DUT
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INYUVESI YASETHEKWINI YEZOBUCHWEPHESHE

FACULTY OF ENGINEERING & THE BUILT ENVIRONMENT

DEPARTMENT OF CIVIL ENGINEERING MIDLANDS

# “Development of a multi-criteria decision-support tool for improving water quality to assist with engineering infrastructure and catchment management.”

Submitted in fulfilment of the academic requirements for the degree of Doctor of Engineering in the Department of Civil Engineering, Faculty of Engineering and the Built Environment, at the Durban University of Technology.

Submitted by: Zesizwe Ngubane

Student Number: 21856807

Supervisors: Prof. Sokolova, Prof. Stenström, and Bloodless. Dzwairo

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As the candidate's supervisor, I do agree to the submission of this thesis.

Main Supervisor: Professor Ekaterina Sokolova

Signature: Date: 2 March 2024

Co-Supervisor: Professor Thor-Axel Stenström

Signature: Date: 2 March 2024

Co-Supervisor: Bloodless Dzwairo

Signature: _ Date:2 March 2024________

# Dedication

I dedicate this thesis to:

My late grandparents: Mr. Mzimuni Simon Ngubane, Mrs. Nomasonto Anastasia (MaZulu) Ngubane, Mr. Zondumuzi Khahlela Dladla, and Mrs. Jabulile (MaKhanyile) Dladla. Though you did not witness my pursuit of a doctoral degree, the imprint you have left on my heart endures. Your love's warmth and the invaluable lessons you imparted continue to shape me, even in your absence.

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I am deeply indebted to my family for their constant encouragement and understanding throughout this academic journey. Their unconditional love and belief in my abilities have been a constant source of strength and motivation. I am grateful for their sacrifices and unwavering support, which have enabled me to pursue my goals with determination and perseverance.

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# Abstract

Research combining water quality modelling, quantitative chemical/microbial risk assessment, and stakeholder engagement to prioritise catchment areas facing water pollution problems to devise effective pollution mitigation strategies are limited. This research therefore aimed to address this gap by providing a practical and comprehensive framework that supports well-informed decision-making processes in water pollution alleviation. By integrating multiple criteria and catchment aspects, this framework can assist infrastructure, operational, and ecological managers within a catchment in prioritising best management practices (BMPs) to reduce pollution and mitigate against potential resultant impacts. Given this context, uMsunduzi catchment, in KwaZulu-Natal, South Africa was chosen as a study site. UMsunduzi River is a major tributary of uMngeni River that is used for water supply to the cities of Pietermaritzburg and Durban.

The study begins with the data synthesis from diverse sources of scientific data to identify chemical and microbial hazards, utilising a water quality modelling tool to map point and non-point source pollution in the catchment. The assessment encompasses the presence of pathogens such as *Cryptosporidium* and *Escherichia coli* (*E. coli*) in the catchment, with rural areas showing a greater contribution from animal sources, while urban areas are affected by impaired wastewater infrastructure.

Quantitative microbial risk assessment (QMRA) was conducted, assuming no water treatment within the catchment. The investigation considered multiple exposure routes, including domestic drinking and recreational activities for both adults and children. The results indicate that the probability of infection from *Cryptosporidium* and *E. coli* exceeds acceptable levels set by South African water quality guidelines and the World Health Organization. The assessment further included a chemical risk assessment on various chemical groups, including organochlorinated pesticides (OCPs), pharmaceuticals and personal care products (PPCPs), heavy metals, nitrates, and phosphates. Elevated carcinogenic risks were observed for most OCPs, while non-carcinogenic pesticide effects pose long-term risks. Heavy metals and PPCPs are within sub-risk levels, but phosphates have notable ecological and health impacts, particularly in Inanda Dam, a key source of potable water for Durban. In this study, a unique contribution is made by incorporating both chemical and microbial risk assessment. Furthermore, the risk assessment methodology not only encompasses various chemical pollutants and exposure pathways but addresses the nuanced issue of water consumption variability between children and adults.

To address these identified risks, a multi-criteria decision analysis methodology is employed to engage stakeholders in the risk management process. Affected, involved, and interested stakeholders, along with economic, environmental, and social criteria, contribute to the selection of Best Management Practices (BMPs). The Simple Multi-Attribute Rating Technique for Enhanced Stakeholder Take-up (SMARTEST) is utilised to identify suitable interventions. The study culminates in the recommendation of BMPs that aim to change behaviour, including public education on livestock grazing management, safe medication disposal, and responsible fertilizer and pesticide use. Pollution management measures, such as solid waste control and river clean-up, are suggested, along with infrastructure management improvements, like sewer system maintenance.

This research strived to bridge the gap in water pollution alleviation by presenting a practical and comprehensive framework designed to support well-informed decision-making processes. This framework, with its integration of multiple criteria and considerations, stands poised to aid infrastructure, operational, and ecological managers within a catchment in prioritising BMPs aimed at reducing pollution and mitigating resultant health impacts.

## Graphical Abstract

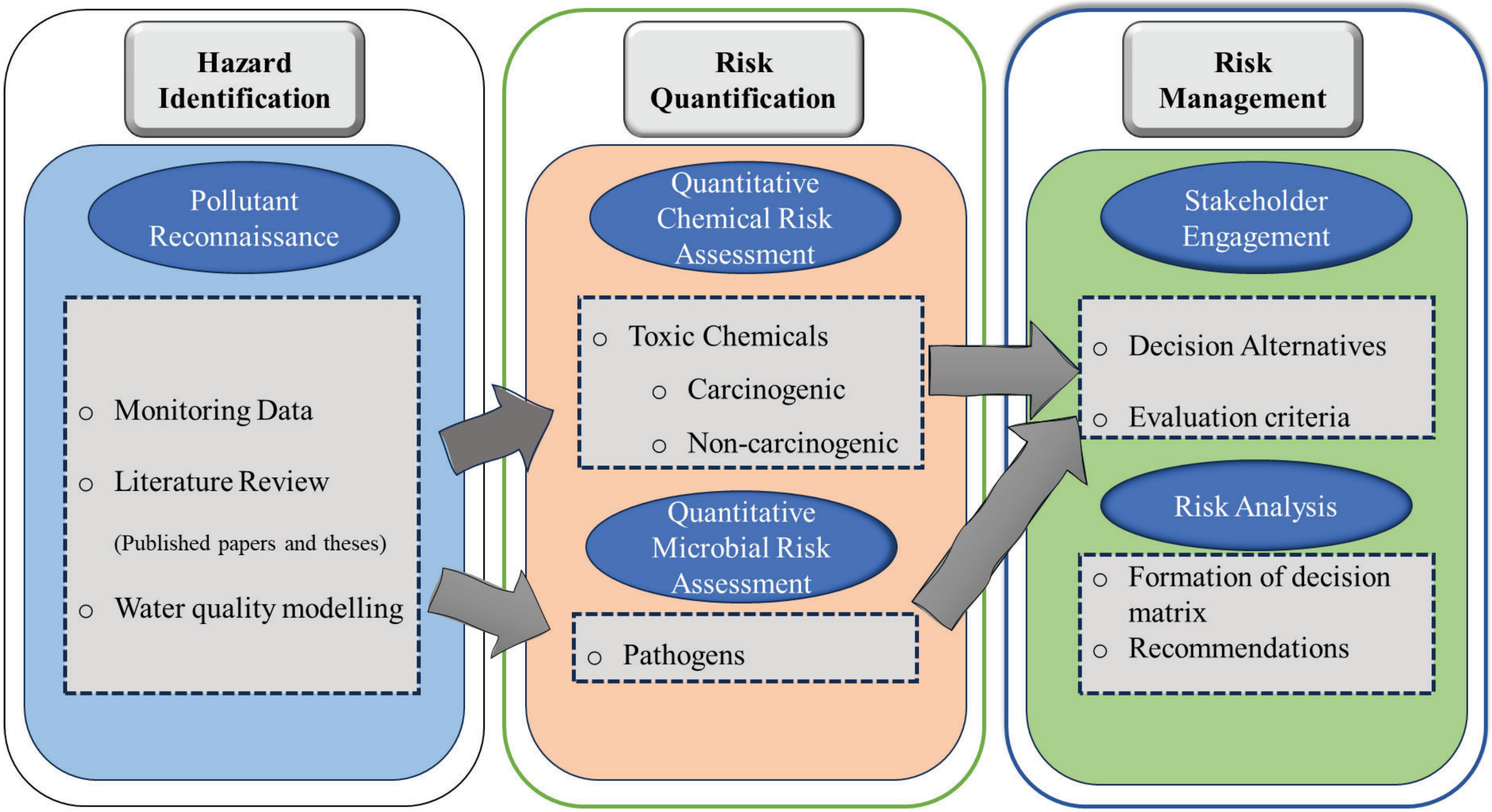

*Figure 0.1 Graphical abstract showing the methodology followed in this study from hazard identification, to risk quantification, and risk assessment.*

## List of Research Outputs

The following outputs have been produced from this thesis:

i) Ngubane, Z., Bergion, V., and Sokolova, E. (2021) ‘Is the uMsunduzi River water, South Africa, safe for drinking?’, in the Global Sustainable Futures (GSF) second annual online conference 2021. https://gmv.gu.se/digitalAssets/1797/1797724_parallel-sessions-summaries.pdf

ii) Ngubane, Z., Bergion, V., Dzwairo, B., Troell, K., Amoah, I. D., Stenström, T. A. and Sokolova, E. (2022) ‘Water quality modelling and quantitative microbial risk assessment for uMsunduzi River in South Africa’, Journal of Water and Health, 20(4). doi: 10.2166/wh.2022.266

iii) Ngubane, Z., Bergion, V., Dzwairo, B., Troell, K., Amoah, I., Stenström, T. A and Sokolova, E. (2022) ‘Water quality modelling to assess sources and transport of pathogens within uMsunduzi catchment, South Africa’, Conference abstract in EGU22. Vienna Austria. doi: https://doi.org/10.5194/egusphere-egu22-2157.

iv) Ngubane, Z., Bergion, V. and Sokolova, E. (2022) ‘Evaluating the health risks associated with drinking uMsunduzi River water, South Africa’, in *IWA-WHO International Conference on Water Safety*. Narvik, Norway, pp. 3–4. doi: 10.2166/wh.2009.101.Smeets.

v) Ngubane, Z., Dzwairo, B., Moodley, B., Stenström, T. A. and Sokolova, E. (2023) ‘Quantitative assessment of human health risks from chemical pollution in the uMsunduzi River, South Africa’, Environmental Science and Pollution Research. doi: https://doi.org/10.1007/s11356-023-30534-4.

vi) Ngubane, Z., Bergion, V., Dzwairo, B., Stenström, T. A. and Sokolova, E. (2023) *Engaging stakeholders in health risk management for the uMsunduzi River, South Africa.*”. Manuscript under review in Scientific Reports journal.

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## List of Abbreviations and Acronyms

| **Abbreviation** | **Explanation** |
|---|---|
| ACRU | Agricultural Catchments Research Unit |
| AGNPS | Agricultural Nonpoint Source Pollution Mode |
| AHP | Analytical Hierarchy Process |
| ANSWERS | Areal Nonpoint Source Watershed Environment Response Simulation Model |
| BMP | Best Management Practice |
| CDI | Chronic Daily Intake |
| CEC | Chemicals of Emerging Concern |
| CR | Cancer Risk |
| CSF | Cancer Slope Factor |
| d | Day |
| DSS | Decision Support Systems |
| DWAF | Department of Water Affairs and Forestry |
| *E. coli* | *Escherichia coli* |
| EDC | Endocrine Disrupting Compounds |
| ELECTRE | Elimination Et Choix Traduisant la Réalité |
| ES | Ecological Services |
| g | gram |
| GIS | Geographic Information System |
| HQ | Hazard Quotient |
| hr | Hour |
| HSPF | Hydrologic Simulation Program Fortran |
| HYPE | Hydrological Predictions for the Environment |
| ILCR | Incremental Lifetime Cancer Risk |
| kg | kilogram |
| L | Litre |

| | |
|---|---|
| MAUT | Multi-Attribute Utility Theory |
| MAVT | Multi-Attribute Value Theory |
| MCDA | Multi-criteria Decision Analysis |
| MCDM | Multi-criteria Decision Making |
| MCDSS | Multi-criteria Decision Support Systems |
| mm | millimetre |
| NSE | Nash-Sutcliffe Efficiency Coefficient |
| OCPs | Organochlorinated Pesticides |
| PBIAS | Percent Bias |
| Pinf | Probability of Infection |
| PLOAD | Pollution-loading Estimator |
| PPCPs | Pharmaceuticals and Personal Care Products |
| QCRA | Quantitative Chemical Risk Assessment |
| QMRA | Quantitative Microbial Risk Assessment |
| R | South African Rand |
| RfD | Reference Dose |
| RII | Relative Importance Index |
| ROC | Rank Order Centroid |
| ROD | Rank Order Distribution |
| RS | Rank-Sum Method |
| RSR | Ratio of the Root Mean Squared Error to the Standard Deviation of Measured Data |
| SAICE | South African Institute of Civil Engineering |
| SANS | South African National Standards |
| SDG | Sustainable Development Goal |
| SMART | Simple Multi-Attribute Rating Technique |
| SMARTER | Simple Multi-Attribute Rating Technique Exploiting Ranks |

| | |
|---|---|
| SMARTEST | Simple Multi-Attribute Rating Technique for Enhanced Stakeholder Take-up |
| SPATSIM | SPatial And Time Series Information Modelling |
| STATSSA | Statistics South Africa |
| SWAT | Soil and Water Assessment Tool |
| SWMM | Stormwater Management Model |
| USEPA | United State Environmental Protection Agency |
| WHO | World Health Organisation |
| WQSAM | Water Quality Systems Assessment Model |
| WRC | Water Research Council |
| WTP | Water Treatment Plant |
| WWTP | Wastewater Treatment Plant |

# Chapter 1 Introduction

## 1.1 Study Background

Globally, water utilities in both rural and urban areas face a daunting challenge in meeting the growing demand for safe water due to the rapid decline of freshwater resources and ageing water infrastructure (Salehi, 2022). To address this issue of declining water quality, the United Nations' 2030 sustainable development goal (SDG) 6 aims to "improve water quality by reducing pollution, eliminating dumping and minimising release of hazardous chemicals and materials, halving the proportion of untreated wastewater and substantially increasing recycling and safe reuse globally" (United Nations, 2015). However, improving water quality is a complex task that requires careful consideration of various factors, including engineering infrastructure and catchment management.

To effectively improve and manage water quality, a holistic understanding of environmental, social, and economic factors is necessary (Wuijts, Driessen and Van Rijswick, 2018). Historically, decision-making processes in water resource projects have often excluded civil society participation, favouring a limited group of experts to maintain control and ensure consistency (Simpungwe, 2006). The challenge identified in this context is the absence of cooperative governance necessary for effective integrated catchment management and water resource protection. As highlighted in Riemann *et al*. (2017), there exists a need to link and address the multiple sources of pollutants that arise due to urbanisation, multiple responsibilities over catchments, and consequently assisting with the required cooperative governance within catchments.

Water sources contaminated with faecal pollution, indicating poor sanitation and hygiene, are still prevalent worldwide, particularly in peri-urban areas of countries in the Global South (Hamilton *et al*., 2018). In the context of South Africa, the issue of water quality presents significant challenges. To illustrate, Oberholster and Ashton (2008), reported that inadequate sewer systems and overloaded treatment plants resulted in a portion of wastewater from urban areas being discharged into the environment without proper treatment. Almost 15 years later, du Plessis (2023) reported that approximately 50% of South Africa's wastewater treatment facilities are in disrepair or critical condition, posing significant risks to human health and contributing to the continued destruction of ecosystems.

Consequently, certain households in South Africa are forced to rely on water sources of compromised microbial and chemical quality for their domestic water needs (Luyt *et al*., 2012; Madilonga *et al*., 2021). Additionally, it has become evident that a lower percentage of households have reliable water supply in the year 2023 than in the year 1994 (du Plessis, 2023). This situation contributes to the prevalence of waterborne diseases, which particularly affect impoverished communities in the Global South (Bampoky, 2013; Chola *et al*., 2015). Diarrhoea, for example, disproportionately impacts socio-economically disadvantaged communities in South Africa and is the second leading cause of infant mortality in countries of the Global South due to inadequate sanitation and the lack of clean water supply (Squire and Ryan, 2017; Ugboko *et al*., 2020).

Additionally, studies have revealed the presence of chemicals of emerging concern (CECs) as pollutants in South African rivers, groundwater systems, and drinking water, suggesting potential chemical misuse and associated health risks (Pieters and Horn, 2020; Horak, Horn and Pieters, 2021). CECs identified in water bodies have been linked to failing wastewater infrastructure and illegal dumps (Agunbiade and Moodley, 2016). The complexity of chemical pollutants lies in their mixture, potentially leading to synergistic effects and long-term health risks with continuous exposure (Petrie, Barden and Kasprzyk-Hordern, 2014; Villanueva *et al*., 2014).

Health risk assessment based on harmful chemical concentrations, pathogen presence, dose-response relationships, exposure estimations, and risk characterisation is vital for managing water quality (World Health Organization, 2016). Through rigorous analysis of various risk factors, such as pollution sources, exposure pathways, and vulnerability assessments, it provides a systematic framework to quantify and prioritise risks based on their potential impact on human health and the environment (USDA/FSIS and USEPA, 2012; Genthe *et al*., 2018; Limaheluw, Medema and Hofstra, 2019). This quantitative approach enables decision-makers within a catchment to identify critical areas requiring immediate attention and allocate resources efficiently. Complementary to the risk assessment, water quality modelling offers a dynamic representation of water systems, considering various sources, processes, and scenarios (Slaughter and Mantel, 2017). Simulation may provide insights into the behaviour of water pollutants, their transport mechanisms (Medema, 2013). In particular, fate and transport models could offer understanding into the behaviour of water pollutants and their transport mechanisms if the models can be tested on monitoring data (Drummond *et al*., 2022). Due to scarcity of scientific data on certain important pollutants like *Cryptosporidium* and *Giardia* (Hamilton *et al*., 2018), water quality modelling can be used to patch the data gap.

Multi-criteria decision support analysis (MCDA) offers a suitable methodological framework for thoughtful assessment, proving to be beneficial in dealing with diverse value aspects associated with shared resources (Saarikoski *et al*., 2016; Huttunen *et al*., 2022). The development of such a decision-support framework can help decision-makers consider and evaluate multiple aspects that impact water quality, such as pollutant sources, pollution health impacts, environmental factors, stakeholder perspectives, and infrastructure requirements. By incorporating these diverse elements, the framework enables a more holistic and integrated approach to water quality improvement. Effective navigation and interpretation of the outputs allow decision-makers to make informed choices and prioritise actions that address water quality challenges efficiently. Tumwebaze *et al*. (2021) stressed the importance of tailoring water resource decision tools to stakeholders' needs, simplifying the tools for ease of use, creating an enabling environment for active stakeholder participation, involving stakeholders early in the decision process, and acquiring local data resources.

So far, there has been limited research focus on combining water quality modelling, quantitative chemical and microbial risk assessment, and stakeholder engagement to prioritise catchment areas facing water pollution problems and devise effective pollution mitigation strategies. This research aimed to address this gap by providing a practical and comprehensive framework that supports well-informed decision-making processes in water pollution alleviation. By integrating multiple criteria and aspects, this framework can assist infrastructure, operational, and ecological managers within a catchment in prioritising best management practices (BMPs) to reduce pollution and mitigate resulting health impacts. These BMPs encompass a range of approaches, from land use management strategies that minimise pollutant wash-off, including public awareness initiatives (non-technical), to engineering structures that intercept and treat runoff (technical). Ultimately, applying the proposed framework in a specific catchment area provides an opportunity to assess its effectiveness and applicability in real-world contexts. This allows for testing, refinement, and validation, ensuring its practicality and relevance in addressing water quality issues specific to the considered catchment.

This study, therefore, focuses on the uMsunduzi catchment in Pietermaritzburg, KwaZulu-Natal, South Africa. The uMsunduzi River serves as a major tributary of the uMngeni River, which ultimately flows into Inanda Dam, the primary water supply for the eThekwini Metropolitan Municipality. The Draft Integrated Development Plan (DIDP) for the financial years 2022 to 2027 (Msunduzi Municipality, 2022) for the uMsunduzi Municipality highlights consistently poor to very poor levels of water quality based on the river monitoring results. An investigation of microbial water quality of the uMsunduzi River revealed pollution with microorganisms associated with waste disposal, including *Salmonella* spp., enterococci, and *E. coli* (Gemmell and Schmidt, 2013). Additionally, inadequate solid waste removal services in certain areas contribute to significant pollution levels, as residents often use riparian areas as convenient dumping sites for refuse (Matongo *et al*., 2015; Ndlela, 2016; Msunduzi Municipality, 2022). Chemical pollution has also been highlighted by studies such as Matongo *et al*. (2015); Moodley, Birungi and Ndungu (2016); Adeyinka *et al*. (2019). Individuals who rely on or utilise water from these highly polluted riparian areas for drinking, cooking, and irrigation purposes face severe health risks (WRC, 2002). As a result, the uMsunduzi River water is unsuitable for human consumption without treatment and largely unfit for recreational use (Gemmell and Schmidt, 2013).

The subsequent sections outline the research hypothesis, aims and objectives, as well as the structure of this thesis.

## 1.2 Research Hypothesis

A practical and comprehensive framework integrating water quality modelling, risk assessment and stakeholder participation in catchment management can facilitate well-informed decision-making processes in water pollution alleviation.

## 1.3 Research Aim and Objectives

The study aimed to develop a multi-criteria decision support framework integrating risk assessment to tackle pollution impacts within a catchment area, while accommodating various decision constraints.

The specific objectives of the study were to:

1. Conduct hazard identification by assessing existing scientific data on river pollution.

2. Utilise a water quality modelling tool to delineate point and non-point source pollution within a selected catchment area, emphasising hotspots and trends.

3. Conduct quantitative risk assessments to analyse potential risks to human health, encompassing both chemical and microbial hazards linked with the identified pollutants and water usage.

4. Develop a multi-criteria decision analysis methodology to foster stakeholder engagement in selecting best management practices for risk management at a catchment scale.

## 1.4 Structure of the Thesis

This thesis comprises six (6) main chapters, structured as follows:

- Chapter 1 Introduction: Introduction of the study gives the background of the overall study, the research hypothesis, aims and objectives, as well the structure of the dissertation.
- Chapter 2 Literature Review and Study Area Background: Presents the literature review of the overarching concepts of the study as well as a brief background of the study area.
- Chapter 3 Microbial Water Quality Modelling and Quantitative Microbial Risk Assessment: This chapter presents a full study performed on the coupled microbial water quality modelling and quantitative microbial risk assessment of the catchment.
- Chapter 4 Quantitative Chemical Risk Assessment: Presents the quantitative chemical risk assessment in relation to chemical pollution for recreational and domestic water users within the uMsunduzi catchment.
- Chapter 5 Multi-criteria Decision Analysis: Presents the application of multi-criteria decision support within uMsunduzi catchment.
- Chapter 6 Synthesis: Discusses the aims and objectives of this study and gives general conclusions drawn in this study as well as any limitations of this study.

References used and cited within the thesis are listed in the reference section.

# Chapter 2 Literature Review and Study Area Background

## 2.1 Literature Review

### 2.1.1 Introduction

Apart from its life-sustaining role, water has an economic value for various in-stream and off-stream uses such as domestic use, agriculture, industry, transportation, recreation, waste assimilation, and ecosystem maintenance (Gibbons, 2013). The South African National Development Plan 2030 (NDP2030) states that food, fuel, and water are interconnected (SA Government, 2015). Within this context, there is increasing concern regarding the quality of South Africa's water resources and the contribution of untreated effluent or poorly treated wastewater discharges to the already deteriorating water quality. Decision makers, investors, and researchers commonly agree that the deteriorating water quality will adversely affect the South African economy, both in the immediate and long-term future (Merwe-Botha, 2009).

This chapter presents a literature review that follows the objectives of this study. The subsections are as follows. Section 2.1.2 presents the literature narrative on factors that affect water quality, water quality assessment, and the effect on human health. Section 2.1.3 presents a review of water resource governance and management. Section 2.2 presents a short background to the study area.

### 2.1.2 Water Quality and Human Health

#### a. Factors affecting water quality

The quality of water is determined by its chemical, physical, and biological attributes, which dictate its suitability for various purposes and the preservation of the health and stability of aquatic ecosystems (Edokpayi, Enitan-folami and Adeeyo, 2020). Water pollution has detrimental effects on aquatic organisms, it exacerbates the spread of waterborne diseases through the pollution of drinking water (Sahoo and Goswami, 2024). Water pollution is a significant environmental challenge that has adverse effects on the quality and accessibility of water resources, consequently endangering the sustainability of ecosystems and human health (Adu and Kumarasamy, 2018; Sahoo and Goswami, 2024). Sustainable development aims to achieve a balance between environmental, social, and economic systems for long-term sustainability (Sahoo and Goswami, 2024). While water quality impairment is a global issue (Wear *et al*., 2021; Sahoo and Goswami, 2024), countries in the Global South are particularly at risk due to limited access to clean water, impacting public health and economic development (Amer *et al*., 2013; Bodager *et al*., 2015).

The overuse of fertilizers, manure, and pesticides to improve crop yield and protect crops can have negative consequences (Horak, Horn and Pieters, 2021) on the environment and rivers. Additionally faecal matter from grazing animals arising from intensive agriculture are examples of diffuse sources of pollution (Neill *et al*., 2018). An instance of this is when excessive amounts of nitrogen and phosphorus from agricultural fields are carried into water bodies through runoff (Isiuku and Enyoh, 2020). The increased nutrient content in the water accelerates eutrophication, leading to the death of fish and other aquatic organisms (Giri and Qiu, 2016; Isiuku and Enyoh, 2020). Pettigrove *et al*. (2023) pointed out that while pesticides are commonly associated with agricultural activities, urban areas utilise larger quantities and varieties of pesticides compared to farms. Urban pesticide use is prevalent due to the need for pest control in gardens, lawns, and even on golf courses.

Mainly, the cities have the highest pollution rate with inadequate waste management systems that have resulted in excessive pollution of receiving waterbodies (Ejigu, 2021). Wastewater discharges due to impaired wastewater infrastructure have been reported to be the major sources of pollution contributing to oxygen demand and nutrient loading of the waterbodies; promoting toxic algal blooms and leading to a destabilised aquatic ecosystem (Morrison *et al*., 2001; WRC, 2002). These have contributed in the increased pathogen and chemical load entering the water sources increased disease burden (Sokolova, 2013; Bergion *et al*., 2018; Odeniji, Okoh and Okoh, 2019). To illustrate, most wastewater treatment facilities in South Africa dispose their effluents directly to nearby rivers or streams, which are used by the surrounding villages for their various water needs (Edokpayi, Enitan-folami and Adeeyo, 2020).

According to a study conducted by Edokpayi, Enitan-folami and Adeeyo (2020), several key factors contribute to the deterioration of wastewater treatment facilities in South Africa and other developing countries. These factors include a lack of sufficient treatment facilities in both urban and rural areas, poor operational conditions of the existing infrastructure, design flaws in the facilities, a lack of expertise in managing the systems, corruption, inadequate funding for wastewater treatment, overburdened capacities of existing facilities, and insufficient monitoring to ensure compliance with recommended guidelines. These assertions find further support from (du Plessis, 2023), highlighting the persistent infrastructure challenges that continue to impact the quality of river water over time.

### b. Water quality assessment

Assessment of water quality and the pollution sources identification are a requirement for efficient water resource management (Talukdar, Kumar and Kulkarni, 2023). Water quality variables are identified using various techniques, including non-spatial land use matrix, riparian zone approach, hydrologic sensitive area, and critical source area (Giri and Qiu, 2016). These variables undergo processing using different indices to capture their unique influence on water quality (Giri and Qiu, 2016). The selected explanatory variables are then further processed using hydrologic/water quality models, statistical modelling, or monitoring data to determine the sources and causes of water quality degradation (Qiu *et al*., 2018). The results of water quality assessment inform watershed managers and policymakers, enabling them to take appropriate actions to safeguard water quality (Qiu *et al*., 2018).

Talukdar and Kumar (2023) asserted that river water quality monitoring paired with water quality models can result in a better assessment of water quality. Water quality modelling is a common approach to increase insight, for example by pin-pointing hotspots of high concentrations or identifying the relative importance of pollutant sources (Vermeulen *et al*., 2019). In response to water-related challenges, water quality modelling has been developed to analyse, understand and explore solutions for sustainable water management, to the decision makers and operational water managers (Abdullah *et al*., 2019). Over the years, hydrologists have developed numerous predictive tools (empirical models, lumped models, distributed models, and statistical models) that aid decision-making with respect to water resources and water quality management (Bougeard *et al*., 2011). Due to a lack of secondary data or constrictive assumptions, applications of more complex models have long encountered challenges (Bougeard *et al*., 2011; Devia, White *et al*., 2017). Nevertheless, panels of tools have recently been successfully applied to compare and display how potential scenarios will impact watersheds, especially with regard to runoff and water quality (Bougeard *et al*., 2011).

Regarding water quality, models can incorporate a heat balance to estimate water temperatures and simulate the behaviour of different types of chemicals (Keller *et al*., 2023). These models can consider conservative chemicals that do not undergo significant reactions with other substances, such as chloride, sodium, and calcium. They can also account for reactive chemicals, including organic compounds that degrade and bind to soils and sediments.

Furthermore, these models can simulate the complex biogeochemical cycles of elements like nitrogen, phosphorus, oxygen, carbon, and mercury. Water quality models vary in their capacity to represent crucial processes such as atmospheric deposition, chemical loading from various land uses (for example pesticides and fertilizers), discharges from both point and non-point sources into water bodies, as well as in-stream processes (Keller *et al*., 2023). In their study, Talukdar, Kumar and Kulkarni, (2023) evaluated water quality models and monitoring techniques with regards to their effectiveness in identifying, categorising, and simulating the transport of pollutants. Their findings can aid in selecting the most appropriate model to reveal the distribution of biochemical pollutants in a surface water. They also emphasised that employing water quality models and algorithms to simulate the dispersion of pollutants from both point and non-point sources can significantly enhance the understanding of pollution control measures.

In point and non-point source pollution modelling, several notable models have been developed, each with its own set of strengths and limitations as noted in Table 2.1. The AGNPS (Agricultural Nonpoint Source Pollution Model) stands out as an event-driven distributed model that encompasses hydrology, chemical, and erosion/sediment transport components. Its notable strengths lie in its ability to simulate the spatial distribution of catchment properties and provide immediate responses compared to experimental data, making it a valuable tool for understanding erosion processes (Leon and Bowen, 2003; Borah and Bera, 2004; Adu and Kumarasamy, 2018; Naik, Kumar and Deshmukh, 2018). However, its use comes with significant demands, including the need for extensive and exclusive data, as well as substantial programming competence. Furthermore, it falls short when it comes to assessing nutrient transformation and in-stream processes (Leon and Bowen, 2003).

On the other hand, the ANSWERS (Areal Nonpoint Source Watershed Environment Response Simulation Model) adopts a distributed parameter, event-oriented approach, incorporating hydrology, evapo-transpiration, infiltration, and overland sediment transport components. Its programmable nature allows for easy modification, and its predicted results align well with observed data (Borah and Bera, 2004; Adu and Kumarasamy, 2018). Nevertheless, ANSWERS has its limitations, as it cannot simulate in-stream processes, sub-processes like snowmelt or pesticide movements, and it necessitates elaborate and detailed input data preparations, which can be computationally intensive (Adu and Kumarasamy, 2018). Additionally, its sensitivity to slight input variable changes poses challenges in the validation process.

The SWAT (Soil and Water Assessment Tool) model operates on a continuous, semi-distributed, basin-scale level, incorporating hydrology, climate, nutrients, and sediment transport (Gassman *et al*., 2007; Liu *et al*., 2019). It excels in considering the long-term impacts of rural and agricultural management practices and conflux and sediment confluence processes. SWAT can also be adapted for specialised applications like bacteria transport (Bergion *et al*., 2017; Löwenström and Hussain, 2017). However, it demands an extensive number of input files, lacks the capability to simulate daily changes in dissolved oxygen in water bodies, struggles to accurately evaluate extreme daily flow occurrences, complex dynamic soil nitrogen and carbon evolution, and runoff yield simulation (Adu and Kumarasamy, 2018). Moreover, the model's database must be modified for different study areas, adding to its complexity.

