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Assessment of the water quality and quantity of the upper Liesbeek River dominated by Cannon Spring discharges: Ecological considerations for the Cannon Spring development

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A thesis submitted in fulfilment of the requirements for the degree of Magister Scientiae in the Department of Earth Science, University of the Western Cape

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ABBREVIATIONS

ADV	Acoustic Doppler Velocimeter
ASPT	Average Score per Taxon
CoCT	City of Cape Town
DO	Dissolved Oxygen
DWAF	Department of Water Affairs and Forestry
EC	Electrical Conductivity
GSM	Gravel, Sand & Mud
ICP-OES	Inductively Coupled Plasma Optical Emission Spectroscopy
ICP	Inductively Coupled Plasma
IHAS	Invertebrate Habitat Assessment System
MV	Marginal vegetation
SANS	South African National Standard
SASS5	South African Scoring System Version 5
SIC	Stones In Current
SOOC	Stones Out Of Current
SWSA	Strategic Waste Source Area
Tiamo	Titration And More
TDS	Total Dissolved Solutes
USA	United States of America

ABSTRACT

Assessment of the water quality and quantity of the upper Liesbeek River dominated by Cannon Spring discharges: Ecological considerations for the Cannon Spring development

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The ecology of spring fed rivers has been under-studied in South Africa. As a result, little is known or documented on the effects of seasonal variation of flows on the species diversity, distribution and abundance of aquatic macroinvertebrates in rivers dominated by spring discharges in, South Africa. In order to expand our understanding of the structure and functioning of spring fed rivers, the study determined and compared the current ecological state of the spring fed Liesbeek River tributary and the non-spring fed Disa River focusing on discharge, water quality and macroinvertebrates.

Two sites were selected from the Liesbeek River tributary representing a direct spring fed river and another two from the Disa River representing a non-spring fed river. The Liesbeek River originates from the natural Table Mountain sandstone catchment above Kirstenbosch and flows through Cape Town surbubs. The Disa River also rises on the western slopes of Table Mountain but above Orangekloof nature reserve and flows through Hout Bay residential areas. Both rivers experience similar weather conditions. Sampling of water quality and river discharge occurred once monthly and quarterly for aquatic macroinvertebrates for a period of a year (August 2018-July 2019). Aquatic macroinvertebrates were sampled following the South African Scoring System (SASS5) method. Onsite river discharge measurements were also determined following the velocity area method using an Acoustic Doppler Velocimeter (ADV) flow tracker2 device. Historical river flow data was obtained from the South African Water & Sanitation Department for the Liesbeek River dating from 1920 – 2005 and the data was also used to confirm onsite river discharge measurements. The habitat integrity was assessed following the Invertebrate Habitat Assessment System method. Water samples were also collected and analysed at the City of Cape Town municipality, Scientific Services laboratory. *In situ* water quality measurements were also conducted using a YSI multi-parameter probe for (pH, electrical conductivity, temperature and dissolved oxygen). The t-test was used to compare different rivers in terms of discharge, water quality and macroinvertebrate community.

The results suggested that there was no significant difference between the river discharge of the Disa and Liesbeek Rivers over the study period. Also, the inflow received from the spring feeding the Liesbeek River was not significantly different from the discharge in the Disa River. The water quality results showed that <50% of the assessed water quality parameters differed significantly between the rivers. The concentrations of all Disa River water quality parameters was generally higher than of the Liesbeek River water quality parameters. These water quality parameters that significantly differed between the rivers and were higher at the Disa River were; iron, chloride, aluminium, sodium and total dissolved solids. The ecological state of both the Disa and Liesbeek Rivers were largely natural characterized by sensitive aquatic macroinvertebrates such as Teloganodidae, Baetidae and Barbarochthoriidae. There was no difference in aquatic macroinvertebrates assemblage composition between the studied rivers with the exception of Amphipoda, which were dominant at the Liesbeek River but did not occur in the Disa River. Despite varying water quality parameters in the study rivers, macroinvertebrates did not show a particular preference for either of these rivers.

Keywords: Spring fed river, non-spring fed river, aquatic macroinvertebrates, flow, water quality, ecological state, habitat, SASS5, t-test.

DECLARATION

FACULTY OF NATURAL SCIENCES

I **Sizeka Felicia Magutywa**, student number **3031064** declare that the thesis titled "*Assessment of the water quality and quantity of the upper Liesbeek River dominated by Cannon Spring discharges: Ecological considerations for the Cannon Spring development* " is my own work, that it has not been submitted for any degree or examination in any other university, and that all the sources I have used or quoted have been indicated and acknowledged by complete references.

Full name: Sizeka Felicia Magutywa Date: 2021 January 29

Signed:

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1. INTRODUCTION

1.1 Background

Several rivers are highly dependent on spring discharges. A spring is where groundwater discharges to the ground surface creating a visible flow. Springs frequently occur in headwater regions (Reiss and Chifflard, 2017) with clear-water ecosystems (Odum, 1957; Heffernan *et al*., 2010). Springs may have perennial or intermittent flows, and can vary widely in size. Spring fed rivers, occurring in natural environments often have low pollution levels that do not fluctuate extensively and have uniform temperature regimes compared to rivers dominated by surface runoff (Steigerwalt, 2005).

Due to stable flows and temperatures, spring fed rivers provide refuge for fishes and invertebrates from adjoining rivers during periods of low flow and extreme temperatures (Hayes *et al*., 2018). Moreover, most spring fed rivers have minimal seasonal variation in discharge as the flows are moderated by groundwater recharge (Lusardi *et al*., 2016). In turn, most spring fed rivers are often deeply incised, have relatively uniform rectangular channel form, and few bars (Allen and Hay, 2011) and most depend on surface flows that drain into the same river. However, water abstraction can greatly reduce the flow thereby disrupting the stable flows of rivers dominated by spring discharges.

Water abstraction from rivers and springs cause alterations of river flow regimes, which may adversely affect aquatic macroinvertebrates (Klein, 1979). Flow components including low flows (base flows), flow pulses, seasonality of flows, and interannual variability create conditions necessary to support natural-assemblage complexity (Poff and Ward, 1989; Poff *et al*., 1997; Richter *et al*., 1997). Alterations of the timing, duration, and magnitudes of flows may adversely affect physical habitats particularly of sensitive aquatic macroinvertebrates (Kennen *et al*., 2010).

Flows determine the physical, chemical and biological processes in rivers including characteristics such as substrate stability, habitat suitability and aquatic community composition (Costigan *et al*., 2017). At the spatial scale, the flow of water creates a hierarchical structure that shapes the different habitats such as pools, riffles and runs available for biota (Frissell et al., 1986). Aquatic macroinvertebrates have many adaptations to flow and some can only live within specific ranges of current velocity (Steigerwalt, 2005).

As a semi-arid country, South Africa receives rainfall with a high spatial and temporal variability with the mean annual rainfall being 500 mm/year, (Mukheibir and Sparks, 2003). This high variability contributes to a diverse range of aquatic ecosystems with species having to adapt to irregular flows in perennial, intermittent, or non-perennial (ephemeral) rivers (Eady *et al*., 2013).

Many studies have been conducted on the effects of seasonal variation of river flows on the diversity and abundance of aquatic macroinvertebrates (Mabidi *et al*., 2017; White *et al*., 2017). Chi *et al*., (2017) demonstrated in a study conducted in Three Gorges catchment in China that the macroinvertebrate communities in rivers varied as a function of seasons. Different hydro-morphological characteristics and water quality during the high discharge period (winter), low discharge period (summer) and normal discharge period (spring and autumn) strongly affected the distribution patterns of macroinvertebrate communities. The seasonal variations of flows resulted in seasonal changes in the abundance of aquatic macroinvertebrates. However, little is known or documented on the effects of seasonal variation on the species diversity, distribution and abundance of aquatic macroinvertebrates of rivers dominated by spring discharges in the City of Cape Town, Western Cape, South Africa.

The mechanisms influencing species abundance and diversity in spring fed rivers are complex, but the need to understand them is urgent because river ecosystems are changing rapidly as a result of land use, water abstractions, and climate change (Stewart *et al*., 2005). Most of the valuable spring water sources are modified, to different degrees by human activity.

1.2 Research problem

The headwaters of most rivers draining mountain areas are spring fed (Wu, 2009). Most spring fed rivers, which are maintained by permanent groundwater discharge, constitute ecosystems with stable flow regimes. Owing to their high stability, spring fed rivers often function as high quality flow refugia for aquatic organisms (Sakai *et al*., 2020). According to Crossman *et al.* (2011), groundwater fed rivers are an important habitat for macroinvertebrate communities within glaciered catchments and studies within alpine areas have also suggested that rivers fed by groundwater may support a higher abundance of taxa than those fed largely by surface snow and ice-melt. The difference in the abundance of taxa is attributed to characteristically high water clarity, and reduced variability in stream temperature and discharge of groundwater fed streams (Brown *et al*., 2003).

The outstanding water clarity, temporal stability, including water chemistry of spring fed rivers supports a high diversity of fauna and flora (Cowell and Dawes, 2008). However, this may not be true for all spring fed rivers as the underlying geology of the spring, and catchment landuse play a pivotal role in groundwater quality. In agricultural catchments, elevated nitrate and pesticide concentrations in groundwater can result from both past and present land use activities, such as commercial or residential fertilizer application (Lawniczak *et al*., 2016). The nutrient-enriched groundwater may enter the surface water system via springs. In many instances, increases in nutrient concentrations in rivers have been linked to changes in autotrophic community composition, plant biomass and an increase of nuisance species. Such changes can, in turn, affect shifts in community structure and alter food web dynamics of a given system (Hershey and Fairbridge 1998; Notestein *et al*., 2003).

Due to exceptional water clarity and quality, spring fed rivers are becoming important water sources especially for urban water supply worldwide (Carrard *et al*., 2019). Dependence on groundwater for public and/or private water supply is a fast increasing phenomenon in developing cities worldwide in response to population growth, accelerating urbanisation and increasing use of water (Foster, 2020). This is aggravated by population growth as more than 50% of people in the world now live in cities and more than 75% live near rivers (Hommann and Lall, 2019). The use of spring fed rivers/groundwater as water sources is also a local phenomenon in Cape Town, Western Cape, South Africa (Wu, 2009; Lapworth *et al*., 2017; Nel *et al*., 2017).

Intensive water abstraction and regulation cause river ecosystems to shift towards non-natural flow regimes, which may have implications for water quality, structure of biotic assemblages and functioning (Sabater, 2018). Dams alter the natural distribution and timing of river flows (Bergkamp *et al*., 2000), water quality, sediment transport and channel structure (Mantel, 2010). This has ecological impacts such as, changes in aquatic systems due to loss of connectivity to the river (Kingsford, 2000), migration of species, and the reduction of flow downstream. Other ecological impacts include the limitation of lateral exchanges of sediments, nutrients and organisms between aquatic and terrestrial areas due to fewer overbank floods (Bednarek, 2001) and the fragmentation of habitat with associated isolation of populations (Benstead *et al*., 1999).

Water abstraction reduces summer flows and available habitat for the aquatic biota (Benejam *et al*., 2010). Water abstraction also alters river flow regimes, which in turn affect aquatic macroinvertebrates and other aquatic animals by altering physical habitat, interrupting life histories, limiting or increasing longitudinal and lateral connectivity (depending on the nature of the alteration), and facilitating the invasion and success of introduced species (Brooks *et al*., 2010).

The effects of flow alteration on river ecosystems have been widely studied worldwide (Poff and Zimmerman 2010; Poff et al 2010; Rolls and Bond 2017). Systematic reviews have found strong evidence of the effects of flow regulation on ecosystems. For example, 92% of the studies examined by Poff and Zimmerman (2010) reported ecological effects associated with flow regulation.

The ecology and hydrology of river ecosystems of both rivers fed by runoff and springs are well documented in many regions in the world (Sear 1999; Fuder et al 2001; Allan and Hay 2011; Yang et al 2012; Hannigan and Quinn, 2013; Lusardi *et al*., 2016). However, the river ecosystems of spring fed rivers have been under-studied in South Africa as most such studies have focused on thermal springs (Olivier *et al*., 2011; Jonker *et al*., 2013; Boekstein, 2014). There is a need to understand how the hydrology and ecological conditions of spring fed river ecosystems compare to runoff fed or non-spring fed rivers.

The ecology of a spring fed river was investigated on Table Mountain in Cape Town, Western Cape. The Liesbeek River tributary in Newslands Forest herewith referred to as the "Liesbeek River" and the Disa River, which both drain from the Table Mountain catchment, were investigated. The Table Mountain area in the Western Cape, Cape Town is rich in groundwater resources like springs and hence the area was selected as a suitable research area for the current study.

In order to expand our understanding on the structure and functioning of spring fed rivers, an investigation of benthos was conducted specifically focusing on the water quality, structure and taxonomic composition and the spatial and temporal distribution of benthic macroinvertebrates. The variation of river discharge or flow was also investigated. It should be noted that even though the Disa River is not fed by a notable spring, many rivers have significant groundwater discharge through diffuse flow along the banks and seepage. Groundwater can also flow to the surface naturally, discharge can occur as seeps, springs or groundwater flowing in or recharging wetlands and rivers (Halenyane, 2017). The Disa River occurs in a Strategic Water Source Area (SWSA) for surface water located in high rainfall areas where baseflow is at least 11-25 mm/annum, evidence of a strong link between groundwater and surface water in the SWSA-surface water. The aquifers sustain baseflow, especially during low-rainfall seasons (le Maitre *et al*., 2018). Therefore, the Disa River has groundwater flow input but no notable springs feeding the river.

The current study will advance existing knowledge on characteristics of aquatic macroinvertebrates, water quality and river discharge in spring fed rivers as opposed to non-spring fed rivers. The study findings may also be used as a way to inform holistic or other Ecological Reserve Determination methods for spring fed rivers. The findings of the study will also contribute to a proposal to abstract water from and develop the Cannon Spring and this study serves as a preliminary to that development. The findings of the study will advance the knowledge of the effects of seasonal flow variation on the aquatic macroinvertebrate community structure of spring fed rivers in Cape Town. Understanding of the existing spring fed river ecological conditions will provide the opportunity for enhancement of the management and protection of spring fed rivers and the opportunity to predict the impact of future events related to progressing anthropogenic pressure.

1.3 Aim

To identify and compare the current ecological state of the spring fed Liesbeek River tributary and the non-spring fed Disa River.

1.3 Objectives

1. To determine aquatic macroinvertebrate abundance, functional feeding group composition, and diversity and their relationship to hydrology/flow and water quality determinants of a spring fed and non-spring fed river.
2. To determine if there are specific aquatic macroinvertebrates that serve as indicators of spring fed rivers as opposed to non-spring fed rivers.

1.4 Research questions

1. How does river discharge, water quality and aquatic macroinvertebrates of a spring fed river compare to that of a non-spring fed river?

2. Do river flow variability and water quality play similar and important roles in structuring benthic macroinvertebrate communities in the non-spring fed Disa River and the spring fed Liesbeek River tributary?

3. Are there specific aquatic macroinvertebrates that serve as indicators of spring fed rivers as opposed to non-spring fed rivers?

1.5 Research outline

Chapter 1 introduces the relationship of river flow characteristics (low, high, floods, base and annual flows) and water abstraction on aquatic macroinvertebrate community structure in a river dominated by spring discharges. This section also elaborates on the research problem, aim and objectives of the study.

Chapter 2 provides a summary of discussions, theories and debates regarding the effects of seasonal variation of river flows on the diversity and abundance of aquatic macroinvertebrates in a spring fed and non-spring fed river are included in this chapter.

Chapter 3 describes the characteristics of the catchment in which the study was conducted and the sites that were selected to conduct the study.

Chapter 4 describes the research design, data collection and analysis methods.

Chapter 5 compares water quality and river discharge of a spring fed river (Liesbeek River) to that of a non-spring fed river (Disa River).

Chapter 6 presents the comparison of aquatic macroinvertebrates, functional feeding groups and indicator aquatic macroinvertebrates for the spring fed Liesbeek River.

Chapter 7 provides the general conclusions, limitations and recommendations of the study.

2. LITERATURE REVIEW

2.1 Introduction

This chapter reviews studies that have investigated the hydrology and ecology of spring fed and non-spring fed river ecosystems. Thereafter, the knowledge gap that will be addressed by this study was established.

2.2. Ecology of spring fed rivers in comparison to non-spring fed rivers

2.2.1 Flow/hydrology of spring fed rivers in comparison to non-spring fed rivers

Rivers are complex, hierarchical systems with three main interlinked components: the geological and geomorphological component which forms the basic physical template; the climatic and hydrological components, which are key abiotic drivers of the system through flow regimes, water quality and the biological component with a suite of species and communities, which have adapted to the conditions created by their interactions with the abiotic components (Le Maitre and Colvin, 2008). Flow regimes are widely regarded as the driving force of biological trends and patterns in lotic ecosystems. Flow regimes of most rivers are dominated by runoff events, but spring fed systems are primarily regulated by groundwater discharge and may show little to no response to local precipitation events (Lusardi *et al*., 2016). Most spring fed rivers are considered to have stable flows in comparison to rivers without spring flows, which is reflective of the underlying aquifer characgteristics (Sear *et al*., 1999).

River flows directly influence the physical habitat of rivers (Kozarek, 2016). As the flow velocity changes along the river due to changes in gradient from the upper to the lower reaches, different sized sediments are transported and deposited. The upper course, which is steep, is characterized by fast flowing water and large substratum as most of the finer material is transported downstream. In the middle and lower courses, finer material is deposited due to the gentle gradient and reduced flow velocity (Bain and Finn, 1988). This results in differences in the physical habitats along a river in terms of sediment particle size and heterogeneity, features of the channel and floodplain morphology and the distribution and extent of geomorphic features (Benson and Thomas, 1996), such as pools or alluvial bars and hydraulic controls, such as riffles and rapids. Spatially, rivers can be classified according to scale, from the drainage basin to localised instream habitats. Different geomorphological processes occur at different temporal and spatial scales. The classification is based on sediment distribution, flow and channel gradient along the river course (Montgometry and MacDonald, 2002).

Gomi *et al*. (2002) separated a river's longitudinal profile into three zones, the upper or headwater zone, where erosion prevails; the transition zone, where equilibrium is reached between erosion and deposition; and the lower zone, where deposition dominates. These zones are characterized by different flow conditions and habitats due to the varying flow speed, gradient and sediment distribution.

Rowntree *et al*. (2000) developed a zonal classification system for South African rivers (Table 1). This zonation is based on the channel slope and other variables, whereby the river channel is classified longitudinally on the basis of the varying slope from the source of the river to the mouth (Rowntree *et al*., 2000). The basin is the largest spatial scale of the landscape and comprises the land that contributes water and sediments to the river. Within the basin, rivers are divided into smaller segments, which are lengths of river channel with similar flow and sediment transport regimes (Rowntree *et al*., 2000). Segments are divided into zones, which comprise reaches. The reach also forms part of the classification system, and is a length of channel with similar gradient (Rowntree *et al*., 2000). The morphology of channel reaches is influenced by the slope of the channel and confinement, bed and bank material, riparian vegetation and the supply of water, sediments and wood from upslope (Allan, 2004). Channel features such as pools, riffles, bars and islands are comprised within channel reaches (Rowntree *et al*., 2000). These are basic structures recognized by fluvial geomorphologists as making up river channel morphology. The hydraulic biotope, which is the last attribute in the classification system, may be defined as a spatially distinct in-stream flow environment characterised by specific hydraulic attributes (Rowntree *et al*., 2000).

Table 1: Geomorphological zonation of river channels (Rowntree *et al*., 2000)

River zone	Gradient	Description
1. Source zone	not specified	Low gradient, upland plateau or upland basin able to store water. Spongy or peaty hydromorphic soils.
2. Mountain headwater stream	0.1 - 0.7	A very steep gradient stream dominated by vertical flow over bedrock with waterfalls and plunge pools. Normally first or second order. Reach types include bedrock fall and cascades.
3. Mountain stream	0.01 - 0.1	Steep gradient stream dominated by bedrock and boulders, locally cobble or coarse gravels in pools. Reach types include cascades, bedrock fall, step-pool, plane bed, pool-rapid or pool riffle. Approximate equal distribution of >vertical= and >horizontal= flow components.
4. Foothills (cobble bed)	0.005 - 0.01	Moderately steep, cobble-bed or mixed bedrock-cobble bed channel, with plane bed, pool-riffle, or pool-rapid reach types. Length of pools and riffles/rapids similar. Narrow flood plain of sand, gravel, or cobble often present.
5. Foothills (gravel bed)	0.001 - 0.005	Lower gradient mixed bed alluvial channel with sand and gravel dominating the bed, locally may be bedrock controlled. Reach types typically include pool- riffle or pool-rapid, sand bars common in pools. Pools of significantly greater extent than rapids or riffles. Flood plain often present.
6. Lowland sand bed or Lowland flood plain	0.0001-0.001	Low gradient alluvial sand bed channel, typically regime reach type. Often confined, but fully developed meandering pattern within a distinct flood plain develops in unconfined reaches where there is an increased silt content in bed or banks.
Additional zones associated with a rejuvenated profile		
7. Rejuvenated bedrock fall / cascades	0.01 - 0.5	Moderate to steep gradient, often confined channel (gorge) resulting from uplift in the middle to lower reaches of the long profile, limited lateral development of alluvial features, reach types include bedrock fall, cascades, and pool-rapid.

