Estimation of Groundwater Recharge and Assessment of Groundwater Quality in Urban Landscapes: A case study of the Wattle Grove area

by

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DEDICATION

First, I dedicate this thesis to our indescribable, uncontainable, all powerful, untameable, incomparable, impossibility specialist, and amazing God for seeing us through thick and thin in the land of Australia. Second, it is to my wonderful kids and wife for their total understanding and patience throughout this eventful but fruitful study. Finally, to my late mum (Nwazuruahu Amakaria Ezemba, Nee Mazi) who toiled day and night to ensure that I acquire the best of education, my late sister (Nkechi Udogwu, Nee Ezemba) and my late brother (Ik Ezemba). All passed away during the later stages of this study and may their souls rest in peace, Amen.

ABSTRACT

United Nations Population Fund estimated that half of the world's population lived in urban areas in 2008 and projected that by the year 2050, the population living in towns and cities would be up by 68%. Furthermore, half of the world's urban population depends on groundwater as the main source of water supply which puts pressure on groundwater as the urban population grows.

The main aim of this research is to estimate the groundwater recharge within urban centres with water table fluctuation (WTF) method and to evaluate physico-chemical quality of groundwater using Wattle Grove area, Sydney, Australia as a case study. The groundwater table depths were continuously monitored on an hourly basis over one year with data loggers installed in the four newly developed boreholes. The groundwater samples were collected on a monthly basis and taken to the university environmental laboratory for physico-chemical analysis.

The concept of rainfall-induced groundwater recharge is taken as groundwater recharge caused solely by rainfall while total groundwater recharge is caused by all other factors including rainfall. Both were estimated by considering the wet and dry periods in the year. On average, during the rainy periods, BH1 had the highest recharge per day of 1.67 mm/day while BH4 with 0.28 mm/day was the least. The variation of recharge estimates across the four sites could be attributed to different surface topography, presence of water bodies and underground water movement.

In spring and summer season, the groundwater-level response to rainfall shows that BH4 recorded the shortest lag time of 3 days while BH2 recorded the longest time of 14 days. The fastest time is as a result of fissure flow while the longest time is attributed to slow matrix flow. The combined analysis of spring and summer also shows that BH4 has the shortest response time and rainfall has a direct impact on groundwater level fluctuations.

For all the four boreholes pH, calcium and potassium concentrations in groundwater were within Australian Drinking Water Guideline (ADWG) and World Health Organisation (WHO) standards. Only BH1 has a high sodium concentration of over 4000 mg/L and a magnesium concentration of 500 mg/L and both exceed the aforementioned guidelines value of (50 – 300 mg/L). Too, the TDS value of BH1 exceeds EPA's guideline value of 1500 – 2600 mg/L and that makes the groundwater unsuitable for irrigation. This indicates that the groundwater may be contaminated by the salt. The source for the salt, at this stage, is

unknown. The groundwater from the four boreholes exceeds the turbidity limit of 5 NTU and it cannot be used directly for drinking.

STATEMENT OF AUTHENTICATION

I, Sylvester Nnamdi Ezemba, declare that all the materials presented in this Master of Philosophy thesis titled 'sustainable management of groundwater recharge and quality in urban landscapes: a case study of the Wattle Grove area' are of my own work, and that any work adopted from other sources is duly cited and referenced as such.

This thesis contains no material that has been submitted previously, in whole or in part, for any award or degree in other university or institution.

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Sylvester Nnamdi Ezemba

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MANUSCRIPTS UNDER PREPARATION

Analysis of water –level response to rainfall and estimation of monthly rainfall-induced recharge in an urban catchment: A case study of the Wattle Grove area.

Evaluation of the four boreholes groundwater suitability for drinking and irrigation purposes.

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

ABBREVIATIONS

ABS	Australian Bureau of Statistics
ADWG	Australian Drinking Water Guidelines
AHD	Australian height datum
ALS	Australian Laboratory Services
ANZECC	Australian and New Zealand Environment Conservation Council
APHA	American Public Health Association
ASR	Aquifer storage and recovery
ASTR	Aquifer storage transfer and recovery
BD	Bulk density
BH1	Borehole one
BH2	Borehole two
BH3	Borehole three
BH4	Borehole four
BOM	Bureau of Meteorology
CFCs	Chlorofluorocarbons
DHA	Defence Housing Australia
DO	Dissolved Oxygen
DOTRD	Department of Transport and regional Development
DS	Dissolved solids
DSITI	Department of Science, Information Technology and Innovation
EC	Electrical conductivity
EOCs	Emerging organic contaminants
EPA	Environmental Protection Agency
GME	Groundwater monitoring events
GWMAs	Groundwater management areas

ICP-OES	Inductively coupled plasma- optical emission spectroscopy
LCC	Liverpool City Council
LDPE	Low density polyethylene (LDPE)
LULC	Land use land cover
MAR	Managed aquifer recharge
mBGL	Metres below ground level
MIT	Moorebank Intermodal Transfer
MRC	Master recession curve
MRS	Magnetic resonance sounding
NEPM	National Environment Protection Measures
NHMRC	National Health and Medical research Council
NSW	New South Wales
NTU	Nephelometric Turbidity Unit
PD	Particle density
PSD	Particle size distribution
SAT	Soil aquifer treatment
SFA	Sectional flight auger
S_r	Specific retention
SS	Suspended solids
S_y	Specific yield
TDS	Total dissolved solids
THMs	Trihalomethanes
TOrCs	Trace organic chemicals
TS	Total solids
TSS	Total suspended solids
USCS	Unified Soil Classification System

USDA	United States Department of Agriculture
USGS	United States Geological Survey
WGL	Wattle Grove Lake
WHO	World Health Organization
WTD	Water table depth
WTF	Water table fluctuation

CHAPTER 1

INTRODUCTION

1.1 Background of the research

More than 50% of the world's population lived in urban areas in 2008 according to United Nations Population Fund. It is estimated by the year 2030, the population of people living in towns and cities would be around 5 billion. The urban population is projected to increase to 68% by the year 2050 (up Urban, 2007, UNFPA, 2018). Urban population growth leads to increased water demand. About half of the world's urban population depends on groundwater as the major source of water supply (Garcia-Fresca and Sharp, 2005). Groundwater is the largest earth's freshwater resource (Scanlon et al., 2005). Groundwater resources are poorly understood and hence poorly managed in many parts of the world. It is estimated that 2.5 billion people depend entirely on groundwater for drinking water supply (Smith et al., 2016). The ability of groundwater to sustain supplies to urban centres is affected by three factors: (1) the reduction of local aquifers capacity; (2) surface sealing or consumptive use of surface water reduces the natural recharge to aquifers; (3) the impact of climate change on water resources (Regnery et al., 2013). Some major chemical elements in groundwater are Ca, Fe, Mg, K and Na (Espinoza-Quiñones et al., 2015). Groundwater is a vital water resource especially for people in arid and semi-arid areas because it is less prone to pollution in comparison to surface water (Sinha et al., 2016). Groundwater plays a key role in the water balance of many freshwater lakes and may also be an important source of nutrients and other chemical constituents to marine coastal waters (Cable et al., 1997). Urban aquifers are affected by a variety of pollution sources such as recharging from the urban runoff polluted by sewerage, polluted rivers or other surface water, seawater intrusion etc. The result of the aforementioned is that a large number and variety of contaminants are found in urban aquifers. Organic pollutants like emerging organic contaminants (EOCs) are also found in urban water environment. The understanding of the processes that determine the biological and chemical quality of urban groundwater helps to address these problems (Tubau et al., 2017). Urban aquifers are crucial to the long term viability of many cities around the world (Thomas and Tellam, 2006). The practice of recharging urban aquifers with stormwater is common in urban areas as it serves a dual purpose of disposal of stormwater and also make up for reduced groundwater recharge. Stormwater infiltration basins minimise reduced groundwater recharge due to urban activities and also enhances the retention and degradation of contaminants in the soil and vadose zone (Datry et al., 2004). Managed aquifer recharge (MAR) is seen as an important method to meet the increasing global demand for water due to rapid population increase and limited water resources in many parts of the world. MAR systems are used to remove organic carbon, total nitrogen, pathogens and a range of trace organic chemicals. MAR systems have the capability to reduce the impact of trace organic chemicals (TOrCs) with lower cost, energy consumption, chemical usage and carbon footprint (Alidina et al., 2015).

Groundwater as an important resource has not been efficiently utilized in many urban settings due to lack of proper management, economies of scale, scientific uncertainties, and public policy promoting the usage of surface waters (Garcia-Fresca and Sharp, 2005). However, the overexploitation of groundwater in recent years has caused geological problems such as land subsidence which has hindered economic development in some countries (Wu et al., 2016). Urban development or urbanization may result in either increases or decreases in net groundwater recharge (Appleyard, 1995). The management of groundwater in urban areas requires a comprehensive knowledge of the hydrogeology of the area and tools for predicting the amount of groundwater and water quality evolution. This is because groundwater is a key tool in the adequate management of urban areas with increasing frequency. The unplanned pumping of groundwater has the tendency to cause ground subsidence, deterioration of the water quality and changes in land use (Tubau et al., 2017). A proper assessment of groundwater quality involves both the quantification of overall recharge and quality assessment of the point sources and non-point sources (Vázquez-Suñé et al., 2010). The estimation of recharge in an urban environment is more complex than in a rural setting. There are more routes for urban recharge than for rural recharge. The urban environment is characterised by abundance of impermeable areas such as roads, and also a dense network of drainage channels and drains that carry away stormwater. Moreover, the numerous pathways associated with urban environment make it difficult to quantify recharge. The various sources of recharge should be identified prior to estimation of urban recharge (Lerner, 2002, Garcia-Fresca and Sharp, 2005).

Sustainable development of groundwater is fundamental to adequate urban planning to ensure that water exploitation does not exceed natural replenishment. Over-exploitation leads to reduced water table by several metres and alteration of the groundwater flow direction (Collin and Melloul, 2001). Sustainable groundwater usage is linked to understanding of groundwater ages, recharge rates and mechanisms, the key processes altering groundwater quality and groundwater's relationship to the wider water-cycle and eco-environments (Currell et al., 2012). According to Yazicigil et al., (2011), the most adverse impact of climate change on groundwater resources will be the changing rate of recharge which is closely related to the changes in precipitation. They found that decreasing groundwater recharge as a result of climate change reduces the sustainable pumping rates by nearly one-third (Yazicigil et al., 2011). Smith et al., (2016), inferred that over the last 50 years, the global abstraction of groundwater has at least tripled and has not abated but continues to increase at an annual rate of between 1 and 2%. A sound groundwater management technique requires a clear understanding of both contaminant and abstraction pressures in addition to interactions with surface water management (Smith et al., 2016).

Sustainability of groundwater can be defined as the ability of present generation to meet their needs without compromising the ability of future generations to meet their needs. The reconciliation of sustainability concept to over-exploitation is problematic (Abderrahman, 2005). The adequate management of groundwater resources requires accurate information about the inputs (recharge) and outputs (pumpage and natural discharge) within each groundwater basin in order to assess its long term sustainable yield. Without a reliable estimate of recharge, the impacts of abstracting groundwater from an aquifer cannot be truly assessed, and the long-term behaviour of an aquifer under various management schemes cannot be reliably estimated (Sophocleous, 2005). The inability to manage groundwater sustainably puts at risk huge benefits for human wellbeing, sustainable development and biodiversity conservation (Smith et al., 2016).

1.2 Research significance

Groundwater plays enormous role in the society as it is used for agricultural, drinking and industrial purposes. It is an important resource not only in urban environment but also in rural environment. Groundwater performs three key roles in our environment: providing base flow which helps to keep most rivers flowing all year long, act as a source of maintaining good river water quality by diluting sewage and effluents, and as an important source of water supply (Lerner and Harris, 2009, Chinnasamy et al., 2018). There are negative consequences associated with uncontrolled use of groundwater such as subsidence, salt water intrusion and alteration of the permeability structures. Both human and natural activities tend to impact on the quality of groundwater and recharge (Oliveira et al., 2017, Scanlon et al., 2005, Garcia-Fresca and Sharp, 2005) . Cases abound where development of groundwater is carried out without adequate understanding of recharge and the relationship between the exploited aquifer and water sources. The implication of this act is that it leads to over-allocation and over-exploitation with its attendant consequences (Adelana, 2011).

For sustainable groundwater development and management, groundwater recharge estimation is a major challenge (Sun et al., 2013, Li et al., 2017, Watson et al., 2018, Sanford, 2002). Different researchers have used different methods to quantify recharge such as, water table fluctuation method (Healy and Cook, 2002, Crosbie et al., 2005, Oliveira et al., 2017, Watson et al., 2018, von Freyberg et al., 2015, Cai and Ofterdinger, 2016), chloride mass balance (Lerner, 2002, Scanlon et al., 2005,), water balances and numerical modelling (Tubau et al., 2017, Ghazaw et al., 2014).

In some instances, a combination of methods has been used to estimate the recharge which yielded different outcomes as observed by (Lerner, 2002 and Scanlon et al., 2005). In the case of water table fluctuation (WTF) method, the key parameter specific yield has been determined by pumping test, aquifer tests, water-budget methods, water table response to recharge, field test, and laboratory method. However, there is a difference in laboratory determined and field determined specific yield values (von Freyberg et al., 2015, Varni et al., 2013, Johnson, 1967). The specific yield value which is necessary in the computation of groundwater recharge by WTF method vary greatly due to variations in geology and depth to water table (Chinnasamy et al., 2018, Zhang et al., 2017). Some researchers have used literature values of specific yield (Cai and Ofterdinger, 2016, Sharma et al., 2015, Oliveira et al., 2017) to estimate recharge while others used varying specific yield with regards to depth (Crosbie et al., 2005) .

Hence, to add to the knowledge of local groundwater status, in this study, soils from the different depths will be analysed for specific retention in the field capacity pressure. Later, specific yield will be evaluated in the laboratory at different depths and the average value will be used with in-situ field measurement of water table depth to estimate recharge to local groundwater system. In addition, the groundwater quality will be monitored through the dedicated boreholes.

1.3 Outline of the thesis

The research undertaken in this study is presented in five chapters. There is also an appendix section where experimental data are included.

Chapter 1 presents a background to this research and its significance.

Chapter 2 presents a literature review that is divided into two sections – groundwater recharge and groundwater quality. The groundwater quality section covers the parameters affecting groundwater quality and its sustainable use in terms of level of urban groundwater abstraction. On the other hand, the groundwater recharge section covers the different types of recharge, factors affecting urban groundwater, routes and pathways of urban groundwater flow and methods of estimating urban groundwater recharge. Finally, the knowledge gaps identified are also highlighted.

Chapter 3 presents the materials and methodology adopted in this study. The chapter includes a description of the study area and hydrogeology, collection of soil samples from the area of study, monthly groundwater collection from the four dedicated boreholes, and sample management in the laboratory, and data acquisition. The equation for estimating groundwater recharge with WTF is also discussed.

Chapter 4 presents the results of daily groundwater recharge estimation. It also explains the impact of daily rainfall over daily groundwater – level fluctuations, and the use of correlation to determine the relationship between first day and last day of the month water table depth difference and monthly rainfall. Cross-correlation technique is used to calculate the response time of rainfall to water table fluctuations known as lag time in days for one year hydrological data and also spring and summer seasons. Finally, the soil profile of each site is discussed and the variation of specific yield values across the sites. Furthermore, the specific yield values obtained from this study are discussed in the context of literature data.

Chapter 5 presents graphical results of physico-chemical analyses, and comparison of the groundwater quality results with those found within the same catchment as study area. In addition, the groundwater quality results are compared with World Health Organisation (WHO) standard, Australian Drinking Water Guidelines (ADWG) and literature values from different countries.

Chapter 6 presents the conclusions and recommendations drawn from the study. The conclusions is divided into two namely groundwater recharge conclusions and groundwater quality conclusions. The recommendations encompass the two in terms of future study.

Appendix A presents details of physico-chemical parameters determined in the laboratory for soil samples collected from site and also groundwater samples.

CHAPTER 2

LITERATURE REVIEW

2.1 Introduction

In Chapter 1, background and research significance are discussed. This chapter covers the relevant issues central to the subject matter. A literature review was carried out to gain full insight into the subject matter with the purpose of ascertaining the current state of art and areas that need to be addressed. In fulfilling this purpose, different websites on the subject were visited, high quality journals, textbooks, governmental reports, and online resources were utilised in the course of the search.

The literature review section is organised into sustainable groundwater management, groundwater quality, types of recharge, managed aquifer recharge, factors affecting urban groundwater, pathways and routes of urban recharge, methods of estimating urban recharge, and knowledge gaps identified.

2.2 Sustainable groundwater management

According to Leahy (2015), 'groundwater is water that seeps into the ground and collects in the spaces between the grains of gravel, sand, silt, or clays, or settles into fractured rock'. Areas with huge volumes of groundwater are known as aquifers. Groundwater basins can be referred to as aquifers. Groundwater basins are recharged either through rainfall, river water, lakes, streams or irrigation water that seeps down through the unsaturated zone to the water table. Overdraft groundwater basin implies that the rate of water withdrawal exceeds the amount of water that recharges the basin over a period of years under average conditions (Leahy, 2015, Smith et al., 2016). Sustainability of groundwater can be defined as the ability of present generation to meet their needs without compromising the ability of future generations to meet their needs. The reconciliation of sustainability concept to over-exploitation is problematic (Abderrahman, 2005). Sustainability issues arise because extraction of groundwater creates a change in the water resource, which, in turn creates an impact on systems dependent on that resource. According to National Water Initiative, sustainable yield is defined as 'the level of water extraction from a particular system that, if exceeded, would compromise key environmental assets, or ecosystem functions and the productive base of the resource'. The major aquifers in Australia have been developed to the point where use is equivalent to or even exceeds the sustainable yield. These very high and possibly unsustainable levels of use may lead to a diminishing resource base and its antecedent environmental impacts. The timeframe and magnitude of groundwater level decline depends on aquifer properties, groundwater pumping regime and recharge (Harrington and Cook 2014). The concept of groundwater sustainability guarantees a level of security of supply and decreases the risk of contamination and ecological harm occurring from over-extraction.

Table 2. 1: Estimates of global water abstraction (Smith et al., 2016)

Continent	**Groundwater Abstraction**					**Compared to Total Water Abstraction**	
	Irrigation (Km^3/y)	**Domestic (Km^3/y)**	**Industrial (Km^3/y)**	**Total (Km^3/y)**	**Total (%)**	**Total water abstraction (Km^3/y)**	**Share of groundwater (%)**
North America	99	26	18	143	15	524	27
Central America and the Caribbean	5	7	2	14	1	149	9
South America	12	8	6	26	3	182	14
Europe (incl. Russia Federation)	23	37	16	76	8	497	15
Africa	27	15	2	44	4	196	23
Asia	487	116	63	676	68	2257	30
Oceania	4	2	1	7	1	26	25
World	**666**	**212**	**108**	**986**	**100**	**3831**	**26**

2.3 Groundwater quality

Groundwater is an essential natural resource which has a range of environmental values such as the provision of drinking water for humans and livestock, cultural and spiritual values, ecosystem values and provision of water flows to groundwater dependent ecosystems. Therefore, it is of paramount importance to ensure the protection of this valuable natural resource. Groundwater quality is not fixed instead it varies both spatially and temporally. The quality of groundwater is influenced by local geology, residence time in the aquifer, groundwater chemistry and the interactions between groundwater and rock formation. Groundwater can have naturally high salinity concentration, high dissolved nutrients and metals. Due to the high variability in groundwater chemistry, it may impact its quality in terms of meeting water quality guidelines set out for some relevant environmental values. Groundwater assessment is based on the comparison of measured groundwater quality indicators against guideline values that usually relate to the potential use of the water if extracted or if it is expressed as surface water (DSITI, 2017). These guideline values are arrived at based on health, taste and environmental impact. The groundwater collected from catchment area is analysed and compared with World Health Organisation (WHO) guidelines, and Australian Drinking Water Guidelines (ADWG). In the absence of guideline values for some parameters in ADWG, then WHO standard and literature values from different countries are used. The table below gives the guidelines values used for the comparison purpose.

Table 2. 2: Guidelines values

Parameters	World Health Organisation (WHO, 2011)	Australian drinking water guidelines (NHMRC, 2011)	Literature values*
pH	6.5 -8.5	6.5 – 8.5	6.5 – 8.5
Na, mg/L	200		200
K, mg/L			10 - 30
Mg, mg/L	100 - 300		50 - 200
Ca, mg/L	100 - 300		200 -300
Turbidity, NTU	5	5	5
Fe, mg/L	0.3	0.3	0.3 - 1
Mn, mg/L	0.4	0.1	0.4 – 0.5
TDS, mg/L	600 - 1000		500 - 2000

*- (Chukwu, 2008, Fisher et al., 2004, Hassen et al., 2016, Jain et al., 2010, Srivastava, 2019, Abbasnia et al., 2018, Longe and Balogun, 2010, Arumugam and Elangovan, 2009).

2.4 Types of recharge

Recharge is categorized into four types, namely: direct (from precipitation), indirect (from surface water bodies and leaky utility systems), localized (through preferential pathways such as sinkholes), and artificial.

2.4.1 Direct recharge

Direct recharge in cities is by means of percolation into unpaved areas, and to some extent through impervious surfaces because paved surfaces are not always impermeable. The method used in recharge calculations is to assume a proportion of impermeable area as permeable. Direct recharge importance wanes as the aridity of the climate or the amount of impervious cover increases (Garcia-Fresca and Sharp, 2005). Direct surface techniques are simple and widely used in the field of groundwater recharge. These techniques involve the movement of water from the land surface to the aquifer by means of simple infiltration (Sakthivadivel, 2007). It can be described as an aerially distributed process that takes place below the point of impact of the precipitation by vertical movement through the vadose zone (Lerner, 2002).

2.4.2 Indirect recharge

Indirect recharge can be described as processes where recharge occurs from runoff into mappable features, such as rivers and sinkholes (Lerner, 2002). Installation of groundwater pumping facilities or infiltration galleries near hydraulically connected surface water bodies such as streams or lakes to lower groundwater levels and induce infiltration from surface water bodies is an indirect recharge method. The effectiveness of induced recharge methods depends on the following factors: number and proximity of surface water bodies, hydraulic conductivity of the aquifer, area and permeability of the streambed or Lake Bottom, and hydraulic gradient created by pumping. One demerit of indirect methods of recharge over direct method is the inability to control the quantity and quality of water (Sakthivadivel, 2007). Water mains must be pressurized to achieve that contaminants do not infiltrate into the mains and also to ensure distribution to the far reaches of the water system. The main cause of leakage in water distribution system is pressure. The percentage of water losses in cities in developed countries varies from cities in less developed countries. It is estimated that developed countries account for approximately 20% to 30% while less developed countries are in the region of 30% to 60%. The leakage from main water supply is a main source of indirect groundwater recharge. Rainfall in more arid regions of the world is short of the amount of water distributed in most cities (Garcia-Fresca and Sharp, 2005).

Leakage of sewage or wastewater causes groundwater contamination and it is widespread. The leakage from wastewater pipes is marginal compared to high leakage rate from water mains due to difference in applied pressure. Leakage from sewer lines above the water table is more serious than sewer lines beneath the water table which could drain groundwater. The inadequacy of effective sewage facilities in many cities makes it possible that most of the supplied water becomes recharge. It is estimated that wastewater pipe leakage accounts for about 5% (Garcia-Fresca and Sharp, 2005).

2.4.3 Localized recharge

Localized recharge is a line or point process where water moves short distances laterally before infiltration (Lerner, 2002). Localized recharge takes place through faults, fractures, etc., and it is dependent on the geologic materials, the structure, and the soil types in each particular area. There is no direct link between localized recharge and urbanization even though it can be affected by it (Garcia-Fresca and Sharp, 2005). The edges of paths and roads where no formal drainage exists promote localized recharge. Localized recharge takes place in arid and semi-arid regions and in many rapidly urbanizing cities ,where there is shortage of storm- drainage infrastructure (Lerner, 2002).

2.4.4 Artificial recharge

Todd and Mays, (2005), defined ‘artificial recharge as a means of augmenting the natural movement of surface water into underground formations by some method of construction i.e., by spreading of water, or by artificially changing natural conditions’. There are several methods of achieving artificial recharge which include water spreading, recharging through pits and wells, and pumping to induce recharge from surface water bodies. The governing factors in choosing any of the methods are local topograph, geologic conditions, soil conditions, the quantity of water to be recharged and the final water use (Todd and Mays, 2005). Man-made structures such as recreational lakes and ponds, soakways, runoff detention ponds, retention basins, artificial infiltration ponds, spreading basins, recharge ditches, and injection wells are designed to reduce flooding, relieve the sewage networks, and promote groundwater recharge (Garcia-Fresca and Sharp, 2005). Artificial recharge is used in other areas such as wastewater disposal, waste treatment, secondary oil recovery, prevention of land subsidence, storage of fresh water with saline aquifers, crop development and stream flow augumentation. Artificial recharge reduces the amount of water lost due to evaporation when compared with similar surface storage systems and also minimises environmental problems that could arise from the use of surface storage facilities (Sakthivadivel, 2007).

Artificial recharge serves the dual purpose of water conservation and overcoming problems associated with overdrafts. One of the most used methods of artificial recharge is known as water spreading. It can be described as the process of releasing water over the ground surface with the aim of increasing the quantity of water infiltrating into the ground and then percolating to the water table. The quantity of water that will enter the soil is dependent on the area of recharge and length of time of recharge. The rate of recharge is used to determine the efficiency of spreading which is expressed as the velocity of downward water movement over the wetted area. The different types of spreading methods are basin, stream channel, ditch and furrow, flooding, and irrigation (Todd and Mays, 2005).

2.4.5 Basin method

Basins formed by construction of dikes or levees or by excavation can be used to recharge groundwater as water is released into the basins. Basin sizes and shapes are made to fit land surface slope. water free of silt materials is valuable in preventing sealing of basins during submergence. Periodic maintenance is required of most basins in order to improve infiltration rates by scarifying, disking, or scrapping the bottom surfaces when dry. A single basin is used to recharge local storm runoff while multiple basins are used for the diversion of streamflow. One of the advantages of using multiple basins is that it allows for the continuity of operation when certain basins are removed for maintenance purpose. Basins are the preffered method of recharge because it is feasible, efficient utilization of space and ease to maintain. Infiltration basins such as SAT (soil aquifer treatment) are widely used for groundwater recharge and removal of municipal waste. The use of SAT systems is spreading globally as a result of its economic viability and low-cost of maintenance (Todd and Mays, 2005).

2.4.6 Stream – channel method

Water spreading in a natural stream channel involves operations which lead to increase of the time and area over which water is recharged from a naturally losing channel. This involves both upstream management of streamflow and channel modifications which enhance the infiltration process. Upstream reservoirs are used to regulate erratic runoff and ideally limit streamflows to rates within the absorptive capacity of downstream channels. Some of the ways of improving stream channels include widening, leveling, scarifying, or ditching to increase infiltration. It is possible to conduct channel spreading without a specific spreading works (Todd and Mays, 2005).

2.4.7 Ditch-and-furrow method

This involves the distribution of water to a series of ditches, or furrows , that are shallow, flat-bottomed, and closely spaced to obtain maximum water-contact area. The three basic layouts which are used are: (1) contour- this is where the ditch follows the ground contour and by means of sharp switchbacks meanders to and fro across the land; (2) tree- shaped – this is the successive branching of main canal into smaller canals and ditches; and (3) lateral – this is the lateral extension of small ditches from the main canal. Any ditch plan should take recognizance of the local configuration of the area. Excess water is conveyed back into the main stream channel via means of a collecting ditch placed at the lower end of the site (Todd and Mays, 2005).

2.4.8 Flooding method

Water is diverted to spread evenly over a large area in flat topograpic areas. The use of canals and earth-distributing gullies aid in releasing the water at intervals over the upper end of the flooding area. The water velocity should be controlled in order to form a thin sheet of water over the land and also avoid disturbing the soil. The highest infiltration rates are found in areas with undistributed vegetation and soil covering. Embankments or ditches are constructed around the flooding area to control the water (Todd and Mays, 2005).

2.4.9 Irrigation method

In irrigated areas, water is intentionally spread by irrigating cropland with excess water during dormant, winter, or nonirrigating seasons. This practice does not attract additional expenses for land preparation as the distribution system is already in place. Irrigation canals that are full will contribute to recharge via seepage from the canals. The leaching action of the percolating water which has the dual effect of carrying salts from the root zone to groundwater and in removing soil nutrients which reduces crop yield should be factored in (Todd and Mays, 2005).

2.5 Managed aquifer recharge (MAR)

Managed aquifer recharge (MAR) is done with the sole pupose of recharging aquifers with water for subsequent recovery or environmental benefit. Aquifers , permeable geological strata that contain water , are replenished naturally via rain soaking through soil and rock to the beneath of aquifer or by means of infiltration from streams. There are three categories of human activities that enhance aquifer recharge namely – (1) Unintentional- acts such as clearing of deep- rooted vegetation, by deep seepage under irrigation areas and by leaks from water pipes and sewers, (2) Unmanaged- the use of stormwater drainage wells and sumps, and septic tank leach fields, for disposal of unwanted water without considering its reuse, (3) Managed- this is achieved through the use of injection wells, and infiltration basins and galleries for rainwater, stormwater, reclaimed water, mains water and water from other aquifers that is recovered for various types of uses (Dillon et al., 2009).

MAR systems could be constructed and operated with a primary target of improving the quality of recharge waters. The augmentation of groundwater supply may be by either direct method which is increasing the amount of water in storage, or indirect method by ameliorating the impacts of groundwater use, which may hinder the utilization of available resources. A good example of indirect method is the strategic recharge of freshwater along coastal areas to control saline- water intrusion. Storage of water in MAR systems can help to overcome the negative effects of climate change (Maliva and Missimer, 2012). It can also be used to harvest abundant water in urban areas that is currently unused (Dillon et al., 2009).
The use of MAR to enhance natural rates of groundwater recharge serves as an important source of water for urban and rural settings. But the focus of this is on urban environment. Stormwater, reclaimed water, mains water, desalinated seawater, rainwater or groundwater from other aquifers can be stored in a MAR system.

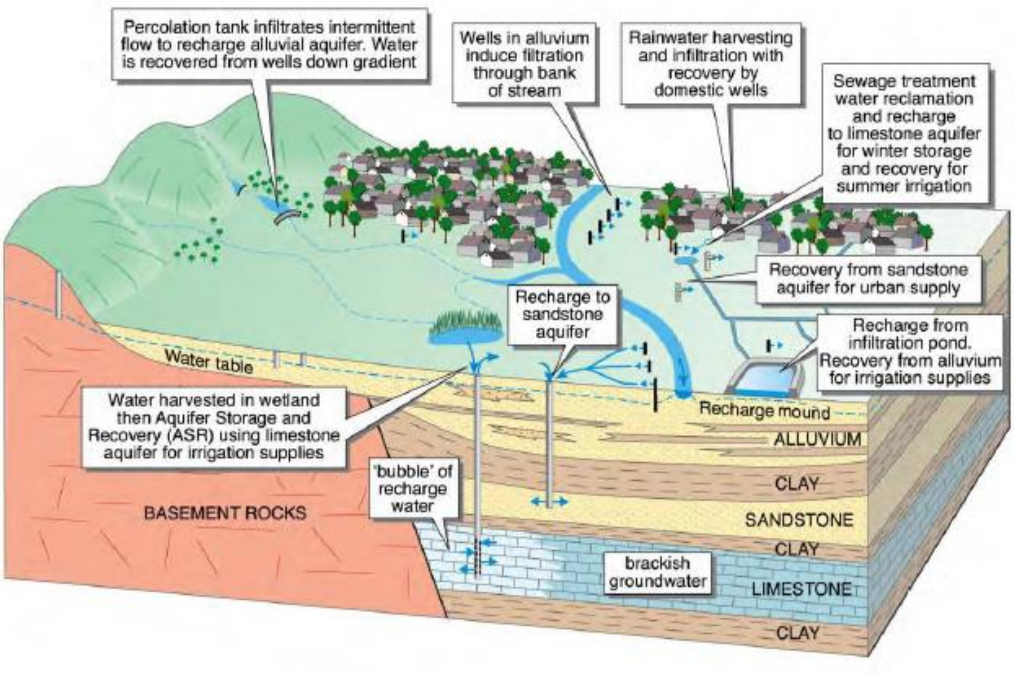

Figure 2. 1: Managed aquifer recharge is adapted to the local situation, and is usually governed by the type of aquifer, topography, land use and intended uses of the recovered water (Dillon et al., 2009).

The common reasons for using MAR are :

I. It secures and enhances water supplies
II. It leads to improvement of groundwater quality
III. It prevents the intrusion of salt water into caostal aquifers
IV. It reduces evaporation of stored water

V. It helps to maintain environmental flows and groundwater- dependent ecosystems

The consequential benefits of MAR are:

I. The improvement of coastal water quality by reducing urban discharges
II. The capacity to mitigate floods and flood damage
III. It facilitates urban landscape improvements that increase the value of land.

2.5.1 Types of managed aquifer recharge systems

MAR systems are grouped into three main categories according to their primary recharge process. They are: (1) Systems that involve subsurface injection using wells, (2) Systems that involve surface or near surface application of water, (3) Systems that result in an enhancement of natural recharge processes (Maliva and Missimer, 2012). But our area of interest is in MAR systems used in urban setting.

2.5.1.1 Aquifer storage and recovery (ASR)

Pyne defined it as "the storage of water in a suitable aquifer through a well during times when water is available, and the recovery of the water from the same well during times when it is needed". From the aforementioned definition, it is crystal clear that three main components are required. First, the water is stored underground. Second, the water is emplaced underground using wells. Third, the water is recovered using the same well as was used for emplacement (Maliva and Missimer, 2012).

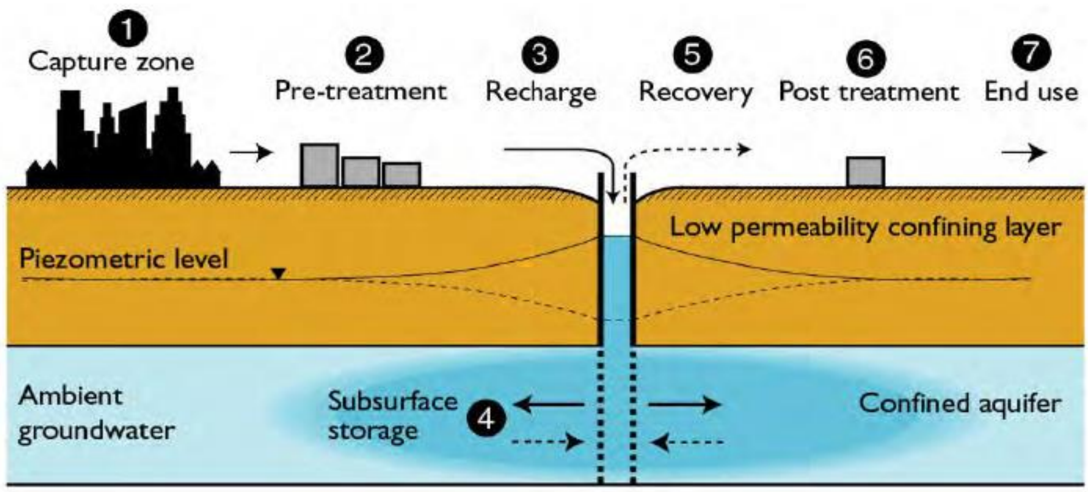

Figure 2. 2: Aquifer storage and recovery system (Dillon et al., 2009).

Figure 2.2 shows the processes involved in ASR. The captured water from different sources undergoes treatment before injection into the aquifer in this case confined aquifer. This

injected water flows radially outward from the injection well, displacing and partially mixing with the ambient groundwater in the storage aquifer. This leads to a development of transition phase between the injected and ambient water. During the recovery phase, the stored water , mixing zone, and ambient water are drawan back towards the ASR well.
The use of the same well as injection and recovery is cost effective when compared to the construction of separate dedicated injection and recovery wells. Well pumps used for recovery can also serve for the rehabilitation of the well. ASR is a storage technology and requires the periodic availability of excess water that can be stored for future use. Water from various sources such as potable water, treated water wastewater, treated surface water, stormwater, and desalinated water is stored in existing ASR systems. The advantages of ASR systems over surface storage options are:

- Lower cost of setting it up
- Lower land requirements
- Avoidance of evaporative losses
- Less prone to contamination

The main shortcomings of ASR systems include that unfavourable hydrogeologic conditions can result in low recoverability of stored water and adverse changes in water quality may occur due to fluid- rock interactions.
There are many types of ASR systems that differ in how they achieve the useful storage of water. Injection is considered to be useful if it results in an additional supply of water at a useable quality that would not otherwise have been available. Chemically bounded ASR systems act by dispacement of poorer water quality in order to achieve useful storage (Maliva and Missimer, 2012).
The second type of ASR system deals with physical storage systems in which injection increases the total volume of water present in an aquifer. Physical storage ASR systems is the injection of freshwater into freshwater aquifers. It is important that physical ASR systems maintain the increase in heads until the time of recovery. If this is not achieved, there would be no net storage. The performance of physical storage ASR systems are evaluated on the basis of water level criteria while chemically-bounded ASR systems performance are evaluated using water quality criteria. It is important that a good aquifer with sufficient lateral and underlying confinement is chosen in order to retain most of the stored water until the time of recovery. Intermontane basins bounded by crystalline bedrock are good medium for physical –storage ASR systems. In the city of Las Vegas, Nevada (USA), there is a large

scale physical storage system in which treated surface water from the Colorado River is stored in an over-draft basin fill aquifer (Maliva and Missimer, 2012).

The regulatory storage systems is the third type of ASR system. This system is common in parts of the United States in which injection of water confers the right to later pump additional groundwater, which would not otherwise be authorised. The system owner may either obtain a 100% credit for injected water or a partial credit depending on the rules in place in the state. The reason for regulatory storage ASR systems is to ensure that both injection and recovery of water does not impact negatively long-term aquifer water levels. Regulatory storage ASR systems may still have negative effects on water resources and environment by increasing groundwater withdrawals during dry seasons.

ASR systems do not have a constant duration of storage. Most of the ASR systems provide storage on a seasonal basis. The performance of ASR systems is dependent upon site-specific hydrogelogy (Maliva and Missimer, 2012). ASR is useful in brackish aquifers, where storage is of paramount importance and water treatment is of secondary importance. A good example is Grange golf course in South Australia (Dillon et al., 2009).

2.5.1.2 Aquifer storage transfer and recovery (ASTR)

This method deals with the injection of water into a well for storage, and recovery of the water from a different well. This helps in the treatment process of the water in the aquifer by extending residence time in the aquifer beyond that of a single well (Dillon et al., 2009). The use of separate injection and recovery wells for the purpose of chemical and microbial contaminant attenuation is equally termed as “aquifer storage transfer and recovery (ASTR).