Lastly, the HSPF (Hydrologic Simulation Program – Fortran) model operates as a long-term continuous simulation lumped model, encompassing hydrology, climate, and sediment transport components. It excels in evaluating runoff flow rates, sediment transport, and nutrient and pesticide concentrations, offering satisfactory predictive capabilities and the ability to simulate in-stream processes (Gericke, 2003; Adu and Kumarasamy, 2018; Xie *et al*., 2019). However, its reliance on numerous empirical relationships to represent physical processes necessitates extensive calibrations and a high level of expertise (Adu and Kumarasamy, 2018). It also lacks consideration for the spatial distribution of watersheds and necessitates substantial datasets for its operation. These variations in strengths and limitations highlight the importance of selecting the most suitable model depending on the specific research objectives and available resources (Xie *et al*., 2019).

Water quality research and modelling in South Africa is relatively underdeveloped due to lack of observed data, technical modelling expertise and funding, despite the rich history of hydrological modelling and systems modelling for flow (Slaughter and Mantel, 2018). Slaughter and Mantel (2018) used the Water Quality Systems Assessment Model (WQSAM) in Buffalo River Catchment within the Eastern Cape, South Africa, with satisfactory results. Slaughter (2017) presented a microbial module for WQSAM for modelling microbial water quality in regions where data is limited. The study findings demonstrated that the model accurately captures the variations observed in the data from a region with limited information. Despite its simplicity, Slaughter (2017) argues that the uncertainty associated with this model is either comparable to or even lower than that of a complex model applied to a similar data-scarce area. WQSAM is run as part of the SPatial And Time Series Information Modelling framework (SPATSIM) as it requires an existing water yield model setup.

*Table 2.1 Summary on non-point source pollution models after (Leon and Bowen, 2003; Borah and Bera, 2004; Adu and Kumarasamy, 2018; Naik, Kumar and Deshmukh, 2018; Xie et al., 2019)*

| ***Model*** | ***Type*** | ***Major components*** | ***Strengths*** | ***Limitations*** |
|---|---|---|---|---|
| AGNPS (Agricultural Nonpoint Source Pollution Model) | Event-driven distributed model | Hydrology, chemical, and erosion/sediment transport | Simulates spatial distribution of catchment properties; gives immediate response compared with experiments; useful in understanding erosion processes | Requires exclusive and extensive data and much work and programming competence;<br>it is unable to assess nutrient transformation and in-stream processes |
| ANSWERS (Areal Nonpoint Source Watershed Environment Response Simulation Model) | Distributed parameter, event-oriented model | Hydrology, evapo-transpiration, infiltration, and overland sediment transport | Program codes are easy to modify and predicted results are comparable with observed data | Unable to simulate in-stream processes and simulate sub-processes like snowmelt or pesticides;<br>requires elaborate and detailed input data preparations and is computationally intensive;<br>output is sensitive to slight changes in input variables and this makes its validation challenging |
| SWAT (Soil and Water Assessment Tool) | Continuous, semi-distributed, basin-scale model | Hydrology, climate, nutrients, sediment transport | Considers long-term impacts of rural and agricultural management practices, and conflux and sediment confluence; can be used for specialised processes such as bacteria transport | Requires large numbers of input files to run the model; unable to simulate daily changes of dissolved oxygen in water bodies and cannot accurately evaluate extreme daily flow occurrences, complex dynamic evolution of soil nitrogen and carbon, and simulation of runoff yield; database must be modified when used<br>in different study areas |
| HSPF (Hydrologic Simulation Program – Fortran) | Long-term continuous simulation lumped model | Hydrology, climate and sediment transport | Effectively evaluates the flow rate of runoff, sediment transport, and nutrient and pesticide concentrations; predicts results satisfactorily and is capable of simulating in-stream processes | Relies on many empirical relationships to represent physical processes and requires extensive calibrations and<br>a high level of expertise; high data sets are required and it does not consider the spatial distribution of watersheds; complex |

Another South African-developed rainfall-runoff model, ACRU (Agricultural Catchments Research Unit) was applied in the uMngeni catchment with satisfactory results, to simulate the flow of water, sediment, *E. coli*, and phosphorus (Dlamini *et al*., 2011) and its sub-model, ACRU-NPS in the Mkhabela subbasin (41 km2), to simulate nitrate, phosphorus and sediments (Kollongei and Lorentz, 2015). In addition, the pollution-loading estimator (PLOAD) was successfully used in the simulation of the total phosphorus export-coefficient at quaternary catchment level (Dabrowski *et al*., 2013). Namugize *et al*. (2017) also reviewed the use of Hydrological Predictions for the Environment (HYPE) model for simulation of water and nutrients in the Upper uMngeni River catchment. An under-estimation of streamflow was identified in the outlet sub-catchments, due to a simplified spatial variation of evapotranspiration processes in the model. HYPE provided acceptable simulations of stream flows, and the good fits between modelled and measured values in four out of the nine subbasins (Namugize *et al*., 2017).

SWAT was used in the Mkhabela subbasin, as a comparative catchment-scale alternative to ACRU-NPS (Gorgens, 2012). In comparison with ACRU-NPS, the SWAT-simulated annual average pollution loadings in the Mkhabela subbasin were lower, indicating under-representation. Primary causes of these discrepancies were attributed to different model representations of site-specific processes of overland versus subsurface discharge and nutrient flux. SWAT adequately incorporate the physics of important hydrological structures, such as wetlands, farm dams and channel roughness (Gorgens, 2012). Löwenström and Hussain (2017) suggested that the model predictions in SWAT for the uMngeni catchment could be improved if more input data and information were available; also, if more observed data of water flow and microbial concentrations in rivers were available, the model results could be calibrated to a greater degree of accuracy.

The SWAT models have also been widely used for pesticides, pathogen, and faecal indicator concentrations at catchment scale (Hofstra *et al*., 2019; Ippolito and Fait, 2019). One example is the coupling of SWAT with other models like AgDRIFT, which helps in improving the estimation of exposure to agrochemicals (Wang *et al*., 2019). Another use of SWAT is providing agrochemical loadings or concentrations as a starting point for ecological risk assessment, cost-effectiveness analysis, optimal BMP selection, and placement. These applications allow for a more comprehensive understanding of the potential impacts of agrochemicals on the environment and enable better decision-making in agricultural management (Wang *et al*., 2019). While there are many models for pesticide entry and behaviour in surface water bodies, the spatio-temporal scale of their application has significant implications for the information provided by these models (Ippolito and Fait, 2019).

### c. Water quality impact on human health assessment

Environmental pollution has long been linked to health problems, such as the spread of diseases like cholera and typhoid, some of which are primarily regarded as waterborne illnesses (Siddiqua *et al*., 2022). The World Health Organization (WHO) has stated that the biggest threat to human health due to water pollution is attributed to the presence of microbial pathogens in the water (World Health Organization, 2016). These organisms could be bacteria, parasites, fungi, or viruses  (Abraham, 2011). It is important to understand the transmission of pathogens via rivers for evaluation of disease risk. Insight in the relative contribution of human versus animal sources, point versus diffuse sources and pathways and effect of control measures is important to design effective management strategies (Vermeulen *et al*., 2019). It is challenging to eliminate every risk in water systems as resources for risk reduction are limited, which necessitates prioritisation of risk-reduction measures when balancing risks, costs, and benefits. The design of a framework that prioritises public health through targets, limits, and estimations of risks, is vital.

Risk assessment is a logical method for identifying all aspects of risks and examining the possible outcomes of accidents on people, materials, products, tool, and the environment (Rahnamay and Elnaz, 2022). The risk assessment estimates the infectious/illness risk of exposure to pollutants, based on dose-response relations between the pollutant concentrations and the health effects (Haas *et al*., 2000; Sunger *et al*., 2012). Risk assessment provides valuable support in the risk analysis, since it predicts the risk of illness, even when the fraction of disease attributable to water is low or difficult to measure directly through public health surveillance or epidemiological studies (Federigi *et al*., 2019).

The risk assessment has been used for drinking water in order to set health targets for water consumption (Galal-Gorchev, 1993). In addition, risk assessment was recommended as the preferred methodology to evaluate the risks associated with greywater or wastewater reuse in agriculture (Angelakis, 2003), or potable reuse in the USA, as described in the recent EPA compendium (USEPA, 2018). In 2003, the risk assessment was suggested by the WHO as a valid methodology for recreational water risk management (Angelakis, 2003), and to date some governments have already integrated risk assessment in their guidelines, such as Australia (Australian Government, 2008) and New Zealand (MfE, 2003) among others.

Possible risks and hazards (pollutants) are identified by the risk assessment, which also examines their root causes and effects before defining risk (Aven, 2011). Additionally, it offers a framework for describing the potential effects of the pollutant under study, determining if the risk is tolerable or acceptable, and selecting the most effective risk management strategy (Aven, 2011). The primary objectives of risk management encompass two key aspects (Aven, 2011). Firstly, it aims to ensure the implementation of sufficient measures to safeguard individuals, the environment, and assets against any undesirable outcomes arising from the activities being carried out. Secondly, it seeks to strike a balance between various considerations, such as safety and financial implications, by effectively addressing and managing potential risks.

Quantitative chemical or microbial risk assessment is a four element process comprising of: problem formulation and hazard identification, exposure assessment, health effects assessment, and risk characterisation (Powell and Danby, 2012; Haas, Rose and Gerba, 2014; WHO, 2015, 2020). Exposure assessment is the quantitative assessment of the probability that water consumers ingest pollutants. Quantitative data are required, including the pollutant concentrations in the water sources utilised and the pollutant's fate in barriers (such as water treatment processes) in both routine and emergency situations. On the other hand, information about human exposure is required (WHO, 2015), such as 1) the size of the population exposed, 2) the nature of the population exposed (such as vulnerable groups), and 3) the frequency of their exposure (such as daily). In the many studies that have looked at risk assessment, this step includes conventional water treatment (Kim, Kabir and Jahan, 2017; Bergion *et al*., 2018; Federigi *et al*., 2019; Maleki and Jari, 2021).

The assessment of the health effects is connected to the degree of exposure to water-borne pathogens. Information on the dose-response relationship between ingested dose and the likelihood of health effects like infection or illness are assessed (WHO, 2015). The dose estimation is the concentration of pollutants consumed per unit time. A risk estimate is created by combining the data from the exposure and effect assessments during the risk characterisation process (Powell and Danby, 2012). One of the key challenges in QMRA is the estimation of pathogen concentrations in water sources. Studies have used a range of approaches to estimate pathogen concentrations (Coffey *et al*., 2013), including traditional culture-based methods (Potgieter *et al*., 2018), molecular-based methods, and models (Seidu *et al*., 2013; Mohammed and Seidu, 2019). Some studies have also explored the use of remote sensing and geographic information systems to estimate the spatial distribution of pathogens in water sources (Pisinaras *et al*., 2010; Fadil *et al*., 2011).

Haas, Rose and Gerba (2014) used QMRA to assess the risks associated with the consumption of produce contaminated with norovirus. The authors found that the risk of infection was dependent on several factors, including the concentration of the virus on the produce, the level of human susceptibility, and the effectiveness of the washing process. In a study by Mohammed and Seidu (2019), the effect of climate-driven precipitation on pathogen infection risks in drinking water treatment plants (WTPs) in Norway was assessed using QMRA. According to the findings of the study, the QMRA models revealed the need for enhanced optimisation of critical treatment steps in WTPs, as well as the introduction of strict rules to protect raw water sources. According to Derx *et al*. (2016), protecting raw water resources necessitates addressing all relevant faecal pollution sources in the catchment under consideration in their study on faecal pollution review of Danube River in Austria. Similarly, Hamilton *et al*. (2018) studied the protozoan pathogens *Cryptosporidium* and *Giardia* in wastewater and surface water environments and they highlight the importance of raw water protection and monitoring strategies that will minimise public health risks.

Gitter *et al*. (2020) estimated gastrointestinal illness risks associated with exposure to waters impacted by human and nonhuman faecal sources. Through this study, it was concluded that identifying the sources contributing to the microbial impairment of water is critical to estimate the human health risk associated with recreation in a waterbody. In an earlier study, Ahmed *et al*. (2018) performed a quantitative microbial risk assessment of microbial source tracking markers in recreational water contaminated with fresh untreated and secondary treated sewage. Both these studies provided a valuable context for water quality managers to evaluate human health risks associated with faecal pollution. Additionally, Owens *et al*. (2020) reviewed 39 international studies on quantitative microbial risk assessment from 2003 to 2019. In their review most of the studies examined the transmission of pathogens from the source water to the consumer through the water supply system. The most frequently used reference pathogen was *Cryptosporidium*, and the only other protozoal reference pathogen considered was *Giardia*. Among the bacterial reference pathogens, *E. coli* was the most used.

Chemical substances that find their way into water resources can pose a significant risk to human health and the environment. Quantitative chemical risk assessment (QCRA) is an important tool for understanding the potential risks posed by chemicals to water resources. This type of assessment uses mathematical models to estimate the likelihood and magnitude of adverse effects on human health and the environment. Using probabilistic risk assessment, various studies have performed chemical risk assessment in different water sources. Risks posed by pollutants such as heavy metals (Muhammad, Shah and Khan, 2011; Anyanwu and Onyele, 2018; da Silva Bonifácio *et al*., 2021), organochlorinated pesticides (Chen *et al*., 2020), and pharmaceuticals and personal care products (Hernando *et al*., 2006; Sengar and Vijayanandan, 2022) in surface water and groundwater systems have been assessed. These studies combined exposure and hazard data to estimate the probability of adverse effects occurring and can be used to prioritise chemicals for further investigation or to guide risk management decisions.

Li and Qian (2011) investigated the human health risk of chemical pollutants in drinking water sources in Shizuishan City, Northwest China, assessing both carcinogenic and non-carcinogenic pollutants. It was found that chemical carcinogens were the primary pollutants in the groundwater. Similarly, in the rural areas of Kurdistan (Iran), Maleki and Jari (2021) evaluated drinking water quality and non-carcinogenic and carcinogenic risk assessment of heavy metals. Here the water resources of some villages were found to have non-carcinogenic effects for children due to nitrate pollution and carcinogenic risk due to arsenic pollution.

In South Africa, Manickum and John (2014) evaluated the occurrence, fate and environmental risk assessment of endocrine disrupting compounds at the wastewater treatment works in the city of Pietermaritzburg. This study concluded that steroid oestrogens potentially contaminate surface waters downstream of wastewater treatment plants due to inadequate removal during wastewater treatment. These studies highlighted the need for various control programs such as reducing and monitoring of pollutants, controlling agricultural runoff, rehabilitation of water resources, and household water treatment to protect the health of consumers (Maleki and Jari, 2021).

Literature on quantitative risk assessment gives foundation to and highlights the need for risk management. Thus, simply identifying risks and assessing their likelihood of occurrence and potential consequences is insufficient. It must be expanded to include identifying solutions and mitigation measures, implementing those mitigation efforts, and constantly monitoring post-mitigation performance for changes and the need for corrective action (Karunathilake and Bakhtavar, 2020). In managing the risks, the perspectives and varying decision priorities of all stakeholder groups have to be considered (Karunathilake and Bakhtavar, 2020).

### 2.1.3 Catchment Management in Relation to Water Quality

#### a. Best management practices

The selection of the mitigation measures also known as the best management practices (BMPs) requires holistic frameworks that integrate information that assists catchment decision-makers in a simplistic and clear format. BMPs can be both physical structural practices and non-structural practices that have direct impacts on the release, transport, or discharge of pollutants (Mishra, Kar and Singh, 2007; VanBergen and Nguyen, 2017). Structural BMPs include a variety of practices that rely on a wide range of hydrologic, physical, biological, and chemical processes to improve water quality and manage runoff (Uniyal *et al*., 2020). Structural BMPs are engineered systems and methods designed to provide temporary storage and treatment of runoff for the removal of pollutants (Muthukrishnan *et al*., 2005).

Structural BMPs are aimed at controlling the total volume and peak discharge rate of stormwater runoff, reducing pollutants in the stormwater via chemical, physical, and/or biological approaches. Common examples of structural BMPs include detention ponds and constructed wetlands (Mishra, Kar and Singh, 2007). Structural BMPs are used to treat the water either at the point of generation or at the point of discharge either to the storm sewer system or to receiving waters (Panagopoulos, Makropoulos and Mimikou, 2012). The selection and successful design of selected structural BMPs for water quality enhancement is the cornerstone of stormwater management in newly developing and redeveloping urban areas. Structural BMPs require commitment of resources for initial construction and continuing operation and maintenance (Panagopoulos, Makropoulos and Mimikou, 2007).

Stormwater ponds are basins that are used to attenuate peak storm discharges, act as water treatment devices, provide amenities and preserve biodiversity (Rohrer, 2014). In the City of Cape Town, South Africa ~800 stormwater ponds used for the management of the city's stormwater runoff exist as were reported by Rohrer (2014). Constructed wetlands are built wetlands that mimic the physical, biological, and geochemical processes seen in natural wetlands to attenuate pollution (Desiderati, 2022). According to Desiderati (2022), the effectiveness of constructed wetlands for attenuating pollutants varies according to factors including flow rate, pollutant loading, maintenance procedures, and design elements.

Non-structural BMPs such as education and source control regulations typically depend on a combination of behavioural change and enforcement (Kosco and Moeller, 2009; Lam, Schmalz and Fohrer, 2011). Non-structural or source control BMPs are practices that prevent pollution by reducing potential pollutants at their source before they come into contact with stormwater (Muthukrishnan *et al*., 2005; Lam, Schmalz and Fohrer, 2011). These BMPs aim to eliminate pollution by preventing their introduction into the environment. Non-structural controls include regulatory controls that prevent pollution problems by controlling land development and land use, as well as source controls that reduce pollutant build-up or lessen its availability for wash off during rainfall (Taylor and Wong, 2002). Examples of non-structural BMPs include public education and outreach, public involvement and participation, and illicit discharge detection and elimination.

A public education and outreach plan provides the municipality with a strategy for involving the public in making catchment management decisions, and for educating its employees, the public and businesses about the importance of protecting stormwater from improper use, storage and disposal of pollutants (Muthukrishnan *et al*., 2005). The results from Brehm and Eisenhauer (2021) in a study performed in Central Illinois, United States of America showed that specific exposure to the public education and outreach plan through a place-based experience supported by a local organisation had a good effect, but it is still unclear to what extent such information can be transferred within a watershed without additional facilitation. Public involvement and participation involve encouraging community participation, forming partnerships, and combining efforts of other groups in the community that encourage everyone to work towards the same goals (USEPA, 2022).

Public education and outreach (and others) were discussed in the reported actions and expenditures against socioeconomic variables to identify the relationships between cost, socioeconomics, and the best management practices (BMPs) used for compliance in Wisconsin, United States of America (McDonald and Naughton, 2019). Illicit discharge detection and elimination refers to discharge of waste that does not consist entirely of stormwater into a municipal storm drain (USEPA, 2022). A flowchart method for source detection of illicit discharges into stormwater drainage systems in Cape Town, South Africa, was developed by Owusu-Asante (2021). While the flowchart method was successful for Cape Town, it's use outside Cape Town may result in 'false positives' or 'false negatives' (Owusu-Asante, 2021).

### b. Multi-criteria decision analysis in water quality management

Improved catchment management provides more practical choices to increase the effectiveness of water use (Kumar *et al*., 2019). Therefore, catchment management needs to consider the holistic urban water cycle in their planning. According to Manny (2023), designing socio-technical fit structures may help improve outcomes in terms of the efficiency of engineering infrastructure and the ensuing environmental effects. Socio-technical fit implies that technical and social systems should align in order to achieve successful, efficient, and sustainable outcomes for water infrastructure (Manny, 2023).

Grabowski *et al*. (2017) discussed two key challenges in infrastructure planning: one is related to the physical aspects and the other to the social aspects. From a physical standpoint, engineering infrastructure must continuously advance in its design, implementation, operations, and maintenance to adapt to a changing world influenced by infrastructure systems and the human activities they support (Grabowski *et al*., 2017). On a social level, the infrastructure managers must recognise that infrastructure systems are inherently intertwined with politics, and as a result, it is crucial to overcome isolated decision-making processes that focus solely on individual systems (Grabowski *et al*., 2017; du Plessis, 2023). Achieving this understanding entails embracing the additional intellectual challenge of comprehending how social perception and values shape the criteria for desirable infrastructure development. In essence, current infrastructure thinking must address both the physical and social dimensions to ensure effective and sustainable progress (Osawe, Grilli and Curtis, 2023). While applying the fundamentals of infrastructure planning, Manny (2023) concluded that socio-technical perspective on social actors and infrastructure elements could help to improve policy design and implementation aiming to achieve more sustainable cities.

In the past, river water environment was neglected in water resources planning, but later it was acknowledged as a limiting factor, and now it is recognised as a significant objective in water resources management (Nohara, Gourbesville and Ma, 2018). The nature of catchment management and environmental decision making necessitates a participatory approach in which regulators, stakeholders, and researchers jointly explore management choices through an iterative process on a common platform (Sun, Miranda and Xu, 2015). A strong decision support system (DSS), according to Silva *et al*. (2023), lowers the complexity of decision-making and enables decision makers to create operational strategies that are economic and address limitations in watershed management and environmental decision-making.

The management of water resources involves the use of diverse and often conflicting criteria, which may include environmental, economic, social, and technical considerations. Multi-criteria decision support (MCDA) tools have emerged as valuable tools for aiding decision-making in water quality management. MCDA techniques enable decision-makers to explore the trade-offs and synergies between multiple criteria and identify optimal solutions to complex problems. Decision-support tools are required to facilitate conflict resolution and consensus reaching in decision making, where fair trade-offs can be made by different stakeholders.

Catchment management done from the top down is unpopular with landowners and rural communities because it generally provides results and recommendations that lack practical significance and broad community support (Prato and Herath, 2007). In many parts of the world, catchment alliances have been formed to reduce the cumulative ecological consequences of land use and water resource management. These alliances include stakeholders such as community members, government agencies, environmental groups, private industry, academic structures, and others. The foundation of these alliances is that assessments of sustainable resource management and the design of policies to alleviate unsustainable resource management should occur at the local level (Prato and Herath, 2007).

Catchment alliances can use multi-criteria decision analysis (MCDA) to evaluate resource management from a social, economic, technical, and ecological perspective (Stewart *et al*., 1997; Razmak and Aouni, 2014; Aceves and Fuamba, 2016). If a catchment alliance or environmental authority determines that resource management is not sustainable, then it is appropriate for them to evaluate alternative policies that encourage sustainable resource management (Prato and Herath, 2007; Liquete *et al*., 2016). Various methods utilised in this context have distinct theoretical underpinnings, including value functions, optimisation algorithms, goal aspiration, outranking, or a blend of these approaches (Marttunen *et al*., 2022). These methods employ diverse techniques for scoring, weighting, and combining data (Marttunen *et al*., 2022).

The MCDA refers to the process of ranking and choosing between alternatives that uses multiple criteria or objectives to decide (Razmak and Aouni, 2014). It is a method for systematically comparing the advantages and disadvantages of different alternatives in support of decision making (Esmail and Geneletti, 2018). Furthermore, some applications allocate budgets or other scarce resources among alternatives, to maximise efficiency (Prato and Herath, 2007). To make the decision-making process easier, computerised systems have been developed to help with applications and the use of the MCDA tools (Belaid and Razmak, 2013), these are termed multi-criteria decision support systems (MCDSS).

Typically, MCDA process consists of the following phases (Saarikoski *et al*., 2016): (i) problem formulation including the identification of the objectives, criteria and measures for them, construction of a decision hierarchy based on all these as well as generation of alternatives, (ii) evaluation of the impacts of the alternatives and creation of a consequence table, (iii) calculation of alternatives' total priorities using tools like Excel, (iv) evaluation of the outcomes, including sensitivity analysis and suggestions. While all of the methods basically adhere to the general steps outlined above, they each use distinct ideas and techniques for gathering and organising information as well as various algorithms for merging it (Saarikoski *et al*., 2016).

Numerous cognitive and motivational biases, including framing bias, anchoring bias, splitting bias, proxy attribute bias, and strategic prejudice, might affect the methodologies used to elicit preferences in both MCDA and stated preference studies (Saarikoski *et al*., 2016). Prior to this study, Montibeller and von Winterfeldt (2015), had discussed these biases, their consequences for multi-criteria decision analysis, and several debiasing strategies for mitigating their impacts. This work is summarised in Table 2.2. The process of constructing models and calculating parameters for MCDA is based on the expert and decision-maker's judgements. However, these judgements are prone to biases that can have a negative impact on the analysis' quality. These biases are the result of defective cognitive processes as well as psychological incentives that favour preferred analysis conclusions (Montibeller and von Winterfeldt, 2015).

Marttunen, Belton and Lienert (2018) investigated whether the biases that are introduced at an early stage of MCDA when building objectives hierarchies and their effect on the weights are observed in real life. Their investigation was based on prior literature. They found that the hierarchy structure and content can substantially influence weight distributions. They found no evidence for the equalising bias. On the other hand, Rezaei, Arab and Mehregan (2022), assessed five Multi-criteria decision making (MCDM) weighting methods, under two structuring formats, hierarchical and non-hierarchical. They concluded that the hierarchical problem structuring leads to a reduction in the equalising bias in all the five methods and that such a reduction significantly varies among the methods.

*Table 2.2 Biases and debiasing methods after Montibeller and von Winterfeldt, 2015*

| **Phase of MCDA** | **Possible bias(es)** | **How to address bias(es)** |
|---|---|---|
| generation of alternatives and objectives | i) Omission bias - Omission of important items.<br>ii) Myopic problem representation bias - generation of incomplete problem descriptions due to over-simplified mental models.<br>iii) Anchoring bias - anchoring alternatives on an initial set.<br>iv) Availability bias - when existence of one alternative prevents the generation of other alternatives.<br>v) Desirability bias - exclusion of alternatives that compete with the preferred one. | i) objective-based strategies, such as presenting one objective at a time and asking for high-value attaining alternatives and designing solutions that perform well on high-weighted objectives.<br>ii) state-based strategies, such as presenting various future states one at a time and soliciting high-value achieving alternatives in each future state.<br>iii) alternative-based strategies including picturing an ideal option and designing alternatives from it, as well as generating new possibilities from current ones. |
| defining attributes | i) contraction bias (underestimating large sizes/differences and overestimating small/size differences).<br>ii) logarithmic response bias (using step-changes in the number of digits used in the response, which fit a log scale).<br>iii) range equalising bias (using most of the range of response whatever is the size of the range of the stimuli).<br>iv) centering bias (producing a symetric distribution of responses centred on the midpoint of the range of stimuli); and<br>v) equal frequency bias (using equally all parts of the response scale). | using natural units as far as possible, where this is not possible, constructed attributes should be used with special attention to scale and its end point. |
| elicitation of utility or value functions over attribute levels | i) anchoring bias,<br>ii) gain-loss bias,<br>iii) certainty effect (people prefer what is guaranteed to gambles with similar expected utilities and will discount the utility of guaranteed things dramatically when there is even a slight degree of uncertainty),<br>iv) desirability bias might distort the utility function in a direction that favours a preferred alternative | This can involve using value functions as approximations of utility functions, deriving utility functions from value functions, or employing standardised shapes for utility functions (for example., linear or exponential).<br>When eliciting utility functions using gambles, analysts should avoid including sure things to prevent the certainty effect from biasing the elicitation process. |
| eliciting weights | i) splitting bias – where objectives that are defined in more detail receive a larger portion of the weights than objectives that are defined in less detail.<br>ii) equalising bias - where decision makers tend to allocate similar weights to all objectives.<br>iii) proxy bias – where objectives are over-weighted when measured by a proxy attribute instead of by an attribute that directly measures a fundamental objective. | i) to avoid elicitation procedures that ask for direct assessments of importance.<br>ii) avoid excessive detail in some objectives and little detail in others.<br>iii) set up the lower and upper anchors of each attribute in a way that they indeed allow similar weights for all objectives.<br>iv) use ranking and ratio weighting methods, coupled with hierarchical |

| Phase of MCDA | Possible bias(es) | How to address bias(es) |
|---|---|---|
| | iv) range insensitivity bias - where weights are insensitive to the range of attribute values.<br>v) desirability bias may lead to the over/under weighting of attributes to favour a preferred alternative. | weighting, which generally produce steeper weights.<br>iv) use of either natural or constructed attributes for fundamental objectives. |

Decision support systems (DSS) encompass a range of formats, including spreadsheet tools, visualisations, and conceptual frameworks, with varying presentation styles tailored to different environments and end-users, effectively promoting group collaboration, multi-stakeholder learning, and catering to specific user needs (Wardropper, 2022). Building consensus among stakeholders or the general public depends heavily on the transparency of the planning and management procedures (Nohara, Gourbesville and Ma, 2018). To enhance the usefulness and usability of DSS, it is important to consider the influence of environmental, sociocultural, and institutional factors on decision-making processes, and one approach to achieve this is by integrating knowledge and perspectives from social sciences into the design and dissemination of DSS (Wardropper, 2022). DSS have evolved from being only engineering tools to becoming systems that offer stakeholder engagement frameworks to direct, inform, and support decision making in a transparent and more sustainable manner (Serrat-Capdevila, Valdes and Gupta, 2011).

In a study by Giupponi and Sgobbi (2013), it was shown that the use of different techniques may indeed lead to contrasting results, even when applied to the same datasets. Wątróbski *et al*. (2019) argue that one of the most important steps in the decision-making process is choosing an MCDA approach that can solve a particular choice problem. In their review, Wątróbski *et al*. (2019) enlist 56 MCDA methods and consequently propose a selection framework that can be found within a web platform available for public use at www.mcda.it. One of the major strengths of MCDA is its ability to support multi-stakeholder processes and bring subjective views into the evaluation (Marttunen *et al*., 2022). MCDA provides a systematic, transparent approach that enhances objectivity and generates results which can be trusted with a reasonable satisfaction (Hassan *et al*., 2015).

The following summary of criticisms of the MCDA techniques is given in Hassan *et al*. (2015):

Aggregation algorithms: Various MCDA techniques produce different results when employed for the same multi-criteria problem. Choosing the suitable MCDA method from a comprehensive range of options is frequently challenging and could potentially influence the outcome of the decision-making procedure.

Compensatory methods: Complete aggregation methods of the additive nature enable balancing between achieving high performance on one criterion while accepting lower performance on other criteria. However, this type of aggregation often results in the loss of significant information. Multi-criteria problems are considered mathematically ambiguous because an action "x" can be superior to action "y" based on one criterion but inferior based on another. This ambiguity arises due to the inherent challenge of fully axiomatizing multi-criteria decision theory.

Elicitation process: The way subjective information (weights and preference thresholds) is elicited is not trivial and is likely to influence the results.

Incomparable options: The primary objective of MCDA is to minimise incomparability, which often leads to the conversion of MCDA problems into single-criterion problems, where a fully optimal solution exists. However, this transformation alters the original structure of the decision problem, which is not realistic. Moreover, the aggregation of data in MCDA frequently reduces alternatives to a single abstract value, resulting in the loss of valuable information. From a layperson's perspective, it may be easier to comprehend the cost of an alternative in monetary terms rather than an abstract value that simply indicates whether option A is better or worse than option B by a value such as 0.45.

Scaling effects: Some MCDA methods derive conclusions based on scales in which evaluations are expressed which is unacceptable.

Problem structuring: Results could be manipulated by omission or addition of some relevant criteria or options. MCDA methods have been reported to suffer from rank reversals by introduction of new options.

Additional required information: Depending on how much additional information is required by the different MCDA methods, “black box” effects are likely to occur thus compromising the ability of the decision-maker to clearly follow the decision process and evaluate the results.

Uncertainty: The presentation of results in MCDA is often misleading as they are often reported with two decimal places, creating a false impression of accuracy. This disregard for the uncertainties in the input data and their propagation errors in the model can be problematic.

Furthermore, the decision-making process itself contains inherent uncertainty, making it challenging to quantify and represent the performance of most options with a single value.

### c. Applications of MCDA in water resource management

The use of MCDA in water resource management is illustrated in several studies as shown in Table 2.3. The reviewed literature showcased a diverse range of applications for multi-criteria decision-support tools in water quality management. These tools have been employed in various contexts, including ecosystem services (McInnes *et al*., 2016), weights for evaluating alternatives in catchment management, prioritising vulnerable areas within catchments, urban water management, stakeholder perceptions.

