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Research Project:

A comprehensive environmental assessment of the Braamfontein Spruit and surrounding tributaries in Northern Johannesburg in 2023 and 2024

Submitted in fulfilment of the requirements for the degree

Master of Science

in

Geography, Archaeology and Environmental Studies

in the Faculty of Science, University of the Witwatersrand, Johannesburg, South Africa

Supervisor:

Professor Craig Sheridan

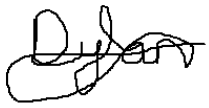
Co-supervisor:

Dr Simone Dahms-Verster

April 2025

Declaration

I Dylan Butler declare that this Dissertation/Research is my own, unaided work. It is being submitted for the Degree of Geography and environmental studies at the University of the Witwatersrand, Johannesburg. It has not been submitted before for any degree or examination at any other University.



____ _____

(Signature of candidate)

05_____day of__April _____2025____in_____

Dedication and Acknowledgements

In appreciation of my supportive parents, Kim Butler and Michael Butler, as well as my supervisor Craig Sheridan and co-supervisor, Simone Dahms-Verster, without whom I could not have completed this work. Thank you.

I would like to thank the Claude Leon Foundation for their generous bursary, helping fund my studies. I would also like to thank my parents for the financial and emotional support throughout my studies. Thank you to my supervisor Craig Sheridan and co-supervisor Simone Dahms-Vorster for their guidance and support.

Abstract

Urban environments are complex systems comprising of interconnected systems consisting of built-up areas, gardens, parks and rivers. These ecosystems provide essential services that support climate resilience, yet urban developments places them under stress, reducing their ability to perform such services. The Braamfontein Spruit in Johannesburg, South Africa, illustrates how urbanisation impacts river health and ecosystem services. This study aimed to assess the spatiotemporal variation in water quality across multiple sites along the Braamfontein Spruit catchment and tributaries, to evaluate the influence of human activity on river health.

Sampling was conducted at five sites during three seven-week periods between April 2023 and March 2024. Sites included the two tributaries (Westdene Spruit and Montgomery Spruit0, the Parkview Golf Course, Delta Park, and the Sandton Riverclub Golf Course. Field measurements were taken using a Merck multivariate probe, while biodiversity was assessed using the SASS5 method. Water and sediment samples were collected and analysed for nutrients (sulphate, nitrite, nitrate, phosphate), microbial indicators (*E. coli* and coliforms), and metals (Fe, Pb, Mn) using standard laboratory methods.

Results revealed variable but consistently concerning contamination across sites and seasons. Nutrient and microbial concentrations frequently exceeded South African Department of Water and Sanitation guidelines, with particularly high *E. coli* and coliform levels at sites along the Braamfontein Spruit. SASS5 biomonitoring scores were low, indicating degraded ecosystems, and negative correlations were observed between nutrient and microbial levels and macroinvertebrate richness.

These findings highlight the cumulative impacts of urban pollution in Johannesburg's rivers and underscore the urgent need for improved water quality management in areas of high recreational and domestic use.

Key Words: Urban rivers, ecosystem services, Braamfontein Spruit, environmental assessment,

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Glossary of abbreviations

E. coli – Escherichia coliforms

pH- Potential of hydrogen

EC- Electrical conductivity

DO- Dissolved oxygen

ASPT- Average Score Per Taxon

DWAF- Department of Water Affairs

SASS5- South African Scoring System volume 5

ISQG- Canadian Interim Sediment Quality Guidelines

TES- Threshold Effect Levels

PEL- Probable effect level

AMD – Acid Mine Drainage

1. Introduction

a. Background

Rivers have historically served as defining features in human settlement and development, but these rivers have, in turn, been degraded by urbanisation (Everard & Moggridge, 2012). Urban development transforms the hydrological system through the process of construction of new passageways, such as stormwater drains, as well as the building of new structures that block or redirect existing water passages (Gurnell et al., 2007). Urban rivers are key natural elements of urban areas, providing ecoservices such as regulating temperatures, facilitating drainage from storms and recreational activities, and contributing to biodiversity growth. When urban rivers are degraded the ecoservices that they provide are diminished, and the polluted rivers can cause harm to humans and biodiversity (Everard & Moggridge, 2012). This study involves close monitoring of five urban river sites in Northern Johannesburg, South Africa through the means of a comprehensive environmental assessment consisting of water sampling, sediment sampling, and macroinvertebrate identification. The importance of urban river monitoring cannot be overstated for key reasons such as the importance of water quality management, which ensures the safety of residents and people who come into contact with the river, ecosystem health, to ensure conservation of biodiversity, urban development and to ensure that regulatory compliance is being followed by local officials in governance. By ensuring that pollution levels within urban river systems are reduced to low levels, the health and resilience of the river improve, and ecosystem services are protected. This has the potential to improve biodiversity and benefit humans who access the river and surroundings. The negligence of sewage transport and treatment systems, as well as negligence of urban rivers by city managers in Johannesburg, resulted in heavily polluted water systems (Mavhungu & Mankga, 2024). These polluted water systems have a negative impact on the inhabitants who use the river for various purposes, such as recreation or food sources.

b. Rationale

There is currently insufficient data on the water quality (and hence environmental health) of urban rivers which results in them having the potential to pose a significant risk to people who reside near and make use of the rivers or come into contact with the water. This exists because there is a lack in consistent assessments of the quality of the water and health of the river. This gap in research is a result of poor governance and lack of accountability by the Integrated Catchment Management (ICM) Unit which falls under Ecosystems Enhancement and Open

Space Management Department (EOD) (Mavasa, 2023). The ICM is responsible for management and maintenance of all the water bodies across the City of Johannesburg along the Jukskei and Klip river and for the state of Johannesburg's urban river systems.

Without consistent and thorough environmental assessments, uncertainty regarding the health of the river and associated human health risks remain a concern for anyone who may encounter the polluted water. Since these urban rivers receive varying amounts of pollutants based on seasonality, the amount of illegal pollution dumping and the state of sewerage infrastructure, regular sampling is required to identify the severity of the impact to the river system. The results from the samples could inform the public about the occurrence and frequency of different pollutants in the rivers, and hence the potential risk they pose to environmental and human receptors. Additionally, there is a possibility of identifying sources which could assist authorities in prosecuting polluters or considering strategies to remediate impact.

c. Research Questions

1. How polluted are the urban rivers in Northern Johannesburg?
2. How much variation in pollution is there over the course of one year?
3. Is it possible to identify sources of pollution for these urban rivers?
4. What parameters are needed to provide a comprehensive environmental assessment of an urban river in Johannesburg?
5. What are the possible impacts on human health that polluted water systems can cause?
6. Are there ways to improve the water quality and health of these urban river systems?

a. Objectives

1. To assess the contamination levels in urban rivers in Johannesburg over a one-year period, through regular water, sediment, and invertebrate sampling across different seasons.
2. To identify the sources of pollution in the Jukskei River and its tributaries (Montgomery and Westdene Spruits), focusing on the spatial and seasonal variations in contamination levels. By identifying areas of increased pollution markers, general sources of pollution are identified.
3. To evaluate the impact of pollutants on aquatic life by analysing the presence and diversity of invertebrates (using the SASS5 method) across the five urban river sites.
4. To measure and quantify key pollutants, including nutrient levels, *Escherichia coli (E. coli)* coliform concentrations, and metals in the water and sediment samples.

5. To assess the potential risks to human health posed by the contaminated rivers, especially for those who come into contact with the water for recreational or subsistence activities, namely the informal communities, as well as people downstream once the river joins with the Crocodile River and flows into Hartbeespoort Dam. This is achieved by identifying and quantifying the levels of pollution across various parameters and comparing the findings to similar studies.
6. To provide recommendations for improving urban river management and reducing pollution levels, based on the findings of the study.

The results of this study can contribute to the body of knowledge around the health of urban rivers in South Africa.

7. Literature review

a. Ecosystems and ecosystem services

Ecosystems are unique places which have interconnected biodiversity and environments which depend on each other's activities in order to properly function (Savard et al., 2000). These unique systems vary, in terms of the types of biodiversity and the aspects of the environment, both spatially and temporally and consist of both biotic (living) and abiotic features (non-living or inanimate) (Savard et al., 2000). Different temperatures, altitudes, latitude and weather all have large impacts on shaping ecosystems (Savard et al., 2000). Urban ecosystems are unique spaces dominated by anthropogenic, or human activities. This has had various impacts on the natural environments which has caused stress and irreversible change on the natural ecosystems that existed before urbanisation (Everard & Moggridge, 2012). Urban ecosystems are built up environments which have green spaces, such as green belts, parks and gardens, and rivers strategically placed within the built-up environment as a way of building resilience to climate change and provide areas of recreation for inhabitants (Savard et al., 2000). These greenspaces and rivers provide ecosystem services which are vital to the survival of the biodiversity, functioning of ecosystems and provide benefits to society (Savard et al., 2000). Ecosystem services are comprised of ecosystem processes and functions and are categorised as provisional (meaning energy outputs from the environment like food, raw materials and water), regulation (services which assist in moderating natural processes like carbon storage and climate regulation), cultural services (benefits for humans such as recreational and aesthetic appeal) and supporting (habitat provisions for species) (Luederitz et al., 2015).

b. Rivers and Riparian environments

Rivers are essential water sources which have important roles for both natural ecosystems and human societies, including water transport, natural purification and supporting biodiversity (Pinto & Maheshwari, 2011). They serve as ecosystems for many vertebrates, invertebrates, bacteria and plants, forming diverse interdependent habitats that support diverse forms of life (Baker & Greenfield, 2019). Beyond their ecological roles, rivers provide essential resources for humans, such as contributing to food production and agriculture and even economic stability (Baker & Greenfield, 2019). These complex and dynamic systems are shaped by a combination of biotic and abiotic factors, making them both diverse and multifaceted (Baker & Greenfield, 2019).

However, the rapid increase in anthropogenic impacts since the industrial revolution has placed unprecedented pressure on rivers, resulting in profound alterations to their structure and function, particularly to riparian ecosystems (Smith et al., 1999). Rivers are amongst some of the most threatened ecosystems in the world (Baker & Greenfield, 2019). This is largely driven by exponential human population growth and growing demands on terrestrial and aquatic ecosystems (Smith et al., 1999). Significant human-induced land transformation, such as deforestation, urban sprawl, agricultural expansion and the construction of dams has occurred as a direct result of human interference with the natural world through land clearing, agricultural activities, urban development, and building dams have fundamentally altered the hydrological cycle, disrupting the delicate balance of river systems (Smith et al., 1999).

These anthropogenic changes have far reaching consequences for the river health and functioning of river ecosystems, impacting aquatic plants, animals, biodiversity, oxygen availability, and essential ecosystem services (Baker & Greenfield, 2019). Additionally, the introduction of invasive species has exacerbated these challenges, disrupting natural biological communities and altering the ecosystem services available (Shousha et al., 2021). In addition to these disruptions, anthropogenic activities have also altered the biochemical cycles of key nutrients, including carbon (C), nitrogen (N) and phosphorus (P), which are integral to maintaining the balance of ecosystems (Shousha et al., 2021). Changes in the levels of nitrogen and phosphorus in aquatic systems can lead to nutrient imbalances, which, in turn, affect the growth and health of aquatic plants as these two nutrients play significant roles in the growth of plants in aquatic systems and on terrestrial environments (Xia et al., 2014). Furthermore, the decay of aquatic plants contributes to the cycling of these nutrients back into the ecosystem, influencing the availability of N, C and O and shaping the overall health of aquatic environments (Xia et al., 2014).

C. Nutrients in urban rivers

i. Nitrates and Nitrites

Polluted water bodies significantly impair an ecosystem's ability to provide essential services, leading to further degradation of water quality and ecosystem health (Shousha et al., 2021). In urban rivers, the presence of inorganic nitrogen is commonly associated with surface runoff and the discharge of effluent, which may be contaminated by fertilizers, organic industrial waste, as well as human and animal excrement (Whitehead et al., 2006). In unimpacted urban rivers, inorganic nitrogen concentrations are generally low, as aquatic plants readily absorb and

convert nitrogen into proteins and organic nitrogen compounds within their cells. However, in highly impacted urban rivers, inorganic nitrogen can exceed natural levels, resulting in excessive aquatic plant growth and algal blooms (Angove et al., 2018).

The primary concern with elevated inorganic nitrogen in urban rivers is its capacity to stimulate the overgrowth of aquatic plants and algae. In South Africa, typical concentrations of nitrogen in unimpacted water bodies are below 0.5 mg/L (Silberbauer, 1996), whereas nutrient-enriched waters can see levels rise to between 5 and 10 mg/L (Silberbauer, 1996). This nutrient enrichment triggers a range of ecological consequences, most notably the promotion of aquatic plant and algae growth, which can disrupt the balance of the ecosystem. Factors such as water temperature, pH, and oxygen availability significantly influence the nitrogen cycle, affecting the conversion processes and the uptake of nitrate by algae and aquatic plants (Shousha et al., 2021; Xia et al., 2014).

Nitrates (NO_3^-) and nitrites (NO_2^-) are important nitrogen species found in freshwater systems from both natural process and anthropogenic activities. These compounds usually enter rivers through stormwater runoff, wastewater discharge and atmospheric deposition (Whitehead et al., 2006). In ecosystems which have elevated nitrogen concentrations, algal blooms reduce clarity, disrupt food webs and deplete oxygen levels during plant decomposition, often leading to anoxic or hypoxic conditions which can stress fish and invertebrates (Banerjee et al., 2023).

In aerobic conditions, nitrifying bacteria oxidize nitrites to form nitrates, which are more abundant in impacted aquatic environments (Brezonik, 2013). In contrast, when anaerobic conditions prevail, denitrifying bacteria reduce nitrates to nitrites, a process that leads to the loss of nitrates from the system (Brezonik, 2013). These microbial processes are central to understanding nitrogen cycling and the impact of altered nitrogen levels on water quality. Additionally, the absorption of inorganic nitrogen from suspended particles is influenced by several chemical bonding processes and the surface charge of sediment particles (Deng et al., 2022). All of these processes are affected by water temperature, pH, and bacterial activity (Deng et al., 2022).

The dynamics of inorganic nitrogen in river systems are further impacted by changes in flow rates. During times of low rainfall, reduced water flow leads to the settling of particulate matter at the riverbed, where inorganic nitrogen can accumulate in the sediments (Deng et al., 2022). When rainfall increases, the sediments are resuspended, reintroducing nitrogen into the water column and potentially triggering nitrification processes that exacerbate the growth of

undesirable aquatic plants and algae (Deng et al., 2022). This process can lead to oxygen depletion in the river, significantly reducing water quality and impacting aquatic life (Zeng et al., 2020).

To accurately assess the significance of inorganic nitrogen in urban rivers, it is crucial to measure average concentrations over a period of time (He et al., 2011). Specifically average summer inorganic nitrogen concentrations are considered the most reliable indicator of nitrogen's presence and its biological consequences within aquatic systems (He et al., 2011). For accurate results, it is suggested that weekly inorganic nitrogen concentrations which have been averaged over the period of at least 4 weeks should be compared to the Target Water Quality Ranges (TWQR) in order to achieve accurate results (Silberbauer, 1996). The nitrite concentration should not exceed 0.1 mg/L, as it can impact the oxygen transport in fish, kill sensitive invertebrate species and is toxic especially in low oxygen or low pH environments (Silberbauer, 1996). Preferably, the aquatic ecosystem should not have nitrite values higher than 0.05 mg/L, to protect more sensitive invertebrate and fish species (Silberbauer, 1996).

The South African Department of Water and Sanitation (DWS) provides a framework for understanding the ecological implications of varying inorganic nitrogen concentrations (Silberbauer, 1996). At concentrations below 0.5 mg/L, rivers experience oligotrophic conditions, characterized by low nutrient availability, moderate species diversity, and minimal aquatic plant or algal blooms. In contrast, when nitrogen concentrations range between 0.5 and 2.5 mg/L, rivers are classified as mesotrophic, supporting productive systems with high species diversity and moderate aquatic plant growth, with a low likelihood of toxic blue-green algae blooms. When inorganic nitrogen concentrations rise between 2.5 and 10 mg/L, eutrophic conditions prevail, leading to low species diversity, excessive plant growth, and the proliferation of harmful blue-green algae (Silberbauer, 1996). In cases where concentrations exceed 10 mg/L, hypertrophic conditions occur, with severe oxygen depletion, a dominance of toxic algae, and significant reductions in biodiversity (Silberbauer, 1996).

ii. Phosphate

Phosphorus enters urban rivers through various pathways, including wastewater discharge, fertilizer runoff and atmospheric depositions, both as dissolved and particulate chemical forms (Zhang et al., 2017). It can exist in numerous inorganic and organic compounds (Zhang et al., 2017). Elemental phosphorus does not occur naturally in the environment, but is instead found as polyphosphates, orthophosphates, metaphosphates, or pyrophosphates and organically

bound phosphates in natural waters. Soluble Reactive Phosphate (SRP), or orthophosphate, is the phosphorus immediately available for uptake by aquatic biota (Mahadevan et al., 2020). Other forms of phosphorus, such as organic or particulate phosphorus, are not immediately available, but can be converted to orthophosphate through natural processes like microbial decomposition and mineralisation (Mahadevan et al., 2020). Once in the river system, phosphorus can be assimilated by aquatic plants and microorganisms which contributes to the primary production (Zhang et al., 2017). As these aquatic plants decompose, they release phosphorus back into the water column in a process called mineralisation (Withers & Jarvie, 2008). Additionally, the phosphorus cycle is influenced by the transfer of phosphorus between the sediment and the water column (Withers & Jarvie, 2008). This exchange is affected by various chemical, physical and biological factors such as pH values, oxygen-dependent redox interactions, sorption processes, mineral-water equilibria and the activities of living organisms (Withers & Jarvie, 2008).

Phosphorus is an essential macronutrient which is accumulated by a variety of living organisms and plays a major role in the synthesis and storage of energy in cells (Zhang et al., 2017). In unimpacted waters, phosphorus is readily utilized by plants and converted into cellular structures by photosynthetic activity (Zhang et al., 2017). It is the primary nutrient controlling the degree of eutrophication in aquatic ecosystems since it is the limiting nutrient as it is least readily available in natural ecosystems (Sterner, 2008).

Sources of phosphorus can be naturally occurring and anthropogenically induced. Natural sources include the weathering and leaching of phosphate salts from rocks into surface waters and the decomposition of organic matter (Akhtar et al., 2021). The regional geography will significantly influence natural phosphorus concentrations, resulting in spatial variation of phosphorus concentrations, even within the same river system (Sterner, 2008). The phosphorus concentration is generally low in mountainous regions and then increases towards the lowland waters with sedimentary deposits due to mountains having less fertile sediments which contain higher concentrations of phosphorus and lowlands having more fertile sediments (Withers & Jarvie, 2008).

In South Africa, phosphorus concentrations in unimpacted waters are generally low, approximately 0,01 mg/l, because of active uptake up by aquatic vegetation (Adams et al., 2019). However, anthropogenic sources contribute significantly to elevated phosphorus levels, often increasing the phosphorus concentration to ranges between 0.1 to 1.0 mg/l (Adams et al.,

2019). Point-source discharges, such as domestic and industrial effluents, and diffuse sources, such as atmospheric precipitation, urban runoff and agricultural drainage where fertilizers have been used introduce excess phosphorus into aquatic systems (Zhang et al., 2017). The availability of phosphorus is further influenced by absorption onto inorganic colloids, organic compounds and particulate material. The availability, mobility and spatial distribution of phosphorus within a river is also influenced by the flow regime of the river, such as flow rate (Withers & Jarvie, 2008). During periods of lower flow rate in dry seasons, the settlement of particulate matter and biotic uptake result in the removal of phosphorus from the water column into the sediments with the river bed acting like a sink for phosphorus (Withers & Jarvie, 2008). Conversely, in rainy seasons, when flow is higher, phosphorus levels can be elevated by runoff from the land, as well as from resuspension and flushing of deposited material from the riverbed to the water column (Withers & Jarvie, 2008). This seasonal variability highlights the need for long-term monitoring to assess the phosphorus trends accurately.

When measuring phosphate concentrations of a river, occasional spikes in the inorganic phosphate above the TWQR are less significant than persistently high concentrations (Silberbauer, 1996). It is recommended that the phosphate levels in aquatic systems should not exceed 0.025 mg/L (Silberbauer, 1996). Single measurements for phosphorus are insufficient for identifying long-term trends, making average seasonal concentrations, particularly during Summer, more useful for predicting biological consequences (Silberbauer, 1996). The stimulation of the growth of aquatic plants is the most significant effect of high phosphorus concentrations in aquatic environments (Zhang et al., 2017). Phosphorus and nitrogen both limit plant growth with phosphorus being the more likely limiting agent in fresh waters, but this also depends on the form of phosphorus in the water, since not all forms are available for uptake by aquatic plants (Bae & Seo, 2021). Factors such as the availability of other nutrients, water temperature and light play a role in limiting plant growth (Bae & Seo, 2021). The growth of algae in particular has serious negative consequences because of its impact on water quality toxicity, eutrophication and can have long term impacts on the economic activities of local businesses, ruining the aesthetics and disrupting recreational activities (Bae & Seo, 2021). The growth of algae in particular has serious negative consequences because of its impact on water quality toxicity, resulting in eutrophication (Bae & Seo, 2021). The eutrophication can have long term impacts on the economic activities of local businesses along the water points, ruining the aesthetics and disrupting recreational activities (Bae & Seo, 2021).

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iii. Sulphate

Sulphate is a naturally occurring anion found in freshwater systems. It often originates from anthropogenic and geological sources (Zak et al., 2021). It is an essential component of biogeochemical cycles, however, excessive sulphate concentrations in rivers can have significant environmental and ecological impacts (Zak et al., 2021). Sulphate plays a role in acidification, salinity changes and interacts with other pollutants, making the regular monitoring of sulphate crucial in urban areas (Vehanen et al., 2022). Sulphate is quantified in three categories with low being less than 50 mg/l, elevated being between 50 and 200 mg/l and high being over 200 mg/l. Elevated sulphate levels are a good indication that there is pollution or contamination in the river from industrial discharges, like chemical manufacturing and textile or dyeing industrial, mining activities, like coal and gold mining, or wastewater effluents (Vehanen et al., 2022). Sulphate contributes to total dissolved solids (TDS) and salinity, which can impact drinking water quality and aquatic life (Walton, 1989). Sulphate is a component of Acid Mine Drainage (AMD) and can lead to severe water quality degradation in mining areas (Villiers & Mkwelo, 2009). High (>200 mg/l) sulphate levels can affect both human and aquatic life, making it essential for regulatory compliance by government officials (Vehanen et al., 2022).

Sulphate enters rivers through various natural and anthropogenic sources (Zak et al., 2021). The natural sources of sulphate in rivers are from geological weathering, since sulphate occurs naturally in soils and rocks, especially in sulphide rich minerals, such as pyrite (FeS_2) (Zak et al., 2021). When the rock is weathered down, the sulphate is released into aquatic environments (Zak et al., 2021). Sulphate also naturally enters rivers through atmospheric deposition such as precipitation, particularly in areas where natural sulphur emissions occur (Zak et al., 2021). In South Africa a large anthropogenic source of sulphate is from AMD released into rivers, especially from coal and gold mining, as the oxidation of sulphide minerals release sulphate (Villiers & Mkwelo, 2009). Paper mills, textile industries and chemical manufacturing plants discharge sulphate-rich wastewater into rivers, polluting the water. Agricultural and urban runoff increase the sulphate levels as they contain fertilizers, pesticides, livestock waste and household waste into rivers (Vehanen et al., 2022). Sewage and wastewater discharge are a large contributor to sulphate levels, as domestic wastewater contain detergents, human waste and other household products which are high in sulphate content (Moreno-Casas et al., 2009). The burning of fossil fuels can release sulphur dioxide (SO_2) into the atmosphere and forms

sulphuric acid (H_2SO_4) which is precipitated into rivers as acid rain, increasing the concentration of sulphate in rivers (Ward, 2009).

Sulphate is highly soluble and stays in the dissolved phase under most conditions, but is influenced by pH levels, redox conditions and interactions with metals (Zak et al., 2021). Under acidic conditions, often induced by the presence of AMD, sulphate ions remain highly mobile and increase water acidity, mobilizing metals (Villiers & Mkwelo, 2009). In low-oxygen environments, sulphate can be converted to hydrogen sulphide (H_2S) by sulphate-reducing bacteria, which is toxic to aquatic organisms. Sulphate can form metal-sulphate complexes when they bond to metals in the river, which will influence the solubility and mobility of metals such as lead, cadmium and iron in river systems (Nieva et al., 2018).

Sulphate has various effects on water chemistry, aquatic biota and invertebrates. (Zak et al., 2021). High levels of sulphate increase salinity in rivers, which can affect aquatic organisms which are more sensitive to change in ionic composition (Zak et al., 2021). Acid mine drainage causes high sulphate concentrations which increase the solubility of metals and degrades the water quality (Villiers & Mkwelo, 2009). High sulphate concentrations can affect fish and macroinvertebrate's ion balance and metabolic processes, causing them stress (Zak et al., 2021). Under anoxic conditions, sulphate is reduced to hydrogen sulphide (H_2S), a toxic gas which can suffocate aquatic organisms and disrupt gill function in fish (Zak et al., 2021). High sulphate concentrations can lead to the decline in sensitive aquatic invertebrate taxa such as mayflies, caddisflies and stoneflies, which are often used in SASS5 as bioindicators (Zak et al., 2021). Decline in sensitive species will cause a change in community composition, reducing the biodiversity of the ecosystem (Zak et al., 2021). Sulphate is not considered to be toxic to humans, however, excessive sulphate concentrations in drinking water can lead to gastrointestinal distress, corrode metal pipes and release metals like lead and copper which have more disastrous effects on humans, and can affect crop yields (Moreno-Casas et al., 2009).

The regulatory standards of sulphate in water are provided by the DWS in the South African Water Quality Guidelines (Silberbauer, 1996). In drinking water, sulphate must be kept below 200 – 400 mg/L, as higher concentrations can cause gastrointestinal effects (Silberbauer, 1996). There are no fixed standards in aquatic ecosystems, but levels above 250 mg/L can affect aquatic organisms (Silberbauer, 1996). Enforcement of these standards remains a challenge in mining areas as AMD results in sulphate concentrations far above these limits (Villiers & Mkwelo, 2009). The methods used for sulphate detection include various forms of analysis of

water samples. Ion Chromatography, spectrophotometry (UV-Vis) and Inductively Coupled Plasma Mass Spectrometry (ICP-MS) provide sensitive sulphate detection in water samples (Divjak & Goessler, 1999).

Sulphate is an important water quality parameter in river health assessments, particularly in urban areas affected by AMD, industrial discharges and wastewater discharges (Villiers & Mkwelo, 2009). Elevated sulphate levels have significant impacts on human health and aquatic ecosystems, making its monitoring essential.

d. Field measurements

When testing water quality, it is important to collect field measurements as it provides real time data of the health of the river, at low cost, with no laboratory testing needed. These results give the reader an understanding of the physiochemical conditions of the aquatic environment. These parameters are crucial as they directly influence the chemical behaviour of pollutants, the health of the river and the ecosystem dynamics. The pH is the measure of hydrogen ion concentration in the water, indicating whether the water is acidic, with a pH value below 7, alkaline or basic, with a pH value higher than 7, or neutral, with a pH value equal to 7 (Wanqing, 2001). Field measurements allow for real-time recording of results using a handheld and transportable probe. In urban rivers, pH is influenced by many different factors such as industrial effluents, urban runoff as well as natural processes, such as decomposition of organic matter, carbon dioxide absorption, algal and plant photosynthesis and weathering of geologic material (Anh et al., 2023). The pH of a river is important for various reasons. It affects the biological communities, with many aquatic organisms having specific pH ranges which they can tolerate with changes in pH leading to stress, toxicity or even mortality of sensitive species (Anh et al., 2023). The pH of water influences alkalinity, hardness and solubility of metals. Alkalinity in water sampling refers to the water's ability to neutralise acids and is an important indicator of water quality. It measures the concentration of bicarbonates (HCO_3^-), carbonates (CO_3^{2-}), and hydroxides (OH^-) in the water (Boyd, 2015). These compounds help maintain a stable pH level in the water by neutralising acids that would otherwise lower the pH, acting as a buffer. This refers to the water body's buffering capacity, which is especially important for protecting aquatic life, since many aquatic organisms have a narrow range of pH (between 6.0 and 9.0) in which they are able to survive (Boyd, 2015). The main sources of alkalinity in

natural waters are the bicarbonates, carbonates and hydroxides (Boyd et al., 2016). Bicarbonates come from the dissolution of carbon dioxide (CO_2) in water, carbonates come from carbonate rocks such as limestone and other mineral formations, such as gypsum, and hydroxides are mostly found in basic waters with high pH levels (Benavente et al., 2024; Boyd et al., 2016). Alkalinity is typically measured in mg/L of calcium carbonate ($CaCO_3$), the standard reference compound for alkalinity. Hardness in rivers refers to the concentration of dissolved minerals, primarily calcium (Ca^{2+}) and magnesium (Mg^{2+}) ions in the water (Boyd et al., 2016). These minerals are typically derived from weathering of rocks caused by the water flow carrying particles which scrape the rocks, weathering them down (Boyd et al., 2016). The rocks which especially cause hardness are limestone, dolomite and other calcium-rich minerals (Boyd et al., 2016). Hardness influences both the ecology of aquatic systems and the useability of the river for human purposes since it affects taste, agricultural use and water treatment (Thompson et al., 2012). Hard water has high concentrations (between 121 to 180 mg/L) of calcium and magnesium ions while soft water has lower concentrations (between 0 to 60 mg/L) and is measured as mg/L of calcium carbonate (Thompson et al., 2012). Solubility of metals refers to the ability of metals to dissolve in water, forming metal ions or complexes (Martínez & Motto, 2000). This is a key factor in determining their concentration and potential impact on water quality, human health and aquatic biodiversity (Martínez & Motto, 2000). Understanding the solubility of metals in water assists in gaining an understanding of their behaviour in water bodies, particularly in urban rivers as they are environments influenced by various forms of pollution (Martínez & Motto, 2000). Various factors affect the pH in urban rivers such as different pollution sources, such as point-source pollution like industrial discharges and sewage leaks, and non-point-source pollution such as runoff from roads, construction sites and urban areas, which potentially introduce acidic or alkaline substances, altering the pH levels (Göbel et al., 2007). Point source pollution is pollution that comes from one specific source at a point along the river while non-point pollution is pollution that enters the river at various points along the river. Water temperature can influence pH by increasing the rate of chemical reactions and affecting solubility. This is often due to thermal pollution that occurs in urban areas (Quevedo et al., 2018). Extreme pH levels, both acidic and alkaline, can cause the release of toxic substances, such as metals, and can also disrupt the metabolic functions of various organisms (Quevedo et al., 2018). Bacterial and algal growth are affected by altering pH levels. In acidic waters, there may be an increase in algal blooms, degrading water quality (Wallace et al., 2016). There are various methods for measuring pH levels in rivers, such as pH meters, indicator papers and spectrophotometry, but there are various challenges when testing pH such as

changes in pH within a sample shortly after sampling due to temperature change or biological activity, varying pH levels within the same position over the course of a day due to urban and industrial runoff and wastewater sewage discharge (Wallace et al., 2016). The levels of pH in an urban river change seasonally and temporally due to factors such as increased rainfall in summer causing more urban runoff, affecting the pH of the river (Xu et al., 2019). Most freshwater systems in South Africa are approximately neutral since they are relatively well buffered and have pH ranges between 6 and 8, but variations in geology can have implications for the river pH (Omarjee et al., 2021). Rivers that are dominated by very dilute sodium chloride are poorly buffered as they contain virtually no bicarbonates or carbonate (Omarjee et al., 2021). If catchments contain certain types of vegetation, such as Fynbos, pH can drop to as low as 3.9 due to the influence of organic acids found in the plants (Omarjee et al., 2021).

Temperature has a substantial impact on the physical properties of water and aquatic chemistry. When temperature increases, the surface tension, viscosity, ionization constant, compressibility and latent heat of vaporization decrease, but it will cause thermal conductivity and vapour pressure to increase (Li et al., 2022). The solubility of H_2, N_2, O_2 and CO_2 will decrease when temperature increases (Kolev, 2012). Temperature affects the rates of chemical reactions and the metabolic rates of organisms; therefore, temperature is one of the major factors influencing the distribution of aquatic organisms (Kazmi et al., 2022). Naturally occurring variations in temperature within aquatic environments throughout the day and across different seasons serve as cues for migration and spawning activities (Kazmi et al., 2022). Anthropogenically altered water temperatures can impact either individual organisms or entire aquatic communities (Kazmi et al., 2022). The temperatures of inland waters in South Africa generally range from 5˚C – 30˚C but are dependent on various features of the catchment area including the latitude of the river, altitude, rate of flow, ground water contributions, climatic factors, as well as structural characteristics of the river and catchment area (Silberbauer, 1996). Surface water temperatures of rivers also change along the length of the river, and where there is deeper water, layers of water closer to the surface are warmer than layers at a deeper depth (Li et al., 2022). The temperatures of rivers can also be altered by anthropogenic activities such as industrial effluent discharge, heated irrigation runoff, removal of riparian vegetation cover resulting in increased solar radiation of the water and inter basin transfers (Poole & Berman, 2001). Changes in the temperature of aqueous environments have various impacts on these environments. Higher temperatures reduce the solubility of dissolved oxygen which decreases the concentration of and availability of dissolved oxygen for aquatic organisms (Kazmi et al.,

2022). Changes in temperature have significant impacts on aquatic organisms, such as the invertebrates found in an urban river (Munyika et al., 2014). Aquatic organisms have upper and lower thermal tolerance limits, optimal temperatures that inhibit growth, preferred temperature ranges and thermal gradients as well as temperature ranges for egg incubation, migration and spawning (Kazmi et al., 2022). All organisms that are associated with freshwater systems, other than birds and mammals, are poikilothermic meaning that they are unable to self-regulate their body temperature and are therefore reliant on the ambient water temperatures, making them susceptible to changes in water temperatures (Fenoglio et al., 2010). Rapid changes in the water temperature can be detrimental to the aquatic organisms, resulting in mass mortality (Kazmi et al., 2022). Changes in temperature that are not rapid or severe can have sub-lethal impacts on aquatic organisms or change the existing aquatic community (Kazmi et al., 2022). The type and number of species change when population shifts occur because of the disappearance of species which are more sensitive to changes in temperature (Kazmi et al., 2022). These species are then replaced by more heat-tolerant species which increase in number, superseding the original species that is found in the water and reducing the biodiversity of the aquatic environment (Brans et al., 2017).

When high organic loads enter the aquatic system, microbial activity increases which causes oxygen concentrations to decline as a result of metabolic activity (Meyer-Reil & Köster, 2000). When the water temperature increases, metabolic rates such as respiration increase causing more oxygen demand from aquatic organisms (Grzybkowska et al., 2017). This increased oxygen demand results in the dissolved oxygen supply decreasing (Grzybkowska et al., 2017). The release of deep waters such as dams causes unnaturally low temperatures to be present in the aquatic system which can suppress the spawning activities of aquatic organisms, which will negatively impact the ecosystem (Grzybkowska et al., 2017). As water temperature increases, the toxicity of most substances and the vulnerability of organisms to toxins increases (Xin & Kinouchi, 2013). Water temperature is measured at each site using a thermometer, of which there are various types, and is expressed in ˚C (Xin & Kinouchi, 2013). It is important to measure aquatic temperatures over various seasons to identify outliers and determine the cause of the temperature change (Xin & Kinouchi, 2013). Over a study period it is important to measure the water temperature at the same time since water temperature varies throughout the day, making it difficult to find consistent results. The effects of temperature change on aquatic organisms will depend on the timing, duration and extent of the changes (Xin & Kinouchi, 2013).

Small changes to temperature over a long period of time can have substantial impacts on the aquatic systems such as changes in community composition, since different organisms have different optimal temperatures in which they thrive. Large and sudden changes in temperature can be lethal to aquatic organisms (Meyer-Reil & Köster, 2000).

Dissolved Oxygen (DO) is a useful measure of the health of aquatic ecosystems since the maintenance of enough DO is necessary for the survival of aquatic biota; all aerobic organisms require it for respiration (Prasad et al., 2014). Oxygen in gaseous form from the atmosphere is dissolved into aqueous systems and is also generated during photosynthesis by aquatic plants and phytoplankton (Prasad et al., 2014). Solubility is the ability of water to dissolve oxygen and is inversely proportional to temperature (Tromans, 1998). It is also a function of salinity, and pressure, as well as other chemical and physical factors that are site-specific (Tromans, 1998). Similarly to pH, DO has a 24-hour cycle which is associated with the 24-hour photosynthesis cycle and respiration by aquatic biota (Howarth & Michaels, 2000). At night the concentrations decline, reaching a minimum before dawn and a maximum by mid-afternoon (Howarth & Michaels, 2000). There are seasonal variations related to DO which arise as a result from changes in temperature and biological activity (Howarth & Michaels, 2000). There are various factors of an aqueous ecosystem which can cause a lowering in the concentration of DO. Sediments which are anoxic, meaning that there is no presence of oxygen which are typically found in deep, stagnant or nutrient rich water bodies, are resuspended due to floods or dredging occurring (Quantin et al., 2005). In deep lakes or reservoirs, turnover, which is when sediments and water from the bottom of the river are brought to the top through the cycling of water across different seasons or because of the mixing of water in rivers, can result in highly deoxygenated water from the bottom of the water body coming to the surface (Quantin et al., 2005). The presence of oxidizable organic matter can either come from natural processes or in wastewater discharges can impact the concentration of DO in surface waters (Lishman et al., 2000). The amount of suspended material in water affects the saturation concentration of DO. This can be done either chemically through the oxygen scavenging attributes of suspended particles or through the reduction of the volume of water available for solution (Best et al., 2007). Natural diffusion of oxygen from the atmosphere into water can increase the concentrations of DO in water bodies. Diffusion continues until the saturation concentration is reached (Best et al., 2007). If turbulence occurs in the water, for example, if there are rapids in a river, the rate of dissolution of oxygen into the water will increase as air from the atmosphere is mixed into the water by a process known as aeration (Best et al., 2007).

Aquatic environments that experience high temperatures and low levels of DO can cause high levels of stress to the aquatic organisms in these environments, especially when there is also presence of toxic substances (Dallas, 2008). Under these conditions, it is observed that there is an increase in toxicity of lead, zinc, cyanide, copper, sulphide and ammonia (Hellawell, 1988). Due to the changes in levels of DO throughout the day, to achieve an accurate reading, it is best to measure the samples at the same time of day each time sampling is done (Mihu-Pintilie et al., 2014). It is also important to measure the DO at the same depth across rivers as the results can vary across different depths (Mihu-Pintilie et al., 2014). DO can be measured as either milligrams per litre (mg/l) or as a percentage of the saturation concentration at the time that sampling is done (Silberbauer, 1996). The significance of DO depletion in aquatic environments depends on the duration, frequency and timing of depletion. When exposure to less than 80% saturation concentration is experienced, harmful effects can be experienced (Silberbauer, 1996). Occasional short-term depletion of oxygen is less significant, but if this exposure is repeated, physiological and behavioural stress will occur (Prasad et al., 2014). If rapid changes in dissolved oxygen depletion and saturation take place, adverse effects on aquatic biota are increased significantly (Howarth & Michaels, 2000). Anoxic or hypoxic conditions can be lethal to aquatic biota within minutes to hours (Howarth & Michaels, 2000). The sensitivity of different species, such as invertebrates and fish, to changes in DO concentrations will depend on the life stages and behavioural changes that occur, such as feeding and reproduction (Franklin, 2014). Juvenile organisms tend to be more sensitive to physiological stress due to depleting DO with secondary effects taking place such as increased vulnerability to predation and disease (Franklin, 2014). Often different species will avoid areas which have depleted DO concentrations if possible (Franklin, 2014). High levels of photosynthesis resulting from nutrient enrichment in the river can lead to supersaturation of dissolved oxygen (DO) in aquatic environments. However, the growth of harmful blue-green algae is not inhibited by this condition, as these organisms exhibit a high tolerance to oxygen supersaturation (Sand-Jensen et al., 2022). The ability of organisms to reverse the toxic effects of oxygen depletion is dependent on the duration, timing and frequency of oxygen depletion (Diaz & Breitburg, 2009). If oxygen depletion is short term, adult and less sensitive organisms can rapidly reverse the physiological stress effects (Diaz & Breitburg, 2009). When DO concentrations are less than 50% saturation over a prolonged period, significant changes in aquatic communities can be observed as more tolerant species will be favoured and less tolerant species will not cope with the experienced stress (Diaz & Breitburg, 2009). The South African DWS has a set of criteria for ensuring the protection of aquatic biota by determining target

quality ranges for dissolved oxygen concentrations in aquatic environments (Silberbauer, 1996). Concentrations within the range of 5-8 mg/L will ensure that life is protected at all stages of life for most southern African aquatic biota which are endemic to or have adapted to warm aerobic water as habitats (Silberbauer, 1996). The minimum allowable concentrations are associated with causing acute toxic effects on aquatic biota. Any measurement less than 60% saturation, measured over a 7 day mean minimum, is considered sub-lethal and any measurement less than 40% saturation, measured over 1 day minimum, is considered lethal and is very likely to have toxic effects on aquatic biota (Silberbauer, 1996). The South African DWS limits apply to any stakeholders involved in water resource management, especially those who regulate water quality such as municipal authorities, industrial and commercial sectors, agricultural sectors, environmental protection agencies and even the general public who use groundwater or lakes (Silberbauer, 1996).

e. *Escherichia coli* and Coliforms

Urban rivers are increasingly subjected to contamination by multiple sources including stormwater runoff, industrial discharge and sewage leaks (Garcia-Armisen & Servais, 2007). *Escherichia col*i (*E. coli*) and coliforms are some of the key indicators of microbial pollution in aquatic environments, but originate from different sources and behave differently in the environment (Some et al., 2021). Total coliforms include a broader group of bacteria that can come from soil, decaying plant material and naturally occur in aquatic environments, while *E. coli* is a specific indicator of contamination originating from human or animal intestines (Neill, 2004). High rainfall and human activity can disturb and resuspend sediments which act as a sink trapping coliforms and *E. coli* (Foppen & Schijven, 2006). When resuspension occurs, the *E. coli* that is resuspended dies off faster than other coliforms, resulting in higher coliform readings than *E. coli* readings (Foppen & Schijven, 2006). The sunlight will also affect the *E. coli* more than other coliforms, resulting in higher coliform concentrations than *E. coli* concentrations (An et al., 2002).

The *E. coli* concentration in rivers serves as proxies for faecal contamination and potential pathogenic risk (Neill et al., 2018). The presence of this bacteria in urban rivers has significant implications for human health and aquatic ecosystems (Neill et al., 2018). Measuring *E. coli* and coliforms is important because coliforms bacteria, specifically *E. coli*, originate from the intestines of warm-blooded animals, including humans. The presence of *E. coli* in water bodies strongly indicates faecal contamination (Some et al., 2021). Monitoring these parameters is crucial for assessing public health risks, since high *E. coli* counts suggests the presence of

pathogenic bacteria, viruses and protozoa (such as *Salmonella enterica*), increasing the risk of waterborne diseases (Toro et al., 2022). The South African Water Quality Guidelines specify limits for *E. coli* in surface water to protect human health, especially in water bodies used for recreation (Silberbauer, 1996). It can be used to trace possible sources of pollution since high levels can indicate leaking sewer infrastructure, illegal sewage connections, agricultural runoff or informal settlements which lack proper sanitation infrastructure (Some et al., 2021).

There are multiple sources of *E. coli* and coliforms in urban rivers, primarily stemming from anthropogenic activities (Neill et al., 2018). Sewage discharge from overloaded wastewater treatment plants and broken sewer lines release untreated and partially treated wastewater contributing significantly to pollution (Schilling et al., 2009). Industrial effluents can contain organic waste that contributes to bacterial growth. Non-point sources include urban stormwater runoff washing contaminants such as pet waste and overflowing toilets and septic tanks into rivers (Schilling et al., 2009). Agricultural runoff containing manure from livestock farms and fertilizers containing organic material can introduce coliforms into water bodies (Schilling et al., 2009). Informal settlements which lack proper sanitation infrastructure leads to direct waste disposal into urban rivers (Schilling et al., 2009). Due to lack of public sanitation infrastructure in urban areas, people without private access to sanitation services defecate in public areas, resulting in E. coli flow into urban rivers from urban runoff when it rains (Verbyla et al., 2021). High *E. coli* and coliform concentrations are often associated with high nutrient loads as a consequence of sediment resuspension which is contaminated with faecal bacteria and nutrient contamination (Shelton et al., 2014). Nutrient loading and high *E. coli* and total coliform concentrations are often linked as a consequence of stormwater runoff washing pollutants into the river from leaking infrastructure (Wang et al., 2018). It is highly likely that this nutrient loading is a consequence of stormwater runoff as they found similar values at in the rainfall seasons with spike of *E. coli* and nutrients like nitrates and nitrites in a study done by Wang et al, (2018) (Wang et al., 2018). Wang et al. (2018) found a very strong correlation between *E. coli* and total coliforms (Wang et al., 2018).

The regulatory standards for *E. coli* have been established to ensure safe water quality. The DWS has set out water quality guidelines to limit the amount of *E. coli* in water for domestic use and for recreational waters (Silberbauer, 1996). Water used for domestic use must be less than 1 MPN/100mL and less than or equal to 130 MPN/100mL with MPN representing the Most Probable Number of *E. coli* bacteria in a sample of water 100mL in size (Silberbauer, 1996). Unlike direct colony-forming unit (CFU) counts from plate methods, MPN provides a

probability-based estimate of bacterial concentration. It is commonly used in water quality testing, especially with the Colilert and other presence-absence tests (Avery et al., 2008). The method uses a statistical based approach based on dilutions and presence / absence results in multiple cells.

Excessive *E. coli* and coliform concentration can have various ecological and environmental impacts. High bacterial loads can disrupt natural microbial diversity, affecting nutrient cycling and decomposition processes (Wu et al., 2021). Faecal contamination leads to high organic matter loads which stimulate microbial activity and reduces the DO levels (Hansen et al., 2008). This has the potential to cause gill infections in fish, leading to fish mortality and reducing invertebrate communities as well as the biodiversity of the river (Hansen et al., 2008). Excessive *E. coli* contamination can have various effects on humans, such as increasing the risk of waterborne diseases like gastrointestinal infections, respiratory illnesses and skin and eye infections (Neill et al., 2018). There are also economic implications since contaminated rivers increase the treatment cost for potable water supply (Neill et al., 2018).

There are various field sampling and culture-based methods used for detecting *E. coli*, such as Membrane Filtration Method which counts the colonies formed after the bacteria are cultured (Avery et al., 2008). Various Molecular and Rapid detection methods also exist for determining the presence of *E. coli* in rivers (Buckalew et al., 2006). Enzyme-Substrate Tests, such as Colilert tests, are used to detect coliforms based on enzyme activity which provides faster results than culture methods (Buckalew et al., 2006). Strategies can be put in place to manage coliform contamination in urban rivers (Neill et al., 2018). Improving sanitation infrastructure by upgrading wastewater treatment plants to ensure effective removal of bacterial contaminants, regular maintenance of sewer networks and providing proper sanitation infrastructure to informal settlements will reduce the amount of coliform contamination in urban rivers (Neill et al., 2018). Stormwater management and green infrastructure can be developed to reduce bacterial loads (Neill et al., 2018). Stricter pollution regulations and community engagement are needed to improve the water quality of urban rivers (Neill et al., 2018).

