OPTIMIZING RAINWATER HARVESTING SYSTEMS IN THE ETHEKWINI MUNICIPALITY

A CASE STUDY OF A PUBLIC SCHOOL

By

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DECLARATION

I, Praval Maharaj, declare that:

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2. This dissertation has not been submitted for any degree or examination at any other university.
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As the candidate's Supervisor I agree/do not agree to the submission of this dissertation.

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- To my family and friends for their thoughts and good wishes.
- To God, for everyone and everything, above and beyond.

"Lead us from untruth to truth, from darkness to light, and from death to immortality."

ABSTRACT

Rainwater harvesting (RWH) provides a unique perspective for water conservation, especially when considering the South African water crisis. Harvested rainwater could potentially be used for activities like toilet flushing, thereby reducing the strain on municipal supply networks. However, the economic and environmental feasibility of such systems needs to be assessed in relation to their water-saving benefits. Therefore, this research aimed to uncover the viability of two types of RWH systems implemented at a school (Duffs Road Primary). The assessment and design of the two systems (pumped and gravity-fed) were performed providing insight into system optimization in the economic and ecological settings. Water harvestings, municipal cost savings, and municipal carbon emission reductions were key aspects of each system's performance. Economic considerations included capital costs and return periods, while the environmental aspects encompassed system carbon footprints (assembly and operation) and carbon emission reduction periods. Life cycle assessments (LCAs) of the system components were also incorporated into the research, acting as an extension of the environmental feasibility analysis. The LCAs were performed using a software-modelling program called SimaPro. It was found that the gravity-fed system outperformed the pumped system in both economic and environmental contexts. Although the pumped system garnered higher harvestings and municipal savings and was also deemed economically feasible with a return period of under **6 years**, the presence of pumps made the system ecologically inviable. On the other hand, the gravity-fed system would allow for yearly benefits of **452.48 kL** in water savings, **R27 850.94** in municipal cost savings, and **185.11 kg CO_2** in municipal carbon emission savings. Including the fact that the return period would be less than **5 years** and the period to reduce the system carbon footprint at just over **10 years**, the system displayed both economic and environmental viability (from a global warming perspective). Besides the gravity-fed system costing less, it would also eliminate environmental emissions that would usually be generated from pump operation. Furthermore, energy usage and costs associated with pump operation would also be non-existent. However, the construction/production of the components of the gravity-fed RWH systems would always result in environmental burdens as assessed using the SimaPro software. Hence, recommendations for alternate materials that are more environmentally friendly may be possible for future endeavours.

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List of Abbreviations

CFC - Chlorofluorocarbon

CO_2 – Carbon Dioxide

CSIR – Council for Scientific and Industrial Research

DRWHS – Domestic Rainwater Harvesting System/Systems

DWS – Department of Water and Sanitation

EWS – eThekwini Water and Sanitation

FBW – Free Basic Water

GHG – Greenhouse Gas/Gases

GIS – Geographic Information System/Systems

GWP – Global Warming Potential

HDPE – High-Density Polyethylene

IAPs – Invasive Alien Plants

IBTs – Inter Basin Transfer Schemes

ISO – International Organization for Standardization

KZN – KwaZulu Natal

LCA – Life Cycle Assessment/Assessments

LCC – Life Cycle Cost/Costs

LCI – Life-Cycle Inventory

LCIA – Life Cycle Impact Assessment

LLDPE – Linear Low-Density Polyethylene

NRW – Non-Revenue Water

NWA – National Water Act

NWRS – National Water Resource Strategy

PP – Polypropylene

PVC – Polyvinyl Chloride

RUE – Rainwater Use Efficiency

RWH – Rainwater Harvesting

RWHS – Rainwater Harvesting System/Systems

SA – South Africa/South African

SUDS – Sustainable Urban Drainage Systems

UV – Ultraviolet

WSAs – Water Source Areas

WSE – Water Saving Efficiency

WWTWs – Wastewater Treatment Works

CHAPTER 1: INTRODUCTION

1.1. Background and Motivation

Water is a limited yet essential resource required for the continuation of life on earth. Its versatile uses and role in the domestic, industrial, and agricultural sectors fortify it as Earth's most precious resource. However, water depletion and scarcity have become quite prevalent globally due to human activities. Many countries already face water scarcity challenges (Morales-Pinzón et al., 2012), with South Africa (SA) also falling into that category (Cole et al., 2018; Donnenfeld et al., 2018). It is estimated that SA will be severely affected by physical water scarcity by 2030 (Fisher-Jeffes et al., 2017) due to many local factors.

Thus, water conservation practices like rainwater harvesting (RWH) are crucial for current and future scarcity challenges in the South African context. Rainwater harvesting is an ancient practice that serves as an effective way of obtaining free water for nearby usage (Rahman, 2017; Torres et al., 2020). Recently, it has become a favourable technology for water conservation practice in attempts to deal with the resultant water strains from population growth, urbanization, climate change, and increasing water demands (Angrill et al., 2012; Ghimire et al., 2014; Mun and Han, 2012; Ward et al., 2012; Yan et al., 2018; Zabidi et al., 2020).

However, merely adopting RWH to address water scarcity may produce other concerns from environmental and economic standpoints (Vieira et al., 2014). Therefore, this research's primary purpose is to optimize the functionality of rainwater harvesting systems (RWHS) in terms of their water performance, economic considerations, and environmental impacts. Furthermore, RWH systems have not been analyzed sufficiently in the schooling context, especially in South Africa. Thus, this study aims to rectify that research gap by assessing the economic and environmental viability of RWH systems implemented at a chosen school in SA and then extending the findings in a wider context. For the purposes of this study, water performance refers to how much rainwater could be collected by an RWHS and then used effectively. Economic considerations indicate the capital and operational costs of said systems, and environmental effects refer to such systems' carbon footprints.

Life cycle assessment (LCA) modelling was also incorporated into the study by use of the SimaPro Software. This inclusion allowed for further assessment of the environmental impacts of RWHS beyond carbon footprints (and climate change) by analyzing each component material, corresponding transport impacts, and electricity impacts from system operation. Furthermore, the viability of RWHS would involve comparative analyses of pumped systems and gravity-fed systems, giving insight into cost outcomes and water performance aspects. Pumped systems would also heighten the RWH energy intensity and since water and energy are intrinsically related, more efficient sources of electricity like wind or solar energy may be adopted for optimization.

1.2. Scope of Research

The scope of this research includes the following key concepts:

- The South African Water Situation
- Rainwater harvesting
- Life cycle assessments

1.3. Research Questions

1. How would rainwater-harvesting systems be implemented and optimized in a school setup?
2. What impacts (financial and ecological) would be introduced due to school RWH implementation and how would these impacts affect system feasibility?

1.4. Aims and Objectives

This research aimed to implement and optimize RWH systems, allowing for economic and environmental viability at a chosen case study. To achieve this, several contributory objectives were needed. The objectives of the study are stipulated as follows:

1. To design potential, complex, and scenario-specific RWH systems for a case study according to their components, placement, and use.

2. To determine the water performance and energy efficiency of a pumped RWHS in comparison to a gravity-fed RWHS thereby indicating system optimization and tradeoffs in the economic and environmental settings.

3. To identify the economic and environmental viability of the proposed RWH systems.

4. To adopt LCAs (cradle-to-gate) of the various RWH components and systems in relation to the scenario-based water harvestings generated over the system lifespans.

5. To assess and compare the environmental impacts of the pumped RWH system during its operational (lifespan) phase and assembly (construction) phase.

1.5. Report Structure

This research is divided into six main chapters, a reference list, and several appendices. All calculations were performed using Microsoft EXCEL and SimaPro. General descriptions for each chapter are provided below.

- **Chapter 1:** Provides an introductory overview of the main concepts of the research and stipulates the aims and objectives of the study.
- **Chapter 2:** Consists of compiled background information pertaining to the area of study in the form of a literature review.
- **Chapter 3:** Presents the description and characteristics of the chosen case study (Duffs Road Primary) in the RWH context.
- **Chapter 4:** Details the methods and procedures used to obtain the research findings and satisfy the investigative aims and objectives
- **Chapter 5:** Displays and deliberates the outcomes of the prescribed methodology by use of graphical and visual representations.
- **Chapter 6:** Draws conclusions from the findings that satisfy the aims and objectives.

CHAPTER 2: LITERATURE REVIEW

2.1. Introduction

This chapter focuses on the theoretical background of water scarcity, rainwater harvesting, and life cycle assessments (LCAs). A brief look at the South African water situation is first presented, which includes the background of rainfall patterns and freshwater resources in the country. The various water threats and challenges are also discussed in the South African context, along with management and conservation practices. Following the local water situation is the detailing of RWH, specifically the types, components, and design considerations of such systems. Life cycle assessments in the RWH context are also discussed, indicating the use of software tools as a path to optimization. A thorough analysis of water performance, economic viability, and environmental viability of RWHS was also undertaken. Other vital aspects of this chapter include the energy and carbon footprints of RWHS and their relation to RWH optimization.

2.2. The South African Water Situation

2.2.1. Rainfall

Rainfall is the first dependent factor when considering RWH implementation and design (Mun and Han, 2012). Logically, it becomes necessary to estimate the amount of rain an area receives to verify if an RWHS would be suitable to capture rainwater in that region. By analyzing historical rainfall data for a particular locale, one can estimate the daily, monthly, or yearly precipitation depths and design an RWHS accordingly (Maharaj, 2020).

On average, it has been found that South Africa receives 495 mm of rain per annum (Colvin et al., 2016), which is below the world average of 860 mm/year (Friedrich et al., 2009a; Mantel et al., 2010). Furthermore, both spatial and temporal rainfall variability exists in SA (Basson, 2011; Kahinda et al., 2008; Muller et al., 2009), resulting in irregular precipitation across the country and throughout the year. This concept refers to SA having uneven rainfall in both space (location) and time (seasonal). **Figures 2-1** and **2-2** highlight the spatial and temporal rainfall variability, respectively.

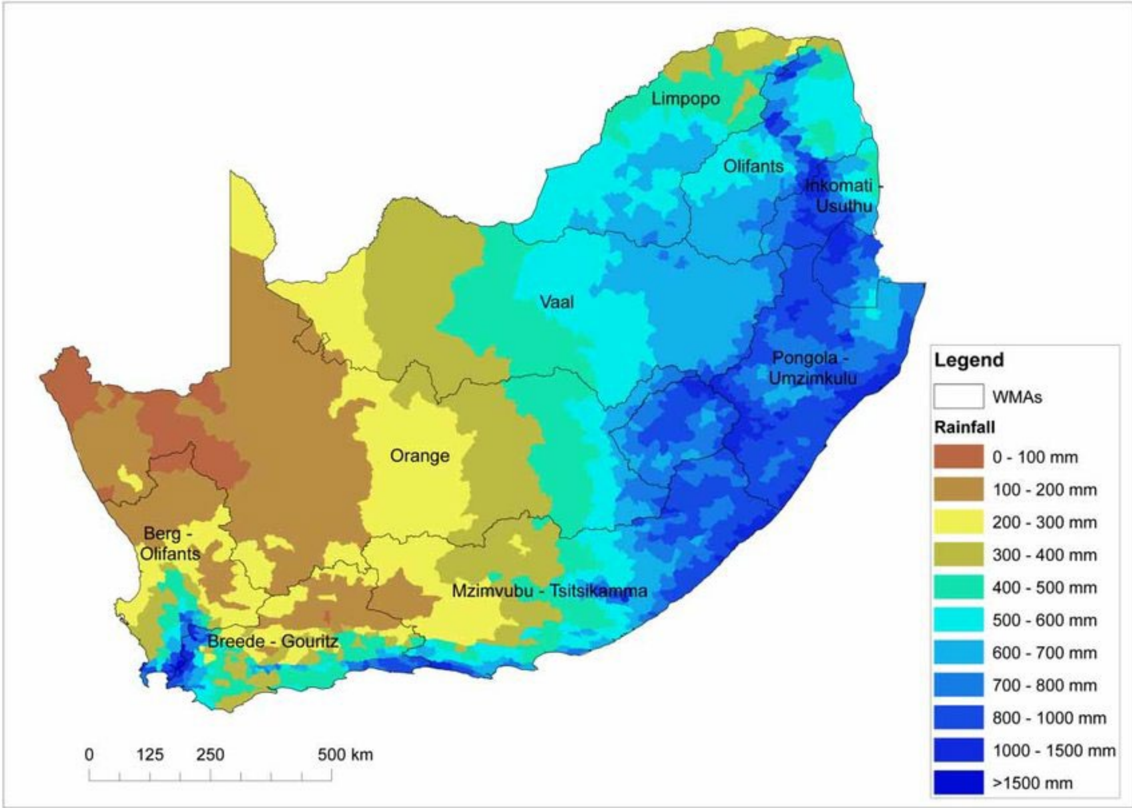

Figure 2-1: Annual Spatial Rainfall Distribution in SA

(Cole et al., 2018)