In two crucial domains, the evidence generally demonstrates that MCDA performs better than cost-benefit analysis and monetary valuation methodologies. Firstly, MCDA can consider several aspects of well-being, including ecological, economic, cultural, and moral facets of a management or policy issue. This larger viewpoint enables a more thorough assessment of the effects and advantages linked to various courses of action. Secondly, by offering a framework that enables a full analysis of the benefits and drawbacks of competing approaches, MCDA promotes open and transparent public discourse. This includes examining how profits and losses would be distributed among ecosystem service recipients. MCDA provides improved skills in capturing multidimensional well-being and encouraging inclusive debates on the relative benefits of various acts, in general.

Marttunen *et al*. (2022) advice on several aspects that should be considered while using MCDA methods where the environment is considered as below:

1. Method selection: Evaluate the case's individual requirements and select MCDA methods accordingly.

2. Stakeholder involvement: Determine the level of stakeholder involvement based on the analysis objectives. Adjust the intensity of the stakeholder involvement process as needed.

3. Criteria selection: When building the MCDA criteria for water management cases, ensure that the criteria framework includes environmental considerations. It is essential that stakeholders carefully analyse the weighting method they choose for a practical watershed management challenge.

4. Preferences of stakeholders: Incorporate and transparently account for stakeholder and expert views.

5. Visualisation: Display the results in a visually appealing manner, emphasising the significance of subjectivity in the decision-making process.

6. Understandability: Create a clear, comprehensible, and theoretically sound technique for eliciting weights. Use of too complex or meaningless questions may result in biased outcomes.

*Table 2.3 Studies depicting use of MCDA in water resource management*

| **Reference** | **Study Aim(s)** | **Study area** | **MCDA Tool/s** | **Lessons learned/Findings** |
|---|---|---|---|---|
| Saarikoski *et al.* (2016) | Comparing Multi-Criteria Decision Analysis and Cost-Benefit Analysis frameworks for integrated valuation of ecosystem services | Finland and Portugal | Analytical Hierarchy Process (AHP) | MCDA does in general perform better than cost-benefit analysis and associated monetary valuation techniques in: 1. the ability to account for multiple dimensions of well-being, including ecological and economic as well as cultural and moral aspects of a policy or management problem; and<br>2. to facilitate open and transparent public debate on the pros and cons of alternative courses of action, including the distribution of gains and losses across beneficiaries of ecosystem services |
| Zardari *et al*. (2014) | developing a comprehensive and logical-based methodology for eliciting relative importance (weights) of targets for managing any watershed | Malaysia | Simple Multi-Attribute Rating Technique (SMART), swing weighting method, Simple Multi-Attribute Rating Technique Exploiting Ranks (SMARTER) | the choice of weighting method for solving a real-world watershed management problem is crucial and much attention of the researchers and decision makers is required before proceeding with next steps in the problem-solving framework |
| Bray (2015) | to contribute towards the development of a more participative method by prototyping a new MCDM technique | Southwest Scotland | SMARTEST | use of SMARTEST was effective in helping stakeholders gain insights into the decision-making process of themselves and each other |
| Jaiswal *et al*. (2015) | application of Saaty's AHP based Multi-criteria decision analysis (MCDA) in prioritising of vulnerable area of watershed | India | AHP | successfully applied |
| McInnes *et al.* (2016) | a payment for ecosystem services (PES) project to evaluate options for delivering good eco- logical status | Gloucestershire, UK | Generic MCDA | MCDA is a useful tool for developing transparent, iterative and participatory water management solutions within the context of a PES scheme |
| Lück and Nyga (2018) | to contribute the experiences of divergent combinations of participatory techniques for MCDA processes in the field of urban water management | Germany | AHP | improvement could be achieved by setting up a structured feedback process for stakeholders for evaluating their perceptions |
| Marttunen *et al.* (2022) | review of how the ecological services concept (ES) has been used in water management projects together with MCDA and to examine the experiences gained and make recommendations to overcome any identified challenges | Varied | Multi-Attribute Value Theory (MAVT)/ Multi-Attribute Utility Theory (MAUT) (10 cases), another generic MCDA method (7 cases), and AHP (4 cases). Other applied method was Elimination Et Choix Traduisant la Réalité (ELECTRE) (1 case) | Choice of the methods: When selecting what MCDA methods to use, carefully consider the needs of the case and make the selection of ES categories based on this.<br>Stakeholder involvement: Plan the intensity of the stakeholder involvement process according to the purpose of the analysis.<br>Criteria selection: Use the ES concept at least as a checklist when creating the MCDA hierarchy in water management cases.<br>Stakeholders' preferences: Utilise and make transparent the use of stakeholder and expert preferences in the evaluation.<br>Visualisation: Visualise the results and present a wide spectrum of preferences to highlight the importance of subjectivity.<br>Understandability: Design a concrete, understandable, and theoretically valid weight elicitation procedure to avoid cognitively too demanding or meaningless questions and biased results.<br>Evaluation of the process. Carefully report the process, and collect, analyse, and report the experiences from participants involved in the evaluation. Reflection on experiences can also benefit other researchers and practitioners in designing how to apply the method and improve MCDA practices in the long term. |
| Ndeketeya and Dundu (2022) | GIS-based MCDA was run to cater for socio-economic factors influencing rainwater harvesting | Johannesburg, South Africa | AHP | The methodology and models used can be adapted to different locations and scales, factoring in location-specific variables. |

7. Evaluation of the process: Thoroughly document the entire process, including participant experiences. Collect and analyse stakeholder comments and reflect on the lessons learned. This evaluation can help to enhance MCDA practices in the future and benefit other researchers and practitioners.

### 2.1.4 Conclusion

This literature review highlights the significance of multi-criteria decision-support tools for improving water quality in the context of engineering infrastructure and catchment management. The reviewed studies demonstrate the potential of these tools in aiding decision-makers by providing a systematic and comprehensive approach for addressing water quality issues. By integrating technical, economic, environmental, and social criteria, these systems provide a holistic perspective for informed decision-making. The integration of water quality modelling, quantitative chemical/microbial risk assessment, and stakeholder engagement to prioritise catchment areas with significant pollution issues and develop pollution mitigation strategies has received limited research attention thus far. Therefore, further research and development are required to overcome the challenges associated with data availability and quality, data integration, stakeholder engagement, improving of socio-economic aspects, enhancing the usability and accessibility of decision-support tools and uncertainty modelling. By advancing the methodologies and enhancing the usability of decision-support tools, water quality management efforts can be significantly improved, leading to sustainable and resilient water systems.

## 2.2 Study Area Background

This research aimed to develop a multi-criteria decision support framework that enables decision-makers to address the impact of pollution in a catchment area while considering multiple decision constraints. To bring this view to life, uMsunduzi catchment was used as a study area. The 875 km$^2$ uMsunduzi catchment shown in Figure 2.1 is in the KwaZulu-Natal province of South Africa, encompassing parts of Msunduzi Municipality 54 600 population in 2023) and Mkhambathini Municipality (63 142 population in 2023). The uMsunduzi River, a 115 km watercourse length, passes through Pietermaritzburg (29°37’S 30°23’E). The uMsunduzi River is a major tributary of the uMngeni River and has been reported to contribute pollution to the uMngeni River (Gemmell and Schmidt, 2013). It flows through rural and urban settlements of the uMsunduzi catchment.

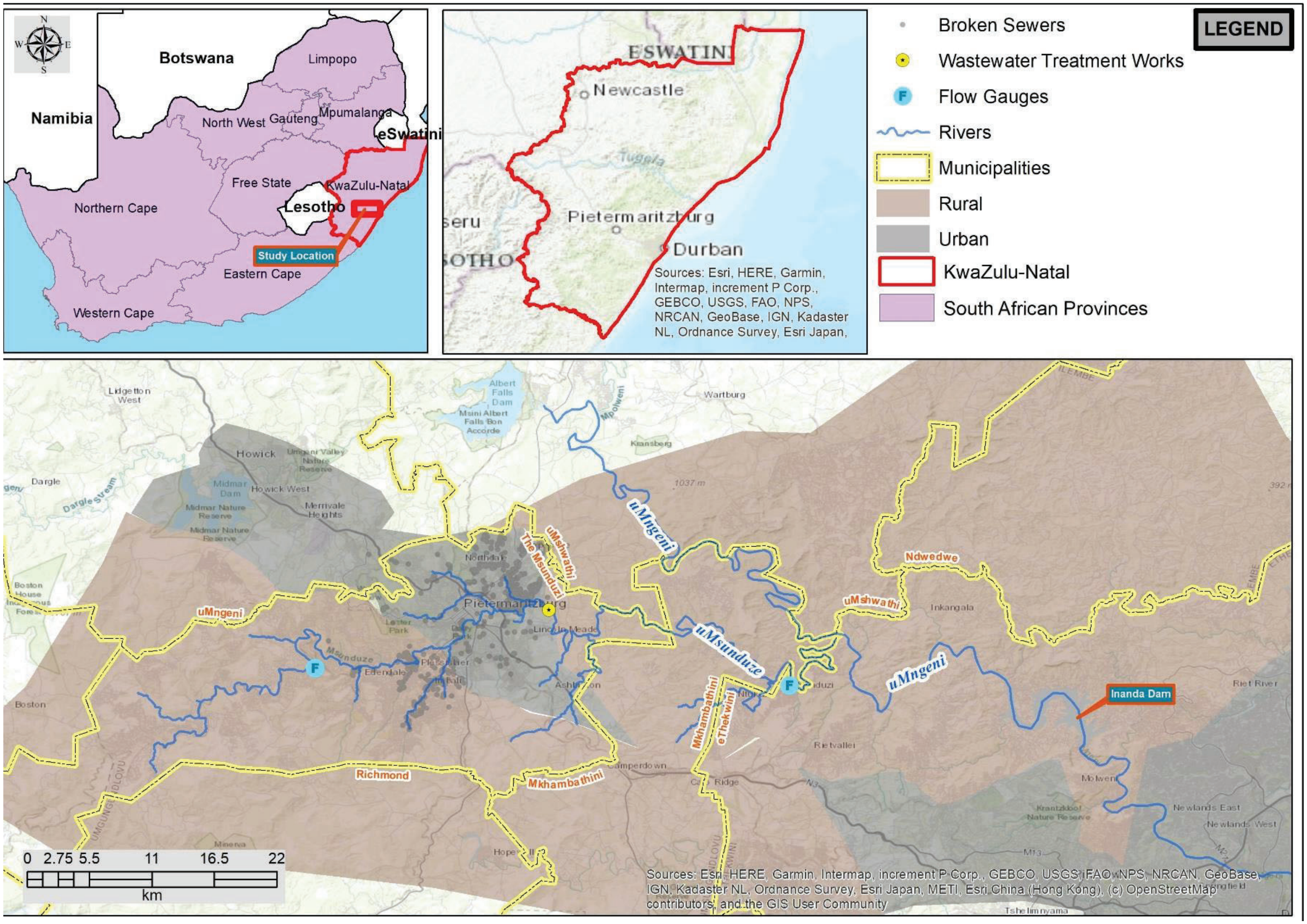
The topography of the uMsunduzi catchment is mountainous and undulating with numerous hills, as well as lowlands, wetlands, marshes, and relatively flatter areas. The mean altitude above sea level (masl) in the catchment varies from 285 m to 1520 m. The average slope of the catchment varies from 2.5 to 33%. The city of Pietermaritzburg has a dry-winter humid subtropical climate (Köppen climate classification: Cwa), with distinct dry and wet seasons. Average annual temperature varies between 16.3°C and 17.9°C and the average rainfall varies between 748 mm and 1017 mm per annum. The dominant soil types in the study area, according to the Food and Agricultural Organisation (FAO, 2015) are Hutton, Griffin, Mispah, and Glenrosa. The main land use types are grasslands. At least six distinct geological soil groups including: Adelaide (mudstone and sandstone), Dwyka (dyamictite, tillite, shale, mudstone, and sandstone), Ecca (sandstone, and carbonaceous shale), Natal (quartzitic sandstone, arkose, and shale), and Thukela and Maphumulo (amphibolite, gneiss, schist and granulite) were identified in the catchment (Hughes *et al*., 2013).

The river is characterised by three distinct units within its catchment: the upper catchment rural area, the urban area in the middle (including Pietermaritzburg), and the lower catchment rural area. The rural areas are primarily engaged in subsistence farming and rural developments, while the urban area consists of a mix of formal city, residential, industrial suburbs, and informal settlements. At Camps Drift, the river enters a canalised reach before flowing through the city in a narrow channel and merging with small tributaries. It also receives treated effluent from the Darvill wastewater treatment plant that services the urban areas of uMsunduzi catchment and has a capacity of 65ML/d (Umgeni Water, 2022). In the lower catchment rural area, the uMsunduzi River confluences with the uMngeni River, eventually flowing into Inanda Dam approximately 15 km downstream. Inanda Dam serves the eThekwini Metropolitan area with drinking water. In the rural areas, the river is typically used for domestic water needs, recreational purposes, and subsistence farming. In urban areas, it is typically used for canoeing and swimming.

In the uMsunduzi River, point and nonpoint source pollution has been reported (Gericke, 2003) in terms of chemical pollution (Shozi, 2015; Agunbiade and Moodley, 2016; Moodley, Birungi and Ndungu, 2016; Adeyinka *et al*., 2019) and microbial pollution (Gemmell and Schmidt, 2013). Table 2.4 shows the main activities within the uMsunduzi catchment, possible pollutants emerging from these activities, and the routes through which humans are exposed to these pollutants. In relation to chemical pollutants, they can be summed up as nitrates, phosphates, pharmaceuticals, and pesticides emanating from agricultural activities, industrial activities, and municipal waste. The microbial pollutants can be bacteria, viruses, protozoa, and others.

*Table 2.4 Configuration of the uMsunduzi catchment and characterisation in terms of activities, possible pollutants, and exposure routes*

| **Location within catchment** | **General Landuse** | **Possible Pollutants** | **Exposure Routes** |
|---|---|---|---|
| Henley Dam (upper catchment rural) Elandskop | Uncontrolled stock grazing and watering | Microbial, Nitrates, Pharmaceutics | Swimming; drinking (Adults and Children) |
| | Laundry (women/children) | Phosphates from detergents | |
| | Subsistence farming (community gardens) | Pesticides, Nitrates | |
| | Small plantations | Pesticides, Nitrates | |
| | Rural developments (houses) | Microbial, Phosphates | |
| Edendale/Imbali/Camps Drift (Dense urban areas/township/informal settlements) | Illegal dumping near the river (domestic refuse including baby diapers) | Microbial, Phosphates, Pharmaceutics | Canoeing Practice, swimming |
| | Edendale Hospital | Medical waste/pharmaceutics | |
| | Caluza Clinic (dumping next to the river and drainpipe on the river side) | Microbial, Phosphates, Pharmaceutics | |
| | Animals grazing next to the river and drinking from the river. | Microbial, Nitrates | |
| | Industries (Hulamin, Oil Refineries, Breweries) | Oil, Nitrates, Heavy Metals | |
| | Subsistence farming | Pesticides, Nitrates | |
| | Broken sewers | Microbial, Nitrates, Pharmaceutics | |
| | Canoeing training (Canoe Club) | | |
| | Slangspruit tributary | Microbial, Nitrates, Pharmaceutics | |
| | Informal settlements (Sanitation) | Microbial, Nitrates, Pharmaceutics | |
| | City streets | Microbial, Nitrates, Pharmaceutics | |
| | Illegal dumping | Microbial, Nitrates, Pharmaceutics, Oil | |
| | New England Landfill site | Microbial, Nitrates, Pharmaceutics | |
| | Four hospitals (Mediclinic, St. Anne's, Royal, East-Boom Clinic) | Microbial, Pharmaceutics | |
| Darvill/Baynespruit/Suburbs/Industries/ Confluence with Umgeni | Broken sewers | Microbial, Nitrates, Pharmaceutics | Swimming |
| | Informal Settlements | Microbial, Nitrates, Pharmaceutics | |
| | Wastewater Works | Microbial, Nitrates, Pharmaceutics | |
| | Northdale Hospital | Microbial, Pharmaceutics | |
| | Industries along Bayenspruit tributary (Willowton Oil, Aluminium) | Microbial, Nitrates, Pharmaceutics, Oil | |
| | Subsistence farming | Pesticides, Nitrates, microbial | |
| Valley of a Thousand Hills, Rural, uMngeni-uMsunduzi River confluence | Subsistence farming including stock | Microbial, Nitrates, Pharmaceutics | Swimming, Drinking |
| | KwaXimba Clinic | Pharmaceutics | |



Sources: Esri, HERE, Garmin, Intermap, increment P Corp., GEBCO, USGS, FAO, NPS, NRCAN, GeoBase, IGN, Kadaster NL, Ordnance Survey, Esri Japan,

Sources: Esri, HERE, Garmin, Intermap, increment P Corp., GEBCO, USGS, FAO, NPS, NRCAN, GeoBase, IGN, Kadaster NL, Ordnance Survey, Esri Japan, METI, Esri China (Hong Kong), (c) OpenStreetMap contributors, and the GIS User Community

*Figure 2.1 Location of uMsunduzi catchment in South Africa. Please click this link  for a full layout*

# Chapter 3 Microbial Water Quality Modelling and Quantitative Microbial Risk Assessment

The contents of this chapter are published in the Journal of Water and Health with the following reference: Ngubane, Z., Bergion, V., Dzwairo, B., Troell, K., Amoah, I. D., Stenström, T. A. and Sokolova, E. (2022) 'Water quality modelling and quantitative microbial risk assessment for uMsunduzi River in South Africa', Journal of Water and Health, 20(4). doi: 10.2166/wh.2022.266.

## 3.1 Introduction

Surface water pollution is strongly associated with land and water uses within a catchment. These include impact from residences with insufficient sanitation infrastructure in the vicinity of streams, poorly operated and maintained sanitation infrastructure, dense settlements in areas prone to surface runoff, and intensive livestock farming (Luyt *et al*., 2012). Low-income countries continue to have a high incidence of diarrhoea among children under five years of age, which is one of the leading causes of death, ill-health, and disability (Naghavi *et al*., 2015). In such communities with poor service and infrastructure, the spread of diarrhoeal diseases is exacerbated by inadequate and unsafe water supplies and inadequate sanitation and hygiene (Araya *et al*., 2016). Shirley, Moonah and Kotloff (2012) found that most attributable cases of moderate-to-severe diarrhoea in the youngest children were due to four pathogens: rotavirus, *Cryptosporidium*, enterotoxigenic *E. coli* producing heat-stable toxin (ST-ETEC; with or without co-expression of heat-labile enterotoxin), and *Shigella*. Surface water pollution, driven by land and water use in catchments, heightens health risks, especially in low-income nations with inadequate sanitation infrastructure and unsafe water supplies, highlighting the critical necessity for enhanced water and sanitation services to protect public health.

*Cryptosporidium* has been identified in South Africa as a causative agent of diarrhoeal diseases in immunocompromised, HIV-positive children and adults (Ojuromi and Ashafa, 2018). In the study performed in KwaZulu-Natal by Jarmey-Swan, Bailey and Howgrave-Graham (2001), *Cryptosporidium* was found to be most prevalent (39.3%) (1100/2800) in the <1-year age group. The quality of South Africa's water resources is linked to the inadequate wastewater infrastructure and the subsequent contribution of untreated or poorly treated wastewater discharges to the already deteriorating water sources (Mema, 2010; Osuolale and Okoh, 2017; Ojuromi and Ashafa, 2018; Sekwadi *et al*., 2019). The lack of infrastructure in South Africa forces rural and peri-urban households to supplement their drinking water intake with water of poor microbial quality (Luyt *et al*., 2012).

In response to water-related challenges, hydrological modelling has been developed to analyse, understand and explore solutions for sustainable water management, to the decision makers and operational water managers (Baffaut and Benson, 2009; Mander *et al*., 2017; Abdullah *et al*., 2019). Using models to understand and manage water quality can make it easier to promote better management. By generating more data over time and space, they can supplement limited observational data and allow for scenarios to explore the impact of different management measures on water quality in a watershed. Catchment-scale water quality models have become a key tool for understanding, evaluating, and predicting the adverse effects of pollution on river water quality. Among others, models that provide refined simulation for water quality include Agricultural Nonpoint Source pollution model (AGNPS/AnnAGNPS), Soil and Water Assessment Tool (SWAT), Stormwater Management Model (SWMM), and Hydrologic Simulation Program Fortran (HSPF). Of these, HSPF and SWAT are highly recommended to users (Yuan, Sinshaw and Forshay, 2020).

In SWAT, there is a microbial sub-model that deals with the fate and transport of both persistent and less persistent microorganisms (Abdullah *et al*., 2019). SWAT can, therefore, simulate microbial transport in catchments as well as identify high-risk areas and estimate peak loads of microorganisms such as *E. coli* and *Cryptosporidium* (Bergion *et al*., 2017; Zhang *et al*., 2019). Output data from a SWAT model was successfully used as input data to conduct quantitative microbial risk assessment (QMRA) by Bergion *et al*. (2020). QMRA assesses the probability of infection due to exposure to microorganisms, based on the dose a person ingests and dose-response models that provide the human response to specific pathogens (Haas, Rose and Gerba, 2014). The risk of being infected by microbial pathogens is dependent on the level of pollution of the water and the amount of polluted water consumed (Müller, Ehlers and Grabow, 2001).

The aim of this study was to utilise SWAT as a water quality modelling tool to map out point and non-point source pollution in the uMsunduzi catchment in KwaZulu-Natal, South Africa, to identify high-risk areas. Concentrations of *Cryptosporidium* and *E. coli* were studied. The concentration outputs from SWAT were used to perform a QMRA to better understand the health implications on communities that use polluted waters for drinking and non-competitive recreation purposes.

## 3.2 Materials and Methods

Figure 3.1 shows the subbasins as demarcated in SWAT, the river flow monitoring gauge positions (owned by Department of Water and Sanitation), water quality monitoring stations (monitored by Umgeni Water), broken sewers (reported by the Duzi-Umgeni Conservation Trust), weather station (monitored by the Weather Bureau of South Africa) within uMsunduzi catchment. This figure also shows the demarcation of the catchment in terms of settlement types.

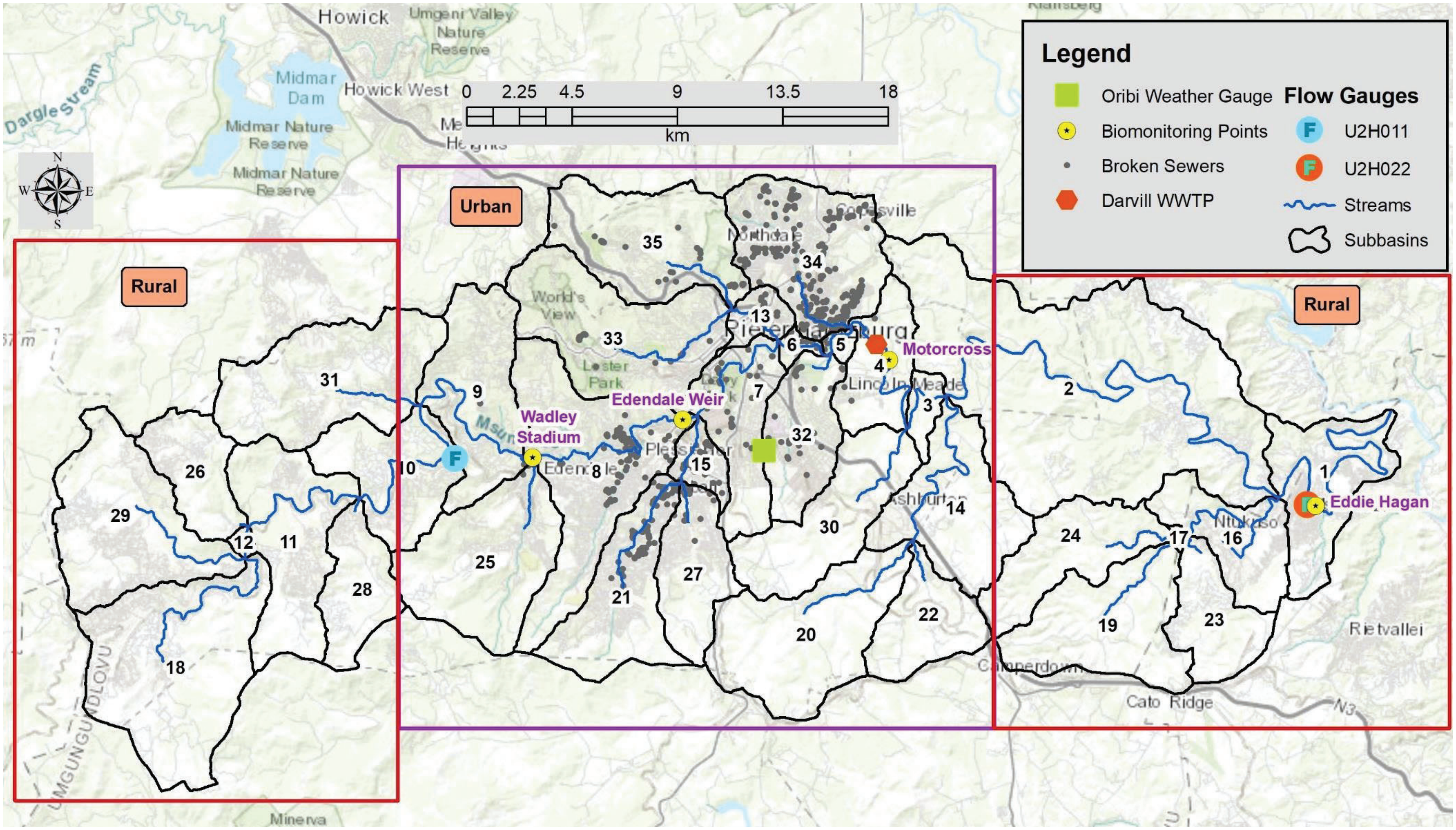


*Figure 3.1 River flow monitoring gauge positions, water quality monitoring stations, broken sewers, weather station within uMsunduzi catchment, as well as subbasins as delineated by SWAT. Please click here to see in full layout*

### 3.2.1 Water Quality Model

QSWAT3 v1.1 for QGIS3, was used to setup the hydrological model for the period 1999-2018, for this study. The years 1999-2003 were reserved for the warm-up period prior to the defined calibration and validation stages to allow the stable performance of the model. Calibration and validation of the model using river flow data were performed for the periods 2004-2011 and 2012-2018, respectively. The input data included topographical, geographical, and geological information as well as meteorological data as indicated in Table 3.1. The uMsunduzi catchment was divided into 35 subbasins. The water quality output of this model was used in the QMRA as discussed in the subsequent sections.

*Table 3.1 Data for the hydrological and water quality model of uMsunduzi River*

| **Data** | **File type** | **Resolution** | **Source** |
|---|---|---|---|
| Digital Elevation Model | Raster | 20 x 20 m | National Geo-spatial Information (NGI) |
| Landuse/landcover 2018 | Raster | 1700 x 1700 m | South African National Biodiversity Institute (SANBI) |
| Soil type | Raster | 250 x 250 m | Food and Agriculture Organization (FAO) |
| Meteorological data | Text | One station, Daily[a] | South African Weather Services (SAWS) |
| River flow data | Text | Two stations, Daily[a] | Department of Water Affairs (DWA) |
| Water quality monitoring data | Text | Four Points, Weekly[a] | Umgeni Water |

Kindly refer to Figure 1 for location within the catchment.

The hydrological parameter optimisation was performed manually. The Nash-Sutcliffe efficiency coefficient (NSE), the coefficient of determination ($R^2$), percent bias (PBIAS), and ratio of the root mean squared error to the standard deviation of measured data (RSR) were used for the evaluation of model performance. These are represented by Equation 3.1, Equation 3.2, and Equation 3.3, respectively.

*Equation 3.1*

$$NSE = 1 - \frac{\sum_i^n (Q_i^m - Q_i^s)^2}{\sum_i^n (Q_i^m - \bar{Q})^2}$$

*Equation 3.2*

$$PBIAS = \left[\frac{\sum_i^n (Q_i^m - Q_i^s) \times 100}{\sum_i^n Q_i^m}\right]$$

*Equation 3.3*

$$RSR = \frac{\sqrt{\sum_i^n (Q_i^m - Q_i^s)^2}}{\sqrt{\sum_i^n (Q_i^m - \bar{Q})^2}}$$

where: Q is discharge, the $\overline{Q}$ is the mean discharge, and the indices m and s stand for measured and simulated, respectively and i/n are the ith and nth measured or simulated data.

Sadeghi and Arnold (2002) developed a SWAT bacterial module for predicting microorganism loads and concentrations in rivers. The SWAT microbial sub-model considers the fate and net transport of microorganisms derived from defined pollution sources. The two microorganisms considered in this study were *E. coli* and *Cryptosporidium*, chosen to represent bacteria and protozoan parasites respectively. Bacteria may be less persistent in the environment due to inactivation by environmental factors compared to the protozoan parasites, whose (oo)cysts are more resistant to environmental conditions (Sadeghi and Arnold, 2002). The input parameter values for both microorganisms are shown in Table 3.2.

The uMsunduzi River is impacted by point and non-point source pollution emanating from rural settlements, informal settlements, townships, formal urban and industrial areas, and the Darvill WWTP (Dabrowski *et al*., 2013). The main faecal sources in the uMsunduzi catchment can be summarised as: Darvill WWTP, broken sewers in the urban area, dry toilets (non-flushing), and faecal droppings from grazing livestock.

To compare the model output with the measured concentrations, four subbasin output points that closely coincide with water quality monitoring points by the Msunduzi Local Municipality and Umgeni Water along the river were considered in the model. These were: uMsunduzi downstream Henley Dam - rural area (subbasin 10); urban areas (subbasin 8); major stream, city, informal settlements, industrial areas, WWTP, and waste dump site (subbasin 4); and rural areas and confluence with the uMngeni River (subbasin 1).

*Table 3.2 SWAT parameter values for E. coli and Cryptosporidium*

| **Microorganism** | **SWAT abbreviation** | **Unit** | **Value** | **Reference** |
|---|---|---|---|---|
| Both | BACTFDDB (Partition coefficient) | Fraction 0≤1 | 0.9 | (Coffey *et al.*, 2013) |
| Both | BACTKDQ (Bacteria soil partitioning coefficient) | Constant | 175 | (Arnold *et al.*, 2012) |
| Both | FRT_SURFACE (Fraction of fertiliser applied to 10 mm of soils) | Fraction 0≤1 | 0.5 | (Tang *et al.*, 2011) |
| *E. coli* | WDLPQ (Die-off factor for less persistent bacteria in soil solution at 20°C) | 1/day | 0.092 | (Westrell, 2004) |
| *E. coli* | WDLPRCH (Die-off factor for less persistent bacteria in streams at 20°C) | 1/day | 0.18 | (Westrell, 2004) |
| *E. coli* | WDLPS (Die-off factor for less persistent bacteria adsorbed to soil particles at 20°C) | 1/day | 0.023 | (Bougeard *et al.*, 2011) |
| *E. coli* | WDLPF (Die-off factor for less persistent bacteria on foliage at 20°C) | 1/day | 0.016 | (Bougeard *et al.*, 2011) |
| *E. coli* | WOF_LP (Fraction of less persistent organisms washed off in rainfall events) | Fraction 0≤1 | 0.5 | (Bougeard *et al.*, 2011) |
| *Cryptosporidium* | WDPQ (Die-off factor for persistent bacteria in soil solution at 20°C) | 1/day | 0.005 | (Westrell, 2004) |
| *Cryptosporidium* | WDPRCH (Die-off factor for persistent bacteria in streams at 20°C) | 1/day | 0.032 | (Westrell, 2004) |
| *Cryptosporidium* | WDPS (Die-off factor for persistent bacteria adsorbed to soil particles at 20°C) | 1/day | 0.003 | (Coffey *et al.*, 2013) |
| *Cryptosporidium* | WDPF (Die-off factor for persistent bacteria on foliage at 20°C) | 1/day | 0.03 | (Tang *et al.*, 2011) |
| *Cryptosporidium* | WOF_P (Fraction of persistent organisms washed off in rainfall events) | Fraction 0≤1 | 0.8 | (Tang *et al.*, 2011) |

The Darvill WWTP serves 51.6% of the 163 993 uMsunduzi Local Municipality households (STATSSA, 2011). The Darvill WWTP is currently being upgraded from 65 000 m$^3$/day to 100 000 m$^3$/day to meet current and future demands (Ramnath, Meier and Mjwara, 2019), which means that it often operated above its capacity and the system was overflowing, releasing untreated wastewater directly to the uMsunduzi river. The wastewater from the Darvill WWTP was put into the SWAT model as a point source with an average discharge of 75 000 m$^3$/day. According to Ramnath, Meier and Mjwara (2019), the inflows to the WWTP were, for a number of months in 2018, below the 12-month moving average (~75 000 m$^3$/day), with the average inflow in October 2018 only ~59 000 m$^3$/day. This is an unseasonal low, and Umgeni Water is concerned that not all the wastewater is reaching the Darvill WWTP. This implies that ~16 000 m$^3$/day could be attributed to losses due to broken or leaking sewers. This volume of wastewater was distributed over the subbasins within the urban section of the catchment in the SWAT model, where there are broken sewers. This data was supplied was supplied by Mr. Vilakazi (pers. comm.) of the Duzi-Umgeni Conservation Trust (DUCT). These leaking sewers were included as a point source contribution in each of these subbasins. For this study, the concentration of *Cryptosporidium* used as input in SWAT was 5 (0-17) oocysts/10L based on 13 samples collected from the Darvill WWTP effluent during 2016-2017 (de Jong, 2020 pers. comm.). *E. coli* concentration in wastewater was estimated to be 50 000 counts/100 mL, a median of 3615 samples from monitoring data supplied by Umgeni Water taken between January 2004 and December 2018.