E. coli and coliforms are key parameters when measuring the health of a river since they indicate biological contamination from faecal matter, which increases waterborne disease risk and poses a threat to human health and aquatic ecosystems. Effective monitoring, improved

sanitation infrastructure and sustainable urban planning are required to mitigate bacterial pollution (Neill et al., 2018).

f. Macroinvertebrates in urban rivers

Macroinvertebrates are insects which do not possess a backbone and are able to be seen without the use of a microscope, and are an important bioindicator of river health (Baker & Greenfield, 2019). They can be either aquatic or terrestrial with the aquatic organisms often being the larval or nymphal forms of terrestrial organisms. Due to the complexity of rivers, the community complexity of aquatic macroinvertebrates has high spatial variability across a river's course, especially in semi-arid regions like South Africa (Baker & Greenfield, 2019). The diversity and abundance or richness of aquatic organisms are determined by the abiotic components of the river such as geomorphology, physio-chemical elements and hydrology (Munyika et al., 2014). A study of invertebrate assemblages can be useful because the analysis of chemical constituents of water samples is deemed insufficient for the complete assessment of aquatic health since it does not take into account the time integrated response to changes that occur within the system (Baker & Greenfield, 2019). Changes such as habitat alterations and destruction result in changes to the natural flow of the river (Baker & Greenfield, 2019). The macroinvertebrates are useful when studying the long-term health of a river because they are consistently present over time. They represent a time-based aggregate sample. Water pollution is often pulse based, meaning that inflows of pollution appear and disappear, thus, unless the pollution spike is caught on the day of occurrence, it is not identified at all (Dahms et al., 2017). The organisms in the river, however, are always there and identifiable, making changes in river health visible as animal assemblages increase when the river is healthy and decrease when unhealthy (Dahms et al., 2017). In many ways, macroinvertebrates act as passive samplers, since they remain in the aquatic environment over time, accumulating chemical pollutants from the environment (Dahms et al., 2017). Biomonitoring of aquatic communities is valuable for determining aquatic health since aquatic organisms integrate interactions and alterations of both physical and chemical environments as well display the cumulative effects of pollution (Munyika et al., 2014). The aquatic macroinvertebrates that are often used when doing a bioassessment of a riverine environment inhabit vegetation that is submerged in the water and the varying substrate in the riverbed (Baker & Greenfield, 2019). The invertebrate assemblages include insect larvae, molluscs, worms and crustaceans. One of the primary motivators for using these macroinvertebrates is that they are short-term indicators of water quality since they are directly

influenced by their surrounding environment (Baker & Greenfield, 2019). In South Africa, the identification of macroinvertebrate taxa is difficult, especially at lower taxonomical levels, since it is significantly underrepresented and poorly documented (Baker & Greenfield, 2019). The limited number of studies which focus on riverine health using macroinvertebrates as bioindicators, such as Dahms et al. (2017) and Sibiya et al. (2025) show that Southern Africa is an underrepresented region in scientific literature of aquatic ecology and requires more comprehensive diversity assessments (Baker & Greenfield, 2019).

i. South African Scoring System version 5 (SASS5)

South African Scoring System Version 5 (SASS5) is a bioassessment tool used to determine the ecological health of a freshwater system (Lebepe et al., 2021). It is commonly used in South Africa to determine the water quality in rivers, streams and other freshwater ecosystems and it is based on the diversity and abundance of macroinvertebrate species (Lebepe et al., 2021). The process of SASS5 sampling involves various steps which focus on the collection, identification and scoring of aquatic macroinvertebrate species from specific sites (Odume & Mgaba, 2016). The method used for SASS5 involves a kick-net which is used to collect the invertebrates from the sediment by disturbing the sediment or substrate and placing the net immediately downstream from the disturbance. A variety of habitats are covered, and the process should be performed for 10-15 minutes (Odume & Mgaba, 2016). The sample is then sorted in a tray and the macroinvertebrates are identified before being released back into the river (Odume & Mgaba, 2016).

ii. SASS5 scoring technique

Each macroinvertebrate taxon has an assigned score based on its tolerance to pollution (De et al., 2004). The scores range from 1 (which is assigned to taxon which are highly pollution tolerant) to 10 (which is assigned to taxon which are highly sensitive to pollution) (Odume & Mgaba, 2016). The Average Score Per Taxon (ASPT) score is determined by dividing the SASS5 score by the number of species identified in the sample (Bere & Nyamupingidza, 2014). The ASPT score is used to determine the water quality based on a predetermined rating scale (Bere & Nyamupingidza, 2014). From the SASS5 results, the water quality classification is determined and can be categorised as either “Excellent,” “Good,” “Fair,” or “Poor” (Dallas, 2007). The importance of SASS5 cannot be understated as it is a useful measure for ecological health, especially when used alongside supporting data such as water samples (Dallas, 2007). It provides insight into the biodiversity and abundance of aquatic life in freshwater systems, which reflects the overall condition of the water body (Munyika et al., 2014). It is used as a

method of pollution monitoring by focusing on the sensitivity of the macroinvertebrates, allowing for the detection of changes in water quality before more obvious identifiers such as fish kills (Munyika et al., 2014). SASS5 sampling allows for the long-term, repeated monitoring of water bodies, which helps identify the trends in water quality over time (Munyika et al., 2014). SASS5 can assist with the conservation and management of freshwater systems by guiding conservation efforts, especially when paired with land use practices, restoration initiatives and pollution control (Odume & Mgaba, 2016).

iii. MINISASS

MINISASS refers to the simplified, user-friendly version of the SASS5 method and is designed for smaller freshwater bodies and limited sampling scenarios (Graham et al., 2004). It is used when resources are limited or a full scale SASS5 assessment is not feasible. It is used for rapid assessments but involves shorter sampling times and less sampling effort (Graham et al., 2004). It has the same scoring method as SASS5, but with a limited key indicator group (Graham et al., 2004). It is a cost effective and time-saving method which has wider applications since it is more useful for citizen science projects and initial screening (Graham et al., 2004). MINISASS has the same implications for conservation efforts and management of freshwater systems as SASS5, such as identifying invertebrate health, identifying areas where biodiversity is at risk and finding solutions to biodiversity problems (Graham et al., 2004).

g. Metal contamination in urban rivers

The assessment of the health of aquatic ecosystems, through water and sediment sampling, is important for various purposes, such as ensuring the water is safe for human interaction or consumption in some cases by detecting signs of pollution and contamination to inform water management and ensure biodiversity and ecosystem health (Dahms et al., 2017). One of the more dangerous forms of contamination that can occur in urban rivers is metal contamination for both humans and ecosystems (Ismail et al., 2013). Metal contamination is more prevalent in urban areas due to human activities such as industrial processes, vehicle emissions, waste disposal and construction (Ismail et al., 2013).

According to Sanders (1997), metals occur in four main reservoirs in an aquatic system including suspended particles, surface water, water between sediment particles called interstitial water and sediment (Dahms et al., 2017). The aquatic environment is a crucial part of the ecosystem, with wetland sediment studies determining the concentration of the metal

contamination being vital to understanding ecosystem health (Dahms et al., 2017). Sediments act as natural sinks for pollution, assisting in the aquatic environment health maintenance by trapping, binding and removing toxic elements from the water system (Dahms et al., 2017). Sediments and metals have different interactions with sediment trapping the metals physically, especially in wetlands where the water is slowed down which allows impurities to settle out (Dahms et al., 2017). Contaminant particles like metals can also interact with sediment by chemically binding to sediment particles, resulting in metals not being detected in water metal analysis (Dahms et al., 2017). Metals in sediments can be either anthropogenically produced or they can be naturally occurring. The study of metals in sediments in important since these metals can be resuspended when there are rainfall events, potentially releasing high levels of toxic metals into the aquatic system (Greenfield et al., 2007). The study of metals in sediments give a time integrated assessment of pollution in an aquatic system since metals accumulate in sediments over time and are released under certain environmental conditions caused by anthropogenic and natural factors (Wang et al., 2007).

Toxic metals can accumulate in benthic organisms (fish, Oligochaeta, molluscs and arthropods) which live in the bottom over rivers, lakes or stream, and can have effects on the entire food web (Wang et al., 2007). Dissolved metals do not degrade, they remain mobile in the water column until they bind to sediment/clay particles and settle to the bottom of the water body (Zheng et al., 2013). This is why they accumulate in sediments and can be released, even after long periods of time (Dahms et al., 2017). Some metals are toxic in low concentrations, but others are essential life sustaining elements in trace amounts (Dahms et al., 2017). Certain metals are essential to life forms but changes in environmental conditions, such as changes in pH, can cause them to become toxic (Dahms et al., 2017). Sediment analysis for metals reveals sites which have significant metal contamination and can help with identifying the sources (Dahms et al., 2017). In Dahms et al., (2017), four risk assessment indices were used to determine the potential risk that sediments pose to the ecology of the Nyl River System. The four risk assessments were Contamination Factor, Pollution Load Index, Geoaccumulation index (Igeo) and Enrichment Factor (Dahms et al., 2017). These methods for measuring the amount of pollution and contamination in rivers are best used as a combination, rather than one method being better than another. This is because each of the indices (CF, PLI, Igeo and EF) measure a slightly different as aspect of contamination, and together they provide a more complete and reliable picture of the degree, sources and ecological risk of metal pollution. While CF quantifies individual metal intensity, the PLI summarises overall site pollution, the

Igeo contextualises contamination relative to natural background levels, and the EF distinguishes anthropogenic inputs from natural sources (Dahms et al., 2017). Using all four methods together provides a more comprehensive assessment of metal pollution in complex urban river systems (Dahms et al., 2017).

i. Manganese (Mn)

Manganese is an essential nutrient for aquatic plants and animals (Barboza et al., 2016). It is important in the assimilation process of nitrate and an essential catalyst of many enzyme systems in animals, plants and bacteria (Barboza et al., 2016). When manganese is not sufficiently present in aquatic systems, the ability of plants to photosynthesise will decrease and yellowing of the leaves will occur (chlorosis) and leaves may fail to properly develop (Marks et al., 2017). A manganese deficiency in vertebrates can lead to skeletal deformities and reduced ability to reproduce (Keen et al., 2000). High manganese concentrations (above 0,18 mg/L according to the DWS target water quality guidelines) are toxic and can lead to disturbances in various metabolic pathways, specifically disturbances of the central nervous system caused by the inhibition of the formation of the dopamine neurotransmitter (Silberbauer, 1996). Manganese is the 8th most abundant metal in nature but does not naturally occur as a metal in aquatic ecosystems; rather it is found in various salts and minerals and is frequently associated with iron compounds (Förstner & Wittmann, 2012). In aquatic environments it may exist in the soluble form manganous (Mn^{2+}) but is readily oxidised to insoluble manganic (Mn^{4+}) form. Sedimentary rocks, metamorphic rocks, soils and sediments are natural sources of manganese (Förstner & Wittmann, 2012). Manganese is also prevalent from many anthropogenic activities and can often be found in industrial discharge, polluting aquatic environments (Marks et al., 2017).

Various industries make use of manganese, its compounds and its alloys in their products or processes, such as the steel industry in the production of dry cell batteries, the fertilizer industry using manganese as a micro-nutrient additive and the chemical industry in paints, glass, dyes, matches and ceramics (Marks et al., 2017). Acid mine drainage is a large contributor to the release of manganese into aquatic environments (Neculita & Rosa, 2019). Manganese is often found in association with iron and is similar to iron in its chemical behaviour (Neculita & Rosa, 2019). The dissolved manganese concentration is influenced by various factors associated with the aquatic body being sampled, such as the amount of organic matter present in the river, pH, changes in redox potential and the dissolved oxygen concentration (Kang et al., 2019). A large proportion of manganese found in natural waters is present in suspended and absorbed forms

(Papina et al., 2017). The suspended manganese particles are particles which are mobile and interact with other particles, sometimes forming clusters, but always in motion (Flores-Guia et al., 2021). Absorbed manganese particles are attached to surfaces such as rocks, sediments aquatic plants and pollution forming bonds with the substrate (Flores-Guia et al., 2021). Increased carbonate decreases manganese solubility and reduce the oxidation rate of manganese.

Manganese is commonly measured using the Atomic Absorption Flame Spectrometry (AAFS) or Inductively Coupled Plasma Optical Emission Spectrometry (ICP-OES). Proper measures must be taken at all stages of sample collection, sample storage, laboratory analysis and data interpretation to ensure accurate results. The dissolved manganese measurements should be within the Target Quality Water Range (TQWR) (below 0.28 mg/l) and below the Chronic Effect Value (CEV) (0.37 mg/l) to ensure the protection of aquatic ecosystems (Silberbauer, 1996). There are no formalised national sediment quality guidelines in South Africa for manganese. The recommended manganese concentration often depends on the background concentrations for the area. Researchers often use similar studies to compare their results to, often relying on background scientific research into developing these standards, such as identifying the concentration that manganese will have negative impacts on the environment (Ogbeibu et al., 2014). The United States Environmental Protection Agency (USEPA) set out a guideline that recommends the manganese in sediments in aquatic environments should remain below 300 mg/kg to minimise ecological risks (Ogbeibu et al., 2014).

ii. Lead (Pb)

The significance of measuring the concentrations of lead in urban rivers is that it has various impacts on human and ecological health. Lead is defined as potentially hazardous to all forms of life by USEPA and is listed in the top ten hazardous materials due to its toxic and non-biodegradable nature which is relatively accessible to aquatic organisms (Ahmad et al., 2022). Some serious health effects are associated with Pb which include kidney failure, cardiovascular disease, liver cirrhosis, mental retardation, heart problems and cancer of the liver and kidney, among many others (Ahmad et al., 2022). Lead is utilised in many applications, such as in water pipes, electroplating, chemical waste, ammunition, paint and lead ion batteries (Ahmad et al., 2022). Lead is a non-essential heavy metal that enters urban rivers through various pathways, many of which are related to anthropogenic activity and increased urbanisation, such as industrial discharge and urban runoff (Sharma et al., 2023). Stormwater runoff is a large contributor to increased lead concentrations in urban rivers since, in older buildings, lead based

paint was used and can degrade over time, especially when buildings are being demolished or renovated (Wijesiri et al., 2019). Historically, vehicle emissions were a major source of lead pollution as leaded petrol was used, distributing it into the environment through emissions and leaks (Ayrault et al., 2012). Although this type of petrol has been phased out, remnants of lead can still be found in urban soils, especially along busy roads where rainwater can wash the lead into the river (Ayrault et al., 2012). Tire wear and road dust are also sources of lead pollution that are introduced into the river by stormwater runoff (Ayrault et al., 2012). Lead can also enter urban rivers through sewage leaks and industrial activities, since certain industries, such as battery manufacturing, metal processing or construction, can release lead into the air, water and soil (Ayrault et al., 2012). This can be directly deposited into the river through wastewater discharge, or indirectly through runoff (Ayrault et al., 2012).

In South Africa, mining activities, particularly in regions like Gauteng and Mpumalanga, significantly contribute to lead contamination in rivers (Ochieng et al., 2010). Once in the water, lead binds to sediments and organic matter, and forms stable complexes that reduce its immediate bioavailability (Sharma et al., 2009). This interaction is influenced by factors like sediment grain size and organic content, with fine-grained sediment and high organic matter increasing lead absorption (Zhao et al., 2010). Some of the lead ions will be freely floating around in the water while others will join with other organic matter and surfaces (Atangana & Oberholster, 2021). Lead exists in aquatic environments in both soluble and particulate forms, with the chemical form or speciation (soluble or particulate) affecting the bioavailability of the lead (Atangana & Oberholster, 2021). In South African rivers systems like the Olifants River, elevated lead concentrations have been reported and attributed to agricultural and industrial runoff and mining activities (Atangana & Oberholster, 2021).

Lead ions (Pb^{2+}) are highly toxic, while lead which is bound to particulates is less bioavailable, but can become remobilised under environmental conditions, such as heavy flow rates from increased rainfall (Jackson et al., 2005). This is because the sediments of a river act as a sink, collecting the free lead ions and storing them, reducing the amount of lead ions available in the water (Jackson et al., 2005). The presence of dissolved organic matter can further alter lead speciation, which will either increase or decrease its bioavailability (Jackson et al., 2005). Low-molecular-weight compounds, such as certain acids and chelating agents, keep the lead in its dissolved form which are more bioavailable, enhancing the uptake by aquatic organisms (Shahid et al., 2012). High-molecular-weight organic compounds, such as humic and fulvic acids, can form stable complexes with lead, reducing its free ion concentration and reducing

its bioavailability (Shahid et al., 2012). Since metallic lead and lead minerals such as sulphates, sulphides, oxides, carbonates and hydroxides are nearly insoluble, levels of dissolved lead are generally low in aquatic ecosystems (Ramessur et al., 2001).

There are multiple physiochemical factors influencing the mobility of lead in a river. At low pH, where the pH is less than 5, lead is more soluble and bioavailable, which is more of a risk to aquatic organisms. Some rivers in South Africa have been heavily impacted by Acid Mine Drainage (AMD), such as the Klip River, and have shown to exhibit increased lead mobility under acidic conditions (Tutu et al., 2008). Higher temperatures can enhance lead solubility and microbial activity, which can potentially increase lead bioaccumulation in aquatic organisms (Shahid et al., 2012). The amount and type of organic matter can decrease or increase the bioavailability of lead, as discussed above (Shahid et al., 2012). Lead toxicity has been shown to increase when there is a decreasing concentration of dissolved oxygen because this can create more reducing conditions in a river, affecting the speciation and mobility of lead (Chakraborty et al., 2016). Under hypoxic conditions (low oxygen), lead can be released from the sediments back into the water column, increasing the bioavailability and toxicity to aquatic organisms (Chakraborty et al., 2016). Additionally, low DO can stress aquatic life, making them more susceptible to lead exposure and the associated toxic affects (Chakraborty et al., 2016).

Lead has several toxicological effects on aquatic and human life (Wang & Rainbow, 2008). Lead bioaccumulates in aquatic organisms and has greater toxic effects experienced by higher trophic levels, so the animals on the top of the food chain will have the most severe effects (Wang & Rainbow, 2008). The effects of metals on different fish and invertebrates depend on the species and resilience and changes between aquatic systems (Wang & Rainbow, 2008). Lead that is absorbed by vertebrates is mainly deposited in the bony skeleton, where there are normally no toxic effects exhibited (Wang & Rainbow, 2008). If stress results in decalcification or deossification of the bones, the lead deposits may result in toxic effects through leaching back into the bloodstream (Wang & Rainbow, 2008). Some of the effects on fish include impacted enzyme activity in the gills and liver tissues, impairing energy metabolism and affecting DNA repair processes (Lee et al., 2019). It can cause synaptic damage and changes in neurotransmitters in fish, causing neurological and behavioural problems (Lee et al., 2019). In South Africa, lead contamination is commonly found in drinking water, especially in informal settlements and around mining areas (Mathee, 2014). Lead is a neurological toxin with links to aggression and violent behaviour (Mathee, 2014). Lead exposure is associated

with reduction in intelligence scores, hearing loss, hyper activity and shortened concentration spans (Mathee, 2014). In some adults, chronic lead exposure is associated with hypertension, kidney damage and increased risk of cardiovascular disease (Mathee, 2014). The duration of lead exposure needed to cause adverse effects depends on the concentration, exposure pathway and biological vulnerability, since children are more susceptible (Mathee, 2014). The DWS has developed regulatory guidelines of lead contamination in surface waters. It has permissible levels in order to protect aquatic ecosystems and human health. The South African Water Quality Guidelines set the recommended maximum lead concentration of 0.01 mg/L for drinking water and 0.0003 mg/L for aquatic ecosystems (Silberbauer, 1996). These standards are the same as the standards set by the World Health Organisation (WHO) and the European Union (EU), in the Document for Development of WHO Guidelines for Drinking-water Quality (2016). Challenges of guideline enforcement still remain in South Africa due to inadequate infrastructure and monitoring capabilities (Mathee, 2014).

The sampling techniques and method for detection of lead in aquatic ecosystems is done through the collection of water, sediment and often biotic components such as fish or invertebrates. The samples are then analysed using a range of techniques such as Inductively Coupled Plasma Mass Spectrometry (ICP-MS), a highly sensitive machine capable of detecting trace lead concentrations, Atomic Absorption Spectrometry (AAS), a less sensitive and cost-effective method of analysing sediment samples for lead and UV-Vis Spectrophotometry, a less precise, but quick field assessment of lead concentration in water (Carter et al., 2018). South Africa does not have formalised national sediment quality guidelines for metals such as lead. Since these are not available, many researchers refer to international standards to assess sediment contamination levels (Muller &. One of the more commonly used set of international guidelines is the Canadian Interim Sediment Quality Guidelines (ISQG) (Gordon & Muller, 2010). These guidelines provide Threshold Effect Levels (TEF) and Probable Effect Levels (PEL) for various metals. The TEL concentrations represent the concentrations at which adverse biological effects are expected to be rare. The PEL concentrations represent the concentrations above which, adverse effects on the ecosystem are expected to occur more frequently. The TEL for lead is 35 mg/kg and the PEL is 91.3 mg/kg (Gordon & Muller, 2010).

Understanding the transport and effects of lead throughout the entire process from source to contamination of life is important for public health and environmental protection. Despite stringent water quality guidelines, lead pollution remains a concern due to industrial, mining

and urban runoff contributions. Advanced detection methods as well as stricter enforcement methods are required to mitigate the long-term effects of lead in urban rivers.

iii. Iron

Iron is a naturally occurring, abundant element which plays a crucial role in the aquatic environments. It is an essential micronutrient for aquatic organisms; however, it can become a contaminant when available in excessive concentrations (Allard et al., 2004). Iron enters aquatic environments through both anthropogenic and natural sources and can undergo complex chemical transformations which are influenced by environmental factors, such as temperature, pH and organic matter presence (Poulton & Raiswell, 2005). In South Africa, iron contamination is a growing concern as industrial processes, stormwater runoff and AMD increases iron levels in urban areas (Tutu et al., 2008).

The natural sources of iron in urban rivers originate from the weathering of iron-rich rocks and sediments (Allard et al., 2004). In South Africa, geological formations such as the Witwatersrand basin contain iron deposits which contribute to the fluxes of natural iron in rivers and wetlands (Tutu et al., 2008). In areas with high iron-bearing minerals, groundwater seepage can introduce dissolved iron into aquatic environments (Tutu et al., 2008). The anthropogenic sources of iron in rivers contribute significantly to the concentration of iron in urban rivers. Mining activities are a major source of iron contamination, with acid mine drainage releasing dissolved iron into waters, increasing water quality issues (Tutu et al., 2008). Industrial effluents are a major source of iron in urban rivers, as metallurgical and steel industries discharge wastewater contaminated with high iron levels (Ntengwe, 2006). Agricultural runoff can increase the iron levels in aquatic environments since the use of iron-containing fertilizers can runoff into nearby waterbodies, contaminating the water (Ntengwe, 2006). The chlor-alkali, household chemical, fungicide and petro-chemical industries all use iron in their processes or products (Ntengwe, 2006). Urban runoff and corrosion of iron pipes and infrastructure can wash iron into urban rivers, accumulating in rivers and wetlands and contributing to iron pollution (Göbel et al., 2007).

Iron undergoes various chemical processes in rivers. It exists in aquatic environments in different oxidation states, primarily ferric iron (Fe^{3+}) and ferrous iron (Fe^{2+}) (Wan et al., 2018). Its behaviour is strongly influenced by pH, redox conditions and the presence of organic matter in the river (Wan et al., 2018). Redox reactions occur in aquatic environments under oxygen-rich (oxidizing) conditions, Fe^{2+} is oxidised to Fe^{3+}, forming insoluble iron oxides and

hydroxides, which precipitate and settle as iron flocs. Conversely, under anoxic (reducing) conditions, Fe^{3+} is reduced to Fe^{2+}, increasing its solubility in the water column (Wan et al., 2018). Ferric iron is insoluble in oxygenated waters resulting in iron concentrations generally being low in the water column (Wan et al., 2018). The pH of a river can influence the behaviour of iron. At low pH, common in the presence of acid mine drainage, iron remains in soluble forms, elevating the concentrations in water as the iron binds less to the sediments and freely floats in the water column (Tutu et al., 2008). Dissolved organic matter can form complexes with iron, increasing its bioavailability and transport into aquatic environments. Humic substances, which are naturally occurring organic compounds which arise from the decomposition of plant and microbial material found in soil and water, in wetlands and estuaries play a significant role in iron cycling by binding with Fe^{3+} and Fe^{2+} (Wan et al., 2018).

High iron concentrations in aquatic environments can have various environmental consequences (Wang et al., 2021a). Excessive iron levels can cause reddish-brown discolouration of the water, reducing its clarity and light penetration, which can affect photosynthetic organisms (Xiao et al., 2015). Precipitated iron can form thick deposits on riverbeds, covering and suffocating benthic organisms like worms and molluscs. Iron can bind with phosphorus, limiting nutrient availability and altering the primary productivity in aquatic ecosystems (Wang et al., 2021b). Iron is toxic to aquatic life in high concentrations, particularly when it precipitates in fish gills, leading to respiratory stress and death (Wang et al., 2021a). Excessive iron accumulation in wetlands can influence microbial communities and biogeochemical cycles, affecting wetland health and ecosystem services (Wang et al., 2021a). High iron levels can have detrimental effects on human health, especially when in drinking water and around areas with mining and industrial areas releasing pollution (Puntarulo, 2005). Elevated iron in drinking water can cause gastrointestinal issues with nausea and stomach discomfort, while also adding a metallic taste to the water and staining plumbing fixtures (Chowdhury et al., 2016). Iron is crucial in the production of haemoglobin, but in excessive concentrations, iron can lead to iron overload disorders such as hemochromatosis, which damages organs like the heart and liver (Chowdhury et al., 2016). Chronic exposure to high iron levels through contaminated water can lead to oxidative stress and cellular damage (Puntarulo, 2005).

The South African Water Quality Guidelines by the DWS recommends a 0.3 mg/L limit for iron in drinking water to prevent health issues and 0.01 in aquatic environments to safeguard aquatic life (Silberbauer, 1996). The data on acute and chronic toxicity effects of iron to both

vertebrates and invertebrates is limited, so only tentative guidelines are given for the Target Water Quality Range (TWQR), stating that the iron should not change more than 10% of the background dissolved iron concentration for a particular site (Silberbauer, 1996). The same methods of detecting lead mentioned earlier are used for detecting iron such as AAS, ICP-MS and UV-Vis Spectrophotometry (Carter et al., 2018). The USEPA does not have a recommended background concentration for iron in aquatic rivers, also stressing the need for background knowledge and concentrations. There are various studies which consider iron concentration in aquatic environments in South Africa, such as the study done by Addo-Bediako et al. (2021) and Dahms et al. (2017), with highly variable ranges of iron being observed. In the Spekboom River in South Africa, a study done by Addo-Bediako et al. (2021) in which activities relating to mining have heavily influenced iron concentrations in rivers, with iron ranging from 35,000 mg/kg to 47,000 mg/kg (Addo-Bediako et al., 2021). A study done on trace metals in the Mvudi River in South Africa by Edokpayi et al. (2016), observed iron concentrations ranging from 4296 to 5557 mg/k. In Dahms et al. (2017), the levels of iron ranged from 3600 mg/kg to 55800 mg/kg throughout the study period, but they did not observe any patterns between sites or seasons (Dahms et al. 2017).

h. Measurement parameters for metal samples

Sediments and soils in rivers play an essential role in trapping and storing metals through various biological, chemical and physical processes (Chon et al., 2012). As metals enter rivers through natural weathering or industrial discharges and urban runoff, they bond to particulate matter and settle at the bottom of the river into sediments (Chon et al., 2012). The accumulation of metals in sediments occurs through adsorption onto mineral surfaces, precipitation as metal oxides and hydroxides, and complexion with organic matter (Foster & Charlesworth, 1996). The mobility and retention of metals in sediments is significantly influenced by redox conditions, pH and organic content. Under oxic conditions, metals tend to precipitate and bind to sediment particles, reducing the bioavailability of metals in the water column (Foster & Charlesworth, 1996). Under anoxic conditions, found in deeper sediments and highly polluted waters, the remobilisation of metals through reduction reactions releases the metals back into the water (Chon et al., 2012). This exchange makes sediments both a sink and a potential source of metal pollution, especially when disturbed by dredging or flooding (Chon et al., 2012). This process can be seen in Figure 1. The long-term effects of bioaccumulation of metals in

sediments can have severe ecological consequences, affecting the benthic communities and resulting in bioaccumulation in aquatic food webs (Heikens et al., 2001). Therefore, the monitoring of metal concentrations in sediments and understanding their geochemical behaviour is important for understanding river health and managing pollution in South African water systems.

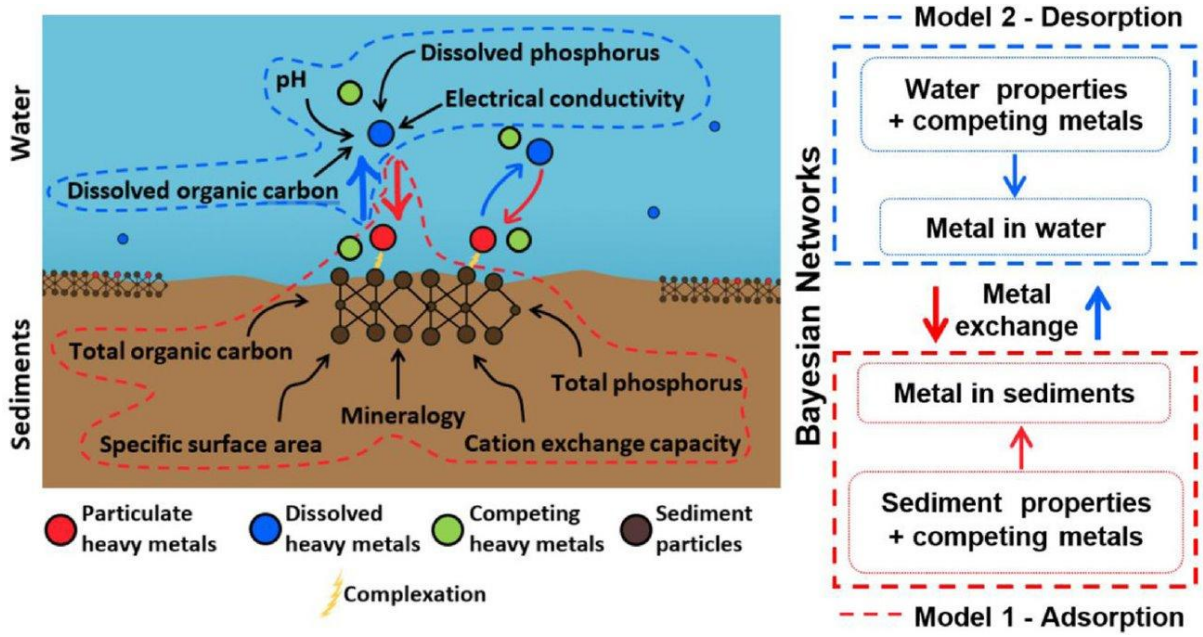


*Figure 1: The relationship between metals and the water and sediment of a river (**Freshwater Ecology. (2021, July).** Water-sediment interface metals diagram [Image]. Freshwater Ecology. Retrieved from https://freshwaterecology.files.wordpress.com/2021/07/wsimetals.jpg?w=1024)*

Pollution and ecological risk indices are tools used by scientists in many different fields and professions involved with environmental issues (Birch, 2023; Fadlillah et al., 2023; Ferreira et al., 2022; Hakanson, 1980). They assist in assessing the contamination from chemical elements in soils and sediments (Ferreira et al., 2022). When measuring the levels of pollution in an urban river by testing the sediment and water for metals, it is useful to make use of Contamination Factor (CF), Geoaccumulation index (I_{geo}), Pollution Load Index (PLI) and Enrichment Factor (EF) (Ferreira et al., 2022). These are parameters used to measure the level of pollution in a particular environment, such as water or sediment (Dahms et al., 2017). These indices are significant in a comprehensive assessment of an urban river because it provides a quantitative assessment of metal pollution and its sources. They are useful alongside SASS5 studies since it incorporates a biological perspective with a quantitative assessment to produce a better understanding of the health of a river (Dahms et al., 2017).

i. Contamination Factor (CF)

The CF is the ratio obtained by dividing the concentration of each metal in the sediment by the baseline, background or control value (Thomilson et al., 1980). Since urban rivers experience a variety of pollutants, CF is used to determine hotspots of contamination. It is especially useful in urban rivers which are more likely to experience various metal contamination loads since it is useful to help researchers and authorities understand pollution sources such as industrial runoff or sewage (Dahms et al., 2017). The CF can vary across different seasons, land uses and geographical locations within the same urban river, indicating pollution sources and management needs (Dahms et al., 2017). It useful since it is easy to apply for assessing contamination levels in specific areas. It is limited since it requires a baseline or background data, which can be challenging to obtain in urban areas which are not always tested for CF (Dahms et al., 2017). CF does not account for the bioavailability of contaminants or their ecological impacts, it is rather used to determine if there is pollution and to what degree of contamination has occurred as the specific sampling site (Dahms et al., 2017).

Equation 1: Contamination Factor (CF)

$$CF = (C_{heavy\ metal}/Cbackground)$$

Where:

- CF is the contamination factor which is being calculated;
- $C_{heavy\ metal}$ is the concentration of the contaminate in the sample;
- $C_{Background}$ is the concentration of the selected baseline sample concentration

The contamination levels may be classified based on their intensities on a scale ranging from 1- 6 where:

- 0 = none,
- 1 = none to medium,
- 2 = moderate,
- 3 = moderately to strong,
- 4 = strongly polluted,
- 5 = strongly to very strongly polluted,
- ≥ 6 = very strongly polluted

ii. Pollution Load Index (PLI)

Pollution load index is used to assess the overall pollution load of a river or water body by combining multiple pollutants into a single index value (Thomilson et al., 1980). This can be used to integrate concentrations of several contaminants, such as metals, to provide a comprehensive pollution assessment. Since urban rivers are subject to a variety of pollution types and concentrations, PLI is a useful measure of pollution load since it considers multiple pollutants which come from multiple sources, such as stormwater runoff, wastewater treatment plants and industrial runoff, and are trapped by the sediment in rivers. PLI is useful for tracking temporal trends, since the same part of the river can be measured multiple times over a study period and tested for the same contaminants (Dahms et al., 2017). This is useful in urban rivers since changing land use and urbanisation can cause stresses and changes to rivers in urban areas over time. PLI is determined by the calculation of the product of the CF. The equation used to determine PLI is:

Equation 2: Pollution Load Index (PLI)

$$PLI = (CF1 \times CF2 \times CF3 \times \cdot\cdot CFn)\, 1/n$$

Where:

- PLI is the Pollution Load Index being measured;
- CF is the Contamination Factor of each metal that was tested in the study;
- n is the number of metals being investigated.

The scale for determining the level of pollution in using PLI is:

- from 0 = Background Concentration,
- 0>PLI≤1 = unpolluted to moderately polluted,
- 1>PLI≤2 = moderately polluted,
- 2> PLI ≤ 3 = moderately to highly polluted,
- 3> PLI ≤ 4 = Highly Polluted,
- PLI≥5 = very highly polluted.

The strengths of using PLI is that it provides an integrated, holistic view of the water and sediment quality, which is especially useful in urban areas which are complex environments where pollution from various sources has impacts on aquatic health (Dahms et al., 2017). The limitations of PLI are that, similarly to CF, the equation relies on accurate background data or

a baseline to compare other rivers to. The PLI also does not describe, identify or explain the ecological impacts of pollutants, such as the toxicity or bioavailability of contaminants (Dahms et al., 2017). This is why it is important to supplement this data with other environmental health, biological indicators or contamination indicators.

iii. Enrichment Factor (EF)

Enrichment factor is used to assess the degree to which the concentration of a particular element or pollutant in sediment or water is above or below the background in that area (Reimann & de Caritat, 2005). It is commonly used to determine the concentration of metals in urban areas as a way of understanding anthropogenic impacts created by various land uses. EF is used to evaluate the impact of anthropogenic activities related to metal abundance in sediments (Reimann & de Caritat, 2005). It is argued that EF is often misused or unreliable because it depends heavily on the choice of reference element and background values, which can vary greatly depending on the geology (Reimann & de Caritat, 2005). A reference element is chosen to normalise natural variations in sediment composition, due to differences in particle size, minerology or lithology (Reimann & de Caritat, 2005) Typically common elements used as reference elements in sediment samples include aluminium (Al), Iron (Fe), Titanium (Ti), Scandium (Sc) and Lithium (Li), since they are major components of the Earth's crust, are largely unaffected by human activity and are chemically stable in elements (Reimann & de Caritat, 2005). The equation for EF is:

Equation 3: Enrichment Factor (EF)

$$EF = \frac{\left(\frac{CX}{CFe1}\right)}{\frac{CX}{CFe2}}$$

Where:

- CX is the concentration of the pollutant being measured
- CFe1 is the reference element from the reference site
- CFe2 is the average concentration of the reference or background pollutant

Calculated EF values are interpreted as:

- EF ≤ 1: no enrichment;
- 1 < EF < 3: minor enrichment;
- 3 < EF < 5: moderate enrichment;
- 5 < EF < 10: moderate-to-severe enrichment;

- 10 < EF < 25: severe enrichment;
- 25 < EF < 50: very severe enrichment;
- EF > 50: extremely severe enrichment.

iv. Geo-accumulation Index (Igeo)

The Igeo is widely used method of assessing the degree of metal contamination in sediments and soils. It is a comparison between the levels of metal pollution found in a system, compared to pre-industrial levels (Muller, 1979). However, in geogenically mineralised systems, naturally high background metal concentrations may result in overestimation of anthropogenic pollution unless site-specific baselines are used (Dahms et al., 2017). A major shortcoming of Igeo calculations is that there is uncertainty with selecting an appropriate background metal concentration (Bn). Natural geological variability and the lack of true pre-industrial data can lead to inaccurate baselines, causing either overestimation or underestimation of contamination (Muller, 1979; Reimann & de Caritat, 2005). The following formula is used to measure Igeo values:

Equation 4: Geo-accumulation Index (Igeo)

$$Igeo = \log 2(\frac{Cn}{1.5Bn})$$

Where:

- Cn is the concentration of the metal in question measured at a specific site
- Bn is the geochemical background metal levels or reference site

Where:

- Fe (iron) is chosen as a natural element of reference

Calculated EF values are interpreted as:

- Class 0 (practically unpolluted): Igeo ≤ 0;
- Class 1 (unpolluted to moderately polluted): 0 < Igeo < 1;
- Class 2 (moderately polluted): 1 < Igeo < 2;
- Class 3 (moderately to heavily polluted): 2 < Igeo < 3;

- Class 4 (heavily polluted): 3 < Igeo < 4;
- Class 5 (heavily to extremely polluted): 4 < Igeo < 5;
- Class 6 (extremely polluted): 5 > Igeo

Table 1: The Description of Contamination factor (CF) (Thomilson et al., 1980), Enrichment Factor (EF) (li et al., 2013), Geo-Accumulation Index (Igeo) (Muller, 1979) and Pollution Load Index (PLI).

Index	Value	Class	Description
CF	CF<1		Low contamination
	1<CF<3		Moderate contamination
	3<CF<6		Considerable contamination
	6<CF		Highly contaminated
EF	EF<1		No enrichment
	1<EF<3		Minor
	3<EF<5		Moderate
	5<EF<10		Moderately severe
	10<EF<25		Severe
	25<EF<50		Very severe
	>50		Extremely severe
Igeo	≤0	0	Unpolluted
	0-1	1	Unpolluted to moderately polluted
	1–2	2	moderately polluted
	2–3	3	Moderately to strongly polluted
	3–4	4	Strongly polluted
	4–5	5	Strongly to extremely polluted
	>6	6	Extremely polluted
PLI	PLI Final < 1		Pollution
	PLI Final > 1		No pollution

i. Similar studies

A study on the Hawkesbury-Nepean River system in New South Wales in Australia used multivariate data to determine the health of the river (Pinto & Maheshwari, 2011). The river

faces various challenges in terms of water quality stemming from its source travelling through farmlands before entering urban, peri-urban and industrial areas. It faces point source pollution from sewerage leaks, mining activities and industrial effluent, as well as diffuse sources of pollution attributed to urban runoff and agricultural activities. Vegetation clearance has caused increased subsurface and agricultural runoff as well as increased sediment loads into the river system. The study used 40 water quality variables to determine the river health. Ten key variables were determined for identifying river quality which revealed anaerobic conditions, microbial quality and effects of eutrophication in the Hawkesbury-Nepean River. The mean values for each parameter were reported and included: Temperature of 21.05^0C, pH of 7.64, DO of 90.44mg/l, EC of 1.78ms/cm, total phosphorus of 0.02mg/l, Total Nitrogen of 0.53mg/l, Iron of 0.43mg/l, Manganese of 0.05mg/l, *E. coli* MPN of 68.12, coliforms MPN of 2230.08. The authors determined anaerobic conditions, microbial quality and eutrophication in the river system were revealed (Pinto & Maheshwari, 2011).

A study done on the Orange River in Namibia used macroinvertebrates and water quality parameters to assess river health (Munyika et al., 2014). Agricultural activities and anthropogenic activities, including informal settlements along the river affect water quality and river health (Munyika et al., 2014). The SASS5 scoring system was applied to eight sites and samples were taken to measure macroinvertebrate presence alongside physio chemical parameters which included pH, EC, temperature, dissolved oxygen and turbidity which included total phosphates, total nitrogen, TDS and chlorophyll, and bacteria presence was measured. Their results from the invertebrate sampling showed fair water quality conditions and a moderately modified system in category C (see table on page 106) (Munyika et al., 2014). This study confirmed the relationship between the presence of macroinvertebrate assemblages and water quality, with thriving macroinvertebrate assemblages indicating river health (Munyika et al., 2014). Statistical analysis was used to determine that there was no significant difference in the mean water quality concentrations among the sampling periods and sites.

Both of these studies contribute valuable insight into river health assessment, but both have noticeable methodological limitations. While the study by Pinto and Maheshwari (2011) provided a comprehensive physio-chemical assessment using multivariate analysis and 40 water quality variables and gave a broad understanding of the pollution in the Hawkesbury-Nepean River. However, the study relied mainly on chemical and microbial indicators, with no incorporation of biological indices such as macroinvertebrates, which limits its ability to capture long-term ecological responses to pollution. In contrast, Munyika et al. (2014) provided

a more integrated approach by combining physio-chemical parameters with biological monitoring (SASS5), offering a more comprehensive ecological perspective. However, the lack of statistically significant variation in water quality parameters across sites and seasons reduce its spatial and temporal resolution. Overall, while both studies advance understanding of river health, they highlight the need for mutli-metric, long-term assessments that integrate biological, chemical and physical indicators for more comprehensive and robust interpretations.

j. Aims and Objectives

The objective of this study was to identify the levels of pollution and contamination within the Jukskei River Catchment System in the Northern Suburban areas of Johannesburg by performing a comprehensive environmental assessment at five different sites along the Braamfontein Spruit and its tributaries. Water samples, sediment samples, field measurements and a SASS5 biodiversity assessment were performed at each of the sites, within the same day, once a week, for seven weeks. This was repeated in two more sampling campaigns, bringing the total number of sampling days to 21. Samples were taken over the course of a year in order to identify seasonal trends. The samples were consistently taken during the morning hours of the day with the first site being sampled around 9am and the fifth site being sampled around 12pm. The combination of different sampling and measurement techniques was chosen in order to comprehensively measure a variety of indicators of river health.

This review has demonstrated that there are various sources of pollution impacting rivers, human health and biodiversity. In order to fully understand the different contaminants, pollutants and effects of anthropogenic activity on aquatic ecosystems, multiple methods of analysis are needed. South African rivers are heavily impacted by anthropogenic activities which come from multiple sources, such as mining, industrial effluent, stormwater runoff and leaking infrastructure. While naturally occurring nutrients and bacteria can influence the measurements done in the study, anthropogenic activities will have significant and noticeable impacts.

There are knowledge gaps in the background of many South African rivers, with a lack of monitoring and continuous health studies being the main cause of this. While numerous studies have been done on sediment metal contamination in South Africa, there still lack national guidelines as to what concentrations are expected, toxic to the environment and toxic to

humans. More studies linking the effects of metal concentration in sediments to the impacts invertebrates is needed.

This study aims to improve understanding of the levels of contamination the urban rivers in Johannesburg. By measuring the contamination using various methods, this study will provide a more comprehensive understanding of the types and levels of pollution and the effects it has on the environment, biodiversity and human health. To identify sources of pollution, regular sampling over a chosen area and over a specific, long-term period is needed. It is anticipated that the study will provide recommendations for improving urban river management and reducing pollution levels.

3. Methodology

a. Study Area

The Northern suburbs of Johannesburg, South Africa were selected as the area of study due to the presence and widespread public use of urban rivers and their proximity to the University of the Witwatersrand Geography Laboratory for sample acquisition and testing. Five sites were selected along the three urban rivers based on accessibility, location within the catchment system and the anthropogenic impact on the rivers. The location of the sites within the cotext of South Africa and the sites within Johannesburg shown on a map in Figure 2.

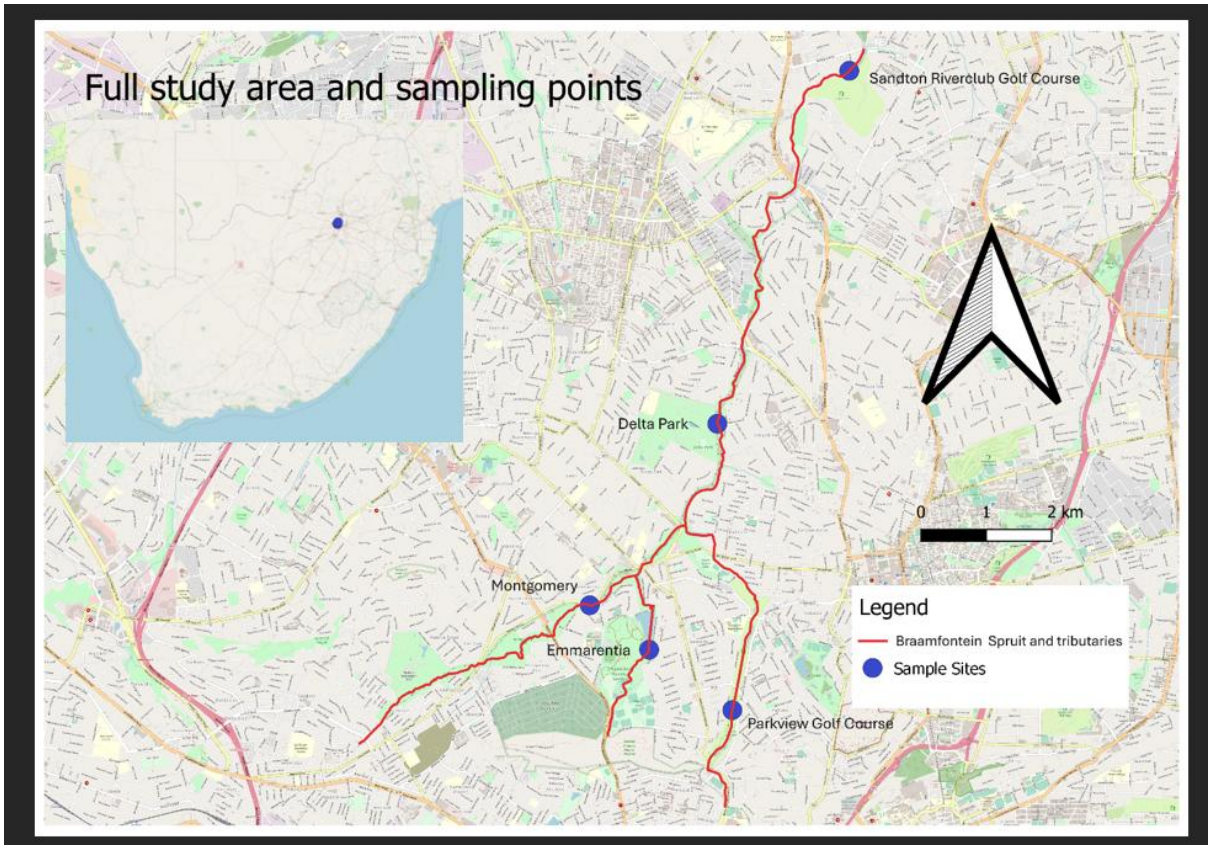


Figure 2: All sampling sites along the Braamfontein Spruit and its tributaries (Westdene and Montgomery Spruit) and the location of the sites within the context of South Africa

Site 1 (Figure 3) (26° 8'59.20"S) (27°59'46.14"E) is located along the Montgomery Spruit, which has its source at Albert's Farm Spring approximately three kilometres upstream of the site. The source is the only Artesian Spring in Gauteng and it flows into a pond where birds and wildlife are able to benefit from it (Davie, 2021). Site 1 is located before the river joins the Westdene Spruit at (26° 8'45.48"S) (28° 0'12.40"E) and the Braamfontein Spruit, following which, they flow into the Jukskei River System near Victory Park and Parkhurst (26° 8'20.18"S)

(28° 0'38.84"E). Site 1 is often used by people for religious and/or spiritual practices, with melted candles often surrounding the riverbanks. Some people also use the river for bathing and washing clothes upstream of Site 1 in various locations between the source in Albert's Farm Spring and the sampling site (26° 9'30.83"S) (27°58'38.84"E).

