time-based rates for a category C employee equate to 17.5 cents per hour for every ZAR 100 or a fraction thereof of the total annual COE of the individual concerned. This guideline was applied to determine hourly rates, culminating in approximately ZAR 1600 for the senior stormwater engineer, ZAR 1500 for the resident engineer, ZAR 615 for the community development officer, and ZAR 550 for the maintenance manager.

The required time for each activity was then estimated based on the data from this study. The derived time and corresponding time rates were used to calculate the costs of stakeholder engagement, project design, and supervision during construction and maintenance. Additionally, material costs were established through market-related figures sourced from quotations procured during the retrofit construction.

The summary of the activities and the associated capital costs if the CoCT implements a retrofit like the School Pond retrofit are presented in Table I-1 in Appendix I. The total capital cost was found to be ZAR 667 000.

11.1.2 Establishing the maintenance cost

The maintenance plan mentioned in Chapter 6 includes a description of the required maintenance work and the frequency thereof to ensure the retrofit functions as designed. The rates for the maintenance work were estimated from industry rates including the COE for the labourers and the supervisor. The rates and the frequency of the maintenance activities were then used to calculate the annual maintenance cost (Table I- 2), which was ZAR 108 700 in 2023.

11.1.3 Evaluating the cost of harvested water

The cost of the harvested water was derived by applying the following considerations and assumptions:

1. The CoCT will implement groundwater abstraction for the CFA as set out in its water strategy (CoCT, 2019b).
2. The CoCT will construct and operate groundwater abstraction sites within the Cape Flats Aquifer at the identified sites (Figure 8-27) and use the already allocated funds as per CoCT (2019).
3. The abstracted water from the retrofitted pond(s) will be pumped from the identified CoCT abstraction sites. Consequently, there will be no requirement for water abstraction at the retrofitted ponds. This means that the installation of additional pipelines and machinery can be avoided, thereby eliminating the need for additional financial investments in these infrastructural elements.
4. The CoCT will price the abstracted groundwater based on the surface water prices plus the groundwater plant's operational cost of ZAR 5/m^3 – see costs for CFA abstraction in CoCT (2019). The water system's capital and operational expenditure are already factored in the surface water tariffs and can be ignored here.
5. The historical per capita demand for the CoCT is 206 L/day (CoCT, 2017). Applying the CoCT water tariff tier system to the demand for a residential non-indigent four-person household results in a water tariff of ZAR 34.5 /m^3.

From the above considerations, the cash inflow from selling 1 m^3 harvested water is estimated to be ZAR 29.5 (derived from ZAR 34.5 less ZAR 5.00).

The mean annual volume of harvested water from the pond was obtained from Scenarios 2 and 4, described in Chapter 8. Scenario 2 is the 'Shallow water table' scenario (named the Shallow WT scenario in this chapter), where the mean annual volume of harvested water was 5905 m^3. Scenario 4 considers the MAR if the CoCT abstracts groundwater from the CFA – as planned – lowering the water table and providing an extended infiltration area. The harvested volume at the pond increases to 9330 m^3 per year in Scenario 4 (named CFA GW scenario in this chapter). However, the pond already had some infiltration potential before the retrofit (2715 m^3); thus, the benefit of the retrofit should be assessed using the additional (net) volume of groundwater harvested. In this study the net harvested volumes were found to be 3190 m^3 for the prototype retrofit 'shallow water table' scenario and 6615 m^3 for the groundwater use scenario.

11.2 School Pond direct economic viability

The capital and operational costs, volume, and value of the net harvested stormwater were then used to calculate the project's BCR, NPV, and URV using the Shallow WT and CFA GW use scenarios. Table 11-1 presents the BCR of the project appraised at a social discount rate of 8%. The calculated BCR for the pond in the context of the shallow water table scenario is 0.55. This value is less than one, indicating that the project would not yield financial advantages for the CoCT if only direct monetary gains were considered. Further, the NPV is negative (- ZAR 822 900), signifying that the CoCT would not receive a financial return on the investment when considering the volume of water harvested using the prototype retrofit at the School Pond. The URV for this scenario was ZAR 28.5 /m^3.

The alternative scenario, where the CoCT abstracts groundwater from the CFA and lowers the water table, results in a positive BCR of 1.14 and a positive NPV of ~ ZAR 247 500 (Table 11-1). The URV in this scenario was R13.7.

Table 11-1: Economic viability for the School Pond retrofit at 8% discount rate

	Shallow WT Scenario	CFA GW Scenario
PV of capital cost	ZAR 667 000	ZAR 667 000
PV of maintenance cost	ZAR 1 152 100	ZAR 1 152 100
PV of all costs	ZAR 1 819 100	ZAR 1 819 1000
PV of cash inflow	ZAR 996 300	ZAR 2 066 600
NPV	- R 822 850	R 247 500
BCR	0.55	1.14
URV	ZAR 28.50	ZAR 13.70

A sensitivity analysis was then performed to assess the impact of varying the discount rate on the project's economic viability. Table 11-2 contains the summary of the sensitivity analysis results. For the given scenarios with varying discount rates, the URV for MAR in the pond ranged from ZAR 26.4 /m^3 to ZAR 31.2 /m^3 in the shallow groundwater scenario and from ZAR 12.7 /m^3 to ZAR15.0 /m^3 in the groundwater use scenario. Meanwhile, the BCR varies from 0.52 to 0.57 in the shallow groundwater scenario and from 1.08 to 1.19 in the groundwater use scenario across 6%, 8%, and 10% discount rates. The BCR for the shallow WT scenario indicates that the project would not be financially viable, regardless of the discount rate used. However, the retrofit is financially viable if the CoCT abstracts groundwater from the CFA, which increases the MAR at School Pond.

As the discount rates increase in both scenarios, both the BCR and URV decline. This is because higher discount rates lead to reduced PV of maintenance costs and cash inflows. This outcome arises due to the reduced principal value when confronted with higher discount rates. However, the capital cost remains constant within this calculation as it is already in PV terms, consequently, the reduction in PV of maintenance costs and cash inflows triggers a corresponding decline in the computed ratios.

Table 11-2: Summary of results of the sensitivity analysis

Discount rate	Shallow WT Scenario			CFA GW use Scenario		
	6%	8%	10%	6%	8%	10%
URV (ZAR/ m^3)	31.2	28.5	26.4	15.0	13.7	12.7
BCR	0.57	0.55	0.52	1.19	1.14	1.08

Subsequently, a comparative analysis was performed on the URV from this study and other augmentation alternatives that the CoCT is considering, namely, water reuse through wastewater treatment for potable purposes at the Faure New water scheme (FNWS), the Berg River to Voelvlei Surface water augmentation scheme (BRVAS), and seawater desalination (CoCT, 2020b). The URVs of the three CoCT schemes were determined using a social discounting rate of 6% (Department of Water and Sanitation, 2021), enabling a direct comparison with the URVs derived at the corresponding 6% discount rate in this study (Figure 11-1). This analysis was conducted for Shallow WT and CFA GW use scenarios.

The BRVAS option has a comparatively low cost of ZAR 4.62 /$m^{3,}$ which underscores its competitiveness as a water augmentation scheme, particularly if cost efficiency is a crucial consideration. Meanwhile, the FNWS water reuse approach demonstrated a URV of ZAR 11.7 /m^3, positioning it as the median value among the augmentation alternatives within the CoCT. Despite its relatively high cost (ZAR 17.3 /m^3), seawater desalination remains an attractive option. Its high cost is offset by its potential to provide a dependable water source, rendering it an enticing prospect during periods of strain on conventional sources. However, such considerations are subject to factors such as environmental impacts, energy demands, which are particularly relevant given South Africa's reported electricity shortage (Mabugu & Inglesi-Lotz, 2022; Nkosi & Govender, 2022), water source reliability, and long-term sustainability.

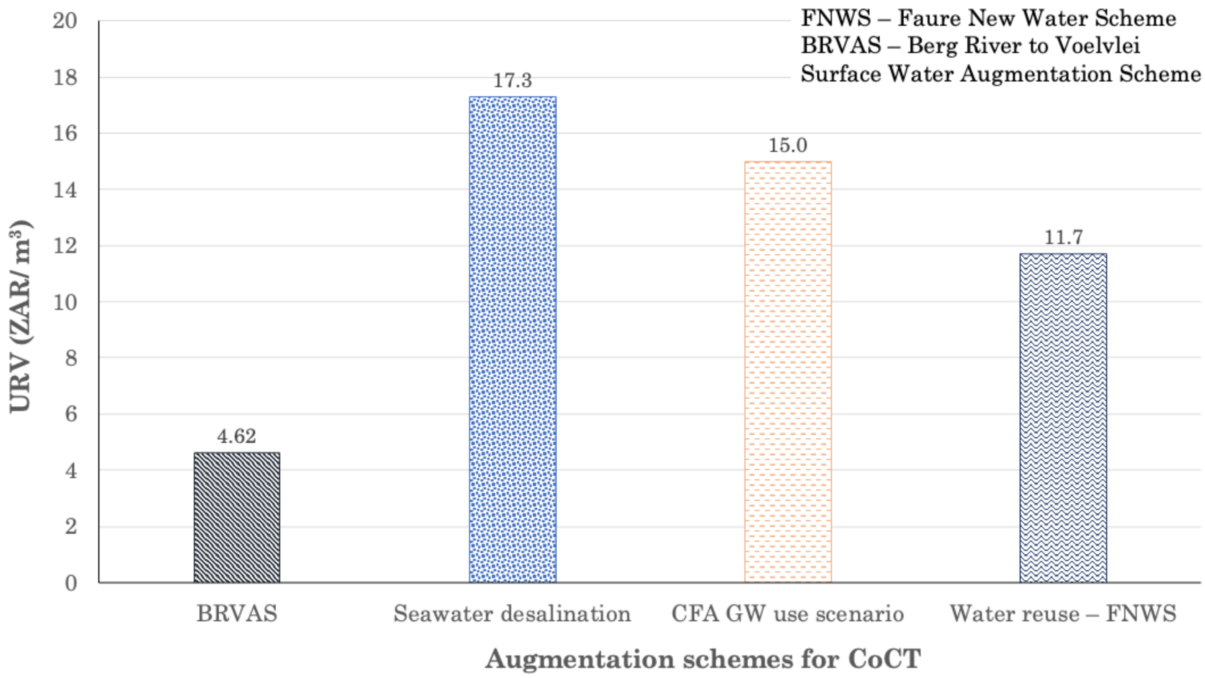

Figure 11-1: URV for augmentation schemes calculated at 6% discount rate

The study's 'CFA GW use' scenario had an URV of ZAR 15.0 /m^3. The 'CFA GW use' scenario is less than the desalination URV, indicating that groundwater use coupled with SWH via MAR is more financially attractive than desalination. Nevertheless, the URV of this scenario remains higher than that of the BRVAS and water reuse alternatives. Moreover, the reliability of these schemes relies on adequate rainfall, whereas the water reuse scheme can theoretically operate even during drought periods owing to the continuous production of wastewater. However, treatment costs may increase because of the increased pollution and concentration in wastewater (Tran *et al.*, 2017).

Although cost is an essential determinant, it is crucial to acknowledge that a comprehensive assessment of water augmentation schemes should encompass diverse dimensions. Therefore, although the findings of this study offer a valuable point of departure, a holistic evaluation remains essential for well-informed choices that guarantee both cost-effectiveness and resource security.

11.3 Chapter summary

The financial feasibility of pond retrofits was evaluated in this chapter, using School Pond as a case study. Two scenarios, the 'Shallow WT' and 'CFA GW use', were analysed. The assessment employed the BCR, NPV, and URV as metrics to gauge the economic viability of the retrofits.

The BCR and NPV results for the shallow WT scenario suggest that the retrofit is not financially viable across all discount rates considered. However, viability is achieved if CoCT abstracts groundwater from the CFA, boosting MAR at the School Pond. A comparison of URV with other augmentation options in CoCT shows that retrofitting in the 'GW use scenario' is a competitive choice.

Overall, it was shown that retrofitted detention ponds exhibit financial viability in some scenarios.

12. The prospect of urban MAR in retrofitted detention ponds

Previous studies have proposed MAR using a spreading method in existing detention ponds to achieve SWH within Cape Town. This approach has the potential to reduce the infrastructure costs associated with SWH. However, the practicality and viability of adapting current detention ponds in Cape Town for MAR are yet to be verified. Consequently, the principal goal of this research was to assess the practicality and viability of converting Cape Town's detention ponds to enable, among other things, urban MAR.

This study used a mixed-methods approach, including desktop studies, laboratory experiments, computational hydraulic simulations, field investigations, and unstructured participant observations. These methodologies were used to evaluate the technical, social, and economic viability of retrofitting detention ponds.

This concluding chapter provides an overview of the thesis and discusses research outcomes, site-specific results, and their broader implications. A practical framework tailored to the CoCT is presented, offering guidance on implementing similar retrofit projects. The chapter also highlights how the study contributes to the current knowledge in this field and concludes with recommendations for future research.

12.1 Stormwater retrofits: A South African perspective

South Africa diverges from Global North countries, which often formulate retrofit guidelines and implement urban MAR in two aspects: 1) socio-economic challenges and 2) stormwater quality. Therefore, the approach to implementing SuDS infrastructure, particularly retrofits, in South Africa might be different. The following section presents the social, technical, and economic perspectives on how SuDS retrofits can be implemented in South Africa.

12.1.1 Understanding and addressing the South African socio-economic perspective

While South Africa has significant socio-economic challenges – for example, crime, high unemployment, increasing inequality, racial tensions, poor access to sanitation services, and increased informal settlements (Taing *et al.*, 2019; Whitehead, 2019; von Fintel & Orthofer, 2020; Swatuk *et al.*, 2021; Pirtle, 2022) – individually these challenges are not necessarily unique to the nation. Other countries in the Global South, like Brazil, Costa Rica, and India, face similar challenges and still manage to implement SuDS retrofits – see Knauer *et al.* (2010); Dillon *et al.* (2014); Diep *et al.* (2019) and Chapa *et al.* (2020). Nevertheless, even in those nations, special provisions must be made to address their challenges. However, the South African context might be further complicated by the ramifications of Apartheid and the performance of the post-Apartheid government with respect to public service delivery which must be delicately addressed in any public-facing projects in the nation (Tanyanyiwa *et al.*, 2023).

Retrofitting SuDS in SA might be different compared to the Global North, where socio-economic factors are usually not at the fore – see Dhakal & Chevalier (2017); Frantzeskaki (2019); Oladunjoye *et al.* (2022). Although there is an emerging appreciation of the importance of community engagement, there is a need for clearly defined engagement processes and frameworks to assist water practitioners in establishing and retrofitting stormwater-harvesting infrastructure, particularly in the Global South.

This study contributes to the development of such frameworks by examining the Mitchells Plain retrofit as a case study. The findings offer insights into two main areas: 1) engaging with South African communities (Sections 6.2.1 and 7.1) and 2) constructing and maintaining public-facing retrofits (Sections 6.2 and 6.3). In addition, an engagement assessment method was devised by adapting an existing approach to monitor and evaluate the level of outreach and engagement within a project.

The adopted mixed-method research philosophy helped determine that the technical aspects of retrofitting detention ponds require minor adjustments within the South African context to effectively and actively addressing social issues. Key among these are: 1) security, which influences the choice of construction materials for pond retrofit, and 2) the extent of community outreach and engagement, which determines the acceptance and support for retrofit initiatives.