8. Rejuvenated foothills	0.001 - 0.01	Steepened section within middle reaches of the river caused by uplift, often within or downstream of gorge; characteristics similar to foothills (gravel/cobble bed rivers with pool-riffle/ pool-rapid morphology) but of a higher order. A compound channel is often present with an active channel contained within a macro-channel activated only during infrequent flood events. A flood plain may be present between the active and macro-channel.
9. Upland flood plain (UFP)	0.0001-0.001	An upland low gradient channel, often associated with uplifted plateau areas as occur beneath the eastern escarpment.

Spring fed rivers have certain characteristics that distinguish them from other types of rivers. Additional characteristics include small catchments, incised channels, relatively uniform rectangular channel form and flow inputs that are moderated by groundwater passage timesand as a result do not tend to experience floods that shape and maintain runoff river channels (Gordon *et al*., 2004; Griffiths *et al*., 2008; Allen and Hay, 2011). The stable flow regime cannot always transport logs and branches falling into the channel (Allen and Hay, 2011). In a study conducted by Whiting and Stamm (1995) on a spring fed river in Oregon Cascades, the results showed that the magnitude of baseflows was about 65% of the bankfull discharge, whereas the respective value for non-spring fed rivers was 10%. Flows exceeded bankfull 20% of the time whereas the typical value for non-spring fed rivers was 2 – 4%. Peak flows occurred in late summer or fall whereas peak flows in non-spring fed rivers in the study areas occurred with the spring snowmelt.

The discharge of groundwater to rivers via springs and seeps involves the outflux of groundwater from an aquifer, whereby the outflux is limited to a small portion of the aquifer. The discharge rate and volume of water from aquifers to rivers via springs are dependent on the type and properties of the aquifer (Mulligan and Charette, 2009). Classification of these is a function of water table location within the subsurface, its structure and hydraulic conductivities are determined by either confined or unconfined aquifers. Confined aquifers often have low permeability and storability while unconfined aquifers have high permeability and greater storability values (Salako and Adepelumi, 2017). Aquifers vary depending on the permeability and porosity of the rock type and include sandstones, conglomerates, fractured limestone and unconsolidated sand, gravels and fractured volcanic rocks (Christelis and Struckmeier, 2011). Some aquifers are characterized by high porosity and low permeability including granites and schist whilst those with high porosity and high permeability include rocks like fractured volcanic rocks (Javaid and Khan, 2018).

Aquifer properties may also influence the rate and timing of groundwater discharge (Alley *et al.,* 2002)). These properties include hydraulic conductivity, porosity and transmissivity. Hydraulic conductivity of a soil or rock or geological formation depends on a variety of physical factors amongst which, includes porosity, particle size and distribution and arrangement of particles (Salako and Adepelumi, 2017). Porosity is determined largely by the packing arrangement of particle sizes and the uniformity of its grain size distribution and connectedness of the pores to allow for water movement (Dippenaar, 2014).

Groundwater levels may also fluctuate depending on the storage of water in the aquifer. The levels vary depending on geology, topography, climatic season and anthropogenic activities (Brassington, 2017). In many semi-arid regions, groundwater can maintain river flows during the dry season (Le Maitre, 2008). Thus, when groundwater levels drop, so to does the groundwater discharge to the river, and if this drop is sustained, ultimately the groundwater will become disconnected from the river and the river will cease to flow in sustained periods of no rainfall (Kelly *et al*., 2019).

Groundwater abstraction/ pumping can reduce streamflow via the capture of groundwater that would have otherwise discharged into a stream. In extreme cases, pumping may even reverse the hydraulic gradient at the stream and induce infiltration from the streambed into the aquifer (Zipper *et al*., 2019).

In a study conducted by Zeng and Cai (2013), sustained groundwater withdrawal has caused water quantity and quality problems in the High Plains aquifer region, in the United States of America. For example, the average groundwater table in High Plains Aquifer has declined by 14 feet since the 1950s, with over 150 feet declines at some sites. The unsustainable groundwater withdrawal in this region has caused stream depletion and water right conflicts between surface water users and groundwater users. In a study conducted by Mukherjee *et al*. (2018), where the effects of groundwater abstraction on baseflow contribution to the Ganges River water in India was quantified, a decrease in discharge rates of groundwater to the Ganges River was reported. Moreover, in a study conducted by Condon and Maxwell (2019) in the United States of America the results showed decreased streamflow in response to groundwater abstraction. Annual volumetric streamflow declines of 10 to 50% were routinely simulated across the western portion of the domain. In the High Plains, where storage losses are greatest, there were also many locations with streamflow declines greater than 50% and a number of small tributaries that dry up completely.

2.2.2 Aquatic macroinvertebrates

Hydrological variability is a key determinant of both habitat structure and macroinvertebrate community composition in lotic ecosystems (Stubbington *et al*., 2010). The composition and structure of aquatic macroinvertebrate communities have been shown to change in response to longitudinal changes in habitat characteristics, such as temperature, current velocity, depth, width, discharge, substratum, turbidity, water chemistry and food availability (Barquin and Death, 2011).

The biotas of spring fed rivers are mostly diverse. Spring fed ecosystems are comprised of diverse complex ecosystems characterised by a varying degree of dependency on groundwater to maintain their composition and function (Boulton and Hancock, 2006). Spring fed rivers are characterized by stable environmental conditions that are coupled with naturally occurring nutrient inputs that may also enhance aquatic macroinvertebrate abundance when compared with runoff fed rivers (Barquín and Death, 2004).

Sear *et al*. (1999) investigated the possibility that a spring fed river has a specific set of ecological characteristics. Aquatic macroinvertebrate data, which included the faunal assemblages were collected from different spring fed rivers. The results of the study revealed that aquatic macroinvertebrates varied with the specific geology of rivers despite the common feature of being spring fed. The soft limestone, chalk and sandstone rivers had features in common but in the final analysis aquatic macroinvertebrates were most influenced by local hydraulic conditions which in turn were affected by local conditions (drift, geology, channel morphology) and distance downstream.

Lusardi *et al*. (2016) found that volcanic spring fed rivers supported seven times greater abundances but lower diversity of aquatic macroinvertebrates when compared with non-spring fed rivers. The differences were attributed to discharge, nutrient availability, and habitat playing particularly important roles. As the river discharge has an inverse relationship with aquatic macroinvertebrates over a certain part of the range of flow, hydraulic stress and sediment mobilization during peak discharge events probably contributed to the seasonal declines in macroinvertebrate abundance in non-spring fed rivers. The nutrient levels were significantly enhanced in spring fed rivers, which contributed to increased macroinvertebrate abundance at these sites as increases in nutrients can enhance primary production with cascading effects on consumers. The lower diversity in the studied volcanic spring fed rivers was attributed to several factors, including increased predation pressure by a dominant amphipod species.

Barquín and Death (2006) found that macroinvertebrate assemblages in New Zealand spring fed rivers were more diverse than those from non-spring fed rivers, and this result was corroborated by Tonkin and Death (2012). Barquín and Death (2006) suggested the pattern was caused by high environmental stability and resource levels in New Zealand spring fed rivers and that predator assemblages in these streams were more diverse than those in Spanish rivers. Furthermore, they suggested that high macroinvertebrate predator diversity in spring fed systems meant that abundant dependent mechanisms, such as predation, played an important role in regulating total macroinvertebrate diversity.

2.2.3 Water quality

Spring fed rivers typically occur in headwater regions of rivers and as a result normally exhibit pristine water quality. This is due to the fact that headwater regions are normally in locations with limited or little development, with minimal or no point sources of pollution that may affect river water quality and ecology. Moreover, water of groundwater origin is believed to be pristine, which may be influenced by the geological characteristics of the formation bearing the water (Wu, 2009).

There are a number of studies conducted on the water quality of spring fed rivers worldwide (Sear, 1999; Jefferson *et al*., 2001; Carrick *et al*., 2007; Brueggen, 2010). Cowell and Dawes (2008) revealed that spring fed rivers have outstanding water clarity and temporal stability, including water chemistry and velocity. However, this may not hold true for all rivers as some spring fed rivers also tend to have naturally low dissolved oxygen (DO) levels. Temperature, DO and flow regime are extremely important for biological communities (Palmer and Ruhi, 2019). Oxygen sensitive species will not be found in stretches of rivers with low oxygen saturation, and stenothermic species will favour habitats with a stable temperature regime. Conductivity, hardness and alkalinity are influenced by the underlying geology; therefore, in limestone areas or karst springs, high levels of hardness will be recorded and may result in the deposition of $CaCO_3$ on benthic substrates as well as on the biota (Hannigan and Quinn, 2014). Moreover, there are springs fed by groundwater and irrigation return flows that discharge saline water to rivers (Warner, 1984).

Thermal springs are the most under-researched and under-utilized of all natural resources worldwide including South Africa (Derso *et al*., 2015; Perry, 2018). The chemical content of the water in hot springs changes according to the chemical composition of rocks situated on path of the hot water flow (Rajapaksha *et al*., 2014). Olivier *et al*. (2011) indicated that assumptions that all spring water is pure should not be made, since many naturally occurring minerals are harmful or even dangerous to human and animal health. A number of studies have found that geothermal water may contain toxic elements such as arsenic and mercury (Mandal and Suzuki, 2002; Romero *et al*., 2003; Churchhill and Clinkenbeard, 2005), radio-active elements (Baradács *et al*., 2001) and pathogenic organisms such as the meningitis-causing *Naegleria fowleri* (Craun *et al*., 2005) and *Legionella pneumonia* (Miyamoto *et al*., 1997). Moreover, Rajapaksha *et al*. (2014). noticed the presence of algal species such as *Oscillatoria formosa*, *Synechococcus curtus, Fischerella thermalis* and *Anabaena* sp at temperature ranges of 53.8 – 55.8 degrees celcius from the hot water spring of Agnano, Italy.

Human activities at the landscape scale can impact river water quality (Allan and Johnson,1997). Rapid human population growth has resulted in worldwide land-use alterations, greatly influencing river ecosystems (Helms *et al*., 2009). According to Cowell and Dawes (2008), land use such as urbanization, farming, and horse and cattle raising, have increased nutrient input into the Rainbow spring fed river in Florida, United States of America. The nutrient concentrations increased more than 12 fold in the Rainbow spring fed river between 1957 and 2000, from 0.08 mg/L to over 1 mg/L. Such increases in nutrient concentrations often support toxic algal blooms associated with the cyanobacteria *Microcystis* and *Anabaena* species, as was the case for the Rainbow River. Thus, the river can no longer be considered pristine. However, Reiser *et al*. (2004) stated that most spring fed rivers are often low in nutrients, and other studies note that the nutrient status is influenced by the underlying geology and type of aquifer that feeds the spring (Biggs and Close, 1989; Biggs and Kilroy, 2004). For example, streams fed via a spring from a shallow unconfined aquifer in an agricultural area would likely have a different chemical composition from those fed via a spring from a deep confined aquifer. Spring fed river systems in agricultural areas can have high nutrient levels, particularly nitrate, given its propensity to leach from soils. Springs in areas with volcanic geology may also be enriched with phosphorus (Reiser *et al*., 2004).

2.3 Landuse and landcover effects on river ecosystems

River ecosystems are affected by characteristics which may be either natural or human made. Natural characteristics include the geology of rocks, climate change and natural disasters including floods, droughts and earthquakes. Human made characteristics includes agricultural, industrial and urbanised areas.

Safe drinking water and sanitation are important for good human health, human survival and development and the conservation of an acceptable aquatic ecosystem health. However, human activities have affected the health of many river catchments worldwide (Pullanikkatil *et al*., 2016) and as a result, rivers have been severely degraded by many factors including the alteration of land use, by agriculture and urbanization, which are the most common in many countries in the world, including South Africa (Helms, 2008). River health is largely impacted by deterioration of water quality, habitat and streamflow alteration, removal of indigenous riparian vegetation, introduction of exotic species and the loss of aquatic species (Roux, 1999; Pullanikkatil *et al*, 2016). Urbanization, which is associated with wastewater treatment works, industries and residential areas (Deng, 2015), also affects patterns of environmental-structure and function (Walsh, 2006).

Agricultural land uses are sources of nitrogen and phosphorous to water bodies (Kitsios, 2004). High loads of nutrients entering the river can lead to eutrophication, altering the food web by increasing production rates and causing a decline in water quality. Alteration of the hydrological cycle by agricultural activities (Moss, 2007) occurs due to the reduced vegetation cover and soil compaction from machinery that reduces infiltration into the soil and therefore increase run off and thus modifying river flow (Alaoui *et al*., 2018).

A characteristic of both, agricultural and urban environments is the alteration of the river by modifying the morphology of the channel. This can include channel modification by dredging of new channels or channel straightening (Yeakley, 2014). Changing the channel morphology of rivers by widening and deepening, increases bankfull capacity and the hydraulic radius of the river channel, thereby either increasing or decreasing the energy and speed of the water flow. Channel straightening or concrete lining reduces channel roughness and increases flows (Fashae, 2015). In urban environments, the purpose for modification of the channel morphology is for flood control, land drainage, navigation, and reduction of erosion (Horsak *et al*., 2009). These changes can result in the loss of instream habitats, thereby reducing the abundance and diversity of the aquatic biota (Gorney *et al*., 2012).

Increased imperviousness is characteristic of urbanized areas as development is progressing due to rapid population growth. Imperviousness increases the occurrence of flooding events, changes the magnitude and frequency of flows to and within a river channel due to reduced infiltration and uptake of water by plants, which also alters the rivers hydrological pattern (Konrad, 2003).

There is widespread evidence that freshwater ecosystems, rivers and wetlands in particular, are amongst the most threatened ecosystems in South Africa due to the impact of land use such as urbanisation and agriculture (Skowno *et al*., 2019).

The health status of several rivers in Cape Town is significantly poor. These rivers are characterized by canalization, poor water quality, modified flows and invasion by alien flora and fauna (Mwangi, 2014). The upper river reaches of most rivers in Cape Town, including the Liesbeek, Lourens and Eerste rivers, are generally in a natural condition as they occur in unhabitated mountainous areas. However, the river health status is mostly poor from the middle to lower river reaches where development intensifies thus impacting on the health of these river reaches.

Canalization, which is one of the impacts affecting Cape Town's rivers, reduces the river's ability to attenuate floods and decompose pollutants and is evident along the Black, Elsieskraal and Keysers Rivers in the City of Cape Town (Collins and Herdien, 2013). The Big and Little Lotus rivers are largely canalized along most of their reaches (Luger, 1998). These rivers have poor water quality as some parts serve as conduits for discharging treated wastewater effluent. Moreover, the water quality of these rivers is further impacted by the stormwater from informal settlements and back-yard dwellings (Collins and Herdien, 2013). The resultant nutrient loading leads to multiple effects, including algal blooms, which are characteristic in some rivers, especially in summer months when algal growth is proliferated.

Altered river flows also affect Cape Town's rivers, due to water abstraction and alien plants, which use more water than native plants in most river catchments including the Diep, Sand, Sir Lowry's Pass and Silvermine Rivers. These flow modifications have reduced habitat availability for aquatic life (Davies and Day, 1998).

2.4 Methods used to investigate river ecosystems

2.4.1 Aquatic macroinvertebrates

Changes in aquatic invertebrates can be assessed using an Index of Biotic integrity, which detects divergence from biological integrity attributable to human actions. The Index of Biotic Integrity, first introduced by Karr (1981), consists of metrics representative of several attributes of assemblage structure, composition and function. These metrics are chosen on the basis of their response to perturbation and ability to discriminate between minimally disturbed reference sites and sites known to have been influenced by perturbation. These methods measure faunal diversity and pollution tolerance and then rank sites against reference conditions. The metrics are scored and added to arrive at an index ranging from 60 (best) to 12 (worse). In an instance where there is nothing living in the water, the index score can have a value of zero.

The South African Scoring System 5 (SASS version 5) is the standard method for the bio-assessment of South African rivers (Dickens and Graham, 2002). It is a rapid bio-assessment method based on benthic macroinvertebrates, whereby each taxon is given a sensitivity or tolerance score according to water quality conditions (Dallas, 1995). Aquatic macroinvertebrates are sampled from different biotopes, scored and the generated tolerance scores are summed to give the total SASS score. The SASS score is an indication of the taxon's sensitivity to pollution. Each of the SASS indices or variables, which include SASS score, number of taxa, and average score per taxon (ASPT), provide significant information about the biological condition of a river. The higher the score, number of taxa and ASPT, the better the biological condition or health of the river (Dickens and Graham, 2002).

Several southern African countries have subsequently utilised SASS for assessing the status of river systems (Mangadze *et al.*, 2019). In Namibia, Botswana and Zambia, SASS has been modified and standardized into the Namibian Scoring System (NASS) (Palmer and Taylor, 2004), Okavango Assessment System (OKAS) (Dallas, 2009) and Zambian Invertebrate Scoring System (ZMSS) (Lowe et al., 2013) to account for additional tropical invertebrate taxa that occur specifically in these regions.

The sampling of aquatic macroinvertebrates can also be conducted using the Surber sampling device (Guild *et al*., 2014). Using this device, macroinvertebrate samples are taken from shallow riffles and or runs because flowing water aids in carrying macroinvertebrates into the collecting net as the benthos is disturbed (Merritt and Cummins1996). The sampler frame is placed on the streambed to collect aquatic macroinvertebrates that inhabit sediment or gravel of stream beds. The collected organisms or species are preserved in ethanol ranging from 70 - 95% and identified in the laboratory.

The River Invertebrate Prediction and Classification System (RIVPACS) is a UK-based statistical model that enables the estimation of ecological health of running water sites (Clarke *et al*., 2003). It is aimed at monitoring water quality and pollution and is based on the assumption that the presence of certain taxonomic groups of invertebrates in rivers will depend on levels of certain chemical and physical variables. This model assesses a river by comparing the observed fauna with the target or expected fauna (Spelleberg, 2005). The reference sites for RIVPACS are chosen to cover short river stretches, which are of high ecological and chemical quality so that a wide range of physical types of running water sites across a geographical region is encompassed.. The method requires identification of aquatic macroinvertebrates to species level. The method has been adapted into various other versions around the world, including AUSRIVAS (Australian River Assessment System) in Australia (Chessman, 2021).

2.5 Summary

Studies on rivers fed by springs have been conducted worldwide (Fuder *et al.*, 1998; Sear, 1999; Allan and Hay, 2011; Lusardi *et al*., 2016). However, most of these studies lacked the integration of the components of the river ecosystems such as assessing the hydrology, water quality or ecology of rivers with spring flows. According to Lusardi *et al*. (2016) the hydrologic and geomorphic nature of spring fed rivers has been documented (Whiting and Stamm, 1995; Whiting and Moog, 2001) but there is limited understanding of how these systems function ecologically. Moreover, according to Hannigan and Quinn (2014), whether or not spring fed rivers host a distinct set of ecological characteristics remains unclear. Fureder *et al*. (2001) also indicated that the hydrology and physico-chemical features of spring fed rivers, have been studied intensively, while, the ecology of these rivers has received little attention.

In South Africa, there is a need to assess the influence of flow/hydrology on the ecology of spring fed rivers for a holistic understanding in order to provide linkages to different river components of the ecosystems of spring fed rivers.

Several methods for assessing the health of freshwater ecosystems and hydrology of rivers exist. For assessing the ecosystem health, aquatic macroinvertebrates are widely investigated using various methods including Index of Biotic Integrity, SASS5, Surber sampling and RIVPACS-like methods.

3. STUDY AREA AND SITE SELECTION

3.1 Introduction

In order to compare the ecology, water quality and hydrology of a spring fed river and of a non-spring fed river, two rivers were selected where one, the upper Liesbeek River, was fed by springs and the other, the Disa River, is without notable spring flows.

The upper Liesbeek River tributary originates from the Newlands area, which is rich in springs (Wu, 2009). The upper Liesbeek River and the Disa River drain Afromontane forests. Both rivers occurred in the same South West Coastal Belt Level 1 ecoregion, which groups areas with similar topography, altitude, slope, rainfall, temperature, geology and potential natural vegetation (Kleynhans *et al*.,2005). Both rivers therefore share similar physical and ecological traits (Ollis, 2005).There are no identifiable direct point sources of effluent, such as stormwater discharge or sewage spills discharging into the study rivers. The studied river reaches also occurred in the same geomorphological zone (mountain streams).

