An analysis of discharge and water quality of an urban river and implications for stormwater harvesting

Fahad Aziz

Research Dissertation for the Degree of:

Master of Science

Department of Environmental and Geographical Science

University of Cape Town

October 2020

Supervised by Dr Kevin Winter

DECLARATION

I, Fahad Aziz, acknowledge that,

1. That plagiarism is wrong. Plagiarism is to use another's work and to pretend that it is one's own.

2. I know the meaning of plagiarism and declare that all of the work in the dissertation, save for that which is properly acknowledged, is my own.

3. I have not allowed, and will not allow, anyone to copy my work with the intention of passing it off as his or her own work.

4. I am presenting this dissertation in fulfilment of the requirements for my degree.

Signed: Signed by candidate Date: 23/10/2020

ACKNOWLEDGEMENTS

I am thankful to my supervisor, Dr Kevin Winter, for all his guidance and support throughout the duration of this thesis. I am grateful for all the things I have learnt through him and for patiently giving me input and advice throughout the course of this thesis.

I would like to extend my gratitude to ERM and the Future Water Institute for their financial support in 2019. I would also like to thank the South African Weather Service for providing me with rainfall data. I am grateful to John Weber for the development of the sensors and for helping me with the maintenance of the sensors throughout the data collection period.

Lastly, I would like to thank my friends and family for their support during my studies.

ABSTRACT

Development in urban catchments often result in rivers being converted into large stormwater canals where stormwater is removed as quickly as possible to prevent flooding. A combination of elevated peak flows, increased nutrients and contaminants and reduced biotic richness are typical features of these urban waterways. This study explored the dynamics of an urban river in Cape Town by using high-resolution monitoring sensors and loggers to analyse and model real-time discharge and water quality data during and after 14 rainfall events. Discharge and water quality data were collected from the Liesbeek River at three sites during the rainfall events. As expected, the upper most sampling site had the lowest discharge and pollution load, compared to sites in the middle and lowest reaches of the river. An analysis showed significant correlations between the discharge and electrical conductivity at all three sampling sites. Rainfall was the primary factor in altering discharge and electrical conductivity. Predictive modelling using selected rainfall designs indicated that average discharge and total volume increases with increasing rainfall. Linear regression analysis for electrical conductivity indicated a strong relationship whereby an increase in discharge resulted in a decrease in electrical conductivity. This study revealed the discharge and water quality of stormwater in the Liesbeek River during rainfall events showed the improved water quality conditions in the river during the rainfall events particularly after the peak discharge. Furthermore, the implications of this study revealed that the Liesbeek River can become a water source for recharging groundwater and aquifers.

CONTENTS

LIST OF FIGURES

LIST OF TABLES

1. Introduction

1.1 Overview

In general, urban catchments are changed from their pre-development state by several factors, *inter alia*, rapid population, increasing water demands and increasing impervious surfaces through development (CoCT, 2012; Fisher-Jeffes *et al.*, 2017) resulting in elevated peak flows, increased contaminants from stormwater, decreasing biodiversity and support for habitat with a concomitant decline in ecological services in their waterways. A combination of these factors and the state of urban waterways is described in the concept of an urban stream syndrome (Walsh *et al.*, 2005; Bernhardt and Palmer, 2007; Bratieres *et al.*, 2008; Walsh *et al.*, 2010). Researchers refer to the 'urban stream syndrome' to describe the conditions that alter the flow, form and function of urban rivers (Walsh *et al.* 2005).

Urban streams are treated as conduits that are designed to protect residential areas from floods, but most often results in unintended consequences that reduce ecological functions and services among others (Walsh *et al.*, 2005). While modified urban rivers and streams become efficient stormwater canals (Bernhardt and Palmer, 2007), they have a reduced capacity for removing pollutants during low flow periods resulting in a general decline in water (Walsh *et al.*, 2005; Bratieres *et al.*, 2008). The degradation of urban streams occurred despite the fact that urban streams have the potential to provide precious natural resources through stormwater harvesting, however, this potential is far from fully realized because drainage systems turn rivers into drains or sewers treating it as a waste product (Walsh *et al.*, 2005). The abrasive use of the urban stream syndrome as a concept that could apply to a terminal condition has led to a shift in research and practice in motivating for a water sensitive approach for regenerating cities that are increasingly vulnerable to water scarcity, climate change and rapid urbanization.

The development of an urban area within catchments has caused major changes in the hydrology of catchments, particularly during rainfall events (Huang *et al.*, 2008). Some of these changes include increased high flows and reduced low flows, making urban waterways vulnerable to flooding and droughts. These changes are exacerbated by climate change. Climate change will have a significant effect on the hydrological cycle and will result in floods and droughts becoming more frequent (IPCC, 2007; van Vliet and Zwolsman, 2008). Cities and urban areas are facing increasing demands for water due to rapid population growth and economic development (Fisher-Jeffes *et al.*, 2017). This is coupled with pollution of freshwater bodies which add further stress on water resources. Urban areas will have to adapt to a reduced rainfall future and diversify their water supply from alternative sources such as aquifers and stormwater (Ziervogel *et al.*, 2010; Fisher-Jeffes *et al.*, 2017; van Mazijk *et al.*, 2018).

Increasing water demands in urban catchments has prompted the need for better management of water resources. The potential benefits of stormwater harvesting have not been fully realised (Walsh *et al.* 2005). Stormwater harvesting has the potential for stormwater to be used as a resource to help alleviate the increasing demand for water in urban areas (Fisher-Jeffes *et al.*, 2017). Harvested stormwater is used for various purposes including non-potable water uses and for recharging groundwater. Stormwater harvesting also has a number of benefits in that it can improve water security, prevent or reduce flooding and improve the condition of urban waterways through flow modulation (Woods-Ballard *et al.*, 2007; Huang *et al.*, 2008; Armitage *et al.*, 2013; Fisher-Jeffes, 2015, Fisher-Jeffes *et al.*, 2017). The utilisation of stormwater requires effective stormwater management. Stormwater management has the potential to serve the purpose of increasing water supply together with improving water quality and protecting the ecological diversity of urban rivers (Wong and Eadie, 2000).

This study examines the potential to augment groundwater supplies and aquifer recharge from stormwater harvesting to alleviate the increasing demand for water in water stressed cities. This research analyses the discharge of an urban river and associated water quality fluxes during rainfall events to determine the suitable conditions for stormwater harvesting by simultaneously measuring flow rate and the volume of stormwater and water quality during rainfall events using sensors and loggers that generate high-resolution continuous data. Monitoring discharge together with water quality demonstrates the relationship between urban rivers, its discharge and quality and ultimately will help inform stormwater management and harvesting.

1.2 Aims and objectives

The aim of this thesis is to analyse the discharge of an urban river and associated water quality fluxes during rainfall events to determine the suitable conditions for stormwater harvesting.

Objectives:

- To measure flow rate and quantify the volume of stormwater in an urban river during rainfall events
- To analyse water quality during rainfall events
- To establish the relationships between rainfall characteristics and discharge; the relationship between rainfall and water quality; and the relationship between discharge and water quality
- To model the discharge in an urban river under various conditions

1.3 Study site

The Salt River catchment is largely an urban catchment in Cape Town that experiences a Mediterranean style climate with wet winters and dry summers. The Liesbeek is a sub-catchment of the Salt River catchment (Figure 1) covering an area of 2600 hectares and abuts the eastern slopes of Table Mountain before it flows into the confluence of the Black River that ultimately discharges into Table Bay in close proximity to the Cape Town Harbour (Brown and Maqoba, 2009). The Liesbeek River is 9km long and is fed by numerous streams from the eastern slopes of Table Mountain. The river flows in a north westerly direction and passes through residential areas with varying population densities (Fisher-Jeffes, 2015). It begins in the foothills of Table Mountain at an elevation of 120masl and is fed by numerous streams from the eastern slopes and springs. The Liesbeek River today still shows some of its natural beauty from its pre-development days although 70% of the entire length of the river is canalised or modified by gabions and other forms of grey infrastructure. The upper reaches of the Liesbeek Catchment are largely natural areas whilst the lower reaches of the river have the highest levels of urban land use.

Development of residential areas began in the 18th century. The construction of the main road between Cape Town and Wynberg in 1807-1811 and the railway connection to Wynberg in 1864 was considered the catalyst towards increasing the residential settlements along or close to the Liesbeek River. Due to the rainfall experienced in the upper reaches of the Liesbeek Catchment and the fact that the gradient of the river smoothens out dramatically in the middle and lower sections, it resulted in the catchment being a flood risk hazard. In the beginning of the 1900s when urbanization rapidly increased, buildings close to or along the river became flooded during the rainy season. It was due to this that the river became canalized between 1942 and 1962. The canalization was to improve the conveyance of water downstream (Brown and Maqoba, 2009).

The Liesbeek Catchment was chosen for this study for several reasons including its Mediterranean climate which allowed for the monitoring of discharge and water quality during the rainfall season. A number of studies has been done on the Liesbeek by students and researchers that created a baseline of data to refer to for this study, in particular, the study from Fisher-Jeffes (2015) which examined the storm and rainwater harvesting potential of the catchment using predictive modelling.

Figure 1: Catchments in the Cape Metropolitan Area, showing the Salt River catchment (3) in relation to the Liesbeek River and the other catchments in Cape Metropolitan Area (Ogutu, 2007).

1.4 Study design and overview of methods

This study aimed to characterize stormwater quality and discharge, and model surface water quality and discharge in an urban river. The study design incorporated two research activities, which examined the surface water quality and discharge to use predictive modelling for selected rainfall scenario events. The monitoring of discharge and water quality took place along the Liesbeek River. The purpose of using a predictive model was to understand how rainfall events, under different scenarios, could affect the flow and water quality. Data were captured at three sites along the river.

Level sensors were placed under bridges along the river to measure the height of the water. For this study, height was measured as a proxy for discharge which is described in greater detail in section 3.3. Water quality sensors were placed in the river to measure pH, DO, EC and temperature which were indicators of water quality that were measured at five-minute intervals during rainfall events. The results were used to understand the relationship between discharge, water quality and rainfall. The purpose of monitoring of discharge and water quality is to understand how the Liesbeek River could become a water source for recharging the groundwater and aquifer by understanding the condition of the water during a rainfall event in terms of its water quality as well as the volume of water that can be abstracted.

The sensors were located upstream (Site 1), midstream (Site 2) and downstream (Site 3). Each site contained one level sensor and one water quality sensor. The location of the sensors was strategically chosen in order to capture the spatial and temporal variability of rainfall in the Liesbeek Catchment. The reason for measuring discharge and water quality in tandem at each site was to create simultaneous readings in order to understand how quality and discharge varies with rainfall and to quantify the stormwater in the river during rainfall events and its associated water quality fluxes. Quantifying and understanding the stormwater in the river during rainfall events will enable the use of the stormwater as a water source for recharging the groundwater and aquifer. Details of the level and water quality sensors are discussed in sections 3.3 and 3.4 respectively. The discharge in the Liesbeek River was undertaken from April 2018 to September 2018 and then again from May to July 2019, whilst data collection for water quality took place from March to July 2019.

1.5 Scope and limitations of study

The study took place in the Liesbeek Catchment which consisted of mostly middle to high income suburbs with no industrial areas, therefore this catchment is not necessarily representative of other urban catchments. This study monitored the discharge and water quality of the Liesbeek River using low cost, high-resolution ultrasonic level sensors and water quality sensors that were placed in and along the Liesbeek River. The study period included two rainfall seasons and were restricted to rainfall events with no data collection during the dry seasons ruling out the comparison between dry and wet periods.

Temperature, Electrical Conductivity (EC), pH and Dissolved Oxygen (DO) were monitored as a proxy for water quality and therefore a limited number of water quality parameters were selected to analyse water quality. The exclusion of chemical and biological water quality parameters such as ammonia and E. coli respectively, limited the extent to fully investigate water quality, however the study investigated how rainfall affects dilution and therefore the selected parameters were deemed sufficient for its purpose.

2. Literature Review

2.1 Urban stream syndrome

Modern infrastructure and rapid urbanization have reduced the size and scale of floodplains and elevated the volume of stormwater runoff that is discharged into urban rivers (Walsh *et al.*, 2005; Bernhardt and Palmer, 2007). These characteristics are cited in the research literature as causal factors that contribute to the urban stream syndrome that feature flashier hydrographs, elevated concentrations of nutrients and contaminants, morphological changes in river channels, reduced biotic richness and the dominance of species that that are tolerant of pollution (Walsh *et al.*, 2005).

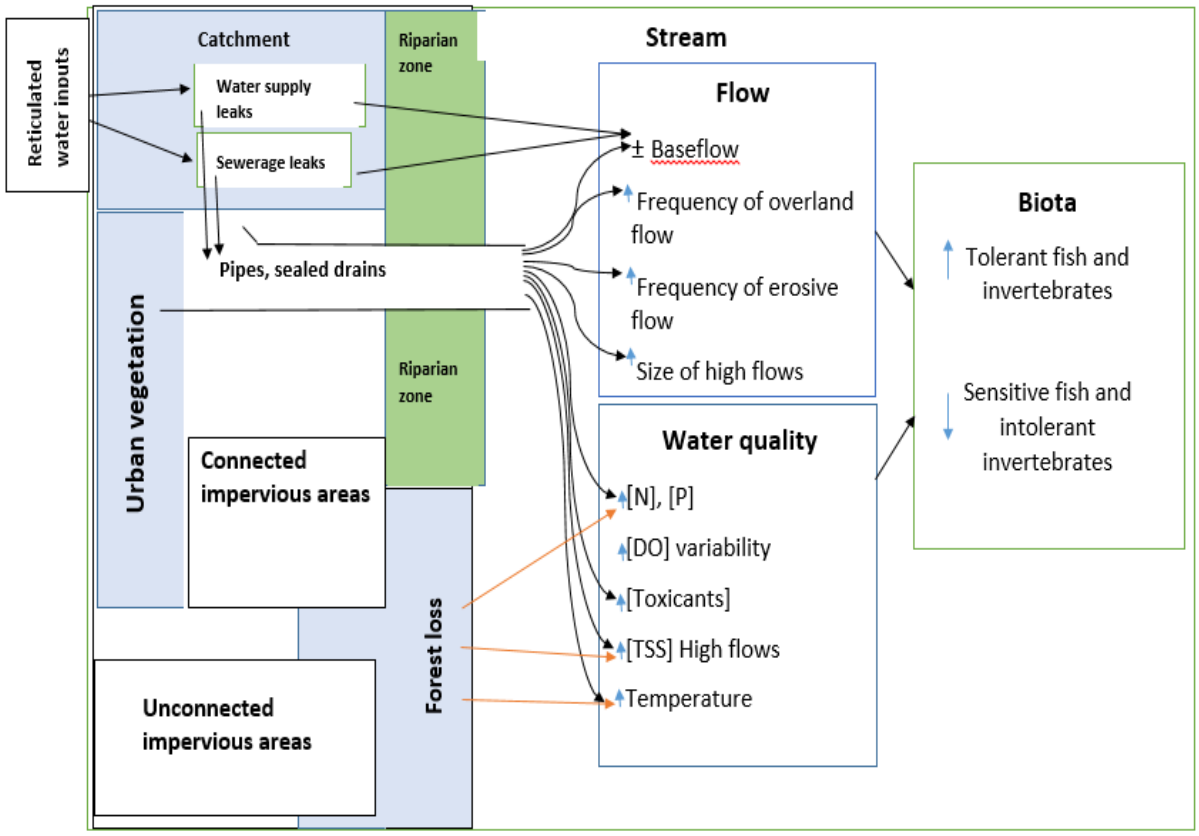

Figure 2: Impacts on the flow and quality of streams in an urban landscape (Adapted from Walsh *et al.*, 2005).

The illustration above shows the different pathways of surface water flow in an urban area. Some water seeps underground from infiltration, but the larger volume is transferred by overland flow and is discharged directly into the river. The resulting effects are shown in Figure 2 which includes the changes in water quality and flow. These changes results in an increase in tolerant species and a decrease in sensitive species. The changes in flow and water quality are further discussed in greater detail below.

2.2 Changes in the hydrology of urban rivers

2.2.1 Introduction

The development of an urban area within catchments causes changes in the hydrology of the catchment, particularly during rainfall events (Huang *et al.*, 2008). In a natural catchment, rainfall infiltrates into the ground where some is taken up by vegetation and eventually transpires whilst the remainder seeps through the soils and into the groundwater where it eventually reaches nearby streams. The water cycle has however, changed due to urban development (Fletcher and Deletic,

2008). The increase in impervious surfaces and simplification of drainage systems for efficient removal of urban runoff for flood protection has caused a rapid runoff in response to rainfall (Fletcher *et al.*, 2013). Urban run-off has the largest impact on flow regimes of urban streams (Walsh *et al.*, 2012).

2.2.2 Land cover changes on the flow regime of urban rivers

The flow regime is determined by a combination of factors such as its magnitude, duration and frequency (Nilsson and Malm-Renöfält, 2008). Changes in land cover affects the overall balance of water in the hydrological system (Ahn and Merwade, 2017). Impervious surfaces and conventional drainage systems are some drivers that impact the hydrology of urban catchments. Urban streams tend to have flashier hydrographs with frequent, larger flow events with increased flow rates, leading to greater channel incision and bank erosion. The increased scouring and changes in sediment supply can alter the width and depth of the river, often making the river wider and deeper with reduced biotic richness (Walsh *et al.*, 2005; Vietz *et al.*, 2015). A reduction in evapotranspiration and infiltration results in an increase in peak discharge and reduced groundwater recharge from impervious surfaces and the removal of vegetation. The decrease in groundwater effectively means that the baseflow is reduced as well (Fletcher *et al.*, 2013). Reduced infiltration decreases the volume of dry-weather flows and increases the volume of wet-weather flows (Walsh et al., 2010).

Previous studies found that land use changes such as urban infrastructure, had a significant effect on the hydrology of catchments. It was noted that built up land replacing vegetation cover can affect the duration and severity of high and low flows, creating higher high flows due to increased run off and lower low flows due to decreased baseflow (Aichele, 2005; White and Greer, 2006; Ahn and Merwade, 2017). This concurs with Walsh *et al.* (2012) that noted that urban catchments had lower baseflows than natural catchments. Furthermore, it was noted that in a forested catchment, water that reaches the river is usually from subsurface flows such as groundwater (Midgley *et al.*, 2001; Ahn and Merwade, 2017). In urban areas however, where efficient conventional drainage systems are in place, streamflow in rivers are mostly from runoff produced by storm events (Walsh *et al.*, 2010; Burns *et al.*, 2012).

Increased impervious areas has increased the runoff volume and peak discharge in urban rivers. It has also decreased the lag time (Hood *et al.*, 2007) by reducing the time it takes for the river to reach its peak flow (Roa and Delleur, 1974). Previous studies have noted that urban catchments produce larger peak flows than natural catchments (Stall and Smith, 1961; Espey *et al.*, 1966; Hood *et al.*, 2007; Huang *et al.*, 2008). Urban land use, stormwater pipes and the substitution of subsurface flow for overland flow has resulted in increased volume and flow rate of stormwater that is being transported in the catchment to the nearest water body. This ultimately causes an earlier and elevated peak in the hydrograph (Figure 3) with a shorter lag time, a familiar problem in urban stormwater management (Hood *et al.*, 2007; Huang *et al.*, 2008).

The lag time of urban hydrographs decreases as the imperviousness of the area increases (Huang *et al.*, 2008), however, Hood *et al.* (2007) showed that there is not much difference in lag times between conventional developments and Low Impact Developments (LID) when storm intensities are increased to durations longer than 4 hours and larger than 25mm (Hood *et al.*, 2007). This shows that whilst imperviousness and basin area influence peak discharge, the magnitude and duration of rainfall plays a role in it as well (Roa and Delleur, 1974). In summary, the peak discharge and lag time depends primarily on two basin characteristics; the area of the basin and the fraction of imperviousness. The peak discharge and lag time also depend on two storm characteristics; the

amount of rainfall and the rainfall duration (Roa and Delleur, 1974). Figure 3 also summarises the changes in flow due to increased development. In Figure 3, the urban hydrograph shows steep rising limbs, elevated peak flows and a shorter lag time. The pre-urban hydrograph has a lower peak flow and has continued subsurface and baseflow.

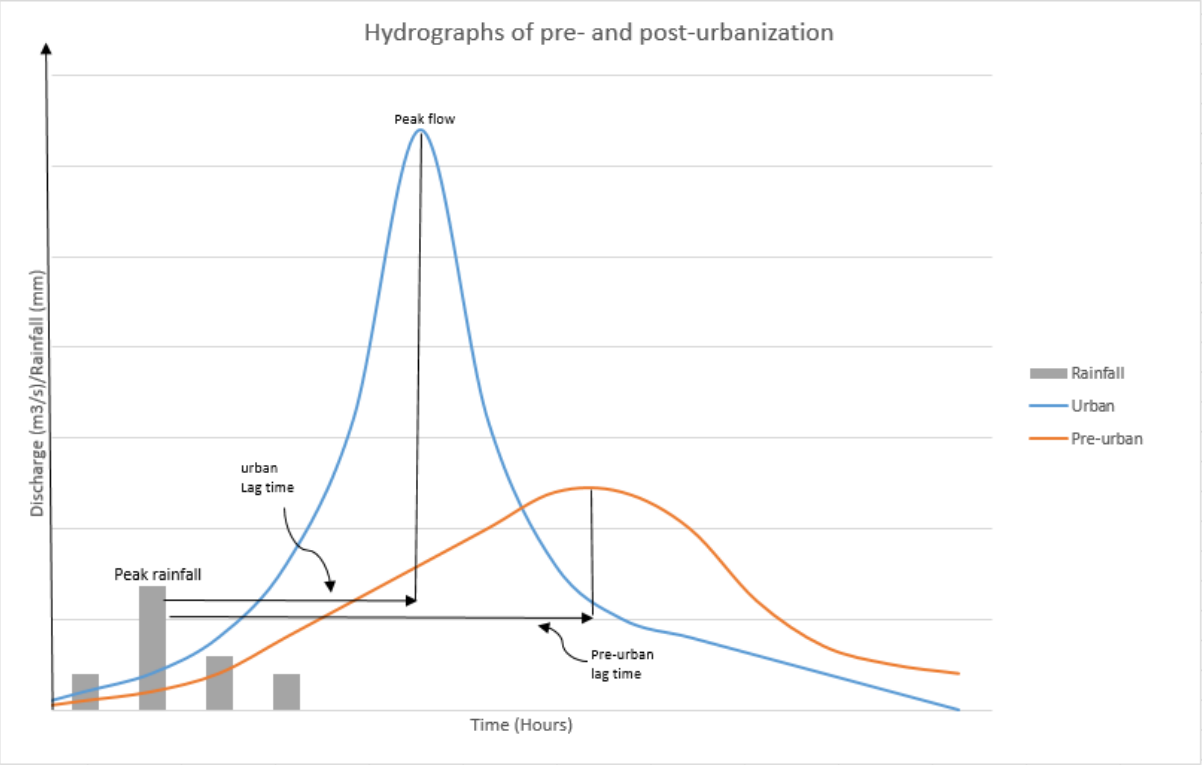

Figure 3: Hydrographs showing flow conditions before and after urban development has taken place (Adapted from Armitage *et al.*, 2013).

Table 1 summarises sections 2.1 and 2.2 showing the changes in the hydrology of urban water bodies due to development and its associated impacts, processes and effects. The table also reiterates the changes in flow and water quality due to urban development, altering the flow, form and function of urban waterways which Walsh *et al.* (2005) alluded to. Urban rivers have become modified which has compromised the function of rivers in supporting habitats and providing ecosystem services. These impacts are important to understand in dealing with stormwater harvesting and its associated challenges.

Table 1: Adapted from Woods-Ballast (2007), summarising the impacts, processes and effects of developments in catchments on receiving water bodies

	Impacts	Processes	Effects
Changes to stream flow	• Reduced infiltration and evapotranspiration • Rapid removal of surface water (Fletcher *et al.*, 2013; Walsh *et al.*, 2005; Woods-Ballard *et al.*, 2007)	• Increased volume of run off, peak flows and flow rates • Increased downstream flooding • Reduced baseflows (Fletcher *et al.*, 2013; Walsh *et al.*, 2005; Woods-Ballard *et al.*, 2007)	• The volume of discharge from a developed catchment that does not attenuate the peak flows can be far greater than natural catchments (Woods-Ballard *et al.*, 2007).
Changes to stream morphology	• Increased erosion • Increased flow rates and flooding • Increased sediment deposition downstream (Vietz *et al.*, 2015; Woods-Ballard *et al.*, 2007)	• Increased channel width • Stream erosion • Changes in channel bed profile (Vietz *et al.*, 2015)	• Channels widen to accommodate increased run off • Higher flow rates will cause more scouring and erosion of banks and cause of a build-up of sandbars downstream (Vietz *et al.*, 2015; Woods-Ballard *et al.*, 2007)
Water quality impacts	• Decomposition of organic matter present in run off • Wash-off of fertilizers, litter, sewer overflows, oil spills, vehicles, household detergents, septic tanks seepages and landfills. (Walsh *et al.*, 2005; Woods-Ballard *et al.*, 2007)	• Reduction of dissolved oxygen in water bodies. • Nutrient rich waters • Increased levels of toxic materials • Increased sediment loads. • Raised temperature levels in water bodies • Increased algal growth (Walsh *et al.*, 2005; Woods-Ballard *et al.*, 2007)	• Reduced dissolved oxygen can kill off fish species • Increased nutrients can result in eutrophication • Water may become toxic and pose a health hazard • Extra costs for the treatment of contaminated water • Polluted waters are aesthetically unattractive (Walsh *et al.*, 2005; Woods-Ballard *et al.*, 2007)

2.2.3 Effect of land cover change on the surface water quality of streams

Water quality is naturally variable over time and space and varies along the course of a river (Nilsson and Malm-Renöfält, 2008). Runoff from urban areas is one of the leading sources of water quality degradation (Hatt *et al.*, 2004) because urban runoff is often polluted (Davis *et al.*, 2001) due to human activities and untreated sewage water (Daniel *et al.*, 2002; Liyanage and Yamada, 2017) which impacts negatively on receiving water bodies. Urban development and conventional drainage system cause an increase in imperviousness and greater hydraulic efficiency causing a quick removal of stormwater from urban areas (Hatt *et al.*, 2004). In urban areas, ecosystems are degraded due to increased volume, intensity and flow rate of runoff that bypasses floodplains and enters directly into streams leading to stream bank erosion and an increase in pollutants (Walsh *et al.*, 2005; Bratieres *et al.*, 2008; Fletcher *et al.*, 2013; Vietz *et al.*, 2015). According to studies by Walsh *et al.* (2005) and Davis *et al.* (2001) a change in water quality was due to high concentrations of nutrients and heavy metals which resulted in oxygen depletion in rivers (Walsh *et al.*, 2005; Davis *et al.*, 2001).

A study by Hatt *et al.* (2004) which sampled water quality variables in fifteen small streams draining independent sub-basins east of Melbourne, Australia, noted that EC and temperature all increased with increased imperviousness. EC in streams and urban density had a strong positive correlation. The salinity in streams in urban areas can be attributed to a combination of sources, *inter alia*, atmospheric deposition, building materials and highways. An increase in imperviousness results in elevated temperatures due to the runoff that is in contact with artificial surfaces (Young *et al.*, 2013). In Brazil, sewage water goes untreated and is discharged into receiving rivers. In the Piracicaba River basin in São Paulo, Brazil, only 16% of their urban sewage and industrial effluents are treated. Most of the urban sewage and industrial effluents are dumped into small streams, rendering it to contamination and lowering the DO to near zero values during low flows (Ometto *et al.*, 2000 and Daniel *et al.*, 2002). Daniel *et al.* (2002) performed a study on the impact of urban sewage discharge into small streams in the Piracicaba River basin (Table 2) looking specifically at water quality parameters which included, *inter alia*, DO and EC. It was noted that the Quilombo and Enxofre streams which drained the most developed catchments were of a poorer water quality than the two streams that were in the least developed catchments. The more developed catchment had lower DO and higher EC values in their streams than the less developed catchments. Selected stream results are summarized in the table below (Daniel *et al.*, 2002).

Table 2: DO and EC results for streams under different urban development percentages (Daniel *et al.*, 2002)

Stream	Built-up land (%)	DO (mg/L)	EC (µS/cm)
Cabras	1	9.1 ± 0.6	53.5 ± 5.6
Guamium	3	6.3 ± 2.4	151.2 ± 44.9
Quilombo	31	2.4 ± 1.5	294.7 ± 107.1
Enxofre	50	1.8 ± 1.6	597.5 ± 159.3

2.3 Relationship between rainfall, discharge and water quality

Urban rivers are known to rapidly increase in discharge following a rainfall event and causes various changes in water quality (van Mazijk *et al.*, 2018). Previous studies have documented the change in the discharge in rivers before and during a rainfall event with its associated water quality fluxes (Jarvie *et al.*, 2001; Li *et al.*, 2007; Sabater and Tockner, 2009; van Mazijk *et al.*, 2018).

Urban stormwater discharge, particularly during rainfall events, is a major contributor to pollution of receiving water bodies (Davis *et al.*, 2001; Lee *et al.*, 2004; Walsh *et al.*, 2005; Bratieres *et al.*, 2008;

Nilsson and Malm-Renöfält, 2008). This is often the case when a rainfall event occurs after a dry period. The existence of a dry period allows for the build-up of pollutants which eventually becomes incorporated into the runoff when the rainfall event occurs. This is sometimes counteracted by the dilution effect of rainfall (Nilsson and Malm-Renöfält, 2008). A study by Lee *et al.* (2004) analysed the stormwater discharge in California during successive wet seasons in which each was preceded with a long dry summer. The results indicated that pollution concentration in the first part of the wet season were the highest at 1.2 to 20 times higher than the pollutant concentrations towards the end of the wet season (Lee *et al.*, 2004). It was also noted that a rainfall event with a longer antecedent dry period was more likely to produce higher pollution loads in the discharge during the initial rainfall period (Li *et al.*, 2007; Yufen *et al.*, 2008).

The initial period of a rainfall event often has the highest levels of pollutants that accompanies the first flush phenomenon (Lee *et al.*, 2004). The first flush phenomenon is defined as the initial period of stormwater runoff during which the concentrations of pollutants are far higher than later periods of stormwater runoff (Gupta and Saul, 1996). Various factors influence the first flush strength including rainfall intensity, storm duration, number of dry days preceding rainfall, percentage of imperviousness and catchment area (Gupta and Saul., 1996; Lee *et al*, 2002 and 2004; Li *et al.*, 2007). Furthermore, it was noted that the first flush phenomenon occurs before the peak flow during a rainfall event. A study by Li *et al.* (2007) which took place in an urban catchment in the city of Wuhan, China, examined the first flush phenomenon. The results showed that elevated concentration of pollutants were observed prior to flow peaks in all rainfall events during the study and that the interval between the concentration peak and the peak flow was shorter for the events that had higher intensities during the initial rainfall period. Figure 4 indicates that the peak concentrations occurred in the initial rainfall period before the peak runoff, indicating a rapidly improving water quality following the peak runoff (Li *et al.*, 2007).

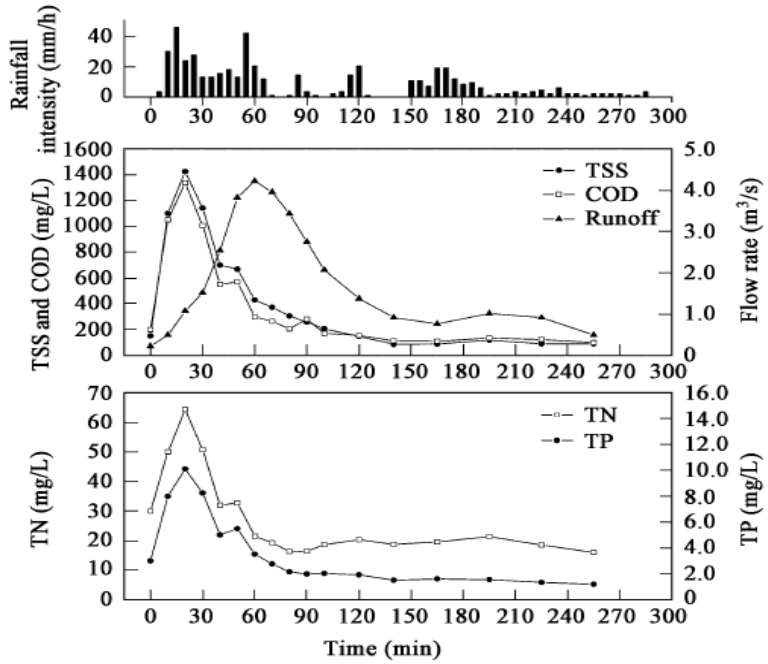

Figure 4: Flow rate and concentration of pollutants for a storm event (41.3mm rainfall) (Li *et al.*, 2007).

The first flush phenomenon was further supported by Bertrand-Krajewski *et al.* (1998), Jarvie *et al.* (2001), Yufen *et al.* (2008) and McGrane *et al.* (2017). All four studies indicated that water quality

decreases with an initial increase in discharge which is then followed by an increase in water quality due to the flushing and dilution of contaminants as the volume of rainfall increases (Jarvie et al., 2001; McGrane *et al.*, 2017).