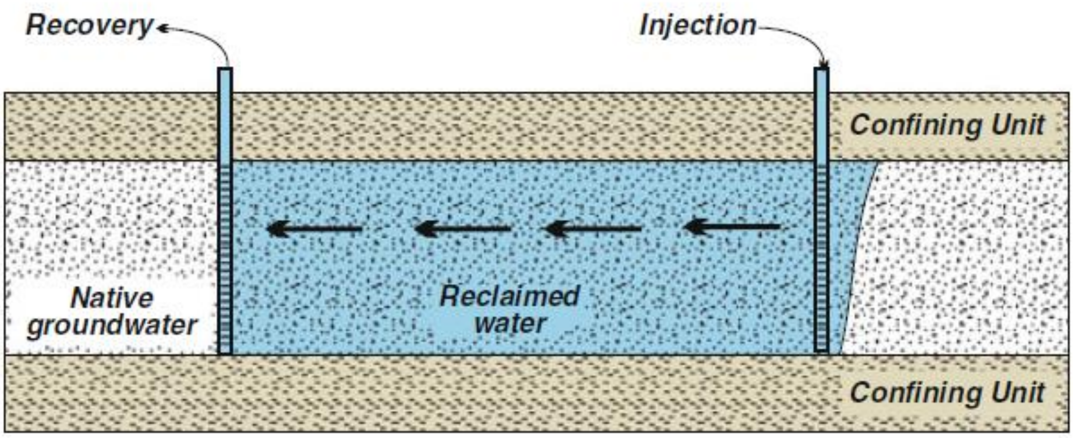

Figure 2. 3: Aquifer storage transfer and recovery system in a confined aquifer (Maliva & Missimer, 2012).

Figure 2.3 shows the improvement of water quality through the natural attenuation of contaminants by physical, chemical and biological processes as the water flows through the aquifer from the injection well to the recovery well.

The underlying feature of ASTR is the intentional use of injected water flow through an aquifer as a treatment mechanism. The determination of the accurate travel times between injection and recovery wells is a key issue for ASTR systems. The reason is that the travel may be more rapid than expected due to the heterogeneity of the aquifer. Methods such as aquifer characterisation, modelling, and tracer testing can be used to assess aquifer travel times. There is a marked difference in water chemistry between the injected water and the native groundwater which can serve as effective natural tracers (Maliva and Missimer, 2012). The El Paso Water Utilities (Texas, USA) Hueco Bolson Recharge Project is an excellent operational example of an ASTR project. Highly –treated reclaimed water from the Fred Hervey Water Reclamation Plant is injected into the upper Hueco Bolson Aquifer and recovered later for potable use. The recharge and recovery wells have a minimum spacing of 782m which is to ensure an adequate aquifer residence time (two year minimum) for complete inactivation of viruses in the recovered water (Maliva and Missimer, 2012). An example of ASTR project in Australia is Parafield Gardens, SA (Dillon et al., 2009).

2.5.1.3 Soil aquifer treatment (SAT)

Treated sewage effluent can be placed in basins, allowing for infiltration into the ground for the purpose of recharging the aquifer. The movement of the treated sewage effluent through the soil and aquifer causes it to undergo significant quality improvements through physical, chemical and biological processes. The sum total of these processes and the water quality improvement obtained are known as soil aquifer treatment (SAT). SAT is based on the concept of infiltration of treated wastewater into the soil and percolation through the unsaturated zone. There are many different mechanisms that can cause improvement in water quality such as infiltration, biological degradation, physical adsorption, ion exchange and precipitation.

There are five major components that make up the SAT system:

1. Pipeline- this carries the treated sewage effluent from the wastewater treatment plant
2. Percolation or infiltration basins- this is where the treated effuent infiltrates into the ground. The infiltration area causes the creation of a local hydraulic mound and the recovery areas (whether natural or engineered) result in a cone of depression that

captures the recharged water. A major design issue is the configuration of the infiltration and recovery locations and rates so that the movement of the plume of effluent is controlled.

3. Soil immediately below the infiltration basins- where natural treatment takes place. This serves as a natural filter by removal of suspended solids, biodegradable organic matter, and pathogenic micro-organisms.There may be a significiant reductions in nutrients and heavy metals by sorption and a variety of biologically mediated reactions.
4. Aquifer- this is where water is stored for a long duration. As the water travels through the aquifer, additional filtration and removal of contaminats may take place.
5. Recovery well- the water is recovered from here for potable and non –potable reuse (Todd and Mays, 2005, Maliva and Missimer, 2012).

SAT , as originally defined , is different from groundwater recharge using sewage effluent in the sense that the recharged water is recovered and its extent in the aquifer is controlled. In a situation where the receiving aquifer contains freshwater, an integral part of the design and operation of the SAT system is controlling the flow of recharged water in the aquifer so that it can be collected instead of migrating away and eventually entering wells used for potable water supply. Both SAT and other types of surface spreading systems share the same natural treatment processes ,irrespective of the recharged water been locally contained or controlled.

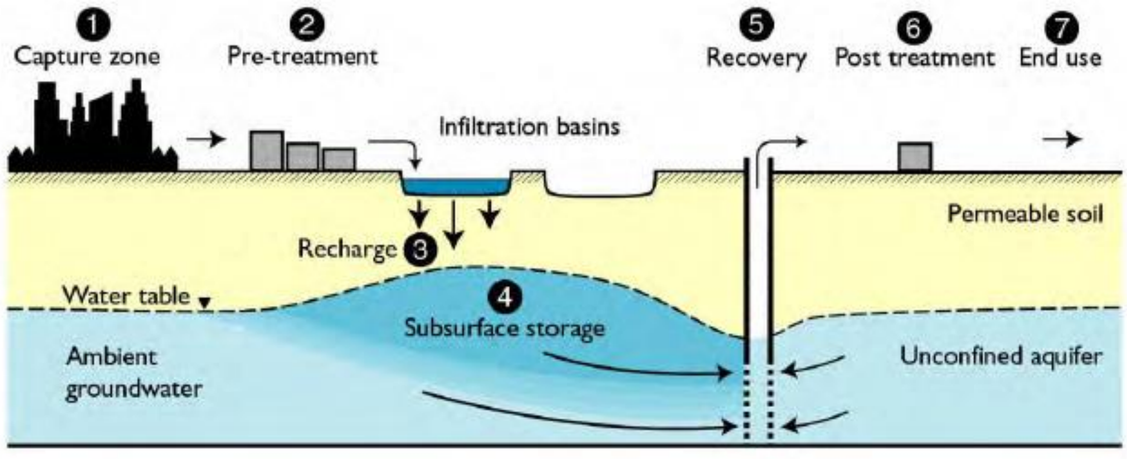

Figure 2. 4: Soil aquifer treatment (Dillon et al., 2009).

Figure 2.4 is a SAT system constructed in an unconfined aquifer. It shows the processes involved right from the capture of the effluent to the final recovery of water in a recovery well. The recovered water maybe treated further depending on its intended use.

SAT systems can be used to provide seasonal and multiannual storage of reclaimed water. One shortcoming of SAT systems are their inability to totally reduce salinity of the water. Salinity may actually increase due to evaporation, leaching of salts in the soil, and the atmospheric deposition of salt as dust and aerosols. SAT systems require large land mass for its construction especially for large capacity systems (Maliva and Missimer, 2012).

SAT systems are technically simple to construct and operate and both the construction and operational costs are low. The performance of the system is dependent on local hydrogeological conditions. It is good for SAT systems to be constructed in areas with granular soils that have sufficient permeability to give high infiltration rates, but yet be fine enough to provide good filtration. The best type of soils suitable for it are in the fine sand, loamy sand, and sandy loam range. Sites with shallow water tables have inadequate thick vadose zone which is not good for SAT construction (Maliva and Missimer, 2012).

SAT system was built in Alice Springs, NT to prevent winter overflows of sewage effluent with the aim of provision of irrigation water for horticultural development. The identified palaeochannel aquifer could store up to 600 ML/yr reclaimed water (Dillon et al., 2009).

2.6 Factors affecting urban groundwater

Groundwater in urban areas is affected by the following: overexploitation, subsidence, seawater intrusion, alteration of the permeability structure, groundwater contamination and changes in recharge and discharge.

2.6.1 Overexploitation

In general, the probability of having aquifers that have been totally depleted is minimal. The increased population of city dwellers put pressure on the water demand which has the consequence to exceed safe, permissive, or sustainable aquifer yields. A major concern is the pumping of water when the water levels drop so low that pumping becomes too exorbitant and water yields are drastically reduced. Shallow unconfined aquifers can act as a secondary source of urban supply during droughts when the water level is drastically reduced. Majority of the cities are dependent on surface waters for their water needs instead of locally available groundwater due to the fact they are situated on navigable rivers, subsidies from federal level encourage building of large dams and other surface- water projects and treatment facilities (Garcia-Fresca and Sharp, 2005).

2.6.2 Subsidence

According to Todd and Mays (2005), land subsidence can be described as the gradual settling or sudden sinking of the earth's surface as a result of the subsurface movement of earth materials. Land subsidence is a global problem and more than 17,000 square miles in 45 states in the United States have been affected by subsidence. The major factors responsible for it are aquifer-system compaction, mining of groundwater, drainage of organic soils, underground mining, hydrocompaction, natural compaction and sinkholes. Approximately 80% of the identified subsidence in the United States is as a result of groundwater exploitation (Todd and Mays, 2005). The overexploitation of groundwater in recent years has caused geological problems such as land subsidence which has hindered economic development in some countries (Wu et al., 2016).

Over 150 areas of contemporary subsidence have been identified. However, in some countries such as Mexico, Japan and the United States, the subsidence is up to 10 m. The forecast is that many more areas are likely to experience subsidence in the future due to accelerated exploitation of natural resources to meet the demands of increasing population and industrial development in many developed countries of the world (Mousavi et al., 2001).

Overexploitation of groundwater in Vietnam for domestic, industrial and agricultural uses has led to a massive reduction of the groundwater tables. A case in point is the Cà Mau province with its increased urban population from about 66 million in 1990 to 90 million in 2013 that has resulted in subsidence that averages 3 cm per year. One of the earliest countries that adopted controlled usage of groundwater to slow subsidence was Japan. The city of Tokyo detected subsidence in the early twentieth century through monitoring of the water table. After World War II, the water table rose as a result of reduced pumping of ground water in the heavily damaged city which slowed subsidence. As a result of economic boom in Japan, there was increased groundwater use and by 1968 subsidence was 24 cm per year in some places. In 2006, the city imposed strict guidelines on groundwater consumption and that has led to reduction of subsidence to about 1 cm per year in the worst affected areas. The subsidence in Bangkok, Thailand was about 12 cm per year in the 1980s due to groundwater consumption that averaged an estimated 1.2 million m^3 per day. In order to bring it under control, the government raised groundwater taxes which consequently reduced the groundwater consumption to 0.8 million m^3 per day. Due to the reduced groundwater consumption, the subsidence has reduced to 1-2 cm per year. In Jakarta, there is unregulated extraction of groundwater and the northwest of Jakarta is expected to have subsidence of 20 cm per year (Schmidt, 2015). Land Subsidence has been observed at many sites in Sweden and Norway and also in other glaciated areas of similar geologic and hydrologic environments as a result of groundwater extraction. Land subsidence has been observed in the Rafsanjan plain in Iran due to excessive exploitation of groundwater. The rate of subsidence is estimated at 50-150 mm for decline of about 1 m in groundwater level (Mousavi et al., 2001) .

Two contrasting environments and mechanisms are chiefly responsible for the phenomenon of subsidence due to groundwater withdrawal. First, carbonate rocks environment overlain by unconsolidated deposits, or old sinkholes filled with unconsolidated deposits, which receive buoyant support from the groundwater body. The lowering of the water table causes the buoyant support to be removed, and the hydraulic gradient increases which cause the downward movement of the unconsolidated material into openings in the underlying carbonate rocks. The result of this is the collapse of the roof. Carbonate terrain prone to sinkhole formation when the water table is lowered is found in many regions of the world. Second, the most extensive occurrence is found in young or semi consolidated clastic sediments of high porosity laid down in alluvial, lacustrine, or shallow marine environments (Mousavi et al., 2001). Low-lying regions are prone to subsidence which is critical because flooding and land loss by coastal retreat are increased by both coastal storms and long-term sea- level fluctuations. Construction and leaky water mains and sewage can induce localized collapse and subsidence. Subsidence can be controlled by regulating the locations and rates of pumping wells. The adverse effect of coastal subsidence can be reduced or totally eliminated by land-use planning. The easiest remedy to coastal subsidence is to limit development in low-lying coastal areas. However, that is not the case as development of such areas continues due to rapid increase in urban populations, the desire to live on the coast, and the fertility of coastal soils (Garcia-Fresca and Sharp, 2005).

2.6.3 Saltwater intrusion

The most common pollutant in fresh groundwater is saline water. Intrusion of saline water is as a result of displacement or mixing of freshwater in an aquifer by saline water. It can occur in deep aquifers, coastal aquifers and shallow aquifers. In deep aquifers, it is known as the upward advance of saline waters of geologic origin, in shallow aquifers is as a result of discharges from surface waste while in coastal aquifers is from an invasion of seawater. Inadvertent human activities are responsible for saltwater intrusion into fresh groundwater formations (Todd and Mays, 2005).

Pumping groundwater can equally lead to abstraction of lesser- quality water. This is obtainable in cities on oceanic islands or in close proximity to the coastline, where salt water underlies or is adjacent to the fresh water. Surface waters in coastal regions are not immune from salt water invasion. Saltwater intrusion can equally take place inland. Example of saltwater intrusion can be found in south of Kansas City, Missouri where overdraft of the Ordovician carbonate aquifers has induced downward intrusion of saline water from the overlying Pennsylvanian clastic rocks. The effect of pumping on the reversal of hydraulic gradients has been observed along the Rio Grande, in El Paso, Texas, and Juarez, Mexico which resulted in the intrusion of poor – quality river water into the Hueco Bolson aquifer (Garcia-Fresca and Sharp, 2005).

According to Todd and Mays, (2005), saline water in aquifers may be derived from any of the several sources mentioned below:

1. Encroachment of seawater in coastal areas
2. Seawater that entered aquifers during past geologic time
3. Salt in salt domes, thin beds, or disseminated in geologic formations
4. Water concentrated by evaporation in tidal lagoons, playas or other enclosed areas
5. Return flows to streams from irrigated land
6. Human saline wastes

The actions that initiate saline water intrusion can be categorized into three namely: (1) Reduction or reversal of groundwater gradients, which makes it possible for denser saline water to displace fresh water, (2) Destruction of natural barriers that separate both fresh water and saline water, (3) Subsurface disposal of saline water into disposal wells, landfills or other waste repositories. Salt water intrusion is a special category of groundwater pollution (Todd and Mays, 2005). There are many ways to tackle saltwater intrusion. One of the methods is desalination which was seen as expensive but it is becoming economic viable in terms of tackling of brackish groundwater. The other methods are importation of water, shifting pumping strategies, limiting the amount of water extracted from groundwater and creation of hydraulic barriers through injection wells or infiltration galleries (Garcia-Fresca and Sharp, 2005).

2.6.4 Alteration of the permeability structure

The urban underground is a complex network of buried structures, pipes, tunnels, etc., which is rapidly evolving. It can be compared to a shallow Karstic system. A study to determine the urban porosity of Quebec City, Canada was carried out. The city of Quebec is built upon

Precambrian metamorphic rocks, Cambrian- Ordovician sedimentary rocks, and unconsolidated alluvial and glacial deposits. The number of large underground openings-subsurface tunnels and other constructions, including water reservoirs and parking garages were documented. A porosity of 0.06% was estimated for Quebec City based on its surface area and the assumption that the tunnels and installations were mostly in the upper five meters of the earth. The value of this estimated secondary porosity is within the range of Karstic aquifer porosities found in literature. However, the unknown estimates of porosity created by smaller utility lines, trenches, pipes and conduits can dominate flow and transport in urban setting (Garcia-Fresca and Sharp, 2005).

The hydrogeological literature on the influence of underground anthropogenic structures is limited. Engineering structures can act as either aquifer discharge routes or barriers to shallow groundwater flow. A study in Germany revealed that sands on which pipes are laid contribute to increasing runoff, and thus reduce groundwater recharge. Higher permeability contributes to declines of groundwater levels as observed in Sweden. Urbanization leads to increase of heterogeneity of permeability and transmissivity. The effects of Utility trenches on groundwater flow and transport are quite high as a result of increase in permeability along utility trenches. High permeability utility trenches change groundwater flow and also cause the development of complex or multiple contaminant plumes arising from a single point source. It is hard to predict the direction and velocity of groundwater flow because of the influence of utility trenches. There is a semblance in the trenches which the utility network lies and the natural fractured systems just as larger underground openings, excavations, and tunnels are similar to natural conduits, caves and channels (Garcia-Fresca and Sharp, 2005).

One adverse effect of urban development on hydrologic systems is the modification, covering, and elimination of major courses. Rivers and streams that have been rendered invisible from the surface and may, in fact have ceased to exist are found in London. An example is the course of the Fleet River that underlies Fleet Street. In Washington, about 75% of the streams that existed in 1880 have disappeared, but the buried channels influence groundwater flow and affect wetlands, construction, and groundwater remediation. The older the city, the more complex is the urban karst, which makes it very difficult to predict groundwater flow and transport. In addition, the development of urban karst is more faster than natural karst (Garcia-Fresca and Sharp, 2005).

2.6.5 Groundwater contamination

The quality of water is the major issue surrounding urban groundwater supply. Shallow aquifers and surface waters in urban environment are prone to pollution by runoff from paved surfaces, leaky storage tanks, surface spills and illegal dumping of dangerous waste, leaky sewage lines, and inadequate provision of sanitation facilities. Contamination of shallow aquifers is a major threat as a result of increase of urbanized area. In many of the developing countries, there is a shortage of sewer systems to meet the growing population. The provision of mains for water supply is not adequate. Sewers are only available to a little part of areas in the centres of cities. The percentage of wastewaters released as a result of unsewered areas accounts for about 90%. These wastewaters are released in pit latrines, cesspools, or septic tanks. These are likely sources of contamination and limit the use of shallow wells as sources of drinking water. The natural hydraulic gradient is altered when there is a decline of water levels in an aquifer. The effect of this is the encroachment of poor-quality water which thus deteriorates the water quality. The extraction of water is shifted to deep aquifers when shallow aquifers have been contaminated. The adverse effect is cross-formational flow and contamination of the deeper aquifers as seen in Sana'a, Yemen (Garcia-Fresca and Sharp, 2005).

In the case of developed countries, urban groundwater quality is impaired resulting from industrial development accompanied by onsite disposal of waste, leachate from landfills and storage areas, leakage from pipelines and storage tanks, accidental spillages, and the demolition of disused or abandoned buildings. The point of contamination is known as either point source or nonpoint source. The urban system is a complex one and the contamination can emanate from multitude of point source that are widespread and diffuse that it becomes difficult to predict the exact source of contamination. Good examples are lawn fertilizers and insecticides used for termite control (Garcia-Fresca and Sharp, 2005, Tompson et al., 1999, Thomas and Tellam, 2006).

2.6.6 Changes in recharge and discharge

It is widely recognized by the hydrologic community that groundwater recharged can be stopped in urban areas as impervious cover enhances runoff and limits infiltration. However, there are several sources of recharge as a result of urban development which include the following: leakage from water and wastewater distribution and collection systems, leaks from storm sewers, and irrigation return flow from lawns, parks, and golf courses. As a result of increased discharge of wastewater, Santa Ana River in Orange County, California has recorded about 350% increase in base flow. A study carried out in the City of Austin, Texas with strontium isotopes showed that streams found in the most urbanized section of the city have over 90% of the flow at normal base-flow conditions was originally treated water from the city's distribution systems. The net recharge to urban areas increases more than the natural recharge rates (Garcia-Fresca and Sharp, 2005).

2.7 Pathways and routes of urban recharge

There are more routes for urban recharge than for rural recharge. The urban environment is characterised by abundance of impermeable areas such as roads, and also a dense network of drainage channels and drains that carry away stormwater. Moreover, the numerous pathways associated with urban environment make it difficult to quantify recharge. The various sources of recharge should be identified prior to estimation of urban recharge. The total recharge in a city is the sum of the direct, indirect, localized, and artificial components (Lerner, 2002, Garcia-Fresca and Sharp, 2005).

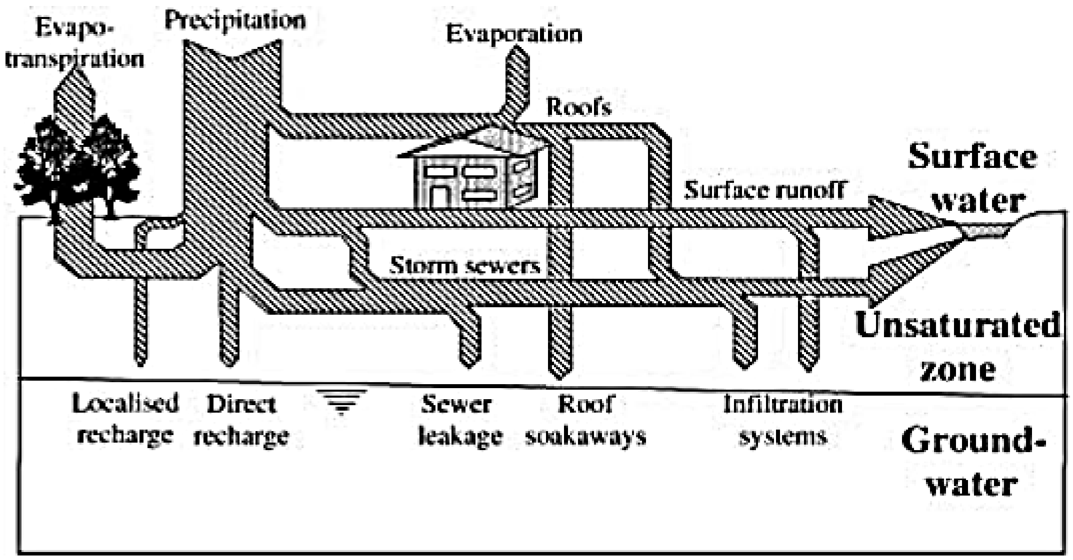

Figure 2. 5: Pathways for precipitation to recharge groundwater in urban areas (Lerner, 2002).

The above Figure 2.5 shows some of the many routes by which precipitation recharges groundwater in urban areas. Parks and gardens are recharged via direct recharge while localised recharge takes place along the edges of paths and roads where no formal storm drainage exists. Roof runoff infiltrates via soakaways, and infiltration basins and boreholes serve to dispose of some storm runoff. There are many more routes for urban recharge than for rural recharge. The multiplicity of locations where each route occurs as a consequence of the high variability of land use and the complexity of the water – carrying infrastructure is not captured in Fig. 2.5. Figure 2.6 summarises how complex and difficult it is to identify and quantify urban recharge when precipitation is considered as a source (Lerner, 2002).

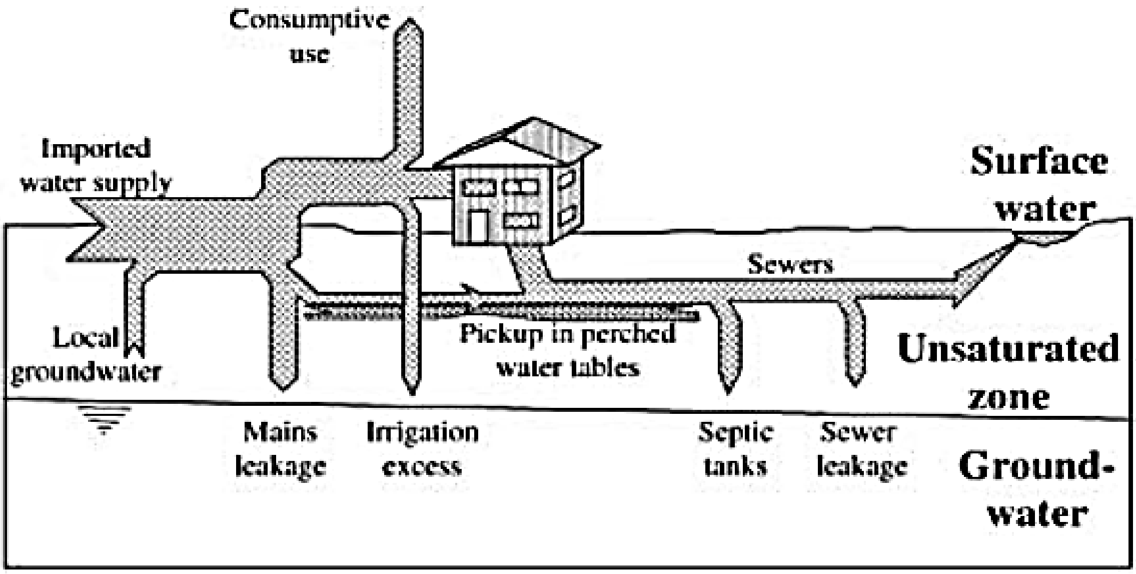

Figure 2. 6: Routes for water supply and sewage to recharge urban groundwater (Lerner, 2002).

Figure 2.6 illustrates the main routes to recharge urban groundwater. In the absence of sewers to take wastewater away, the most important recharge route would be the infiltration of wastewater from large numbers of septic tanks, latrines, and soakaways. Under this circumstance, majority of the imported water recharges the aquifer. If sewers are used to remove effluents, much of the imported water is re-exported and does not become recharge. There could be leakage from sewers which enhances groundwater recharge. Over-irrigation of parks and gardens in high –income arid climates cause the excess water to recharge the groundwater. The determination of losses from water –supply systems is also difficult as a result of the complexity of the infrastructure (Lerner, 2002).

Large urban areas have a meso-climate which can change rainfall and evapotranspiration rates. These are likely second order effects on recharge when compared to the changes caused by surface coverings which reduce infiltration, and increase and accelerate runoff. This runoff is carried in facilities such as storm sewers, drains, or other artificial waterways. Sewers and drains which are designed to carry water out of the city have the tendency to recharge. This is due to the fact that they do leak to the environment. A good example of such is in Hong Kong where sewers are known to leak. A study carried out at Liverpool, UK showed that storm water leaked from sewers into the groundwater system, more than replacing the reduction in direct recharge due to impermeable cover. At times, storm water is intentionally recharged. There is no water supply network that does not leak. Few authorities have achieved a success rate of reducing leakage below 10% of water supply. The leakage from water supply network can generate a potential recharge of up to 3000 mma^{-1} . This type of leakage exists in Hong Kong and Lima, Peru. These were deduced from hydrochemical evidence of mains water in groundwater and piezometric evidence under completely paved areas. In the city of Birmingham, UK trihalomethanes have contaminated groundwater due to leakage from mains. Recharge rates of 100 -300 mma^{-1} are common in water supply network.

Groundwater pollution below cities by sewage is as result of leakage of sewers. Sewers are few in urban areas as most rely on septic tanks and soakaways to dispose of effluents- this water must recharge groundwater. For instance in Bermuda, which has an average rainfall of 1460 mma^{-1}, water supply is from roof catchments and sewage is disposed to septic tanks. The combination of these and soakaways for storm drainage increase recharge from 365 mma^{-1} in rural areas to 575 mma^{-1} in urban areas (Lerner, 1990).

In sewered cities, leak from water mains is more important than the effects of sewers. In some cities, rates up to 50% have been recorded, causing large amounts of recharge. In Tomsk, Russia, it is reported that 4-11 leaks develop annually per kilometre of water main, thereby causing leakage rates of 15 – 30%. In Goteborg, Sweden, the leakage rate is about 26% (Lerner, 1990).

Table 2. 3: Relative sizes of inputs to the urban hydrological networks (Lerner, 1990).

City	Area (Km^2)	Date	Precipitation	Imports	Local groundwater	Units
Urban Sweden	4024	1970	701	235		mm
Mexico City		1980	86	14		%
Hong Kong	1046	1971	1912	1310	64	mm
Hong Kong	0.61-0.35	1980	2000	650-7500	0	mm
Sydney	1035	1962-1971	1150	333	16	mm
Vancouver	0.21	1982	1215	576	0	mm
Lima	400	1978	10	1650	950	mm
Doha, Qatar	294	1981-1982	167	175	27	mm
Birmingham	500	1985	730	675	30	mm

NOTE:-Many of the areas are for a supply zone and include rural and semi-rural land

Natural groundwater chemical evolution is dominated by water – rock interactions. Therefore, there is the need for effective identification and quantification of different recharge sources which requires substances or tracers that can better discriminate between anthropogenic from natural sources (Rueedi et al., 2009). Piezometry and chemical signatures are groundwater-based methods for detecting recharge at a local scale. The quantification of urban recharge at a regional scale involves identifying sufficient individual sources of, for example, water-main leakage exist to have an impact on overall urban recharge rather than on individual points of recharge. The only viable means of quantifying it at this scale is by chemical signatures and water balances (Lerner, 2002).

2.8 Current methods for urban recharge estimation

There is variety of methods for estimating groundwater recharge and the problem associated with that is determining which technique that is likely to provide robust and reliable estimates. The factors that impact on recharge should be taken into account in choosing a method of quantifying recharge. A good understanding of the attributes of the different techniques is critical. Recharge estimates are obtained over varying space and time scales and that should guide in the choice of method. Recharge techniques are classified into physical, tracer, or numerical-modelling (Scanlon et al., 2002).

2.8.1 Groundwater modelling

Modelling groundwater recharge in rural systems is daunting because of the uncertainties associated with quantifying of indirect and localized recharge, as well as the complexity of the processes that take place in the vadose zone. The estimation of recharge in an urban environment is more complex than in a rural setting as a result of the complexity of the routes and pathways (Lerner, 2002). The estimation of total recharge can be accomplished with groundwater modelling but it requires a specific approach to identify the contribution of each particular source to the total recharge. The steps involved are: 1) Definition of the conceptual model, 2) Implementation into a numerical model, 3) Parameterization of areal recharge, 4) Calibration, and 5) Iteration (Tubau et al., 2017).

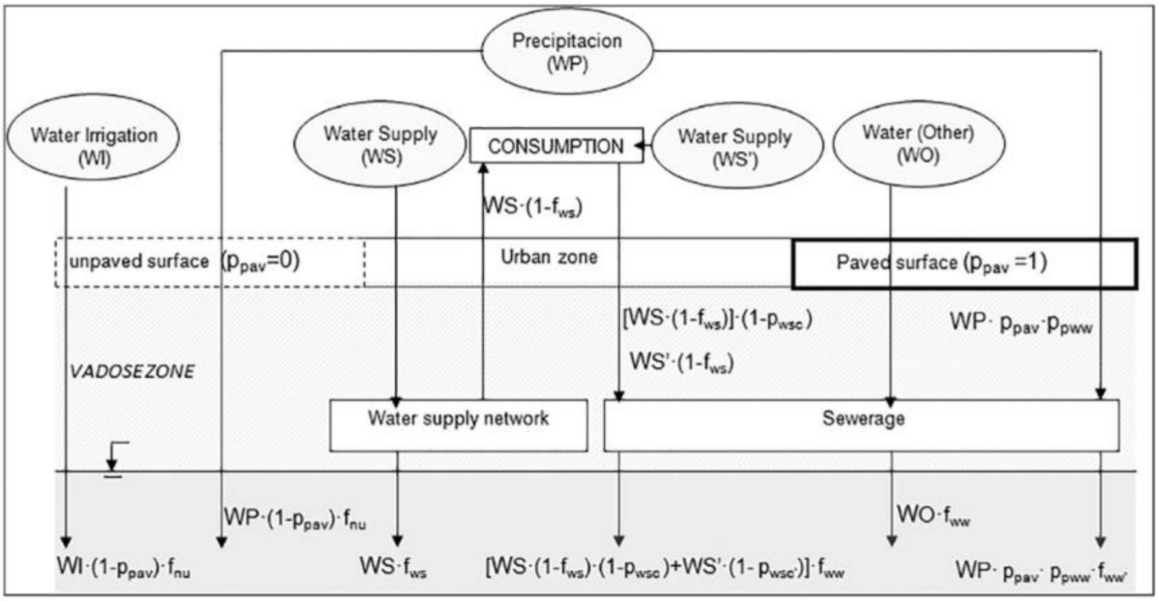

Figure 2. 7: A typical conceptual model to assess groundwater recharge in urban areas (Tubau et al., 2017).

2.8.2 Water balances

It is possible in certain situations to confirm the existence of a source of recharge by a water balance on the source water. For instance, leakage rates of 25% from water mains would have a big impact in the water balance of the water supply. When the recharge is a small proportion of the water balance, such a method does not work coupled with the uncertainty of data, or when alternative destinations exist for the water (Lerner, 2002). The traditional method used to estimate urban recharge is a combination of water balance with groundwater modelling. Due to the nature of the urban environment, most examples do not consider all the sources and routes of recharge (Yang et al., 1999).

Recharge is estimated from the different components of the hydrologic budget. On the impact of precipitation, part of precipitation on the basins infiltrates through the soil zone to the water table and becomes groundwater. Some of this groundwater is discharged to the streams as baseflow while some is lost to the atmosphere via evapotranspiration. In a given period of time, precipitation reaching the water table (recharge) is balanced by baseflow (groundwater discharge to the stream), seepage to deeper aquifer units, and evapotranspiration, plus or minus changes in groundwater storage which is stated mathematically as shown below:

$$R = BF + ET + S + St \qquad 2.1$$

Where :

R= Groundwater recharge

BF= Groundwater discharge

ET= Evapotranspiration

S= Subsurface seepage out of the basin

St= Change in groundwater storage

2.8.3 Chemical signatures

It is a tracer method of recharge estimation. Chemical signature of water is defined as the concentrations of various solutes in water, or the ratios between solute concentrations. The use of such signatures to identify waters and their origins in the field of hydrogeology is wide. The markers can be used to identify various sources of urban recharge. An ideal recharge marker is an easily analysed solute that can be used only to identify one source and its pathway, at a constant concentration in the source, and non-reactive in all conditions species (Lerner, 2002). The four categories of potential marker solutes are:

I. Inorganic –they are further grouped into major cations and anions, nitrogen species, metals and other minor ions. The elements that make up major cations (Ca, Mg, K, and Na) and anions (HCO_3, SO_4 and Cl), nitrogen species (NO_3 and NH_4), metals (Fe, Mn, and trace metals), and other minor ions (B, PO_4, Sr, F, Br, and CN).

II. Organic –the most relevant of this group of potential markers are chlorofluorocarbons (CFCs); trihalomethanes (THMs); faecal compounds, such as coprostanol and 1-aminopropanone; detergent- related compounds, such as optical brighteners and EDTA; and industrial chemicals, including chlorinated solvents and many hydrocarbons.

III. Particulate – this includes faecal microbiological species and various colloidal particles.

IV. Isotopic- this type is made up of mostly the stable isotopes (^{2}H, ^{15}N, ^{18}O and ^{35}S) (Lerner, 2002).

Table 2.1: Sources of possible marker species (Lerner, 2002).

Group of marker species	Potential sources of solutes					
	Atmosphere	**Geological materials**	**Agriculture**	**Mains water**	**Sewage**	**Industrial and commercial sites**
Major cations and anions	X	X	X	X	X	X
N species (NO_3,NH_4)	X		X	X	X	X
B and P		X			X	X
Other minor ions	X	X		X	X	X
Heavy metals		X			X	X
Chlorofluorocarbons (CFCs)	X			X	X	X
Trihalomethanes (THMs)				X	X	X
Faecal organic compounds					X	
Organics in detergents					X	X
Industrial organic chemicals					X	X
Microbiological species					X	
Colloidal particles					X	X

Table 2.4 shows both the potential sources of these solutes in groundwater and that multiple sources exist for most potential tracers. There are no universally applicable tracers known for identifying recharge sources. The use of a tracer, particularly isotopes is to identify a source of urban recharge. The combination of multiple tracers is used to increase confidence in the outcome, despite the uncertainties involved in determining the origin of each individual tracer (Lerner, 2002, Barrett et al., 1999).

2.8.4 Piezometry

Piezometry method is a physical method of groundwater estimation. Continuously flowing point sources of recharge, for example leaking mains, cause local, steady mounding of the water table. Intermittent recharge sources, such as storm-water infiltration systems, cause transient,brief responses in piezometric levels. Both types of signature are detected by means of sufficient density of measurements. One problem associated with this method is to know where to put the piezometers, because the locations of the recharge points are not usually known in advance. Also, there is the problem of getting access in urban areas in order to install piezometers. To acquire such access requires the goodwill of land owners and a thorough knowledge of the subsurface infrastructure in order to avoid damage to utilities. Lerner (1986b) successfully used piezometers to detect some urban-recharge sources in the heavily built-up Mid-Levels area of Hong Kong Island. These mechanisms were partially under the control of piezometric levels and recharge sources. He installed more than 400 piezometers to record heads. The leakage from the water- supply was identified by the unexpectedly high water-table levels. Transient response of piezometers was attributed to heavy rainfall, vertical recharge, soakaway drainage, downslope throughflow along preferential pathways and leaking storm drains. It is easier to interpret transient responses in ordinary aquifers because vertical recharge responses can be separated and rapid downslope flow is not present (Lerner, 2002).

2.8.5 Water table fluctuation method

Water table fluctuation method is a physical technique of calculating recharge. The WTF method relies on the fact that rises in groundwater levels in unconfined aquifers are due to recharge water arriving at the water table (Scanlon et al., 2002, Watson et al., 2018). This method uses groundwater-level fluctuations over time to estimate recharge and it is applicable only to unconfined aquifers (Healy and Cook, 2002) . WTF method is best suited to shallow water tables that display sharp rises and declines in water levels over short time periods. Analysis of water-level fluctuations can be used to determine the magnitude of long-term changes in recharge due to climate or land-use change (Scanlon et al., 2002).

WTF is mathematically stated as:

$$R = S_y dh/dt = S_y \Delta h/\Delta t \qquad 2.2$$

Where:

R= recharge

S_y = specific yield (dimensionless).

$\Delta h/\Delta t$ is the observed change in water-table elevation.

The above equation can be seen as a linear correlation between groundwater table rise and groundwater recharge with the coefficient (specific yield). The derivation of Eq. (2.2) is under the assumption that water arriving at the water-table goes at once into storage. This implies that the impact of the lateral groundwater flow on water-level decline during a recharge event (drainage effect) is neglected, which could underestimate the actual recharge rate (Cai and Ofterdinger, 2016, von Freyberg et al., 2015). The application of Eq. (2.2) gives the total or gross recharge for each individual water- level rise. To obtain a total recharge estimate, Δh is the difference between the peak of the rise and low point of the extrapolated antecedent recession curve at the time of the peak as shown in figure 2.8.

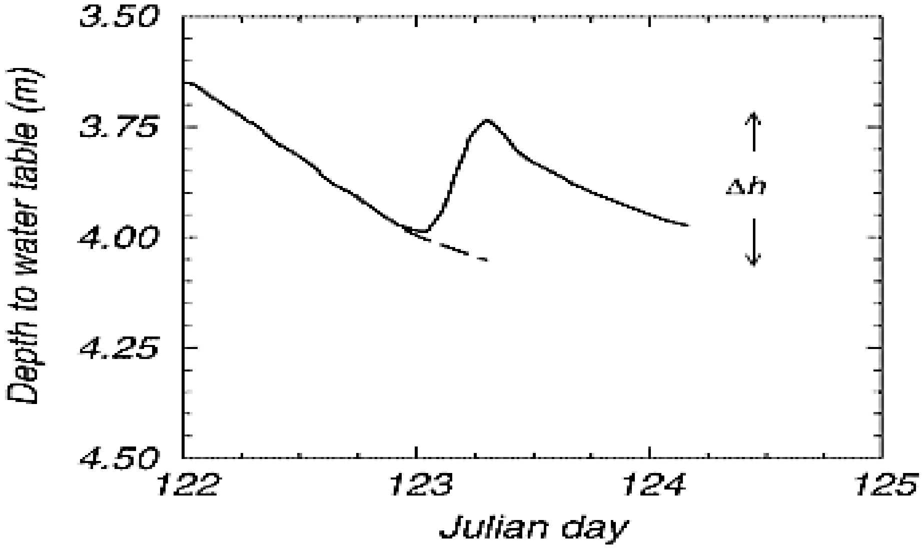

Figure 2. 8: Hypothetical water-level rise in well in response to rainfall (Δh is equal to the difference between the peak of the rise and low point of the extrapolated antecedent recession curve (dashed line at the time of the peak) (Healy and Cook, 2002).