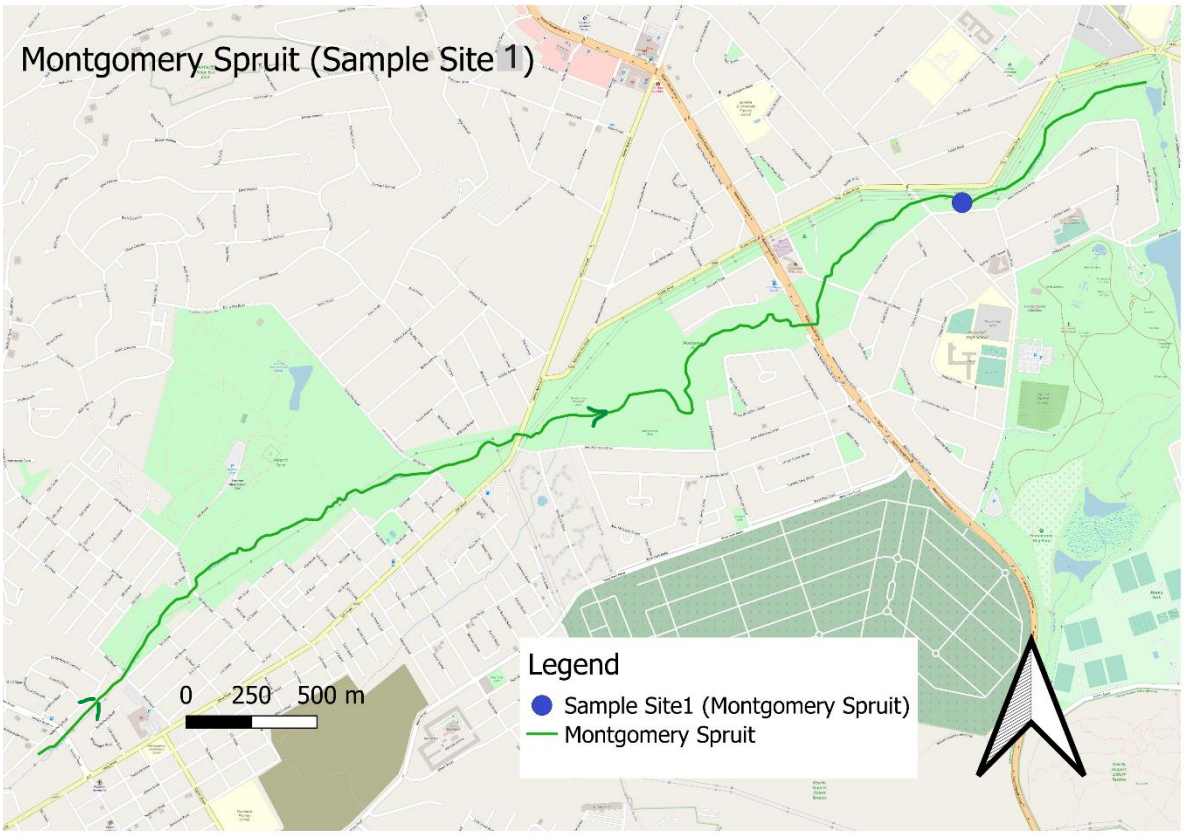


Figure 3: Site 1, The Montgomery Spruit

Site 2 (Figure 4) (26° 9'20.73"S) (28° 0'17.49"E) is located on the Westdene Spruit in the Emmarentia Botanical Gardens, which is a few kilometres downstream from the Westdene Dam (26°10'52.61"S) (27°59'17.87"E), a publicly accessible dam. The river flows through the Melville Koppies before entering the Emmarentia Botanical Gardens and travels through two small dams and vegetated areas before reaching Sampling Site 2. This Site is located at a common recreational area and is often accessed by dogs on walks and people who use it. The Westdene Spruit joins the Montgomery Spruit first and then the Braamfontein spruit upstream of Delta Park.

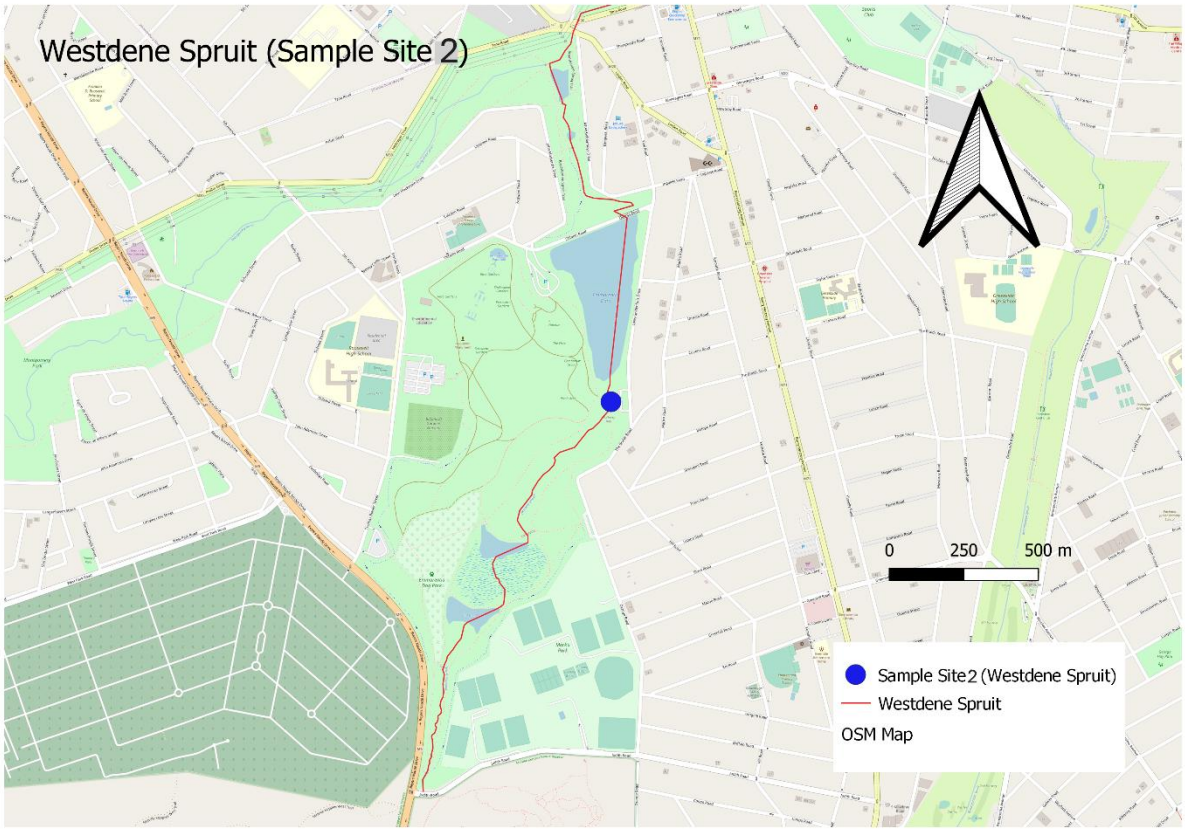


Figure 4: Site 2, The Westdene Spruit at Emmarentia Botanical Gardens

Site 3 (Figure 5) (26° 9'50.93"S) (28° 1'4.53"E) is located in the Parkview Golf Course near the source of the Braamfontein Spruit. This site is important as it has not been influenced by the water flowing from the Montgomery Spruit and Westdene Spruit. This river flows through the Northern Suburbs of Johannesburg before it joins the combined Montgomery Spruit and Westdene Spruit, located upstream of Delta Park (26° 8'20.18"S) (28° 0'38.84"E).

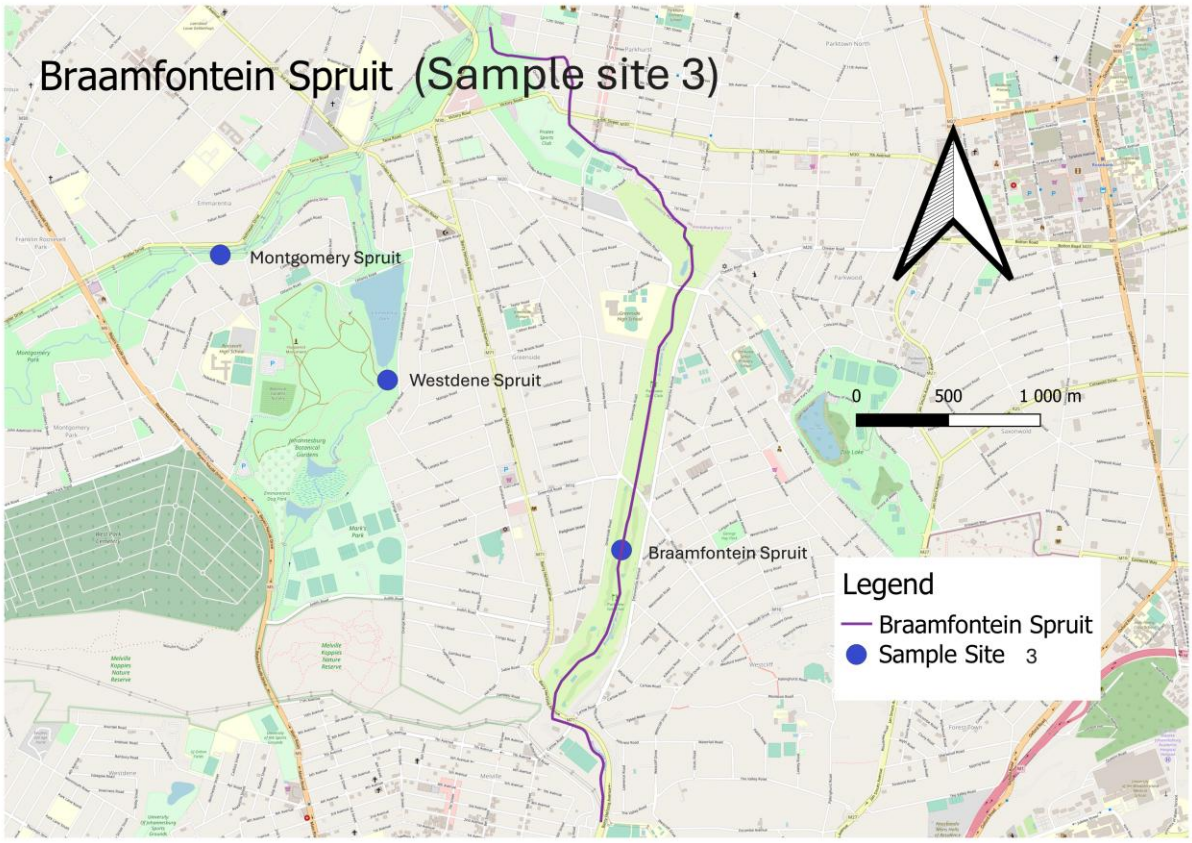


Figure 5: Site 3, The Braamfontein Spruit at Parkview Golf course (Site 1 and 2 included as reference)

Site 4 (Figure 6) (26° 7'29.94"S) (28° 0'56.77"E) is located in Delta Park and is a commonly used section of the urban river for recreational purposes such as dog walking but is also used by some people for bathing and clothes washing. The relevance of this site is that it is a combination of the water flowing from Site 1, Site 2 and Site 3. Many people make use of the surrounding recreational areas around the river and can interact with the contaminated water causing waterborne diseases, rashes and sickness (Toro et al., 2022). It is important to the community to ensure that the pollution levels in this river are kept within the Target Water Quality Ranges.

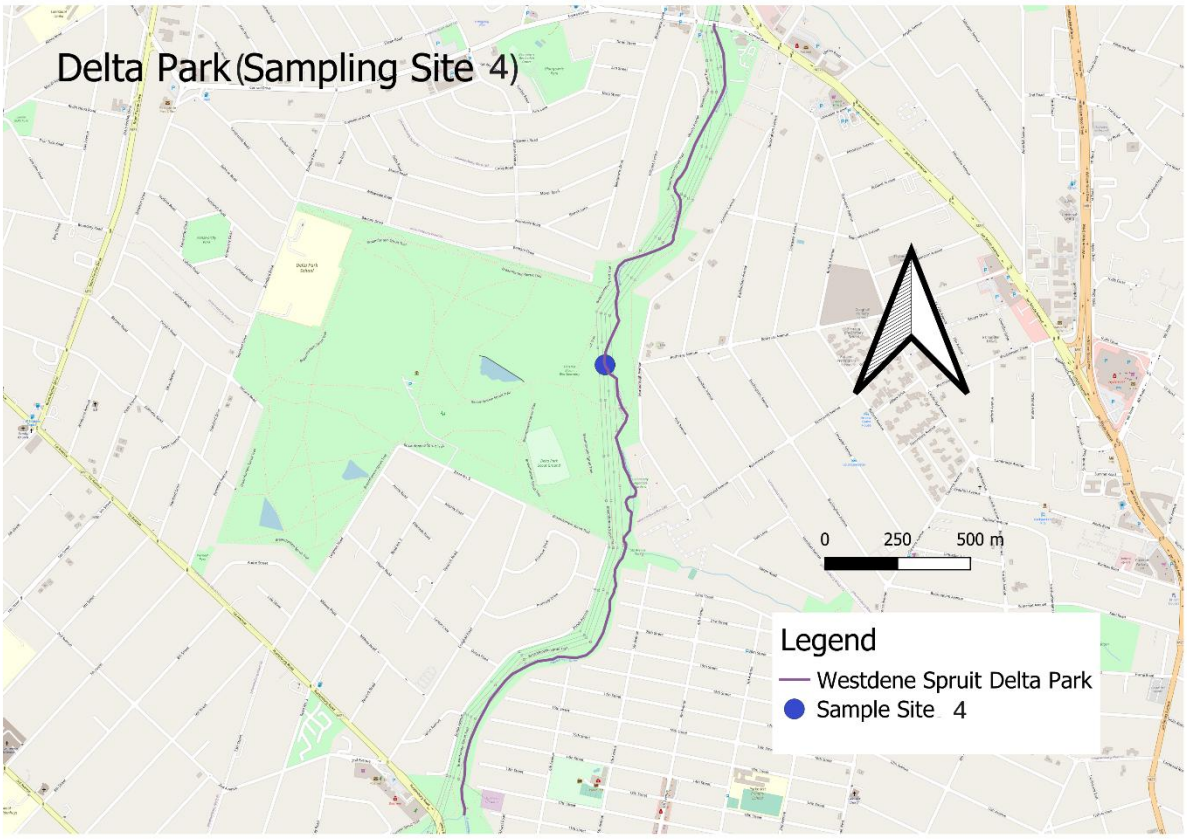


Figure 6: Site 4, The Braamfontein Spruit at Delta Park

Site 5 (Figure 7) (26° 4'34.15"S) (28° 2'9.57"E) is located downstream of Delta Park at the Sandton Riverclub Golf Course. This site was chosen to demonstrate either environmental amelioration or accumulation of contaminants from this dense suburban area of Johannesburg.

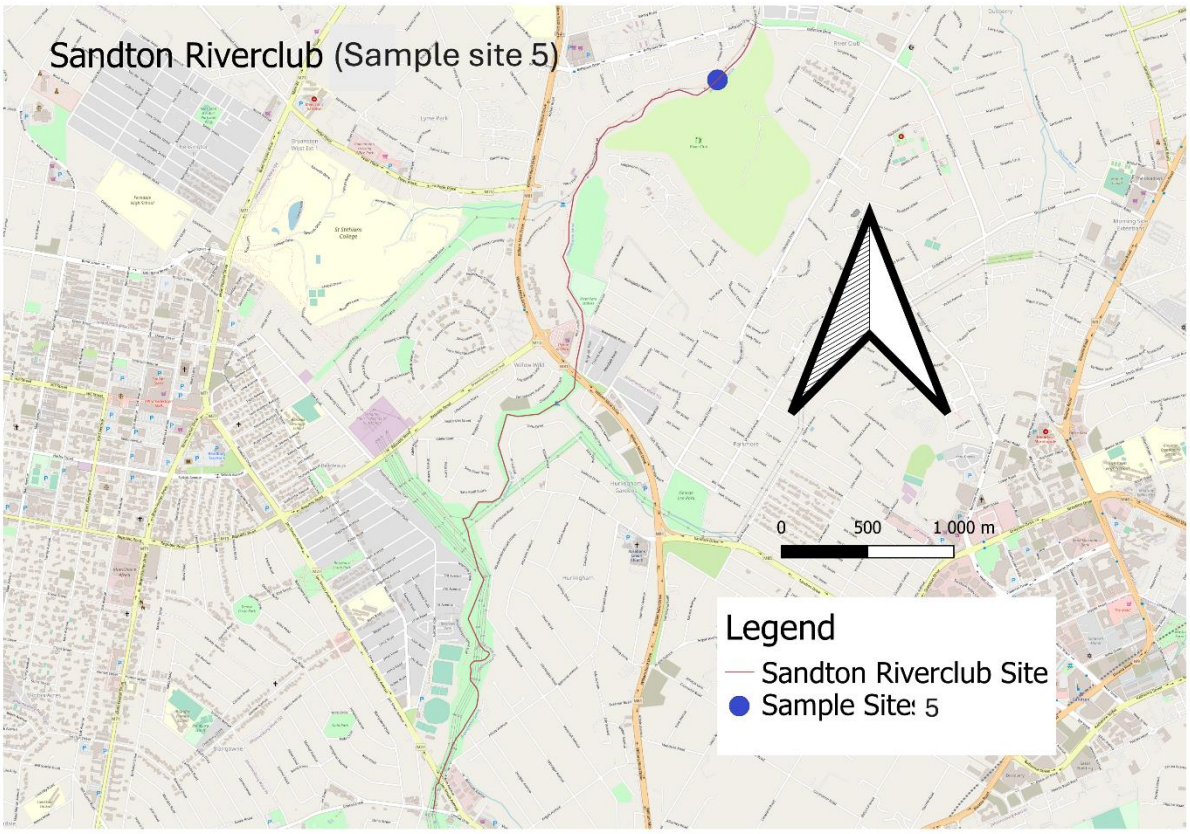


Figure 7: Site 5, The Braamfontein Spruit at Sandton Riverclub

b. Sampling strategy



Figure 8: The waders used to take samples and the YSI used to take in-situ measurements



Figure 9: The multivariate probe used to take in-situ measurements:

Field measurements were taken at each site using a YSI multivariate probe, shown in Figure 8 and Figure 9, by dropping the probe into the water approximately 30-40cm deep, depending on the depth of the river at the sampling sites, in order to measure the pH, Electrical Conductivity (EC), Dissolved Oxygen (DO) and temperature of the water in real time.



Figure 10: The bottles and gloves used to take water (left) and sediment (right) samples

The sediment sampling consisted of taking 100 mL sediment samples from each site under the water by using 100 mL plastic sealable containers to scrape sediment from the bottom of the river as close to the current and as deep as possible without losing the container. These bottles are shown in Figure 10. This was achieved by wearing waders and gloves, walking into the river channel and reaching down into the water to reach the sediment below, approximately 10-20 cm below the water level through all seasons. Site 3 in Parkview consisted of mainly bedrock below the water meaning that in order to retrieve the sediment sample, rocks had to be moved, and small pockets of sediment had to be located each sampling trip. This was done to measure the concentration of metals trapped in the sediment at each site. The samples were sealed and then stored inside of a cooler box for the remainder of the sampling trip before being placed into a freezer, at approximately -15˚C, in the Wits Geography Laboratory.



Figure 11: the Merck lab equipment used to test the water samples

Water samples were taken in 500 mL acid cleaned polyethylene bottles and stored in a cooler box before being taken to the Wits Geography Laboratory. Figure 11 shows the samples in the lab being tested for sulphate, nitrate, nitrite, phosphate, *E. coli* and coliforms. When taking water samples, waders were worn to reach deeper channels of flowing water and avoid any stagnant water near the riverbanks. The samples were taken at an approximate depth of 10-20 cm, making sure to rinse out the bottles with river water twice before filling up the sample bottles to ensure that there are no cross contaminants from residue left in the sampling bottles. Water samples were taken at each site each week to be measured for nutrients, and approximately every second week a portion of the same samples were measured for *E-coli*. Once measuring was complete, the samples were stored in a freezer at approximately -15˚C in the Wits Geography Laboratory.



Figure 12: A net used to capture macroinvertebrates

Benthic macroinvertebrate samples were collected at each site using a standard 30x30 cm, shown in Figure 12, 1 mm mesh size square dip net and a 40x30 cm tray to identify the invertebrates. The standardized "kick-stir-sweep' method, adapted from the South African Scoring System version 5 (SASS5), was used for the collection of the macroinvertebrate samples from all habitats (Dickens & Graham, 2002). This method is a refinement of the South African Scoring System (SASS) method developed by Chutter (1994). The same amount of effort and time (close to 10 minutes) spent sampling the invertebrates was done at each of the sample sites to ensure consistency. The samples were then placed in the tray so that they could be easily identifiable, with each sample being inspected for as long as it took to ensure thorough identification of each species was obtained and counted. The net and tray were then rinsed in the river to ensure that all invertebrates were safely returned to the river where they were removed. The literature used to identify the various species sampled at each of the sites was the Aquatic Invertebrates of South African Rivers Field Guide (A. Gerber & M. Gabriel, 2002).

i. Chemical Analysis

In order to test the concentration of metals in the sediment samples, sample preparation had to occur by digesting the samples in nitric acid and Hydrogen Peroxide. This method is the EPA Standard Method for sediment digestion. The samples were left outside the freezer to dry overnight. Using a laboratory scale (Accuris Precision Balance), 1g of sediment was measured, placed into a glass digestion tube and labelled. A Thermoline Scientific Laboratory uncirculated water bath was heated to 90˚C. 5 mL of 1:1 nitric acid was added to each sample and the digestion tubes were placed inside the bath, covered with a watch cover and left to heat for 15 minutes, not allowing the samples to dry out. The samples were then removed from the bath and cooled for 2 minutes before 2.5mL of nitric acid was added to each sample. They were then placed back into the bath for 30 minutes before repeating the process of removing, cooling and adding another 2.5mL of nitric acid to the samples. This process is repeated as many times as it takes until there were no brown fumes are given off by the samples when removing them from the bath, indicating that the reaction with the nitric acid is complete. The sample is then cooled and 0.5mL of 30% hydrogen peroxide was slowly added to the samples, which would induce effervescence in the sample. They were then placed back inside the bath until the effervescence subsided which took approximately 10 minutes. The sample was then removed from the bath and left to cool before adding another 1mL of 30% hydrogen peroxide to each sample, repeating the same process two more times resulting in four total cycles of adding hydrogen peroxide to the samples. The samples were then removed from the bath and allowed to cool, the watch covers were removed, and the samples were placed in 50ml falcon tubes which were filled to 50mL by adding deionised water to the samples. These samples are then filtered using 0.45 µm pore size Whatmann filter paper and a vacuum filter. These samples were tested at the Wits School of Chemical and Metallurgical Engineering using Atomic Absorption Spectroscopy (AAS) to measure the concentration of manganese, iron and lead in the sediment. A sample of deionised water was also measured as the control for the study and all the metals in the sample were below level of detection.

On the days where *E. coli* was tested, water samples were immediately placed into a cooler box when the sample was taken to reduce microbial change of the water due to exposure to heat. The Colilert method was used to measure the concentration of *E. coli* at each of the sample sites (Buckalew et al., 2006). Once in the laboratory, 100 mL of the sample is poured into an IDEXX sample vial and the Colilert reagent, which is made up of two key components, the

first being Ortho-nitrophenyl-β-D-galactopyranoside (ONPG): This substrate detects Total Coliforms produce an enzyme that breaks down ONPG, resulting in a yellow colour. The second key component added to the sample is MUG (4-methylumbelliferyl-β-D-glucuronide). MUG detects *E. coli*. The *E. coli* produces the enzyme β-glucuronidase, which breaks down MUG, causing fluorescence under UV light, however, the exact concentrations are not publicly available as they are proprietary to IDEXX. The sample is then poured into an IDEXX tray which is made up of 49 large cells and 48 small cells, which was then sealed. The sample is then incubated at 35˚C for 18 hours in a laboratory incubator. When the samples are taken out of the incubator, the tray is inspected to identify the number of large cells and small cells that are yellow, which was placed in the IDEXX calculator tool to determine the most probable number of coliforms in the sample. The tray is then placed under an ultraviolet light, at a wavelength of 365nm, to inspect the amount of large and small cells that are fluorescent. The count of fluorescent blocks and the dilution used is entered into the IDEXX calculator to determine the most probable number of *E. coli* concentration in the sample. In order to calibrate and provide a quality control, deionised water was measured using the same method.

The water samples were measured according to the Merck test kit's instructions for each nutrient. The Merck kits used were 1.09713.0002 for nitrates, 1.14776.001 for nitrites, 1.14842.0001 for phosphates and 1.02537.0001 for sulphates. A Merck prove 600 Spectroquant spectrometer was then used to measure the concentration of each nutrient throughout the study. Deionised water was used as a blank sample to test the accuracy of the results and showed no nutrients or bacterial contamination.

The samples were sent to Aquatico Laboratories, a SANAS accredited laboratory, to be tested for the quantification of selected metals, namely manganese, lead and iron. The samples were analysed in the laboratory using their internal method ALM 31, which is a validated procedure under SANAS. This method determines the quantity of dissolved and total metals in water samples and is based on internationally recognised standards utilising Inductively Coupled Plasma Optical Emission Spectrometry (ICP-OES). This method is widely used for the simultaneous determination of multiple metals in environmental samples, offering high sensitivity and precision. Prior to analysis, water samples were preserved using nitric acid

ii. Data Analysis

The results for the *E. coli*, coliforms, nutrients, field measurements and metals in both the sediment and water samples were recorded in an excel table and graphs were created using the data, expressing the results across various parameters, within the same graph, for each river.

The SASS5 monitoring was recorded by taking notes of which taxa of invertebrates were found and assigning them a category depending on the number of invertebrates that were found for each taxon. The categories are A (if there were 1-10 invertebrates), B (if there were 11-100 invertebrates) and C (if there were over 100 invertebrates). The results were then recorded in a SASS5 scoring sheet (shown in Figure 62 in the appendix) along with the date, site name and the type of environment sampled. The scoring sheet provided the SASS score, which was divided by the number of different taxa found in that sample to acquire the average score per taxon (ASPT). The ASPT and SASS score were used to determine the quality of the water and the health of the river by comparing the results to a standardised categorisation sheet. The SASS5 score sheet used in this study can be found in the appendix.

4. Results

a. Montgomery Spruit

The field measurements from Montgomery Spruit are presented in Figure 13 and Figure 14. The original data is presented in Table 5 in the appendix. The pH, Dissolved Oxygen, temperature and EC are presented from the period 11 May 2023 to 9 April 2024. The pH results from the Montgomery Spruit seen in Figure 14 was seen to be circumneutral throughout the study. The temperature increases from a minimum in June to a maximum in February and the range for temperature is from 7.9˚C to 20.6˚C. The DO varied slightly with an increase in colder temperatures and a decrease in warmer temperatures, however the river remains somewhat oxygenated throughout the study. This shows a negative correlation between temperature and dissolved oxygen shown in Figure 52 in the appendix. The range for DO was between 6.6 mg/L and 14.2 mg/L, and the range for temperature is between 7.9˚C and 20.6˚C. There is a positive correlation between pH and DO shown in Figure 53 in the appendix, this could potentially indicate that acidification is not a major concern in the river. The removal of CO_2 reduces carbonic acid (H_2CO_3) in the water, resulting in a higher pH value (Reeder, 2011). While nutrients themselves don't inherently increase pH, phosphate driven algal productivity can raise the pH during the day, while nitrification and organic decomposition tend to lower it (Nimick et al., 2003). The EC increases and is at its peak in February of 442 µs/cm. There was a positive correlation between temperature and EC shown in Figure 54 in the appendix, with an increase in EC when temperature increased. This could be on account of warmer temperatures increasing the solubility and mobility of dissolved ions, leading to a higher EC. This could also be because there are higher rainfall patterns in summer, increasing the urban runoff and carrying salts, industrial effluents and sewage, increasing the EC. Since there are no significant spikes, these results are not concerning. The data from this graph

indicates low levels of impact from urban activity. The pH remained circum-neutral throughout the study period with a pH of 8.64 being the maximum and 7.84 being the minimum.

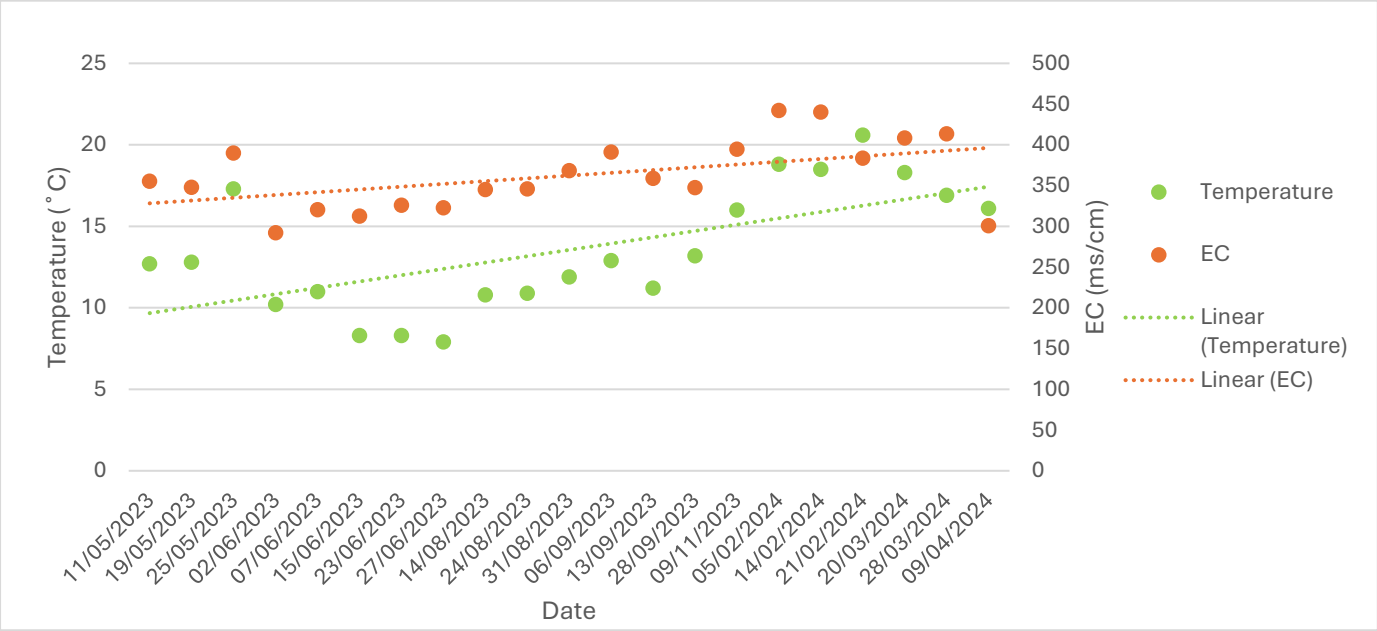


Figure 13: Temperature and EC analysis from the Montgomery Spruit.

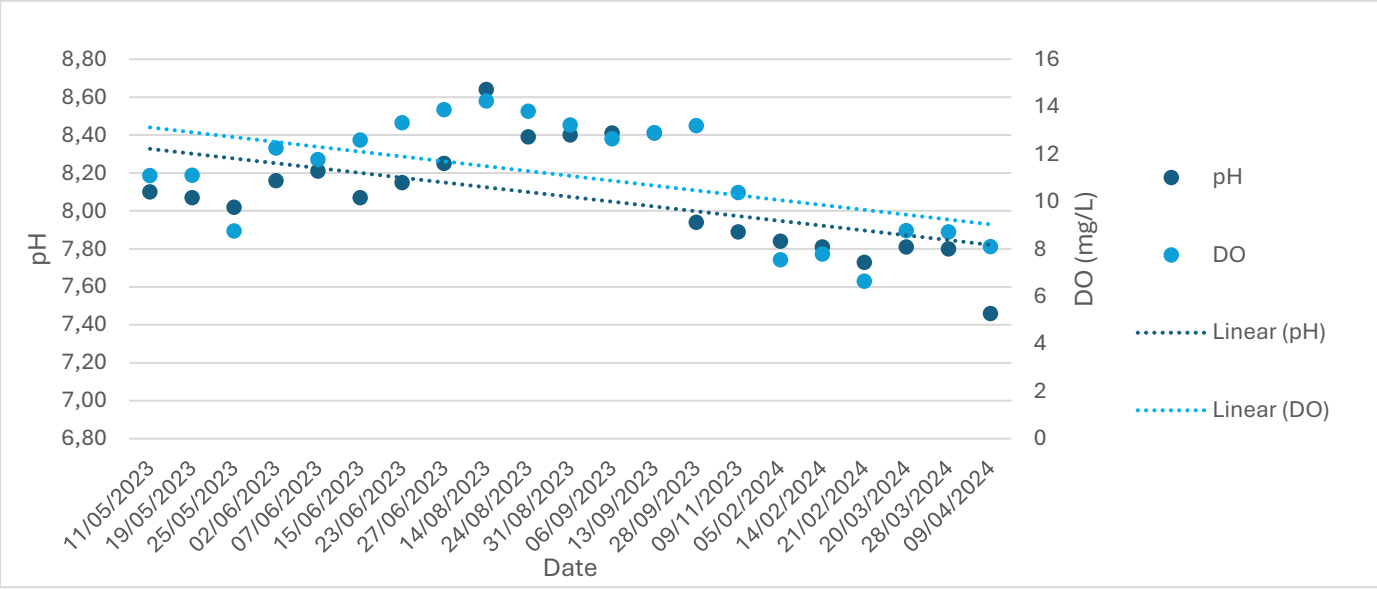


Figure 14: pH and DO analysis from the Montgomery Spruit.

The nutrient results from Montgomery Spruit are presented in Figure 15 and Figure 16. The original data is presented in Table 5 in the appendix. The sulphate, nitrate, nitrite and phosphate are presented from the period 11 May 2023 to 9 April 2024. The data from the field measurements does not present concerning levels of pollutions, however, there is a nitrate peak and a nitrite peak (1.4 mg/L) which is seen in some months and corresponds with *E. coli*, seen in Figure 15 and Figure 16, in the same period. It is highly likely that these nutrients originate from untreated sewage entering the river as peaks

overlap. Whilst there are high sulphate values specifically in the latter part of the year (30 and 16 mg/L) the concentrations are generally low, and therefore, they are not of concern. Phosphate concentrations were constant throughout the study period with a spike in phosphate concentration recorded on the 25th of May and two significant increases over 25 mg/L recorded on the 6th and 13th of September 2023, before returning to below 5 mg/L for the remainder of the study.

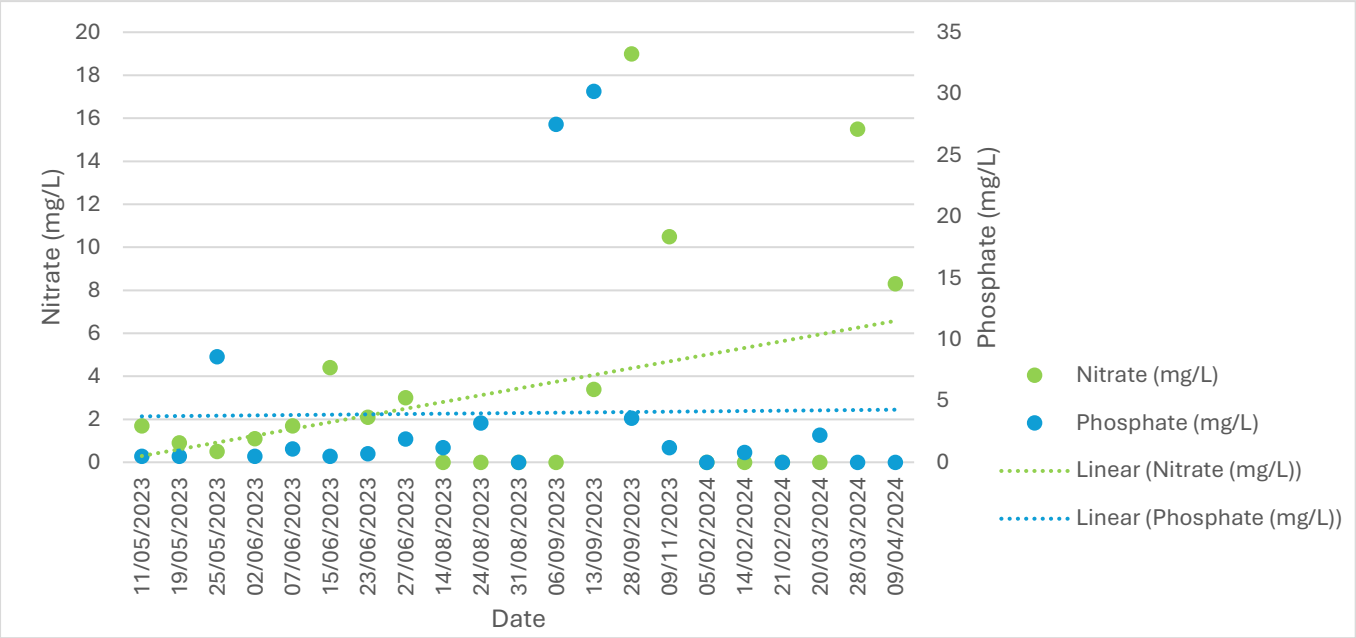


Figure 15: Nitrate and Phosphate analysis from samples results collected from the Montgomery spruit.

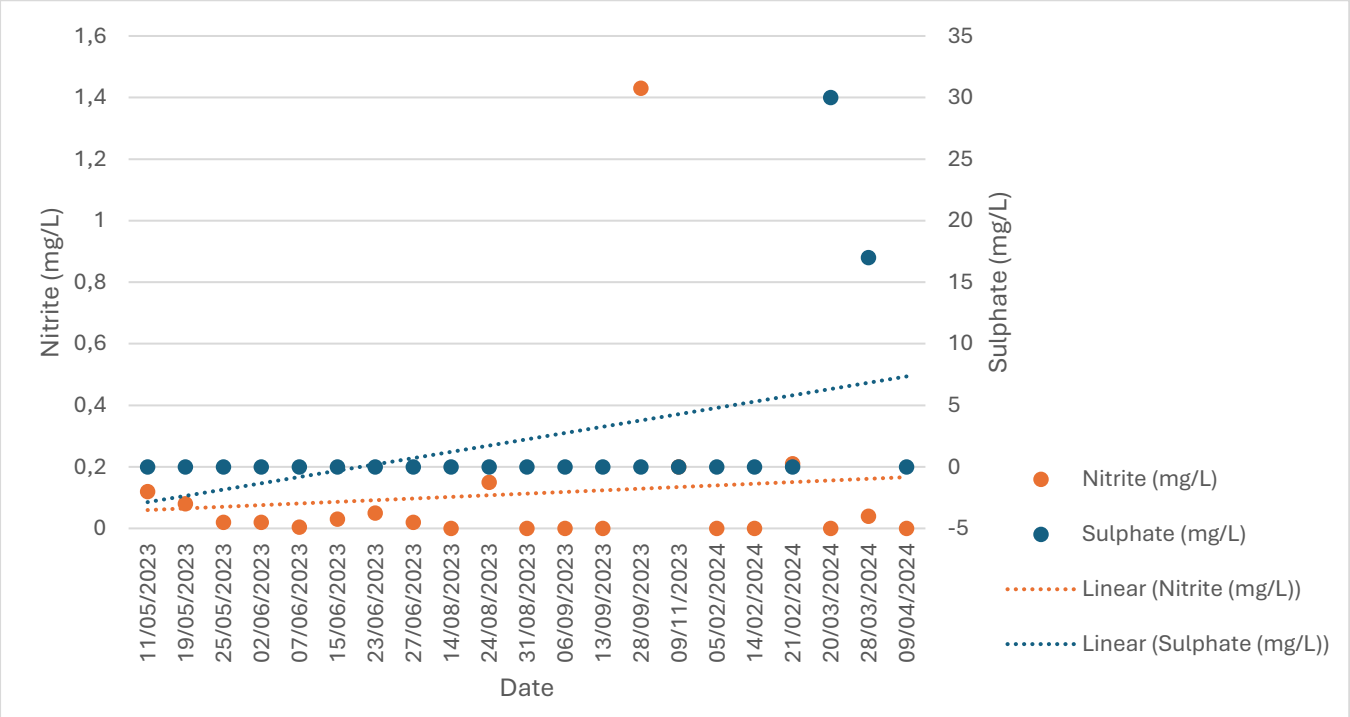


Figure 16: Nitrite and Sulphate analysis from samples results collected from the Montgomery spruit.

The *E. coli* results from Montgomery Spruit are presented in Figure 17 from the period 11 May 2023 to 9 April 2024. The original data is presented in Table 5 in the appendix. The *E. coli* and coliform concentrations that are shown in Figure 17 are significant in two periods, September 2023 and April 2024. According to Wang et al. (2018), in the literature review, it is highly likely that this nutrient loading is a consequence of stormwater runoff because they found similar values at in the rainfall seasons with spike of *E. coli* and nutrients like nitrates and nitrites (Wang et al., 2018). They also found a correlation between *E. coli* and total coliforms, which was observed in this study on the Braamfontein Spruit (Wang et al., 2018). According to Shelton et al, (2014), the increased *E. coli* and nutrient concentration in the river could be a result of resuspension of sediments containing nutrient and faecal bacteria. Interestingly, there is a negative correlation between pH and coliforms (Figure 55, in the appendix). This could be a consequence of coliform bacteria, including *E. coli*, thriving in neutral water and slightly decreasing pH causing organic matter to breakdown, releasing more nutrients and promoting microbial growth (Nyieku et al., 2022). *E. coli* was found consistently throughout the stud period, indicating persistent microbial contamination, rather than isolated pollution events. Three distinct *E. coli* events were observed on the 31st of August, 28th of September and 9th of April. The September *E. coli* peaks occurred shortly after high phosphate concentrations were recorded, possibly suggesting a link between nutrient enrichment and microbial proliferation.

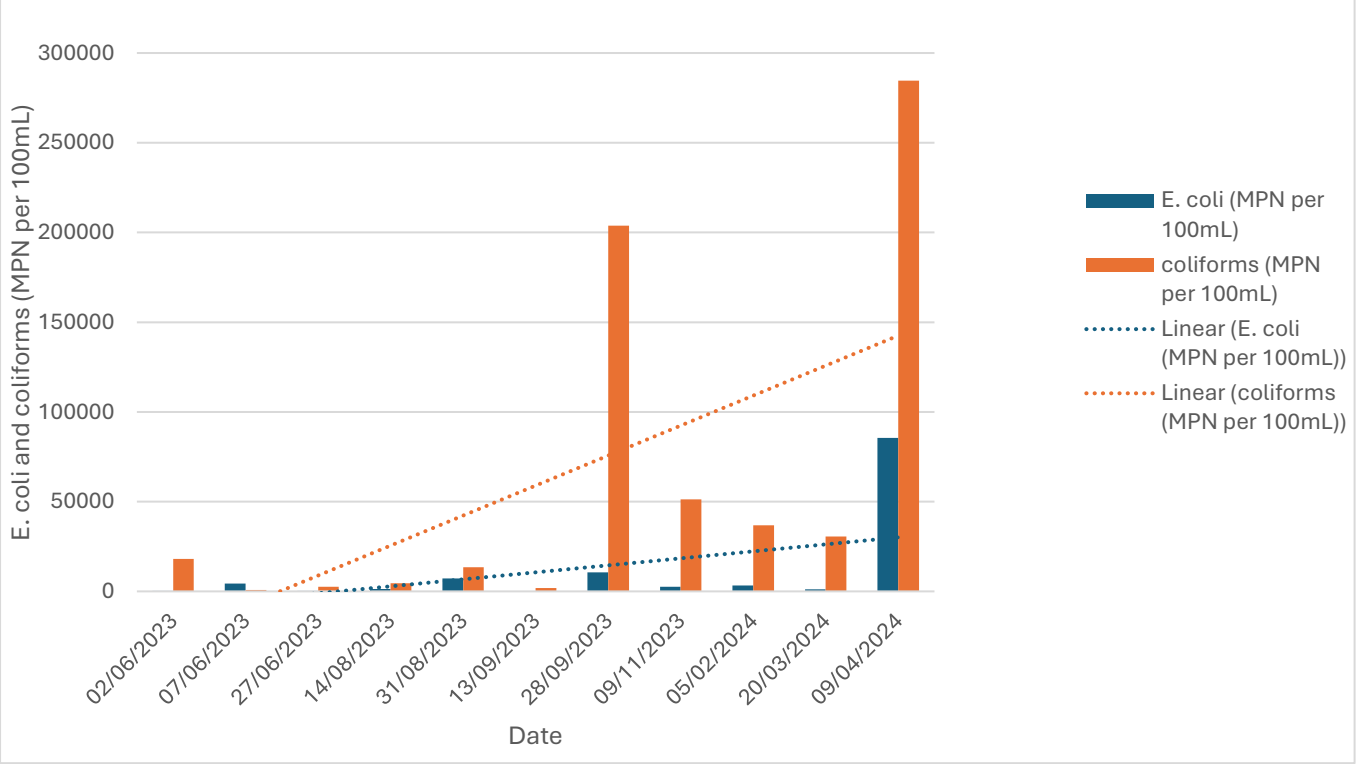


Figure 17: The E. coli and coliforms results from Montgomery Spruit

The metals from both the water and sediment samples, specifically manganese, iron and lead, are presented in Table 6 in the appendix. The concentration of each metal found in Figure 18 are well below

the thresholds set out by the South African DWS to preserve aquatic life of 0.01 mg/L, indicating that there was no contamination observed during the sampling period. The data in the sediment samples, indicated in Figure 16 is highly variable with manganese, iron and lead all exceeding their recommended thresholds in May and June 2023 and April 2024. This data, specifically manganese and lead exceed aquatic environment sediment quality guidelines for protection of the environment set out by the US Government EPA (1999), providing a limit of 300 mg/kg for Manganese and the ISQG providing a limit of 35 mg/kg for lead. There was over 900 mg/kg of manganese, almost 12000 mg/kg of iron and almost 300 mg/kg of lead recorded on the same day. This could have detrimental effects on the environment, biodiversity and human health. Since there are no set out guidelines available for iron concentrations since it relies on background data, a threshold of 10 000 mg/kg was selected based off similar studies done in South Africa (Newman & Watling, 2007). The iron concentration appears to be high through the end of winter into spring before decreasing over summer and increasing towards Autumn again. This could be due to rainfall events washing the iron concentrations further downstream, or dilution from fresh sediment inputs being washed into the sampling sites, as there was higher rainfall in the summer periods than in the winter periods, verified on Weather Directory (Johannesburg Precipitation & Humidity by Month | Rain & Snow Averages). The results for the metals concentrations in the Montgomery Spruit were extremely low for all metals throughout the sampling period, showing no significant metal contamination in the river water. The Manganese results from the sediment sample showed consistent manganese presence in the sediment with significant increases being observed on the 25th of May 2023, the 7th of June 2023 and the 9th of April 2024. Iron concentration in the sediment was consistently high, with a significant trough recorded on the 14th of August 2023. The lead concentration in the sediment remained consistent throughout the sampling period with a significant trough being observed on the 14th of February 2024. The manganese concentrations observed throughout the study show that the manganese exceeded concentrations three times throughout the study. The lead exceeded the TEL guidelines every week that the sediments were measured for lead but were low on the 14th of February 2024 compared to the other days. The patterns of pollution and contamination indicate that while metal sulphate concentrations remained low, the Montgomery Spruit faces greater risks from diffuse microbial pollution likely driven by surface runoff and organic nutrient loading.

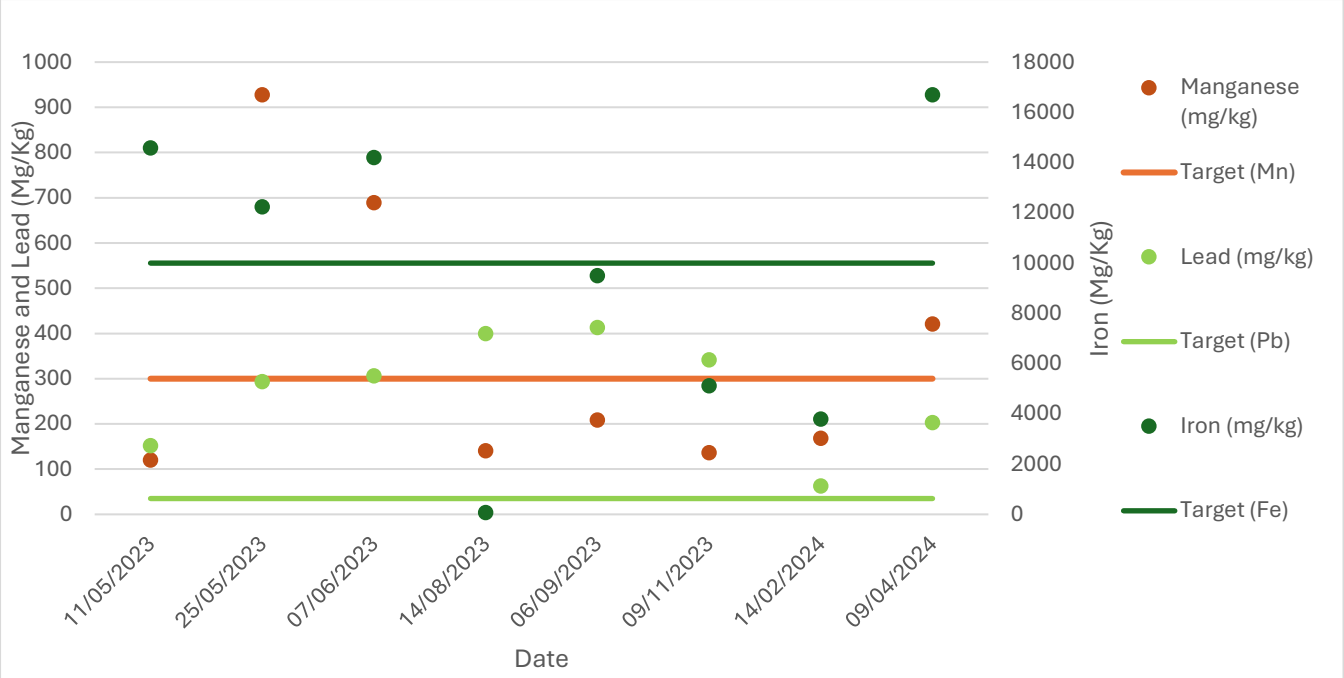


Figure 18: The manganese, iron and lead concentrations from the sediment samples taken from Montgomery Spruit

b. Emmarentia

The field measurements from Westdene Spruit are presented in Figure 19 and 20. The original data is presented in Table 7 in the appendix. The pH, DO, temperature and EC are presented from the period 11 May 2023 to 9 April 2024. The pH in Figure 20 was seen to be circumneutral throughout the study. The temperature increases from a minimum in June to a maximum in February and the range for temperature is from 8.7˚C to 20.4˚C. The DO varied slightly with an increase in colder temperatures and a decrease in warmer temperatures, however the river remains somewhat oxygenated throughout the study. The EC increases and is at its peak in June 2023. The DO has a positive correlation to pH (Figure 56 in the appendix) and the pH has a negative correlation with temperature (Figure 57 in the appendix), with higher pH being recorded in winter. The data from this graph indicates low levels of impact from urban activity.