The study by Jarvie *et al.* (2001) measured flow, pH and conductivity at 15-minute intervals in the River Dee, Scotland. The study showed the changes in water quality during rainfall events whereby pH experienced a decline during rainfall events followed by a more prolonged recovery period following the peak flow (Figure 5). Conductivity however, did not react to changes in the flow as abruptly as pH. Conductivity showed an increase during the initial period of increased stream flow (Jarvie *et al.*, 2001). This was followed by a decrease in conductivity as the stream flow continued to increase. In a study performed by McGrane *et al.* (2017) that monitored catchments with various attributes, it was noted that DO decreased moderately during the rising limb of a flow hydrograph but increased during peak flows. It was also noted that observed EC showed a pre-event peak which was followed by a dilution during the peak flow (McGrane *et al.*, 2017).

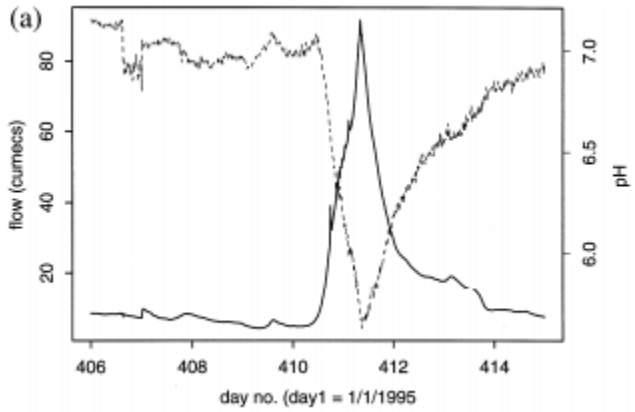

Figure 5: Tendency for pH to become more acidic during a rainfall event (Jarvie *et al.*, 2001).

Various studies such as Betrand-Krajewski *et al.* (1998) and Li *et al.* (2007) have attempted to quantify the first flush phenomenon. The study by Bertrand-Krajewski *et al.* (1998) (Figure 6) analysed 197 rainfall events in 12 separate and combined sewer systems. The results indicated that in combined sewer systems, 80% of the total pollutant mass is transported in the first 79% of the total volume for 50% of the rainfall events. In separate sewer systems, 80% of the total pollutant mass is transported in the first 74% of the total volume for 50% of the rainfall events (Bertrand-Krajewski *et al.*, 1998). According to Li *et al.* (2007) the first 30% of run off volume is highly polluted. The fractions of pollution load transported by the first 30% of run off volume in an urban catchment located in China were 62.4% of TSS, 59.4% of chemical oxygen demand, 46.8% of total nitrogen, and 54.1% of total phosphorus.

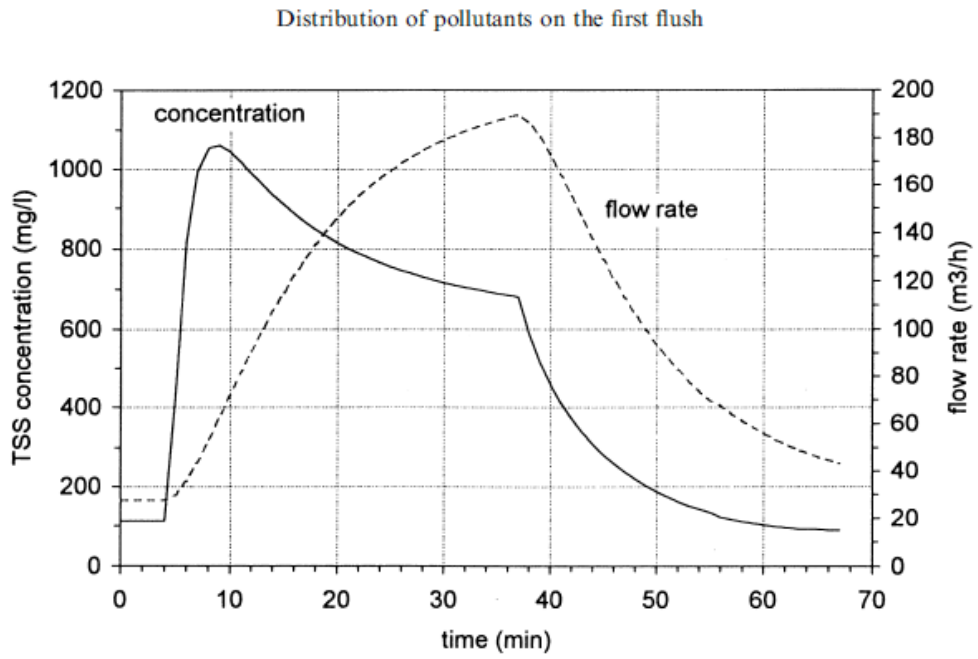

Figure 6: Flow rate and Total Suspended Solids (TSS) respectively over a period of 70-minutes (Bertrand-Krajewski *et al.*, 1998).

This section described the relationships between rainfall, flow and water quality. The studies suggested that after the initial runoff volume created from a rainfall event (approximately the first 30% of volume) the pollutant concentrations are reduced by almost half, showing improved water quality thereafter. Monitoring discharge and water quality at a fine granularity will provide a good indication on how water quality and discharge varies with rainfall to help inform decision making in stormwater harvesting.

2.4 Monitoring of discharge and water quality

2.4.1 River discharge monitoring

Discharge of rivers are one of the most accurately measured aspects of the hydrological cycle. The collection and distribution of the data are however limited (Fekete and Vörösmarty, 2007) particularly in developing countries (Fekete *et al.*, 2012).

River discharge data forms the basis of the Global Runoff Data Centres (GRDC) data archives which was established in 1988 and operates under the World Meteorological Organization (WMO) and is regarded as the most complete global discharge data set. According to the data archive of the GRDC, operating discharge monitoring stations are in rapid decline (Figure 7) (Fekete and Vörösmarty, 2007). Hannah *et al.* (2011) pointed out that monitoring infrastructure are relatively in place in developed countries (Hannah *et al.*, 2011) with sparse gauging stations in developing countries (Vörösmarty *et al.*, 2000). There are large gaps in discharge measurements in Saharan Africa due to insufficient monitoring infrastructure. It was also noted that the number of monitoring stations reached its peak in the 1980s which coincided with increased concerns surrounding population growth and environmental degradation, but soon after, as the focus became more concentrated on climate change, the monitoring of *in situ* networks declined (Hannah *et al.*, 2011; Fekete *et al.*, 2012). South Africa is no different with the number of flow gauges and rainfall stations in decline (Pitman, 2011). Runoff and discharge of South African rivers are measured using flow gauging stations. These gauging stations consist of compound gauging weirs. The stations are however,

restricted to lower flows and become inaccurate when high discharges occur because they are not built large enough to manage high flows. This creates a gap in South Africa's river discharge data with inaccurate to no discharge data during high flows (Meyer *et al.*, 2000).

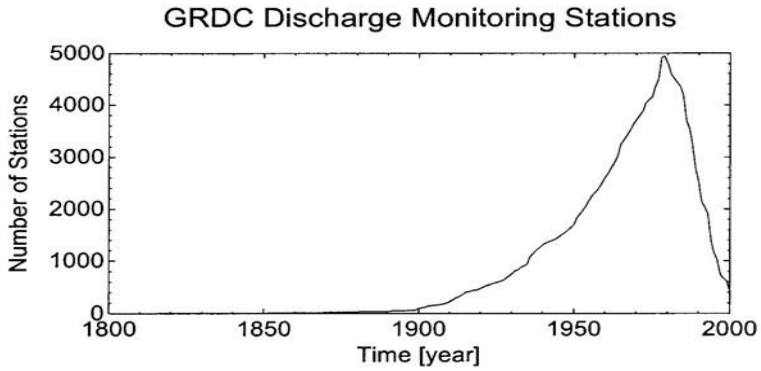

Figure 7: Number of operating discharge monitoring stations according to the GRDC data archive (Fekete and Vörösmarty, 2007).

2.4.2 Water quality monitoring

Deterioration of water quality has prompted the need for water quality monitoring systems all over the world (Dong *et al.*, 2015). The deterioration of water quality through multiple stresses such as, population growth, urbanization and increased pollution has resulted in the need for effective and efficient water quality monitoring programs (Ijaradar and Chatterjee, 2018).

Previously, water quality data was largely restricted to field measurements in the form of on-site grab samples which were costly and labour intensive. The data was then analysed off-site and in doing so was time consuming and slows down the process of disseminating important information (Ijaradar and Chatterjee, 2018). Due to the time consuming nature of analysing hand grab samples of water quality, it has been typically restricted to individual storm events (Kirchner *et al.*, 2004). The laboratory analyses of single samples only provide a snapshot of the concentration levels of the pollutants at the time of sampling. This does not provide adequate information for water quality variables that vary over time and an episodic pollution event could be missed (Vrana *et al.*, 2005). This technique of measuring water quality is limited to the temporal and spatial scales which hinder continuous monitoring and analyses. According to O'Flynn *et al.* (2010) it is unlikely that on-site field grab samples can provide an accurate estimate of the maximum, mean and minimum values of different water quality variables nor can it give accurate trends and fluctuations with temporal variability (O'Flynn *et al.*, 2010).

2.4.3 Comparisons between grab sampling and continuous monitoring

The importance of maintaining water quality standards increased the need for advanced equipment to monitor and manage water resources (O'Flynn *et al.*, 2007). The realization of the shortcomings of traditional river discharge and water quality measurements has resulted in more continuous data collection by *in situ* sensors. River discharge and water quality data is important and the monitoring of hydrological data that is in real-time, continuous and at a fine granularity underpins effective management of rivers as it increases the accuracy of the data which can effectively capture hydrological variability at high-resolutions (Kawanisi *et al.*, 2010). This allows for constant surveillance of biological and physiochemical parameters in water quality in which the data can be accessed remotely in real-time.

A study by O'Flynn *et al.* (2010) indicated the difference between on-site grab samples and continuous monitoring and showed the amount of detail and information that is obtained with continuous monitoring as opposed to on-site grab samples. O'Flynn *et al.* (2010) showed that on-site grab sampling does not capture the diurnal signal that is present in pH. The diurnal signal was only picked up with the 10-minute interval sampling. Furthermore, Figure 8 illustrates the dramatic loss of information that occurs at lower sampling frequencies. Much like what O'Flynn *et al.* (2010) illustrated, Kirchner *et al.* (2004) illustrated that monthly, weekly and daily measurements conceal the rapid response of conductivity to increased flow rates. It also shows conductivity's weak response to flow rate initially, followed by a stronger response to flow rate after day 390 (Kirchner *et al.*, 2004).

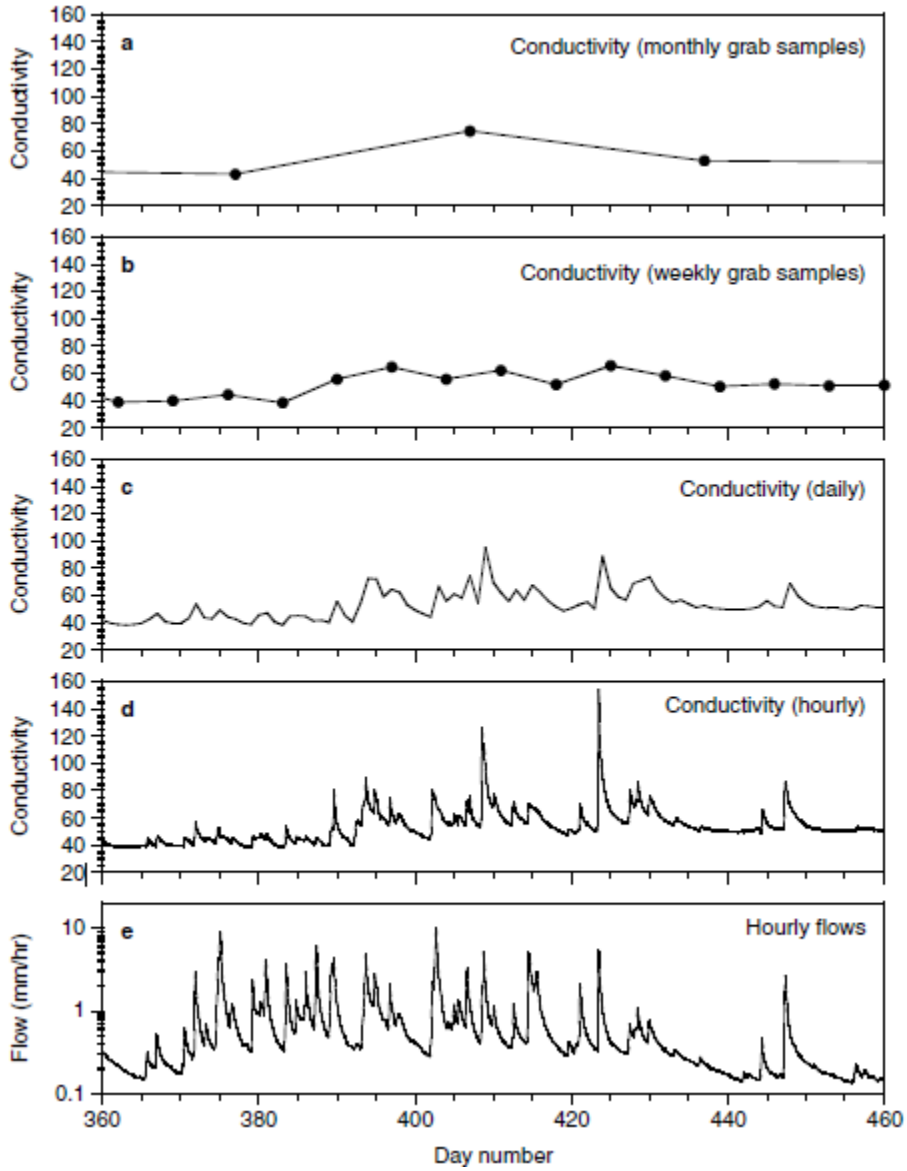

Figure 8: Monthly, weekly, daily and hourly measurements of conductivity with hourly measurements of flow over a period of 460 days at Hore stream, Wales (Kirchner *et al.*, 2004).

Continuous monitoring provides high-resolution data which enables enhanced accuracy. The more monitoring that takes place the less uncertainty there is as Fekete *et al.* (2012) pointed out that

discharge calculations have greater uncertainty when it is under-sampled compared to discharge calculations that has greater monitoring (more regularly sampled). This reinforces that continuous monitoring will further increase the accuracy of measuring discharge in rivers (Fekete *et al.*, 2012) as the reliability of collecting and evaluating river discharge is considered a pressing issue (Kawanisi *et al.*, 2010). High-resolution continuous monitoring will provide new insights into catchment behaviours and will be able to capture hydrological variations (Kirchner *et al.*, 2004).

2.4.4 Conclusion

As land use changes continue to deteriorate urban rivers and its water quality, high-resolution monitoring becomes important in building a robust set of data to inform models and decision making and ultimately stormwater management. Continuous water monitoring will provide new insights into the rhythms and patterns of urban rivers and the kind of pollutants that are transferred by these rivers. The robust set of high-resolution data can ultimately help inform decision making on how to better manage our water resources.

2.5 Hydrological modelling

Climate and land use changes can affect the hydrological cycle including the quality and quantity of water (El-Khoury *et al.*, 2015). It can, *inter alia*, change the runoff within catchments and the streamflow in rivers as well as change the transport of pollutants in water (Tu, 2009). Hydrological modelling is a useful tool for studying the effects of climate and land use changes on water resources and being able to project the impacts of future scenarios on future changes (Praskievics and Chang, 2009). Hydrological models provide a framework to understand and investigate the relationships between climate, hydrological processes and human activities. Hydrological models are designed to investigate climate and land use changes in tandem or separately and can predict the effects of land use and climate changes (Legesse *et al.*, 2003).

2.5.1 Model complexity

Ultimately modelling aims to construct a component of the real world and therefore it is a simpler representation of the real world (Silberstein, 2006). Modelling is probabilistic with uncertainty increasing at every stage of the process from the input of parameters to the simulation of the hydrological processes. Further uncertainty can arise from errors in the model structure and the input of parameters (Praskievicz and Chang, 2009). In theory, the more data available, the more reliable the model should be (James, 2005), however, Silberstein (2006) argues that if the model is flawed, no addition of parameters will reduce the structural uncertainty (Silberstein, 2006). James (2005) mentions that an increase in model complexity does not always lead to increased model reliability, especially when there are uncertainties (James, 2005). James (2005) further mentions that when uncertainty is prevalent, a simple model should be used. A simpler model however, implies a lower spatial resolution. A lower spatial resolution requires modellers to make assumptions on how to aggregate catchment properties that will still provide a reasonably sound model that can mimic the real world. Krebs *et al.* (2014) and Lee *et al.* (2017) recommended a weighted average approach to aggregating land use properties (Krebs *et al.*, 2014; Lee *et al.*, 2017).

Choosing the correct level of complexity is a difficult but crucial part of modelling. It is difficult to estimate the required level of complexity, as it is based on the user's objectives (James, 2005). It is recommended that models should not be too complex or too simple as overly complex models will increase uncertainty however, too simple models may be an unrealistic representation of the real world.

2.5.2 Rainfall data

Using rainfall data that is accurate and representative of an entire catchment can be difficult if there are high levels of spatial and temporal variabilities in the catchment. This is typically the case in catchments where mountainous topography is present. This was evident in a study performed by Fisher-Jeffes (2015) where the mean annual precipitation in a 2600 hectares catchment varied from 600 to 1500 mm/yr. This indicates that when a modelling time step is reduced to a high-resolution time interval, the spatial variability becomes significant because the distribution of rainfall intensities during an event can affect the timing and volume of peak flows during a storm event (Fisher-Jeffes, 2015).

The approaches to investigate rainfall scenarios in models vary from using the Delta-change method which consists of choosing arbitrarily rainfall variations (either an increase or decrease in rainfall) with reference to historical data. The other approach is to use the outputs of global and regional climate models (El-Khoury *et al.*, 2015; Tu, 2009). According to Praskievics and Chang (2009) both approaches have their associated disadvantages. Whilst choosing the former approach can reduce the uncertainty associated with Global Climate Models (GCM), the arbitrary chosen rainfall variations may be unrealistic changes associated with climate change. The latter approach includes the uncertainty associated with GCMs which can vary in their projections (Denault *et al.*, 2006; Praskievics and Chang, 2009).

2.5.3 Event vs continuous modelling

In the past, event-based modelling was commonly used however, in recent times, continuous modelling has become more prevalent. Tan *et al.* (2008) showed that an event-based calibration approach was better for reproducing the overall shape of a hydrograph, peak flow and time to peak. Continuous-event calibration however, provided more accurate runoff volume. An event-based model requires an initialized period when running the model or that certain parameters at the start of the simulation be assumed because one of the shortcomings of event-based modelling is that the antecedent conditions are not taken into account (Wanielista *et al.*, 1991). Boughton and Droop (2003) indicated that the choice of event-based or continuous modelling remains a personal preference. The choice between the two types of modelling could be dictated by the focus of the modeller or the availability of data.

2.5.4 Sensitivity analysis

A sensitivity analysis is performed before calibration takes place. This is done to understand the sensitivity of different parameters in the model. This allows for the calibration to focus on the most sensitive parameters. A common sensitivity analysis is the factor perturbation method which involves keeping all parameters fixed whilst varying a single parameter. James (2005) outlined the shortcomings of this method which includes the fact that the sensitivity gradient is assumed to be linear. This is not always the case as most hydrological parameters are thought to be nonlinear. This method also estimates the sensitivity of one parameter by keeping the other parameters at an expected value.

2.5.5 Calibration and validation

Once the sensitivity analysis is complete, the model is ready to be calibrated. The model is calibrated against known data – typically called observed data. Only a portion of the data is needed to be calibrated whilst the rest is used for validation. The selection ratio of calibration and validation events in modelling has not been consistent. Fisher-Jeffes (2015) used the 1:2 ratio (Fisher-Jeffes,

2015). According to a number of other studies, the ratio used was 1:1.3, 1:6 and 1:2 (Ashbolt *et al.*, 2013; Krebs *et al.*, 2014; Mancipe-Munoz *et al.*, 2014; Palla and Gnecco, 2015).

2.6 Stormwater Harvesting (SWH)

2.6.1 Introduction

The growing demand for water in urban areas together with limited water resources has seen an increase in the interest in Stormwater Harvesting (SWH) (Fisher-Jeffes, 2015). SWH is seen as an option to increase water supply and decrease the demands on potable water. SWH is also seen as a solution to dealing with increased runoff and peak flows into urban waterways (Fletcher *et al.*, 2008). SWH, specifically in urban areas has not been widely practised largely because stormwater runoff is considered highly polluted which can have a significant impact on receiving water bodies (Walsh *et al.*, 2005; Lim *et al.*, 2011). The impacts urban runoff has on urban waterways has also increased the acceptance of SWH as an alternate water supply as the effects of polluted run off on receiving waterbodies have become well documented. SWH is the collection and storage of runoff which is eventually reused. The uncertainty of the effects of climate change makes SWH necessary for water security but it also adds to the complexity of urban water management (Fisher-Jeffes *et al.*, 2017). The benefits of SWH is discussed in greater detail below.

2.6.2 Stormwater Harvesting has the potential to improve water security

A study by Fisher-Jeffes (2015) demonstrated that SWH had the potential to reduce the potable water demand in the Liesbeek Catchment by approximately 20%. This can have significant implications for Cape Town in terms of water security. SWH has been practised in Singapore and was one of the first countries in the world to use stormwater harvesting from its urban catchments to supplement its water supply. Singapore is a densely populated area that receives abundant rainfall, however, due to its limited land area, this makes it difficult to collect and store rainfall. Due to Singapore's increasing water demand, it has turned its urban areas into urban water catchments that harvests its rainfall in the catchments. Pollutant source management strategies were put into place in order to ensure that the water quality of the runoff is acceptable for drinking water (Lim *et al.*, 2011). Fletcher and Deletic (2008) noted that a city in Australia called Ballarat had a typical water usage of 10 400 Million Litres (ML) per year under their stage 4 restrictions. Fletcher and Deletic (2008) also noted that during pre-urbanization, Ballarat's annual runoff was 10 000ML whilst after urbanization the run-off increased to 20 000ML per year. This excess runoff of 10 000ML following urbanization is equal to the total water usage in the town and therefore has the potential to overcome the water shortages faced in the area as well as in many other towns in Australia because in most major cities in Australia the total volume of stormwater exceeds the urban water use for that city each year. SWH in urban areas are more reliable than rural areas because it requires larger rainfall events to produce runoff in forested catchments whilst urbanized areas can produce runoff with small rainfall events (Fletcher and Deletic, 2008).

2.6.3 Stormwater harvesting reduces flooding

SWH is seen as a method to attenuate peak flows, reduce runoff volumes and ultimately prevent floods (Woods-Ballard *et al.*, 2007; Huang *et al.*, 2008; Armitage *et al.*, 2013; Fletcher *et al.*, 2013). SWH systems makes use of several storage systems which includes, *inter alia*, retention ponds, detention ponds, rain gardens and wetlands to store the harvested stormwater. In this way, runoff is detained and reduced downstream. This results in less flooding downstream of urban catchments (Fisher-Jeffes *et al.*, 2017). Fisher-Jeffes (2015) demonstrated the effects SWH could have in the reduction of flooding in the figure below (Figure 9). Whilst the attenuation of peak flows has shown

to be a consistent beneficiary of SWH (Hatt *et al.*, 2006; Fletcher *et al.*, 2013), most studies that has shown this, done so through modelling with little to no monitoring data available (Fisher-Jeffes, 2015).

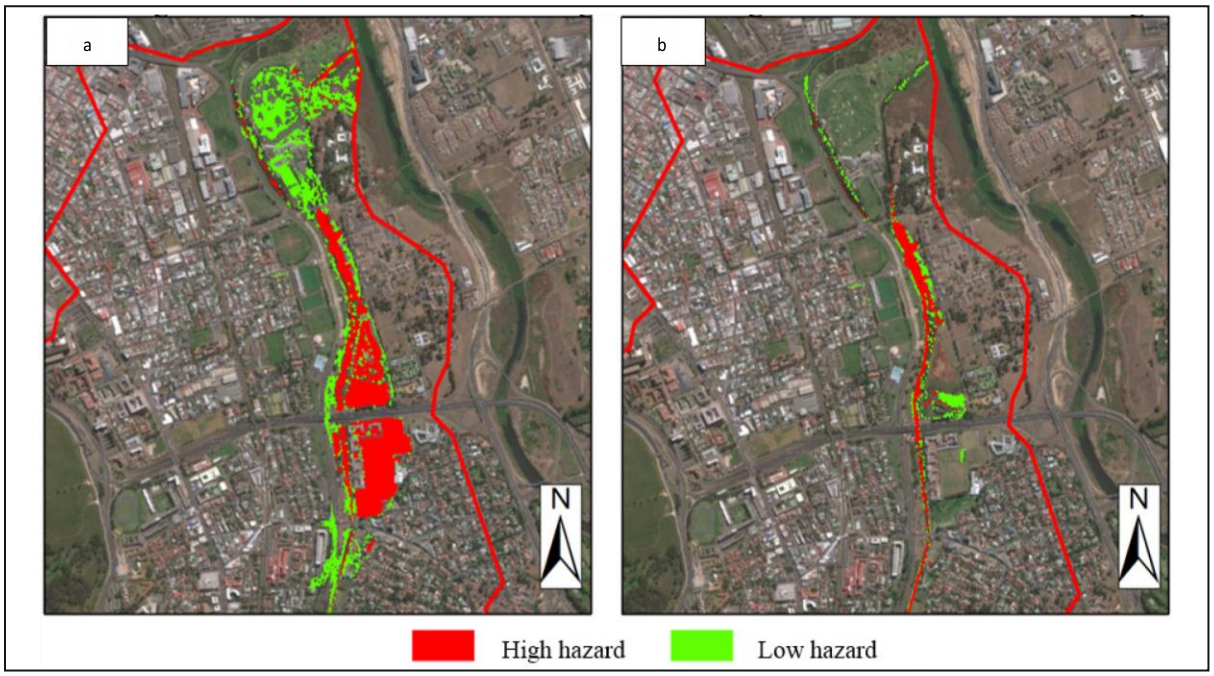

Figure 9: Flooding in the Liesbeek River Catchment on 12 July 2009 shown (a) without and (b) with stormwater harvesting (Fisher-Jeffes, 2015).

2.6.4 Stormwater harvesting improves urban rivers

Urbanization degrades urban rivers, removes riparian zones, changes the water quality and flow regime of rivers and replaces natural channels with piped drainages (Davis *et al.*, 2001; Walsh *et al.*, 2005, 2012; Bernhardt and Palmer, 2007; Fletcher *et al.*, 2007). It has been shown by Fisher-Jeffes *et al.* (2017) that stormwater harvesting can be a viable alternative to conventional water supplies with benefits such as providing water security and prevent flooding, however a study by Fletcher *et al.* (2007) demonstrated that SWH could also improve urban rivers by reducing the runoff volume and contaminants that enter urban rivers. This resulted in a flow regime that is representative of a pre-development hydrograph. Fletcher *et al.* (2007) used the Model for Urban Stormwater Improvement Conceptualisation (MUSIC) to investigate the impacts of SWH on two Australian cities; Brisbane and Melbourne. Whilst it was shown that the runoff and contaminants were reduced that entered its urban waterways in Melbourne, this was not the case in a high density development catchment of Brisbane that received almost triple the amount of rainfall that Melbourne received. It resulted in longer periods of runoff as the SWH harvesting ponds became flooded and overflowed. It was also shown that if the demand of SWH is too high and the capacity of the storage ponds are too low, it can result in over-abstraction, causing lower levels of flow than a pre-development flow regime (Fletcher *et al.*, 2007). A balance needs to be met with regards to the demand for SWH, the storage capacity and the minimum amount of flow needed in urban rivers to adequately allow the ecosystems to operate. This reinforces the importance of monitoring and building a robust set of data to understand the behaviour of urban rivers in terms of its volume and water quality under varying conditions.

2.6.5 Conclusion

SWH offers an alternative water supply to conventional surface water schemes which could ensure improved water security in a city that experiences droughts. It can also help attenuate peak flows, lowering the volume of flow downstream which effectively would prevent flooding. It can also provide amenity values through the creation of detention ponds, wetlands and parks that are regularly maintained.

2.7 Stormwater management

2.7.1 Introduction

Traditionally stormwater management practices were designed to remove stormwater as quickly as possible to reduce flooding and damage to property. This often results in stormwater entering directly into streams, bypassing floodplains and riparian zones. Urban development has also resulted in the removal of floodplains and riparian zones, decreasing the attenuation of stormwater causing an increase in erosion and bank instability (Vietz *et al.*, 2016). The removal of floodplains and riparian zones, as well as other vegetation has resulted in a decrease in evapotranspiration. This has resulted in decreased water quality and ecological diversity in urban rivers. Previously, stormwater management techniques to deal with geomorphic change in urban streams were limited to altering the flow pattern rather than attenuating the excess stormwater volume (Vietz *et al.*, 2016). The realization that urban stormwater has led to environmental damage inspired new stormwater management techniques with multiple objectives.

2.7.2 Stormwater control measures (SCMs)

In recent decades, the urban runoff has been identified as the main contributor to stream degradation and therefore treating urban hydrology in the form of Stormwater Control Measures (SCMs) has been used for restorative treatments. An integrated approach is needed, one that incorporates the need to restore ecological aspects of the pre-development flow regime (Burns *et al.*, 2012) that mimics the volume balance of evaporation, runoff and infiltration (Walsh *et al.*, 2016). In recent decades, SCMs included reducing the peak flows and extending flow duration. Recently stormwater management also includes incorporating the stormwater system into the landscape thus, aiming to mimic the natural hydrological cycle. This has been done by using small scale detention ponds, biofiltration ponds and porous pavements. These techniques are aimed at minimising the impacts on receiving water bodies, making restoration more effective (Bernhardt and Palmer, 2007). However, Burns *et al.* (2012) argues that end-of-catchment load reduction approaches such as constructed wetlands do little in reducing the frequency of high flow events. Constructed wetlands can reduce pollutants and increase detention time but is ineffective in reducing the volume of stormwater through infiltration causing a further decrease in baseflow (Burns *et al.*, 2012).

Understanding the processes of urban stormwater and the impacts it has on receiving urban streams are a prerequisite for developing management strategies (Fletcher *et al.*, 2013). Burns *et al.* (2012) argues that flow-regime management can successfully restore ecological restoration and return the hydrograph to a pre-development flow regime which includes having stormwater retained in a distributed manner at small scales spread throughout the catchment thus covering the entire catchment unlike in end-of-catchment load reduction approaches (Burns *et al.*, 2012). Previous studies by Elliot *et al.* (2010); Tillinghast *et al.* (2012) and Hogan *et al.* (2013) has suggested that SCMs can reduce stormwater volumes but not to the extent of pre-urban development flows (Elliot

et al., 2010; Tillinghast et al., 2012 and Hogan et al., 2013) because the physical form of modified urban streams are different from their natural condition (Burns *et al.*, 2012).

There is a growing interest in the control of stormwater management with the use of Low Impact Development (LID), Water Sensitive Urban Design (WSUD) and Sustainable Drainage Systems (SuDS) (Allen *et al.*, 2004; Ellis *et al.*, 2006; Van Roon, 2007; Armitage *et al.*, 2014) and is being widely promoted to control runoff and prevent downstream floods (Hood *et al.*, 2007). Examples of these stormwater control techniques include wetlands, rain gardens, treatment trains, filter strips and porous pavements (Hood *et al.*, 2007; Armitage *et al.*, 2013) In urban areas, streams are designed to act as drains or sewers with the purpose of moving stormwater away from urban areas to prevent flooding (Armitage *et al.*, 2013). The broad philosophy of WSUD and SuDS is aimed at minimizing peak flows and extend flow duration whilst reducing runoff through pervious surfaces and treat stormwater as close to its source as possible, acting as an on-site drainage (Bernhardt and palmer, 2007; Armitage *et al.*, 2013; Carlson *et al.*, 2014). It is aimed at returning the flow regime to pre-development and restore water quality and the health of receiving water bodies (Fletcher *et al.*, 2013). This technique involves integrating urban stormwater into the landscape, thus preserving the landscape and the hydrological cycle. This will help treat the stormwater as a resource and not a waste product (Carlson *et al.*, 2014), because the primary reason for deterioration of water bodies is the disruption of the natural water cycle (Bernhardt and palmer, 2007; Armitage *et al.*, 2013). The slow nature of the process of protecting streams is due to the lack of understanding of urban stormwater runoff (Walsh *et al.*, 2012). Therefore, a paradigm shift is needed, one that involves catchment scale solutions that provides a fluvial corridor and reduces the peak flow events to create a feasible environment for urban rivers (Kondolf, 2011). Despite this, the field is still poorly understood in terms of the quantification stormwater management practices has on the water cycle and water budget of a system (Fletcher *et al.*, 2013).

2.7.3 Conclusion

To develop new approaches to urban stormwater management, a better understanding is required to efficiently protect and restore urban streams to a pre-development catchment (Burns *et al.*, 2014). Management of urban stormwater remains a complex challenge. Stormwater has the potential to be used as a resource and therefore stormwater management could serve the purpose of improving water quality and protecting the ecological diversity of urban rivers (Wong and Eadie, 2000). Integrated approaches that incorporates the entire water cycle into decision making, for example, an approach that views stormwater as a resource rather than a waste product has received significant attention in recent years, however, more research in this field is still required to understand this intricate complex system (Fletcher *et al.*, 2013).