The security concerns can be addressed using low-value materials such as sandbags from recycled materials such as stone check dams to act as litter traps. This approach contributes towards producing resilient retrofits.

This study also found a need to establish good, mutually beneficial relationships with local stakeholders to obtain the community's informed consent. Informed consent is obtained by a continuous outreach process that is cognisant of the different actors in the community and can identify and navigate competing interests (Sections 6.2.1 and 7.1). However, the process of community engagement can be time-consuming and demands expertise that is not typically possessed by stormwater engineers. This engagement must also be an ongoing effort throughout the project's implementation to foster engaged communities. These engaged communities are essential for maintaining pond retrofits and ensuring their sustainability. The South African context means that communities may not fully enjoy the amenities provided by pond retrofits because of the high level of crime. Nonetheless, the Mitchells Plain retrofit suggests that engaging the communities that live near and around retrofitted ponds from the outset can aid in creating water-sensitive communities that value their ecosystem and are more likely to protect municipal efforts to improve the management of urban water (Chapter 10).

12.1.2 Assessing and addressing stormwater quality concerns

The quality of stormwater and associated treatment processes play a crucial role in the implementation of stormwater harvesting. Various design guidelines, such as those proposed by Schueler *et al.* (2007) and Payne *et al.* (2015), Woods-Ballard *et al.* (2015), and the Bureau of Environmental Services (2016), offer typical pollutant removal efficiencies for different types of SuDS designs. Some guidelines specify the desired effluent quality based on whether the receiving body is surface or groundwater. However, the existing stormwater management policy for the City of Cape Town (CoCT, 2009) only provides target removal efficiencies without explicitly considering the water quality of the receiving body.

This study postulated that stormwater in Cape Town might exhibit higher pollution levels than regions in the Global North due to significant anthropogenic activity disparities. Factors such as the prevalence of inadequately drained informal settlements and poor sanitation services contribute to the potential degradation of the stormwater quality in the city. Consequently, relying solely on target removal efficiencies may result in the use of poor-quality water for MAR systems, thereby introducing potential contamination risk. This study thus evaluated the stormwater quality in residential areas of Cape Town and compared the stormwater characteristics against the South African water quality guidelines for recreational use (DWAF, 1996c) – see Table 5-2. The stormwater quality in Cape Town residential areas was also compared to other Global North and South countries. This evaluation of stormwater quality characteristics from catchments in Cape Town contributes to the existing knowledge on water quality in Cape Town by establishing links between water quality and land use while providing additional statistical measures of inference.

The stormwater quality assessment indicated that the stormwater quality in Cape Town's residential areas fell below the South African water quality guidelines for recreational use (DWAF, 1996c), necessitating treatment before direct mixing with groundwater. Overall, the mean stormwater characteristics from the residential areas in the assessed catchments were more polluted than those in developed countries (Singapore, the Netherlands, and the USA). However, the stormwater quality in two of the four analysed catchments was generally better than that in a comparable developing country (India). These variations among the assessed countries and analysed catchments signify the importance of site-specific factors in stormwater characteristics. Furthermore, socioeconomic status does not appear to be the sole determinant of stormwater quality, thus necessitating retrofitting strategies that consider the unique characteristics of each area.

An investigation of the treatment efficacy of retrofitted ponds was conducted upon recognising the notable disparities and generally poor-quality stormwater within the CoCT. The analysis evaluated the anticipated effluent quality following the filtration of stormwater through the CFA media. The treatment efficiency was established by comparing the effluent and influent concentrations. Treatment efficiency was examined across varying vadose zone depths. Furthermore, because of the observed variable nature of stormwater quality, the treatment efficiency of retrofitted ponds using stormwater that is five times more polluted than a typical Cape Town residential catchment was also investigated. Column studies packed with media from the CFA and silica sand were used to infiltrate synthetic stormwater containing organic contaminants (TP, TN, and TOC) and heavy metals (Cr, Cu, Pb, Ni, and Zn).

The mean concentrations of the effluent during both infiltration and saturation cycles were evaluated to assess the acceptability of the effluent. The mean effluent concentrations were compared against the CFA groundwater baseline (the mean characteristics of the groundwater in the aquifer) and established water quality standards from domestic water use and aquatic ecosystems.

The investigation revealed that the columns generally improved SSW quality. The columns with media from the CFA and silica sand produced effluents with mean concentrations below the CFA baselines for two of the three organics assessed (TP and TOC). However, the treatment for TN was insufficient, and all the columns at all depths produced effluents above the CFA baseline. This result indicates a risk when using stormwater for MAR in CFA as it may introduce TN contamination. Nevertheless, the columns produced effluents below the domestic water use limits for TP, TN, and TOC. Furthermore, there was a significant improvement in the treatment efficiency for all organic contaminants in the saturated zone. This indicates the prospect of further reducing the contaminant concentrations within the aquifer.

The CFA media, as well as silica sand (representing a worst-case scenario), demonstrated a significant ability to lower the concentrations of heavy metals in both 'typical' and highly polluted stormwater (five times higher than typical contaminant concentrations). This was achieved even when the vadose zone depth was limited to 0.5 m. The columns generated effluent concentrations below the CFA baseline for all metals except for Zn (observed in all columns and vadose depths). However, the mean effluent concentration of Zn remained within the recommended standards for domestic water usage. These results indicate that the effluent produced by infiltrating typical and highly polluted stormwater using CFA media aligns with established recommendations (effluent quality should not surpass the baseline level of the aquifer) for four of the five contaminants analysed. Although Zn concentration exceeded the CFA baseline, there was no immediate risk associated with domestic water use.

There was a statistically significant correlation between vadose zone depth and treatment efficiency. Thus, the worst-case scenario analysis can be conducted by considering the treatment efficiency of silica sand in the 0.5 m vadose zone with stormwater, which is five times more polluted than stormwater from a typical residential area in Cape Town. For the 0.5 m vadose zone depth, the silica sand columns reduced the concentrations of most of the assessed contaminants; however, for TN, Cu, and Zn, the silica sand columns could not treat the SSW to concentrations below the CFA baseline. Poor treatment implies that infiltrating poor-quality stormwater might contaminate shallow aquifers, a concern also raised by Gxokwe (2017) and Mauck (2017) for the CFA. Consequently, retrofitting ponds with shallow water tables for MAR without additional treatment may not be appropriate without site-specific investigations.

12.1.3 The viability of MAR via retrofitted detention ponds

One significant finding from Okedi (2019) was that stormwater harvesting via MAR would lead to a 30% increase in groundwater infiltration into the CFA. The MAR was proposed to occur in 61 ponds found in the Zeekoe catchment, producing an additional mean annual infiltration volume ranging between 9 – 12 Mm^3. Okedi's assessment considered MAR as the additional infiltration resulting from introducing LIDs in 61 ponds at a catchment scale. However, this study defined MAR as the water volume contributing to the CFA, considering losses due to surface evaporation, evapotranspiration, and groundwater outflow. Further, Okedi (2019) did not investigate the influence of water table fluctuations on infiltration volumes, which significantly impact infiltration volumes as shown in Section 8.7.1. Thus, the MAR values reported by Okedi (2019) may have been overestimated. Further, it is difficult to objectively compare this study's findings to those of Okedi (2019) because of the difference in scales (catchment vs neighbourhood), MAR definitions, and geo-hydraulic conditions.

The short- and long-term hydrological impacts of the School Pond retrofit were evaluated using a calibrated and validated geo-hydraulic PCSWM model that accounted for surface-groundwater interactions.

This study revealed that detention ponds within a sandy aquifer, like the CFA, possess a certain degree of infiltration capacity. However, the implementation of pond retrofitting increased the MAR within these ponds. A shallow water table ranging from 0.6 to 1.8 mbgl limits the potential infiltration achievable through retrofits. Nonetheless, even under these conditions, the School Pond exhibited a 120% increase in MAR, translating to an additional 3200 m^3/year. Furthermore, the study examined a scenario in which the water table was lowered owing to water abstraction by the CoCT from the CFA. In this scenario, the mean annual MAR in the School Pond increased to 290%, harvesting an extra 7800 m^3/year. These MAR values are comparatively modest and do not meaningfully contribute to the COCT water demand. However, the expected increase in MAR resulting from retrofitting is expected to extend to other ponds above CFA. This is primarily due to the evident dominant influence of CFA characteristics compared to the other factors considered in the model, except for the thickness of the LID soil layer (vadose zone). Consequently, integrating an infiltration swale into a pond overlying the CFA is expected to yield similar increases in MAR, as observed in this study. This assertion was supported by the sensitivity analysis conducted in Section 8.5.2, which highlighted ten of the 11 parameters influencing the groundwater response were associated with the modelled CFA parameters.

The climate change scenarios presented in this thesis also indicate that retrofitted ponds are expected to harvest less water via MAR owing to decreased precipitation and higher temperatures.

The financial viability of the School Pond retrofit was assessed through two scenarios: 'Shallow WT' and 'CFA GW use', with economic metrics including the benefit-cost ratio (BCR) and unit reference value (URV) utilised for evaluation. The BCR for the pond retrofit under the 'Shallow WT' scenario, was 0.55, indicating that the project would not yield direct monetary benefits for the CoCT. In contrast, the 'CFA GW use' scenario involving groundwater abstraction and a lowered water table demonstrated a positive BCR of 1.14, indicating the financial viability of this scenario.

A comparative analysis was performed to assess the URVs from this study and other CoCT augmentation options. The augmentation options considered were wastewater treatment for potable use at the Faure New water scheme (FNWS), the Berg River to Voelvlei Surface water augmentation scheme (BRVAS), and seawater desalination. In the 'Shallow WT' scenario from this study, the URV was ZAR 31.2 /m^3, making it the least cost-effective option compared to other alternatives. This is owing to inadequate MAR storage, which affects financial viability. Nevertheless, the retrofitted pond provides other benefits such as amenities, biodiversity, and improved water quality for residents (Chapter 10).

The URV of the 'CFA GW use' scenario was lower than seawater desalination, indicating that combining groundwater use and SWH via MAR is more financially attractive. However, the URV remained higher than that of the BRVAS and water-reuse alternatives. The reliability of MAR and BRVAS depends on the amount of rainfall. The FNWS water reuse scheme can operate even during droughts, although treatment costs may increase owing to increased pollution and concentration in wastewater (Tran *et al.*, 2017).

12.2 A practical approach to retrofitting Cape Town's detention ponds

'…in theory, there is no difference between theory and practice, while in practice there is' – Brewster (1882)

A practical 'middle-out' approach for pond retrofitting ponds in the CoCT is introduced in this section. The term 'middle-out' is adapted from various fields such as public health, education, and human-computer interaction (Cummings *et al.*, 2005; Fredericks *et al.*, 2016; Mindell *et al.*, 2021). In this context, it represents a socio-technical approach that combines traditional top-down approaches, primarily focused on community outreach, with bottom-up approaches, which emphasise community engagement. By merging these two approaches, the aim is to improve operational efficiency and project feasibility, while enhancing social acceptability and buy-in. This middle-out approach leverages findings from literature, laboratory studies, computational simulations, and a full-scale pond retrofit constructed in Mitchells Plain, Cape Town.

First, it is advocated that if the main goal for pond retrofits in the CoCT's detention ponds is MAR, then the CoCT officials ('the City') would be best served by focusing on formal residential areas on the CFA which accommodate 84% of the dry detention ponds in this part of the CoCT. While this practical approach was developed for the CoCT, its principles can be applied to similar South African cities. The practical, empirical middle-out approach is described below:

- Identify technically suitable ponds for retrofitting as infiltration basins. A multi-criteria decision tool that includes, *inter alia*, aquifer type; mean, maximum, and minimum annual water table depths; and inlet and outlet positions, as presented in Chapter 6, can be used.
- If the pond is in a residential area, conduct community outreach at or near the identified pond to inform residents of the potential retrofits if preliminary studies are positive.
- Develop pamphlets in local languages that explain the goals and objectives of the retrofit project and how the project might impact the residents. Residents could also be invited to share their concerns and questions.
- As far as possible, a transdisciplinary team that mirrors the community's racial composition must be used in the outreach step. This is an important step which recognises the legacies of Apartheid and is further addressed in Tanyanyiwa *et al.* (2023).
- The transdisciplinary team should encompass landscape architects, social scientists, stormwater engineers, water quality scientists, and local leaders/ward councillors. The necessary technical expertise is available (as of 2023) across various CoCT departments: Roads and Stormwater, Public Participation, Urban Planning and Design, Scientific Services, and Recreation and Parks Department.
- Local labour recruited from communities immediately adjacent to the pond of interest should be used to construct a pond retrofit. This is necessitated by residents' concerns over who get to work in their spaces (Chapter 6). A socially acceptable and impartial system, such as a modified Expanded Public Works Programme (EPWP) job portal, can be used to select applicants.
- After the initial outreach, a preliminary technical study should be conducted to determine the water table position post-winter and at the peak of summer. Preliminary water table information can be obtained by hand auguring the lowest and highest points of the pond up to a depth of 2 m.
- The augured wells should be used to verify whether the data correlates with the available measured or modelled catchment scale data. A topographical survey can also be conducted if none exists.
- Draft retrofit designs can be developed if the preliminary study results are positive. At this stage, co-design workshops with residents must be conducted

to understand the community's needs regarding the use of the space; and design elements that should be incorporated, including the type of vegetation that can be planted in the pond. This exercise could be used to obtain informed consent from the community. This can be a slow process, and multiple attempts may be required, depending on the residents' past interactions with the city and their interaction with the identified pond. Competing interests can be identified and navigated through this process.

- The significance of the co-design process cannot be overstated, as the insights drawn from each pond will strengthen institutional transformation and possibly diminish institutional inertia towards community-centred projects.
- Residents should be encouraged to establish pond committees as anchors – accessible conduits to the residents. However, this could lead to affinity groups which do not necessarily reflect the needs and voices of the community, as a community can have multiple communities.
- Once the co-design workshops have been finalised, draft designs that include the community's recommendations within the allocated budget can be produced. The retrofit's primary aim, available budget and constraints should be disclosed to the residents, which will aid the prioritisation of elements to include in the design. This step is based on findings from Chapter 6 and has also been successfully implemented in Brazil (Knauer *et al.*, 2010).
- The designs and work schedules should then be presented to the residents. The design team should ensure that a mole barrier is included in the design, with a preference for environmentally friendly barriers if mole-rats are present near the site.
- An accredited training program can be created and implemented to incentivise engagement and improve public knowledge of the value of stormwater infrastructure. The project team can then share the selected residents who will work at the site based on applications submitted via the preferred portal.
- Groundwater monitoring wells should be installed during the construction of retrofit elements to assess the impact of infiltrated water on the groundwater quality.
- A maintenance schedule should be developed alongside the established pond committee. The maintenance plan should aim to utilise paid resident-derived labour. It is recommended that the project strive to achieve full community engagement, which can be tracked using the tool introduced in Chapter 7.

- The geo-hydraulic performance of the retrofitted pond can be evaluated using the hydraulic models for each pond. These models can facilitate the development of a complete digital twin model for a city's stormwater infrastructure.

This practical approach is summarised in Table 12-1 with suggestions of existing CoCT departments (in 2023) that can be used to implement pond retrofits and reference chapters from this document. The need for a flexible approach is crucial, as theory and practice ought to be the same; however, as suggested by Brewster (1882), they are different in practice. Thus, guidelines and practical approaches such as those presented here may still deliver different results, even based on empirical studies. Furthermore, although a trial-and-error retrofit approach may yield favourable outcomes, it is argued that the principled and practical approach outlined here offers a faster route. The practical 'middle-out' approach presented here is a positive step towards understanding and documenting the factors influencing pond retrofits in Cape Town.