The remaining 48.4% of the households within the municipality use dry toilets or are without any toilet at home, a small percentage of which may have septic tanks (STATSSA, 2011). Data showing the distribution of these dry toilets were not available, however, these toilets are used mainly in informal and rural settlements. Defaecating in open spaces could also be a human source of microorganisms in this catchment. It was, however, not considered in this study due to difficulties in quantifying its contribution in the runoff.

The SWAT grazing function was used to represent livestock manure production during grazing. Only cattle and sheep/goats were considered for this study as the numbers of other animals were low, and relevant prevalence and infectivity data in Africa are scarce. Due to the warm climate and general subsistence farming practices in Pietermaritzburg, these animals are grazing all year round. No data could be found that account for the distribution of the livestock herds over the catchment, thus, the estimated produced faecal matter was distributed evenly over the modelled grassland landuse area of 15 943 ha. To account for different faecal production depending on age, the fraction of animals that are young was determined using a guide by Vermeulen *et al*., (2017). The average *Cryptosporidium* concentrations in livestock faeces were calculated from the prevalence of infection and the concentration in faeces of infected animals. The assumed input data for *E. coli* and *Cryptosporidium* are presented in Table 3.3. The *E. coli* concentrations shown in Table 3.3 do not differentiate between pathogenic and non-pathogenic *E. coli*.

*Table 3.3 Cryptosporidium and E. coli concentration estimates for SWAT input*

| Host | Livestock in catchment [#][a] | Age differentiation [#][b] | Animal density in grazing area [#/ha] | Manure production [kg/d/1000kg][b] | Animal mass [kg][b] | Manure production [kg/ha/day][b] | *Cryptosporidium* prevalence[b] | *Cryptosporidium* concentration infected livestock [oocysts/g][b] | *Cryptosporidium* concentration total livestock [oocysts/g][c] | *E. coli* concentration [#/g][d] |
|---|---|---|---|---|---|---|---|---|---|---|
| Cattle | 35 345 | 32 517 | 2.04 | 86 | 250 | 32.07 | 0.17 | 1.0E+02 | 1.7E+01 | 2.0E+05 |
| Calves |  | 2 828 | 0.18 |  | 125* | 1.39 | 0.29 | 7.9E+04 | 2.3E+04 | 4.2E+05 |
| Goats | 29 560 | 25 126 | 1.58 | 41 | 30 | 1.42 | 0.16 | 2.0E+02 | 3.2E+01 | 6.6E+04 |
| Kids |  | 4 434 | 0.28 |  | 15 * | 0.13 | 0.11 | 2.5E+04 | 2.7E+03 | 6.6E+04 |
| Sheep | 9 920 | 8 432 | 0.53 | 40 | 28 | 0.43 | 0.25 | 1.6E+02 | 4.0E+01 | 6.6E+04 |
| Lambs |  | 1 488 | 0.09 |  | 14* | 0.04 | 0.13 | 2.0E+05 | 2.6E+04 | 6.6E+04 |

a Data sourced from STATSSA (2011)
b Calculations and values after Vermeulen *et al*. (2017)
c Calculated using concentration in infected livestock and prevalence
d After Coffey *et al*., 2013
*0.5 of the adult animal was used to calculate body mass of the respective young as no data could be found

### 3.2.2 Quantitative Microbial Risks Assessment (QMRA)

The QMRA tool version 2018-06-07 was used for microbial risk assessment to consumers accessed via this link. This tool was developed by Abrahamsson, Ansker and Heinicke (2009) and is managed by the Centre for Drinking Water Research "DRICKS" on behalf of the Swedish drinking water industry. For this study, water treatment steps were omitted in the QMRA tool, to represent untreated water ingested during recreational activities and direct domestic use.

#### a. Source Water Characterisation and Exposure

The outputs from the SWAT model in terms of concentration of the selected microorganisms along the river were used as input values for the QMRA. Considering that not all *E. coli* are pathogenic, 8% of *E. coli* concentration was assumed as pathogenic following various studies such as Haas *et al*. (2000) and Mbanga *et al*. (2020). The exposure routes investigated were direct ingestion of the uMsunduzi river water during recreational swimming, canoeing training, and drinking. These exposure scenarios are detailed below.

Recreational swimming: This exposure scenario was considered for subbasins 1, 4, 8, and 10. This is because it was observed during the study that the population along this stretch of the catchment habitually swim in the river during warm periods. The exposed population was categorised into children and adults. The volume ingested during swimming was estimated based on the ranges of 37-47 mL for children and 16-24 mL for adults per 45 minute event as reported in Dufour *et al*. (2006) as shown in Table 3.4.

Canoeing: This exposure scenario was considered for subbasin 8 as training takes place in this stretch of the river. Training information was obtained from Mr. Z. L. Mthalane (pers. comm., 2019), a coach at two canoe clubs and a seasoned Dusi Canoe Marathon and Dusi Non-stop participant. The exposed population was categorised into three: juvenal (10 - 14 years old), junior (15 - 18 years old), and senior (>18 years old) based on the competition categories and the required hours for training. These weekly training hours are 3 to 5 (juvenal), 6 to 10 (junior), and 11 to 14 (senior). Based on these schedules, 0.5 to 1 hr (juvenal), 1 to 1.5 hours (junior), and 1.5 to 2 hours (senior) per day were assumed. These hours, together with ingestion volume of 5.8 mL per 45 minute event, were used to estimate the ingested volumes during canoeing events as reported in Dorevitch *et al*. (2011). The subsequently calculated daily minima and maxima are presented in Table 3.4.

Drinking Water: This exposure scenario was considered for subbasins 10 and 1, which are in the upper and lower rural parts of the catchment, respectively. The exposed population was categorised into three: juvenal, junior, and senior. Table 3.4 shows daily ingestion volumes for South Africans adapted from Steyn, Jagals and Genthe (2001).

*Table 3.4 Water volumes ingested during swimming, canoeing, and drinking*

| **Ingested during swimming (L/d)**[a] | | |
|---|---|---|
| Category | Min | Max |
| Children | 0.0493 | 0.0627 |
| Adults | 0.0213 | 0.0320 |
| **Ingested during canoeing (L/d)**[a] | | |
| Category | Min | Max |
| Juvenal | 0.0039 | 0.0077 |
| Junior | 0.0077 | 0.0116 |
| Senior | 0.0116 | 0.0154 |
| **Ingested during drinking (L/d)**[a] | | |
| Category | Min | Max |
| Juvenal | 0.0013 | 0.63 |
| Junior | 0.63 | 0.773 |
| Senior | 0.773 | 0.952 |

*a* In the QMRA dose-response modelling the volume was assigned a uniform distribution with minima (Min) and maxima (Max).

### b. Dose-response modelling

The dose-response functions provide the important link between exposure dose and the likelihood of occurrence of a negative consequence like infection, illness, severe illness, or death (Smeets *et al*., 2010; Haas, Rose and Gerba, 2014). The ingested dose (μ) was estimated using Equation 3.4.

*Equation 3.4*

$$\mu = v \; x \; c$$

where μ is the ingested dose of pathogens, v is the ingested volume, and c is the concentration of the targeted pathogens. Ingested volume was based on the values in Table 3.4, using a uniform distribution with minima and maxima, according to the values reported in the table. Concentration was defined as the fitted lognormal distribution based on the simulated daily pathogen concentrations output from the SWAT modelling for each subbasin. For each subbasin, fitting was conducted using the @RISK 8 software, and the lognormal distribution was one of the top three ranking distributions according to the Akaike Information Criterion (AIC).

In the QMRA tool, the Exponential dose-response model was used to calculate the probability of infection by pathogenic *E. coli* (O157:H7) and *Cryptosporidium* (Equation 3.5).

*Equation 3.5*

$$P_{(inf)} = 1 - e^{-r\mu}$$

where $P_{(inf)}$ was the probability of infection, μ the pathogen dose, and r was a Beta function. In the QMRA tool, applying Monte Carlo simulations and defining a Beta function for r in the Exponential function approximates an exact Beta-Poisson model. The Beta function (α, β) was used for *Cryptosporidium* (0.115, 0.176) (Teunis, Chappell and Okhuysen, 2002) and pathogenic *E. coli* (0.37, 37.65) (Teunis, Takumi and Shinagawa, 2004).

## 3.3 Results and Discussion

### 3.3.1 Water Quality Model

The SWAT model's predictions for streamflow were in good agreement with monitored data as shown in Table 3.5. The good agreement is corroborated by the visual comparison between simulated and observed flow for calibration and validation periods, using monthly data (Figure 3.2). All evaluation metrics shown in Table 3.5, that is, NSE > 0.5, RSR ≤ 0.7 and PBIAS ≤ ±0.25, except for the PBIAS for validation for subbasin 10, are referred to as satisfactory, with the PBIAS only slightly out of range (Moriasi *et al*., 2007). The PBIAS possibly deviated from its expected range because of data quality issues. Parameters optimised during the calibration process are shown in Table 3.6.

*Table 3.5 SWAT calibration and validation results for streamflow*

| | **Calibration** | | **Validation** | |
|---|---|---|---|---|
| **Index** | **Subbasin 1** | **Subbasin 10** | **Subbasin 1** | **Subbasin 10** |
| $R^2$ | 0.58 | 0.69 | 0.64 | 0.73 |
| NSE | 0.57 | 0.61 | 0.58 | 0.59 |
| RSR | 0.31 | 0.29 | 0.25 | 0.28 |
| PBIAS | -0.05 | 0.24 | 0.14 | 0.27 |

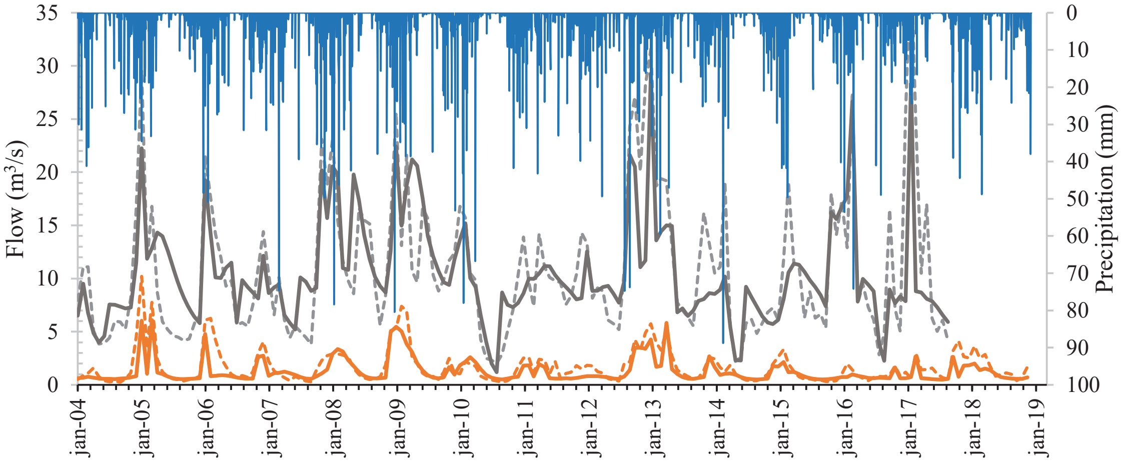


*Figure 3.2 Graphical comparison of simulated streamflow against observed flow for subbasins 1 and 10 calibration (Jan 2004 – Dec 2011) and validation (Jan 2012 – Dec 2018) periods. Subbasin 1 is depicted in grey and subbasin 10 in orange. Continuous and dashed lines represent simulated and observed data, respectively. Blue line in secondary vertical axis shows the daily precipitation*

*Table 3.6 Parameters optimised during SWAT calibration*

| **Variable** | | **Description** | **Default Value(s)** | **Calibrated Value** | **Input file** |
|---|---|---|---|---|---|
| Flow | $CN_2$ | Initial SCS runoff curve number for moisture condition | 45 - 85 | 49 | .mgt |
| | SOL_K | Saturated hydraulic conductivity | 2.6 - 52 mm/hr | 7.54 mm/hr | .sol |
| | SOL_BD | Soil moist bulk density | 0.9 - 2.5 g/cm$^3$ | 1.36 g/cm$^3$ | .sol |
| | ESCO | Soil evaporation compensation factor | 0.95 | 0.9018 | .sol |
| | SOL_AWC | Available water capacity of the soil | 0.12-52 mm $H_2O$/mm Soil | 0.127 mm $H_2O$/mm Soil | .sol |
| | MSK.$CO_2$ | Coefficient for impact of low flow | 3.5 | 3.65 | .bsn |
| | CANMX | Maximum water storage in canopy | 0 mm | 0.3 mm | .hru |
| Base flow | ALPHA-BF | Groundwater recession factor | 0.084 d | 0.27 d | .gw |
| | GW-DELAY | Groundwater delay | 31 d | 28 d | .gw |

Figure 3.3 and Figure 3.4 show the daily simulated concentrations of *Cryptosporidium* and *E. coli* in subbasins 10, 8, 4, and 1. In Figure 3.4, these are plotted against weekly monitored *E. coli* concentrations between January 2004 and December 2018. The output from the SWAT model shows slightly low variability, a standard artefact observed in microbiological modelling (Iqbal and Hofstra, 2019a). In general, the trend in the SWAT model simulations follows the observed data patterns in most subbasins. The model may not have captured some of the extremities observed in the monitoring data, however, the simulated peaks are one order of magnitude less than those of the monitoring data. Comparing the monitored and simulated mean of the *E. coli* concentration shows that for subbasins 4 and 1 the SWAT model overestimates, while for subbasin 10 and 8 it slightly underestimates the concentrations. If the model overestimates, it might lead to unnecessary or excessive interventions, potentially

wasting resources and causing undue concern among the public. Conversely, if the model underestimates, it could fail to identify areas of concern, allowing pollution or other hazards to go unaddressed, thereby jeopardising public health.

Figure 3.4 shows that subbasin 4 has the highest *Cryptosporidium* concentration, followed by subbasin 1, then subbasins 8 and 10, respectively. The trends are generally such that concentrations are higher downstream the WWTP than upstream. *Cryptosporidium* concentrations analysed between September 2015 and March 2016 by Adeyemo (2019) in effluents from the wastewater treatment plants in Durban show a range of 50 - 70 oocysts/10L (six samples) was detected in the effluent. In the study by Dungeni and Momba (2010) in the Gauteng province (South Africa), the concentrations were <0.1 - 4 oocysts/10L, in 14 samples taken between January and April 2008, weekly. In a study performed between March 2016 and March 2017 with fortnightly sampling, Razzolini *et al*. (2020) detected *Cryptosporidium* concentrations of <0.003- 2.6 oocysts/10L in wastewater effluent in Sao Paulo, Brazil. In Shanghai China, Ma *et al*. (2016) detected 0-0.1 oocyst/10L *Cryptosporidium* concentrations in wastewater effluent from three WWTPs. In this study, values provided by de Jong (2020, pers. comm.), 5 oocysts/10L, were used as these were from the Darvill WWTP specifically. Due to a short time-series, there is a possibility of underestimation. More continuous monitoring can help reduce this uncertainty. Robertson, Johansen and Kifleyohannes, (2020) suggest that the likelihood of *Cryptosporidium* pollution of drinking water in Africa is probably greater than in many other countries, due to water supplies being limited and a general lack of catchment control for securing the supply of water, as surface waters are both used as drinking water and for livestock drinking.

In reference to *Cryptosporidium*, DWAF (1996a) for domestic use, states that 1 oocyst/10L (0.1 oocyst/L) is enough to pose the risk of infection. No guidelines are given for protozoa in recreational use.

In the DWAF (1996b) guidelines for recreational use, the effects of *E. coli* on human health based on their concentration are outlined as follows: The concentrations of 0-130 counts per 100 mL may pose a low risk of gastrointestinal illness for full-contact recreational water use; negligible effects are expected if these levels occur in isolated instances only. The concentrations of 200-400 counts per 100 mL may pose some risk of gastrointestinal effects, particularly if this occurs frequently. Above 400 counts per 100 mL will pose increasing risk of gastrointestinal effects. The criteria proposed for full-contact recreation (swimming) are also recommended for intermediate contact (canoeing) (DWAF, 1996b).

In the DWAF (1996a) guidelines for domestic use, the effects of faecal coliforms on human health based on their concentration are as follows: 0 will pose negligible risk of microbial infection; 0-10 counts per 100 mL will pose a slight risk of microbial infection with continuous exposure and negligible effects with occasional or short-term exposure; 10-20 counts per 100 mL will pose risk of infectious disease transmission with continuous exposure and slight risk with occasional exposure; and >20 counts per 100 mL will pose significant and increasing risk of infectious disease transmission.

In subbasins 10 and 8, animal sources contribute more than human sources to the combined *E. coli* concentrations, on the other hand, subbasins 4 and 1 show high human source impact. The model has successfully captured the trends of different sources in different settlements within the catchment. For subbasin 10 (upper rural), human contributions are lower than animal contributions, highlighting the intensive and uncontrolled livestock farming. As subbasin 8 is a transition between rural and urban areas, human contributions are still lower than animal contributions in these parts. Broken sewers do contribute in these parts, albeit low. In subbasin 4, there is a significant contribution from human sources (Figure 3.4) from the WWTP and broken sewers. Gemmell and Schmidt (2013) pointed out that the water quality of the uMsunduzi River is poor, mainly due to faecal pollution, and deteriorates as the river passes through the city.

The data in Figure 3.3 and Figure 3.4 show that *E. coli* concentrations in the uMsunduzi River render the water unsuitable for drinking, swimming, and canoeing due to the high possibility of contracting gastrointestinal diseases, according to DWAF (1996a, 1996b).

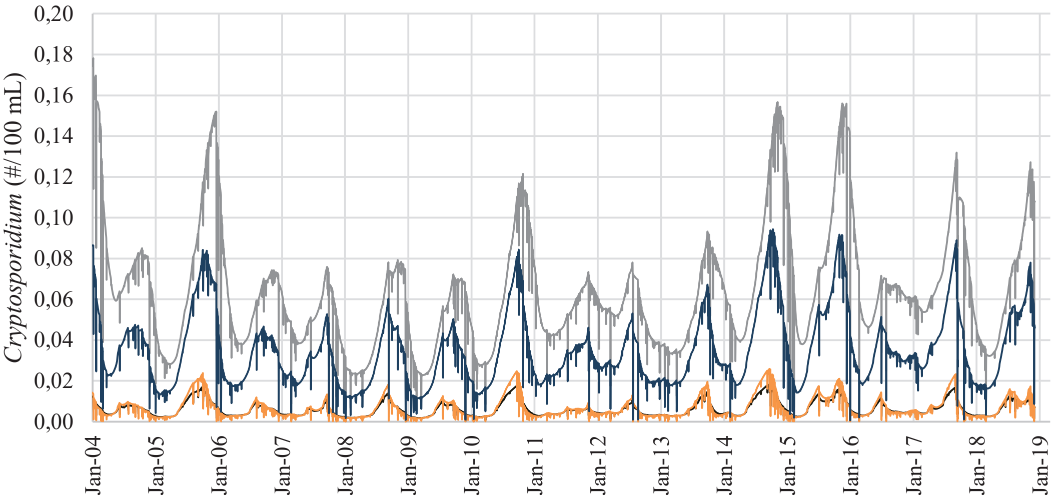


*Figure 3.3 Simulated Cryptosporidium concentrations in basin 1, 4, 8, and 10. The black line represents subbasin 10, grey line represents subbasin 4, orange line represents subbasin 8, and the blue line represents subbasin 1*

There are several limitations to this study in terms of the microbial estimates in the SWAT model. *Cryptosporidium* loads in the effluent from the WWTP were represented as a point value rather than using a time-series. The quantification of the contribution from broken sewers was an estimate since the reporting of this did not include any information on discharge. There are also limitations in available data on pathogen parameters, for example, the proportion of pathogenic *E. coli*. In the townships and rural areas, pit latrines are used as sanitation infrastructure, but in the study area, their contribution to the surface waters could not be quantified. Also, microbial contributions from direct stream deposits from livestock and wildlife were not estimated. Defaecating in open spaces could also be a human source of microorganisms in this catchment. It was, however, not considered in this study due to difficulties in quantifying its contribution in the runoff.

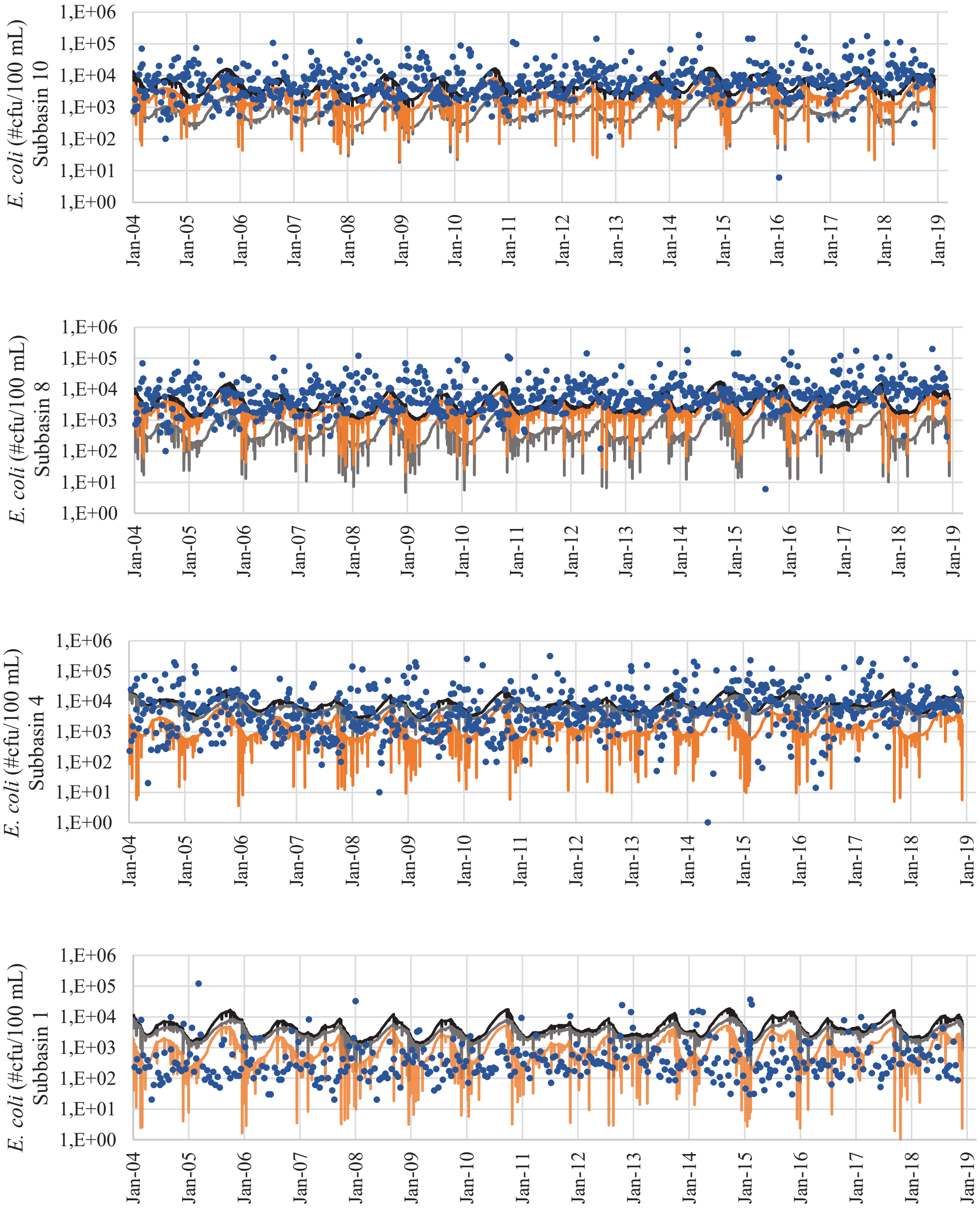


*Figure 3.4 Simulated and monitored (#cfu/100mL) E. coli concentrations in basins 10, 8, 4, and 1 (respectively from top to bottom). The black line represents combination of sources, grey line represents human sources, orange line represents animal sources, and the blue circles represent monitored E. coli*

### 3.3.2 Quantitative Microbial Risk Assessment Results

Table 3.7 shows the probability of infection ($P_{inf}$), 50th and 95th percentile, from different exposure scenarios to *Cryptosporidium* and pathogenic *E. coli*, respectively. Table 3.7 also shows $P_{inf}$ calculated using monitored *E. coli*. The $P_{inf}$ calculated using simulated data and using the monitoring data generally fall into the same order of magnitude. To put the results of this study into the context of the existing guidelines, the concentrations stated by DWAF were recalculated into the probability of infection. In the DWAF (1996a) guidelines for domestic use, the effects of faecal coliforms on human health based on their concentration is 0 counts/100 mL for negligible risk of microbial infection. In terms of QMRA inputs, uncertainty lies with the ingested volumes for exposure routes.

A value of 0.8 counts per L (based on 1 count/100 mL), accounting for pathogenic portion, was used together with different ingestion scenarios in Table 3.4, to calculate the associated probability of infection. For domestic use, the recalculated 50th percentile risk values were 6/10 000 (juvenile), 2/1000 (junior), and 3/1000 (senior). For swimmers and canoeists, according to DWAF (1996b), low risk of gastrointestinal illness is considered as below 8/1000.

For *Cryptosporidium*, for domestic use, the guidelines by DWAF (1996a) state that 1 oocyst/10L is enough to pose the risk of infection; and no guidelines are given for protozoa in recreational use. Therefore, a concentration of 1 oocyst/10L as a point value was used together with different ingestion scenarios in Table 3.4 to calculate the probability of infection. The 50th percentile by DWAF (1996b) for swimming is 9/10 000 children and adults 4/10 000 adults with respect to *Cryptosporidium*. For canoeing; 8/100 000 Juvenile, 2/10 000 Junior, and 2/10 000 Seniors were calculated based on the DWAF (1996b).

The findings of this study are summarised below in terms of the 50th percentile of probability of infection calculated using the SWAT output data. Recreational swimming: In reference to *Cryptosporidium*, subbasins 1, 4, and 8 show generally higher probability of infection than the calculated 50th percentile by DWAF (1996b). In reference to pathogenic *E. coli*, the probability of infection is above the DWAF "low risk category", with the highest risks in subbasin 4.

Canoeing: In subbasin 8, the probability of infection with *Cryptosporidium* is lower than the recalculated values from DWAF (1996b) and higher for Junior and Senior canoeists. The probability of infection with pathogenic *E. coli* during canoeing in subbasin 8 was found to be higher than the risk suggested by DWAF (1996b) of 8/1000 canoeists across all scenarios.

Drinking Water: QMRA results for *Cryptosporidium* indicate that the probability of infection, to the population that drink untreated river water in subbasin 1 is higher than DWAF (1996a) recommendations for protozoa in drinking water. This risk may be reduced by boiling the water before drinking. On the contrary, in subbasin 10, for all groups, the probability of infection is lower than the thresholds in DWAF (1996a). Based on pathogenic *E. coli*, QMRA results show that in subbasin 1 and 10 the probability of infection is higher than the criteria set in DWAF (1996a). Smeets *et al*. (2010) used 2.74E-07 per person per day as the drinking risk benchmark, based on the work by Signor and Ashbolt (2009). Both subbasins 1 and 10, across all drinking scenarios, are above the 2.74E-07 per person per year.

Mbanga *et al*. (2020) quantified *E. coli* pollution and the potential health hazards that workers at the Darvill WWTP and nearby informal communities may face after exposure to waterborne pathogenic *E. coli* in the WWTP and the uMsunduzi River. The authors found that the daily probability of infection with pathogenic *E. coli* following intentional uptake of 100 mL of the river water upstream and downstream from Darvill is 97.6 and 90.8%, respectively.

*Table 3.7 Probability of infection, $P_{inf}$, by Cryptosporidium and pathogenic E. coli across subbasins*

| **Exposure Scenario (per day)** | **Age Group** | **Location** | **$P_{inf}$ 50th (95th) Percentile *Cryptosporidium*** | **Pinf 50th (95th) Percentile Modelled *E.coli*** | **Pinf 50th (95th) Percentile Monitored *E.coli*** |
|---|---|---|---|---|---|
| Swimming | Adults | Sub 10 | 1.78E-04 (2.59E-03) | 3.23E-01 (9.99E-01) | 6.34E-02 (9.83E-01) |
| | | Sub 8 | 2.4E-03 (1.91E-02) | 2.51E-01 (9.94E-01) | 3.02E-01 (1.00E+00) |
| | | Sub 4 | 1.96E-03 (2.37E-02) | 4.58E-01 (1.00E+00) | 2.15E-01 (1.00E+00) |
| | | Sub 1 | 1.01E-03 (1.41E-02) | 2.99E-01 (9.98E-01) | 2.12E-02 (5.10E-01) |
| | Children | Sub 10 | 3.17E-04 (5.35E-03) | 5.79E-01 (1.00E+00) | 1.18E-01 (1.00E+00) |
| | | Sub 8 | 5.08E-03 (3.87E-02) | 4.33E-01 (1.00E+00) | 5.38E-01 (1.00E+00) |
| | | Sub 4 | 3.90E-03 (4.99E-02) | 7.55E-01 (1.00E+00) | 4.04E-01 (1.00E+00) |
| | | Sub 1 | 2.44E-03 (2.90E-02) | 5.23E-01 (1.00E+00) | 4.41E-02 (7.99E-01) |
| Canoeing | Juvenal | Sub 8 | 3.14E-05 (6.52E-04) | 5.99E-02 (6.7E-01) | 7.18E-02 (8.99E-01) |
| | Junior | Sub 8 | 6.34E-05 (1.13E-03) | 1.00E-01 (8.54E-01) | 1.24E-01 (9.74E-01) |
| | Senior | Sub 8 | 9.25E-05 (1.58E-03) | 1.38E-01 (9.23E-01) | 1.69E-01 (9.94E-01) |
| Drinking | Juvenal | Sub 10 | 1.34E-03 (3.49E-02) | 9.65E-01 (1.00E+00) | 7.78E-01 (1.00E+00) |
| | | Sub 1 | 8.15E-03 (1.78E-01) | 9.54E-01 (1.00E+00) | 1.71E-01 (1.00E+00) |
| | Junior | Sub 10 | 4.00E-03 (6.47E-02) | 1.00E+00 (1.00E+00) | 9.90E-01 (1.00E+00) |
| | | Sub 1 | 2.81E-02 (3.10E-01) | 1.00E+00 (1.00E+00) | 4.22E-01 (1.00E+00) |
| | Senior | Sub 10 | 5.51E-03 (7.87E-02) | 1.00E+00 (1.00E+00) | 1.00E+00 (1.00E+00) |
| | | Sub 1 | 3.28E-02 (3.63E-01) | 1.00E+00 (1.00E+00) | 5.06E-01 (1.00E+00) |

Several local studies (WRC, 2002; Gemmell and Schmidt, 2013; Mbanga *et al*., 2020) highlight that the water quality of the uMsunduzi River is poor to very poor, with apparent faecal pollution which impacts on Inanda Dam which serves the eThekwini Municipality with potable water. The findings of this study agree with these studies where they concluded that the uMsunduzi River water did not meet the requirements of DWAF for drinking and recreational uses, in terms of *E. coli*.