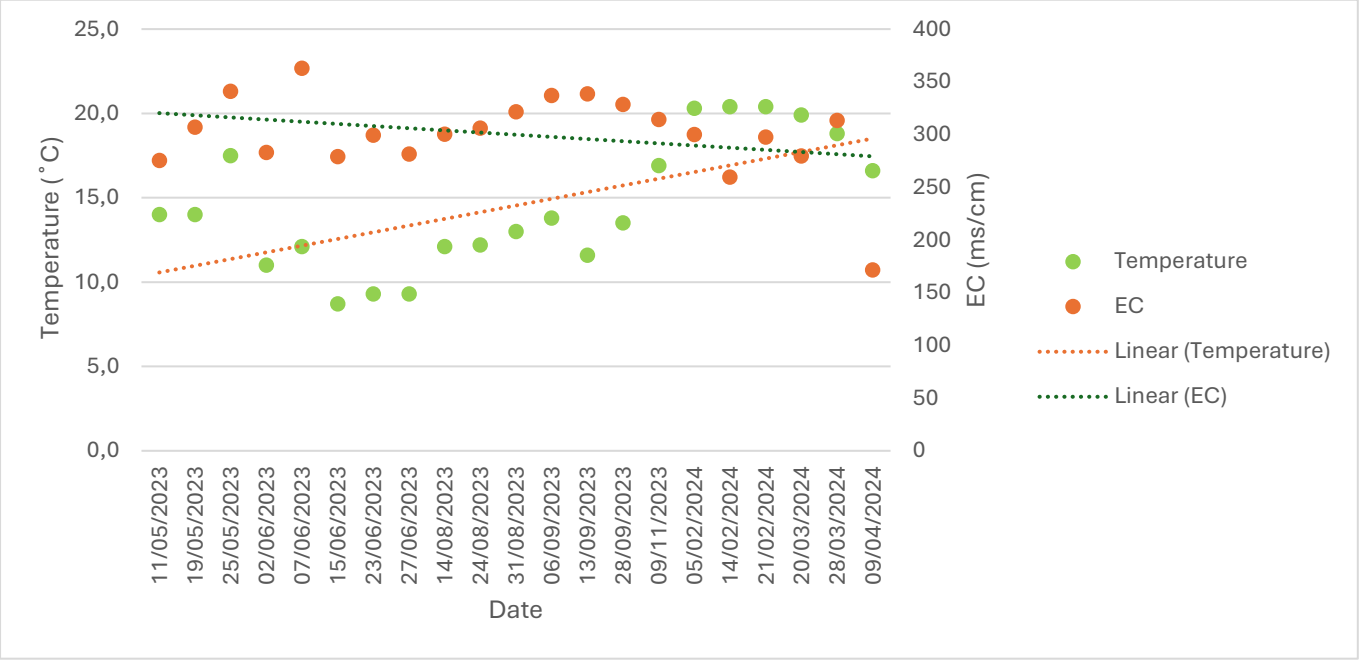


Figure 19: Temperature and EC analysis from the Westdene Spruit.

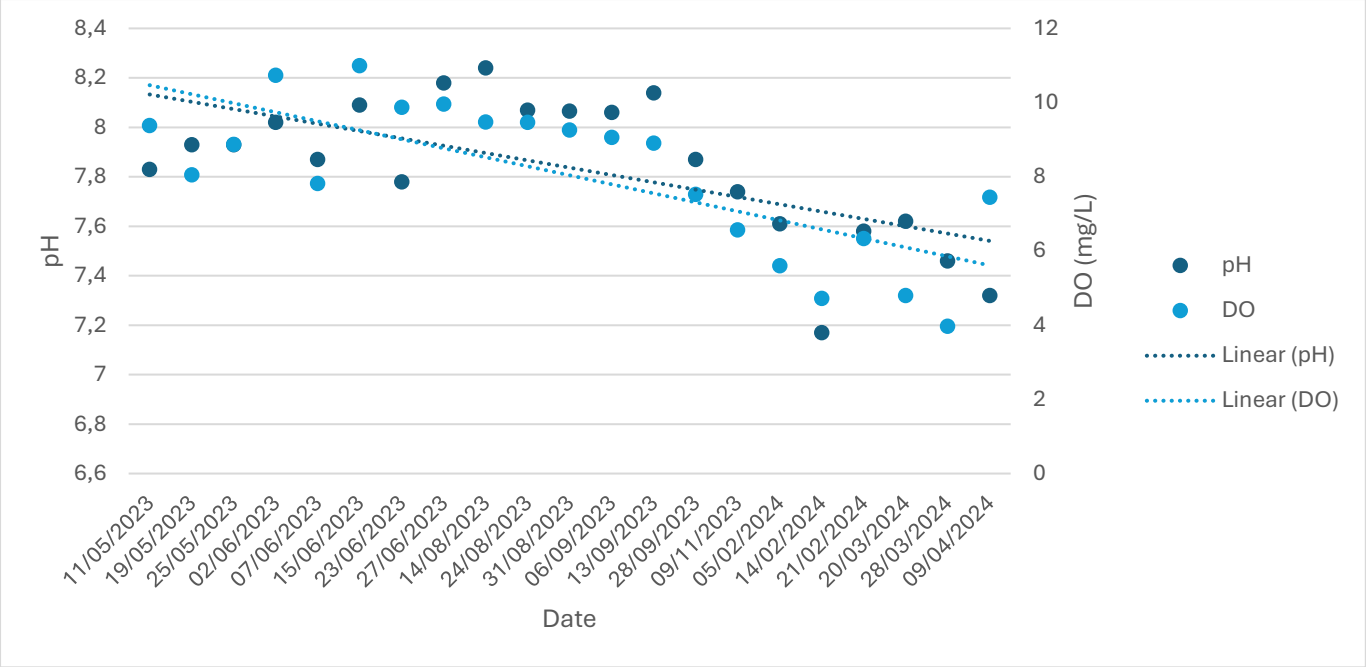


Figure 20: pH and DO analysis from the Westdene Spruit.

The nutrient results from Westdene Spruit are presented in Figure 21 and Figure 22. The original data is presented in Table 7 in the appendix. There were no nutrients recorded in the first half of the study in the Westdene Spruit. The data is highly variable with two significant increases of nitrate being observed on the 6th and 28th of September 2023 of over 1 mg/L and over 2 mg/L, and very low nitrate concentrations through the rest of the study. The maximum concentration for nitrates was 15 mg/L, which is not a concern. The nitrite values were very low throughout the first half of the study, before an increase was observed from September 2023, but are below DWS limits (1996) and do not cause any concern. The phosphate values were highly variable, with no phosphate being recorded in the first half of the study and two significantly high values being recorded on the 14th of August 2024 and the 6th of September 2023 in comparison to the other values recorded in this river. The values are still much lower than the phosphate concentrations recorded in other rivers, but still greatly exceed the DWS guidelines for phosphate in aquatic ecosystems (Silberbauer, 1996). The sulphate had three high values recorded from August to September 2023 and two significantly high concentrations recorded in March 2024.

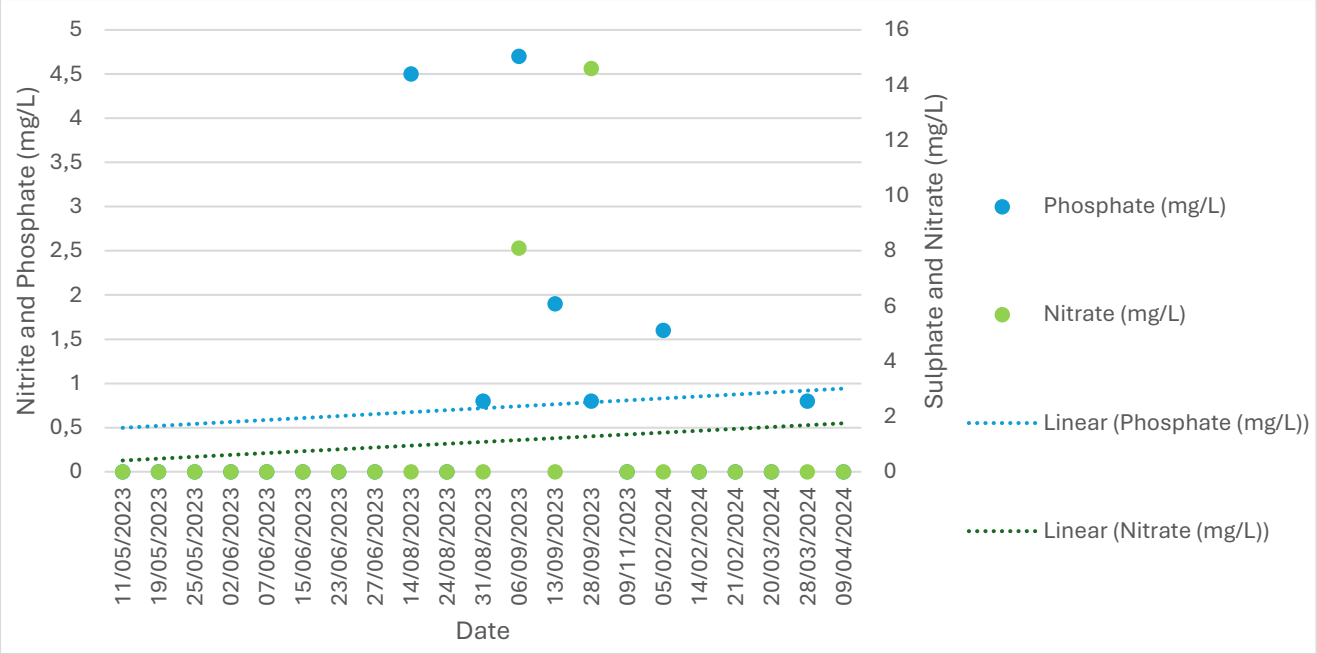


Figure 21: The nitrate and phosphate results from The Westdene Spruit at the Emmarentia Botanical Gardens

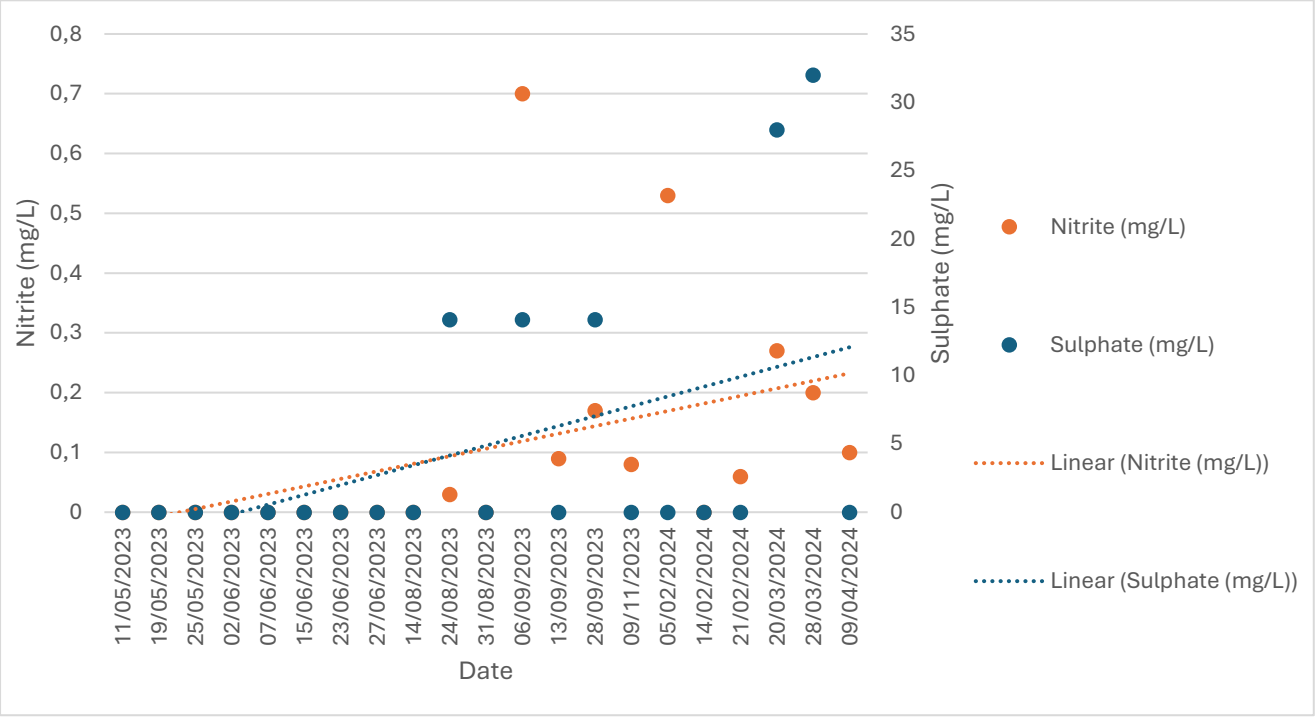


Figure 22: The nitrite and sulphate results from The Westdene Spruit at the Emmarentia Botanical Gardens

The *E. coli* results from Westdene Spruit at Emmarentia are presented in Figure 23 The original data is presented in Table 7 in the appendix. Unlike the Montgomery Spruit, in Figure 17, there is an *E. coli* and coliform peak, but there is no nutrient increase observed in the same period indicated in Figure 21 and Figure 22. On the 28th of September 2023, there was an increase in *E. coli* and coliforms, as well as an increase in nitrate and sulphate. It could also be likely that these nutrients originate from untreated sewage entering the river as peaks overlap, or as a result of dog or bird droppings entering the river

when sampling occurred. Whilst there are high sulphate values in the middle of the year, but specifically in the latter part of the year, the concentrations are generally low, and therefore, they are not of concern. The *E. coli* and coliform concentrations that are shown in Figure 23 are high on the 28th of September, matching high nutrient results on the same day, and significantly high on the 9th of April 2024, which had low nutrient concentration recordings. It is hypothesised that this contamination is not necessarily caused by systemic sewage ingress from leaky infrastructure, but rather, it is a consequence of indigent people using the Spruit as an open toilet, or stormwater drainage from unplanned settlements (Vollmer & Grêt-Regamey, 2013).

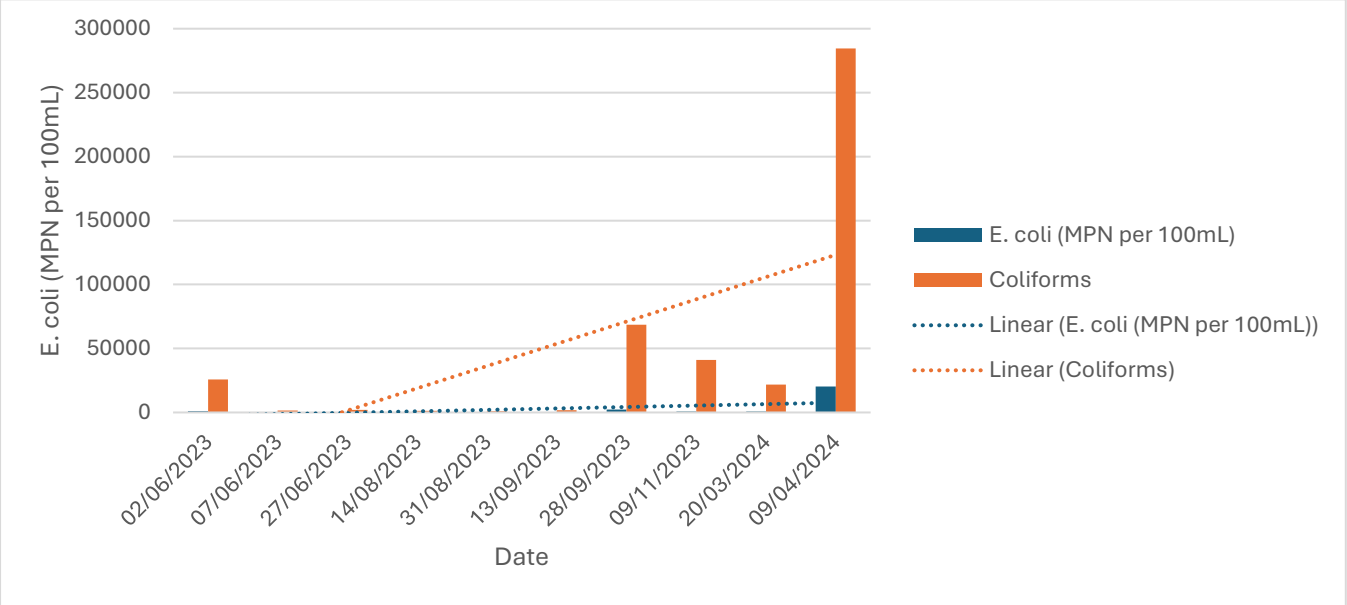


Figure 23: The E. coli and coliforms results from the Westdene Spruit in Emmarentia

The metals specifically manganese, iron and lead are presented in figure 24. The original data is presented in Table 8 in the appendix. The concentration of each metal found in Figure 24 are well below the thresholds set out by the South African DWS to preserve aquatic life of 0.01 mg/L, indicating that there was no contamination observed during the sampling period (Silberbauer, 1996). The data in the sediment samples, indicated in Figure 20, is highly variable with manganese exceeding 680 mg/kg in August 2023 and lead exceeding 390 mg/kg on the same day. This data, specifically manganese and lead exceed aquatic environment sediment quality guidelines for protection of the environment set out by the US Government EPA (1999), providing a limit of 300 mg/kg for Manganese and the ISQG providing a limit of 35 mg/kg for lead. The Manganese concentration from the sediment samples was inconsistent throughout the study with significant increases on the 25th of May 2023, 14th of August 2023 and 9th of September 2023. Every week that the sediment samples were tested for lead, the samples exceeded the TEL guidelines, except for the 14th of February 2024, which was below the limit. The Montgomery Spruit experienced its lowest concentration of lead on the same day. Since there is no background concentration for iron at these sites, and there is not a guideline for iron limits, it cannot be

stated that the iron concentration was exceeding natural limits. However, based off other studies done in South Africa on iron concentration, a tentative limit of 10 000 mg/kg was used for this study as a reference point. Based off this limit, iron was observed to exceed these limits twice, once in May 2023 and once in November 2023 and was on the limit in June and August 2023. The results for the metals concentrations in the Westdene Spruit were extremely low for all metals throughout the sampling period, showing no significant metal contamination in the river water. There were consistently high iron concentration levels in the sediment samples with two significant increases on the 25th of May and the 9th of November 2023. The lead concentration remained consistent throughout the study period with a significant trough on the 14th of February 2024 and a low concentration on the 9th of April 2024.

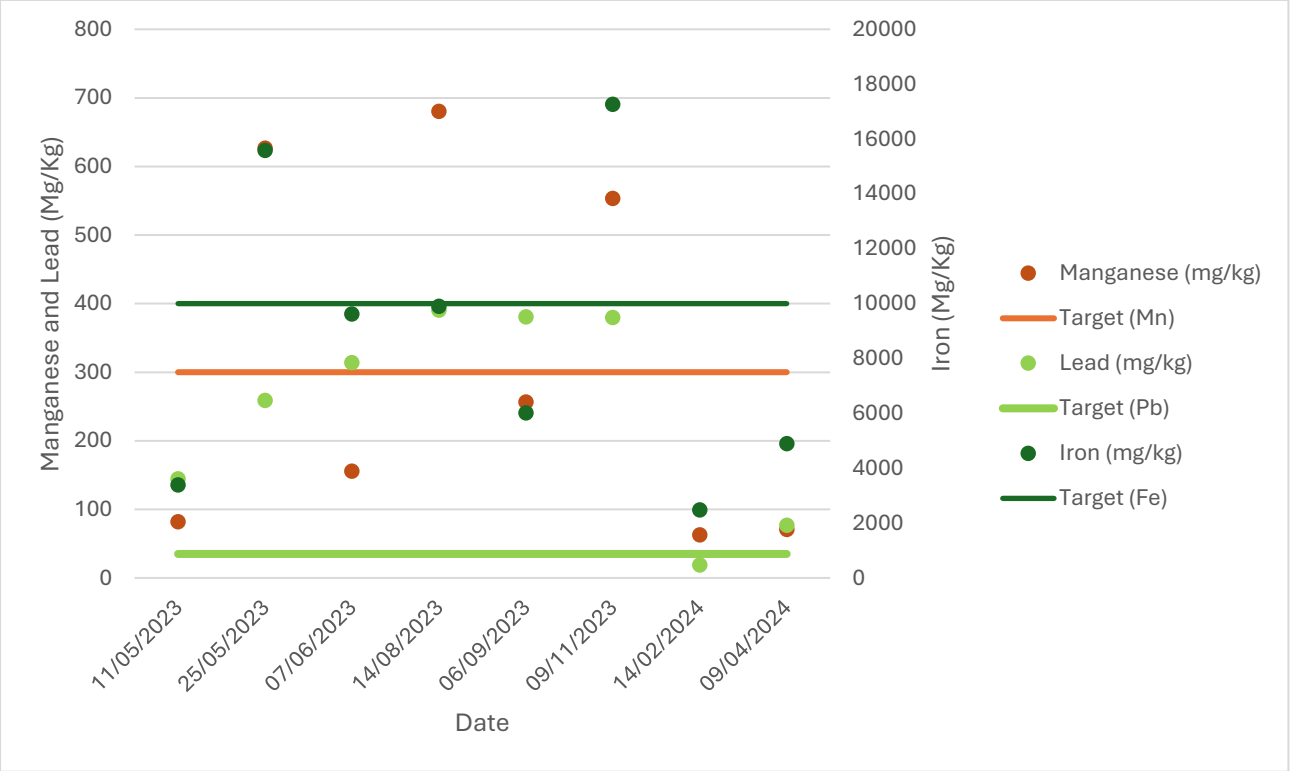


Figure 24: The manganese, iron and lead concentrations from the sediment samples taken from the Westdene Spruit

c. Parkview

The field measurements from Braamfontein Spruit at Parkview Golf Course are presented in Figure 25 and Figure 26. The original data is presented in Table 9 in the appendix. The pH, DO, temperature and EC are presented from the period 11 May 2023 to 9 April 2024. Similarly to the Westdene Spruit and the Montgomery Spruit, there are no outliers or unexpected data for this site over the sampling period. The ranges of the data are very similar in that the pH is circumneutral, with dissolved oxygen ranges between 6 and 10 mg/L, temperature ranges between 12 and 22˚C and EC ranges from 250 to 400 µs/cm. The EC and DO are again inversely proportional, with DO decreasing when EC increases

towards the end of 2023 and start of 2024. The pH and DO have a positive correlation (Figure 58 in the appendix), with pH having a negative correlation to temperature and EC (Figure 59 in the appendix). EC and DO have a negative correlation (Figure 60 in the appendix).

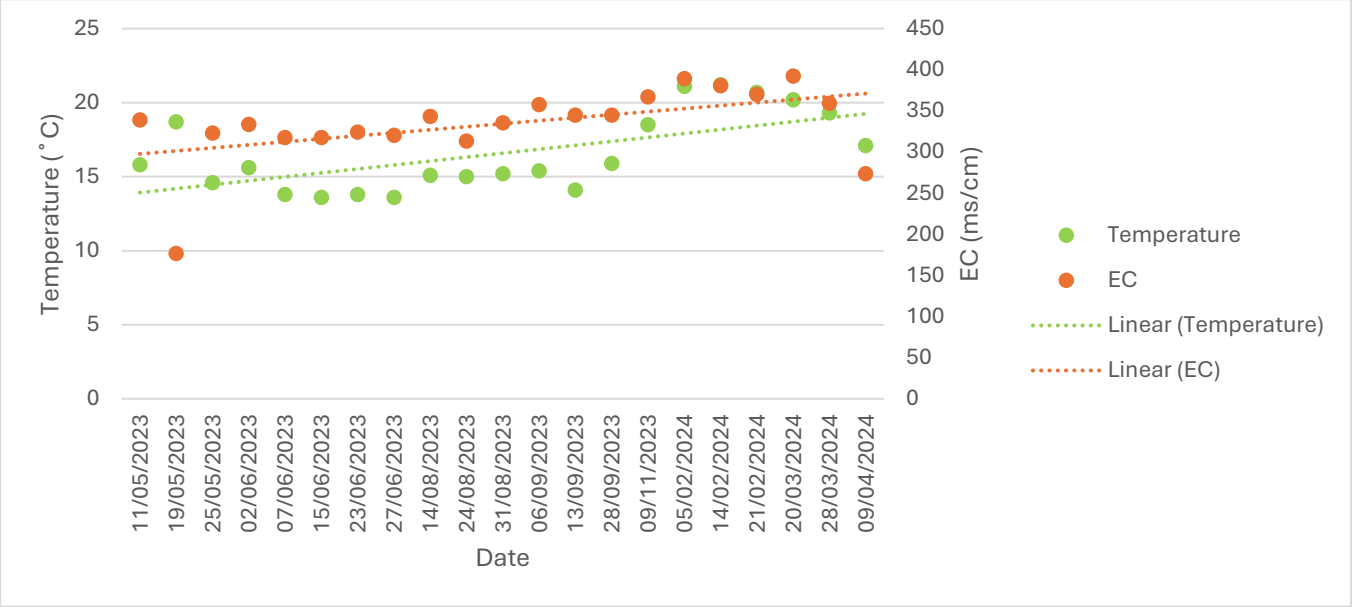


Figure 25: The Temperature and EC from The Braamfontein Spruit at the Parkview Golf Course

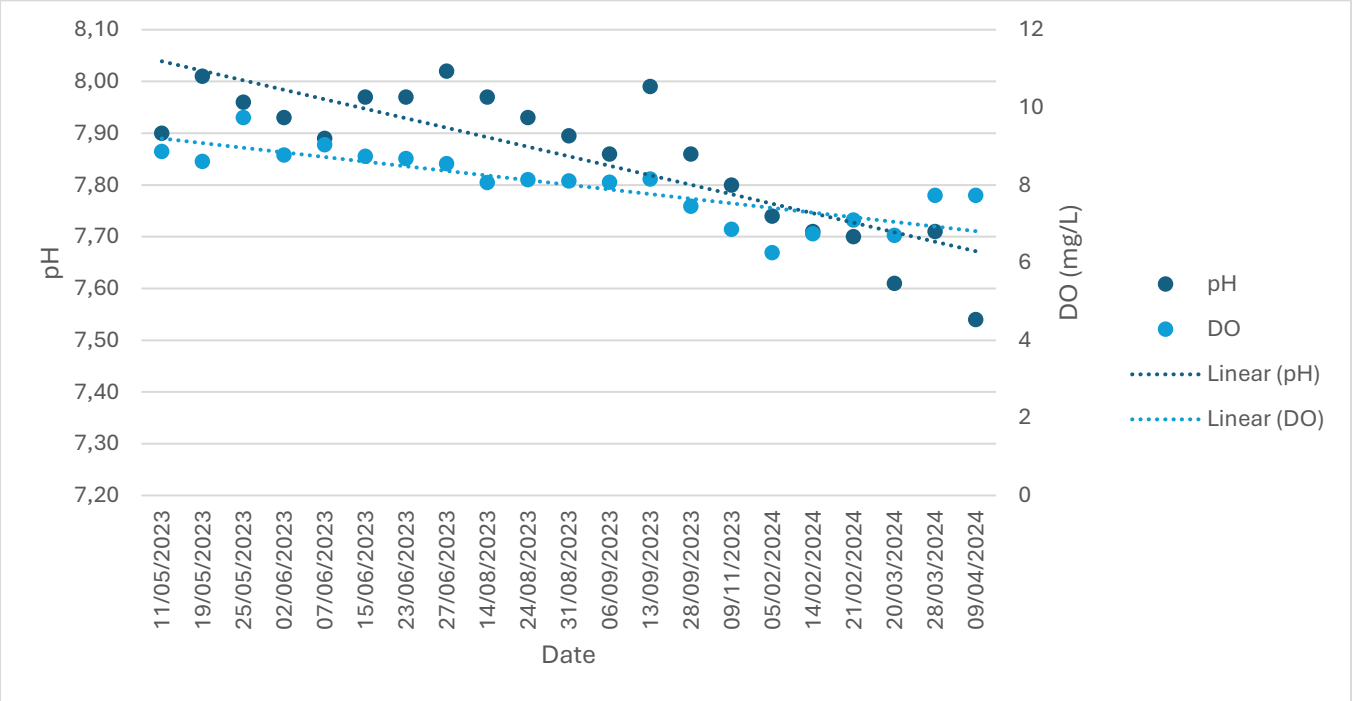


Figure 26: The pH and DO from The Braamfontein Spruit at the Parkview Golf Course

The nutrient results from Braamfontein Spruit at Parkview Golf Course are presented in Figure 27 and 28. The original data is presented in Table 9 in the appendix. The pH, DO, temperature and EC are presented from the period 11 May 2023 to 9 April 2024. In Figure 22, the nutrient loadings are presented

are similar to the findings from the other sites with low nitrite and nitrate values, with observable increases in nitrate being observed in September and November 2023, reaching a maximum of almost 20 mg/L. Phosphate had significant concentrations observed, up to 7 mg/L, greatly exceeding the DWS guidelines (Silberbauer, 1996). The sulphate results are not linear, with high values being recorded at the start of the study and again towards the end of the study, with lower concentrations recorded in between. The sulphate reached a maximum of slightly over 50 mg/L in May 2023 and April 2024, indicating a large possibility of pollution entering the stream that day.

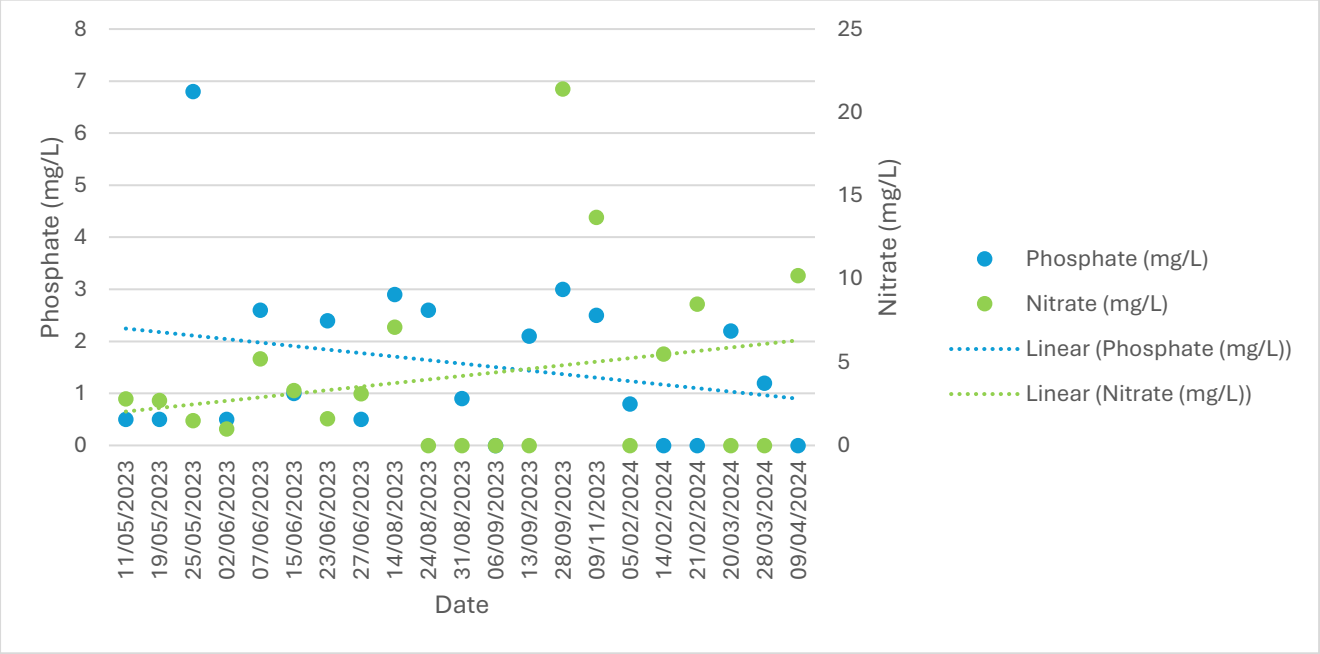


Figure 27: The Phosphate and Nitrate results from The Braamfontein Spruit at the Parkview Golf Course

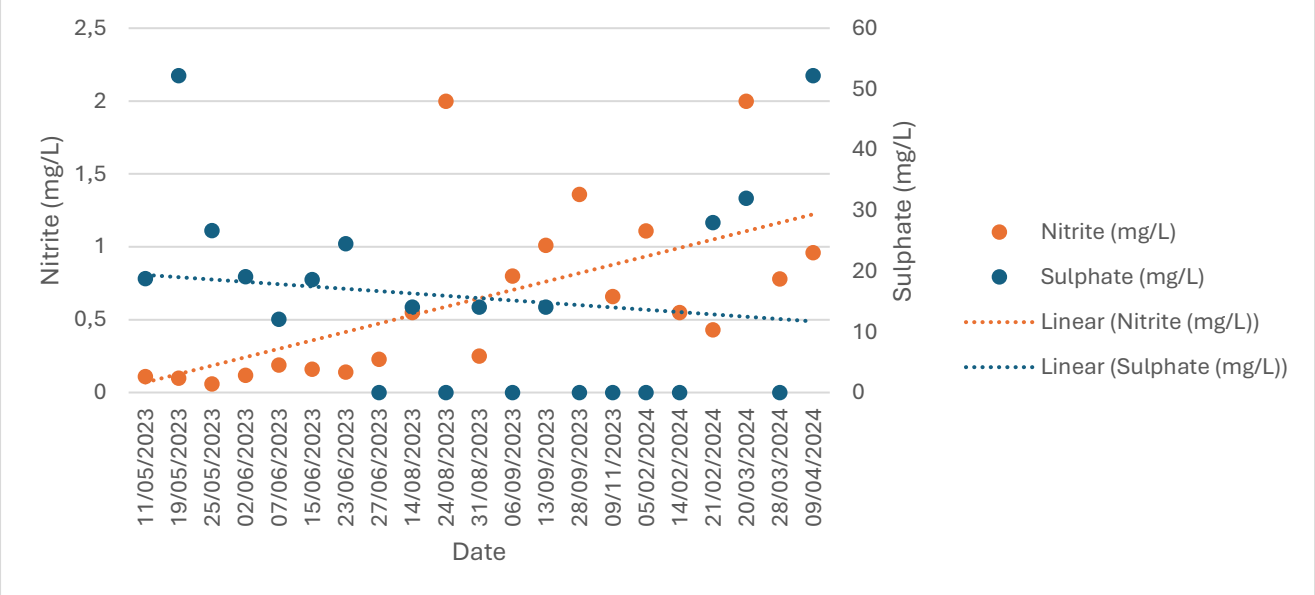


Figure 28: The Nitrite and Sulphate results from The Braamfontein Spruit at the Parkview Golf Course

The *E. coli* results from Braamfontein Spruit at Parkview Golf Course are presented in Figure 29. The original data is presented in Table 9 in the appendix. The pH, DO, temperature and EC are presented from the period 11 May 2023 to 9 April 2024. Surprisingly, the *E. coli* and total coliform values, presented in Figure 23, do not correspond to a similarly observable increase in the nutrient loading spread on the same day at the same site. As shown in Figure 23, the *E. coli* and coliform concentrations are significant with faecal coliforms regularly exceeding 300000 MPN per 100 mL, and the total coliforms reaching as much as 800000 MPN per 100 mL for substantial periods of the year.

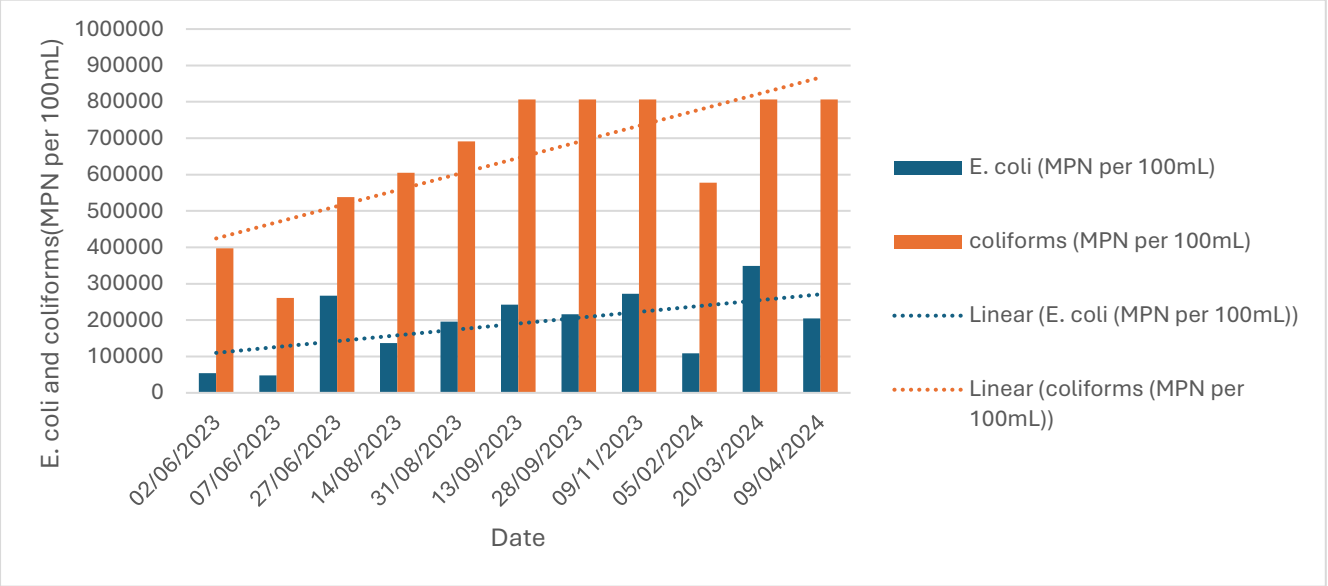


Figure 29: The E. coli and coliform results from The Braamfontein Spruit at the Parkview Golf Course

The metals in the water, specifically, manganese, iron and lead sampled from the Braamfontein Spruit are presented in table 10 in the appendix. The concentration of each metal found in Figure 30 was very low, indicating that there was no contamination observed during the sampling period. The data in the sediment samples, indicated in Figure 24 is lower than the Mongomery and Westdene Spruits, with the maximum manganese concentration being only 220 mg/kg in February 2024. The lead concentration was highly variable throughout the study period with an obvious increasing trend between May and September 2023, exceeding 380 mg/kg at the maximum concentration in September. The lead concentration then reduced significantly from November 2024 to the end of the study period, reaching its minimum in February 2024, similarly to all other sites, before slightly increasing again. The lead concentrations greatly exceed aquatic environment water quality guidelines for protection of the environment set out by the US Government EPA 1999. The iron concentrations were somewhat consistent throughout the sampling period with a maximum exceeding 4900 mg/kg in May 2023 and a minimum in September 2023 of 1615 mg/kg. When compared to the Westdene Spruit and the Montgomery Spruit, these concentrations are significantly lower. Since there is no background concentration for iron at these sites, and there is not a guideline for iron limits, it cannot be stated that

the iron concentration was exceeding natural limits. There may also be iron-rich rocks in the stream which increase the natural iron concentration in the river.

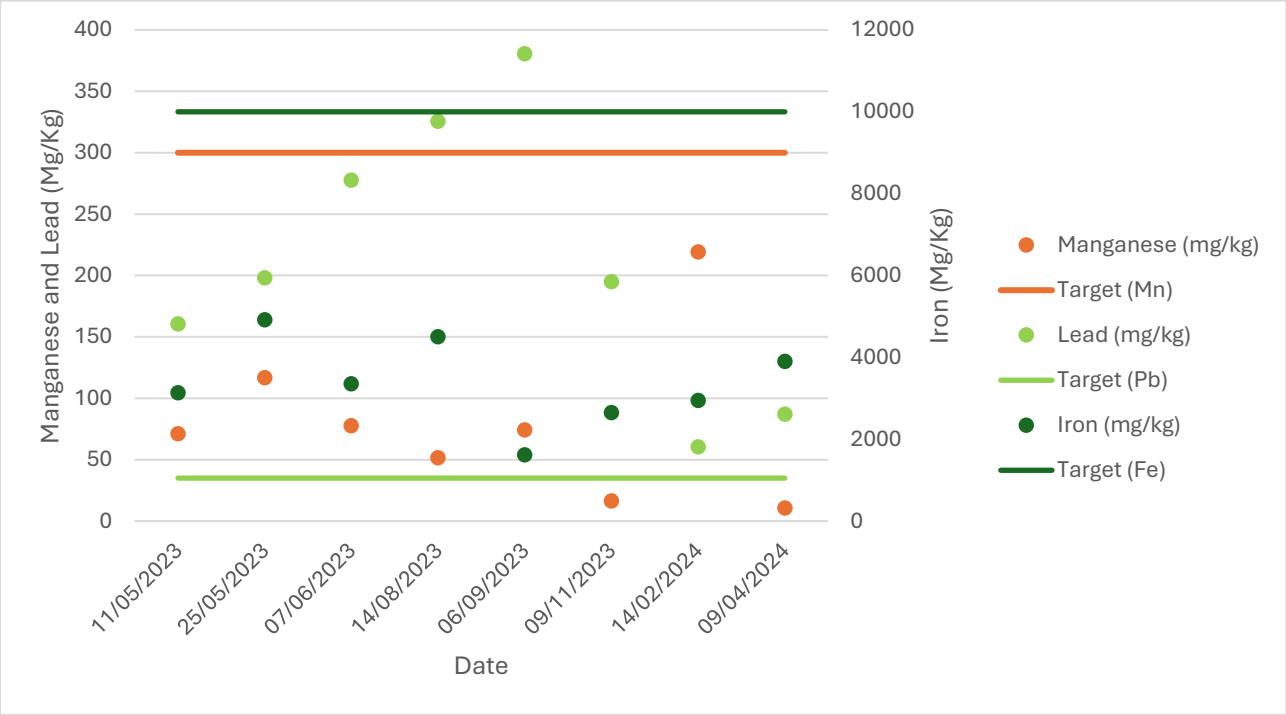


Figure 30: The manganese, iron and lead concentrations from the sediment samples taken from the Braamfontein Spruit at Parkview Golf Course

d. Delta Park

The field measurements from Braamfontein Spruit at Delta Park are presented in Figure 31 and Figure 32. The original data is presented in Table 11 in the appendix. The pH, DO, temperature and EC are presented from the period 11 May 2023 to 9 April 2024. Similarly to the other sites, the pH remains circumneutral throughout the study period, the DO increases in the colder winter months, with a maximum of 15 mg/L and a minimum in late summer of just over 6 mg/L. The pH and DO have a positive correlation (Figure 61 in the appendix). The temperature changes are consistent with the expected seasonal variations and ranges from 10° to just above 20°C. The EC, however, is variable, with inconsistent measurements throughout the study and low values being recorded in February and April 2024. The EC varies between 200 and 400 µs/cm and the water is oxygenated changing from a maximum of almost 15 mg/L corresponding to the midwinter value with the minimum in the late summer with a value of just over 6 mg/L.

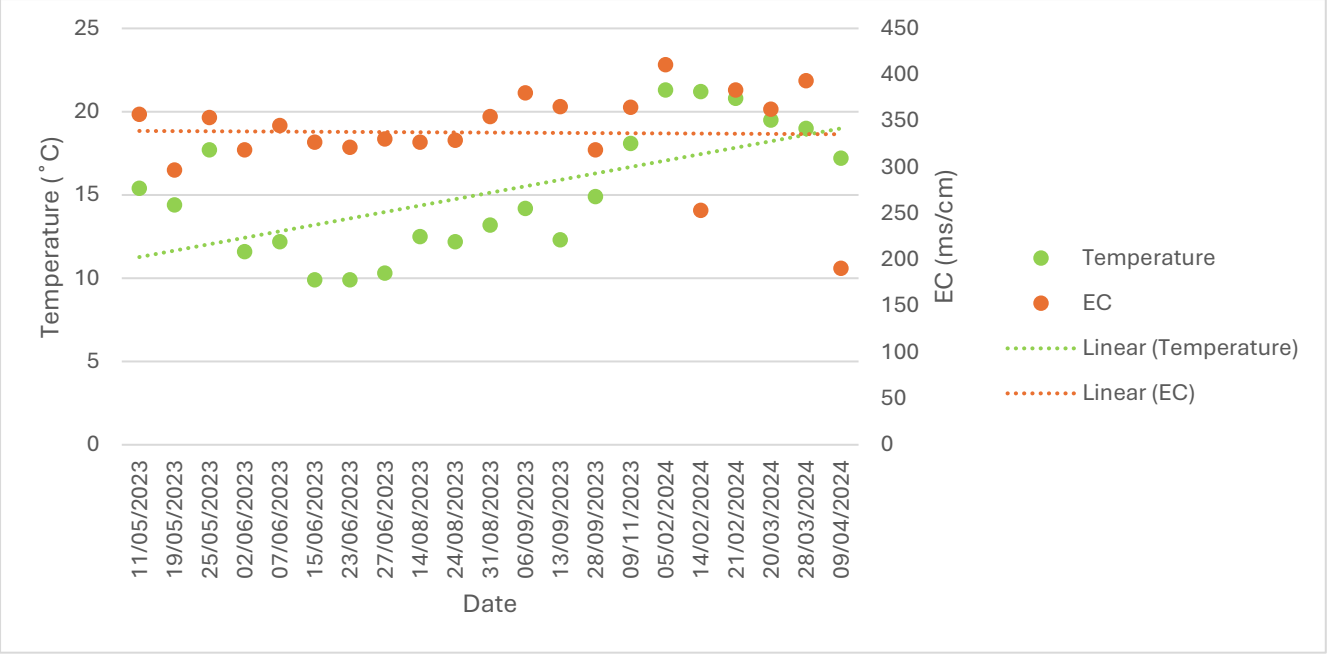


Figure 31: the Temperature and EC results from The Braamfontein Spruit at Delta Park

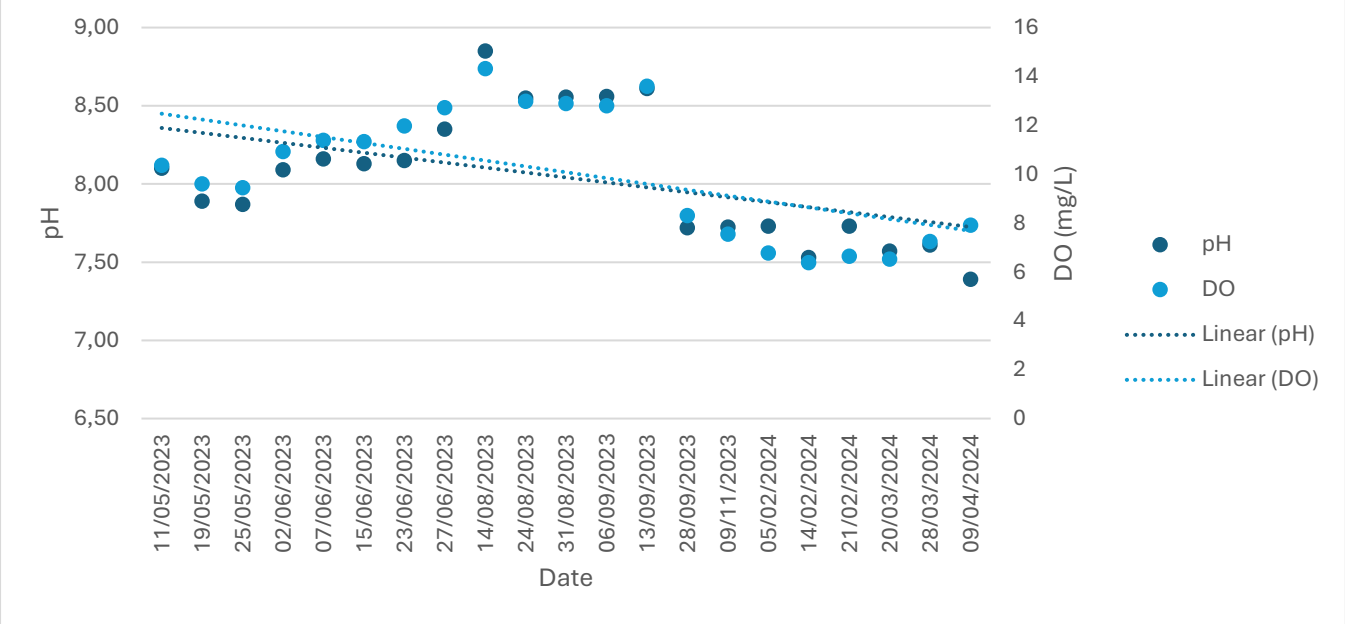


Figure 32: The pH and DO results from The Braamfontein Spruit at Delta Park

The nutrient results from Braamfontein Spruit at Delta Park are presented in Figure 33 and Figure 34. The sulphate concentrations range from zero to just above 50 mg/L. This is the highest sulphate value (52 mg/L) recorded and is significantly higher than the Montgomery and Westdene Spruits. The nitrite and phosphate concentrations were low compared to other sites with a maximum nitrite value less than 1 mg/ L and a maximum phosphate value of just over 3 mg/L. This is lower than many of the other sites but still exceeds DWS guidelines for aquatic ecosystems. The nitrate values were low compared to Parkview (below 3 mg/L at Delta Park and up to 7 mg/L at Parkview), perhaps due to dilution from the

Westdene and Montgomery Spruits. The nitrate results indicated peaks of higher than 50 mg/L and in this instance these data correspond to *E. coli* values.