3. Methods

3.1 Study design

This study aims to characterize stormwater quality and discharge, and model surface water quality and discharge in an urban river using high-resolution data. The study design incorporated two research activities, which was to explore surface water in the present through water quality and discharge analysis and the future through surface water modelling. The monitoring of discharge and water quality took place along the Liesbeek River. The purpose of the monitoring was to understand the water quality and discharge in an urban river to determine the suitable range for harvesting stormwater. The purpose of the model was to understand the hydrological conditions and the catchment response to rainfall events under various conditions. Three level sensors were placed under bridges along the river which measured the height of the water. For this study, the height of the water level was used to calculate the discharge. The calculations are described in greater detail in section 3.3.6. Three water quality sensors were placed in the river for measuring pH, DO, EC and temperature to measure a limited set of water quality parameters. The combined set of data from these sensors were used to understand the relationship between discharge, water quality and rainfall.

Six (three level and three water quality) sensors were used for the purpose of this study which measured the water level of the river and the water quality. The level and water quality sensors were located upstream (Site 1), midstream (Site 2) and downstream (Site 3) (Figures 18, 19 and 20). The location of the sensors was strategically chosen in order to capture the spatial and temporal variability of rainfall in the Liesbeek Catchment as indicated in Figure 12. Details of the flow and water quality sensors are discussed in sections 3.3.6 and 3.4.1 respectively.

3.2 The Liesbeek Catchment

The Liesbeek Catchment covers an area of 2600 hectares and is situated along the eastern slopes of Table Mountain (Figure 10). The Liesbeek River is 9km long and is fed by numerous streams from the eastern slopes of Table Mountain. The river flows approximately in the north to north-nest direction and runs through a number of middle to high-income residential areas (Fisher-Jeffes, 2015). The headwaters of the Liesbeek River flow above Kirstenbosch Botanical Gardens from the eastern slopes of Table Mountain and empties out in Table Bay. The Liesbeek River begins on the foothills of Table Mountain at an elevation of 120masl which drains numerous streams on its eastern slopes. The Protea Stream is the southernmost source of the Liesbeek River. Protea Stream is joined by Window Stream, Hiddingh Stream which arises at the top of Mountain, joins the Liesbeek River. The remaining streams are joined into the Liesbeek River through piped stormwater drains in developed areas. Finally, the Liesbeek River joins the Black river 2km from their mouth in Table Bay (Brown and Maqoba, 2009).

Development of residential areas began in the 18th century. The construction of the main road between Cape Town and Wynberg in 1807-1811 and the railway connection to Wynberg in 1864 was considered the catalyst towards increasing the residential settlements along or close to the Liesbeek River. The Liesbeek River today still shows some of its natural beauty albeit being affected by severe urbanization with much of the lower sections canalised. Due to the rainfall experienced in the upper reaches of the Liesbeek Catchment (Figure 12) and the fact that the gradient smoothens out dramatically over the course of the river (Figure 11), it resulted in the catchment being a flood risk hazard. In the beginning of the 1900s when urbanization rapidly increased, buildings close to or along the river became flooded during the rainy season. It was due to this that the river became canalized between 1942 and 1962. The canalization was to improve the conveyance of water

downstream from Claremont and Rondebosch (Brown and Maqoba, 2009). Currently, 50% of the catchment is considered urban. The remainder of the catchment that is not urbanized consists of Kirstenbosch Botanical Gardens, forestry plantations and Table Mountain National Park. The upper reaches of the Liesbeek River are natural areas whilst the lower reaches of the catchment have the highest levels of built-up land with more compact urban infrastructure and a higher population density. The upper reaches also receives close to three times more rainfall than the lower reaches (Figure 12) (Fisher-Jeffes, 2015).

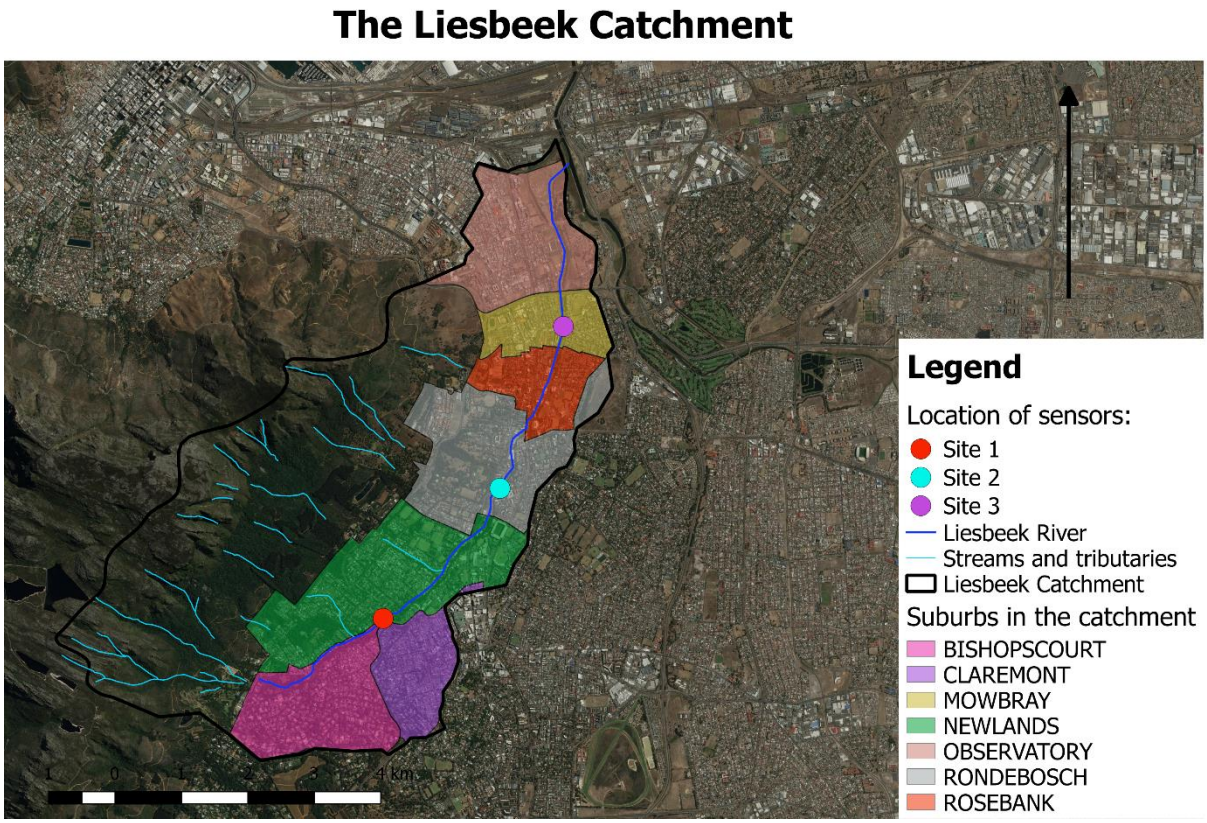

Figure 10: The Liesbeek Catchment

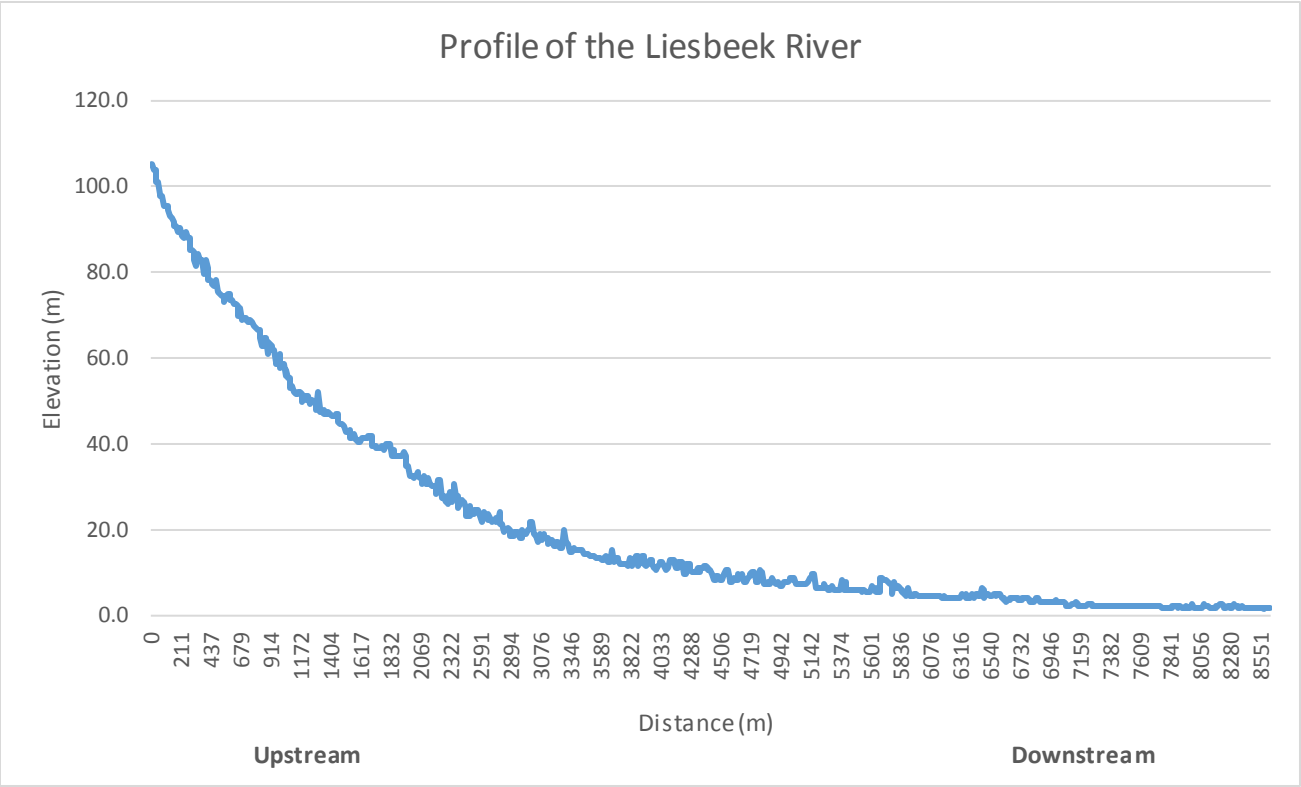

Figure 11: Gradient of the Liesbeek River

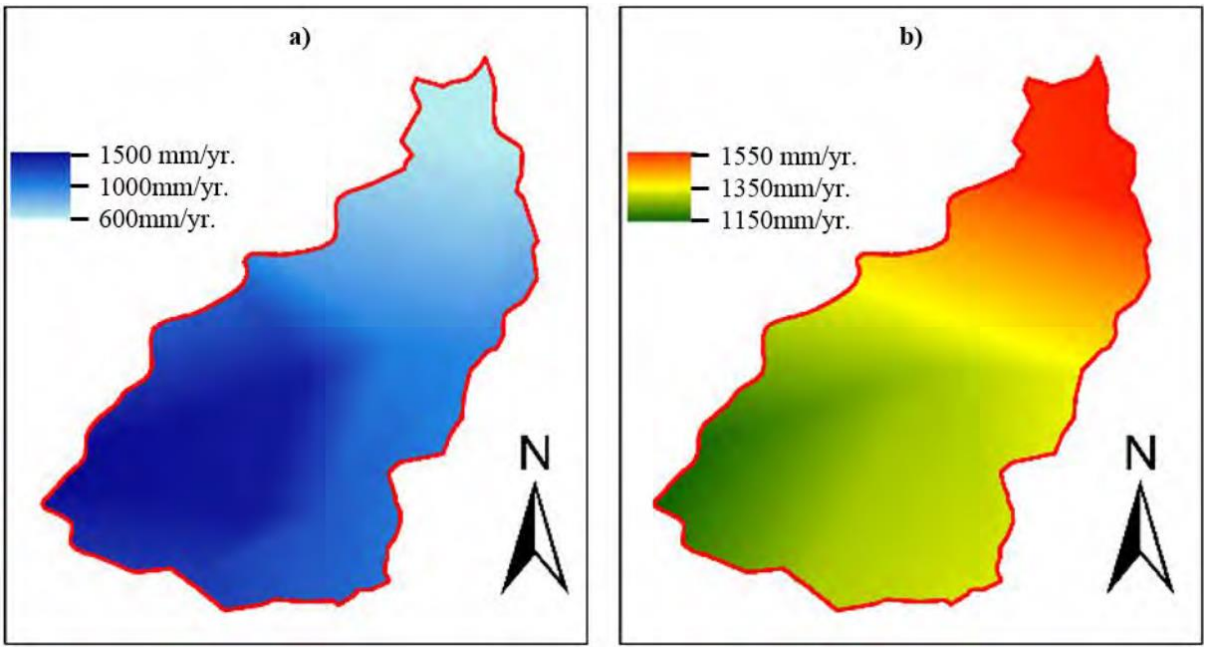

Figure 12a & b: a) Annual average rainfall and b) annual average evaporation over the Liesbeek Catchment (Fisher-Jeffes, 2015).

3.3 Hydrological modelling: Data collection and processing

3.3.1 Selection of software

Personal Computer Storm Water Management Model (PCSWMM) is a commercial software package that improves the usability and functionality of the United States (US) Environmental Protection Agency (EPA) software SWMM (Armitage *et al.*, 2014). The PCSWMM software was chosen for

hydrological modelling due to the researcher's familiarity with the software and its familiar interface with Quantum GIS (QGIS). Furthermore, PCSWMM is ideal for research purposes and hydrological modelling. PCSWMM is also widely used in South Africa and internationally recognized as a standard for modelling stormwater.

3.3.2 Rainfall, temperature and evaporation data

Rainfall data was obtained from the South African Weather Service (SAWS) for two rainfall stations in the Liesbeek Catchment. The two rainfall stations were located in Kirstenbosch and Observatory (Figure 13). Due to the high variability of rainfall within the Liesbeek Catchment (from 600 to 1500 mm/yr), only using two rainfall stations was not considered ideal, however these were the only two rainfall stations that provided accurate daily as well as five-minute interval rainfall data. The rainfall data from these two rainfall stations were used as inputs into PCSWMM. The sub-catchments in the model that were not located in Observatory and Kirstenbosch were then assigned to the rainfall stations based on Figure 12 and their proximity to the rainfall station (Figure 13). The sub-catchments that fall within the regions that receive 1000-1500mm/yr of rainfall were assigned to the Kirstenbosch rain gauge. Whilst the sub-catchments that falls within the region that receives less than 1000mm/yr of rainfall were assigned to the Observatory rain gauge (see figure 12). This gave a spatial representation of how the rainfall in the catchment might have been.

Daily (minimum and maximum) and five-minute temperature data located in Kirstenbosch and Observatory were obtained from SAWS. Evapotranspiration data for the hydrological model was then calculated in PCSWMM using Hargreaves' method (Rossman, 2008). According to Allen *et al.* (1998) the Hargreaves method has shown reasonable results and is accepted globally. One of the main advantages of this method is its simplicity with low data requirements.

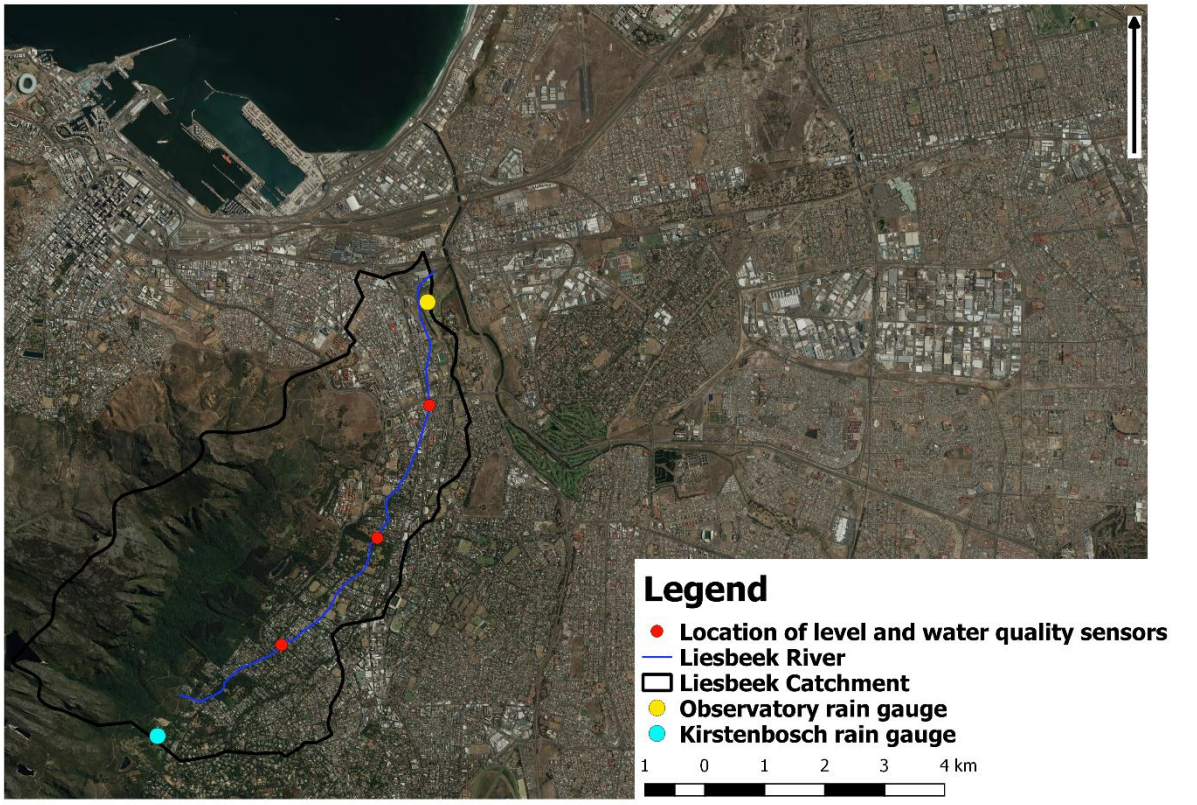

Figure 13: The location of the Kirstenbosch and Observatory rain gauges in relation to sampling sites.

3.3.3 Topography

Elevation data for the Liesbeek Catchment was obtained from the City of Cape Town (CoCT) in the form of Light Detection And Ranging (LiDAR). The LiDAR data had a resolution of 2m and were based on ground points only. This allowed QGIS to create an accurate Digital Elevation Model (DEM). The 2m DEM was used to calculate catchment slopes and node elevations in PCSWMM. A 2m contour file was also obtained from the CoCT and was used to supplement the DEM data.

3.3.4 Land use and soils data

Land uses in the Liesbeek catchment were manually delineated using aerial imagery and land use ESRI shapefiles on QGIS. The land uses were classified into three dominant groups (Figure 14); Nature and conservation, green open space and residential areas. This is a simplistic representation of the land uses in the Liesbeek Catchment. Due to the nature of the study, high-resolution detail in the catchment was not needed whereas Fisher-Jeffes *et al.* (2015) that looked at rainwater harvesting, needed high-resolution data to account for property and roof sizes. Soil types were obtained from the CoCT. Infiltration modelling is required to model runoff from pervious surfaces. The Gren-Ampt infiltration calculation was used for this study, therefore, Green-Ampt parameters were used as inputs to the model, which are based on soil types.

3.3.5 Stormwater network data

The stormwater pipe networks of the Liesbeek Catchment were obtained in GIS files from the CoCT. The pipe sizes from these files were used in PCSWMM. The Liesbeek Catchment has several short stormwater pipes that feeds into the main larger stormwater pipes that enters the Liesbeek River. It was decided it would not be practical to model all the short stormwater pipes but rather a select few that feeds into the main pipes. This resulted in more water being channelled into fewer pipes and therefore the small stormwater pipes were given a slightly larger diameter in the model.

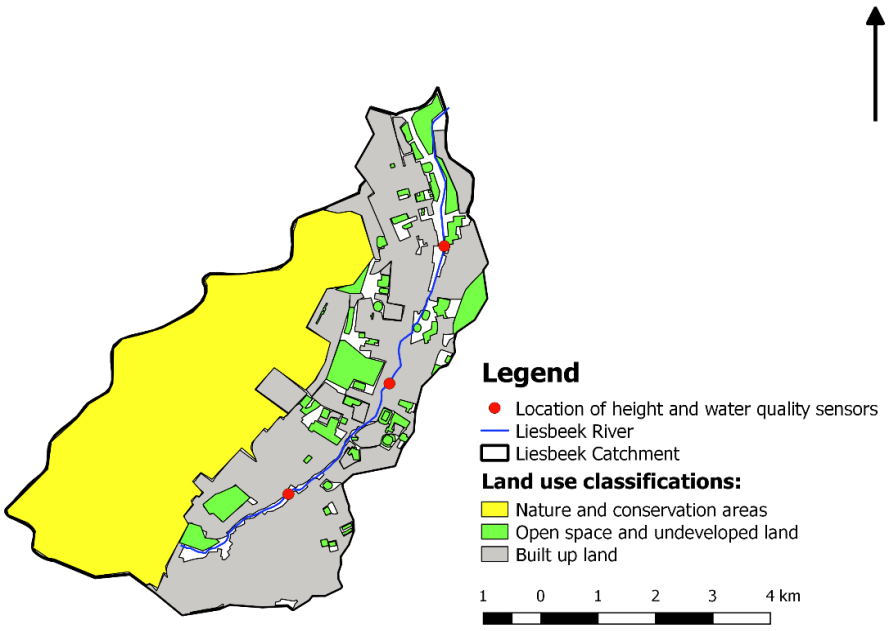

Figure 14: Land use classification in the Liesbeek Catchment

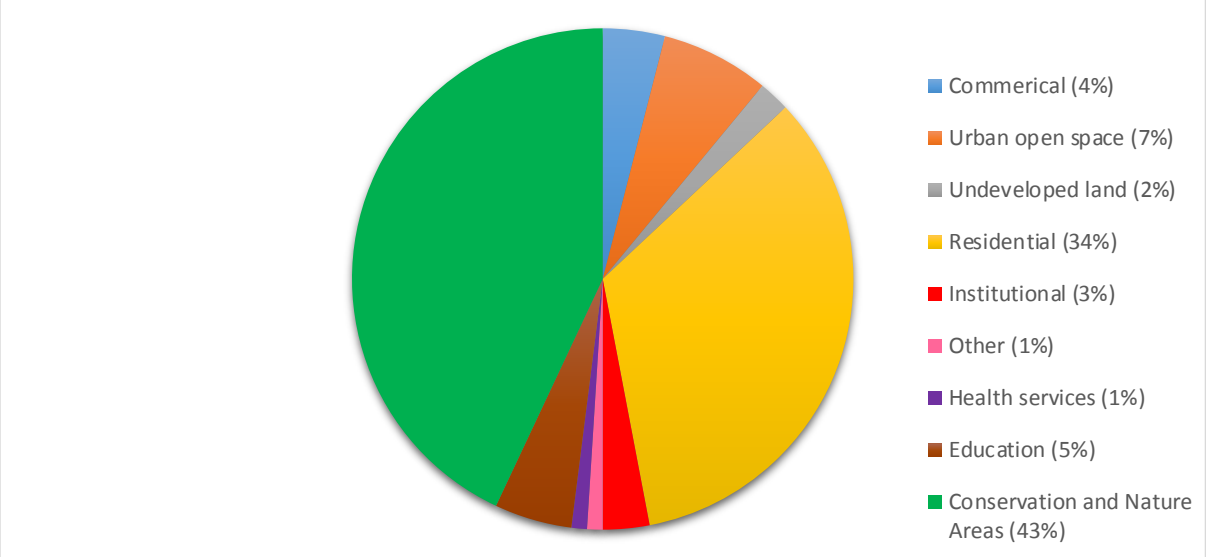

Figure 15: Division of land use categories in the Liesbeek Catchment (Fisher-Jeffes, 2015).

3.3.6 Discharge data

River levels were measured *in situ* from April to September 2018 and then again from May to July 2019 at three locations across the Liesbeek River. Site 1 and 2 collected level data from May to July 2019, whilst Site 3 collected level data from April to September 2018 and again from May to July 2019. All three level sensors were located in canalized, rectangular sections of the Liesbeek River. Sites 2 and 3 were selected on the basis that flow would be uniform and thus discharge could be calculated using Manning's equation. Upstream however, did not have a suitable location that fits the conditions for Manning's equation due to the numerous weirs that obstructed the river's flow. Instead Site 1 fitted the conditions for a broad-crested weir from which discharge could be calculated. The discharge data obtained from Site 3 during 2018 was used to calibrate the PCSWMM model.

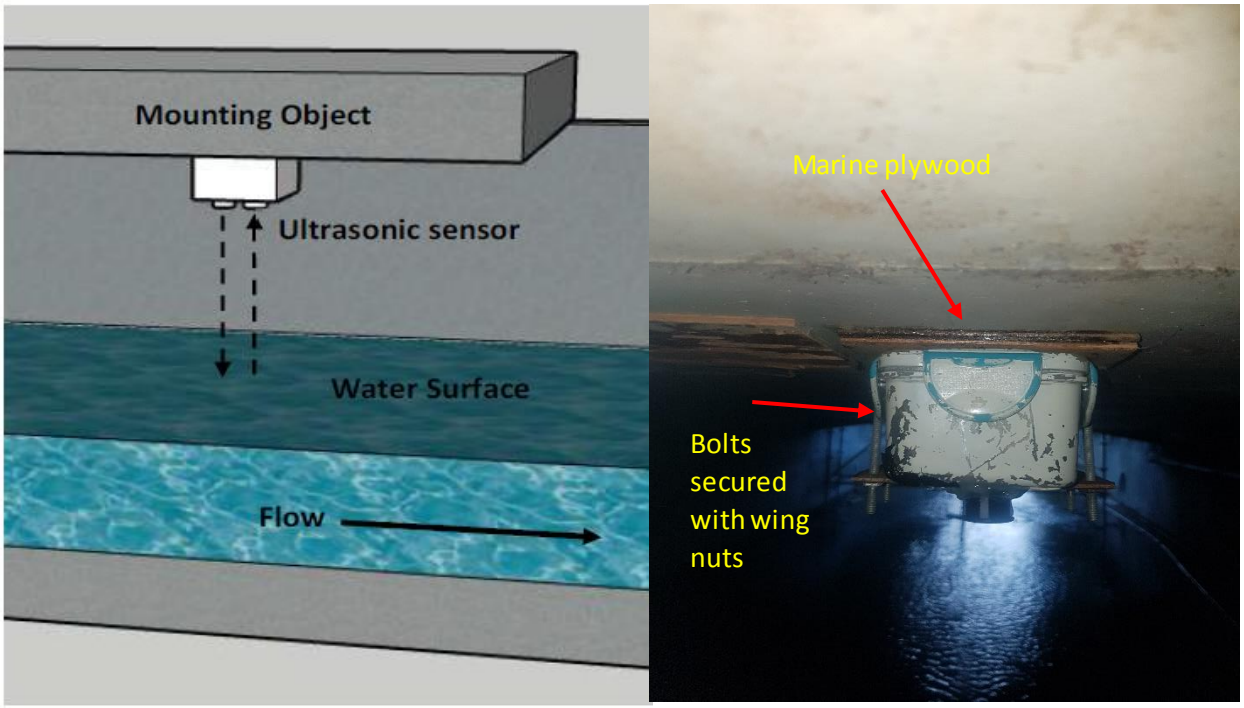

Figure 16: Water level sensor: Diagram (left) and photograph (right) (Adapted from Fell *et al.*, 2017).

The level sensors are low cost measuring devices that were developed by researchers at the University of Cape Town (Fell *et al.*, 2017). The sensors consisted of ultrasonic devices which measured the distance from the sensor to the water level by sending an ultrasonic sound wave to the water. The sound wave is then returned and the sensor reads the echo. The sensor calculates the distance from the sensor to the top of the water using the speed of sound. Water depth was then calculated by subtracting the recorded distance from the distance between the sensor and the canal base. The ultrasonic level sensor recorded measurements every five-minutes and uploaded them onto an open source website, from which high-resolution data records were downloaded. The data was validated by using a measuring tape to measure the distance between the sensor and the top of the water of the river and comparing it to the data sent to the server.

The sensors were housed in plastic containers that were kept within a marine plywood structure. Glue was used to secure the structure on the underside of the bridges at all sites. The marine plywood structure housed the plastic containers using bolts secured by wing nuts (Figure 16). The wing nuts could easily be loosened to remove the lunch box for maintenance purposes such as changing the batteries. These sensors collected data from April to September 2018 at all three Sites. It was later discovered that Sites 1 and 2 did not fit the conditions for Manning's equation, rendering the data at Sites 1 and 2 useless. Thereafter, all three level sensors broke. This prompted the development of three new level sensors which were housed in similar plastic containers and structures (Figure 17). The new level sensors collected data from May to July 2019.

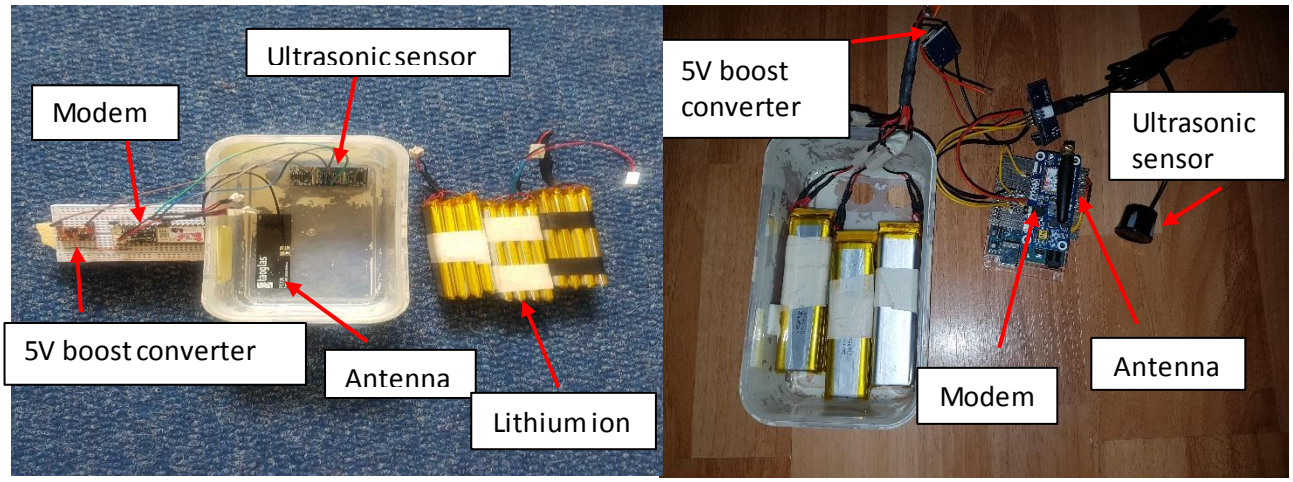

Figure 17: Old (left) and new (right) level sensors.

Once the height of the river was determined, the flow rate at Site 2 and 3 was calculated using manning's equation which is given by the following formula: $Q = \frac{1}{n}\frac{A^{\frac{5}{3}}}{P^{\frac{2}{3}}}S^{\frac{1}{2}}$

Whereby:

Q = Flow rate; N = Manning's roughness coefficient; A = Area; P = Wetted perimeter; S = Slope

The height of the river at Site 1 was used to calculate flow rate using a broad-crested weir equation which is given by the following formula: $Q = CLH^n$

Whereby: Q = Flow rate; C = Discharge coefficient; L = Width of the weir; H = Height of water above weir; n = 1.5

The geometric properties needed for both equations were measured on site and was used for the calibration of constants for the broad crested weir. The slope was calculated using the 2m DEM as well as from a land survey undertaken by the researcher in October 2018.

Figure 18: Site 1 (Upstream)

Figure 19: Site 2 (midstream)

Figure 20: Site 3 (downstream)

3.4 Water Quality

3.4.1 Water quality selection and data

The water quality parameters that were chosen for this study were temperature, pH, Electrical Conductivity (EC) and Dissolved Oxygen (DO). The selected parameters were chosen because they offer signals in water quality variations and they are also widely used. This allowed for comparisons with other studies.

Water quality parameters such as pH, EC, DO and Temperature (Figure 21) were used in this study as a proxy to measure water quality. The water quality instruments were purchased from Atlas Scientific and were calibrated based on the instructions from the manual. These sensors measured water quality in real-time from March to July 2019. The water quality sensors were located at Sites 1-3. The water quality sensors however, had a higher battery consumption than the level sensors and therefore required a battery with a much higher capacity. This did not allow the water quality sensors to be attached under the bridges as they were too heavy. The water quality sensors were buried along the sides of the canal of the river under the ground in a water proof box with the cable probes coming out of the box and into the river through a pipe that was glued to the wall of the river (Figure 22). The holes in which the cables came out from the box were then sealed using silicone sealant.

Initially the water quality sensors captured the data onto an SD card in five-minute intervals. This, however, proved problematic as the SD card was inconsistent, which resulted in missing data. Subsequently the water quality sensors were changed to send the data onto an open source website and took measurements every five-minutes from which high-resolution data records were downloaded. The water quality probes were calibrated using their respective calibration solutions based on the instructions from their manuals. The water quality data was validated using handheld water quality probes that measured pH, DO, EC and temperature. The handheld probes were placed

in a beaker filled with water from the Liesbeek River on site. The water quality data from the hand held probes were then compared to the continuous water quality probes in the river as verification.