Risk of infection during swimming risk is highest in the urban parts of the catchment for both children and adults, considering both pathogens. This part receives contribution from sewers, WWTP, as well as animals. Canoeing was considered only for training events in this study. Canoeing competitions do take place in the uMsunduzi catchment. One of these is the historic Dusi Canoe Marathon taking place in February each year, along the uMsunduzi River to the uMngeni River all the way to Durban. It attracts 900-1600 paddlers each year. Hay (2017) reported that a survey done in 2016, immediately after the race, revealed that 40% of the paddlers contracted mild to severe gastro-intestinal infections. This illness has been termed the "Dusi guts" by the paddlers. It is also reported that seasoned paddlers take antibiotics before the race, as a preventative measure.

Those who drink the water from the river are at the highest risk of infection by both studied pathogens. The rural communities and those in peri-urban areas are affected the most. These communities supplement the lack/shortage of treated water with the water from the polluted uMsunduzi. These communities are not always aware of the risks and how they can protect themselves.

This study focused on exposure per event for risk calculations and did not account or relate to the frequency of these events. To extend this approach to annual risk, the frequency of drinking river water may be considered to occur on a daily basis. However, canoeists may not train every day of the year, the training may intensify closer to competitions compared to outside competitions. Additionally, recreational swimming may be more intense during hotter days of the year (September to March) as opposed to the colder days. These aspects are important to consider if results are reviewed and compared to an annual risk level. Nonetheless, the effect of risk mitigation can be evaluated on a per event or a daily risk level as well (Signor and Ashbolt, 2009).

This study opted for the lognormal distribution dose-response model due to its robust statistical properties and its ranking as one of the top three distributions according to the AIC. However, the variations in dose-response relationships and parameters could potentially impact the outcomes of the QMRA. Future studies could delve deeper into exploring alternative dose-response models and assessing the sensitivity of the QMRA results to changes in these relationships and parameters, thereby providing a more comprehensive understanding of the uncertainties inherent in our QMRA approach.

## 3.4 Conclusions

The aim of this chapter was to utilise SWAT and QMRA to highlight areas of high pollution and the risk that the water may pose to the health of consumers. Based on the water quality modelling results of this study, it can be concluded that the uMsunduzi River is highly polluted with pathogens, and the use of untreated water from the river may result in a high risk of infection.

In the urban section, human sources contribute more than animal sources, indicating to the contributions from broken sewers. In terms of the QMRA, people who use the river for domestic and recreational purposes are at risk of infection by *Cryptosporidium* and pathogenic *E. coli*.

Investing in water treatment facilities, regulation of livestock practices, and safe sanitation systems for communities in need are likely to provide long-term and reliable improvement in the uMsunduzi River water quality. Indirect reuse through the river is not often considered when talking about wastewater reuse. The lack of treated water in rural areas and informal settlements usually forces people to reuse water without giving it much thought. Supporting rural communities without reticulated water is only possible if the water is not directly drawn from the river. Controlling how children access the river and educating the people about boiling their drinking water will help reduce the exposure to pathogens.

# Chapter 4 Quantitative Chemical Risk Assessment

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## 4.1 Introduction

According to the World Health Organisation (WHO, 2020), 24% of deaths worldwide are caused by environmental factors that can be modified, including exposure to toxic chemicals. Chemicals have an impact on aquatic environments through different sources, including through wastewater and industrial discharges, agricultural residues, and land runoff. The conventional treatment processes offered by water or wastewater treatment plants cannot generally remove or reduce these substances to acceptable low values. Ecological studies, animal models, human clinical observations, and epidemiological studies confirm the importance of chemicals impacting wildlife and humans (Pironti *et al.*, 2021). Several studies have studied the occurrence of these chemicals in wastewater (for example, Adeyinka *et al.*, 2019; Nyamukamba *et al.*, 2019), surface water (for example, Matongo *et al.*, 2015; Sengar and Vijayanandan, 2022), sediments (for example Shozi, 2015; Adeyinka *et al.*, 2019), and groundwater (Zhai *et al.*, 2017; Mohammadi *et al.*, 2019).

In the surface waters of the uMsunduzi River, South Africa, organochlorinated pesticides (OCPs), pharmaceuticals and personal care products (PPCPs), heavy metals, and nitrates and phosphates have been detected in variable concentrations (Matongo *et al.*, 2015; Shozi, 2015; Adeyinka *et al.*, 2019). In addition, chemical spills have been observed in the uMsunduzi River and its tributaries. To illustrate, in the year 2019, an oil spill from an oil company was reported; the spill affected the majority of the uMsunduzi River downstream of the city of Pietermaritzburg (Mdletshe, 2019). Fish, aquatic animals, and livestock who drank from the river died as a result of this spill (Mdletshe, 2019). Additionally, the different settlements within the catchment use this river for recreational activities and drinking without treatment.

In their assessment of forty research studies that investigated agrochemicals in freshwater aquatic habitats in South Africa from 2011 to 2020, Horak, Horn and Pieters (2021) examined rivers such as the uMngeni, Vaal, Olifants, Buffalo, Lourens, and uMsunduzi. According to these research studies, agrochemicals were found in all of South Africa's provinces, including along the Indian Ocean coast. OCPs are a group of chlorinated chemicals historically used as insecticides with some classified as persistent organic pollutants based on their extended half-lives in the environment (Adeyinka *et al.*, 2019; Wolmarans *et al.*, 2021) and toxicity even at low concentrations (de Souza *et al.*, 2020). These pesticides were banned or severely restricted by the Stockholm Convention on Persistent Organic Pollutants in the year 1983 (UNEP, 2020). Agrochemicals can impair organisms' (including humans) ability to produce hormones normally, which can result in several endocrine disrupting impacts (Qu *et al.*, 2015), including intersex, decreased spermatogenesis, asymmetric urogenital papillae, testicular lesions, and infertile eggs (Horak, Horn and Pieters, 2021). As a result, OCPs continue to pose a hazard to both the ecological environment and human health (Chen *et al.*, 2020). In humans, OCPs have also been listed as supposed carcinogens (Wexler, 2014).

According to Adeleye *et al.* (2022), PPCPs are introduced into urban wastewater systems through hospitals, PPCPs producers, and agricultural sources in addition to human excretion and typical household usage. Based on the data obtained from wastewater treatment plants (WWTPs) in South Africa, analgesics, antibiotics, and stimulants are the most abundant PPCPs in raw wastewater (Adeleye *et al.*, 2022). Regulation of pharmaceuticals and their treatment are not as stringent in African countries as compared to developed economies; and the current wastewater treatment systems were not designed with the intent of managing pharmaceuticals as pollutants (Agunbiade and Moodley, 2016) but will, to a varying degree, reduce their concentrations (Matongo *et al.*, 2015; Faleye *et al.*, 2019). Different investigations confirm the presence of PPCPs in natural waters around the world, including freshwater (such as rivers, streams, lakes), marine and estuarine environments, groundwater, and sediment (Agunbiade and Moodley, 2016; Kong *et al.*, 2021; Adeleye *et al.*, 2022). The presence of PPCPs in receiving waters and sediments has been shown in the KwaZulu-Natal province, South Africa (Matongo *et al.*, 2015; Agunbiade and Moodley, 2016). In the environment, PPCPs have been reported to be enriched highly in organisms and amplified with the food chain (Lin *et al.*, 2023). In humans, PPCPs have different side effects upon prolonged exposure and overdosage (Wexler, 2014).

Heavy metal ions are among the most released pollutants, and for this reason as well as their persistence they are particularly concerning. About 40% of lakes and rivers on Earth are polluted with heavy metals, the sources of which are both natural and anthropogenic (Zamora-Ledezma *et al.*, 2021). Particularly in recent decades, human activities such as urbanisation, industrialisation, and pollution have increased the concentration of these pollutants (Zamora-Ledezma *et al.*, 2021). Mean concentrations of heavy metals in water samples from South Africa exceeded the World Health Organization guidelines for safe levels of intake, according to the study performed by Genthe *et al.* (2018). That study followed the death of a dozen crocodiles in the Olifants River catchment near the South Africa-Mozambique border, where it was found that the death was due to anthropogenic pollution (Genthe *et al.*, 2018). In humans, these chemicals have such effects as liver damage, reduced lung function, and thyroid disorder with some heavy metals having probable carcinogenic effects (Wexler, 2014).

Agricultural runoff, industrial effluents, and municipal wastewater systems may lead to excess nitrogen and phosphorus load in water environments. As a result, algal overgrowths or the presence of noxious algal species can become a nuisance and interfere with the desirable uses of a water body (DWAF, 1996b). Waterbody eutrophication in turn impacts human health by causing conjunctivitis, dermatological conditions, and gastrointestinal illnesses (Oberholster and Ashton, 2008).

The human health risk assessment is the process used to estimate the nature and probability of adverse health effects among people exposed to hazardous environmental substances now or in the future (Genthe *et al.*, 2018). To understand the nature, magnitude, and probability of an adverse health or environmental effect of a chemical, a chemical risk assessment is required (Nyamukamba *et al.*, 2019; Moloi, Ogbeide and Voua Otomo, 2020). Health risk assessments are primarily intended to protect consumers against serious adverse effects of toxicants in food or water (Taiwo, 2019) and comprise four main stages: hazard identification, exposure assessment, dose-response characterisation, and risk characterisation (Fryer *et al.*, 2006). Chen *et al.* (2020) performed an ecological and health risk assessment of OCPs in an urbanised river network of Shanghai, China. This study found that OCPs in river waters did not pose significant non-carcinogenic health hazards for a majority of the people. Sengar and Vijayanandan (2022) assessed ecological and human health risks of PPCPs detected in surface waters and wastewater in India. This study found very high RQs for few pharmaceuticals were obtained, signifying a great potential of the detected PPCPs in causing severe health concern. Carcinogenic and non-carcinogenic health risks posed by heavy metals were assessed in groundwater water in rural Iran (Maleki and Jari, 2021), in wastewater discharge in the Vaal River Basin in South Africa

(Moloi, Ogbeide and Voua Otomo, 2020), and in drinking water in Khorramabad, Iran (Mohammadi *et al.*, 2019). The health impact of nitrate pollution in groundwater was investigated by Zhai *et al.* (2017) in Songnen Plain of Northeast China, and in this study it was concluded that risk levels generally followed the pattern of being highest for infants, followed by children, adult females, and then adult males.

The goal of chapter was to demonstrate the importance of riverine chemical pollution by undertaking a quantitative chemical risk assessment for consumers of untreated water engaged in domestic and recreational activities. This strategy helps identify important areas of concern while also highlighting the need for mitigation measures. The aim of this chapter therefore was to quantify the carcinogenic and non-carcinogenic risks that chemical pollutants pose to the population of the uMsunduzi catchment through consumption of the water for drinking (without treatment) and inadvertent ingestion during swimming and canoeing. The specific objectives were to (i) review existing literature on chemical concentrations within the uMsunduzi River, (ii) access and evaluate river monitoring data, and (iii) quantify the probability of developing both adverse non-carcinogenic health effects (hazard quotient: HQ) and cancer risk (CR).

## 4.2 Materials and Methods

### 4.2.1 Chemicals Detected Within uMsunduzi Catchment

#### a. Organochlorinated Pesticides

Adeyinka *et al.* (2019) evaluated the concentrations of OCPs in the sediments, soil, and surface water of the uMsunduzi catchment, and at the Darvill wastewater treatment plant. The pesticides detected in the uMsunduzi River were: hexachlorobenzene (HCB), hexachlorocyclohexane (HCH), heptachlor, aldrin, dichlorodiphenyldichloroethane (o,p’-DDD, p,p’-DDD), dichlorodiphenyltrichloroethane (o,p’-DDT, p,p’-DDT), dieldrin, endrin, and mirex (Adeyinka *et al.*, 2019). Two samples (per location) were collected, and the concentrations are presented in Table 4.2 as reported by the authors. A grab sampling technique was used to collect wastewater or surface water samples from a depth of 1–2 cm from the water surface.

### a. Pharmaceuticals and Personal Care Products

The PPCPs in the uMsunduzi catchment were measured by Agunbiade and Moodley (2016) and Matongo *et al.* (2015), their concentrations are shown in Table 4.4.

These include therapeutic classes such as antipyretics (acetaminophen, aspirin, diclofenac, ibuprofen, ketoprofen); stimulants (caffeine); anti-epileptics (carbamazepine); antipsychotics (clozapine); antibiotics (ampicillin, ciprofloxacin, erythromycin, metronidazole, nalidixic acid, sulfamethoxazole, sulfamethazine, trimethoprim); and antihyperlipidemic (bezafibrate), as shown in Table 4.3. These were selected based on statistics of drug usage in South Africa (Agunbiade and Moodley, 2016). Sampling was performed over spring, summer, autumn, and winter with one sample collected per location for each season in one year. A grab sampling technique was used to collect wastewater or surface water samples from a depth of 1–2 cm from the water surface. The presented concentrations show the mean of the four samples and the standard deviation for the work of Matongo *et al.* (2015) as reported by the authors. For Agunbiade and Moodley (2016), the mean concentrations are shown, as their work only reported mean concentrations for surface water. Where both Matongo *et al.* (2015) and Agunbiade and Moodley (2016) have the same PPCP, for instance, ibuprofen, the higher of the two values was used to calculate risk.

The recommended reference dose (RfD) shown in the Table 4.3 derived from the National Department of Health of South Africa (2020) shows that seven of the drugs are not recommended for children.

*Table 4.1 Organochlorinated pesticides detected in the uMsunduzi catchment; their human effects, status in South Africa (SA), environmental persistence, cancer slope factors (where applicable), and reference doses are presented below*

| **Pesticide name** | **Human effects if the reference dose is exceeded.** | **Current Status in South Africa** | **Environmental Persistence (Half-life in soil)** (Wexler, 2014) | **CSF**[1] **(1/mg/kg/day)** (ATSDR, 2005, 2007, 2020, 2021b, 2022) | **Reference Dose (RfD)**[2] **(mg/kg/day)** (ATSDR, 2005, 2007, 2020, 2021b, 2022) |
|---|---|---|---|---|---|
| aldrin ($C_{12}H_8Cl_6$) | Myoclonic jerk, dizziness (Rani *et al.*, 2021). The target organs are the central nervous system and the liver (WHO, 2017). **Probable carcinogen** (ATSDR, 2021a) | Banned (1983) (DEA, 2011) | 2-15 years (changes to dieldrin) | 17 | 0.00003 |
| dieldrin ($C_{12}H_8Cl_6O$) | Vomiting, disorder in muscle movement (Rani *et al.*, 2021).<br>**Probable human carcinogen** (ATSDR, 2021a) | Banned (1983) (DEA, 2011) | 5 years (Quinn *et al.*, 2011) | 16 | 0.00005 |
| *o,p'*-DDE, *p,p'*-DDE | **Probable human carcinogen** (ATSDR, 2022) | Banned (1983) (DEA, 2011) | 2-30 years | 0.34 | 0.0003 |
| *o,p'*-DDD, *p,p'*-DDD | **Probable human carcinogen** (ATSDR, 2022) | Banned (1983) (DEA, 2011) | 2-30 years | 0.24 | 0.00003 |
| *o,p'*-DDT ($C_{14}H_9Cl_5$) | Nausea, seizures (Rani *et al.*, 2021).<br>**Probable human carcinogen** (ATSDR, 2022) | Banned (1983) (DEA, 2011) | 2-30 years | 0.34 | 0.0005 |
| *p,p*-DDT ($C_{14}H_9Cl_5$) | Nausea, seizures (Rani *et al.*, 2021), breast cancer (Wexler, 2014).<br>**Probable human carcinogen** (ATSDR, 2022) | Banned (1983) (DEA, 2011) | 2-30 years | 0.34 | 0.0005 |
| endrin ($C_{12}H_8Cl_6O$) | Nausea, damage of CNS (Rani *et al.*, 2021) (WHO, 2017).<br>**Not carcinogenic** (ATSDR, 2021b) | Withdrawn in 1980 (DEA, 2011) | 12 years | - | 0.0003 |
| heptachlor ($C_{10}H_5Cl_7$) | damage to the liver and central nervous system toxicity (WHO, 2017). **Probable human carcinogen** (ATSDR, 2007) | Banned (1983) (DEA, 2011) | 0.3-2.5 years (Quinn *et al.*, 2011) | 4.5 | 0.0005 |
| hexachlorobenzene (HCB) ($C_6Cl_6$) | impairment of several organ systems including the kidneys, blood cells, and immune, endocrine, develop- mental, and nervous systems. In babies porphyria cutanea tarda, poor growth, arthritis, and enlarged thyroids. (Wexler, 2014). **Probable human carcinogen** (ATSDR, 2015) | Banned (1983) (DEA, 2011) | 3-6 years | 1.6 | 0.0008 |
| hexachlorocyclohexane (Lindane) (HCH) ($C_6H_6Cl_6$) | Liver and kidneys. **Probable human carcinogen** (ATSDR, 2005) | Banned (1983) (DEA, 2011) | 3-6 years | 6.3 | 0.0003 |
| mirex ($C_{10}Cl_{12}$) | Damage liver, CNS, and reproductive system (Rani *et al.*, 2021)<br>**Probable human carcinogen** (ATSDR, 2020) | Was never registered in South Africa (DEA, 2011) | ~10 years | 0.93 | 0.0002 |

*Table 4.2 Concentrations of organochlorinated pesticides presented as mean (SD) (mg/L) detected in selected sub-basins of the uMsunduzi River surface water by Adeyinka et al. (2019). (NGV = No guideline value)*

| **Pesticide** | **Subbasin 10** | **Subbasin 8** | **Subbasin 4** | **Subbasin 1** | **DWAF Guidelines (mg/L) (DWAF, 1996a; Horak, Horn and Pieters, 2021)** | **WHO Guidelines (mg/L) (WHO, 2017)** |
|---|---|---|---|---|---|---|
| HCB | 3.31 (0.86) | 0.76 (0.02) | 1.73 (0.49) | 1.13 (1.99) | NGV | NGV |
| HCH | 8.77 (1.15) | 11.30 (0.50) | 17.63 (0.66) | 6.67 (2.75) | 0.015 | 0.002 |
| Heptachlor | 2.55 (1.14) | 6.42 (0.31) | 0.06 (1.86) | 3.44 (0.12) | <0.39 | NGV |
| Aldrin | 3.72 (0.80) | 7.71 (0.27) | 1.89 (2.16) | 4.64 (0.09) | 0.01 | 3.00x10-5 |
| o,p'-DDE | 9.17 (0.30) | 6.73 (0.32) | 7.48 (0.47) | 7.48 (0.62) | 0.0015 | 0.001 |
| p.p'-DDE | 14.36 (0.29) | 12.6 (0.44) | 15.06 (0.51) | 11.81 (0.10) | 0.0015 | 0.001 |
| o.p'-DDD | 16.36 (0.23) | 14.24 (0.05) | 15.14 (0.43) | 14.55 (0.12) | 0.0015 | 0.001 |
| Dieldrin | 8.39 (0.19) | 6.54 (0.28) | 8.14 (0.64) | 6.84 (0.18) | 0.0015 | 0.001 |
| Endrin | 2.95 (1.32) | 10.15 (0.26) | 0.37 (1.31) | 2.04 (0.83) | 0.005 | 3.00x10-5 |
| p,p'-DDD | 6.71 (1.21) | 12.35 (0.96) | 2.93 (1.55) | 4.82 (1.13) | 0.0015 | 0.001 |
| o,p'- DDT | 1.94 (2.16) | 7.76 (1.50) | 0.98 (0.24) | 1.52 (1.50) | 0.0015 | 0.001 |
| p,p'-DDT | 13.24 (0.23) | 13.53 (0.31) | 2.09 (0.52) | 18.84 (0.50) | 0.0015 | 0.001 |
| Mirex | 16.79 (0.27) | 24.98 (0.41) | 12.08 (1.33) | 19.06 (0.13) | 0.001 | NGV |

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[1] CSF is the cancer slope factor, and it is defined as the risk generated by a lifetime average amount of one mg/kg/day of carcinogen chemical.

[2] RfD is the reference dose, and it is defined as maximum acceptable oral dose of a toxic substance.

*Table 4.3 Pharmaceuticals and personal care products detected in the uMsunduzi catchment; their human effects, and reference doses are presented below. (NR= Not recommended)*

| **Group** | **Pharmaceutical** | **RfD** (The National Department of Health of South Africa, 2020) **Adults (mg/kg/day)** | **RfD** (The National Department of Health of South Africa, 2020) **Children (mg/kg/day)** | **Human effects if the RfD is exceeded** (Wexler, 2014) |
|---|---|---|---|---|
| Antipyretics | Acetaminophen ($C_8H_9NO_2$) | 4000 | NR | renal failure, liver failure, non-carcinogenic |
| | Aspirin ($C_9H_8O_4$) | 300 | NR | acute lung injury and pulmonary edema, acute renal failure, acute liver injury, and coagulopathies, non-carcinogenic |
| | Diclofenac | 200 | 4 | Use of diclofenac tablets or capsules frequently or in high quantities, causes the risk of developing a stomach or intestinal ulcer. If there is a tiny chance of developing heart failure or kidney failure. |
| | Ibuprofen | 2400 | 900 | gastritis, renal dysfunction, non-carcinogenic |
| | Ketoprofen | 300 | NR | may result in stomach or intestinal ulcers, bleeding, or holes. |
| Stimulants | Caffeine ($C_8H_{10}N_4O_2$) | 2500 | NR | nausea, vomiting, hematemesis, diarrhoea, and fever, metabolic acidosis, respiratory alkalosis, ketosis, hypokalaemia, and hyperglycaemia, non-carcinogenic |
| Anti-epileptics | Carbamazepine ($C_{15}H_{12}N_2O$) | 1600 | 15 | mouth sores, swollen lymph nodes, persistent vomiting, severe stomach/abdominal pain, yellowing eyes/skin, dark urine, change in the amount of urine, persistent or severe headache, fainting, fast/slow/irregular heartbeat, non-carcinogenic |
| Psychotics | Clozapine | 900 | NR | sedation, dizziness, insomnia, agitation, tardive dyskinesia, dysphoria, dystonic reactions, tachycardia, syncope, anorexia, nausea, vomiting, constipation, diarrhoea, dyspepsia, non-carcinogenic |
| Antibiotics | Ampicillin | 2000 | 800 | severe stomach pain, diarrhoea; blisters, ulcers, or soreness in the mouth; skin rash, redness, or itching; fever, chills, sore throat, swollen glands, joint pain, or not feeling well; pale skin, cold hands and feet, non-carcinogenic |
| | Ciprofloxacin ($C_{17}H_{18}FN_3O_3$) | 1500 | 1500 | neurotoxicity, altered mental status, hematologic effects, bleeding disorders, altered kidney function, crystalluria, increases in liver enzymes, fatal hepatic failure, not carcinogenic |
| | Erythromycin ($C_{37}H_{67}NO_{13}$) | 4000 | 20 | gastric issues, profound cardiovascular toxicity, hepatic damage, not carcinogenic |
| | Metronidazole ($C_6H_9N_3O_3$) | 1500 | 240 | headache, dry mouth, nausea, and vomiting. dizziness, vertigo, encephalopathy, seizures, and ataxia, not carcinogenic |
| | Nalidixic acid | 1200 | 30 | increased intracranial pressure with bulging anterior fontanel, papilledema, and headache, not carcinogenic |
| | Sulfamethoxazole | 480 | 80 | severe stomach pain, diarrhoea; skin rash; yellowing skin or eyes; a seizure; joint pain; increased or decreased urination, not carcinogenic |
| | Sulfamethazine | 200 | NR | not carcinogenic |
| | Trimethoprim | 480 | NR | diarrhoea, nausea, vomiting, stomach upset, loss of appetite, changes in taste, and headache, not carcinogenic |
| Antihyperlipidemic | Bezafibrate | 600 | 160 | dizziness, muscle pain, tenderness, or weakness, not carcinogenic |

*Table 4.4 Concentrations of pharmaceuticals and personal care products presented as mean (SD) (µg/L) detected in selected sub-basins of the uMsunduzi River surface water by Matongo et al. (2015)A and Agunbiade and Moodley (2016)B (ND = Not Detected)*

| **Therapeutic Class** | **Pharmaceutical** | **Subbasin 10** | **Subbasin 8** | **Subbasin 4** | **Subbasin 1** | **Reference** |
|---|---|---|---|---|---|---|
| **Antipyretics** | Acetaminophen | 0.99 (5.35) | 1.29 (0.57) | 1.26 (3.47) | 1.74 (4.35) | A |
| | Aspirin | 13.70 | 14.54 | 13.84 | 25.35 | B |
| | Diclofenac | 0.88 | 8.17 | 0.60 | 2.08 | B |
| | Ibuprofen | 0.55 | 0.66 | 0.70 | 0.45 | B |
| | | 84.6 (6.65) | 27.6 (0.63) | 4.7 (1.43) | 2.58 (0.76) | A |
| | Ketoprofen | 0.39 | ND | 0.45 | ND | B |
| **Stimulants** | Caffeine | 0.11 (3.45) | ND | ND | 3.32 (0.98) | A |
| **Anti-epileptics** | Carbamazepine | 1.26 (7.65) | 3.24 (0.67) | 0.29 (3.95) | 0.32 (2.54) | B |
| **Psychotics** | Clozapine | 8.89 (4.56) | 5.59 (0.33) | 2.18 (0.57) | 2.48 (7.65) | A |
| **Antibiotics** | Ampicillin | 3.68 | 4.05 | 3.87 | 3.21 | B |
| | Ciprofloxacin | 2.63 | 12.99 | 14.33 | 2.40 | B |
| | Erythromycin | 0.06 (13.56) | ND | ND | ND | A |
| | Nalidixic acid | 19.42 | 12.48 | 14.90 | 20.66 | B |
| | Sulfamethoxazole | ND | 1.22 (3.75) | 4.32 (0.56) | ND | A |
| | Sulfamethazine | ND | ND | ND | 1.09 | A |
| | Trimethoprim | 0.29 (0.48) | ND | ND | ND | A |
| **Antihyperlipidemic** | Bezafibrate | 0.23 | ND | 0.31 | ND | B |

### b. Heavy Metals and Nutrients

The heavy metals chosen for risk analysis were primarily determined by the studies referenced, but it is advisable to consider additional heavy metals when assessing risks. The concentrations of heavy metals and nutrients were measured in selected subbasins of the uMsunduzi River surface water by Shozi (2015) and Umgeni Water respectively (Table 4.6). The metals detected by Shozi (2015) were copper, lead, and zinc during a once-off sampling event in September 2013, when one sample was collected per chosen site; these data were presented as a single value in Table 4.6 below. A grab sampling technique was used to collect wastewater or surface water samples from a depth of 1–2 cm from the water surface. As part of Umgeni Water monitoring programme, concentrations of nitrates and soluble reactive phosphates have been quantified in the uMsunduzi River at different sites. The mean concentrations and their standard deviations for nitrates and phosphates (Table 4.6) were calculated over the period between January 1990 and December 2018; sampling frequency varied, with number of samples ranging from 149 to 1490 between sampling stations and chemicals. Some possible human effects due to exposure to these heavy metals and nutrients are listed in Table 4.5 after Wexler (2014).

*Table 4.5 Heavy metals and nutrients in the uMsunduzi catchment; their human effects, cancer slope factors (where applicable), and reference doses are presented below*

| **Chemical** | **Human effects** (Wexler, 2014) | **CSF (1/mg/kg/day)** (USEPA, 1988) | **RfD (mg/kg/day)** (SANS, 2015) |
|---|---|---|---|
| Copper ($Cu^{2+}$) | nonspecific toxic symptoms, a metallic taste, nausea, and vomiting. Not classifiable as carcinogenic to human | n/a | 0.04 |
| Lead ($Pb^{2+}$) | Anorexia, vomiting, malaise, and convulsions. Lead can cause neurological impairment in foetuses and young children (DWAF, 1996). Probable carcinogen | 0.0085 | 0.036 |
| Zinc ($Zn^{2+}$) | Liver damage. Not carcinogenic. | n/a | 0.005 |
| Nitrate ($NO_3^-$) | relative risk of thyroid disorder and goitre rates in pregnant women, not carcinogenic | n/a | 0.36 |
| Phosphates ($PO_4^{3-}$) | reduced lung function, aggravated asthmatic symptoms, and increased risk of emergency department visits, hospitalisations, and death in people who have chronic heart or lung diseases, not carcinogenic | n/a | 0.00002 |

*Table 4.6 Concentrations of heavy metals and nutrients detected in selected subbasins of the uMsunduzi River surface water by Shozi (2015) for heavy metals and Umgeni Water for nutrients presented as mean (SD) (mg/L). (NGV = No Guideline Value and ND = Not Detected)*

| **Chemical** | **Subbasin 10** | **Subbasin 8** | **Subbasin 4** | **Subbasin 1** | **DWAF Guidelines (mg/L)** (DWAF, 1996a) | **WHO Guidelines (mg/L)** (WHO, 2017) |
|---|---|---|---|---|---|---|
| Copper | $5.92 \times 10^{-5}$ | $1.87 \times 10^{-5}$ | ND | - | 30 | 2 |
| Lead | $4.53 \times 10^{-3}$ | $1.47 \times 10^{-4}$ | $5.65 \times 10^{-5}$ | - | 0.01 | 0.01 |
| Zinc | $8.10 \times 10^{-4}$ | $3.13 \times 10^{-4}$ | ND | - | 10 | NGV |
| Nitrates | 0.93 (0.41) | 1.06 (0.49) | 1.85 (2.18) | 2.37 (1.30) | 10 | 10 |
| Phosphates | 0.007 (0.013) | 0.02 (0.04) | 0.21 (0.30) | 0.18 (0.21) | NGV | NGV |

### 4.2.2 Risk Assessment

Studies, locally and abroad, that have quantified chemical risk via ingestion have looked at treated water, while this study assumed no treatment of the water, as this is the normality in the studied communities. The chemical concentrations in the surface waters of the uMsunduzi River used in the risk assessment calculations were based on the following sources: organochlorinated pesticides (Adeyinka *et al.*, 2019), pharmaceuticals and personal care products (Matongo *et al.*, 2015; Agunbiade and Moodley, 2016), heavy metals (Shozi, 2015), and nitrates and phosphates (Umgeni Water monitoring data). Ngubane *et al.* (2022), our earlier study in which we estimated microbial hazards in this watershed, provided the exposed population and the exposure routes employed in this research. However, for the sake of simplicity, only maximum ingestion rates (IR) were used in the current analysis. The exposure routes investigated were direct ingestion of the uMsunduzi River water during recreational swimming, canoeing training, and drinking.

Recreational swimming was considered for subbasins 1, 4, 8, and 10. This is because it was observed during the study that the population along this stretch of the catchment habitually swim in the river during warm periods. The exposed population was categorised into children and adults. The volume ingested during swimming was estimated based on the ranges of 37-47 mL for children and 16-24 mL for adults per 45 minute event as reported by Dufour *et al.* (2006). The values used in the current study were 0.0627 (children) and 0.032 (adults) L/day. Based on observation in the study area, swimming was assumed to take place 50 times a year of an hour's swim, during warm periods.

Canoeing was considered for subbasin 8 as training takes place in this stretch of the river. Training information was obtained from Mr. Z. L. Mthalane (*pers. comm.*, 2019), a coach at two canoe clubs and a seasoned Dusi Canoe Marathon and Dusi Non-stop participant.

Approximately 1000-2000 paddlers enter the Dusi Canoe Marathon annually. The exposed population was categorised into two: children (10 - 18 years old) and adults (>18 years old) based on the competition categories and the required hours for training. Based on the training schedules, 0.5 to 1.5 hours (children), and 1.5 to 2 hours (adults) per day were assumed. Forty days of canoeing were assumed per year. The ingestion volume of 5.8 mL per 45 minute event (Dorevitch *et al.*, 2011) was used as a baseline to estimate the ingested volumes during canoeing events. The values used in this study were 0.0116 (children) and 0.0154 (adults) L/day.