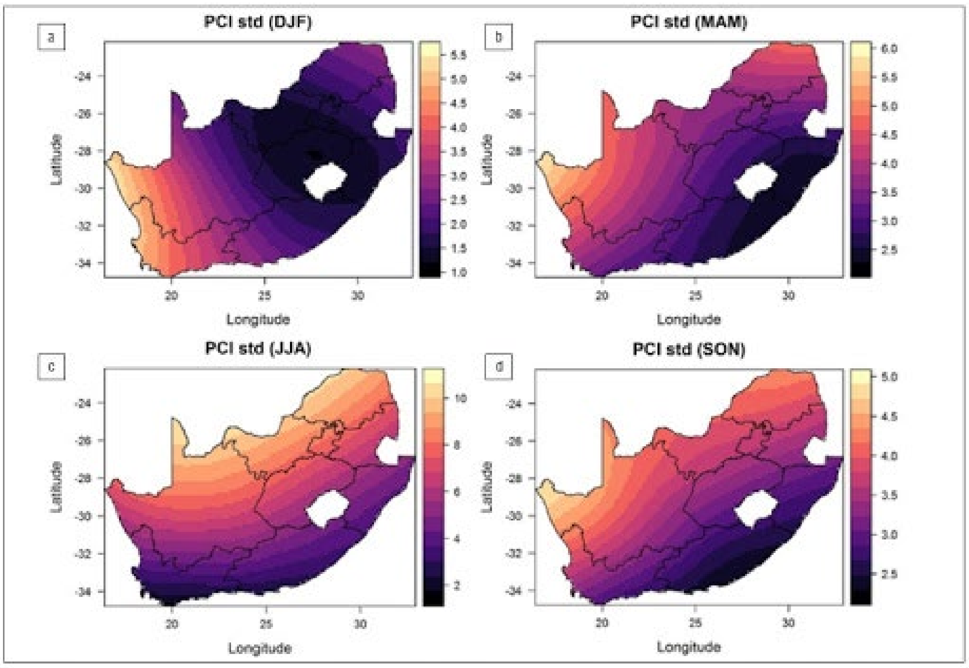

Figure 5: Spatial variability of seasonal precipitation concentration index (PCI) for (a) December–January–February (DJF), (b) March–April–May (MAM), (c) June–July–August (JJA), (d) September–October–November (SON).

Figure 2-2: Annual Temporal Rainfall Distribution in SA

(Botai et al., 2018)

Figure 2-1 shows that rainfall depths increase from west to east; however, wetter regions do occur along the southern boundary. The spatial rainfall variability also displays commonalities with the temporal rainfall distribution presented in **Figure 2-2**. In this figure, the precipitation concentration index is shown for summer (December, January, February - DJF), autumn (March, April, May - MAM), winter (June, July, August - JJA), and spring (September, October, November - SON). Hence, it can be seen that higher precipitation concentrations occur on the eastern side of the country during summer, autumn and spring as opposed to the wintertime. Western parts of the country display relatively dry periods all year round, with higher rainfall during the winter months.

2.2.2. Freshwater Resources

South Africa, like most countries, commonly relies on its freshwater resources to provide municipal, centralized water to urban users (Colvin et al., 2016; Fisher-Jeffes et al., 2017; Marteleira and Niza, 2018). Freshwater resources (mostly rivers in SA) are replenished by rainfall (Colvin et al., 2016). However, their renewability is counteracted by climate change, urbanization, population growth, and increasing water demand (Ghimire et al., 2014). **Figure 2-3** shows the primary water sources in the country.

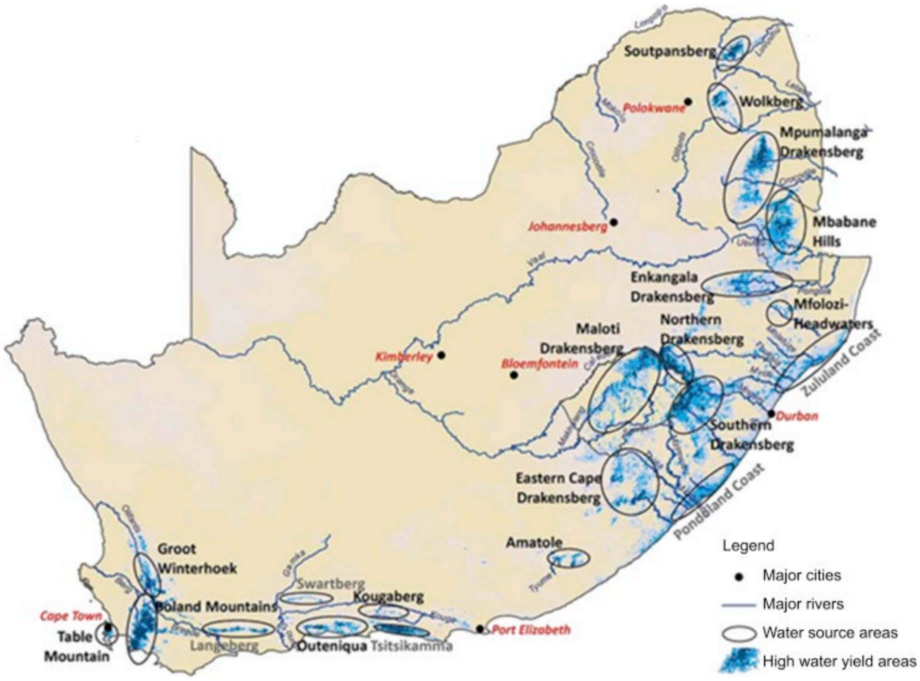

Figure 2-3: Water Source Areas in SA

(Cilliers and Rohr, 2019)

Since rainwater has a chief role in the manifestation of land runoff and freshwater resources, specific locations become valuable sources for water use. In SA, several regions have high natural runoff due to rainfall and topography and are known as water source areas (Nel et al., 2017). According to the Council for Scientific and Industrial Research (CSIR), which is an entity of the SA Department of Science, these water source areas (WSAs) should be valued as they supply substantial amounts of the country's water (CSIR, 2017). Mudombi and Montmasson-Clair (2020) substantiate this point by considering WSAs as regions of national importance.

The WSAs are grouped into 21 zones (Colvin et al., 2016), reflecting the major river basins (Muller et al., 2009), and are shown in **Figure 2-3**. When looking at **Figure 2-3**, one can recognize the correlation between WSAs and the spatial rainfall distribution in **Figure 2-1**. Since freshwater resources are sustained by rainfall, the WSAs are located in the eastern and southern SA regions, where higher precipitation occurs. The surface WSAs may only form 8 – 10 % of the total land area but provide 50% of the water volume in rivers (Colvin et al., 2016; CSIR, 2017; Mudombi and Montmasson-Clair, 2020; Nel et al., 2017). Therefore, it can be seen that a tiny percentage of the land generates half the mean annual runoff needed to sustain society and ecosystems. Hence, water concerns do arise.

2.2.3. Water Threats and Challenges

The notions of variable rainfall distribution and limited freshwater resources form the root of South Africa's major water threats and challenges. SA is classified as semi-arid and water-stressed because of the aforementioned elements (Basson, 2011; Cole et al., 2018; Fisher-Jeffes et al., 2017). As shown by the WSAs in **Figure 2-3**, spatial rainfall variability results in the uneven distribution of freshwater resources across the land. This occurrence leads to disproportional water availability relative to demand centres (Fisher-Jeffes et al., 2017; Muller et al., 2009). Hence, water scarcity becomes a significant issue.

2.2.3.1. Physical Water Scarcity

Larger population sizes demand more resources, so looking at the available water supply in relation to the demand becomes crucial. According to the SA Department of Statistics (Maharaj, 2020), the total populace of SA is approximately 59.62 million. Recalling that the average rainfall is 495 mm, only 9% of it replenishes rivers (Colvin et al., 2016). The rainwater also produces 49 billion m^3 of mean surface runoff per annum (Basson, 2011;

Colvin et al., 2016). However, approximately 30 – 31 % of the mean surface runoff is safely allocated for usage (Colvin et al., 2016; Muller et al., 2009). This amounts to 14.7 – 15.2 billion m^3 of water distributed at a high assurance of supply to national water users (the current demand). It is estimated that future demand will increase by 32% (to 17.7 billion m^3) by 2030 (Colvin et al., 2016), leaving a deficiency of around 3 billion m^3. Mudombi and Montmasson-Clair (2020) substantiate these claims by stating a 17% water deficit by 2030.

Another factor responsible for physical water scarcity is evaporation levels (Malema et al., 2016). According to Basson (2011) and Cole et al. (2018), evaporation rates in SA are comparatively higher than global evaporation rates, with the average rate of potential evaporation being about 1800 mm per year (Colvin et al., 2016; Mantel et al., 2010). This point suggests that water is evaporated at a rate almost four times the precipitation rate (495 mm/annum). Thus, physical water scarcity in SA is exacerbated by both the climatic and societal conditions in the country.

2.2.3.2. Economic Water Scarcity

Water scarcity in SA also presents itself from another social perspective. Economic water scarcity, which is essentially the inability of water users to pay for or receive centralized water (Maharaj, 2020; Muller et al., 2009), is a consequence of past water inequities of the apartheid era (Colvin et al., 2016; Muller et al., 2009). However, even with the abolishment of the regime, previously disadvantaged communities still experience the after-effects of unequal water access (Cole et al., 2018; Colvin et al., 2016; Sutherland et al., 2014). It is estimated that 12 – 14 million people in SA did not have safe water access at the start of the democracy (Hellberg, 2014). Colvin et al. (2016) argue that as much as 15.9 million South Africans lacked proper water supplies; hence, the municipal challenge of providing to so many people becomes apparent.

It is predicted that less than half (45 – 46 %) of the South African population have piped water access inside dwellings (Cole et al., 2018), with as much as 9% having no piped water access (Colvin et al., 2016). The remainder of the population has varying forms of channeled water service ranging from near (12%), far (6%), and yard access (27%) (Colvin et al., 2016). Studies by Cole et al. (2018) and Colvin et al. (2016) showed that the regions with the lowest water access were in KwaZulu Natal (KZN) and Eastern Cape.

Water is a constitutional human right that allows every SA citizen daily access to 25 litres of free basic water (FBW) per day as stipulated by the Water Services Act of 1997 (Colvin et al., 2016; Crous et al., 2013; Hellberg, 2014). According to Cole et al. (2018), the average water use per capita per day (l/c/d) is 208 l/c/d. Thus, the tremendous task faced by municipalities is to extend water services and resources to a growing population while maintaining previous water responsibilities to agriculture and industry (Colvin et al., 2016; Sutherland et al., 2015).

2.2.3.3. Water resource exploitation and contamination

Three sectors are generally responsible for depleting water resources, namely the agricultural, domestic and industrial sectors (Blignaut and Van Heerden, 2009; Donnenfeld et al., 2018). According to Blignaut and Van Heerden (2009), the agricultural sector consumes 62% of the available water resources. Donnenfeld et al. (2018) corroborate this fact, suggesting that 9.7 km^3 of water by volume is used to support agriculture and food production. The domestic and industrial sectors are responsible for 27% and 11% of the water withdrawals, which amount to 4.2 km^3 and 1.6 km^3. The addition of these values is in line with the total national water supply of 15 km^3 (or 15 billion m^3), as highlighted previously.

Although agriculture and industry form the backbone for economic development, these activities produce by-products, effluent, and discharge that further contaminate freshwater resources (Basson, 2011; Donnenfeld et al., 2018). Electricity generation, food production, and domestic uses all require clean water yet also contribute to water quality degradation (Colvin et al., 2016). Contaminated water used by citizens (usually rural) causes further concerns like water-borne diseases (Mudombi and Montmasson-Clair, 2020). This issue links back to unequal water provision highlighting the correlation between impoverished communities and lower water quality (Cole et al., 2018; Colvin et al., 2016; Maharaj, 2020).

Thus, with the growing demands and thinly spread resources, overexploitation of freshwater resources occurs (Basson, 2011; Muller et al., 2009). Donnenfeld et al. (2018) state that over 60% of South African rivers are over-exploited, with only one-third functioning in good condition. According to Colvin et al. (2016), 44% of SA's rivers are critically threatened by human activities. The over-exploitation of freshwater resources also creates other challenges like the disruption of ecosystems and river flow. Rivers that experience reduced flow rates are unable to absorb the by-products of industrial discharge and fertilizer runoff effectively and may become contaminated to a point beyond use (Basson, 2011; Donnenfeld et al., 2018; Muller et al., 2009).