3.1. Location of study area

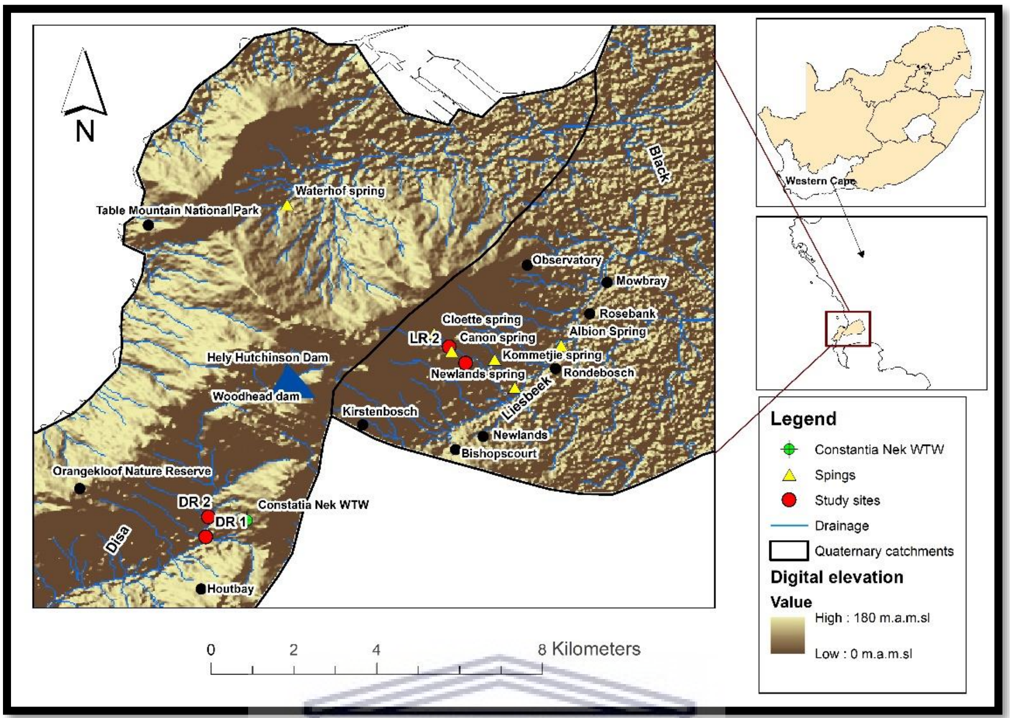

Figure 1: The location of the study area within the Western Cape Province of South Arica, the Liesbeek and Disa River catchments within the quaternary catchment G22C and G22B with springs and sampling sites on the Liesbeek and Disa Rivers

The Liesbeek River tributary drains into the Liesbeek River from the Newslands Forest (Figure 1). There is limited literature on this tributary, which has an average width of 1.6 metres and length 1.1 km. The Liesbeek River drains from Table Mountain and flows through the Newlands suburbs area, in Cape Town, South Africa. The Liesbeek River has the longest history of being urbanised in South Africa (Evans, 2007). According to Luger (1998), in 1652, Jan van Riebeeck recorded in his journal that the eastern slopes of Table Mountain, stretching down to the Liesbeek River, were covered by extensive forests, which were so dense from the top to the bottom, close to the river, that no opening could be found. Furthermore, the Liesbeek valley was described as “the finest and richest arable and pasture land in the world, wide and level, through which countless fresh rivulets wind, the largest of which was half as wide and quite deep”. The Liesbeek River occurs in quaternary catchment G22C and has a catchment area of 327 km^2 and is approximately 9 km long, fed by numerous streams flowing down the eastern slopes of Table Mountain, including the tributary fed by Canon Spring (Jeffess *et al*., 2017) (Figure 1). Quaternary catchments are hydrological units that are hierarchically nested from the primary drainage basin, through to secondary, tertiary and quaternary level (Nel *et al*., 2011). The headwaters of the Liesbeek River flow from the eastern slopes of Table Mountain where the vegetation is largely indigenous and undisturbed. The course of the Liesbeek River follows in a north-north-east striking fault zone (Brown and Magoba, 2009).

The Disa River rises on the western slopes of Table Mountain (Figure 1). The main stream of the Disa River is approximately 12 km in length and has a catchment area of approximately 34 km^2 (Grindley 1988), occurring in quaternary catchment G22B (Figure 1). The river flows in a south-westerly direction, through the Orange Kloof Reserve and thereafter through residential areas before discharging into the sea at Hout Bay (Ollis, 2005). The river is typical of the rivers of the Fynbos Biome, which are acidic, short, steep and fast flowing (Dawson, 2003).

The Liesbeek River tributary has clear water because it is almost entirely spring fed, and differs from the upper Disa River within the same geographic and catchment area, which is dominated by surface runoff originating as precipitation. Even though the Disa River has no notable groundwater inflow through springs, it may be receiving groundwater through seepage.

3.2 Drainage and groundwater

The Liesbeek River rises from a number of small streams that drain the eastern slopes of Table Mountain (Figure 1) (Brown and Magoba, 2009). A number of springs are found in the catchment, including the Cannon Spring, which discharges into the Liesbeek River tributary, which the study addresses. The most notable spring in the catchment is the Albion Spring, which discharges into the Liesbeek River and provides water to the South African Breweries site in Newlands (Figure 1). This spring is used to augment the City’s potable water supply and provides 4 Mℓ/day (Wu, 2009). Additional springs include the Newlands Spring, which has been used by the

South Africa Breweries for beer production since 1889. The Kommetjie Spring is shared by Breweries Company and local schools. The Waterhof Spring is used by the local community for domestic purposes. The Cloette spring, which also discharges into the Liesbeek River tributary upstream of the study sites Visagie (1995). The catchment also has additional groundwater sources and the private extraction of groundwater from boreholes also occurs (Wu, 2009).

The headwaters of the Disa River are controlled by the the Hely-Hutchinson and the Woodhead Dams situated on Table Mountain (Figure 1). During dry summer months, there is no release mechanism from the dams to the river. However, in winter the dams overtop into the Disa River.

Groundwater on Table Mountain occurs in two main aquifers. The Table Mountain Group (TMG) is highly fractured (Lin *et al*., 2007), and the quartzites of the Peninsula Formation make up a secondary porosity aquifer with high yields (>10 L/s) in boreholes and from springs (Diamond and Harris, 2019). The TMG Aquifer extends over 248,000 km^2 , of which 37,000 km^2 is exposed at the surface with a vertical thickness ranging from 900 to 4000 m (Wu, 2005). The TMG aquifer ranges between 50 – 2000 metres in depth (Harilall, 2020).

According to Rosewarne *et al*. (2002), groundwater associated with the TMG in mountainous regions is characterised by low pH. The low pH causes minerals to be readily dissolved resulting in high iron and manganese concentrations, i.e. >1 mg/l. The water quality of the Peninsula formation tends to be oligotrophic (low in nutrients), acidic and low in salinity, which is characteristic of water flowing through or over TMG formations (Waters *et al*., 2003).

The movement of groundwater in the TMG aquifer is highly variable and controlled by structure and lithology. Parts of the TMG aquifer are unfractured rock and therefore have low hydraulic conductivity and while the highly fractured rock have very high hydraulic conductivity (Waters *et al*., 2003). According to Rosewarne *et al*. (2002), a large percentage of the TMG aquifer's total thickness of >2 000 m consists of quartzitic sandstones of Ordovician to Silurian age (500 My) and because of their age and mild regional metamorphism essentially possess zero primary hydraulic conductivity.

The mean annual precipitation over mountains in the Western Cape exceeds 1000 mm/year. Wu (2005) conducted a detailed overview of recharge estimation in TMG area and found that recharge rates vary greatly, 0.3% to 83% of rainfall.

3.3 Geology and soils

The upper catchment of the Disa River is underlain by the Table Mountain sandstone, which overlies a granite base with a narrow band of shale at approximately 200 m (Grindley, 1984). These sandstones were deposited 320 million years ago and are the dominant rock in the Cape Peninsula (Grindley, 1984). Outcrops of the underlying Cape Granite rock are very sparse throughout the catchment area (Ollis, 2005). The Table Mountain sandstone above Kirstenbosch (Van Mazijk *et al*., 2018) also underlies the Liesbeek River.

According to Wu (2009) soils derived from Table Mountain sandstone are usually coarse sand and sandy clay. Soils produced by weathering are acidic, sandy and poor, chiefly due to the lack of feldspar. Quartz pebbles are released from the sandstone and occurs in the soil. The soil is generally shallow, and less than 0.5m thick (ref).

3.4 Climate

The Liesbeek and Disa Rivers fall in a mediterranean-type climate characterized by mild, wet winters and dry, warm summers (Rohli and Vega, 2011). In winter, the City of Cape Town (CoCT) receives approximately 100 mm of rain per month and in summer, less than 20 mm of rain per month is received (Figure 2). The average annual rainfall measured at a rainfall station at Newlands (Kirstenbosch) was 1245.6 mm and from a station in the Table Mountain region was 1522.2 mm was recorded as for the period 1998 to 2007 (Figure 2).

However, rainfall and evaporation are highly variable across the CoCT due to the mountainous landscape. Mountainous areas receive more rainfall than flatter landscapes, which often exceeds 2000 mm/annum. Lower rainfall areas such as the West Coast receive an average annual rainfall of only 300 mm/annum (Du Plessis, 2017).

The source of the Liesbeek River is affected by the presence of the Peninsula mountain chain to the west (Rohli and Vega, 2011)

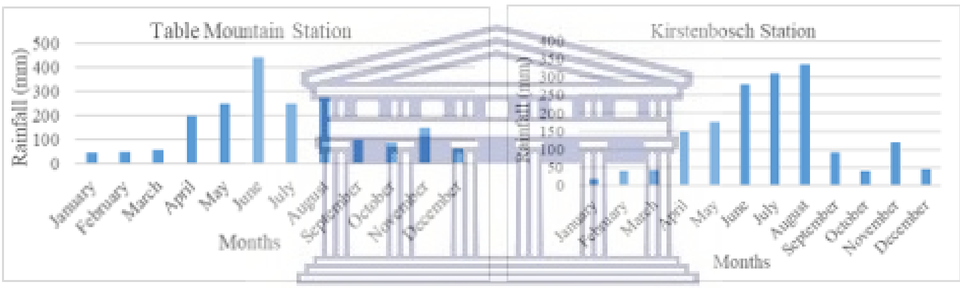

Figure 2: The records of two rainfall stations showing average precipitation patterns in the Newlands (Kirstenbosch) and Table Mountain regions for the period 1998 to 2007 adapted from (Wu, 2009).

3.5 Landuse

The study sites are all above surbuban developments (Figure 3). Approximately 50% of the Liesbeek River catchment is urbanized (Figure 3). Urbanization is dominant downstream of Kirstenbosch where the catchment is suburban with little natural vegetation (Luger, 1998). Urban land use within the catchment include residential housing, commercial offices, recreational sportsfields and small, light industries (Crisp, 2016). The middle to lower reaches of the catchment encompass the suburbs of Cape Town including Bishopscourt, Rondebosch, Newlands, Rosebank, Mowbray and Observatory with a large proportion of the catchment comprising impervious surfaces.

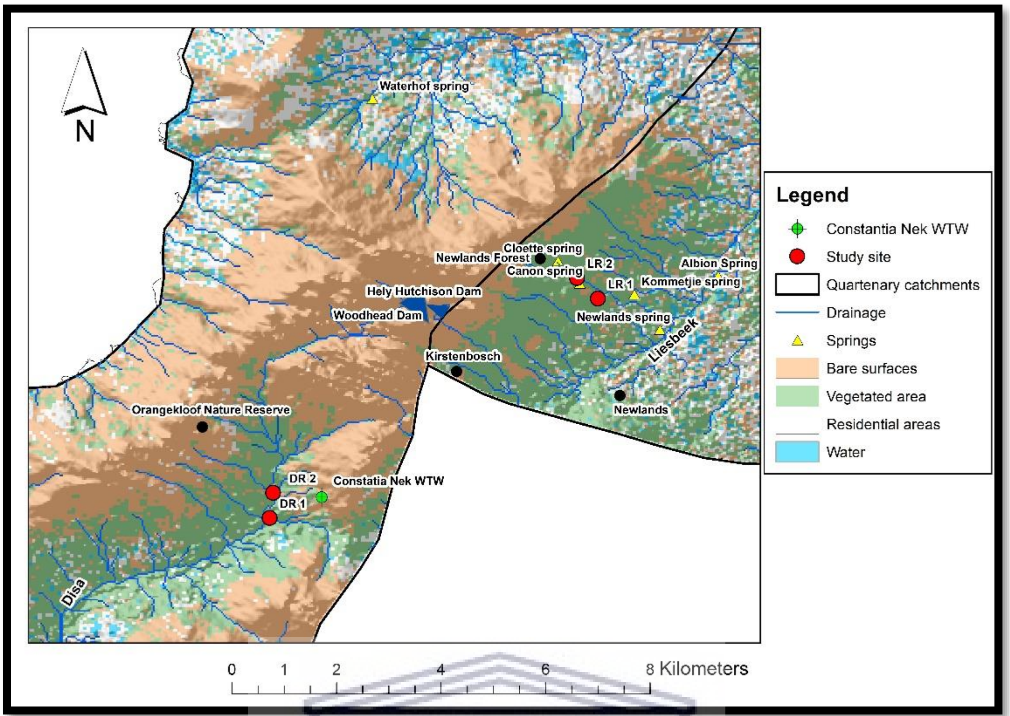

Figure 3: Landuse map of the study area.

The upper reaches of the Disa River flow through the Orange Kloof Reserve with the middle to lower reaches of the Disa River dominated by large peri-urban and urban properties. The Constantia Nek Water Treatment Work located at Table Mountain is a small seasonal plant that supplies drinking water to Hout Bay (Figure 3). The plant makes use of the following coagulants; aluminium sulphate, sodium aluminate and silica (sodium silicate and sulphuric acid mix). The sludge is discharged through a desludge pipeline that runs to sprayers located in an open field that is located in Orangekloof.

The lower Disa River catchment has gradually changed from agricultural to residential use. Hout Bay Village comprises of residential and commercial development, the informal settlement, and gardens and horse paddocks that extend to the rivers edge (Whithers, 2003).

3.6 Vegetation

The mountainous upper catchment of the Disa River is not developed and the river runs through natural vegetation and indigenous forest in Orangekloof

(Grindley, 1984). The natural vegetation along the upper reaches of the Hout Bay River is primarily Mountain Fynbos (in the Table Mountain National Park) and Afromontane Forest (in Orangekloof Reserve), with Sand Plain Fynbos grading into Dune Thicket in the middle to lower reaches below the study sites (Ollis, 2005).

The upper catchment of the Liesbeek River tributary is dominated by forestry plantations in the Newlands Forest and the Table Mountain National Park (Luger, 1998). The vegetation consists of a diverse array of Protea, Erica, geophyte and daisy species, as well as some endemic species. In the wetter areas, the Ericas predominate over the other plant groups. Along with the Granite Fynbos, this is by far the most diverse and richest in species of the ecosystems at Newlands Forest.

3.7 Sampling sites selection and description

The Liesbeek River (Figure 1) is comprised of two study sites referred to in the map as “(LR1)” abbreviated for Liesbeek River site 1,which is downstream of the point of discharge of the Canon spring and “(LR2)” abbreviated for Liesbeek River site 2 , which is the upstream site. Thus the study site was suitable to represent a river reach dominated by spring discharges. It should be noted the there is an additional spring, Cloette, feeding the Liesbeek River tributary furher above LR 2. The study sites served as the experimental sites of the study. The Disa River sites “(DR 1)” abbreviated for Disa River site 1 and “(DR 2)” abbreviated for Disa River site 2 flow through the Orangekloof Nature Reserve and represent non-spring fed river, thus serving as the control sites of the study.

The sites were all in the upper reaches of both streams, in a near pristine environment with the land use being a forested area or nature reserve. This enabled the comparison of the river reaches.The sites were also accessible. The sites at Disa River were below the Woodhead Dam, located at the headwaters of the Disa River (Figure 1). The sites are described below and sampling site characteristics are summarized in (Table 2).

Liesbeek River site 1 (LR1)

The reach assessed at this site flows through Newlands Forest. The site was characterized by a variety of substrates including bedrock, boulders, cobbles, pebbles and gravel (Figure 4). There was no marginal vegetation but patches of algae on some bedrock. The riparian vegetation on both river banks comprised of the following species: *Castanea sativa*, *Ulmus alata*, *Tradescantia fluminensis*, *Ehretia anacua*, *Quercus phellos*, *Elaeagnus pungens*, *Chiococca alba*, *Pinus pinea*, *Ligustrum lucidum*, *Cissus Antarctica.*

Figure 4: Liesbeek River site 1 (a) looking upstream and (b) looking downstream.

Liesbeek River site 2 (LR2)

The reach assessed at this site was flowing through the Newlands Forest, upstream of site LR1 (Figure 1). The site was also characterized by a variety of substratum including bedrock, boulders, cobbles, pebbles and gravel

(Figure 5). There was no marginal vegetation as well but only patches of algae on some bedrock. The riparian vegetation on both river banks comprised of the following species: *Castanea sativa*, *Ulmus alata*, *Ehretia anacua*, *Quercus phellos*, *Elaeagnus pungens*, *Pinus pinea*, *Ligustrum lucidum*, *Cissus Antarctica.*

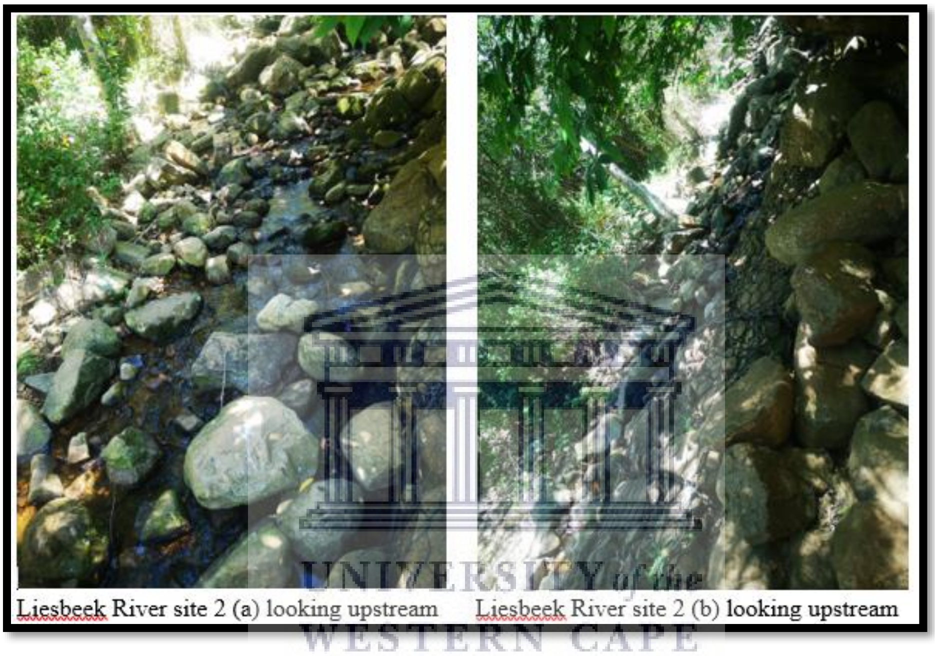

Figure 5: Liesbeek River site 2 a) looking upstream and (b) looking downstream.

<u>Cannon Spring</u>

Figure 6: Cannon spring flowing into Liesbeek River tributary.

Disa River site 1 (DR1)

The reach assessed at this site was flowing through Orange Kloof Nature Reserve and serves as one of the reference sites of the study as seen on (Figure 1). The site comprised a variety of substrates including bedrock, boulders, cobbles, pebbles and gravel. The water had a clear tea colour and aquatic, marginal and riparian vegetation was present (Figure 7). With the riparian vegetation on the right river bank largely dominated by grass and on the left dominated by trees and shrubs. The following species were present: *Prunus caroliniana, Dryopterix filix, Lonicera japonica, Ehretia anacua, Triadica sebifera, Prunus laurocerasus, Quercus robur, Pteridium aquilinum, Sorbus domestica, Carya cordiformis, Quercus agrifolia, Ulmus minor, Elaeagnus pungens, Lingustrum ovalifolium, Juglans nigra.*

Figure 7: Disa River site 1 a) looking upstream and (b) looking downstream.