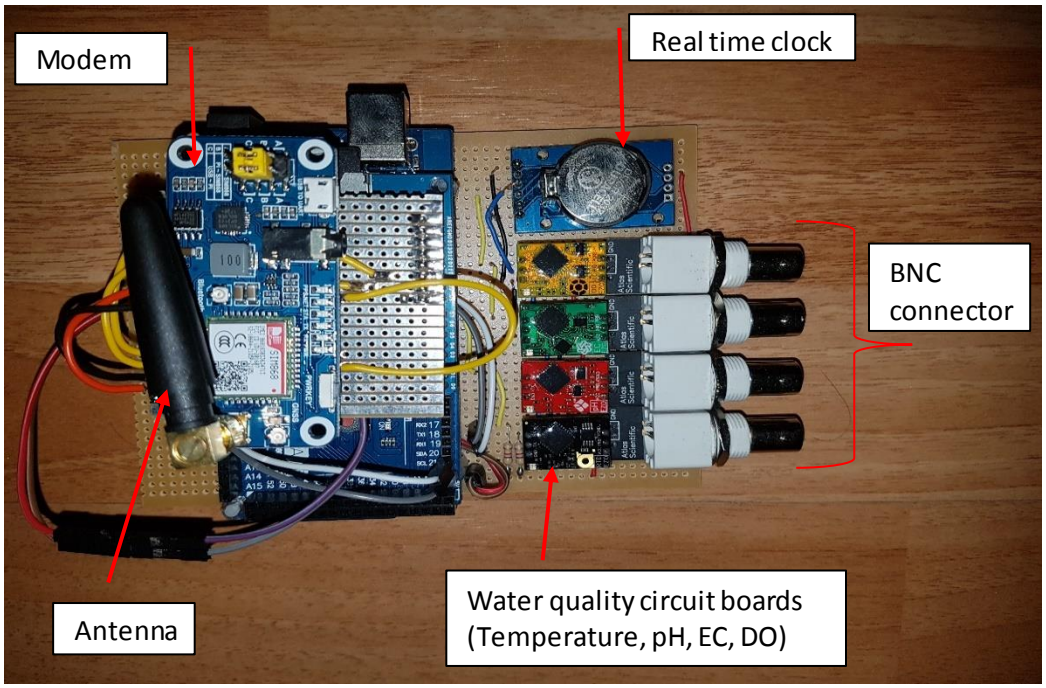

Figure 21: Water quality sensors.

Figure 22: Water quality logger housing (left) and water quality sensors attached to the canal (right)

3.4.2 Water Quality parameters

3.4.2.1 pH

The pH is a measure of the acidity of the water. The pH is determined by many factors such as rainfall quantity, temperature, run off, microbial activity and the interaction of hydrogen ions with other ions in the water, therefore, it is not easy to allocate changes in pH with a specific event making it a highly variable parameter (DWAF, 1996a). Changes in pH could be due to rainfall or long periods without rain, human perturbation, temperature and daily fluctuations. pH is an important

parameter in determining the state of a riverine system. pH is an important component in the chemical and biological systems of natural waters (US EPA, 1986). The pH can determine whether a river is suitable for certain species or not. Most aquatic species can survive in a riverine system with a pH range of 5.0 to 9.0. However, the optimal pH of water in the river for aquatic organisms is from 6.5 to 8.5 (Al-Asadi, 2016).

3.4.2.2 DO

Dissolved Oxygen (DO) is the measure or weight of oxygen molecules dissolved in water. DO is a variable parameter that is influenced by many factors such as temperature, time of day, altitude and flow rate. Changes in DO is due to photosynthesis by plants, interactions with the atmosphere through diffusion and through respiration of aquatic species. The maintenance of a sufficient amount of DO (4-6mg/L) in rivers are crucial for aquatic species. Therefore, the DO provides a good indication of aquatic health. There is a natural cycle of DO in which it coincides with the cycle of photosynthesis and respiration. The DO will decrease throughout the night and reach its minimum by morning, and then rise to a maximum by mid-afternoon. High levels of dissolved oxygen in a riverine system are healthier and will consist of a greater species diversity. Lower levels of dissolved oxygen in a riverine system will not be able to support sensitive or intolerable species (DWAF, 1996f). Sewage run off and organic matter in rivers can reduce the dissolved oxygen due to decomposition by microorganisms which uses the oxygen and converts ammonia into nitrate (Watt, 2000). The DO in unpolluted water bodies are close to but less than 10 mg/L (Chapman and Kimstach, 1996).

3.4.2.3 Temperature

Temperature affects the rate of chemical reactions and metabolic rates of organisms and therefore temperature plays a major role in the distribution of aquatic organisms. Water bodies show daily as well as seasonal variations in temperature however, artificially induced changes in temperature can affect aquatic organisms (DWAF, 1996f).

3.4.2.4 EC

Electrical Conductivity (EC) in water is a measure of the ability to pass electrical flow which is related to the ions and salts present in the water (DWAF, 1996f). Conductivity and salinity are directly related to each other and are important in water bodies as it can determine its health, because high salinity values are associated with low DO values (Miller et al., 1988). Conductivity in water remains constant and therefore, any change in the conductivity can be a clear indication of pollution and can have detrimental impacts on the water body as most aquatic organisms can only survive in a specific salinity range (Wetzel, 2001). Salinity levels increase as water moves downstream because it continuously accumulates due to natural and anthropogenic sources. Evaporation also contributes to high EC values as salts are left behind when water is evaporated, thus increasing the ratio of salts to water volume. However, rainfall and other freshwater inputs tends to dilute the water and decreases the EC value (DWAF, 1996f). EC ranges from 10 - 1000 µS/cm in fresh water bodies but can exceed 1000 if the water is polluted or drains large amounts of run off from an urbanized catchment (Chapman and Kimstach, 1996).

3.5 Data analysis

The discharge and water quality obtained from the sensors during rainfall events were comprised of the following analyses:

- Descriptive statistics including graphs, pollutographs, tables and box plots to examine discharge and water quality.

- Correlation analysis between the water quality parameters, discharge and rainfall to examine their relationships.

The descriptive statistics were used as a way to visualise the data in a meaningful way and allows for visual comparisons. The correlation analysis further investigated the comparisons between the sites by allowing the relationships between the variables at each site to be quantified.

- A PCSWMM model to model surface water under various rainfall designs
- Regression analysis to predict EC values based on modelled discharge data from PCSWMM.

PCSWMM is useful for modelling rainfall scenarios within the catchment, whilst the regression analysis allows for the predictive modelling of water quality.

3.7 Limitations of the study

Only three sensors were placed out in the field that measured discharge and water quality at a specific point, therefore it gave no indication of the discharge and water quality of the river between the sensors. The sensors were battery operated and therefore when the battery was low, the sensors were taken out of the field and the batteries were placed on charge. This resulted in missing data on the days when the sensors were removed from the field. Whenever the signal at the locations was low, it would not send a reading to the website, resulting in fragmented data. In response to these limitations, the sensors were removed from the field to be placed on charge during a dry period when the height of the water and water quality remains constant with little to no runoff, therefore ensuring that rainfall events were captured during the rainfall season. Furthermore, there were several of limitations in the PCSWMM model which is discussed in Chapter 4.

4. Results and Discussion

This chapter discusses stormwater quality and the rate of discharge in the Liesbeek River and the implications for stormwater management as a water resource during peak flow. As discussed earlier stormwater harvesting has a number of benefits such as improving water security, preventing or reducing flooding and improving the condition of urban waterways through flow modulation (Woods-Ballard *et al.*, 2007; Huang *et al.*, 2008; Armitage *et al.*, 2013; Fisher-Jeffes, 2015, Fisher-Jeffes *et al.*, 2017). This study assessed the quality of stormwater and the implications for abstraction and reuse by using high-resolution data of the patterns and trends observed in the discharge and water quality during rainfall events.

4.1 Rainfall events in the Liesbeek Catchment

A total of 14 rainfall events were recorded during the study however, not all sites recorded all 14 rainfall events due to equipment failures including poor signal transmission at various times. Insufficient data were collected at Site 1, and therefore the results are limited. A selection of rainfall events were discussed in this section with the full record of rainfall events available in Appendix A (See tables A1-A14).

The upper reaches of the Liesbeek Catchment are dominated by vegetation of low shrub and grasses on the eastern slopes of Table Mountain. The upper reaches of the catchment have a lower infrastructure and population density compared to the lower reaches of the catchment that is highly impacted by more compact urban areas (Fisher-Jeffes, 2015). Rainfall is unevenly distributed over the catchment with the upper reaches receiving three times more rainfall than the lower reaches of the catchment. The uneven distribution of rainfall and the changes in land use could be responsible for some of the differences in the responses to rainfall at each site. Furthermore, the contour lines shown in figure 23 illustrates the steep mountain slopes which receives the highest amount of rainfall in the catchment. Site 1 was situated upstream of the catchment and experienced higher rainfalls than Site 2 and 3 that were situated in highly urbanized sections of the catchment and experienced lower rainfalls (See figure 23 and figure 12).

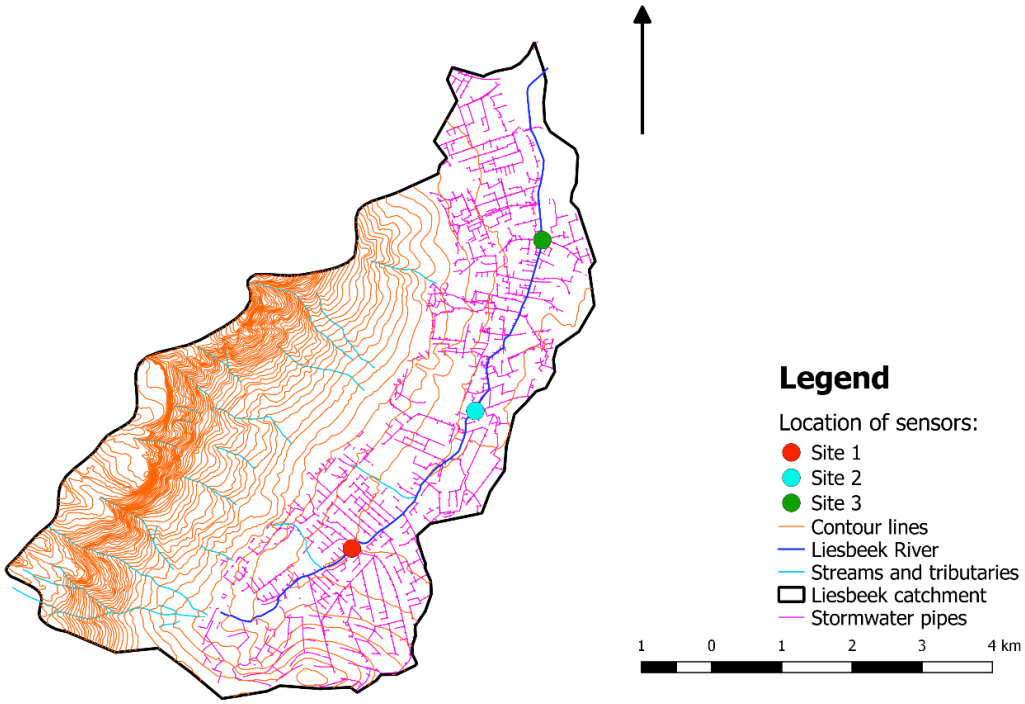

Figure 23: The Liesbeek Catchment showing contour lines and stormwater pipes.

4.1.1 Comparisons between the sites

4.1.1.1 Discharge

According to Table 3, the discharge in the Liesbeek River increased from upstream (Site 1) to downstream (Site 3). Site 1 had an average discharge of 0.6 m^3/s, whilst Site 2 had an average discharge of 1.2 m^3/s and Site 3 had an average discharge of 2.1 m^3/s. According to Fisher-Jeffes (2015) the upper catchment has a lower urban density and more vegetation cover and is therefore expected that Site 1 would have the lowest peak discharges and longest lag times. Site 2 experienced the highest peak discharges and shortest lag times. This was caused by the runoff produced around Site 2 due to high urban density and imperviousness downstream. This was also noted in studies by Aichele (2005) and White and Greer (2006) whereby an urbanized area resulted in higher stream discharge with shorter lag times. The lag and attenuation of peak flows for site 3 was mainly due to channel hydraulic characteristics. Despite Site 3 having a higher average discharge, Site 2 had higher peak discharges which were shorter in duration (Figure 24). The peak discharges for Site 2 was close to double the peak discharge for Site 3 for rainfall events that were above 42.2mm at Kirstenbosch (Table 4). The higher peak discharges for Site 2 indicate the quick rainfall to runoff response which may be due to the impermeable surfaces in the vicinity. Even though Site 2 had higher peak discharges than Site 3 for high rainfall events, the total volume during the rainfall events were still higher at Site 3.

Table 3: Average ± standard deviation (STD) and range for discharge data across Sites 1-3.

Discharge		
Site	**Average ± STD**	**Range**
Site 1	0.6 ± 0.7	0.0003 – 6.7
Site 2	1.2 ± 1.9	0.004 – 28.1
Site 3	2.1 ± 1.0	0.9 – 15.2

Table 4: Comparison of peak discharge between Site 2 and 3.

Date	Rainfall (mm)		Peak discharge (m^3/s) Site 2	Peak discharge (m^3/s) Site 3
	Kirstenbosch	Observatory		
19-20/05/19	88.2	22.4	15.3	7.5
4-5/06/19	71.6	44.4	28.1	15.2
9/06/19	17.8	9.2	2.1	2.7
21-22/06/19	56.6	20.8	9.5	5.1
27-28/06/19	41.2	11.8	4.1	4.0
18-19/07/19	30.2	4.0	4.4	3.7
22-23/07/19	74.2	13.4	17.8	9.1
29-31/07/19	42.2	14.4	4.3	3.4

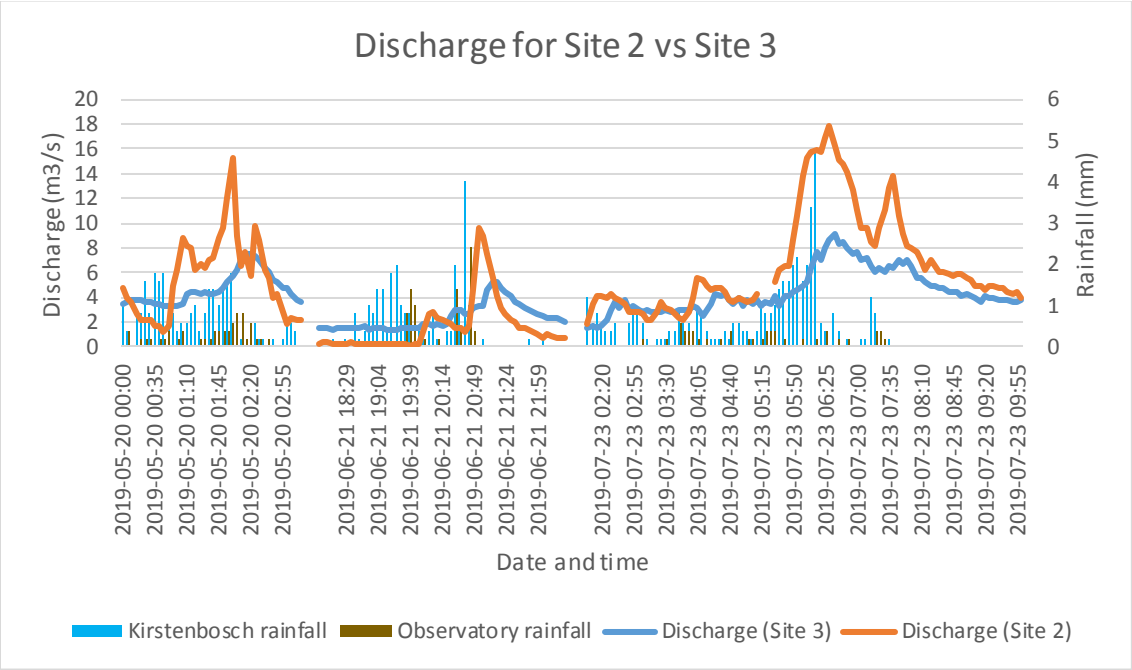

Figure 24: Discharge at Site 2 vs Site 3 for three rainfall events.

The highest rainfall event, measured at the Kirstenbosch rain gauge site (19-20th May 2019), did not coincide with the highest peak discharge (Figure 25 and 26) during the study. The most probable reason is that it occurred at the beginning of the rainfall season and therefore soils were drier, resulting in large amounts of infiltration before producing runoff. Some support for this explanation was found in a study by Beighley *et al.* (2003) who observed catchments can absorb rainfall at the start of the rainfall season before producing a hydrological response, and then as the rainy season continues, there is an increase in runoff per unit of rainfall. In the case of the Liesbeek the next recorded rainfall event (4-5th June 2019) produced the highest peak discharges (Figure 27 and 28) at all sampling sites. This rainfall event was the third highest recorded rainfall for discharge at Kirstenbosch (71.6mm) and the highest rainfall at Observatory (44.4mm).

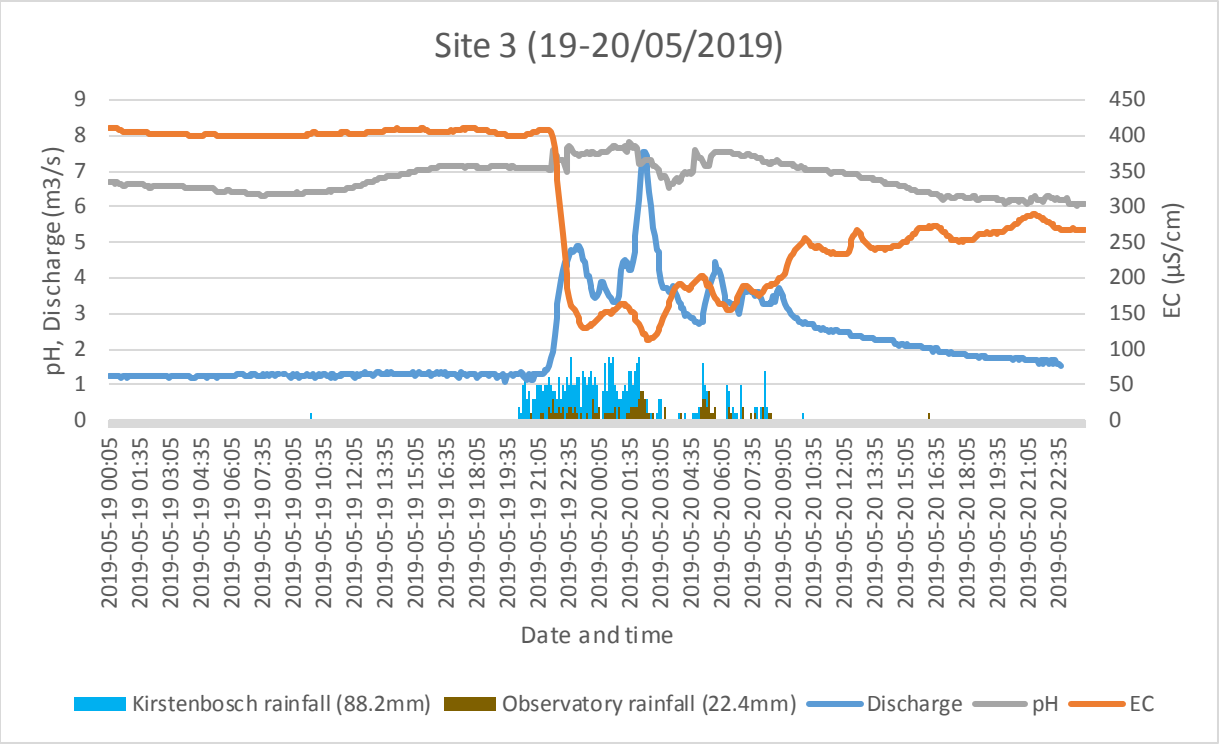

Figure 25: Changes in discharge and water quality at Site 3 for the highest rainfall recorded at Kirstenbosch (19-20/05/2019).

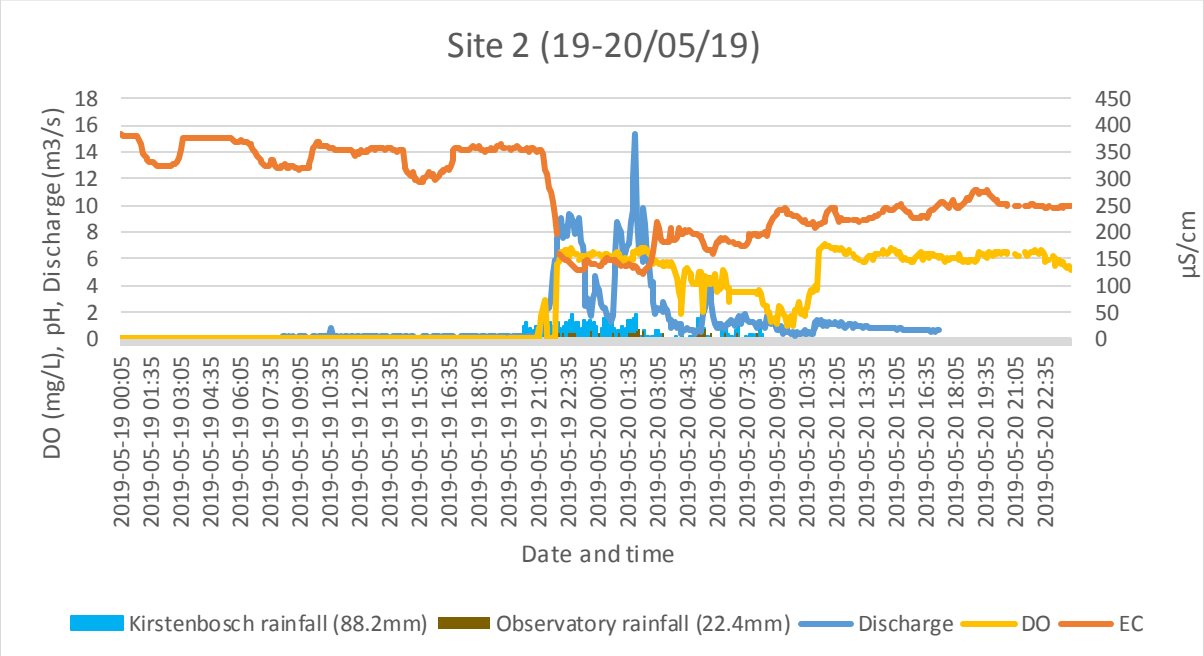

Figure 26: Changes in discharge and water quality at Site 2 for the highest rainfall event recorded at Kirstenbosch (19-20/05/2019).

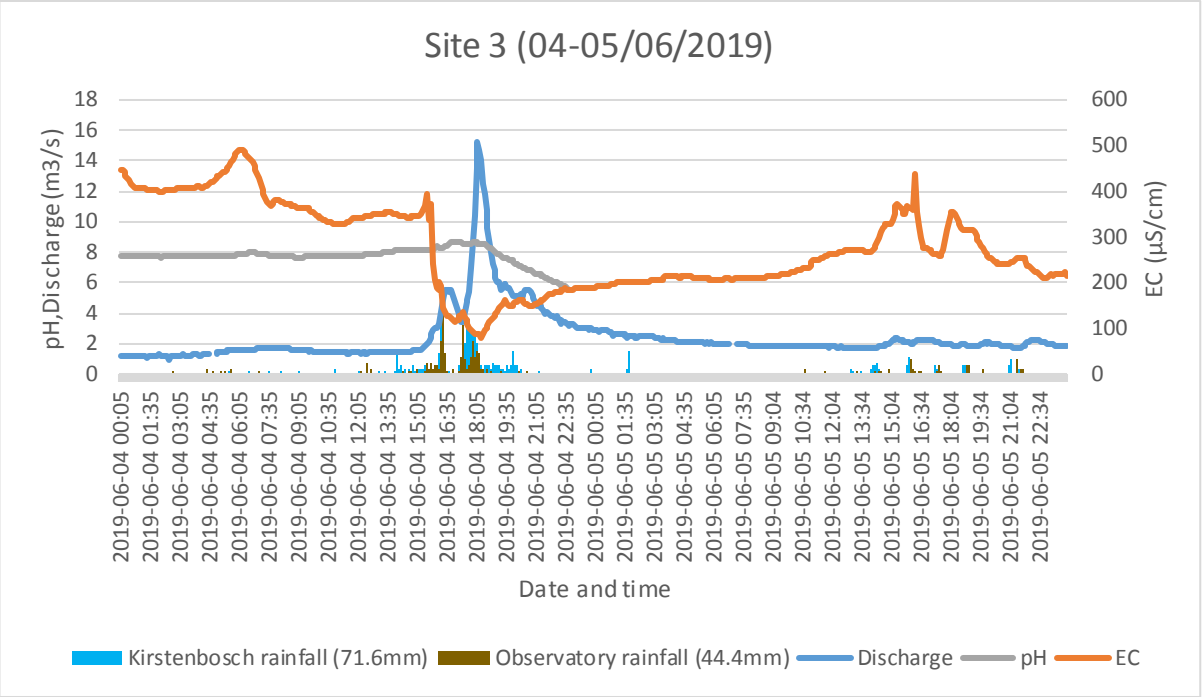

Figure 27: Changes in discharge and water quality at Site 3 for the highest rainfall event recorded at Observatory (04-05/06/2019).

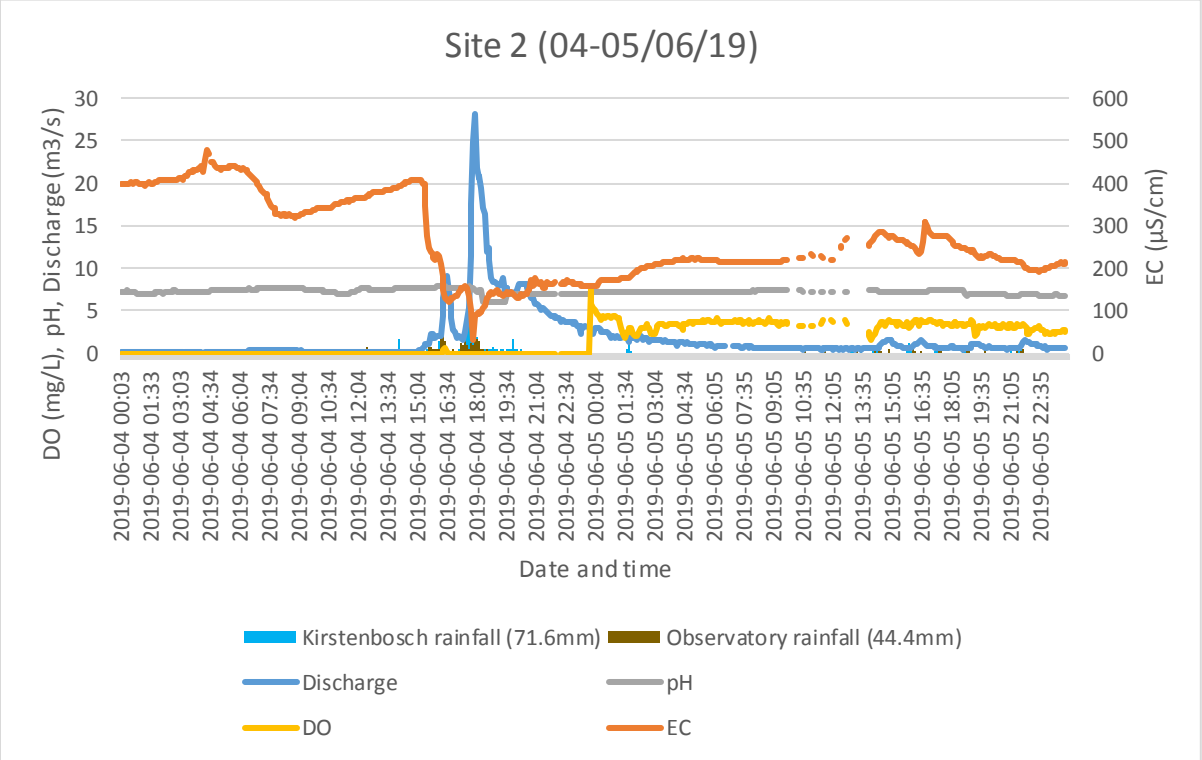

Figure 28: Changes in discharge and water quality at Site 2 for the highest recorded rainfall event at Observatory (04-05/06/2019).

4.1.1.2 Electrical Conductivity

According to Table 5, Site 1 had the lowest EC values with an average EC of 122 µS/cm compared to Site 2 and 3 that had EC averages of 244 µS/cm and 237.1 µS/cm respectively. The average EC values at all sampling sites were below the domestic use threshold according to the South African Water Quality Guidelines (SAWQG). It was also within the range that can support diverse aquatic systems (DWAF, 1996a; DWAF, 1996f) The EC values in the Liesbeek River were also much lower than the EC in the Philippi farming area on the Cape Flats Aquifer (CFA). The borehole water in the Philippi farming area ranged between 850 µS/cm and 2840 µS/cm for EC (Aza-Gnandji *et al.*, 2013).

Table 5: Average ± standard deviation (STD) and range for Electrical Conductivity data across Sites 1-3.

Electrical Conductivity		
Site	**Average ± STD**	**Range**
Site 1	122.8 ± 55.5	222.2 – 43.5
Site 2	244.1 ± 71.3	520.2 – 30.1
Site 3	237.1 ± 96.9	490 – 67.1

The largest range in EC in a single rainfall event at Sites 2 and 3 occurred on the 4-5th of June 2019 (Figure 27 and 28). This rainfall event was the highest rainfall recorded for Observatory (44.4mm). The lag times for EC were based on the time between the peak rainfalls and the lowest concentration values. The longest lag times for EC at Site 3 occurred on the 2-3rd of May 2019. This was the lowest rainfall recorded at Observatory and the second lowest recorded rainfall at Kirstenbosch for Site 3. For Site 2, the longest lag time occurred on the 29-31st of July 2019 (Figure 29). The lag time for EC at Site 1 were longer than the lag times at Site 2 and Site 3. The pattern of EC during rainfall events at all sites were that the average EC values dropped with time with the lowest EC values recorded on the last rainfall event of the data collection period. In general, the peak discharge did not coincide with the lowest EC concentration.

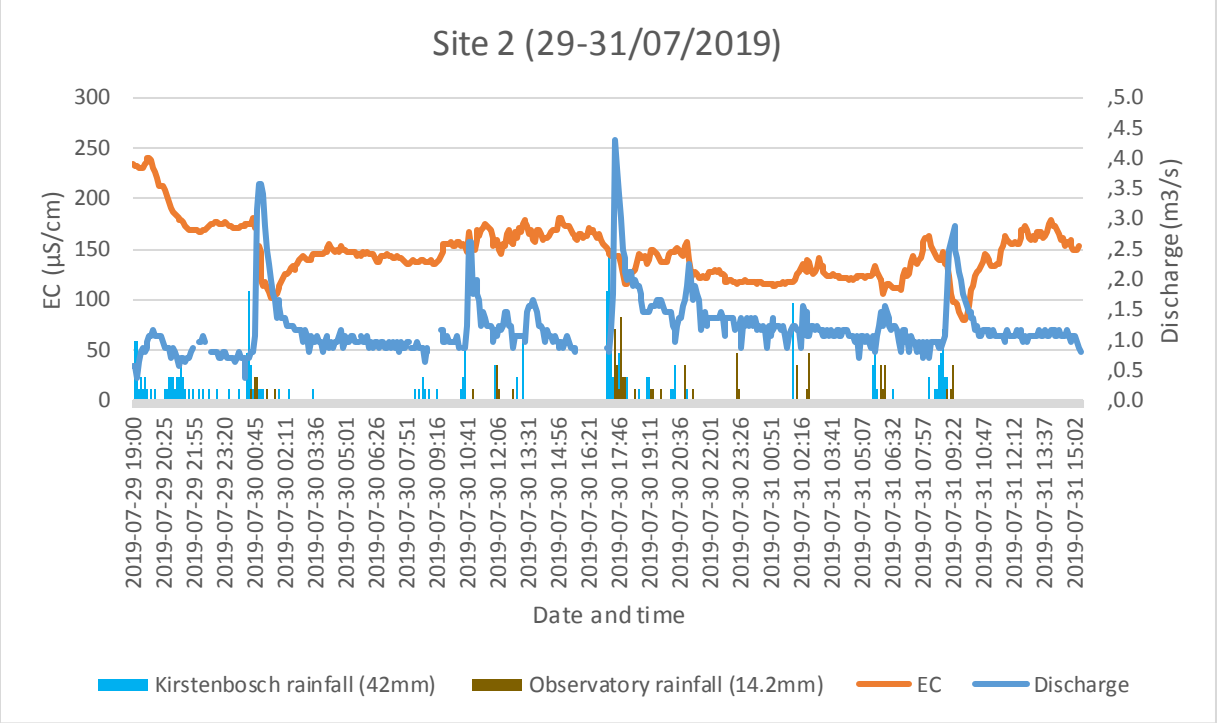

Figure 29: Changes in discharge and water quality for Site 2 during a rainfall event (29-31/07/2019).