Table 12-1a: A practical approach to retrofitting Cape Town's detention ponds

Action	Responsible Department/Person(s)	Reference Chapter
Identify technically suitable ponds to retrofit as infiltration basins. A multi-criteria decision support tool should be used.	Roads and Stormwater Scientific Services	Chapter 4.8 Chapter 11
If the identified pond is in a residential area, then community outreach is required.	Parks and Recreation Public Participation Roads and Stormwater Scientific Services Urban Planning and Design	Chapter 2.5.1 Chapter 6.2.1
Establish and brief multidisciplinary team. Develop outreach material.		
Contact local leader – ward councillor	Project lead	Chapter 6.2.1
Conduct initial community outreach exercise and obtain informed consent.	Parks and Recreation Public Participation Roads and Stormwater Scientific Services Urban Planning and Design	
Conduct topographical and geotechnical surveys – Use of local workforce via the Expanded Works Programme (EPWP) is strongly advised	Local representative Public Participation Roads and Stormwater Scientific Services	Chapter 4.9
If preliminary studies are positive, produce preliminary retrofit designs.	Roads and Stormwater Urban Planning and Design	Chapter 6.2.2

Table 11-1b: A practical approach to retrofitting Cape Town's detention ponds (continued)

Action	Responsible Department/Person(s)	Reference Chapter
Conduct co-design workshops with community. Identify community desires and map potential points of conflict	Parks and Recreation Public Participation Roads and Stormwater Scientific Services Urban Planning and Design	Chapter 6.2.1
Establish Pond committee	Local representative Public Participation	Chapter 6.2.1
Produce draft designs and work schedule. Prioritise elements that further the primary aim of the retrofit. Workforce for retrofit must be derived from local community via the EPWP portal	Local representative Parks and Recreation Public Participation Roads and Stormwater Scientific Services Urban Planning and Design	Chapter 6.2.2 Chapter 6.2.2.2
Install groundwater monitoring wells	Roads and Stormwater Scientific Services	
Retrofit the pond	Local representative Parks and Recreation Public Participation Roads and Stormwater Scientific Services Urban Planning and Design	Chapter 6.2.2

Table 11-1c: A practical approach to retrofitting Cape Town's detention ponds (continued)

Action	Responsible Department/Person(s)	Reference Chapter
Pond Maintenance	Pond committee Parks and Recreation Roads and Stormwater Urban Planning and Design	Chapter 6.3
Monitoring and simulation	Roads and Stormwater & Scientific Services	Chapter 8 Chapter 9

12.3 Study contribution to knowledge

This study contributes to the ongoing discourse around water resource augmentation for the City of Cape Town, particularly stormwater harvesting, by presenting an evidence-based practical approach for retrofitting detention ponds to facilitate SWH via MAR in Cape Town. This study found that technical retrofitting guidelines require minor modifications and that the key to successful pond retrofits in a South African context is handling social issues through extensive community outreach and, eventually, engagement. Summarily, this study found that:

- Cape Town's detention ponds with similar characteristics to the School Pond can be successfully retrofitted by leveraging engaged communities and using low-value materials to increase MAR volumes to varying extents depending on the water table depth. Pond characteristics such as soil type, aquifer type, mean, max and minimum annual water table depths, inlet and outlet positions and security concerns play a significant role in the pond selection process.
- Retrofitting a detention pond to enhance infiltration increases MAR via infiltration in a pond by ~ 120%. If the CoCT abstracts groundwater from the CFA, the resultant water level wound will increase MAR in the pond by ~ 290%. Thus, retrofitted ponds can contribute to the sustainability of CoCT water supply by topping up the CFA. In climate change scenarios, the retrofitted ponds will harvest less water, 24 – 37% less than the 2005-2022 observed climatic conditions – due to reduced precipitation and increased evapotranspiration.
- Cape Town's residential stormwater quality is poor according to South African water quality standards for recreational use. Further, the stormwater is more polluted compared to developed countries assessed in this study.
- Media from the CFA can reduce the pollutant concentrations of heavy metals from a typical Cape Town residential catchment even at limited vadose depths (0.5 m) without pre-treatment. The pollutant reduction of nutrients varies but improves in the saturated zone (aquifer). An extended residence time of the infiltrated stormwater can improve water quality to levels below the CFA baseline.
- Stormwater harvesting via MAR is a financially viable exercise if done in conjunction with CFA abstraction and is not financially viable if there is a shallow water table (< 1.2 mbgl). However, even when the MAR benefit-cost ratio is negative, the retrofitted ponds can provide other SuDS benefits such as water quality improvement, provision of amenities, and biodiversity.

12.4 Recommendations for further research

This research focused on investigating the feasibility and viability of retrofitting Cape Town's detention ponds and developed a practical approach to retrofitting. It was limited to one full scale operational study, laboratory studies and a hydrological model. There is scope to expand on the approach presented here by addressing the study's limitations. Thus, the following areas are recommended for future research:

- The long-term water quality improvement in the retrofitted pond remains unknown, mainly due to the difficulty of installing instrumentation at the School Pond. This necessitated the use of laboratory experiments and hydraulic models as surrogates. However, laboratory experiments and models can differ from field experiments; thus, it is recommended that the School Pond be monitored for an extended period to better understand the actual treatment efficiencies and hydraulic potential, which can be used to improve models.
- A limitation of this study was the absence of contaminant breakthrough curves. Thus, future research should develop breakthrough curves for heavy metals using media sourced from ponds within the CFA. This will provide knowledge of the loading capacity of these soils and inform optimal maintenance strategies, including the frequency of media washing, in retrofitted ponds.
- This study found that while retrofits increase MAR, the volumetric contribution from one pond does not meaningfully contribute to the CoCT's water demand. However, it is suggested that that similar MAR gains can be achieved in other ponds with similar characteristics in the CFA. This is akin to the Okedi (2019) study; however, Okedi did not investigate the impact of high groundwater tables, and thus their values are possibly overestimates. Thus, the author recommends modelling the impact of such retrofits in larger catchment areas while accounting for surface-groundwater interactions. The effects of a variable aquifer – can be modelled using a coupled two-way model, e.g., a SWMM – MODFLOW model.
- This study primarily focused on MAR, prioritising the valuation of water over ecosystem services and socioeconomic considerations. While attempts to utilise Eurocentric tools such as the CIRIA B£ST were made, the absence of region-specific data hindered their meaningful adaptation within this study's scope. Therefore, future research should address these aspects by deriving and validating region-specific values. This could be achieved using methods such as benefits transfer, stated preference, damage avoidance, and 'willingness to

pay' surveys (Hagedoorn *et al*., 2021; Bockarjova *et al*., 2022). Such research would strengthen the case for retrofitting detention ponds in South Africa.

- It has been proposed that retrofitted ponds can contribute to mitigating the heat island effect via evaporative cooling. It is thus recommended that the broader impact of pond retrofits (with respect to the urban heat island effect) at the neighbourhood and catchment scale be investigated.

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Appendices

Appendix A: Publications and conference presentations

Some of the findings from this study have been submitted to (or published) in international journals and presented at local and international conferences as listed below:

Publications

Tanyanyiwa, C.T., Abrams, A.L., Carden, K., Armitage, N.P., Schneuwly, R., Mguni, P., Byskov Herslund, L. & Mclachlan, J. (2023). Managing Stormwater in South African neighbourhoods: when engineers and scientists need social science skills to get their jobs done. *AQUA - Water Infrastructure, Ecosystems and Society*. (April 8): jws2023173. DOI: 10.2166/aqua.2023.173.

Mclachlan, J., **Tanyanyiwa, C.T**., Schneuwly, R., Carden, K., Armitage, N.P., Abrams, A., Mguni, P. & Herslund, L.B. (2023). Pathways to water resilient South African cities – from mono-functional to multi-functional stormwater infrastructure. *Scientific African*. 20:e01674. DOI: 10.1016/j.sciaf.2023.e01674.

Mguni, P., Byskov Herslund, L., Abrams, A.L., Carden K., **Tanyanyiwa, C.T**, Mclachlan, J., Schneuwly, R., Armitage, N.P. (under review). Scaling deep at the margin: Reflecting on the coproduction of Nature-based Solutions as decolonial praxis in Cape Town. *Npj Urban Sustainability*

Conference Presentations

Tanyanyiwa, C.T, Armitage, N.P., Okedi, J., Schneuwly, R. (2023) – Evaluating the hydrogeological and treatment performance of retrofitted infiltration basins for urban MAR in Cape Town, South Africa. Novatech 11th International conference, July 3 – 7, 2023, Lyon, France. July 3rd – 7th, 2023.

Tanyanyiwa, C.T, Armitage, N.P., Okedi, J. (2022) – Towards urban managed aquifer recharge in Cape Town's retrofitted multifunctional stormwater ponds. *17th Water Institute of Southern Africa Biennial Conference*, September 28 - 30, 2022, Sandton, South Africa. September 28th – 30th, 2022.

Tanyanyiwa, C.T, Abrams, A.L., Carden K., Armitage, N.P., Schneuwly, R., Mguni, P., Byskov Herslund, L., Mclachlan, J. (2022) – Managing stormwater in South African neighbourhoods: When engineers and scientists need social science skills to get their jobs done. *World Water Congress & Exhibition,* September 11 – 15, Copenhagen, Denmark. 11th – 15th September 2022.

Tanyanyiwa, C.T, Armitage, M.P, Dzurume, T (2021). Cape Town's Stormwater Quality – A GIS-aided case study of the Zeekoe Catchment. *15th International Conference on Urban Drainage*, October 25 - 28, 2021, Melbourne, Australia. 25th – 28th October 2021.

Appendix B: Electrical resistivity survey method

B1. Introduction

Utilising direct current (DC) resistivity techniques, also known as electrical resistivity, 2D resistivity imaging, or electric resistivity tomography (ERT), involves assessing the earth's resistivity. This is achieved by introducing a DC signal into the ground and subsequently measuring the resulting potentials that arise (Figure B–1). The electrical properties of the sub-surface are derived from this data.

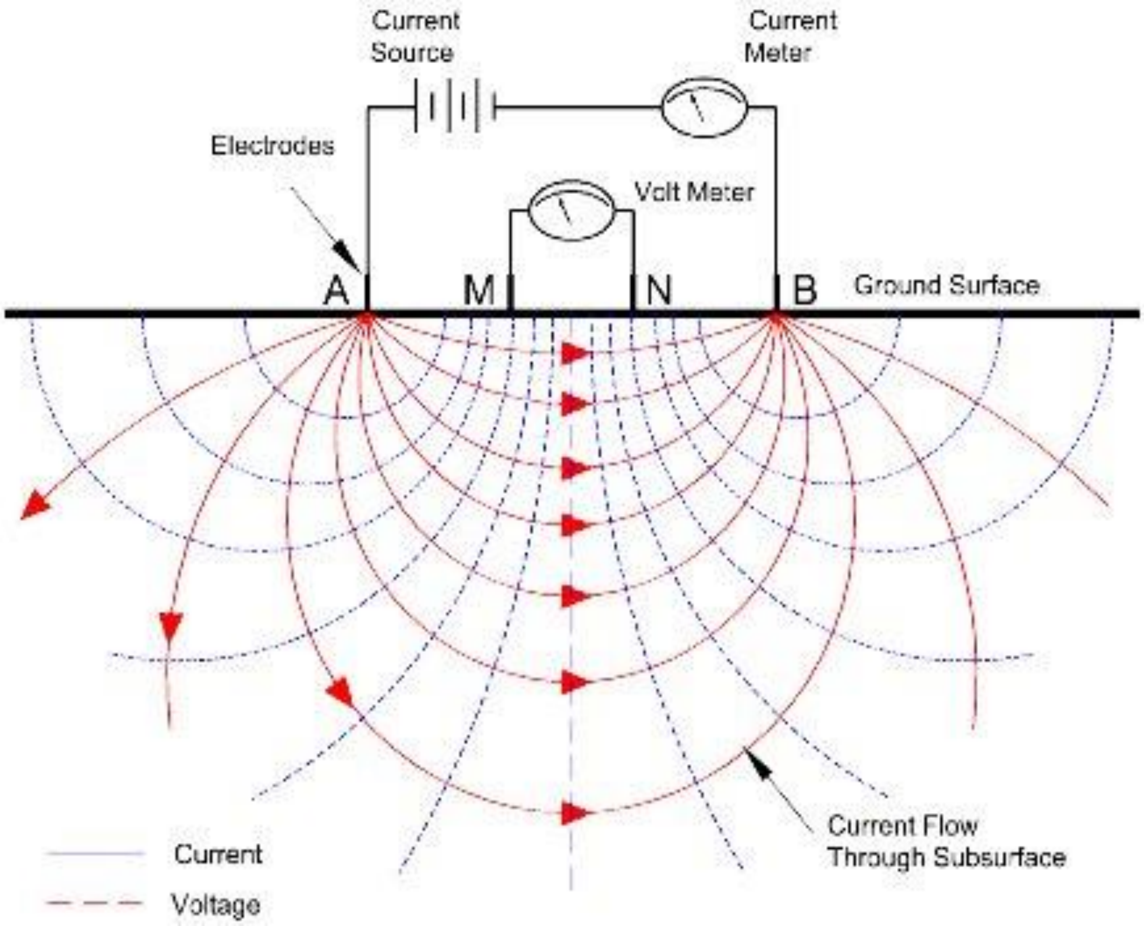

Figure B–1: Schematic of the resistivity concept
(Marescot *et al*., 2008)

The ER varies across geological materials owing to changes in the water content and dissolved ions in groundwater. This variability allows resistivity assessments to differentiate zones with distinct electrical characteristics akin to diverse geological strata. It is synonymous with the specific resistance, which is the inverse of the conductivity. Common mineral-based soils and rocks exhibit high resistivity when dry. Consequently, resistivity depends on water quantity, quality, fractures, and tropical weathering. This variability is often confined to a specific geological region. Within soil or rock types, resistivity alterations mirror shifts in physical attributes. For instance, the lower resistivities of sandstone and limestone suggest saturated pore spaces, whereas the highest values indicate consolidated rocks or dry formations. Low resistivity can arise from saturated sand, gravel, or sedimentary rock pore spaces.

Fresh crystalline rocks typically exhibit high resistivity even if they contain conductive ore minerals. However, weathering often yields conductive, clay-rich saprolite. Variability within geological material types mandates the calibration of resistivity data using geological documentation, such as surface mapping, test pit exposures, or drilling. This requirement holds for all geophysical approaches.

The resistivity is influenced by the degree of saturation. Above the groundwater level, the resistivity surpasses that below, assuming material consistency. Consequently, this technique gauges the water table depth when it is clearly distinguished. However, significant fine-grained content can lead to water above the groundwater surface, held by hygroscopic and capillary forces, to overpower the electrical behaviour of the material. Pore water resistivity depends on the ion concentration, ion type, and temperature. Clay minerals notably impact sediment and weathered rock resistivity. These minerals act as conductive particles, facilitating ion and water molecule absorption and release through ion-exchange processes.

B2. Field survey method

Three ERT-2D traverses were conducted during March 2022 at the School Pond area near Philippi, Cape Town. Traverse coordinates were obtained with a handheld GPS unit. An Abem SAS 1000 Terrameter and ES 10-64 switching unit were used in the field survey. Traverse-1 was conducted with the Wenner measuring protocol and a 4m electrode spacing which yields an investigation depth of 24m. Traverses 2 and 3 were conducted with the Wenner 32SX measuring protocol and a 2m electrode spacing. This measuring protocol and electrode spacing yield a maximum investigation depth of approximately 12 m.