Drinking water was considered for subbasins 10 and 1, which are in the upper and lower rural parts of the catchment, respectively. The exposed population was categorised into children and adults. The daily ingestion volumes for South Africans were adapted from Steyn, Jagals and Genthe (2001) through changing units to fit this study's requirements, and the resulting values used were: 0.773 (children) and 0.952 (adults) L/day.

The risk to develop adverse health effects due to exposure to chemical substances is estimated using hazard quotient (HQ) for harmful non-carcinogenic effects. HQ is the ratio of Chronic Daily Intake (CDI) to Reference Dose (RfD) as calculated using Equation 4.1 (Pieters and Horn, 2020). A value of HQ below 1 means that the exposed population is unlikely to experience adverse health effects, and an HQ value greater than 1 represents a potential health risk to the exposed population (USEPA, 2009). CDI is the potential exposure to a substance, and RfD is the level at which no adverse effects are expected. CDI was calculated using Equation 4.2.

*Equation 4.1*

$$HQ = \frac{CDI}{RfD}$$

Where CDI is the Chronic Daily Intake (mg/kg/day) via ingestion, and RfD is the recommended dose.

*Equation 4.2*

$$CDI = \frac{Cw \times IR \times EF \times ED}{BM \times AT}$$

Where Cw is the concentration of the chemical (mg/L) in ingested water, IR is the ingestion rate (L/day), EF is the exposure frequency (day/year), ED is the exposure duration (years), BM is the human body mass (kg), AT is the average time (days).

In this study, RfD values for OCPs were based on the Agency for Toxic Substances and Disease Registry (ATSDR, 2005, 2007, 2015, 2020); for PPCPs they were based on the South African National Department of Health guidelines as published in the National Department of Health of South Africa (2020); for heavy metals, nitrates and phosphates, RfD values were based on the South African National Standards (SANS, 2015). The body mass (BM) of 66 kg for adults and 35 kg for children was based on the local study by Pieters and Horn (2020); with the corresponding 70 years and 12 years exposure duration (ED). The Exposure frequency (EF) was set at 50, 365, and 40 days per year for swimming, drinking, and canoeing, respectively.

According to WHO (2010), incremental lifetime cancer risk (ILCR) refers to the incremental risk a person faces over a lifetime because of exposure to a given concentration of a carcinogenic agent averaged over a lifetime. ILCR is estimated using Equation 4.3 after WHO (2010). The World Health Organization (WHO, 2008) and several countries worldwide have set their acceptable cancer risk level at $10^{-5}$ for 70 years life expectancy.

*Equation 4.3*

$$ILCR = CDI \times CSF$$

Where, CSF $(mg/kg/day)^{-1}$ is the cancer slope factor and is defined as the risk generated by a lifetime average amount of one mg/kg/day of carcinogenic chemical and is pollutant specific (Qu *et al.*, 2015). In this study, CSF values for chemicals with potential carcinogenic effects were based on various toxicological data as per Table 4.1, Table 4.3, and Table 4.5.

## 4.3 Results

Figure 4.1 shows HQ values representing non-carcinogenic risks of all chemicals calculated using the mean concentrations considering swimming, drinking, and canoeing as exposure routes. Since PPCPs, heavy metals, and nitrates had HQ values below 1, the exposed population is unlikely to experience adverse non-carcinogenic health effects due to ingestion of these chemicals. In contrast, for pesticides, HQ values exceeded 1 for all exposure routes, suggesting possible non-carcinogenic health risks, which may lead to increased risk of hospitalisations and death (Wexler, 2014). The health risks associated with consuming water containing high phosphate levels are typically influenced by the overall water quality and the presence of other harmful substances, rather than being solely attributed to phosphates. For instance, algal blooms are caused by the simultaneous presence of N and P (Chen *et al.*, 2020). Algal overgrowths or the presence of noxious algal species can, however, become a nuisance and interfere with the desirable uses of a water body (DWAF, 1996b). Eutrophication of waterbodies has a negative effect on human health, causing gastrointestinal diseases, dermatological disorders, and conjunctivitis (DWAF, 1996b). For PPCPs, OCPs, and nutrients an upper bound estimate for HQ was also calculated using mean measured concentration plus standard deviation (Figure 4.4), but the same could not be performed for heavy metals due to the lack of data. The HQ increased, overall, with HQ for carbamazepine exceeding 1. This means that, if the variability in the concentrations is considered, as opposed to when only mean values are considered, PPCPs can pose non-carcinogenic health effects to the exposed population.

The results for cancer risk, using the mean concentrations, are shown in Figure 4.3. Apart from endrin, all pesticides have the probability to cause cancer to the exposed population in all subbasins and exposure routes, with cancer risk greater than $10^{-5}$. Dieldrin and aldrin pose the highest risk in all exposure scenarios in subbasins 8 and 10. For the swimming exposure route, the cancer risk trend for OCPs is such that it is the highest in subbasin 10 followed by subbasin 8, subbasin 4, and subbasin 1, in decreasing order. Due to transboundary effects and long-distance transportation, it is possible that OCPs were transported from the upper course of the river to other sampling sites. Similar to the calculations of non-carcinogenic risk, upper bound cancer risk was also calculated using the mean concentration plus standard deviation (Figure 4.4).

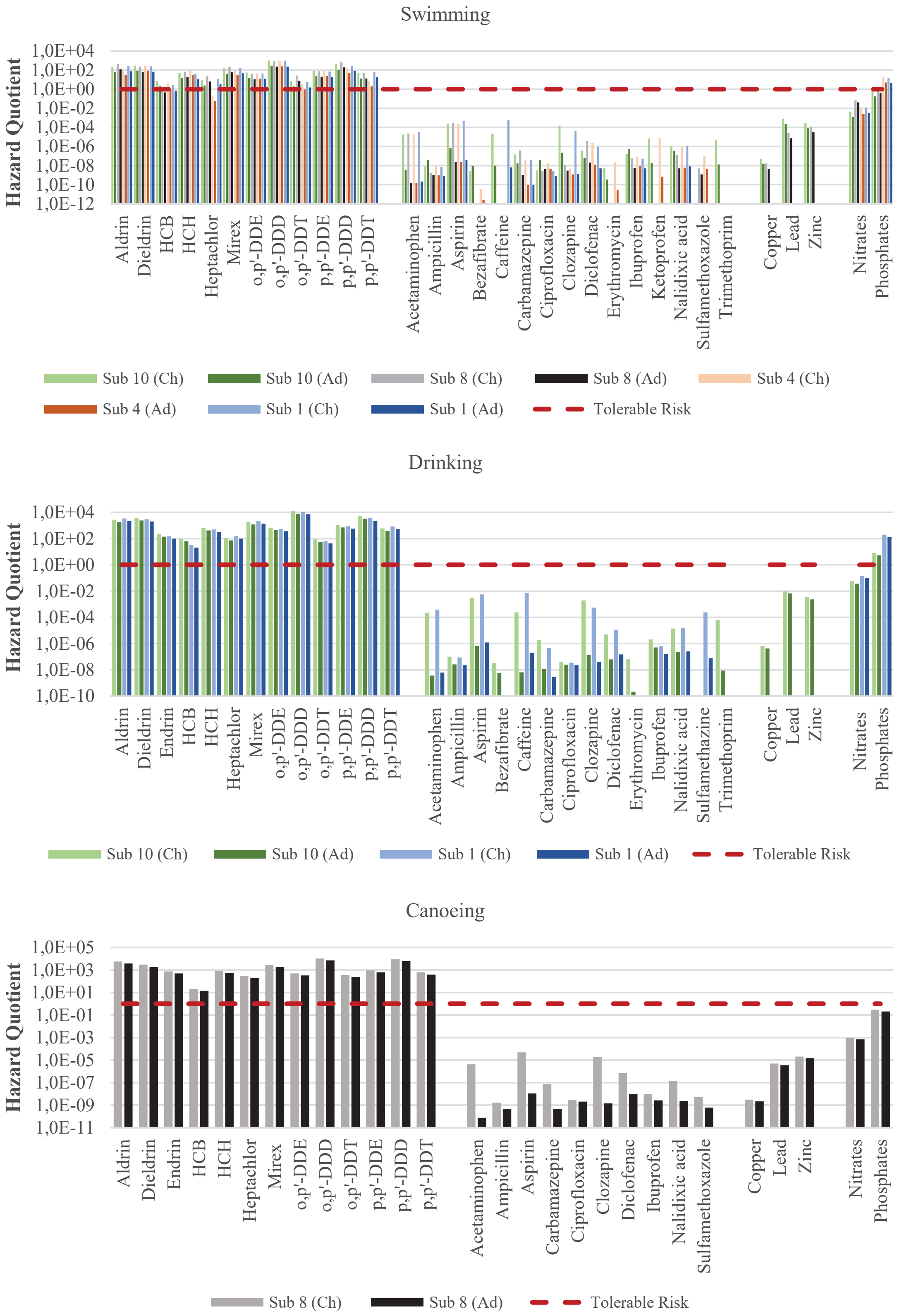

*Figure 4.1 Hazard quotient calculated using the mean concentrations for non-carcinogenic risk during swimming, drinking, and canoeing (top to bottom, respectively) for both children (Ch) and adults (Ad)*

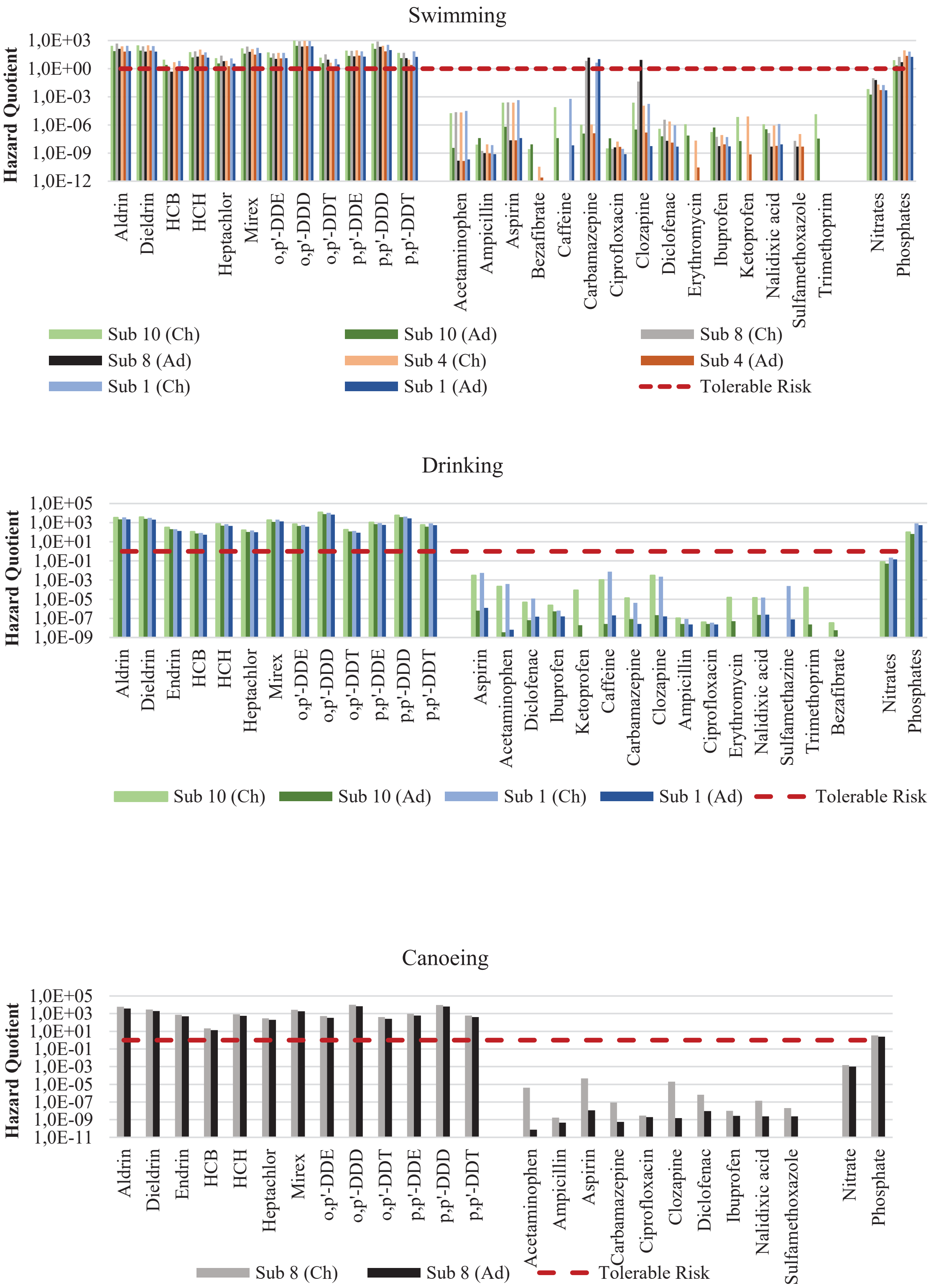

*Figure 4.2 Hazard quotient calculated using the mean plus standard deviation concentrations for non-carcinogenic risk during swimming, drinking, and canoeing (top to bottom, respectively) for both children (Ch) and adults (Ad)*

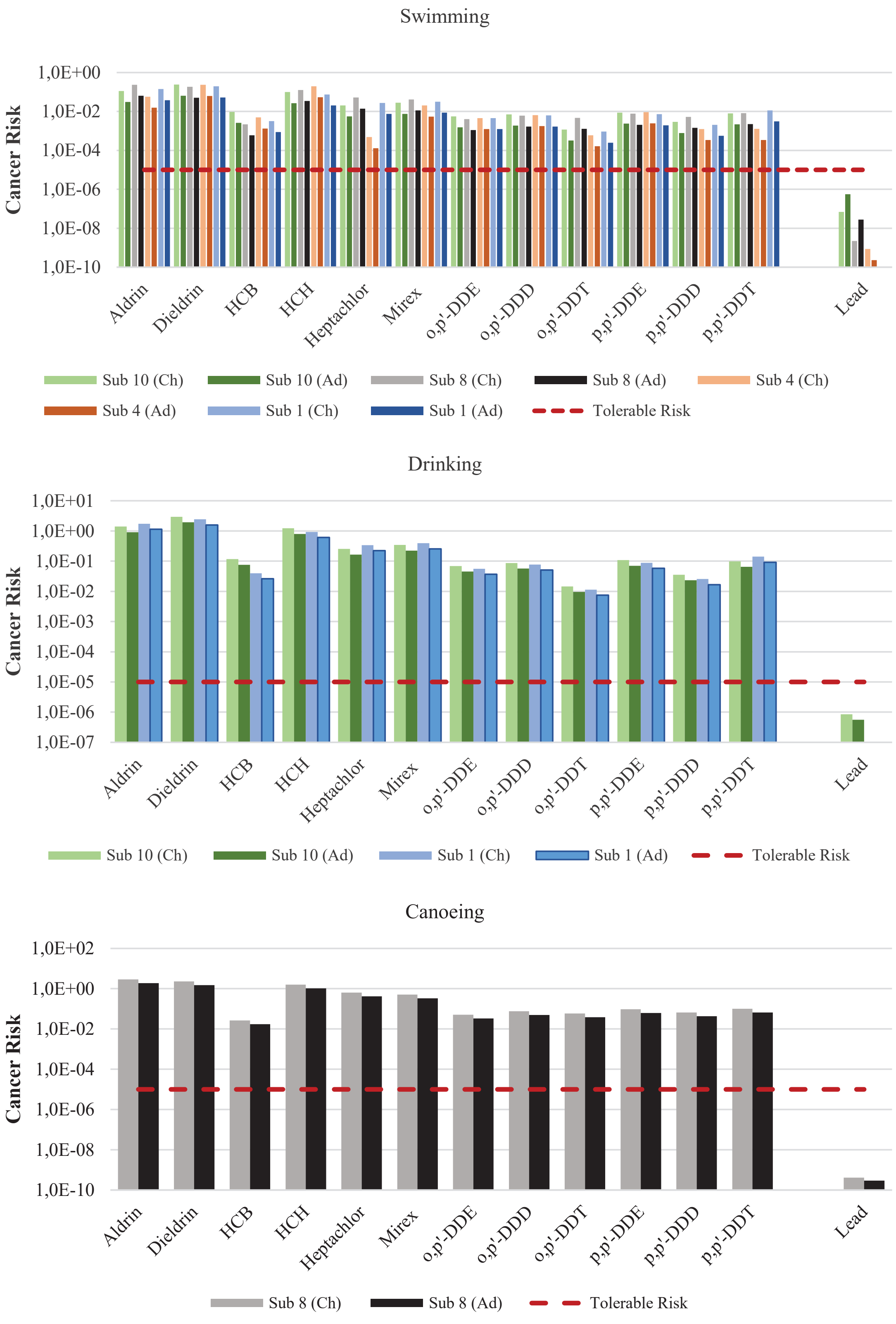

*Figure 4.3 Cancer risk calculated using mean concentrations for carcinogenic risk during swimming, drinking, and canoeing (top to bottom, respectively) for both children (Ch) and adults (Ad)*

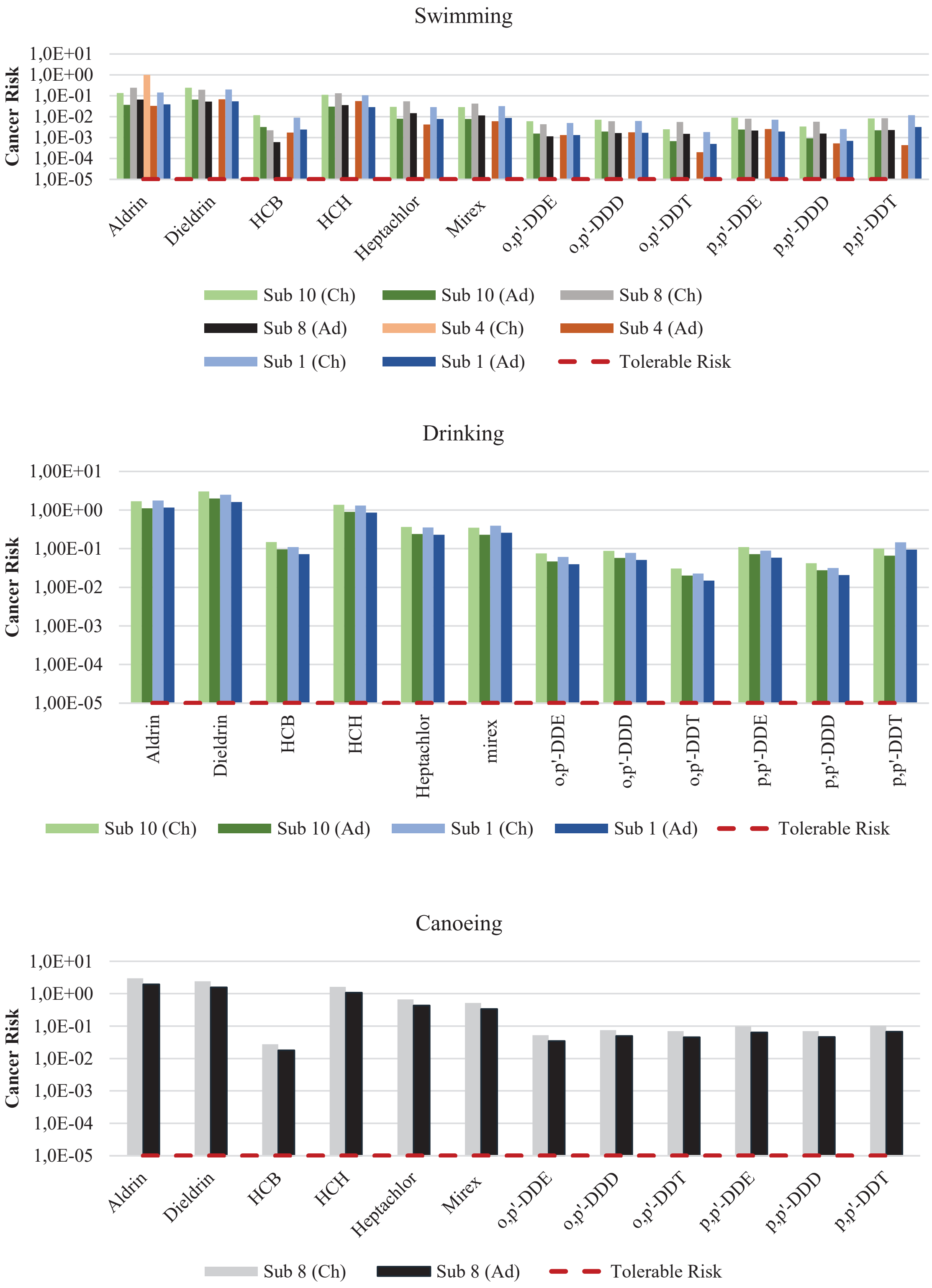

*Figure 4.4 Cancer risk calculated using mean plus standard deviation concentrations for carcinogenic risk during swimming, drinking, and canoeing (top to bottom, respectively) for both children (Ch) and adults (Ad)*

## 4.4 Discussion

Water is an essential part of all lives, for recreational, domestic, and agricultural uses. Oftentimes populations in rural and informal settlements in developing countries rely directly on the rivers for their everyday water uses. This water is generally used without prior treatment. Moreover, in these communities, people use water for recreational purposes. The uMsunduzi River in South Africa is one of the rivers with this kind of backdrop. Subbasins 1 and 10 are both in the rural parts of the uMsunduzi catchment, with subbasin10 in the headwaters and subbasin 1 by the confluence with the uMngeni River, ~15 km from the Inanda Dam. The Inanda Dam supplies Durban Metropolitan Municipality with a population estimated at ~ 3,228,000 with drinking water, and water quality concerns impact the cost of treatment.

### 4.4.1 Key Pollutants and their International Significance

Overall, pesticides and nutrients have shown the highest non-carcinogenic risk in the studied catchment. Moreover, in all four subbasins (1, 4, 8, and 10), pesticides have shown cancer risk potential. Historically, in Africa, agriculture has been reported as the largest polluter, even more so than industries and municipalities (Olowu *et al.*, 2010). Similar to the uMsunduzi catchment, the iSimangaliso Wetland Park and the uPhongolo floodplains, South Africa's largest floodplains and largest wetlands, have been found polluted with agricultural chemicals, further highlighting the threat to the country's biodiversity (Pieters and Horn, 2020). As the country with the largest use of pesticides in sub-Saharan Africa, South Africa has likely misused many of the chemicals that are now classified as endocrine disrupting compounds (EDCs) (Horak, Horn and Pieters, 2021); as evidenced by studies that have detected (anti-)oestrogenic and (anti-)androgenic activity in South African rivers, groundwater systems, and drinking water (Pieters and Horn, 2020). In addition to being a concern to human health, impacts of EDCs have been discovered in wildlife species (Pieters and Horn, 2020). Qu *et al.* (2015) discovered high potential of carcinogenic risk for humans exposed to OCPs in Ningde, Southeast China. In the study by Chen *et al.* (2020) in Shanghai, the ranking of the cancer risk caused by mistaken oral intake of OCPs was such that the risk was higher in adults than children. In the context of current study, the ranking is such that the risk in children is higher than in adults. The highest cancer risk was from drinking in subbasin 10 for children due to dieldrin, consistent with their exposures.

Hazard quotients for PPCPs showed that the exposed population is unlikely to experience adverse health effects due to these chemicals, based on the concentrations used in this study.

In the study by Kong *et al.* (2021), while the non-carcinogenic risk of antibiotics in drinking water was negligible, the ecological risks were high, based on the antibiotics concentrations in surface water around Lake Luoma in the north of Jiangsu province, China. The major concern with PPCPs is the endocrine disruption caused by natural and synthetic steroids and an increase in antibiotic resistance among microorganisms (Manickum and John, 2014). Issues of antibiotic resistance, for instance, have been raised by many studies such as Cizmas *et al.* (2015); Adegoke, Faleye and Stenström (2018), and Ben *et al.* (2019). Additionally, as an ecotoxicological risk, antibiotic residues in aquatic environments may pose threats to a variety of organisms at different trophic levels, with algae considered particularly sensitive to many antibiotics (da Le *et al.*, 2021). The chemicals continually added to the aquatic environment almost become "persistent" pollution (or pseudo-persistent), even if their half-lives are short, because their supply is continually replenished (Hernando *et al.*, 2006; Patel *et al.*, 2019), and PPCPs are an example of this. In the review by Adeleye *et al.* (2022), antibiotics and analgesics are the most frequently detected PPCPs in freshwater. This review by Adeleye *et al.* (2022) also shows that the analgesics represent the majority of the highest PPCP concentrations reported in Africa, including 107 µg/L of acetaminophen in the Ngong River, Kenya; 85 µg/L of ibuprofen in the uMsunduzi River, South Africa; and 62 µg/L of ibuprofen in the Umgeni River, South Africa. Analgesics (such as aspirin) are not recommended for children since they may affect liver functioning (Wexler, 2014; The National Department of Health of South Africa, 2020).

Madilonga *et al.* (2021) performed a risk assessment of heavy metals in the Mutangwi River in the Limpopo Province of South Africa. In that study, the non-carcinogenic risks were found to be lower than 1, for both children and adults, agreeing with a study done in Pakistan by Mohammadi *et al. (*2019) as well as the current study. Similarly, in Songnen Plain, Northeast China, Zhai *et al.* (2017) evaluated the non-carcinogenic risks associated with nutrients, explaining that the risk potential for children is higher than that for adults. There was a significant increase in phosphorus and nitrogen loadings at the Inanda Dam inflow between 2016 and 2020, and this is currently on an upward trend (Umgeni Water, 2022). Due to nutrient enrichment, autotrophic growth occurs on a large scale, which has several consequences, such as biodiversity loss, oxygen depletion, algal toxin production, and taste/odour generation (Oberholster and Ashton, 2008). Freshwater eutrophication is often caused by phosphate enrichment, and phosphate limitation is commonly used to control it (Isiuku and Enyoh, 2020).

### 4.4.2 Sources of Uncertainty and their Impact on Study Results

Even when the quantitative chemical risk assessment indicates that no adverse health effects are likely, it does not mean that the exposure to these chemicals is completely harmless. It is possible to develop new acute/chronic medical conditions or even exacerbate existing chronic conditions due to environmental exposure to pharmaceuticals. Moreover, the mixing of various chemical compounds may create a more toxic mixture than any one compound alone. De Souza *et al.* (2020) assert that a cocktail of pesticides exists in nature and could pose more toxic and adverse consequences to humans and animals exposed to them than a pesticide containing a single component. Such issues as ecotoxicity were not quantified in this study, and more work may be required on that subject. Riva *et al.* (2019) performed an environmental risk assessment of a mixture of emerging pollutants in surface water in a highly urbanised area in Italy using Risk Quotients. Risk Quotients consider the ratio of the expected exposure to the hazards of the mixture. Their findings indicated a potential cumulative risk for the substances that individually could be considered safe, highlighting the importance of taking the whole mixture of pollutants into account.

The chemical risk assessment in the current study was based on assumptions such as the exposure frequency and ingestion rates, which are likely to vary between individuals. Moreover, the concentrations considered in the risk assessment represent the water column and ignore the role of sediments from which the chemicals may re-enter the water column. The pollutant concentrations, with exception of the extensive long-term data for nutrients, are based on a very small number of samples that were gathered during focused campaigns, providing a snapshot in time. While for PPCPs, OCPs, and nutrients an upper bound risk estimate was calculated using the mean concentrations plus standard deviation, the same could not be performed for heavy metals.

### 4.4.3 Exposure Routes and Vulnerabilities

Unequal access to water of reliable quality is both a cause and consequence of poverty in developing regions such as Africa (Olowu *et al.*, 2010). Communities in rural subbasins (subbasins 10 and 1) of the uMsunduzi catchment are the most affected by pollution in the catchment due to their dependence on the river for domestic and recreational use. The highest HQ for children and adults exposed during drinking was found in Subbasin 1 due to exposure to phosphates. These communities are also exposed to a high risk of cancer due to pesticides.

As the resources for risk reduction are limited, it is necessary to prioritise risk-reduction measures by balancing risks, costs, and benefits. Techniques based on ecological engineering are preferable due to their high economic, environmental and ecological benefits, ease of maintenance, and because they are free from secondary pollution (Anawar and Chowdhury, 2020). Constructed wetlands, microbial dosing, ecological floating beds, and biofilm technologies are the most widely applicable ecological techniques (Anawar and Chowdhury, 2020). To control nutrient and chemical loads in catchments, chemical control technologies for agricultural runoff and household wastewater can be used (Shortle *et al.*, 2020). Discussing the challenges and opportunities, including social, policy, institutional, and financial considerations, with all stakeholders will accelerate the adoption of reliable technologies to achieve system-level outcomes (Shortle *et al.*, 2020).

This study makes a unique contribution by employing a quantitative chemical risk assessment methodology that not only considers a wide range of chemical pollutants and exposure pathways but also addresses the nuanced issue of water consumption variability between children and adults. This methodology has not been previously applied to this specific problem or within this study area, thus hindering the development of proper mitigation strategies. Consequently, this research has successfully provided valuable insights into the quantitative characterisation of chemical risks in surface waters.

## 4.5 Conclusions

The population in the catchment of South Africa's uMsunduzi River is exposed to health risks through drinking untreated water from the river (in the rural areas), swimming (in the entire catchment), and canoeing (in the urban area). Organochlorinated pesticides were found to pose elevated cancer risks (except endrin), as well as cause long-term non-carcinogenic effects, in all subbasins. Heavy metals and pharmaceuticals and personal care products occurred at sub-risk levels. Phosphates could have ecological and health impacts, especially near the Inanda Dam. These findings aid catchment managers in prioritising high-risk areas when reducing chemical pollution in the uMsunduzi River.

# Chapter 5 Engaging Stakeholders in Health Risk Management

The contents of this chapter are in the manuscript under review in the Scientific Reports Journal with details as follows: Ngubane Z., Bergion V., Dzwairo B., Stenström T. A. and Sokolova E. 'Multi-criteria decision analysis framework for engaging stakeholders in river pollution risk management', Manuscript ID d93e9148-c762-4950-a14a-6b456827ebb8.

## 5.1 Introduction

The issue of river pollution has become a concern in today's world, with detrimental effects on ecosystems (Anawar and Chowdhury, 2020), human health (Cullis *et al.*, 2019), and overall environmental well-being. The pollution of rivers may pose a significant threat to aquatic life, water quality, and the sustainability of natural resources. Addressing this requires urgent attention and effective measures to mitigate pollution sources, restore affected ecosystems, and ensure the long-term health and viability of rivers. The concept of catchment management strategies has been widely used to alleviate pollution (da Silva Bonifácio *et al.*, 2021). These strategies, known as best management practices (BMPs), may include specific schedules, bans, guidelines, and other measures to prevent or reduce water pollution (Shortle *et al.*, 2020). These BMPs have included structural interventions, for instance, constructed wetlands (Anawar and Chowdhury, 2020), buffer strips, retention ponds (Rohrer, 2014), or porous pavements (Aladesote, 2022). They have also included non-structural BMPs, for instance, animal waste management (WHO, 2013), advanced tillage systems (Shao *et al.*, 2017), nutrient management plans (Lam, Schmalz and Fohrer, 2011; Shao *et al.*, 2017), pesticide management plans (Shastri *et al.*, 2021), planned grazing systems on pasture and rangeland (Lam, Schmalz and Fohrer, 2011), erosion control (Wu *et al.*, 2022), and public education (Lam, Schmalz and Fohrer, 2011). While the structural and non-structural management plans have been well studied, their implementation is often inhibited by limited resources, multiple conflicting criteria, cost effectiveness, and technical feasibility under specific circumstances (Jing *et al.*, 2013; Marttunen *et al.*, 2022).

In order to successfully decrease river pollution, it is crucial to engage a diverse group of stakeholders who have an interest in or are impacted by changes in the catchment area. These stakeholders should come from various sectors, including social, policy, institutional, and financial domains. The collective aim of the stakeholders is to identify the most effective strategies for reducing pollution (Shortle *et al.*, 2020). Effectively utilising the capabilities and dedication of stakeholders can greatly enhance the capacity to safeguard, develop, conserve, and manage water resources (Schreiner, 2013). These stakeholders could be community-based organisations, water user associations, catchment management forums, non-governmental organisations (NGOs), academic and scientific communities, and the private sector (Schreiner, 2013). Adom and Simatele (2022) conducted a study that confirmed the importance of stakeholder engagement in water resource management in South Africa for improving the collective comprehension of the decision and policy making process. This understanding plays a pivotal role in decision-making and has a positive influence on the sustainable management of water resources.