2.2.3.4. Biological Concerns

Pollutants, primarily nitrogen and phosphates (Colvin et al., 2016) introduced into water sources, may result in eutrophication (nutrient enrichment). Algal bloom and further water quality deterioration through cyanobacteria and cyanotoxins (Colvin et al., 2016; Van Ginkel, 2011). According to Muller et al. (2009), toxic cyanobacteria had been found in 11 WSAs at the time, posing severe health risks to humans and animals (Van Ginkel, 2011). Typically, the most common contaminants include poorly treated sewage (urban and rural), mine waste (from acid mine drainage), and industrial and agricultural effluents, as discussed previously (Colvin et al., 2016).

Another major threat to freshwater resources comes in the form of invasive alien plants (IAPs). Generally, IAPs consume more water than indigenous species and lower water availability by up to 4% (Colvin et al., 2016). According to Blignaut and Van Heerden (2009) and Colvin et al. (2016), IAPs have the potential to consume 16% of total water resources, which would otherwise be better served towards ecosystems and society. Mudombi and Montmasson-Clair (2020) suggest IAPs reduce the mean annual runoff by 3 – 7 %. These percentages amount to a range of 1.4 – 3.3 billion m^3 of water resources lost. Colvin et al. (2016) verify this information by stating that a 1.44 billion m^3 water loss due to IAPs could sustain 3.38 million households with four inhabitants for one year.

Alternatively, this amount of water lost could irrigate 120 000 Hectares (Ha) to supplement food production (Colvin et al., 2016). Hence, IAPs not only threaten and disrupt ecosystems, biological diversity, and water security (Colvin et al., 2016) but also have severe economic impacts. Mudombi and Montmasson-Clair (2020) estimate a loss of R6.5 billion in ecosystem services every year due to the IAPs situation.

2.2.3.5. Non-Revenue Water

The many water threats and challenges in SA are compounded by the occurrence of non-revenue water (NRW). This is the amount of water that generates no returns for municipalities due to leakages, billing errors (commercial losses), and illegal use (Mudombi and Montmasson-Clair, 2020; Tsitsifli et al., 2017). It is estimated that between 35 – 37 % of urban-piped water is non-revenue, primarily on account of the ageing water networks and infrastructure (Colvin et al., 2016; Mudombi and Montmasson-Clair, 2020). These percentages amount to roughly 1.5 km^3 of water lost, making SA one of the worst countries experiencing water scarcity and NRW (Donnenfeld et al., 2018; Moahloli et al., 2019).

NRW presents negative impacts from both environmental and economic perspectives. Water consumers pay higher costs for water that is already scarce (Tsitsifli et al., 2017) yet, this does not cover municipal expenses. According to Colvin et al. (2016), NRW has municipal economic losses of R7.2 billion per year, although Mudombi and Montmasson-Clair (2020) argue that R9.9 billion worth of losses occurs per annum (1.66 billion m^3 or 1.66 km^3).

When analyzing the physical losses due to leakage, Moahloli et al. (2019) estimate the water loss component of NRW to be 25%. This means that a quarter of the total water available in SA is lost to leakages alone and, while this is on par with the global average, climatic and social conditions exacerbate the local situation (Colvin et al., 2016; Donnenfeld et al., 2018; Moahloli et al., 2019). Thus, it can be seen that the consequences of human activity and population growth worsen the strains placed on water resources. Hence, proper management and conservation of water resources are vital.

2.2.4. Water Management and Conservation

The significant threats and challenges discussed in Section 2.2.3 highlight the importance of efficient water management. This section aims to review the management and conservation of water resources in SA and within the eThekwini Municipality. Countrywide legislation like the National Water Act (NWA) of 1998 and the Water Services Act (1997) provide the foundation for how water is allocated, distributed, and used in the country (Basson, 2011; Colvin et al., 2016).

2.2.4.1. National Regulations

Generally, social equity is a significant driver in policy-making due to past injustices (Mantel et al., 2010; Muller et al., 2009), but many other factors are also deliberated based on local circumstances. Hence, tactical objectives stipulated in the National Water Resource Strategy (NWRS) (under the NWA) aim to allow for the fair and proper management, control, and protection of water resources (Colvin et al., 2016). Basson (2011) lists the priorities of the NWRS in descending order of importance as follows: provision for the Ecological Reserve, international agreements and obligations, social equality (domestic usage), economic uses (within agriculture and industry), and convenience and recreation.

It must be noted that SA allocates a portion of water (known as the Ecological Reserve) back to natural ecosystems to ensure their sustainability (Colvin et al., 2016). International contracts refer to water resources shared with neighboring countries since four main rivers (or 60% of the river basins) are joined to other countries (Basson, 2011; Colvin et al., 2016). Water resources are then distributed through a series of engineered networks and infrastructure to demand locations (Colvin et al., 2016). **Figure 2-4** represents the generalized planned infrastructure of SA water systems.

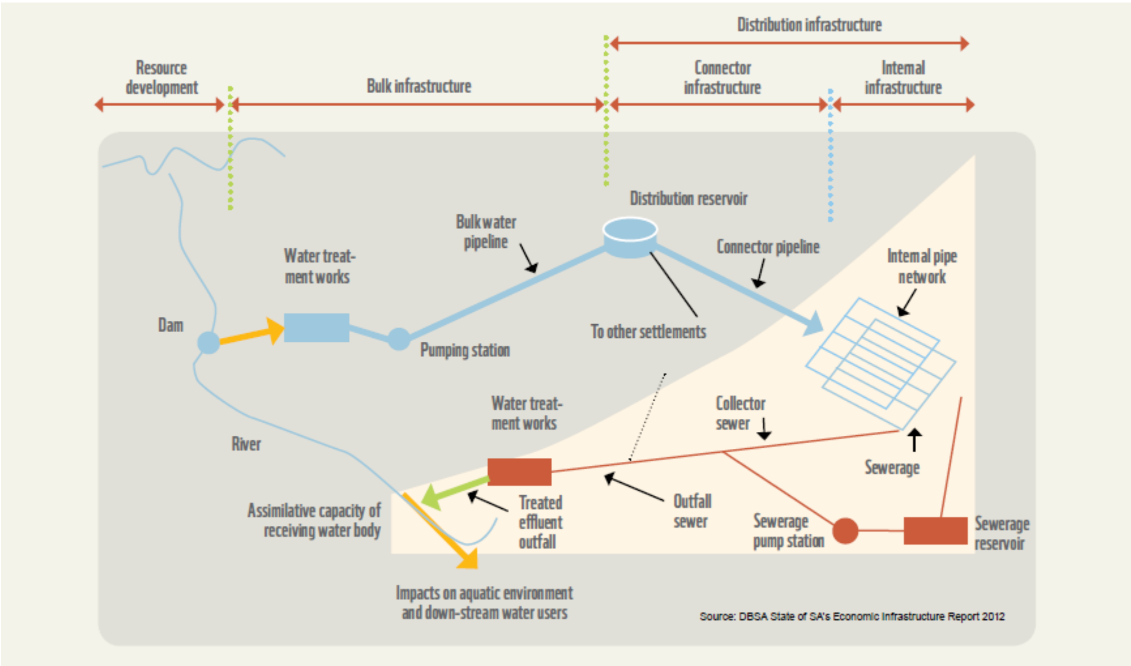

Figure 2-4: Engineered Infrastructure of SA Water Networks *(Colvin et al., 2016)*

As mentioned previously, centralized, urban water is supplied from the impoundments of rivers; however, 60% of the rivers in SA are commonly characterized by reduced flows due to over-exploitation (Donnenfeld et al., 2018). Therefore, protecting the WSAs is vital for future water security, but only 13 – 18 % of these areas are officially protected as parks or nature reserves (Colvin et al., 2016; CSIR, 2017; Nel et al., 2017). The lack of protection for WSAs is cause for concern since a study by Nel et al. (2017) suggests that they support 51% of SA's population and 64% of the economy when linked to downstream urban centres.

Research by Colvin et al. (2016) and Mudombi and Montmasson-Clair (2020) reiterate the importance of conserving ecological infrastructure like WSAs. Water stewardship (which is efficient and sustainable water usage from social, economic, and environmental aspects) has become a central practice in many companies and businesses in SA to safeguard resources, albeit in informal water governance (Colvin et al., 2016; Mudombi and Montmasson-Clair, 2020). Hence, it becomes critical to link water stewardship practices into formal governmental legislation. Donnenfeld et al. (2018) and Mudombi and Montmasson-Clair (2020) believe that stricter water supply and demand measures need to be initiated at a governmental level to avoid a similar crisis to the energy situation. This idea encompasses proactive legislation promoting efficient water use on social, economic, and environmental levels.

2.2.4.2. Water Association and Infrastructure

Several studies have been conducted on the relationship between water and energy (Albrecht et al., 2018; Dai et al., 2018; Valek et al., 2017). This idea (commonly termed the water-energy nexus) refers to the connection/dependency between water and energy production (Marteleira and Niza, 2018; Simpson and Berchner, 2017). However, the association between water and energy often leads to environmental concerns based on how centralized water is provided (i.e., its physical and operating infrastructure). In SA, coal-based energy plants primarily produce electricity and, in doing so, are inevitable sources of carbon dioxide (CO_2) emissions (Knegtel and Naidoo, 2014). Furthermore, electricity generation requires large amounts of water, thereby ushering drives towards more renewable energy sources (Colvin et al., 2016; Donnenfeld et al., 2018; Mudombi and Montmasson-Clair, 2020). Since water security is dependent on sustainable water resources (Colvin et al., 2016), renewable energy would allow for lower industrial demand, decreased carbon emissions, and minimized water contamination from coal (Donnenfeld et al., 2018; Valek et al., 2017).

Traditionally, the increased demands, limited resources, and variable rainfall have propelled the development of engineered infrastructure like dams and inter-basin transfer schemes (IBTs) to increase water storage and supply at demand centres (Colvin et al., 2016). Historically, South Africa formed economic and social hubs around mining operations, often situated away from freshwater resource areas (Maharaj, 2020; Wall, 2018). Hence, IBTs were introduced to convey water from surplus regions to areas of deficit (Basson, 2011; Colvin et al., 2016; Muller et al., 2009). Furthermore, engineered infrastructure like dams provides increased water storage from the rainwater runoff (Maharaj, 2020). According to Colvin et al. (2016), water infrastructure owned by the Department of Water and Sanitation (DWS) is valued at R63 billion, partially consisting of 305 dams with a total capacity of 29.2 billion m^3. While dam infrastructure aims to increase water security, it alters ecosystems and environmental flows (Colvin et al., 2016; Mantel et al., 2010). A study by Mantel et al. (2010) showed that the amassed effect of small dams in SA affects the water quantity and quality of rivers. Water infrastructure also requires upkeep and maintenance (and its associated costs of R1.4 billion per annum) to ensure efficiency (Colvin et al., 2016; Mudombi and Montmasson-Clair, 2020). Furthermore, Moahloli et al. (2019) state that suitable sites for dam construction are becoming scarce with increases in water demand, particularly in the domestic sector.

2.2.4.3. Economic Undertakings

A study by Cole et al. (2018) showed that social factors (like water access and income) influence water usage per capita more than natural factors like rainfall and runoff. This observation is particularly relevant within the urban environment. It is projected that water demand in the domestic sector increased from 22% to 27% from 2003 to 2013 and would continue growing at 1.2% per annum over the next ten years (Moahloli et al., 2019). The migration of people from rural to urban areas (especially within the eThekwini Municipality) compounds the issue of domestic water demands (Donnenfeld et al., 2018; Gumbi and Rangongo, 2018). Mudombi and Montmasson-Clair (2020) have stated that municipal water use provides 58% of the national revenue giving insight into its economic relevance.

Nonetheless, economic shortcomings are also prevalent amidst the rising water demands. The DWS estimate that a capital investment of R70 billion is needed for water supply infrastructure and R20 billion for sanitation, wastewater collection, and wastewater treatment (Mudombi and Montmasson-Clair, 2020). Currently, available funding is around R57 billion for water and sanitation; hence, a R33 billion deficit. Muller et al. (2009) also highlight the uneven progress in terms of water supplies and sanitation, which could be linked to the uneven funding described. Furthermore, sanitation is dependent on water supply infrastructure; hence, the sanitation lag could be attributed to backlogs in water supplies (Department of Cooperative Governance and Traditional Affairs, 2020)

While water supply issues are apparent, challenges still exist once the supply is implemented, e.g., NRW. According to Mudombi and Montmasson-Clair (2020), the first avenue to improve water and sanitation in SA is addressing NRW. By reducing NRW, the gap between supply and demand can be minimized without relying on new infrastructure and its high associative costs (Colvin et al., 2016).