The first flush phenomenon of increased EC at the beginning of the rainfall event was observed however, the increase in EC caused by the first flush was negligible compared to the decrease in the EC caused by dilution. Figure 30 shows an example of the first flush phenomenon at Site 3 whereby there is an increase in EC and a decrease in DO during the initial period of the rainfall event. The peak concentration for EC preceded the peak discharges. The peak discharges then preceded the minimum EC concentration. This was also observed by previous studies (Jarvie *et al.*, 2001; Li *et al.*, 2007; McGrane *et al.*, 2017). The EC values for Site 2 and Site 3 were 1.3 and 1.8 times higher respectively, in the first half of the rainy season compared to the second half. This showed that water in the Liesbeek River was more polluted in the beginning of the rainfall season. This was also observed by Lee *et al.* (2004) that analysed the stormwater discharge in California over successive wet seasons. The results indicated that pollutant concentrations in the first part of the wet season were the highest at 1.2 to 20 times higher than the pollutant concentrations towards the end of the wet season (Lee *et al.*, 2004).

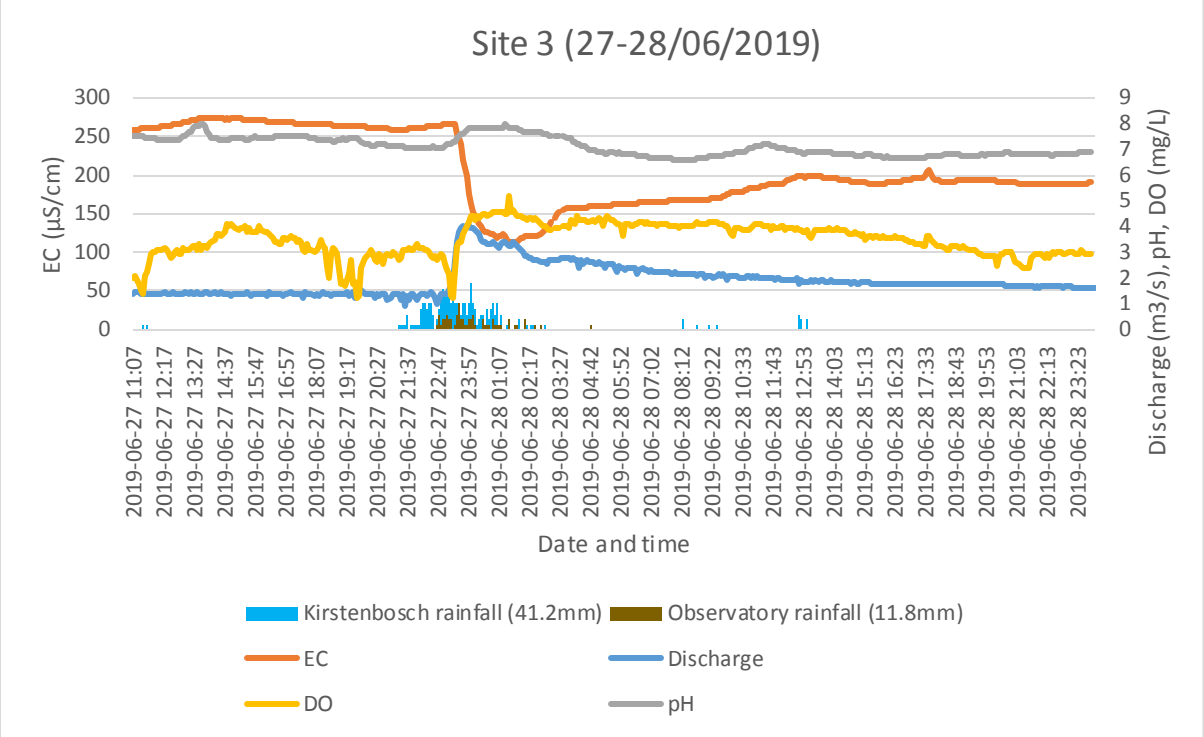

Figure 30: Changes in discharge and water quality for Site 3 during a rainfall event (27-28/06/2019).

4.1.1.3 Dissolved Oxygen

No DO data were recorded for Site 1 as the DO probe at Site 1 did not work. Low DO values are not uncommon in a river draining an urban catchment as shown by Daniel *et al.* (2002). The urbanized catchments in a study by Daniel *et al.* (2002) had average DO values of 1.8 and 2.4 mg/L. According to Chapman and Kimstach (1996), DO levels below the value of 5mg/L adversely affect the functioning and survival of aquatic communities. According to Table 6, Site 2 had a higher DO average of 6.0 mg/L compared to Site 3's 4.7 mg/L. Site 2 had a maximum DO of 10.1 mg/L whilst Site 3 had a maximum DO of 8.6 mg/L. In some cases DO values dropped close to anoxic conditions reaching values of 0.6 mg/L at Site 3 and 1.0 mg/L at Site 2. In both cases, these low DO values occurred during the beginning of the rainfall season.

Table 6: Average ± standard deviation (STD) and range for Dissolved Oxygen data across Sites 2 and 3.

Dissolved Oxygen		
Site	**Average ± STD**	**Range**
Site 2	6.0 ± 1.7	10.1 – 1.0
Site 3	4.7 ± 1.6	8.3 – 0.6

The higher DO values at Sites 2 and 3 occurred during the final stages of the study period. The DO was slightly higher during the rainfall events in July as opposed to the beginning of the rainy season. The average DO values in July were 7.1 and 5.5 mg/L for Site 2 and 3 respectively compared to before June where the average values were 5.5 and 4.4 mg/L for Site 2 and 3 respectively. The general trend was that DO increased with rainfall. This could be due to the flushing and diluting effects of rainfall, inhibiting pollutants that may deplete DO through processes such as nitrification (Jianlong and Ning, 2004). In a study performed by McGrane *et al.* (2017), it was noted that DO

experiences a moderate decrease during the rising limb of a discharge hydrograph but increases during the peak discharge (McGrane *et al.*, 2017). This was observed at most rainfall events at Site 3 (Figure 30) that recorded DO, however, it was not consistently observed during the rainfall events at Site 2.

4.1.1.4 pH

According to Table 7, Site 1 had the most acidic pH with an average of 6.6. This could be due to the rain fed streams originating from the Table Mountain which are slightly acidic due to the native fynbos coupled with the mixing of rainwater that has a pH of 5.6 (Malmqvist and Rundle, 2002). The pH increases to neutral by Site 2 which can be attributed to anthropogenic causes such as the incorporation of alkali pollutants into the stormwater. The pH at Site 3 then becomes slightly acidic again which could be attributed to the rainfall. A study by Jarvie *et al.* (2001) observed that rainfall causes a decrease in pH followed a prolonged recovery period (Jarvie *et al.*, 2001). The study by Jarvie *et al.* (2001) also observed that pH reacted more dramatically to rainfall than EC. The opposite was observed in the Liesbeek River whereby the EC was more responsive to rainfall than pH.

Table 7: Average ± standard deviation (STD) and range for pH data across Sites 1-3.

pH		
Site	**Average ± STD**	**Range**
Site 1	6.6 ± 0.5	7.4 – 5.8
Site 2	7.0 ± 0.5	8.6 – 6.0
Site 3	6.8 ± 0.67	8.7 – 5.6

The largest range in pH at Site 3 occurred on the 4-5th of June 2019 which was the highest rainfall recorded for Observatory (Figure 27). The large range of 8.7 to 5.7 could be explained by the 8 antecedent dry days which allowed for pollution build up before it got washed off during the rainfall. This was then followed by continuous dilution by rainfall resulting in a decrease in pH. The largest range in pH at Site 2 did not occur during the same rainfall event but the range was still substantial at 7.8 – 6.0. The pH at Site 2 remained above 6.0. At site 3, June experienced the highest pH values, with the values showing a drop to slightly acidic conditions in the July rainfall events.

4.1.2 Comparison between Sites 2 and 3

There were observable similarities and differences between Site 2 and Site 3 in the discharge and water quality data. A nonparametric test (Mann-Whitney U test) was performed to determine whether the differences in discharge and water quality variables between Site 2 and Site 3 were statistically significant. The Mann-Whitney U test was used because the distribution was not normal. The Mann-Whitney U tests for this study (Table 8), showed that there is enough evidence to support a statistically significant ($\alpha = 0.05$) difference between Site 2 and Site 3 for all parameters.

The average discharge for Site 3 is almost double that of Site 2. During periods of no rainfall, the discharge at Site 2 drops to as low as 0.07 m^3/s making it close to 20 times lower than the discharge at Site 3. The means for EC at Site 2 and 3 were similar, however, the Mann-Whitney U test showed that they are significantly different because the Mann-Whitney U test is rank-based and compares the median values rather than the means.

Table 8: Mann-Whitney U test for water quality variables and discharge

EC	
Mann-Whitney U	14758028
Significance (2-tailed)	2.05×10^{-11}
pH	
Mann-Whitney U	731987.5
Significance (2-tailed)	4.09×10^{-58}
DO	
Mann-Whitney U	1542837.5
Significance (2-tailed)	7.02×10^{-124}
Discharge	
Mann-Whitney U	1936913
Significance (2-tailed)	1.0×10^{-200}

Concluding remarks

The Liesbeek River demonstrated the characteristics of a typical urban river showing flashy hydrographs with steep rising limbs, particularly in the more urbanized sections of the river compared to upstream. Rainfall resulted in significant responses in all variables with significant levels of dilution. The rainfall improved the water quality after the peak discharge and peak rainfall. The improved levels of water quality was greater during the higher rainfall events, owing to the dilution factor. The results also provided clues in understanding the behaviour of the river in terms of its discharge and water quality that would help in determining the suitable range for stormwater abstraction.

4.2 Relationship between rainfall, discharge and water quality

4.2.1 Correlation analysis

It was observed that an increase in rainfall and discharge, resulted in a decrease in EC and an increase in DO. However, an increase in rainfall initially resulted in an increase in EC and decrease in DO (first flush phenomenon), before being taken over by the dilution factor, causing a decrease in EC and an increase in DO. To investigate these relationships, correlation analyses were performed in order to test the strength of the relationship between the variables. Spearman rank-order correlations (Spearman *R* coefficient) were used to study the correlation structure between variables as the data showed abnormal distribution of water quality parameters (Wunderlin *et al.* 2001). Table 9 provided the correlation matrix for water quality variables, discharge and rainfall for Sites 2 and 3. Temperature had a significantly strong positive correlation with EC for Sites 2 (0.68) and 3 (0.60). EC and discharge had significantly strong negative correlations for Sites 2 (-0.76) and 3 (-0.62). EC was also the most responsive variable to changes in discharge. DO and pH had a correlation of 0.34 at Site 2, however, the correlation was 0.003 at Site 3. This indicated a very weak relationship existed between pH and DO. Discharge had a positive correlation of 0.44 with DO at Site 3, whilst that correlation was only 0.18 at Site 2. Discharge at Site 2 had a stronger positive correlation with rainfall than at Site 3. This can be expected due to Site 2's quicker response to rainfall resulting in its steeper rising and falling limbs on the hydrographs.

The weak correlation between rainfall and the other parameters is an indication that rainfall itself does not fully explain the variation in water quality in the Liesbeek River. Other factors which has been included for correlation analysis in other studies include antecedent dry days, first flush strength, rainfall duration and intensity, land use, impervious area and catchment area (Gupta and

Saul, 1996; Lee *et al.*, 2002; Soller *et al.*, 2005; Han *et al.*, 2006; Kang *et al.*, 2008; He *et al.*, 2010; Hathaway *et al.*, 2012).

Table 9: Spearman's *R* correlation analyses for Site 2 and 3.

Spearman *R* correlation analysis for site 3							
Variables	Temperature	pH	EC	DO	Discharge	Kirstenbosch rainfall	Observatory Rainfall
Temperature	1						
pH	0.009	1					
EC	0.60**	-0.13**	1				
DO	0.18**	0.003	-0.61**	1			
Discharge	-0.04*	-0.13**	-0.62**	0.44**	1		
Kirstenbosch rainfall	-0.40**	0.23**	-0.14**	0.10**	0.18**	1	
Observatory rainfall	0.018	0.19**	-0.08**	0.10**	0.24**	0.38**	1
Spearman *R* correlation analysis for site 2							
	Temperature	pH	EC	DO	Discharge	Kirstenbosch rainfall	Observatory rainfall
Temperature	1						
pH	N/A	1					
EC	0.68**	0.12**	1				
DO	0.14*	0.34**	-0.33**	1			
Discharge	-0.67**	-0.12**	-0.76**	0.18**	1		
Kirstenbosch rainfall	-0.22**	0.009	-0.14**	0.06*	0.22**	1	
Observatory rainfall	-0.19**	0.21**	-0.17**	0.007	0.27**	0.38**	1

* = Correlation is significant at the 0.05 level (2-tailed)

** = Correlation is significant at the 0.01 level (2-tailed)

Further analysis (Table 10) were performed to investigate the relationship between water quality and rainfall with 30 and 60-minute lag times. At both sites the correlations between rainfall and water quality becomes weaker with an increase in the lag time. The only exception is DO at Site 2 whereby, after the 30-minute lag time in rainfall the positive correlation increased. The general decrease in correlations for water quality and rainfall as the lag time increases indicates that there is a quick rainfall to runoff response in both sites with the strongest correlations occurring without a lag, indicating that changes in the characteristics of the discharge and water quality occur in conjunction with the rainfall rather than a delayed response.

Table 10: Spearman's *R* correlation analyses for water quality and discharge with lagged intervals in rainfall.

Site 2						
	Rainfall		**Rainfall 30min lag**		**Rainfall 60min lag**	
	Kirstenbosch	Observatory	Kirstenbosch	Observatory	Kirstenbosch	Observatory
Temperature	-0.22**	-0.19**	-0.19**	-0.18**	-0.18**	-0.17**
pH	0.009	0.21**	-0.002	0.21**	0.006	0.20**
EC	-0.14**	-0.17**	-0.08**	-0.09**	-0.05**	-0.06**
DO	0.06*	0.007	0.04	-0.02	0.03	-0.02
Discharge	0.22**	0.27**	0.14**	0.18**	0.09**	0.11**
Site 3						
	Rainfall		**Rainfall 30min lag**		**Rainfall 60min lag**	
	Kirstenbosch	Observatory	Kirstenbosch	Observatory	Kirstenbosch	Observatory
Temperature	-0.40**	0.02	-0.35*	0.02	-0.31*	0.025
pH	0.23**	0.19**	0.22**	0.17**	0.21**	0.16**
EC	-0.14**	-0.08**	-0.12**	-0.06**	-0.09**	-0.04**
DO	0.10*	0.10*	0.069**	0.079**	0.05**	0.06**
Discharge	0.18**	0.24**	0.13**	0.16**	0.09**	0.12**

* = Correlation is significant at the 0.05 level (2-tailed)

** = Correlation is significant at the 0.01 level (2-tailed)

Table 11: Spearman's *R* correlation analysis for sites 1-3 with a 15 and 30-minute lag time for Site 2.

Spearman *R* for discharge between sites 1-3					
Discharge	Site 3	Site 1	Site 2	Site 2 with 15min lag	Site 2 with 30min lag
Site 3	1	0.90**	0.90**	0.93**	0.94**
Site 1	0.90**	1	0.92**	0.89**	0.87**

Furthermore, correlation analysis (Table 11) was performed to investigate the relationship in discharge across the three sites with 15 and 30-minute lag times for Site 2. Site 2 was chosen to have the lag times given its relatively quicker rainfall to runoff response. All sites had significantly strong relationships (0.90**) with each other with respect to discharge. The discharge relationship between Site 1 and 2 decreases as the lag increases. This means that the changes in their discharge is closely matched, reaching their peak discharge roughly at the same time. Despite Site 2 being more impervious compared to Site 1, Site 1 was situated close to Kirstenbosch where the rainfall arrives earlier than at Observatory and also approximately three times the volume. This could be the reason why the discharges coincide without a lag. The discharge relationship between Site 2 and Site 3 increases from 0.90 to 0.94 with a 30-minute lag for Site 2. This indicates that Site 3 reaches its peak discharge approximately between 15-30-minutes after Site 2.

Multivariate analysis was performed in the form of Principal Component Analysis (PCA). Results from the PCA (Appendix B) indicated that most of the variation in the water is explained by the combination of discharge, DO and EC with substantial contributions from rainfall and minor

contributions from pH and temperature. The results from the PCA also reinforces the findings from the correlation analysis that rainfall itself does not fully explain the variation in water quality in the Liesbeek River.

Concluding remarks

The relationships between rainfall, discharge and water quality were established using descriptive, bivariate and multivariate analysis. Rainfall influences all hydrological processes (Niemczynowicz, 1999) but rainfall alone does not fully explain the variations in discharge and water quality. This was evident by the weak correlations between rainfall and the water quality parameters. This was further indicated by the PCA analysis whereby discharge, EC and DO were considered as the main contributors.

The general trend amongst all sites were that as discharge increases, EC and pH decreases, whilst DO increases. These trends were confirmed by the correlation analysis. Discharge had the quickest response to rainfall than the other parameters, therefore, the peak discharge preceded the minimum EC and maximum DO concentrations. This becomes important when evaluating the suitable range for stormwater abstraction during rainfall events which will be further discussed in the next section.

4.3 Suitable range for stormwater abstraction and potential for aquifer recharge

This study has established that the peak discharge and minimum EC concentration do not coincide directly with a rainfall event. It has also been established that the peak discharge precedes the minimum EC concentration. This means that the highest volume and improved water quality occurs slightly after the peak discharge. The water quality for Sites 1-3 were benchmarked against the South African Water Quality Guidelines (SAWQG) (DWAF, 1996a-f), US EPA, and various other studies to determine the suitability of the water quality. The volume and water quality of the rising and falling limb were compared to determine the suitable range for stormwater abstraction in the Liesbeek River. The suitable range for stormwater abstraction was based on the various SAWQG.

The potential for aquifer recharge on the CFA was explored using a study performed by Mauck (2017) showing the aquifer recharge potential and flow direction. The water quality was also compared against the water quality in the CFA, in specific, the groundwater in the Philippi farming area. The reason for this location in the CFA is because the aquifer is used for agricultural purposes, with its crops intensively irrigated with groundwater drawn from the CFA. In the past the CFA has been affected by salinity issues (directly related to the EC content). Moreover, the CFA also presents an opportunity as an important resource that could potentially be used to augment municipal water supply in times of drought (Aza-Gnandji *et al.*, 2013).

4.3.1 Water quality guidelines

The fitness of water quality for various uses require different criteria (Cordoba *et al.*, 2010) and is discussed further in this section. For this study, the chemical and physical variables investigated for water quality included pH, EC, DO and temperature. The SAWQG were developed by the Department of Water Affairs and Forestry (DWAF). These guidelines were used as the primary source of judging the fitness of water for various uses based on its physical, chemical, biological and aesthetic properties. The quality criteria consists of a Target Water Quality Range (TWQR) which describes the range at which a variable would have no known adverse effects on the suitability of the water when used continually.

4.3.1.1 Electrical Conductivity

According to the SAWQG for EC (Appendix C, Table C1), the EC ranges at all the sampling sites fall within the target of 0-700 µS/cm for domestic water use (DWAF, 1996a). The EC values, with the exception of one event at Site 2, meets the criteria (<500 µS/cm) to support diverse aquatic organisms (Behar *et al.*, 1996). Close to 50% of the EC values at both sites were within the range of 150-300 µS/cm which is capable of causing minor damage as a result of scaling and corrosion to equipment for industrial uses.

The quality of water for irrigation is determined by its effects on soil and plants. Water that is too saline can damage salt sensitive crops. In total 4 out of 12 and 7 out of 14 rainfall events at Site 2 and Site 3 respectively, had values above 400 µS/cm which is regarded as the limit whereby no damage to salt sensitive crops will occur for irrigation purposes. In all events however, the rainfall diluted the EC values to below 400 µS/cm (DWAF, 1996d). The observations above 400 µS/cm occurred in the first half of the rainy season. Only 2.4% and 8.8% of the data at Site 2 and Site 3 respectively, were above 400 µS/cm. According to a study performed by Aza-Gnandji *et al.* (2013), the EC values in the Philippi farming area of the CFA ranged between 850 µS/cm and 2840 µS/cm for borehole water and between 990 µS/cm and 2840 µS/cm for pond water in the months of February, April, June and August. None of these values are below the threshold for the SAWQG on irrigation (400 µS/cm). Barring one rainfall event at Site 2 (4th of March 2019) and Site 3 (7th of March 2019), the average EC values in the Liesbeek River during rainfall events were below the SAWQG for irrigation. The rainfall events on the 4th and 7th of March 2019 were the lowest recorded rainfall depth for each site during the study and therefore did not cause enough dilution to lower the EC to applicable standards. The minimum EC values however, for all rainfall events at both sites were below the threshold for the SAWQG on irrigation. This implies that not only is the stormwater in the Liesbeek River lower than that of the Philippi farming area, but it is also below the threshold of the SAWQG for irrigation. It is therefore a plausible scenario that abstracted stormwater from the Liesbeek River during rainfall events could be used to replenish the aquifer with lower EC water.

Figure 31 displays the EC for Site 2 and Site 3 throughout the study with the various water quality thresholds. The graph show that the EC stays below the domestic use and aquatic systems thresholds except for one event at Site 2 that was slightly above 500 µS/cm. The graph also showed that most of the EC values were below 400 µS/cm. It is also interesting to note the gradual decrease in EC as time continued showing lower EC values towards the end of the study. This gradual decrease in EC can be attributed to the dilution of rainfall with minimal antecedent dry days in between.

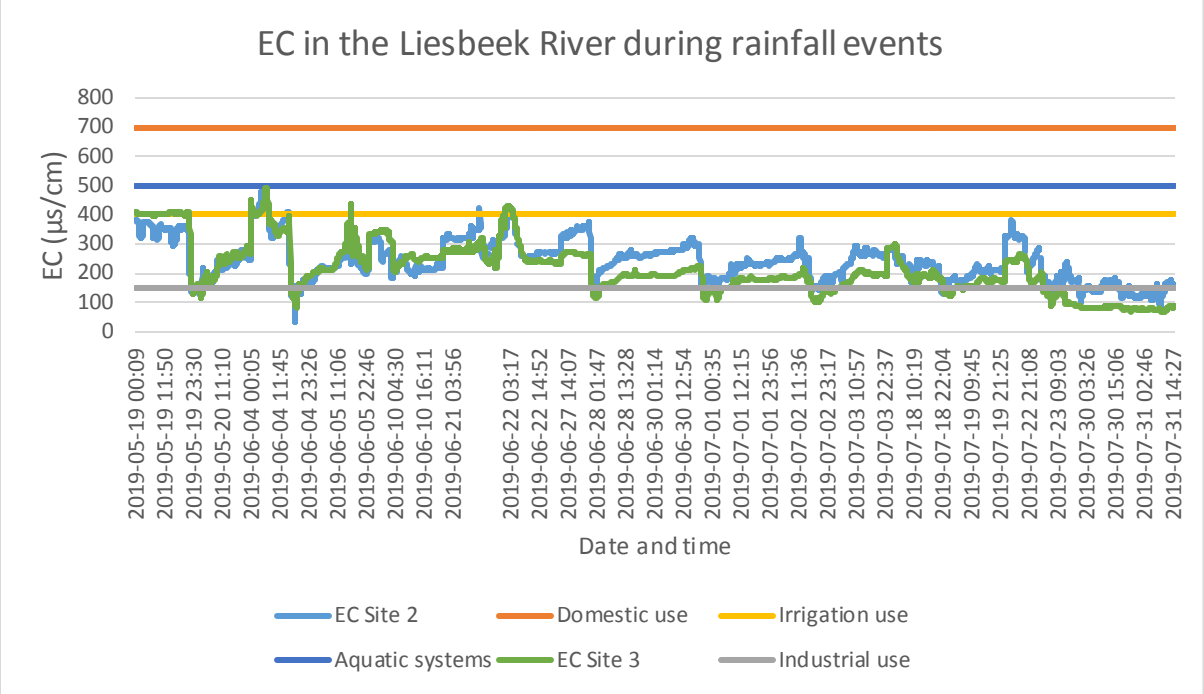

Figure 31: EC in the Liesbeek River during rainfall events.

4.3.1.2 DO

For Site 3, 35% of its DO values were below 4 mg/L and 18% of DO values for Site 2 was below 4 mg/L. Chapman and Kimstach (1996) stated that DO levels below the value of 5 mg/L affects the functioning and survival of aquatic communities. According to the SAWQG for DO (Appendix C, Table C2), a range of between 5-9 mg/L for DO can accommodate cold, intermediate and warm water species. Based on the range given by SAWQG, 61% of the data from Site 3 and 22% of the data from Site 2 are below 5mg/L. With an average DO of 6mg/L, the water at Site 2 is more suitable for aquatic systems than at Site 3. Figure 32 displays the DO for Site 2 and Site 3 throughout the data collection period with the range for aquatic systems marked on the graph. Most of the DO at Site 3 was below the target range for aquatic systems for the first half of the rainy season. The second half of the rainy season saw a gradual increase in DO that fell within the target range. The gradual increase in DO throughout the rainy season can be attributed to the continuous dilution from the rainfall events and flushing out of pollutants.

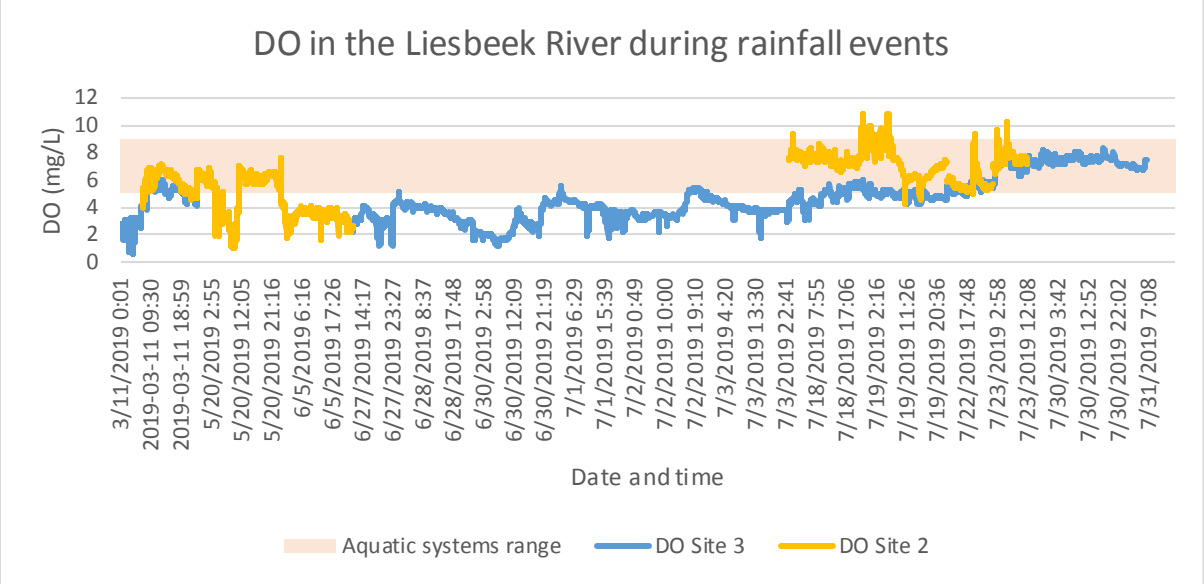

Figure 32: DO in the Liesbeek River during rainfall events.

4.3.1.3 pH

According to the SAWQG for pH (Appendix C, Table C3), most of the data meets the criteria for domestic and recreational uses and aquatic systems with only 1% of the data above 8.5, and 9% below 6.0 at Site 3. The situation improves at Site 2 with no values below 6.0 and only 0.2% of pH above 8.5. At Site 1 no values were above 8.0 and only 13% of the data were below 6.0. Most aquatic species can survive in a riverine system with a pH range of 5.0 to 9.0. However, the optimal pH of water in the river for aquatic organisms is from 6.5 to 8.5 (Al-Asadi, 2016). For industrial use criteria, a range of 7.0-8.0 causes little to no tendency of corrosion and scaling. Only around 50% of the data meets this criterion. Most of the data meets the criteria for irrigation purposes (6.5-8.4) with 35% of the data falling out of this range at Site 3. The situation improves at Site 2 with only 13% of the date falling out of the range. According to a study performed by Aza-Gnandji *et al.* (2013), the pH values in the Philippi farming area of the CFA ranged between 6.6 and 7.8 for the borehole water. In the ponds however, some of the values went above 8.4. The pH in the Liesbeek River falls within a similar range as the Philippi farm area on the CFA and therefore using the abstracted water from the river for aquifer recharge will pose no serious threat with regards to pH levels.

Figure 33 displays the pH for Site 2 and Site 3 throughout the data collection period with the range for domestic use and aquatic systems marked on the graph. Almost all the data fell within the target ranges for domestic use and aquatic systems as mentioned previously.

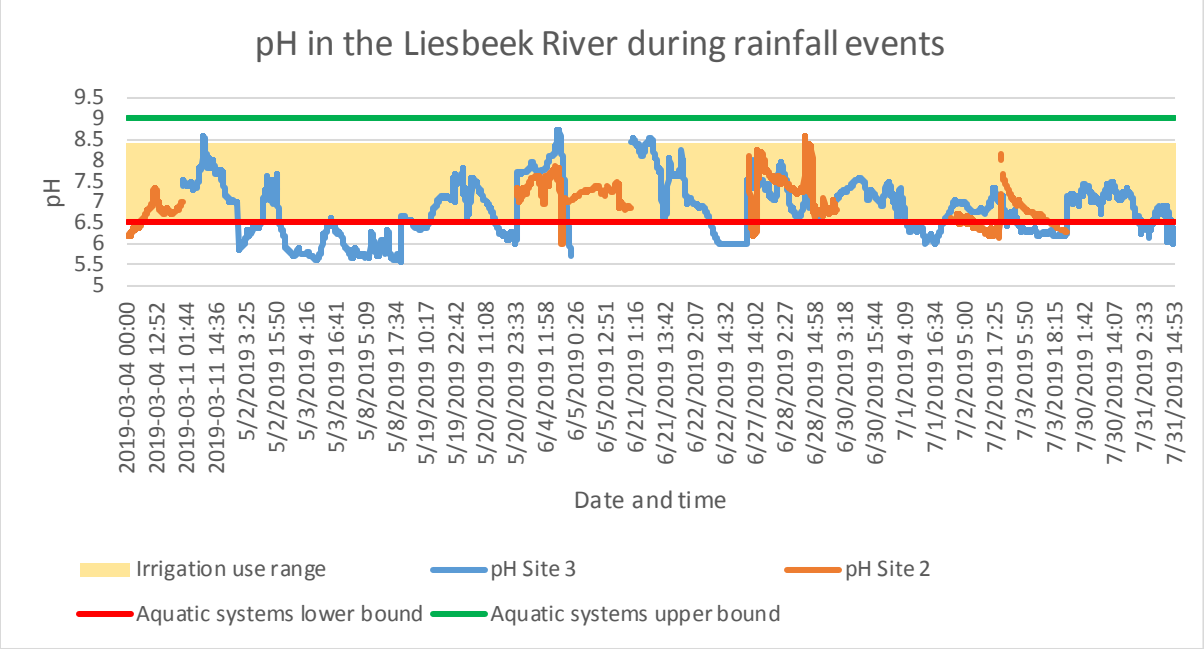

Figure 33: pH in the Liesbeek River during rainfall events.

4.3.2 Falling limb vs rising limb

It has been established that the peak discharge does not coincide with the peak DO and minimum EC concentration. This means that the highest available volume of stormwater does not necessarily coincide with the best water quality during a rainfall event. Therefore, the volume and water quality for the rising limb and falling limbs for each hydrograph at Site 2 (12a) and Site 3 (12b) were investigated and compared. The volume of water is the total amount of water measured in cubic metres that passes over time.

4.3.2.1 Site 2 (Table 12a)

In total, 7 out of the 8 rainfall events recorded for discharge had larger volumes during the falling limb except for the 19-20th May 2019 rainfall event. This was the first major rainfall event of the rainy season and produced the highest rainfall depth in Kirstenbosch at 88.2mm. The volume for the rising limb was 19194 m^3 as opposed to the falling limb's volume of 18979 m^3. 7 out of the 8 rainfall had lower EC averages for the falling limb. The one exception was the 22-23rd July 2019 rainfall event. The reason for this could be due to the light rain that was present for half a day before finally reaching the peak rainfall. This may have led to a continuous dilution of EC and therefore reached its minimum EC slightly sooner than expected. This same rainfall event also had a higher DO average during the rising limb which can be attributed to the continuous dilution effect and flushing out pollutants, therefore reaching its peak DO slightly sooner than expected. The average DO during the falling limb was above 5mg/L and therefore was in the range to support aquatic systems. 6 out of the 8 falling limbs had EC averages of below 200 µS/cm which is below the domestic use, aquatic and irrigation use limits for EC and within the industrial use range that will only cause minor scaling and corrosion. The EC average during the falling limb, for example was up to 10 times lower than the EC values in the Philippi farming area on the CFA which is a horticultural area supplying vegetables to Cape Town (Aza-Gnandji *et al.*, 2013).

Table 12a: Comparison in discharge and water quality between falling and rising limbs for Site 2.