Disa River site 2 (DR2)

The reach assessed at this site also flows through the OrangeKloof Nature Reserve and served as one of the reference sites of the study as seen in (Figure 1). The site comprised a variety of substrates including bedrock, boulders, cobbles, pebbles and gravel. The water also had a clear tea colour and aquatic, marginal and riparian vegetation was present as well (Figure 8). The vegetation on the right bank largely dominated by grass and on the left river bank by shrubs, trees, herbs and graminoids. The following species were present: *Prunus caroliniana, Dryopterix filix, Lonicera japonica, Ehretia anacua, Prunus laurocerasus, Quercus robur, Carya cordiformis, Quercus agrifolia, Ulmus minor, Elaeagnus pungens, Lingustrum ovalifolium, Juglans nigra.*

Figure 8: Disa River site 2 a) looking upstream and (b) looking downstream

Table 2: summary of study site characteristics

Site	GPS coordinates	Average discharge (m^3/s)	Substrates	Geomorphological zone	Stream order	Landuse impacts	Geormophic characteristics	Average stream (m)	Vegetation type	Disturbance
Liesbeek River site 1	-33.971111 18.447222	0.013	bedrock, boulders, cobbles, pebbles and gravel	Mountain stream	2	Natural forest	Step-pool morphology	1.64	Indigenous vegetation (mostly trees)	Instream woody debris present Localised erosion?
Liesbeek River site 2	-33.967555 18.443638	0.012	bedrock, boulders, cobbles, pebbles and gravel	Mountain stream	2	Natural forest	Step-pool morphology	1.57	Indigenous vegetation (mostly trees)	Instream woody debris present
Disa River site 1	-34.004580 18.391349	0.008	bedrock, boulders, cobbles, pebbles gravel and sand	Mountain stream	1	Natural forest	Step-pool morphology	1.53	Indigenous vegetation (mostly trees)	Instream woody debris present
Disa River site 2	-34.008915 18.390775	0.009	bedrock, boulders, cobbles, pebbles gravel	Mountain stream	1	Natural forest	Step-pool morphology	1.48	Indigenous vegetation (mostly trees)	Instream woody debris present

4. METHODS

4.2 Research design

Direct field measurements of water quality parameters, river discharge/flow and aquatic macroinvertebrates were made to achieve the study objectives. Aquatic macroinvertebrates were sampled following the the SASS5 method, attemping to ensure that macroinvertebrates and available habitats were evenly sampled (Dickens and Graham, 2002). Water quality and river discharge were measured at pre-selected sampling points providing a good representation of the river conditions. River discharge was measured at regular intervals across the river channel and this method has been used extensively in most water quality monitoring programs, usually because it is relatively simple, easy and the sample population is evenly sampled (Gilbert, 1987; Sharma, 2017).

4.2.1 Justification for river components assessed

4.2.1.1 Water quality

The understanding of the spatial and temporal variations in physico-chemical, biological and microbiological parameters is imperative for effective river monitoring and management (Vadde *et al*., 2018). The water quality information aids in water resource management and may also contribute to international environmental quality measurement programmes (Helmer, 1994). Selected water quality parameters included some parameters monitored for drinking water. This was done due to the fact that the findings of the study will be useful for any proposal to use water from the Cannon Spring for domestic water supply. The water quality results were assessed using the Department of Water and Sanitation water quality standards for aquatic ecosystems (DWAF, 1996).

In order to assess the water quality of the Liesbeek and Disa Rivers, selected water quality determinants were assessed (Table 3) with justification for selection provided.

Table 3: Water quality determinants assessed for the Liesbeek and Disa Rivers

Physical determinants	Water chemical determinants
Temperature	Iron as (Fe)
Conductivity	Lead as (Pb)
Total dissolved solids (TDS)	Aluminium as (Al)
Turbidity	Nitrate as (N)
Dissolved Oxygen (DO)	Sulfate as (SO_4^{2-})
pH	Ammonia as (N)
	Chloride as (Cl-)
	Sodium as (Na)

Temperature

All aquatic organisms have optimal temperature ranges for different life stages (Dallas and Day, 2004). Water temperature influences physical, chemical and biological processes in water bodies as the rate of chemical reactions and the metabolic rate of aquatic organisms' increase and decrease in response to changes in temperature (Dallas, 2008). Changes of water temperature adversely affect aquatic organisms. Anthropogenic temperature changes in river systems can be due effluent dispoals, stream regulation and changes in riparian vegetation. An increase in water temperature decreases oxygen solubility and may increase the toxicity of certain chemicals, both which result in increased stress in the associated organisms (Dallas and Day, 2004).

Total Dissolved Solids and Electrical Conductivity

Total dissolved solids (TDS) represent the total quantity of dissolved material, organic and inorganic, ionized and unionized in a water sample (Dallas and Day, 2004). The TDS affects freshwater communities, and determine assemblage composition, along with other factors like temperature, substrate composition, flow and the type of food available (Olson and Hawkins, 2017). Anthropogenic activities such as industrial effluents, irrigation and water re-use lead to increases in TDS. Very little information is available about the tolerances of freshwater organisms to increased TDS (Dallas and Day, 2004).

Conductivity is a measure of the ability of water to conduct an electrical current and TDS and conductivity usually correlate closely for a particular type of water (Dallas and Day, 2004). The electrical conductivity indicates the presence of salts or ions, which carry an electrical charge and is a useful measure of dissolved solids in water (Hur, 2012). Salts and other substances affect the quality of water used for irrigation or drinking. They also have a critical influence on aquatic biota, and every kind of organism has a typical salinity range that it can tolerate.

Turbidity

The turbidity of water affects clarity of water. An increase in turbidity or suspended solids affects light penetration, which may have far-reaching consequences for aquatic biotas (Dallas and Day 2004).Turbidity and transparency of water are determined by the concentration and nature of Total Suspended Solids (TSS) (Feroz and Bahnemann, 2021).

A continuous increase in turbidity due to high-level inputs may have very serious consequences for the riverine biota (Dallas and Day, 2004). Effects at each trophic level are mortality, reduced physiological function, and avoidance; however, decreases in available food at trophic levels also result in depressed rates of growth, reproduction, and recruitment (Henley *et al*., 2000). Increases in turbidity results in reduced light penetration, decreases primary production and food availability to organisms higher in the food chain (Dallas and Day, 2004). Suspensoids that settle out may smother and abrade riverine plants and animals. Community composition may change, depending on which organisms are best able to cope with this alteration in habitat. Predator-prey interactions are affected by the impairment of visually hunting predators. Nutrients, trace metals, biocides and other toxins adsorb to suspended solids and are transported in this form (Dallas and Day, 2004).

Dissolved Oxygen (DO)

The maintenance of adequate dissolved oxygen concentrations is critical for the survival and functioning of aquatic biotas (Dallas and Day, 2004). The determination of dissolved oxygen concentration is a fundamental part of water quality assessments because oxygen influences nearly all chemical and biological processes within water bodies (DWAF, 1996). Factors causing an increase in DO include atmospheric re-aeration, increasing atmospheric pressure, decreasing temperature and salinity, and photosynthesis by plants (Dallas and Day, 2004). Factors causing a decrease in DO include increasing temperature and salinity, respiration of aquatic organisms, decomposition of organic material by microorganisms, chemical breakdown of pollutants, re-suspension ofanoxic sediments and release of anoxic bottom water.The significance to aquatic biota of dissolved oxygen depletion depends on the frequency, timing and duration of such depletion. Continuous exposure to concentrations of less than 80% of saturation is harmful, and is likely to have acute effects, whilst repeated exposure to reduced concentrations may lead to physiological and behavioural stress effects (Dallas and Day, 2004).

pH

pH is an important water quality indicator. For example, humans require drinking water at a near neutral range while the pH of freshwaters ranges between 5 and 9, the range in which most aquatic organisms survive (Mwangi, 2014). Major changes in pH adversely affect aquatic organisms by disrupting the ionic and osmotic balance of an organism’s body tissue by altering the rate and type of ion exchange across body surfaces (DWAF, 1996). Some streams are naturally far more acidic than others and their biotas are adapted to these conditions (Dallas and Day, 2004).

Freshwaters draining catchments containing certain types of vegetation (e.g. fynbos, some forest types), may naturally have the pH drop to as low as 3.9 owing to the influence of organic acids (e.g. humic and fulvic acids and other polyphenol-rich compounds) leaching from the vegetation (Dallas and Day, 2004).

Iron

Iron concentration affects both aquatic and non-aquatic organisms. In rivers, iron serves as one of the significant nutrients for algae and other organisms (Vouri, 1995). High iron concentrations in rivers are considered an environmental problem with negative effects to the aquatic life. The negative effects of high concentrations of iron in rivers includes disturbing the normal metabolism and osmoregulation and by changing the structure of habitats and food sources, thereby leading to the decrease in the species diversity and abundance of fish, aquatic macroinvertebrate and periphyton (Alsaffar *et al*., 2016).

Lead

Lead is one of the toxic heavy metals most abundant in the environment and an emerging global concern due to its potential hazards on public health and aquatic life (Kumar *et al*., 2007). Lead and its compounds, if present in aquatic environments in high concentrations, can cause acute or chronic toxicity to aquatic organisms. However, the level of toxicity is determined by bioavailability factors such as water chemistry, solubility, salinity and organic matter content. Many of the adverse effects are reversible once exposure levels decline.

Aluminium

The presence of aluminium in rivers in high concentrations is known as a toxic agent to aquatic freshwater organisms. In aquatic animals breathing by gills such as fish and invertebrates, aluminium causes loss of plasma and haemolymph ions leading to osmoregulatory failure (Roseland *et al*., 1990). At high concentrations of aluminium, gills become clogged with colloidal forms of hydroxo aluminium complexes, whereas at lower concentrations the permeability of epithelium in fish-gills increases, which results in the loss of osmoregulation equilibrium of an organism (Bezak-Mazur *et al*., 2001).

Nitrates and Ammonia

Various plant nutrients are required for normal plant growth and reproduction. Most nutrients are not toxic (exceptions include nitrite and ammonia), even in high concentrations, but when present in aquatic systems in these high concentrations, they may have a significant impact on the structure and functioning of biotic communities (Dallas and Day, 2004).

Sources of nitrates in rivers include runoff from agricultural land, stormwater, treated wastewater effluent discharge and poorly functioning septic systems (Nkambule, 2016). Nitrates are one of the nutrients, which contribute to eutrophication and when nitrates are in excessive concentrations, may result in algal blooms and oxygen depletion in rivers. The oxygen depletion results in mortality of many aquatic organisms including fish and aquatic macroinvertebrates (Nkambule, 2016). Exposure to high levels of nitrate over a long time can affect aquatic macroinvertebrates, fishes and amphibians (Camargo *et al*., 2005). For example, an increased nitrate concentration from agricultural activities around Amala and Nyangores tributaries of the Mara River in Kenya resulted in a decline in the aquatic macroinvertebrate taxa diversity at downstream sites (Nyoh, 2015).

Ammonia can have adverse effects on aquatic macroinvertebrates when present in water at high concentrations as it is toxic to to fish and other aquatic life and may cause lower reproduction and growth, or death (DWAF, 1996). It may affect the respiratory system; reduce hatching and growth rates of different aquatic organisms (Nyoh, 2015).

Sulphate

Sulphate commonly occurs at elevated concentrations in wastewaters from industrial processes, in runoff from agricultural areas, and in natural waters draining areas of high mineralization (Elphick *et al*., 2010). Sulphates can contribute to an undesirable taste in water when in high concentrations (Moreno, 2009). Moreover, sulphate promotes methylation of mercury to its most toxic and bio accumulative form, methylmercury, which also promotes the release of nutrients from sediments resulting in eutrophication. Sulphate also enhances biodegradation of organic soils.

Chloride

Chloride is one of the essential elements of life with its concentration highly dependent on the geological, geographical and ecological conditions (Guedens *et al*., 2018). Chloride from anthropogenic sources is increasingly identified as a significant pollutant of rivers (Kauschal *et al*., 2005). High chloride concentrations may act corrosively and be harmful to plant life. High chloride concentrations can affect the aquatic environment by causing changes in water chemistry thereby affecting the health of aquatic organisms (Shambaugh, 2008).

Sodium

Sodium is a common element in the environment and occurs widely in soils, plants, water, and foods (Advisory, 2003). Sodium ion is ubiquitous in water, owing to the high solubility of its salts and the abundance of sodium containing mineral deposits. There are a number of anthropogenic sources of sodium that can contribute significant quantities of sodium to surface water, including road salt, water treatment chemicals, domestic water softeners, and sewage effluents. Sodium and other salts in drinking water may produce a laxative effect and reduce the suitability of a water supply for grazing animals (DWAF, 1996). Sodium is also one of the salts, which in excessive concentrations in water, can have drastic effects on the fitness and survival of freshwater organisms. According to Cañedo-Argüelles *et al*. (2019), the species richness declines along the salinity gradient in inland waters and laboratory toxicity tests show that most freshwater species are extirpated once a certain threshold of salinity is exceeded.

4.2.1.2 River discharge

The measurement of river discharge is an important component of most water quality monitoring projects (Meals and Dressing, 2008). Flooding, river geomorphology, and aquatic life are all directly influenced by streamflow. As the flow changes in speed along the river course or length due to varying gradients influencing the speed of flow from the upper course to the lower course, different sizes of sediments are transported and deposited(Bentley, 2012). River discharge measurements in natural watercourses are performed in order to determine the value of the surface outflow of a basin, its temporal variability, and the outflow characteristics (Tazioli, 2011).

4.2.1.3 Aquatic macroinvertebrates

Remarkable advancements in ecosystem monitoring have resulted in assessment methods that characterize the integrity of water bodies by sampling and summarizing the structure of associated faunal/floral communities/assemblages through selected biological indices (Karr, 1991; Resh, 2008). A comprehensive review of the existing biomonitoring literature showed that benthic macroinvertebrates are the most commonly cited (64% of sources) taxon for bioassessment of aquatic ecosystem health (Resh 2008), but algae (Chessman *et al*., 1999) and fish are also used (Resh, 2008).

Aquatic macroinvertebrates play a significant role in the functioning of freshwater ecosystems. These roles can include regulating the rates of primary production, decomposition, water clarity, thermal stratification and nutrient cycling (Giese *et al*., 2009). Aquatic macroinvertebrates can also be used for continuous monitoring of the water they inhabit enabling analysis of regular and intermittent discharges and variable concentrations and these organisms also integrate the effects of short term environmental variations and thus serve as indicators of biological integrity (Giese *et al*., 2009).

4.3 Data collection methods

Water samples were collected once monthly for a period of a year (August 2018- July 2019) and analysed following standard methods for the sampling and analysis of various water quality parameters (APHA, 2005). The samples were analysed at the Scientific Services Laboratory (SSB) of the City of Cape

Town municipality. The SSB is an accredited laboratory under the ISO/IEC17025 international standard. The sample bottles were collected from the designated laboratory sections (Biological and Analytical laboratories) prior to sampling to ensure compliance with the Laboratory accreditation system.

4.3.1 Water quality

4.3.1.1 Chemical and physical analysis

Clean, labelled one-liter polypropylene sample bottles were used to collect samples for chemical and physical determinants as in (Table 3). The sample bottles were cleaned being soaked in detergent solution for 30 minutes and scrubed with a soft brush or sponge and rinsed thoroughly with water.

The sample bottles were first rinsed with sample water then submerged 10 – 15 cm below the water surface. The sampling bottles were filled to the brim, sealed to prevent contamination and transported to the SSB laboratory in a cooler box with ice packs.

In situ measurements were conducted seasonally for electrical conductivity, temperature, pH and dissolved oxygen. Measurements were made using a YSI ProDSS multiparameter probe with the accuracy for each sensor specified in (Table 4). These water quality parameters as well as discharge were measured during aquatic macroinvertebrate sampling.

Table 4: Accuracy and detection limits for each variable measured using various instruments.

Determinand/water quality parameter	Accuracy	Lower detection limit
Electrical Conductivity	0 to 100 mS/cm (±0.5% of reading or 0.001 mS/cm, whichever is greater) 100 to 200 mS/cm (±1% of reading)	0
Total dissolved solids	(Not available)NA	NA
Turbidity	±7.7% of reading	0 NTU
pH	±0.2 units	0
Temperature ºC	±0.2°C	-5 ºC
Dissolved Oxygen DO mg/L	0 to 20 mg/L (±0.1 mg/L or 1% of	0 mg/L

	reading, whichever is greater) 20 – 50 mg/L (±8% of reading)	
Nitrate as N	±4.3% of reading	0.03 mg/L
Sulfate as SO_4^{2-}	NA	NA
Ammonia as N	NA	NA
Chloride as Cl^-	±2,5% of reading	0.02 mg/L
Sodium as Na	±2,8% of reading	0.751 mg/L
Iron as Fe	±3.3% of reading	3.837 µg/L
Lead as Pb	±4.1% of reading	0.037 µg/L
Aluminum as Al	±2,6% of reading	2.186 ug/L

4.3.1.2 Laboratory analysis

Enumeration of physical determinants

The total dissolved solids were analysed using the gravimetric method where a known volume of a well-mixed sample was filtered through a standard glass-fibre filter and the filtrate collected. The filtrate was evaporated to a constant weight condition in an oven maintained at a temperature of 180°C to remove mechanically occluded water. The mass of the dried dissolved solids was determined and used to calculate the concentration of total dissolved solids in the sample.

Other physical determinants such as chlorides, nitrates and sulphates were analysed using the discrete analyser machine (Aquakem 250: Thermo Scientific Aquakem Lab medics, Finland) with the accuracy and lower detection limit specified in (Table 4). A discrete analyzer is an automated chemical analyzer that performs tests on samples kept in discrete cuvettes in contrast to a continuous flow analyzer (SFA and/or FIA) that uses a peristaltic pump for a continuous stream of reagents.

Enumeration of Chemical determinants

The Inductively Coupled Plasma Optical Emission Spectroscopy (ICP-OES) Spectroblue FMX36 (Spectro/AMETEK, Germany) was used for the analysis of metal ions using a plasma matrix. The metals included, Iron (Fe), Lead (Pb), Sodium (Na) and Aluminium (Al).

4.3.2 River discharge

The river discharge was measured once every month for a period of a year, July 2018 to June 2019 following the velocity area method. The width of the river channel is divided into at least 20 vertical sections, with each section having no more than 10% of the total flow (Gravelle, 2015). The velocity is measured at one or more points in each vertical by a current meter and an average velocity determined in each vertical. The discharge is derived from the sum of mean stream velocity, and the channel cross-sectional area (Herschy, 1998). Discharge was measured for two sites/river reaches in both studied rivers using an electronic flow meter, the FlowTracker2 (FT2) handheld Acoustic Doppler Velocimeter (ADV®).

At each site, along each assessed 20-meter reach, one cross section was established. The cross section was divided into equidistant points whereby 20 readings of flow velocity were recorded at each equidistant point from one river bank to the other covering the morphological units present along the cross section including; pools, riffles and runs (Couperthwaite, 1997). The criteria used to choose a suitable spot was in accordance with (Rantz, 1982; Sauer and Turnipseed, 2010) and included the following conditions:

- A straight stretch of water with the horizontal velocity vectors running parallel to the stream bank.
- A stable, even streambed without large rocks, weeds and protruding obstructions that create turbulence and interfere with sensor performance.
- A level streambed configuration to reduce variation in the vertical components of velocity.
- A water depth of at least2-3 cm across most of the transect. The sensor will not work correctly unless the water depth is at least 2-3 cm. The probe geometries of the Flow tracker2 instrument are particularly well-suited to measure shallow depths (2-3 cm) of water.

All four of these conditions are seldom satisfied. Nevertheless, the best possible reach using these criteria were selected (Rantz, 1982; Turnipseed and Sauer, 2010).

A tape measure was stretched from one river bank to the other where the flow velocity measurements at equidistant points were measured. A total of 20 flow readings were taken using the handheld electronic flow tracker. The overall river discharge of the site was automatically computed by the Flow tracker 2 ADV instrument using the mean and mid-section method. The cross sections were only used for river discharge readings.

Historical river flow data (1920 – 2005) was also estimated/modelled by the National Water Resources program for the upper Liesbeek River downstream of Newlands Spring (33°58'33.01"S; 18°27'38.67"E). Modelled flow data was used due to the absence of a gauging station to measure actual flows. Moreover, monthly actual river discharge was measured for the Liesbeek and Disa Rivers. The estimated historical river discharge produced as part of national water resources assessment for the Liesbeek River dating from 1920-2005 was used to confirm or validate the monthly river discharge measurements of the Liesbeek River. National Water Resources is a Department of Water and Sanitation (DWS) national programme with its own database where data can be extracted for flow-modelling purposes. It is a broad national assessment of the water resources of South Africa at a quaternary catchment scale. The main products of these studies are modelled monthly estimates of actual and naturalized streamflow per catchment from 1920 onwards.

4.3.3 Aquatic macroinvertebrates

Aquatic macroinvertebrates were sampled once seasonally for a period of a year (August 2018- July 2019) following the South African Scoring System 5 (SASS5) method, which is the standard method for rapid bioassessment of South African rivers (Dickens and Graham, 2002). The method forms the backbone of the River Eco-status Monitoring Programme (REMP) and is also included in the determination of the Ecological Reserve as required by the National Water Act (No. 36 of 1998) (Graham and Dickens, 2010). In addition, the method may be used to assess the ecological state of aquatic ecosystems, assessing the impacts of development and the spatial trends in ecological state (Dallas, 2007).