Multi-criteria decision analysis (MCDA) has found extensive application in water resource management. It offers a systematic and transparent approach that can be trusted to acknowledge stakeholder values (Marttunen *et al.*, 2022). For instance, MCDA has been utilised to prioritise vulnerable areas within catchments, evaluate ecosystem services(Marttunen *et al.*, 2022), facilitate stakeholder engagement (Lück and Nyga, 2018), and optimise rainwater harvesting strategies (Ndeketeya and Dundu, 2022). MCDA provides a comprehensive means to evaluate water resource management from a social, economic, technical, and ecological perspectives (Razmak and Aouni, 2014). Given the wide range of available MCDA techniques, it becomes crucial to carefully evaluate the nature of the decision problem, the available data, the decision-maker's preferences, and the unique characteristics of each technique while selecting the most appropriate MCDA approach. Nevertheless, to maintain control and predictability, decision-making procedures in water resource projects have traditionally limited participation of civil society and favoured a small number of specialists (Simpungwe, 2006). This poses a challenge that needs to be addressed to ensure more inclusive and effective water resource management.

In the uMsunduzi catchment in South Africa (comprised of rural, urban, and informal settlements), a variety of water pollutants have been identified, including potentially toxic chemicals (Adeyinka *et al.*, 2019) and pathogens (Msunduzi Municipality, 2022). Furthermore, the risk to human health has been quantified for pathogens (Ngubane *et al.*, 2022) and toxic chemicals (Ngubane *et al.*, 2023) . In these studies, scenarios of ingestion through domestic and recreational uses (swimming and canoeing) were investigated, and the water was found unsuitable for these uses. This highlights the importance of alleviating pollution in the uMsunduzi River to reduce the effects on human health and the environment.

Using the uMsunduzi catchment in South Africa, this chapter aims to provide watershed managers with a framework for selecting BMPs to reduce pollution and the related risk to river users, while also including the perspectives of key catchment stakeholders. The specific objectives were to: 1) identify key stakeholders within the catchment; 2) consult the identified stakeholders to solicit BMPs to alleviate pollution and the BMP evaluation criteria; 3) apply the MCDA methodology to compare and prioritise the BMPs.

## 5.2 Methods

### 5.2.1 MCDA and stakeholder involvement

The MCDA refers to the process of ranking and choosing between alternatives based on multiple criteria or objectives (Razmak and Aouni, 2014). It is a method for systematically comparing the advantages and disadvantages of different alternatives in support of decision making (Esmail and Geneletti, 2018). Furthermore, some applications allocate budgets or other scarce resources among alternatives, to maximise efficiency (Prato and Herath, 2007). Typically, MCDA process consists of the following phases (Saarikoski *et al.*, 2016): (i) problem formulation including the identification of the objectives, criteria and measures for criteria and generation of alternatives; (ii) evaluation of the impacts of the alternatives and creation of a consequence table; (iii) integration of stakeholders' preferences and opinions on the significance of the objective and the weighting of the criteria; (iv) calculation of alternatives' total priorities using software such as Excel, for example; and (v) evaluation of the outcomes, including sensitivity analysis and recommendations.

The Simple Multi-Attribute Rating Technique for Enhanced Stakeholder Take-up (SMARTEST) is an MCDA methodology tailored by Bray (2015) to engage stakeholders in as many stages as possible during the decision-making process, without being onerous. The SMARTEST method was developed from the Simple Multi-Attribute Rating Technique Extended to Ranking (SMARTER) method developed by Edwards and Barron(Edwards and Barron, 1994). The SMARTER method employs a ranking approach to determine criteria weighting, which was considered less burdensome than the Swing method used in earlier versions of the method (Edwards and Barron, 1994). SMARTER uses the rank order centroid (ROC) method to convert ranks into weights, however Roberts and Goodwin (2003) argued that the rank order distribution (ROD) method is a better choice, even though it is complex to calculate. Furthermore, Roberts and Goodwin (Roberts and Goodwin, 2003) suggested the Rank-Sum (RS) method as a simpler approximation. In the SMARTEST, the ranks are therefore converted to weights using the RS method, which is simple and closely agrees with the ROC weighting method used in SMARTER (Bray, 2015).

There were two stages to stakeholder consultation in this study (depicted as 4 & 5 in Figure 5.1). Stage 1 was performed to solicit decision alternatives and evaluation criteria from stakeholders. In Stage 2 stakeholders were asked to refine the alternatives and criteria solicited from them in Stage 1, as well as to rank the criteria. The ranking of the criteria helps ensure that the decision-making process fairly reflects the preferences and values of all stakeholders involved (Bray, 2015).The methodology delineated in Figure 5.1 spanned a duration of eleven months. The data collection for stage 1 survey was conducted during November 2022-March 2023, while data collection for stage 2 survey was conducted during July – August 2023. The data from Stage 1 was analysed before Stage 2 commenced (April – May 2023) in order to share the findings with stakeholders during data collection for Stage 2.

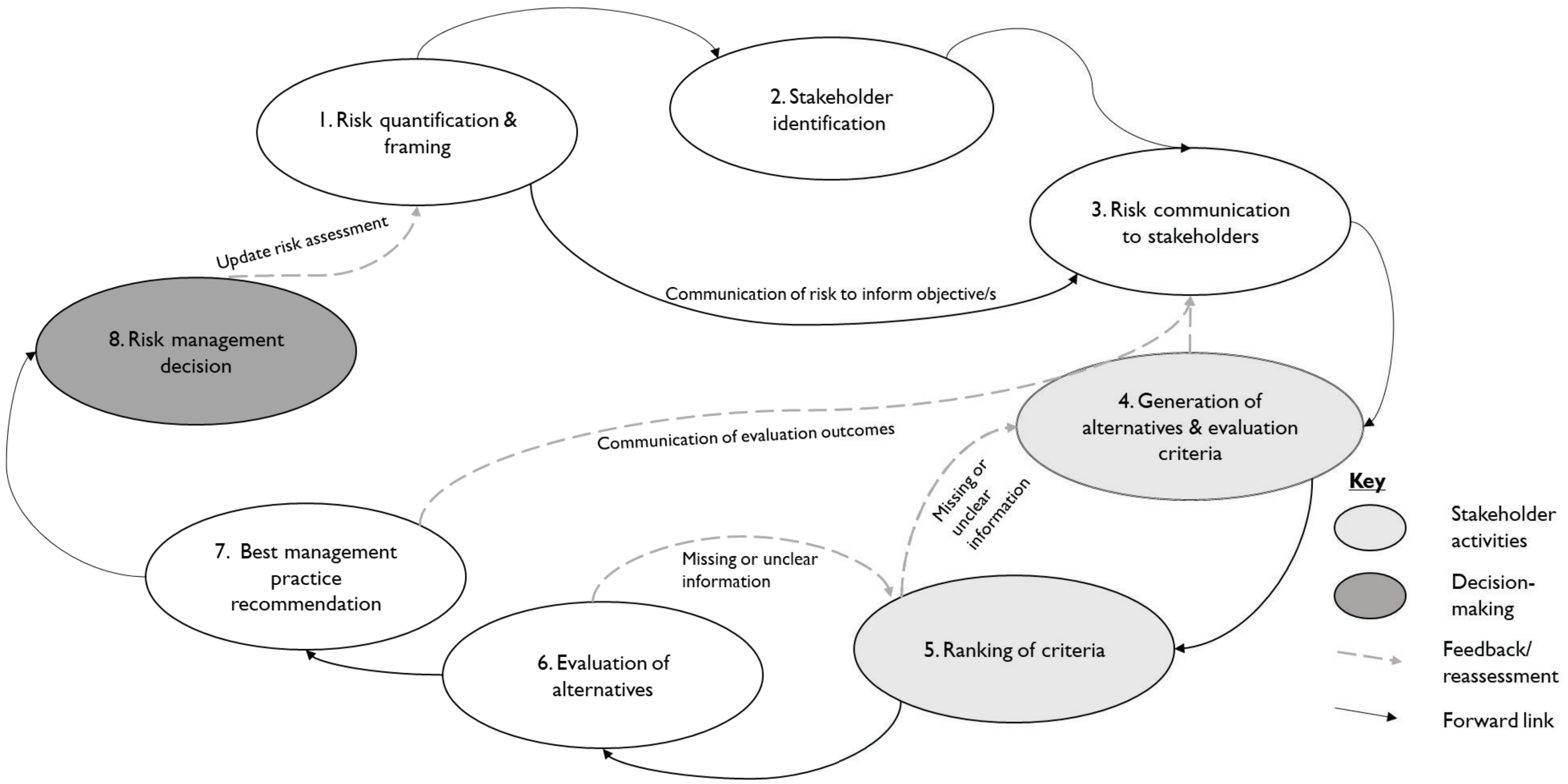

*Figure 5.1. Conceptual framework for stakeholder engagement in the multi-criteria decision analysis process. The clear parts of the framework may be performed without the stakeholders.*

A combination of stratified random sampling (Singh, Masuku and Department, 2011) and purposive sampling (Rai and Thapa, 2019) was used in this study to ensure a representative sample. This approach aimed to capture diverse perspectives from stakeholders across different subbasins of the catchment and levels of involvement in river use and management. Stakeholders identified for participation in this study fall into the following groups as defined by the Department of Water Affairs of South Africa (DWAF, 2001):

- Affected parties: Those that are directly affected by the implementation of the strategy and its outcomes. In this study, these are communities within the uMsunduzi catchment, including professionals such as scientists, engineers, and athletes who use the river. Populace outside of this catchment, was excluded from this study in this category.
- Involved parties: Those involved in catchment management like local government, and those financially and legally involved. Representation of all involved municipalities, including the eThekwini (Durban) Metropolitan were considered in this study. Individuals in this category represented their respective organisations.
- Interested parties: Those who have an interest in broader developments (for example, environmentalists, other developers, and the interested academics. Individuals in this category represented their respective organisations.

The engagement of the identified stakeholders was performed partly with the guide of DWAF (DWAF, 2001), namely: i) informing stakeholders, ii) meeting with stakeholders, iii) feedback to and from stakeholders, and iv) monitoring and evaluation. The different stakeholder groups were informed, and ethical clearance was sought before commencement, as guided by DWAF (DWAF, 2001). The experimental protocol was approved by the local Tribal Councils, the South African Institute of Civil Engineering (SAICE), and the involved municipalities. The Gatekeeper Letter and the Letter of Consent were translated into isiZulu, which is the predominant local language, particularly in rural areas. Prior to data collection, informed consent was obtained from participants. This involved explaining the purpose of the survey, how the data will be used, and any potential risks or benefits associated with participation. The study was carried out in accordance with relevant guidelines and regulations.

### 5.2.2 Development of decision alternatives and evaluation criteria (Stage 1)

In Stage 1 of stakeholder engagement, the primary activities involved soliciting decision alternatives and evaluation criteria. These activities encompassed several key steps: first, requesting consent and obtaining the necessary Gatekeeper letters, with additional clarification provided as needed; followed by a concise presentation of the study to inform stakeholders about identified risks based on QMRA and QCRA findings (Ngubane *et al.*, 2022, 2023). Stakeholders were then provided with a user-friendly survey form, which they could complete at their own pace to ensure their comfort with the process.

This survey form was designed on the Microsoft Forms platform for ease of data collection and collation. The survey link was distributed to participants through materials shared during the meetings and, where applicable, via email. The survey form was designed to not automatically collect information such as names, addresses, and contact details through anonymisation to protect participants' privacy. The raw data remains stored securely for five years and it is password protected to prevent unauthorised access and tampering. This data will be disposed of securely when it is no longer needed. This survey had nine questions, which were designed as a mix of short questions, Likert scale, multiple choice, and long answer questions. The specific questions are shown in Table 5.1. Where applicable and requested, the questions were translated to isiZulu during meetings, making the survey a structured interview instead.

The results of the survey were analysed using Microsoft Excel and IBM SPSS version 29 for descriptive statistics, in preparation for Stage 2. The Relative Importance Index (RII) was used to assess the ranked degree of importance (Tholibon *et al.*, 2021). The RII of each factor *i* was calculated using Equation 5.1 where $r_j$ was $\mathrm{j}^{\mathrm{th}}$ respondents rating of the factor; A was the highest possible rating/ranking; and N was the number of respondents. RII (Equation 5.1) was used to analyse the responses to question 5 and 6 of Stage 1 survey.

*Equation 5.1*

$$RII_i = \frac{\sum_j^N r_j}{A \times N}$$

*Table 5.1. Questions in Stage 1 of stakeholder engagement survey*

| Question | Options/sub-questions |
|---|---|
| 1. Please select the stakeholder group to which you belong. | Affected parties (resident/sport participant within uMsunduzi catchment)<br>Involved parties (municipality) |

| Question | Options/sub-questions |
|---|---|
|  | Interested parties (environmentalists, hydrologists, water utilities, NGOs, NPOs, etc). |
| 2. Contact details |  |
| 3. Please provide the name of the organisation that you represent or the name of the area you live in within the catchment. |  |
| 4. Please select how you or anyone in your household use the river water. You may select multiple options. | Bathing,<br>Swimming,<br>Cooking,<br>Washing clothes,<br>Discard waste,<br>Canoeing,<br>I do not use the river at all,<br>Religious/cultural reasons,<br>Other |
| 5. Please indicate how important the following (indicators) factors are in deciding whether river water quality is good or bad.<br>Colour,<br>Taste,<br>Smell,<br>Presence of aquatic plants in the river,<br>Presence of waste (faeces, plastics) around the riverbank or in the river | Likert options were:<br>Not at all important,<br>A little important,<br>Neutral,<br>Considerably important,<br>Very important. |
| 6. How important is it to you that the water quality in the river improves? | Likert options were;<br>Extremely important,<br>Somewhat important,<br>Neutral,<br>Somewhat not important,<br>Extremely not important. |
| 7. Please enlist possible strategies that you would like to see being adopted to improve water quality. |  |
| 8. What is important to consider when selecting the best strategy? List any parameters that you think can be useful. |  |
| 9. Please use this part to give any other thoughts you have at this stage based on the presented information and the questions above. |  |

### 5.2.3 Ranking and weighing of evaluation criteria (Stage 2)

The aim of Stage 2 survey was to provide feedback on Stage 1 survey to stakeholders and to solicit ranks for evaluation criteria from them. After Stage 1, responses to questions 7 and 8 in Table 5.1 were categorised, coded, themed, and collated for feedback to stakeholders during Stage 2. The stakeholders were given a chance to assess the themed alternatives and add what they feel was omitted or underrepresented or remove what they felt was irrelevant using question 4 in Stage 2 survey. The feedback to stakeholders was done over emails, telephone or in person meetings, based on the stakeholder preferences. The stakeholders were requested to rank the criteria in order of importance from highest to lowest. Table 5.2 shows the seven specific questions that were asked in Stage 2 of stakeholder engagement.

*Table 5.2. Questions in Stage 2 of stakeholder engagement survey.*

| **Question** | **Options/sub-questions** |
|---|---|
| 1. Please select the stakeholder group to which you belong. | Affected parties (resident/sport participant within uMsunduzi catchment)<br>Involved parties (municipality)<br>Interested parties (environmentalists, hydrologists, water utilities, NGOs, NPOs, etc). |
| 2. Contact details | |
| 3. Please provide the name of the organisation that you represent or the name of the area you live in within the catchment. | |
| 4. The following list represents the strategies for pollution alleviation that were listed by the stakeholders in stage 1. Please think about whether you would like to add or remove any of them and indicate that below. | i) Public education and outreach<br>(Livestock grazing management and education on the impact of livestock on the river water and vice versa.<br>Educate people on safe disposal of medication.<br>Fertilizer and pesticide management.)<br>ii) Fixing of sewer system<br>iii) Constructed wetlands<br>iv) Runoff control<br>(Through detention ponds, river buffer etc.)<br>v) Solid waste control<br>(Through recycling, river cleanup etc.) |
| 5. Please select the projects that would be acceptable to you if they were to be chosen as the best strategy. You may select more than one. | i) Public education and outreach<br>ii) Fixing of sewer system<br>iii) Constructed wetlands<br>iv) Runoff control<br>v) Solid waste control |
| 6. When stakeholders were asked to select parameters for selecting the best strategy in stage 1, the following were listed. Please rank them from the most important to the least important. | a) Project funding/capital costs (costs of implementation, operation, and maintenance)<br>b) Socio-economic benefits (potential of job creation)<br>c) Feasibility (can it be done in this catchment?)<br>d) Community acceptance (acceptance by affected stakeholders)<br>e) Aesthetics (potential creation of scenic values)<br>f) Sustainability (pollutant load reduction/potential long-term effects) |
| 7. If you wish to add more parameters for selecting the best strategy, please use the space below. | |

Questions 5 and 6 in Table 5.2 were analysed using the RII method as shown in Equation Equation 5.1 to get the ranks for each factor. Subsequently, to derive the weights from the ranks, the Rank Sum (RS) method was used in this study using Equation Equation 5.2). In this equation, $w_{\text{i}}$ was the weight of the $i^{\text{th}}$ criteria of n, and $r_i$ was its ranking as determined for question 6 in Table 5.2 using the RII value to rank the criteria(Bray, 2015).

*Equation 5.2*

$$w_{\text{i}} = \frac{2\,(n\ +\ 1 - r_i)}{n(n+1)}$$

### 5.2.4 Assessment of decision alternatives

The aggregate scores for decision alternatives were derived through Equation Equation 5.3) (Bray, 2015).

*Equation 5.3*

$$v(a) = \sum w_i\, v_i(a)$$

where $v(a)$ was the aggregate score of alternative $a$, $v_i\,(a)$ was the performance score of alternative $a$ on criteria $i$, and $w_i$ was the weight of criterion $i$ as derived from the RS method in Equation Equation 5.2).

The criteria used to assess the decision alternatives are shown in Table 5.3. The performance scores were compiled using a combination of literature and expert opinion. Literature was used to establish the cost ranges for the criterion “Project funding/capital cost”, and expert opinions were used to assign the performance scores for the criteria “Socio-economic benefits”, “Aesthetics”, “Sustainability”, “Feasibility”, for each decision alternative. The performance scores for the criterion “Community acceptance” were based on Stage 2 survey question 5 in Table 5.2. The criteria were classified into two categories: "benefit" and "cost". Under the “benefit” criterion, a score of 1 was assigned to low, 2 to medium, and 3 to high. Conversely, for the "cost" criteria, high was scored 1 and low was scored 3. Project funding is shown in South African Rands and converted to Euro and US Dollar in brackets based on exchange rates in August 2023 (1 US$ = R19, and 1€ = R20).

*Table 5.3. Evaluation criteria used to assess the decision alternatives.*

| Score definition | Criteria definition | Performance score |
|---|---|---|
| **Socio-economic benefits** | | |
| Low | Minimal increase in job creation | 1 |
| Medium | Moderate increase in job creation | 2 |
| High | Significant increase in job creation | 3 |
| **Project funding/capital costs** | | |

| Score definition | Criteria definition | Performance score |
|---|---|---|
| High | >R20M (1 070 583 USD) (980 088 Euro) | 1 |
| Medium | R10M-R20M | 2 |
| Low | <R10M (535 304 USD) (490 044 Euro) | 3 |
| **Community acceptance** | | |
| Low | Low community support | 1 |
| Medium | Limited community support | 2 |
| High | Widespread community support | 3 |
| **Aesthetics** | | |
| Low | No scenic value potential | 1 |
| Medium | Moderate scenic value potential | 2 |
| High | Significant scenic value potential | 3 |
| **Sustainability** | | |
| Low | No noticeable reduction in pollution levels | 1 |
| Medium | Moderate reduction in pollution levels | 2 |
| High | Substantial and lasting reduction in pollution levels | 3 |
| **Feasibility** | | |
| Low | Requires significant morphological changes | 1 |
| Medium | Achievable with some morphological advancements | 2 |
| High | Easily achievable in current catchment morphology | 3 |

In the context of sensitivity analysis, the robustness of the results was assessed through examining how variations in the criteria ranks could affect the outcome in the decision-making model through scenario analysis. Stakeholders were not a part of this process. Nine random possible rank combinations (RC1-9) were considered as depicted in Table 5.4. The resulting scores of the alternatives were compared with the scores from the original combinations.

*Table 5.4 Rank combinations (RC) used for sensitivity analysis.*

| Criteria | Original | RC 1 | RC 2 | RC 3 | RC 4 | RC 5 | RC 6 | RC 7 | RC 8 | RC 9 |
|---|---|---|---|---|---|---|---|---|---|---|
| Project funding/capital costs | 1 | 2 | 3 | 4 | 5 | 6 | 1 | 1 | 1 | 1 |
| Feasibility | 2 | 1 | 2 | 2 | 2 | 2 | 2 | 2 | 2 | 6 |
| Socio-economic benefits | 3 | 3 | 1 | 3 | 3 | 3 | 3 | 3 | 6 | 3 |
| Community acceptance | 4 | 4 | 4 | 1 | 4 | 4 | 4 | 6 | 4 | 4 |
| Sustainability | 5 | 5 | 5 | 5 | 1 | 5 | 6 | 5 | 5 | 5 |
| Aesthetics | 6 | 6 | 6 | 6 | 6 | 1 | 5 | 4 | 3 | 2 |

## 5.3 Results

During the initial phase (Stage 1) of the study, 39 individuals took part, comprising three stakeholder categories. Specifically, there were 30 affected stakeholders, one representative from a municipality as an involved stakeholder, and eight interested stakeholders. Among the 30 affected stakeholders, nine were from rural areas, seven from the suburban areas, seven from townships, five from informal settlements next to an industrial park, one was a canoe club representative, and one was from an industrial park. Of the eight interested stakeholders, there were NGOs, university members, environmentalists, and civil engineers.

The data collected regarding the river's utilisation demonstrated that affected stakeholders depend on it for a range of activities, including cooking, washing clothes, bathing (the act of washing one's body), swimming, and engaging in religious and cultural practices, as illustrated in Figure 5.2. All participants responded to this question, with the affected stakeholders indicating a wide range of use, the involved stakeholder indicated no use of the river at all, and only two interested stakeholders indicated some use of the river. Moreover, under the "Other" category, affected stakeholders mentioned that their livestock also drink from the river.

When stakeholders were requested to express the significance of certain indicators in determining the quality of river water, they chose the presence of waste in the water or on the riverbanks as the most important factor in deciding whether the quality of water is good or bad, as depicted in Figure 5.3.

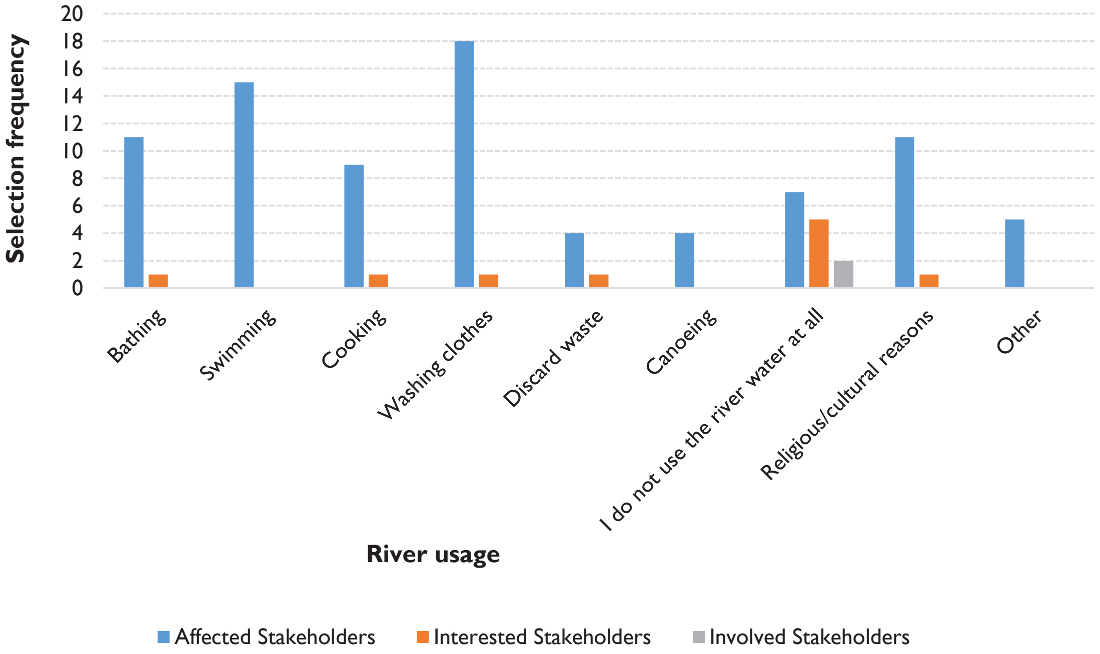

*Figure 5.2. UMsunduzi River usage by stakeholders based on question 4 of survey Stage 1 (39 participants). The number of responses has been presented as the selection frequency on the y-axis expressed in terms of different stakeholders.*

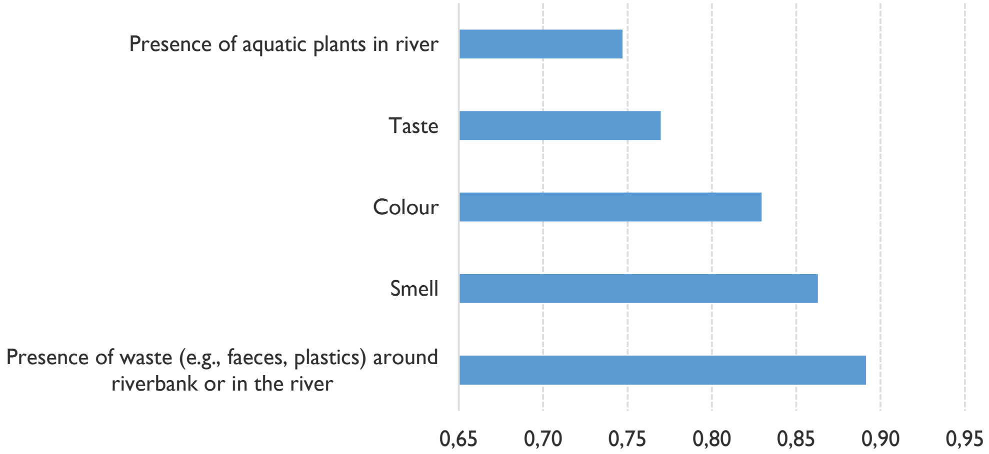

*Figure 5.3. Relative importance index of physical characteristics of water to participants based on question 5 of survey Stage 1 (39 participants).*

During the second phase (Stage 2) of the study, input was obtained from 21 participants, consisting of 17 affected stakeholders, one municipality representative as an involved stakeholder, and three interested stakeholders. Of the 17 affected stakeholders, five were from suburban areas, four from rural areas, five from townships, two from informal settlements, and one was a canoeist representing their canoeing club. Amongst the interested stakeholders, two were from NGOs and one was an environmental scientist representing their respective research and education institution. While the total number of participants decreased in the second stage, the different stakeholder groups were still well-presented.

In Stage 2, stakeholders were presented with a list of alternatives that were identified during Stage 1. When stakeholders were asked to add or remove any of the alternatives (Question 4 of Stage 2 survey), for inclusion, they listed: "*Landowner stewardship; Erosion control measures; Maintenance and management of roads; Livestock exclusion from streambeds; Maintenance of riparian zones to serve as buffers; and Compliance monitoring checks*". No one wanted to remove anything from the list.

Stakeholders were asked in Stage 1 to propose criteria that can be used to evaluate the alternatives. The criteria were grouped as economic, environmental, and social. Overall, six sub-criteria were presented for ranking. Table 5.5 shows the criteria ranks given by stakeholders and the subsequent weightings deduced using the Rank Sum method Equation 5.2).

*Table 5.5. Ranking of evaluation criteria by stakeholders in Stage 2 (21 participants) and the corresponding weights derived through Rank Sum method.*

| Criterion | Ranking | Weighting |
|---|---|---|
| Project funding/capital costs (costs of implementation, operation, and maintenance) | 1 | 0.2857 |
| Feasibility (can it be done in this catchment?) | 2 | 0.2381 |
| Socio-economic benefits (potential of job creation) | 3 | 0.1905 |
| Community acceptance (acceptance by affected stakeholders) | 4 | 0.1429 |
| Sustainability (pollutant load reduction/potential long-term effects) | 5 | 0.0952 |
| Aesthetics (potential creation of scenic values) | 6 | 0.0476 |

To assess the performance of each decision alternative on each evaluation criterion (Table 5.3), a performance matrix was formulated, as detailed in Table 5.6. Public education and outreach as well as runoff control scored lower than the other alternatives on the socio-economic benefit criterion. Fixing the sewers was the only alternative with a high capital cost, while the other alternatives were deemed inexpensive and scored low on this criterion. Most alternatives scored high on community acceptance and environmental criteria, except for constructed wetlands and runoff control, while constructed wetlands was the only alternative scoring high on aesthetics.

*Table 5.6. Performance matrix for decision alternatives for the defined criteria.*

| | **Economic criteria** | | **Social criteria** | | **Environmental criteria** | |
|---|---|---|---|---|---|---|
| **Decision Alternative** | **Socio-economic benefits** | **Project funding/capital costs** | **Community acceptance** | **Aesthetics** | **Sustainability** | **Feasibility** |
| Public Education and Outreach | Medium (2) | Low (3) | High (3) | Medium (2) | High (3) | High (3) |
| Fixing of sewer systems | High (3) | High (1) | High (3) | Medium (2) | High (3) | High (3) |
| Solid waste control | High (3) | Low (3) | High (3) | Medium (2) | High (3) | High (3) |
| Constructed wetlands | High (3) | Low (3) | Low (1) | High (3) | Medium (2) | Medium (2) |
| Runoff control | Medium (2) | Low (3) | Low (1) | Medium (2) | Medium (2) | Medium (2) |

Table 5.7 illustrates the aggregated scores (Equation 3) that account for the weight of the criteria for the decision alternatives, revealing that solid waste control emerges as the top-performing BMP, closely followed by public education and outreach, with sewer system repairs as well as constructed wetlands coming in third. These aggregated scores, based on the

performance score of each alternative and the weight of each criterion, indicate the importance of non-structural BMPs that influence behavioural change within this catchment. In the sensitivity analysis of the MCDA model, the impact of varying criteria ranks on the final scores of the decision alternatives was systematically assessed (Table 5.7). Public education and outreach as well as solid waste control are in the top three alternatives for all sensitivity scenarios, reinforcing the results of the analysis. Solid waste control was the top alternative and runoff control was the bottom alternative in all scenarios, that is, irrespective of the criteria ranks.

*Table 5.7. Aggregated scores of decision alternatives without criteria weights, alongside weights assigned based on stakeholder ranking, and sensitivity analysis of criteria ranking.*

| | | | **Sensitivity analysis of criteria ranking (weights)** | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| **Decision Alternatives** | **Without weights** | **With weights** | **RC 1** | **RC 2** | **RC 3** | **RC 4** | **RC 5** | **RC 6** | **RC 7** | **RC 8** | **RC 9** |
| Public education and outreach | 2.7 | 2.8 | 2.8 | 2.7 | 2.8 | 2.8 | 2.5 | 2.7 | 2.7 | 2.8 | 2.6 |
| Fixing of sewer systems | 2.5 | 2.4 | 2.5 | 2.6 | 2.7 | 2.8 | 2.6 | 2.3 | 2.3 | 2.2 | 2.2 |
| Solid waste control | 2.8 | 3.0 | 3.0 | 3.0 | 3.0 | 3.0 | 2.7 | 2.9 | 2.9 | 2.8 | 2.8 |
| Constructed wetlands | 2.3 | 2.4 | 2.3 | 2.4 | 2.1 | 2.2 | 2.4 | 2.4 | 2.6 | 2.4 | 2.6 |
| Runoff control | 2.0 | 2.1 | 2.1 | 2.0 | 1.9 | 2.0 | 1.9 | 2.1 | 2.2 | 2.1 | 2.1 |

## 5.4 Discussion

In the context of the uMsunduzi catchment in South Africa, this research sought to offer watershed managers support for the selection of BMPs aimed at mitigating pollution and associated risks to river users. This was achieved by applying the comprehensive framework (Figure 5.1) that incorporates the valuable perspectives of catchment stakeholders. In this study a framework for stakeholder engagement in pollution alleviation and risk management was proposed and exemplified for uMsunduzi River in South Africa. This focus on stakeholder involvement in decision-making, was advocated by Sharpe, Harwell and Jackson (2021) to underscore the significance of local community participation in preventing project immobilisation and improving decision-making. This approach further aligns with the Department of Water Affairs of South Africa's (DWAF, 2001) categorisation of water resource management stakeholders into affected, involved, and interested parties, prioritising those directly affected by development projects, local government, and stakeholders with broader interests. Furthermore, Sharpe, Harwell and Jackson (2021) proposed comprehensive criteria for stakeholder selection, considering factors such as level of interest, influence, impact, probability, proximity, economic interest, rights, fairness, and underrepresented populations. In the current study, this DWAF approach provides a contextually appropriate and detailed stakeholder categorisation. However, there are limitations related to stakeholder participation, evident from the reduced number of participants in Stage 2, which may reflect their areas of interest and available time, emphasising the need to balance stakeholder expectations with project timelines. Nevertheless, the achieved sample size, coupled with the sampling strategies implemented, facilitated a multifaceted comprehension of stakeholder perspectives. This exemplifies the breadth and depth of insights necessary to achieve the study's objectives.