B3. Data Analysis

The RES2Dinv (GEOTOMO) version 3.52 inversion software was used to derive true subsurface resistivity values from the measured apparent resistivities along the survey line (traverse). This inversion method relies on the smoothness-constrained least-squares approach method (deGroot-Hedlin & Constable, 1990; Sasaki, 1992). A distinctive advantage of this method is its flexibility to tailor the damping factor and flatness filters to various data types.

Appendix C: The state of Cape Town's stormwater quality

C1. Description the Diep and Sand River catchments

The Diep River catchment is located in the northeastern section of Cape Town and extends from the northern boundary of Cape Town with Atlantis to the northwest, Paarl to the east, Parrow to the south, and Bloubergstrand to the southwest. The catchment covers a total area of 1495 km^2. However, a section of the catchment extends outside the Cape Town boundary towards Malmesbury to the northeast of Atlantis and was not included in the analysis; thus, the area assessed in this study covers 726 km^2. The Diep River, which drains the catchment, has one major tributary: the Mosselbank River (Figure C-1).

The catchment has 21 sampling points along its water bodies (Figure C-1). Agricultural land use dominates the catchment area, accounting for 65% of the total area. Natural vegetation accounts for 24% of the area, while residential land use covers 5.3% – 4.7% formal and 0.6% informal residential land use. Wetlands occupy 2.5% of the catchment, while other land uses (industrial land, commercial, water bodies, barren, parks) occupy 3.2%.

The Sand River catchment is bound by the Muizenberg Mountains to the South, Silvermine Dam to the South-West, Constantia Berg to the West, Cecilia Forest to the North-West, Wynberg to the North and the Princess Vlei to the East. The Keysers and Sand rivers are the longest watercourses in the catchments and converge at the Zandvlei (Figure C-2).

The catchment has 53 sampling points; however, only 23 points from the dataset had complete records of stormwater water quality parameters as measured by the CoCT (Figure C-2). The catchment is highly urbanised and its land use comprises: residential (formal and informal), commercial, industrial, agricultural, naturally vegetated, waterbodies, and wetlands. Formal residential areas comprise most of the catchment's land use composition (42%) – followed by naturally vegetated areas (31%). Land-use catergorised as commercial covers 9.3% of the area while industrial activities are limited to 1.4% of the catchment area. Barren lands and informal residential areas are minimal – 0.8% and 0.7%, respectively. While other land uses were evaluated, this study presents the characteristics from residential areas as 84% the detention ponds over the CFA are in residential areas.

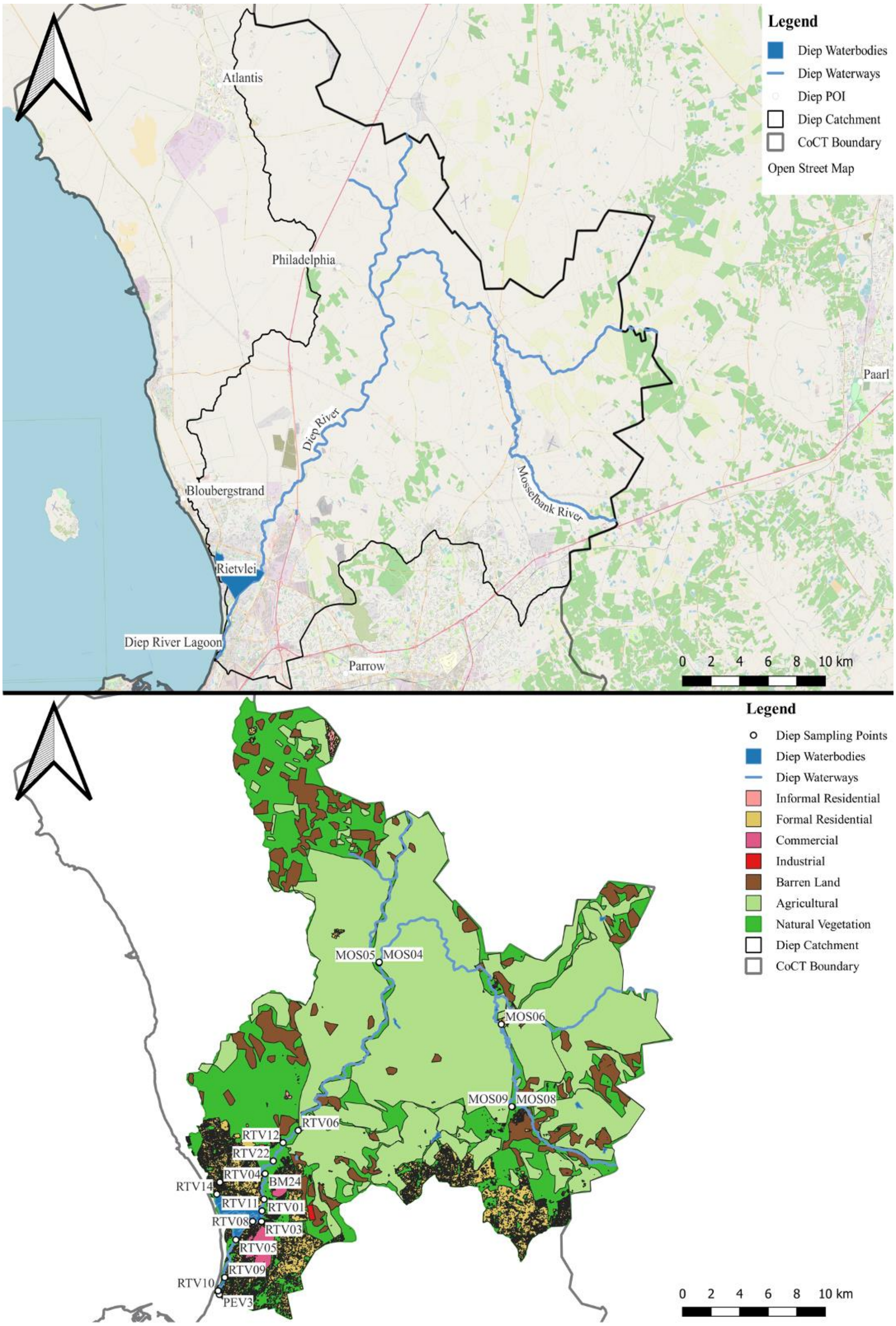

Figure C-1: Diep River catchment locality plan (top); LULC composition and sampling points (bottom)

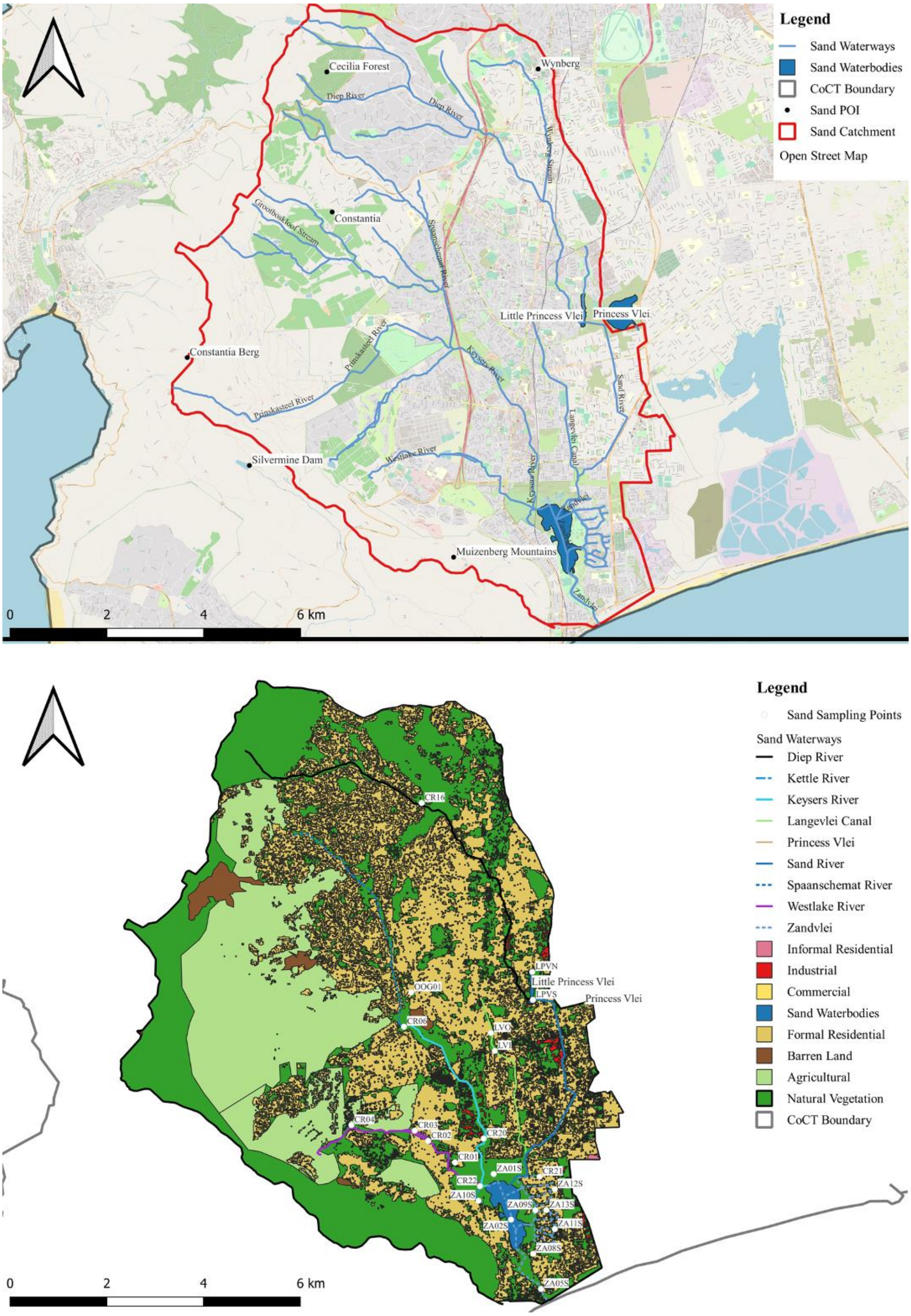

Figure C-2: Sand River catchment locality plan (top); LULC composition and sampling points (bottom)

C2. Evaluated stormwater characteristics

C2.1 Diep River Catchment

Table C-1 displays the water quality characteristics of residential land use in the Diep catchment. The median values indicate generally poor stormwater quality, with Ammonia, *E. coli*, and TP median values classified as unacceptable according to South African recreational-use standards. Additionally, the mean and 95th percentile values for Ammonia, *E. coli*, TN, and TP are categorised as poor to unacceptable. These findings suggest potential contamination from agricultural activities such as fertilizer use – given the catchment's predominantly agricultural nature, and the possibility of sewer pipe leaks or bursts (Engel, 2022; Human, 2022) resulting in higher mean and 95th percentile concentrations. These findings indicate that stormwater from the catchment may be unsuitable for direct use via MAR without pre-treatment.

Table C-1: Stormwater quality characteristics for residential land use in the Diep catchment

Parameters	Diep River (n = 301)			Evaluation of the median value
	Mean	Median	95th percentile	
Ammonia (mg/L)	3.98	0.79	26.4	Unacceptable
DO (mg/L)	9.99	6.49	12.55	Fair
E. coli (CFU/100 mL)	2.12E+05	4.10E+03	4.30E+05	Unacceptable
EC (mS/m)	938.1	398.0	4183.0	–
pH	7.79	7.80	8.40	Natural
Temperature (°C)	18.9	18.8	25.0	Target
TN (mg/L)	5.3	2.37	28.2	Fair
TP (mg/L)	1.50	0.73	5.82	Unacceptable
TSS (mg/L)	28.76	11.00	80.0	–

The mean values of the five parameters in the Diep River were compared with those of studies from five countries (Table C-2). The Diep River had the highest mean concentrations of Ammonia and TP among the compared datasets. This is likely due to the predominant agricultural land use in the Diep River catchment, resulting in the wash-off of fertilisers into the system, which are then picked up in residential sampling points along the river. However, the mean *E. coli* count was lower than that of India and the USA. Moreover, the Diep River exhibited the second lowest

TSS values but had the second-highest TN concentration. These findings suggest that stormwater from residential areas in the Diep River catchment is more contaminated compared to most of the evaluated countries.

Table C-2: Comparing Diep River residential stormwater with selected countries

	Ammonia (mg/L)	*E. coli* (CFU/100 mL)	TN (mg/L)	TP (mg/L)	TSS (mg/L)
Diep River	3.98	2.12E+05	5.3	1.5	28.76
India	–	4.44E+06	9.2	0.7	27.8
Singapore	–	–	1.2	0.1	31.9
Malaysia	0.14	–	–	0.73	37
USA	0.3	7.00E+02	1.4	–	49
The Netherlands	–	1.90E+04	1.9	0.4	17

C2.2 Sand River Catchment

Table C-3 presents the mean, median, and 95th percentile characteristics of residential land use in the Sand River Catchment, along with the evaluation of stormwater quality based on South African recreational-use standards. The analysis of the water quality from the residential informs that only the TP median values are classified as poor – from the parameters analysed. However, the mean and 95th percentile values for Ammonia, *E. coli* and TP are classified as unacceptable

A comparative analysis was conducted on the mean stormwater quality characteristics of India, Singapore, Malaysia, the USA, and the Netherlands, contrasted with those obtained from residential areas within the Sand River catchment (Table C-4). The ammonia concentration in the Sand River catchment residential area was higher than in Malaysia and the USA. However, it displayed a lower *E. coli* count than India and the Netherlands. Arora & Reddy (2013) linked the high *E. coli* counts in the Indian dataset to public urination and open defecation in residential areas, whereas the Netherlands study (Boogaard, 2015) lacked such an explanation. The Sand River catchment had higher TN concentrations than the other countries, except India. Comparatively, the TP concentration in the Sand River catchment was lower than that in all countries except Malaysia, and it displayed the lowest TSS concentrations. These results suggest that the stormwater quality in the Sand River catchment is more polluted than that of most developed countries assessed but generally better than that of other developing countries examined in this study.