Site 2										
Date	Rainfall		Total volume (m^3)		EC average (µS/cm)		DO average (mg/L)		pH average	
	K	O	Rising	Falling	Rising	Falling	Rising	Falling	Rising	Falling
19-20/05	88.2	22.4	19194	18979	136	129	5.9	6.5		
4-5/06	71.6	44.4	22472	62906	128	124			7.6	6.3
9-10/06	17.8	9.2	1082	3831	260	186				
21-22/06	56.6	20.8	7601	12877	300	235				
27-28/06	41.2	11.8	5272	18198	265	177			7.5	7.5
18-19/07	30.2	4	5249	8212	181	130	5.8	7.2		
22-23/07	74.2	13.4	35789	35958	120	136	9	7.9		
29-31/07	42	14.2	2720	3648	249	247				

4.3.2.2 Site 3 (Table 12b)

In total, 9 out of the 10 rainfall events had larger volumes during the falling limb except for the 2-3rd July 2019 rainfall event. This was the second largest rainfall event recorded at Kirstenbosch at 85.2mm. A total 8 out of the 10 rainfall events had lower EC averages during the falling limb. The two exceptions were 4-5th June and 2-3rd July 2019. The 2-3rd July 2019 rainfall event had the minimum EC concentration before the peak discharge. This was also the rainfall event that had a higher volume during the rising limb and therefore the lower EC could be due to the dilution and flushing out of pollutants during the high voluminous period of the rising limb. The DO during June had higher DO averages in the falling limb however, for all the events in July, the DO was higher in the rising limb during the month of July. All EC averages during the falling limb were below the domestic, aquatic and irrigation use thresholds. The EC was also up to 10 times lower than that of the EC in the Philippi farming area.

Table 12b: Comparison in discharge and water quality between falling and rising limbs for Site 3.

Site 3										
Date	Rainfall		Total volume (m^3)		EC µS/cm		DO (mg/L)		pH	
	K	O	Rising	Falling	Rising	Falling	Rising	Falling	Rising	Falling
19-20/05	88.2	22.4	16478	17610	149	122			7.5	7.1
4-5/06	71.6	44.4	24711	38519	112	115				
9-10/06	17.8	9.2	11581	14790	313	216				
21-22/06	56.6	20.8	10533	12490	252	229			7.9	7.4
27-28/06	41.2	11.8	7343	18929	260	130	2.5	4.4	7.2	7.8
30/06-1/07	55.6	17.8	8016	20227	141	115	4.5	4.9	6.7	7.1
2-3/07	85.2	6.6	47065	39619	108	128	5.3	5.0	7.0	6.7
18-19/07	30.2	4	6902	10858	182	138	5.6	5.3		
22-23/07	74.2	13.4	17071	21975	117	99	7.7	7.2		
29-31/07	42	14.2	5337	9851	79	74	7.8	7.3	7.2	7.0

4.3.3 Potential for Cape Flats Aquifer recharge.

A study by Mauck (2017) modelled and mapped the capacity of the CFA and indicated that the highest potential for aquifer recharge occurred in areas that have the highest topographic elevation, such as in the Table Mountain region (Figure 34). Figure 34 shows the high potential of recharge of the CFA around the Table Mountain region in the western side of the CFA. This is also the same region in which the Liesbeek Catchment occurs. There are higher groundwater elevations in the western parts of the CFA model around the Table Mountain region and therefore higher rates of groundwater flow. The topography and groundwater levels around the Table Mountain regions of the CFA allows water to flow towards the east therefore recharging the central regions of the CFA.

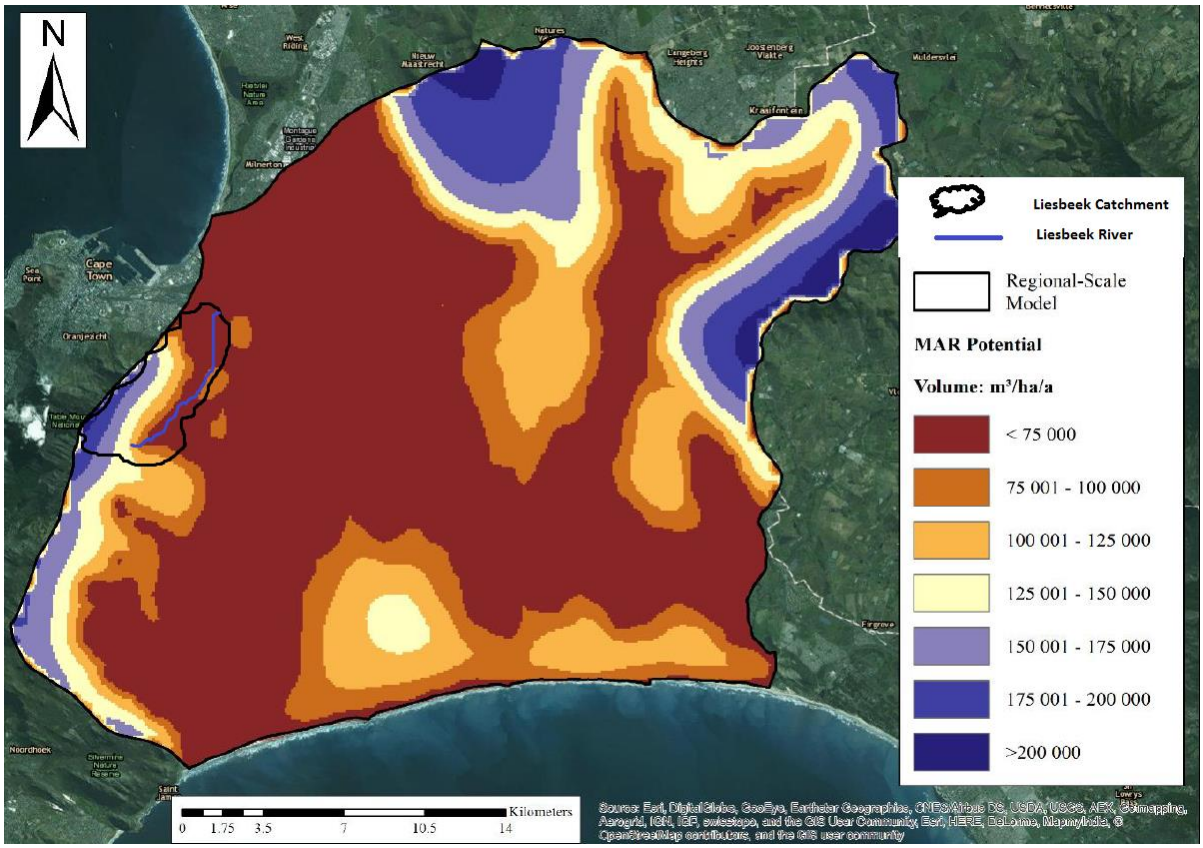

Figure 34: Managed Aquifer Recharge potential of the CFA (Adapted from Mauck, 2017).

Section 4.3 has shown that the EC in the Liesbeek River was within the domestic and aquatic systems threshold. It was not always suitable for irrigation purposes, however, once it rained, the water is sufficiently diluted to meet the irrigation limits. The EC averages for both sites in the falling limb were all below 200 µS/cm except for two events at each site. The EC averages for the falling limb were low enough for domestic, irrigation, aquatic and industrial use. These values were also lower than the EC values in the Philippi farming area on the CFA, and in some cases, up to 10 times lower. This shows the relatively 'clean' nature of the falling limb's stormwater with respect to its EC in the Liesbeek River and creates a plausible scenario of potentially using that stormwater for aquifer recharge in the CFA.

The DO values at Site 3, were not suitable for aquatic systems during the beginning of the rainfall. As the rainy season progressed, the DO values increased to within the suitable range for aquatic systems. The DO averages all fell within the aquatic systems range during the falling limb except for the two rainfall events in June at Site 3. The pH averages during the falling limb fell within the

domestic use and aquatic systems ranges except for one event at Site 2 which had an average pH of 6.3. After comparing the rising and falling limb for both sites, it was observed that the falling limb has a higher volume and better water quality. This implies that the falling limb is more suitable for stormwater abstraction in terms of water quality and volume.

This type of information was only made possible due to the high-resolution data that monitored discharge, water quality and rainfall in five-minute intervals. This provided a deeper understanding of the changes in discharge and water quality in response to rainfall and how water quality varies with discharge during rainfall events.

4.4 Modelling

4.4.1 The Liesbeek Catchment model

4.4.1.1 Sub-catchment delineation

145 Sub-catchments in the Liesbeek Catchment were delineated manually based on the stormwater pipes as well as a 2m contour file. Areas that were not urbanized (e.g on the mountain) had bigger catchments (15-160 ha) and their boundaries were dictated by the 2m contour lines. The urbanized areas had smaller sub-catchments (1-15 ha) and their areas were dictated by the stormwater pipes and road network. The calculation of imperviousness was based on typical land use characteristics and so did not necessarily represent local conditions. According to Fisher-Jeffes (2015) this is acceptable for high level planning such as stormwater harvesting, rather than rainwater harvesting which requires details on a much finer resolution. The attributes and input parameters for each land use was obtained from GIS and in some cases estimated based on recommendations from Rossman (2008) such as Manning's *n* and depth of depression storage. Using PCSWMM's area weighting tool, weighted averages of the attributes and input parameters were calculated for each sub-catchment.

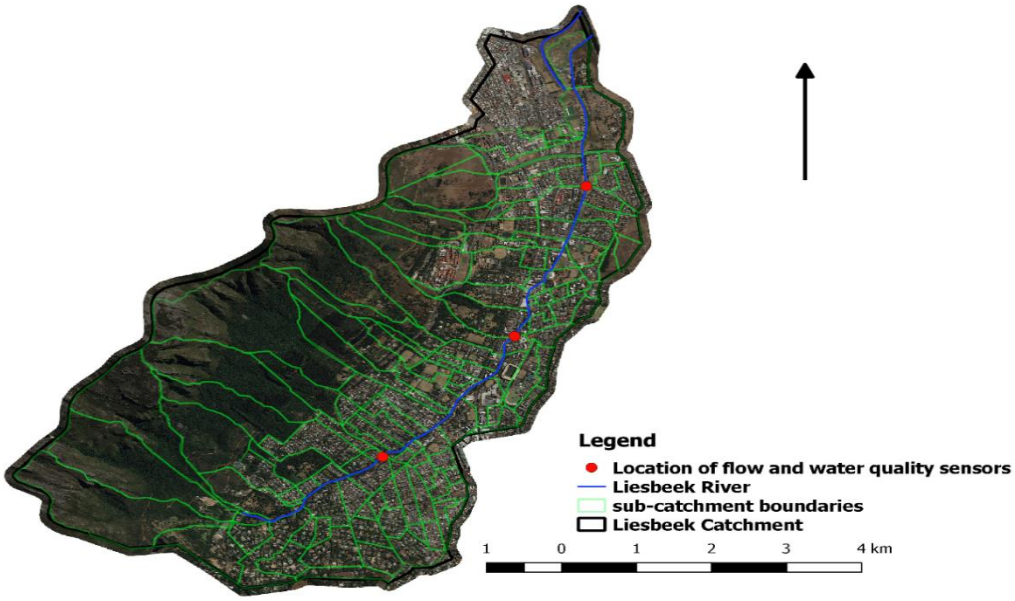

Figure 35: Delineated sub-catchments in the Liesbeek Catchment.

4.4.1.2 Model sensitivity

A sensitivity analyses was performed on the model in order to determine the sensitivity of the input parameters. PCSWMM has a built in sensitivity analysis called the Sensitivity-Based Radio Tuning Calibration (SRTC) tool. This tool uses the factor perturbation method which involves keeping all the

parameters at a fixed value whilst varying a single factor. The sensitivity analyses did not include parameters with a high degree of certainty such as area.

Parameters tested for their sensitivity were flow length, slope, imperviousness, Manning's N, depression storage (D) for impervious and pervious surfaces, and percentage of impervious surfaces that has no depression storage. Green-Ampt parameters were also tested. This included conductivity, suction head, and initial deficit. The sensitivity analyses was used as a means to determine the most sensitive parameters and therefore used for calibration. For the sensitivity analyses, discharge rate was used as a proxy to test the parameters. The sensitivity analyses took place at Site 3 for two storm events recorded from 2018. The first storm event experienced 7.8mm rainfall in Kirstenbosch and 0.6mm in Observatory. This event did not produce a hydrograph and thus all parameters rendered insensitive. The second rainfall event experienced 33.4mm rainfall in Kirstenbosch and 13.6mm in Observatory. The model was most sensitive to changes (in order): Imperviousness, flow length, Manning's N for imperviousness, Depression storage (impervious) and slope. The model was least sensitive to changes (in order): Manning's N for perviousness, Depression Storage (pervious), Green-Ampt parameters (Conductivity, initial deficit, suction head) and percentage of the impervious area that has no depression storage.

4.4.1.3 Model calibration and validation

It is essential for models to be calibrated to ensure its reliability (James, 2005). For the model calibration, 2/3 of the rainfall events were used for calibration and 1/3 used for validation. This resulted in 7 out of the 11 rainfall events used for calibration and 4 rainfall events used for validation.

Calibration was undertaken at Site 3 in the Liesbeek Catchment. Discharge for 11 rainfall events were recorded in 2018. From the 11 rainfall events recorded, 7 events were used for calibration and 4 used for validation. For the calibration, a continuous simulation was run which included seven days preceding the rainfall event to reduce the impacts associated with the assumptions of antecedent conditions. The same parameters as the sensitivity analyses were used, whereby the most sensitive parameters were the main focus for calibration. The SRTC tool was used for the calibration. This tool allows the user to toggle buttons to assess the impacts that changes in parameters will have. Discharge was modelled and compared to observed discharge data. The performance of the model was assessed by three functions, namely; Integrated Squared Error (ISE), Nash-Sutcliffe Efficiency (NSE) and coefficient of determination (R^2). ISE measures the difference between the modelled and observed data, therefore, the lower the difference (closer to zero) the better the correlation between the modelled and observed discharge. NSE values vary from $-\infty$ to 1. If the NSE value is less than zero, it means that using the mean discharge value of the observed data would be more accurate than the modelled discharge. On the other hand, if the NSE value is close to 1, the better the correlation between the observed and modelled discharges. The coefficient of determination measures the variance of modelled discharges from observed discharges. The closer the R^2 is to 1, the better the correlation between discharges. These three functions were used as an indication of the performance of the model and its acceptability with regards to the degree of uncertainty.

The discharge response to rainfall in the Liesbeek Catchment was measured for 11 rainfall events with 7 used for calibration and 4 used for validation. The results for the model calibration and validation are presented in Table 13a-b. The calibration for all rainfall events were acceptable with an ISE rating of excellent and one event with an ISE rating of very good.

Table 13a: Results of calibration for the Liesbeek Catchment.

Event date Calibrations	Rainfall (mm)		Integrated Squared Error (ISE)	Nash-Sutcliffe Efficiency (NSE)	Coefficient of determination (R^2)	Peak discharge error (%)	Total volume error (%)
	K	O					
7/05/2018	15	2.8	Excellent (0.6)	0.24	0.70	-6.5	12
24/05/2018	48.8	16.2	Excellent (1.58)	0.62	0.85	33.4	-12
27/06/2018	29.8	11.8	Excellent (2.86)	-0.49	0.82	63.9	-1
1-2/07/2018	83.6	35.6	Excellent (1.83)	0.25	0.50	-8	-13
5-7/08/2018	55.4	9.6	Excellent (0.83)	0.31	0.50	3.4	-7
12-13/08/2018	7.8	0.6	Excellent (0.32)	-1.35	0.13	-8.35	21
16-17/08/2018	33.4	13.6	Very good (3.55)	-1.81	0.55	96	13

Table 13b: Results of validation for the Liesbeek Catchment.

Event date Validations	Rainfall (mm)		Integrated Squared Error (ISE)	Nash-Sutcliffe Efficiency (NSE)	Coefficient of determination (r^2)	Peak discharge error (%)	Total volume error (%)
	K	O					
28-31/05/2018	83.4	15.4	Excellent (1.05)	0.44	0.64	3.1	-12
14-16/06/2018	38.8	25.6	Excellent (1.72)	-7.26	0.82	130	6
29-30/06/2018	51.4	20	Excellent (1.38)	0.1	0.4	66.9	-13
31/07/2018 – 01/08/2018	14	2.4	Excellent (0.35)	0.810	0.60	-1.0	-7.2

All rainfall events tended to produce respectable coefficient of determination (R2) values (0.5>) except for the rainfall event on the 29-30th June and 12-13th August. This high coefficient of determination (R2) values shows good correlation with observed data. The lowest coefficient of determination (R2) value was produced for the lowest rainfall event (7.8mm in Kirstenbosch and 0.6mm in Observatory). The highest discrepancy in total volume error occurred during the lowest rainfall event which can be attributed to the catchment response in the model as well as the inaccuracies in the level sensors. The level sensors pick up a lot of 'noise' during low flow which can be attributed to the speed of the sound wave which is dependent on temperature. Other inaccuracies can also result from the level sensor measuring the height of the crest of a wave producing spikes in the level sensor's data. These inaccuracies however are not observed during high discharges as discrepancies of 1-2cm become negligible. The high peak discharge errors for the 16-17th August and the 14-16th June was a result of their very high model peak discharges of 13.89 m^3/s

and 11.91 m^3/s respectively compared to their observed peak discharges of 7.1 m^3/s and 4.9 m^3/s. However their total volume errors were low (13% and 6% respectively).

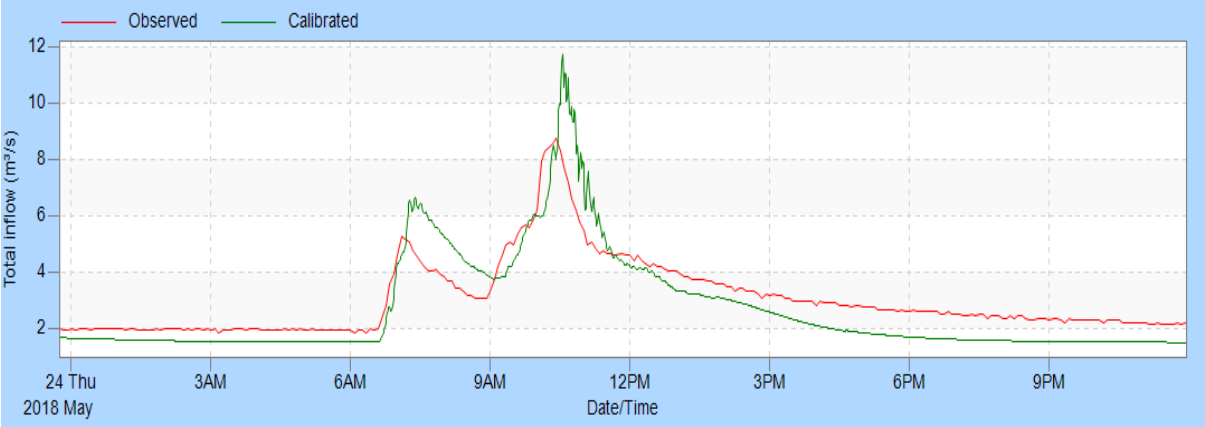

Figure 36a: Site 3 - observed vs modelled discharge (Calibration)

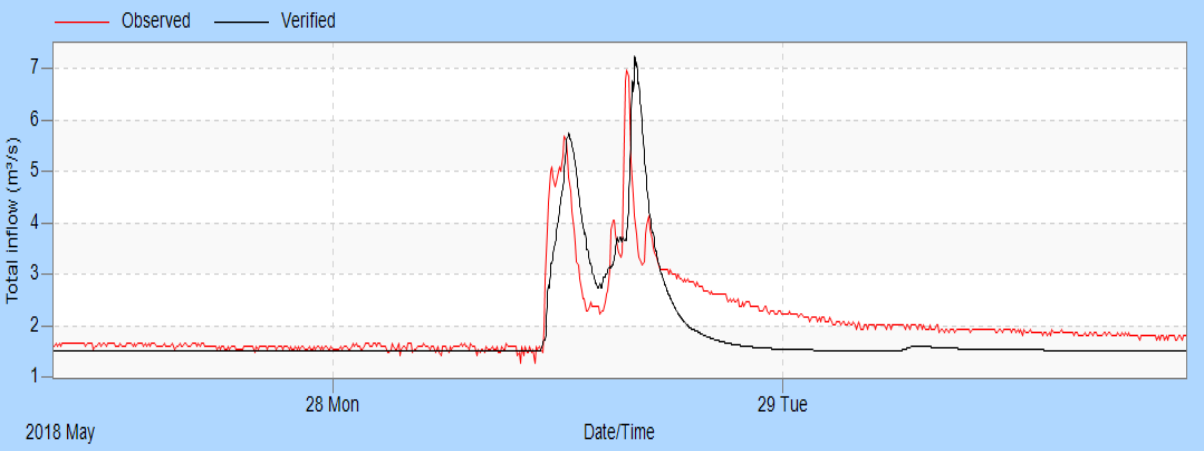

Figure 36b: Site 3 – observed vs modelled data (Verification)

4.4.1.4 Limitations of the PCSWMM model

Only one site was used for calibration and therefore no indication of the accuracy of the model further upstream was observed. For this model, Hargreave's method was used for evapotranspiration which was based primarily on temperature and wind speed. Evapotranspiration however, was not calibrated against measured data. The simplicity of disaggregating land use data into three dominant groups with weighted averages for land use and soil data introduces further uncertainty. The DEM data which was used to extract node elevations, catchment slope and irregular cross sections were limited to 2m grid accuracy. The sizes of the stormwater pipes were obtained from the GIS information supplied from the CoCT, however, due to the nature of the study, not all pipes were modelled and therefore certain pipes were made slightly larger to account for the ones that were not included. Only 11 rainfall events were used for calibration and therefore there is still uncertainty with regards to the parameters in the model and its accuracy. Baseflow and groundwater plays a role in the Liesbeek Catchment however, it was not included in the model.

All the above mentioned has introduced uncertainty in the model at varying degrees. If the model is used for further research, it is recommended that more rainfall events be included on a continuous scale that accounts for the 'noise' in the data that impacts the discharge particularly during low flow

events. It is also recommended that higher resolution of DEM data, ideally higher than 1m grid accuracy be used with a higher resolution of land use data.

4.4.1.5 Hypothetical rainfall designs

There are various ways of investigating rainfall scenarios as mentioned in section 3.3. For this study, the chosen approach was the Delta-change method which consists of choosing arbitrarily rainfall variations with reference to historical data. The reason for choosing this approach was because the study was focused on determining the harvesting potential based on different rainfall designs. The purpose of modelling the Liesbeek Catchment was to understand the hydrological conditions and the catchment response to rainfall events under various conditions. The section below explores hypothetical rainfall scenarios in the Liesbeek catchment.

For the PCSWMM modelling exercise, the rainfall designs consisted of 20mm increments for the Kirstenbosch rain gauge (20mm, 40mm, 60mm, 80mm and 100mm). It has been observed that the Observatory rain gauge records approximately 3 times less rainfall than the Kirstenbosch rain gauge and therefore the rainfall designs were 3 times lower for Observatory (7mm, 13mm, 20mm, 27mm and 33mm). The rainfall designs were all 24hour storm events. The approach of increasing rainfall designs in increments was used by previous studies as well (Waters *et al*., 2003; Shrestha *et al*., 2014). The results from the modelling exercise was taken at the same location where Site 3 is located and is summarized in table 14. The reason for the modelling exercise was to model discharge in the Liesbeek Catchment in order to predict peak discharges under hypothetical rainfall designs. The peak discharges in the modelling exercise were then used to predict EC based on the linear regression equations in figures D1-D6 (Appendix D). The EC vs Discharge scatter plots in figures D1-D6 were transformed via log10 to show a more linear relationship for the linear regression equations. There were no consideration of catchment conditions such as impervious surface coverage, soils and topography in the linear regression equations, which are known to have important impacts on the generation of surface runoff and, in turn, river discharge and pollutant concentrations (Miller *et al*. 2014; McGrane *et al*. 2017). The lack of catchment data entered into the regression models means the results must be interpreted with caution, beyond the healthy scepticism that should already be attached to models and the use of scenarios (Silberstein 2006).

Table 14: Hypothetical rainfall designs showing average and peak discharge, volume and nodes flooded at Site 3.

Event	Rainfall (mm)		Average discharge (m^3/s)	Peak discharge (m^3/s)	Volume (m^3)	Nodes flooded
	Kirstenbosch	Observatory				
1	20	7	1.9	4.2	171900	4
2	40	13	2.4	6.5	212800	13
3	60	20	2.9	9.7	254200	22
4	80	27	3.4	10.7	290800	29
5	100	33	3.7	12.2	319600	33

Table 15a: Predicted EC values for May, June and July based on modelled discharges (Site 2).

Event	Rainfall (mm) site 2		Modelled peak discharge (m^3/s)	Predicted EC in May (µS/cm)	Predicted EC in June (µS/cm)	Predicted EC in July (µS/cm)
	Kirstenbosch	Observatory				
1	20	7	4.2	161.1	158.7	129.8
2	40	13	6.5	147.4	143.2	117.2
3	60	20	9.7	136.0	130.4	106.7
4	80	27	10.7	133.3	127.4	104.3
5	100	33	12.2	129.8	123.6	101.1

Table 15b: Predicted EC values for May, June and July based on modelled discharges (Site 3).

Event	Rainfall (mm) site3		Modelled peak discharge (m^3/s)	Predicted EC in May (µS/cm)	Predicted EC in June (µS/cm)	Predicted EC in July (µS/cm)
	Kirstenbosch	Observatory				
1	20	7	4.2	156.9	133.2	142.5
2	40	13	6.5	112.5	100.4	122.1
3	60	20	9.7	82.9	77.5	106.0
4	80	27	10.7	77.0	72.8	102.4
5	100	33	12.2	69.7	66.8	97.7

Table 14 showed that as the rainfall depth increased, the average discharge, peak discharge, volume and nodes flooded increased. The modelled peak discharges were slightly higher than the observed peak discharges for rainfall events that were of similar depth. The reason for this was because all the design rainfalls were 24 hours only with a symmetrical shape (Appendix E, Figure E1). The observed rainfalls did not necessary occur over a 24 hour period with the same rainfall intensity. Table 15a-b showed the predicted EC values for May, June and July based on the peak discharges from the model. The reason for splitting up the linear regression models into each month was because each rainfall event had some sort of impact on the next rainfall event either due to the number of antecedent dry days, the depth of the rainfall or the rainfall intensity. This led to the gradual decrease in EC through the data collection period that the average EC in May was very different to the average EC in July and therefore splitting up the EC into three separate months created stronger linear regression models. It was evident in table 15a-b that the EC decreases as the rainfall and discharge increases for all months. This was expected as EC and discharge has a strong negative relationship as mentioned in section 4.2. It was also evident that the EC values were the lowest in July compared to June and May for Site 2. This was expected as well due to the accumulation of rainfall events that resulted in dilution and flushing out of pollutants as we progressed through the rainy season. The peak discharges for all the designed rainfall depths, diluted the EC to meet the domestic use, industrial use, agricultural use and aquatic systems thresholds. The designed rainfall depths also produced high quantities of volume which can be used for stormwater abstraction. In reality however, not all of the volume can be used, as water will need to be left in the river to maintain the ecological reserve and the suitable water quality range mostly only occurs during the falling limb.

The results for the modelling section reveal that an increase in discharge causes an increase in dilution and therefore lower EC values. The results also indicate that more surface flooding is

expected with an increase in discharge. The number of nodes and pipes performing over their capacity increased due to the increased rainfalls. It is therefore a plausible scenario that whilst an increase in discharge can result in cleaner water, it can also contribute to flooding, particularly in the highly urbanized sections of the Liesbeek Catchment. This was evident in the PCSWMM model whereby most of the flooded nodes occurred downstream. A study performed by Shrestha *et al*. (2014), used PCSWMM to model a peri-urban catchment using hypothetical increases in rainfall. The study showed that the number of flooded nodes increased as well as the surface flooding frequency and the duration of floods. Waters *et al*. (2003) also modelled a catchment with increasing increments of rainfall and indicated that there are increases in discharge, peak discharge, volume and surcharging of pipes.

It is expected that land use and rainfall changes will continue to have an impact in urbanized catchments with increased stormwater runoff and flooding occurring together with an increase in the demand for water (CoCT, 2012; Fisher-Jeffes *et al*., 2017). Stormwater harvesting could help diversify the city's water resources, increasing its resilience to changes in future rainfall. Stormwater harvesting systems makes use of a number of storage systems which includes, *inter alia*, retention ponds, detention ponds, rain gardens and wetlands as well as sub-surface storages such as aquifer recharge to store the harvested stormwater. In this way, runoff is detained and reduced downstream. Stormwater abstraction has the potential to reduce flooding and ease the demands for non-potable water uses. Fisher-Jeffes *et al*. (2017) noted in a study that stormwater harvesting can reduce flooding downstream in an urban catchment and that the retained stormwater should be used in a closed system for purposes such as aquifer recharge to prevent evaporation losses (Fisher-Jeffes, 2015; Fisher-Jeffes *et al*., 2017).

5. Conclusion

5.1 Key findings

This study aimed to analyse discharge of an urban river and associated water quality fluxes during rainfall events. The study addressed the primary objectives of measuring discharge and water quality using high-resolution monitoring, quantitatively establishing the relationships between discharge, water quality and rainfall, and lastly, modelling the discharge in the river. This study illustrated the changes in discharge and water quality in response to various rainfall events at 3 sites in the Liesbeek River. The results showed that upstream (Site 1) had the lowest discharge rate as well as the lowest EC values and pH. Site 2 and 3 had elevated EC values and faster discharges with neutral pH values during the rainfall events. The Liesbeek River overall had a quick response to rainfall, with changes in discharge and water quality occurring within 30-minutes after the beginning of the rainfall. Site 1 had the longest lag times, whilst Site 2 had the shortest lag times. This was indicative of the land use changes in the Liesbeek Catchment from upstream to downstream. Site 2's peak discharges were close to double that of Site 1 for rainfall events that exceeded 42mm in Kirstenbosch.

Rainfall resulted in significant responses in all variables with significant levels of dilution. Rainfall ultimately resulted in an increase in discharge and DO, whilst causing a decrease in EC. The dilution factor of rainfall was intensified during high rainfall events. This was evident during the highest rainfall event recorded at Observatory (44.4mm) whereby the EC decreased by around 400 µS/cm for Site 2 and Site 3. The dilution factor of rainfall resulted in a gradual decrease in EC and a gradual increase in DO throughout the progression of the rainy season with the lowest EC values and highest DO values occurring towards the end of the study. On average, the DO at Site 2 was higher than the DO Site 3. The EC at Site 2 and 3 were quite similar, however, a Mann Whitney U test indicated that Site 2 and 3 were statistically significantly different. The Liesbeek River demonstrated the characteristics of a typical urban river showing flashy hydrographs with steep rising limbs, particularly in the more urbanized sections of the river compared to upstream. The rainfall also caused an improvement in water quality after the peak discharge and peak rainfall. The improved levels of water quality was greater during the higher rainfall events, owing to the dilution factor.

Descriptive, bivariate and multivariate analyses were completed to establish the relationships between the variables during rainfall events. The general trend amongst all sites were that as discharge increases, EC and pH decreases, whilst DO increases. These trends were confirmed by the correlation analysis. The correlation analysis also indicated that discharge had the quickest response to rainfall than the other parameters, therefore, the peak discharges preceded the minimum EC and maximum DO concentrations.

The water quality in the Liesbeek River was compared against the SAWQG for each parameter measured to assess its suitability for various uses. The water quality and volume for the rising and falling limb was also compared. The results indicated that the EC was below the domestic use and aquatic systems threshold for most of the study. The EC was below the agricultural threshold for all rainfall events during the falling limb. The EC values were also lower than the EC in the Philippi farming area on the CFA, and in some cases, up to 10 times lower. This showed the relatively 'clean' nature of the falling limb's stormwater with respect to its EC in the Liesbeek River and creates a plausible scenario of potentially using that stormwater for aquifer recharge in the CFA.

The DO at Site 3 initially was not suitable for aquatic systems, particularly during the beginning of the rainy season. As the rainy season progressed, the DO values increased to within the suitable range for aquatic systems at Site 2 and 3. Furthermore, it was shown that the falling limb had a

higher volume and better water quality. This implies that the falling limb was more suitable for stormwater abstraction in terms of water quality and volume.

In addition, a PCSWMM model exercise, with rainfall design scenarios in 20mm increments, were used to predict volume and discharge in the Liesbeek River for specific rainfall events. The peak discharges from the modelling exercise were used to predict EC in the linear regression equations. The results for the modelling section reveal that an increase in discharge causes an increase in dilution and therefore lower EC values. The results also indicated that more surface flooding is expected with an increase in discharge. It is therefore a plausible scenario that whilst an increase in discharge can result in cleaner water, it can also contribute to flooding, particularly in the highly urbanized sections of the Liesbeek Catchment.

The utilisation of stormwater in the Liesbeek River will require effective stormwater management. Stormwater management has the potential to serve the purpose of increasing water supply together with improving water quality and protecting the ecological diversity of urban rivers (Wong and Eadie, 2000). This study, with the emphasis on high-resolution monitoring has helped inform decision making for stormwater management and harvesting.

5.2 Recommendations for further research

The degradation of urban rivers, increasing water supply and the threat of reduced future rainfall have been identified as some of the key issues in urban catchments. In this study the results of water quality and discharge has been assessed and a hydrological model has been built. These two activities have provided insight into an urban river and its catchment. Future key research activities have been identified that can improve this study in the Liesbeek Catchment:

- Measuring discharge and water quality at multiple sites (more than 3) will provide greater insight into the behaviour of the river
- Due to the high spatial variability of rainfall in the Liesbeek Catchment, additional rain gauges in the Liesbeek Catchment that measures rainfall in five-minute intervals will help spatially represent the rainfall more accurately.
- Continuously measure discharge and water quality during dry and wet periods over a longer time frame
- Due to the quantitative nature of this study, no stakeholder engagement was included. Stakeholder engagement is an important aspect particularly in the management process of dealing with urban catchments.
- A study of this nature should include other urban catchments to incorporate the abovementioned recommendations.