The antecedent recession curve is the trace that well hydrograph would have followed if there is no rise producing precipitation (Healy and Cook, 2002, Oliveira et al., 2017). The antecedent recession curves are extrapolated manually by means of visual inspection of the whole data set. There are other factors that can cause water table rise such as electrical surges, changes in barometric pressure, cessation of pumping, earth tide effects, entrapped air, temperature variations, and manual adjustment to the water-level measuring instruments (USGS, 2007).

To apply Eq. (2.2), the specific yield (S_y) is required. It is defined as the volume of water released from storage by an unconfined aquifer per unit surface area of aquifer per unit decline of the water table. It is related to total porosity (Johnson, 1967).

Some of the methods commonly used to determine S_y are laboratory methods, aquifer tests, water-budget methods and water table response to recharge (Varni et al., 2013, Chinnasamy et al., 2018). Laboratory methods involve the measurement of porosity and specific retention values. There is a wide discrepancy between field obtained values and laboratory measured values. Aquifer tests provide S_y values over large areas depending on the distance between observation wells and pumping well(Johnson, 1967). The water-budget method is a water balance method where the change in groundwater storage is solved, and knowing the water table variation, S_y can be estimated (Chinnasamy et al., 2018). Finally, the response of water table to recharge is the calculation of the ratio of water table rise to total rainfall for all the registered events in the area of interest. The height of water table rise measured after a rainfall event gives an estimate of the amount of open pore space available in the vadose zone (i.e. S_y). The method is appropriate for a shallow water table (Varni et al., 2013).

Groundwater levels based techniques are one of the most widely- applied methods for estimating recharge rates. This could be attributed to the availability of groundwater – level data and the simplicity of estimating recharge rates from temporal fluctuations or spatial patterns of groundwater levels (Scanlon et al., 2002) .

2.8.5.1 Methods of determination of $\boldsymbol{\Delta h/\Delta t}$

The three approaches used for the determination of $\Delta h/\Delta t$ are (Nimmo et al., 2015):

- rise method;
- graphical method; and
- Master recession curve (MRC).

The rise method is applied when a given record of water level at equal time intervals, the rise for a given interval is the amount by which the water table at the end of that interval is higher than the previous interval. A rise of zero shows a water table decline (Nimmo et al., 2015). The rise method does not extrapolate for continuation of a hypothetical recession while the water table is increasing (USGS, 2007). The graphical method approach is used for hydrologic episodes instead of fixed time intervals. One advantage of this is that it can estimate and correct for unrealized recession. The effective rise due to a recharge episode is obtained as the difference between the peak water-table position and the extrapolated recession at the time of the peak. The antecedent recession curves are extrapolated manually by visual inspection of the whole data set. When precipitation is considered as the factor solely responsible for recharge, rises that are not caused by precipitation are identified and eliminated from the recharge calculations. Different users would produce slightly different recession curves as the approach is based on subjectivity unlike other WTF approaches. It is difficult to use the graphical method when water levels are fluctuating rapidly and are followed closely by multiple recharge events such as in fractured rock settings (USGS, 2007, Nimmo et al., 2015) Finally, the MRC is used to express the rate of water table decline as a function of H. It is a complex setup unlike the rise and graphical methods. It is very sensitive to measurement frequency (Nimmo et al., 2015). Both the rise and MRC can be applied only at sites where there is continuous water level monitoring (USGS, 2007). This research will adopt the rise method in calculating $\frac{\Delta h}{\Delta t}$.

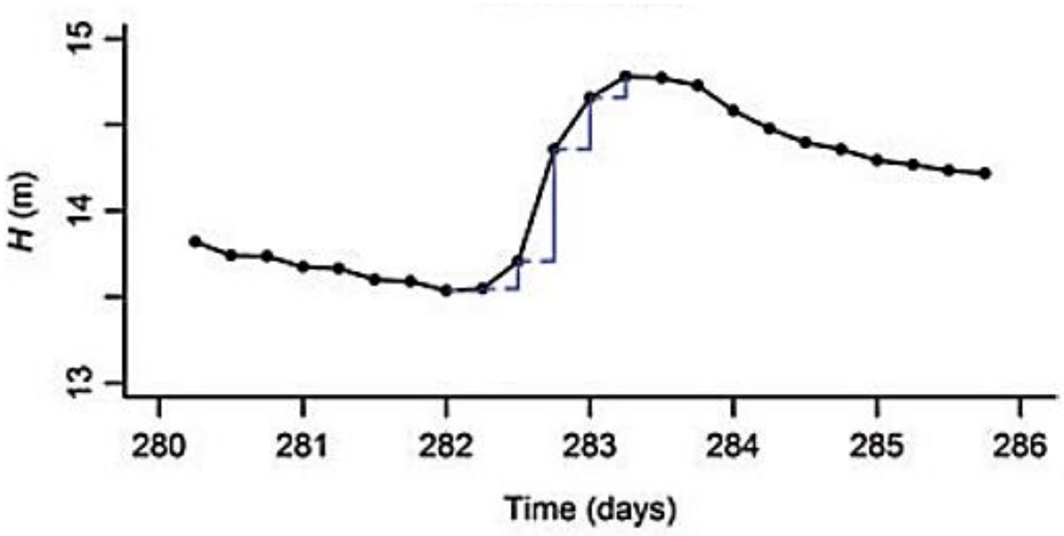

Figure 2. 9: Rise method for determining Δh/Δt (Nimmo et al., 2011).

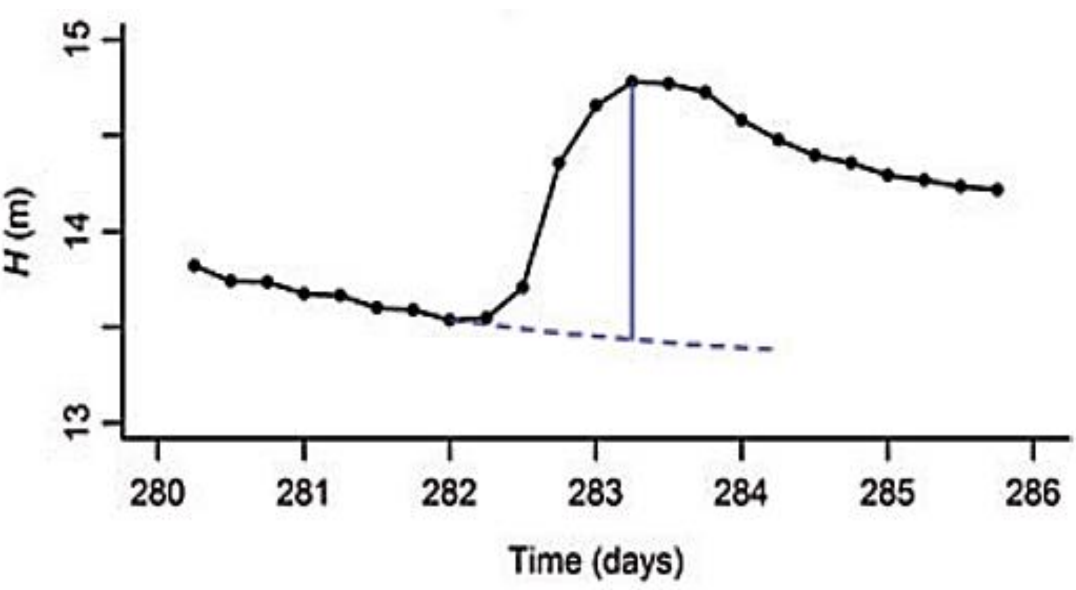

Figure 2. 10: Graphical method for determining Δh/Δt (Nimmo et al., 2011).

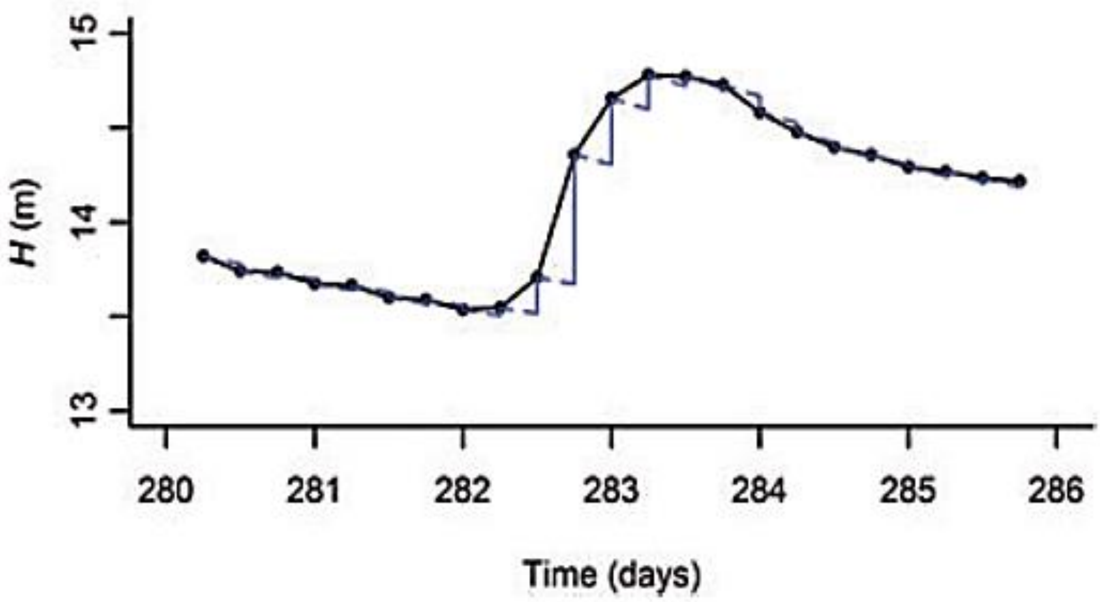

Figure 2. 11: MRC method for determining Δh/Δt (Nimmo et al., 2011).

2.9 Knowledge gaps

A review of available literature on the use of water table fluctuation (WTF) method in estimating recharge has revealed that some of the studies (Cai and Ofterdinger, 2016, Varni et al., 2013, von Freyberg et al., 2015) assumed that rainfall alone will impact the recharge. Some others assumed that head setting alone is an indication of recharge. On the contrary, it is not a sufficient criterion to quantify recharge as it may be influenced by boundary conditions (Tubau et al., 2017). Most Studies on recharge estimation via WTF have not accounted properly for lateral flow and runoff (Cai and Ofterdinger, 2016, Varni et al., 2013, von Freyberg et al., 2015, Healy and Cook, 2002, Oliveira et al., 2017). But the four boreholes were sited to account for the effect of lateral flow and runoff due to precipitation. The specific yield values which are necessary in the computation of groundwater recharge by the aforementioned method vary greatly due to variations in geology and depth to water table (Chinnasamy et al., 2018). Therefore, in order to account for these observable differences, it is necessary to determine the specific yield values at different depths and use the average value as a true representative value. This study aims to use average specific yield to estimate the recharge in an urban area.

2.10 Aim and objectives of current research

The overall aim of this research is to estimate groundwater recharge in an urban area and to study groundwater pollution at selected points. The specific objectives are:

1. To estimate the urban recharge via WTF by incorporating a method of specific yield determination that will reduce the effects of variations in geology and depth to water table.
2. To determine the lag time between the change in water table depth and rainfall by the use of cross- correlation technique.
3. To collect grab samples from the boreholes, and test for various water quality parameters which are assessed against established guidelines for groundwater use.

2.11 Scope of research

The scope of this research comprised:

- Marking out of the four monitoring points with a GPS.
- Certifying of the four marked points free from underground utilities by a professional utility locating company (Onpointlocating).
- Engaging a professional drilling company (Matrix) for the bores drilling.
- Collection of water table depths data from the boreholes using data loggers
- Samples from the boreholes were collected on a monthly basis and tested for various water quality parameters.
- Laboratory determination of specific yield and soil classification
- Recharge estimation using water table fluctuation (WTF) method

CHAPTER 3

MATERIALS AND METHODS

3.1 Introduction

This chapter details the methodology used which includes:

- Collection of soil samples at different depths during drilling and soil sample preparation for analysis
- Soil analysis methods such as particle size determination, sedimentation test, volumetric ring test and specific retention analysis
- Development of boreholes for groundwater monitoring
- Groundwater sampling and physico-chemical analysis of groundwater such as electrical conductivity, pH, dissolved oxygen, turbidity, total solids, total dissolved solids, suspended solids and cations (Na^+,Ca^{2+}·mg^{2+}, K^+, Fe^{2+})
- Data acquisition (water table depth and rainfall)
- Statistical analysis of ground water-level response to rainfall
- Application of WTF to estimate Wattle Grove urban catchment recharge

3.2 Study area, soil collection and soil analysis

3.2.1 Study area

The area of study is located between 33°56'57.7"S to 33°56'56.2"S latitude and 150°56'25.4"E to 150°56'22.6"E longitude (Maps, 2019) . Wattle Grove is a suburb in inner Sydney of New South Wales, Australia. It is located about 36 Km south west of the Sydney CBD and approximately 5 Km from the Liverpool CBD. Wattle Grove is managed by Liverpool City Council (LCC). It provides an easy access to the M5 motorway which is on the Northern Side (DHA, 2015). It is surrounded by suburbs of Holsworthy, Moorebank and Hammondville (ABS, 2016). The red border line of Figure 3.1 denotes the extent of catchment under study.

Figure 3. 1: Wattle Grove catchment map (Google, 2019).

Surrounding development consists of freestanding residential dwellings of similar design and vintage. There are shopping facilities, schools and parklands that are located within the catchment (DHA, 2015). There is a man-made urban lake known as Wattle Grove Lake that is located within the residential suburb of Wattle Grove. The lake is managed by Liverpool City Council (LCC). The lake was constructed between 1992 – 1993 and it is about 2 m deep. The area was previously used by Australian Defence force before the lake was constructed. The lake is to serve a dual purpose of improving the stormwater quality and provide some flood water storage. The lake and associated parkland is about 2.5 hectares and the catchment area containing the lake is roughly 95 hectares, with approximately 1,022 residential properties. The lake discharges excess water into the nearby Anzac creek. The parkland is landscaped and maintained for recreational activities such as exercising and walking by residents and other lake visitors. Liverpool City Council has installed 3 aerators and 2 fountains to improve the lake's water quality and make the surrounding area more aesthetically pleasing. The lake does not allow for primary contact by visitors because it is majorly used as a treatment system for stormwater (Hagare et al., 2015, Natarajan et al., 2018).

3.2.1.1 Hydrogeology

In Australia, all major aquifers are grouped by upper, middle and lower levels based on age and stratigraphic position. According to Bureau of Meteorology (BoM) groundwater insight map, Wattle Grove catchment is located within the Moorebank catchment. The principal hydrogeology can be described as porous, extensive aquifers of low to moderate productivity. The catchment aquifer belongs to the lower groups. The lower aquifer is composed of sedimentary rocks. The salinity change of the lower aquifer is stable (BOM, 2019).

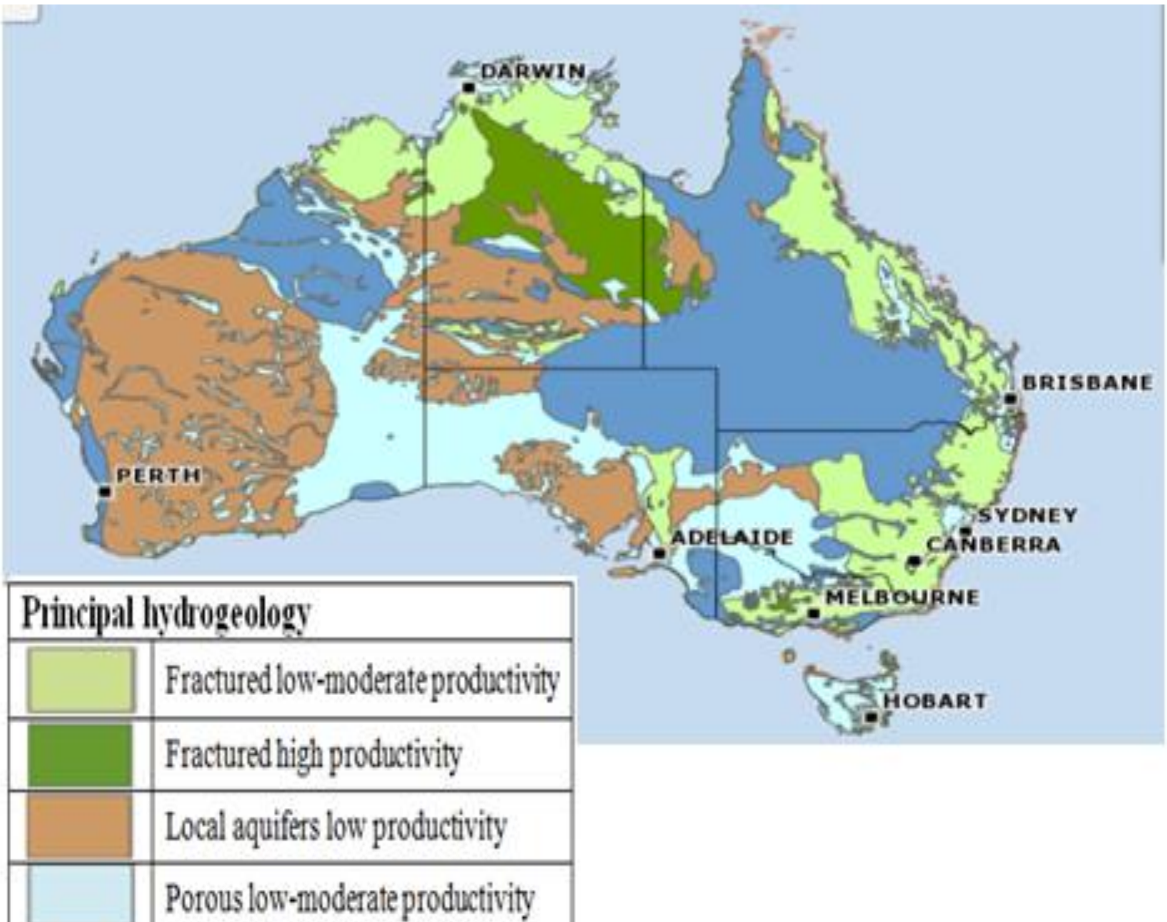

Figure 3. 2: The principal hydrogeology map of Australia showing only the productivity of the main aquifers at each location (BOM, 2019).

The site is underlain by alluvial sands, silts and clays overlying shale of the Wianamatta Group and Hawkesbury Sandstone. The groundwater flow beneath the site is towards the north-east in the direction of the Georges River as revealed by the local topography. Shales of the Wianamatta Group are typically black to dark grey shales and laminates from the Triassic period. The site lies at about 10 m Australian Height Datum (AHD). Groundwater can be inferred to be present in the alluvium and shale. Alluvial deposits are found in valleys, creeks and river beds in the region. The alluvial deposits are characterised as shallow, discontinuous and relatively permeable. It is also responsive to rainfall and stream flow. The shallow alluvium is hydraulically connected to the Georges River. Ecosystems that depend on groundwater are sustained by groundwater from the alluvium. Groundwater within the shale is more saline. Shale has a low hydraulic conductivity which makes it to behave like an aquitard thus restricting groundwater flow into the underlying Hawkesbury Sandstone unit (Brinckerhoff, 2014b). The Hawkesbury Sandstone is a dual porosity regional aquifer system found across the whole of the Sydney Basin. Groundwater flow is variable throughout the Hawkesbury Sandstone, and is generally dominated by secondary porosity and fracture flow such as faults and fracture zones. Its primary porosity rock matrix is low (Brinckerhoff, 2014a). A result of resistivity imaging conducted on Hawkesbury Sandstone reveals that it is divided into three different litho-stratigraphic units. Both the upper third and basal third of the Hawkesbury Sandstone are chiefly composed of clean quartz-dominant units while the middle unit is significantly more silty. Geological logs also reveal that the thickness of the Hawkesbury Sandstone aquifer is about 140 to 160m (Milne-Home, 2009). The Hawkesbury Sandstone has low matrix permeability and that implies water yielding potential is dependent upon fractures and other structural features that improve permeability. The yields from bores in the sandstone are within 0.2 to 11.3 litres per second, with an average yield of 1.3 litres per second. The water quality is generally good and salinity ranges from 100 – 1000 mg/L TDS (DoTRD, 1997). Figure 3 shows the geologic features of the aquifer in the study area.

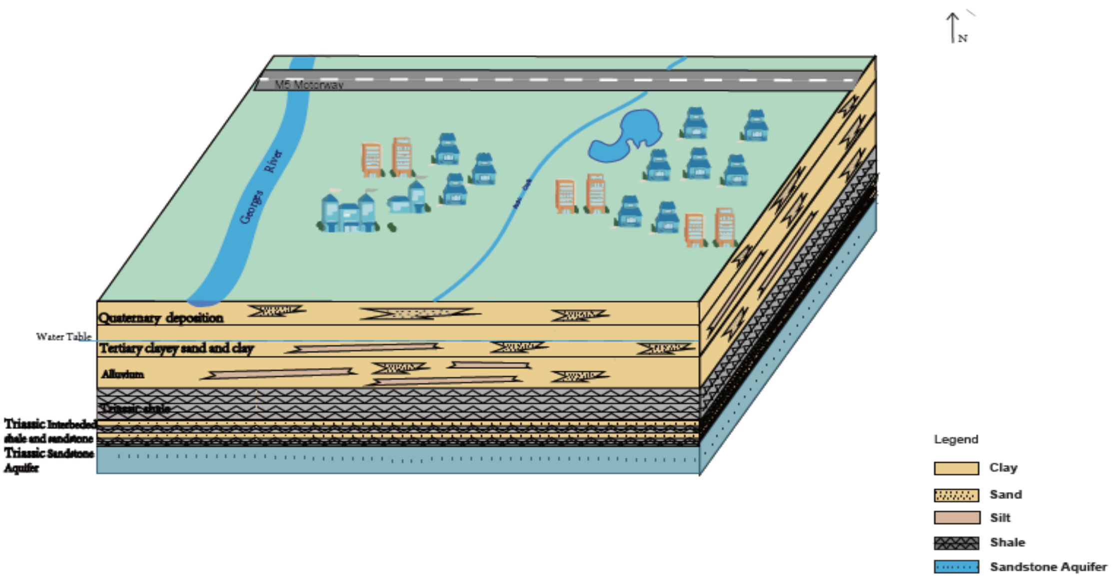

Figure 3. 3: Schematic diagram of geologic features of the aquifer in the study area (Dake, 2019)

Table 3.1: Summary of regional Permo-Triassic geological stratigraphy (EMM, 2016)

Period	**Group**	**Sub-group**	**Formation**	**Description**	**Average thickness (m)***
Quaternary			Alluvium	Quartz and lithic fluvial sand, silt and clay	< 20
Tertiary			Alluvium	High level alluvium	
Triassic	Wianamatta Group		Bringelly Shale	Shale, carbonaceous claystone, laminate, lithic sandstone, rare coal	80 (top eroded)
			Minchinbury Shale	Fine to medium-grained lithic sandstone	
			Ashfield Shale**	Black to light grey shale and laminate	
			Mittagong Formation	Dark grey to grey alternating beds of shale laminate, siltstone and quartzose sandstone	11
			Hawkesbury Sandstone	Massive or thickly bedded quartzose sandstone with siltstone, claystone and grey shale lenses up to several metres thick	173
	Narrabeen group	Gosford Sub-group	Newport Formation	Fine-grained sandstone (less than 3 m thick) interbedded with light to dark grey, fine-grained sandstones, siltstones and minor claystones.	35
			Garle Formation	Cream, massive, Kaolinite-rich pelletal claystone, which grades upwards to grey, slightly carbonaceous claystone containing plant fossils at the base of the Newport Formation.	8
		Clifton Sub-group	Bald Hill Claystone**	Massive chocolate coloured and cream pelletal claystones and mudstones, and occasional fine-grained channel sand units.	34
			Bulgo Sandstone	Thickly bedded sandstone with intercalated siltstone and claystone bands up to 3 m thick	251
			Stanwell Park Claystone**	Red-green-grey shale and quartz sandstone	36
			Scarborough Sandstone	Quartz-lithic sandstone, pebbly in part	20
			Wombarra Claystone**	Grey shale and minor quartz-lithic sandstone	32

Note: * - Average thickness from available well data within Camden Gas Project (CGP)

** - Aquitard or aquiclude

3.2.1.2 Boreholes location

The four points to be drilled were marked out with the aid of GPS and pegged with wooden pegs. These boreholes are named as BH1, BH2, BH3 and BH4. The BH4 is located on the downstream of the Wattle Grove Lake while the BH1, BH2 and BH3 are located on the upstream side of the lake. As the standard practice in Australia, dial before you dig was contacted before a professional drilling company (Matrix) was engaged. They furnished us with all the necessary details including maps and companies with their associated utilities in the catchment. We sieved through all that and engaged the services of a registered utilities searching company (Onpointlocating) to conduct search around our marked points. Utilities were not found in BH4 and BH3 but in BH1 and BH2. Due to the presence of utilities in BH1 and BH2, we moved a little bit away to new BH1 and BH2. The locations of these boreholes are shown in Fig. 3.5 while the GPS coordinates are shown in Table 3.2 which also gives brief description of the location in which the boreholes are located.

Figure 3. 4: Confirmed marked point free from utilities

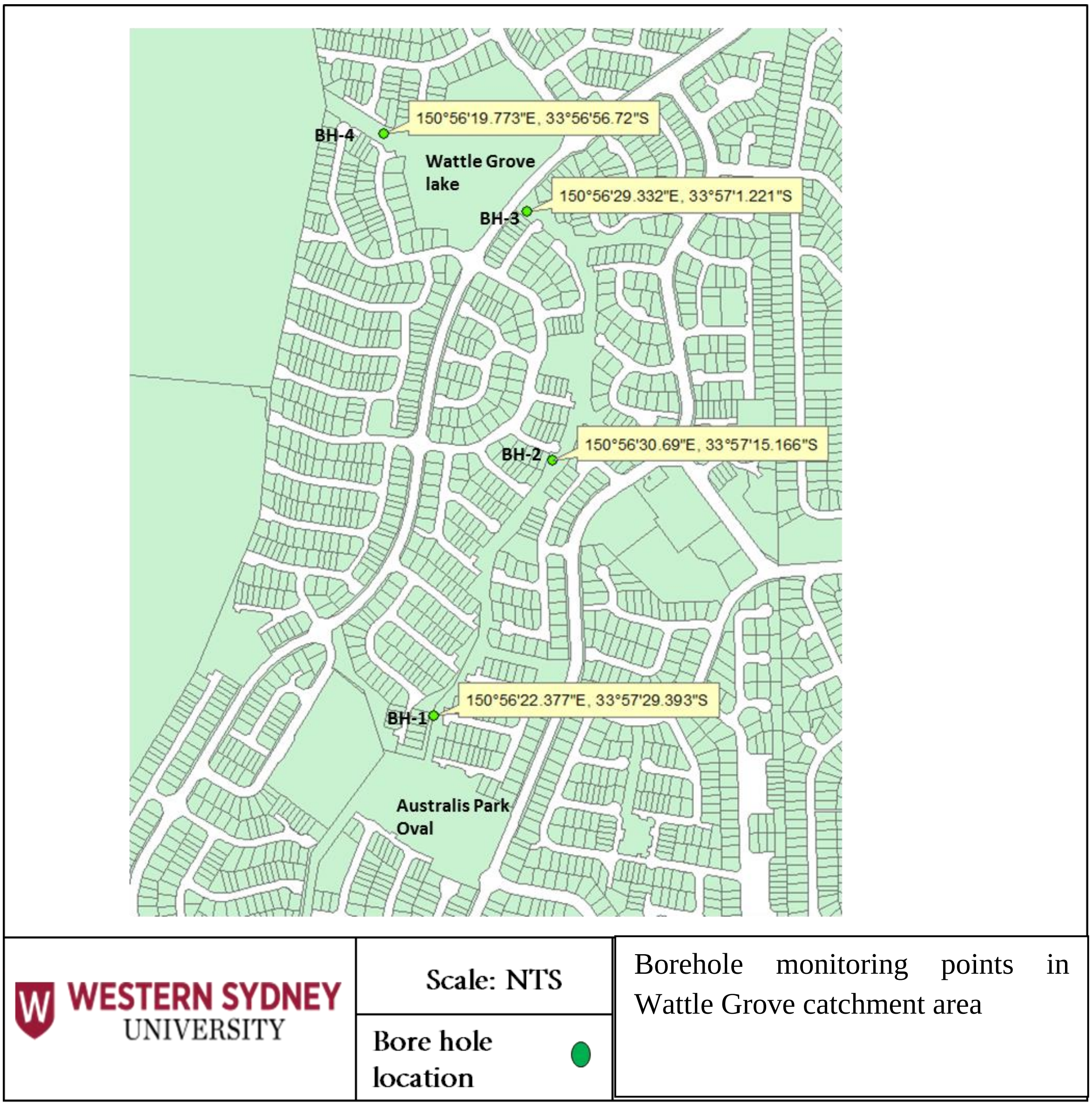

Figure 3. 5: Location of boreholes in the Wattle Grove catchment

Table 3.2: GPS coordinates of the four borehole points

Name	North (°)	East (°)	Elevation (m)	Landmark
BH1	6240566.436	309614.595	15.011	It is furthest from the lake (about 120m) and is closer to Australis park.
BH2	6241201.568	3098733.34	12.5	It is about 60m from BH1 and 50m from BH3.
BH3	6241445.934	309781.393	11.86	It is about 10m from the edge of the lake.
BH4	6241560.538	309541.895	9.075	This is closest to the edge of the lake by about 4m.

3.2.1.3 Groundwater flow direction

The groundwater flow direction within an aquifer is determined by measuring static groundwater elevations at different points within the aquifer. The force of gravity causes groundwater to flow from points of higher static groundwater elevation to lower static groundwater elevation.

Table 3.3: Boreholes hydraulic head

Borehole	**North (º)**	**East(º)**	**Elevation (m) AHD**	**Borehole top to ground(m)**	**Water table from borehole top (m)**	**Elevation+ Borehole top (m)**	**Reduced level of groundwater table(m)**
BH1	6240566.436	309614.595	15.011	0.83	3.41	15.841	12.431
BH2	6241201.568	3098733.34	12.5	0.67	3.6	13.17	9.57
BH3	6241445.934	309781.393	11.86	0.57	3.64	12.43	8.79
BH4	6241560.538	309541.895	9.075	0.45	1.03	9.525	8.495

From the reduced level, it shows that the groundwater flows from BH1 towards BH4 that is towards Georges River on the north-west side of the lake

3.2.1.4 Bore drilling and development

Sectional Flight Auger (SFA) drilling method was used for the drilling. Monuments were installed in each of the boreholes and locked up with padlock in order to prevent vandalism. According to Sundaram et al., (2009), 'all groundwater bores should be drilled, cased and equipped according to national construction standards defined in Minimum Construction Requirements for Water Bores in Australia' (Sundaram et al., 2009). Drilling, construction and development of bores can have a significant impact on the quality of groundwater samples extracted from them, through the introduction of physical or chemical effects or unwanted residues. The use of rotary auger in drilling of boreholes can cause smearing of borehole walls with the tendency to transport geological formation materials and drilling fluids into different zones. The effect of this is the blockage of groundwater and contaminant pathway which invariably excludes contamination from the monitored material. Drilling fluids are used during the drilling process in order to remove cuttings from the borehole, to clean and cool the drilling bits, reduce friction between the drill string and the sides of the borehole and to hold the borehole open during the drilling operation. Some of the drilling fluids used include air, water and mud. They all have impact on the groundwater quality and the effects of each are stated below.

1. Air can cause oxidation and precipitation of analytes of interest, such as dissolved metals and may cause serious disturbance of hydrochemical profiles in highly permeable formations.
2. Water can cause bore dilution or flush groundwater near the bore thereby changing the groundwater chemistry and also cause precipitation of minerals which results in blocking of contaminant and groundwater pathways.
3. Mud may enter the formation and seal preferential groundwater pathways.

Figure 3. 6: Sectional flight auger drilling

The standard practice is that the act of developing the bore may reduce the impacts of the drilling technique. Bore development is the process of removing fine sand, silt and clay from the aquifer around the bore screen and breaking down drilling mud on the borehole wall. The aim of bore development is to maximise the hydraulic connection between the bore and the formation. It involves the pumping out of water from the borehole until it is visibly clean and of a constant quality. The development process for monitoring bores should not introduce unwanted substances such as water or other materials into the aquifer (Victoria, 2000).

Before the four boreholes were developed, samples were taken and tested for NEPM metal suites by ALS laboratory Smithfield. The samples were collected on the same day they were taken to ALS laboratory Smithfield for analysis to minimise the variation of groundwater properties. The results of the laboratory analysis were checked against Australian Drinking Water Guidelines (ADWG) for health and aesthetic (ANZECC, 2000). The concentration of Manganese in BH1 as highlighted in red colour exceeded both the aesthetic and health guideline while the other analytes in all boreholes are below the threshold of health and aesthetic guidelines. The results are given in Table 3.4 while the bore log data is given in Table 3.5.

Table 3.4: Concentration of NEPM metals in groundwater

Analyte	BH1	BH2	BH3	BH4	Health guideline (mg/L) (ANZEEC, 2000)	Aesthetic guideline (mg/L) (ANZEEC, 2000)
Arsenic	<0.001	<0.001	0.001	<0.001	0.01	
Beryllium	<0.001	<0.001	<0.001	<0.001		
Barium	0.062	0.163	0.087	0.088	2	
Cadmium	<0.001	<0.001	<0.001	<0.001		
Chromium	0.002	<0.001	<0.001	0.002	0.05	
Cobalt	<0.001	0.004	0.004	0.015		
Copper	0.032	0.014	0.017	0.009	2	1
Lead	0.003	0.002	<0.001	0.001	0.01	
Manganese	2.25	0.219	0.145	0.043	0.5	0.1
Nickel	0.005	0.006	0.011	0.019	0.02	
Selenium	<0.001	<0.01	<0.01	<0.01		
Vanadium	<0.01	<0.01	<0.01	<0.01		
Zinc	0.722	0.4	0.464	0.542		
Boron	<0.05	<0.05	<0.05	<0.05		
Mercury	<0.0001	<0.0001	<0.0001	<0.0001		

The value in red indicates that the concentration of Manganese in BH1 is above ANZEEC (2000) guideline value.

Table 3.5: Bore log

Name	Hole diameter (mm)	Cased hole diameter (mm)	Metres drilled (m)	Total well depth (m)	Sand (m)	Bentonite (m)	Elevation (m) AHD	North ⁰	East ⁰
BH1	125	50	12	12	7	2.5	15.011	6240566.436	309614.595
BH2	125	50	10.5	10	7.5	2	12.5	6241201.568	309873.334
BH3	125	50	10.5	10	7.5	2	11.86	6241445.934	309781.393
BH4	125	50	9	8.8	7	2	9.075	6241560.538	309541.895

Figure 3. 7: Bore development with a twister pump

3.2.2 Soil sampling and management

The soil samples were collected at different depths (0 m – 10 m) during drilling of the boreholes with sectional flight auger (SFA) method and stored in protective plastic bags. They were then transported to the soil laboratory located at Kingswood campus of Western Sydney University and preserved effectively for analysis. The soil samples were air dried at room temperature for a minimum of three days depending on the wetness of the soil samples and weather. Finally, all lumps were broken into small particles before passage through a 2.36 mm mesh sieve for analysing soil specific retention at field capacity.

3.3 Soil analysis

3.3.1 Mechanical analysis of soil samples

Mechanical analysis is a method of determining the size range of particles present in a soil and it is expressed as a percentage of total dry weight. The two common methods are sieve analysis which is applied to soil particles larger than 0.075 mm in diameter while hydrometer analysis is for particles less than 0.075 mm in diameter. First, the soil samples were dried in the oven and about 200 g of each soil borehole depth weighed for sieve analysis. It was passed through a series of sieves in descending order of 6.70 mm, 4.75 mm, 2.36 mm, 1.18 mm, 0.6 mm, 0.425mm, 0.300mm, 0.150mm, 0.075mm and pan. This was done according to AS 1289.3.6.1(Australia, 2009a). The mass of soil retained on the pan was used for sedimentation test which was carried out according to AS 1289.3.6.3 (Australia, 2009b). The results obtained from the aforementioned two tests were used to classify the soil according to Unified Soil Classification System (USCS) and United States Department of Agriculture (USDA) textural soil classification.

Figure 3. 8: Mechanical analysis of a soil sample

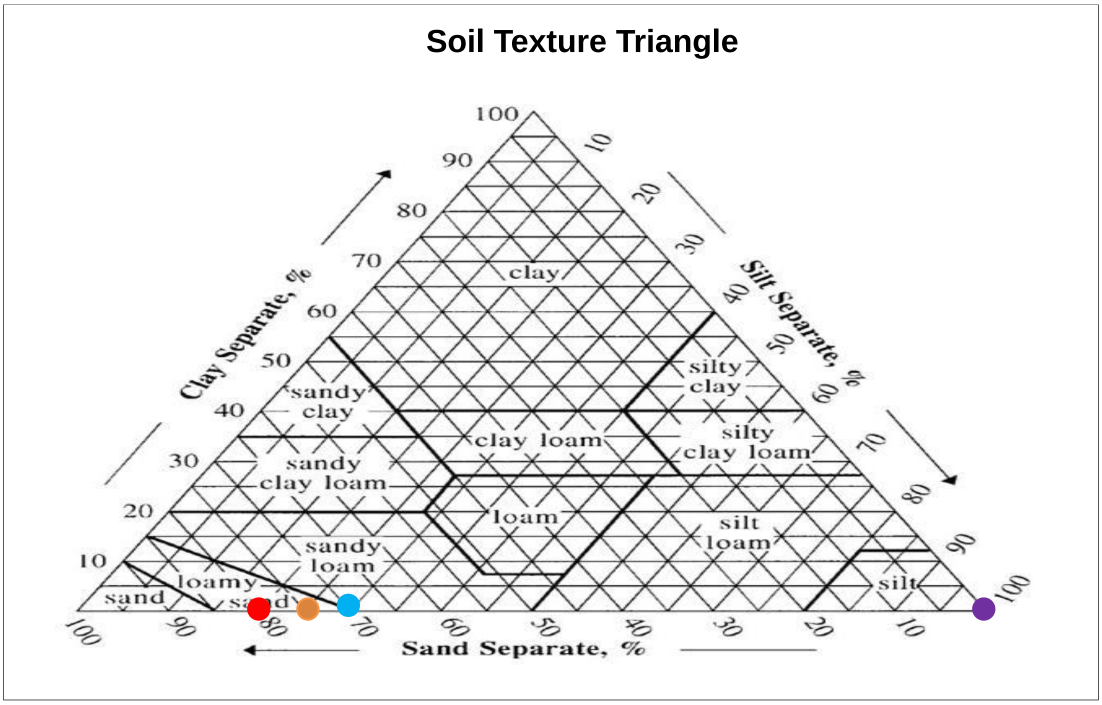

*Figure 3. 9**:** USDA soil texture triangle*

3.3.2 Determination of soil specific retention (Sr)

Specific retention is the quantity of water that a soil or rock will retain against the pull of gravity if it is drained after having been saturated. It is expressed as the ratio of the retained water to the total volume of material (Johnson, 1967). The pressure plate method was used to determine the specific retention of the soils collected at different depths during borehole drilling. The pressure that the plate was subjected to is the field capacity pressure. Field capacity is defined as the amount of water held in soil after excess water has drained away, and the rate of downward movement has decreased. It usually takes place within two to three days after a rain event or irrigation in pervious soils of uniform structure and texture. It is determined at a pressure of 0.3 bar to 0.5 bar (Johnson, 1967, Cong et al., 2014) There are three stages involved in the determination of specific retention by means of pressure plate: soil preparation, soil soaking and applying a pressure plate extractor to the saturated soil.