Table C-3: Stormwater quality characteristics for residential land use in the Sand catchment

Parameters	Sand River (n = 105)			Evaluation of the median value
	Mean	Median	95th percentile	
Ammonia (mg/L)	0.34	0.09	1.44	Fair
DO (mg/L)	7.36	7.30	11.1	Target
E. coli (CFU/100 mL)	1.55E+03	2.00E+02	8.84E+03	Fair
EC (mS/m)	61.0	63.0	86.8	–
pH	7.85	7.70	8.98	Natural
Temperature (°C)	18.4	18.3	24.7	Target
TN (mg/L)	3.05	2.63	6.90	Fair
TP (mg/L)	0.19	0.16	0.43	Poor
TSS (mg/L)	12.1	9.00	30.4	–

Table C-4: Comparing Sand River residential stormwater with selected countries

	Ammonia (mg/L)	*E. coli* (CFU/100 mL)	TN (mg/L)	TP (mg/L)	TSS (mg/L)
Sand River	0.34	1.55E+03	3.05	0.19	12.1
India	–	4.44E+06	9.2	0.7	27.8
Singapore	–	–	1.2	0.1	31.9
Malaysia	0.14	–	–	0.73	37
USA	0.3	7.00E+02	1.4	–	49
The Netherlands	–	1.90E+04	1.9	0.4	17

Appendix D: School Pond maintenance plan

Name of Pond:	School Pond
Location:	Rondevlei Park, Mitchells Plain, Cape Town
Coordinates:	-34.034047, 18.584698
Type of retrofit:	Infiltration swale for MAR and WQ improvement

Remedial action	Frequency	Accountable person/s	Key Performance Indicator	Contact
Clearing up of litter at inlets and forebay	Monthly and after every storm	CoCT and Pond Committee/School Environmental Club	Minimal litter in pond	Private information
Clearing of weeds in riprap	Every second month	Pond Committee	No weeds in riprap	
Maintenance of riprap	Monthly in winter	CoCT and the Pond Committee	No weed in riprap and stones rearranged	
Mowing grass on the slopes and up to the swampy area	Every 3 months	CoCT Recreation and Parks Department	Short grass in those areas throughout the year	
Trimming the grass and shrubs in swampy area	Once a year	CoCT Recreation and Parks Department	Shrub height less than 1 m	
Clearing outlet canal	Monthly in winter	CoCT Catchment, Stormwater & River Management	No litter in canal	
Cleaning litter traps and reinforcing/ replacing rocks	Monthly in winter	Pond Committee/School Environmental Club	Minimal litter on litter traps and forebay	
Plugging Mole holes	Once a year before winter	CoCT and Pond Committee	No short circuiting in infiltration swale	
Infiltration test	Annual	CoCT Catchment, Stormwater & River Management	Rate > 100 mm/hr*	

* Payne *et al*. (2015)

Appendix E: Objective function equations

Table E-1: Objective functions for model calibration

Integral Square Error (ISE)	$ISE = \frac{\sqrt{\sum(y^i{}_{obs} - y^i{}_{comp})^2}}{\sum y^i{}_{obs}}$	**Equation E-1**
Nash–Sutcliff Efficiency (NSE)	$NSE = \frac{\sum(y^i{}_{obs} - y^i{}_{comp})^2}{\sum(y^i{}_{obs} - \overline{y^i{}_{obs}})^2}$	**Equation E-2**
Coefficient of Determination (R^2)	$R^2 = \left(\frac{\sum(y^i{}_{obs} - \overline{y^i{}_{obs}})(y^i{}_{comp} - \overline{y^i{}_{comp}})}{\sqrt{(y^i{}_{obs} - \overline{y^i{}_{obs}})^2 \sum(y^i{}_{comp} - \overline{y^i{}_{comp}})^2}}\right)^2$	**Equation E-3**
Standard Error Estimation (SEE)	$SEE = \frac{\sqrt{\sum_{i=1}^{n}(y^i{}_{obs} - y^i{}_{comp})^2}}{n}$	**Equation E-4**
Least Square Error (LSE)	$LSE = \sum(y^i{}_{obs} - y^i{}_{comp})^2$	**Equation E-5**
Root Mean Squared Error (RMSE)	$ISE = \sqrt{\frac{\sum(y^i{}_{obs} - y^i{}_{comp})^2}{n}}$	**Equation E-6**

Where, $y^i{}_{obs}$ is the observed value for the i^{th} observation, $y^i{}_{comp}$ is the computed/simulated value for the i^{th} observation, $\overline{y^i{}_{obs}}$ is the mean of observed values, and n is the number of observations.

Appendix F: Community engagement assessment tool

Figures F-1, F-2 and F- 3 show the community engagement tool by NCEI (2018)

Q: WHAT KIND OF RELATIONSHIP DO YOU HAVE WITH COMMUNITY MEMBERS?

OUTREACH	UNSURE WHICH WE ARE DOING	DOING PRIMARILY OUTREACH	BEGINNING TO TALK ABOUT MOVING TO CE	WORKING TOWARD CE	DOING CE	COMMUNITY ENGAGEMENT
• Relationships are primarily TRANSACTIONAL, for the purpose of completing a project.						• Relationships are FOUNDATIONAL, continually built between and among people and groups. Staff/institutions continually build the relationships they need to know their community.
• Relationships are often NOT INCLUSIVE of all racial or cultural groups in the community.						• Relationships reflect the DIVERSITY within the community.
• Relationships can be LIMITED to a few community members, often giving influence to those with the loudest voices.						• Relationships are built not just with current leaders, but also with people with an interest and/or POTENTIAL TO BE LEADERS.
• Relationships are SHORT-TERM, so staff have to rebuild them as other projects or issues come up.						• Relationships are transformational and LONG-TERM, so community leaders/members can engage in projects and issues as they come up.

Q: WHY ARE YOU ENGAGING PEOPLE?

OUTREACH	UNSURE WHICH WE ARE DOING	DOING PRIMARILY OUTREACH	BEGINNING TO TALK ABOUT MOVING TO CE	WORKING TOWARD CE	DOING CE	COMMUNITY ENGAGEMENT
• To accomplish a project or a SPECIFIC GOAL defined by the organization.						• To create space for people to CONNECT, RAISE CONCERNS, BUILD POWER and ACT IN THEIR OWN INTERESTS.
• To SEEK BUY-IN OR APPROVAL of something the organization has already planned.						• To CREATE SPACE for the community's assets to be recognized and utilized.

Figure F-1: Questions 1 and 2 of the assessment tool

Q: HOW ARE YOU GETTING PEOPLE INVOLVED? WHEN?						
OUTREACH	UNSURE WHICH WE ARE DOING	DOING PRIMARILY OUTREACH	BEGINNING TO TALK ABOUT MOVING TO CE	WORKING TOWARD CE	DOING CE	COMMUNITY ENGAGEMENT
• Primary activities with community include FLYERING, SURVEYS, FOCUS GROUPS, WORKSHOPS, etc.						• Primary activities with community include LISTENING SESSIONS, ONE-TO-ONE MEETINGS, CELEBRATIONS, LEADERSHIP DEVELOPMENT, COMMUNITY-BUILDING PROJECTS, etc.
• Information is given or feedback is requested AFTER A PROJECT IS PLANNED.						• Planning is done WITH THE COMMUNITY from the beginning

Q: HOW DO IDEAS GET GENERATED?						
OUTREACH	UNSURE WHICH WE ARE DOING	DOING PRIMARILY OUTREACH	BEGINNING TO TALK ABOUT MOVING TO CE	WORKING TOWARD CE	DOING CE	COMMUNITY ENGAGEMENT
• STAFF/ INSTITUTIONS GENERATE IDEAS they think the community will support.						• Staff/institutions SUPPORT COMMUNITY MEMBERS in generating their own ideas.
• Staff/institutions generate SOLUTIONS TO A PROBLEM they have defined.						• Staff/institutions engage in CONTINUAL SELF-REFLECTION to respond to and incorporate people's ideas, feedback, talents, and challenges into the work.

Figure F-2: Questions 3 and 4 of the assessment tool

Q: HOW DO YOUR ORGANIZATIONAL POLICIES AND STRUCTURES SUPPORT ENGAGEMENT?						
OUTREACH	UNSURE WHICH WE ARE DOING	DOING PRIMARILY OUTREACH	BEGINNING TO TALK ABOUT MOVING TO CE	WORKING TOWARD CE	DOING CE	COMMUNITY ENGAGEMENT
• The organizational culture is primarily focused on OBTAINING SPECIFIC OUTCOMES.						• The organizational culture is focused on learning and it values EMERGENT AND LONG-TERM OUTCOMES.
• Board and staff may NOT REPRESENT the community.						• Board and staff REFLECT the community.
• The organization ADHERES TO WAYS OF OPERATING that reflect the DOMINANT CULTURE, such as using Robert's Rules for meetings, prioritizing staff to speak, etc.						• The organization CREATES SPACE FOR DIFFERENT CULTURAL WAYS, such as offering cultural foods and social spaces/ times, giving elders a special role, etc.
• Racism and power may not be discussed or may be DEALT WITH SUPERFICIALLY.						• The organizational culture supports discussions to UNDERSTAND AND DISMANTLE structural racism, to help heal historical trauma and to claim individual and community power.
• The organization adheres to ORGANIZATION-DRIVEN policies and structures.						• The organization demonstrates a willingness to revisit organizational policies and structures to RESPOND TO COMMUNITY NEEDS AND IDEAS.

Figure F-3: Question 5 of the assessment tool

Appendix G: Column treatment perfomance

G1. Summary of p-values

Table G-1: Summary of p-values from ANOVA tests for organics

Organics	Column Pair	Pairwise p-values		
		1.5 m vadose zone	1.0 m vadose zone	0.5 m vadose zone
TN	A	0.740	0.838	0.963
	B	0.063	0.324	0.775
	C	0.669	0.745	0.828
	D	0.980	0.403	0.739
TP	A	0.051	0.076	0.359
	B	0.449	0.742	0.387
	C	0.698	0.749	0.890
	D	0.793	0.969	0.674
TOC	A	0.139	0.090	0.747
	B	0.808	0.870	0.918
	C	0.570	0.936	0.948
	D	0.980	0.021	0.310

Key:

A – CFA media dosed with typical SSW that contains added carbon.

B – CFA media dosed with highly contaminated SSW that contains added carbon.

C – CFA media dosed with highly contaminated SSW without carbon added.

D – Silica sand dosed with highly contaminated SSW that contains added carbon.

Table G-2: Summary of p-values from ANOVA tests for all heavy metals

Heavy metals	Column Pair	Pairwise p-values		
		1.5 m vadose zone	1.0 m vadose zone	0.5 m vadose zone
Chromium	A	0.209	0.542	0.889
	B	0.835	0.122	0.716
	C	0.710	0.191	0.901
	D	0.925	0.369	0.803
Copper	A	0.209	0.542	0.889
	B	0.835	0.122	0.716
	C	0.710	0.191	0.901
	D	0.925	0.369	0.803
Lead	A	0.430	0.250	0.149
	B	0.894	0.620	0.640
	C	0.555	0.944	0.980
	D	0.819	0.853	0.860
Nickel	A	0.242	1.000	0.334
	B	0.393	1.000	0.847
	C	0.133	1.000	1.000
	D	0.332	1.000	1.000
Zinc	A	0.362	0.736	0.693
	B	1.000	0.192	0.161
	C	0.889	0.512	0.319
	D	0.510	0.785	0.614

Key:

A – CFA media dosed with typical SSW that contains added carbon.

B – CFA media dosed with highly contaminated SSW that contains added carbon.

C – CFA media dosed with highly contaminated SSW without carbon added.

D – Silica sand dosed with highly contaminated SSW that contains added carbon.

G2. Vadose zone treatment performance graphs

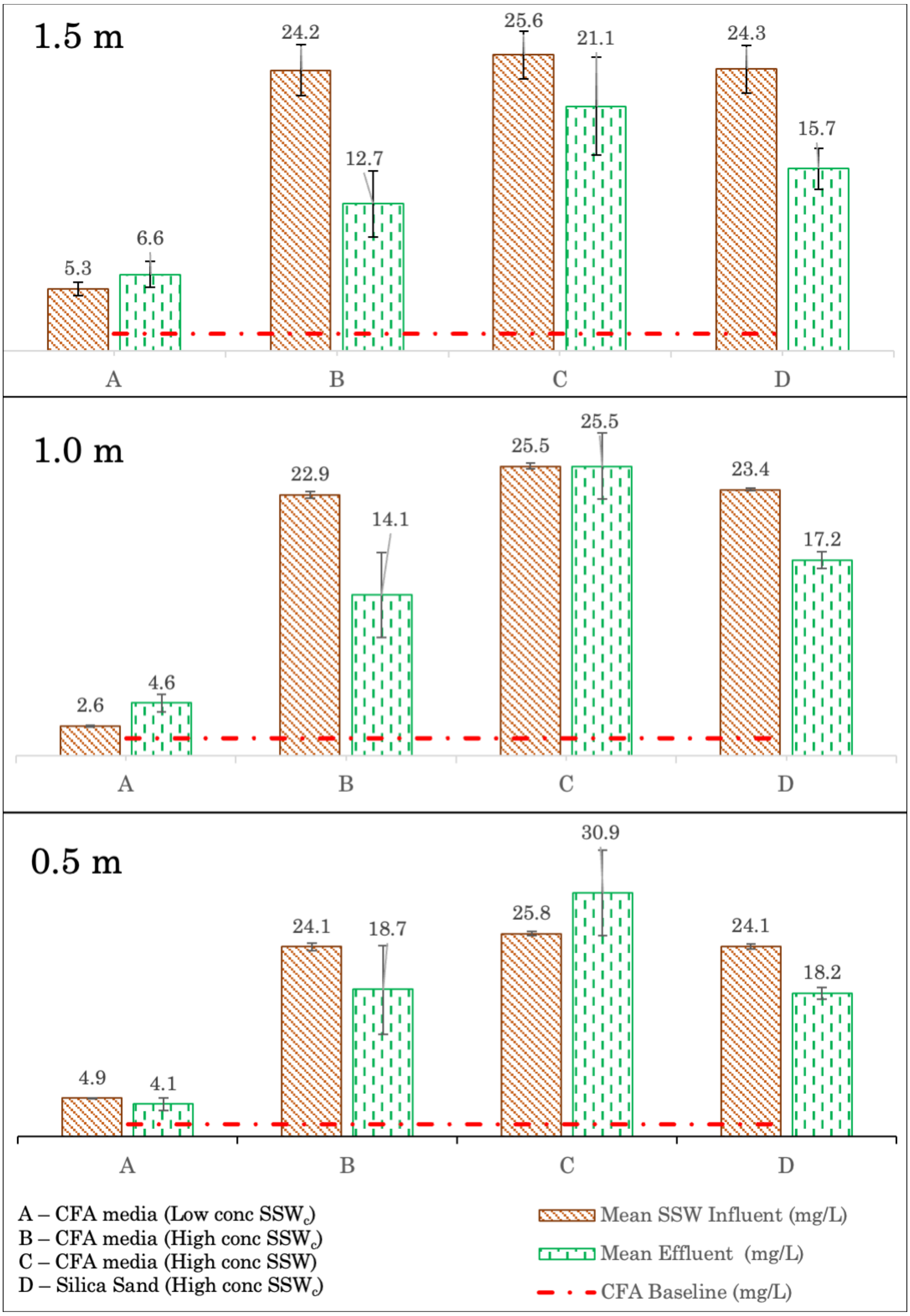

Figure G–1: Treatment performance for TN at 1.5, 1.0 and 0.5 m vadose zone depths

Figure G–2: Treatment performance for TP at 1.5, 1.0 and 0.5 m vadose zone depths