The SASS5 method is quick, cost effective, easy to use and produces valuable results. Limitations of the SASS5 method are that it is not applicable in wetlands, impoundments, estuaries and other lentic habitats and habitat variability and high flows may lead to the incorrect interpretation of results (Dickens and Graham, 2002).

Sampling of aquatic macroinvertebrates was conducted from the following biotopes:, stones biotope, which consists of the stones in current (SIC) which are free or loose stones such as pebbles and cobbles, stones out of current (SOOC) such as bedrock or any solid object out of current, marginal/aquatic vegetation (MV) biotope, gravel biotope and the sand/mud (GSM) biotope. Each biotope was sampled using a kick net, which was held downstream to collect dislodged macroinvertebrates from the various biotopes. Each biotope type was sampled according to the methods described by Dickens and Graham (2002) and Dallas (2007).

Each sample was washed down to the bottom of the net then carefully tipped into three separate trays, one for each of the three different biotopes. The sampled organisms were viewed and identified to family level over a period of 15 minutes per biotope. At each site, sampled aquatic macroinvertebrates were identified, counted and recorded on the SASS5 data sheet.

Three indices calculated, namely, SASS5 Score, Number of Taxa and Average Score per Taxon (ASPT). All data at each site was quantified in terms of the total SASS score, the Average Score per Taxon (ASPT) and total abundance per family per biotope indicative of the diversity and abundance of aquatic macroinvertebrates.

A quality score based on its susceptibility to pollution was allocated for each taxon per sample. The score attributed to benthic macroinvertebrates varies between 1 and 15. Higher scores are attributed to organisms with greater sensitivity to pollution and the lower scores correspond to tolerant organisms (Dallas, 2000). The estimated abundance range is indicated in categories from A - D, where category A shows a range of 2 - 10, B shows a range of 10 -

100, C 100 - 1000 and D a range of greater than 1000 macroinvertebrates recorded per site (Dallas, 1995). When there is only one macroinvertebrate found in a sample, it is recorded as 1 on a data sheet. The sensitivity scores of identified taxa were then added to calculate the SASS5 score.

The total number of taxa found in a sample corresponds to the sum of number of taxa per sample, while the Average Score per Taxon (ASPT) was calculated by dividing the SASS5 scores by number of taxa for each sample at each site (Dickens and Graham, 2002).

At each sampling site, on each sampling occasion, an assessment of the diversity and quality of the habitat available for aquatic macroinvertebrates was conducted using the Integrated Habitat Assessment System (IHAS) (Ollis *et al.*, 2006). This was conducted due to the sensitivity of the South African Scoring System (SASS) to biotope availability (Dickens and Graham, 2002). The IHAS method aims to summarize numerically and reflect the quantity, quality and diversity of biotopes available for habitation by macroinvertebrates at a sampling site (McMillan,1998; Dallas, 2000). The scoring system is based on a total of 100 points, split into two sections: Sampling Habitat (55 points) and Stream Condition/Characteristics (45 points). The Sampling Habitat section is further divided into three sub-sections: Stones-in-Current (20 points), Vegetation (15 points), and Other Habitat (20 points), including stones-out-of-current, gravel, sand and mud. The Stream Condition section provides an evaluation of a site in terms of its physical characteristics and the degree of disturbance present, including estimates of aspects such as stream width, depth and velocity. The maximum Total IHAS score is 100 (representing a percentage). Total IHAS scores of greater than 75 is representative of excellent macroinvertebrate habitat conditions, whilst Total scores of between 65 and 75 indicate adequate habitat conditions (McMillan, 1998).

The frequency of occurrence of aquatic macroinvertebrate taxa throughout the sampled period was determined from the SASS5 data in order to show common or rare taxa found at the studied rivers. This was done in accordance to a method recommended by (Eady *et al*., 2013). Using this method, the taxa were categorised as common or rare. Macroinvertebrate taxa were classified according to the number of times taxa were present at sites per seasons. If present over many seasons at the same site, taxa were categorized as common, whereas if taxa were rare either between seasons, taxa were categorized as rare. This would then provide an indication of typical macroinvertebrate taxa occurring in the Liesbeek River with flow discharges from the Cannon spring.

4.4 Data analysis

The t-test was used to test for any significant differences between the rivers with regards to water quality determinants, river discharge and aquatic macroinvertebrates. The statistical tests were conducted using monthly data. The tests were conducted at the 5% significance level. The Lavene's test was used to test for the homogeneity of variances. The Levene's test uses an *F*-test to test the null hypothesis that the variance is equal across groups. The chi-squared goodness of fit test on excel was used to check if the data was normally distributed. Pearson correlation was used to determine the correlation of aquatic macroinvertebrates between the studied rivers. Pearson correlation measures the existence (given by a p-value) and strength (given by the coefficient r between -1 and +1) of a linear relationship between two variables. If the outcome is significant it can be concluded that a correlation exists. According to Cohen (1988) an absolute value of r of 0.1 is classified as small, an absolute value of 0.3 is classified as medium and of 0.5 is classified as large.

5. WATER QUALITY AND RIVER DISCHARGE

5.1 Introduction

The chapter aimed to evaluate the hypothesis that there is no statistical significant difference between the river discharge and water quality determinants of a spring fed and non-spring fed river. Moreover, this chapter also addresses an aspect of objective 1 of the study, which is:

1. To determine and compare the river discharge and water quality of the non-spring fed Disa River, and spring fed Liesbeek River tributary.

5.2 Results and Discussion

Catchment hydrology, which is strongly influenced by climate, geology, and soil type also influences water quality (Lintern *et al*., 2018). The section presents the results of river discharge and water quality of the Disa and Liesbeek Rivers. The causes of similarities and differences in water quality determinands and river discharge between the Disa and Liesbeek rivers are also discussed.

4.5.1 Hydrology/River discharge

River discharge measurements were performed after the 2016-2018 drought. The discharge of both rivers followed a similar trend throughout the study period (Figure 9).

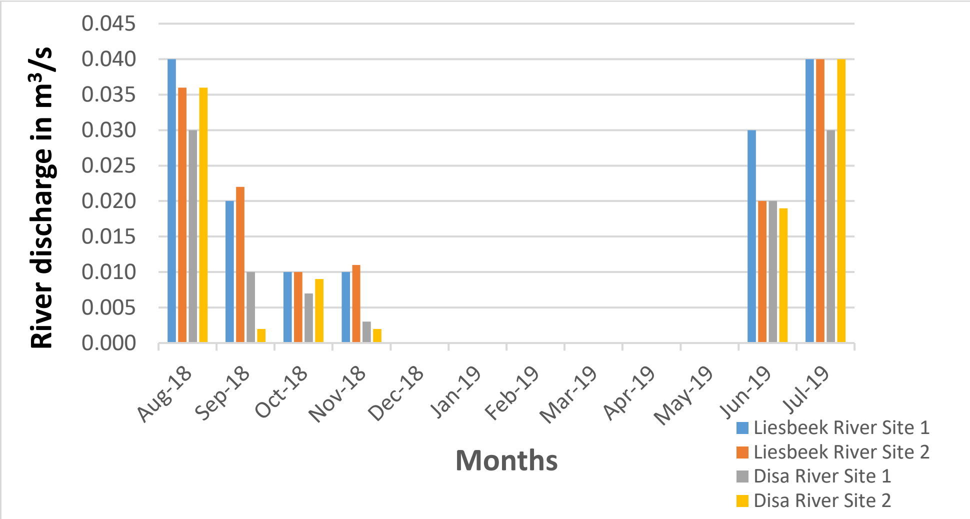

Figure 9: Comparison of the monthly river discharge measurements of the Liesbeek and Disa Rivers for period July 2018 to June 2019.

The highest discharges in both rivers were measured during the winter season with a value of 0.04 m^3/s for the Liesbeek River site 1 and site 2 and Disa River site 2 in July 2019 and Aug 2018 at Liesbeek River site 2. The higher river discharge during July and August is attributed to rainfall during the winter season, which is typical for the study area as seen in (Figure 2). During the winter and spring seasons (September, October, November), both rivers had as expected higher flows than in summer and autumn. This seasonal pattern is expected due to winters being wet while summers are dry. The rivers were dry from December to May. It should be noted that during this period, isolated patches of standing water/pools with no continuous flow characterized the stream channel. There was no statistically significant difference in the annual river discharge between the Liesbeek and Disa Rivers, p value 0.39.

The similar discharge between the Disa and Liesbeek Rivers also provided an indication that the volume of flow received from the spring feeding the Liesbeek River did not result in higher discharge at the Liesbeek River that was significantly different when compared to the river discharge of the Disa River. Moreover, even though there were no notable springs identified feeding the Disa River, it should be noted that groundwater may be discharging as diffuse flows and not through a spring into the upper Disa

River. As most studies worldwide have shown that groundwater contributes substantially to streamflow in many high mountain catchments. However, the contribution of groundwater to streamflow is highly dependent on the geology, climate, topography and spatial scale (Somers and McKenzie, 2020). The upper catchment of the Disa River is underlain by the Table Mountain sandstone which overlies a granite base with a narrow band of shale at approximately 200m (Grindley, 1984). According to Le Maitre and Colvin (2008) granite rock types have limited volumes of groundwater and discrete discharge to rivers. Also, the size of the Liesbeek River catchment is a total of 26km^2 according to Jeffes *et al*. (2017) and the size of the catchment upstream of the spring is 10.1km^2. It may be possible that while the spring is discharging, due to the downstream area that the spring is discharging into being large, the spring discharges downstream does not make a significant contribution to total flows.

Spring fed rivers are characterized by reduced variability of river discharge (Crossman *et al.,* 2012). This may be due to the fact that most spring fed rivers flow throughout all seasons as groundwater inflow provides flow during the dry season. According to Le Maitre *et al*. (2008) groundwater discharge is dominates dry season flows in perennial river systems and to sustain aquatic biodiversity. Lusardi *et al.* (2016) further indicated that spring fed rivers are primarily regulated by groundwater discharge and may show little to no response to local precipitation events. However, the findings of the current study contradicted these findings. The river discharge of the spring fed river Liesbeek River varied during the seasons as the river had low to no flows during the summer and autumn and had higher flows during winter and spring. The historical river discharge data obtained from DWS also coincided with the river discharge results of the current study following a similar trend (Figure 10).

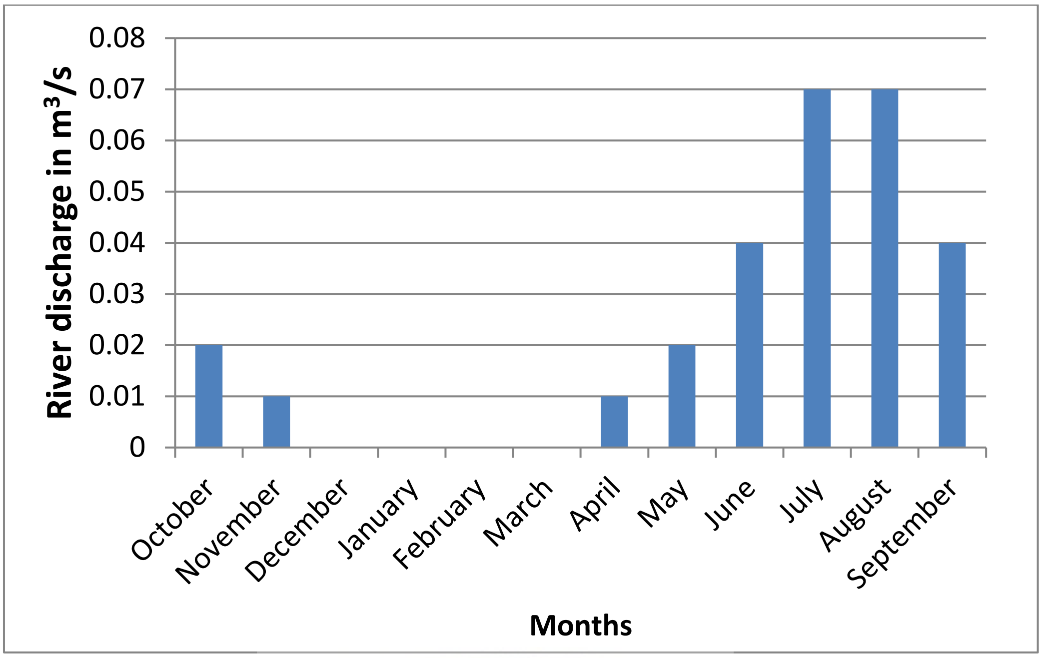

Figure 10: Average monthly historical river discharge of the Liesbeek River for period (1920-2005) based on National Water Resources by Department of Water and Sanitation.

The summer months (December – March) were characterised by a river discharge of 0 m^3/s (Figure 10) and autumn (April and May) was characterized by flow with an average discharge of 1.5 m^3/s, whereas the current study's autumn season had no flow with a river discharge of 0 m^3/s. This may be attributed to the natural variability of precipitation over the years and climate change. A study conducted by Du Plessis and Schloms (2017), which made use of historical rainfall data for the Western Cape Mediterranean climatic region, revealed a shift in the onset of the rainy season and longer dry seasons. The winter season was characterized by high flow, which slightly decreased during the spring season as the current study's river discharge pattern illustrates.

The variation of river discharge with seasons in a spring fed river was also substantiated by Hanson and Benedict (1984) who indicated that shallow wells near rivers often fluctuate with the seasons and even more than the river itself. During the wet season, the aquifer may fill up and flow into the river thus producing a water level in the well, which is higher than in the water surface. During the dry season, groundwater may also flow from the aquifer into the river, dominating the dry season flows. This pattern is not necessarily followed in all rivers or even in parts of the rivers, as some have constant inflow from aquifers and some constant outflow to aquifers, whereas some rivers lose too much water to aquifers causing them to run dry during the dry season.

4.5.2 Water quality

The water quality results are presented in (Table 5). The physical and chemical water quality compliance of the Liesbeek and Disa River was determined using the South African Water Quality Guidelines for Aquatic Ecosystems (DWAF, 1996). The t-test results for water quality parameters, which showed a significant difference ($p<0.05$) in the concentration of physical and chemical concentration between the Liesbeek and Disa Rivers are presented in (Table 6). It should be noted that even though these water quality parameters varied significantly between the studied rivers, they all complied with the set guidelines for aquatic ecosystems (DWAF, 1996), with the exception of aluminum. The monthly concentration of the water quality parameters that significantly varied and were statistically significant are presented in (Table 7 and Table 8) to show the detailed variation of the water quality parameters. Moreover, the monthly concentration of the water quality parameters of the studied river sites are all presented in (Appendix 1) for Disa River site 1 and site 2 and (Appendix 2) for Liesbeek River site 1 and site 2.

Table 5: Annual means and standard deviation of the water quality parameters of the Liesbeek and Disa Rivers sampled over a period of a year, n = 12.

Determinants/parameters	Aquatic ecosystems (1996e &1996a)	Liesbeek River Site 1	Liesbeek River Site 2	Disa River Site 1	Disa River Site 2
Electrical Conductivity (MSM)	NA	9.52 ± 0.35	9.25 ± 0.34	13.38 ± 1.74	13.11 ± 0.73
Total dissolved solids (mg/L)	200-1100	57.35 ± 6.16	60.30 ± 4.73	86.90 ± 8.33	86.98 ± 9.18
Turbidity (NTU)	<10% background value	0.95 ± 0.36	0.83 ± 0.41	1.29 ± 1.39	1.26 ± 1.08
pH	5-9	6.35 ± 0.40	6.47 ± 0.22	5.51 ± 0.18	5.95 ± 0.34
Temperature (°C)	5-30	15.71 ± 1.89	16.37 ± 1.62	14.27± 2.59	14.82± 2.09

Dissolved Oxygen DO (mg/L)	80 % - 120 % of saturation	11.47 ± 2.40	10.17 ± 1.52	9.02 ± 0.42	8.41 ± 0.47
Nitrate as N (mg/L)	<0.5 oligotrophic 0.5-2.5 mesotrophic	0.13 ± 0.04	0.10 ± 0.02	0.13 ± 0.07	0.16 ± 0.04
Sulfate as SO42- (mg/L)	NSL	3.05 ± 1.38	3.56 ± 0.42	2.75 ± 1.33	2.64± 1.14
Ammonia as N (mg/L)	0-0.007	0.02 ± 0.01	0.03 ± 0.01	0.02 ± 0	0.02 ± 0
Chloride as Cl- (mg/L)	NSL	19.08 ± 2.42	21.75 ± 1.35	33.75 ± 5.06	32.16 ± 4.72
Sodium as Na (mg/L)	NSL	11.48 ± 1.17	11.37 ± 1.07	17.69 ± 2.00	18.21 ± 2.05
Iron as Fe (mg/L)	NSL	0.07 ± 0.04	0.06 ± 0.02	0.23 ± 0.05	0.23 ± 0.02
Lead as Pb (mg/L)	0.0002	0 ± 0	0.00 ± 0.00	0 ± 0	0 ± 0
Aluminum as Al (mg/L)	0-0.005	0.09 ± 0.08	0.14 ± 0.15	0.24 ± 0.05	0.26 ± 0.08

No standard limits = **NSL.** All data are presented as means ± standard deviation (SD) & those that did not comply with aquatic ecosystems guidelines (1996a & 1996a) are highlighted.

Table 6: Summary of t-test results for water quality parameters between the Liesbeek and Disa Rivers that indicated a significant difference over the study period.

Water quality parameter	P-value	F-ratio variances
Aluminum	0.000	3.113
Chloride	0.000	4.247
Iron	0.001	1.813
Sodium	0.000	3.308
Total dissolved solids	0.000	2.291

The results showed that (93.75%) of the assessed water quality parameters complied with the South African Water Quality Guidelines for Aquatic Ecosystems (DWAF, 1996) with the exception of aluminum (6.25%), in both Disa and Liesbeek Rivers. The non-compliance of the aluminum determinant in both rivers could be attributed to leaching from minerals containing this element (Guibaud and Gauthier, 2003).

The aluminum concentration of the Disa River ranged from 0.15 mg/L measured in the month of June 2019 at site 2 to 0.41 mg/L measured in the month of August 2018 at site 2 (Table 7). The aluminum concentration ranged from 0.04 mg/L, measured in February and March 2019 at the Liesbeek River site 1 and site 2 in May 2019 to 0.6 mg/L measured in August 2018 at Liesbeek River site 2 as depicted in (Table 8).

Table 7: Disa River monthly concentration of the water quality determinants that were significantly different compared to Liesbeek River water quality paramers.

Month	Aluminium (mg/L)		Chloride (mg/L)		Iron ugl (mg/L)		Sodium (mg/L)		Total Dissolved Solids (mg/L)	
	Site 1	Site 2	Site 1	Site 2	Site 1	Site 2	Site 1	Site 2	Site 1	Site 2
18-Aug	0.35	0.41	21	19	0.27	0.26	13.8	14.1	68.3	62.6
18-Sep	0.23	0.28	32	34	0.26	0.25	17.3	16.9	84.4	79.8
18-Oct	0.25	0.24	35	35	0.27	0.27	17	17.2	85.1	88.3
18-Nov	0.28	0.31	38	36	0.26	0.22	18.1	19	84	89
18-Dec	0.25	0.26	36	32	0.23	0.25	17	18.7	91	86
19-Jan	0.18	0.18	39	34	0.24	0.25	17.6	17.3	89	92
19-Feb	0.2	0.19	38	38	0.2	0.24	18	17.6	93.8	91.6
19-Mar	0.21	0.24	33	33	0.1	0.2	18.3	18.3	84.4	84.4
19-Apr	0.29	0.31	36	31	0.22	0.22	19	21.2	81	87
19-May	0.23	0.21	31	30	0.18	0.19	17	19.1	95	98
19-Jun	0.16	0.15	37	33	0.29	0.25	22.6	21.9	101.8	97
19-Jul	0.32	0.39	29	31	0.24	0.21	16.6	17.1	85.1	88.1

Table 8: Liesbeek River monthly concentration of the water quality determinants that were significantly different compared to Disa River water quality parameters.