Initiatives such as the 'War on Leaks' campaign were also developed and aimed to employ 15 000 people as plumbers and artisans to deal with NRW (Colvin et al., 2016; Donnenfeld et al., 2018). However, Mudombi and Montmasson-Clair (2020) have argued that the 'War on Leaks' campaign had some shortcomings like mismanagement and uneven implementation. Other suggestions to reduce NRW included timely water meter replacement (Gumbi and Rangongo, 2018; Moahloli et al., 2019). In proactively doing so, water meters would be replaced before they become dysfunctional, and hence, NRW and utility losses could be reduced. A study by Tsitsifli et al. (2017) offers an integrated approach for water service providers to deal with NRW by use of the Decision Support System Tool. In doing so, evaluating water distribution networks under particular water service jurisdiction could be performed, thereby reducing NRW and providing a better overview of said networks' operation and functionality.

2.2.4.4. Groundwater Usage

With all the strains and challenges of surface water resources, and municipal extraction and distribution, other water sources are often overlooked or underutilized (Knüppe, 2011). Donnenfeld et al. (2018) estimate that around 4.8 km^3 (4.8 billion m^3) of groundwater is potentially usable in SA though further research would be needed on its quality (Maharaj, 2020). Studies from Blignaut and Van Heerden (2009) stated that groundwater usage has increased at a rate of 3.4% per year since 1999. This upsurge is evident from the fact that over 300 towns already use groundwater (Colvin et al., 2016).

Since groundwater is also dependent on rainfall, with about 4% of the rainwater allowing for rechargeability of aquifers (sediment layers storing water), practices such as water banking are common (Colvin et al., 2016). This water-banking concept refers to artificially recharging underground water areas in socially, environmentally, and economically feasible locations to lessen water supply strains (Colvin et al., 2016).

Hence, groundwater usage would be advantageous in staving off water demands for agriculture and food production (Donnenfeld et al., 2018). Research by Basson (2011) and Muller et al. (2009) have already suggested inefficient water use in the agricultural sector, with a lack of metered consumption being quite prevalent (Mudombi and Montmasson-Clair, 2020). Hence, groundwater potential offers a unique opportunity for reducing water strains and demands.

2.2.4.5. Biological Control

Water conservation and management have also involved responses to biological threats like IAPs through manual interventions. According to Colvin et al. (2016), 2.7 million hectares (Ha) of land has been cleared of invasive plants. This figure is equivalent to 27000 km^2 of land, highlighting the intensive manual labour required for the task. Thus, it can be seen that the removal of IAPs not only improves water security but also initiates job creation (Mudombi and Montmasson-Clair, 2020). Furthermore, waste wood generated through these removal programs is recycled for compost and furniture manufacturing, thereby mitigating fire and flooding risks (Colvin et al., 2016).

Several schemes have been implemented to deal with IAPs and water security (Colvin et al., 2016). According to Mudombi and Montmasson-Clair (2020), the ‘Working for Water’ initiative, partly led by DWS, has successfully eradicated IAPs. Other solutions include biological controls that are employed to restore ecosystem balance (Van Ginkel, 2011). By introducing natural threats to invasive species, SA now has 10 IAPs under complete control and 18 IAPs under substantial control, thereby mitigating economic and environmental damages (Colvin et al., 2016).

Furthermore, 106 biological control agents have been introduced to 48 IAPs since 1913, allowing for R6.5 billion in savings every year (Colvin et al., 2016). Thus, natural means to solve water security problems have been very effective in garnering more innovative ideas to protect water at the source.

2.2.4.6. Contamination Reduction

It is apparent the natural state of river basins allows for runoff to be generated; however, it also allows for agricultural and industrial seepage to flow into surface water, causing contamination (Colvin et al., 2016). Since large water consumers like mining and industry are often thought to compensate for the adverse effects of significant withdrawals by providing high economic value (Muller et al., 2009), regulations are not consistently followed to ensure proper effluent disposal (Wall, 2018). This inconsistency is problematic since legislation for effluent exists but may not be enforced adequately at governmental levels (Wall, 2018). Therefore, Colvin et al. (2016) have proposed natural vegetation to be used as buffers. In this way, agricultural and industrial seepage can be diverted away from WSAs, thereby reducing contamination and eutrophication.

Another avenue for improved water management and conservation is wastewater treatment. It is estimated that 54% of municipal wastewater is treated (Donnenfeld et al., 2018), potentially exacerbating ecosystem conditions downstream when discharged into rivers. While the effluent discharge of a certain quality into river bodies is legal, proper monitoring and enforcement of correct regulations are lacking (Wall, 2018). Initiatives such as the Blue Drop and Green Drop Programs do exist to compare and verify wastewater performance in SA but serve mainly as a rating tool (Colvin et al., 2016).

It was found that wastewater treatment works (WWTWs) are in disrepair based on evaluations from the Green Drop Program (Colvin et al., 2016; Donnenfeld et al., 2018). It was assessed that 58% of WWTWs nationwide are in a high-to-critical danger of failure, with only 16% at low risk (Mudombi and Montmasson-Clair, 2020). KZN, in particular, had about half of its WWTWs categorized as medium-risk (Colvin et al., 2016). Challenges such as funding and unskilled operation for new technologies also arise; however, increasing energy efficiency in WWTWs can reduce metropolitan energy outputs by 358 460 MWh/year, saving R216 million per annum (Mudombi and Montmasson-Clair, 2020). The energy output reduction would also have a positive impression on the water-energy nexus.

2.2.4.7. Performance by the eThekwini Municipality

The challenges and threats faced at a national level are often filtered down to the municipal levels in varying degrees. In KwaZulu Natal (KZN), the eThekwini Municipality forms the most prominent metropolitan municipality (eThekwini Municipality, 2012) that conventionally provides water from the impoundments of dams and rivers (Friedrich et al., 2009b; Sutherland et al., 2014). Specifically, the eThekwini Water and Sanitation Unit (EWS) is responsible for water service provision in the greater Durban metropolitan area (Sutherland et al., 2014).

According to the SA Department of Cooperative Governance and Traditional Affairs (2020), eThekwini Municipality occupies 2556 km^2 and is home to 3.99 million people. In 2015, the eThekwini region consisted of 3.6 million people (Sutherland et al., 2015), highlighting the 1.1% growth of the population in five years. Sutherland et al. (2014)

estimate a migration of 150 000 people every year. Hence, water service provision to this rising populace results in backlogs and shortcomings that could take 5 – 15 years to address, based on the current funding, subsidies, and population growth (Department of Cooperative Governance and Traditional Affairs, 2020).

Despite this scenario, it was found that the EWS had better water service delivery in the domestic sector compared to the rest of the province and country. Statistics showed that 98.3% of households in eThekwini had access to water services higher than the 83.35% access in KZN and 86.2% in SA (Department of Cooperative Governance and Traditional Affairs, 2020). The EWS is often recognized as a pioneer in water and sanitation policy in SA (Sutherland et al., 2014), with the aforementioned statistics substantiating this point. However, other challenges still exist based on previous inequities and township development during apartheid (Sutherland et al., 2014).

Pressures from the current rapid urbanization compound the spatial and socio-economic inequalities in eThekwini (Cole et al., 2018; Sutherland et al., 2015). This issue has been discussed at the national level, along with the migration issue. Nonetheless, how the EWS has responded to new and increasing demands correlates with spatial and socio-economic factors. Generally, water service delivery by the EWS takes three forms that are implemented based on settlement type (Sutherland et al., 2014). The three service levels include ground tank, semi-pressure supply, and full-pressure supply systems (Hellberg, 2014)

Impoverished rural communities are usually provided with the ground tank systems (supplying, at minimum, the FBW) while urban areas receive full-pressure connections from the supply networks (Hellberg, 2014; Sutherland et al., 2014). Interim measures like communal ablution blocks and semi-pressure supply systems are usually implemented in informal settlements (Crous et al., 2013; Hellberg, 2014). Thus, the different levels of water service delivery based on settlement type can be seen.

It is clear from the literature that the EWS attempts to cater to people from all occupations and incentivizes efficient urban water use through tiered pricing (eThekwini Municipality, 2021d). Generally, water tariffs are regarded as cost-recovery based (Basson, 2011; Blignaut and Van Heerden, 2009), lowering the incentive to save water. Tiered pricing offers a way to encourage efficient water use by charging lower water rates for essential needs and higher water rates for more extravagant usage (Donnenfeld et al., 2018). The tiered pricing for eThekwini Municipality is shown in **Table 2-1**.

Table 2-1: Water Tariffs (eThekwini Municipality, 2021d)

Range (kL)	Roof-tank semi-pressure supply	Full-pressure (low income)	Full pressure (middle/high)	Other (industry, commerce etc.)
0 – 6	nil	nil	R23.42	R36.52
6 – 25	R18.87	R27.70	R27.70	R36.52
25 – 30	R25.81	R36.90	R36.90	R36.52
30 – 45	R56.98	R56.91	R56.91	R36.52
> 45	R62.58	R62.58	R62.58	R36.52

Thus, with all the water threats and challenges present in SA at national and municipal levels, looking at other alternative methods for procuring water is necessary. While means like desalination hold more potential in coastal areas (Maharaj, 2020), rainwater harvesting is a promising and flexible solution to the growing water crisis.

2.3. Rainwater Harvesting (RWH)

2.3.1. General

The numerous water threats and challenges have initiated various management and conservation techniques, as discussed in **Sections 2.2.3** and **2.2.4**. Issues of variable rainfall, population growth, increased demand, water scarcity, and unequal access highlight the need for environmentally and economically sustainable technologies like rainwater harvesting (GhaffarianHoseini et al., 2016; Zabidi et al., 2020).

Rainwater harvesting (RWH) is often regarded as an efficient tool for on-site water usage (Kahinda et al., 2007; Helmreich and Horn, 2009). It is defined as the collection and storage of rainwater that can be used for various water purposes in agricultural, industrial, commercial, and domestic frameworks (Shaari, 2020). RWH has been practiced for generations and recently re-emerged as a solution for increased water demands and water scarcity (Hafizi Md Lani et al., 2018; Semaan et al., 2020; Torres et al., 2020). In order to understand RWH, the types, components, and design considerations are presented.

2.3.2. Types and Components of RWHS

2.3.2.1. Types

The categorization of RWHS has some consensus in the literature. RWHS are usually implemented at three levels (small, medium, and large scales) which relate to domestic and agricultural purposes (Che-Ani et al., 2009). Kahinda et al. (2008) and Malema et al. (2016) classify RWHS into three groups: in-field, ex-field, and domestic rainwater harvesting systems. The descriptions of these key forms of RWH correlate to the catchment surfaces and the household and agricultural sectors.

In-field RWH refers to the in-situ collection of rainwater for irrigation purposes, whereas ex-field or external RWH is achieved by capturing rainwater away from cultivation sites (Bello et al., 2020; Helmreich and Horn, 2009). Thus, both in-field and ex-field RWH is used for agriculture. Domestic rainwater harvesting systems (DRWHS) utilize roofs and courtyards to collect rainfall (Helmreich and Horn, 2009; Malema et al., 2016) for various water purposes.

2.3.2.2. Components/Elements/Subsystems

RWH systems consist of several subsystems or components, allowing for their coherent functionality. Rainwater first requires capture from a chosen surface before being conveyed and filtered to a storage unit. Water must then be distributed from the storage point to the application point. Hence, the various components of RWHS allow for this process.

According to Mun and Han (2012), an RWHS is subdivided into six major components: the catchment area, filtration, storage tank, supply facility, piping, and an overflow unit. GhaffarianHoseini et al. (2016) and Shaari (2020) similarly categorize the components highlighting the catchment area and conveyance mechanisms as a part of the collection subsystem. However, only five major subsystems were listed in these studies, including the collection, treatment, storage, distribution, and municipal backup components.

Research by Vieira et al. (2014) corroborates the classification by GhaffarianHoseini et al. (2016), highlighting the treatment subsystem as a means for pre-filtration, i.e., removing larger contaminants before storage. Furthermore, valve-control mechanisms were also discussed, which would enable switchback to municipal water use should the rainwater be depleted. A summary of the RWHS components is provided in **Table 2-2**.

Table 2-2: RWHS Components (Bello et al., 2020; Che-Ani et al., 2009; GhaffarianHoseini et al., 2016; Stec and Zeleňáková, 2019; Vieira et al., 2014)

Component/Subsystem	Function	Examples
Catchment Surface/Area	Allows for the initial capture of rainwater	Roofs, courtyards
Conveyance	Transfers rainwater from the catchment point to the storage point	Gutters, downpipes
Pre-filtration	Removes larger, unwanted rainwater contaminants before storage	First-flush diverter, leaf eater, mesh/tank screen
Storage Unit	Amasses filtered rainwater for potable and non-potable uses	Storage tanks (above or below ground)
Distribution network	Supplies rainwater from the storage unit to the water application point	Polycop piping, pumps
Municipal Backup	Allows for centralized water use should the demand be above the RWHS capacity	Valves, controller apparatus

2.3.3. RWHS Design Considerations

The design of rainwater harvesting systems is multi-faceted and draws a parallel with the RWH components described in Section 2.3.2. According to Mun and Han (2012) and Ghimire et al. (2019), the main design parameters for RWH include rainfall, catchment area, collection efficiency, tank volume, and water demand, thereby allowing for system functionality and steady water supply. Similarly, Fisher-Jeffes et al. (2017) state that RWH modelling is dependent on water demand volume, volumetric reliability (ratio of supply to demand), runoff that generates supply, and the municipal cost of water per kL.