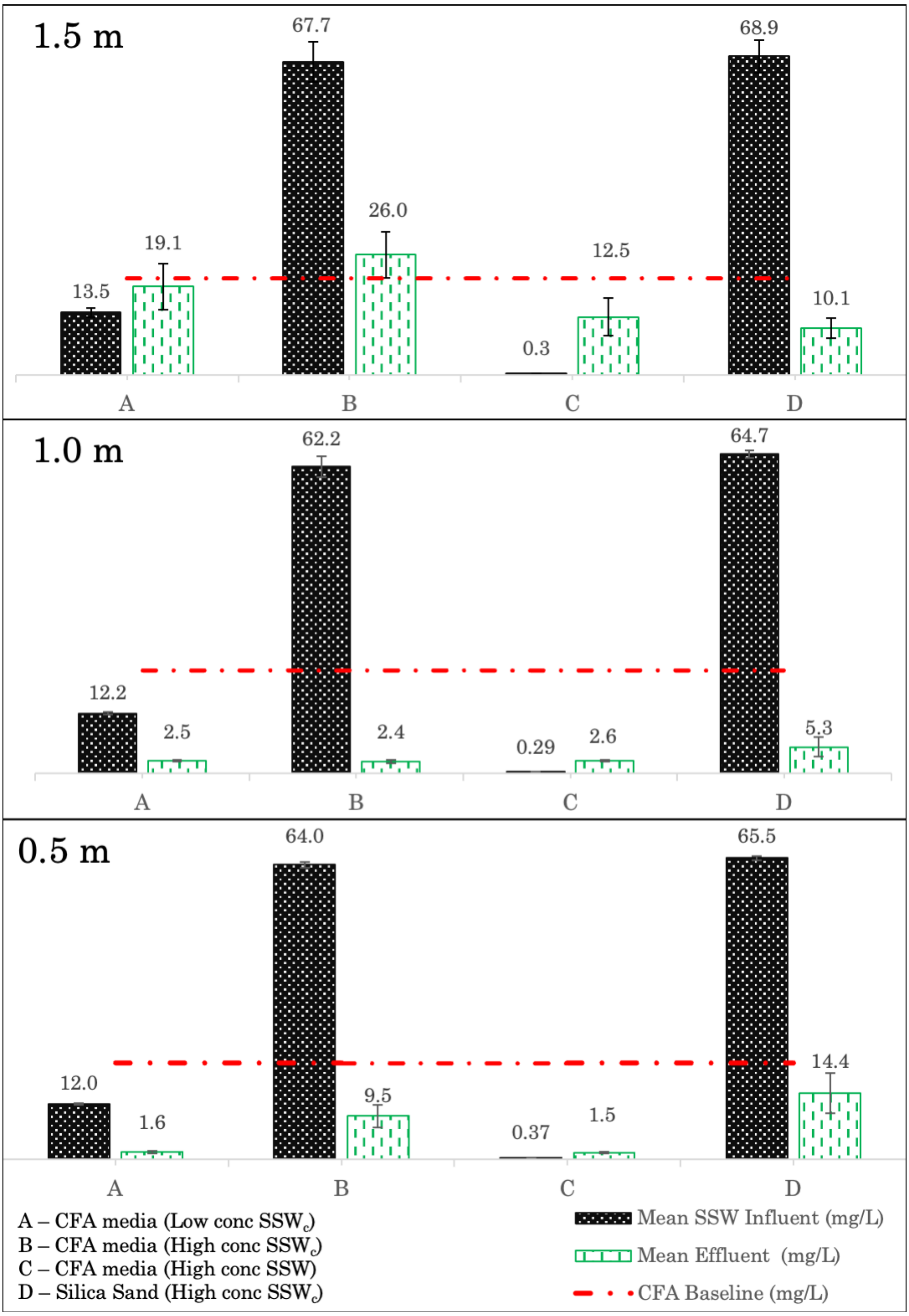

Figure G–3: Treatment performance for TOC at 1.5, 1.0 and 0.5 m vadose zone depths

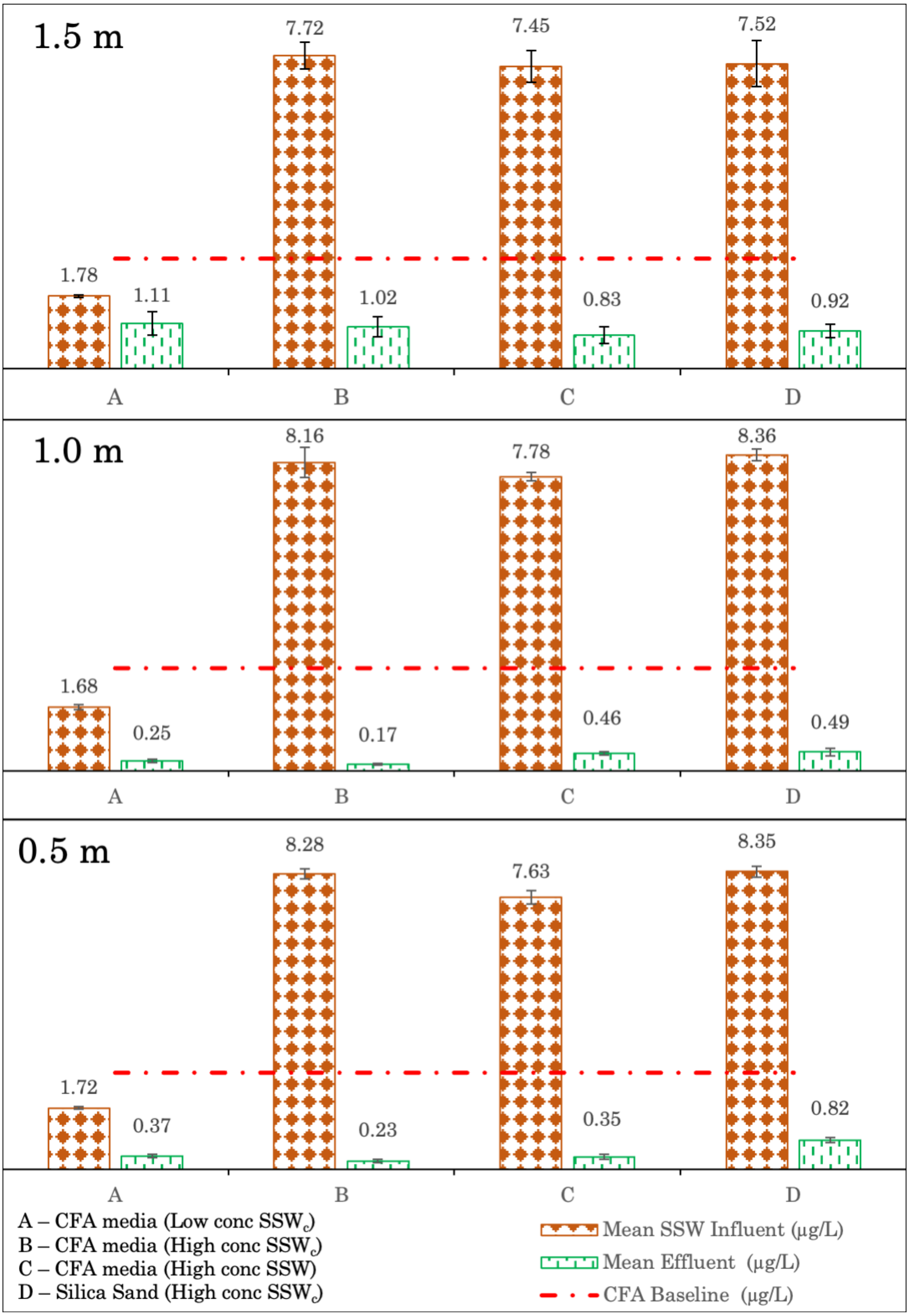

Figure G–4: Treatment performance for Cr at 1.5, 1.0 and 0.5 m vadose zone depths

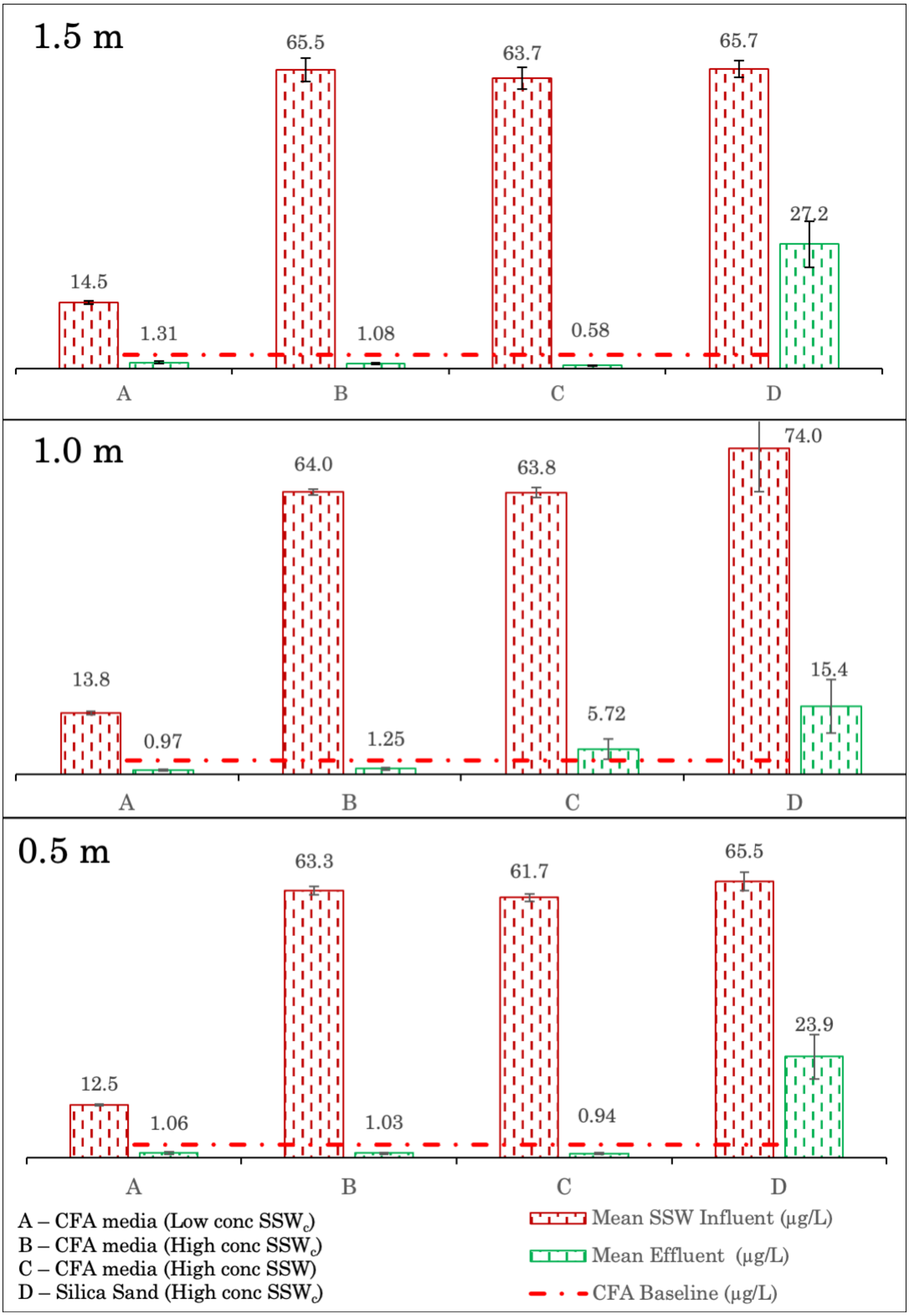

Figure G–5: Treatment performance for Cu at 1.5, 1.0 and 0.5 m vadose zone depths

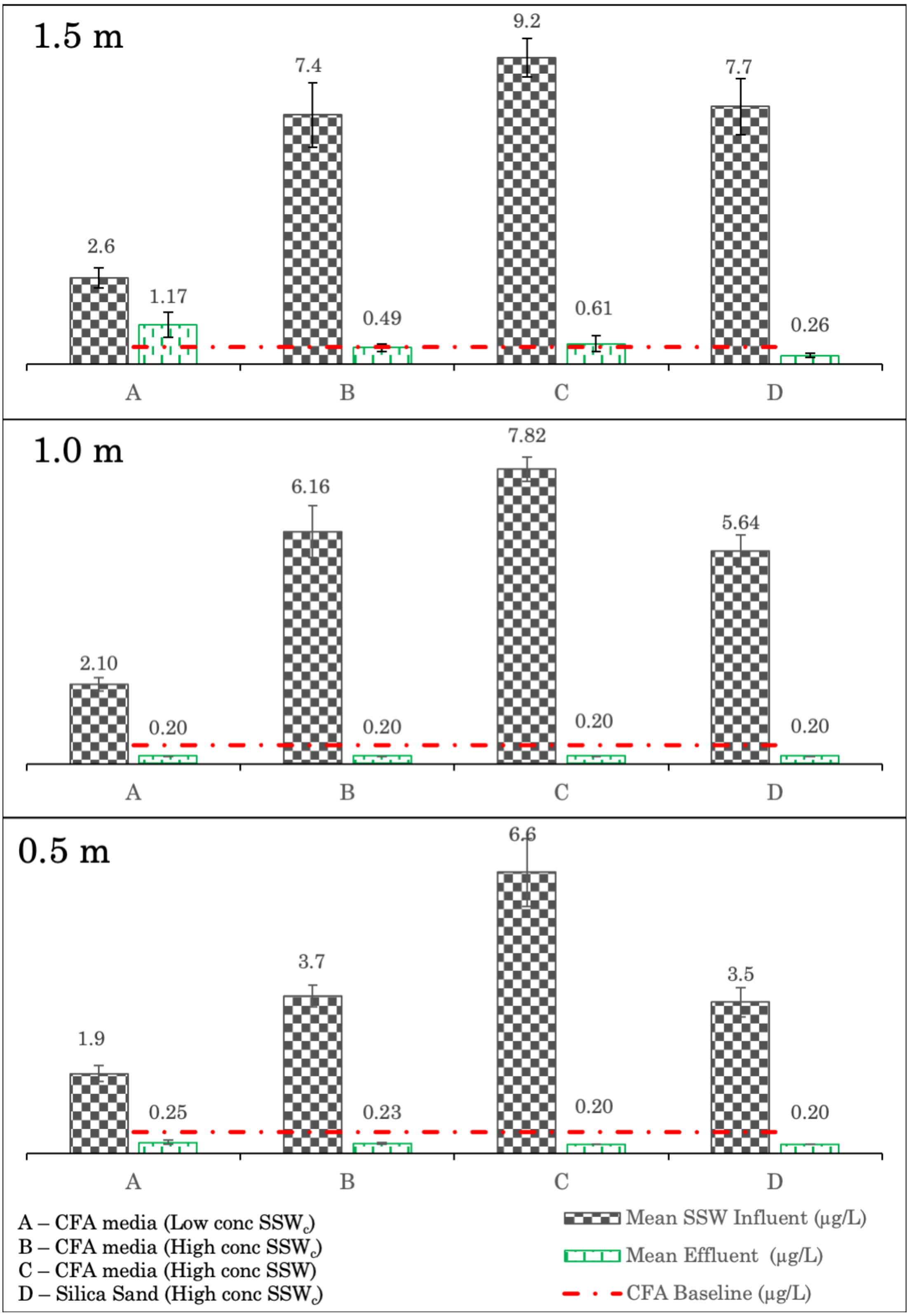

Figure G–6: Treatment performance for Pb at 1.5, 1.0 and 0.5 m vadose zone depths