The following steps were followed in carrying out the analysis:

1. The soil was sieved through a 2 mm mesh sieve.
2. 50 g minimum of soil was needed to do duplicate pressure setting as each sample ring holds about 25 g.
3. Scoop about 25 g of the soil sample and pack in 1 cm high by 5.5 cm in diameter sample rings.

4. Fill the sample rings with water to saturate completely the soil samples. It is advisable to leave it overnight to ensure complete saturation (Schelle et al., 2013).
5. The sample ring was covered with plastic to prevent evaporation of the water as shown in Figure 3.10.
6. The sample ring containing saturated soil was transferred to the pressure plate at the desired pressure.
7. The suction was connected to an inverted burette in order to read drained water. When the water level in the burette is static, it shows equilibrium has been reached. It may take more than one week to reach equilibrium at low pressures (Ford, 1997).
8. The moist soil samples were transferred to moisture tins and weighed.
9. It was then oven dried at 105^{o}c for 24 hours and reweighed (Ford, 1997, Carter and Gregorich, 2008).

Figure 3. 10: Sample rings covered with plastic to inhibit evaporation after soaking

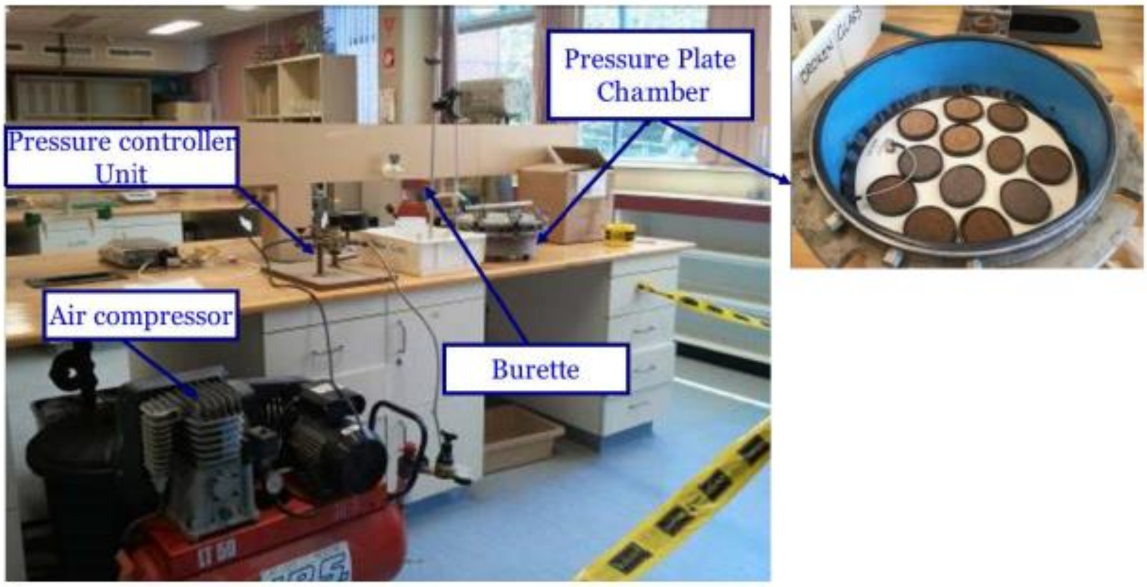

Figure 3. 11: *Setup of pressure plate extractor*

3.3.3 Porosity determination (Φ)

Porosity is the percentage of the soil volume that is occupied by pore spaces. Pore spaces are filled with air or water, or with air and water (Matko, 2003, Carter and Gregorich, 2008). The two most important parameters in assessing anthropogenic change in soil are soil density and porosity. The porosity of a soil can be obtained from the soil bulk density. The bulk density includes all pore spaces in the soil. The volumetric ring method was carried out to determine the soil bulk density which was invariably used to calculate the soil porosity. The soil surface was scrapped off with a shovel and the volumetric ring driven into the soil with a hammer. Compaction of the soil was reduced to the barest minimum because it affects porosity values. The volumetric ring (metal rod) of 50 mm internal diameter, 1.5 mm of thickness and 240 mm of height was used as shown in Figure 3.11. The exact depth of the ring was determined by measuring the height of the ring above the soil. Four different measurements were taken of the height from the soil surface to the top of the ring and the average was used as the height of the ring above the soil. The excess soil from the ring was removed with a flat-bladed knife and the sample pushed into a plastic sealable bag and marked with the different boreholes names. The volume of the ring for each collected sample was determined from its dimension. These samples were taken to the laboratory for analysis. Each bag was weighed and the collected samples were dried in soil sampler rings. The samples were put in a 105^{o}c oven and allowed to dry until a constant weight was reached. The minimum time for the drying of the soil samples was 24 hours. The removed soil sample from the ring was weighed to obtain dry weight (Carter and Gregorich, 2008, Pedrotti et al., 2005). Then the bulk density and porosity were determined from below equations:

$$Bulk\ density(g/cm^3) = \frac{Md}{V} \qquad 3.1$$

Where:

Md is the dry mass of soil sample (g)

V is the volume of ring (mm^3)

$$V = \pi \times r^2 \times h \qquad 3.2$$

Where:

π is a constant (3.142)

r is the radius of the ring

h is the height of the ring

$$\%\ solid\ space = (\frac{BD}{PD}) * 100 \qquad 3.3$$

Where:

BD is the bulk density (g/cm^3)

PD is the particle density which is universally accepted as $2.65 g/cm^3$

$$Porosity\ (\%) = 100 - (\%\ solid\ space) \qquad 3.4$$

Figure 3. 12: Soil sample collection at field for porosity determination

3.3.4 Determination of specific yield (S_y)

The specific yield (S_y) used was laboratory determined. Both specific retention (Sr) and soil porosity (Φ) have been determined earlier and the two values were used mathematically to obtain S_y. The mathematical equation is stated below:

$$Sy = \phi - Sr \qquad 3.5$$

Where S_y is specific yield; ϕ is porosity (dimensionless); and Sr is specific retention (dimensionless). However, the S_y used in Eq. (2.1) is the average specific yield ($\overline{S}_y$) which is determined by the formula given by Jinxi and Xunhong (2010) as:

$$\overline{S}y_i = \frac{\sum S_{y_{i,j}} . L_{i,j}}{\sum L_{i,j}} \qquad 3.6$$

Where $\overline{S}y_i$ is the average specific yield value for the soil samples within total test depth at site i; $S_{y_{i,j}}$ is the value of specific yield for individual soil samples within interval depth of j at site i; Li,j is the length of the soil samples within interval depth of j at site i. i refers to BH1, BH2, BH3 and BH4 while j refers to 0- 0.5 m, 1 m,...

3.4 Groundwater sampling and physico-chemical analysis

The groundwater samples were collected with the aid of a twister pump connected to LDPE tube and powered with a 12 V battery. Sterilized polyethylene bottles (1 L) were used to collect samples on each monthly visit. On site readings were taken for electrical conductivity (EC), pH, and dissolved oxygen (DO) by the use of (HQ40d, HACH, USA) probe while turbidity was measured with (2100Q Portable Turbidimeter, HACH, USA). The collected groundwater samples were taken to the university environmental laboratory and preserved accordingly for other parameters analysis. Total dissolved solids (TDS), total solids (TS), and suspended solids (SS) were performed according to standard methods given by the American Public Health Association (APHA, 2005). Ca, Fe, Mg, K and Na the five major elements which are present in groundwater were analysed with inductively coupled plasma- optical emission spectroscopy (Agilent 700 Series, Australia) by the following steps:

- 3 % HNO_3 was prepared by adding 85.7 mL HNO_3 in a 2 L volumetric flask and made up with Milli- Q (MQ) water
- The standards were prepared from multi element stock solution supplied by Agilent technologies that contain the analytes sought (Ca, Fe, Mg, K and Na).

- Ten different standards of varying concentrations not exceeding 10 mg/L as upper limit and the blank solution made up of 3 % HNO_3 were used to calibrate ICP-OES.
- The first standard placed in the ICP-OES rack is the blank followed by the other prepared standards.
- The collected groundwater samples were filtered first with 45 µm filter paper
- The filtered samples were diluted to various amounts and added few drops of HNO_3 acid.
- Finally, they were placed in the racks as well as the standards and ran in ICP-OES.
- The results obtained were multiplied with the dilution factor and the average of each analyte was used as the concentration value of the sample.

Figure 3. 13: Labelled groundwater samples collected from the boreholes

3.5 Acquisition of data

The four monitoring wells known as BH1, BH2, BH3 and BH4 were instrumented with data loggers to record hourly variation of groundwater levels. BH1 was instrumented with levelogger edge™ and barologger edge™ (Solinst, 2020). On the other hand, BH2, BH3 and BH4 were instrumented with odyssey^{R} depth and temperature data loggers (Dataflow, 2020). The odyssey depth and temperature logger is lowered in the groundwater. The pressure sensor of odyssey depth and temperature logger gives a resolution of approximately 2 mm while the temperature sensor is 0.02 degrees centigrade. The vent tube supplied with odyssey depth and temperature logger removes atmospheric pressure variations from the water height measurement (Dataflow, 2020). The Levelogger is lowered in the groundwater while the

Barologger is lowered above the groundwater. The resolution of Levelogger is quite high and with an accuracy of 0.05 % FS. It has the capability to store up to 120,000 readings. The essence of the Barologger is to compensate for atmospheric pressure fluctuations during changes in water level (Solinst, 2020). The data was logged on a monthly basis. A linear interpolation method was used to fill missing data gaps due to any unexpected data logger malfunctioning (Cai and Ofterdinger, 2016). Since the loggers are logging on one hour intervals, rainfall measurements are obtained from the nearest weather observation station Holsworthy Aerodrome Aws (066161) in one hour intervals (BOM, 2019). The distance of the station to study area is about 4.58 km. The loggers readings were calibrated against water table measurement carried out with magnetic tape on the day of logging. The hourly readings were converted to daily readings by averaging to capture the variation of water table with time. The daily rainfall and groundwater level data over one hydrological year was used in this study.

Figure 3. 14: Different loggers deployed in the boreholes

3.6 Time series analysis

According to Lee et al., (2006), 'Cross- correlation is a time series technique which can be used to evaluate the statistical correlation between two sets of data at different time lags'. The cross-correlation of daily rainfall and change in groundwater level can unmask the importance of water table response to rainfall after a given number of days. Furthermore, it can allow the time taken for the first water table response to rainfall to be calculated. Cross-correlations were calculated by the relationship:

$$\rho_{\gamma}(K) = \frac{E\{(X_t - \mu_x)(Y_{t+k} - \mu_y)\}}{Б_x Б_y} \quad 3.7$$

Where $\rho_{\gamma}(K)$ = cross-correlation at time lag k, k = 0, ± 1, ± 2,...± n time lag between the two series in days, Xt = observed rainfall at time t, y_t = observed water level at time t, μ_x= mean of rainfall series, μ_y = mean of water level series, $Б_x$ = standard deviation of rainfall series, $Б_y$ = standard deviation of water level series. Significant correlations achieved at 95% confidence are greater than the standard error ~ $2/\sqrt{N}$ where N is the number of values in the data set (Lee et al., 2006). The cross-correlation analysis shows the inter-relationship between the input and output time-series. Cross –correlation function that exhibits a maximum or minimum for a positive lag means that the input signal has some effects on the output signal. The response time is obtained from the time lag which corresponds to the maximum of the correlation function (Cai and Ofterdinger, 2016).

To account for the seasonal variation of groundwater level response to rainfall, the sliding window cross- correlation method was used. It involves the separation of the entire input and output time series data into sets of three –month windows. In our own case, we considered the wet season in Australia which occurs between October – April and that falls within summer and spring seasons. Additionally, combined spring and summer season was analysed by using a subset of six-month data to show the temporal variability of groundwater-level response to rainfall (Cai and Ofterdinger, 2016).

3.7 Application of WTF to estimate urban recharge

Equation (3.8) below is a modified version of Eq. (2.1) by replacing the specific yield (S_y) to average specific yield ($\overline{S}_y$). The below equation was used to estimate recharge.

$$R = \overline{S}y dh/dt = \overline{S}y \, \Delta h/\Delta t \quad 3.8$$

The rise method by Nimmo et al.,(2015) was used to determine Δh/Δt. $\overline{S}_y$ was determined from Eq. (3.6) as given by Jinxi and Xunhong (2010). The focus of the study was to estimate daily groundwater recharge across the four monitoring sites. The following procedures were adopted to calculate daily groundwater recharge; (i) net storage during no or less rainfall was determined by the difference in groundwater level between the first day of the month of February, 2018 and last day of the month of September, 2018, (ii) net storage during rainfall period was obtained by the groundwater level difference between first day of the month of October, 2018 and last day of January 2019, (iii) net storage per day (mm/day) during no or less rainfall was calculated by dividing the net storage during no or less rainfall by the total number of days in the months of February to September, (iv) net storage per day (mm/day) during rainfall periods was obtained by dividing the net storage during rainfall periods by the total number of days in the months of October to January, (v) the change in water table height was obtained with Eq. (3.12), (vi) the daily groundwater recharge was obtained by multiplying the average specific yield with the change in water table height as given in Eq. (3.8). In addition, recharge per day per amount of effective rainfall was determined as given in Eq. (3.11).

$$storage = (Inflow\ - Outflow) + Recharge \qquad 3.9$$

$$Recharge/_{day} = \frac{Recharge}{day} \qquad 3.10$$

$$\frac{Recharge}{day/_{Effective\ rainfall}} = \frac{Eq.4.0}{Effective\ rainfall} \qquad 3.11$$

The change in water table depth between the beginning of the month and end of the month during rainy period was calculated with the equation below:

$$\frac{Delta\ H}{Delta\ t\ during\ rainy\ period} = abs(net\ storage\ during\ no\ or\ less\ rainfall\ period) + net\ storage\ during\ rainfall\ period \qquad 3.12$$

CHAPTER 4

RECHARGE ESTIMATION

4.1 Groundwater recharge estimation and analysis

4.1.1 Estimate of daily groundwater recharge

The method used in estimating daily groundwater recharge is given in section 3.7. First, the change in water table height was calculated with Eq. (3.12) and the recharge with Eq. (3.8). The inflow and outflow were duly considered as shown in Eq. (3.9). BH1 recharge per day was 1.67 mm/day followed by BH3 with 1.16 mm/day and BH2 was 0.36 mm/day while the least was BH4 with 0.28 mm/day. It is worthy to note that there was a data logger failure between the month of February and April 2018 for BH2 and the said months were not used in estimating the recharge per day. The groundwater flow direction in the area is from BH1 to BH4 and the sites were located to capture runoff. The low recharge obtained for BH2 and BH4 which is less than 1 mm/day for each borehole could be as a result of drier soils with higher moisture holding capacity and smaller rainfall events. During the dry season (May – September) there were several rainfall events that did not lead to an increase in the water table depth especially having thicker unsaturated zone. Shallow water table limits groundwater recharge as a result of reduced capacity to capture more infiltrating rainfall as observed more especially in BH4. The soil conditions in lower elevation during wet season (October – April) could be described as having several continuous weeks of near saturation which restricts recharge during these periods particularly in the case of BH4 (Fan et al., 2014). Generally, the differences in recharge estimates across the sites are due to the sites having different surface topography, underground movement of water and the presence of water bodies.

The recharge per day per amount of effective rainfall was determined also for the four boreholes by using the methodology given in section 3.7. BH1 was 0.5% followed by BH3 with 0.4%; BH4 was 0.2% while BH2 was least with 0.1%. In conclusion, it shows the impact of effective rainfall on the recharge rates estimated with varied outcomes across the four sites.

4.1.2 Daily groundwater- level fluctuations

The daily groundwater-level fluctuations of all four boreholes and daily rainfall data obtained from BOM site location number 066161over one year is shown in Figures 4.1 to 4.4. The distribution of rainfall across Australia varies from one place to another as a result of different physiographic and climatic setting. The daily groundwater - level data and rainfall were interpreted graphically for the purpose of understanding the dynamics of the groundwater level and rainfall (Shalini et al., 2012). The various patterns seen in a hydrograph is influenced by the physical characteristics of the groundwater flow system, the pattern of rainfall and the interrelation between recharge to and discharge from an aquifer, land use change and other groundwater management strategies (Ferdowsian and Pannell, 2009).The annual change in groundwater levels was obtained by the difference between the highest water table depth and lowest water table depth recorded (Cai and Ofterdinger, 2016).

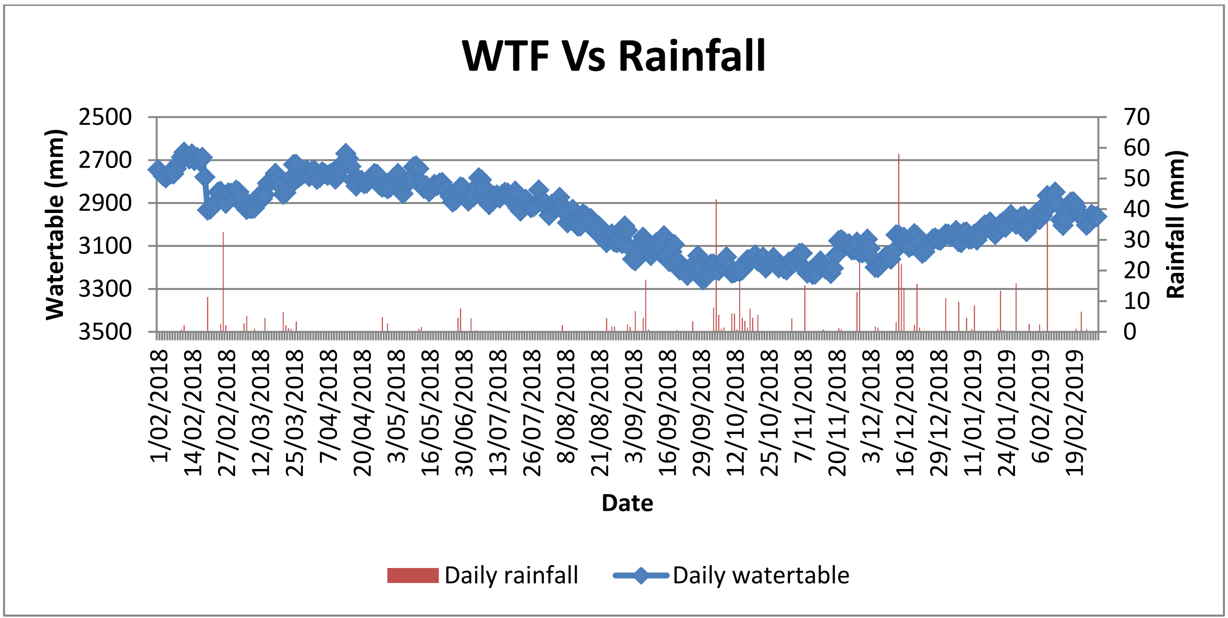

Figure 4.1 : Daily rainfall and groundwater-level data for BH1

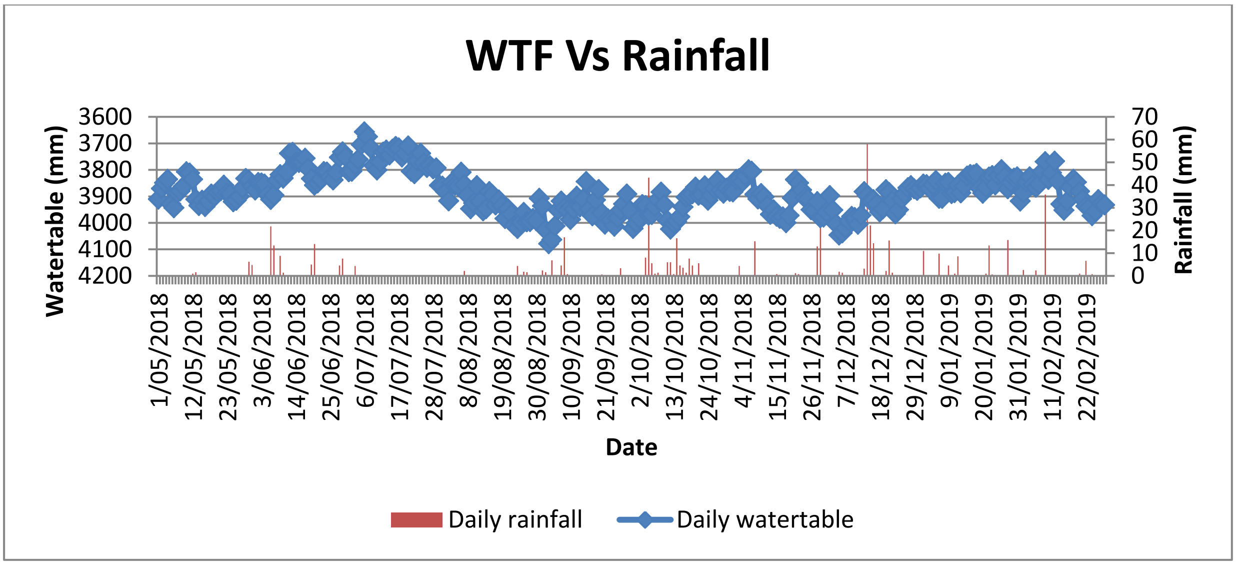

Figure 4. 2: Daily rainfall and groundwater-level data for BH2

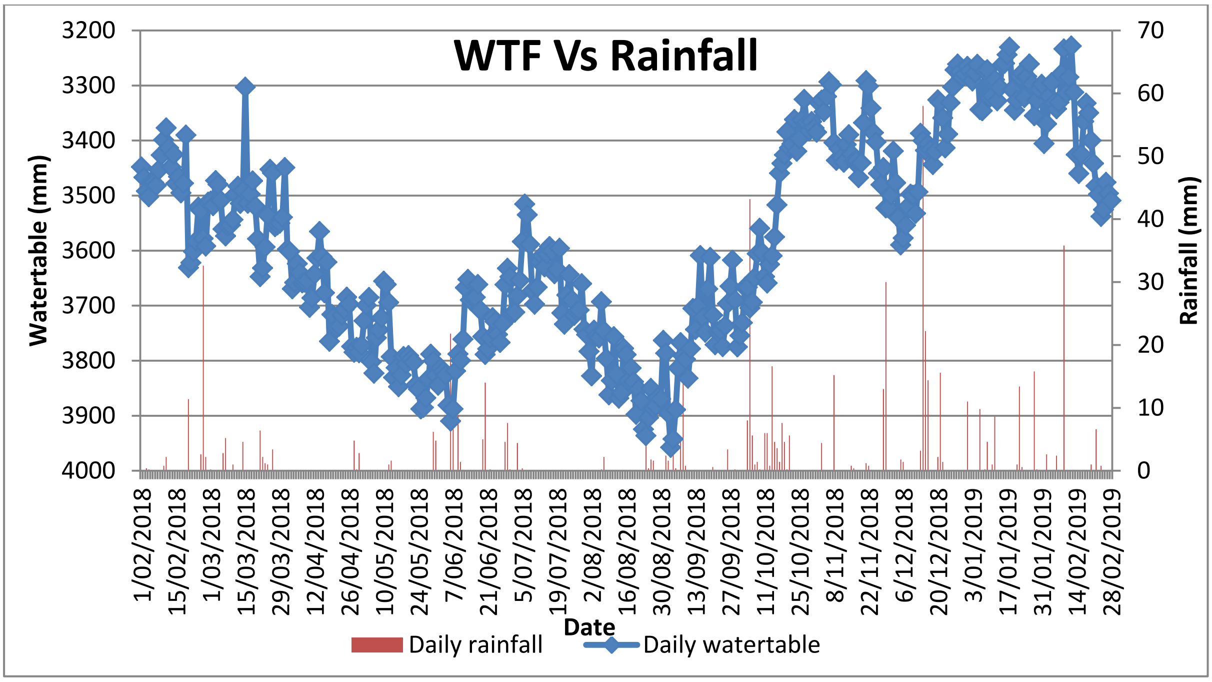

Figure 4. 3: Daily rainfall and groundwater-level data for BH3

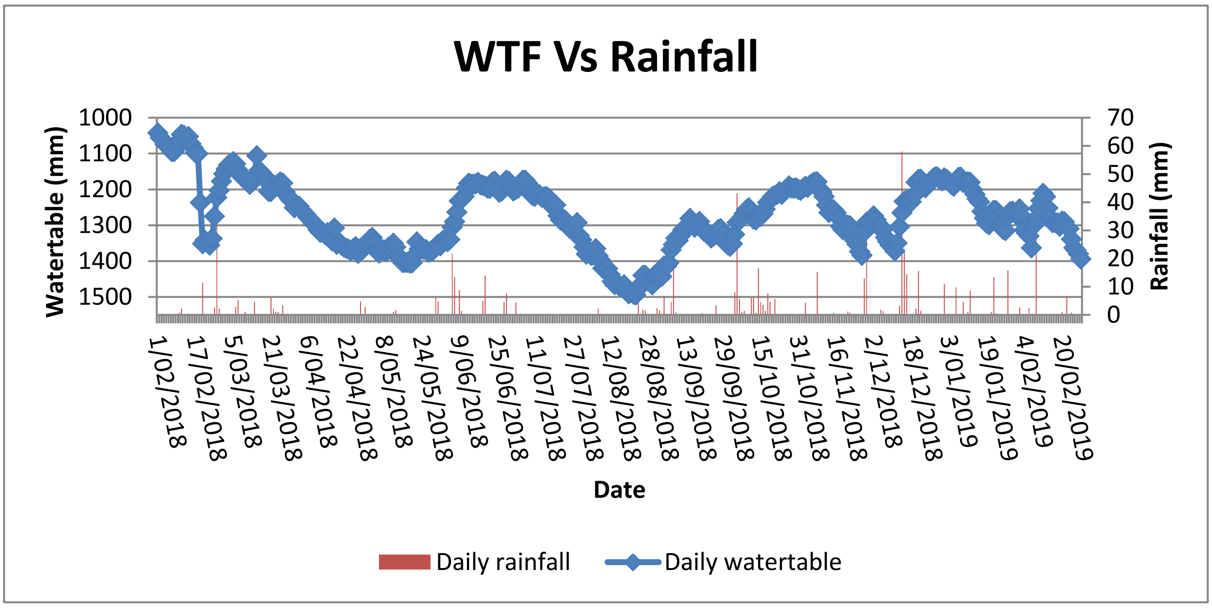

Figure 4. 4: Daily rainfall and groundwater-level data for BH4

As can be seen in (Fig. 4.1), generally the groundwater –level fluctuation of BH1 exhibits a steady decline till the end of September before it continues with a gradual increase. However, the final increase is greater than the initial by about 280 mm. The maximum water table is 2664 mm and the minimum is 3259 mm. The annual variation of groundwater level is 594 mm. It can be said that the overall response of groundwater - level to rainfall depicts a distinctive pattern for BH1 in the sense that the hydrograph exhibited smooth and seasonal change. It means that the recharge taking place in BH1 bedrock unit is dominated by the slow flow pathways. Matrix flow is responsible for these slow flow pathways (Cai and Ofterdinger, 2016). In the case of BH2, it shows an erratic pattern of increase and decrease and vice versa with flashy points. Its maximum water table is 3657 mm and the minimum is 4079 mm with an annual water table variation of 421 mm. One interesting feature of (Fig. 4.2) is that the initial water table matches the final water table value. BH2 generally showed a smooth and seasonal of change of hydrograph and also a few distinctive flashy hydrographs especially in wet season. The observed flashy hydrographs is an indication of rapid responses to individual rainfall events which is linked to a good hydraulic connection among the hydrological units (Cai and Ofterdinger, 2016).

Fig. 4.3 shows BH3 water table fluctuation as been more erratic than BH2 with more flashy points. It has a wider difference between the water table increase and decrease. The initial water table value is less than the final water table value by approximately 30 mm. BH3 minimum water table is 3228 mm and maximum is 3958 with an annual water table variation 0f 729 mm. BH3 hydrograph depicts an erratic pattern of water table increase and decrease and vice versa with series of flashy points. These flashy points could be attributed to fissures altering the soil storage capacity (Krzeminska et al., 2014). Fig. 4.4 depicts BH4 groundwater – level fluctuation which is similar to BH1. However, the difference between the two is that BH4 has more crests and troughs while BH1 clearly shows fewer. The final water table value is about 370 mm greater than the initial water table value. The maximum water table value is 1494 mm while the minimum is 1042 mm with an annual water table variation of 452 mm. The transition between water table depth increase and decrease and vice versa could be described as a smooth and seasonal change of hydrograph. The dominant flow in this bedrock unit is matrix flow due to the slow flow pathways as exhibited by the groundwater - level fluctuation (Cai and Ofterdinger, 2016).

In conclusion, BH3 has the highest annual water table variation followed by BH1, and BH4 while the least is BH2. BH3 and BH4 which are located at lower elevation show a clear declining trend of groundwater - level fluctuation unlike the other two boreholes which are located at higher elevations. The apparent groundwater-level hydrographs observed across the four monitoring sites show that different hydrogeological regimes impact the groundwater flow and storage. The overall pattern of groundwater level hydrographs shows that not only rock permeability influence groundwater-level response to rainfall but also topography and others (Cai and Ofterdinger, 2016).

4.1.3 Beginning and end of month change in water table depth (WTD)

The first and last day of the month change in water table depth was estimated for the four boreholes by the rise method. Furthermore, correlation was carried out to determine the dependence between the first and last day of the month change in water table depth against total monthly rainfall. The figures below show the results obtained:

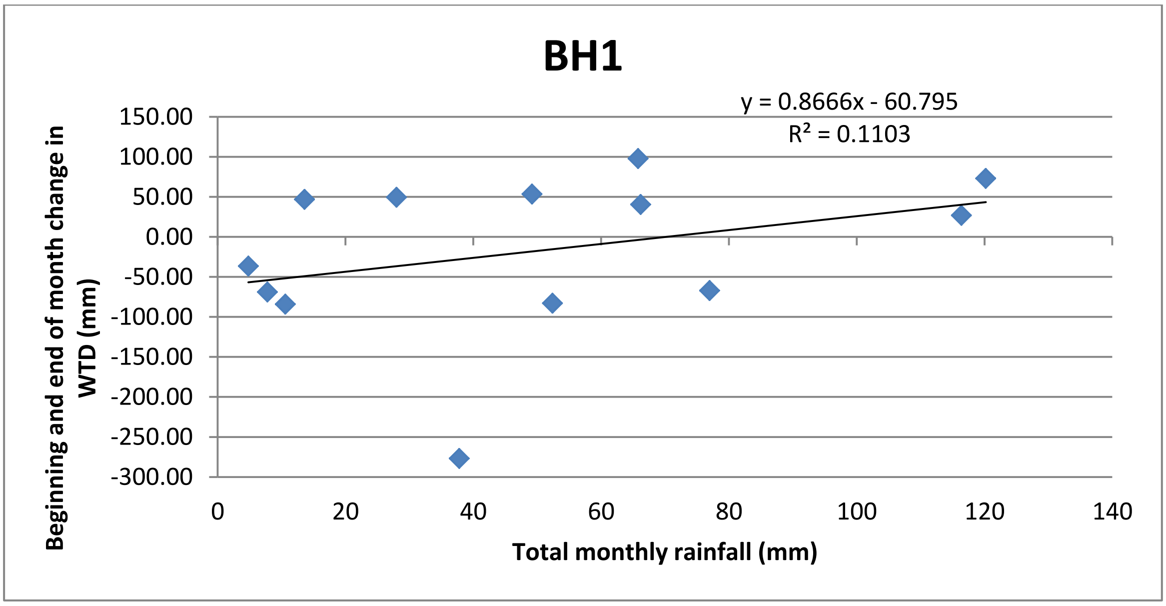

Figure 4. 5: BH1 Monthly beginning and end of month groundwater-level difference against total monthly rainfall

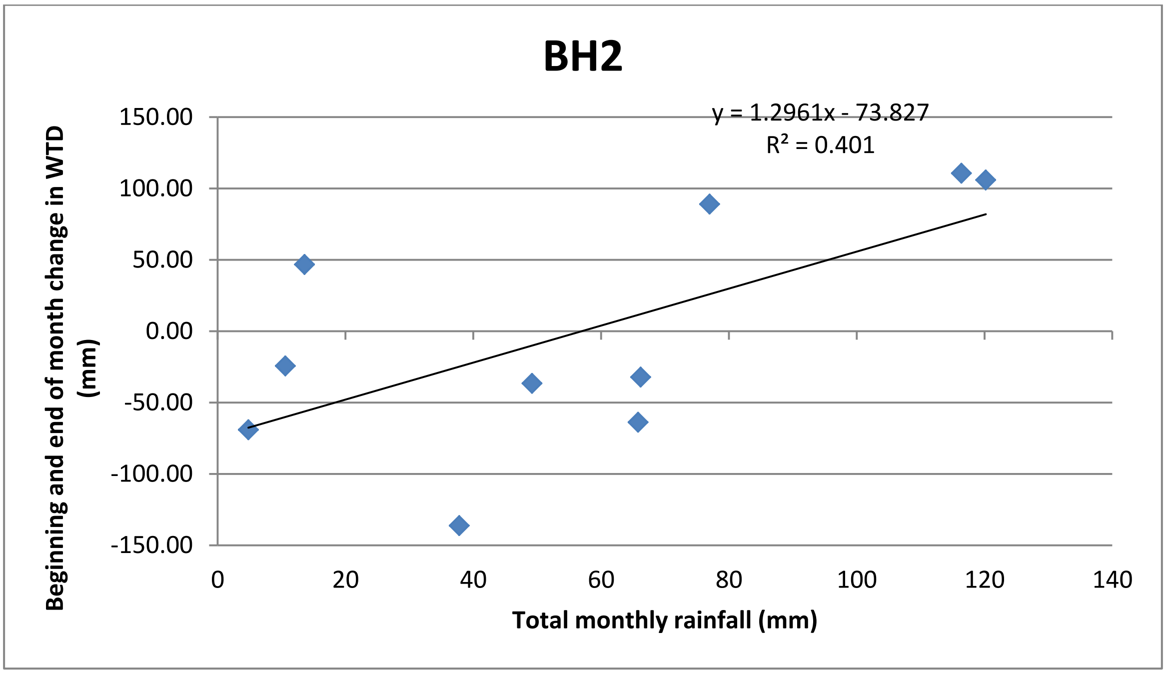

Figure 4. 6:BH2 Monthly beginning and end of month groundwater-level difference against total monthly rainfall

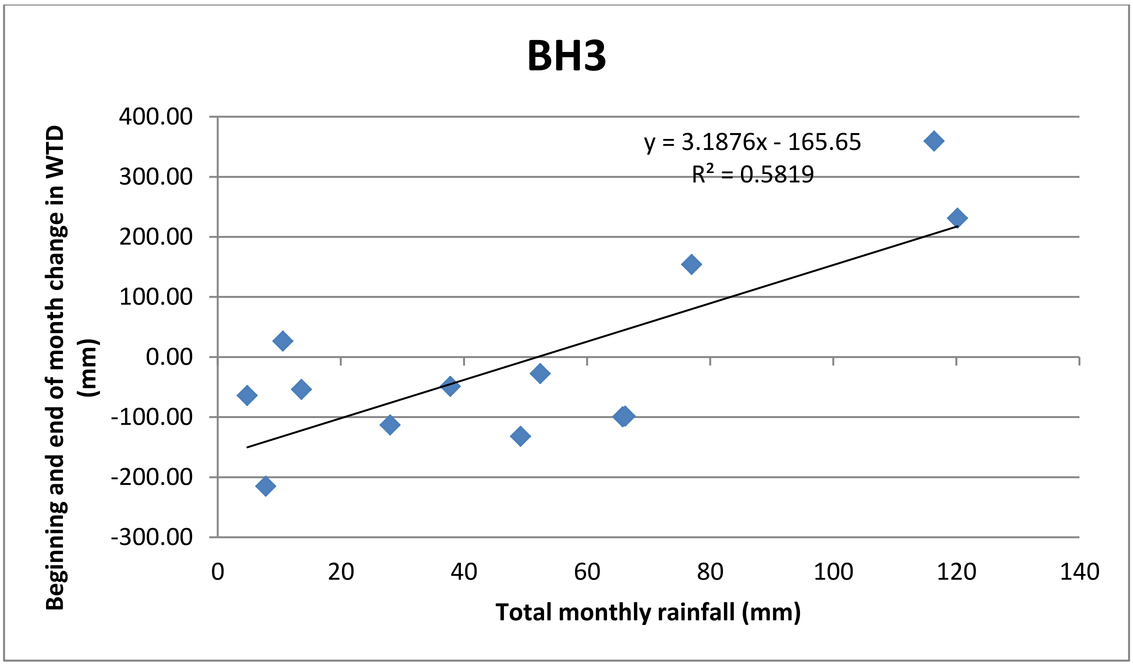

Figure 4. 7: BH3 Monthly beginning and end of month groundwater-level difference against total monthly rainfall

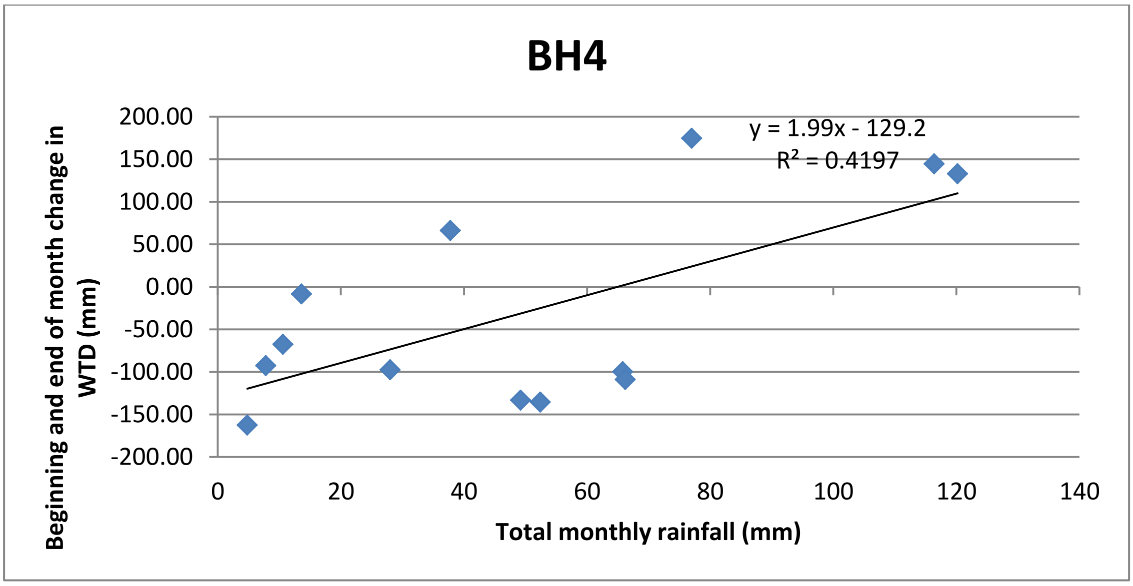

Figure 4. 8:BH4 Monthly beginning and end of month groundwater-level difference against total monthly rainfall

A positive correlation indicates the extent to which those variables (monthly beginning and end of month groundwater – level difference, rainfall) increase or decrease in parallel; a negative correlation indicates the extent to which one variable increases as the other decreases while a nil correlation shows lack of dependence. The strongest dependence is observed for BH3 (76%) followed by BH4 (65%), BH2 (63%) and the least is BH1 (33%). The positive correlation obtained for all four boreholes is an indication that rainfall accounts for more recharge than loss. Some of the factors responsible for the variations in the data are:

(i) Percolation characteristics - precipitation that falls on the earth surface does not spread to a single direction but multiple directions in different ways. Some of them will flow as runoff and some partially into the ground as water infiltration and percolation (Wessolek et al., 2008). Rainfall that exceeds the soil infiltration capacity results in much of the precipitation being flushed away as surface runoff instead of being available for groundwater recharge (Lee et al., 2019). Changes made to land use such as development of residential, industrial and urban facilities have the tendency to disrupt the hydrological cycle chain. The percolation characteristic is directly proportional to soil gradation (sand content). As the sand content increases it leads to increase of percolation rate. The percolation characteristic is inversely proportional to rainfall intensity. As the intensity of rainfall increases so does the percolation rate decrease. Both preferential flow and a decreasing water storage capacity increase the annual percolation rate in autumn (Wessolek et al., 2008). The increase in groundwater table depth during rainy season is due to groundwater recharge through precipitation percolation. However, vertical recharge may be impeded in the unsaturated layer and also human activities could influence the water level responses to rainfall events during dry season (Lee et al., 2019).