2.3.3.1. Harvesting Potential

It is apparent that the success of RWHS is primarily dependent on the uncontrolled rainfall factor. Hence, it is crucial to design and implement such systems in areas where rainfall quantity is sufficient based on historical data (Hafizi Md Lani et al., 2018; Semaan et al., 2020). When considering the eThekwini Municipality, rainfall in this region is much higher; hence RWHS may be better suited relative to drier parts of the country.

Research has shown that rainwater volume dictates the available supply of RWHS, which can be contained in appropriately sized tanks that aim to match the supply and improve system efficiency (Semaan et al., 2020; Stec and Zeleňáková, 2019). The product of local catchment areas, precipitation depths, and runoff coefficients enable the rainwater volume to be calculated (Ghimire et al., 2019; Kisakye and Van der Bruggen, 2018; Marteleira and Niza, 2018). This type of analysis is known as the empirical method that compares the available rainwater supply with the municipal water demand to approximate tank size (Semaan et al., 2020). Marteleira and Niza (2018) have also represented RWH supply and demand using a simple relationship. It states that the final water volume is equivalent to the summation of the initial volume in the tank and accumulated rainwater minus the water demand.

Other analysis methods include stochastic (probability) relationships and the mass/water balance approach (Campisano et al., 2017; Fisher-Jeffes et al., 2017; Marteleira and Niza, 2018; Semaan et al., 2020). Stochastic techniques are used for RWHS implemented at a broader scale to better represent the design parameters for those case studies (Fisher-Jeffes et al., 2017) or where data may be missing or incomplete (Semaan et al., 2020). The water/mass balance approach is similar to the empirical method as it accounts for system inflow (supply), outflow (demand), and water losses for the quantification of tank size (Campisano et al., 2017; Marteleira and Niza, 2018).

Thus, it can be seen that supply and demand are significant considerations in RWH design, the ratio of which dictate the volumetric reliability of the system (Fisher-Jeffes et al., 2017). Therefore, RWHS that allow larger supply would be more reliable than systems displaying supply deficits. According to Che-Ani et al. (2009), supply and demand also govern tank size selection along with other factors like dry season, catchment surface, aesthetics, personal preference, and available budget. Besides the demand-side approach, tank sizing may also be computed via graphical or computer-based simulation/behavioral models, increasing the design complexity and accuracy (Khan et al., 2017).

2.3.3.2. Existing Features

RWHS are usually retrofitted to existing structures (Che-Ani et al., 2009); hence, the catchment area's design should maximize rainwater supply while also considering catchment materials and runoff coefficients (Worm, 2006). Rainwater captured on roof surfaces often accumulates pollutants that are then conveyed into the storage units. Recently, RWHS has been enhanced by adopting green roofing materials to decrease contamination and improve water quality (Hafizi Md Lani et al., 2018). The inclusion of pre-filtration mechanisms like first-flush diverters, leaf eaters, and tank screens are also adopted to improve water quality (Kisakye and Van der Bruggen, 2018; Maharaj, 2020)

The usage of runoff coefficients gives an estimation of the amount of rainfall that will be collected for supply (Worm, 2006). Alternatively, it approximates the rainwater losses from the catchment area, with typical coefficient ranges being 0.8 – 0.95 (Kisakye and Van der Bruggen, 2018). **Table 2-3** highlights the common roofing materials and their corresponding runoff coefficients.

Table 2-3: Typical Roof Materials and their Runoff Coefficients (Worm, 2006)

Roof Material	Runoff Coefficient
Galvanized iron sheets	> 0.9
Tiles (glazed)	0.6 – 0.9
Aluminium sheets	0.8 – 0.9
Flat cement roof	0.6 – 0.7
Organic (thatched)	0.2

Limited research entailing conveyance subsystems has been done, although various methods from the Southern African Steel Construction Handbook can be used to size gutters and downpipes (Maharaj, 2020; Southern African Institute of Steel Construction, 2013). Worm (2006) suggests the use of splash guards on channels to avoid overshooting water losses. Overall, the gutter and downpipe analysis verifies if a proposed RWHS would effectively manage the rainwater transference for a chosen case study (Maharaj, 2020) but is not mandatory. The proper design of RWHS ensures efficient system functionality and considers other essential concepts like energy outputs, economic and environmental impacts, and water performance (Campisano et al., 2017; Devkota et al., 2015; Ward et al., 2012). Studies related to RWH have also incorporated life cycle assessments (LCA); therefore, it is necessary to understand the LCA concept and its link to RWH optimization.

2.4. Life Cycle Assessments

2.4.1. Background and Overview

A life cycle assessment (LCA) is a tool measuring the ecological impacts of a product or service (Devkota et al., 2015; Ecochain, 2020; Russell et al., 2005; Silva et al., 2017). It was developed in the late 1960s and early 1970s across Europe and the USA and involves the analysis of the creation/extraction, production/usage, and disposal (i.e., life cycles) of products, processes, and activities and their impacts on human health and the environment (Ghimire et al., 2017; Hauschild and Huijbregts, 2015; Unger et al., 2004).

In general, there are five steps of a product life cycle: raw material extraction, manufacturing and processing, transportation and distribution, usage and retail, and waste disposal (Ecochain, 2020; Goga, 2020; Yan et al., 2018). These five steps indicate a cradle-to-grave approach for assessing the environmental and human health impacts according to the stipulations by the International Organization for Standardization (ISO) (Goga et al., 2019; Ghimire et al., 2019; Russell et al., 2005).

An LCA consists of four phases that utilize inputs and outputs like raw materials, energy, water, and ecological emissions to assess environmental impacts (Angrill et al., 2012; Ecochain, 2020; Goga et al., 2019). Briefly, these four phases include the goal outline, inventory analysis, impact assessment, and result interpretation.

With the first LCA phase being the goal and scope definition (Ecochain, 2020), this stage attempts to define the purpose of the analysis and its intended application (Goga, 2020). The first phase incorporates the five steps of the product life cycle system while also using a functional unit for comparison (Goga, 2016). Hence, functional units act as the reference point for the inputs and outputs (Goga et al., 2019).

The second phase of an LCA is the life cycle inventory (LCI) analysis, which quantifies the elementary flows (or environmental inputs and outputs) of the system in relation to the functional unit (Ecochain, 2020; Hauschild and Huijbregts, 2015; Russell et al., 2005). This phase allows for data collection, which is then assigned to impact categories or indicators. Common impact categories include climate change, human toxicity, eco toxicity, ozone depletion, acidification, eutrophication, fossil fuel depletion, freshwater depletion, ionizing radiation, land transformation, land usage, and particulate matter formation. **Figure 2-5** shows the midpoint and endpoint impact categories often used in an LCA.

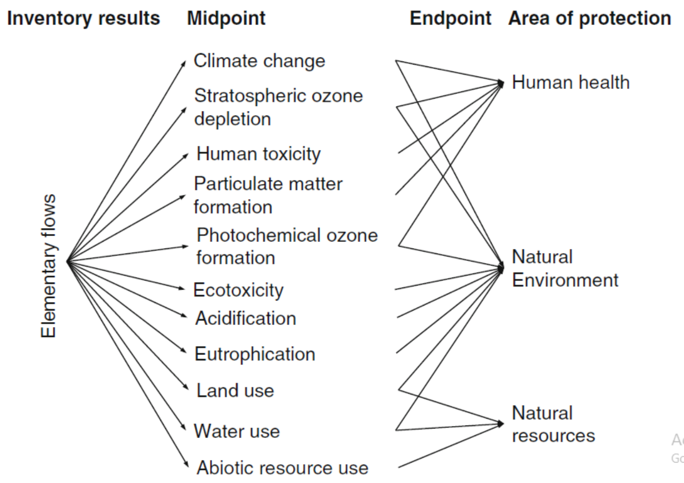

Figure 2-5: Impact Categories *(Hauschild and Huijbregts, 2015)*

The underlying principle of midpoint and endpoint impact categories is that midpoint indicators occur along the impact pathway, whereas endpoint indicators highlight protection areas (Hauschild and Huijbregts, 2015). Therefore, endpoint indicators are consequences of the midpoint indicators, which have themselves resulted from the elementary flows.

Given the vast array of products, processes, and activities for which an LCA can be used, organization and standardization of LCA methodologies became a challenge (Russell et al., 2005), with approaches only being standardized in a few impact categories/indicators as shown in **Figure 2-5** (Ecochain, 2020; Hauschild and Huijbregts, 2015).

A study by Ghimire et al. (2017), which compared a commercial RWHS to the municipal supply network, adopted impact indicators such as acidification, energy demand, eutrophication, fossil depletion, freshwater withdrawal, global warming, human health criteria, metal depletion, ozone depletion, smog, and evaporated water consumption; some of which, are similar to those presented in **Figure 2-5**. Another study by Angrill et al. (2012) on urban RWHS utilized slightly different indicators like abiotic depletion potential and photochemical ozone creation potential, among others.

Therefore, while impact categories vary according to literature-specific content and relevance, ISO still mandates them for any LCA analysis (Goga et al., 2019; Hauschild and Huijbregts, 2015). Impact categories are then integral to the third phase of an LCA: the life cycle impact assessment (LCIA). The LCIA phase aims to relate the product life cycle system (phase 1) and the potential environmental impacts based on the inventory setup (phase 2) using the chosen impact categories and a functional unit (Goga et al., 2019).

This phase incorporates the selection of impact indicators, the classification according to the LCI and impact categories, and finally, the impact measurement/characterization using impact category totals (Ecochain, 2020; Goga et al., 2019). Russell et al. (2005) have noted that LCI methodology can be attributional as described previously or consequential depending on possible changes in the chosen life cycle. Once the LCIA has been performed, interpretation of the final results can be made (Goga et al., 2019). Data interpretation can also be performed at any point during the methodology; however, when completed after the LCIA, the interpretation can be more conclusive (Ecochain, 2020).

2.4.2. LCA Software Tools

The concept of LCA is both intricate and comprehensive, often requiring in-depth analysis of vast amounts of data (Silva et al., 2017). Recently, LCA has become more prominent in literature and commerce, leading to the development of several software tools, which are used to facilitate and expedite the various processes timeously (Hafizi Md Lani et al., 2018; Silva et al., 2017; Unger et al., 2004). Moreover, software tools can be used in advance to optimize the design (Vieira et al., 2014).

These software tools include SimaPro, GaBi, openLCA, and Umberto, which provide a framework to perform life cycle assessments. Research on the development, methodology, and comparison of software tools is limited. According to Unger et al. (2004), several technical and methodological requirements are needed for the proper functioning of LCA software tools and are described subsequently.

The first requirement pertains to the structure and display of processes. Software tools usually contain databases and interfaces that model interlinking inputs and outputs to assess the environmental impacts (Unger et al., 2004). A study by Silva et al. (2017) also highlighted the importance of databases and interfaces as criteria for the qualitative comparison of LCA software tools. Unger et al. (2004) further suggest that software tools would be more efficient if they allowed for more than one output follow-up instead of being limited to one output scenario.

Other important considerations for software efficiency include transparency, flexibility, and user-friendliness of the tool (Unger et al., 2004). Support facilities are of paramount importance to the user as they provide ways to navigate the software for better understanding (Silva et al., 2017). The software's structure/layout may also aid with ease of use and should have options to display data graphically (Unger et al., 2004). Software compatibility with other applications is also necessary as it allows data to be transferred and presented in a conventional manner (Unger et al., 2004).

The different software tools can give result discrepancies according to the impact indicators (Silva et al., 2017). Calculation methods may also differ slightly between software tools, with specific devices incorporating more in-depth methodologies like uncertainty and sensitivity/variability analyses (Unger et al., 2004). Overall, software tools are handy for wide-ranging applications, including rainwater harvesting.

2.4.3. Life Cycle Assessments and Rainwater Harvesting

The theory of LCA discussed in Sections 2.4.1 and 2.4.2 outlines the benefits of such analyses and tools. Therefore, many studies have integrated LCA and RWH methodologies to gauge the environmental impacts of RWHS components.

According to Angrill et al. (2012) and Ghimire et al. (2017), an LCA provides a comprehensive representation of the planning, design, and decision-making for RWHS. Impact indicators form the LCA basis and are dictated by storage tank sizing, water collection rates, and pumping energy (Ghimire et al., 2019). Hence, to prove the effectiveness of RWHS, several studies incorporate LCAs specifically to assess and compare the environmental impacts of RWHS and centralized networks.