Care was taken in the study's approach towards stakeholder involvement by ensuring that participants could engage comfortably by offering survey instructions in English and facilitating discussions with stakeholders in rural areas, townships, and informal settlements in isiZulu, fostering a deeper understanding of scientific evidence and instructions. This approach aligns with local cultural context and enhances inclusivity as emphasized by Behr(Behr, 2018). If this methodology were applied on a larger scale, translating all materials into the relevant language would be advisable to further enhance understanding and response rates, ultimately promoting more effective and meaningful stakeholder participation. Although this study maintained isolation between different stakeholder groups, future research could benefit from integrating workshops guided by a moderator in the consultation process. Workshops have the potential to expose participants to new information and perspectives from other stakeholders, broadening their views and positively influencing their preferences, in line with findings by Marttunen, Mustajoki and Dufva (2013) . Additionally, setting up a structured feedback mechanism from stakeholders to assess their perceptions of the process, as suggested by Lück and Nyga (2018), can further enhance the engagement approach.

The Stage 1 survey findings shed light on the factors that respondents consider most critical in assessing water quality. Notably, solid waste and faecal matter emerged as the top concerns, with taste ranking as the least important in their collective assessment. This aligns with the research conducted by Okumah, Yeboah and Bonyah (2020) in Ghana, which underscores the significance of clean and hygienic surroundings for water resources. In such environments, free from solid and liquid waste, water is more likely to be perceived as safe for consumption and recreational activities. However, a contrasting perspective was revealed by Rangecroft *et al.*

(2023) in their study of water quality perceptions in the Santa basin, Peru. Their research highlighted the pivotal role of organoleptic properties, such as taste, smell, and visual aspects, in shaping local perceptions of water quality, along with traditional ecological knowledge and water usability. This striking difference underscores the notion that local perceptions of water quality are intricately linked to the specific uses and cultural contexts of the water.

In the Stage 1 survey, Likert scales were used to assess the factors influencing river water quality (Question 5) and the importance of improving water quality (Question 6). The Likert scales differed due to oversight during the survey creation and during the piloting of the survey. Despite diverse stakeholder perspectives on river utilisation and the identified inconsistency with the questions, all participants unanimously agreed on the river's inadequate condition, highlighting a consensus on the urgency of improving water quality. The possible alternatives brought up by stakeholders were public education and outreach, fixing of sewer system, constructed wetlands, runoff control, and solid waste control. The identification of alternatives for the decision problem should be based on case-specific information (Freitas and Magrini, 2013). Authors judge that the stakeholder responses in the uMsunduzi case all provide potentially sustainable solutions that can satisfy the problem objectives. Even though there may be additional alternatives that would meet the objectives, the authors concluded that the derived alternatives presented a wide enough range of possible solutions. The evaluation criteria selected by stakeholders could be grouped as project funding/capital costs (costs of implementation, operation, and maintenance), socio-economic benefits (potential of job creation), feasibility (can it be done in this catchment?), socio-economic benefits (potential of job creation), sustainability (pollutant load reduction/potential long-term effects), and aesthetics (potential creation of scenic values).

Solid waste management emerged as the preferred BMP, aligning with the prominent water quality indicator, presence of waste (faeces, plastics) around the riverbank or in the river, identified by stakeholders in Stage 1. Effective solid waste management, sometimes challenging (Idowu *et al.*, 2019; Ayeleru, Okonta and Ntuli, 2021), can be an important part in mitigating water pollution (Ayeleru, Okonta and Ntuli, 2021; Bangani *et al.*, 2023; Sahoo and Goswami, 2024). Challenges include increasing waste volumes, financial constraints, limitations within existing containment systems (Idowu *et al.*, 2019) as well as limited public awareness about recycling, a shortage of engineering expertise in waste management, substandard service delivery, and a lack of effective educational campaigns (Ayeleru, Okonta and Ntuli, 2021). In essence, while addressing solid waste is important to promote environmentally friendly practices, it is essential to recognise that dealing with chemical and microbial pollution requires additional efforts beyond solid waste management alone.

Interestingly, the second management plan was public education and awareness, highlighted as vital for addressing water pollution (Sahoo and Goswami, 2024). Osawe, Grilli and Curtis (2023) proposed community-based initiatives for behavioural change to address declining water quality. This aligns with the study by Brehm and Eisenhauer (2021), emphasising the positive impact of public education in pollution mitigation, underscoring the growing importance of community engagement and collaboration in addressing water pollution. It is imperative for public education to encompass subjects such as livestock management, given its established role in faecal pollution and the resulting microbial health risks for humans (Ngubane *et al.*, 2022). Moreover, the proper disposal of medical waste requires control and public education can play a significant role, possibly with involvement from the pharmaceutical industry.

Another prominent management plan involves repairing and maintaining the sewer systems. Research has emphasised insufficient wastewater treatment plant performance in South Africa, underscoring negative impact on the environmental and public health consequences(du Plessis, 2019, 2023). In Ngubane *et al.* (Ngubane, Bergion and Sokolova, 2022) human faecal sources were found to be more prominent in urban areas owing to the major contributions from wastewater infrastructure. A report by South African Institute of Civil Engineering(SAICE, 2022) further emphasised the pressing need for sewer system improvements, as 34% of sanitation systems face high or critical risk of failure. South Africa's water infrastructure challenges are evident, with just 40% of wastewater effluent meeting microbial water quality standards and 23% of wastewater effluent meeting chemical water quality standards(SAICE, 2022). Specifically, the uMsunduzi sewer pipeline, which spans over 1450 kilometres, has aged considerably, with approximately 60% of it being between 30 to 50 years old (Msunduzi Municipality, 2022).

Constructed wetlands mimic natural ecosystems in purifying water, preserving water quality, and providing habitats for wildlife and recreation (Sahoo and Goswami, 2024). Research has shown that constructed wetlands efficiently remove a wide range of pollutants, including nutrients, heavy metals, organic compounds, and PPCPs (Salah *et al.*, 2023). These pollutants have been detected in the uMsunduzi River with their human health risk quantified by Ngubane *et al.* (Ngubane *et al.*, 2023). The last management plan involves runoff control measures, including detention ponds, designed to reduce stormwater peak flow and capture sediment and nutrients (Kaini, Artita and Nicklow, 2012; Rohrer, 2014). Key processes for reducing pollutants are sedimentation and biological processes (Rohrer, 2014), and these ponds help manage stormwater surges by delaying runoff to nearby rivers (Kaini, Artita and Nicklow, 2012). The effectiveness depends on factors like size, retention time, and local rainfall intensity(Kaini, Artita and Nicklow, 2012). Detention ponds have been reported to reduce total nitrogen, nitrate, ammonium, total phosphorus, orthophosphate, and faecal coliform bacterial counts (Kaini, Artita and Nicklow, 2012; Rohrer, 2014).

The order of the decision alternatives with and without accounting for stakeholder ranking of the evaluation criteria is very similar (Table 5.7). Moreover, the sensitivity analysis demonstrated that while the top (solid waste management) and the bottom (runoff control) alternatives remained the same, the order of the other alternatives may change depending on stakeholder ranking of the criteria. As suggested by Giupponi and Sgobbi (2013), when variations in factor weights during sensitivity analysis do not result in substantial alterations in the model's outcomes, it can be inferred that the model yields more objective and consistent results. In this study the top alternative remains the same, regardless of sensitivity scenarios. Despite the acknowledged limitations of the case study, the model offers a comprehensive perspective on the impacts being examined.

The choice of solid waste management as the stakeholders' primary decision alternative aligns with its immediate practical importance, while the emphasis on education (stakeholders' secondary choice to improve the uMsunduzi water quality) acknowledges the need for a sustainable, long-term solution. These results should guide the development of comprehensive BMPs that combine practical infrastructure improvements with educational initiatives (Bergion *et al.*, 2020). Additionally, broadening the stakeholder base and refining the methodology through using workshops for stakeholder engagement could further enhance decision-making in this context. Future use of this framework within the study area should consider if criteria and alternatives proposed here are complete, or whether additional criteria/alternatives need to be developed.

The choice of stakeholders, their representation, and the level of their engagement can influence the weighting of criteria, the evaluation of alternatives, and the final decision outcomes (Intelisano *et al.*, 2022), or even the choices of criteria and alternatives. To overcome parts of subjectivities and provide less biased information, one option is to integrate cost-benefit analysis in future studies to diversify and broaden the available decision support. This addition, as presented in, for example, (Bergion *et al.*, 2018), could provide more comprehensive economic criteria for MCDA and facilitate comparisons between different mitigation measures. Specifically, the use of social cost-benefit analysis was suggested to assess the societal benefits of individual projects and identify the most economically viable pollution alleviation strategies (Bergion *et al.*, 2018). The social cost-benefit analysis additionally considers the non-financial effects (Bergion *et al.*, 2018).

The current study introduces a novel risk management framework, depicted in Figure 5.1, which fills a crucial gap in literature (Akdogan and Guven, 2023) by integrating risk assessment within the MCDA framework for watershed-scale pollution. This innovative approach lays the groundwork for a more holistic and robust strategy in pollution risk management. With the focus on developing countries with diverse land uses within catchments, the framework underwent practical testing in the uMsunduzi catchment, South Africa, with potential global applicability. To ensure adaptability and transferability, key aspects such as assessing data availability and collaborating with local stakeholders are paramount. Furthermore, the framework's flexibility allows for accommodation of environmental variations and regulatory frameworks.

## 5.5 Conclusions

In the context of the uMsunduzi River catchment in South Africa, this study aimed to provide a framework for watershed managers to select effective BMPs for pollution reduction, addressing the associated risks to human health, while incorporating the perspectives of key stakeholders. Multi-criteria decision analysis using the SMARTEST method proved to be a valuable tool for enhancing stakeholder engagement in collaborative risk management. Furthermore, the study's outcomes are based on the perspectives of stakeholders at the time of data collection and may evolve over time. These limitations highlight the need for tailored approaches when applying the study's recommendations in other regions or considering long-term sustainability.

The MCDA results revealed that solid waste control emerged as the top-ranked BMP, closely followed by public education and outreach. This study provides a valuable novel approach for informed decision-making in water quality management incorporating local stakeholder perspectives, with the potential for broader application in similar contexts. The framework offers a robust foundation for developing strategies to enhance river water quality and reduce pollution, emphasising the importance of stakeholder involvement, effective BMP selection, and health risk management in environmental sustainability efforts.

# Chapter 6 Synthesis

This synthesis serves to show that the objectives of this study were not isolated activities but rather intricately linked components of a framework. Figure 6.1 represents a graphical view of the framework that is proposed by this study. The hazard identification and modelling informed the risk assessments, which, in turn, guided the formulation of evaluation criteria within the decision analysis. In concert, hazard identification, risk assessment, and multicriteria decision analysis have forged a comprehensive decision support framework that transcends disciplinary boundaries, allowing for the holistic management of water quality within catchment areas. This integration stands as a testament to the power of interdisciplinary approaches in addressing complex environmental challenges and positions this study at the forefront of informed and sustainable decision-making practices.

In this study, a unique contribution is made by incorporating both chemical and microbial risk assessment. Furthermore, the risk assessment methodology not only encompasses various chemical pollutants and exposure pathways but addresses the nuanced issue of water consumption variability between children and adults. This chapter is dedicated to the study's objectives, general conclusions, the limitations encountered, and recommendations derived from the findings.

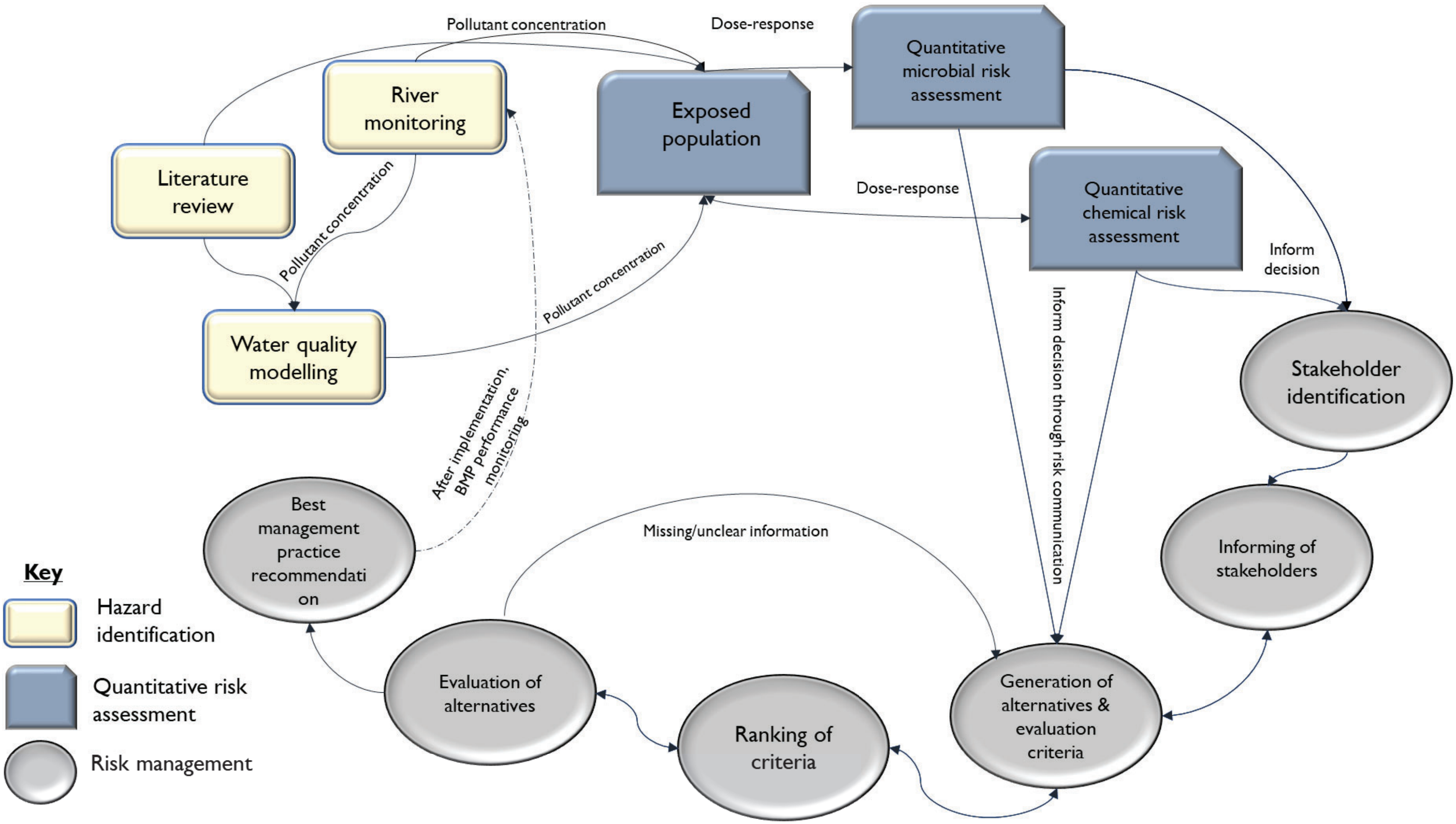

*Figure 6.1 Multi-criteria decision-support framework for improving water quality to assist with engineering infrastructure and catchment management*

## 6.1 Hazard Identification

The sustainability of ecosystems and human health are at risk due to water pollution, which has negative effects on the quality and accessibility of water resources (Adu and Kumarasamy, 2018). While the presence of pollution in the uMsunduzi catchment area was previously acknowledged through various studies (Gemmell and Schmidt, 2013; Matongo *et al.*, 2015; Adeyinka *et al.*, 2019; Mdletshe, 2019; Msunduzi Municipality, 2022), this study sought to deepen the understanding of how it impacts human health. This commenced with hazard identification of pathogens and potentially toxic chemicals in the river water through literature review and perusal of Umgeni Water's water quality monitoring data. This was performed in both Chapter 3 and Chapter 4.

To identify the hazards, the current research combined water quality modelling with an extensive review of published data sources, including journal articles, theses, and dissertations. This concept aligned with research conducted by scholars like Talukdar, Kumar and Kulkarni (2023), which underscored the importance of combining river water quality monitoring with water quality models to achieve a more thorough assessment of water quality. This approach assists decision-makers in addressing water-related challenges (Baffaut and Benson, 2009; Mander *et al.*, 2017; Abdullah *et al.*, 2019), making catchment-scale water quality models an indispensable tool for comprehending, evaluating, and forecasting the detrimental consequences of pollution on river water quality.

In Chapter 3, the study utilised the Soil and Water Assessment Tool (SWAT) to simulate the transport and fate of *Escherichia coli* (*E. coli*) and *Cryptosporidium*, chosen as indicators of faecal and protozoan pathogens, respectively, in the uMsunduzi catchment. Limaheluw, Medema and Hofstra (2019) found that surface water consumption in sub-Saharan Africa may cause an estimated 43.1 million annual cryptosporidiosis cases. To gain insights into microbial pollution in rivers, typically, indicator organisms like faecal coliforms and *E. coli* are employed to estimate the presence of both pathogenic and non-pathogenic microorganisms (Iqbal and Hofstra, 2019a). This is because assessing *E. coli* is a more cost-effective and time-efficient method compared to other pathogenic microorganisms like *Cryptosporidium* and *Rotavirus* (Iqbal and Hofstra, 2019b, 2019a).

To effectively manage water resources, it is necessary to assess the water quality and identify the sources of pollution (Vermeulen *et al.*, 2019; Talukdar, Kumar and Kulkarni, 2023). The model effectively identified pathogen sources in different settlements, emphasising the prevalence of animal sources upstream of the city and the greater impact of human sources downstream. This highlights the influence of livestock farming and urbanisation on water quality. Regarding *Cryptosporidium*, the model indicated higher concentrations downstream of the wastewater treatment plant. However, limited data made direct comparisons with observed data challenging. These findings highlight the role of animals in rural areas and the significance of human sources in urban areas, primarily due to wastewater infrastructure. This aligns with Hofstra *et al.* (2019) observation that human sources typically contribute more to pollution than animal sources in most regions.

Nevertheless, this study had limitations in assessing microbial pollutants. These include the approximation of *Cryptosporidium* loads from the wastewater treatment plant, the estimation of broken sewer contributions, and data constraints on pathogen parameters. Additionally, the impact of pit latrines in townships and rural areas on surface waters was challenging to quantify. The study also did not account for microbial contributions from direct stream deposits, wildlife, and open-space defecation due to difficulties in quantification within the catchment. These limitations underscore the complexity of assessing microbial pollutants in a real-world context.

Additionally, literature has highlighted the deterioration of wastewater treatment facilities in developing countries, such as South Africa, and their contribution to deteriorating river quality (Medema and Schijven, 2001; WRC, 2002; du Plessis, 2019; Edokpayi, Enitan-folami and Adeeyo, 2020; SAICE, 2022). These include the lack of sufficient wastewater treatment infrastructure, operational deficiencies, design flaws, a shortage of qualified personnel, corruption, funding constraints, overburdened systems, and inadequate monitoring for compliance with environmental standards. These challenges require a multifaceted approach, involving government policies, public-private partnerships, and community engagement, to enhance wastewater treatment and environmental sustainability in these regions.

For chemical pollutants, literature was used to supplement river monitoring data, covering various pollutants such as organochlorinated pesticides (OCPs), pharmaceuticals and personal care products (PPCPs), heavy metals, nitrates, and phosphates. These findings were evaluated for quantitative chemical risk assessment (QCRA) input, emphasising potential adverse effects and the need for future water quality modelling where sufficient data is available. Water modelling, as illustrated by Wang *et al.* (2019), offers a cost-effective and comprehensive approach for understanding chemical transport and fate at the watershed scale, benefitting researchers and stakeholders in catchment management.

## 6.2 Risk Quantification

Risk quantification was performed in Chapter 3 & 4. It is important to recognise that while risk quantification alone cannot completely eliminate risks or address trade-offs (Aven, 2011), it does provide valuable insights for principled risk management and reduction. The focus on risk assessment in the current study has been extensive, encompassing both chemical and microbial risk assessment for domestic and recreational uses of the river by both adults and children. Research, such as that by Bergion *et al.* (2018) and Cantoni *et al.* (2021), has delved into risk-based cost-benefit analyses and assessments for emerging pollutants in drinking water supplies. However, the application of risk assessment to fresh water used for recreation remains relatively limited, with only a handful of studies, including Peng *et al.* (2016) and Ahmed *et al.* (2018), exploring this aspect. Consequently, this study's examination of exposure routes for drinking, recreational swimming, and non-competitive canoeing contributes a unique perspective to the field.

To ensure water safety, quantitative data is indispensable, encompassing pollutant concentrations and their behaviour within water sources and treatment processes (Galal-Gorchev, 1993). Researchers such as Kim, Kabir and Jahan (2017); Bergion *et al*. (2018); Federigi *et al*. (2019); Maleki and Jari, (2021) have underscored the significance of quantitative insights for a comprehensive understanding of water quality and safety. The risk assessment in this study considered no water treatment before use, mirroring actual conditions within the uMsunduzi catchment. It encompassed a diverse target population, including adults and children, recognising differences in water consumption and activities.

Quantitative microbial risk assessment (QMRA) was performed for *Cryptosporidium* and pathogenic *E. coli* in Chapter 3, while quantitative chemical risk assessment (QCRA) was conducted for OCPs, PPCPs, heavy metals, nitrates, and phosphates in Chapter 4. Owens *et al.* (2020) conducted a comprehensive review of 39 international studies on QMRA, focusing on pathogen transmission from source water to consumers through water supply systems.

*Cryptosporidium* emerged as the most frequently used reference pathogen, with *Giardia* being the sole other protozoal reference pathogen considered. Among bacterial reference pathogens, pathogenic *E. coli* was the most commonly utilised (Owens *et al.*, 2020). The results of the QMRA for *Cryptosporidium* and pathogenic *E. coli* in the current study revealed that the probability of infection for most users exceeded the acceptable levels outlined in the South African Department of Water Affairs water quality guidelines (DWAF, 1996a, 1996b) and the World Health Organization (WHO, 2017, 2021) for both drinking and recreational activities. The findings showed elevated risks, emphasising the vulnerability of individuals consuming river water, particularly in rural and peri-urban communities with limited access to treated water.

Various studies have employed probabilistic risk assessment methods to evaluate chemical risks in different water sources. These assessments have considered risks associated with pollutants like heavy metals (Muhammad, Shah and Khan, 2011; Anyanwu and Onyele, 2018; da Silva Bonifácio *et al*., 2021), OCPs (Chen *et al*., 2020), and PPCPs (Hernando *et al*., 2006; Sengar and Vijayanandan, 2022) in both surface water and groundwater systems. In the current study OCPs presented elevated carcinogenic risks across the entire catchment, along with persistent non-carcinogenic pesticide effects. Notably, Pheiffer *et al.* (2018) estimated both carcinogenic and non-carcinogenic OCP effects through fish consumption, highlighting significant risks to human health in the Klip River catchment in Gauteng, South Africa. It is noteworthy that urban areas are emerging as significant users of pesticides in Africa, underscoring the urgency of risk management. This was most likely attributed to long distance transport and historic production and formulation of OCPs in the areas and use of insecticides (Pheiffer *et al.*, 2018; Pieters and Horn, 2020).

While heavy metals and PPCPs were found to be present at sub-risk levels in the current study, phosphates raised concerns due to potential ecological and health impacts, particularly in the Inanda Dam area. The ecological and health implications of antibiotic residues, including their potential to disrupt the endocrine system and enhance antibiotic resistance among microorganisms, were highlighted. In a global review on the abundance, fate, and effects of pharmaceuticals and personal care products (PPCPs) in aquatic environments, Adeleye *et al.* (2022) identified antibiotics and analgesics as the most frequently detected PPCPs in freshwater. In a separate study, Madilonga *et al.* (2021) conducted a risk assessment of heavy metals in South Africa's Mutangwi River, revealing predominantly low risks. However, it is essential to acknowledge that specific areas within the uMsunduzi catchment exceeded recommended risk thresholds.

The risk assessment approach in this study focused on exposure per event for risk calculations and did not address event frequency. Extending this approach to annual risk assessment would necessitate considering factors such as the frequency of drinking river water, canoeist training schedules, and seasonal variations in recreational swimming. Nonetheless, the findings emphasise the importance of proactive risk management measures. Quantitative risk assessment, therefore, remains a cornerstone of effective risk management and the implementation of mitigation measures. Stakeholder perspectives and decision priorities should be considered for continuous improvement and remedial action.

## 6.3 Risk Management

Following the quantitative risk assessment, this study utilised a multi-criteria decision analysis methodology to foster stakeholder engagement in Chapter 5. Tumwebaze *et al*. (2021) underscored the importance of customising water resource decision-making tools, streamlining them for user-friendliness, and promoting active stakeholder engagement. Stakeholders offered several alternative solutions, including public education and outreach, sewer system improvements, constructed wetlands, runoff control, and solid waste management. The selection of these alternatives was based on site-specific information such as risks quantified, the target pollution, and water uses within the uMsunduzi catchment, aligning with the guidance of Freitas and Magrini (2013). The study's findings indicate that stakeholder responses in the uMsunduzi case offer potentially sustainable solutions that align with the problem objectives. The balanced consideration of these solutions is fundamental to ensuring the long-term sustainability of the chosen BMPs, as underscored in the work of Sahoo and Goswami (2024).

In the field of water quality programmes, addressing nonpoint source pollution is an enduring challenge. To enhance the efficiency of water quality enhancing solutions, "targeting" specific areas has been a recognised strategy for quite some time, with Morton (2008) introducing the concept of concentrating conservation and preservation efforts in areas that yield the greatest benefits at the lowest cost. This approach assumes that these targeted landscape areas are either especially sensitive to the impact of human activities or ideally situated for mitigating these effects (Arabi, Govindaraju and Hantush, 2006; Fleming *et al.*, 2022). Given the variability of pollutant sources in this study, it may be advantageous to replicate this approach on a more localised scale. For example, in urban areas, where human-related faecal pollutants dominate, prioritising structural BMPs, such as sewer system repairs, may outweigh the impact of public education and outreach efforts. Conversely, in less urbanised subbasins, focusing on BMPs like environmental education and controlling livestock grazing may yield greater benefits. The World Health Organization underscores the importance of addressing risks at the pollution source (WHO, 2011).

Achieving an inclusive and collaborative approach to river governance hinges on a comprehensive understanding of the social behaviours and perceptions regarding pollution among diverse stakeholders in the catchment area. Such insights serve as the foundation for improved education and empowerment initiatives. Workshops and focus groups can prove invaluable in fostering a collaborative learning environment, especially considering the social and academic disparities among stakeholder groups. This collaborative approach, aligning with the insights of Marttunen and Hämäläinen (2008), significantly improves the quality and efficiency of the collaborative planning process by enhancing communication and understanding within the decision-making group. It was particularly effective in resolving conflicts of interest and developing consensus solutions in challenging cases. To fully realise the advantages of the MCDA approach, interactive preference elicitation is considered vital.

To enhance decision support in the context of BMPs, the use of a cost-benefit analysis proves valuable for a more detailed evaluation of economic criteria within the MCDA framework. However, it is important to acknowledge that, as pointed out by Quah, Tan and Lee (2021), conducting cost-benefit analyses in developing nations presents intricate challenges, often necessitating unique approaches due to factors such as incomplete tracking of activities in labour, goods, and financial markets. Additionally, this analysis allows for the comparison of various mitigation measures (Bergion *et al.*, 2018), enabling the assessment of societal benefits for individual projects and the identification of the optimal BMP or combination of BMPs that would yield the highest net present value, thus establishing the highest-priority pollution mitigation strategy.

The achievement of this study's overarching aim relied on the integration of three distinct yet interconnected objectives. As depicted in Figure 6.1, the process begins with hazard identification, providing a comprehensive understanding of pollution sources and trends within the catchment area. This foundation, reinforced by insights from water quality modelling, enables the identification of critical hotspots and evolving pollution patterns. Subsequently, quantitative risk assessments estimates potential human health risks from both chemical and microbial hazards. Finally, the development of a multi-criteria decision analysis framework acts as a cornerstone, facilitating stakeholder engagement and inclusive decision-making in the selection of best management practices for mitigating risks at a catchment scale.

## 6.4 Recommendations for Future Studies

Further research could consider integrating the following into the proposed framework:

- Quantify the contribution of poor sanitation, including pit latrines and open defecation, as sources of pollution in water quality studies.
- Expand the scope of QMRA to encompass additional pathogens, such as viruses, for a more comprehensive assessment of health risks.
- Explore the application of QMRA and QCRA in relation to events like the Duzi Canoe Marathon to assess potential health and environmental impacts.
- Incorporate chemical pollutant concentrations into water quality models to enable future predictions in areas where sufficient data is available for model calibration and validation.
- Enhance risk assessments by further differentiating the population by gender, recognising their distinct roles and behaviours, particularly in rural areas and informal settlements.
- Consider conducting environmental risk assessments using measures such as risk quotients to evaluate the potential ecological impacts of pollutants.
- Investigate the risk associated with emerging pollutants like microplastics, extending risk assessment methodologies to address these evolving environmental concerns.

## 6.5 Conclusions

This research strived to bridge the gap in water pollution alleviation by presenting a practical and comprehensive framework designed to support well-informed decision-making processes. This framework, with its integration of multiple criteria and considerations, stands poised to aid infrastructure, operational, and ecological managers within a catchment in prioritising BMPs aimed at reducing pollution and mitigating resultant health impacts. These BMPs encompass a spectrumg of approaches, ranging from land use management strategies that minimise pollutant wash-off and encompass public awareness initiatives (non-technical) to engineering structures designed to intercept and treat runoff (technical). The application of this proposed framework in a specific catchment area provides an invaluable opportunity to gauge its practicality and relevance in real-world scenarios as applied in the case of uMsunduzi River.

The methodologies employed in this study, including SWAT, literature analysis, and water quality monitoring, have been instrumental in identifying areas of heightened pollution. The water quality modelling results unequivocally reveal the uMsunduzi River's significant pollution with pathogens and potentially toxic chemicals. Subsequent quantitative chemical and microbial risk assessments bring to light the substantial health risks associated with the use of untreated water from the river for different population groups. This study also underscored the often-overlooked facet of indirect water reuse through the river, which becomes a necessity in rural areas and informal settlements due to the scarcity of treated water.

Incorporating diverse water uses and engaging a heterogeneous population of water users in this study is a strength that enables a more comprehensive approach. This approach aligns with MCDA, allowing various stakeholders' perspectives to be heard and catering to different groups of people. Given South Africa's socio-economic diversity, it is essential to consider these various uses and users to effectively serve the population and address societal inequalities, a factor often overlooked.

In conclusion, this research has successfully validated the hypothesis that the integration of water quality modelling, quantitative risk assessment (both chemical and microbial), and MCDA can yield a comprehensive system for decision-making. By uniting these diverse elements, this study has demonstrated its ability to effectively harmonise various catchment activities and factors, providing valuable insights for well-informed decision-making. This finding underscores the critical significance of adopting a holistic approach when evaluating and managing catchment areas, ultimately leading to more sustainable and effective decision-making practices.

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