Month	Aluminium (mg/L)		Chloride (mg/L)		Iron ugl (mg/L)		Sodium (mg/L)		Total Dissolved Solids (mg/L)	
	Site 1	Site 2	Site 1	Site 2	Site 1	Site 2	Site 1	Site 2	Site 1	Site 2
18-Aug	0.3	0.6	17	22	0.2	0.12	8.3	10	43.6	60.3
18-Sep	0.09	0.15	20	20	0.05	0.08	11.6	12.1	51.6	53.6
18-Oct	0.13	0.16	19	22	0.07	0.04	10.7	11.2	52.3	60.3
18-Nov	0.05	0.09	21	20	0.05	0.06	11.3	9.9	57.6	53.6
18-Dec	0.09	0.12	22	21	0.09	0.07	12.5	10.5	60.3	60.3
19-Jan	0.05	0.08	20	22	0.07	0.05	12.2	10.7	55.6	60.3
19-Feb	0.04	0.05	20	22	0.06	0.06	12.2	11.2	59	60.3
19-Mar	0.04	0.07	20	21	0.05	0.09	12.4	13.1	60.3	60.3
19-Apr	0.05	0.06	21	22	0.05	0.07	11.9	12.8	61	60.3
19-May	0.07	0.04	13	25	0.09	0.08	11.5	11.5	57.6	73.7
19-Jun	0.05	0.07	19	23	0.05	0.06	10.8	10.9	61	60.3
19-Jul	0.21	0.19	17	21	0.09	0.05	12.4	12.6	68.3	60.3

The higher concentration of aluminum in both rivers during the winter season could be attributed to the mobilization of aluminum associated with erosion from the land with specific storm events (Ingerman *et al*., 2008). According to Wu (2009) soils derived from Table Mountain Sandstone are usually coarse sand and sandy clay. In soils, aluminum normally binds to soil constituents, clay in particular. Moreover, soils produced by weathering are acidic, sandy and poor, chiefly due to the lack of feldspar. Acidic soils normally contain high aluminum concentrations (Jaiswal *et al*., 2020). The highest precipitation was recorded during June and August at the Table Mountain weather station and August and July at the Kirstenbosch weather station as depicted on (Figure 2). The mobilization of aluminum from terrestrial to aquatic environments results from environmental acidification, which results in the increased aluminum concentrations (Krewski *et al*., 2007).

The aluminum concentration was significantly greater in the Disa River than in the Liesbeek River (Table 5, Table 7 and Table 8). The higher aluminium concentration in the Disa River could be attributed the fact that the Constantia Nek Water Treatment Work discharges sludge in an open field close to this river, which due to runoff, can drain into the Disa River. Moreover, in water, the concentration of total aluminium increases with a decrease in pH (close or lower than 5) and an increase in organic matter (Senze *et al.*, 2015). At the Disa River, the average pH concentration was lower than 6, which may have been a contributing factor leading to a higher aluminium concentrations at the Disa River.

Both rivers flow through undisturbed forests with no significant human activites affecting water quality (Ollis, 2005 and Crisp, 2016). This was evident in the low concentrations of nitrates, 0.1 – 0.2 mg/L at the Liesbeek River all the sites on both rivers. The ammonia concentration ranged between 0.01 – 0.04 mg/L at Liesbeek River site 1 and 0.01 – 0.07 mg/L at Liesbeek River site 2 (Appendix 2). At the Disa River, the ammonia concentration ranged between 0.01 – 0.04 mg/L site 1 and 0.01-0.03 mg/L at site 2 (Appendix 1). This was a clear indication that the water was not contaminated by anthropogenic sources that would proliferate nutrient enrichment. Furthermore, the concentration of dissolved ions such as chlorides , sulphates and sodium ranged between 13-22 mg/L, 0.3-4 mg/L and 8.3-12.5 mg/L respectively for the Liesbeek River site 1 and 20-25 mg/L. 2.5-4 mg/L and 9.9-13.1 mg/L for Liesbeek River site 2 (Table 8 and Appendix 2). The concentration of dissolved ions for the Disa River such as chlorides, sulphates and sodium were determined as 21-39 mg/L, 0.1-5 mg/L and 13.8-22.6 mg/L for the site 1 and 19-38 mg/L, 0.8-4.2 mg/L and 14.1-19.1 mg/L at site 2 (Table 7 and Appendix 1).

The chloride concentration of the Liesbeek River ranged from 13 mg/L measured in May 2019 at Liesbeek River site 1 to 25 mg/L in May 2019 at Liesbeek River site 2 (Table 8). The chloride concentrations in the Disa River ranged from 21 mg/L in t August 2018 at site 1 to 39 mg/L in January 2019 at site 2. During the wet season, runoff dilutes chlorides in rivers while the reduction of flows and evaporation cause elevated concentrations levels (Figure 9). (Jadhav and Jadhav, 2019).

The chloride level was higher at the Disa River than the Liesbeek River (Table 5, Table 7 and Table 8). All natural water bodies contain chloride in varying degrees with its content increasing as mineral content increases (Brandt *et al*. 2017). Sources of chloride in rivers may be natural or anthropogenic. Natural sources are principally atmospheric deposition and from precipitation and aerosols, leaching of rocks and leaching of evaporine sediments (Albek, 1999). Chloride in precipitation and dry deposition originates from marine aerosols or volcanic gases. Naturally occurring chloride concentrations in rainwater and snowmelt can be several mg/L near the coastal regions due to the contribution of seawater aerosols (Kelly *et al*., 2012). The Disa River may receive seawater aerosols from the Hout Bay beach or ocean, which may have been one of the contributing factors to the higher chloride in Disa River. Verma (2012) also reported that high chloride concentration in water might be due to a high rate of evaporation or due to organic waste of animal origin. Pal and Chakraborty (2017) also indicated that higher chloride concentrations in water might be due to animal faeces and sewage inflow. The Disa River at the Orangekloof Reserve is a recreational site and popular for dog walking. Dog waste/faeces washed into the river system may have also contributed to a higher chloride concentration at the Disa River (Table 5 and Table 7).

The iron concentration of Liesbeek River water ranged from 0.05 mg/L measured in September 2018 at site 1 to 0.12 mg/L also measured in September 2018 at site 2 (Table 8). The iron concentration of the Disa River ranged from 0.1 mg/L measured in the month of March 2019 at site 1 to 0.29 mg/L measured in the month of June 2019 at site 1. The highest iron concentration were measured during winter (June 2019 & August 2018) both rivers, which was characterized by the highest river discharge in both rivers (Figure 9). As depicted in (Figure 2), the average measured precipitation was higher during the winter season (June, July, and August). According to Ekstrom *et al*. (2016), higher precipitation should lead to increasing iron export from soils to ground and surface waters. Both rivers drain from Table Mountain sandstone, which is characterized by coarse sand and sandy clay (Wu, 2009). According to Stucki (2006) & Carroll (1958), iron is ubiquitous in clay soils. Consequently, as a result of higher precipitation during the winter month, runoff from soils with iron content may have resulted in the increased iron concentration in both rivers during the winter season.

The iron concentrations were significantly higher at the Disa River than the Liesbeek River (Table 5, Table 7 and Table 8). The source of iron in water can either be geogenic or via industrial effluents and domestic waste (Kumar *et al*., 2017). The higher iron concentration at the Disa River was mainly due to the natural conditions as iron primarily comes from the products of weathered rocks and soil around the river catchment (Xing and Liu, 2011). Forest soils often yield more iron than minerogenic soils (Ekstrom *et al*., 2016) and the studied rivers are in forested areas. The pH of the Disa River was low and the aluminium concentration of the Disa River was higher compared to the Liesbeek River, which may have contributed to the higher iron concentration at the Disa River, as iron has an inverse correlation with pH and a close positive correlation with aluminium (Zhu, 2006).

The sodium concentration of the Liesbeek River ranged from 8.3 mg/L measured in August 2018 at site 1 to 13.1 mg/L measured in July 2019 at site

2 (Table 8). The sodium concentration of the Disa River ranged from 13.8 mg/L measured in the month of August 2018 at site 1 to 22.6 mg/L measured in the month of June 2019 at site 1 as well. The sodium concentration was significantly higher in the Disa River than the Liesbeek River. This couldpossibly be due to a high rate of evaporation or due to organic waste of animals (Brandt *et al*., 2001) (Table 5, Table 7 and Table 8). Recreational activities including dog walking may have contributed to faecal contamination or organic waste of animal origin at the Disa River (Sasakova *et al*., 2018). Additional sources of sodium concentration at the Disa River may include natural sources such as rock-water interactions and major atmospheric contributions (Panno *et al*., 2002). Seawater aerosols from the Hout Bay coastal area may also have contributed to a higher sodium concentration at the Disa River in comparison with the Liesbeek River, as seawater spray is one of the factors known to increase sodium concentration (Priadarshi, 2005).

The TDS concentrations of Liesbeek River water ranged from 43.6 mg/L in August 2018 at site 1 to 73.7 mg/L measured in May 2019 at site 2 (Table 8). The TDS concentration of the Disa River ranged from 62.6 mg/L measured in the month of August 2018 at site 2 to 101.8mg/L measured in the month of June 2019 at site 1. As depicted in (Figure 2), the average measured precipitation was higher during the winter season. Higher level of TDS during the rainy seasons are more likely due to the surface runoff, which may contain increased sediment load (Ioryue *et al*., 2018). The TDS levels was also significantly higher at the Disa River than the Liesbeek River (Table 5, Table 7 and Table 8). The measured values of EC were proportional to the TDS concentration as the EC was also higher (Appendix 1 & Appendix 2) at the Disa River. The higher TDS at the Disa River may be attributed to organic sources such as leaves, silt and plankton, as the site was dominated by instream leaf litter. The TDS is generally comprised of inorganic salts and small amounts of organic matter present in solution in water (Islam *et al*., 2017). Additional sources of the TDS at the Disa River also include the organic material from the soil, contributing to an increased TDS concentration (Butler and Ford, 2018).

Water quality parameters that did not vary significantly

Differences in concentrations of electrical conductivity, turbidity, pH, temperature, dissolved oxygen, nitrate, ammonia, lead, *E.coli* and total coliforms were not statistically significant between the Disa and Liesbeek Rivers (Table 9).

Table 9: Summary of T test results for water quality parameters between the Liesbeek and Disa Rivers that did not indicate a significant difference over the study period.

Water quality parameter	P-value	Degrees of freedom (Df)	F-ratio variances
Electrical conductivity (mS/m)	0.071	7	1
Turbidity	0.082	46	9.9
pH	0.224	7	0.7
Temperature ºC	0.858	14	1.5
Dissolved oxygen	0.385	7	13.7
Nitrate	0.210	46	2.5
Ammonia	0.323	23	2.5
Lead	0.462	46	1.6

The similarity in electrical conductivity in the Disa and Liesbeek River water could be attributed to the fact that both rivers were draining the same geological rock formations (Table Mountain sandstone) and in both rivers there were no identifiable sources of pollution that would affect the EC concentration. The rocks in the catchment is the source of the ions that act as conductors of electricity (Olson, 2012). Geological heterogeneity is one of the factors affecting the variation of electrical conductivity in rivers in the same catchment or ecoregion (Griffith, 2014). Rock weathering, other natural sources and anthropogenic drivers account for majority of the dissolved ions in river water (Hamid *et al*., 2020). The similarity of turbidity in the studied rivers could be attributed to the similar vegetated land cover geology and soils. Disturbance of riparian vegetation results in increased sedimentation of rivers, which may proliferate turbidity in rivers (Khatri and Tyagi, 2014). Sources of turbidity in rivers include eroded material, clay, silt, organic matter and plankton (Henley *et al*., 2000). Similarity of temperature between the studied rivers could be attributed to the fact that both studied rivers experience the same Mediterranean climate, which controls the temperature of the catchment. Forested land cover with both rivers having tree canopy cover at the studied rivers may have also contributed to both rivers having similar temperatures. According to Dallas *et al*. (2008), at the catchment scale, differences in temperature are driven by variation in climate, geography, topography and vegetation (Dallas, 2008). The similarity of ammonia and nitrates in both studied rivers could be attributed to both rivers being in a near natural condition with no point sources of pollution that would influence the concentration of these nutrients.

The similarity in the DO in both studied rivers may be attributed to both rivers being in mountain areas, similar temperature, similar in stream channel characteristics such the dominance of bedrock and boulders that would promote aeration with contributes to DO, river flow and altitude. According to He *et al*. (2011), the amount of oxygen that can be dissolved in water depends on several factors, including water temperature, the amount of dissolved salts present in the water (salinity), and atmospheric pressure. Aquatic plants and algae also contribute dissolved oxygen to water bodies during daylight hours through photosynthesis.

The similarity of pH between the studied rivers may be attributed to the mineral composition of the geology and soil as both rivers drain a catchment with similar geology and soils (Lintern *et al*., 2018). Bedrock mineralogy influences the pH of water by chemical (dissolution of minerals by the action of water and its solutes) and physical weathering (Kamenik *et al*., 2001). The presence of aquatic vegetation also influences pH of rivers as photosynthesis by aquatic plants during the daylight removes carbon dioxide (CO_2) from the medium hence pH would increase. At night, respiratory processes of aquatic organism's release CO_2 into the medium and pH declines (Araoye, 2009). Both rivers were characterized by aquatic vegetation, which may have been another reason for the similar pH in both rivers.

The similarity of lead concentration in both studied rivers could be attributed to the same geology or rock type of the catchment, land use/land cover and the fact that both sites had no identifiable sources of pollution. The major natural sources for mobilisations of lead from the earth's crust is weathering of rocks (Obasi and Okudinobi ,2020). An increased amount of lead in our environment comes from human activities including burning fossil fuels, mining, and manufacturing, industries and treated wastewater effluent (Tiwari *et al*., 2013), none of which occurs in the two study areas.

5.3 Summary

There was no difference in the discharge of the Disa and Liesbeek Rivers over the study period. This was attributable to the fact that both studied river reaches were in the same catchment with same climate and thus receiving similar rainfall, which contributed to river flows. The similarity in discharge in both rivers may also be an indication that even though the Liesbeek River is fed by a notable spring, groundwater may be contributing to the Disa River through diffuse flow along the banks and the bed and by seepage. Moreover, the inflow received from the spring feeding the Liesbeek River was not significantly different from the river discharge of the Disa River.

The water quality results showed that the water quality of the Liesbeek and Disa River upper reaches mostly complied with the DWS Aquatic ecosystem guidelines with only a few water quality parameters that did not comply with the guidelines attributable to recreational activities. The compliance of the water quality of these rivers is attributable to land use/land cover as the river reaches are in a natural forest with minimal developments or identifiable point sources of pollution. Moreover, <50% of the assessed water quality parameters significantly varied with most having similar trends but the Disa River water quality values were generally higher than those in the Liesbeek River.

6. AQUATIC MACROINVERTEBRATES

Hydrodynamics is known to play an important role in shaping macroinvertebrate communities in rivers and streams (Reaver *et al*., 2019). Higher velocities can cause some macroinvertebrates to be dislodged from the substrate or be prevented from colonizing. Water quality also plays a pivotal role in aquatic macroinvertebrates abundance, diversity and assemblage composition where certain water quality parameters including temperature, electrical conductivity, pH and dissolved oxygen greatly influence these organisms (Sekiranda *et al*., 2004).

This section presents the findings of the aquatic macroinvertebrates monitored throughout the sampling period and the habitat scores of the sampled sites. The aquatic macroinvertebrates were sampled seasonally over a 1-year period (Aug 2018- July 2019). Data was collected from the Disa and Liesbeek Rivers to determine the SASS5 score, number of taxa and ASPT to improve the understanding of the ecological conditions of the rivers. Moreover, functional feeding groups of sampled aquatic macroinvertebrates were also determined. The chapter aimed to evaluate the hypothesis that there is no statistically significant difference between the aquatic macroinvertebrates diversity and abundance of a spring fed and non-spring fed river. Moreover, this chapter also addresses an aspect of objective 1 and objective 2 of the study, which were:

1. To determine macroinvertebrate abundance, functional feeding group composition, and diversity and their relationship to hydrology/flow and water quality determinants.
2. To test the use of aquatic macroinvertebrates as indicators of spring fed river water quality.

6.1 Comparison of aquatic macroinvertebrates in the Disa and Liesbeek rivers

<u>**SASS5 score**</u>

The SASS5 scores of the Disa River varied over the seasons, with the lowest SASS5 score of 82 recorded during the summer season at site 2 and the highest SASS5 score of 132 recorded during the spring season at site 1(Table 10). The SASS5 scores of the Liesbeek River also varied with the lowest SASS5 score of 92 recorded during the winter season at both sites and the highest recorded during the spring season at site 2 (Table 11).

Table 10: Seasonal aquatic macroinvertebrates abundance, diversity and habitat scores recorded from the Disa (D) River.

Metrics	Winter		Spring		Summer		Autumn		Standard deviation	Standard error
	Site 1	Site 2	Site 1	Site 2	Site 1	Site 2	Site 1	Site 2		
Total SASS score	92	97	132	102	98	82	104	94	14.5	5.1
No of taxa	15	15	17	14	16	13	16	16	1.2	0.4
ASPT	6	6.4	7.7	7.2	6.1	6.3	6.5	5.8	0.6	0.2
IHAS scores (%)	82	87	94	92	75	77	82	79	6.9	2.4

Table 11: Seasonal aquatic macroinvertebrates abundance, diversity and habitat scores recorded from the Liesbeek (L) River.

Metrics	Winter		Spring		Summer		Autumn		Standard deviation	Standard error
	Site 1	Site 2	Site 1	Site 2	Site 1	Site 2	Site 1	Site 2		
Total SASS score	92	92	107	128	93	95	102	100	12.1	4.2
No of taxa	12	12	14	16	15	14	16	14	1.5	0.5
ASPT	7.6	7.6	7.6	8	6.2	6.7	6.3	7.1	0.6	0.2
IHAS scores (%)	92	94	92	94	72	79	84	78	8.5	3

The SASS5 scores of the winter and summer season at the Disa River were also very similar, with the winter SASS5 score being 92 at both sites and the summer season being 98 at site 1 and 82 at site 2 (Table 10). Both seasons had lower SASS5 scores compared to the other seasons. The highest SASS5 scores of both rivers were recorded during the spring season. The trend or pattern observed of higher SASS5 scores during the spring season at both rivers and lower scores in winter at the Disa River coincides with the fact that aquatic macroinvertebrates thrive during the spring season and would therefore be comprised of a greater diversity of aquatic macroinvertebrates resulting in a higher SASS5 score. According to Stark and Phillips (2009) in New Zealand the highest diversity of aquatic macroinvertebrates is often recorded during the spring season, which has moderate temperatures and river flows. The average river discharge during the winter season was recorded as 0.02 m^3/s, which was higher than the average river discharge recorded during the spring season 0.006 m^3/s. The faster flows during the winter season, which often dislodges aquatic macroinvertebrates and the fact that many taxa are univoltine and emerge before winter resulted in fewer taxa compared to the spring season. Faster flows change the habitat of aquatic macroinvertebrates as the shear stress on the river increases, removing organisms or forcing them to seek refuge among substrates or in the drift (Death, 2008). Consequently, significant decreases in the abundance of benthic macroinvertebrates have been recorded after bed moving floods (Brewin *et al*., 2000; Bogan and Lytle, 2007; Mesa, 2010).

The main mechanisms behind the declines observed in the studied streams were likely to be catastrophic substrate mobilization. Increased shear stress from high flows removes the macroinvertebrates into the water column and produces a catastrophic drift of individuals (Mesa, 2012). Moreover, it should be noted that apart from environmental factors, temporal variability of aquatic macroinvertebrates occurs as a result of life history features such as emergence, feeding and growth (Dallas, 1995).

The lower SASS5 score during the summer season at the Liesbeek and Disa Rivers may be attributed to the fact that, during the summer season both rivers were characterized by a lack of discharge (Figure 9) and the river channel was comprised of isolated patches of pools. According to Eady *et al*. (2013) low flows are accompanied by shrinking habitats, which was the case for the Liesbeek and Disa Rivers during the summer season, also indicated by the lowest IHAS score 72% at Liesbeek River site 1, 79% at site 2 (Table 11) and 75% at the Disa River site 1 and 77% at site 2 compared to other seasons (Table 10). Increased water temperatures may have also contributed to lower

SASS5 scores of both rivers during the summer season (Appendix 1 & Appendix 2). Temperature is a key factor that influences the abundance and diversity of aquatic macroinvertebrates (Kemp *et al*., 2014). During the summer period, aquatic macroinvertebrates tolerant to water pollution with low sensitivity scores became dominant at the Liesbeek River. These aquatic macroinvertebrates included Turbellaria, Oligochaeta and chironomids, which are associated with slow-moving waters (Mesa, 2012). There was also an absence of aquatic macroinvertebrates that are highly sensitive to water pollution and thus have high sensitivity scores, during the summer season at both rivers. These included Helodidae, Pyralidae and Amphipoda at the Liesbeek River and Leptophlebiidae at the Disa River. According to a study conducted by Thirion (2016), Leptophlebiidae have a preference for fast flowing 0.3 - 0.6 m^3/s water over large gravel to small cobbles, although they also occur at other velocity categories and substratum types.

Number of taxa

The number of taxa at the Disa River ranged from 13 recorded at site 2 during the summer season to 17 recorded at site 1 during the spring season. At the Liesbeek River, the number of taxa ranged from 12 recorded during the winter season at both sites to 16 recorded during the spring and autumn at both site 1.