This study revealed the relatively 'clean' nature of the water in the Liesbeek River and that during the entire rainfall event the EC was much lower than the EC found in the Philippi farming area on the CFA. The Liesbeek River is situated on the foothills of Table Mountain which is on the western side of the CFA. The direction of groundwater flow in the region indicated that groundwater flows towards the east and therefore recharging the central areas of the CFA. The location of the Liesbeek River on the western side of the CFA has the potential for various WSUD practices such as wetlands and detention ponds to capture stormwater and over time recharge the groundwater of the CFA on the western side with relatively clean stormwater.

It is recommended that a similar study be undertaken in some of Cape Town's more polluted catchments or a river that is closer to the central areas of the CFA. An example of a river closer to the central regions of the CFA is the Kuils River and presents itself as an attractive option for aquifer

recharge. The catchment is also characterized by various wetland ecosystems that play important economic and ecological roles. Those wetlands are important in maintaining the CFA, as well as reduce flooding and improving water quality. It also presents itself as an opportunity to store water. It is therefore recommended that a study of this nature takes place in a catchment that is situated closer to central regions of the CFA if aquifer recharge using stormwater is being considered.

References

Aichele, S. (2005). Effects of urban land-use change on streamflow and water quality in Oakland County, Michigan, 1970-2003, as inferred from urban gradient and temporal analysis, date posted: June 8, 2005. *US geological survey scientific investigations report*, *5016*.

Al-Asadi, S. (2016). A study of pH values in the Shatt Al-Arab River (southern Iraq). International Journal of Marine Science, 6(29).

Alberto, W., del Pilar, D., Valeria, A., Fabiana, P., Cecilia, H. and de los Ángeles, B. (2001). Pattern Recognition Techniques for the Evaluation of Spatial and Temporal Variations in Water Quality. A Case Study: Suquía River Basin (Córdoba–Argentina). *Water research*, *35*(12), pp.2881-2894.

Ahn, K. and Merwade, V. (2017). The effect of land cover change on duration and severity of high and low flows. *Hydrological processes*, *31*(1), pp.133-149.

Allen, R., Pereira, L., Raes, D. and Smith, M. (1998). Crop evapotranspiration-Guidelines for computing crop water requirements-FAO Irrigation and drainage paper 56. *Fao, Rome*, *300*(9), p.D05109.

Allen, M., Argue, J., Geiger, W., Johnston, L., Pezzaniti, D. and Scott, P. (2004). Water sensitive urban design: basic procedures for 'source control' of stormwater: a handbook for Australian practice.

Armitage, N., Vice, M., Fisher-Jeffes, L., Winter, K., Spiegel, A. and Dunstan, J. (2013). Alternative technology for stormwater management. The South African Guidelines for Sustainable Drainage Systems. *Report TT*, *558*, p.13.

Armitage, N., Fisher-Jeffes, L., Carden, K., Winter, K., Naidoo, V., Spiegel, A., Mauck, B. and Coulson, D. (2014). Water Sensitive Urban Design (WSUD) for South Africa: FRAMEWORK AND GUIDELINES. WRC Report No. TT 588/14. Urban Water Management Research Unit.

Ashbolt, S., Aryal, S., Petrone, K., McIntosh, B., Maheepala, S., Chowdhury, R. and Gardner, T. (2013). Can stormwater harvesting restore pre-development flows in urban catchments in South East Queensland?. *Water Science and Technology*, *67*(2), pp.446-451.

Aza-Gnandji, C.D.R., Xu, Y., Raitt, L. and Levy, J. (2013). Salinity of irrigation water in the Philippi farming area of the Cape Flats, Cape Town, South Africa. *Water SA*, *39*(2), pp.199-210.

Behar, S., Byrne, J. and Dickason, C. (1996). Testing the waters: chemical and physical vital signs of a river.

Beighley, R., Melack, J. and Dunne, T. (2003). Impacts of California's climatic regimes and coastal land use change on streamflow characteristics. *JAWRA Journal of the American Water Resources Association*, *39*(6), pp.1419-1433.

Bernhardt, E. and Palmer, M. (2007). Restoring streams in an urbanizing world. *Freshwater Biology*, *52*(4), pp.738-751.

Bertrand-Krajewski, J.L., Chebbo, G. and Saget, A. (1998). Distribution of pollutant mass vs volume in stormwater discharges and the first flush phenomenon. *Water research*, *32*(8), pp.2341-2356.

Boughton, W. and Droop, O. (2003). Continuous simulation for design flood estimation—a review. *Environmental Modelling & Software*, *18*(4), pp.309-318.

Bratieres, K., Fletcher, T., Deletic, A. and Zinger, Y. (2008). Nutrient and sediment removal by stormwater biofilters: A large-scale design optimisation study. *Water research*, *42*(14), pp.3930-3940.

Brown, C., and Magoba, R. (Eds.). (2009). Rivers and Wetlands of Cape Town: Caring for Our Rich Aquatic Heritage (TT 376/08.). South African Water Research Commission. Cape Town. ISBN: 9781770057777.

Burns, M., Fletcher, T., Walsh, C., Ladson, A. and Hatt, B. (2012). Hydrologic shortcomings of conventional urban stormwater management and opportunities for reform. *Landscape and Urban Planning*, 105(3), pp.230-240.

Burns, M., Fletcher, T., Walsh, C., Ladson, A. and Hatt, B. (2014). Flow-regime management at the urban land-parcel scale: test of feasibility. *Journal of Hydrologic Engineering*, *20*(12), p.04015037.

Carlson, C., Barreteau, O., Kirshen, P. and Foltz, K. (2014). Storm water management as a public good provision problem: survey to understand perspectives of low-impact development for urban storm water management practices under climate change. *Journal of Water Resources Planning and Management*, *141*(6), p.04014080.

Chapman, D and Kimstach, V. (1996). Selection of water quality variables. In Chapman, D (Ed.). Water quality assessments: a guide to the use of biota, sediments, and water in environmental monitoring. Spon Press, Cambridge

COCT (City of Cape Town). (2012). *Trends and Change - 10 years: Census 2001 – Census 2011*. Cape Town: Information and Knowledge Management Department using 2011 Census data supplied by Statistics South Africa.

Cordoba, E., Martinez, A. and Ferrer, E. (2010). Water quality indicators: Comparison of a probabilistic index and a general quality index. The case of the Confederación Hidrográfica del Júcar (Spain). *Ecological Indicators*, *10*(5), pp.1049-1054.

Daniel, M., Montebelo, A., Bernardes, M., Ometto, J., De Camargo, P., Krusche, A., Ballester, M., Victoria, R. and Martinelli, L. (2002). Effects of urban sewage on dissolved oxygen, dissolved inorganic and organic carbon, and electrical conductivity of small streams along a gradient of urbanization in the Piracicaba river basin. *Water, Air, and Soil Pollution*, *136*(1-4), pp.189-206.

Davis, J. (1975). Minimal Dissolved Oxygen Requirements of Aquatic Life with Emphasis on Canadian Species: a Review. *Journal of the Fisheries Research Board of Canada*, 32(12), pp.2295-2332.

Davis, A., Shokouhian, M., Sharma, H. and Minami, C. (2001). Laboratory study of biological retention for urban stormwater management. *Water Environment Research*, *73*(1), pp.5-14.

Denault, C., Millar, R. and Lence, B. (2006). Assessment of possible impacts of climate change in an urban catchment. *JAWRA Journal of the American Water Resources Association*, *42*(3), pp.685-697.

Dong, J., Wang, G., Yan, H., Xu, J. and Zhang, X. (2015). A survey of smart water quality monitoring system. *Environmental Science and Pollution Research*, 22(7), pp.4893-4906.

DWAF. (1996a) South African Water Quality Guidelines Volume1: Domestic Use. Department of Water Affairs and Forestry, Pretoria, South Africa.

DWAF. (1996b) South African Water Quality Guidelines Volume 2: Recreational Use. Department of Water Affairs and Forestry, Pretoria, South Africa

DWAF. (1996c) South African Water Quality Guidelines Volume 3: Industrial Use. Department of Water Affairs and Forestry, Pretoria, South Africa

DWAF. (1996d) South African Water Quality Guidelines Volume 4: Agricultural Use: Irrigation. Department of Water Affairs and Forestry, Pretoria, South Africa

DWAF. (1996e) South African Water Quality Guidelines Volume 6: Agricultural Use: Aquaculture. Department of Water Affairs and Forestry, Pretoria, South Africa

DWAF. (1996f) South African Water Quality Guidelines Volume 7: Aquatic Ecosystems. Department of Water Affairs and Forestry, Pretoria, South Africa

El-Khoury, A., Seidou, O., Lapen, D., Que, Z., Mohammadian, M., Sunohara, M. and Bahram, D. (2015). Combined impacts of future climate and land use changes on discharge, nitrogen and phosphorus loads for a Canadian river basin. *Journal of environmental management*, *151*, pp.76-86.

Elliott, A., Spigel, R., Jowett, I., Shankar, S. and Ibbitt, R. (2010). Model application to assess effects of urbanisation and distributed flow controls on erosion potential and baseflow hydraulic habitat. *Urban Water Journal*, 7(2), pp.91-107.

Ellis, J., Deutsch, J., Legret, M., Martin, C., Revitt, D., Scholes, L., Seiker, H. and Zimmerman, U. (2006). The DayWater decision support approach to the selection of sustainable drainage systems: A multi-criteria methodology for BMP decision makers. *Water practice and technology*, *1*(1), p.wpt2006002.

Espey Jr, W., Morgan, C. and Masch, F. (1966). *Study of some effects of urbanization on storm runoff from a small watershed*. Texas Water Development Board.

Fatema, K., Maznah, W. and Isa, M. (2014). Spatial and temporal variation of physico-chemical parameters in the Merbok Estuary, Kedah, Malaysia. *Tropical life sciences research*, *25*(2), p.1.

Fekete, B. and Vörösmarty, C. (2007). The current status of global river discharge monitoring and potential new technologies complementing traditional discharge measurements. *IAHS publ*, *309*, pp.129-136.

Fekete, B., Looser, U., Pietroniro, A. and Robarts, R. (2012). Rationale for monitoring discharge on the ground. *Journal of Hydrometeorology*, *13*(6), pp.1977-1986.

Fell, J., Pead, J. & Winter, K. (2017). Affordable digital water monitoring technology. Institute of Electronics Engineers Consumer Electronics Magazine. (special issue on “Electronics for Sustainable Development” (in press).

Fisher-Jeffes, L. (2015). The viability of rainwater and stormwater harvesting in the residential areas of the Liesbeek River Catchment, Cape Town. PhD thesis, University of Cape Town.

Fisher-Jeffes, L., Carden, K., Armitage, N. and Winter, K. (2017). Stormwater harvesting: Improving water security in South Africa's urban areas. *South African Journal of Science*, *113*(1-2), pp.1-4.

Fletcher, T., Mitchell, V., Deletic, A., Ladson, T. and Seven, A. (2007). Is stormwater harvesting beneficial to urban waterway environmental flows?. *Water Science and Technology*, *55*(4), pp.265-272.

Fletcher, T. and Deletic, A. (2008). An urban water shortage solution—right under our feet. *ReNew: Technology for a Sustainable Future*, (105), pp.54-57.

Fletcher, T., Deletic, A., Mitchell, V. and Hatt, B. (2008). Reuse of urban runoff in Australia: a review of recent advances and remaining challenges. *Journal of Environmental Quality*, *37*(5_Supplement), pp.S-116.

Fletcher, T., Andrieu, H. and Hamel, P. (2013). Understanding, management and modelling of urban hydrology and its consequences for receiving waters: A state of the art. *Advances in Water Resources*, 51, pp.261-279.

Gupta, K. and Saul, A. (1996). Specific relationships for the first flush load in combined sewer flows. *Water research*, *30*(5), pp.1244-1252.

Han, Y., Lau, S., Kayhanian, M. and Stenstrom, M. (2006). Characteristics of highway stormwater runoff. *Water Environment Research*, 78(12), pp.2377-2388.

Hannah, D., Demuth, S., van Lanen, H., Looser, U., Prudhomme, C., Rees, G., Stahl, K. and Tallaksen, L. (2011). Large-scale river flow archives: importance, current status and future needs. *Hydrological Processes*, *25*(7), pp.1191-1200.

Hatt, B., Fletcher, T., Walsh, C. and Taylor, S. (2004). The influence of urban density and drainage infrastructure on the concentrations and loads of pollutants in small streams. Environmental management, 34(1), pp.112-124.

Hatt, B., Deletic, A., & Fletcher, T. (2006). Integrated treatment and recycling of stormwater: a review of Australian practice. Journal of Environmental Management, 79(1), p102–113.

Hathaway, J., Tucker, R., Spooner, J. and Hunt, W. (2012). A traditional analysis of the first flush effect for nutrients in stormwater runoff from two small urban catchments. *Water, Air, & Soil Pollution*, 223(9), pp.5903-5915.

He, J., Valeo, C., Chu, A. and Neumann, N. (2010). Characterizing physicochemical quality of storm-water runoff from an urban area in Calgary, Alberta. *Journal of Environmental Engineering*, 136(11), pp.1206-1217.

Hogan, D., Jarnagin, S., Loperfido, J. and Van Ness, K. (2013). Mitigating the Effects of Landscape Development on Streams in Urbanizing Watersheds. *JAWRA Journal of the American Water Resources Association*, 50(1), pp.163-178.

Hood, M., Clausen, J. and Warner, G. (2007). Comparison of Stormwater Lag Times for Low Impact and Traditional Residential Development. *Journal of the American Water Resources Association*, 43(4), pp.1036-1046.

Huang, H., Cheng, S., Wen, J. and Lee, J. (2008). Effect of growing watershed imperviousness on hydrograph parameters and peak discharge. *Hydrological Processes*, 22(13), pp.2075-2085.

Ijaradar, J. and Chatterjee, S. (2018). Real-Time Water Quality Monitoring System. *International Research Journal of Engineering and Technology,* 5(3), pp.1166-1171.

IPCC. (2007). Summary for policymakers. In Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K., Tignor, M., Miller, H. (Eds.). Climate change 2007: the physical science basis. Contribution

of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, and New York.

James, W. (2005). Rules for responsible modeling. 4th ed. Guelph, Ontario: CHI.

Jarvie, H., Neal, C., Smart, R., Owen, R., Fraser, D., Forbes, I. and Wade, A. (2001). Use of continuous water quality records for hydrograph separation and to assess short-term variability and extremes in acidity and dissolved carbon dioxide for the River Dee, Scotland. *Science of The Total Environment*, 265(1-3), pp.85-98.

Jianlong, W. and Ning, Y. (2004). Partial nitrification under limited dissolved oxygen conditions. *Process Biochemistry*, 39(10), pp.1223–1229.

Kang, J., Kayhanian, M. and Stenstrom, M. (2008). Predicting the existence of stormwater first flush from the time of concentration. *Water Research*, 42(1–2), pp.220–228.

Kawanisi, K., Razaz, M., DeCosta, G. and De Costa, C. (2010). Continuous monitoring of water discharge, temperature, and salinity in a tidal spillway using fluvial acoustic tomography. *Journal of Hydrology (New Zealand)*, pp.61-68.

Khaledian, Y., Ebrahimi, S., Natesan, U., Basatnia, N., Nejad, B., Bagmohammadi, H. and Zeraatpisheh, M. (2018). Assessment of water quality using multivariate statistical analysis in the Gharaso River, Northern Iran. In *Urban ecology, water quality and climate change* (pp. 227-253). Springer, Cham.

Kirchner, J., Feng, X., Neal, C. and Robson, A. (2004). The fine structure of water-quality dynamics: The (high-frequency) wave of the future. *Hydrological Processes*, *18*(7), pp.1353-1359.

Kondolf, M. (2011). Setting goals in river restoration: when and where can the river "heal itself"? *Stream Restoration in Dynamic Fluvial Systems*, pp.29-43.

Krebs, G., Kokkonen, T., Valtanen, M., Setälä, H. & Koivusalo, H. (2014). Spatial resolution considerations for urban hydrological modelling. *Journal of Hydrology*. 512:482‐497.

Lee, J., Bang, K., Ketchum Jr, L., Choe, J. and Yu, M. (2002). First flush analysis of urban storm runoff. *Science of the Total Environment*, *293*(1-3), pp.163-175.

Lee, H., Lau, S., Kayhanian, M. and Stenstrom, M. (2004). Seasonal first flush phenomenon of urban stormwater discharges. *Water Research*, *38*(19), pp.4153-4163.

Lee, J.G., Nietch, C.T. & Panguluri, S. (2017). SWMM Modeling Methods for Simulating Green Infrastructure at a Suburban Head watershed: User's Guide. Cincinatti, Ohio: U.S. Environmental Protection Agency.

Legesse, D., Vallet-Coulomb, C. and Gasse, F. (2003). Hydrological response of a catchment to climate and land use changes in Tropical Africa: case study South Central Ethiopia. *Journal of Hydrology*, *275*(1-2), pp.67-85.

Li, L., Yin, C., He, Q. and Kong, L. (2007). First flush of storm runoff pollution from an urban catchment in China. *Journal of Environmental Sciences*, *19*(3), pp.295-299.

Lim, M., Leong, Y., Tiew, K. and Seah, H. (2011). Urban stormwater harvesting: A valuable water resource of Singapore. *Water Practice and Technology*, *6*(4), p.wpt20110067.

Liyanage, C. and Yamada, K. (2017). Impact of Population Growth on the Water Quality of Natural Water Bodies. *Sustainability*, *9*(8), p.1405.

Malmqvist, B. and Rundle, S. (2002). Threats to the running water ecosystems of the world. *Environmental conservation*, 29(2), pp.134-153.

Mancipe-Munoz, N., Buchberger, S., Suidan, M. and Lu, T. (2014). Calibration of rainfall-runoff model in urban watersheds for stormwater management assessment. *Journal of Water Resources Planning and Management*, *140*(6), p.05014001.

Mauck, B. (2017). *The capacity of the cape flats aquifer and its role in water sensitive urban design in Cape Town* (Doctoral dissertation, University of Cape Town).

McGrane, S., Hutchins, M., Miller, J., Bussi, G., Kjeldsen, T. and Loewenthal, M. (2017). During a winter of storms in a small UK catchment, hydrology and water quality responses follow a clear rural-urban gradient. *Journal of hydrology*, *545*, pp.463-477.

Meyer, N., Rooseboom, A., Retief, M. (2000). Discharge measurement in terms of pressure differences at bridge piers. WRC Report No 980/3/00

Midgley, J., Scott, D. and Harris, C. (2001). How do we know how much groundwater is stored in southwestern Cape mountains? research in action. *South African Journal of Science*, *97*(7-8), pp.285-286.

Miller, R., Bradford, W. and Peters, N. (1988). *Specific conductance: theoretical considerations and application to analytical quality control*. US Government Printing Office.

Miller, J., Kim, H., Kjeldsen, T., Packman, J., Grebby, S. and Dearden, R. (2014). Assessing the impact of urbanization on storm runoff in a peri-urban catchment using historical change in impervious cover. *Journal of Hydrology*, *515*, pp.59-70.

Mishra, A. (2010). Assessment of water quality using principal component analysis: a case study of the river Ganges. *Journal of Water chemistry and Technology*, *32*(4), pp.227-234.

Niemczynowicz, J. (1999). Urban hydrology and water management–present and future challenges. *Urban water*, *1*(1), pp.1-14.

Nilsson, C. and Malm-Renöfält, B. (2008). Linking flow regime and water quality in rivers: a challenge to adaptive catchment management. *Ecology & society*, *13*(2), p.18.

O'Flynn, B., Martinez, R., Cleary, J., Slater, C., Regan, F., Diamond, D. and Murphy, H. (2007). October. SmartCoast: a wireless sensor network for water quality monitoring. In *Local Computer Networks, 2007. LCN 2007. 32nd IEEE Conference on* (pp. 815-816).

O'Flynn, B., Regan, F., Lawlor, A., Wallace, J., Torres, J. and O'Mathuna, C. (2010). Experiences and recommendations in deploying a real-time, water quality monitoring system. *Measurement Science and Technology*, 21(12), p.124004.

Ogutu, C. (2007) Water Demand Management Options for Cape Metropolitan Area – South Africa. M.Tech Thesis, Tshwane University of Technology.

Ometto, J., Martinelli, L., Ballester, M., Gessner, A., Krusche, A., Victoria, R. and Williams, M. (2000). Effects of land use on water chemistry and macroinvertebrates in two streams of the Piracicaba river basin, south-east Brazil. *Freshwater Biology*, *44*(2), pp.327-337.

Palla, A. and Gnecco, I. (2015). Hydrologic modeling of Low Impact Development systems at the urban catchment scale. *Journal of hydrology*, *528*, pp.361-368.

Pejman, A., Bidhendi, G., Karbassi, A., Mehrdadi, N. and Bidhendi, M. (2009). Evaluation of spatial and seasonal variations in surface water quality using multivariate statistical techniques. *International Journal of Environmental Science & Technology*, *6*(3), pp.467-476.

Pitman, W. (2011). Overview of water resource assessment in South Africa: current state and future challenges. *Water SA*, *37*(5), pp.659-664.

Praskievicz, S. and Chang, H. (2009). A review of hydrological modelling of basin-scale climate change and urban development impacts. *Progress in Physical Geography*, *33*(5), pp.650-671.

Rao, R. and Delleur, J. (1974). Instantaneous unit hydrographs, peak discharges and lag times in urban basins. *Hydrological Sciences Bulletin*, 19(2), pp.185-198.

Rossman, L. (2008). Storm Water Management Model user's manual, version 5.0. National Risk Management Research Laboratory, Office of Research and Development, U.S. Environmental Protection Agency. United States of America. Retrieved from http://nepis.epa.gov/Exe/ZyPURL.cgi?Dockey=P10011XQ.txt.

Sabater, S. and Tockner, K. (2009). Effects of hydrologic alterations on the ecological quality of river ecosystems. In *Water scarcity in the Mediterranean* (pp. 15-39). Springer, Berlin, Heidelberg.

Shrestha, A., Chaosakul, T., Priyankara, D., Chuyen, L., Myat, S., Syne, N., Irvine, K., Koottatep, T. and Babel, M. (2014). Application of PCSWMM to explore possible climate change impacts on surface flooding in a Peri-urban area of Pathumthani, Thailand. *Journal of Water Management Modeling*.

Silberstein, R. (2006). Hydrological models are so good, do we still need data?. *Environmental Modelling & Software*, *21*(9), pp.1340-1352.

Soller, J., Stephenson, J., Olivieri, K., Downing, J. and Olivieri, A. (2005). Evaluation of seasonal scale first flush pollutant loading and implications for urban runoff management. *Journal of environmental management*, 76(4), pp.309-318.

Stall, J. and Smith, H. (1961) A comparison of urban and rural runoff. Paper presented at the Hyd. Diu. Conf. of the Am. Soc. Civ. Engrs., Urbana, Illinois.

Tan, S., Chua, L., Shuy, E., Lo, E. and Lim, L. (2008). Performances of rainfall-runoff models calibrated over single and continuous storm flow events. *Journal of Hydrologic Engineering*, *13*(7), pp.597-607.

Tillinghast, D., Hunt, F., Jennings, D. and D'Arconte, P. (2012). Increasing stream geomorphic stability using storm water control measures in a densely urbanized watershed. *Journal of Hydrologic Engineering*, *17*(12), pp.1381-1388.

Tu, J. (2009). Combined impact of climate and land use changes on streamflow and water quality in eastern Massachusetts, USA. *Journal of Hydrology*, *379*(3-4), pp.268-283.

United States Environmental Protection Agency (1986). *Quality criteria for water*. Washington: Environmental Protection Agency.

Van Mazijk, R., Smyth, L., Weideman, E. and West, A. (2018). Isotopic tracing of stormwater in the urban Liesbeek River. *Water SA*, *44*(4), pp.674-679.

Van Roon, M. (2007). Water localisation and reclamation: Steps towards low impact urban design and development. *Journal of environmental management*, *83*(4), pp.437-447.

Van Vliet, M. and Zwolsman, J. (2008). Impact of summer droughts on the water quality of the Meuse river. *Journal of Hydrology*, *353*(1-2), pp.1-17.

Vietz, G., Walsh, C. and Fletcher, T. (2015). Urban hydrogeomorphology and the urban stream syndrome. Progress in Physical Geography, 40(3), pp.480-492.

Vietz, G., Rutherfurd, I., Fletcher, T. and Walsh, C. (2016). Thinking outside the channel: Challenges and opportunities for protection and restoration of stream morphology in urbanizing catchments. Landscape and Urban Planning, 145, pp.34-44.

Vörösmarty, C., Green, P., Salisbury, J. and Lammers, R. (2000). Global water resources: vulnerability from climate change and population growth. *science*, *289*(5477), pp.284-288.

Vrana, B., Allan, I., Greenwood, R., Mills, G., Dominiak, E., Svensson, K., Knutsson, J. and Morrison, G. (2005). Passive sampling techniques for monitoring pollutants in water. *TrAC Trends in Analytical Chemistry*, *24*(10), pp.845-868.

Walsh, C., Roy, A., Feminella, J., Cottingham, P., Groffman, P. and Morgan II, R. (2005). The urban stream syndrome: current knowledge and the search for a cure. *Journal of the North American Benthological Society*, *24*(3), pp.706-723.

Walsh, C., Fletcher, T., Hatt, B. and Burns, M. (2010). New generation stormwater management objectives for stream protection: implementation at multiple scales to restore a small stream. In *Proceedings of the 2010 Stormwater Industry Association National Conference*.

Walsh, C., Fletcher, T. and Burns, M. (2012). Urban Stormwater Runoff: A New Class of Environmental Flow Problem. *PLoS ONE*, 7(9), p.e45814.

Walsh, C., Booth, D., Burns, M., Fletcher, T., Hale, R., Hoang, L., Livingston, G., Rippy, M., Roy, A., Scoggins, M. and Wallace, A. (2016). Principles for urban stormwater management to protect stream ecosystems. *Freshwater Science*, 35(1), pp.398-411.

Wanielista, M., Yousef, Y., Harper, G., Lineback, T. and Dansereau, L. (1991). Section 2, Design Curves for the Reuse of Stormwater. *State Department of Environmental Protection*, *8*.

Waters, D., Watt, W., Marsalek, J. and Anderson, B. (2003). Adaptation of a storm drainage system to accommodate increased rainfall resulting from climate change. *Journal of Environmental Planning and Management*, 46(5), pp.755-770.

Watt, M. (2000). *A hydrologic primer for New Jersey watershed management* (No. 2000-4140). US Geological Survey.

Wetzel, R. (2001). *Limnology: lake and river ecosystems*. Gulf Professional Publishing.

White, M. and Greer, K. (2006). The effects of watershed urbanization on the stream hydrology and riparian vegetation of Los Penasquitos Creek, California. *Landscape and urban Planning*, *74*(2), pp.125-138.

Wong, T. and Eadie, M. (2000). Water sensitive urban design: a paradigm shift in urban design. In *10th World Water Congress: Water, the World's Most Important Resource* (p. 1281). International Water Resources Association.

Woods-Ballard, B., Kellagher, R., Martin, P., Jefferies, C., Bray, R., Shaffer, P., and Wallingford, H. (2007). *The SuDS manual*.

Young, D., E. Afoa, K. Meijer, A. Wagenhoff, and C. Utech. (2013). Temperature as a Contaminant in Streams in the Auckland Region, Stormwater Issues and Management Options. Technical Report TR2013/044. Auckland Council, Environment Waikato and Hawkes Bay Regional Council.

Yufen, R., Xiaoke, W., Zhiyun, O., Hua, Z., Xiaonan, D. and Hong, M. (2008). Stormwater runoff quality from different surfaces in an urban catchment in Beijing, China. *Water Environment Research*, *80*(8), pp.719-724.

Ziervogel, G., Shale, M., & Du, M. (2010). Climate change adaptation in a developing country context: The case of urban water supply in Cape Town. *Climate and Development*, *2*(2), 94-110.

Appendix A: Rainfall events in the Liesbeek River

Rainfall events site 1:

Table A1: Rainfall and discharge data for rainfall events showing total rainfall, peak and average discharge, lag time and antecedent dry days

Discharge Site 1					
Date	**Total rainfall (mm)**	**Peak discharge (m^3/s)**	**Average discharge (m^3/s)**	**Lag time (minutes)**	**Antecedent dry days**
19-20/05/19	88.2	5.3	0.71	5	11
4-5/06/19	71.6	6.7	0.72	105	8
21-22/06/19	56.6	4.0	0.49	10	8
18-19/07/19	30.2	1.8	0.47	-105	1

Table A2: Rainfall and EC data for rainfall events showing total rainfall, average ± standard deviation (st dev), range, lag time and antecedent dry days

EC Site 1					
Date	**Total rainfall (K)**	**Average ± st dev (µS/cm)**	**Range (µS/cm)**	**Lag time**	**Antecedent dry days**
8/05/19	12	130.7 ± 12.9	164.7 – 108.1	11h	4
19-20/05/19	88.2	118.7 ± 65.2	222.2 – 43.57	5hr	11

Table A3: Rainfall and pH data for rainfall events showing total rainfall, average ± standard deviation (st dev), range, lag time and antecedent dry days

pH Site 1				
Date	**Total rainfall (K)**	**Average ± st dev**	**Range**	**Antecedent dry days**
8/05/19	12	7.1 ± 0.1	7.4 – 6.9	4
19-20/05/19	88.2	6.4 ± 0.4	7.0 – 5.8	11

Table A4: Rainfall and temperature data for rainfall events showing total rainfall, average ± standard deviation (st dev), range, lag time and antecedent dry days

Temperature Site 1				
Date	**Total rainfall (K)**	**Average ± st dev (°C)**	**Range (mg/L)**	**Antecedent dry days**
8/05/19	12	17.1 ± 0.7	18.2 – 15.9	4
19-20/05/19	88.2	20.6 ± 4	24.0 – 16.8	11

Rainfall events site 2:

Table A5: Rainfall and discharge data for rainfall events showing total rainfall, peak and average discharge, lag time and antecedent dry days

Discharge Site 2							
Date	**Total rainfall (mm)**		**Peak discharge (m^3/s)**	**Average discharge (m^3/s)**	**Lag time (minutes)**		**Antecedent dry days**
	K	**O**			**K**	**O**	
19-20/05/19	88.2	22.4	15.3	1.2	0	-5	11
4-5/06/19	71.6	44.4	28.1	1.7	25	100	8
9/06/19	17.8	9.2	2.1	0.5	30	300	1
21-22/06/19	56.6	20.8	9.5	0.8	10	5	8
27-28/06/19	41.2	11.8	4.1	0.9	-20	10	3
18-19/07/19	30.2	4.0	4.4	0.8	-100	-10	1
22-23/07/19	74.2	13.4	17.8	2.9	20	-260	2
29-31/07/19	42.2	14.4	4.3	1.2	20	-15	3

Table A6: Rainfall and EC data for rainfall events showing total rainfall, average ± standard deviation (st dev), range, lag time and antecedent dry days

EC Site 2							
Date	**Total rainfall**		**Average ± st dev (µS/cm)**	**Range (µS/cm)**	**Lag time**		**Antecedent dry days**
	K	**O**			**K**	**O**	
4/03/19	2.6	9.4	430.4 ± 32.5	520.2 ± 361.4	275	300	2
11/03/19	47.6	18.8	294.6 ± 72.8	426.3 ± 149.8	20	-10	1
19-20/05/19	88.2	22.4	273.6 ± 76.1	383.7 – 123.4	25	20	11
4-5/06/19	71.6	44.4	267.1 ± 96.4	478.3 – 30.1	20	95	8
9/06/19	17.8	9.2	237.6 ± 38.9	327.1 – 179.6	60	>300	1
21-22/06/19	56.6	20.8	302.6 ± 39.3	425.5 – 218.3	25	20	8
27-28/06/19	41.2	11.8	272.9 ± 55.2	376.2 – 163.5	105	135	3
30/06/19 – 1/06/19	55.6	17.8	236.9 ± 45.1	321.6 – 137.5	45	-30	1
2-3/07/19	85.2	6.6	233.5 ± 37.9	322.3 – 136.7	31	10	1/16
18-19/07/19	30.2	4.0	208.7 ± 32.9	285.5 – 126.3	60	205	1
22-23/07/19	74.2	13.4	241.1 ± 74.9	378.8 – 104.9	-5	-300	2
29-31/07/19	42.2	14.4	146.5 ± 26.2	240.6 – 79.8	>5hrs	>5hrs	3

Table A7: Rainfall and pH data for rainfall events showing total rainfall, average ± standard deviation (st dev), range, lag time and antecedent dry days

pH Site 2					
Peak rainfall	**Total rainfall**		**Average ± st dev (mg/L)**	**Range (mg/L)**	**Antecedent dry days**
	K	**O**			
4/03/19	2.6	9.4	6.7 ± 0.4	7.4 – 6.1	11
4-5/06/19	71.6	44.4	7.2 ± 0.3	7.8 – 6.0	8
27-28/06/19	41.2	11.8	7.3 ± 0.5	8.6 – 6.2	3
2-3/07/19	85.2	6.6	6.6 ± 0.35	8.1 – 6.1	1/16

Table A8: Rainfall and temperature data for rainfall events showing total rainfall, average ± standard deviation (st dev), range, lag time and antecedent dry days