An example of this comparison is seen in the research by Devkota et al. (2015), which incorporated a life cycle-based evaluation for an RWHS implemented at a university dormitory in North America. The LCA study showed that the energy and greenhouse gas (GHG) emissions of the proposed RWHS displayed mostly favorable outcomes compared to the municipal supply. Similarly, Ghimire et al. (2017) performed an LCA of a commercial RWH in contrast to the municipal supply network in Washington, D.C. Here, the RWHS was shown to outperform the municipal water network in all impact categories except ozone depletion.

Research by Ghimire et al. (2014), which compared a domestic RWHS (DRWHS) and an agricultural RWHS to the municipal water delivery infrastructure, offers a further understanding of the environmental effects of RWH. The study found that a gravitationally induced DRWHS, used for non-potable purposes, surpassed the municipal network in every impact category except ecotoxicity. Hence, the LCA revealed the proposed RWHS to be more environmentally efficient than the municipal infrastructure (except in terms of adverse ecosystem effects).

The LCA comparison of a DRWHS to the municipal water supply was also undertaken for a case study in France. According to Vialle et al. (2015), RWHS implemented at a smaller scale (e.g., a household) have more favorable outcomes from an environmental standpoint. Despite this declaration, the study found that the DRWHS had a slightly higher ecological impact than the existing drinking water system. This impact was attributed to the electricity required for pumping.

Another study by Yan et al. (2018) evaluated the ecological impacts of an RWHS using an LCA. This research was also performed for a university building (office block); however, unlike the previous study, the focal point was adopting RWHS for potable uses. Hence, a more intensive treatment method was required for a smaller scale system. Contrary to the findings by Ghimire et al. (2017), Ghimire et al. (2014), and Devkota et al. (2015), this investigation by Yan et al. (2018) suggested that the planned RWHS would be less environmentally efficient than the current municipal supply setup due to the electricity consumption of the system. This finding is similar to the pump/electricity effects discussed by Vialle et al. (2015). Recommendations to improve the environmental performance emphasized the use of renewable energy resources.

Thus, it can be seen that several studies have compared a case-specific RWHS to a local municipal water network to evaluate the environmental efficacy of RWH. The outcomes of these investigations have varied, subsequently underlining the case-specific efficiency of RWHS. Furthermore, LCAs are not necessarily confined to the comparisons between RWHS and municipal supply networks. However, few studies examine LCA and RWH outside of this context. Fundamentally, through the research reviewed, LCAs have proven to be valuable tools in implementing and optimizing RWHS.

2.5. Optimization of RWHS

2.5.1. Performance of RWHS

RWH habitually considers cost, reliability, performance, and design for its optimization (Semaan et al., 2020). Correspondingly, Marteleira and Niza (2018) surmise that RWHS studies tend to incorporate tank sizing, hydrological investigations, energy impacts, GHG emissions, and financial deliberations to provide conclusive evidence for or against system implementation.

Rainwater harvesting systems do offer a multitude of benefits, with several studies highlighting their potential for urban flood reduction (Freni and Liuzzo, 2019; Torres et al., 2020), especially if implemented at a broader scale (Hafizi Md Lani et al., 2018; Słyś and Stec, 2020). Other benefits include the reduced strains on municipal supply networks, which allows for increased infrastructure lifespans and the conservation of freshwater resources (Hafizi Md Lani et al., 2018; Vialle et al., 2015; Vieira et al., 2014). In this way, decentralized RWHS provide an alternative supply source and could allow for partial or complete independence from centralized networks (Hafizi Md Lani et al., 2018; Shaari, 2020; Stec and Zeleňáková, 2019; Sunar et al., 2019)

However, the underlying reason for RWH usage is in the water-cost saving potential for the end-user, which acts as the incentive for domestic, agricultural, and commercial implementation. Therefore, it becomes imperative to assess the water performance of RWHS in terms of its saving efficiency, applications, and quality. Rainwater stored may be allocated for several water activities, with greater collections and efficient usage correlating to higher water savings (Maharaj, 2020). Several studies have analyzed the water performance of RWHS by considering the water-saving efficiency (WSE) for specific water activities.

2.5.1.1. Water Saving Efficiency

According to Mun and Han (2012), WSE is a vital operational parameter for an RWHS. It indicates the proportion of the total water demand that is satisfied by the collected rainfall. The research employed a ratio between the tank volume (V) and the catchment area (A) to ascertain the optimal design for rainwater use efficiency (RUE), i.e., the effective use of rainwater.

The ratio of V/A was found to be proportional to both WSE and RUE. Logically, collecting more rainwater and using it effectively would increase the savings. Thus, the study was able to show that an RWHS (implemented at a Korean university dormitory) would be optimized by expanding the rainwater applications, thereby increasing the RUE. Furthermore, the authors stated that the design and evaluation methods used could be applied to other regions irrespective of water abundance or scarcity since RWH improves water security as a whole.

Another study by Stec and Zeleňáková (2019) assessed the water-saving potential of RWHS that would be implemented at student dormitories in Poland and Slovakia. It was found that the Slovakian facility offered higher potential water savings (about 29%), while 18% in water savings was achieved at the Polish dormitory. It was further noted that the RWHS would be limited to toilet flushing alone; hence, if the water uses were to be extended to other activities, the RWHS could have potentially higher water savings. However, much debate about potable and non-potable usage is present (Hofman-Caris et al., 2019)

2.5.1.2. Water Quality

In analyzing the activities for which harvested rainwater is used, water quality becomes a crucial consideration. Słyś and Stec (2020) have specified that the harvested rainfall quality depends on factors like air quality, catchment condition, and general maintenance of the RWHS. Nguyen and Han (2017) have also stated that a properly designed, well-maintained RWHS has the potential to be a source of potable water without treatment. Conversely, Malema et al. (2016) have argued the inevitability of harvested rainwater contamination, especially during the collection and storage stages.

While various methods are available to cleanse captured rainwater, Stec and Zeleňáková (2019) estimate that 50% of domestic water usage does not need extreme treatment methods. Therefore, activities like toilet flushing, garden irrigation, and car washing require less extensive treatment as opposed to potable uses (Campisano et al., 2017; Hafizi Md Lani et al., 2018; Vieira et al., 2014).

According to Helmreich and Horn (2009) and Campisano et al. (2017), preliminary treatment occurs through the pre-filtration mechanisms like first flush diverters, leaf eaters, and tank screens. However, these mechanisms are insufficient against microbial contaminants, which can cause severe health risks if untreated rainwater is ingested (Malema et al., 2016; Malema et al., 2018). Therefore, some post-storage treatment options include disinfection and filtration (Campisano et al., 2017; Hafizi Md Lani et al., 2018).

Disinfection is commonly achieved using chlorination tablets with typical dosages ranging from 0.2 – 0.5 milligrams per litre (mg/l) of water (Helmreich and Horn, 2009;

Malema et al., 2016). Other disinfection methods include solar and ultraviolet (UV) lamp techniques that render microorganisms ineffectual (Malema et al., 2016). Treatment by filtration usually involves slow sand filtration (Campisano et al., 2017); however, this method may only reduce microbial contaminants and not eliminate them (Helmreich and Horn, 2009).

The treatment of harvested rainwater often holds more significance in rural areas and developing countries (Pachpute et al., 2009; Shaari, 2020). Impoverished communities tend to rely on RWH for potable water consumption (Kahinda et al., 2007; Kisakye and Van der Bruggen, 2018; Malema et al., 2016; Malema et al., 2018), while people from urban areas usually adopt RWHS for supplementing and non-potable uses (Malema et al., 2016; Marteleira and Niza, 2018). Hence, quantity and quality needs vary between developed and developing countries (Kisakye and Van der Bruggen, 2018).

Likewise, research by Malema et al. (2018) accentuated rural RWHS adoption and its relation to antimicrobial (antibiotic) resistant E. coli. The study found that RWHS that are not adequately treated, monitored, and maintained have the potential to become breeding grounds for pathogenic diseases. Thus, in terms of quality, the water performance of RWHS relies on effective management by its water users.

2.5.1.3. Improved Systems

Although RWH has also been implemented to improve water security in a changing climate, system functionality depends entirely on rainfall. A study by Kisakye and Van der Bruggen (2018) outlined this concept by suggesting that the water savings from RWHS could become less effective in certain months due to irregular rainfall patterns in Uganda. Climate change is a pivotal driver for erratic rainfall, water security, and scarcity, with compounding effects in the rural context. Since many developing countries adopt RWHS for water access and supply, unpredictable climate concerns can shift rainfall patterns and heighten scarcity. In turn, this may require increased water storage times to allow for water availability during drought periods. Hence, the research suggested a seasonal analysis to optimize RWH design, factoring in the dry periods as the worst-case scenarios.

The optimization of water performance within RWHS can also be undertaken by comparing existing/conventional RWHS to improved systems. An example of this is a study by Roman et al. (2017), which aimed to improve urban RWHS through a continuous monitoring and adaptive control approach. The research showed that an enhanced RWHS would primarily minimize sewer discharge and reduce municipal water for irrigation. The significant findings also highlighted better water performance by the improved RWHS, allowing for higher runoff capture and retention. More specifically, it was found that the enhanced RWHS would be able to capture and retain 76.6% of the roof runoff. In comparison, the conventional moisture and timer-based RWHS would only capture and retain 14.8% and 41.3% of the runoff, respectively.

It has been revealed that the adoption of RWHS on a broader scale would increase system sustainability to counteract municipal strains and urban flooding. According to Adugna et al. (2018), public institutions may have increased merit for RWHS due to larger catchment areas. Furthermore, such facilities could promote the widespread usage of RWHS. Therefore, the study concluded that RWHS could replace 2.3% of the urban water supply in large public institutions in Addis Ababa, Ethiopia.

Furthermore, urban water supply by conventional, centralized systems tends to be more energy demanding (Marteleira and Niza, 2018). Likewise, Chiu et al. (2015) have stated that centralized urban water supply is not energy-efficient, thereby compounding the water-energy nexus. Hence, Chiu et al. (2015) focused on implementing RWHS on a city-scale in the Taipei metropolitan area in Taiwan. In terms of water performance only, their findings indicated that RWHS would allow for approximately 21% in domestic water savings. The proposed RWHS would be used for non-potable purposes, thereby simplifying treatment and distribution. However, the study acknowledged RWHS constraints due to spatial and temporal rainfall variability.

To address the variability of rainfall in time and space, Kahinda et al. (2008) proposed the idea of suitability maps for RWH in South Africa. The research's premise was to identify specific rainfall locations that would be most suitable for agricultural RWH implementation. Six vital factors were considered when determining the potential RWH sites. They included climate, hydrology (runoff and watercourses), topography, crop characteristics, soil detail, and socio-economic considerations (Velasco-Muñoz et al., 2019).

Rainfall variability has also been considered in the urban context. Torres et al. (2020) proposed a computational geographic information system (GIS) to ascertain the optimal placement points of RWHS. The research highlights the possibility of collaborative GIS frameworks and stochastic methodologies to incorporate the uncertainty of rainfall. Hence, optimally placed RWHS would maximize benefits such as municipal water savings and reduced urban runoff. The study results (tested on a Colombian university campus) indicated reductions in potable water consumption and total runoff volume by up to 50% and 67%, respectively. Hence, RWH was considered an efficient component of sustainable urban drainage systems (SUDS).

2.5.1.4. RWH for toilet flushing

As global domestic water usage has been on the upsurge in recent years (Donnenfeld et al., 2018), a common point of study for RWHS has been in the household toilet flushing setup. According to Mudombi and Montmasson-Clair (2020), toilet flushing accounts for 40% of domestic water use on average. Hence, RWHS designed to cater to and supplement toilet-flushing demands may exhibit better water performance (in terms of water-saving potential and system efficiency).

An older study by Vialle et al. (2011) found that the WSE for a DRWHS (used for toilet flushing) would be 87%. This percentage translated to a potential water saving of 42 kL (m^3) per annum by the centralized network. Similarly, research by Maharaj (2020) assessed RWH in a domestic context and found that the WSE for a toilet flushing RWHS was 46.1%, amounting to 50.2 kL/year in municipal water savings.

Słyś and Stec (2020) extended the WSE evaluation by considering centralized and decentralized DRWHS. The study found that when utilizing harvested rainwater for toilet flushing alone, the WSE was 80% and 79% for the decentralized and centralized systems, respectively. The WSE then dipped to 57% (for the decentralized system) and 54% (for the centralized system) when irrigation was also considered. Hence, this research outcome showed that WSE decreased when more water applications/activities were proposed, provided that the same storage considerations were used.