(ii) Surface characteristics - it impacts on the groundwater dynamics. A comparison of an area with uniform precipitation and infiltration rate over an undulating surface lends credence that a groundwater system will develop a water table that is a replica of the land surface. In areas with steep terrain, there is the tendency for water level to increase earlier in upland wells than in foot slope sites. Topography is not considered a dominant factor in spatial variation of groundwater levels in flat regions. The amount of water percolating into the water table is reduced as a result of evapotranspiration (Bista, 2019). Land surface affects the amount of water that infiltrates in to the ground. Porous surface materials such as sand and gravel allow easily water to infiltrate through the ground. It is estimated that about 40 to 50 % of rain and snow melt may seep in to the ground in porous surface materials while the seepage may range from 5 to 20 % in less porous surface material. The rest is lost to the atmosphere through evaporation (Abdullahi et al., 2015).

(iii) Groundwater delay - groundwater flow systems have differing capacities to retain and transport water, and the residence times of groundwater could vary from days to tens of thousands of years. Warmer and drier weather conditions elongate the residence time due to changes in hydraulic properties in the aquifer after long droughts (Chen et al., 2004). The lag time measures groundwater-level response to rainfall. Short lag times indicate fast response of groundwater to rainfall as a result of fissure flow. However, long lag times point to a delay groundwater response due to matrix flow (Cai and Ofterdinger, 2016). The obtained lag time means that the precipitation received today will not significantly affect the groundwater - level at that well location for about the lag time (Chen et al., 2004).

(iv) Groundwater table depth - the groundwater table depth in any period is influenced by the corresponding rainfall in that period and the rainfall and groundwater table depth in the previous period. The antecedent groundwater table depth effect is more pronounced than that of rainfall and antecedent rainfall. This phenomenon could be attributed to over-exploitation of the groundwater in the basin. The primary source of recharge for many aquifers is rainfall hence variations of rainfall and groundwater depth are closely related. Correlation of groundwater table depth and rainfall could be imperfect because differences in rainfall intensity and distribution produce different amounts of recharge for the same amount of rainfall (Ismail et al., 2010). Warming temperature speed up evaporation, thus reducing the recharge rate to the groundwater resource and causes a drop in the groundwater table. Warmer and drier weather conditions cause less groundwater recharge due to more evaporative loss of surface water and less precipitation in the winter and spring (Chen et al., 2004). Despite the consistency of the rainfall a considerable amount of groundwater table depth decrease may be due to Land Use Land Cover (LULC) or any other human activities (Lee et al., 2019).

(v) Lithology - the differing responses are due to differences in lithology and the percentage of highly permeable rock strata in the aquifer recharge areas. Aquifers that have recharge areas that are dominated by highly permeable lithological strata tend to show high correlations at medium and long times scales. On the other hand, those that have low permeability strata exhibit high correlations at short time scales. The aquifer recharge areas in which clays dominate do not show a strong correlation between groundwater level depth and precipitation which is in contrast with those aquifers where there are no clays present in the recharge areas. The effect of having clays in aquifer recharge areas is that the relationship between precipitation and aquifer storage is affected thus makes these aquifers less sensitive to climatic variability. The occurrence of strong surface and subsurface runoff processes decrease the amount of aquifer recharge and promote the transport of water out of the aquifer recharge area. The observed responses of groundwater levels to precipitation variability could be as a result of water management, varying climate and lithological features. Densely vegetated areas influence the response of groundwater level to precipitation variability by reducing the amount of water available for recharge. Dense urbanization could reduce the amount of water available for aquifer recharge by generating high runoff rates during intense precipitation episodes (Lorenzo-Lacruz et al., 2017).

(vi) Soil water movement - the movement of water in the soil is impacted heavily by water repellency. The wetting rates of dry soils are reduced which leads to induced preferential flow. In water-repellent soils, water bypasses regions of the unsaturated soil domain, but it only flows through some parts of the soil matrix. The soil moisture affects the soil profile in the sense that some parts of the soil profile are dry while the others show a successive remoistening after heavy rain. The wetting of the profile is clearly not a homogeneous infiltration process from the topsoil down to the subsoil. Preferential flow tends to be dominant in such a scenario. Water-repellent zones and wettable regions react differently to precipitation infiltration. A wettable soil leads to homogeneous flow and it occurs in spring. Water-repellent occurs in autumn and it leads to preferential flow. The soil moisture tends to increase rapidly during heavy rainfall while other parts are dry which results in much spatial variation of moisture at all depths (Wessolek et al., 2008).

4.1.4 Rise method determined water table response to rainfall

The method used here is also given in section 3.7 where positive water table changes attributed to rainfall by the rise method were separated from the negative water table changes as well. The reason was to determine the correlation between the water table increases and decreases with daily rainfall. Correlation is a statistical tool that allows us to determine the extent to which two or more variables fluctuate together. Figures 4.9 to 4.12 show the combined water table increases and decreases versus daily rainfall.

Based on correlation, the water table increases as a result of rainfall is greater than water table decreases also due to rainfall for BH4. It means rainfall impacts more on its recharge than loss. The water table increases due to rainfall for BH1 shows it has a negative linear relationship while the water table decreases has a positive relationship. The summary is that rainfall contributes minimally to recharge as against loss. BH2 water table increases and decreases exhibit similar pattern to BH1. Both BH3 water table increases and decreases show a negative correlation. The implication of it is that more loss than recharge occurs in BH3.

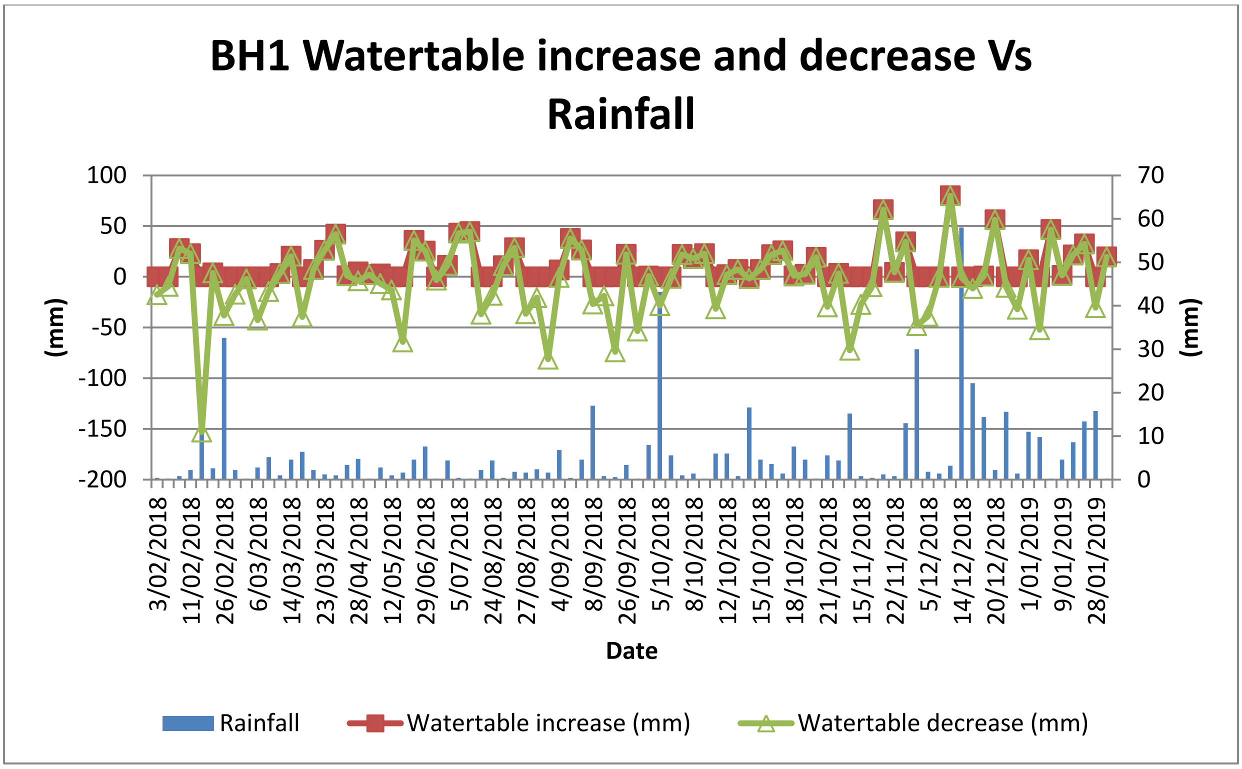

Figure 4. 9: Combined water table increases and decreases against rainfall for BH1

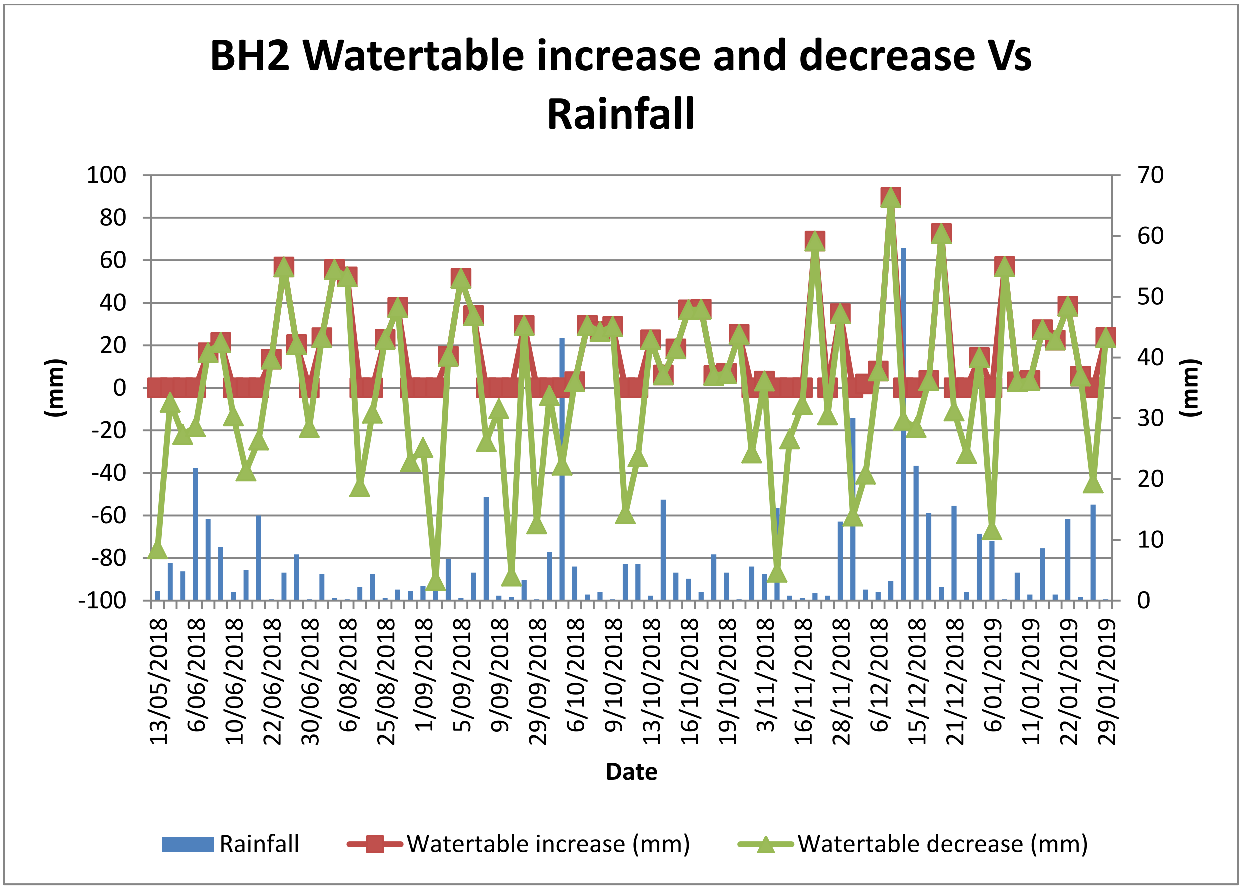

Figure 4. 10: Combined water table increases and decreases against rainfall for BH2

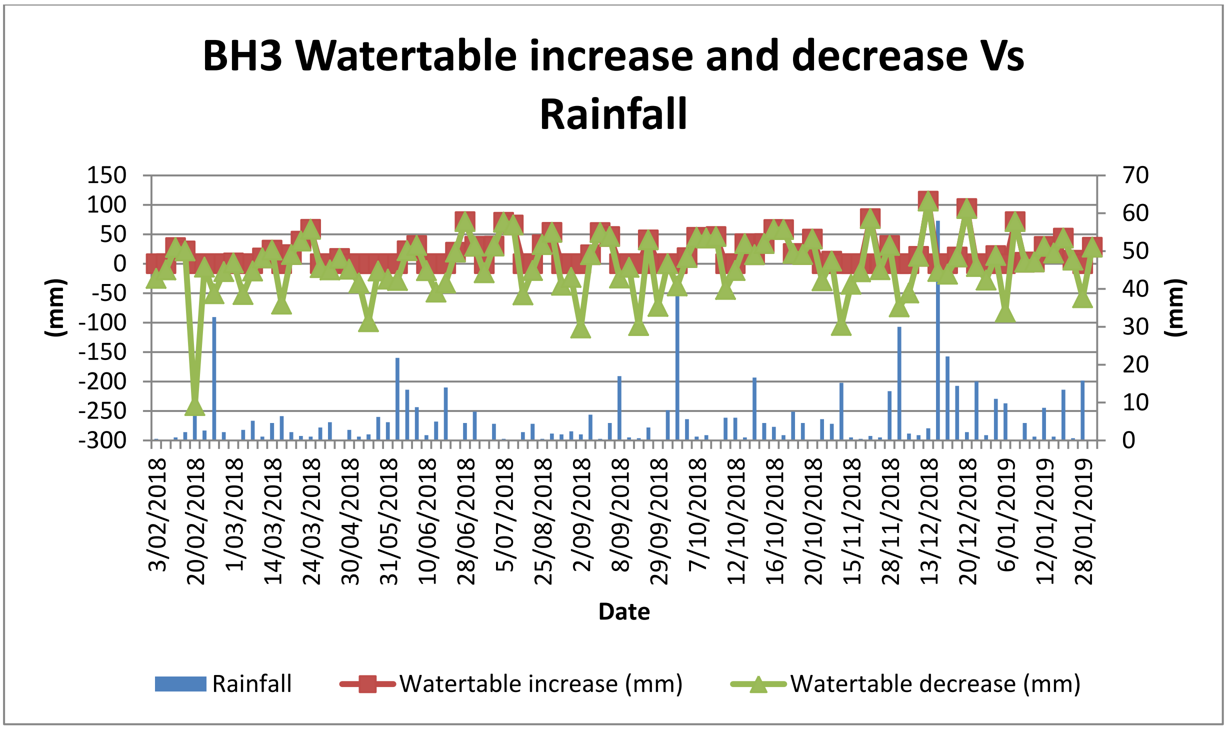

Figure 4. 11: Combined water table increases and decreases against rainfall for BH3

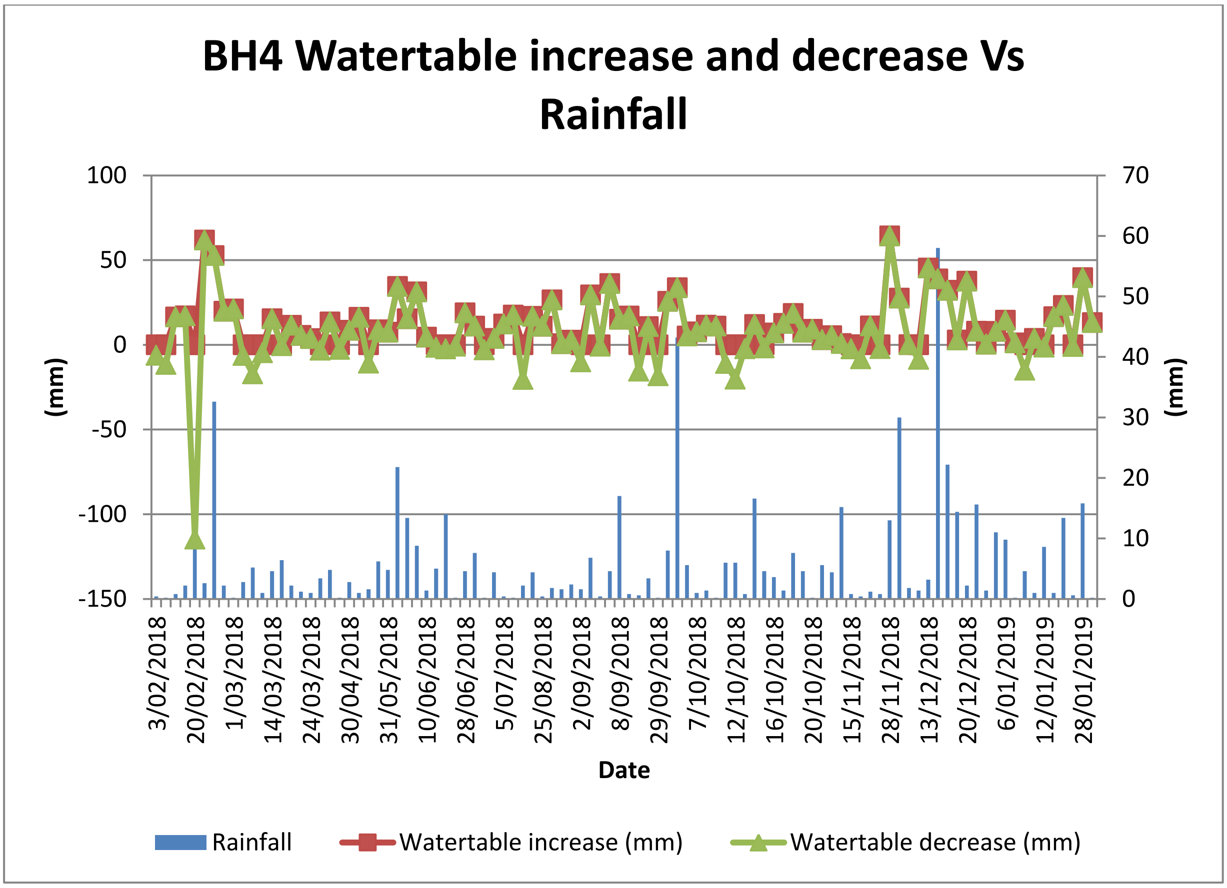

Figure 4. 12: Combined water table increases and decreases against rainfall for BH4

4.1.5 Cross-correlation of groundwater-level response to rainfall

This was performed according to the methodology section 3.6. The determination of groundwater-level response time (day) to rainfall was carried out with cross- correlation analysis by using the appropriate time-series data as the input and output signal. The lag time (day) is the absolute maximum correlation value obtained from the cross- correlation function (Cai and Ofterdinger, 2016). The cross-correlation of the data was performed with Minitab™ software. Our response time is in days because we used both daily groundwater –level data and rainfall. The wet season in Australia which occurs between October – April was used and it falls within spring and summer seasons. The variation of the spring and summer seasons response times were determined by the sliding windows cross-correlation technique which uses subsets of three- month data from the entire dataset. However, subset of six – month data was used to analyse the combined spring and summer seasons. The result of the analysis is presented in Table 4.1.

In spring season, both BH3 and BH4 response time was 3 days followed by BH1 with 11 days and BH2 was 14 days. However, BH4 lag time was 3 days while the others were 8 days each in summer season. The combined analysis of summer and spring season shows BH4 still gave a lag time of 3 days and BH1 and BH2 was 9 days each while BH3 was 13 days. It is evident from both spring and summer lag time obtained for BH4 that rainfall had a more direct impact on groundwater - level fluctuation because of its shorter lag time than the others. The quick response time of groundwater-level to rainfall is indicative of fast flow pathways taking place in BH4. In spring season alone, rainfall had the same impact on groundwater - level fluctuation of BH3 and BH4 while there was a delayed response for BH1 and BH2 groundwater – level fluctuation. The long response time recorded for BH1 and BH2 are caused by slow flow matrix storage and diurnal tidal force. Matrix pore flow is very slow as a result of its low hydraulic conductivity and high porosity (Lee et al., 2006). BH4 recorded the quickest time in the combined summer and spring season analysis as a result of fissure flow. However, the delayed response of the other three boreholes is due to the effect of slow matrix storage as it takes time for rainfall to impact on groundwater – level fluctuation (Cai and Ofterdinger, 2016, Lee et al., 2006). In addition, the low permeability of the matrix means more time is needed for the aquifer response to be observed. The rock matrix takes longer time to respond after or during rainfall events which in turn causes a delay or lag in the response of groundwater - level from rainfall events (Cook, 2003).

Table 4. 1: Groundwater level response time (days)

Lag time (days)				
Seasons	**BH1**	**BH2**	**BH3**	**BH4**
Spring	11	14	3	3
Summer	8	8	8	3
Combined spring and summer	9	9	13	3

4.2 Soil analysis results

A summary of the results from the mechanical analysis and specific yield are presented in this sub-section. The results of two replicate analyses are shown in appendix section (Table A.0.1 – A.0.47).

4.2.1 Soil textural classification

The methodology used is given in section 3.3.1. The results obtained from the sieve analysis and sedimentation tests were used to classify the soil samples according to Unified Soil Classification System (USCS) and United States Department of Agriculture (USDA) (García-Gaines and Frankenstein, 2015) textural soil classification. The texture classification of soil is used in water resources engineering to determine the quantity of water that can infiltrate a given soil. The spatial variability in soil permeability is linked to variations in soil texture. Both topography and soil texture are related. Soil textures tend to control the rates of recharge (Scanlon et al., 2010). Figure 4.13 shows the combined soil profile at BH1, BH2, BH3 and BH4 bores.

Figure 4. 13: Combined soil profile at BH1, BH2, BH3 and BH4 bores

The texture of the soil across the four monitoring sites at a depth of 0 - 1 m is generally made up of sand. Both BH1 and BH4 at a depth of 1 - 2 m is loamy sand while BH2 and BH3 at a depth of 1 - 3 m is loamy sand. Sandy loamy is found for BH1 at a depth of 2 - 4 m while it is found for BH4 at the depth of 2 – 5 m. On the other hand, BH2 soil at a depth of 3 – 5 m is composed of sandy loamy while BH3 at a depth of 3 – 6 m is equally made up of sandy loamy. At a soil depth of 5 + m, all three of BH1, BH2 and BH4 are composed of loamy sand while BH3 at a soil depth of 6 + m is made up of loamy sand as well.

4.2.2 Specific yield estimation

The methodology used in estimating the different boreholes specific yield values is given in section 3.3.2, 3.3.3, and 3.3.4. The estimation of specific yield is a valuable parameter in the adequate management of groundwater resources (Jinxi and Xunhong, 2010, Johnson, 1967). The values of specific yield obtained from the four monitoring sites at different depths showed it varied with depth (Table 4.2). At monitoring site BH1, the S_y ranges from 0.37 to 0.46. The S_y at the bottom depth had the highest value. The S_y values for BH2 ranges from 0.23 to 0.37, BH3 ranges from 0.33 to 0.43 while BH4 ranges from 0.23 to 0.37. At monitoring sites BH2 and BH4, the surface S_y value had the highest value compared with others within the sample total depth. On the other hand, monitoring site BH3 had two S_y

values within the sample total depth that were higher than surface S_y value. Average S_y values for the monitoring sites at BH1, BH2, BH3 and BH4 were 0.41, 0.31, 0.38 and 0.31.

S_y depends on time and depth to water table. The dependence of S_y on time is linked to the slow drainage of soil water from pores above the water table. Shallow water table aquifers are associated with low specific yields. The S_y is zero when the depth to water table is less than the pore size distribution index and it implies that no water will be released unless the water table depth is larger than the soil air – entry pressure (Nachabe, 2002). The variability for the S_y with interval depth among the four monitoring sites is quite different. This could be attributed to variation of grain size, grain shape, sorting and compaction of sediments (Jinxi and Xunhong, 2010).

According to Heliotis and DeWitt (1987), the inverse of S_y shows the water table response which is the rise of water table for a certain water input. When there is a higher S_y it is an indication of a larger pore size which leads to a reduced thickness of the capillary fringe. Some of the factors that cause whether a rapid or capillary response will occur during a water table response to rainfall are soil pore size, water table depth at the onset of rainfall and the magnitude of rainfall (Heliotis and DeWitt, 1987). The values of S_y are not definitive due to the fact that the quantity of water drained by gravity is dependent on duration of drainage, temperature, mineral composition of the water and other physical characteristics of the rock or soil being considered (Johnson, 1967, Crosbie et al., 2005). S_y varies from one to another location in both the horizontal and vertical direction in an aquifer. It is an over-simplification to use a single S_y to represent the capacity of the extractable water in an unconfined aquifer (Jinxi and Xunhong, 2010). The method used in the determination of S_y could have an impact on the result as different methods give different results. Laboratory determined values are quite different from field determined values. Crosbie et al. (2005) noted that the values of S_y obtained from laboratory are about 2 – 3 times higher than values obtained with the pump test method (Crosbie et al., 2005).

Table 4. 2: Specific yield values of boreholes at different depths

Soil depth (m)	Specific Yield (S_y)			
	BH1	BH2	BH3	BH4
0	0.40	0.37	0.41	0.37
0.5	0.38	0.37	0.40	0.36
1	0.37	0.35	0.43	0.25
2	0.37	0.29	0.37	0.31
3	0.41	0.33	0.33	0.27
4	0.42	0.35	0.36	0.23
5	0.42	0.36	0.35	0.33
6	0.44	0.27	0.37	0.25
7	0.43	0.27	0.39	0.34
8	0.42	0.23	0.41	0.33
8.5	0.46	0.26	0.35	0.34
9		0.28	0.34	0.34
10				0.33
Mean	0.41	0.31	0.38	0.31

4.2.3 Literature values of specific yield

This section discusses a compilation of some of the S_y values collected from literature in relation to S_y obtained in this study. Nwankwor et al., (1984) values of S_y obtained with the laboratory volume balance method were in the range of 0.02 – 0.25. The values were increasing as the pumping rate was increased. On the other hand, the type-curve methods provided values of 0.07 and 0.08. The early values of volume balance method at early times were similar to values obtained with the type-curve methods. This calls into question the rationale of using type-curve methods values for analysis of the long-term yield characteristics of the aquifer (Nwankwor et al., 1984). Varni et al., (2013) used a graphical procedure of fitting water table variations in relations to rainfall events and the method was applied to determine the S_y of River Azul aquifer in Argentina. The maximum value of S_y

was obtained from the inverse of the slope of a line drawn through the origin to just above all of the measured points. The maximum S_y value obtained was 0.09. The advantages of using such a method are the availability of extensive data records of water table rise events for the analysed aquifer and the fast response of the water table during recharge events. Another method of considering the water level response towards selected rainfall events for the same aquifer yielded a S_y of 0.07 (Varni et al., 2013). Chinnasamy et al., (2018) estimated S_y based on crop irrigation water use and changes in the pre and post irrigation season water table depths in a hardrock aquifer system of Rajasthan, India. The average S_y was found to be 0.02. However, another study conducted within the same geological environment but by using double water-table fluctuation method found S_y to be between 0.002 - 0.038. The variation in S_y could be as a result of variations in geology and depth to water table. As the water table depth decreases, it leads to closure of fractures and groundwater dominant flowpaths which inhibits some layers from being active in groundwater flow (Chinnasamy et al., 2018). Kotchoni et al., (2019) used Magnetic Resonance Sounding (MRS) technique and applied it to estimate the S_y at three monitoring sites under sedimentary and crystalline aquifers in Benin, West Africa. The variation in depth of the MRS water content θ_{MRS} is one of the primary output parameters generated from the interpretation of MRS measurements. Finally, the Sy was obtained by the relationship between θ_{MRS} and S_y based on studies carried out in several sites of the same geological environment. The values of Sy ranged between 0.004 – 0.162. The main advantage of this technique over other geo-physical methods such as electrical resistivity tomography is the direct measurement of signals that are generated by subsurface water itself. The low value of S_y obtained in one of the aquifers is as a result of its very low specific capacity (Kotchoni et al., 2019). Three methods namely pump test, rainfall - water table response and Tempe cell were used to estimate S_y and the results varied. Pump test gave very low S_y values while the S_y from both Tempe cell and rainfall –water table response were reasonable. The values of S_y obtained with the Tempe cell were in the range of 0.230 to 0.363, water table response to rainfall was 0.120 to 0.230 while pump test was 0.065 – 0.136. The S_y obtained from the laboratory method is 2 to 3 times higher than the one obtained with the pump test (Crosbie et al., 2005).

Table 4. 3: Specific yield of various porous media (Beretta and Stevenazzi, 2018).

Lithotype	No. of determinations	Specific yield		
		Maximum	Minimum	Average
Clay	15	0.05	0	0.02
Silt	16	0.19	0.03	0.08
Sandy clay	12	0.12	0.03	0.07
Fine sand	17	0.28	0.10	0.21
Medium sand	17	0.32	0.15	0.26
Coarse sand	17	0.35	0.20	0.27
Gravelly sand	15	0.35	0.20	0.25
Fine gravel	17	0.35	0.21	0.25
Medium gravel	14	0.26	0.13	0.23
Coarse gravel	14	0.26	0.12	0.22

The Table 4.3 shows a compilation of S_y values obtained with different methods at various locations of the globe but mainly in the United States. The maximum value of S_y range between 0.30 – 0.35 and it occurs in medium sand, coarse sand, gravelly sand and fine gravel (Beretta and Stevenazzi, 2018). Beretta and Stevenazzi (2018) used the below algorithm to estimate Sy.

$$S_y = -0.0014(In\ k)^2 - 0.0003In\ k\ + 0.2973 \qquad 4.1$$

Where S_y is the specific yield and K is the hydraulic conductivity (m/s) which is derived from Kr (radial hydraulic conductivity). The algorithm is a statistical correlation between hydraulic conductivity and S_y. Field data obtained from pumping test was used to estimate S_y of a shallow unconfined aquifer in the area of Milan and its surroundings. The range obtained for S_y was in the order of 0.14 – 0.30 (Beretta and Stevenazzi, 2018).

Table 4. 4*: Thickness of sediment cores and respective specific yield values at the test sites (Jinxi and Xunhong, 2010).*

Test sites	Thickness of sediment cores, m		Total depth, m	Specific yield			Test number of sediment cores
	Range	Average		Range	Average	Standard deviation	
A	0.69 –1.36	1.05	19.50	0.02- 0.12	0.09	0.028	11
B	0.42 –0.86	0.65	21.00	0.01- 0.13	0.06	0.043	13
C1	0.26 –1.01	0.59	21.00	0.01- 0.15	0.07	0.046	14
C2	0.33 –1.07	0.69	9.00	0.02- 0.18	0.07	0.057	6
D	0.49 - 0.98	0.72	21.00	0.06 - 0.13	0.12	0.022	13

Jinxi and Xunhong (2010) used the drainage method to estimate the S_y of an alluvial aquifer of the Platte River valley, Nebraska, USA and the results are presented in Table 4.4. The method is a two-step process. The first involves the re-saturation of the sediments in water. This is continued for more than 24 hours to ensure that the sediments are fully saturated. The second is after the water is fully drained by force of gravity and the estimation of S_y is by using the following formula:

$$S_y = \frac{V_{wb} - V_{wt}}{V_s} \qquad 4.2$$

$$V_{wt} = \pi r^2 L_{wt} \qquad 4.3$$

$$V_s = \pi r^2 L_s \qquad 4.4$$

Where S_y is the value of specific yield for the sediment core; V_{wb} is the water volume in the bucket; V_{wt} is the water volume above the sediments in the tube just before the start of drainage; V_s is sediment volume; π is circumference ratio; r is the tube radius; L_w is depth of water above the sediments in the tube before the drainage start; and L_s is the length of sediments in the tube. The average values of S_y for the sediment cores was between 0.06 – 0.12. The magnitude of vertical hydraulic conductivity (K_v) affects the pace of drainage in the draining experiment. This test is more cost effective than pumping test because pumping test requires the installation of observation and pumping wells. However, a drawback of some drainage experiments is the inability to keep intact the core original sedimentary structures and particle framework during sampling and repacking (Jinxi and Xunhong, 2010).

The proposed methodology in this study is cost effective when compared with other methods such as pumping test. Furthermore, it provides a quick evaluation of the specific yield when constrained by technical and economic resources. The obtained S_y values were within literature values but it is difficult to make direct comparisons between the S_y values obtained in this study with those reported elsewhere in literature due to the differences in depths in which soil samples were collected, methods of soil collection, methods of S_y determination, variation in aquifer properties and geographical differences. Different hydro-geological conditions will lead to complete different results. Moreover, there is paucity of relevant data in the study area such as pump tests to compare this result with. Too, the implicit heterogeneity of aquifers impact on a reasonable estimate of aquifer S_y. The average value was used to account for S_y depth variation with respect to geology in addition to having a single logger deployed in each borehole to record groundwater table fluctuation.

CHAPTER 5

GROUNDWATER QUALITY

This section covers the results obtained from the monitoring of groundwater quality across the four monitoring sites over the course of the study. Detailed results from the physico-chemical analyses of the collected groundwater samples from the sites are presented in Tables in appendix section. However, graphical results across the different months of samples collection are presented here in order to understand the temporal variation of groundwater with time.

5.1 Catchment dataset

A search of groundwater quality results within the same catchment was conducted to enable us understand how our own results compare with them. The found catchment results are presented in Table 5.1 while Table 5.2 shows the range of observed values for BH1, BH2, BH3, and BH4.

Table 5. 1: Groundwater quality results from the same catchment

Parameters	**Denham Court (EMM, 2016)**	**Glenlee Road (EMM, 2016)**	**Menangle Park (EMM, 2016)**	**Heathcote (ES, 2015)**
pH	6.46 – 9.85	7.31 – 8.35	5.15 – 10.06	5.31 -5.69
DO (mg/L)				0.50 -2.45
TDS (mg/L)	4,580 – 8,510	3091 - 3182	525 -693	
SS (mg/L)	5 - 1,350	< 5	5 -41	
Ca (mg/L)	7-337	36 - 111	3- 91	
Mg (mg/L)	3 - 91	64 - 77	2 -30	
Na (mg/L)	1,400 – 2,470	991 - 999	100 -187	
K (mg/L)	12 - 29	34 - 38	1 -13	
Fe (mg/L)	0.05 – 7.59	0.14 – 0.76	0.05 -3.7	
EC (µS/cm)	7058- 13,090	4755 - 4887	782 – 1,066	286- 580

Table 5. 2: Range of concentrations of physico-chemical parameters in groundwater

Parameters	BH1	BH2	BH3	BH4
EC, µS/cm	8,000 – 17,970	310 – 1010	780 – 2850	300 – 3720
DO, mg/L	1.5 – 5.5	2.3 – 4.6	1.9 – 5.8	1.6 – 6.9
Turbidity, NTU	95 – 1,000	26 – 395	121 – 626	8 – 119
pH	5.8 – 7.3	5.1 – 7.1	5.3 – 7.2	4.3 – 6.6
Total solids (TS), mg/L	12,360 – 18,606	305 – 6,334	579 – 7,335	257 – 7,816
Total dissolved solids (TDS), mg/L	12,082 – 15,974	277 – 613	419 – 1,550	2,042 – 2,590
Total suspended solids (TSS), mg/L	108 – 567	26 – 533	78 – 571	3 – 84
Sodium (Na), mg/L	3034 – 4047	66 – 202	95 – 305	15 – 138
Calcium (Ca), mg/L	130 – 233	4 – 8	2 – 7	7 – 17
Magnesium (Mg), mg/L	565 – 841	11 – 40	13 – 20	95 – 107
Potassium (K), mg/L	22 – 30	2 – 3	2 – 5	2 – 3
Iron (Fe), mg/L	1.3 – 5.7	1.1 – 4.7	0.2 – 4.8	0.5 – 1.7

5.1.1 Electrical conductivity (EC)

Electrical conductivity (EC) is defined as the ability of a substance to conduct an electric current and it is widely used to assess water quality (Marandi et al., 2013). EC measurements are also used to determine the salinity, ionic strength, major solute concentrations and total dissolved solids concentrations of natural waters (McCleskey et al., 2012). The use of electrical conductivity in groundwater monitoring is to show the presence of ions of chemical substances that may reflect the natural variations in water quality or the presence of contamination. The change in conductivity has a strong dependence on types of dissolved ions (Marandi et al., 2013). Increased EC have an adverse effect on aquatic organisms and irrigated crops by impairing cell function (DoTRD, 1997).

Figure 5.1 shows the electrical conductivity of the analysed samples. As shown in the figure, the groundwater in BH1 has the highest electrical conductivity while BH2 has the least. According to (Rhoades et al., 1992), BH2 can be classified as slightly saline, BH3 and BH4 are moderately saline while BH1 is highly saline. Higher EC in the groundwater of BH1, indicate that there could be some localised sources of contamination that has resulted in higher EC levels such as fluid migration into the aquifer from nearby formations as the groundwater flow in the study area is from BH1 to BH4. Also , increase in dissolved solids and the presence of metallic ions may be responsible for high EC (Chukwu, 2008). Enrichment of salt as a result of evaporation effect and leaching also cause high EC in groundwater (Kumar et al., 2007). Comparing the EC values from the four boreholes against values in Table 5.1, it shows that only BH2, BH3 and BH4 maximum and minimum values are within the range. The maximum value of EC in BH1 is 17970 µs/cm and it exceeds it. Only the minimum EC value of BH1 is in the range of the catchment threshold.