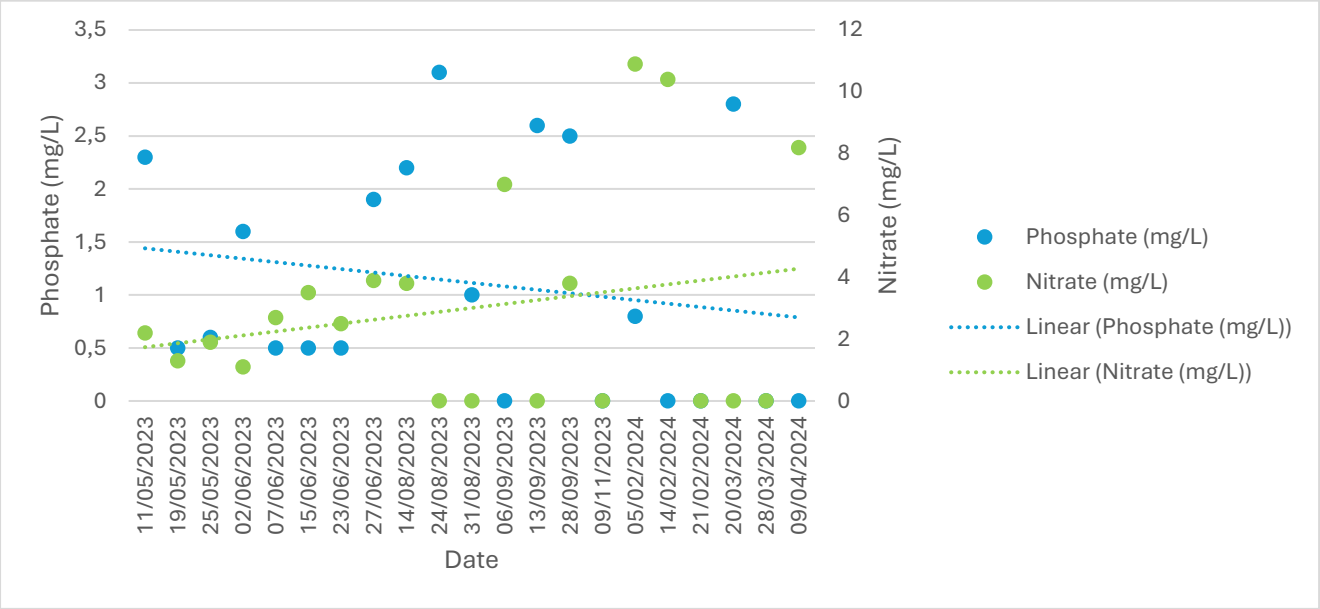


Figure 33: The Phosphate and Nitrate results from The Braamfontein Spruit at Delta Park

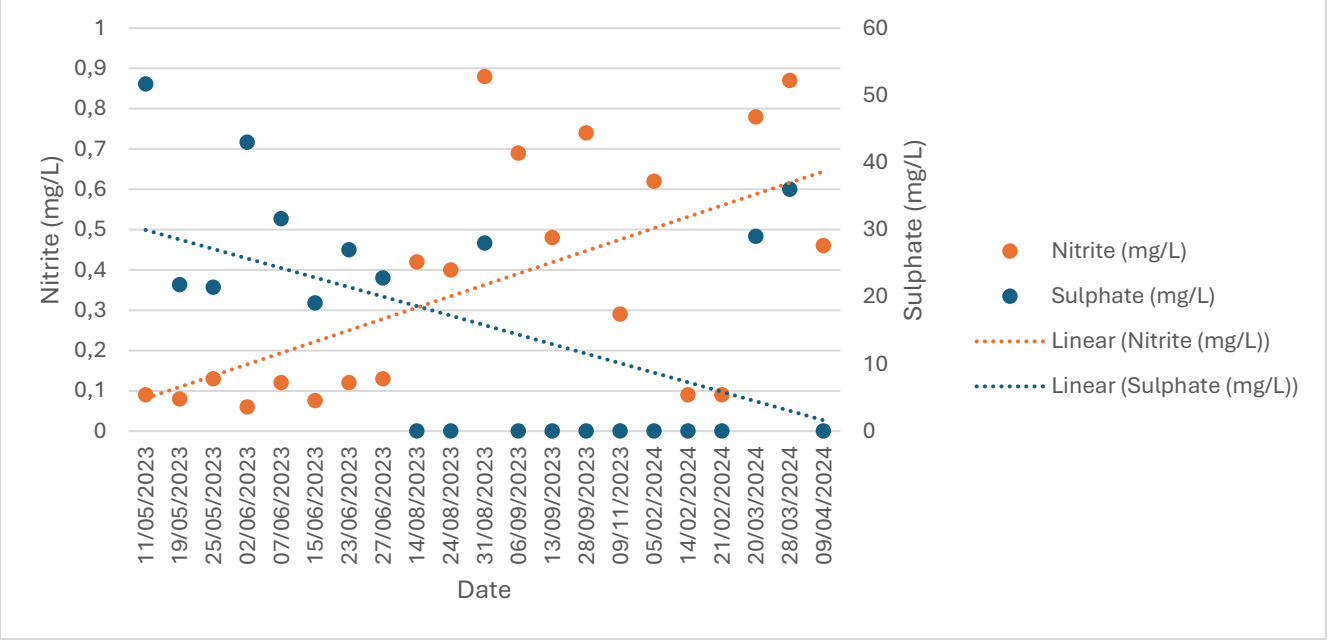


Figure 34: The Nitrite and Sulphate results from The Braamfontein Spruit at Delta Park

The *E. coli* results from Braamfontein Spruit at Delta Park are presented in Figure 35. The original data is presented in Table 11 and shows the results from 11 May 2023 to 9 April 2024. The *E. coli* was consistently present throughout the study period, but not significantly high. However, there was an increase observed in June, and a substantial increase observed in September 2023. The coliforms were high at this site (three values recorded at 300000 MPN per 100 mL), and the *E. coli* was not always high with total coliforms were high, this could be due to the presence of natural coliforms from decaying

plant matter and sediments. The E. coli peaks at 200,000 MPN per 100 milliliters but never gets especially low again whilst the total coliforms are well above 300,000 per 100 milliliters. This data indicates that this water would be unsuitable for swimming as risk of ingestion could cause illness, but general recreation around the dam is suitable.

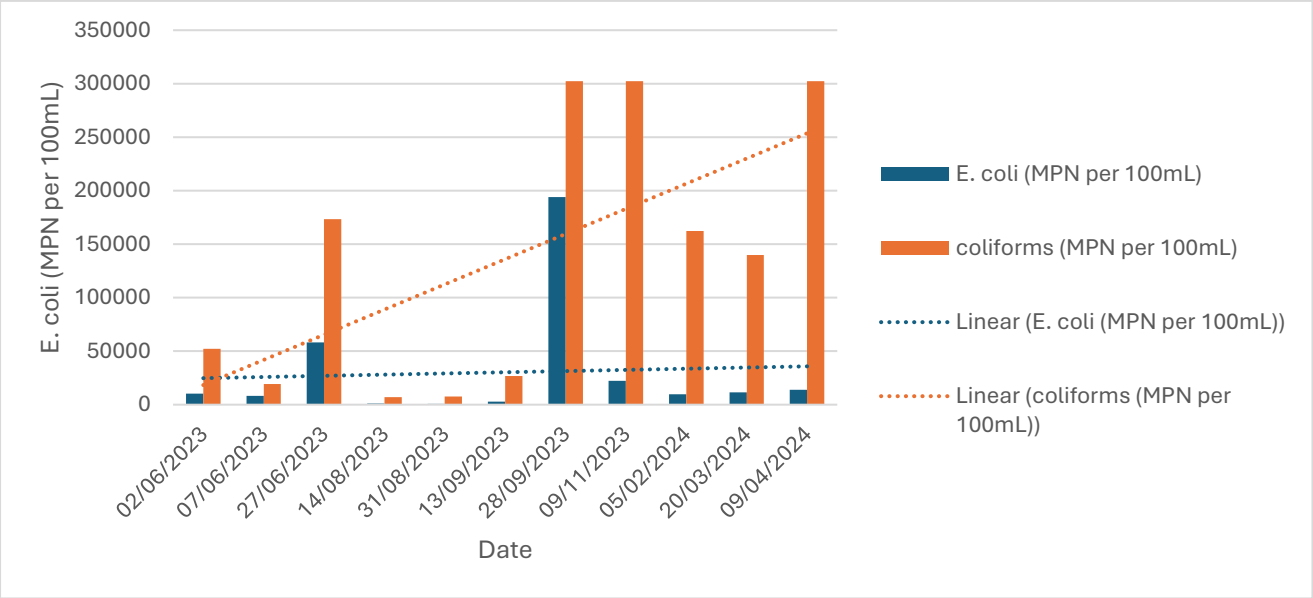


Figure 35: The E. coli and coliform results from The Braamfontein Spruit at Delta Park

The metals in the water, specifically, manganese, iron and lead sampled from the Braamfontein Spruit at Delta Park are presented in table 12 in the appendix and in Figure 36. The concentration of each metal found in the water table was very low, indicating that there was no contamination observed during the sampling period. The metal contaminations in the sediment samples are indicated in Figure 28. The iron and manganese concentrations were well below the defined target of 10,000 mg/kg and 300 mg/kg respectively at this site. The lead, however, is significantly higher than the recommended target quality of 35 mg/kg set out by the ISQG. This could have severe ecological and human health impacts and is cause for concern.

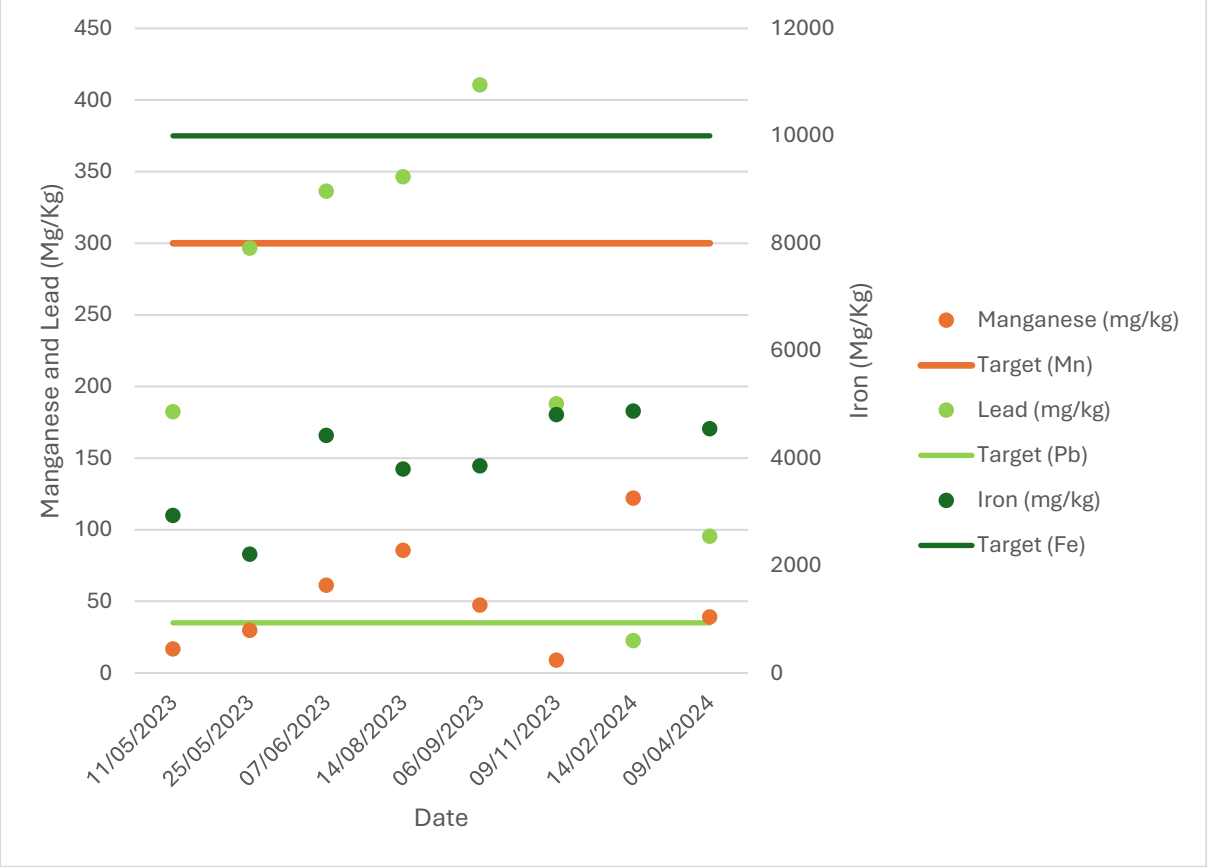


Figure 36:The manganese, iron and lead concentrations from the sediment samples taken from the Braamfontein Spruit at Delta Park

e. Sandton Riverclub

The field measurements from Braamfontein Spruit at Sandton River Club are presented in Figure 37 and Figure 38. The original data is presented in Table 13 in the appendix. The pH, DO, temperature and EC are presented from the period 11 May 2023 to 9 April 2024. The field measurements measured from the Braamfontein Spruit at Sandton River Club indicate the same trends as the other sites with a circumneutral pH (with a maximum of 9.23 of 14 August 2023), consistent temperatures with the expected variations across seasons, DO that increases in the colder temperatures and decrease when there are warmer temperatures, but never reaching anoxic conditions. The EC measurements were very similar to the EC measurements taken at Delta Park with consistent EC throughout the study, but a spike was recorded on the 5th of February, and a low result was recorded on the 4th of April 2024.

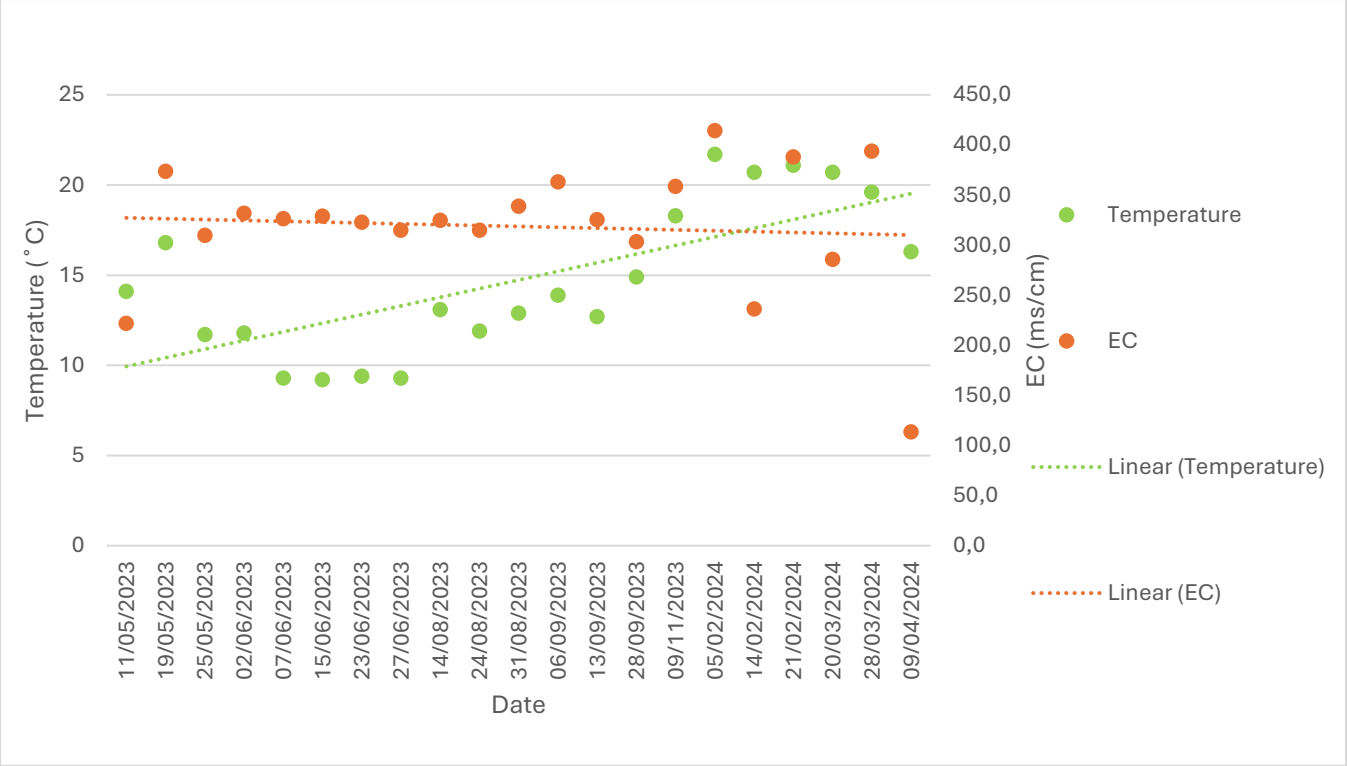


Figure 37: The Temperature and EC results from the Braamfontein Spruit at Sandton Riverclub Golf Course

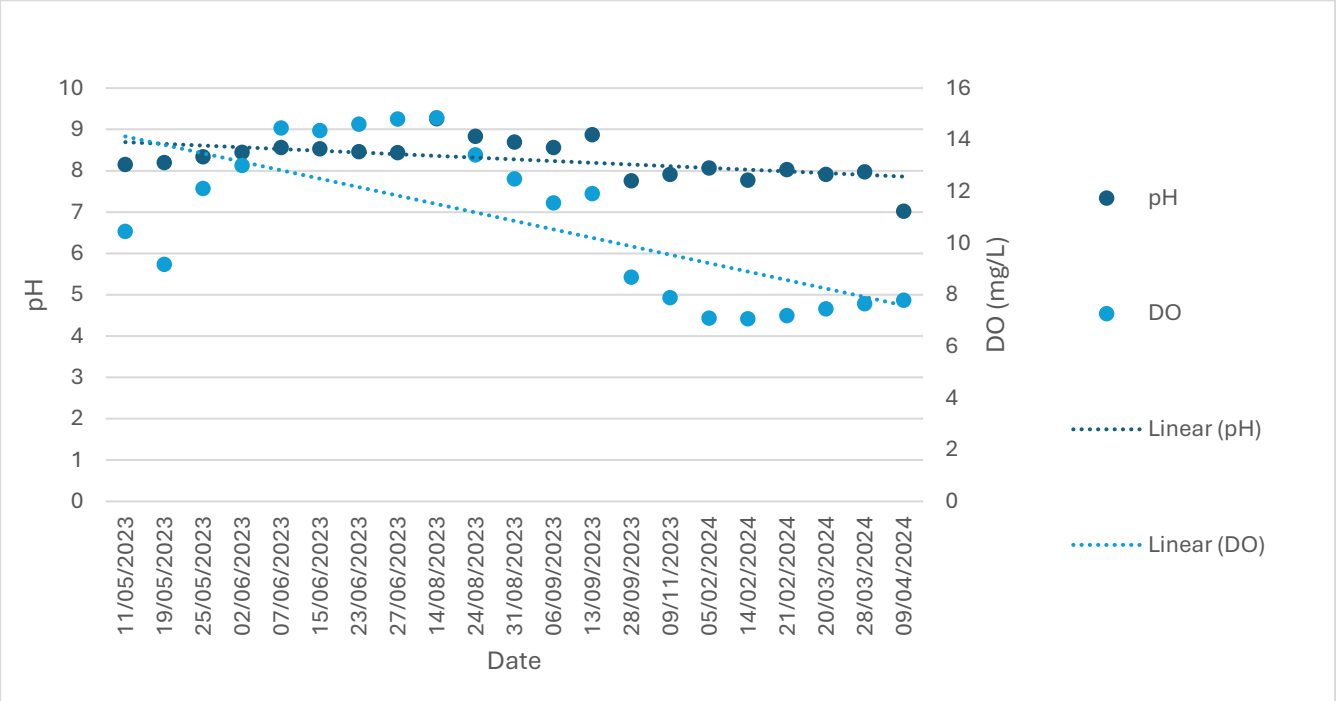


Figure 38: The pH and DO results from the Braamfontein Spruit at Sandton Riverclub Golf Course

The nutrient results from Braamfontein Spruit at Sandton River Club are presented in Figure 39 and Figure 40. The original data is presented in Table 13 in the appendix. The data is presented from the period 11 May 2023 to 9 April 2024. The nutrient results, however, indicate significant nutrient loading and pollution in the river, with high sulphate concentrations being consistently recorded in the first half

of the study, reaching a significantly high concentration of 50 mg/L in June 2023, and decreasing through the second half of the study with only two days where sulphate was recorded in the river. The nitrates followed a similar trend in that consistent nitrates were found in the river in the first half of the study before a large period of zero nitrates were recorded and two days near the end of the study recorded nitrate in the river. There was consistently low nitrite values recorded in the river throughout the study. The phosphate concentrations were variable during the study period with phosphate presence being recorded on the majority of sampling days throughout the study period, consisting of various spikes in concentration in September 2023 and February 2024. These concentrations greatly exceed DWS guidelines for aquatic ecosystems (Silberbauer, 1996).

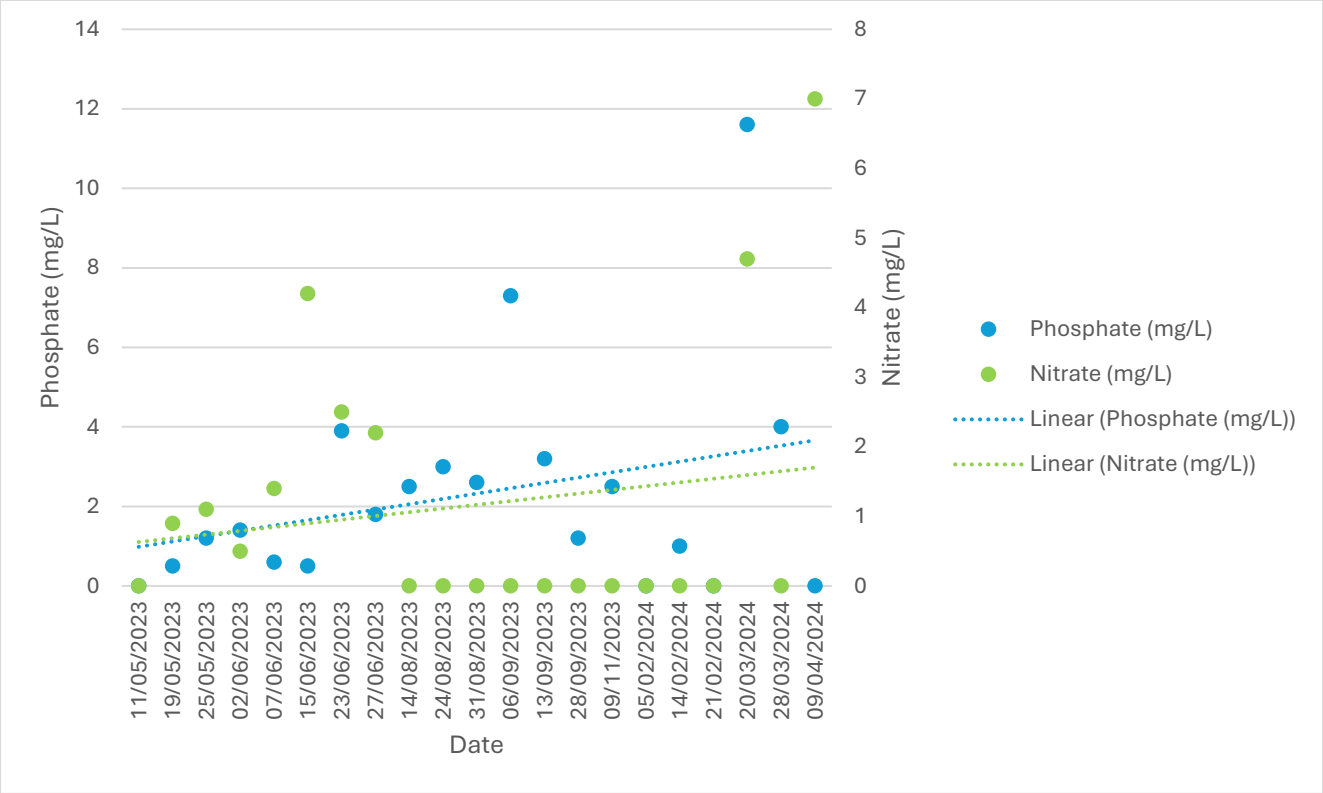


Figure 39: The Phosphate and Nitrate results from the Braamfontein Spruit at Sandton Riverclub Golf Course

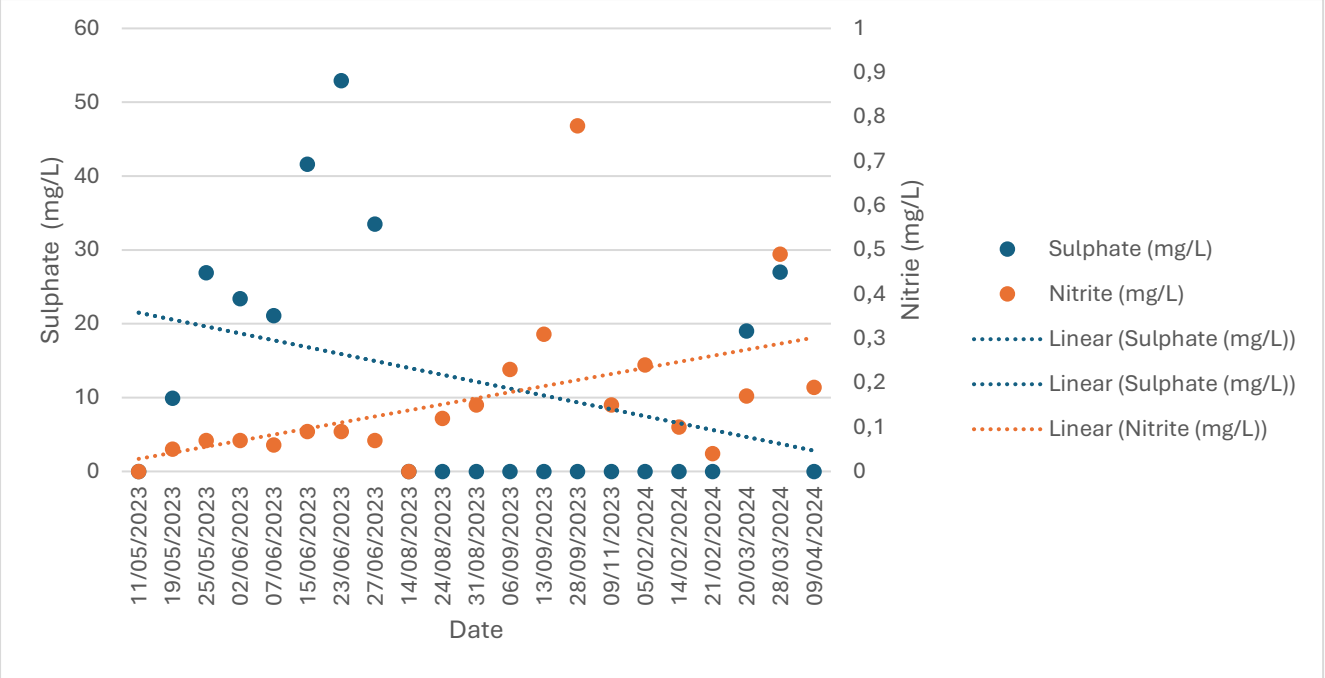


Figure 40: The Sulphate and Nitrite results from the Braamfontein Spruit at Sandton Riverclub Golf Course

The *E. coli* and total coliform results from Braamfontein Spruit at Sandton River Club are presented in Figure 31. The original data is presented in Table 13 in the appendix. The data is presented from the period 11 May 2023 to 9 April 2024. These results indicate significant concentrations of both *E. coli* and total coliforms. In fact, an *E. coli* value of 300,000 was observed in February 2024, which is extremely high, indicating significant faecal contamination.

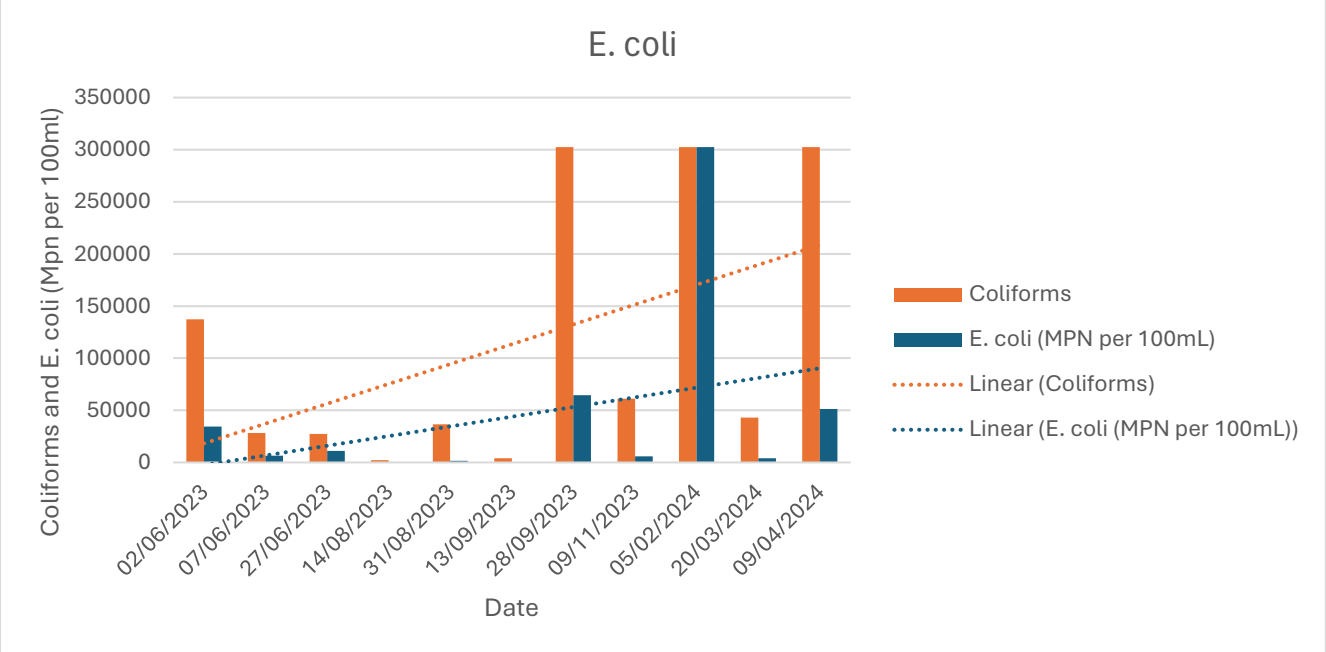


Figure 41: The E. coli and total coliforms results from the Braamfontein Spruit at the Sandton Riverclub Golf Course

The metal analysis for this site, shown in Figure 42 and Table 14, indicates very low concentrations of metals in the water column, however, there are high concentrations of metals found in the sediments of the river. Manganese was specifically high in this river with a concentration of 678 mg/kg was observed on the 14th of August 2023, which is well above the threshold set out by the USEPA (1996). The manganese had its lowest concentration on the 9th of November 2023. On the 7th of June 2023, a maximum lead concentration of almost 1600 mg/kg was observed on the same day that iron concentrations were high, with a maximum of 24,000 mg/kg, despite iron not being toxic. Similarly to all other sites, the lowest lead concentration was recorded on the 14th of February 2024. Iron is not an issue because it's not toxic but elevated levels of iron can however pose an issue during flooding because it may lead to reduced levels of light penetration in the water body.

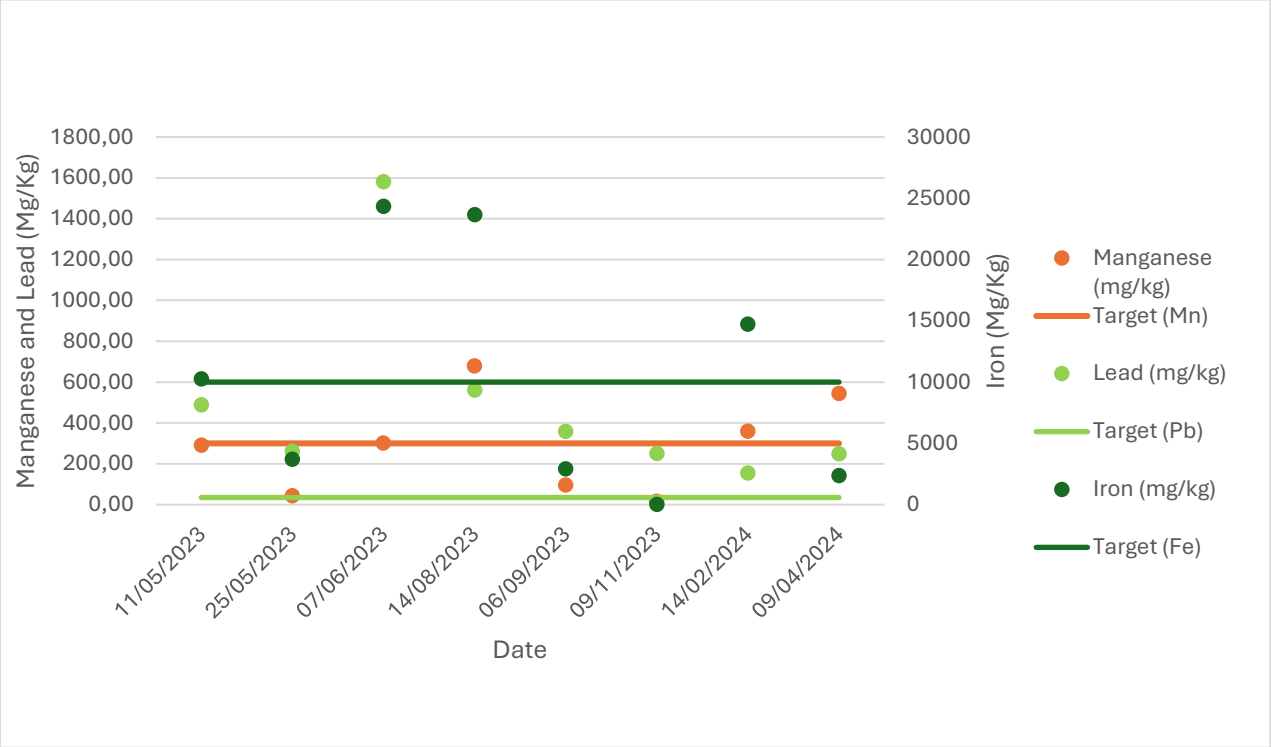


Figure 42:The manganese, iron and lead concentrations from the sediment samples taken from the Braamfontein Spruit at Sandton River Club

Consolidated summary of all results

Across all sites surveyed from 11 May 2023 to 9 April 2024, physio-chemical conditions were generally stable and indicative or moderate urban river conditions rather than extreme degradation. pH values remained circumneutral at all sites, temperature varied seasonally as expected, EC showed some variation but remained within moderate ranges and DO followed the expected inverse relationship with temperature.

Nutrient concentrations were typically low for much of the study period at most sites, but multiple episodic spikes were observed, especially in nitrate, phosphate and sulphate, which aligns with the faecal indicator bacteria peaks (*E. coli* and coliforms). These spikes suggest intermittent outputs of sewage or stormwater runoff (notably during September 2023 and April 2024).

Microbial contamination was consistently present across all sites, with especially high E. coli and coliform counts recorded at the Braamfontein Spruit at Parkview Golf Course with E. coli values of 300000 MPN per 100mL and coliforms of 800000 MPN per 100mL being recorded. These findings point to chronic faecal contamination in some urban areas, rather than isolated events. However, in most cases, the microbial peaks did not coincide perfectly with nutrient spikes, indicating multiple pollution pathways such as direct faecal input, sediment resuspension and stormwater runoff.

Metal concentrations in the water column were uniformly very low across all sites and below South African guideline thresholds for aquatic life. In contrast, sediment metal contaminations exhibited greater variability and several exceedances of international guideline thresholds (eg. Lead >35 mg/kg ISQG, manganese >300 mg/kg USEPA) especially in some areas along the Braamfontein Spruit and at the Montgomery Spruit site. These exceedances indicate localised metal accumulation hotspots in sediments and thus pose ecological risk despite health water column signals.

Overall, while the water-column physio-chemical and nutrient data suggest low to moderate impact from urban activity for much of the sampling period, the recurrent spikes in nutrients and microbial indicators and the sediment metal exceedances highlight episodic contamination events and legacy accumulation in urban river sediments that require management.

f. Spatial Variation of pollution identifiers at the various sample sites

Figure 33 displays the average values recorded at each site for the field measurements taken throughout the sample period. The pH remains the most consistent parameter across each site, with the averages remaining circumneutral across each of the sites. The temperature was consistent across the various sites with Parkview having the highest temperature and Montgomery Spruit having the lowest temperature. The dissolved oxygen is inversely proportional to temperature, with DO decreasing when temperature increases resulting in Parkview having the lowest DO and Montgomery spruit having the highest. The EC was fairly consistent across the sites with Montgomery Spruit having the highest value and Emmarentia having the lowest.

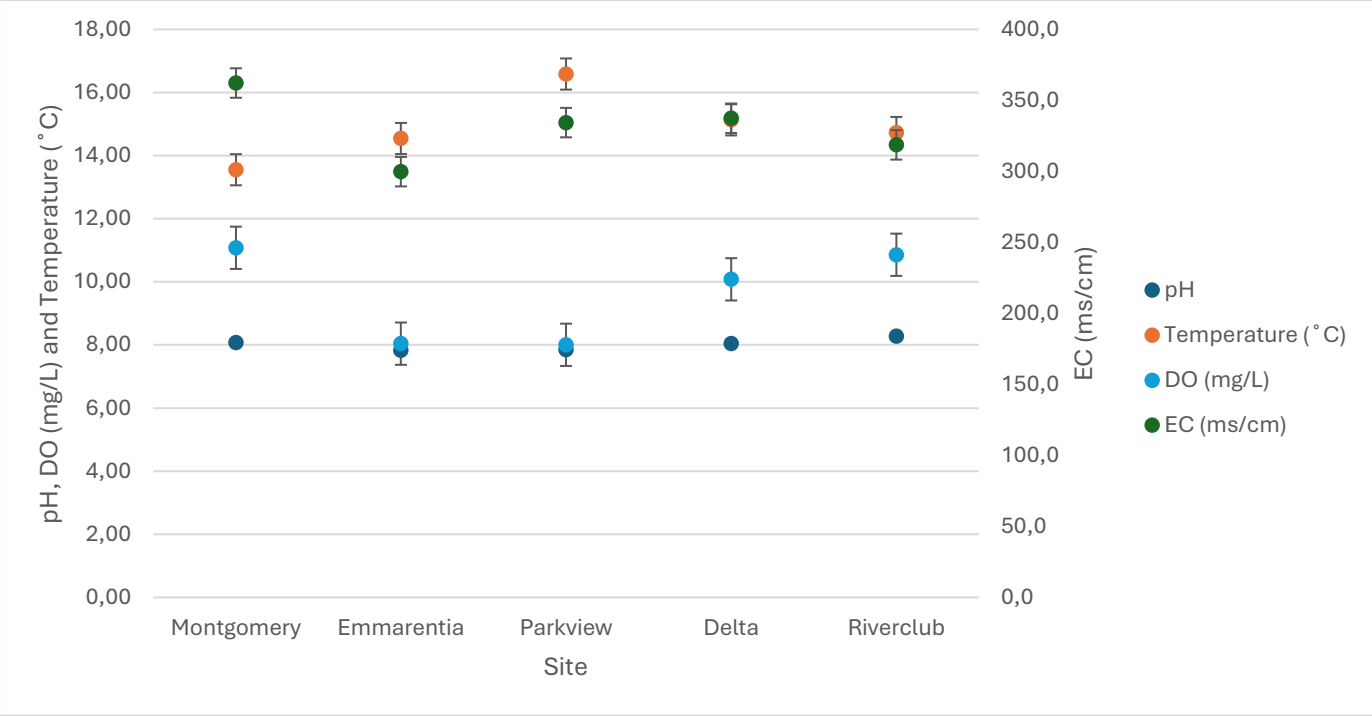


Figure 43: The average values recorded at each site for pH, DO, EC and temperature at each site

The average nutrient results over the study period and across the various sample sites are shown in Figure 34. The nitrite values for each site were relatively low and consistent with Parkview having the highest nitrite average. The average nitrate concentration was fairly consistent across the sites, with Parkview having the highest average, and Riverclub and Emmarentia having lower averages. The average sulphate values at Emmarentia and Montgomery are significantly lower than the averages recorded at the other three sites. There were significant levels of sulphate found at Parkview, Delta Park and Sandton Riverclub showing that pollution ingress of sulphate in occurring from the Braamfontein Spruit and not the tributaries Westdene and Montgomery Spruits. The phosphate, however, shows significant contamination at the Montgomery Spruit compared to the other rivers. This could be attributed to the many people using the river to wash clothes both upstream and at the sampling site or urban runoff and leaking gutter systems. The phosphate values at Emmarentia are low compared to the other

sites, as expected, since the areas upstream of the sample site are mostly fenced off. The phosphate value decreases between Parkview and Delta Park, which could be attributed to the inflow from the tributaries which dilute the phosphate concentrations, or natural process that remove phosphate from the water column. The phosphate is then seen to increase again at the Sandton Riverclub site, which could also be attributed to urban runoff, leaking gutters and sinks, and people using the river to wash their clothes. Despite variable phosphate concentrations across the sites and over time, the average phosphate concentrations for each site was over the DWS guideline for aquatic ecosystems which is set out to preserve aquatic life (Silberbauer, 1996).

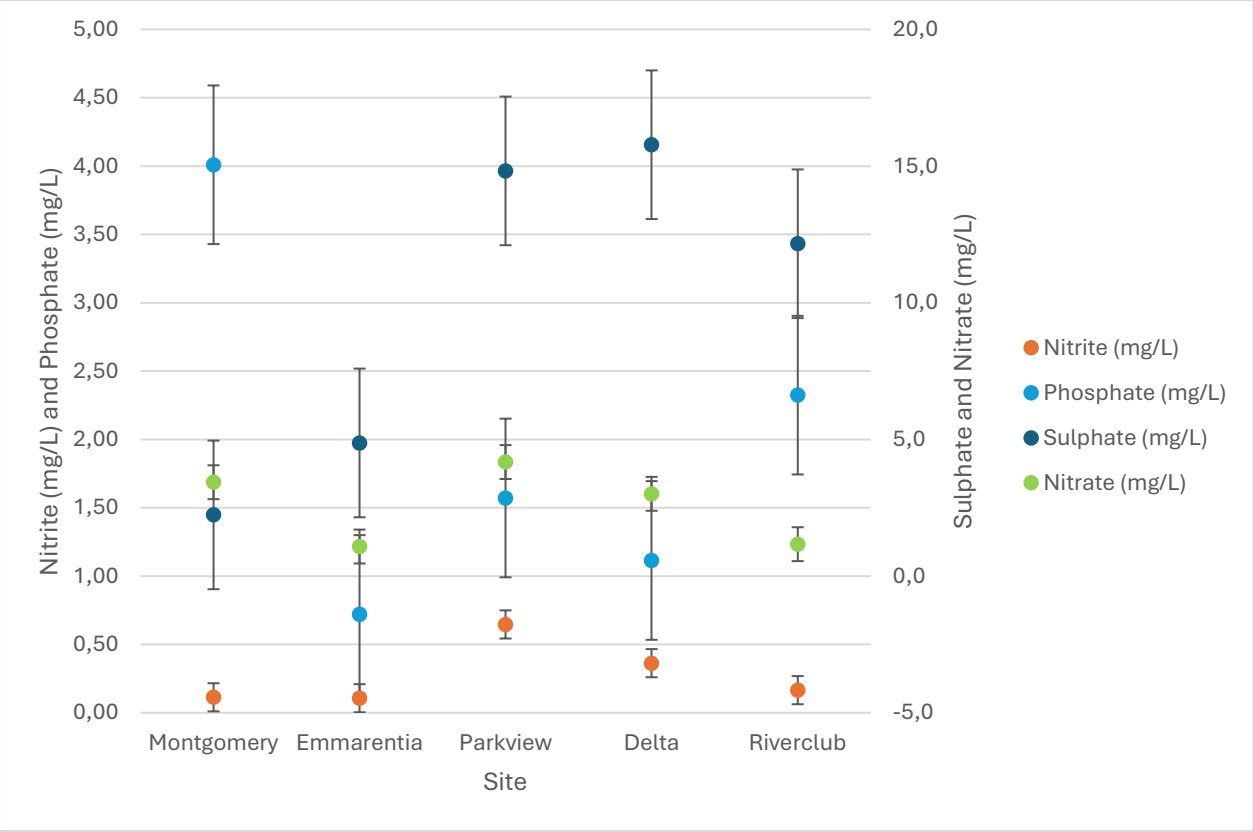


Figure 44: The average values recorded at each site for sulphate, nitrate, nitrite and phosphate at each site

There was *E. coli* identified at each of the sampling sites throughout the sampling period, as shown in Figure 35. This could mean that leaking sewage systems are releasing a constant flow of faecal contamination into the rivers causing damage to the health of the rivers. While Montgomery and Emmarentia are significantly lower than Parkview, the presence of faecal matter is still an issue that needs to be addressed. The *E. coli* and total coliforms averages are significantly higher than all the other sites, showing that there were high concentrations of faecal matter being released into the river near the source. The concentrations of both *E. coli* and total coliforms decrease significantly between Parkview and Delta Park, which could indicate that the inflow from Emmarentia

and Montgomery Spruit are diluting the contaminated water, or that natural processes were removing the bacteria from the water before the Delta Park Site. The concentrations for both parameters remain consistent between Delta Park and Sandton Riverclub, perhaps indicating that no new leaking sewage pipes occur downstream of Delta Park.

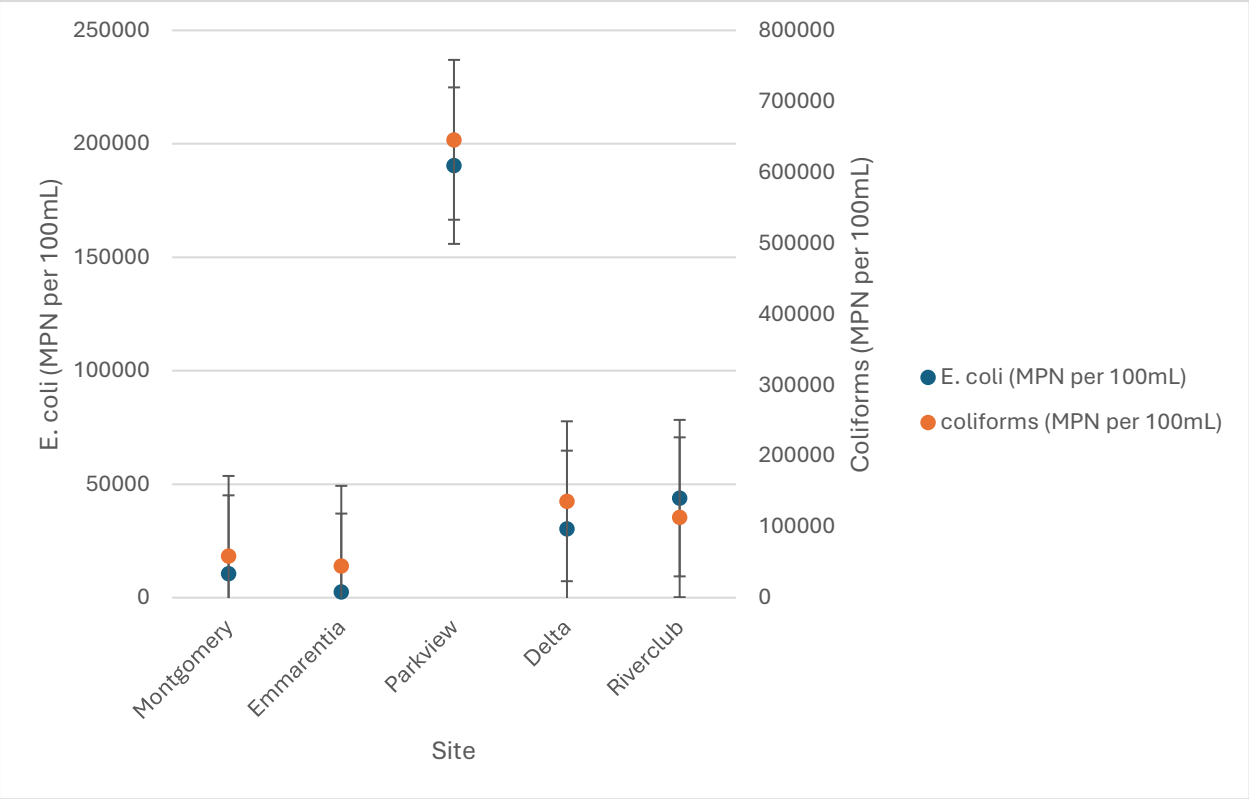


Figure 45: The average values recorded at each site for E. coli and Coliforms at each site

The average concentration recorded for manganese, iron and lead measured from the sediment samples are displayed in Figure 36. The iron was significantly higher at Montgomery, Emmarentia and Sandton River than it was at Parkview and Delta Park, this is, however, not a concern as iron is not toxic to humans or the environment. The manganese followed a similar trend in that the concentration was much higher at Montgomery and Emmarentia than it was at Parkview and Delta Park. The high average lead concentrations are a concern for the environment and humans. There was significant lead concentrations found at each of the sites with Sandton Riverclub having a significantly higher concentration than the other sites. This is a concern as lead presence in aquatic environments has detrimental effects on fish and invertebrates, and can cause serious illness in humans, especially toddlers.