2.5.1.5. Tank sizing

Storage is an essential aspect of the RWH design; hence, optimal sizing improves the system's capability. Optimization based on sizing is generally a compromise or trade-off between harvested rainwater use, system reliability/functionality (i.e., its ability to satisfy user needs), and spatial layout for installation (case variable) (Marteleira and Niza, 2018). However, optimization of RWHS is not limited to tank sizing alone (Semaan et al., 2020).

A study by Freni and Liuzzo (2019) found that RWH tanks (with a capacity of 5 kL or more) have the potential to reduce peak flows in a flood-susceptible residential area in Southern Italy, especially if implemented on a wide scale. However, the authors stated that each household might have case-specific properties; hence, the optimization of RWHS through tank sizing could vary accordingly. This correlates with the writings by Semaan et al. (2020), which states that optimal tank sizing is more favourably adopted in the local setting as opposed to global conditions. Semaan et al. (2020) further state that storage sizing is the most critical objective for optimization as an oversized or undersized tank reduces system efficiency/water performance.

A study by Khan et al. (2017) also utilized software tools for optimal tank sizing in relation to variable catchment areas and rainfall intensities in Bangladesh. Historical rainfall data and a seasonal analysis were incorporated into the study. Historical rainfall records are often used to estimate tank size; however, Semaan et al. (2020) state that future studies will need to deliberate the changing climate and water demands to optimize RWHS. This concept relates to the study by Kisakye and van der Bruggen (2018), which outlined the fluctuating efficiency of RWHS due to climate change and sporadic rainfall patterns.

Thus, from the analysis of the selected literature, it can be seen that RWHS are water-reliable investments that allow for municipal water savings. While the water-saving potential varies according to local factors, some research has shown RWH to substitute up to 80% of potable municipal supply (Torres et al., 2020). Moreover, RWH seems to offer opportune advantages based on the climatic and social conditions within eThekwini. According to the NWA (1998), SA citizens have the right to practice RWH (Colvin et al., 2016), although; clearer legislation is needed on a national level (Adugna et al., 2018; Kahinda et al., 2007; Malema et al., 2016).

2.5.2. Economic Analysis of RWHS

While rainwater harvesting provides ample benefits in terms of physical water collections, usage, and savings, capital investment is required for its implementation. Economic viability may vary according to local characteristics, with many studies displaying conflicting results (Rahman, 2017). Hence, Christian Amos et al. (2016) suggest the need for standardization of economic analyses in the RWH setting.

2.5.2.1. Economic Feasibility Factors

Semaan et al. (2020) estimate that (on average) the capital costs of RWH account for 80 – 82% of the system life cycle costs. Other associative expenses include maintenance and operation from pumping and disinfection. According to Rahman (2017), the overall feasibility of RWH implementation can be underestimated when considering monetary savings only. Therefore, studies tend to relate the initial investment costs with municipal cost savings to determine the repayment period of a proposed RWHS (Campisano et al., 2017; Maharaj, 2020; Marteleira and Niza, 2018; Semaan et al., 2020). The price of municipal water is a central factor for the system financial analysis (Campisano et al., 2017); hence, harvested rainwater may offer greater economic benefits for countries with higher water tariffs (Hafizi Md Lani et al., 2018; Stec and Zeleňáková, 2019; Vieira et al., 2014; Zabidi et al., 2020).

Other factors contributing to economic feasibility include frequent rainfall and proper tank sizing since greater water collections are proportional to higher cost savings (Christian Amos et al., 2016). RWH could also incorporate energy inputs (in kilowatts-hour) from pumping, which have economic (and environmental) impacts. Furthermore, research has suggested that RWHS are more economically viable when considering both alternative water supply and flood control (Hafizi Md Lani et al., 2018). Therefore, it can be seen that the economic considerations are multi-faceted, drawing benefits beyond cost-savings. Generally, the components of an RWHS (Section 2.3.2) form the basis of capital investigations, whereas operation and maintenance (which is the onus of the end-user) are not usually considered due to negligible effects on economic feasibility (Semaan et al., 2020; Sunar et al., 2019; Ward, 2012). Depending on the component materials, initial costs could be reduced; however, defective materials may lead to pollutant and treatment issues in the future (Hafizi Md Lani et al., 2018), incurring further costs.

2.5.2.2. Municipal supply vs. harvested rainwater

A study by Christian Amos et al. (2016) aimed to assess the economic aspects of DRWHS in urban and peri-urban environments. The financial analysis was performed through a life cycle cost (LCC) analysis with general findings indicating that harvested rainwater is more expensive than municipal water in Australia (urban environment). However, the study also concluded that should capital costs be covered by government subsidies and rebates, financial benefits would be achieved for water users carrying the operation and maintenance costs of the systems. Hence, this finding shows the negligible influence of operation and maintenance expenses on economic viability. Furthermore, RWHS that are fully or partially funded by governmental schemes may hold economic viability, especially in the rural context where people may not be able to afford it (Hafizi Md Lani et al., 2018).

According to Hafizi Md Lani et al. (2018), it is essential to consider treatment costs for sustainability and human health. The study by Hofman-Caris et al. (2019) broaches this concept while also comparing the costs of harvested rainwater to the municipal supply. The research found that similar prices (per m^3 of water) occurred for RWHS and centralized supply networks in a city district in the Netherlands.

However, the proposed RWHS would only cater to half the total water demand due to harvested volumes. When considering an individual household, the costs were much higher (approximately 60 – 110 Euros/m^3 or R1030 – R1888/m^3). Furthermore, the research showed that treating the harvested rainwater to a potable standard compounded the economic impracticality. Thus, the authors recommended an RWHS as a supplementing source of water.

2.5.2.3. Tank Considerations

Słyś and Stec (2020) also showed RWHS to be economically unviable in their research. Recalling that their study compared centralized and decentralized RWHS for a single household, it was found that co-financing investments ranging from 25% - 50% would be needed for economic viability. This is similar to the findings by Christian Amos et al. (2016), which showed that financial aids would improve economic feasibility for the end-user. Słyś and Stec (2020) also indicated that the tank capacity would factor in improving economic practicality. Given that the cost of the tank is usually the highest portion of total capital expenses (Khan et al., 2017; Semaan et al., 2020; Worm, 2006), an optimally sized tank catering for water performance savings and cost efficiency is essential (Rahman, 2017).

A study by Fernandes et al. (2015) assessed an RWHS for a waste treatment facility in Mirandela, Portugal. In this research, tanks were deemed oversized and reduced to capacities that provided cost and system efficiency. From an economic standpoint, the study concluded that RWHS have the potential to save 1300 – 2100 Euros per annum (approximately R22 327 – R36 067 per year) with reasonable payback periods ranging from 5 – 8 years. Another Portuguese study by Matos et al. (2015) evaluated the cost efficiency of an RWHS for a commercial building in Braga. The research also highlighted low payback periods of 2- 6 years as a significant indicator for economic feasibility.

2.5.2.4. Energy Aspects

Energy considerations under RWHS are also indicators of economic viability. Chiu et al. (2015) stated that RWHS might be more economically feasible when considering both water and energy savings due to the water-energy nexus. Recalling that their research focused on GIS-based RWH design in metropolitan Taiwan, the water savings of 307.173 m^3/year resulted in 561.438 kWh/year energy savings. Thus, the research evaluated the long-term water and energy benefits to warrant the system investment. Alternatively, some studies may analyze the direct energy usage of RWHS and relate energy to costs.

Depending on case-specific circumstances and considerations, pumps may be needed to transfer harvested rainwater to the application point. Hence, some studies analyze the energy impact of the pump (Maharaj, 2020; Ward et al., 2012). Hafizi Md Lani et al. (2018) suggest that the energy efficiency of a pump increases when the storage and application points are in close proximity. Thus, it becomes essential to place an RWHS in an optimal location. Optimization of the RWHS would also be achieved through a gravity-fed system that eliminates the energy, cost, and carbon footprint from pump operation and maintenance (Hafizi Md Lani et al., 2018; Maharaj, 2020).

In a study by Maharaj (2020), a pumped DRWHS was proposed for a middle-income house in Durban, South Africa. It was found that the R9854 system would prompt R1176 in water-cost savings per annum, giving a reasonable return period of 8.4 years. Furthermore, the pump contribution to the existing energy usage was deemed insignificant since it would only increase the electrical expenses by 0.4% per year. Hence, the researchers concluded that the proposed DRWHS would be financially viable.

2.5.2.5. Domestic Setting

The implementation of RWHS are popularly adopted at university campuses. It has been reiterated that the financial feasibility of RWHS varies from case to case. Stec and Zeleňáková (2019) studied this concept by evaluating the profitability of two RWHS implemented at different student dormitory complexes in Poland and Slovakia. It was found that the proposed RWHS for the Polish dormitory would be economically impractical for all the stipulated scenarios, mainly due to the extensive payback periods of more than 30 years. Conversely, the Slovakian student dormitory offered favourable economic assessments for RWH. Hence, within one study, the economic perspectives on RWH were shown to differ.

Another study by Marteleira and Niza (2018) showed that an RWHS implemented at a Lisbon university would be economically viable for non-potable water uses. Potable uses of harvested rainwater were not widely accepted, especially if functional and affordable urban networks were available. The research used a holistic tool (RaINvest) to assess the system sizing, energy outputs, and overall feasibility.

The study showed that implementation by the local water utility was viable but would be more economically feasible from the user perspective (i.e., users should provide for the costs of implementation to realize the benefits in the long term). This outcome is somewhat contrary to the research by Christian Amos et al. (2018), and Słyś and Stec (2020), which encouraged government subsidies for the capital costs of implementation. The investment return period was calculated as 12 years (deemed satisfactory), and the energy efficiency of the proposed RWHS was shown to be better than the existing centralized system. The future scope of this research was implementing RWH at a broader, urban scale to verify if the financial benefits would also be upscaled.

Assessing the viability of RWHS in the urban setting has been a common theme in the selected literature. A study by Fisher-Jeffes et al. (2017) followed this trend by evaluating the implementation of DRWHS in the Liesbeek River Catchment, Cape Town. The study showed RWH as an unreliable means for urban flooding in the Liesbeek River Catchment, which is contrary to claims by Freni and Liuzzo (2019), and Torres et al. (2020). However, similar to the research by Kisakye and Van der Bruggen (2018), this research emphasized fluctuating rainfall, runoff volumes, and volumetric reliability due to climate change. Furthermore, economic viability would only be present for a minority of the property owners, mainly due to their income status. Therefore, the research also recommends grants and subsidies for more impoverished households to improve economic viability. Alternatively, the authors proposed increased water tariffs (by three to four times) to incentivize RWH for the case study.

2.5.3. Environmental Analysis of RWHS

The water performance and economic benefits (case-specific) often serve as the major enticements for RWH implementation. Depending on local characteristics, RWH has been found to be economically viable in certain studies. However, it also becomes necessary to assess the environmental performance of these systems. RWHS that offer benefits like water savings and cost savings can be sullied by any detrimental ecological impacts. Hence, the environmental considerations for RWH have garnered much research, with selected case studies presented in this section.

The environmental performance of RWHS is often researched in two settings: greenhouse gas (GHG) emissions and energy contributions. While some studies compare the GHG emissions of RWH operation to the centralized networks (Devkota et al., 2015; Maharaj, 2020; Valdez et al., 2016), others adopt LCAs to ascertain the ecological impacts of RWHS throughout their life cycles (Ghimire et al., 2017; Vialle et al., 2015). Energy-related aspects from pumping may also present environmental impacts, mainly in the form of carbon dioxide emissions (Ward et al., 2012). Hence, studies tend to identify the carbon emission savings as a verification tool for environmental viability (Maharaj, 2020)

2.5.3.1. Energy and Carbon Footprints of RWHS

According to Vieira et al. (2014), the energy intensity (or unit energy used per unit water) of an RWHS is vital for assessing its environmental feasibility, particularly within the water-energy nexus. Gauging the energy outputs of RWHS may also factor into system optimization through new and improved technologies (Ward et al., 2012). Recalling the water performance of RWHS in Section 2.5.3, the physical savings introduced by RWHS may also correlate to carbon emission savings by the use of a carbon emission factor (Friedrich et al., 2009a; Maharaj, 2020). Therefore, it can be seen that energy also plays a vital role in the environmental feasibility of RWHS.

In a study by Ward et al. (2012) that utilized LCA methodologies for an office building, it was found that UV disinfection and pumping provided the core operational energy input for the significant carbon emissions. The research offered an improved method to estimate energy consumption and carbon emissions by analyzing pump start-up and operation energy consumption differences. The study showed that a more straightforward approach could underestimate energy consumption and carbon emissions by 60%; hence, the need for an improved method.