Hussain (2012) observed that severe high water levels are associated with reduced aquatic macroinvertebrates diversity in rivers and hence the slightly lower number of taxa during the winter season. This was also substantiated by an earlier study by Moffett (1935), which showed that floods completely wiped out, aquatic macroinvertebrates but that recovery started soon after. in In the current study even though, aquatic macroinvertebrates were not wiped out, a slightly lesser number of taxa during the winter season was recorded as compared to the spring season.

Average score per taxon

The ASPT was the lowest during the autumn season 5.8 at site 2 and the highest during spring 7.7 at site 1 (Table 10) at the Disa River. However, it should be noted that even though the ASPT was lower during spring it still indicated good water quality (Dallas, 2007). Chutter (1998) also points out that ASPT is a more reliable measure of the health of good/acceptable water quality rivers as opposed to poor quality rivers. At the Liesbeek River, the lowest ASPT was recorded during the summer 6.2 at site 1 the highest ASPT score of 8 recorded during the spring season (Table 11). This suggests that samples in the autumn and summer seasons were comprised of a larger number of taxa that were tolerant to environmental disturbances than those found during winter and spring seasons.

Overall sampled aquatic macroinvertebrate taxa

Aquatic macroinvertebrate taxa present at both rivers are highlighted in (Table 12).

Table 12 :Overall aquatic macroinvertebrate taxa sampled in the Liesbeek and Disa Rivers.

TAXA	DISA RIVER	LIESBEEK RIVER
	Species present (Yes/No)	
TURBELARIA	No	Yes
ANNELIDA		
Oligochaeta (Earthworms)	Yes	Yes
CRUSTACEA		
Amphipoda	No	Yes
Potamonautidae* (Crabs)	Yes	Yes
EPHEMEROPTERA		
Baetidae> 2 sp	Yes	Yes
Leptophlebiidae	Yes	Yes
Heptagenidae	Yes	No
Tricorythidae	Yes	Yes
Teloganodidae SWC	Yes	Yes
ODONATA		
Coenagrionidae (Sprites and blues)	Yes	No
Aeshnidae (Hawkers & Emperors)	Yes	Yes
Gomphidae	Yes	No
Libellulidae	Yes	No
LEPIDOPTERA		
Crambidae	No	Yes
HEMIPTERA		
Corixidae* (Water boatmen)	Yes	No
Gerridae	Yes	No
Naucoridae	Yes	No
Notonectidae* (Backswimmers)	Yes	No

Pleidae* (Pygmy backswimmers)	Yes	No
Veliidae/M...veliidae* (Ripple bugs)	Yes	Yes
TRICHOPTERA		
Barbarochthoriidae SWC	Yes	Yes
Pisuliidae	Yes	No
Philopotamidae	Yes	No
Leptoceridae	Yes	No
COLEOPTERA		
Scirtidae	Yes	Yes
Dytiscidae	Yes	No
Hydraenidae	No	Yes
Elmidae	Yes	Yes
DIPTERA		
Athericidae	Yes	No
Chironomidae (Midges)	Yes	Yes
Psychodidae	Yes	No
Ceratopogonidae (Biting midges)	No	Yes
Dixidae	No	Yes
Tipulidae	Yes	No
Simuliidae (Blackflies)	Yes	Yes

At the Disa River, 29 taxa were collected throughout the sampling period and 19 taxa at the Liesbeek River (Table 12). The Disa River comprised all SASS5 habitats required for sampling aquatic macroinvertebrates, including aquatic and marginal vegetation habitats, and hence supported a higher number of taxa than the Liesbeek River. According to Khudhair *et al*. (2019), aquatic vegetation provides shelter against vertebrate predation of vulnerable prey species such as macroinvertebrates and small fish. In addition, aquatic vegetation provides more surface area attachment for periphyton, a major component in the diet of many macroinvertebrate primary consumers (Khudhair *et al*., 2019).

Certain aquatic macroinvertebrate taxa were present throughout the seasons as depicted in (Appendix 3) at both rivers. These included Oligochaeta, Potamonautidae, Ashnidae, Simulidae and Chironomidae. Additional taxa such as Coenagrionidae, Veliidae, Philopotamidae and Athericidae were also present throughout the seasons at the Disa River. All these taxa were either tolerant or moderately tolerant to environmental disturbances. Barbarochthoniidae, Telagonodidae and Baetidae were also present at the Liesbeek River throughout the seasons. These taxa are highly sensitive to environmental disturbances and perturbations, with the exception of

Chironomidae and Simulidae, which also occurred at the Liesbeek River, together with other species that are tolerant to environmental perturbations. The prevalence of sensitive taxa in a spring fed river supported findings elsewhere (Fudere *et al*., 2001; Lusardi *et al*., 2016).

There were also taxa such as Baetidae and Simulidae that were present in abundance in three or more of the sampling seasons at both rivers and Amphipoda at the Liesbeek River. According to Ferreira (2015) some species of the family Baetidae such as *B. harrisoni* are common in South Africa and hence can be present in abundance in sites with conditions suitable for sensitive taxa. The Baetidae do not have a specific velocity preference (Thirion, 2016). Hence the Baetidae were present in abundance during summer and autumn when the river was characterized by low flow and no measurable discharge at both rivers and during the spring season where the river discharge was 0.006m^3/s in the Disa river and 0.013 m^3/s in the Liesbeek River.

In general, some species of the Simuliidae family are highly tolerant organisms found in a wide variety of habitats and ecological conditions in running water (Palmer and de Moor, 1998; Craig *et al*., 2012). Their high adaptability and tolerance limits enables these organisms to dominate in a variety of habitats (Palmer and de Moor, 1998). Simuliids attach to organic or inorganic substrata in flowing water (Picker, 2012). These organisms occur in fast flowing rivers or parts of the river. However, there are species of the family that have adapted to slower flowing water. The Simulidae family was present at both studied rivers in abundance in all seasons, attributed to its adaptability to both fast flowing and slow flowing water. The studied rivers were both characterized by a variety of substrates to which the Simulidae are typically attached.

Amphipods have been found to dominate most spring habitats worldwide (Gooch and Glazier, 1991; Webb *et al*., 1998). The variety of habitats at the site offered a wide variety of ecological conditions favourable to the proliferation of these taxa. Amphipods found in rivers in the south-western

Cape are often adapted to highly oxygenated, shallow and still waters (Thirion, 2016) where they often bury themselves under any type of substratum. The average annual DO of the Liesbeek River was 11.45 mg/L indicative of highly oxygenated water.

Invertebrate habitat assessment score

The IHAS scores ranged between 75% measured at the Disa River during the summer season and 95% measured during the spring season. The IHAS scores of both rivers followed a similar trend where the scores were lower during the summer and autumn seasons and higher during the winter and spring seasons. The lower IHAS scores observed during the summer and autumn seasons may be attributed to the fact that during these seasons, the river channel was characterized by isolated patches of pools and consequently the aquatic macroinvertebrates habitats were reduced. The marginal vegetation and stones-in-current were also limited or reduced during the summer and autumn seasons due to low flows.

Habitat structure is a key factor determining the occurrence and distribution of aquatic macroinvertebrates in rivers (Verdonschot and Verdonschot, 2012). Sediment type, vegetation type, and physical and chemical parameters making up the habitat influence the diversity and abundance of aquatic macroinvertebrates in rivers (Khudhair, 2019). The substrate provides places for food and refuge for aquatic macroinvertebrates. Therefore, habitat influences the diversity, abundance, and distribution pattern of aquatic invertebrates (Ali *et al*., 2007). Hence a greater diversity of habitats in a river results in a greater diversity and abundance of aquatic macroinvertebrates, except if the water quality of the river is poor (Barnes *et al*., 2013). However, the IHAS scores for both rivers indicate excellent aquatic macroinvertebrate habitat conditions. As according to McMillan (1998), total IHAS scores of greater than 75 indicate excellent macroinvertebrate habitat conditions; whilst total scores of between 65 and 75 indicate adequate habitat conditions.

6.2 Comparison of aquatic macroinvertebrates results using t-test and Pearson correlation.

There was no significant difference in the SASS5 scores for the Disa and Liesbeek rivers p value 0.73. Moreover, Pearson's correlation indicated a moderate correlation of the SASS5 score of both rivers r value 0.48 (Appendix 5) thus indicating no distinct variation between macroinvertebrate assemblage or composition between a spring fed and non spring fed river.This may be attributed to the fact that these rivers have comparable physical characteristics. Both rivers also fall in the same ecoregion, South West Coastal Belt, and therefore share similar physical and ecological traits (Ollis, 2005). These include; deep incised river channels, with similar habitats including variable substratum composition and flow patterns. The similar river physical and chemical characteristics resulted in the presence of similar aquatic macroinvertebrates adapted to such environments. Both rivers had good water quality and therefore provided a conducive environment for the associated aquatic macroinvertebrates.

There was no significant difference in the number of taxa recorded in the Disa and Liesbeek rivers p value 0.09 and Pearson's correlation indicated a moderate correlation in the number of taxa in both rivers r value 0.50 (Appendix 5). The findings of the current study contradict the results of studies done in the USA such as Barquin (2004), which revealed that spring fed rivers had a subsequently lower number of taxa compared to non-spring fed rivers thereby revealing differences between the types of rivers. The current study revealed no significant differences in the number of taxa as they were similar rivers with similar habitats due to similar geology, geomorphology, vegetation etc. The water contained in the two studied rivers were therefore of similar origin, even if one was spring fed.

There was no significant difference in the average score per taxon recorded in the Disa and Liesbeek rivers and Pearson's correlation indicated a moderate correlation in the ASPT of both rivers r value 0.45. The Average Score per Taxon (ASPT) is one of the indices that represents the average tolerance or pollution sensitivity of all sampled taxa. The comparable ASPT score between both rivers further indicates that there was no variation of macroinvertebrates assemblages between the spring fed and non-spring fed rivers.

6.3 Macroinvertebrate functional feeding groups

The sampled aquatic macroinvertebrates were classified into functional feeding categories as described by Cummins and Klug (1979) viz. shredders, scrapers/grazers, filter feeders, deposit feeding collectors, predators, herbivore piercers, grazers and generalist (Table 13).

Functional feeding groups (FFG) are a classification approach that is based on morphological behavioral mechanisms of food acquisition rather than taxonomic group. The major functional feeding groups are scrapers/grazers, which consume algae and associated material. Shredders, which consume leaf litter or other coarse particulate organic matter (CPOM), including wood. collector-gatherers, which collect fine particulate organic matter (FPOM) from the stream bottom; 4) collectors-filterers, which collect FPOM from the water column using a variety of filters; and 5) predators, which feed on other consumers. A sixth category, other, includes species that are omnivores, or simply do not fit neatly into the above categories.

Table 13: Macroinvertebrates sampled at the Liesbeek and Disa Rivers and their functional feeding group classification.

TAXA	DISA RIVER	LIESBEEK RIVER
	Functional feeding groups	
TURBELARIA	No	Predators
ANNELIDA		
Oligochaeta (Earthworms)	Gathering collectors	Gathering collectors
CRUSTACEA		
Amphipoda	No	Shredders
Potamonautidae* (Crabs)	Shredders	Shredders
HYDRACARINA (Mites)	Predators	No
EPHEMEROPTERA		
Baetidae> 2 sp	Gathering collectors	Gathering collectors
Leptophlebiidae	Gathering collectors	Gathering collectors
Heptagenidae	Scrapers	No
Tricorythidae	Gathering collectors	Gathering collectors
Teloganodidae SWC	Gathering collectors	Gathering collectors
ODONATA		
Coenagrionidae (Sprites and blues)	Predators	No
Aeshnidae (Hawkers & Emperors)	Predators	Predators
Gomphidae	Predators	No
Libellulidae	Predators	No
LEPIDOPTERA		
Crambidae	No	Grazers
HEMIPTERA		
Corixidae* (Water boatmen)	Scrapers	No
Gerridae	Predators	No
Naucoridae	Predators	No
Notonectidae* (Backswimmers)	Predators	No
Pleidae* (Pygmy backswimmers)	Predators	No
Veliidae/M...veliidae* (Riffle bugs)	Predators/Scrapers	Predators/Scrapers
TRICHOPTERA		
Pisuliidae	Shredders	No
Philopotamidae	Filtering collectors	No
Leptoceridae	Gathering collectors/filtering collectors	No
COLEOPTERA		
Scirtidae	Shredders	Shredders
Dytiscidae	Predators	No
Hydraenidae	No	Predators/gathering collectors/scrapers
Elmidae	Gathering collectors	Gathering collectors
DIPTERA		
Athericidae	Predators	No
Chironomidae (Midges)	Predators/gathering collectors/filtering collectors	Predators/gathering collectors/filtering collectors
Psychodidae	Gathering collectors	No
Ceratopogonidae (Biting midges)	No	Predators
Dixidae	No	Gathering collectors
Tipulidae	Predators, shreders & gathering collectors	No
Simuliidae (Blackflies)	Filtering collectors	Filtering collectors

No= not present/recorded

The Liesbeek River was comprised of predators, gathering collectors, filtering collectors, shredders and herbivore piercers. The Disa River was comprised of the same functional feeding groups as the Liesbeek River with the exception of herbivore piercers, Pyralidae were not present at the Disa River. This may be attributed to the high occurrence (48%) of predators at the Disa River (Figure 11), which may have resulted in the preying of the Pyralidae taxa. According to Pabis (2018), larvae of some bettles and dragonflies are known to prey on Pyralidae taxa and the Disa River was characterized by beetles/bugs from different taxa including Gerridae, Naucoridae, Veliidae, Pleidae and Notonectidae (Table 13).

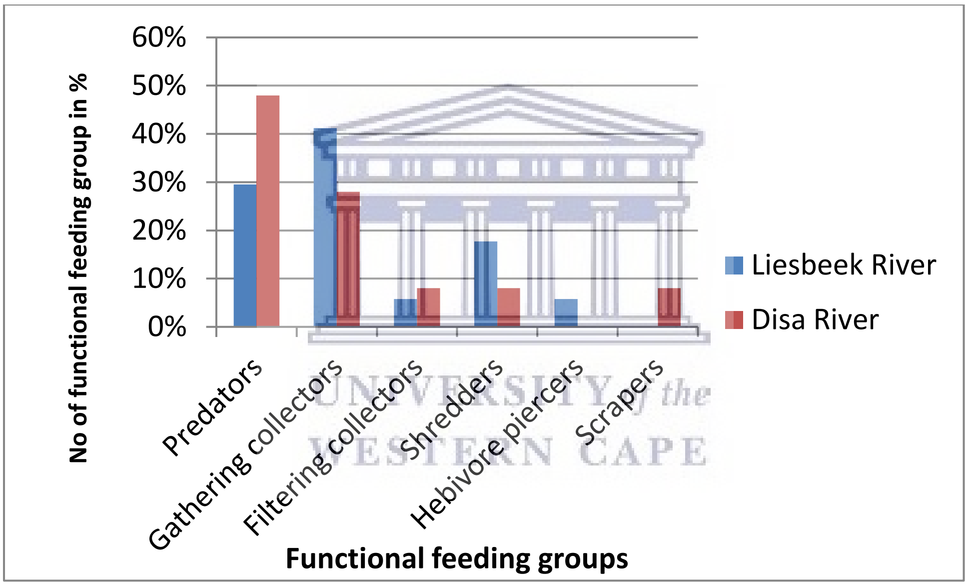

Figure 11:The number of functional feeding groups in percentage found at the Disa and Liesbeek Rivers.

The Liesbeek River comprised of 29.5% of predators, 41.2% of gathering collectors, 5.8% of filtering collectors, 17.7% of shredders and 5.8% of herbivore piercers (Figure 11). The Disa River comprised of 28% of gathering collectors, 48% of predators and 8% of scrapers, shredders and filtering collectors. Predators were the dominant functional feeding group at the Disa River with the Coenagrionidae being the most abundant. The higher abundance of the predators may be attributed to the availability of their prey (Vannote and Sweeny, 1980). Gathering collectors were the dominant functional feeding group in the Liesbeek River with Baetidae being the most abundant taxon. This could be related to the availability of food resources. Rotten logs, leaves and other organic matters, which were characteristic at the Liesbeek River may have increased the soft sediments in the river which are favourable to several gathering collectors (Moreyra and Fonseca, 2015).

6.4 An indicator aquatic macroinvertebrate taxa of a spring fed river.

Aquatic macroinvertebrate taxa that were found in the Liesbeek River with a frequency of occurrence score of three or four and dominant in all sampling seasons but not found at the Disa River were regarded as indicator aquatic macroinvertebrates for the spring fed river, Liesbeek River.

There was only one taxon (Amphipoda) that met the criterion of an indicator for the Liesbeek River. However, it should be noted that in many lotic systems, certain taxa may be highly seasonal while the community as a whole may be less affected by seasonal variation (Thompson and Townsend, 1999; Gibbins *et al*., 2001). Moreover, in some rivers with high seasonal abiotic variability and large numbers of seasonal taxa, taxa common in all seasons can occur (Bogan and Lytle, 2007).

Amphipods have been found in a wide variety of habitats, which includes surface and subterranean habitats. Amphipods were also found to dominate many spring habitats around the world (Gooch and Glazier 1991; Webb *et al*., 1998). A study conducted in New Zealand revealed that there was a high level of diversity of amphipods, which included groundwater and surface water forms, with springs as an area of overlap and, hence, greater diversity (Scarsbrook *et al*., 2007). Barquin (2004) reported that communities dominated by amphipods have also been found in limestone springs elsewhere in Europe and in USA.

The dominance of the amphipoda taxa at the Liesbeek River may be attributed to the fact that in South Africa, amphipods are known to inhabit mountain streams such as the headwater region of the Liesbeek River tributary studied (Wellborn and Cothran, 2015). Their predominance at the

Liesbeek River as opposed to the Disa River may also be attributed to the fact that amphipods are often the most prevalentmacroinvertebrate taxa of freshwater in spring environments (Glazier, 2009). These organisms also have high oxygen requirements and are usually restricted to waters of highdissolved oxygen concentrations (Sutcliffe, 1984). The average DO of the Liesbeek River was 11.45 mg/L, which was higher than of the Disa River (Appendix 1 & Appendix 2). The bed substrate of the Liesbeek River was dominated by different sized stones ranging from sand grains, cobbles to bedrock thereby proliferating the dissolved oxygen content and in turn providing favourable water and habitat conditions for amphipods.

6.5 Implication of the study's results for catchment management

Managing and or mitigating the effects of land use activities on water quality requires the identification and quantification of sources of pollutants linked to specific land use activities. Identifying and quantifying pollution derived from non-point sources in particular is very challenging. This is because nonpoint source pollution is derived from a large surface area, which is often heterogeneous in terms of the land use activities taking place in the catchment (Dabrowski *et al.*, 2013). Based on the study's results, the water quality of both rivers was in a near pristine condition substantiated by the diversity of moderately to sensitive aquatic macroinvertebrates to water pollution. Moreover, the river flows were natural at the Liesbeek River with no identifiable disturbance such as abstraction with the Disa River being dammed upstream. Both studied river reaches were within protected areas, the Liesbeek River sites in Newlands Forest and the Disa River sites in the Orangekloof Nature reserve where access is granted and monitored by the South African National Parks (SANParks). There were no identifiable point sources of pollution in the study area with the exception of anthropogenic activities including open defecation by dogs along the river bank which, should be avoided and desludge from the Constantia Nek water treatment plant that is sprayed in an open lawn in Orangekloof nature reserve.

The preservation or maintenance of existing water quality and instream habitat to avoid significant changes in aquatic macroinvertebrates abundance and diversity is essential. Awareness programmes may be organized using print and electronic media to stop the malpractices of defecation along the river banks (Mariya *et al*., 2019). Community-based projects that would include all stakeholders that affect the catchment of the Disa River need to be developed and implemented. One already exists for the Liesbeek River termed "Friends of the Liesbeek". The study revealed that during the summer and autumn months, the river comprises low flows and no abstraction should occur. The low flows should be maintained to sustain existing aquatic macroinvertebrates during the summer months. During the winter season, the river is characterized by higher flows and abstraction can occur while maintaining natural flows in order to avoid significant changes in aquatic macroinvertebrates adapted to high flows.

The near pristine water quality conditions of the studied rivers are mainly attributable to the officially protected areas in which they occur. Maintaining and preservation of current land use/land cover will ensure that essential ecosystem services are maintained including the provision of good water quality and ecological river health. The alteration of land cover has caused deforestation, one of the biggest issues in recent decades. According to Zukilfi *et al*. (2017), forest ecosystems are crucial and serve different functions and ecosystem services such as supporting soil development, supporting the nutrient and water cycle, providing fresh water supply, regulating erosion, and water purification. Statutory resource and land use plans, including river management plans, should assess and control potentially deleterious impacts on these ecosystems at catchment scales.

6.5 Summary

The findings of the study revealed that the Disa and Liesbeek River environments were largely natural, being characterized by high-scoring sensitive macroinvertebrate taxa This was further substantiated by water quality results that complied with the set DWAF (1996) aquatic ecosystem guidelines.