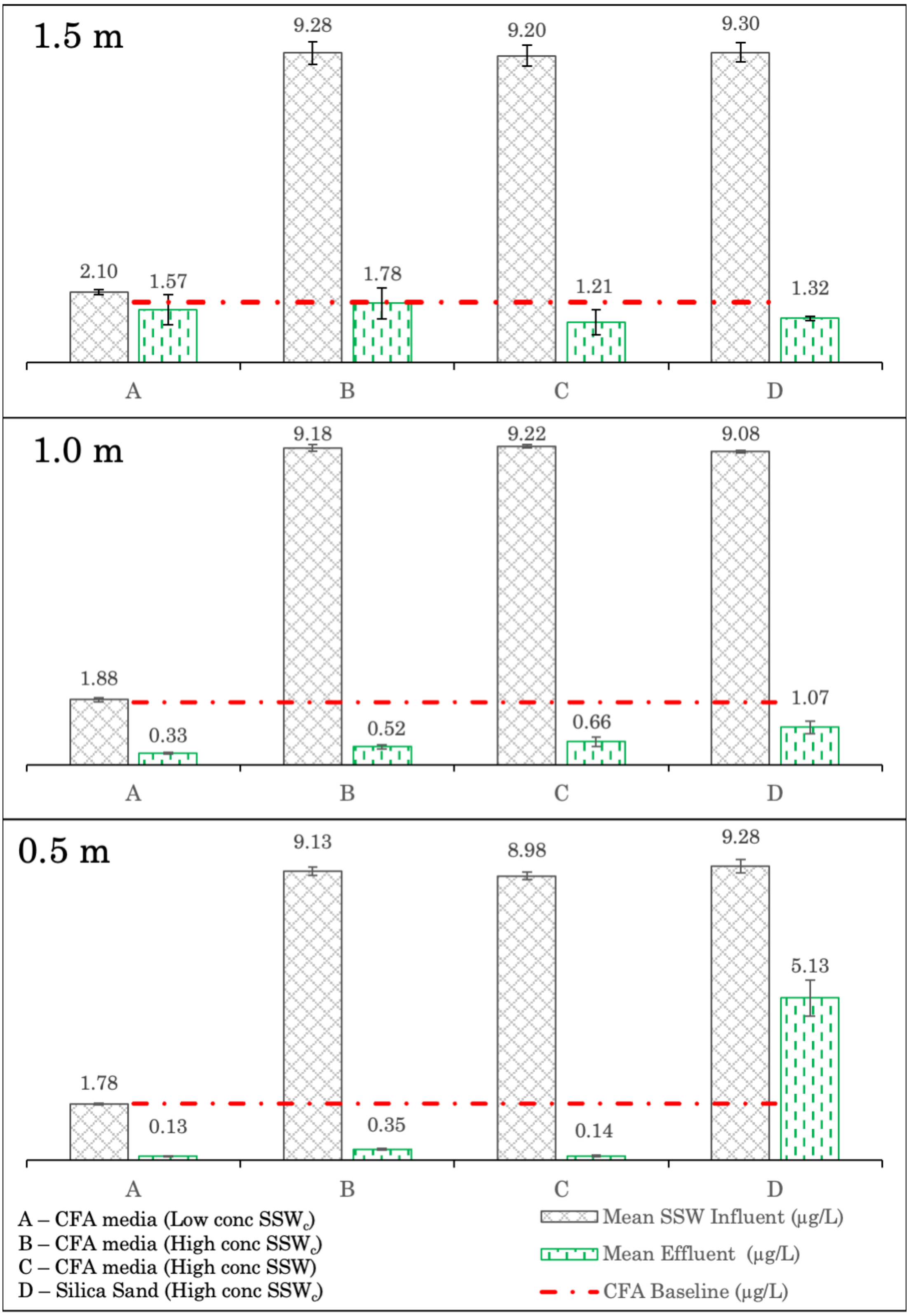

Figure G–7: Treatment performance for Ni at 1.5, 1.0 and 0.5 m vadose zone depths

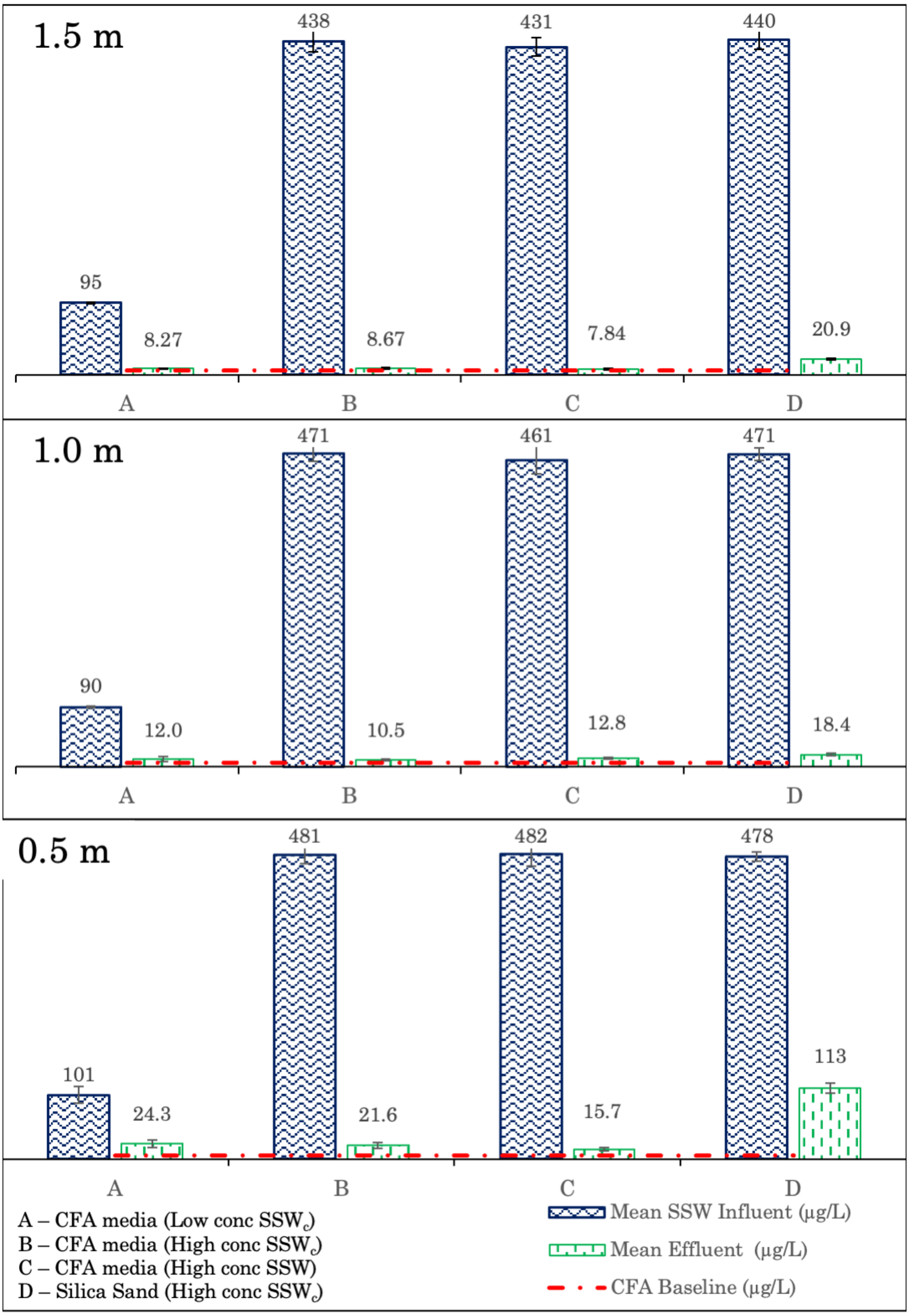

Figure G–8: Treatment performance for TOC at 1.5, 1.0 and 0.5 m vadose zone depths

G3. Saturated zone treatment performance graphs

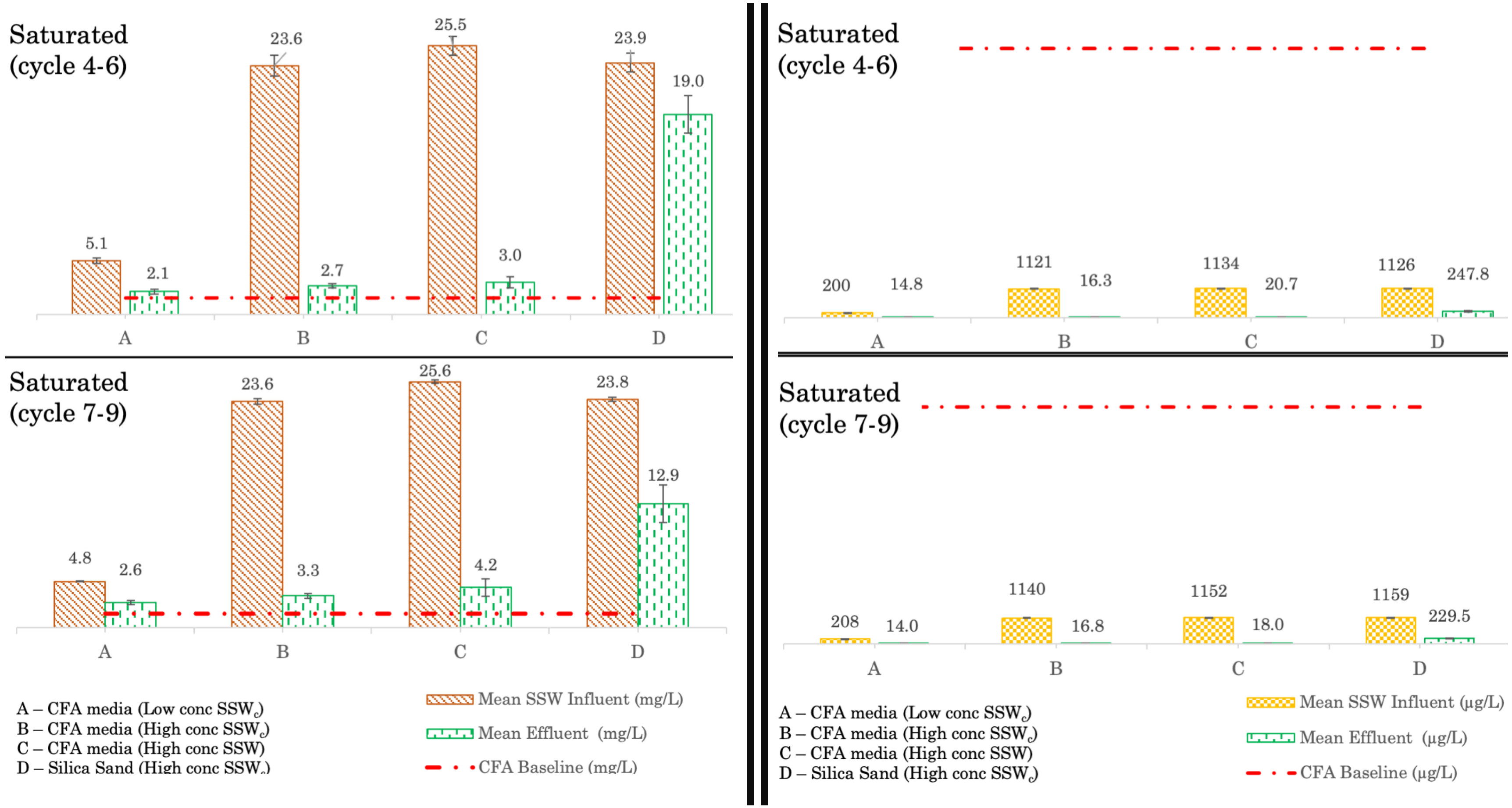

Figure G–9: Treatment performance for TN (left) and TP (right) in saturated zone

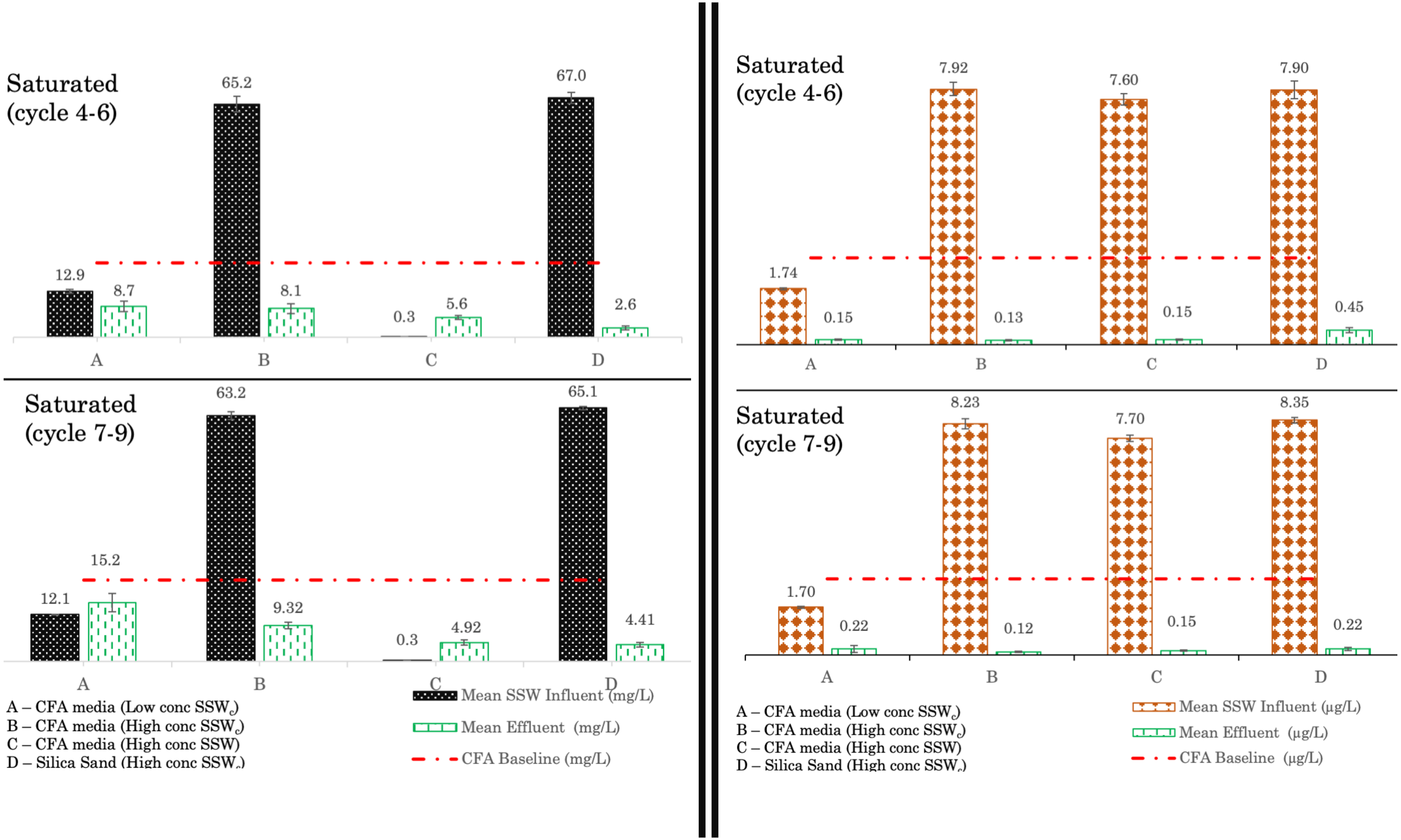

Figure G–10: Treatment performance for TOC (left) and Cr (right) in saturated zone