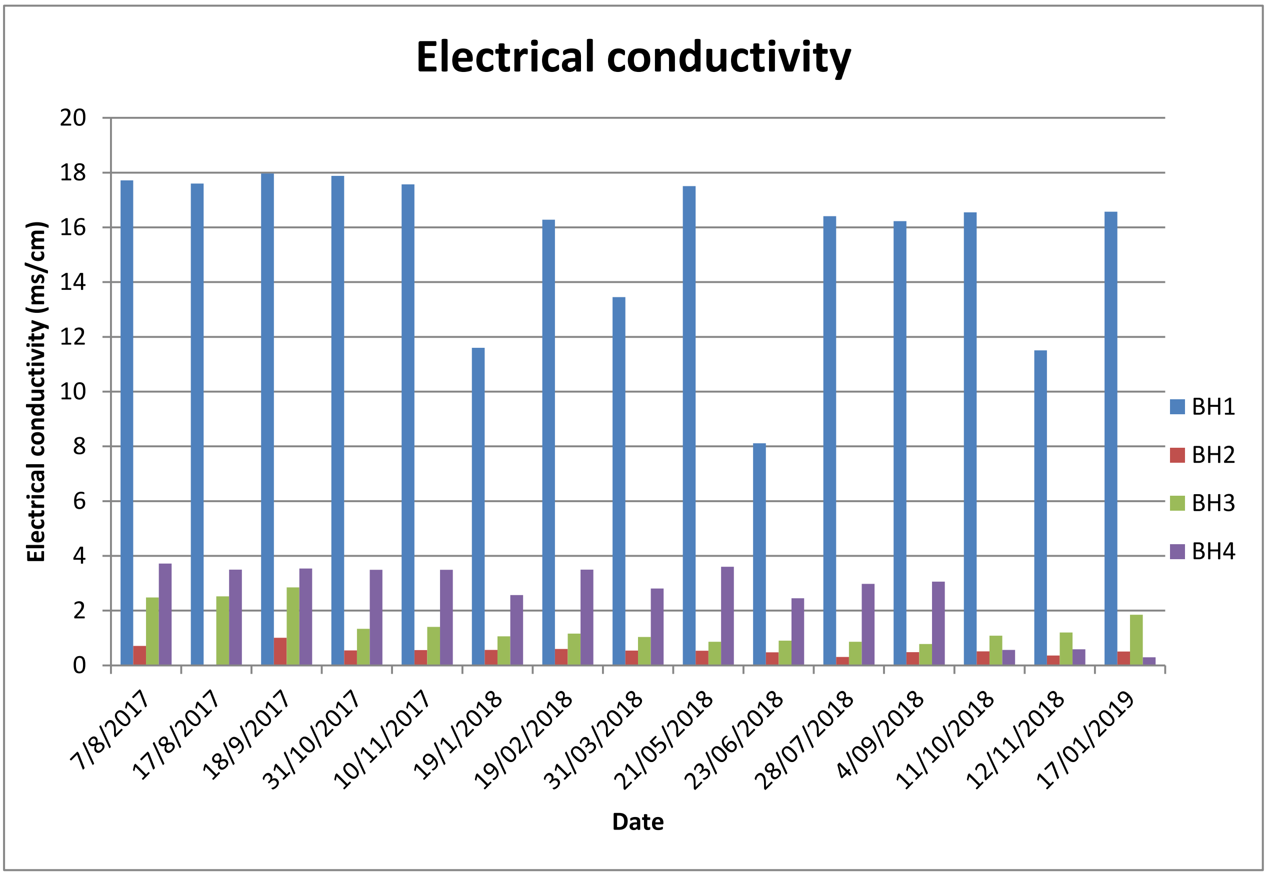

Figure 5. 1: Variation in the electrical conductivity (EC) in groundwater

5.1.2 Dissolved oxygen (DO)

It is the amount of oxygen contained in water. Low DO concentration has a negative effect on many aquatic organisms which rely on oxygen dissolved in water for efficient functioning. DO concentrations depend on temperature, salinity, biological activity and rate of transfer from the atmosphere. Oxygen has limited solubility in water which ranges from 6 to 14 mg/L (Palani et al., 2009). Groundwater has a lower DO than surface waters because of absence of contact with the atmosphere (WHO, 2011). Figure 5.2 shows the DO measurement taken onsite for all four boreholes. As expected, the DO levels are generally below 4 mg/L. However, in recent analysis, BH1 has exceeded the 4 mg/l threshold from 21/05/2018 till date. This is mainly due to the fact that there are very little opportunities for the groundwater to come in contact with atmospheric oxygen (Addy and Green, 1997). WHO (2011) has no recommended health based guidance values for DO. High levels of DO quicken corrosion of metal pipes (WHO, 2011). The minimum DO values of all four boreholes are within Table 5.1 values and the maximum values of all four exceed it.

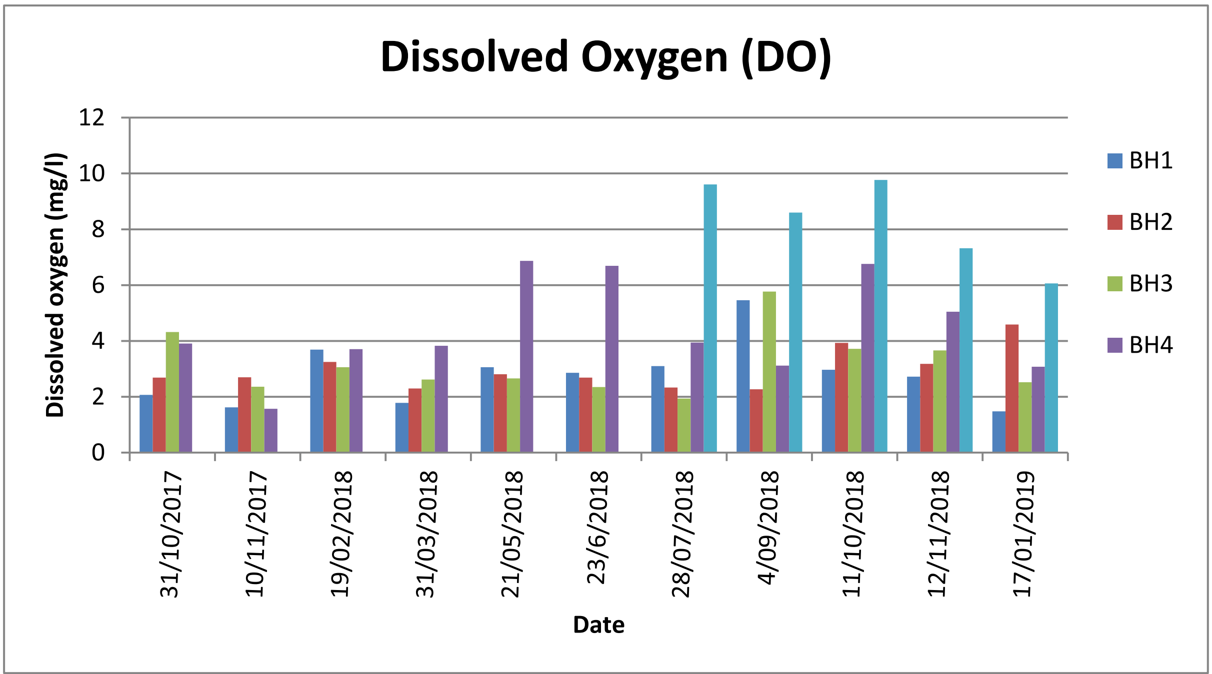

Figure 5. 2: Variation of dissolved oxygen (DO) in groundwater.

5.1.3 Turbidity

Turbidity readings give an indication of how light is scattered by suspended particulate materials in the water, which is then used to indicate the amount of turbid material present in water. Inorganic or organic matter or a combination of the two can cause turbidity. The presence of turbidity in some groundwater is as a result of inert clay and other oxides when water is pumped from waters without oxygen. High levels of turbidity are an indication of possible presence of contaminants. If the groundwater is to be used directly for drinking, it shows all four are not suitable as they exceeded the 5 NTU limit (WHO, 2011, NHMRC, 2011) and also 5 NTU limit literature values from different countries (Chukwu, 2008, Fisher et al., 2004, Hassen et al., 2016, Jain et al., 2010, Srivastava, 2019, Abbasnia et al., 2018, Longe and Balogun, 2010, Arumugam and Elangovan, 2009). Highly turbid waters impair aesthetic and recreational value of water (DoTRD, 1997). High turbidity of water is also linked with high levels of disease causing organisms such as viruses, parasites and bacteria. These organisms are responsible for symptoms such as nausea, cramps and diarrhoea (Kaur et al., 2017). On average of the sampling results, BH1 has the highest turbidity level. This means that the groundwater contains the colloidal particles.

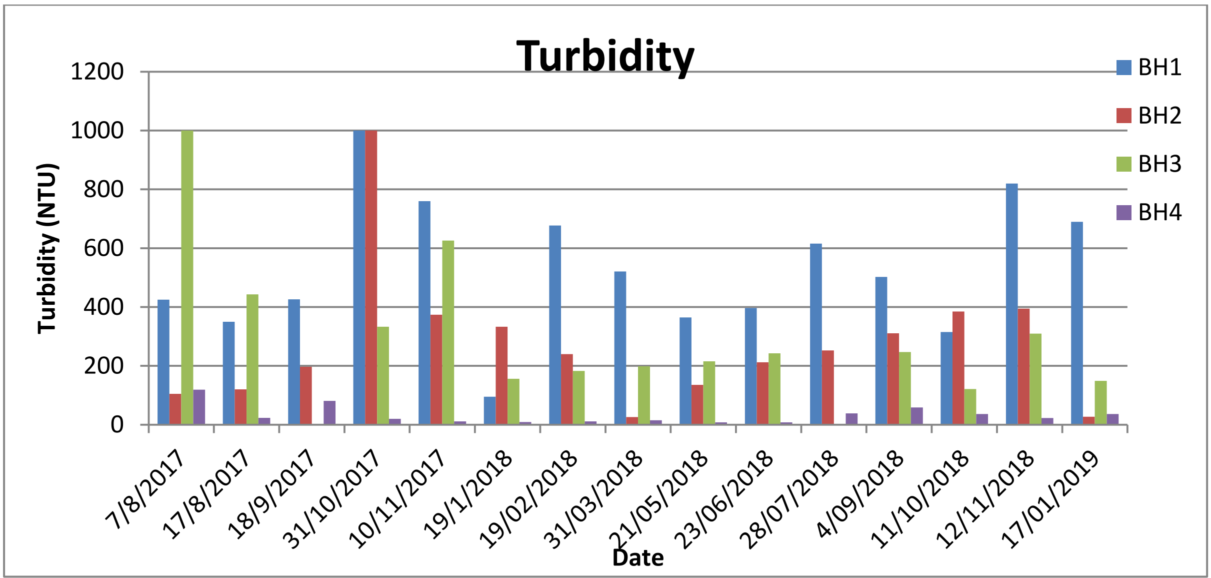

Figure 5. 3: *Variation of turbidity in groundwater.*

5.1.4 pH

Taste and odour of a substance is influenced greatly by pH, when it controls the equilibrium concentration of the neutral and ionized forms of a substance in solution (Otieno et al., 2012). Figure 5.4 shows the pH values in the groundwater from all the four boreholes. As shown in the figure, the pH appears to be varying between 5 and 7. This means that the groundwater is slightly acidic. Also, the pH levels do not exceed the guideline range of 6.5- 8.5 found in literature (Hassen et al., 2016, Srivastava, 2019, Abbasnia et al., 2018, Longe and Balogun, 2010, Jain et al., 2010, Arumugam and Elangovan, 2009, NHMRC, 2011, NWQMS, 2013, WHO, 2011) which is given in Table 2.2. Water with pH values of less than 6.5 are considered too acidic and not suitable for human consumption as it can cause health problems such as acidosis and also is generally corrosive (Ackah et al., 2011). On average, BH1 has the highest pH value while BH4 has the least. Low pH values are as a result of acidification, declining water table due to low annual rainfall, and dewatering to allow for peat excavation

(Appleyard et al., 2004). Only BH4 pH minimum value is outside the values given in Table 5.1.

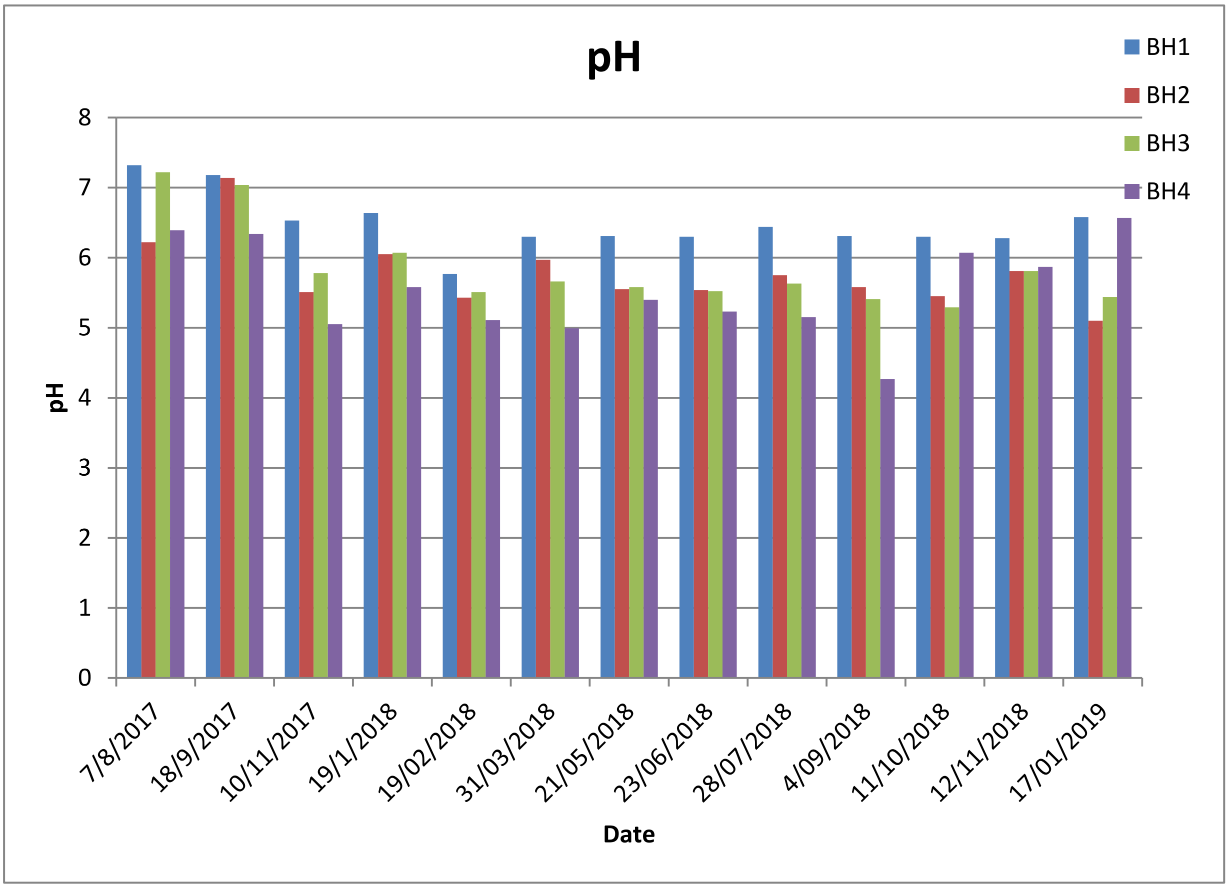

Figure 5. 4: *Variation of pH in groundwater*.

5.1.5 Total solids

Total solids are a sum of all the suspended, colloidal and dissolved solids in water. Total solids also include dissolved salts such as sodium chloride (NaCl) and solid particles such as silt and plankton. There are a lot of factors that contribute to the total solids in water of which soil erosion is a large contributor. Dissolved solids often make a huge contribution to the amount of total solids in water. The presence of excessive solids in water may be as a result of agricultural activities and geological parameters. The presence of excessive solids in water indicates the occurrence of pollution which can have a laxative effect (Karunakaran et al., 2009). Total solids impact on water clarity. Figure 5.5 shows the variation of total solids in all the groundwater. As can be seen in the figure, the groundwater from BH1 has the highest total solids. This may be due to the high dissolved salts. This observation is consistent with the high EC observed in Figure 5.1.

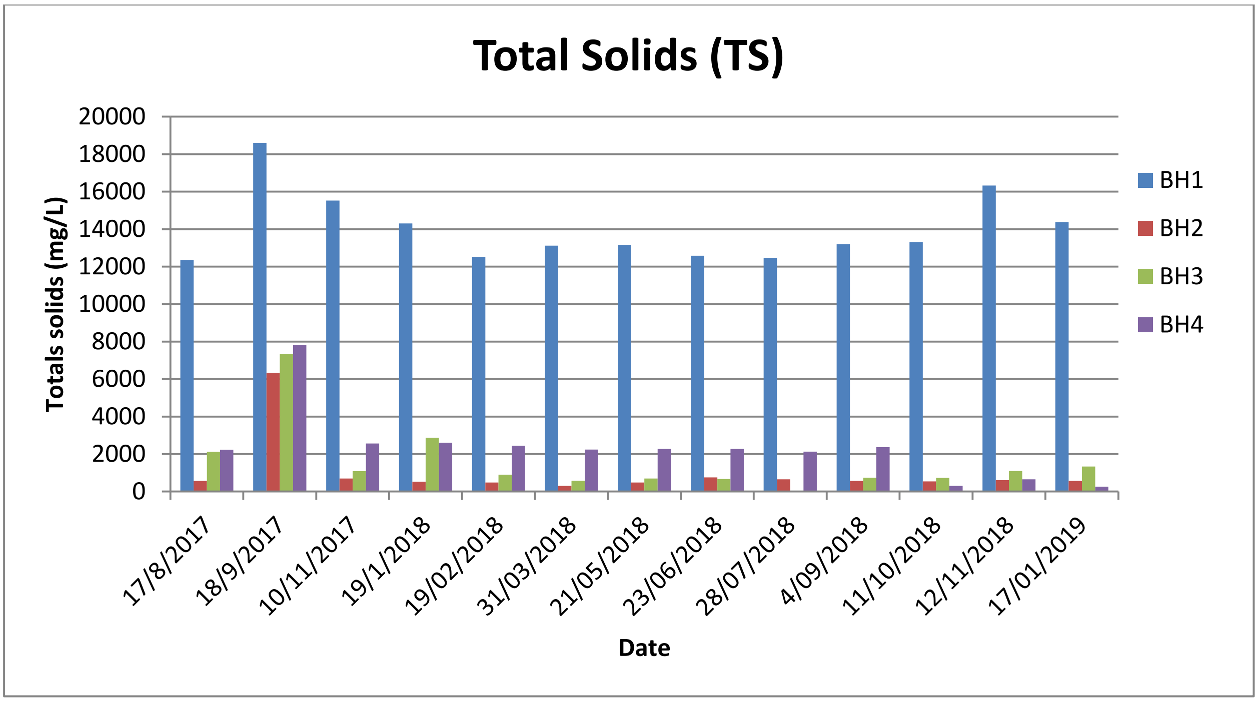

Figure 5. 5: *Variation of total solids in groundwater.*

5.1.6 Total dissolved solids (TDS)

Total dissolved solids have been obtained by subtracting suspended solids from total solids. The total dissolved solids are the total mass of dissolved constituents in water, which typically represent dissolved salt (comprising cations and anions) for groundwater samples. According to EPA's secondary drinking water guidelines (Harter, 2003), water with a TDS above 500 mg/L is not recommended for use as drinking water. Water with a TDS above 1,500 to 2,600 mg/L is not good for irrigation use on crops with low or medium salt tolerance (Harter, 2003). WHO recommends that water with a TDS less than 600 mg/L is good for drinking while above 1000 mg/L is not good for drinking. Corrosion is associated with high TDS levels (WHO, 2011). There is a relationship between TDS and EC of water (Thirumalini and Joseph, 2009, Walton, 1989, McCleskey et al., 2012).

Figure 5.6 shows the variation of TDS for all the boreholes. Groundwater from BH2 has the least TDS and, on the other hand, the groundwater from BH1 has the highest during all the sampling periods. The groundwater from BH1 and BH4 appear to be unfit even for irrigation purpose. There may be some contamination that is resulting in high TS and TDS values for the groundwater from BH 1. The minimum and maximum values of TDS obtained from BH2, BH3 and BH4 do not exceed values from similar catchment as given in Table 5.1. On the other hand, both the minimum and maximum values obtained from BH1 are outside the values given in Table 5.1. Only BH2 is good for drinking according to WHO guidelines and it has a low salinity. Groundwater flows from BH1 to BH4 according to the hydraulic gradient. This means that BH1 is the first receptor of sediments flowing through the groundwater and that is why it has the highest TDS and also EC which is related to TDS.

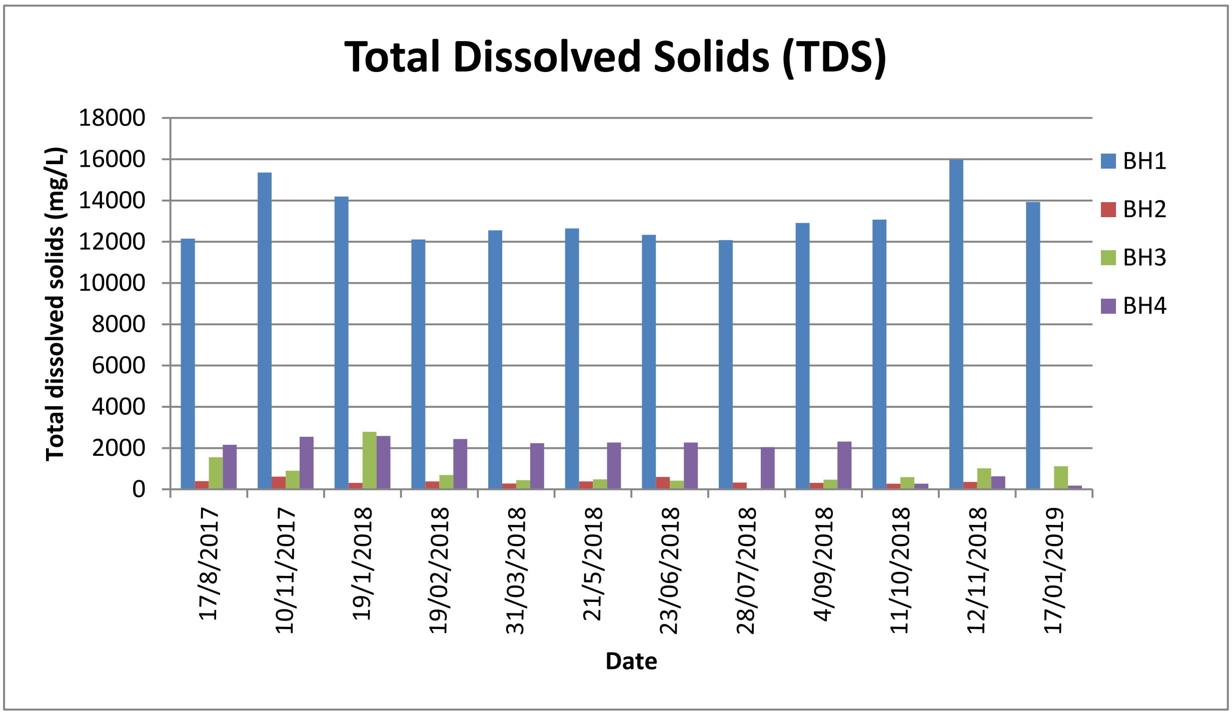

Figure 5. 6: *Variation of total dissolved solids (TDS) in groundwater.*

5.1.7 Total suspended solids (TSS)

Suspended solids may consist of inorganic fraction such as, silts and clays, and organic fraction such as, algae and plant materials, which are carried along by water as it runs off the land. These suspended particles become sediments when they settle to the bottom of a water body. The inorganic portion is exceedingly higher than the organic component. High TSS can reduce water clarity, degrade habitats, and reduce photosynthetic activity and cause an increase in water temperatures (Harter, 2003, Johnson et al., 2015). Also, a major impact of TSS on water is its ability to severely degrade its aesthetic value.

Figure 5.7 shows the variation of TSS in all the boreholes. As can be seen in the figure, generally the suspended solids in BH1 water appear to be significantly high and the suspended solids in BH4 appear to be low. There appears to be a large difference between the levels of TSS of these two waters. In addition, the TSS for BH2 and BH3, appear to be high as compared to the typical groundwater samples.

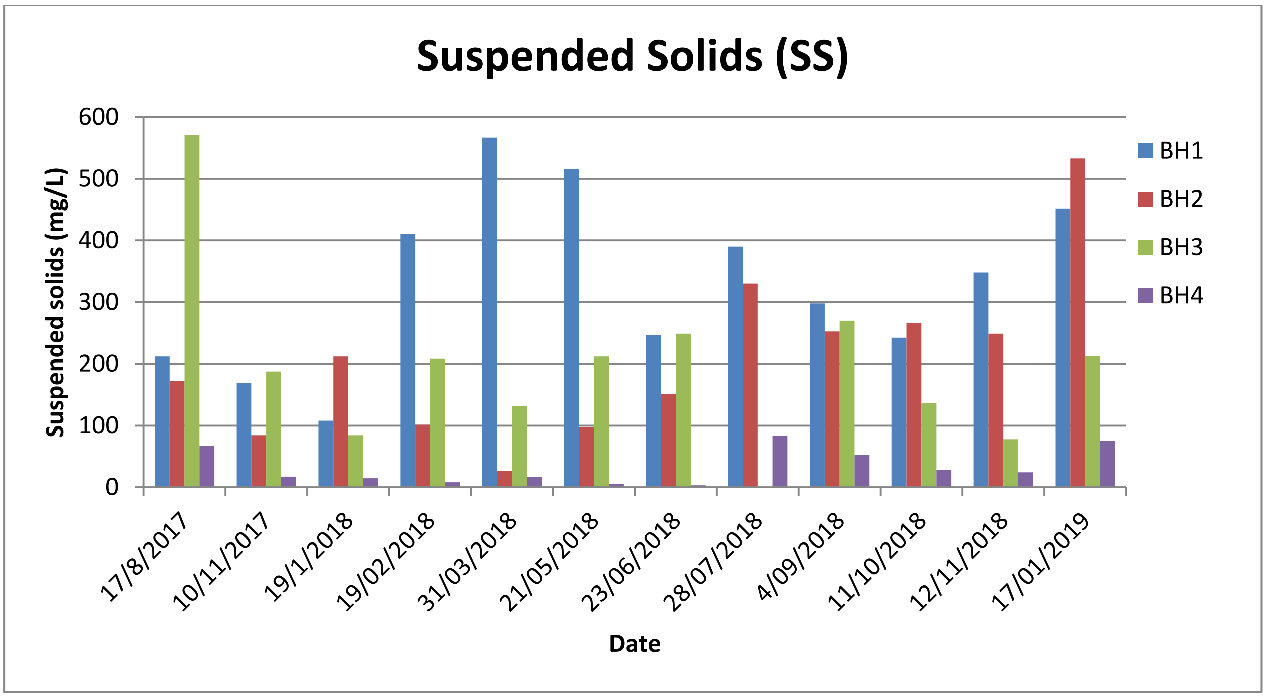

Figure 5. 7: *Variation of suspended solids (SS) in groundwater.*

The following could be some of the reasons for high suspended solids in the groundwater:

- Turbulent water flow within the aquifer, which can transport fine material such as clays and particulate organic material.
- The site may have been used for waste (domestic/ commercial/ industrial) dumping (Krishnan et al., 2007).
- Water from shallow wells (less than 60 m) generally has higher total suspended solids than from deeper wells.
- Potentially due to suspended clays and, organic and inorganic contaminants.
- Presence of unconsolidated to poorly consolidated sands, silts, and clays in the area (Fisher et al., 2004)

5.1.8 Cations

The analysis of the four major cations (Mg^{2+}, Ca^{2+}, K^{+}, and Na^{+}) was carried out as detailed in methodology section 3.4. The results presented here in graphical form are those that are within our 10% acceptable coefficient of variation (CV). The four major cations found in groundwater are Mg^{2+}, Ca^{2+}, K^{+}, and Na^{+}. The dominant cation among the four is Na followed by Mg, Ca and K respectively. The concentrations of these cations in groundwater are usually greater than 1 mg/L (Ackah et al., 2011, Adimalla and health, 2019). Both Mg and Ca ions contribute to water hardness but do not pose any health threat. The ions engage in reactions that leave insoluble mineral deposits (WHO, 2011). Potassium (K^+) is generally low in fresh water (< 10 mg/L). Soils that have been affected by the dominance of sodium (Na^+) are known as 'sodic soils' and it has an adverse effect on the growth and yield of most crops (Qadir et al., 2005). When the amount of sodium concentration is higher relative to both calcium and magnesium it affects the water supply needed by crops by reducing the soil permeability (Kaur et al., 2017). Calcium and magnesium help to maintain a state of equilibrium in most waters. The higher the proportion of Mg^{2+} in waters, the more it will impact on the soil quality thereby converting it to alkaline and reduces crop yields (Gowd, 2005). Figures 5.8 to 5.11 show the measured cations among the four boreholes. As can be seen from the figures, the groundwater from BH1 appears to have very high cation concentrations. Next high concentrations are in BH4 groundwater samples. Particularly, groundwater in BH1 seems to be containing very high levels of sodium (Na^+). The literature value for calcium is 200 - 300 mg/L and potassium is between 10 - 30 mg/L as given in Table 2.2. However, all four boreholes do not exceed them. The standard value for magnesium is 50 - 200 mg/L and only BH1 exceeds it. The literature established standard for sodium is 200 mg/L and only BH2 is within it. In terms of groundwater values from similar catchment as given in Table 5.1, both calcium and potassium from the four boreholes do not exceed the values. Both minimum and maximum values of magnesium and sodium from BH2, BH3 and BH4 are within the catchment values while the minimum and maximum values of BH1 are far greater than the catchment values.