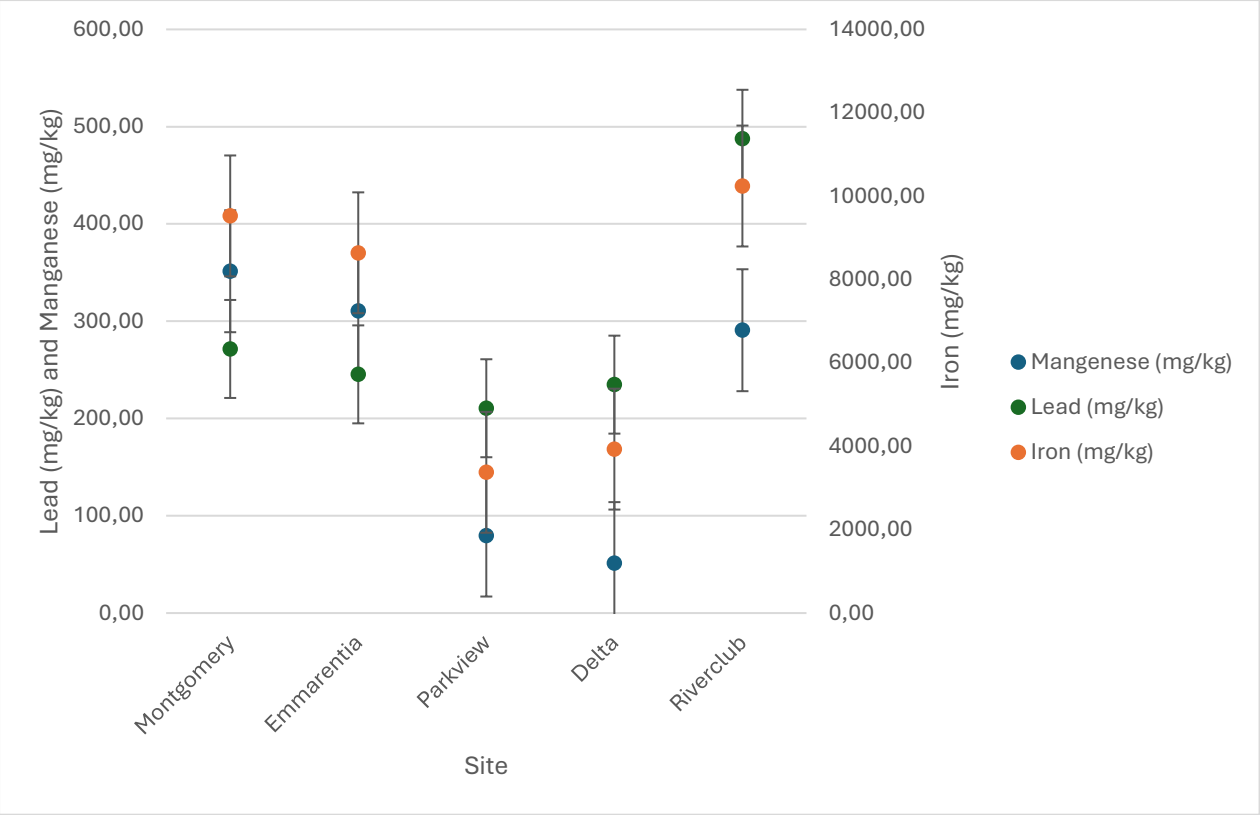


Figure 46: The average concentration recorded for manganese, iron and lead in the sediment at each site

g. Metal statistical analysis

Table 2 shows the results from the sediment metal analysis including the Contamination Factor, Pollution Load Index, Geo-accumulation index and Enrichment Factor for lead, iron and manganese. There is no background information available for iron in the Braamfontein Spruit of its tributaries. However, the average

iron concentration from the Montgomery Spruit was selected as the reference metal when calculating enrichment factor for all sites and metals. The ecological risk indices were applied to the data, including Contamination Factor, Pollution load index, Geo-accumulation index and Enrichment Factor.

i. Lead

According to this scale for CF in the sediment in Table 2, Montgomery Spruit experienced low Pb contamination on 3 out of the 8 sampling days and moderate contamination on 5 out of the 8 sampling days tested. Emmarentia experienced low contamination on 4 of 8 days and moderate on 4 of 8 days. Parkview experienced low contamination on 5 of 8 days and moderate contamination on 3 of 8 days. Delta experienced low contamination on 4 of 8 days and moderate contamination on 4 of 8 days. Riverclub experienced low contamination on 4 of 8 days, moderate contamination on 2 of 8 days and considerable contamination on 2 of 8 days.

The Igeo assessment in Table 2 indicated that all sites were in Class 1 and were classified as unpolluted to moderately polluted by Pb as all values were between 0 and 1.

According to the EF in Table 2, the Montgomery Spruit experienced minor lead enrichment on 2 of the 8 days and extremely severe enrichment on one of the days. The Westdene Spruit at Emmarentia experienced 3 days of minor lead enrichment. The Braamfontein Spruit at Parkview Golf Course experienced minor lead enrichment on 5 of the 8 days and moderate to severe lead enrichment on 1 of the days. The Braamfontein Spruit at Delta Park experienced minor enrichment on 5 of the 8 days and moderate lead enrichment on 1 of the 8 days. The Braamfontein Spruit at Sandton River Club experienced minor lead enrichment on 4 of the 8 days, moderate enrichment on 1 of the 8 days and extremely severe enrichment on 1 of the 8 days.

ii. Iron

According to the same scale for Iron CF in Table 2, Montgomery experienced low contamination on 3 of 8 days and moderate contamination on 5 of 8 days. Emmarentia experienced low contamination on 4 of 8 days and moderate on 4 of 8 days. Parkview experienced low contamination on

all days. Delta experienced low contamination on all days. Riverclub experienced low contamination on 4 of 8 days, moderate contamination on 2 of 8 days and considerable contamination on 2 of 8 days.

The Igeo assessment in Table 2 indicated that all sites were in Class 1, and categorised as unpolluted to moderately polluted by Fe as all values were between 0 and 1.

According to the EF in Table 2, none of the rivers were enriched with iron. This site was used as the reference site, thus, no EF was detected.

iii. Manganese

According to the same scale for Manganese CF in Table 2, Montgomery experienced low contamination on 5 of 8 days and moderate contamination on 2 of 8 days and considerable contamination on 1 of 8 days. Emmarentia experienced low contamination on 4 of 8 days and moderate on 4 of 8 days. Parkview experienced low contamination on all days. Delta experienced low contamination on all days. Riverclub experienced low contamination on 4 of 8 days, moderate contamination on 2 of 8 days and high contamination on 2 of 8 days.

The Igeo assessment in Table 2 indicated that all sites were in Class 1, and categorised as unpolluted to moderately polluted by MN as all values were between 0 and 1.

According to the EF in Table 2, the Montgomery Spruit experienced minor manganese enrichment on 1 of the 8 days and extremely severe enrichment on 1 of the 8 days. The Westdene Spruit at Emmarentia experienced minor manganese enrichment on 1 of the 8 days. The Braamfontein Spruit at Parkview Golf Course experienced minor enrichment on 1 of the 8 days. The Braamfontein Spruit at Delta Park experienced no manganese enrichment. The Braamfontein Spruit at Sandton River Club experienced moderate manganese enrichment 1 of the 8 days and extremely severe enrichment on 1 of the 8 days.

iv. Pollution Load Index

According to this scale in Table 2, Montgomery Spruit experienced metal pollution on 2 of 8 days, Emmarentia on 1 of 8 days, none of the days Parkview or Delta was tested, and 2 of 8 days that Riverclub were tested.

Table 2: The CF, PLI, Igeo and EF of each site over the study period

	Pb (CF)	Fe (CF)	Mn (CF)	PLI Final	Pb (Igeo)	Fe (Igeo)	Mn (Igeo)	Pb (EF)	Fe (EF)	Mn (EF)
Montgomery										
11/05/2023	0,56	1,53	0,34	0,006	0,018	0,001	0,013	0,27	0,75	0,17
25/05/2023	1,08	1,29	2,64	0,007	0,02	0,001	0,019	0,63	0,75	1,54
07/06/2023	1,13	1,49	1,96	0,007	0,02	0,001	0,018	0,57	0,75	0,99
14/08/2023	1,47	0,01	0,40	0,007	0,021	0	0,014	139,7	0,75	37,95
06/09/2023	1,52	1,00	0,59	0,007	0,021	0,001	0,015	1,14	0,75	0,45
09/11/2023	1,26	0,54	0,39	0,007	0,021	0,001	0,013	1,76	0,75	0,54
14/02/2024	0,23	0,40	0,48	0,005	0,015	0,001	0,014	0,44	0,75	0,9
09/04/2024	0,75	1,75	1,20	0,006	0,019	0,001	0,017	0,32	0,75	0,51
Emmarentia										
11/05/2023	0,53	0,36	0,23	0,006	0,018	0,001	0,012	1,12	0,75	0,49
25/05/2023	0,95	1,64	1,78	0,007	0,02	0,001	0,018	0,44	0,75	0,82
07/06/2023	1,16	1,01	0,44	0,007	0,02	0,001	0,014	0,86	0,75	0,33
14/08/2023	1,44	1,04	1,94	0,007	0,021	0,001	0,018	1,04	0,75	1,4
06/09/2023	1,40	0,63	0,73	0,007	0,021	0,001	0,015	1,67	0,75	0,87
09/11/2023	1,40	1,81	1,57	0,007	0,021	0,001	0,017	0,58	0,75	0,65

14/02/2024	0,07	0,26	0,18	0,003	0,01	0,001	0,011	0,2	0,75	0,51
09/04/2024	0,28	0,51	0,20	0,005	0,015	0,001	0,012	0,41	0,75	0,29
Parkview										
11/05/2023	0,59	0,33	0,20	0,006	0,018	0,001	0,012	1,35	0,75	0,46
25/05/2023	0,73	0,52	0,33	0,006	0,019	0,001	0,013	1,06	0,75	0,48
07/06/2023	1,02	0,35	0,22	0,007	0,02	0,001	0,012	2,18	0,75	0,47
14/08/2023	1,20	0,47	0,15	0,007	0,02	0,001	0,011	1,9	0,75	0,23
06/09/2023	1,40	0,17	0,21	0,007	0,021	0,001	0,012	6,2	0,75	0,93
09/11/2023	0,72	0,28	0,05	0,006	0,019	0,001	0,008	1,94	0,75	0,13
14/02/2024	0,22	0,31	0,62	0,005	0,015	0,001	0,015	0,54	0,75	1,52
09/04/2024	0,32	0,41	0,03	0,005	0,016	0,001	0,006	0,59	0,75	0,06
Delta										
11/05/2023	0,67	0,31	0,05	0,006	0,018	0,001	0,008	1,64	0,75	0,12
25/05/2023	1,09	0,23	0,08	0,007	0,02	0,001	0,009	3,53	0,75	0,27
07/06/2023	1,24	0,46	0,17	0,007	0,021	0,001	0,011	2	0,75	0,28
14/08/2023	1,28	0,40	0,24	0,007	0,021	0,001	0,012	2,4	0,75	0,46
06/09/2023	1,51	0,41	0,13	0,007	0,021	0,001	0,011	2,8	0,75	0,25
09/11/2023	0,69	0,51	0,03	0,006	0,019	0,001	0,006	1,03	0,75	0,04
14/02/2024	0,08	0,51	0,35	0,004	0,011	0,001	0,013	0,12	0,75	0,51
09/04/2024	0,35	0,48	0,11	0,005	0,016	0,001	0,01	0,55	0,75	0,18
Riverclub										
11/05/2023	1,80	1,08	0,83	0,007	0,022	0,001	0,016	1,25	0,75	0,58
25/05/2023	0,96	0,39	0,12	0,007	0,02	0,001	0,01	1,86	0,75	0,24
07/06/2023	5,82	2,56	0,86	0,009	0,026	0,001	0,016	1,71	0,75	0,25
14/08/2023	2,06	2,48	1,93	0,007	0,022	0,001	0,018	0,62	0,75	0,58
06/09/2023	1,32	0,31	0,27	0,007	0,021	0,001	0,012	3,22	0,75	0,67

09/11/2023	0,92	0,00	0,04	0,007	0,02	0	0,007	3877,32	0,75	179,39
14/02/2024	0,57	1,55	1,02	0,006	0,018	0,001	0,016	0,28	0,75	0,5
09/04/2024	0,91	0,25	1,55	0,007	0,02	0,001	0,017	2,75	0,75	4,65

h. SASS5 Sampling Results

The ASPT values across all sites range from 1,5 to 4, with averages between 2.62 and 3.04. These values indicate “critically modified” (E/F) conditions, suggesting extreme pollution and loss of sensitive aquatic species. This is significantly worse than other studies such as (Lebepe et al., 2021). Lebepe et al (2021) study also consisted of SASS5 sampling in urban rivers in Durban, South Africa. Their study had a minimum ASPT score of 4, almost equalling the max ASPT score from this study done in Johannesburg. Lebepe’s study had ASPT scores ranging from 4 to 8, showing that spatial distribution influences the ASPT scores as different rivers and sections of river experience different pollution loads.

There is a low taxon richness in this study. The number of taxon groups recorded is consistently low, averaging between 2.53 and 3.11 per sample. This is significantly lower than expected in healthy river systems, where taxon richness should be above 5 for moderately impacted and above 7 for healthy systems.

There are noticeable seasonable trends across the study, with deterioration of water quality, and thus, taxon richness over time. There were slightly higher scores over the winter months, May to August 2023, particularly in June and August. This could be due to lower temperatures and reduced microbial decomposition, leading to improved dissolved oxygen levels. There was a sharp decline in February to March in 2024, where some of the lowest scores were recorded, some as low a 2 at some sites. This could suggest severe water quality deterioration, potentially due to increased urban sewage pollution, urban runoff of seasonal algal blooms.

Table 3: The SASS5 sampling results

	Montgomery			Emmarentia			Parkview			Delta			Riverclub		
Date	SASS score	Taxon	ASPT	SASS score	Taxon	ASPT	SASS score	Taxon	ASPT	SASS score	Taxon	ASPT	SASS score	Taxon	ASPT
19/05/2023	11	3	3.67	7	3	2.33	7	3	2.33	7	3	2.33	9	3	3
25/05/2023	14	3	4,5	8	4	2	7	3	2,33	12	3	4	7	3	2,33
02/06/2023	12	3	4	11	3	3,66	7	3	2,33	7	3	2,33	11	3	3,66
07/06/2023	7	3	2,33	7	3	2,33	12	4	3	12	4	3	5	3	2,5
15/06/2023	6	2	3	6	2	3	8	4	2	3	2	1,5	6	2	3
23/06/2023	12	3	4	8	4	2	8	4	2	9	3	3	9	3	3
27/06/2023	6	2	3	6	2	3	8	4	2	7	4	1,75	4	3	1,33
14/08/2023	8	4	2	7	3	2.33	8	4	2	10	4	2.5	8	4	2
24/08/2023	10	3	3.33	15	5	3	12	4	3	12	3	4	8	4	2
31/08/2023	12	3	4	8	4	2	7	4	1.75	8	2	4	10	4	2.5
06/09/2023	6	2	3	10	3	3.33	6	2	3	10	4	2.5	8	4	2
13/09/2023	11	4	2.75	7	3	2.33	10	3	3.33	8	2	4	7	3	2.33
28/09/2023	7	3	2.33	7	3	2.33	6	2	3	12	4	3	10	4	2.5
09/11/2023	7	3	2.33	10	3	3.33	10	3	3.33	9	3	3	10	4	2.5
05/02/2024	2	1	2	6	2	3	6	2	3	11	3	3.67	8	2	4
14/02/2024	2	1	2	4	1	4	6	2	3	8	2	4	4	1	4
20/03/2024	6	2	3	6	3	2	6	2	3	8	4	2	9	3	3
28/03/2024	6	2	3	10	4	2.5	10	3	3.33	7	3	2.33	7	3	2.33
09/04/2024	2	1	2	3	2	1.5	6	2	3	3	2	1.5	7	3	2.33
Averages	7,74	2,53	2,99	7,68	3	2,73	7,89	3,05	2,62	8,58	3,05	3,04	7,74	3,11	2,76



Figure 47: An Oligochaete, Hirudinae (Leech) and Baetidae caught from The Westdene Spruit at Emmarentia Botanical Gardens



Figure 48: A Leech caught from the Braamfontein at Parkview Golf Course during SASS5.



Figure 49: A Nepidae caught from the Braamfontein Spruit at Delta Park during SASS5



Figure 50: A coenagrionidae caught from the Braamfontein Spruit at Sandton River Club



Figure 51: An image of a Libellulidae caught during SASS5 in the Westdene Spruit at Emmarentia

Table 3:The list of taxa found at each of the sites throughout the sampling period

Mont	Emma	Parkview	Delta	Riverclub
Chironomidae	Chironomidae	Chironomidae	Chironomidae	Chironomidae
Oligochaeta	Oligochaeta	Oligochaeta	Oligochaeta	Oligochaeta
Baetidae	Baetidae	Baetidae	Baetidae	Baetidae
Culicidae	Culicidae	Culicidae	Culicidae	Culicidae
Hydropsychidae	Leech	Leech	Nepidae	Leech
	Libellulidae	Hydropsychidae	Leech	Hydropsychidae
	Hydropsychidae		Hydropsychidae	
			Potamonautidae	

Table 4:Guideline SASS5 scores and ASPT values for interpreting SASS5 data in relation to river health

Ecological Category	SASS5 score	ASPT score	Ecological category name	Description
E/F	0-42	0-4.2	Critically modified	Critically modified
D	>42-53	>4.2-4.5	Poor	Largely modified
C	>53-71	>4.5-4.8	Fair	Moderately modified
B	>71-104	>4.8-5.8	Good	Largely natural with few modifications
A	>104-180	5.8-8	Natural	unmodified

Many of the species found throughout the study can be seen in Figures 37 to 41. According to the guideline for SASS5 scores and ASPT values for interpreting the data in relation to river health all the results collected throughout this study with the exception of Montgomery Spruit on 25/05/2023 show that the rivers have very low invertebrate biodiversity and those organisms that are found are typically those present only where water quality is classified as polluted. The river is critically modified by human interaction and through contamination with human waste specifically sewage which leads to a low SASS5 score in this environment. Those species that are left are those that are tolerant of changes in water quality. The sample from the Montgomery Spruit on 25/05/2023 indicates a largely modified aquatic environments with poor water quality. While Delta Park had the largest species diversity, the invertebrates found were inconsistent with only two unique taxa being found there on two specific days (14/08/2023 and 06/09/2023) these species also had a low SASS5 score due to their ability to withstand low water quality environments. The data here confirm the findings of Munyika et al (2014), who used many of the same methods to determine water quality.

5. Discussion

This study had the following objectives:

1. To assess the contamination levels in urban rivers in Johannesburg over a one-year period, through regular water, sediment, and invertebrate sampling across different seasons.
2. To identify the sources of pollution in the Jukskei River and its tributaries (Montgomery and Westdene Spruits), focusing on the spatial and seasonal variations in contamination levels.
3. To evaluate the impact of pollutants on aquatic life by analysing the presence and diversity of invertebrates (using the SASS5 method) across the five urban river sites.
4. To measure and quantify key pollutants, including nutrient levels, E-coli, coliform concentrations, and metals in the water and sediment samples.
5. To assess the potential risks to human health posed by the contaminated rivers, especially for those who come into contact with the water for recreational or subsistence activities.
6. To provide recommendations for improving urban river management and reducing pollution levels, based on the findings of the study.

Sampling was carried out over three sampling trips in the course of a year with an 8-week period in the start of 2023, a 7-week period towards the end of 2023 and a 6-week period at the start of 2024, spanning across the different seasons to identify seasonal variation. A total of 5 sites were chosen to be sampled based on their accessibility and relevance to the study. The sites consisted of three points along the Braamfontein Spruit and two points along two separate tributaries, the Montgomery Spruit and the Westdene Spruit, flowing into the Braamfontein Spruit. Field measurements, water samples, sediment samples and SASS5 monitoring were carried out at each of the sites each week that sampling took place. The findings of this study show that there are significant and variable loads of pollution entering the river throughout the course of a year. The types and quantities are variable both spatially and temporally.

This study was especially interested in the spatial and seasonal variations of pollution and contamination. The seasonal variation was identified by doing regular and consistent sampling across various seasons to identify how the river health is impacted, and to identify if pollution loads increase in certain seasons. The spatial variation was identified by performing the same

sampling techniques across different sites, which are all connected, to identify how the quality of the river is impacted by pollution and contamination from closer to the source, to further away from the source of the river. The mean of each pollution parameter was calculated for each of the sites and compared to each other in Figures 43-46. This assisted in identifying the level of contamination at each site to identify which rivers are more contaminated and the type of pollution they are likely experiencing. The pH remained consistent throughout the sampling period at each site, showing very little variation across time and space, indicating that there was no contamination affecting the acidity or alkalinity of the rivers, such as AMD discharge which can lower pH in the river. The slight variation in pH could be on account of different rainfall patterns affecting the dilution of pH in the river, with higher rainfall patterns in the summer periods and lower rainfall in the winter, verified through the weather directory. The average EC was consistent throughout the sites with averages ranging between 280 µs/cm (at Emmarentia) and 360 µs/cm (at Montgomery Spruit). According to the South African DWS (1996), the recommended ranges for EC should be between 30 and 500 µs/cm, indicating that all results recorded throughout the study period fall within the limits and there is no cause for concern surrounding the EC values (Silberbauer, 1996). High levels of pollution have the ability to significantly reduce the available oxygen in rivers, affecting the natural processes of a river and the biodiversity of the river, especially when anoxic conditions occur. The DO was consistent throughout the sampling period and across each site with no significant outliers being identified. According to the DWS, the DO in an aquatic environment should be greater than 5 mg/L as this is the minimum acceptable concentration for supporting a balanced aquatic ecosystem, especially for fish and invertebrates. Emmarentia was the only site where DO values less than 5 mg/L were identified. On the 14th of February, 20th of March and 28th of March, the lowest DO concentrations were recorded, which were less than 5 mg/L. This indicates that this site experiences periodic stress for fish and invertebrates, and could impair their reproduction, mobility and survival (Franklin, 2014). Low DO values could be a consequence of sediment oxygen demand, sewage or greywater presence, high nutrient levels or algal blooms and subsequent dying off of the aquatic plants. However, since other parameters such as *E. coli* and nutrients were measured, some of these reasons could be discarded. It is likely that the DO was low at this site because of stagnant water flow, also resulting from the construction of a dam upstream of the sample site towards the end of the sampling period. Since measurements were taken during the day, it could mean that aquatic plants and algae photosynthesise, consuming CO_2 and producing oxygen, raising DO levels. It cannot be discarded that these values were a result of plant matter decay or algal blooms

requiring more dissolved oxygen, or that groundwater is entering the river and impacting the results. The temperature values recorded across each of the sites were insignificant, with the expected seasonal variations across summer and winter. The DO and temperature at each of the sites were inversely proportional, which is expected because was water holds less oxygen than cold water since oxygen molecules move faster when temperatures increase, making it harder for them to stay in the water, since the intermolecular bonds between the DO and water are weak. Warmer temperatures also accelerate microbial activity and decomposition and in polluted waters, microbes use more oxygen to break down organic matter, resulting in DO being consumed faster (Chakraborty et al., 2016). There is a positive correlation between pH and DO, this could potentially indicate that acidification is not a major concern in the river. Since measurements were taken during the day, it could mean that aquatic plants and algae photosynthesise, consuming CO_2 and producing oxygen, raising DO levels.

Nitrite and nitrate concentrations across the 5 sites demonstrated visible temporal and spatial variability, which is aligned to the patterns described in the literature, regarding nutrient loads and its consequences. According to the DWS, the nitrite concentration should not exceed 0.1 mg/L, as higher concentration can impact the oxygen transport in fish, kill sensitive species of invertebrates and fish, and is toxic, especially in low oxygen or low pH waters (Silberbauer, 1996). While the averages of the nitrites for Emmarentia and Montgomery are on the 0.1 mg/L limit, Parkview, Delta Park and Sandton Riverclub were all well above the stipulated limit and are subject to the impacts that high nitrite concentrations can have on a river. The Nitrate values were variable across the sites with values typically below the 0.5 mg/L threshold set out by the DWS (Silberbauer, 1996). The Montgomery Spruit had elevated nitrite and nitrate concentrations on the 28th of September 2023 of 1.43 mg/L and 19 mg/L respectively. This suggests that a potential contamination event, likely linked to leaky sewer systems, stormwater runoff or resuspension of nitrogen-rich sediments following rainfall (Deng et al., 2022). The nitrite maximum greatly exceeds the recommended ecological guideline of 0.1 mg/L (Silberbauer, 1996). This could indicate that either incomplete nitrification under aerobic conditions, or denitrification under localised anoxic zones. These are both possibilities as they are common in urban streams with fluctuating oxygen levels (Deng et al., 2022). Emmarentia showed a substantial absence of nitrites and nitrates in most of the samples, with exceptions on the 6th of September 2023 (8.1 mg/L nitrate and 0.7 mg/L nitrite) and 28th of September 2023 (14.6 mg/L nitrate and 0.17 mg/L nitrite). These values were recorded towards the end of winter, perhaps indicating that the increased nutrient concentration is a consequence of first-

flush runoff from fertilisers, or legacy nutrient release from the sediment (Deng et al., 2022; Kato et al., 2009). Parkview consistently indicated moderate to high nitrate conditions with extreme concentrations on the 28th of September 2024 (21.4 mg/L) and 9th of November 2023 (13.7 mg/L). Nitrite levels at Parkview also occasionally exceeded the sensitive ecosystem threshold of 0.05 mg/L with a particularly high reading on the 28th of September 2023 (1.36 mg/L). These continuous spikes of nitrogen, often exceeding 10 mg/L indicate a chronically nutrient enriched environment. This is most likely attributed to leaking infrastructure and urban runoff carrying fertilisers, showing the significance of urban influence on river systems. This may indicate that Parkview is a key hotspot for nutrient loading into the broader river system and could perhaps be linked to nitrogen-rich fertilisers used on the golf course (Whitehead et al., 2006). Parkview also had the highest average nitrate value of over 4 mg/L. The nitrate profile at Delta Park had higher levels of fluctuation, with peaks between June and August 2023 and again from February to June 2024. The nitrite at Delta Park was often elevated above the 1 mg/L limit. These nitrogen loads suggest a combination of seasonal sediment resuspension (which is expected due to the high sediment loads at the site), stormwater inputs and possibly diffuse runoff from the surrounding urban areas. This pattern is explained in Deng et al. (2022), which states that summer months bring both nutrient accumulation in sediments and higher bacterial activity (Deng et al., 2022). Sandton Riverclub displayed comparatively lower nitrate concentrations than other sites with modestly high values recorded on 20th of March 2024 (4.7 mg/L) and 9th of April 2024 (7 mg/L). Nitrite values, however, were consistently above the 0.05 mg/L preferred guideline set out by the DWS, with a maximum of 0.49 mg/L recorded in March 2024 (Silberbauer, 1996). These nutrient values could point to attenuated nutrient flow from upstream, or effective buffering from vegetation between Riverclub and Delta Park. The elevated nitrite level, however, may indicate poor oxygenation or microbial hotspots, which pose a threat to aquatic invertebrate species (Banerjee et al., 2023). Montgomery, Parkview and Delta Park all had averages that place these rivers in the eutrophic conditions category, which leads to low species diversity, excessive plant growth and the possibility of toxic blue-green algae presence in the river (Silberbauer, 1996). Emmarentia and Sandton River club had lower nitrate values between 0.5 mg/L and 2.5 mg/L, indicating that mesotrophic conditions are prevailing, which is associated with productive systems with high species diversity and moderate aquatic plant growth. A point of interest to make is that the SASS5 sampling results showed Emmarentia and Parkview both did not have the highest taxon or ASPT scores out of all the sites, perhaps indicating that nitrite is not the main cause of the low species diversity at these sites.

The phosphate concentrations had significant variability across sites and seasons, indicating prevalent risks of eutrophication, anthropogenic nutrient loading and sediment interactions in the Braamfontein Spruit. According to Silberbauer (1996), who is from the DWS, a single spike of phosphate is not as concerning as persistently elevated levels of phosphate. However, many of the sites sampled in this study displayed both. The Montgomery Spruit showed several isolated peaks in phosphate readings, especially on the 25th of May 2023 (8.6 mg/L), 6th of September 2023 (27.5 mg/L) and 13th of September 2023 (30.2 mg/L). These concentrations greatly exceed the typical ecological threshold levels of 0.025mg/L, and indicate that severe nutrient enrichment which has led to eutrophic and hypertrophic conditions (Silberbauer, 1996). These peaks did not coincide with the elevated nitrate levels, perhaps indicating that different nutrient inputs, such as greywater. This could also be a result of preferential uptake and cycling of nitrogen by algae or aquatic plants. Since phosphate readily binds to sediment particles, the observed spikes could be linked to disturbed sediment or wastewater discharges (Foppen & Schijven, 2006). Emmarentia generally had low phosphate levels with isolated elevations from August to September 2023, coinciding with elevated nitrate levels. An elevated concentration on the 14th of August 2023 (4.5 mg/L) and the 6th of September 2023 (4.7 mg/L) suggest seasonal accumulation which is potentially influenced by dry season low-flow conditions which allow phosphorus to build up in sediments before being resuspended (Withers & Jarvie, 2008). Other than this period of high concentrations, phosphate remained below 1 mg/L, indicating that this site may benefit from better upstream nutrient retention, or lower urban surface runoff and industrial effluent inflow. Parkview was one of the most enriched sites in terms of phosphate concentration. The phosphate results showed elevated concentrations over extended periods of time. The phosphate values at Parkview were greater than 2 mg/L in 9 of the 21 samples, with a major spike identified on the 25th of May 2023 (6.8 mg/L). This site is likely exposed to a range of pollutants from chronic fertiliser inputs and urban runoff to leaking sewer systems, which would also explain the recurring nitrate spikes at this site. The high nutrient availability, especially in winter and spring, suggest a high risk of eutrophication and potential toxic algal blooms (Zhang et al., 2017). Delta Park showed moderate variability of phosphate concentrations, with levels typically between 0.5 mg/L and 2.5 mg/L. These values are consistent with mesotrophic to eutrophic conditions. A notable increase was identified on the 20th of March 2024 (2.8 mg/L), which is likely a result of sediment resuspension and warmer seasonal temperatures. It is important to note that phosphate and nitrate peaks did not always coincide, indicating seasonal nutrient partitioning in which phosphorus accumulated during dry spells and is released when storms wash it into the river

(Withers & Jarvie, 2008). The phosphate concentrations at Sandton Riverclub indicated some of the most variable trends with increases being observed between August and September 2023 (up to 7.3 mg/L) and again in March 2024 (11.6 mg/L). These concentrations greatly exceed DWS guidelines (1996) for aquatic ecosystems and are a concern to ecosystem and may indicate a build-up of phosphorus rich sediment which is resuspended in rainy seasons when river discharge and sediment disturbance is higher (Silberbauer, 1996). These high phosphate concentrations will have severe implications for aquatic life, such as macroinvertebrates, reducing biodiversity as less tolerant species are unable to survive under these conditions (Silberbauer, 1996). The sharp spike observed in March coincides with elevated levels of nitrates and nitrites, indicating a multi-nutrient influx. This could be attributed to sewage leaks, illegal dumping or stormwater inflows. The episodic nature of these events aligns with patterns of urban stormwater contamination observed in Zhang et al. (2017) and Bae & Seo (2021).

The sulphate concentrations across the five sampling sites show both spatial and temporal variation, indicating different degrees and sources of contamination. As described by Zak et al. (2021), sulphate is a naturally occurring anion but can reach elevated concentrations in urban rivers from anthropogenic inputs such as urban runoff, wastewater discharge and AMD. This contamination is evident in the data captured during this study. The sulphate was significantly lower at Montgomery and Emmarentia than the other sites, mostly below 5 mg/L at Montgomery and peaking just below 15 mg/L at Emmarentia across the study period. These sites therefore likely experience less direct anthropogenic discharge or could perhaps have better natural attenuation processes. However, there are still periodic spikes in sulphate concentrations, reflecting episodic pollution, which could be a result of urban runoff from rainfall events, which is known to transport sulphate-rich materials into rivers (Vehanen et al., 2022). Parkview exhibited moderate sulphate conditions with concentrations ranging from 12.1 to 52.2 mg/L. There are frequent spikes noted at this site throughout the sampling period, which could be attributed to localised pollution events such as leaking wastewater infrastructure. There is a high possibility that these elevated sulphate levels are caused by the wastewater since there were elevated *E. coli* concentrations at this site throughout the study. This aligns with Zak et al (2021) who mention that sulphate from domestic wastewater and urban runoff is common in developing countries. Delta Park consistently displayed higher sulphate concentrations with several readings above 40 mg/L and a maximum of 51.7 mg/L. The persistently high concentrations, especially in mid-winter and spring suggest significant upstream contamination. This contamination is likely from a combination of sewer leaks, stormwater

runoff and informal settlements (Villiers & Mkwelo, 2009). Villiers and Mkwelo (2009) explain that sulphate levels above 40 mg/L are an indication of anthropogenic influence and could be a result of mining or industrial activity. Sandton River Club presented highly variable sulphate concentrations, ranging from 0 to 52.9 mg/L. The peaks observed could possibly reflect industrial or urban runoff inputs. The lack of sulphate presence in some months could suggest either seasonal dilution during high rainfall periods, or possible errors in sample testing. These irregular patterns highlight the need for frequent monitoring in urban areas, as emphasised by Walton (1989) as there is fluctuation in pollution loads. The temporal trends reflect an upward trend in sulphate concentrations, particularly in Delta Park and Riverclub. The increases are in line with seasonal runoff patterns and potentially reduced flow in drier months, concentrating pollutants. Zak et al. (2021) describes sulphate as highly soluble, making it persist in water and contribute to long-term salinity changes which can affect aquatic life and water quality. Under anoxic or acidic conditions, sulphate can form hydrogen sulphide (H_2S), a toxic gas which affect fish and invertebrates, as described by Zak et al. (2021). This is particularly concerning in warmer months and areas with high organic loads where bacterial activity is heightened. Elevated sulphate levels also mobilise metals which impact both ecosystem and human health (Villiers & Mkwelo, 2009).

The health of these urban rivers in Johannesburg, which were assessed through SASS5 techniques, reveals substantial ecological degradation across all sampling sites. SASS5 sampling was conducted in order to identify the impact of pollutants on the invertebrate biodiversity across all sites from May 2023 to April 2024. The results from the invertebrate sampling were recorded in a SASS5 scoring sheet which provided the SASS score, number of taxon and the Average Score Per Taxon (ASPT). There are many possible causes of poor ecological health that could potentially be causing low scores to be recorded throughout the study. High pollution levels with the presence of *E. coli* and coliform counts in the physiochemical dataset aligns with the low SASS5 scores. High levels of nitrates, phosphates and Electrical Conductivity suggest pollution from sewage, stormwater or agricultural runoff. Agricultural runoff is not highly likely in an urban area, however fertilisers from gardens may be causing similar impacts. The low taxon richness could potentially indicate habitat degradation such as channelisation, sedimentation or loss of vegetation (Munyika et al., 2014). This is relevant as sites such as Riverclub and Delta Park had high levels of sedimentation at the sample sites. Parkview Golf Course had very low levels of sedimentation, as it was higher upstream and closer to the source, consisting mostly of rock and gravel with little to no

vegetation around. Some sites experienced low flow rates (although not measured for this study) which could potentially lead to stagnation and further degradation. Across the entire sampling period, ASPT scores consistently fell within the 'critically modified' (E/F) category, indicating that these ecosystems are under severe stress with significant loss of macroinvertebrate taxa. The low ASPT values indicate an absence of more pollution-sensitive species such as stoneflies and Amphipoda which are usually found in healthier rivers. Only a few taxa that can tolerate pollution are found on the list of species identified in this study. According to Table 4 under the results section, there were low SASS5 scores across all sites compared to other studies (Lebepe et al., 2021) (Munyika et al., 2014). In these other studies, SASS5 sampling results have identified categories ranging mostly from A to C, showing natural and unmodified to fair and moderately modified. Some results did show D and E/F categories, but these were temporary lows with the rivers quickly recovering to the A-C range. The results in this study speak to the degree of modification that has occurred to these urban rivers over years of land use change, with the SASS5 scores range between 2 and 15, with an average between 7.68 and 8.58 across all sites. According to the scoring criteria, a score below 42 places the water quality in the "critically modified" (E/F) category. The average score per site were all placed within the "critically modified category" (E/F) confirming severe ecological degradation across all the sampled sites. Given these scores, and the locality of the river, it is it is not surprising to find such low scores because these rivers would be considered critically modified as urban rivers in a major conurbation such as Johannesburg. The results from this study are similar to those found by Lepebe at al. (2021), who applied SASS5 to urban rivers in Durban, South Africa. Their ASPT values, however, ranged from 4 to 8, reflecting less severe ecological modification. The difference between these two studies underlines Johannesburg's greater contamination and pollution from infrastructure failure, urban runoff and sewage contamination. These findings align with those in a study done by Munyika et al. (2014) in similar urban environments.

Delta Park had the highest SASS5 score and taxon richness, which could be attributed to its higher vegetation on the riverbanks, higher sediment quantities and dilution from less polluted streams. The SASS5 scores still fell below the critically modified range, indicating significant urban stressors reducing invertebrate biodiversity. Seasonally, higher scores were recorded in winter months, such as May to August 2023, when the temperatures were cooler. Since cooler temperatures reduce microbial decomposition and nutrient loading, which temporarily stabilises DO levels and benefits invertebrate life. In contrast, summer rainy seasons, between

February and March 2024, coincided with the steepest decline in invertebrate assemblages. This aligns with findings from Baker & Greenfield (2019) on seasonal stress in urban rivers.

This study shows how macroinvertebrate biodiversity is influenced by complex interconnections of physiochemical stressors, microbial contamination and metal toxicity in sediments. The critically modified classification across all sites signals that an urgent need to urban river management intervention is needed, such as stormwater drainage and sewage infrastructure upgrades, regular monitoring of microbial contamination, source identification of metal pollution, particularly at the Montgomery Spruit and Sandton Riverclub sites, as they are hotspots for sediment metal contamination. Another method that should be implemented is riparian restoration to increase habitat complexity and filtration capacity (Baker & Greenfield, 2019). The SASS5 results confirm and provide biological validation of the chemical and microbial assessments made in this study. The dominance of tolerant invertebrates and lack of sensitive taxa indicate that there are acute pollution events occurring and cumulative ecological stress, resulting in deteriorating macroinvertebrate communities and sediment quality.

There are various links between the results of the SASS5 sampling and the physio-chemical results. The macroinvertebrates identified consistently throughout the study, such as Chironomidae, Oligochaeta, Culicidae and Baetidae, are known to tolerate highly polluted environments with low oxygen (Odume & Mgaba, 2016). The absence of pollution-sensitive taxa such as Trichoptera and Ephemeroptera families which have higher SASS scores, indicates poor water quality. These findings are consistent with low DO levels observed throughout the warmer months, especially between February and March 2024, which recorded some of the lowest SASS5 scores and ASPT values. The macroinvertebrate declines in these months corresponds to spikes in nutrient levels, particularly nitrate and phosphate at Montgomery Spruit, Parkview and Delta Park. It also corresponds with elevated *E. coli* and total coliforms, particularly at Delta Park and Sandton Riverclub, likely indicating sewage and urban runoff pollution. These stressors act in combination, and are a characteristic of heavily urbanised catchments, and have been linked to low ASPT scores with degraded ecological conditions in other studies done in South Africa (Dallas, 2007; Munyika et al., 2014). The presence of metals in sediments are another factor contributing to degradation of invertebrate assemblages in these rivers. According to the ecological risk indices, Montgomery, Emmarentia and Riverclub all showed moderate lead contamination, with Riverclub having been categorised as having extremely severe enrichment on certain days. There was significant manganese concentrations recorded at Riverclub and Montgomery, with several exceeding the USEPA threshold of 300

mg/kg. The iron concentration was more widespread, but largely insignificant according to EF and Igeo. The presence of lead and manganese in the sediments aligns with the study done by Dahms et al. (2017), where sediment contamination was linked to benthic invertebrate toxicity and shifts in their community. Chronic exposure to lead has been shown to reduce invertebrate survival and impair their reproduction and development (Ahmad et al., 2022; Atangana & Oberholster, 2021). The metal persists in the sediments and potentially remobilise during rainfall events, increasing the seasonal water quality degradation (Greenfield et al., 2007).

The *E. coli* and total coliform counts revealed substantial bacterial contamination throughout the Braamfontein Spruit during the study period. There were alarming spikes particularly during and after the wetter seasons. The results from this study strongly align with the literature, suggesting that contamination is caused by faecal contamination from leaking sewer systems, stormwater runoff, informal settlement presence and sediment resuspension as dominant pollution pathways in urban rivers (Neill et al., 2018; Wang et al., 2018). The South African DWS and DWAF have stipulated guidelines (1996) for *E. coli* concentration in aquatic environments. They describe water that is less than 130 MPN/100mL as suitable for full-contact recreation, between 131-250 MPN/100mL as acceptable for some activities and anything higher than 250 MPN/100mL as not recommended for recreation. It is important to note that these guidelines are for geometric mean values on at least 5 samples over 30 days. They stipulate that any single sample over 500 MPN/100mL may trigger adverse health impacts (Silberbauer, 1996). There are no stipulated guidelines for total coliforms in aquatic environments as they naturally occur at different concentrations in rivers. According to these guidelines, there were significant *E. coli* bacterial loads observed at each of the sites from this study. The Montgomery Spruit exhibited a relatively low concentration of *E. coli* and total coliforms compared to other sites. However, there were also days when the concentration spiked to significantly higher than the guideline, such as a recording of 10,658 on the 28th of September 2023 85529 MPN/100mL on the 9th of April 2024. These levels suggest recent faecal contamination, likely from leaking sewer infrastructure or illegal discharges. These results correlate with elevated phosphate and nitrate during the same periods, indicating combined microbial and nutrient pollution at this site. Emmarentia generally displayed lower bacterial concentrations than the other sites. However, occasional spikes such as 25920 coliforms on the 2nd of June 2023 and 208,870 coliforms on the 9th of April 2024. Furthermore, *E. coli* increases are observed, peaking at 20270 MPN/100mL on the 9th of April 2024 as well. Interestingly, the *E. coli* concentrations were significantly lower than the total coliform

concentrations, aligning with Foppen and Schijven (2006), who suggest that coliforms persist for longer in sediments compared to *E. coli*, especially under shaded or low-UV conditions. This could potentially indicate that less fresh contamination is present, and more sediment-associated bacteria are present in the river. The Braamfontein Spruit had significantly higher results recorded throughout the study period at each of the sites with *E. coli* values measuring all the way up to 350,000 MPN/100mL at the Parkview Golf Course. This data confirms the findings of Butler (2022), who found significant faecal contamination in the Braamfontein Spruit in that study, with *E. coli* values up to 79,000 MPN (Butler, 2022). It is hypothesised that this contamination is not necessarily caused by systemic sewage ingress from leaky infrastructure, but rather, it is a consequence of indigent people using the Spruit as an open toilet, or stormwater drainage from informal settlements (Vollmer & Grêt-Regamey, 2013). This usage would lead to high *E. coli* values being recorded which correspond to low nutrient loadings (Gannon & Busse, 1989).

Escherichia Coliform concentrations that are this high can have extremely detrimental effects on human health. These values indicate the presence of recent faecal contamination which may contain pathogenic bacteria such as Salmonella and Shigella, viruses such as norovirus and hepatitis, and protozoa such as *Giardia*. These organisms can cause a range of illnesses in humans such as gastroenteritis, skin infections, raspatory infections, urinary tract infections and hepatitis, especially for high-risk groups such as children, the elderly and pregnant women. Parkview consistently recorded extremely high bacterial loads, particularly from the 27th of June 2023 onwards, with *E. coli* reaching 267,400 MPN/100mL and coliforms reaching over 500,000 MPN/100mL. From September 2023 to April 2024, coliform counts remained around 806,000 MPN/100mL, likely indicating chronic contamination from upstream sewer systems or informal settlements. These periods also showed high phosphate and high nitrate concentrations, further supporting the hypothesis of untreated sewage inputs. Parkview is a major public health concern. Delta Park showed intermittent but significant bacterial contamination. *E. coli* exceeded 100,000 MPN/100mL on the 20th of March 2024, and the coliform concentration consistently measured above 300,000 MPN/100mL from September 2023 onward. The findings from Delta Park suggest seasonal sediment disturbance and poorly managed stormwater infrastructure, which are likely exaggerated during rainy seasons in summer and spring. Similarly to the other sites, high bacteria concentrations often coincide with elevated nutrient concentrations. Sandton Riverclub also displayed concerning levels of contamination, including *E. coli* concentrations of 64,650 MPN/100mL on the 28th of

September 2023 and 51,325 MPN/100mL on the 9th of April 2024. These values both exceed the DWS limits significantly. Coliform counts followed the same trend, peaking at 302,450 MPN/100mL. The temporal patterns at Sandton Riverclub suggest that pulses of contamination from urban runoff or leaking sewage infrastructure. This aligns with the findings of similar studies which found strong correlations between rainfall, nutrient surges and faecal bacteria concentrations (Wang et al., 2018). These values indicate that in these recreational areas, people should be made aware of the dangers of high *E. coli* values which are present in these rivers.

In order to measure and quantify key pollutants, including nutrient levels, E-coli, coliform concentrations, and metals in the water and sediment samples, various analytical procedures were followed to identify a range of pollutants and contaminants that are impacting the river.

The physical field measurements showed surprisingly little variation. Other than the expected seasonal changes, there were very few outliers and there was nothing unexpected or significantly different from the expected variations. However, the chemistry of the water showed significant variation, and in many instances, there were clear linkages between nutrient loading and presence of faecal coliforms. This study therefore demonstrates that whilst sewage infrastructure leakages are not always seen or reported, the impact of the leakages and damaged infrastructure on the environment is significant, both in terms of causing eutrophication of still water bodies, and its associated river and human health issues, and damage to human health, especially in recreational areas like Emmarentia Dam.

The risks to human health and aquatic ecosystems are identified through quantification of the results from the river and by comparing the results to guidelines set out by governing bodies like the Department of Water and sanitation (DWS) or by comparing the results to other studies and observing what they have concluded and the reasons behind these conclusions. The DWS has set out Target Water Quality Ranges which were developed according to the effects that contamination at these levels and higher have on aquatic ecosystems and humans.

In aquatic health studies, it is critical to study the quality of sediments as sediments in aquatic ecosystems act as a sink where impurities are stored, making it an important component of the study of the overall health of an urban river. When the metal results from this study are compared to similar studies, the results varied. There is no formal guideline for sediment metal concentration in sediments in South Africa, therefore, other studies often use USEPA (1996) and Canadian guidelines (1999) as a reference for sediment contamination. Metal contamination in urban rivers is a growing environmental concern because of the persistence,

toxicity and the bioaccumulation potential of metals such as iron, manganese and lead. Urban rivers in Johannesburg are particularly at risk since there is increase anthropogenic activity, such as stormwater runoff, industrial discharges, aging infrastructure and historical mining activities. As Dahms et al. (2017) and Ismail et al. (2013) observed, metal pollutants accumulate in sediment over time, making sediment analysis a critical portion of river health assessments. Sediments not only reflect long-term contamination, but also act as potential secondary sources of pollution since they store contaminants and release them into the river during rainfall events or when the sediments are resuspended or disturbed (Greenfield et al., 2007; Wang et al., 2007). This study's sediment sampling revealed that concerning levels of Mn, Pb and Fe were observed across all sites over the sampling period. Site specific variations reflect local land use, pollution sources and environmental conditions. Various risk indices were applied to the results to assess the extent and origin of contamination.

Manganese is essential in trace amounts but becomes toxic when concentrations exceed ecological thresholds. As mentioned, there are no South African sediment quality guidelines, however, USEPA sets a sediment threshold of 300 mg/kg (Ogbeibu et al., 2014). Across the five sites, Montgomery, Emmarentia and Riverclub results showed Mn levels frequently approaching or exceeding the guideline target. Montgomery indicated considerable contamination on one occasion (where CF was 2.64 on the 25th of May 2023) and exceeded the enrichment threshold (where EF was 37.95 on the 14th of August 2023). This indicates possible anthropogenic input linked to industrial activity upstream and stormwater runoff. Riverclub indicated signs of moderate contamination on two separate occasions with extremely severe enrichment (where EF was 179.39) on one sampling day. This could possibly indicate urban runoff in the area and possible infrastructure deterioration. Parkview and Delta Park consistently recorded low Mn levels where CF < 1 and no enrichment detected as EF <1, indicating minimal Mn input from anthropogenic activities. The Igeo for Mn across all sites remained below 1, classifying these sites as unpolluted to moderately polluted. Compared to a study done on the Nyl river in Limpopo by Dahms et al. (2017), where the Mn concentration was widespread but not particularly extreme, the levels in Johannesburg's urban rivers are within a similar, but more variable range.