There was no significant difference in aquatic macroinvertebrates assemblage composition between the studied rivers with the exception of the amphipods dominant in the Liesbeek River. The amphipoda taxa has been found to dominate mountain streams and freshwater spring environments worldwide (Glazier, 2009; Wellborn and Cothran, 2015). Seasonality also affected aquatic macroinvertebrate diversity and abundance with the low flow summer season characterized by some tolerant aquatic macroinvertebrates to water pollution such as Oligochaeta and Turbellaria and the spring season comprising of a diversity of high-scoring pollution-sensitive aquatic macroinvertebrates taxa.

7. CONCLUSION

The comparison of the two rivers enabled the differentiation between the ecological status of the spring fed Liesbeek River and the non-springfed river, Disa River. In light of the research objectives outlined in section **1.3 Objectives**, the following conclusions can be made:

1. There was no significant difference in the river discharge of the spring fed Liesbeek River and non-spring fed Disa River. The discharge of both rivers was similar thus indicating that the flow received from the spring feeding the Liesbeek River was not significantly different when compared to the river discharge of the Disa River.

   The water quality of the spring fed Liesbeek River and of the non-spring fed Disa River significantly varied. Of the 14 water quality parameters that were monitored, five water quality parameters (iron, sodium, chloride, aluminium and total dissolved solids) significantly varied between the Disa and Liesbeek Rivers, with the concentration of each determinant increased at the Disa River compared to the Liesbeek River. The variation of some of the water quality determinants were mostly attributed to natural variability such as weathering of rocks and leaf litter and Constantia Nek WTW sludge that is sprayed at the Orangekloof lawn where runoff may cause an increase in certain determinants including aluminium.

   There was no significant difference in aquatic macroinvertebrates associated with either the spring fed and non-spring fed rivers, as similar aquatic macroinvertebrates were recorded across the two rivers. However, there were some ecological differences between the Disa and Liesbeek Rivers such as the dominant occurrence of amphipods, known to inhabit mountain streams such as the freshwater spring environment of the Liesbeek River.

2. A greater diversity and abundance of aquatic macroinvertebrates was observed during the spring season where the river flows were moderate and favourable for aquatic macroinvertebrates in both rivers. There was a lesser diversity and abundance of aquatic macroinvertebrates during the winter season owing to higher flows resulting in drifting of aquatic macroinvertebrates. In summer the lesser macroinvertebrate abundance and diversity observed was attributed to shrinking habitats due to lower river flows and isolated pools.

   The current study also revealed that although the Disa and Liesbeek Rivers were characterized by a diverse abundance of highly sensitive aquatic macroinvertebrates, a few tolerant aquatic taxa were also recorded. The dominance of predators at the Disa River could be attributed to the availability of prey and site-specific habitat conditions. Gathering collectors at the Liesbeek River particularly Baetidae were dominant and attributed to the availability of food resources, such as rotten logs, leaves and other organic matters that were characteristic at the Liesbeek River, which also accounted for the increased number of shredders compared to the Disa River. However, the presence of pollution-tolerant macroinvertebrates in the Disa River was not an indication of water pollution due to low abundances and due to increased abundance of highly sensitive taxa. This was further substantiated by continually occurring high SASS5 and ASPT scores.

   The findings of the study revealed that the ecological state of the Disa and Liesbeek Rivers were largely natural, in an unmodified state as both rivers were characterized by high scoring sensitive aquatic macroinvertebrates indicative of near natural or pristine river water quality. This was further substantiated by water quality results that complied with the DWAF (1996) aquatic ecosystem guidelines.

3. There were only one aquatic macroinvertebrate taxa (Amphipoda) that met the criteria of indicator aquatic macroinvertebrate taxa for the Liesbeek River known to be dominant in freshwater springs mountain streams.

7.1 Limitations of the study

The study was intended to provide or enhance knowledge about the ecology, water quality and river flows of a spring fed river in comparison to a non-spring fed river. The following specific limitations pertained:

- Taxonomic identifications of aquatic macroinvertebrates were only performed to family level as per the SASS5 method.
- The study did not assess other ecological components such as fish, zooplankton or algae and riparian vegetation.
- There was no historical river flow data available for the Disa River to confirm or validate the obtained river discharge of the current study. The river discharge was also only measured during SASS5 sampling events and not continoulsy as no gauging weir was present in either of the rivers assessed.
- The spring fed river is a groundwater system and no groundwater data was collected or data to assess the surface water groundwater interactions and the associated freshwater ecosystems potentially supported by this.
- The study was also of a very limited period. Longer-term flow data and ecological data such as macroinvertebrate, fish, algae and vegetation will be required for a more holistic view of ecological functioning.
- A longer monitoring period is required to draw meaningful conclusions from the statistical analyses.

7.2 Recommendations

- Further research on spring fed rivers is required, which should employ a holistic approach such as investigating riparian and instream vegetation, channel morphology, bed substrate, fish, zooplankton and algae. All these components are interlinked and would provide a greater understanding to advance knowledge on the ecological functioning and characteristics of spring fed rivers.
- Should the spring be developed, a comprehensive and effective monitoring programme need to be initiated and conducted on the Liesbeek River to ensure resource protection.
- Groundwater monitoring (levels and quality, etc.) should be implemented as the spring fed river is of groundwater origin.

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9. APPENDICES

Appendix 1

Monthly water quality results for the Disa River site 1 for the study period.

Month	Aluminium (mg/L)	Ammonia (mg/L)	Chloride (mg/L)	Electrical conductivity (MSM)	Iron ugl (mg/L)	Lead (mg/L)	Nitrate (mg/L)	pH	Sodium (mg/L)	Sulphate (mg/L)	Total Dissolved Solids (mg/L)	Turbidity (NTU)	Temperature ⁰C	Dissolved Oxygen (mg/L)
Aug-18	0.35	0.02	21		0.27	0.002	0.2		13.8	0.1	68.3	1.55		
Sep-18	0.23	0.02	32	13.7	0.26	0.001	0.2	5.7	17.3	2	84.4	0.61	14.8	9.5
Oct-18	0.25	0.03	35		0.27	0.002	0.2		17	1.7	85.1	0.7		
Nov-18	0.28	0.02	38		0.26	0.003	0.1		18.1	1.5	84	1.7		
Dec-18	0.25	0.01	36	16	0.23	0.001	0.1	5.6	17	2	91	0.9	17.8	8.9
Jan-19	0.18	0.02	39		0.24	0.01	0.1		17.6	3	89	0.6		
Feb-19	0.2	0.01	38		0.2	0.002	0.2		18	4.7	93.8	0.65		
Mar-19	0.21	0.01	33	11.25	0.1	0.017	0.2	5.5	18.3	4	84.4	0.6	16.1	8.5
Apr-19	0.29	0.03	36		0.22	0.02	0		19	3	81	0.8		
May-19	0.23	0.01	31		0.18	0.001	0		17	3	95	1		
Jun-19	0.16	0.03	37	12.6	0.29	0.002	0.1	5.2	22.6	5	101.8	5.59	12	9.2
Jul-19	0.32	0.04	29		0.24	0.018	0.2		16.6	3	85.1	0.88		

Monthly water quality results for the Disa River site 2 for the study period.

Month	Aluminium (mg/L)	Ammonia (mg/L)	Chloride (mg/L)	Conductivity (MSM)	Iron ugl (mg/L)	Lead (mg/L)	Nitrate (mg/L)	pH	Sodium (mg/L)	Sulphate (mg/L)	Total Dissolved Solids (mg/L)	Turbidity (NTU)	Temperature ºC	Dissolved Oxygen (mg/L)
Aug-18	0.41	0.03	19		0.26	0.002	0.2		14.1	0.8	62.6	1.1		
Sep-18	0.28	0.02	34	12.4	0.25	0.002	0.2	6.1	16.9	3.1	79.8	0.77	13.8	8.7
Oct-18	0.24	0.03	35		0.27	0.002	0.2		17.2	1.9	88.3	1.07		
Nov-18	0.31	0.01	36		0.22	0.004	0.1		19	2.1	89	1.6		
Dec-18	0.26	0.02	32	13.4	0.25	0.019	0.2	5.5	18.7	3.6	86	0.99	16.9	8.1
Jan-19	0.18	0.03	34		0.25	0.01	0.1		17.3	2.7	92	0.8		
Feb-19	0.19	0.01	38		0.24	0.001	0.2		17.6	3.9	91.6	0.62		
Mar-19	0.24	0.04	33	12.6	0.2	0.017	0.2	5.9	18.3	4.2	84.4	0.7	16.2	7.9
Apr-19	0.31	0.03	31		0.22	0.023	0.1		21.2	2.4	87	0.8		
May-19	0.21	0.01	30		0.19	0.002	0.1		19.1	1	98	1.2		
Jun-19	0.15	0.03	33	14	0.25	0.002	0.2	6.3	21.9	4	97	4.6	12.4	8.9
Jul-19	0.39	0.02	31		0.21	0.021	0.2		17.1	2	88.1	0.88		

Appendix 2

Monthly water quality results of the Liesbeek River site 1 for the study period.

Month	Aluminium (mg/L)	Ammonia (mg/L)	Chloride (mg/L)	Conductivity (MSM)	Iron (mg/L)	Lead ugl (mg/L)	Nitrate (mg/L)	pH	Sodium (mg/L)	Sulphate (mg/L)	Total Dissolved Solids (mg/L)	Turbidity (NTU)	Temperature ºC	Dissolved Oxygen (mg/L)
Aug-18	0.3	0.04	17		0.2	0.002	0.2		8.3	0.3	43.6	1.1		
Sep-18	0.09	0.02	20	9.1	0.05	0.001	0.2	5.8	11.6	4.5	51.6	0.67	16	13
Oct-18	0.13	0.01	19		0.07	0.002	0.1		10.7	3.8	52.3	1.12		
Nov-18	0.05	0.03	21		0.05	0.003	0.1		11.3	3.4	57.6	0.75		
Dec-18	0.09	0.01	22	9.4	0.09	0.001	0.1	6.3	12.5	3.8	60.3	0.76	17.4	9
Jan-19	0.05	0.01	20		0.07	0.01	0.1		12.2	3.6	55.6	0.7		
Feb-19	0.04	0.04	20		0.06	0.002	0.1		12.2	3.6	59	1.69		
Mar-19	0.04	0.02	20	9.7	0.05	0.017	0.1	6.6	12.4	3.5	60.3	1.52	16.4	9.9
Apr-19	0.05	0.01	21		0.05	0.02	0.1		11.9	3.9	61	0.73		
May-19	0.07	0.04	13		0.09	0.001	0.2		11.5	1.6	57.6	1.2		
Jun-19	0.05	0.02	19	9.9	0.05	0.002	0.1	6.7	10.8	4	61	0.6	13	14
Jul-19	0.21	0.01	17		0.09	0.018	0.2		12.4	0.7	68.3	0.6		

Monthly water quality results of the Liesbeek River site 2 for the study period.

Month	Aluminium (mg/L)	Ammonia (mg/L)	Chloride (mg/L)	Conductivity (MSM)	Iron (mg/L)	Lead ugl (mg/L)	Nitrate (mg/L)	pH	Sodium (mg/L)	Sulphate (mg/L)	Total Dissolved Solids (mg/L)	Turbidity (NTU)	Temperature ºC	Dissolved Oxygen (mg/L)
Aug-18	0.6	0.01	22		0.12	0.003	0.1		10	3.4	60.3	1.06		
Sep-18	0.15	0.03	20	8.8	0.08	0.002	0.2	6.2	12.1	2.5	53.6	0.6	16.8	11.9
Oct-18	0.16	0.04	22		0.04	0.001	0.1		11.2	3.7	60.3	0.6		
Nov-18	0.09	0.03	20		0.06	0.004	0.1		9.9	3.4	53.6	0.66		
Dec-18	0.12	0.02	21	9.2	0.07	0.003	0.1	6.6	10.5	4	60.3	0.74	17.7	8.7
Jan-19	0.08	0.03	22		0.05	0.01	0.1		10.7	3.6	60.3	0.6		
Feb-19	0.05	0.07	22		0.06	0.01	0.1		11.2	3.7	60.3	2.01		
Mar-19	0.07	0.02	21	9.4	0.09	0.019	0.1	6.7	13.1	4.2	60.3	0.6	17	9.1
Apr-19	0.06	0.03	22		0.07	0.009	0.1		12.8	3.7	60.3	1.09		
May-19	0.04	0.01	25		0.08	0.004	0.1		11.5	3.8	73.7	0.6		
Jun-19	0.07	0.03	23	9.6	0.06	0.006	0.1	6.4	10.9	3.6	60.3	0.84	14	11
Jul-19	0.19	0.03	21		0.05	0.009	0.1		12.6	3.2	60.3	0.6		

Appendix 3

Aquatic macroinvertebrate diversity and abundance of the Liesbeek River.

Taxon	Sensitivity Score	Winter		Spring		Summer		Autumn	
Rivers: Liesbeek River (L)		**Site 1**	**Site 2**	**Site 1**	**Site 2**	**Site 1**	**Site 2**	**Site 1**	**Site 2**
TURBELARIA	**3**	0	0	A	B	B	B	A	A
ANNELIDA									
Taxon	**Sensitivity Score**	**Winter**		**Spring**		**Summer**		**Autumn**	
Oligochaeta (Earthworms)	**1**	A	A	A	A	C	C	A	A
CRUSTACEA									
Potamonautidae* (Crabs)	**3**	A	A	A	A	A	A	A	A
Amphipoda	**13**	B	C	B	C	A	B	A	B
HYDRACARINA (Mites)	**8**	0	0	0	0	0	0	0	0
EPHEMEROPTERA									
Baetidae> 2 sp	**12**	B	B	B	C	B	B	B	B
Leptophlebiidae	**9**	B	B	B	B	0	0	1	0
Heptagenidae	**13**	0	0	0	0	0	0	0	0
Teloganodidae SWC	**12**	A	A	A	A	A	B	A	A
Tricorythidae	**9**	A	A	A	B	0	0	0	0
ODONATA									
Coenagrionidae (Sprites and blues)	**4**	0	0	0	0	0	0	0	0

Aeshnidae (Hawkers & Emperors)	**8**	A	A	A	B	A	A	A	A
Gomphidae	**6**	0	0	0	0	0	0	0	0
Libellulidae	**4**	0	0	0	0	0	0	0	0
LEPIDOPTERA									
Crambidae	**12**	0	0	A	A	0	0	0	0
HEMIPTERA									
Corixidae* (Water boatmen)	**3**	0	0	0	0	0	0	0	0
Gerridae	**5**	0	0	0	0	0	0	0	0
Naucoridae	**7**	0	0	0	0	0	0	0	0
Notonectidae* (Backswimmers)	**3**	0	0	0	0	0	0	0	0
Pleidae* (Pygmy backswimmers)	**4**	0	0	0	0	0	0	0	0
Veliidae/M...veliidae* (Riffle bugs)	**5**	0	0	A	1	B	B	A	A
Taxon	**Sensitivity Score**	**Winter**		**Spring**		**Summer**		**Autumn**	
TRICHOPTERA									
Barbarochthoriidae SWC	**13**	A	A	B	B	A	B	A	A
Pisuliidae	**10**	0	0	0	0	0	0	0	0
Philopotamidae	**10**	0	0	0	0	0	0	0	0
Leptoceridae	**6**	0	0	0	0	0	0	0	0
COLEOPTERA									
Scirtidae	**12**	0	0	0	0	0	0	0	0
Dytiscidae	**5**	0	0	0	0	0	0	0	0

Elmidae	**8**	0	0	0	0	B	B	0	1
Hydraenidae	**8**	0	0	0	1	A	A	0	0
DIPTERA									
Athericidae	**10**	0	0	0	0	0	0	0	0
Chironomidae (Midges)	**2**	A	A	A	A	1	A	A	A
Psychodidae	**1**	0	0	0	0	0	A	A	A
Tipulidae	**5**	0	0	0	0	0	0	0	0
Dixidae	**10**	0	0	0	0	A	A	A	1
Ceratopogonidae (Biting midges)	**5**	A	A	0	0	A	A	A	A
Simuliidae (Blackflies)	5	B	B	B	B	B	B	B	B
Total SASS score		92	92	107	128	93	95	102	100
No of taxa		12	12	14	16	15	14	16	14
ASPT		7.6	7.6	7.6	8	6.2	6.7	6.3	7.1

Appendix 4

Aquatic macroinvertebrate diversity and abundance of the Disa River.

Taxon	Sensitivity Score	Winter		Spring		Summer		Autumn	
Rivers: Disa River (D)		Site 1	Site 2	Site 1	Site 2	Site 1	Site 2	Site 1	Site 2
TURBELARIA	**3**	0	0	0	0	0	0	0	0
ANNELIDA									
Taxon	**Sensitivity Score**	**Winter**		**Spring**		**Summer**		**Autumn**	
Oligochaeta (Earthworms)	**1**	A	A	1	A	A	A	1	A
CRUSTACEA									
Potamonautidae* (Crabs)	**3**	A	A	A	A	A	A	A	1
Amphipoda	**13**	0	0	0	0	0	0	0	0
HYDRACARINA (Mites)	**8**	0	0	1	0	0	0	0	0
EPHEMEROPTERA									
Baetidae> 2 sp	**12**	0	A	B	B	B	B	B	B
Leptophlebiidae	**9**	B	B	B	B	0	0	0	0
Heptagenidae	**13**	B	B	0	0	A	A	B	A
Teloganodidae SWC	**12**	0	0	B	0	0	0	0	0
Tricorythidae	**9**	0	0	0	0	0	0	0	0
ODONATA									
Coenagrionidae (Sprites and blues)	**4**	A	A	B	A	A	A	1	A
Aeshnidae (Hawkers & Emperors)	**8**	A	A	A	A	A	A	A	A

Gomphidae	**6**	A	A	0	0	0	0	A	A
Libellulidae	**4**	0	0	0	0	1	A	0	0
LEPIDOPTERA									
Crambidae	**12**	0	0	0	0	0	0	0	0
HEMIPTERA									
Corixidae* (Water boatmen)	**3**	A	A	0	0	0	0	A	A
Gerridae	**5**	0	0	0	0	A	A	0	0
Naucoridae	**7**	A	0	0	0	0	0	0	0
Notonectidae* (Backswimmers)	**3**	0	0	0	0	0	0	1	A
Pleidae* (Pygmy backswimmers)	**4**	0	0	0	0	0	0	B	B
Veliidae/M...veliidae * (Riffle bugs)	**5**	A	A	A	A	A	A	A	A
Taxon	**Sensitivity Score**	**Winter**		**Spring**		**Summer**		**Autumn**	
TRICHOPTERA									
Barbarochthoriidae SWC	**13**	0	0	B	A	0	0	0	0
Pisuliidae	**10**	0	0	1	0	0	0	0	0
Philopotamidae	**10**	A	A	A	A	1	0	A	A
Leptoceridae	**6**	B	A	0	0	0	0	0	0
COLEOPTERA									
Scirtidae	**12**	0	0	A	A	0	0	0	0
Dytiscidae	**5**	0	0	0	0	1	0	1	A
Elmidae	**8**	0	0	B	A	0	0	0	0
Hydraenidae	**8**	0	0	0	0	0	0	0	0
DIPTERA									

Athericidae	**10**	A	A	A	A	A	A	A	A
Chironomidae (Midges)	**2**	B	A	B	B	A	A	1	A
Psychodidae	**1**	0	0	0	0	0	0	0	0
Tipulidae	**5**	0	0	0	0	A	A	0	0
Dixidae	**10**	0	0	0	0	0	0	0	0
Ceratopogonidae (Biting midges)	**5**	0	0	0	0	0	0	0	0
Simuliidae (Blackflies)	**5**	B	B	C	B	B	B	B	B
Total SASS score		92	97	132	102	98	82	104	94
No of taxa		15	15	17	14	16	13	16	16
ASPT		6	6.4	7.7	7.2	6.1	6.3	6.5	5.8

Appendix 5

Correlation results of aquatic macroinvertebrates of the Disa and Liesbeek rivers.

	SASS score Liesbeek	*SASS score Disa*	*No of taxa Liesbeek*	*No of taxa Disa*	*ASPT liesbeek*	*ASPT Disa*	*IHAS scores (%) Liesbeek*	*IHAS scores (%) Disa*
SASS score Liesbeek	1							
SASS score Disa	0.487955271	1						
No of taxa Liesbeek	0.636545328	0.201636478	1					
No of taxa Disa	-0.10342423	0.718394759	0.53083819	1				
ASPT liesbeek	0.474710654	0.264095519	-0.374951081	-0.14517823	1			
ASPT Disa	0.672362579	0.806965877	0.315576421	0.173762012	0.45422	1		
IHAS scores (%) Liesbeek	0.394613406	0.379460012	-0.318753167	-0.06842715	0.853322	0.562568677	1	
IHAS scores (%) Disa	0.655906302	0.724455567	0.01998003	0.177488628	0.780454	0.86410261	0.854937221	1