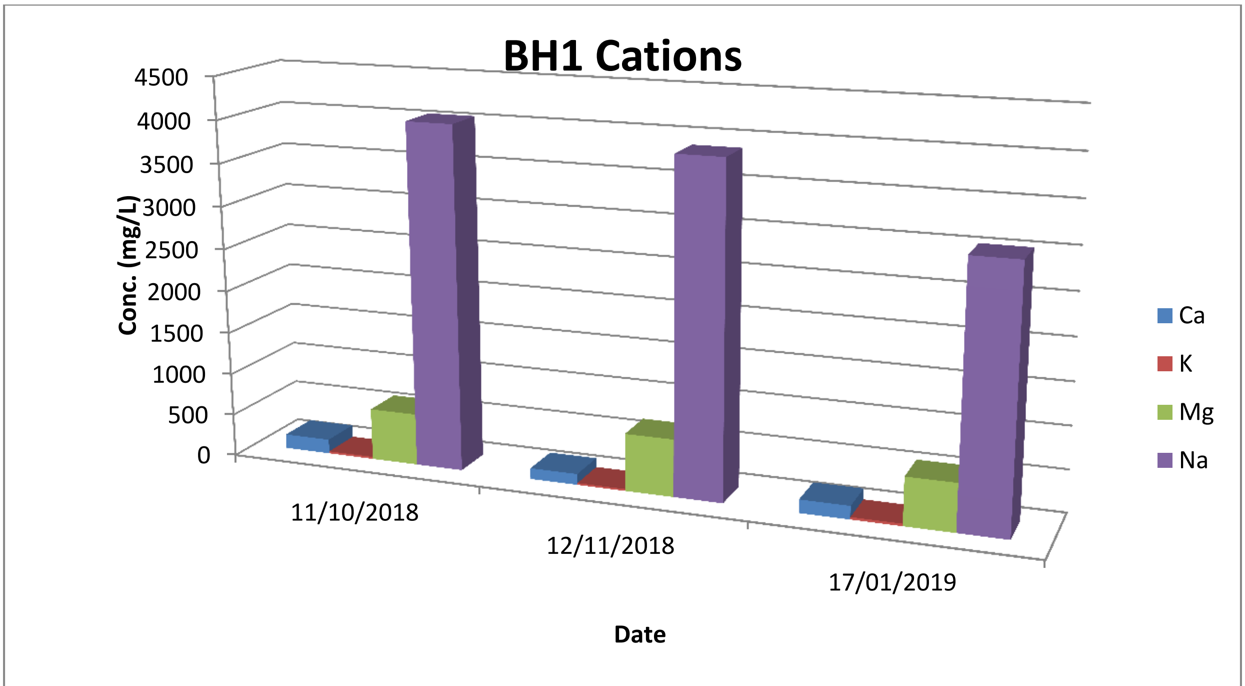

Figure 5. 8: *Concentration of cations in BH1.*

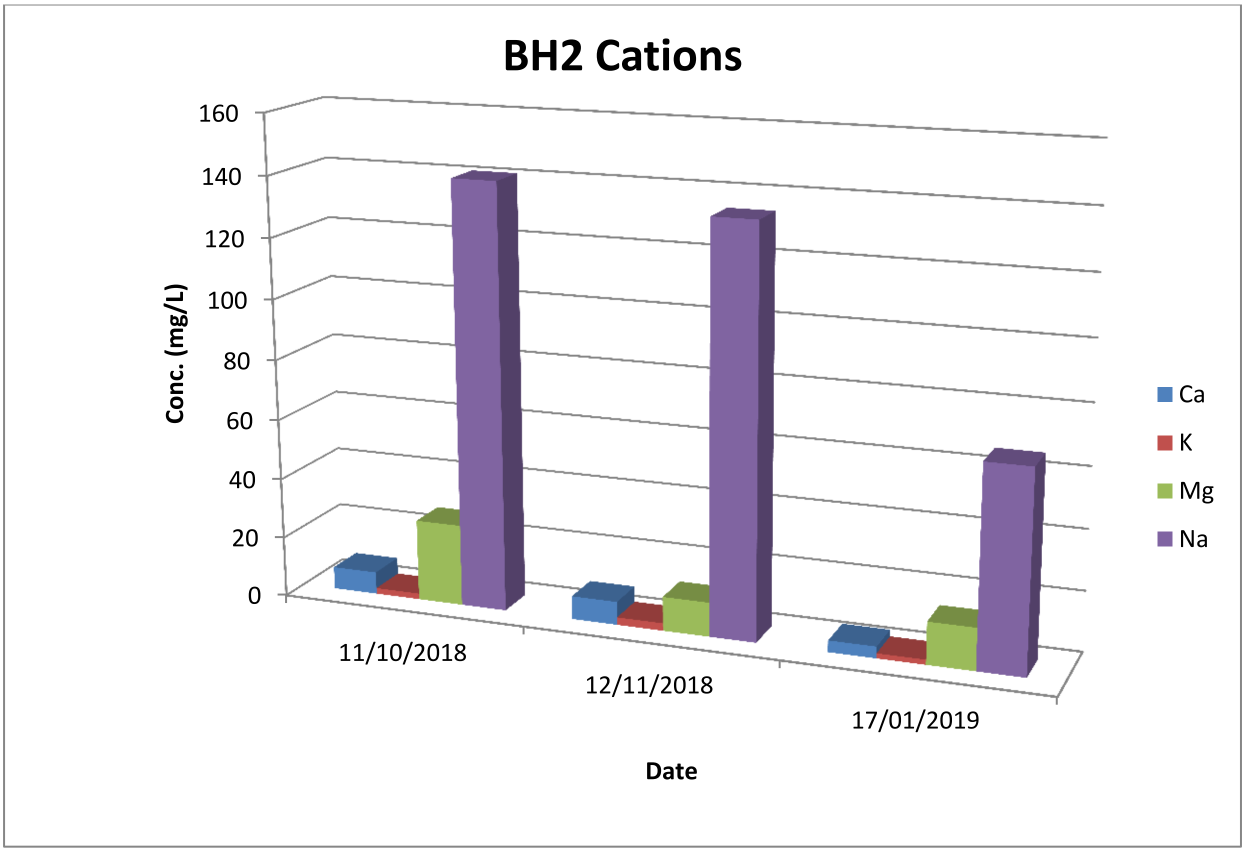

Figure 5. 9: *Concentration of cations in BH2.*

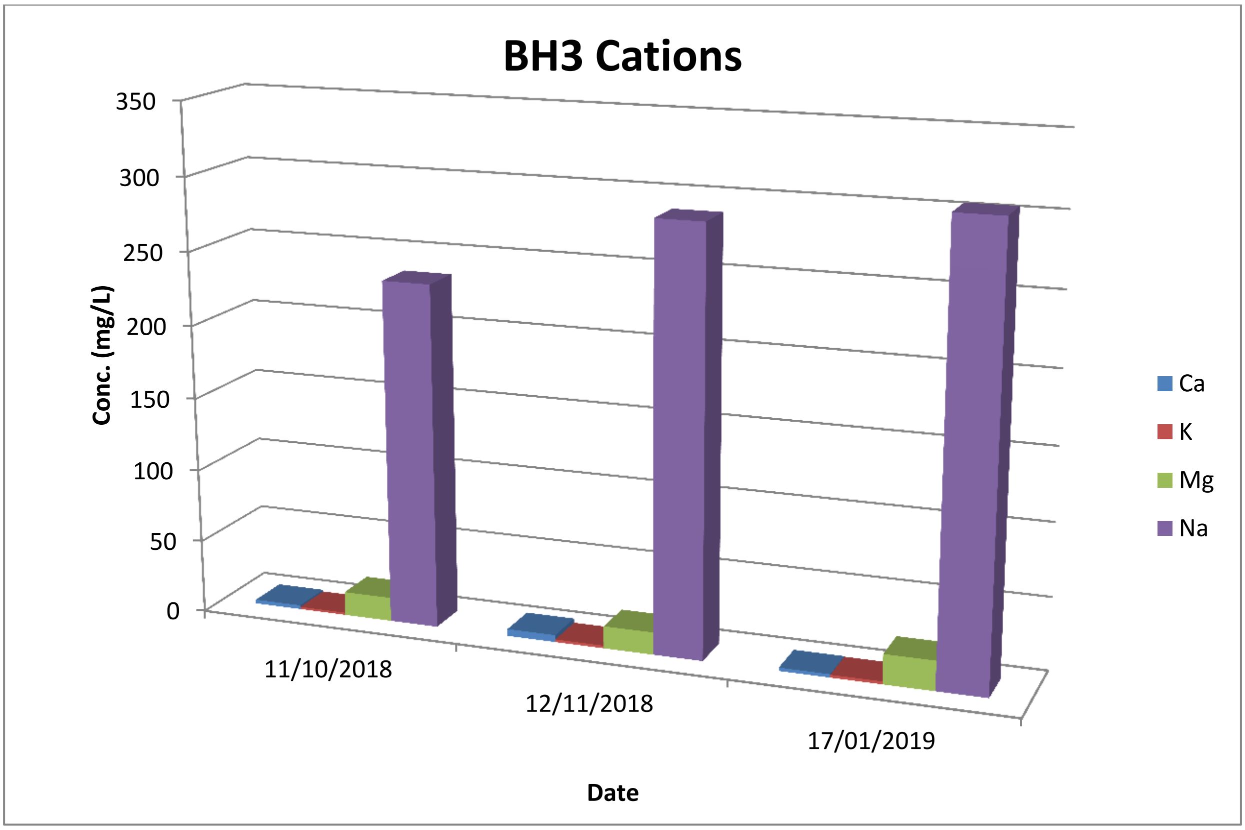

Figure 5. 10: *Concentration of cations in BH3.*

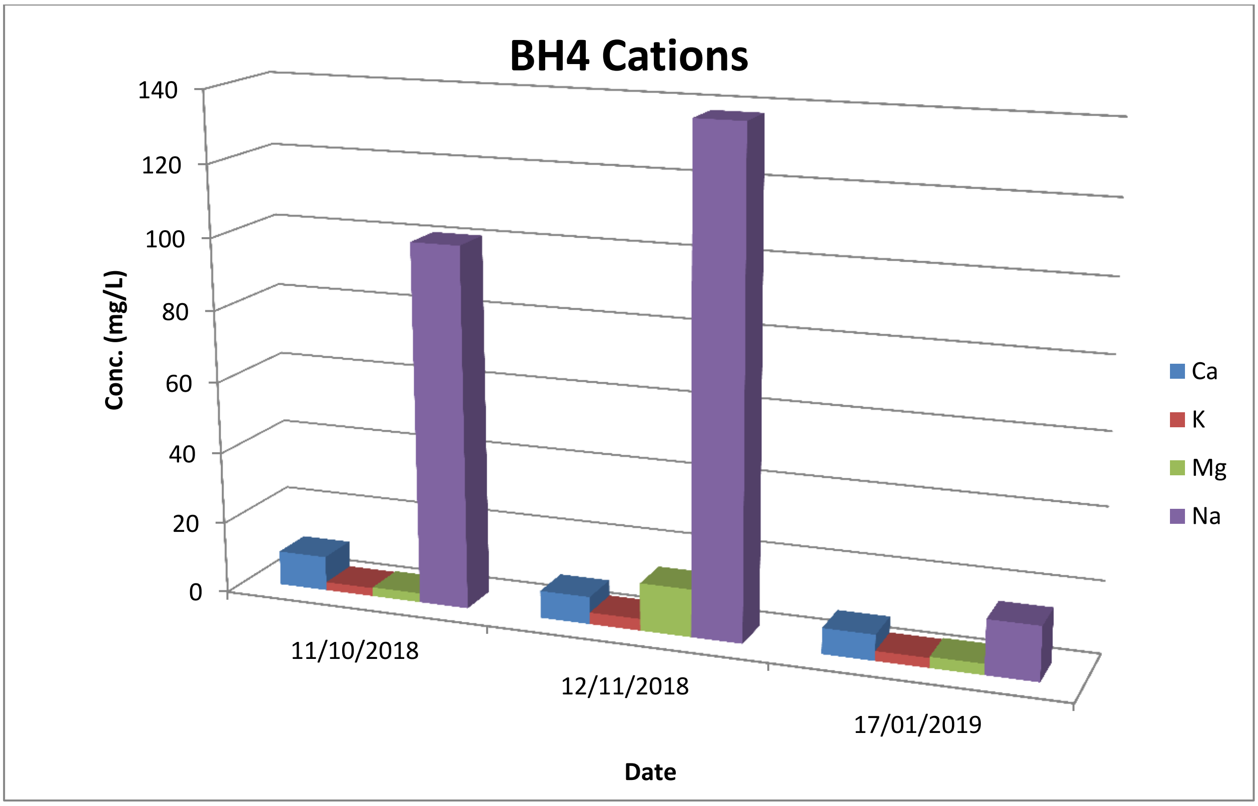

Figure 5. 11: *Concentration of cations in BH4.*

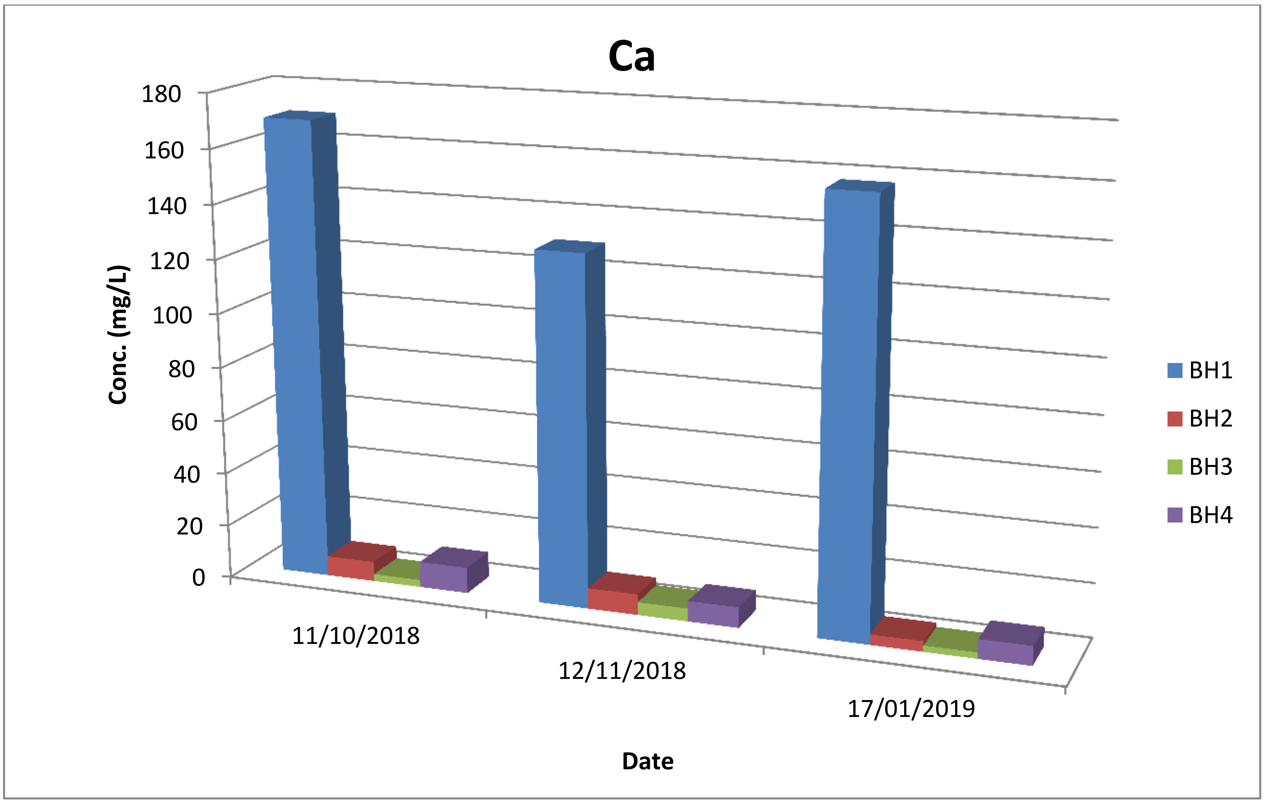

Figure 5. 12: *Variation of Calcium concentrations in the four boreholes.*

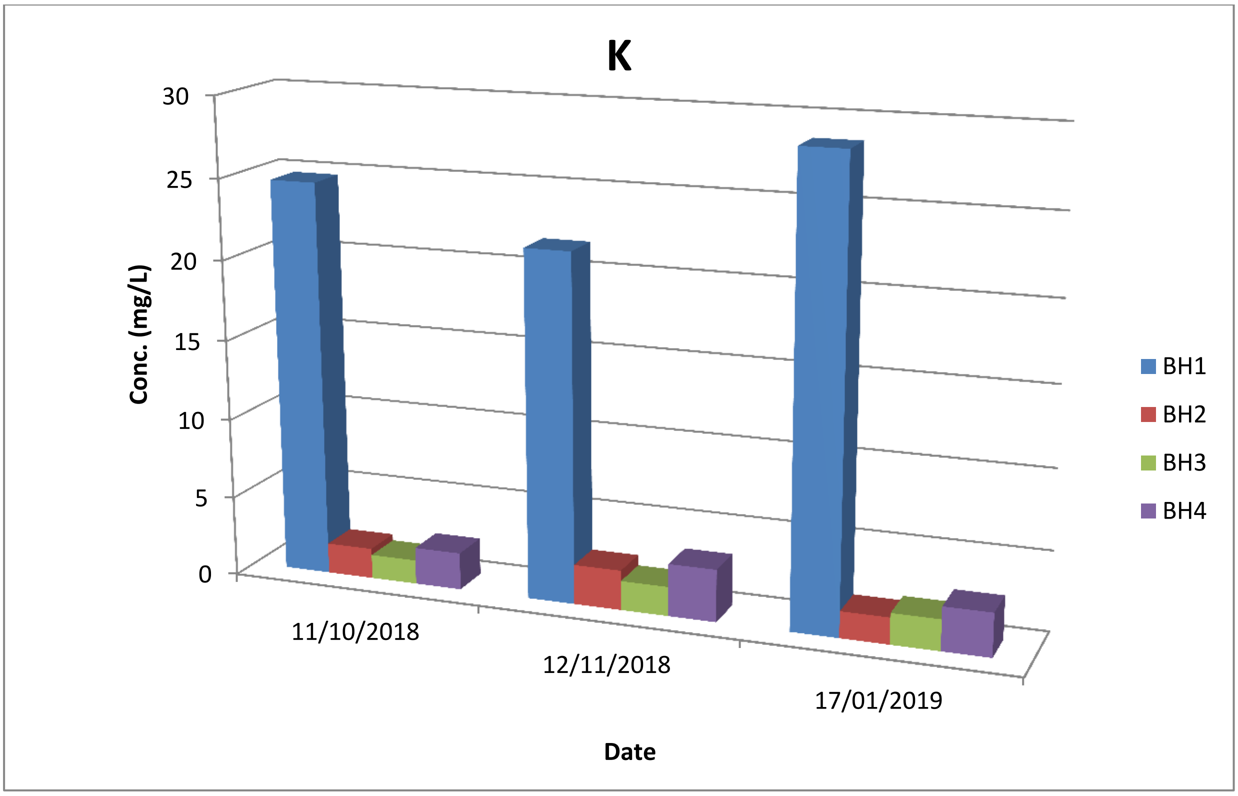

Figure 5. 13: *Variation of Potassium concentrations in the four boreholes.*

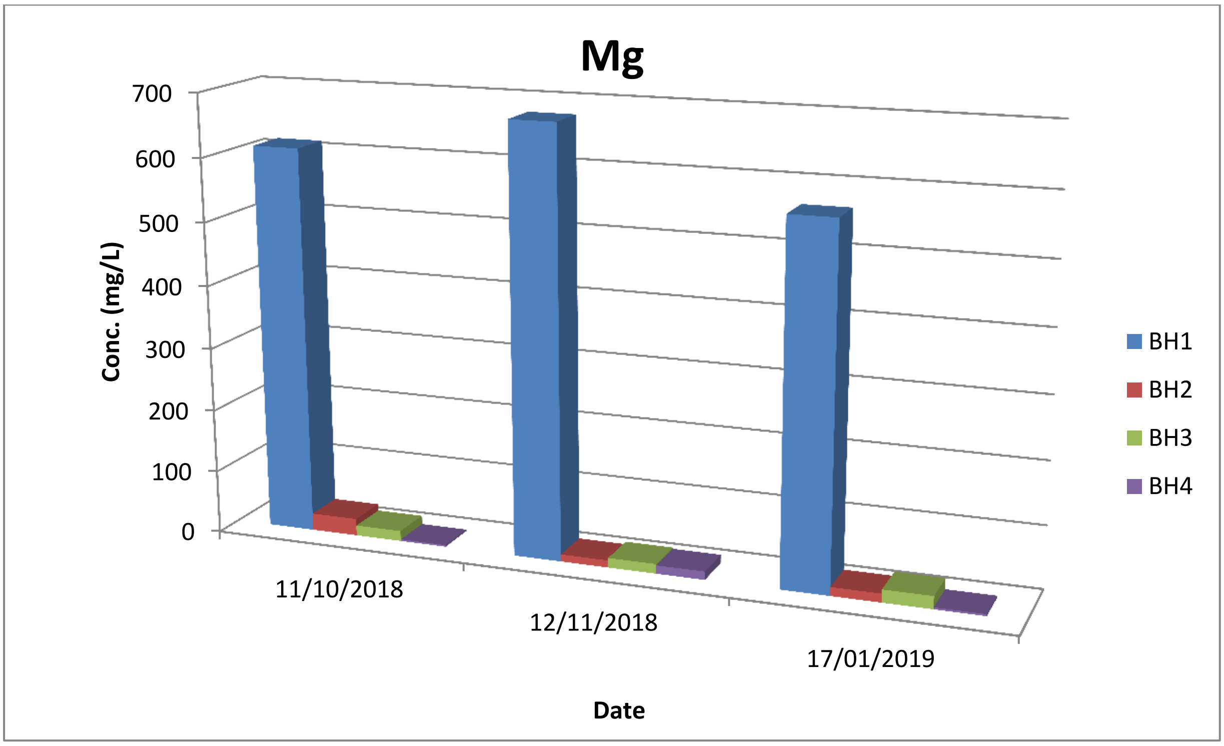

Figure 5. 14: Variation of Magnesium concentrations in the four boreholes

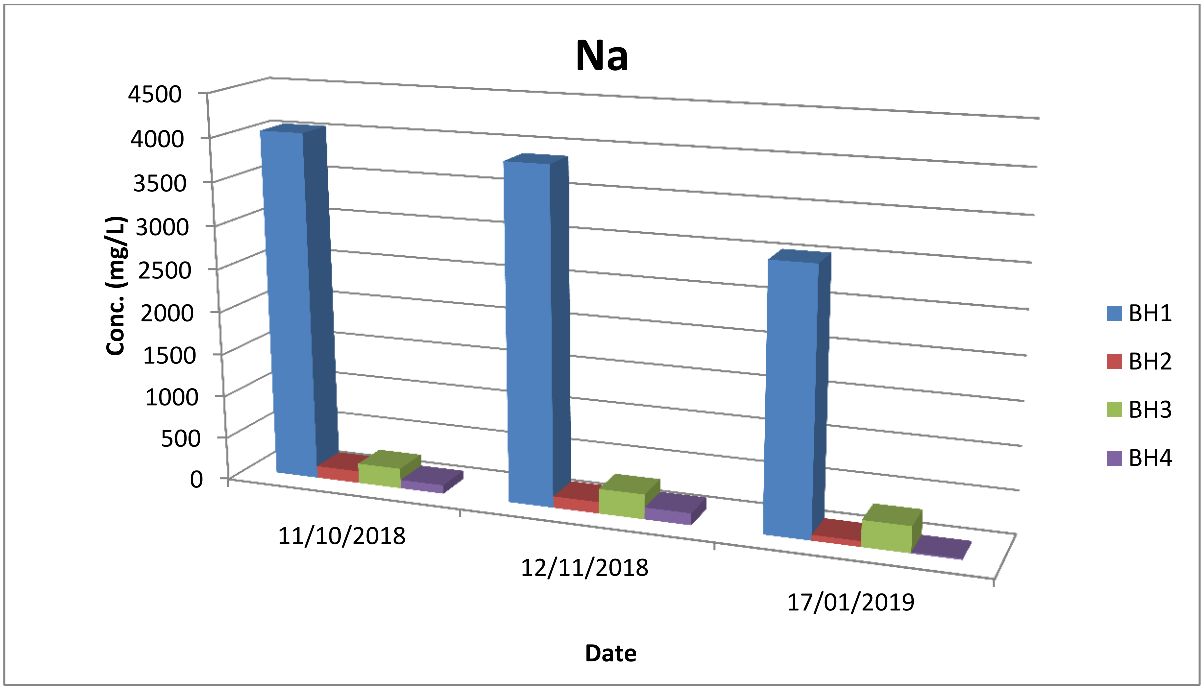

Figure 5. 15: *Variation of Sodium concentrations in the four boreholes.*

5.1.9 Iron (Fe)

Iron (Fe) is a naturally occurring metal and it is widely present in groundwater. Iron is the second most abundant metallic element in the Earth's outer crust. The source of iron in groundwater may originate from a variety of mineral sources, and several sources of iron may be present in a single aquifer system. Iron can exist in either of the two states namely as oxidized state (Fe^{3+}) or reduced state (Fe^{2+}) (Fisher et al., 2004). Iron can be found in all rocks, soil and sand. The most common type is ferrous iron (Idoko, 2010). Season has an influence on groundwater quality variability with respect to iron. The concentration of iron in groundwater during rainy season is higher than during dry season. The coefficient of variation (CV) shows it is lesser in rainy season than in dry season. This could be attributed to influence of rainfall infiltrating and dissolving mineral in rocks and soil which are leached into groundwater sources (Idoko, 2010). Seasonal variation shows that the high iron concentration observed during pre-monsoon may be due to flushing/dissolution of lithogenic and non-lithogenic materials by infiltrating water (Rajmohan et al., 2005). Significant amount of Fe may be leached through the soil with the percolating water in the monsoon season leading to high concentration of iron in the aquifer. High concentrations of iron are attributed to its release in conjunction with arsenic during the reduction of arsenopyrites by oxygen – deficient groundwater (Oinam et al., 2011).

Iron is a vital mineral in the human body and is needed for oxidative energy metabolism, red blood cell production and oxygen transport, plus other important functions. Dietary food sources are the source of majority of iron that is absorbed and utilized in the body. The diets consumed in many parts of the developing countries are low in iron thus cannot provide adequate amounts of iron to meet daily requirements (Karakochuk et al., 2015). The metabolic activity of bacteria impacts on the concentration of iron found in groundwater. Because there are no identified threats posed by iron, there is no EPA primary drinking water standard for iron in water. But, a secondary standard of 0.3 mg/L is established for iron because iron concentrations above this level may produce bad odour, colour, scaling and corrosion (Fisher et al., 2004). There are no established health-based guidelines values for the concentration of iron in drinking water. The ADWG and WHO recommended guideline value of iron for drinking water purposes is 0.3 mg/L (WHO, 2003, NHMRC, 2011, WHO, 2011). High iron concentrations are commonly found in shallow wells of less than 30 m deep than in deeper wells (Fisher et al., 2004). Groundwater lacking in oxygen may contain iron (II) at concentrations up to several mg/L without discolouration or turbidity in the water when pumped directly from a well. Iron concentrations below 0.3 mg/L does not show any noticeable taste, however turbidity and colour may develop in piped systems at levels above 0.05 – 0.1 mg/L (WHO, 2003).

There are variety of methods used for the removal of iron from groundwater such as oxidation-precipitation-filtration, lime softening, ion-exchange, sub-surface iron removal and membrane processes. Stabilisation with phosphate or silicates is applied also in order to avoid the oxidation or precipitation of iron. The most used method among the techniques mentioned above is aeration or chemical oxidation followed by rapid sand filtration (Sharma et al., 2005). Water that contains iron does not have any harmful effect when consumed by human beings. Long term consumption of drinking water with high iron concentration could cause liver disease. Communities can reject groundwater as a source of water supply when the water is coloured due to high iron concentration (Idoko, 2010).

Figure 5.16 shows the concentrations of iron in the groundwater for all the four boreholes. As can be seen from Figure 5.16, the groundwater from BH1 seems to have highest iron concentrations and the groundwater from BH3 appears to be having the lowest iron concentrations. The concentrations of iron found in BH3 and BH4 are lower compared with that found in BH2 and BH1. BH3 and BH4 are located in-between Wattle Grove Lake (WGL). Lower concentrations of terrace elements are associated with recharge of fresh water from the lakes (Rajmohan et al., 2005). The possible source for the iron may be from the soil around the boreholes. All four boreholes values are within the catchment values given in Table 5.1. The results of the samples collected between 19/07/2018 to 4/02/2019 for BH2, BH3 and BH4 were outside the calibrated standard concentrations and was not used in Figure 5.16. In the case of BH1, the iron concentrations were within the calibrated standard concentrations for all sampled groundwater. The analysis was repeated for BH2, BH3 and BH4 in order to understand the seasonal impact on their iron concentrations.

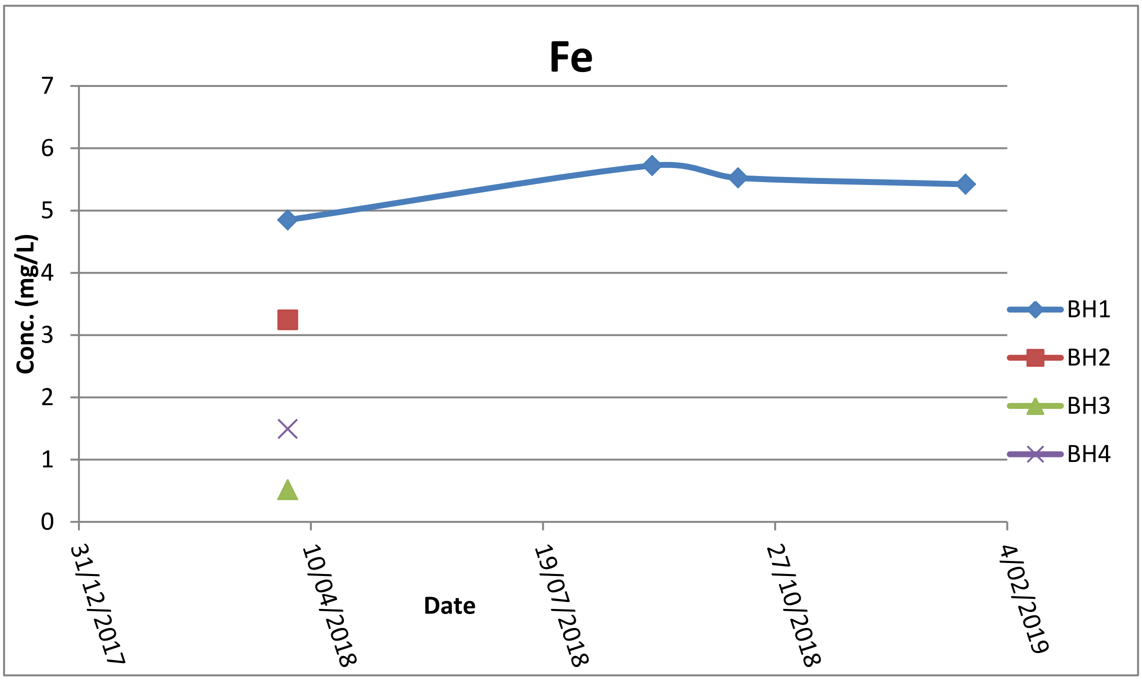

Figure 5. 16: *Concentration of iron in the four boreholes.*

CHAPTER 6

CONCLUSIONS AND RECOMMENDATIONS

6.1 Conclusions

This thesis focuses on 'estimation of groundwater recharge and assessment of groundwater quality in urban landscapes. The concept was applied to: a case of Wattle Grove area, located near Liverpool, Sydney, Australia'. To facilitate recharge estimation and groundwater quality assessment four newly developed boreholes were established. The conclusions from this study are divided into two sections namely: groundwater recharge and groundwater quality.

6.1.1 Groundwater recharge conclusions

The groundwater recharge per day was estimated for all four monitoring wells by the use of water table fluctuation method. However, the average specific yield was used in the WTF equation and the rise method was used to determine the change in water table height. The recharge per day for each borehole was estimated by considering wet and dry seasons. BH1 had the highest recharge per day of 1.67 mm/day while the least was BH4 with 0.28 mm/day. Both BH1 and BH4 recharge per day were less than 1 mm/day and it could be as a result of drier soils with higher moisture holding capacity and smaller rainfall events. The different recharge estimates obtained at the sites are due to the sites having different surface topography, impact of water bodies and underground movement of water. Analysis of rainfall impact on recharge estimation shows that not all amount of rainfall contribute to recharge as a result of canopy interception, the amount of storage available above and below ground and specific yield. The variation of S_y with depth was calculated and it showed that S_y varies greatly with depth. On correlating S_y with depth, it revealed that S_y depends strongly on depth.

This study also examined the correlations between rainfall events and groundwater-level fluctuations. Cross- correlation technique was used to identify the response time of rainfall to groundwater - level fluctuations known as lag time in days. The seasonal cross-correlation known as the sliding window principle was used to find out how rainfall impacts on considered seasons in terms of groundwater-level fluctuations. The considered daily groundwater table and daily rainfall obtained from BOM station (066161) were used in carrying out the analysis. In both spring and summer season, BH4 had the fastest response time of 3 days while BH2 had a longer lag time of 14 days. The fastest time is as a result of fissure flow while the longer lag time is attributed to slow matrix flow and the rainfall response not in phase with the groundwater - level fluctuations. The combined analysis of spring and summer as a season revealed that BH4 lag time was 3 days and both BH1 and BH2 recorded 9 days each while BH3 was 13 days. Rainfall had a more direct impact on the groundwater – level fluctuations of BH4 followed by BH1 and BH2 and the least was BH3.

6.1.2 Groundwater quality conclusions

This study has shown that the spatial distribution of groundwater quality and of the changes in time that occur, either naturally, or as a result of human activities among the four monitored boreholes. The groundwater from BH1 is highly turbid which is also confirmed from its high dissolved solids content. Water samples of all boreholes successfully meet the guidelines of Australian Drinking Water Guideline, World Health Organization and literature values in terms of pH, calcium, and potassium concentrations. BH1 has the highest sodium and magnesium content which are significantly above the literature values as well as other boreholes. This indicates that the BH1 water may have been particularly contaminated by the salt present either in the soil or disposal of waste that may have been dumped in the surrounding area. If the groundwater is to be used directly for drinking, it shows all four are not suitable as they exceeded the turbidity standard level of 5 NTU. However, if EPA's secondary drinking water guideline of TDS not exceeding 500 mg/L and WHO's TDS value of less than 600 mg/L are used, only BH2 is suitable. Furthermore, BH1 is unfit for irrigation as it exceeded the TDS level of EPA's guideline of 1,500 to 2,600 mg/L.

6.2 Recommendations

The following recommendations are suggested for further study:

1. The use of longer period of hydrological data to better reflect on the characteristics of the study site.
2. The synchronisation of the data loggers' readings in minutes or hours with a weather station set-up in the site that will record in tandem with the loggers so that the recharge and rainfall response time to groundwater - level fluctuations can be obtained in the same units.
3. The location of BH1 is very close to Holsworthy barracks and its water quality is the worst among the three. Therefore, the source of this contamination must be investigated.

4. Finally, longer monitoring of the groundwater is required to draw objective conclusions in regards to its quality and sustainable use.
5. Carryout comprehensive groundwater modelling to determine the recharge rate and the contaminant movement.
6. Comprehensive seasonal analysis of iron concentrations to reveal why iron was found in BH1 throughout the four sampled times but only once in the other three boreholes.
7. The correlation of groundwater table and rainfall could be improved by using the overall daily trend as opposed to water table depth difference between first and last day of each month and monthly rainfall as applied here.
8. Small – scale fluctuations should be filtered out as not all fluctuations are caused by rainfall in order to improve the correlation coefficient.

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APPENDIX A PHYSICO-CHEMICAL PARAMETERS OF SOIL SAMPLES AND GROUNDWATER SAMPLES FROM THE FOUR MONITORING POINTS

Table A.0.1: BH1 specific retention analysis

Sample ID	**Pressure (bars)**	**Tin Weight with Lid(g)**	**Tin, Lid and wet soil (g)**	**Tin, Lid and dry soil (g)**	**Wet soil Minus Tin (g)**	**Dry soil Minus Tin (g)**	**% Moisture**
BH1-Surface	0.5	59.2746	88.373	83.06	29.0984	23.7854	22.33723
BH1-0.5M	0.5	61.992	92.803	86.664	30.811	24.672	24.88246
BH1- 1M	0.5	59.075	90.071	83.776	30.996	24.701	25.4848
BH1-2M	0.4	59.2746	87.374	81.666	28.0994	22.3914	25.49193
BH1-3M	0.4	62.827	94.054	88.599	31.227	25.772	21.16638
BH1-4M	0.35	59.075	93.072	87.15	33.997	28.075	21.0935
BH1-4.5M	0.4	60.35	91.633	86.294	31.283	25.944	20.57894
BH1-5M	0.35	60.091	93.896	88.471	33.805	28.38	19.11557
BH1-6M	0.35	60.787	90.704	85.787	29.917	25	19.668
BH1-7M	0.35	61.992	85.759	81.699	23.767	19.707	20.60182
BH1-8-9M	0.5	60.35	85.612	82.012	25.262	21.662	16.61896
Average							21.54905

Table A.0.2: BH2 specific retention analysis

Sample ID	**Pressure (bars)**	**Tin Weight with Lid(g)**	**Tin, Lid and wet soil (g)**	**Tin, Lid and dry soil (g)**	**Wet soil Minus Tin (g)**	**Dry soil Minus Tin (g)**	**% Moisture**
BH2-Surface	0.5	58.2096	87.441	84.303	29.2314	26.0934	12.02603
BH2- 0.5M	0.5	60.787	90.626	87.204	29.839	26.417	12.95378
BH2-1M	0.5	61.417	87.733	84.396	26.316	22.979	14.52195
BH2-2M	0.5	60.686	91.334	86.062	30.648	25.376	20.77554
BH2-3M	0.4	59.5932	88.642	84.619	29.0488	25.0258	16.07541
BH2-4M	0.4	58.2096	87.048	83.349	28.8384	25.1394	14.71395
BH2- 5M	0.4	60.686	90.863	87.321	30.177	26.635	13.29829
BH2- 5-6M	0.4	61.992	89.363	84.249	27.371	22.257	22.97704
BH2-6M	0.4	60.444	90.829	85.326	30.385	24.882	22.11639
BH2-7M	0.5	60.35	89.654	83.563	29.304	23.213	26.23961
BH2-(8-9M)	0.35	60.686	87.157	82.178	26.471	21.492	23.16676
BH2-9+M	0.35	61.417	90.377	85.275	28.96	23.858	21.38486
Average							18.35413

Table A.0.3: BH3 specific retention analysis

Sample ID	**Pressure (bars)**	**Tin Weight with Lid(g)**	**Tin, Lid and wet soil (g)**	**Tin, Lid and dry soil (g)**	**Wet soil Minus Tin (g)**	**Dry soil Minus Tin (g)**	**% Moisture**
BH3- Surface	0.5	59.5932	88.299	85.333	28.7058	25.7398	11.52301
BH3- 0.5M	0.5	60.686	90.397	87.179	29.711	26.493	12.1466
BH3- 1M	0.5	60.091	86.847	84.531	26.756	24.44	9.476268
BH3-2M	0.5	60.444	87.01	83.51	26.566	23.066	15.17385
BH3-3M	0.5	62.827	87.842	83.725	25.015	20.898	19.70045
BH3- 4M	0.4	60.091	89.553	85.265	29.462	25.174	17.03345
BH3-5M	0.5	59.5932	88.212	83.935	28.6188	24.3418	17.5706
BH3-6M	0.5	58.2096	85.054	81.518	26.8444	23.3084	15.1705
BH3-7M	0.35	59.2746	86.309	83.103	27.0344	23.8284	13.45453
BH3-(7-8M)	0.35	60.35	86.46	83.756	26.11	23.406	11.55259
BH3-9+M	0.35	58.2096	87.52	83.044	29.3104	24.8344	18.02339
BH3-10M	0.5	59.2746	82.647	78.967	23.3724	19.6924	18.68741
Average							14.95939

Table A.0.4: BH4 specific retention analysis

Sample ID	Pressure (bars)	Tin Weight with Lid(g)	Tin, Lid and wet soil (g)	Tin, Lid and dry soil (g)	Wet soil Minus Tin (g)	Dry soil Minus Tin (g)	% Moisture
BH4- Surface	0.5	62.827	88.842	85.098	26.015	22.271	16.8111
BH4-0.5M	0.5	60.444	88.281	84.006	27.837	23.562	18.14362
BH4- 1M	0.5	61.417	93.046	85.936	31.629	24.519	28.99792
BH4- 2M	0.4	59.075	90.191	84.347	31.116	25.272	23.12441
BH4- 3M	0.4	60.787	91.28	84.728	30.493	23.941	27.36728
BH4- 4M	0.5	59.075	89.689	82.42	30.614	23.345	31.13729
BH4-5M	0.4	61.417	92.433	86.945	31.016	25.528	21.49796
BH4- 6M	0.5	60.787	92.231	85.114	31.444	24.327	29.25556
BH4-7M	0.5	61.992	96.615	90.699	34.623	28.707	20.60821
BH4-8M	0.5	60.091	91.778	86.185	31.687	26.094	21.43405
BH4-(8-9M)	0.35	62.827	90.684	86.059	27.857	23.232	19.90789
BH4-9M	0.35	59.5932	87.606	82.837	28.0128	23.2438	20.5173
BH4-10M	0.35	60.444	85.772	81.394	25.328	20.95	20.89737
Average							23.05384

Table A.0.5: Calculation of the boreholes porosity values

Boreholes	Wt. of core sampler (g)	Avg. Wt. of tins (g)	Avg.Wt. of tins+moist soil (g)	Avg. Wt. of moist soil (g)	Avg. Wt. of dry soil+tins (g)	Avg. Wt. of dry soil (g)	Soil bulk density (g/cm^3)	Soil porosity (%)
BH1	1497.17	64.719	105.921	41.201	100.432	35.713	0.990	62.637
BH2	1497.40	56.468	113.512	57.043	108.794	52.326	1.338	49.518
BH3	1498.02	40.997	112.897	71.900	108.841	67.844	1.257	52.564
BH4	1496.96	30.162	106.013	75.849	98.7523	68.590	1.211	54.297

Table A.0.6: PSD of BH1 surface soil

Size of sieve (mm)	Mass of empty sieve (g)	Mass of Sieve +Soil retained (g)	Soil retained (g)	Percent retained	Cumulative retained %	Percent passing
6.7	435.43	440	4.57	2.309831	2.309831	97.69017
4.75	428.08	432	3.92	1.981299	4.29113	95.70887
2.36	396.68	398	1.32	0.667172	4.958302	95.0417
1.18	363.05	367.53	4.48	2.264342	7.222643	92.77736
0.6	323.25	346.99	23.74	11.99899	19.22163	80.77837
0.425	340.69	380.43	39.74	20.08592	39.30756	60.69244
0.3	301.07	342.76	41.69	21.07152	60.37908	39.62092
0.15	274.04	312.87	38.83	19.62598	80.00505	19.99495
0.075	264.76	280.48	15.72	7.945413	87.95047	12.04953
0	271.36	295.2	23.84	12.04953	100	
			197.85			

Table A. 0.7: PSD of BH1 @ 0.5 m

Size of sieve (mm)	Mass of empty sieve (g)	Mass of Sieve +Soil retained (g)	Soil retained (g)	Percent retained	Cumulative retained %	Percent passing
6.7	435.43	436.41	0.98	0.510045	0.510045	99.48996
4.75	428.08	429.22	1.14	0.593317	1.103362	98.89664
2.36	396.68	397.96	1.28	0.666181	1.769543	98.23046
1.18	363.05	368.07	5.02	2.612678	4.382221	95.61778
0.6	323.25	340.72	17.47	9.092329	13.47455	86.52545
0.425	340.69	367.15	26.46	13.77121	27.24576	72.75424
0.3	301.07	336.83	35.76	18.61143	45.85719	54.14281
0.15	274.04	324.33	50.29	26.17362	72.03081	27.96919
0.075	264.76	289.22	24.46	12.7303	84.76111	15.23889
0	271.36	300.64	29.28	15.23889	100	
			192.14			

Table A.0.8: PSD of BH1 @ 1 m

Size of sieve (mm)	Mass of empty sieve (g)	Mass of Sieve +Soil retained (g)	Soil retained (g)	Percent retained	Cumulative retained %	Percent passing
6.7	435.43	468.23	32.8	17.53354	17.53354	82.46646
4.75	428.08	428.58	0.5	0.26728	17.80082	82.19918
2.36	396.68	399.79	3.11	1.662479	19.4633	80.5367
1.18	363.05	373.39	10.34	5.527343	24.99065	75.00935
0.6	323.25	338.81	15.56	8.317742	33.30839	66.69161
0.425	340.69	354.09	13.4	7.163094	40.47148	59.52852
0.3	301.07	325.63	24.56	13.12878	53.60026	46.39974
0.15	274.04	318.37	44.33	23.69701	77.29727	22.70273
0.075	264.76	284.63	19.87	10.62169	87.91896	12.08104
0	271.36	293.96	22.6	12.08104	100	
			187.07			

Table A.0.9: PSD of BH1 @ 2 m

Size of sieve (mm)	Mass of empty sieve (g)	Mass of Sieve +Soil retained (g)	Soil retained (g)	Percent retained	Cumulative retained %	Percent passing
6.7	435.43	448.2	12.77	6.566228	6.566228	93.43377
4.75	428.08	439.98	11.9	6.118881	12.68511	87.31489
2.36	396.68	440.94	44.26	22.75812	35.44323	64.55677
1.18	363.05	408.38	45.33	23.30831	58.75154	41.24846
0.6	323.25	348.18	24.93	12.8188	71.57034	28.42966
0.425	340.69	351.01	10.32	5.306458	76.8768	23.1232
0.3	301.07	309.94	8.87	4.56088	81.43768	18.56232
0.15	274.04	287.33	13.29	6.833608	88.27129	11.72871
0.075	264.76	274.23	9.47	4.869395	93.14068	6.859317
0	271.36	284.7	13.34	6.859317	100	
			194.48			

Table A.0.10 : PSD of BH1 @ 3 m

Size of sieve (mm)	Mass of empty sieve (g)	Mass of Sieve +Soil retained (g)	Soil retained (g)	Percent retained	Cumulative retained %	Percent passing
6.7	435.43	444.95	9.52	5.871107	5.871107	94.12889
4.75	428.08	434.77	6.69	4.125809	9.996916	90.00308
2.36	396.68	415.03	18.35	11.31668	21.3136	78.6864
1.18	363.05	392.48	29.43	18.14986	39.46346	60.53654
0.6	323.25	348.26	25.01	15.42399	54.88745	45.11255
0.425	340.69	351.7	11.01	6.790009	61.67746	38.32254
0.3	301.07	311.09	10.02	6.179463	67.85692	32.14308
0.15	274.04	291.44	17.4	10.7308	78.58773	21.41227
0.075	264.76	278.76	14	8.633981	87.22171	12.77829
0	271.36	292.08	20.72	12.77829	100	
			162.15			

Table A.0.11: PSD of BH @ 4 m

Size of sieve (mm)	Mass of empty sieve (g)	Mass of Sieve +Soil retained (g)	Soil retained (g)	Percent retained	Cumulative retained %	Percent passing
6.7	435.43	468.23	32.8	17.53354	17.53354	82.46646
4.75	428.08	428.58	0.5	0.26728	17.80082	82.19918
2.36	396.68	399.79	3.11	1.662479	19.4633	80.5367
1.18	363.05	373.39	10.34	5.527343	24.99065	75.00935
0.6	323.25	338.81	15.56	8.317742	33.30839	66.69161
0.425	340.69	354.09	13.4	7.163094	40.47148	59.52852
0.3	301.07	325.63	24.56	13.12878	53.60026	46.39974
0.15	274.04	318.37	44.33	23.69701	77.29727	22.70273
0.075	264.76	284.63	19.87	10.62169	87.91896	12.08104
0	271.36	293.96	22.6	12.08104	100	
			187.07			