The lead contamination in urban rivers poses a significant risk due to its non-essential and highly toxic nature (Ahmad et al., 2022). Urban sources include historical use of lead-based paints, vehicle emissions sewage leaks and stormwater runoff. All sites displayed Pb levels which exceed the Canadian TEL of 35 mg/kg and occasionally exceed the PEL of 91.3 mg/kg.

The Montgomery Spruit and Emmarentia exhibited moderate Pb concentration in over half of the sampling events, displayed where CF values are frequently less than 1. They also had enrichment factors which occasionally indicated anthropogenic input, displayed where EF was less than 1. Sandton Riverclub was the most severely impacted site by Pb, with a CF recording greater than 3 and extremely severe EF of 3877.32 on the 9th of November 2023. This indicated significant pollution inputs which are likes a consequence of high-density urban and industrial activities. Parkview and Delta Park showed minor to moderate Pb enrichment where EF was between 1.35 and 6.2. This is consistent with diffuse urban sources such as stormwater runoff from roads and sewers. All sites maintained an Igeo of less than 1, indicating unpolluted to moderately polluted condition in the rivers. This is valuable information as invertebrates and other aquatic species are still impaired, even in low Igeo waters (Wang & Rainbow, 2008). These findings align with Atangana and Oberholser (2021), who documented elevated Pb in the sediments of South African rivers and attributed them to diffuse anthropogenic sources.

The iron in aquatic environments is typically of low ecological risk, especially when naturally derived. However, excessive concentrations can result from anthropogenic activities such as mining, industrial effluent release and urban runoff (Tutu et al., 2008). There are no South African sediment guidelines that exist, however, values exceeding 10000 mg/kg are often considered elevated (Addo-Bediako et al., 2021; Edokpayi et al., 2016). Sandton Riverclub recorded consistently high Fe concentrations in the sediment, with values exceeding 10000 mg/kg. This site also had CF values reaching 2.56 and high PLI values over 3, confirming elevated contamination. Montgomery and Emmarentia showed moderate Fe contamination where CF was between 1.29 and 1.81. This is likely due to a variety of sources of iron such as infrastructure corrosion, urban runoff and background concentrations. Parkview and Delta Park recorded low Fe contamination where CF was less than 1, suggesting more stable conditions or lower natural background levels. Despite these elevated concentrations, all Igeo values for iron were less than 1 and all EF values were 0.75, since iron was used as the background or reference metal. This indicates that no anthropogenic enrichment of iron is occurring at these sites. This supports the idea that Fe in sediments may be more naturally derived in some regions, despite their high total concentrations. The results in this study could potentially be impacted by iron-rich rocks and other environmental factors that may not have been taken into consideration.

The PLI summarises the cumulative metal impacts at each site. Sandton Riverclub displayed the highest pollution loads, particularly on the 7th of June 2023 where PLI was equal to 4.25

and 14th of August 2023 where PLI was equal to 3.29, classifying this site as highly polluted. Montgomery and Emmarentia also experienced moderate pollution days where PLI was greater than 1, while Parkview and Delta Park remained consistently unpolluted, since PLI was less than 1. The PLI data corroborate the sediment metal trends and indicate that there are areas of concern for long-term management within the study area. These trends align with international findings from other studies. In a study on the Hawkesbury-Nepean River, Pinto and Maheshwari (2011) connected pollution to peri-urban development and stormwater runoff, similar to what was observed in the Johannesburg urban river systems. Similarly, the lead concentrations in the sediment in the Mvudi River (Edokpayi et al., 2016) and the Nyl River (Dahms et al., 2017) showed comparable patterns to Riverclub and Emmarentia, expressing the role of urbanisation and land-use pressures in metal enrichment of rivers.

All sites experienced their lowest concentration of lead on the 14th of February 2024. This could be due to rains which were experienced in the week leading up to this day, resulting in movement of lead particles further downstream.

Since there is no background iron concentration for these sites or guidelines for the presence of iron contamination in sediments, Montgomery Spruit was selected as the reference site as there was low metal concentrations expected from this site.

The PLI, EF, Igeo and CF were compared to the results from another study (Dahms et al., 2017). The findings of this study are in line with the findings by Dahms et al (2017) who reported that increased metal contamination in sediments downstream from anthropogenic sources in the Nyl River System in Limpopo. Similarly, the results from their samples indicated that sites such as the Nylsvley floodplain and Moorddrift Dam exceeded sediments quality guidelines (SQGs) and exhibited high PLI. This study found that elevated concentrations of metals, particularly manganese and iron, at urban-impacted sites. Both studies made use of sediment risk assessment indices (CF, EF, PLI and Igeo), which consistently indicated moderately to high ecological risk at multiple sites as a result of cumulative impacts from sewage effluent, agricultural inputs and urban runoff.

The data recorded throughout the study indicates that there are periods where the quality of the water is not polluted beyond DWS guideline limits (1996), however, without ongoing monitoring it would be erroneous to say that the water is safe for recreational use. This statement is true, even for the rivers like the Westdene Spruit in Emmarentia, since without ongoing monitoring, new pollutants and contaminants can enter the river without anyone

knowing, causing disastrous effects on the environment and human health. Most water bodies in Johannesburg, and certainly all of the urban rivers in this study should have warning signs, indicating that there could possibly be faecal bacteria in the water, and metal contamination beyond toxic levels in the river sediments. These signs should indicate to people passing by that they should not enter the water due to potentially toxic health impact. It is recommended that an ongoing monitoring campaign is implemented in order to understand the temporal data over longer timeframes not just for one season. These types of surveys would typically have to be conducted by the City Council. The study provides scientific evidence that there is an impact on public water bodies by sewage and through other anthropogenic activities which are not specified which renders water bodies toxic to humans, and these should therefore be avoided.

The limitations of this study are the low frequency of sampling: due to budgetary constraints sampling could not be conducted weekly rather it was done on a biweekly or monthly basis. It would be far better if the study were done daily or perhaps weekly. The *E. coli*, total coliforms and metals in the water and sediments could not be measured each time sampling occurred due to budgetary restraints. It would have provided more accurate results with a higher number of metal measurements and *E. coli* results. SASS5 was not conducted at each site for a full 15 minutes. This was shortened to just under 10 minutes, but was kept consistent at each site, in order to reduce the time spent at each site as weather conditions were bad and not all of the sites were completely safe to be around for too long. The third sampling campaign was one week shorter than the others as time constraints towards the end of the sampling campaign restricted the sampling trips. It would have been beneficial for this study if there was a sampling site located further downstream from the Sandton River Club to identify what changes in water quality occur between there and the new site.

The Braamfontein, Westdene and Montgomery Spruits display the symptoms of an impacted urban river with disturbances occurring at various points along the rivers. Disturbances are negatively impacting the water quality and habitat availability for biodiversity. It was found that the degraded water quality has had altering effects on species diversity of the riverine aquatic macroinvertebrate communities. Representatives of the mayfly family Oligoneuriidae, a highly sensitive macroinvertebrate family, were found at each of the sites, but not consistently throughout the study, suggesting influxes of pollution reducing their population.

On the basis of cost restrictions, we had to choose which aquatic elements to analyse. Due to the known risks associated with lead, this was chosen. Notwithstanding this, manganese and iron may also have toxic effects and where therefore considered in this study.

6. Conclusion

This study assessed the ecological health of urban rivers in Johannesburg, South Africa, using an integrated, multidisciplinary approach. By analysing the in-filed measurements (pH, EC, DO and temperature), nutrient (nitrate, nitrite, phosphate and sulphate), microbial (*E. coli* and total coliforms), biological (macroinvertebrate communities through SASS5 sampling) and water and sediment-based metal contamination (Pb, Mn and Fe) indicators across five sites. The results of this study provide a robust evaluation of riverine conditions in a highly urbanised catchment.

The results confirm that these urban rivers are critically modified systems which are heavily impacted by a combination of anthropogenic nutrient enrichment, bacterial contamination and metal pollution in sediments. Most of the parameters measured in this study exceed the limits of the South African Department of Water and Sanitation (DWS) guidelines (1996) for aquatic ecosystem protection multiple times and at all sites (Silberbauer, 1996).

Inorganic nutrients, specifically nitrate and phosphate, were frequently elevated above DWS (1996) limits, particularly at Montgomery, Emmarentia and Riverclub. These nutrient spikes are likely driven by surface runoff, leaking sanitation infrastructure and wastewater discharge. They correlate with seasonal changes and are especially prominent during the warmer months. Nutrient enrichment was linked to eutrophication symptoms such as algal growth and oxygen depletion, as discussed in the literature by Silberbauer (1996) and Shousha et al. (2021). These symptoms negatively affected aquatic biodiversity and overall river health.

Microbial contamination, specifically *E. coli* and total coliforms, was widespread and severe across most sites, but substantially higher at Parkview, which constantly exhibited the highest *E. coli* and coliform levels. Most sights, however, recorded values consistently surpassing the DWS water quality guideline limits for recreational and domestic use. These results suggest chronic exposure to intense faecal matter pollution, likely from multiple sources such as stormwater drainage, urban runoff from nearby residential areas, leaking sewerage infrastructure, and presence of unplanned settlements or illegal dumping upstream. Parkview's readings frequently exceeded those of Riverclub and Delta Park, which were also highly contaminated, confirming the intense microbial pollution affecting the Braamfontein Spruit from its source above Parkview all the way downstream to Sandton Riverclub. It must be taken into consideration that the droppings from birds and dogs could potentially impact the bacterial results of this study. The significant decrease in bacterial loads between Parkview and Delta

Park were attributed to a vegetation buffer filtering the water, dilution from groundwater or baseflow, or dilution from the Montgomery and Westdene Spruits. The high bacterial loads are consistent with elevated nutrient loads observed at the sites on some days, supporting the findings of Wang et al. (2018) that nutrient enrichment and faecal contamination are often strongly correlated in urban rivers.

The conclusion of the SASS5 sampling suggests that the river is in a "critically modified" (E/F) ecological state. There are indication of persistent pollution and urban impacts have severely degraded the macroinvertebrate community. Worsening scores and values in early 2024 suggest increase pollution inputs. SASS5 scores validate findings from the water chemistry data, with particularly high microbial contamination and nutrient pollution. Urgent conservation and rehabilitation efforts are needed to restore ecological integrity of these rivers to a state that is not dangerous to human health. People who use these recreational waters and come into contact with the water need to be made aware of the dangers they are exposed to. Signs need to be put up warning people of the possibility of high *E. coli* concentrations and the possible effects they can have.

Sediment analysis revealed persistent metal contamination, particularly of lead and manganese, at multiple sites throughout the study period. Contamination Factor (CF), Enrichment Factor (EF), Geoaccumulation index (Igeo) and Pollution Load Index (PLI) values confirmed moderate to high ecological risk, especially at Sandton Riverclub, which showed extremely severe enrichment of both lead and manganese on particular days. These findings are aligned with the broader literature on sediment-bound metal pollution, and the potential for sediment and pollution resuspension during rainfall events (Dahms et al., 2017; Villiers & Mkwelo, 2009). The sediment metal data from this study show that iron is broadly present across all sites, but manganese and lead are more site specific and attributed to anthropogenic influence, such as vehicle emissions and brake wear which enter the river through urban runoff. Riverclub and Montgomery emerge as the most affected sites by sediment metal contamination, exhibiting moderate to high pollution and evidence of metal enrichment. These findings are supported by the ecological risk indices and are consistent with national and international literature. Understanding the sediment metal contamination is essential for effective urban river management. Continued monitoring and the development of national sediment quality guidelines in South Africa remains critical steps in managing metal contamination and protecting aquatic ecosystems.

The combination of field measurements, water samples, sediment samples and SASS5 sampling provide a holistic view of river health in Johannesburg. This study demonstrates the value of integrating chemical, microbial, and biological assessments when evaluating the health of urban rivers. While individual spikes in various parameters may reflect short-term conditions, the macroinvertebrate data provided a time-integrated reflection of sustained ecological stress, which is consistent with the findings of Dallas (2007) and Baker and Greenfield (2019). The cumulative impacts observed in this study show the chronic nature of urban pollution in Johannesburg's urban river systems, which are exacerbated by rapid urbanisation, old, decaying and failing infrastructure, and limited enforcement of environmental regulations.

In order to address the issues of urban river contamination, it is recommended that urgent upgrades to wastewater and stormwater infrastructure, especially along the Braamfontein Spruit where microbial and nutrient contamination is persistent. Targeted sediment remediation strategies are implemented, especially at Riverclub and Montgomery Spruit where metal concentrations were highest. Projects targeting riparian zone rehabilitation and green infrastructure implementation to improve natural filtration of urban runoff and reduce direct inflows. Continued biomonitoring using SASS5 sampling techniques, paired with microbial and chemical testing, is needed to track changes and recovery trends, informing adaptive management of invertebrate assemblages. Officials need to establish a background sediment metal baseline and develop standardised South African specific sediment quality guidelines to use for future assessments.

This research contributes to the growing body of evidence on the state of urban river health in South Africa and highlights the need for holistic, long-term water resource management. By integrating traditional water quality data with biomonitoring and sediment analysis, this study shows the interconnections of chemical, physical and biological stressors on urban rivers, as well as the necessity of multi-faceted solutions to restore and preserve urban freshwater ecosystems.

References

Adams, J. B., Taljaard, S., van Niekerk, L., & Lemley, D. A. (2019). *Nutrient enrichment as a threat to the ecological resilience and health of South African microtidal estuaries: African Journal of Aquatic Science: Vol 45, No 1-2*. https://www.tandfonline.com/doi/abs/10.2989/16085914.2019.1677212

Addo-Bediako, A., Nukeri, S., & Kekana, M. (2021). Heavy metal and metalloid contamination in the sediments of the Spekboom River, South Africa. *Applied Water Science*, *11*(7), 133. https://doi.org/10.1007/s13201-021-01464-8

Ahmad, K., Shah, H.-R., Khan, M. S., Iqbal, A., Potrich, E., Amaral, L. S., Rasheed, S., Nawaz, H., Ayub, A., Naseem, K., Muhammad, A., Yaqoob, M. R., & Ashfaq, M. (2022). Lead In drinking water: Adsorption method and role of zeolitic imidazolate frameworks for its remediation: A review. *Journal of Cleaner Production*, *368*, 133010. https://doi.org/10.1016/j.jclepro.2022.133010

Akhtar, N., Syakir Ishak, M. I., Bhawani, S. A., & Umar, K. (2021). Various Natural and Anthropogenic Factors Responsible for Water Quality Degradation: A Review. *Water*, *13*(19), Article 19. https://doi.org/10.3390/w13192660

Allard, T., Menguy, N., Salomon, J., Calligaro, T., Weber, T., Calas, G., & Benedetti, M. F. (2004). Revealing forms of iron in river-borne material from major tropical rivers of the Amazon Basin (Brazil)1. *Geochimica et Cosmochimica Acta*, *68*(14), 3079–3094. https://doi.org/10.1016/j.gca.2004.01.014

An, Y.-J., Kampbell, D. H., & Peter Breidenbach, G. (2002). *Escherichia coli* and total coliforms in water and sediments at lake marinas. *Environmental Pollution*, *120*(3), 771–778. https://doi.org/10.1016/S0269-7491(02)00173-2

Angove, C., Norkko, A., & Gustafsson, C. (2018). Assessing the efficiencies and challenges for nutrient uptake by aquatic plants. *Journal of Experimental Marine Biology and Ecology*, *507*, 23–30. https://doi.org/10.1016/j.jembe.2018.07.005

Anh, N. T., Can, L. D., Nhan, N. T., Schmalz, B., & Luu, T. L. (2023). Influences of key factors on river water quality in urban and rural areas: A review. *Case Studies in Chemical and Environmental Engineering*, *8*, 100424. https://doi.org/10.1016/j.cscee.2023.100424

Atangana, E., & Oberholster, P. J. (2021). Using heavy metal pollution indices to assess water quality of surface and groundwater on catchment levels in South Africa. *Journal of African Earth Sciences*, *182*, 104254. https://doi.org/10.1016/j.jafrearsci.2021.104254

Avery, L. M., Williams, A. P., Killham, K., & Jones, D. L. (2008). Survival of *Escherichia coli* O157:H7 in waters from lakes, rivers, puddles and animal-drinking troughs. *Science of The Total Environment*, *389*(2), 378–385. https://doi.org/10.1016/j.scitotenv.2007.08.049

Ayrault, S., Roy-Barman, M., Le Cloarec, M.-F., Priadi, C. R., Bonté, P., & Göpel, C. (2012). Lead contamination of the Seine River, France: Geochemical implications of a historical perspective. *Chemosphere*, *87*(8), 902–910. https://doi.org/10.1016/j.chemosphere.2012.01.043

Bae, S., & Seo, D. (2021). Changes in algal bloom dynamics in a regulated large river in response to eutrophic status. *Ecological Modelling*, *454*, 109590. https://doi.org/10.1016/j.ecolmodel.2021.109590

Baker, N. J., & Greenfield, R. (2019). Shift happens: Changes to the diversity of riverine aquatic macroinvertebrate communities in response to sewage effluent runoff. *Ecological Indicators*, *102*, 813–821. https://doi.org/10.1016/j.ecolind.2019.03.021

Banerjee, P., Garai, P., Saha, N. C., Saha, S., Sharma, P., & Maiti, A. K. (2023). A critical review on the effect of nitrate pollution in aquatic invertebrates and fish. *Water, Air, & Soil Pollution*, *234*(6), 333. https://doi.org/10.1007/s11270-023-06260-5

Barboza, N. R., Guerra-Sá, R., & Leão, V. A. (2016). Mechanisms of manganese bioremediation by microbes: An overview. *Journal of Chemical Technology & Biotechnology*, *91*(11), 2733–2739. https://doi.org/10.1002/jctb.4997

Benavente, D., Ruiz, M. C., Garcia-Martinez, N., Vergara, M., Fernandez-Cortes, A., & Sanchez-Moral, S. (2024). *Estimation of Mineral Saturation and Co2 Partial Pressure in Natural Waters Using Electrical Conductivity and Ph: A Fast and Versatile On-Site Tool*. https://doi.org/10.2139/ssrn.5066615

Bere, T., & Nyamupingidza, B. B. (2014). Use of biological monitoring tools beyond their country of origin: A case study of the South African Scoring System Version 5 (SASS5). *Hydrobiologia*, *722*(1), 223–232. https://doi.org/10.1007/s10750-013-1702-7

Best, M. A., Wither, A. W., & Coates, S. (2007). Dissolved oxygen as a physico-chemical supporting element in the Water Framework Directive. *Marine Pollution Bulletin*, *55*(1), 53–64. https://doi.org/10.1016/j.marpolbul.2006.08.037

Birch, G. F. (2023). A review and critical assessment of sedimentary metal indices used in determining the magnitude of anthropogenic change in coastal environments. *Science of The Total Environment*, *854*, 158129. https://doi.org/10.1016/j.scitotenv.2022.158129

Boyd, C. E. (2015). pH, Carbon Dioxide, and Alkalinity. In C. E. Boyd (Ed.), *Water Quality: An Introduction* (pp. 153–178). Springer International Publishing. https://doi.org/10.1007/978-3-319-17446-4_8

Boyd, C. E., Tucker, C. S., & Somridhivej, B. (2016). Alkalinity and Hardness: Critical but Elusive Concepts in Aquaculture. *Journal of the World Aquaculture Society*, *47*(1), 6–41. https://doi.org/10.1111/jwas.12241

Brans, K. I., Jansen, M., Vanoverbeke, J., Tüzün, N., Stoks, R., & De Meester, L. (2017). The heat is on: Genetic adaptation to urbanization mediated by thermal tolerance and body size. *Global Change Biology*, *23*(12), 5218–5227. https://doi.org/10.1111/gcb.13784

Brezonik, P. L. (2013). DENITRIFICATION IN NATURAL WATERS. In S. H. Jenkins (Ed.), *Proceedings of the Conference on Nitrogen As a Water Pollutant* (pp. 373–392). Pergamon. https://doi.org/10.1016/B978-1-4832-1344-6.50030-7

Buckalew, D. W., Hartman, L. J., Grimsley, G. A., Martin, A. E., & Register, K. M. (2006). A long-term study comparing membrane filtration with Colilert® defined substrates in detecting fecal coliforms and *Escherichia coli* in natural waters. *Journal of Environmental Management*, *80*(3), 191–197. https://doi.org/10.1016/j.jenvman.2005.08.024

Butler, D. (2022). *The temporal assessment of three urban sperate urban rivers in Johannesburg*.

Carter, S., Clough, R., Fisher, A., Gibson, B., Russell, B., & Waack, J. (2018). Atomic spectrometry update: Review of advances in the analysis of metals, chemicals and materials. *Journal of Analytical Atomic Spectrometry*, *33*(11), 1802–1848. https://doi.org/10.1039/C8JA90039F

Chakraborty, P., Chakraborty, S., Jayachandran, S., Madan, R., Sarkar, A., Linsy, P., & Nath, B. N. (2016). Effects of bottom water dissolved oxygen variability on copper and lead fractionation in the sediments across the oxygen minimum zone, western continental margin of India. *Science of The Total Environment*, *566–567*, 1052–1061. https://doi.org/10.1016/j.scitotenv.2016.05.125

Chon, H.-S., Ohandja, D.-G., & Voulvoulis, N. (2012). The role of sediments as a source of metals in river catchments. *Chemosphere*, *88*(10), 1250–1256. https://doi.org/10.1016/j.chemosphere.2012.03.104

Chowdhury, S., Mazumder, M. A. J., Al-Attas, O., & Husain, T. (2016). Heavy metals in drinking water: Occurrences, implications, and future needs in developing countries. *Science of The Total Environment*, *569–570*, 476–488. https://doi.org/10.1016/j.scitotenv.2016.06.166

Dahms, S., Baker, N. J., & Greenfield, R. (2017). Ecological risk assessment of trace elements in sediment: A case study from Limpopo, South Africa. *Ecotoxicology and Environmental Safety*, *135*, 106–114. https://doi.org/10.1016/j.ecoenv.2016.09.036

Dallas, H. (2008). Water temperature and riverine ecosystems: An overview of knowledge and approaches for assessing biotic responses, with special reference to South Africa. *Water SA*, *34*(3), Article 3. https://doi.org/10.4314/wsa.v34i3.180634

Dallas, H. F. (2007). *SOUTH AFRICAN SCORING SYSTEM (SASS) DATA INTEPRETATION GUIDELINES*.

Davie, L. (2021). *Gauteng's only artesian spring | The Heritage Portal*. https://www.theheritageportal.co.za/article/gautengs-only-artesian-spring?utm_source=chatgpt.com

De, la R. P. A., Taylor, J. C., Laas, A., Van, R. L., & Vosloo, A. (2004). Determining the possible application value of diatoms as indicators of general water quality: A comparison with SASS 5. *Water SA*, *30*(3), 325–332. https://doi.org/10.10520/EJC116169

Deng, Z., Huang, D., He, Q., & Chassagne, C. (2022). Review of the action of organic matter on mineral sediment flocculation. *Frontiers in Earth Science*, *10*. https://doi.org/10.3389/feart.2022.965919

Diaz, R. J., & Breitburg, D. L. (2009). Chapter 1 The Hypoxic Environment. In J. G. Richards, A. P. Farrell, & C. J. Brauner (Eds.), *Fish Physiology* (Vol. 27, pp. 1–23). Academic Press. https://doi.org/10.1016/S1546-5098(08)00001-0

Dickens, C., & Graham, P. M. (2002). The South African Scoring System (SASS) Version 5 Rapid Bioassessment Method for Rivers. *African Journal of Aquatic Science - AFR J AQUAT SCI*, *27*, 1–10. https://doi.org/10.2989/16085914.2002.9626569

Divjak, B., & Goessler, W. (1999). Ion chromatographic separation of sulfur-containing inorganic anions with an ICP–MS as element-specific detector. *Journal of Chromatography A*, *844*(1), 161–169. https://doi.org/10.1016/S0021-9673(99)00409-4

Edokpayi, J. N., Odiyo, J. O., Popoola, O. E., & Msagati, T. A. M. (2016). Assessment of Trace Metals Contamination of Surface Water and Sediment: A Case Study of Mvudi River, South Africa. *Sustainability*, *8*(2), Article 2. https://doi.org/10.3390/su8020135

Everard, M., & Moggridge, H. (2012). Rediscovering the Value of Urban Rivers. *Urban Ecosystems*, *15*. https://doi.org/10.1007/s11252-011-0174-7

Fadlillah, L. N., Utami, S., Rachmawati, A. A., Jayanto, G. D., & Widyastuti, M. (2023). Ecological risk and source identifications of heavy metals contamination in the water and surface sediments from anthropogenic impacts of urban river, Indonesia. *Heliyon*, *9*(4), e15485. https://doi.org/10.1016/j.heliyon.2023.e15485

Fenoglio, S., Bo, T., Cucco, M., Mercalli, L., & Malacarne, G. (2010). Effects of global climate change on freshwater biota: A review with special emphasis on the Italian situation. *Italian Journal of Zoology*, *77*(4), 374–383. https://doi.org/10.1080/11250000903176497

Ferreira, S. L. C., da Silva, J. B., dos Santos, I. F., de Oliveira, O. M. C., Cerda, V., & Queiroz, A. F. S. (2022). Use of pollution indices and ecological risk in the assessment of contamination from chemical

elements in soils and sediments – Practical aspects. *Trends in Environmental Analytical Chemistry*, *35*, e00169. https://doi.org/10.1016/j.teac.2022.e00169

Flores-Guia, T. E., Cano Salazar, L. F., Martínez-Luévanos, A., & Claudio-Rizo, J. A. (2021). Manganese Oxides: Synthesis and Application as Adsorbents of Heavy Metal Ions. In O. V. Kharissova, L. M. Torres-Martínez, & B. I. Kharisov (Eds.), *Handbook of Nanomaterials and Nanocomposites for Energy and Environmental Applications* (pp. 2409–2428). Springer International Publishing. https://doi.org/10.1007/978-3-030-36268-3_153

Foppen, J. W. A., & Schijven, J. F. (2006). Evaluation of data from the literature on the transport and survival of *Escherichia coli* and thermotolerant coliforms in aquifers under saturated conditions. *Water Research*, *40*(3), 401–426. https://doi.org/10.1016/j.watres.2005.11.018

Förstner, U., & Wittmann, G. T. W. (2012). *Metal Pollution in the Aquatic Environment*. Springer Science & Business Media.

Foster, I. D. L., & Charlesworth, S. M. (1996). Heavy metals in the hydrological cycle: Trends and explanation. *Hydrological Processes*, *10*(2), 227–261. https://doi.org/10.1002/(SICI)1099-1085(199602)10:2<227::AID-HYP357>3.0.CO;2-X

Franklin, P. (2014). Dissolved oxygen criteria for freshwater fish in New Zealand: A revised approach. *New Zealand Journal of Marine and Freshwater Research*, *48*(1), 112–126. https://doi.org/10.1080/00288330.2013.827123

Gannon, J. J., & Busse, M. K. (1989). *E. coli* and enterococci levels in urban stormwater, river water and chlorinated treatment plant effluent. *Water Research*, *23*(9), 1167–1176. https://doi.org/10.1016/0043-1354(89)90161-9

Garcia-Armisen, T., & Servais, P. (2007). Respective contributions of point and non-point sources of *E. coli* and enterococci in a large urbanized watershed (the Seine river, France). *Journal of Environmental Management*, *82*(4), 512–518. https://doi.org/10.1016/j.jenvman.2006.01.011

Göbel, P., Dierkes, C., & Coldewey, W. G. (2007). Storm water runoff concentration matrix for urban areas. *Journal of Contaminant Hydrology*, *91*(1), 26–42. https://doi.org/10.1016/j.jconhyd.2006.08.008

Gordon, A. K., & Muller, W. J. (2010). *(PDF) DEVELOPING SEDIMENT QUALITY GUIDELINES FOR SOUTH AFRICA PHASE 1: Identification of international best practice and applications for South Africa to develop a research and implementation framework. Report to the Water Research Commission*. ResearchGate. https://www.researchgate.net/publication/266678828_DEVELOPING_SEDIMENT_QUALITY_GUIDELINES_FOR_SOUTH_AFRICA_PHASE_1_Identification_of_international_best_practice_and_applications_for_South_Africa_to_develop_a_research_and_implementation_framework_Repor

Graham, P. M., Dickens, C. W., & Taylor, R. J. (2004). miniSASS — A novel technique for community participation in river health monitoring and management. *African Journal of Aquatic Science*, *29*(1), 25–35. https://doi.org/10.2989/16085910409503789

Greenfield, R., Vuren, J. H. J. van, & Wepener, V. (2007). Determination of sediment quality in the Nyl River system, Limpopo Province, South Africa. *Water SA*, *33*(5), Article 5. https://doi.org/10.4314/wsa.v33i5.184090

Grzybkowska, M., Kucharski, L., Dukowska, M., Takeda, A. M., Lik, J., & Leszczyńska, J. (2017). Submersed aquatic macrophytes and associated fauna as an effect of dam operation on a large

lowland river. *Ecological Engineering*, *99*, 256–264. https://doi.org/10.1016/j.ecoleng.2016.11.023

Gurnell, A., Lee, M., & Souch, C. (2007). Urban Rivers: Hydrology, Geomorphology, Ecology and Opportunities for Change. *Geography Compass*, *1*(5), 1118–1137. https://doi.org/10.1111/j.1749-8198.2007.00058.x

Hakanson, L. (1980). An ecological risk index for aquatic pollution control.a sedimentological approach. *Water Research*, *14*(8), 975–1001. https://doi.org/10.1016/0043-1354(80)90143-8

Hansen, D. L., Clark, J. J., Ishii, S., Sadowsky, M. J., & Hicks, R. E. (2008). Sources and Sinks of *Escherichia coli* in Benthic and Pelagic Fish. *Journal of Great Lakes Research*, *34*(2), 228–234. https://doi.org/10.3394/0380-1330(2008)34[228:SASOEC]2.0.CO;2

He, B., Kanae, S., Oki, T., Hirabayashi, Y., Yamashiki, Y., & Takara, K. (2011). Assessment of global nitrogen pollution in rivers using an integrated biogeochemical modeling framework. *Water Research*, *45*(8), 2573–2586. https://doi.org/10.1016/j.watres.2011.02.011

Heikens, A., Peijnenburg, W. J. G. M., & Hendriks, A. J. (2001). Bioaccumulation of heavy metals in terrestrial invertebrates. *Environmental Pollution*, *113*(3), 385–393. https://doi.org/10.1016/S0269-7491(00)00179-2

Hellawell, J. M. (1988). Toxic substances in rivers and streams. *Environmental Pollution*, *50*(1), 61–85. https://doi.org/10.1016/0269-7491(88)90185-6

Howarth, R. W., & Michaels, A. F. (2000). The Measurement of Primary Production in Aquatic Ecosystems. In O. E. Sala, R. B. Jackson, H. A. Mooney, & R. W. Howarth (Eds.), *Methods in Ecosystem Science* (pp. 72–85). Springer. https://doi.org/10.1007/978-1-4612-1224-9_6

Ismail, Z., Salim, K., Othman, S. Z., Ramli, A. H., Shirazi, S. M., Karim, R., & Khoo, S. Y. (2013). Determining and comparing the levels of heavy metal concentrations in two selected urban river water. *Measurement*, *46*(10), 4135–4144. https://doi.org/10.1016/j.measurement.2013.08.013

Jackson, R. N., Baird, D., & Els, S. (2005). The effect of the heavy metals lead (Pb2+) and zinc (Zn2+) on. *Water SA*, *31*(1), Article 1. https://doi.org/10.4314/wsa.v31i1.5127

Kang, M., Tian, Y., Peng, S., & Wang, M. (2019). Effect of dissolved oxygen and nutrient levels on heavy metal contents and fractions in river surface sediments. *Science of The Total Environment*, *648*, 861–870. https://doi.org/10.1016/j.scitotenv.2018.08.201

Kato, T., Kuroda, H., & Nakasone, H. (2009). Runoff characteristics of nutrients from an agricultural watershed with intensive livestock production. *Journal of Hydrology*, *368*(1), 79–87. https://doi.org/10.1016/j.jhydrol.2009.01.028

Kazmi, S. S. U. H., Wang, Y. Y. L., Cai, Y.-E., & Wang, Z. (2022). Temperature effects in single or combined with chemicals to the aquatic organisms: An overview of thermo-chemical stress. *Ecological Indicators*, *143*, 109354. https://doi.org/10.1016/j.ecolind.2022.109354

Keen, C. L., Ensunsa, J. L., & Clegg, M. S. (2000). Manganese Metabolism in Animals and Humans Including the Toxicity of Manganese. In *Metal Ions in Biological Systems*. CRC Press.

Kolev, N. I. (2012). Solubility of O2, N2, H2 and CO2 in water. In N. I. Kolev (Ed.), *Multiphase Flow Dynamics 4: Turbulence, Gas Adsorption and Release, Diesel Fuel Properties* (pp. 209–239). Springer. https://doi.org/10.1007/978-3-642-20749-5_11

Lebepe, J., Khumalo, N., Mnguni, A., Pillay, S., & Mdluli, S. (2021). Data of macroinvertebrates assemblage across different stretches of an urban Palmiet River in Durban, South Africa. *Data in Brief*, *39*, 107493. https://doi.org/10.1016/j.dib.2021.107493

Lee, J.-W., Choi, H., Hwang, U.-K., Kang, J.-C., Kang, Y. J., Kim, K. I., & Kim, J.-H. (2019). Toxic effects of lead exposure on bioaccumulation, oxidative stress, neurotoxicity, and immune responses in fish: A review. *Environmental Toxicology and Pharmacology*, *68*, 101–108. https://doi.org/10.1016/j.etap.2019.03.010

Li, L., Stewart, B., Zhi, W., Sadayappan, K., Ramesh, S., Kerins, D., Sterle, G., Harpold, A., & Perdrial, J. (2022). *Climate Controls on River Chemistry—Li - 2022 - Earth's Future - Wiley Online Library*. https://agupubs.onlinelibrary.wiley.com/doi/full/10.1029/2021EF002603

Lishman, L. A., Legge, R. L., & Farquhar, G. J. (2000). Temperature effects on wastewater treatment under aerobic and anoxic conditions. *Water Research*, *34*(8), 2263–2276. https://doi.org/10.1016/S0043-1354(99)00393-0

Luederitz, C., Brink, E., Gralla, F., Hermelingmeier, V., Meyer, M., Niven, L., Panzer, L., Partelow, S., Rau, A.-L., Sasaki, R., Abson, D. J., Lang, D. J., Wamsler, C., & von Wehrden, H. (2015). A review of urban ecosystem services: Six key challenges for future research. *Ecosystem Services*, *14*, 98–112. https://doi.org/10.1016/j.ecoser.2015.05.001

Mahadevan, H., Krishnan, K. A., Pillai, R. R., & Sudhakaran, S. (2020). Assessment of urban river water quality and developing strategies for phosphate removal from water and wastewaters: Integrated monitoring and mitigation studies. *SN Applied Sciences*, *2*(4), 772. https://doi.org/10.1007/s42452-020-2571-0

Marks, B., Peters, A., & McGough, D. (2017). Aquatic environmental risk assessment of manganese processing industries. *NeuroToxicology*, *58*, 187–193. https://doi.org/10.1016/j.neuro.2016.04.011

Martínez, C. E., & Motto, H. L. (2000). Solubility of lead, zinc and copper added to mineral soils. *Environmental Pollution*, *107*(1), 153–158. https://doi.org/10.1016/S0269-7491(99)00111-6

Mathee, A. (2014). Towards the prevention of lead exposure in South Africa: Contemporary and emerging challenges. *NeuroToxicology*, *45*, 220–223. https://doi.org/10.1016/j.neuro.2014.07.007

Mavasa, W. (2023). *Water and Sanitation releases 2023 full Blue Drop Report | South African Government*. https://www.gov.za/news/media-statements/water-and-sanitation-releases-2023-full-blue-drop-report-05-dec-2023?utm_source=chatgpt.com

Mavhungu, T. C., & Mankga, L. T. (2024). The Impact of Sewage Water Overflow (SWO): An Analysis of Zandspruit River in Cosmo City, Johannesburg, South Africa. *Asian Journal of Water, Environment and Pollution*, *21*(4), 63–71. https://doi.org/10.3233/AJW240047

Meyer-Reil, L.-A., & Köster, M. (2000). Eutrophication of Marine Waters: Effects on Benthic Microbial Communities. *Marine Pollution Bulletin*, *41*(1), 255–263. https://doi.org/10.1016/S0025-326X(00)00114-4

Mihu-Pintilie, A., Romanescu, G., & Stoleriu, C. C. (2014). The seasonal changes of the temperature, pH and dissolved oxygen in the Cuejdel Lake, Romania. *Carpathian Journal of Earth and Environmental Sciences*, *9*, 113–123.

Moreno-Casas, P., Aral, H., & Vecchio-Sadus, A. (2009). *ENVIRONMENTAL IMPACT AND TOXICOLOGY OF SULPHATE*.

Muller, G. (1979). *Index of geoaccumulation in sediments of the Rhine River*. 108–118.

Munyika, S., Kongo, V., & Kimwaga, R. (2014). River health assessment using macroinvertebrates and water quality parameters: A case of the Orange River in Namibia. *Physics and Chemistry of the Earth, Parts A/B/C*, *76–78*, 140–148. https://doi.org/10.1016/j.pce.2015.01.001

Neculita, C. M., & Rosa, E. (2019). A review of the implications and challenges of manganese removal from mine drainage. *Chemosphere*, *214*, 491–510. https://doi.org/10.1016/j.chemosphere.2018.09.106

Neill, A. J., Tetzlaff, D., Strachan, N. J. C., Hough, R. L., Avery, L. M., Watson, H., & Soulsby, C. (2018). Using spatial-stream-network models and long-term data to understand and predict dynamics of faecal contamination in a mixed land-use catchment. *Science of The Total Environment*, *612*, 840–852. https://doi.org/10.1016/j.scitotenv.2017.08.151

Neill, M. (2004). Microbiological Indices for total coliform and *E. coli* bacteria in estuarine waters. *Marine Pollution Bulletin*, *49*(9), 752–760. https://doi.org/10.1016/j.marpolbul.2004.05.016

Newman, B. K., & Watling, J. (2007). Definition of baseline metal concentrations for assessing metal enrichment of sediment from the south-eastern Cape coastline of South Africa. *Water SA*, *33*(5), 675–692.

Nieva, N. E., Borgnino, L., & García, M. G. (2018). Long term metal release and acid generation in abandoned mine wastes containing metal-sulphides. *Environmental Pollution*, *242*, 264–276. https://doi.org/10.1016/j.envpol.2018.06.067

Nimick, D. A., Gammons, C. H., Cleasby, T. E., Madison, J. P., Skaar, D., & Brick, C. M. (2003). Diel cycles in dissolved metal concentrations in streams: Occurrence and possible causes. *Water Resources Research*, *39*(9). https://doi.org/10.1029/2002WR001571

Ntengwe, F. W. (2006). Pollutant loads and water quality in streams of heavily populated and industrialised towns. *Physics and Chemistry of the Earth, Parts A/B/C*, *31*(15), 832–839. https://doi.org/10.1016/j.pce.2006.08.025

Nyieku, F. E., Essandoh, H. M. K., Armah, F. A., & Awuah, E. (2022). Modelling the interaction between physico-chemical and bacteriological characteristics of oilfields produced water from a waste management facility. *Cleaner Waste Systems*, *3*, 100054. https://doi.org/10.1016/j.clwas.2022.100054

Ochieng, G. M., Seanego, E. S., & Nkwonta, O. I. (2010). Impacts of mining on water resources in South Africa: A review. *Sci. Res. Essays*.

Odume, N., & Mgaba, N. (2016). Statistical analysis of macroinvertebrate assemblage structure in relation to river-health assessment of an urban river, Eastern Cape, South Africa. *Aquatic Ecosystem Health & Management*, *19*, 420–430. https://doi.org/10.1080/14634988.2016.1255098

Ogbeibu, A., Omoigberale, M., Maxwell, I., Eziza, J., & Igwe, J. (2014). Using Pollution Load Index and Geoaccumulation Index for the Assessment of Heavy Metal Pollution and Sediment Quality of the Benin River, Nigeria. *Natural Environment*, *2*, 1. https://doi.org/10.12966/ne.05.01.2014

Omarjee, A., Taljaard, S., Ramjukadh, C.-L., & van Niekerk, L. (2021). pH variability in catchment flows to estuaries – A South African perspective. *Estuarine, Coastal and Shelf Science*, *262*, 107605. https://doi.org/10.1016/j.ecss.2021.107605

Papina, T. S., Eirikh, A. N., Serykh, T. G., & Dryupina, E. Yu. (2017). Space and time regularities in the distribution of dissolved and suspended manganese forms in Novosibirsk Reservoir water. *Water Resources*, *44*(2), 276–283. https://doi.org/10.1134/S0097807817020105

Pinto, U., & Maheshwari, B. L. (2011). River health assessment in peri-urban landscapes: An application of multivariate analysis to identify the key variables. *Water Research*, *45*(13), 3915–3924. https://doi.org/10.1016/j.watres.2011.04.044

Poole, G. C., & Berman, C. H. (2001). An Ecological Perspective on In-Stream Temperature: Natural Heat Dynamics and Mechanisms of Human-CausedThermal Degradation. *Environmental Management*, *27*(6), 787–802. https://doi.org/10.1007/s002670010188

Poulton, S. W., & Raiswell, R. (2005). Chemical and physical characteristics of iron oxides in riverine and glacial meltwater sediments. *Chemical Geology*, *218*(3), 203–221. https://doi.org/10.1016/j.chemgeo.2005.01.007

Prasad, B. S. R. V., Srinivasu, P. D. N., Varma, P. S., Raman, A. V., & Ray, S. (2014). Dynamics of Dissolved Oxygen in Relation to Saturation and Health of an Aquatic Body: A Case for Chilka Lagoon, India. *Journal of Ecosystems*, *2014*(1), 526245. https://doi.org/10.1155/2014/526245

Puntarulo, S. (2005). Iron, oxidative stress and human health. *Molecular Aspects of Medicine*, *26*(4), 299–312. https://doi.org/10.1016/j.mam.2005.07.001

Quantin, C., Joner, E. J., Portal, J. M., & Berthelin, J. (2005). PAH dissipation in a contaminated river sediment under oxic and anoxic conditions. *Environmental Pollution*, *134*(2), 315–322. https://doi.org/10.1016/j.envpol.2004.07.022

Quevedo, L., Ibáñez, C., Caiola, N., & Mateu, D. (2018). Effects of thermal pollution on benthic macroinvertebrate communities of large Mediterranean River. *JOURNAL OF ENTOMOLOGY AND ZOOLOGY STUDIES*, *6*.

Ramessur, R. T., Parry, S. J., & Ramjeawon, T. (2001). The relationship of dissolved Pb to some dissolved trace metals (Al, Cr, Mn, and Zn) and to dissolved nitrate and phosphate in a freshwater aquatic system in Mauritius. *Environment International*, *26*(4), 223–230. https://doi.org/10.1016/S0160-4120(00)00109-4

Reeder, B. C. (2011). Assessing constructed wetland functional success using diel changes in dissolved oxygen, pH, and temperature in submerged, emergent, and open-water habitats in the Beaver Creek Wetlands Complex, Kentucky (USA). *Ecological Engineering*, *37*(11), 1772–1778. https://doi.org/10.1016/j.ecoleng.2011.06.018

Reimann, C., & de Caritat, P. (2005). Distinguishing between natural and anthropogenic sources for elements in the environment: Regional geochemical surveys versus enrichment factors. *Science of The Total Environment*, *337*(1), 91–107. https://doi.org/10.1016/j.scitotenv.2004.06.011

Sand-Jensen, K., Riis, T., & Martinsen, K. T. (2022). Photosynthesis, growth, and distribution of plants in lowland streams—A synthesis and new data analyses of 40 years research. *Freshwater Biology*, *67*(7), 1255–1271. https://doi.org/10.1111/fwb.13915

Savard, J.-P. L., Clergeau, P., & Mennechez, G. (2000). Biodiversity concepts and urban ecosystems. *Landscape and Urban Planning*, *48*(3), 131–142. https://doi.org/10.1016/S0169-2046(00)00037-2

Schilling, K. E., Zhang, Y.-K., Hill, D. R., Jones, C. S., & Wolter, C. F. (2009). Temporal variations of *Escherichia coli* concentrations in a large Midwestern river. *Journal of Hydrology*, *365*(1), 79–85. https://doi.org/10.1016/j.jhydrol.2008.11.029

Shahid, M., Pinelli, E., & Dumat, C. (2012). Review of Pb availability and toxicity to plants in relation with metal speciation; role of synthetic and natural organic ligands. *Journal of Hazardous Materials*, *219–220*, 1–12. https://doi.org/10.1016/j.jhazmat.2012.01.060

Sharma, A., Grewal, A. S., Sharma, D., & Srivastav, A. L. (2023). Chapter 3 - Heavy metal contamination in water: Consequences on human health and environment. In S. K. Shukla, S. Kumar, S. Madhav, & P. K. Mishra (Eds.), *Metals in Water* (pp. 39–52). Elsevier. https://doi.org/10.1016/B978-0-323-95919-3.00015-X

Sharma, M. K., Jain, C. K., Singhal, D. C., & Choubey, V. K. (2009). Kinetics of sorption of lead on bed sediments of River Hindon, India. *Environmental Monitoring and Assessment*, *157*(1), 11–21. https://doi.org/10.1007/s10661-008-0510-4

Shelton, D. R., Pachepsky, Y. A., Kiefer, L. A., Blaustein, R. A., McCarty, G. W., & Dao, T. H. (2014). Response of coliform populations in streambed sediment and water column to changes in nutrient concentrations in water. *Water Research*, *59*, 316–324. https://doi.org/10.1016/j.watres.2014.04.019

Shousha, S., Maranger, R., & Lapierre, J.-F. (2021). Different forms of carbon, nitrogen, and phosphorus influence ecosystem stoichiometry in a north temperate river across seasons and land uses. *Limnology and Oceanography*, *66*(12), 4285–4298. https://doi.org/10.1002/lno.11960

Silberbauer, M. (1996). *WATER QUALITY GUIDELINES - VOL 7*. *7*. https://www.dws.gov.za/iwqs/wq_guide/Pol_saWQguideFRESHAquaticecosystemsvol7.pdf

Smith, V. H., Tilman, G. D., & Nekola, J. C. (1999). Eutrophication: Impacts of excess nutrient inputs on freshwater, marine, and terrestrial ecosystems. *Environmental Pollution*, *100*(1), 179–196. https://doi.org/10.1016/S0269-7491(99)00091-3

Some, S., Mondal, R., Mitra, D., Jain, D., Verma, D., & Das, S. (2021). Microbial pollution of water with special reference to coliform bacteria and their nexus with environment. *Energy Nexus*, *1*, 100008. https://doi.org/10.1016/j.nexus.2021.100008

Sterner, R. W. (2008). On the Phosphorus Limitation Paradigm for Lakes. *International Review of Hydrobiology*, *93*(4–5), 433–445. https://doi.org/10.1002/iroh.200811068

Thomilson, D. C., Wilson, D. J., Harris, C. R., & Jeffrey, D. W. (1980). *Problem in heavy metals in estuaries and the formation of pollution index*. *33*(1–4), 566–575.