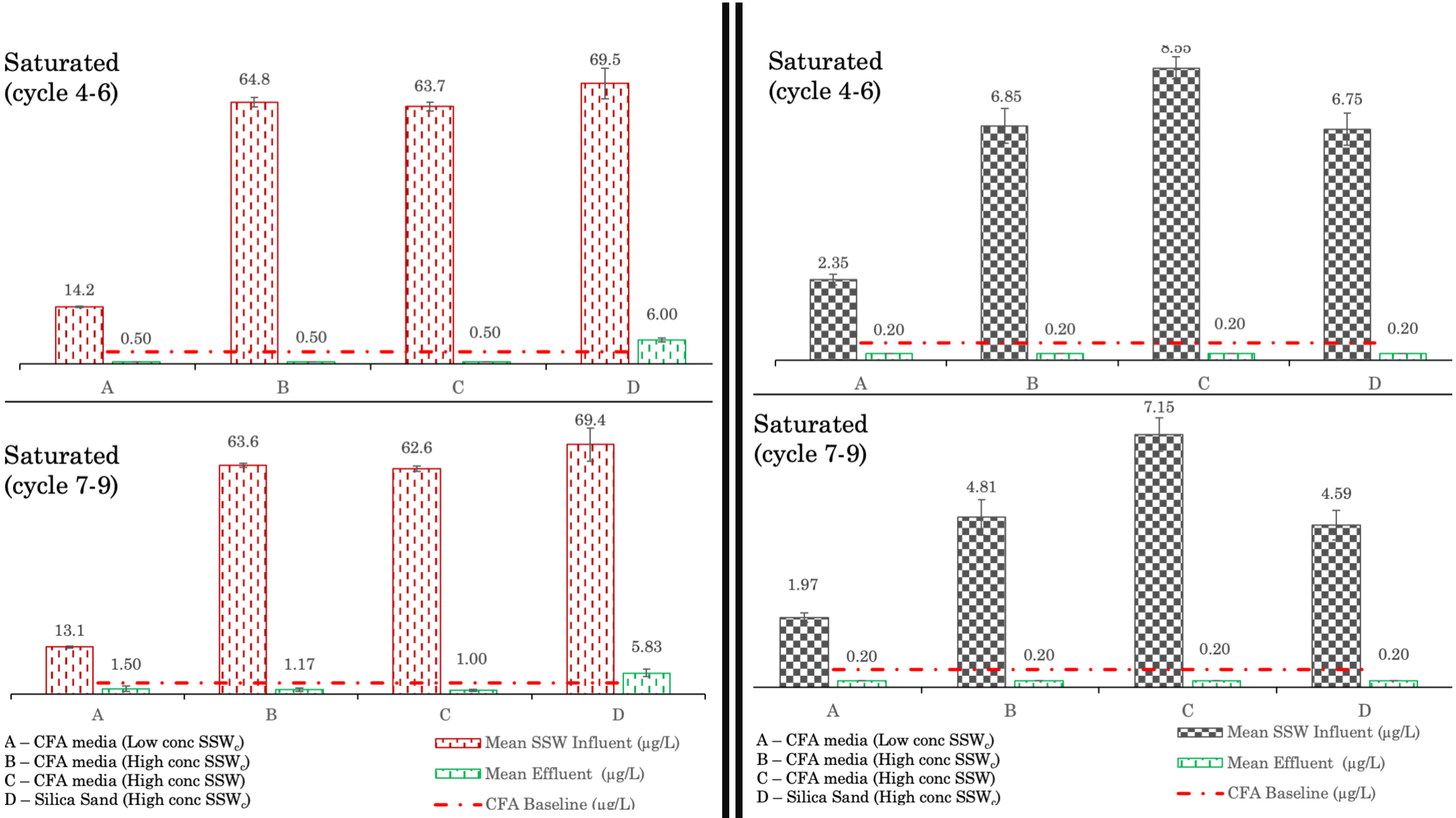

Figure G–11: Treatment performance for Cu (left) and Pb (right) in saturated zone

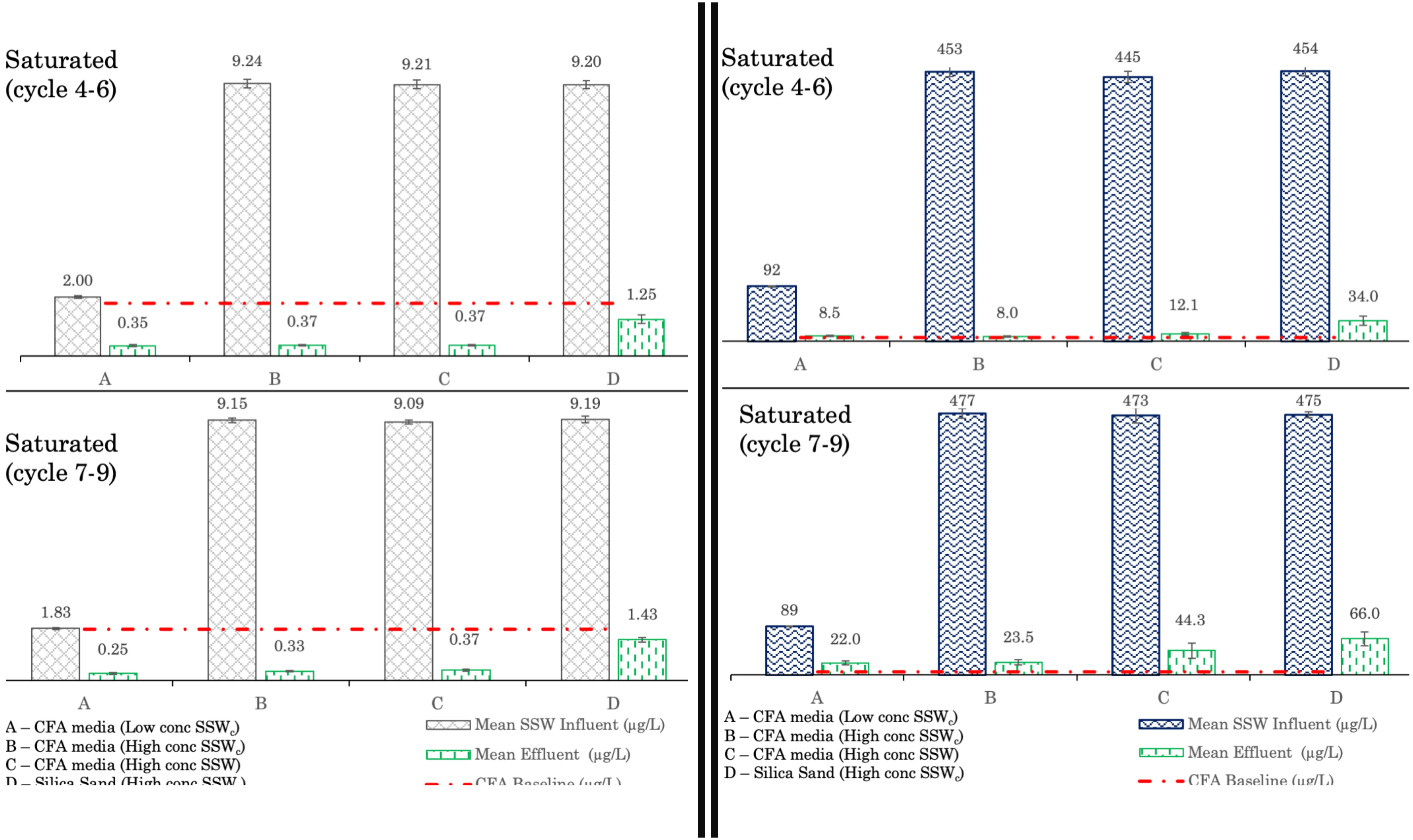

Figure G–12: Treatment performance for Ni (left) and Zn (right) in saturated zone

Appendix H: Riprap design equations

The design of the School Pond ripraps factored in the required sediment deposition velocity of ~ 0.3 m/s (Federal Highway Administration (FHWA), 2006) and the expected maximum velocity of the incoming stormwater This maximum velocity was calculated by considering the inlet pipe diameter, slope and material and then applying Manning's equation (Equation H-1) – assuming full pipe flow (an overestimation). The pipe inlets' diameter, material type, and slope were obtained from the as-built drawings supplied by the CoCT stormwater department and the topographical survey.

$$V = \frac{1}{n} \cdot R^{\frac{2}{3}} \cdot S^{\frac{1}{2}}$$ **Equation H-1**

Where V is the stormwater velocity, *n* is the coefficient of roughness or friction applied to the flow by the channel, R is the hydraulic radius, and S is the pipe slope.

Next, the flow in the pipe was calculated using Equation H-1 and multiplying the resultant velocity by the pipe area. The riprap and apron sizing were based on the design recommended by the Federal Highway Administration in their hydraulic structures design guidelines (FHWA, 2006). The riprap size can be determined by applying Equation I-2.

$$D_{50} = 0.2D \cdot \left(\frac{Q}{D^{2.5} \cdot \sqrt{g}}\right)^{\frac{4}{3}} \cdot \frac{D}{TW}$$ **Equation H- 2**

D_{50} is the riprap stone size, Q is the design discharge (m^3/s), D is the pipe diameter (m), TW is the tailwater depth (m), and g is the acceleration due to gravity (m/s^2).

The TW depth was established for the two inlets to be 0.9D (FHWA, 2006). The Froude's number (Fr) for the flow was calculated, and the flow regime was established as supercritical flow (Fr > 1), which meant D' (Equation H-3) was used in Equation H-2 rather than D.

$$D' = \frac{(D + y_n)}{2}$$ **Equation H-3**

Where D pipe diameter and y_n is the normal depth. The design steps differ according to the type of riprap to be designed.

Appendix I: Capital and maintenance cost for School Pond retrofit

Table I-1: School Pond capital cost (in 2023)

Item	Unit	Quantity	Rate (ZAR)	Cost (ZAR)
Inception and Engagement	Hr	80	615	49 200
Design cost	Hr	40	1600	63 900
Construction supervision	Hr	240	1500	357 000
Labour for construction (6 labourers)	Hr	1440	38	54 700
Provision for PPE	No.	6	880	5 300
Construction materials	Sum	1	63 000	63 000
Topographical and land surveys	Hr	16	850	13 300
Subtotal				606 700
Contingencies		10%		60 700
Total Construction Cost				**667 000**

Table I- 2: School Pond annual maintenance cost (in 2023)

Remedial action	Frequency	Workforce (duration of work)	QTY	Rate (ZAR)	Amount (ZAR)
Clearing up of litter at inlets and forebay	Monthly	6 People (One day)	72	R300	R21 600
Clearing of weeds in riprap	Every second month	6 People (One day)	36	R300	R10 800
Maintenance of riprap	Monthly in winter	6 People (One day)	24	R300.00	R7 200
Mowing grass on the slopes and up to the swampy area	Quarterly	6 People (One day)	24	R300	R7 200
Trimming the grass and shrubs in the swampy area	Twice a year	6 People (One day)	12	R300	R3 600
Clearing canal	Monthly in winter	2 People (One day)	6	R300	R1 800
Cleaning litter traps, reinforcing/replacing rocks	Monthly in winter	2 People (One day)	6	R300	R1 800
Plugging Mole holes	Annual before winter	6 People (One day)	6	R300	R1 800
Maintenance supervision	As required	1 Person (12 times a year)	12	R4 400	R1 800
Total					**108 700**

Appendix J1: Ethics in research clearance (Personal)

Application for Approval of Ethics in Research (EiR) Projects
Faculty of Engineering and the Built Environment, University of Cape Town

ETHICS APPLICATION FORM

Please Note:
Any person planning to undertake research in the Faculty of Engineering and the Built Environment (EBE) at the University of Cape Town is required to complete this form **before** collecting or analysing data. The objective of submitting this application *prior* to embarking on research is to ensure that the highest ethical standards in research, conducted under the auspices of the EBE Faculty, are met. Please ensure that you have read, and understood the EBE **Ethics in Research Handbook** (available from the UCT EBE, Research Ethics website) prior to completing this application form: http://www.ebe.uct.ac.za/ebe/research/ethics1

APPLICANT'S DETAILS		
Name of principal researcher, student or external applicant		Craig Tinashe Tanyanyiwa
Department		Civil Engineering
Preferred email address of applicant:		craigtanyaryiwa@gmail.com
If Student	Your Degree: e.g., MSc, PhD, etc.	PhD in Civil Engineering
	Credit Value of Research: e.g., 60/120/180/360 etc.	360
	Name of Supervisor (if supervised):	Prof. Neil P. Armitage
If this is a researchcontract, indicate the source of funding/sponsorship		DANIDA
Project Title		Pathways to water resilient South African cities

I hereby undertake to carry out my research in such a way that:
- there is no apparent legal objection to the nature or the method of research; and
- the research will not compromise staff or students or the other responsibilities of the University;
- the stated objective will be achieved, and the findings will have a high degree of validity;
- limitations and alternative interpretations will be considered;
- the findings could be subject to peer review and publicly available; and
- I will comply with the conventions of copyright and avoid any practice that would constitute plagiarism.

APPLICATION BY	Full name	Signature	Date
Principal Researcher/ Student/External applicant	Craig Tinashe Tanyanyiwa	Signed by candidate	29/05/2020
SUPPORTED BY	Full name	Signature	Date
Supervisor (where applicable)	Neil Philip Armitage	Signed by candidate	29 May 2020

APPROVED BY	Full name	Signature	Date
HOD (or delegated nominee) Final authority for all applicants who have answered NO to all questions in Section 1; and for all Undergraduate research (Including Honours).			
Chair: Faculty EIR Committee For applicants other than undergraduate students who have answered YES to any of the questions in Section 1.	Prof H von Blottnitz	Signed by candidate	16/05/2021

Page 1 of 1

Appendix J2: Ethics in research clearance (PaWS Team)

Application for Approval of Ethics in Research (EiR) Projects
Faculty of Engineering and the Built Environment, University of Cape Town

ETHICS APPLICATION FORM

Please Note:
Any person planning to undertake research in the Faculty of Engineering and the Built Environment (EBE) at the University of Cape Town is required to complete this form **before** collecting or analysing data. The objective of submitting this application *prior* to embarking on research is to ensure that the highest ethical standards in research, conducted under the auspices of the EBE Faculty, are met. Please ensure that you have read, and understood the **EBE Ethics in Research Handbook** (available from the UCT EBE, Research Ethics website) prior to completing this application form: http://www.ebe.uct.ac.za/ebe/research/ethics1

APPLICANT'S DETAILS		
Name of principal researcher, student or external applicant		Dr Kirsty Carden
Department		Future Water research institute
Preferred email address of applicant:		Kirsty.carden@uct.ac.za
If Student	Your Degree: e.g., MSc, PhD, etc.	
	Credit Value of Research: e.g., 60/120/180/360 etc.	
	Name of Supervisor (if supervised):	
If this is a research contract, indicate the source of funding/sponsorship		Danida Fellowship Centre, Denmark
Project Title		Pathways to water resilient South African cities

I hereby undertake to carry out my research in such a way that:
- there is no apparent legal objection to the nature or the method of research; and
- the research will not compromise staff or students or the other responsibilities of the University;
- the stated objective will be achieved, and the findings will have a high degree of validity;
- limitations and alternative interpretations will be considered;
- the findings could be subject to peer review and publicly available; and
- I will comply with the conventions of copyright and avoid any practice that would constitute plagiarism.

APPLICATION BY	Full name	Signature	Date
Principal Researcher/ Student/External applicant	Kirsty Jane Carden	Signed by candidate	06/09/2019
SUPPORTED BY	Full name	Signature	Date
Supervisor (where applicable)			

APPROVED BY	Full name	Signature	Date
HOD (or delegated nominee) Final authority for all applicants who have answered NO to all questions in Section 1; and for all Undergraduate research (Including Honours).			
Chair: Faculty EIR Committee For applicants other than undergraduate students who have answered YES to any of the questions in Section 1.	R Behrens	Signed by candidate	06 Sep 2019