Table A.0.12: PSD of BH1 @ 5 m

Size of sieve (mm)	Mass of empty sieve (g)	Mass of Sieve +Soil retained (g)	Soil retained (g)	Percent retained	Cumulative retained %	Percent passing
6.7	435.43	448.2	12.77	6.566228	6.566228	93.43377
4.75	428.08	439.98	11.9	6.118881	12.68511	87.31489
2.36	396.68	440.94	44.26	22.75812	35.44323	64.55677
1.18	363.05	408.38	45.33	23.30831	58.75154	41.24846
0.6	323.25	348.18	24.93	12.8188	71.57034	28.42966
0.425	340.69	351.01	10.32	5.306458	76.8768	23.1232
0.3	301.07	309.94	8.87	4.56088	81.43768	18.56232
0.15	274.04	287.33	13.29	6.833608	88.27129	11.72871
0.075	264.76	274.23	9.47	4.869395	93.14068	6.859317
0	271.36	284.7	13.34	6.859317	100	
			194.48			

Table A.0.13: PSD of BH1 @ 6 m

Size of sieve (mm)	Mass of empty sieve (g)	Mass of Sieve +Soil retained (g)	Soil retained (g)	Percent retained	Cumulative retained %	Percent passing
6.7	435.43	439.67	4.24	2.232401	2.232401	97.7676
4.75	428.08	432.1	4.02	2.116569	4.348971	95.65103
2.36	396.68	417.29	20.61	10.85137	15.20034	84.79966
1.18	363.05	411.72	48.67	25.62523	40.82557	59.17443
0.6	323.25	367.88	44.63	23.49813	64.3237	35.6763
0.425	340.69	357.29	16.6	8.740062	73.06376	26.93624
0.3	301.07	312.92	11.85	6.239141	79.3029	20.6971
0.15	274.04	289.39	15.35	8.081925	87.38483	12.61517
0.075	264.76	275.02	10.26	5.40199	92.78682	7.213184
0	271.36	285.06	13.7	7.213184	100	
			189.93			

Table A.0.14: PSD of BH1 @ 7 m

Size of sieve (mm)	Mass of empty sieve (g)	Mass of Sieve +Soil retained (g)	Soil retained (g)	Percent retained	Cumulative retained %	Percent passing
6.7	435.43	439.2	3.77	1.996399	1.996399	98.0036
4.75	428.08	429.27	1.19	0.630163	2.626562	97.37344
2.36	396.68	410.52	13.84	7.328956	9.955518	90.04448
1.18	363.05	419.69	56.64	29.99365	39.94916	60.05084
0.6	323.25	373.69	50.44	26.71044	66.65961	33.34039
0.425	340.69	357.16	16.47	8.721669	75.38128	24.61872
0.3	301.07	312.43	11.36	6.015675	81.39695	18.60305
0.15	274.04	287.73	13.69	7.249523	88.64647	11.35353
0.075	264.76	273.21	8.45	4.474688	93.12116	6.878839
0	271.36	284.35	12.99	6.878839	100	
			188.84			

Table A.0.15: PSD of BH1 @ 8-9 m

Size of sieve (mm)	Mass of empty sieve (g)	Mass of Sieve +Soil retained (g)	Soil retained (g)	Percent retained	Cumulative retained %	Percent passing
6.7	435.43	444.95	9.52	5.871107	5.871107	94.12889
4.75	428.08	434.77	6.69	4.125809	9.996916	90.00308
2.36	396.68	415.03	18.35	11.31668	21.3136	78.6864
1.18	363.05	392.48	29.43	18.14986	39.46346	60.53654
0.6	323.25	348.26	25.01	15.42399	54.88745	45.11255
0.425	340.69	351.7	11.01	6.790009	61.67746	38.32254
0.3	301.07	311.09	10.02	6.179463	67.85692	32.14308
0.15	274.04	291.44	17.4	10.7308	78.58773	21.41227
0.075	264.76	278.76	14	8.633981	87.22171	12.77829
0	271.36	292.08	20.72	12.77829	100	
			162.15			

Table A.0.16 : PSD of BH2 @ surface

Size of sieve (mm)	Mass of empty sieve (g)	Mass of Sieve +Soil retained (g)	Soil retained (g)	Percent retained	Cumulative retained %	Percent passing
6.7	435.43	442.16	6.73	3.564807	3.564807	96.43519
4.75	428.08	435.96	7.88	4.17395	7.738757	92.26124
2.36	396.68	404.71	8.03	4.253403	11.99216	88.00784
1.18	363.05	373.37	10.32	5.466391	17.45855	82.54145
0.6	323.25	349.4	26.15	13.85137	31.30992	68.69008
0.425	340.69	370.32	29.63	15.69469	47.00461	52.99539
0.3	301.07	328.3	27.23	14.42343	61.42804	38.57196
0.15	274.04	308.87	34.83	18.44907	79.87711	20.12289
0.075	264.76	282.4	17.64	9.343715	89.22083	10.77917
0	271.36	291.71	20.35	10.77917	100	0
			188.79			

Table A.0.17: PSD of BH2 @ 1 m

Size of sieve (mm)	Mass of empty sieve (g)	Mass of Sieve +Soil retained (g)	Soil retained (g)	Percent retained	Cumulative retained %	Percent passing
6.7	435.43	436.41	0.98	0.510045	0.510045	99.48996
4.75	428.08	429.22	1.14	0.593317	1.103362	98.89664
2.36	396.68	397.96	1.28	0.666181	1.769543	98.23046
1.18	363.05	368.07	5.02	2.612678	4.382221	95.61778
0.6	323.25	340.72	17.47	9.092329	13.47455	86.52545
0.425	340.69	367.15	26.46	13.77121	27.24576	72.75424
0.3	301.07	336.83	35.76	18.61143	45.85719	54.14281
0.15	274.04	324.33	50.29	26.17362	72.03081	27.96919
0.075	264.76	289.22	24.46	12.7303	84.76111	15.23889
0	271.36	300.64	29.28	15.23889	100	
			192.14			

Table A. 0.18: PSD for BH2 @ 2 m

Size of sieve (mm)	Mass of empty sieve (g)	Mass of Sieve +Soil retained (g)	Soil retained (g)	Percent retained	Cumulative retained %	Percent passing
6.7	435.43	443.18	7.75	6.274288	6.274288	93.72571
4.75	428.08	431.68	3.6	2.914508	9.188795	90.8112
2.36	396.68	401.39	4.71	3.813148	13.00194	86.99806
1.18	363.05	375.16	12.11	9.80408	22.80602	77.19398
0.6	323.25	341.32	18.07	14.62921	37.43523	62.56477
0.425	340.69	354.91	14.22	11.51231	48.94754	51.05246
0.3	301.07	315.18	14.11	11.42325	60.37079	39.62921
0.15	274.04	293.97	19.93	16.13504	76.50583	23.49417
0.075	264.76	276.74	11.98	9.698834	86.20466	13.79534
0	271.36	288.4	17.04	13.79534	100	0
			123.52			

Table A. 0.19: PSD for BH2 @ 3 m

Size of sieve (mm)	Mass of empty sieve (g)	Mass of Sieve +Soil retained (g)	Soil retained (g)	Percent retained	Cumulative retained %	Percent passing
6.7	435.43	448.36	12.93	6.837652	6.837652	93.16235
4.75	428.08	433.1	5.02	2.65468	9.492332	90.50767
2.36	396.68	407.71	11.03	5.832893	15.32522	84.67478
1.18	363.05	388.78	25.73	13.60656	28.93178	71.06822
0.6	323.25	354.36	31.11	16.45161	45.3834	54.6166
0.425	340.69	363.86	23.17	12.25278	57.63617	42.36383
0.3	301.07	323.24	22.17	11.72396	69.36013	30.63987
0.15	274.04	300.55	26.51	14.01904	83.37916	16.62084
0.075	264.76	278.05	13.29	7.028027	90.40719	9.592808
0	271.36	289.5	18.14	9.592808	100	
			189.1			

Table A.0.20: PSD for BH2 @ 4 m

Size of sieve (mm)	Mass of empty sieve (g)	Mass of Sieve +Soil retained (g)	Soil retained (g)	Percent retained	Cumulative retained %	Percent passing
6.7	435.43	468.23	32.8	17.53354	17.53354	82.46646
4.75	428.08	428.58	0.5	0.26728	17.80082	82.19918
2.36	396.68	399.79	3.11	1.662479	19.4633	80.5367
1.18	363.05	373.39	10.34	5.527343	24.99065	75.00935
0.6	323.25	338.81	15.56	8.317742	33.30839	66.69161
0.425	340.69	354.09	13.4	7.163094	40.47148	59.52852
0.3	301.07	325.63	24.56	13.12878	53.60026	46.39974
0.15	274.04	318.37	44.33	23.69701	77.29727	22.70273
0.075	264.76	284.63	19.87	10.62169	87.91896	12.08104
0	271.36	293.96	22.6	12.08104	100	
			187.07			

Table A. 0.21: PSD for BH2 @ 5 m

Size of sieve (mm)	Mass of empty sieve (g)	Mass of Sieve +Soil retained (g)	Soil retained (g)	Percent retained	Cumulative retained %	Percent passing
6.7	435.43	441.08	5.65	2.723679	2.723679	97.27632
4.75	428.08	440.41	12.33	5.943887	8.667567	91.33243
2.36	396.68	408.38	11.7	5.640185	14.30775	85.69225
1.18	363.05	379.63	16.58	7.992673	22.30042	77.69958
0.6	323.25	357.27	34.02	16.39992	38.70035	61.29965
0.425	340.69	373.89	33.2	16.00463	54.70497	45.29503
0.3	301.07	328.9	27.83	13.41593	68.1209	31.8791
0.15	274.04	307.67	33.63	16.21192	84.33282	15.66718
0.075	264.76	284.81	20.05	9.665445	93.99826	6.001735
0	271.36	283.81	12.45	6.001735	100	
			207.44			

Table A. 0.22: PSD for BH2 @ 6 m

Size of sieve (mm)	Mass of empty sieve (g)	Mass of Sieve +Soil retained (g)	Soil retained (g)	Percent retained	Cumulative retained %	Percent passing
6.7	435.43	439.67	4.24	2.232401	2.232401	97.7676
4.75	428.08	432.1	4.02	2.116569	4.348971	95.65103
2.36	396.68	417.29	20.61	10.85137	15.20034	84.79966
1.18	363.05	411.72	48.67	25.62523	40.82557	59.17443
0.6	323.25	367.88	44.63	23.49813	64.3237	35.6763
0.425	340.69	357.29	16.6	8.740062	73.06376	26.93624
0.3	301.07	312.92	11.85	6.239141	79.3029	20.6971
0.15	274.04	289.39	15.35	8.081925	87.38483	12.61517
0.075	264.76	275.02	10.26	5.40199	92.78682	7.213184
0	271.36	285.06	13.7	7.213184	100	
			189.93			

Table A.0.23: PSD for BH2 @ 7 m

Size of sieve (mm)	Mass of empty sieve (g)	Mass of Sieve +Soil retained (g)	Soil retained (g)	Percent retained	Cumulative retained %	Percent passing
6.7	435.43	439.2	3.77	1.996399	1.996399	98.0036
4.75	428.08	429.27	1.19	0.630163	2.626562	97.37344
2.36	396.68	410.52	13.84	7.328956	9.955518	90.04448
1.18	363.05	419.69	56.64	29.99365	39.94916	60.05084
0.6	323.25	373.69	50.44	26.71044	66.65961	33.34039
0.425	340.69	357.16	16.47	8.721669	75.38128	24.61872
0.3	301.07	312.43	11.36	6.015675	81.39695	18.60305
0.15	274.04	287.73	13.69	7.249523	88.64647	11.35353
0.075	264.76	273.21	8.45	4.474688	93.12116	6.878839
0	271.36	284.35	12.99	6.878839	100	
			188.84			

Table A.0.24: PSD for BH2 @ 8 m

Size of sieve (mm)	Mass of empty sieve (g)	Mass of Sieve +Soil retained (g)	Soil retained (g)	Percent retained	Cumulative retained %	Percent passing
6.7	435.43	444.95	9.52	5.871107	5.871107	94.12889
4.75	428.08	434.77	6.69	4.125809	9.996916	90.00308
2.36	396.68	415.03	18.35	11.31668	21.3136	78.6864
1.18	363.05	392.48	29.43	18.14986	39.46346	60.53654
0.6	323.25	348.26	25.01	15.42399	54.88745	45.11255
0.425	340.69	351.7	11.01	6.790009	61.67746	38.32254
0.3	301.07	311.09	10.02	6.179463	67.85692	32.14308
0.15	274.04	291.44	17.4	10.7308	78.58773	21.41227
0.075	264.76	278.76	14	8.633981	87.22171	12.77829
0	271.36	292.08	20.72	12.77829	100	
			162.15			

Table A.0.25: PSD for BH2 @ 9 m

Size of sieve (mm)	Mass of empty sieve (g)	Mass of Sieve +Soil retained (g)	Soil retained (g)	Percent retained	Cumulative retained %	Percent passing
6.7	435.43	456.87	21.44	11.42187	11.42187	88.57813
4.75	428.08	433.45	5.37	2.860796	14.28267	85.71733
2.36	396.68	416.99	20.31	10.81988	25.10255	74.89745
1.18	363.05	393.92	30.87	16.44558	41.54813	58.45187
0.6	323.25	349.42	26.17	13.94172	55.48985	44.51015
0.425	340.69	354.72	14.03	7.474295	62.96415	37.03585
0.3	301.07	314.84	13.77	7.335784	70.29993	29.70007
0.15	274.04	295.43	21.39	11.39524	81.69517	18.30483
0.075	264.76	278.08	13.32	7.096052	88.79122	11.20878
0	271.36	292.4	21.04	11.20878	100	
			187.71			

Table A.0.26: PSD for BH3 @ surface

Size of sieve (mm)	Mass of empty sieve (g)	Mass of Sieve +Soil retained (g)	Soil retained (g)	Percent retained	Cumulative retained %	Percent passing
6.7	435.43	468.23	32.8	17.53354	17.53354	82.46646
4.75	428.08	428.58	0.5	0.26728	17.80082	82.19918
2.36	396.68	399.79	3.11	1.662479	19.4633	80.5367
1.18	363.05	373.39	10.34	5.527343	24.99065	75.00935
0.6	323.25	338.81	15.56	8.317742	33.30839	66.69161
0.425	340.69	354.09	13.4	7.163094	40.47148	59.52852
0.3	301.07	325.63	24.56	13.12878	53.60026	46.39974
0.15	274.04	318.37	44.33	23.69701	77.29727	22.70273
0.075	264.76	284.63	19.87	10.62169	87.91896	12.08104
0	271.36	293.96	22.6	12.08104	100	
			187.07			

Table A.0.27: PSD for BH3 @ 1 m

Size of sieve (mm)	Mass of empty sieve (g)	Mass of Sieve +Soil retained (g)	Soil retained (g)	Percent retained	Cumulative retained %	Percent passing
6.7	435.43	439.67	4.24	2.232401	0.510045	97.7676
4.75	428.08	432.1	4.02	2.116569	2.626614	95.65103
2.36	396.68	417.29	20.61	10.85137	13.47798	84.79966
1.18	363.05	411.72	48.67	25.62523	39.10321	59.17443
0.6	323.25	367.88	44.63	23.49813	62.60134	35.6763
0.425	340.69	357.29	16.6	8.740062	71.3414	26.93624
0.3	301.07	312.92	11.85	6.239141	77.58054	20.6971
0.15	274.04	289.39	15.35	8.081925	85.66247	12.61517
0.075	264.76	275.02	10.26	5.40199	91.06446	7.213184
0	271.36	285.06	13.7	7.213184	98.27764	
			189.93			

Table A.0.28: PSD for BH3 @ 2 m

Size of sieve (mm)	Mass of empty sieve (g)	Mass of Sieve +Soil retained (g)	Soil retained (g)	Percent retained	Cumulative retained %	Percent passing
6.7	435.43	439.67	4.24	2.232401	2.232401	97.7676
4.75	428.08	432.1	4.02	2.116569	4.348971	95.65103
2.36	396.68	417.29	20.61	10.85137	15.20034	84.79966
1.18	363.05	411.72	48.67	25.62523	40.82557	59.17443
0.6	323.25	367.88	44.63	23.49813	64.3237	35.6763
0.425	340.69	357.29	16.6	8.740062	73.06376	26.93624
0.3	301.07	312.92	11.85	6.239141	79.3029	20.6971
0.15	274.04	289.39	15.35	8.081925	87.38483	12.61517
0.075	264.76	275.02	10.26	5.40199	92.78682	7.213184
0	271.36	285.06	13.7	7.213184	100	
			189.93			

Table A.0.29: PSD for BH3 @ 3 m

Size of sieve (mm)	Mass of empty sieve (g)	Mass of Sieve +Soil retained (g)	Soil retained (g)	Percent retained	Cumulative retained %	Percent passing
6.7	435.43	448.36	12.93	6.837652	6.837652	93.16235
4.75	428.08	433.1	5.02	2.65468	9.492332	90.50767
2.36	396.68	407.71	11.03	5.832893	15.32522	84.67478
1.18	363.05	388.78	25.73	13.60656	28.93178	71.06822
0.6	323.25	354.36	31.11	16.45161	45.3834	54.6166
0.425	340.69	363.86	23.17	12.25278	57.63617	42.36383
0.3	301.07	323.24	22.17	11.72396	69.36013	30.63987
0.15	274.04	300.55	26.51	14.01904	83.37916	16.62084
0.075	264.76	278.05	13.29	7.028027	90.40719	9.592808
0	271.36	289.5	18.14	9.592808	100	
			189.1			

Table A.0.30: PSD for BH3 @ 4 m

Size of sieve (mm)	Mass of empty sieve (g)	Mass of Sieve +Soil retained (g)	Soil retained (g)	Percent retained	Cumulative retained %	Percent passing
6.7	435.43	468.23	32.8	17.53354	17.53354	82.46646
4.75	428.08	428.58	0.5	0.26728	17.80082	82.19918
2.36	396.68	399.79	3.11	1.662479	19.4633	80.5367
1.18	363.05	373.39	10.34	5.527343	24.99065	75.00935
0.6	323.25	338.81	15.56	8.317742	33.30839	66.69161
0.425	340.69	354.09	13.4	7.163094	40.47148	59.52852
0.3	301.07	325.63	24.56	13.12878	53.60026	46.39974
0.15	274.04	318.37	44.33	23.69701	77.29727	22.70273
0.075	264.76	284.63	19.87	10.62169	87.91896	12.08104
0	271.36	293.96	22.6	12.08104	100	
			187.07			

Table A.0.31: PSD for BH3 @ 5 m

Size of sieve (mm)	Mass of empty sieve (g)	Mass of Sieve +Soil retained (g)	Soil retained (g)	Percent retained	Cumulative retained %	Percent passing
6.7	435.43	441.08	5.65	2.723679	2.723679	97.27632
4.75	428.08	440.41	12.33	5.943887	8.667567	91.33243
2.36	396.68	408.38	11.7	5.640185	14.30775	85.69225
1.18	363.05	379.63	16.58	7.992673	22.30042	77.69958
0.6	323.25	357.27	34.02	16.39992	38.70035	61.29965
0.425	340.69	373.89	33.2	16.00463	54.70497	45.29503
0.3	301.07	328.9	27.83	13.41593	68.1209	31.8791
0.15	274.04	307.67	33.63	16.21192	84.33282	15.66718
0.075	264.76	284.81	20.05	9.665445	93.99826	6.001735
0	271.36	283.81	12.45	6.001735	100	
			207.44			

Table A.0.32: PSD for BH3 @ 6 m

Size of sieve (mm)	Mass of empty sieve (g)	Mass of Sieve +Soil retained (g)	Soil retained (g)	Percent retained	Cumulative retained %	Percent passing
6.7	435.43	448.2	12.77	6.566228	6.566228	93.43377
4.75	428.08	439.98	11.9	6.118881	12.68511	87.31489
2.36	396.68	440.94	44.26	22.75812	35.44323	64.55677
1.18	363.05	408.38	45.33	23.30831	58.75154	41.24846
0.6	323.25	348.18	24.93	12.8188	71.57034	28.42966
0.425	340.69	351.01	10.32	5.306458	76.8768	23.1232
0.3	301.07	309.94	8.87	4.56088	81.43768	18.56232
0.15	274.04	287.33	13.29	6.833608	88.27129	11.72871
0.075	264.76	274.23	9.47	4.869395	93.14068	6.859317
0	271.36	284.7	13.34	6.859317	100	
			194.48			

Table A.0.33: PSD for BH3 @ 7 m

Size of sieve (mm)	Mass of empty sieve (g)	Mass of Sieve +Soil retained (g)	Soil retained (g)	Percent retained	Cumulative retained %	Percent passing
6.7	435.43	439.67	4.24	2.232401	2.232401	97.7676
4.75	428.08	432.1	4.02	2.116569	4.348971	95.65103
2.36	396.68	417.29	20.61	10.85137	15.20034	84.79966
1.18	363.05	411.72	48.67	25.62523	40.82557	59.17443
0.6	323.25	367.88	44.63	23.49813	64.3237	35.6763
0.425	340.69	357.29	16.6	8.740062	73.06376	26.93624
0.3	301.07	312.92	11.85	6.239141	79.3029	20.6971
0.15	274.04	289.39	15.35	8.081925	87.38483	12.61517
0.075	264.76	275.02	10.26	5.40199	92.78682	7.213184
0	271.36	285.06	13.7	7.213184	100	
			189.93			

Table A.0.34: PSD for BH3 @ 8 m

Size of sieve (mm)	Mass of empty sieve (g)	Mass of Sieve +Soil retained (g)	Soil retained (g)	Percent retained	Cumulative retained %	Percent passing
6.7	435.43	441.08	5.65	2.723679	1.996399	97.27632
4.75	428.08	440.41	12.33	5.943887	7.940286	91.33243
2.36	396.68	408.38	11.7	5.640185	13.58047	85.69225
1.18	363.05	379.63	16.58	7.992673	21.57314	77.69958
0.6	323.25	357.27	34.02	16.39992	37.97307	61.29965
0.425	340.69	373.89	33.2	16.00463	53.97769	45.29503
0.3	301.07	328.9	27.83	13.41593	67.39362	31.8791
0.15	274.04	307.67	33.63	16.21192	83.60554	15.66718
0.075	264.76	284.81	20.05	9.665445	93.27098	6.001735
0	271.36	283.81	12.45	6.001735	99.27272	
			207.44			

Table A.0.35: PSD for BH3 @ 9 m

Size of sieve (mm)	Mass of empty sieve (g)	Mass of Sieve +Soil retained (g)	Soil retained (g)	Percent retained	Cumulative retained %	Percent passing
6.7	435.43	444.95	9.52	5.871107	5.871107	94.12889
4.75	428.08	434.77	6.69	4.125809	9.996916	90.00308
2.36	396.68	415.03	18.35	11.31668	21.3136	78.6864
1.18	363.05	392.48	29.43	18.14986	39.46346	60.53654
0.6	323.25	348.26	25.01	15.42399	54.88745	45.11255
0.425	340.69	351.7	11.01	6.790009	61.67746	38.32254
0.3	301.07	311.09	10.02	6.179463	67.85692	32.14308
0.15	274.04	291.44	17.4	10.7308	78.58773	21.41227
0.075	264.76	278.76	14	8.633981	87.22171	12.77829
0	271.36	292.08	20.72	12.77829	100	
			162.15			

Table A.0.36: PSD for BH3 @ 10 m

Size of sieve (mm)	Mass of empty sieve (g)	Mass of Sieve +Soil retained (g)	Soil retained (g)	Percent retained	Cumulative retained %	Percent passing
6.7	435.43	436.41	0.98	0.510045	11.42187	99.48996
4.75	428.08	429.22	1.14	0.593317	12.01519	98.89664
2.36	396.68	397.96	1.28	0.666181	12.68137	98.23046
1.18	363.05	368.07	5.02	2.612678	15.29405	95.61778
0.6	323.25	340.72	17.47	9.092329	24.38638	86.52545
0.425	340.69	367.15	26.46	13.77121	38.15759	72.75424
0.3	301.07	336.83	35.76	18.61143	56.76902	54.14281
0.15	274.04	324.33	50.29	26.17362	82.94264	27.96919
0.075	264.76	289.22	24.46	12.7303	95.67294	15.23889
0	271.36	300.64	29.28	15.23889	110.9118	
			192.14			

Table A.0.37: PSD for BH4 @ surface

Size of sieve (mm)	Mass of empty sieve (g)	Mass of Sieve +Soil retained (g)	Soil retained (g)	Percent retained	Cumulative retained %	Percent passing
6.7	435.43	448.2	12.77	6.566228	2.630847	93.43377
4.75	428.08	439.98	11.9	6.118881	4.78892	87.31489
2.36	396.68	440.94	44.26	22.75812	7.586629	64.55677
1.18	363.05	408.38	45.33	23.30831	13.32666	41.24846
0.6	323.25	348.18	24.93	12.8188	27.87141	28.42966
0.425	340.69	351.01	10.32	5.306458	44.35174	23.1232
0.3	301.07	309.94	8.87	4.56088	59.49719	18.56232
0.15	274.04	287.33	13.29	6.833608	78.86979	11.72871
0.075	264.76	274.23	9.47	4.869395	88.68124	6.859317
0	271.36	284.7	13.34	6.859317	100	
			194.48			

Table A.0.38: PSD for BH4 @ 1 m

Size of sieve (mm)	Mass of empty sieve (g)	Mass of Sieve +Soil retained (g)	Soil retained (g)	Percent retained	Cumulative retained %	Percent passing
6.7	435.43	456.87	21.44	11.42187	0.510045	88.57813
4.75	428.08	433.45	5.37	2.860796	3.370841	85.71733
2.36	396.68	416.99	20.31	10.81988	14.19072	74.89745
1.18	363.05	393.92	30.87	16.44558	30.6363	58.45187
0.6	323.25	349.42	26.17	13.94172	44.57802	44.51015
0.425	340.69	354.72	14.03	7.474295	52.05232	37.03585
0.3	301.07	314.84	13.77	7.335784	59.3881	29.70007
0.15	274.04	295.43	21.39	11.39524	70.78334	18.30483
0.075	264.76	278.08	13.32	7.096052	77.87939	11.20878
0	271.36	292.4	21.04	11.20878	89.08817	
			187.71			

Table A.0.39: PSD for BH4 @ 2 m

Size of sieve (mm)	Mass of empty sieve (g)	Mass of Sieve +Soil retained (g)	Soil retained (g)	Percent retained	Cumulative retained %	Percent passing
6.7	435.43	443.18	7.75	6.274288	6.274288	93.72571
4.75	428.08	431.68	3.6	2.914508	9.188795	90.8112
2.36	396.68	401.39	4.71	3.813148	13.00194	86.99806
1.18	363.05	375.16	12.11	9.80408	22.80602	77.19398
0.6	323.25	341.32	18.07	14.62921	37.43523	62.56477
0.425	340.69	354.91	14.22	11.51231	48.94754	51.05246
0.3	301.07	315.18	14.11	11.42325	60.37079	39.62921
0.15	274.04	293.97	19.93	16.13504	76.50583	23.49417
0.075	264.76	276.74	11.98	9.698834	86.20466	13.79534
0	271.36	288.4	17.04	13.79534	100	
			123.52			

Table A.0.40: PSD for BH4 @ 3 m

Size of sieve (mm)	Mass of empty sieve (g)	Mass of Sieve +Soil retained (g)	Soil retained (g)	Percent retained	Cumulative retained %	Percent passing
6.7	435.43	439.67	4.24	2.232401	6.837652	97.7676
4.75	428.08	432.1	4.02	2.116569	8.954221	95.65103
2.36	396.68	417.29	20.61	10.85137	19.80559	84.79966
1.18	363.05	411.72	48.67	25.62523	45.43082	59.17443
0.6	323.25	367.88	44.63	23.49813	68.92895	35.6763
0.425	340.69	357.29	16.6	8.740062	77.66901	26.93624
0.3	301.07	312.92	11.85	6.239141	83.90815	20.6971
0.15	274.04	289.39	15.35	8.081925	91.99008	12.61517
0.075	264.76	275.02	10.26	5.40199	97.39207	7.213184
0	271.36	285.06	13.7	7.213184	104.6053	
			189.93			

Table A.0.41: PSD for BH4 @ 4 m

Size of sieve (mm)	Mass of empty sieve (g)	Mass of Sieve +Soil retained (g)	Soil retained (g)	Percent retained	Cumulative retained %	Percent passing
6.7	435.43	444.95	9.52	5.871107	17.53354	94.12889
4.75	428.08	434.77	6.69	4.125809	21.65935	90.00308
2.36	396.68	415.03	18.35	11.31668	32.97604	78.6864
1.18	363.05	392.48	29.43	18.14986	51.1259	60.53654
0.6	323.25	348.26	25.01	15.42399	66.54989	45.11255
0.425	340.69	351.7	11.01	6.790009	73.3399	38.32254
0.3	301.07	311.09	10.02	6.179463	79.51936	32.14308
0.15	274.04	291.44	17.4	10.7308	90.25016	21.41227
0.075	264.76	278.76	14	8.633981	98.88414	12.77829
0	271.36	292.08	20.72	12.77829	111.6624	
			162.15			

Table A.0.42: PSD for BH4 @ 5 m

Size of sieve (mm)	Mass of empty sieve (g)	Mass of Sieve +Soil retained (g)	Soil retained (g)	Percent retained	Cumulative retained %	Percent passing
6.7	435.43	441.08	5.65	2.723679	2.723679	97.27632
4.75	428.08	440.41	12.33	5.943887	8.667567	91.33243
2.36	396.68	408.38	11.7	5.640185	14.30775	85.69225
1.18	363.05	379.63	16.58	7.992673	22.30042	77.69958
0.6	323.25	357.27	34.02	16.39992	38.70035	61.29965
0.425	340.69	373.89	33.2	16.00463	54.70497	45.29503
0.3	301.07	328.9	27.83	13.41593	68.1209	31.8791
0.15	274.04	307.67	33.63	16.21192	84.33282	15.66718
0.075	264.76	284.81	20.05	9.665445	93.99826	6.001735
0	271.36	283.81	12.45	6.001735	100	
			207.44			

Table A.0.43: PSD for BH4 @ 6 m

Size of sieve (mm)	Mass of empty sieve (g)	Mass of Sieve +Soil retained (g)	Soil retained (g)	Percent retained	Cumulative retained %	Percent passing
6.7	435.43	448.4	12.97	6.643446	6.566228	93.35655
4.75	428.08	439.91	11.83	6.05952	12.62575	87.29703
2.36	396.68	440.99	44.31	22.69631	35.32205	64.60073
1.18	363.05	408.89	45.84	23.48	58.80205	41.12073
0.6	323.25	348.24	24.99	12.80029	71.60234	28.32044
0.425	340.69	351.01	10.32	5.286073	76.88841	23.03437
0.3	301.07	309.94	8.87	4.543359	81.43177	18.49101
0.15	274.04	287.33	13.29	6.807355	88.23913	11.68366
0.075	264.76	274.23	9.47	4.850689	93.08982	6.832966
0	271.36	284.7	13.34	6.832966	99.92278	
			195.23			

Table A.0.44: PSD for BH4 @ 7 m

Size of sieve (mm)	Mass of empty sieve (g)	Mass of Sieve +Soil retained (g)	Soil retained (g)	Percent retained	Cumulative retained %	Percent passing
6.7	435.43	439.67	4.24	2.232401	2.232401	97.7676
4.75	428.08	432.1	4.02	2.116569	4.348971	95.65103
2.36	396.68	417.29	20.61	10.85137	15.20034	84.79966
1.18	363.05	411.72	48.67	25.62523	40.82557	59.17443
0.6	323.25	367.88	44.63	23.49813	64.3237	35.6763
0.425	340.69	357.29	16.6	8.740062	73.06376	26.93624
0.3	301.07	312.92	11.85	6.239141	79.3029	20.6971
0.15	274.04	289.39	15.35	8.081925	87.38483	12.61517
0.075	264.76	275.02	10.26	5.40199	92.78682	7.213184
0	271.36	285.06	13.7	7.213184	100	
			189.93			

Table A.0.45: PSD for BH4 @ 8 m

Size of sieve (mm)	Mass of empty sieve (g)	Mass of Sieve +Soil retained (g)	Soil retained (g)	Percent retained	Cumulative retained %	Percent passing
6.7	435.43	449.2	13.77	8.51735	1.996399	91.48265
4.75	428.08	432.1	4.02	2.486547	4.482946	88.9961
2.36	396.68	400.52	3.84	2.375209	6.858154	86.62089
1.18	363.05	389.69	26.64	16.47801	23.33617	70.14288
0.6	323.25	373.69	50.44	31.19936	54.53552	38.94353
0.425	340.69	357.16	16.47	10.18742	64.72294	28.75611
0.3	301.07	312.43	11.36	7.026659	71.7496	21.72945
0.15	274.04	287.73	13.69	8.467867	80.21747	13.26158
0.075	264.76	273.21	8.45	5.226696	85.44416	8.034886
0	271.36	284.35	12.99	8.034886	93.47905	
			161.67			

Table A.0.46: PSD for BH4 @ 9 m

Size of sieve (mm)	Mass of empty sieve (g)	Mass of Sieve +Soil retained (g)	Soil retained (g)	Percent retained	Cumulative retained %	Percent passing
6.7	435.43	456.89	21.46	11.40943	11.42187	88.59057
4.75	428.08	433.47	5.39	2.865649	14.28752	85.72492
2.36	396.68	417.01	20.33	10.80866	25.09618	74.91626
1.18	363.05	393.94	30.89	16.42299	41.51917	58.49327
0.6	323.25	349.46	26.21	13.93482	55.45399	44.55846
0.425	340.69	354.78	14.09	7.491095	62.94508	37.06736
0.3	301.07	314.87	13.8	7.336913	70.28199	29.73045
0.15	274.04	295.49	21.45	11.40412	81.68611	18.32633
0.075	264.76	278.15	13.39	7.118932	88.80504	11.2074
0	271.36	292.44	21.08	11.2074	100.0124	
			188.09			

Table A.0.47: PSD for BH4 @ 10 m

Size of sieve (mm)	Mass of empty sieve (g)	Mass of Sieve +Soil retained (g)	Soil retained (g)	Percent retained	Cumulative retained %	Percent passing
6.7	435.43	444.95	9.52	5.871107	5.871107	94.12889
4.75	428.08	434.77	6.69	4.125809	9.996916	90.00308
2.36	396.68	415.03	18.35	11.31668	21.3136	78.6864
1.18	363.05	392.48	29.43	18.14986	39.46346	60.53654
0.6	323.25	348.26	25.01	15.42399	54.88745	45.11255
0.425	340.69	351.7	11.01	6.790009	61.67746	38.32254
0.3	301.07	311.09	10.02	6.179463	67.85692	32.14308
0.15	274.04	291.44	17.4	10.7308	78.58773	21.41227
0.075	264.76	278.76	14	8.633981	87.22171	12.77829
0	271.36	292.08	20.72	12.77829	100	
			162.15			

Table A1.0.48: Statistical summary of boreholes cations concentrations

Date sampled	Boreholes	Cations	Mean (mg/l)	Std. Dev	CV (%)
19/02/2018	BH1	Ca	159.339	6.28335	3.94
		K	14.67465	0.23214	1.58
		Mg	525.855	20.1653	3.83
		Na	*		
	BH2	Ca	*		
		K	*		
		Mg	21.7803	1.97889	9.09
		Na	84.45927	3.77932	4.42
	BH3	Ca	*		
		K	*		
		Mg	*		
		Na	169.5616	5.93051	3.50
	BH4	Ca	*		
		K	*		
		Mg	51.56595	0.47256	0.91
		Na	413.7228	33.7406	8.15
21/05/2018	BH1	Ca	*		
		K	28.7135	2.2581	7.86
		Mg	*		
		Na	*		
	BH2	Ca	*		
		K	*		
		Mg	24.4285	1.8301	7.49
		Na	*		
	BH3	Ca	*		
		K	4.22737	0.3896	9.21
		Mg	*		
		Na	206.225	2.6124	1.27
	BH4	Ca	*		
		K	*		
		Mg	94.6246	9.5027	10
		Na	*		
23/06/2018	BH1	Ca	*		
		K	*		
		Mg	*		
		Na	*		
	BH2	Ca	*		
		K	*		
		Mg	15.1959	1.226	8.05
		Na	*		
	BH3	Ca	*		
		K	*		
		Mg	16.2061	0.423	2.61
		Na	83.2951	4.329	5.20
	BH4	Ca	16.2286	1.0358	6.38
		K	*		
		Mg	89.9498	3.1123	3.46
		Na	*		

*-Values obtained were outside the calibration concentration

Table A2.0.49: Statistical summary of boreholes cations concentrations

28/07/2018	BH1	Ca	206.201	6.2271	3.02
		K	29.94	0.6084	2.03
		Mg	*		
		Na	*		
	BH2	Ca	*		
		K	*		
		Mg	*		
		Na	163.319	6.2628	3.83
	BH3	Ca	*		
		K	*		
		Mg	*		
		Na	*		
	BH4	Ca	*		
		K	*		
		Mg	102.031	7.3758	7.23
		Na	*		
04/09/2018	BH1	Ca	229.38	0.91	0.40
		K	*		
		Mg	831	37.839	5.55
		Na	*		
	BH2	Ca	*		
		K	*		
		Mg	25.476	0.7269	2.85
		Na	113.84	8.7312	7.67
	BH3	Ca	*		
		K	*		
		Mg	*		
		Na	*		
	BH4	Ca	*		
		K	*		
		Mg	102.76	5.3409	5.20
		Na	873.58	62.742	7.18
11/10/2018	BH1	Ca	165.13	1.0769	0.65
		K	*		
		Mg	606.69	35.118	5.79
		Na	*		
	BH2	Ca	*		
		K	*		
		Mg	25.896	1.5942	6.16
		Na	132.73	11.969	9.02
	BH3	Ca	*		
		K	*		
		Mg	15.451	0.4537	2.94
		Na	224.53	11.391	5.07
	BH4	Ca	*		
		K	*		
		Mg	*		
		Na	94.52	7.1303	7.54

*- Values obtained were outside the calibration concentration

Table A3.0.50: Statistical summary of boreholes cations concentrations

12/11/2018	BH1	Ca	125.08	8.4079	6.72
		K	20.493	0.9982	4.87
		Mg	671.92	1.212	0.18
		Na	*		
	BH2	Ca	*		
		K	*		
		Mg	9.2605	0.0992	1.07
		Na	85.198	2.8502	3.35
	BH3	Ca	*		
		K	*		
		Mg	11.308	0.2721	2.41
		Na	*		
	BH4	Ca	5.6372	0.2691	4.78
		K	*		
		Mg	11.308	0.2721	2.41
		Na	130.12	7.9654	6.12
17/01/2019	BH1	Ca	155.09	3.3323	2.15
		K	22.216	1.5271	6.9
		Mg	558.83	1.2961	0.2319
		Na	3028.5	103	3.4011
	BH2	Ca	2.0127	0.2031	10
		K	*		
		Mg	13.208		
		Na	54.145	2.3232	4.29
	BH3	Ca	*		
		K	*		
		Mg	17.967	0.7336	4.08
		Na	295.78	3.9647	1.34
	BH4	Ca	5.4659	0.4706	8.61
		K	*		
		Mg	2.1654	0.1211	5.60
		Na	*		

*- Values obtained were outside the calibration concentration