Thompson, M. Y., Brandes, D., & Kney, A. D. (2012). Using electronic conductivity and hardness data for rapid assessment of stream water quality. *Journal of Environmental Management*, *104*, 152–157. https://doi.org/10.1016/j.jenvman.2012.03.025

Toro, M., Weller, D., Ramos, R., Diaz, L., Alvarez, F. P., Reyes-Jara, A., Moreno-Switt, A. I., Meng, J., & Adell, A. D. (2022). Environmental and anthropogenic factors associated with the likelihood of detecting *Salmonella* in agricultural watersheds. *Environmental Pollution*, *306*, 119298. https://doi.org/10.1016/j.envpol.2022.119298

Tromans, D. (1998). Temperature and pressure dependent solubility of oxygen in water: A thermodynamic analysis. *Hydrometallurgy*, *48*(3), 327–342. https://doi.org/10.1016/S0304-386X(98)00007-3

Tutu, H., McCarthy, T. S., & Cukrowska, E. (2008). The chemical characteristics of acid mine drainage with particular reference to sources, distribution and remediation: The Witwatersrand Basin, South Africa as a case study. *Applied Geochemistry*, *23*(12), 3666–3684. https://doi.org/10.1016/j.apgeochem.2008.09.002

Vehanen, T., Sutela, T., Aroviita, J., Karjalainen, S.-M., Riihimäki, J., Larsson, A., & Vuori, K.-M. (2022). Land use in acid sulphate soils degrades river water quality – Do the biological quality metrics respond? *Ecological Indicators*, *141*, 109085. https://doi.org/10.1016/j.ecolind.2022.109085

Verbyla, M. E., Calderon, J. S., Flanigan, S., Garcia, M., Gersberg, R., Kinoshita, A. M., Mladenov, N., Pinongcos, F., & Welsh, M. (2021). An Assessment of Ambient Water Quality and Challenges with Access to Water and Sanitation Services for Individuals Experiencing Homelessness in Riverine Encampments. *Environmental Engineering Science*, *38*(5), 389–401. https://doi.org/10.1089/ees.2020.0319

Villiers, S. de, & Mkwelo, S. T. (2009). Has monitoring failed the Olifants River, Mpumalanga? *Water SA*, *35*(5), Article 5. https://doi.org/10.4314/wsa.v35i5.49193

Vollmer, D., & Grêt-Regamey, A. (2013). Rivers as municipal infrastructure: Demand for environmental services in informal settlements along an Indonesian river. *Global Environmental Change*, *23*(6), 1542–1555. https://doi.org/10.1016/j.gloenvcha.2013.10.001

Wallace, J., Champagne, P., & Hall, G. (2016). Multivariate statistical analysis of water chemistry conditions in three wastewater stabilization ponds with algae blooms and pH fluctuations. *Water Research*, *96*, 155–165. https://doi.org/10.1016/j.watres.2016.03.046

Walton, N. R. G. (1989). Electrical Conductivity and Total Dissolved Solids—What is Their Precise Relationship? *Desalination*, *72*(3), 275–292. https://doi.org/10.1016/0011-9164(89)80012-8

Wan, X., Xiang, W., Wan, N., Yan, S., Bao, Z., & Wang, Y. (2018). Complexation and reduction of iron by phenolic substances: Implications for transport of dissolved Fe from peatlands to aquatic ecosystems and global iron cycling. *Chemical Geology*, *498*, 128–138. https://doi.org/10.1016/j.chemgeo.2018.09.019

Wang, C., Schneider, R. L., Parlange, J.-Y., Dahlke, H. E., & Walter, M. T. (2018). Explaining and modeling the concentration and loading of *Escherichia coli* in a stream—A case study. *Science of The Total Environment*, *635*, 1426–1435. https://doi.org/10.1016/j.scitotenv.2018.04.036

Wang, F., Dong, W., Zhao, Z., Wang, H., Li, W., Chen, G., Wang, F., Zhao, Y., Huang, J., & Zhou, T. (2021). Heavy metal pollution in urban river sediment of different urban functional areas and its influence on microbial community structure. *Science of The Total Environment*, *778*, 146383. https://doi.org/10.1016/j.scitotenv.2021.146383

Wang, Q., Liao, Z., Yao, D., Yang, Z., Wu, Y., & Tang, C. (2021). Phosphorus immobilization in water and sediment using iron-based materials: A review. *Science of The Total Environment*, *767*, 144246. https://doi.org/10.1016/j.scitotenv.2020.144246

Wang, W.-X., & Rainbow, P. S. (2008). Comparative approaches to understand metal bioaccumulation in aquatic animals. *Comparative Biochemistry and Physiology Part C: Toxicology & Pharmacology*, *148*(4), 315–323. https://doi.org/10.1016/j.cbpc.2008.04.003

Wang, X.-C., Feng, H., & Ma, H.-Q. (2007). Assessment of Metal Contamination in Surface Sediments of Jiaozhou Bay, Qingdao, China. *CLEAN – Soil, Air, Water*, *35*(1), 62–70. https://doi.org/10.1002/clen.200600022

Wanqing, L. (2001). The characterization of hydrogen ion concentration in sequential cumulative rainwater. *Atmospheric Environment*, *35*(35), 6219–6225. https://doi.org/10.1016/S1352-2310(01)00235-7

Ward, P. L. (2009). Sulfur dioxide initiates global climate change in four ways. *Thin Solid Films*, *517*(11), 3188–3203. https://doi.org/10.1016/j.tsf.2009.01.005

Whitehead, P. G., Wilby, R. L., Butterfield, D., & Wade, A. J. (2006). Impacts of climate change on in-stream nitrogen in a lowland chalk stream: An appraisal of adaptation strategies. *Science of The Total Environment*, *365*(1), 260–273. https://doi.org/10.1016/j.scitotenv.2006.02.040

Wijesiri, B., Liu, A., He, B., Yang, B., Zhao, X., Ayoko, G., & Goonetilleke, A. (2019). Behaviour of metals in an urban river and the pollution of estuarine environment. *Water Research*, *164*, 114911. https://doi.org/10.1016/j.watres.2019.114911

Withers, P., & Jarvie, H. (2008). *(PDF) Delivery and Cycling of Phosphorus in Rivers: A Review*. https://www.researchgate.net/publication/23269506_Delivery_and_Cycling_of_Phosphorus_in_Rivers_A_Review?utm_source=chatgpt.com

Wu, J., Gu, L., Hua, Z., Li, X., Lu, Y., & Chu, K. (2021). Effects of *Escherichia coli* pollution on decomposition of aquatic plants: Variation due to microbial community composition and the release and cycling of nutrients. *Journal of Hazardous Materials*, *401*, 123252. https://doi.org/10.1016/j.jhazmat.2020.123252

Xia, C., Yu, D., Wang, Z., & Xie, D. (2014). Stoichiometry patterns of leaf carbon, nitrogen and phosphorous in aquatic macrophytes in eastern China. *Ecological Engineering*, *70*, 406–413. https://doi.org/10.1016/j.ecoleng.2014.06.018

Xiao, Y.-H., Räike, A., Hartikainen, H., & Vähätalo, A. V. (2015). Iron as a source of color in river waters. *Science of The Total Environment*, *536*, 914–923. https://doi.org/10.1016/j.scitotenv.2015.06.092

Xin, Z., & Kinouchi, T. (2013). Analysis of stream temperature and heat budget in an urban river under strong anthropogenic influences. *Journal of Hydrology*, *489*, 16–25. https://doi.org/10.1016/j.jhydrol.2013.02.048

Xu, G., Li, P., Lu, K., Tantai, Z., Zhang, J., Ren, Z., Wang, X., Yu, K., Shi, P., & Cheng, Y. (2019). Seasonal changes in water quality and its main influencing factors in the Dan River basin. *CATENA*, *173*, 131–140. https://doi.org/10.1016/j.catena.2018.10.014

Zak, D., Hupfer, M., Cabezas, A., Jurasinski, G., Audet, J., Kleeberg, A., McInnes, R., Kristiansen, S. M., Petersen, R. J., Liu, H., & Goldhammer, T. (2021). Sulphate in freshwater ecosystems: A review of sources, biogeochemical cycles, ecotoxicological effects and bioremediation. *Earth-Science Reviews*, *212*, 103446. https://doi.org/10.1016/j.earscirev.2020.103446

Zeng, Y., Bi, C., Jia, J., Deng, L., & Chen, Z. (2020). Impact of intensive land use on heavy metal concentrations and ecological risks in an urbanized river network of Shanghai. *Ecological Indicators*, *116*, 106501. https://doi.org/10.1016/j.ecolind.2020.106501

Zhang, W., Swaney, D. P., Hong, B., & Howarth, R. W. (2017). Anthropogenic Phosphorus Inputs to a River Basin and Their Impacts on Phosphorus Fluxes Along Its Upstream-Downstream Continuum. *Journal of Geophysical Research: Biogeosciences*, *122*(12), 3273–3287. https://doi.org/10.1002/2017JG004004

Zhao, H., Li, X., Wang, X., & Tian, D. (2010). Grain size distribution of road-deposited sediment and its contribution to heavy metal pollution in urban runoff in Beijing, China. *Journal of Hazardous Materials*, *183*(1), 203–210. https://doi.org/10.1016/j.jhazmat.2010.07.012

Zheng, S., Wang, P., Wang, C., Hou, J., & Qian, J. (2013). Distribution of metals in water and suspended particulate matter during the resuspension processes in Taihu Lake sediment, China. *Quaternary International*, *286*, 94–102. https://doi.org/10.1016/j.quaint.2012.09.003

Appendix

Tables used to create graphs

Tables 5-14 show the results of the various parameters measured over the course of three sampling trips at each of the sites. The *E. coli* and total coliforms were not measured everyday as it is a timely and expensive process which is not required everyday in order to achieve an understanding of the bacterial loads entering the river. On the days where *E. coli* and total coliforms were not measured, Did Not Measure (DNM) is written in the table in place of the measurement.

Table 5:The field measurements, nutrients, E. coli and coliform results collected from Montgomery Spruit throughout the sampling period

Date	pH	Temperature (˚C)	EC (µs/cm)	DO (mg/L)	Sulphate (mg/L)	Nitrate (mg/L)	Nitrite (mg/L)	Phosphate (mg/L)	*E. coli* (MPN per 100mL)	coliforms (MPN per 100mL)
11/05/2023	8,10	12,7	355,5	11,09	0	1,7	0,12	0,5	DNM	DNM
19/05/2023	8,07	12,8	347,8	11,10	0	0,9	0,08	0,5	DNM	DNM
25/05/2023	8,02	17,3	390,0	8,75	0	0,5	0,02	8,6	DNM	DNM
02/06/2023	8,16	10,2	292,0	12,25	0	1,1	0,02	0,5	200	18120
07/06/2023	8,21	11,0	320,4	11,76	0	1,7	0	1,1	4366	866
15/06/2023	8,07	8,3	312,7	12,59	0	4,4	0,03	0,5	DNM	DNM
23/06/2023	8,15	8,3	326,0	13,32	0	2,1	0,05	0,7	DNM	DNM
27/06/2023	8,25	7,9	322,6	13,87	0	3,0	0,02	1,9	333	2633
14/08/2023	8,64	10,8	345,2	14,24	0	0	0	1,2	1323	4538
24/08/2023	8,39	10,9	345,8	13,81	0	0	0,15	3,2	DNM	DNM
31/08/2023	8,40	11,9	368,4	13,23	0	0	0	0	7164	13376
06/09/2023	8,41	12,9	391,0	12,64	0	0	0	27,5	DNM	DNM

13/09/2023	8,41	11,2	359,0	12,90	0	3,4	0	30,2	117	1835
28/09/2023	7,94	13,2	347,5	13,20	0	19,0	1,43	3,6	10658	203870
09/11/2023	7,89	16,0	394,8	10,37	0	10,5	0,2	1,2	2623	51200
05/02/2024	7,84	18,8	442,0	7,54	0	0	0	0	3282	36823
14/02/2024	7,81	18,5	440,3	7,78	0	0	0	0,8	DNM	DNM
21/02/2024	7,73	20,6	383,6	6,64	0	0	0,21	0	DNM	DNM
20/03/2024	7,81	18,3	408,2	8,77	30	0	0	2,2	1141	30623
28/03/2024	7,80	16,9	413,3	8,71	17	15,5	0,04	0	DNM	DNM
09/04/2024	7,46	16,1	300,8	8,09	0	8,3	0	0	85529	284658

Table 6: The metals results from water and sediment samples from the Montgomery Spruit

	Water samples			Sediment samples		
Date	Manganese (mg/L)	Iron (mg/L)	Lead (mg/L)	Manganese (mg/kg)	Iron (mg/kg)	Lead (mg/kg)
11/05/2023	0,004	0,004	0,004	119,85	14585	151,5
25/05/2023	0,001	0,004	0,004	927,5	12245	293,5
07/06/2023	0,001	0,004	0,004	689	14200	306,5
14/08/2023	0,001	0,004	0,004	140,45	75,25	399,5
06/09/2023	0,001	0,004	0,004	208,4	9495	413
09/11/2023	0,001	0,004	0,004	136,65	5120	342
14/02/2024	0,001	0,004	0,004	168	3795	63
09/04/2024	0,001	0,004	0,004	421,25	16695	203

Table 7:The field measurements, nutrient, E. coli and coliform results collected from Westdene Spruit throughout the sampling period

Date	pH	Temperature (˚C)	EC (µs/cm)	DO (mg/L)	Sulphate (mg/L)	Nitrate (mg/L)	Nitrite (mg/L)	Phosphate (mg/L)	*E. coli* (MPN per 100mL)	coliforms (MPN per 100mL)
11/05/2023	7,83	14,0	275,2	9,38	0	0	0	0	DNM	DNM
19/05/2023	7,93	14,0	307,0	8,05	0	0	0	0	DNM	DNM
25/05/2023	7,93	17,5	341,0	8,87	0	0	0	0	DNM	DNM
02/06/2023	8,02	11,0	282,8	10,74	0	0	0	0	1040	25920
07/06/2023	7,87	12,1	362,9	7,82	0	0	0	0	181	1563
15/06/2023	8,09	8,7	279,1	11,00	0	0	0	0	DNM	DNM
23/06/2023	7,78	9,3	299,4	9,87	0	0	0	0	DNM	DNM
27/06/2023	8,18	9,3	281,5	9,96	0	0	0	0	181	2100
14/08/2023	8,24	12,1	300,2	9,48	0	0	0	4,5	166	1250
24/08/2023	8,07	12,2	306,1	9,47	14,1	0	0,03	0	DNM	DNM

31/08/2023	8,07	13,0	321,6	9,27	0	0	0	0,8	125	925
06/09/2023	8,06	13,8	337,0	9,06	14,1	8,1	0,7	4,7	DNM	DNM
13/09/2023	8,14	11,6	338,5	8,91	0	0	0,09	1,9	100	1730
28/09/2023	7,87	13,5	328,4	7,53	14,1	14,6	0,17	0,8	2260	68670
09/11/2023	7,74	16,9	314,2	6,57	0	0	0,08	0	740	41060
05/02/2024	7,61	20,3	300,0	5,60	0	0	0,53	1,6	DNM	DNM
14/02/2024	7,17	20,4	259,5	4,72	0	0	0	0	DNM	DNM
21/02/2024	7,58	20,4	297,4	6,34	0	0	0,06	0	DNM	DNM
20/03/2024	7,62	19,9	279,5	4,80	28	0	0,27	0	741	21882
28/03/2024	7,46	18,8	313,3	3,97	32	0	0,2	0,8	DNM	DNM
09/04/2024	7,32	16,6	171,4	7,45	0	0	0,1	0	20270	284658

Table 8:The metals results from water and sediment samples from the Westdene Spruit

	Water samples			Sediment samples		
Date	Manganese (mg/L)	Iron (mg/L)	Lead (mg/L)	Manganese (mg/kg)	Iron (mg/kg)	Lead (mg/kg)
11/05/2023	0,001	0,004	0,004	81,7	3383	144,5
25/05/2023	0,001	0,004	0,004	626,5	15585	258,5
07/06/2023	0,001	0,004	0,004	155,4	9620	314
14/08/2023	0,001	0,004	0,004	680	9900	390,5
06/09/2023	0,001	0,004	0,004	256,3	6010	380,5
09/11/2023	0,001	0,004	0,004	553	17260	379,5
14/02/2024	0,001	0,004	0,004	62,8	2480	19

09/04/2024	0,001	0,004	0,004	70,35	4897,5	76,5

Table 9: The field measurements, nutrient, E. coli and coliform results collected from The Braamfontein Spruit at The Parkview Golf Course throughout the sampling period

Date	pH	Temperature (˚C)	EC (µs/cm)	DO (mg/L)	Sulphate (mg/L)	Nitrate (mg/L)	Nitrite (mg/L)	Phosphate (mg/L)	*E. coli* (MPN per 100mL)	coliforms (MPN per 100mL)
11/05/2023	7,90	15,8	338,9	8,86	37,0	2,8	0,11	0,5	DNM	DNM
19/05/2023	8,01	18,7	176,6	8,61	18,8	2,7	0,1	0,5	DNM	DNM
25/05/2023	7,96	14,6	323,2	9,74	52,2	1,5	0,06	6,8	DNM	DNM
02/06/2023	7,93	15,6	333,4	8,77	26,7	1,0	0,12	0,5	54000	397260
07/06/2023	7,89	13,8	317,5	9,04	19,1	5,2	0,19	2,6	47622	261022
15/06/2023	7,97	13,6	317,6	8,74	12,1	3,3	0,16	1,0	DNM	DNM
23/06/2023	7,97	13,8	324,2	8,68	18,6	1,6	0,14	2,4	DNM	DNM
27/06/2023	8,02	13,6	320,3	8,55	24,5	3,1	0,23	0,5	267400	537688
14/08/2023	7,97	15,1	343,1	8,06	0	7,1	0,55	2,9	136875	604900
24/08/2023	7,93	15,0	313,1	8,14	14,1	0	2,0	2,6	DNM	DNM
31/08/2023	7,90	15,2	335,3	8,11	0	0	0,25	0,9	196200	691314
06/09/2023	7,86	15,4	357,5	8,07	14,1	0	0,8	0	DNM	DNM
13/09/2023	7,99	14,1	344,9	8,15	0	0	1,01	2,1	242333	806533
28/09/2023	7,86	15,9	344,7	7,45	14,1	21,4	1,36	3,0	216266	806533
09/11/2023	7,8	18,5	366,9	6,86	0	13,7	0,66	2,5	272133	806533
05/02/2024	7,74	21,1	389,0	6,26	0	0	1,11	0,8	108500	577633

14/02/2024	7,71	21,2	380,5	6,74	0	5,5	0,55	0	DNM	DNM
21/02/2024	7,70	20,7	370,0	7,10	0	8,5	0,43	0	DNM	DNM
20/03/2024	7,61	20,2	392,3	6,70	28,0	0	2,0	2,2	348733	806533
28/03/2024	7,71	19,3	359,3	7,73	32,0	0	0,78	1,2	DNM	DNM
09/04/2024	7,54	17,1	273,5	7,73	0	10,2	0,96	0	204336	806533

Table 10: The metals results from water and sediment samples from the Braamfontein Spruit at the Parkview Golf Course

	Water samples			Sediment samples		
Date	Manganese (mg/L)	Iron (mg/L)	Lead (mg/L)	Manganese (mg/kg)	Iron (mg/kg)	Lead (mg/kg)
11/05/2023	0,056	-0,004	-0,004	71,15	3134	160,5
25/05/2023	-0,001	-0,004	-0,004	116,6	4916,5	198
07/06/2023	0,064	-0,004	-0,004	77,7	3350,5	277,5
14/08/2023	-0,001	-0,004	-0,004	51,45	4499	325,5
06/09/2023	-0,001	-0,004	-0,004	74,25	1615,5	380,5
09/11/2023	-0,001	-0,004	-0,004	16,35	2651,2	195
14/02/2024	-0,001	-0,004	-0,004	219,25	2942	60,5
09/04/2024	-0,001	-0,004	-0,004	10,7	3901	87

Table 11: The field measurements, nutrient, E. coli and coliform results collected from The Braamfontein Spruit at Delta Park throughout the sampling period

Date	pH	Temperature (˚C)	EC (µs/cm)	DO (mg/L)	Sulphate (mg/L)	Nitrate (mg/L)	Nitrite (mg/L)	Phosphate (mg/L)	*E. coli* (MPN per 100mL)	coliforms (MPN per 100mL)
11/05/2023	8,10	15,4	357,2	10,36	51,7	2,2	0,09	2,3	DNM	DNM
19/05/2023	7,89	14,4	297,0	9,60	21,8	1,3	0,08	0,5	DNM	DNM
25/05/2023	7,87	17,7	353,8	9,45	21,4	1,9	0,13	0,6	DNM	DNM
02/06/2023	8,09	11,6	318,7	10,92	43	1,1	0,06	1,6	10420	52060
07/06/2023	8,16	12,2	345,1	11,39	31,6	2,7	0,12	0,5	8280	19180
15/06/2023	8,13	9,9	327,0	11,33	19,1	3,5	0,076	0,5	DNM	DNM

23/06/2023	8,15	9,9	321,5	11,98	27	2,5	0,12	0,5	DNM	DNM
27/06/2023	8,35	10,3	330,5	12,72	22,8	3,9	0,13	1,9	58180	173280
14/08/2023	8,85	12,5	327,0	14,32	0	3,8	0,42	2,2	1040	6900
24/08/2023	8,55	12,2	329,2	12,98	0	0	0,4	3,1	DNM	DNM
31/08/2023	8,56	13,2	354,9	12,89	28	0	0,88	1	866	7733
06/09/2023	8,56	14,2	380,5	12,80	0	7	0,69	0	DNM	DNM
13/09/2023	8,61	12,3	365,5	13,60	0	0	0,48	2,6	2725	26775
28/09/2023	7,72	14,9	318,6	8,31	0	3,8	0,74	2,5	194137	302450
09/11/2023	7,73	18,1	364,7	7,54	0	0	0,29	0	22312	302450
05/02/2024	7,73	21,3	410,8	6,77	0	10,9	0,62	0,8	9700	162462
14/02/2024	7,53	21,2	253,4	6,38	0	10,4	0,09	0	DNM	DNM
21/02/2024	7,73	20,8	383,6	6,64	0	0	0,09	0	DNM	DNM
20/03/2024	7,57	19,5	362,6	6,52	29	0	0,78	2,8	11662	139987
28/03/2024	7,61	19,0	393,5	7,25	36	0	0,87	0	DNM	DNM
09/04/2024	7,39	17,2	190,7	7,91	0	8,2	0,46	0	13987	302450

Table 12: The metals results from water and sediment samples from the Braamfontein Spruit at Delta Park

	Water samples			Sediment samples		
Date	Manganese (mg/L)	Iron (mg/L)	Lead (mg/L)	Manganese (mg/kg)	Iron (mg/kg)	Lead (mg/kg)
11/05/2023	0,001	0,004	0,004	16,85	2931	182,5
25/05/2023	0,001	0,004	0,004	29,65	2210,75	296,5
07/06/2023	0,001	0,004	0,004	61,3	4425	336,5

14/08/2023	0,001	0,004	0,004	85,65	3800	346,5
06/09/2023	0,001	0,004	0,004	47,3	3859,5	410,5
09/11/2023	0,001	0,004	0,004	9,05	4811,5	188
14/02/2024	0,001	0,004	0,004	122	4875	22,5
09/04/2024	0,001	0,004	0,004	39,2	4547,5	95,5

Table 13: The field measurements*, nutrient, E. coli and coliform results collected from The Braamfontein Spruit at Sandton Riverclub Golf Course throughout the sampling period*

Date	pH	Temperature (˚C)	EC (µs/cm)	DO (mg/L)	Sulphate (mg/L)	Nitrate (mg/L)	Nitrite (mg/L)	Phosphate (mg/L)	*E. coli* (MPN per 100mL)	coliforms (MPN per 100mL)
11/05/2023	8,15	14,1	222,0	10,45	0	0	0	0	DNM	DNM
19/05/2023	8,2	16,8	373,8	9,18	9,9	0,9	0,05	0,5	DNM	DNM
25/05/2023	8,34	11,7	309,8	12,11	26,9	1,1	0,07	1,2	DNM	DNM
02/06/2023	8,45	11,8	331,8	13,00	23,4	0,5	0,07	1,4	34520	137340
07/06/2023	8,56	9,3	326,5	14,46	21,1	1,4	0,06	0,6	6488	28311
15/06/2023	8,53	9,2	329,0	14,36	41,6	4,2	0,09	0,5	DNM	DNM
23/06/2023	8,46	9,4	322,9	14,60	52,9	2,5	0,09	3,9	DNM	DNM

27/06/2023	8,44	9,3	315,0	14,80	33,5	2,2	0,07	1,8	11022	27446
14/08/2023	9,26	13,1	324,9	14,85	0	0	0	2,5	444	2177
24/08/2023	8,83	11,9	314,8	13,42	0	0	0,12	3	DNM	DNM
31/08/2023	8,70	12,9	339,0	12,49	0	0	0,15	2,6	1433	36450
06/09/2023	8,56	13,9	363,1	11,56	0	0	0,23	7,3	DNM	DNM
13/09/2023	8,87	12,7	325,6	11,91	0	0	0,31	3,2	387	4037
28/09/2023	7,76	14,9	303,3	8,68	0	0	0,78	1,2	64650	302450
09/11/2023	7,92	18,3	358,9	7,89	0	0	0,15	2,5	5912	61050
05/02/2024	8,07	21,7	414,4	7,10	0	0	0,24	0	302450	302450
14/02/2024	7,77	20,7	236,4	7,07	0	0	0,1	1	DNM	DNM
21/02/2024	8,03	21,1	388,0	7,19	0	0	0,04	0	DNM	DNM
20/03/2024	7,91	20,7	285,8	7,46	19	4,7	0,17	11,6	3912	43100
28/03/2024	7,97	19,6	393,7	7,65	27	0	0,49	4	DNM	DNM
09/04/2024	7,02	16,3	113,5	7,79	0	7	0,19	0	51325	302450

Table 14:The metals results from water and sediment samples from the Braamfontein Spruit at Sandton Riverclub Golf Course

Date	Manganese (mg/L)	Iron (mg/L)	Lead (mg/L)	Manganese (mg/kg)	Iron (mg/kg)	Lead (mg/kg)
11/05/2023	0,001	0,004	0,004	290,76	10243,17	487,57
25/05/2023	0,001	0,004	0,004	43,35	3692	261,5
07/06/2023	0,001	0,004	0,004	301	24340	1581
14/08/2023	0,001	0,004	0,004	678	23650	560
06/09/2023	0,001	0,004	0,004	95,65	2923,5	357,5

09/11/2023	0,001	0,004	0,004	15	1,7	250,5
14/02/2024	0,001	0,004	0,004	358,85	14720	154,5
09/04/2024	0,001	0,004	0,004	543,5	2375	248

Figures used to draw conclusions about correlations

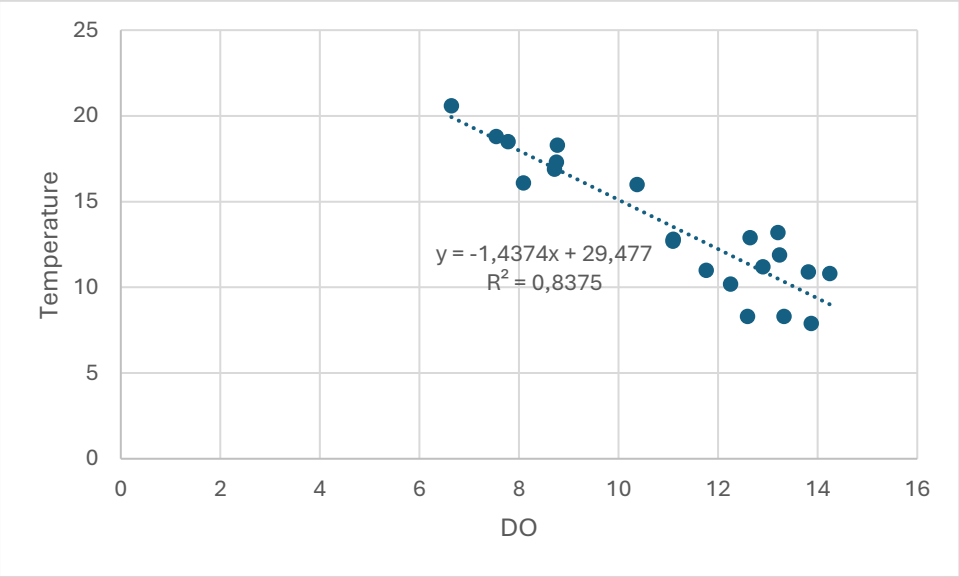


Figure 52: A graph showing the correlation between temperature and DO in the Montgomery Spruit

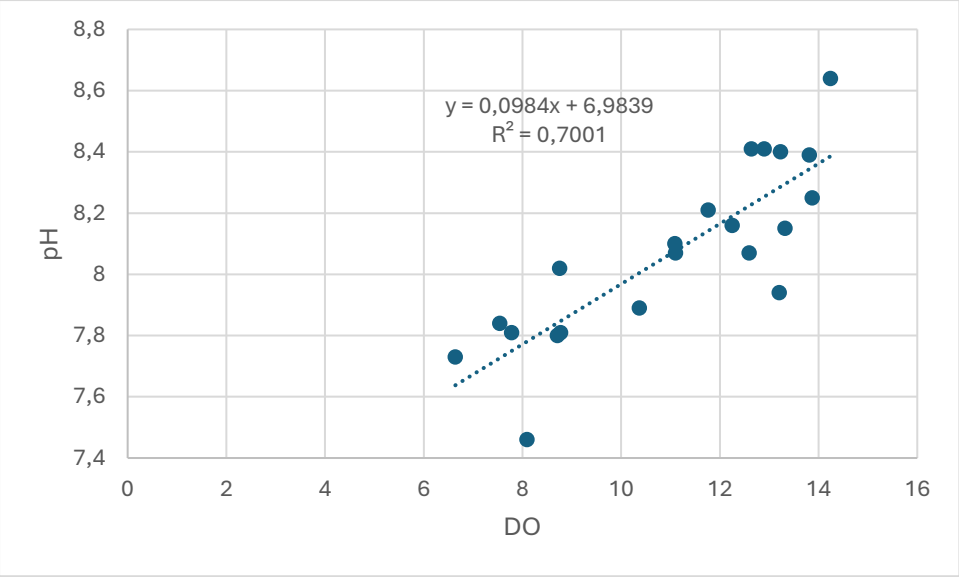


Figure 53: A graph showing the correlation between pH and DO in the Montgomery Spruit

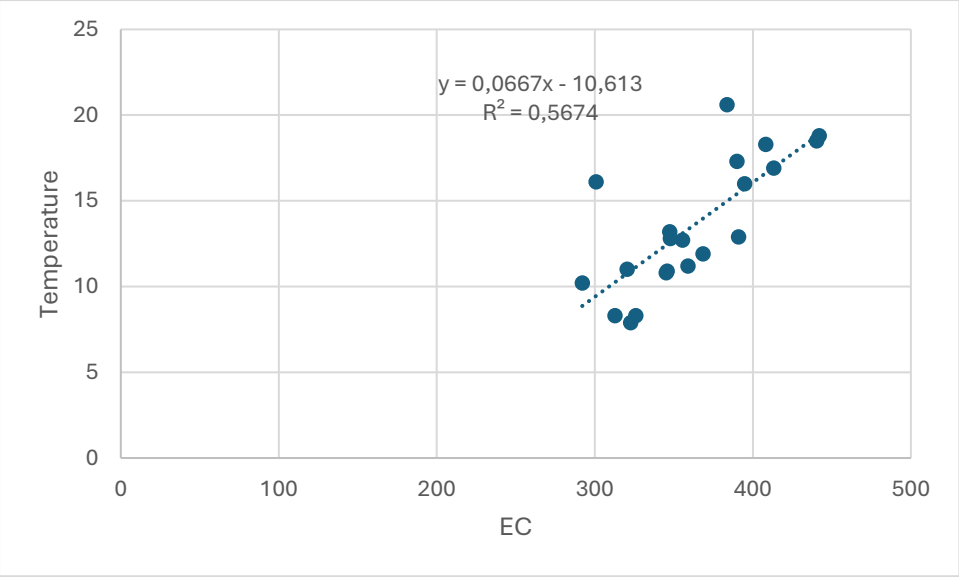


Figure 54: A graph showing the correlation between temperature and EC in the Montgomery Spruit

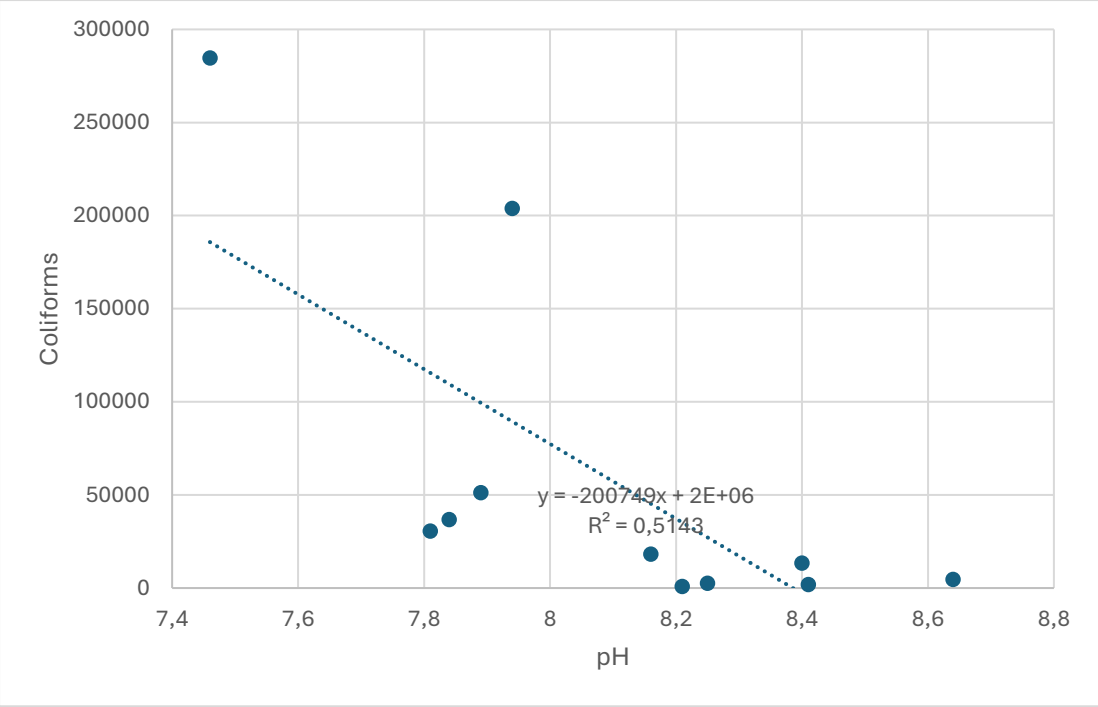


Figure 55: A graph showing the correlation between pH and coliforms in the Montgomery Spruit

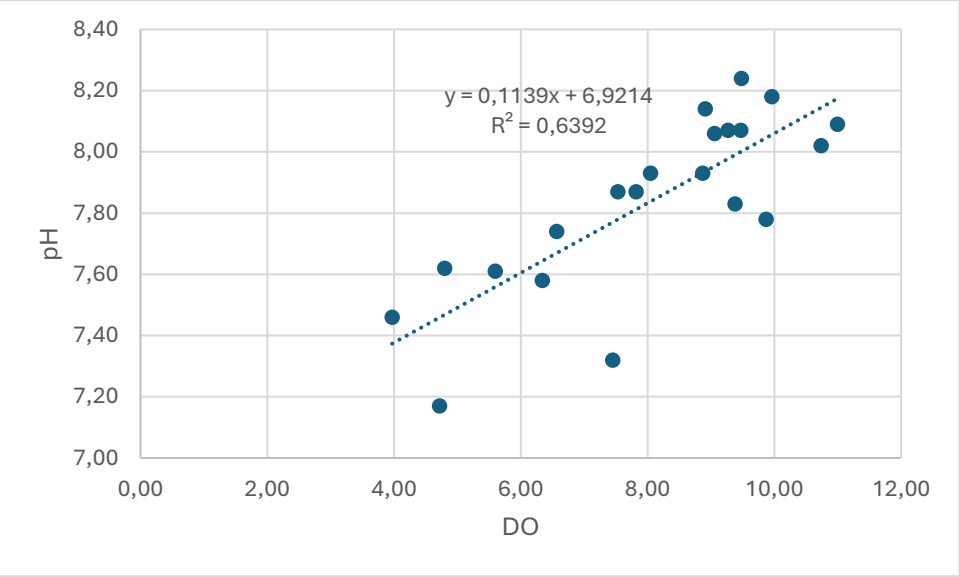


Figure 56: A graph showing the correlation between pH and DO in the Westdene Spruit.

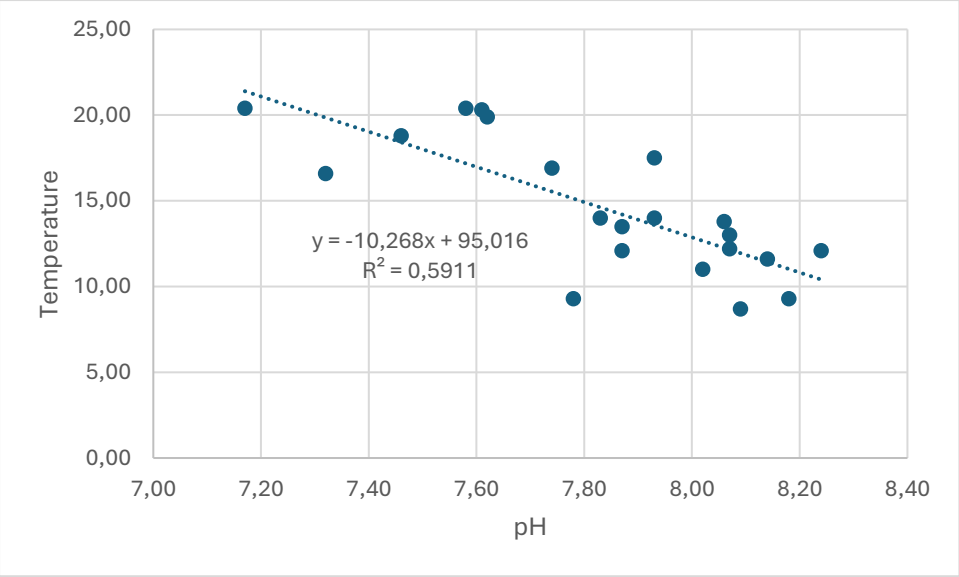


Figure 57: A graph showing the correlation between temperature and pH in the Westdene Spruit

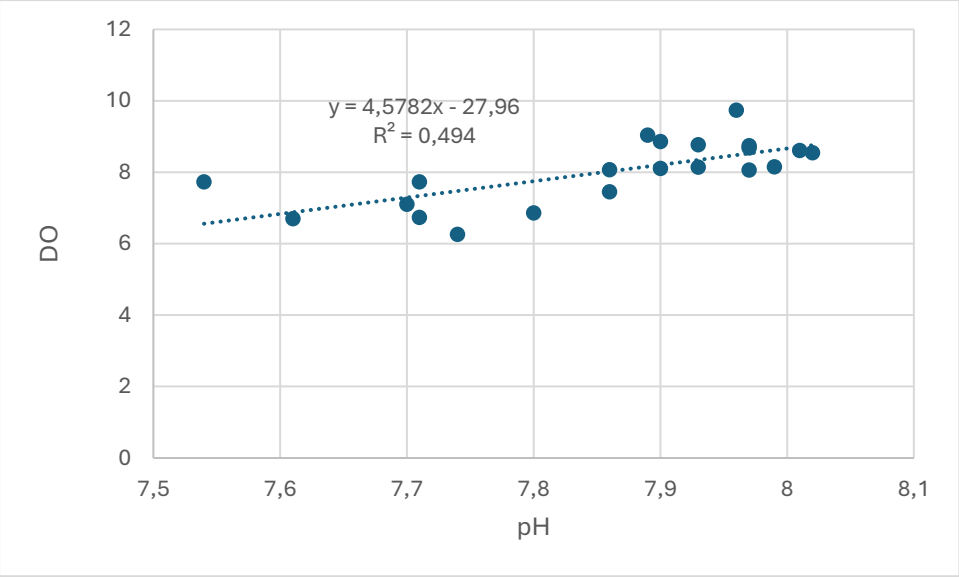


Figure 58: A graph showing the correlation between pH and DO in the Braamfontein Spruit at Parkview Golf Course

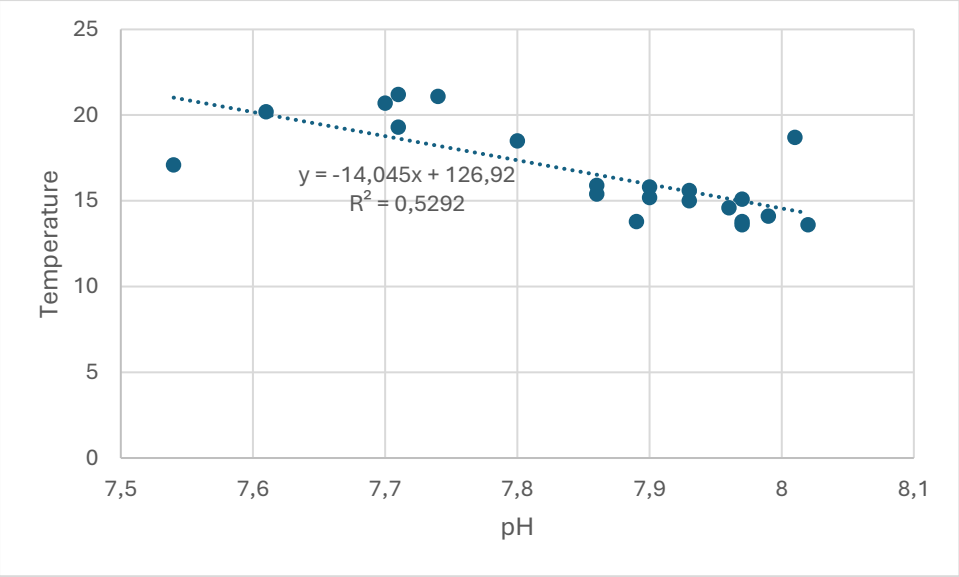


Figure 59: A graph showing the correlation between temperature and pH in the Braamfontein Spruit at the Parkview Golf Course

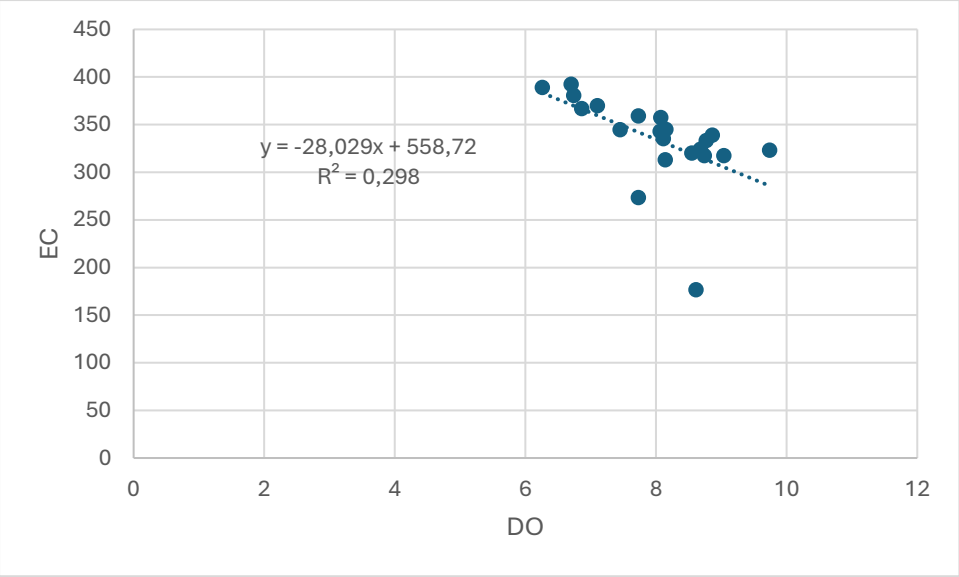


Figure 60: A graph showing the correlation between EC and DO in the Braamfontein Spruit at the Parkview Golf Course

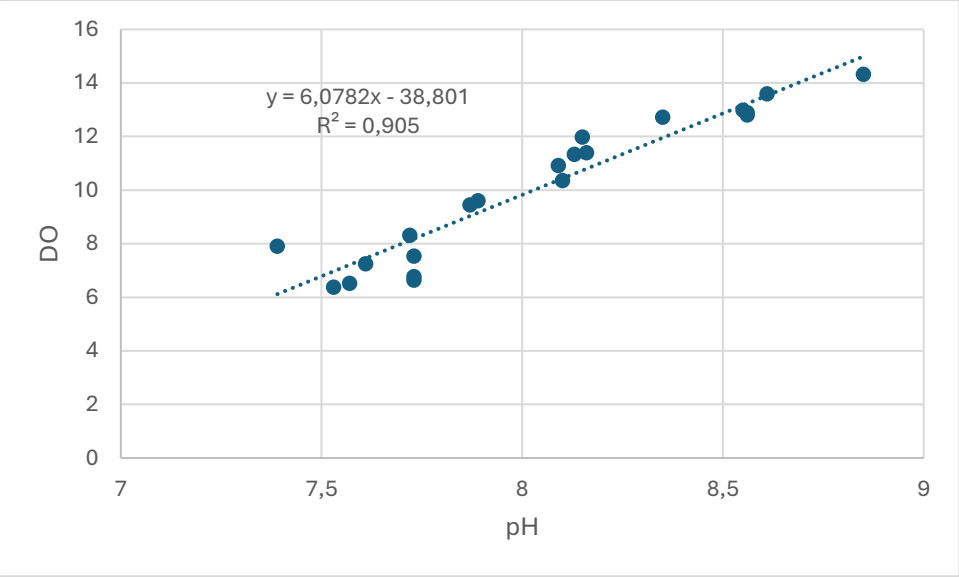


Figure 61: A graph showing the correlation between pH and DO in the Braamfontein Spruit at Delta Park

SASS Version 5 Score Sheet

Date: / /200__

Collector:

Grid Reference: WGS-84 Cape datum
S: ° ' " E: ° ' "

Site code:

River:

Site description:

Weather Condition:

Temp: °C pH:

DO: mg/l Cond: mS/m

Biotopes sampled:

SIC Time minutes
SOOC Time minutes
Average size of stones cm
Bedrock
Aquatic veg'n. Dom. sp.
MvegIC Dom. sp.
MvegOOC Dom. sp.
Gravel Sand
Mud
Hand picking/Visual observation

Flow: Low/Medium/High/Flood
Turbidity: Low/Medium/High
Riparian land use:

Disturbance in the river: eg. sandwinning, cattle drinking point, floods etc.

Observations: eg. smell and colour of water, petroleum, dead fish, etc.

Taxon		S	Veg	GSM	TOT
PORIFERA	5				
COELENTERATA	1				
TURBELLARIA	3				
ANNELIDA					
Oligochaeta	1				
Leeches	3				
CRUSTACEA					
Amphipoda	13				
Potamonautidae*	3				
Atyidae	8				
Palaemonidae	10				
HYDRACARINA	8				
PLECOPTERA					
Notonemouridae	14				
Perlidae	12				
EPHEMEROPTERA					
Baetidae 1sp	4				
Baetidae 2 sp	6				
Baetidae > 2 sp	12				
Caenidae	6				
Ephemeridae	15				
Heptageniidae	13				
Leptophlebiidae	9				
Oligoneuriidae	15				
Polymitarcyidae	10				
Prosopistomatidae	15				
Teloganodidae SWC	12				
Tricorythidae	9				
ODONATA					
Calopterygidae ST,T	10				
Chlorocyphidae	10				
Chlorolestidae	8				
Coenagrionidae	4				
Lestidae	8				
Platycnemidae	10				
Protoneuridae	8				
Aeshnidae	8				
Corduliidae	8				
Gomphidae	6				
Libellulidae	4				
LEPIDOPTERA					
Pyralidae	12				

Taxon		S	Veg	GSM	TOT
HEMIPTERA					
Belostomatidae*	3				
Corixidae*	3				
Gerridae*	5				
Hydrometridae*	6				
Naucoridae*	7				
Nepidae*	3				
Notonectidae*	3				
Pleidae*	4				
Veliidae/M...vellidae*	5				
MEGALOPTERA					
Corydalidae	8				
Sialidae	6				
TRICHOPTERA					
Dipseudopsidae	10				
Ecnomidae	8				
Hydropsychidae 1 sp	4				
Hydropsychidae 2 sp	6				
Hydropsychidae > 2 sp	12				
Philopotamidae	10				
Polycentropodidae	12				
Psychomyiidae/Xiphocen	8				
Cased caddis:					
Barbarochthonidae SWC	13				
Calamoceratidae ST	11				
Glossosomatidae SWC	11				
Hydroptilidae	6				
Hydrosalpingidae SWC	15				
Lepidostomatidae	10				
Leptoceridae	6				
Petrothrincidae SWC	11				
Pisulidae	10				
Sericostomatidae SWC	13				
COLEOPTERA					
Dytiscidae*	5				
Elmidae/Dryopidae*	8				
Gyrinidae*	5				
Haliplidae*	5				
Helodidae	12				
Hydraenidae*	8				
Hydrophilidae*	5				
Limnichidae	10				
Psephenidae	10				

Taxon		S	Veg	GSM	TOT
DIPTERA					
Athericidae	10				
Blepharoceridae	15				
Ceratopogonidae	5				
Chironomidae	2				
Culicidae*	1				
Dixidae*	10				
Empididae	6				
Ephydridae	3				
Muscidae	1				
Psychodidae	1				
Simulidae	5				
Syrphidae*	1				
Tabanidae	5				
Tipulidae	5				
GASTROPODA					
Ancylidae	6				
Bulininae*	3				
Hydrobiidae*	3				
Lymnaeidae*	3				
Physidae*	3				
Planorbinae*	3				
Thiaridae*	3				
Viviparidae* ST	5				
PELECYPODA					
Corbiculidae	5				
Sphaeriidae	3				
Unionidae	6				
SASS Score					
No. of Taxa					
ASPT					

Sample collection effort exceeds method? .....

Other biota including juveniles:

Comments:

Procedure: 'Kick SIC & bedrock for 2 mins, max. 5 mins; Kick SOOC & bedrock for 1 min; Sweep marginal vegetation (IC & OOC) for 2m total and aquatic veg 1m²; Stir & sweep gravel, sand, mud for 1 min total; * = airbreathers; Hand picking & visual observation for 1 min — record in biotope where found; Score for 15 mins/biotope but stop if no new taxa seen after 5 mins; "Estimate abundances: 1 = 1, A = 2–10, B = 10–100, C = 100–1 000, D = >1 000; S = Stone, rock & solid objects; Veg = All vegetation; GSM = Gravel, sand, mud; SWC = South Western Cape; T = Tropical; ST = Sub-tropical; Rate each biotope sampled: 1 = very poor (i.e. limited diversity), 5 = highly suitable (i.e. wide diversity)

Figure 62: The SASS5 scoring sheet (Dickens & Graham, 2002)