Temperature Site 2					
Date	**Total rainfall**		**Average ± st dev (°C)**	**Range (°C)**	**Antecedent dry days**
	K	**O**			
4/03/19	2.6	9.4	21.3 ± 0.7	23.4 – 20.4	2
19-20/05/19	88.2	22.4	18.3 ± 1.1	22.5 – 16.0	11

Table A9: Rainfall and DO data for rainfall events showing total rainfall, average ± standard deviation (st dev), range, lag time and antecedent dry days

DO Site 2							
Date	**Total rainfall**		**Average ± st dev (mg/L)**	**Range (mg/L)**	**Lag time**		**Antecedent dry days**
	K	**O**			**K**	**O**	
11/03/19	47.6	18.8	5.9 ± 0.7	7.23 – 4.0	260	230	1
19-20/05/19	88.2	22.4	5.3 ± 1.5	7.1 – 1.0	>300	>300	11
4-5/06/19	71.6	44.4	3.4 ± 0.63	7.59 – 1.66	N/A	N/A	8
18-19/07/19	30.2	4.0	5.4 ± 1.0	8.8 – 2.8	-50	40	1
22-23/07/19	74.2	13.4	6.5 ± 1.1	10.18 – 4.92	-5	-300	2

Rainfall events site 3:

Table A10: Rainfall and discharge data for rainfall events showing total rainfall, peak and average discharge, lag time and antecedent dry days

Discharge Site 3							
Date	**Total rainfall (mm)**		**Peak discharge (m^3/s)**	**Average discharge (m^3/s)**	**Lag time (minutes)**		**Antecedent dry days**
	K	**O**			**K**	**O**	
19-20/05/19	88.2	22.4	7.5	2.2	15	5	11
4-5/06/19	71.6	44.4	15.2	0.95	35	110	8
9/06/19	17.8	9.2	2.7	1.7	55	325	1
21-22/06/19	56.6	20.8	5.1	1.8	30	25	8
27-28/06/19	41.2	11.8	4.0	1.9	-15	15	3
30/06/19 – 1/07/19	55.6	17.8	5.1	2.1	25	15	1
2-3/07/19	85.2	6.6	10.4	3.3	50	20	1/16
18-19/07/19	30.2	4.0	3.7	1.8	-65	12	1
22-23/07/19	74.2	13.4	9.1	2.7	20	-240	2
29-31/07/19	42.2	14.4	3.4	2.0	55	20	3

Table A11: Rainfall and EC data for rainfall events showing total rainfall, average ± standard deviation (st dev), range, lag time and antecedent dry days

EC Site 3							
Date	**Total rainfall**		**Average ± st dev (µS/cm)**	**Range (µS/cm)**	**Lag time**		**Antecedent dry days**
	K	**O**			**K**	**O**	
7/03/19	7.4	9.4	423.5 ± 31.2	468.7 – 324.8	45	70	2
11/03/19	47.6	18.8	296.7 ± 72.1	441 – 155.4	75	45	1
2-3/05/19	15.4	2.4	371.4 ± 32.1	434.8 – 321.2	>300	>300	1
8/05/19	10.6	8.8	325.9 ± 54.2	403.5 – 235.8	-240	85	4
19-20/05/19	88.2	22.4	303.9 ± 98.9	411.9 – 111.8	30	25	10
4-5/06/19	71.6	44.4	276.7 ± 94.7	490.5 – 80.7	45	120	8
9/06/19	17.8	9.2	274 ± 43.5	350.7 – 205.2	115	>300	1
21-22/06/19	56.6	20.8	280.4 ± 52.7	432.5 – 217.8	60	55	8
27-28/06/19	41.2	11.8	237 ± 56.1	274.1 – 112.3	90	120	3
30/06/19 – 1/06/19	55.6	17.8	177.3 ± 28.3	227 – 104.4	35	25	1
2-3/07/19	85.2	6.6	173.1 ± 28.6	216.4 – 102.1	-105	-75	1/16
18-19/07/19	30.2	4.0	184 ± 40.3	297.7 – 123.1	150	230	1
22-23/07/19	74.2	13.4	173.7 ± 44.2	266.7 ± 88.15	20	-240	2
29-31/07/19	42.2	14.4	81 ± 6.3	98.9 – 67.16	250	220	3

Table A12: Rainfall and pH data for rainfall events showing total rainfall, average ± standard deviation (st dev), range, lag time and antecedent dry days

pH Site 3					
Date	**Total rainfall**		**Average ± st dev**	**Range**	**Antecedent dry days**
	K	**O**			
7/03/19	7.4	9.4	6.1 ± 0.2	7.6 – 5.7	2
11/03/19	47.6	18.8	7.5 ± 0,4	8.5 – 6.9	1
2-3/05/19	15.4	2.4	6.2 ± 0.5	7.6 – 5.6	1
8/05/19	10.6	8.8	5.8 ± 0.2	6.3 – 5.6	4
19-20/05/19	88.2	22.4	6.8 ± 0.4**	7.8 – 6.0**	11
4-5/06/19	71.6	44.4	7.7 ± 0.3	8.7 – 5.7	8
21-22/06/19	56.6	20.8	7.1 ± 0.9	8.54 - 6	8
27-28/06/19	41.2	11.8	7.4 ± 0.2	8.0 – 6.5	3
30/06/19 – 1/06/19	55.6	17.8	6.7 ± 0.4	7.5 – 6	1
2-3/07/19	85.2	6.6	6.6 ± 0.3	7.3 – 6.2	1/16
29-31/07/19	42.2	14.4	6.9 ± 0.3	7.4 – 5.9	3

** = Missing data

Table A13: Rainfall and temperature data for rainfall events showing total rainfall, average ± standard deviation (st dev), range, lag time and antecedent dry days

Temperature Site 3					
Date	**Total rainfall**		**Average ± st dev (°C)**	**Range (°C)**	**Antecedent dry days**
	K	**O**			
7/03/19	7.4	9.4	20.6 ± 1.2	22.1 – 17.7	2
11/03/19	47.6	18.8	18.1 ± 0.6	19.4 – 17.2	1
2-3/05/19	15.4	2.4	17.3 ± 0.9	19.8 – 15.8	1
8/05/19	10.6	8.8	16.7 ± 0.5	17.3 – 15.7	4
19-20/05/19	88.2	22.4	16.8 ± 0.8	18.3 – 14.9	11
4-5/06/19	71.6	44.4	18.6 ± 1.3	21.5 – 15.9	8
9/06/19	17.8	9.2	16.2 ± 0.5	17 – 15	1
21-22/06/19	56.6	20.8	14.3 ± 0.8	16.2 – 12.6	8
27-28/06/19	41.2	11.8	14.8 ± 0.96	16.6 – 13.1	3
30/06/19 – 1/06/19	55.6	17.8	13.8 ± 0.7	16.1 – 13.0	1
2-3/07/19	85.2	6.6	14.3 ± 0.4	15.2 – 13.5	1/16
18-19/07/19	30.2	4.0	14.8 ± 0.7	15.9 – 13.1	1
22-23/07/19	74.2	13.4	13.5 ± 0.7	14.4 – 11.7	2

Table A14: Rainfall and DO data for rainfall events showing total rainfall, average ± standard deviation (st dev), range, lag time and antecedent dry days

DO Site 3							
Date	**Total rainfall**		**Average ± st dev (mg/L)**	**Range (mg/L)**	**Lag time**		**Antecedent dry days**
	K	**O**			**K**	**O**	
7/03/19	7.4	9.4	1.5 ± 0.5	2.7 – 0.63	85	60	2
11/03/19	47.6	18.8	4.4 ± 1.3	6.48 – 0.67	70	40	1
27-28/06/19	41.2	11.8	3.3 ± 0.7	5.2 – 1.2	90	120	3
30/06/19 – 1/06/19	55.6	17.8	3.2 ± 1.0	5.5 – 1.2	30	20	1
2-3/07/19	85.2	6.6	3.9 ± 0.6	5.4 – 1.8	10	-10	1/16
18-19/07/19	30.2	4.0	4.9 ± 0.4	5.9 – 3.0	-70	20	1
22-23/07/19	74.2	13.4	6.1 ± 0.7	8.25 – 4.83	15	-245	2
29-31/07/19	42.2	14.4	7.4 ± 0.3	8.31 – 6.79	50	15	3

Appendix B: Principal Component Analysis

Principal Component Analysis (PCA) methods were used to extract key factors in the dataset. The PCA method starts with the covariance matrix describing the dispersion of the original variables, and extracting the eigenvalues and eigenvectors. Eigenvectors are coefficients that is multiplied by the original correlated variables to obtain the Principal Components (PCs). Therefore a PC is the product of the original data and an eigenvector. PCA is a reduction variable method with PCs providing information on the most meaningful parameters. This allows for data reduction with minimal loss of information. In this study, eigenvalues greater than 1 were used as a cut-off value to determine the number of factors. PCA is a powerful technique used for reducing the dimensionality of large data sets without loss of information (Wunderlin *et al.*, 2001; Fatema *et al.*, 2014; Khaledian *et al.*, 2018).

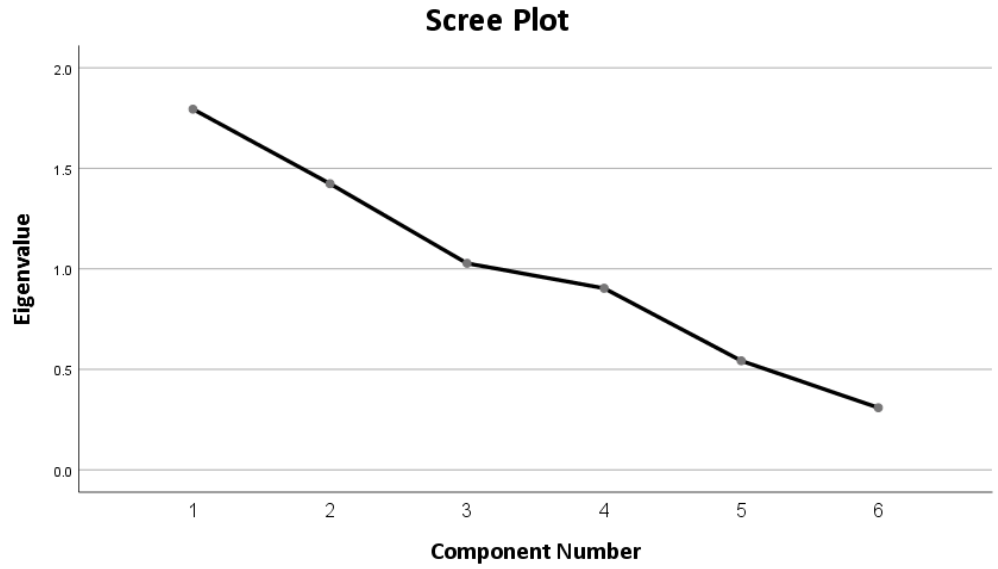

Figure B1: Scree plot of eigenvalues of principal components for Site 2.

Table B1: Total variance of components 1-3 explained for Site 2

	Initial Eigenvalues			**Extraction sums of squared loadings**			**Rotation sums of squared loadings**		
Component	Total	% Variance	Cumulative %	Total	% Variance	Cumulative %	Total	% Variance	Cumulative %
1	1.79	29.91	29.91	1.79	29.91	29.91	1.72	28.71	28.71
2	1.42	23.72	53.63	1.42	23.72	53.63	1.38	23.03	51.70
3	1.03	17.12	70.75	1.03	17.12	70.75	1.14	19.01	70.75

Table B2: Principal Component (PC) matrixes (rotated) for site 2

	Component		
Parameters	**PC 1**	**PC 2**	**PC 3**
EC	-0.87	0.19	0.16
Discharge	0.79	0.32	
DO	0.27	0.82	
pH	-0.47	0.74	-0.16
Rainfall (Observatory)			0.86
Rainfall (Kirstenbosch)	0.15	-0.10	0.57

Tables B2 and B4 presents the loading of the variables under each PC. PC1 represents the most important processes controlling the composition of water quality in the river. It also has the highest eigenvalue (Figure B1 and B2) and therefore the highest variance.

For Site 2, three PC's explained 70.75% of the total variance for the rainfall events (Table B1). PC1 explained 29.91% of the total variance and is highly negatively contributed by EC whilst it is also highly positively contributed by discharge. PC2 explained 23.72% of the total variance and is highly positively contributed by DO and pH. PC3 explained 18.73% of the total variance, showing a strong positive loading for rainfall.

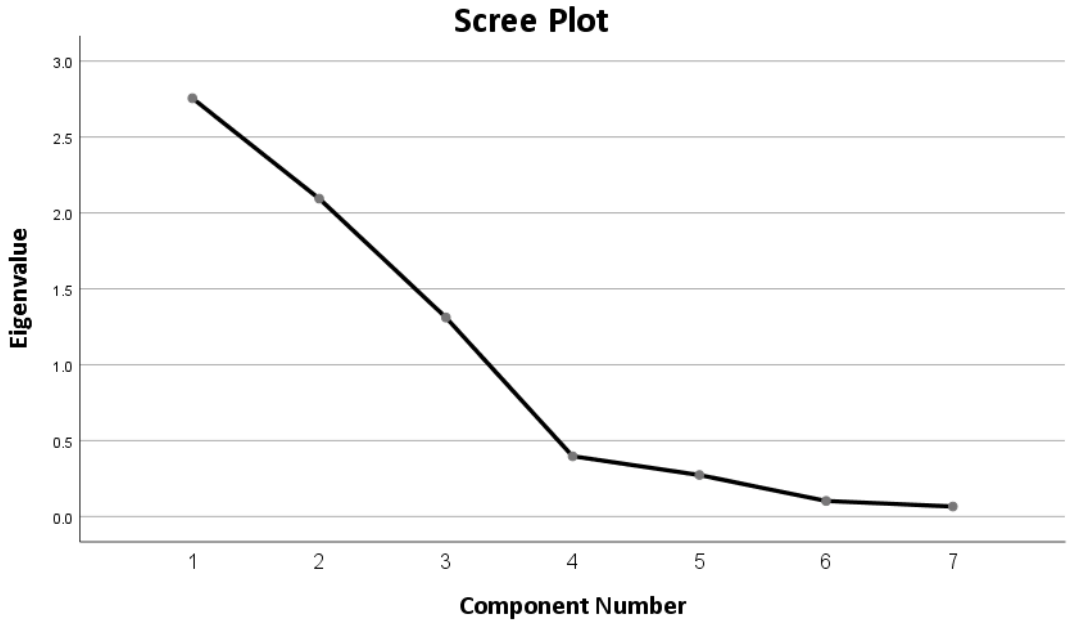

Figure B2: Scree plot of eigenvalues of principal components for Site 3.

Table B3: Total variance of components 1-3 explained for Site 3

	Initial Eigenvalues			**Extraction sums of squared loadings**			**Rotation sums of squared loadings**		
Component	Total	% Variance	Cumulative %	Total	% Variance	Cumulative %	Total	% Variance	Cumulative %
1	2.75	39.36	39.36	2.75	39.36	39.36	2.42	34.53	34.53
2	2.09	29.92	69.28	2.09	29.92	69.28	1.93	27.62	62.15
3	1.31	18.73	88.01	1.31	18.73	88.01	1.81	25.85	88.01

Table B4: Principal Component (PC) matrixes (rotated) for site 3

	Component		
Parameters	**PC 1**	**PC 2**	**PC 3**
Discharge	0.895	0.363	
DO	0.894		
EC	-0.839		0.479
Rainfall (Kirstenbosch)		0.923	
Rainfall (Observatory)		0.907	
Temperature			0.908
pH		0.309	0.852

For Site 3, the first three PC's had eigenvalues greater than 1 and explained 88.01% of the total variance of the data set for the rainfall events (Figure B2 and Table B3). PC1 explained 39.36% of the total variance and is highly positively contributed by discharge and DO whilst it is also highly negatively contributed by EC. This may be due to the runoff and dilution from the rainfall causing an increase in discharge. The dilution and faster discharges inhibit dissolved organic matter from consuming oxygen, whilst faster discharges also increases DO. Furthermore, the dilution causes a decrease in EC. A study by Mishra (2010) however, found that DO negatively correlated whilst EC was positively correlated in PC1. This was due to the high levels of organic matter found in the urban wastewater which consumed DO (Mishra, 2010). PC2 explained 29.92% of the total variance and is highly positively contributed by rainfall. PC3 explained 18.73% of the total variance, showing a strong positive loading for temperature and pH.

Considering the results from the PCA, most of the variation in the water is explained by discharge, DO and EC with substantial contributions from rainfall and minor contributions from pH and temperature. According to Pejman *et al*. (2009) water quality parameters that had a high correlation coefficient value (>0.75), were considered to be significant contributors to variations in water quality.

Appendix C: South African Water quality Guidelines

Table C1: Water quality guidelines for EC.

EC		
South African Water Quality standards	**Water quality target range**	**Comment**
Domestic use (DWAF, 1996a)	≤700 µS/cm	No obvious health effects are likely above this limit but the water will have a saltier taste to it
Industrial use (DWAF, 1996c)	≤150 µS/cm	No damage due to scaling or corrosion below this limit
Agricultural use: irrigation (DWAF, 1996c)	≤400 µS/cm	No damage to salt-sensitive crops below this limit
Aquatic systems (DWAF, 1996f)	150 – 500 µS/cm (Behar *et al.*, 1996)	This is the acceptable range for freshwater streams to support diverse aquatic life

Table C2: Water quality guidelines for DO.

DO		
South African Water Quality standards	**Water quality target range**	**Comment**
Agricultural use: Aquaculture (DWAF, 1996e)	5.0 – 9.0	This range can accommodate cold, intermediate and warm water species
Aquatic systems (DWAF, 1996f)	• 4-5 mg/L for warm water biota and 5-6 mg/L for cold water biota (Davis, 1975) • 80% - 120% saturation	This range can accommodate all species without causing stress on the organisms

Table C3: Water quality guidelines for pH.

pH		
South African Water Quality standards	**Water quality target range**	**Comment**
Domestic use (DWAF, 1996a)	6.0 – 9.0	Non-toxic as metal ions do not readily dissolve in this range
Recreational use (DWAF, 1996b)	6.5 – 8.5	Minimal eye and ear irritation in this range
Industrial use (DWAF, 1996c)	7.0 – 8.0	Little to no tendency to cause corrosion and scaling in this range
Agricultural use: irrigation (DWAF, 1996d)	6.5 – 8.4	No problems with unavailability of plant nutrients in this range
Agricultural use: Aquaculture (DWAF, 1996e)	6.5 – 9.0	Majority of species can tolerate and reproduce within this range
Aquatic systems (DWAF, 1996f)	6.5 – 9.0 (US EPA, 1986)	The pH value should not change by more than 5% of the background value for a specific site and time of day

Appendix D: EC vs Discharge scatter plots

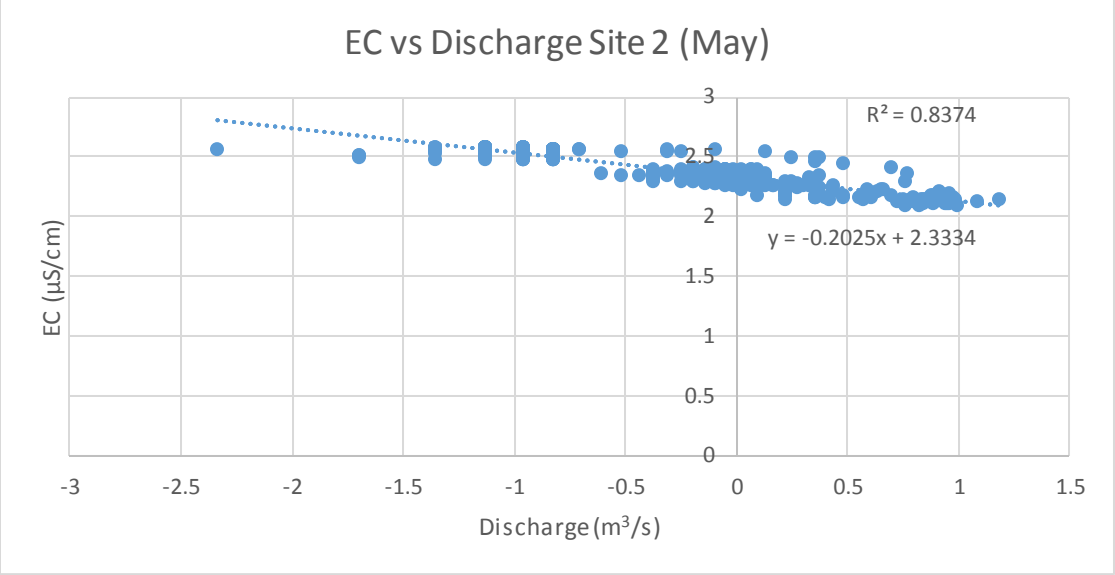

Figure D1: Log10 of EC vs Discharge for site 2 (May)

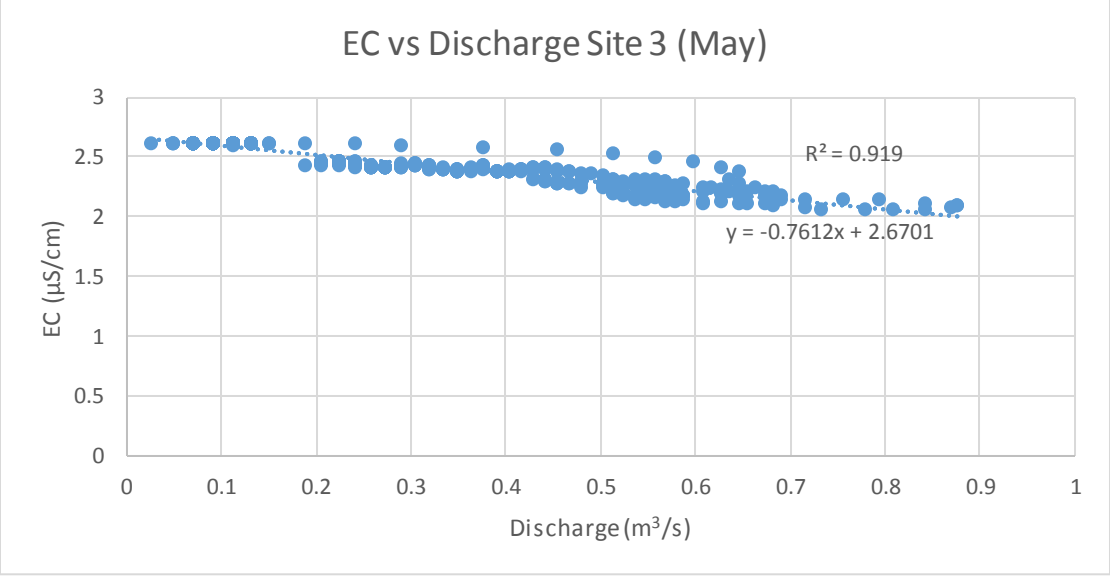

Figure D2: Log10 of EC vs Discharge for site 3 (May)

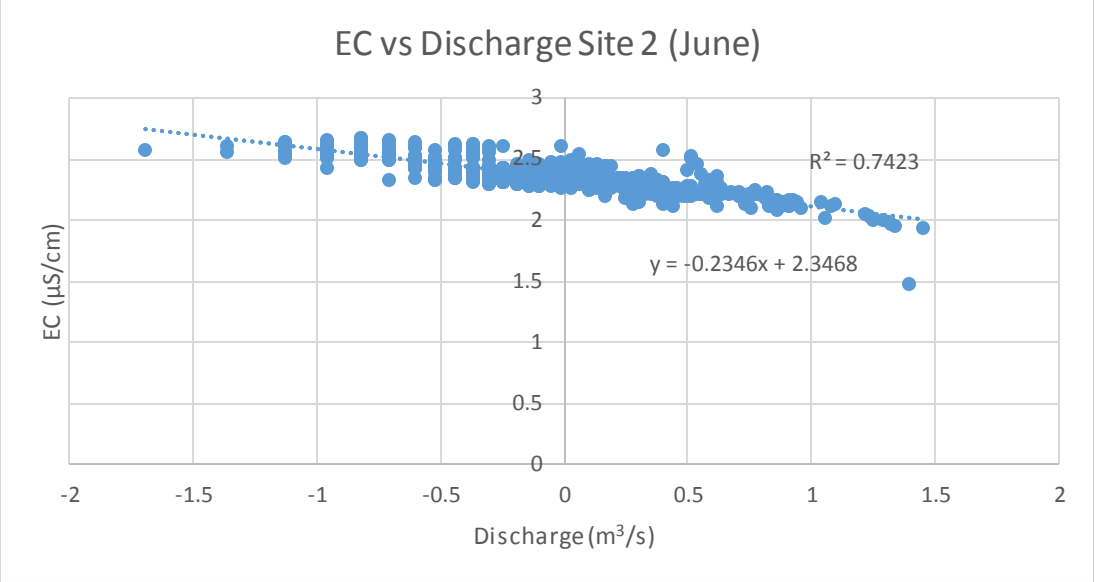

Figure D3: Log10 of EC vs Discharge for site 2 (June)

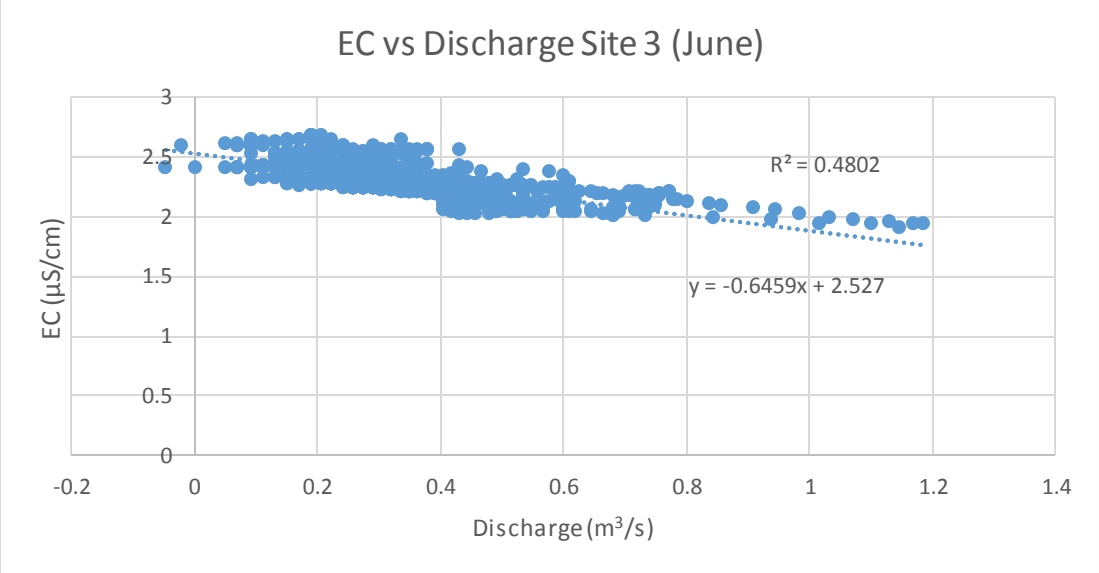

Figure D4: Log10 of EC vs Discharge for site 3 (June)

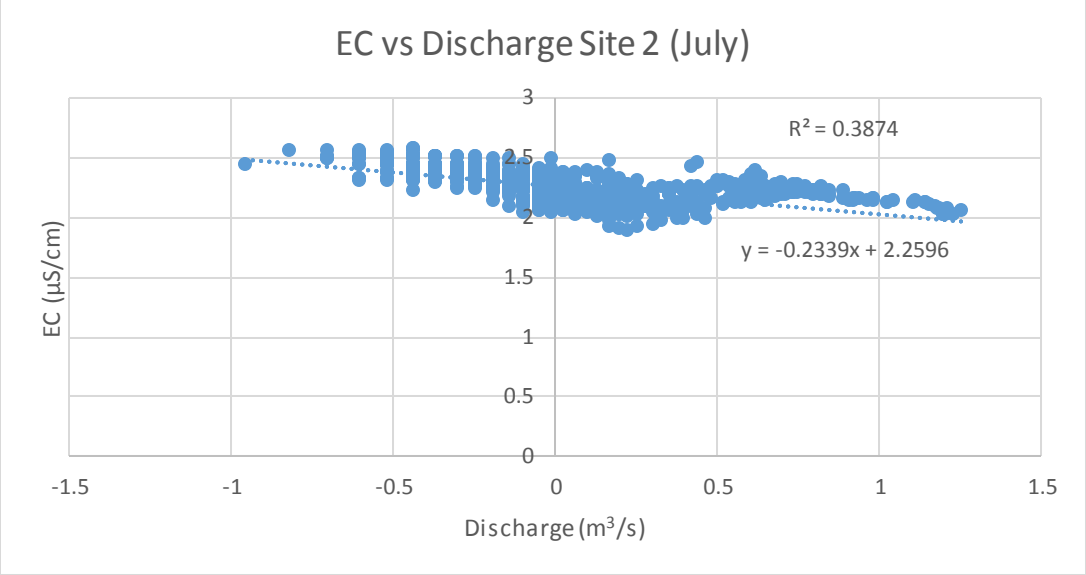

Figure D5: Log10 of EC vs Discharge for site 3 (July)

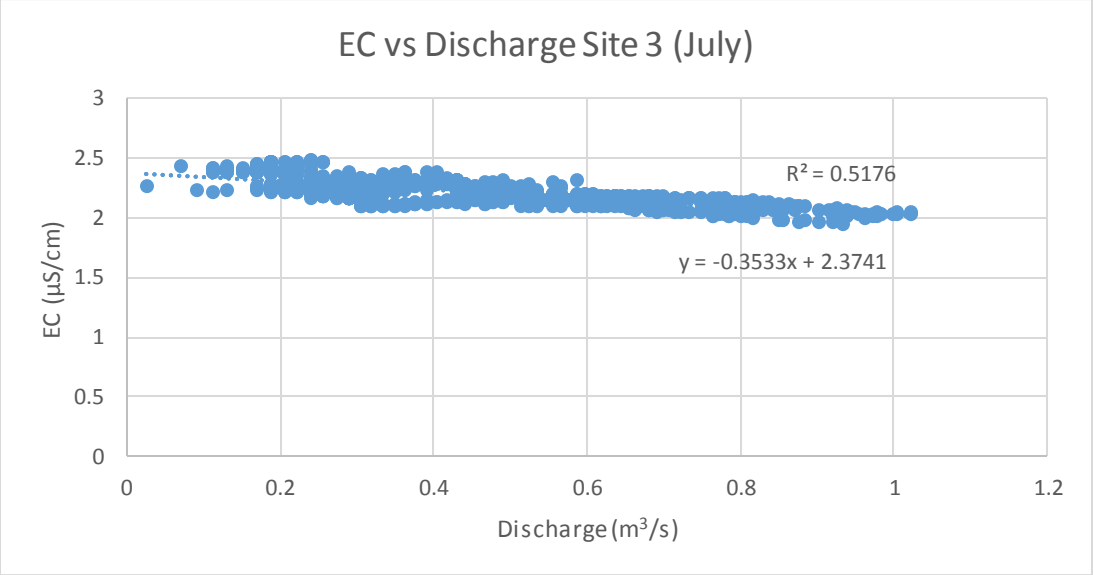

Figure D6: Log10 of EC vs Discharge for site 3 (July)

Appendix E: Rainfall design example used in PCSWMM

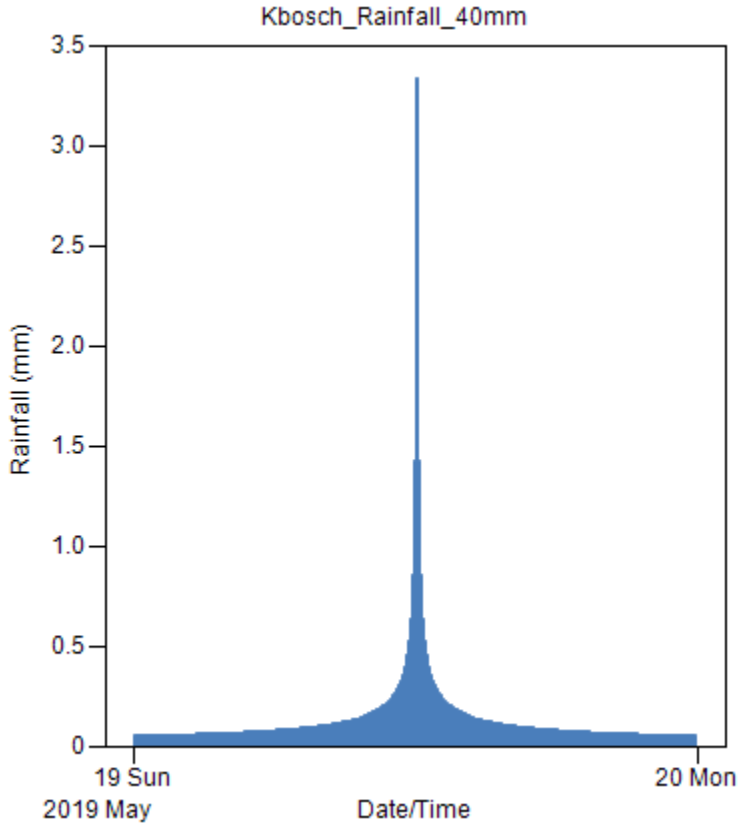

Figure E1: Example of a rainfall design used in the model for the Kirstenbosch rain gauge.