The research also found that for every cubic metre of harvested rainwater pumped (utilized rainwater), the energy consumed would be 0.54 kWh. Similarly, the CO_2 equivalent emissions were calculated as 0.56 kg $CO_2e/m^{3,}$ meaning the electrical output of the RWHS produced 0.56 kg of CO_2 equivalents whenever a cubic metre of rainwater was pumped. Thus, the study concluded that the energy of the proposed RWHS was marginal in comparison to the existing energy consumption of the office, utilizing 0.07% of the total energy consumption. Furthermore, a gravity-fed system was recommended to increase the energy efficiency of the RWHS, thereby improving environmental feasibility.

Another study that follows similar themes to that of Ward et al. (2012) is the research by Vieira et al. (2014). For their study, the authors reviewed the energy intensity of RWHS and highlighted the underestimation of pump energy consumption in empirical contexts. Hence, RWHS could potentially be optimized by reducing the energy outputs of the pumping component. Recalling that pump energy consumptions can be underestimated by up to 60% (Ward et al., 2012), the actual pumping consumption would serve as a reference point towards energy efficiency and optimization. Furthermore, poorly selected oversized pumps are more energy-rigorous; therefore, proper design considerations should be enacted.

The study also reviewed the energy intensity of RWHS in comparison to conventional municipal supply networks. It was found that non-optimized RWHS could be three times more energy-intensive than urban supply networks. However, the study accentuated local characteristics like rainwater demand, building type, sub-system design, urban network design, and water energy intensity as representative factors for environmental and economic viability.

Furthermore, the rainwater consumption from user demand may influence tradeoffs between water consumption and energy consumption. Thus, the authors offer recommendations to improve energy efficiency, thereby optimizing RWHS. An example is an optimization via the catchment area. Properly maintained RWHS would not require rigorous treatment (Nguyen and Han, 2017), thus mitigating the need for energy-intensive treatment methods like UV disinfection. Gravity systems and header tanks were also suggested to improve energy efficiency that emulates urban water supply systems. Enhanced planning and design for RWH optimization were laid out as the future scope.

The optimization of RWHS through optimal design has been alluded to previously. RWHS, if designed correctly, are capable of producing CO_2 emissions that are lower than municipal supply emissions (Marteleira and Niza, 2018; Zabidi et al., 2020). Correspondingly, Hafizi Md Lani et al. (2018) suggest RWHS have lower carbon footprints than other water supply systems. Since energy and GHG emissions are intricately related, a more energy-intensive system will likely have higher carbon emissions generated from energy production. Hence, the claims by Hafizi Md Lani et al. (2018), Marteleira and Niza (2018), and Zabidi et al. (2020) seem to contrast the findings by Vieira et al. (2014), who stated RWHS could be up to three times more energy-intensive than municipal networks.

2.5.3.2 Life cycle assessments in the RWH context

In Section 2.4.3, several studies pertaining to municipal supply networks, LCA, and RWH were discussed (Devkota et al., 2015; Ghimire et al., 2014; Ghimire et al., 2017; Vialle et al., 2015; Yan et al., 2018). The study by Yan et al. (2018) showed environmental impracticality by using the LCA methodology. In contrast, the studies by Ghimire et al. (2014), Ghimire et al. (2017), and Ghimire et al. (2019) highlighted RWHS as being environmentally feasible. According to Ghimire et al. (2019), RWH can reduce the environmental impacts of centralized systems and is more effective when implemented with other water conservation techniques and green practices.

LCAs adopted in research also offer versatile approaches that can be tailored to assess the environmental impacts from any or all stages of the life cycle process, i.e., material extraction, processing, transportation, operation, and usage. An example of this is the study by Valdez et al. (2016), which incorporated an LCA for every stage of the RWHS life cycle. Valdez et al. (2016) were able to show that the emissions from raw materials and construction accounted for less than 7% of the total life cycle impacts.

Once again, the energy consumption and GHG emissions of the municipal water grid were compared to various RWHS configurations in Mexico City. It was found that municipal water provision consumed 1.23 kWh for every cubic metre of water supplied. Hence, the study showed that RWH could reduce GHG emissions from the centralized network in Mexico City for specific building configurations, thereby reducing global warming potential (GWP). Since electricity consumption was shown to be the most significant GWP contributor, gravity-fed RWHS could potentially reduce GWP by between 0.9% and 18.4% for the 11 building scenarios.

Recalling the study by Devkota et al. (2015), which modelled five scenarios for RWH in the University of Toledo, North America, an LCA showed RWH to be environmentally viable for most of the simulated models. However, other human activities outside of water supply were shown to produce greater energy consumptions and GHG emissions, suggesting a slight reduction in global GHG emissions through RWH. Furthermore, while the costs of the proposed system were not viable for certain scenarios, realistic energy and GHG emission payback periods were achievable, thus substantiating the local environmental expediency.

This finding is contrary to the study by Maharaj (2020), which established economic capability but not environmental viability for a DRWHS. Due to the presence of a pump, carbon emissions of 35.8 kg CO_2/year counteracted and exceeded the system carbon emission savings of 20.5 kg CO_2/annum, making the proposed RWHS environmentally unviable. Once more, a gravity-fed system was recommended to eliminate the need for the pump, thus removing the carbon emissions to attain environmental feasibility.

Another study in the domestic RWH context is by Vialle et al. (2015). This research was briefly discussed in Section 2.4.3, as it incorporated an LCA to assess the environmental impacts of a DRWHS. When comparing the municipal network and DRWHS, the study showed RWH to have a slightly higher ecological implication. This was attributed to the pump energy consumption. Furthermore, the authors stated that specific disinfection methods (like UV lamps) could increase the negative environmental impacts. Hence, non-potable uses like toilet flushing would be more favourable in this regard.

In other research by Rashid et al. (2016), the authors compared the environmental effects of a gravity-fed RWHS to a pumped RWHS using an LCA. Logically, the findings indicated fewer ecological impacts from the gravity-fed system since pumping required energy input. The research also emphasized environmental impacts from tank sizes since a larger tank constitutes more processed material. Certain materials like high-density polyethylene (HDPE) are not environmentally friendly; hence, recommendations to replace tank materials with more ecologically suitable materials could potentially lower the environmental effects and optimize the RWHS. Angrill et al. (2012) reiterate tank materials and their life cycles as critical factors towards the ecological optimization of RWHS. Nonetheless, the literature review by Rashid et al. (2016) indicated that RWHS that are financially viable might not be environmentally feasible. This finding was also present in the research by Maharaj (2020). Hence, the link highlights the tradeoff between economic and environmental features.

2.5.3.3. Context Specificity

According to Devkota et al. (2015), water applications in particular settings may regulate the cost, energy, and GHG viability. For their study, harvested rainwater used for irrigation was more economically feasible for renovation projects, whereas harvested water for toilet flushing showed practicability in a new construction project situation. Angrill et al. (2012) revealed a similar verdict in the research, in that RWHS implemented with new infrastructure development displays fewer carbon emissions as opposed to retrofitted RWHS.

The research by Angrill et al. (2012) aimed to identify the most environmentally efficient RWHS in Mediterranean urban contexts under different population densities. The study concluded that environmental viability incorporated economic, social and technological factors. Design technologies would need to integrate ecological impacts to optimize RWHS. In terms of the two population densities considered (i.e., compact and diffuse/spread-out), the research found that compact urban density models generated lower environmental impacts. Finally, the future scope recommended by the authors included optimization of RWHS via tank placement and proximity to water usage points, as well as economic integration and social acceptance aspects.

In the South African context, the economic and environmental viability of RWHS may hold special status. SA already has substantial CO_2 emissions, primarily from coal-based energy plants (Knegtel and Naidoo, 2014). Thus, environmentally feasible RWHS should be implemented to avoid compounding the carbon emission effects. SA is also a developing country with many citizens living in impoverished conditions. Subsidized RWH schemes could cater to such communities, making RWH more widely available (Kahinda et al., 2007). Extensive implementation would also lead to water-stress alleviation since SA suffers from physical and economic water scarcity (Donnenfeld et al., 2018). According to Colvin et al. (2016), RWHS specifically fitted for schools has enormous water-saving potential. It is estimated that a 90mm downpour rainfall event over 48 hours can yield 73 kL of rainwater, given a roof catchment area of 376 m^2. Overflows would be used for irrigation purposes, thus encouraging green principles. Furthermore, the implementation of RWHS in educational settings may instill RWH ethos amongst the youth, thereby safeguarding future water conservation and environmental sustainability.

2.6. Summary

Water conservation and demand management have become increasingly important with population growth, urbanization, and industrialization. Human activity, in general, has accelerated climate change leading to variable rainfall conditions and strained water resources. In attempts to deal with the water scarcity crises from municipal supply networks, dam construction, desalination, and rainwater harvesting are a few of the engineered technologies that have been favoured. RWH, in particular, offers multiple benefits like free water usage, municipal water and cost savings, relief on metropolitan supply strains, and on-site water storage for a variety of building types.

However, water quality issues that may limit water applications often juxtapose the convenience of RWHS. Potable water uses from harvested rainwater necessitate more intensive treatment that may not be economically feasible. The economic viability of RWHS is an essential consideration as this factor tends to promote or reject system implementation. RWHS already consist of several components or subsystems, and while some of these features may already be present in buildings, the costly subsystems like the storage tanks require financial input.

Thus, many studies have opted for governmental financial aids and subsidizations as ways to promote usage and financial practicality for more impoverished communities. In this way, more widespread RWH usage would offer increased benefits in the long term. Ultimately, the economic considerations are only a part of the RWH viability. Environmental impacts also play a critical role in RWH implementation. In this regard, LCAs are helpful assessment tools that gauge the ecological effects of RWH subsystems throughout their life cycles, i.e., from raw material extraction to construction and processing and, finally, usage and disposal.

It is often found that pumping and treatment display the inept energy and carbon footprints of RWHS; hence, reducing the effects of these facets promotes environmental efficiency and viability. In light of the ecological benefits, inevitable economic tradeoffs and compromises could be experienced, thereby complicating optimum RWHS. Optimization of RWH extends further into design and placement considerations, particularly in terms of the tank size. Thus, it can be seen from the reviewed literature that the implementation and optimization of RWHS is not a straightforward solution but requires methodical planning and analysis under specific circumstances.

Therefore, to better understand system optimization, RWHS should be implemented in a case-specific context. Thus, this research opted to design and optimize RWHS in a school setup, primarily focusing on toilet flushing as the main water-saving activity. LCA methods would also be adopted as part of the environmental analysis of the RWHS. The details of the chosen case study are presented in the next chapter (Chapter 3), and the procedure for the research is specified in Chapter 4.

CHAPTER 3: CASE STUDY

3.1. Introduction

Rainwater harvesting is a complex endeavor that offers many benefits, particularly in non-agricultural and non-industrial environments. Hence, the primary case study chosen for this research was a public school (Duffs Road Primary) falling under the jurisdiction of the eThekwini Municipality. The water consumption in a school setup is mainly attributed to toilet flushing, hand washing, showering, and irrigation (Morote et al., 2020). However, not all schools have the same resources and facilities; hence, water consumption (demand) may vary depending on affluence or poverty circumstances. Thus, this chapter details the specific features of the case study, including the municipality description, the case study site description, location, layout, water usage, and proposed RWH considerations. The case study limitations conclude this chapter.

3.2. eThekwini Municipality

3.2.1. General

The eThekwini Municipality is one of eight category A (metropolitan) municipalities in the country (Republic of South Africa, 2021). The municipality operates in KwaZulu Natal, with the topography in this region, including many hills and ravines (KZN Online, 2020; Municipalities of South Africa, 2021). Durban forms the largest city under the jurisdiction of the eThekwini Municipality, with many other towns and suburbs also present.

The performance of the eThekwini Municipality has already been discussed in **Section 2.2.4.7** of Chapter 2, highlighting the need for water conservation practices like RWH. Since the municipality is located along the eastern side of SA, which is characterized as a summer rainfall region (Friedrich et al., 2009a), most parts of the region receive higher rainfall from November to March (Weather and Climate, 2021). Conversely, the lowest precipitation values occur in the winter months of June and July. Hence, when adopting RWH within the eThekwini Municipality, it is essential to consider this seasonal rainfall variability as it indicates the months in which such systems would be optimized. A summary of the eThekwini Municipality characteristics is shown in **Table 3-1**.

Table 3-1: eThekwini Municipality General Characteristics (Department of Cooperative Governance and Traditional Affairs, 2020; KZN Online, 2020; Municipalities of South Africa, 2021)

Area	2556 km^2
Population Size	3.99 million
Main Economic Sectors	Finance (22%) Manufacturing (22%) Community Services (18%) Trade (16%) Transport (16%) Construction (3%) Electricity (2%)

3.2.2. Layout

Figure 3-1 shows the location of the eThekwini Metropolitan Municipality relative to the ten district municipalities and 43 local municipalities in KZN.

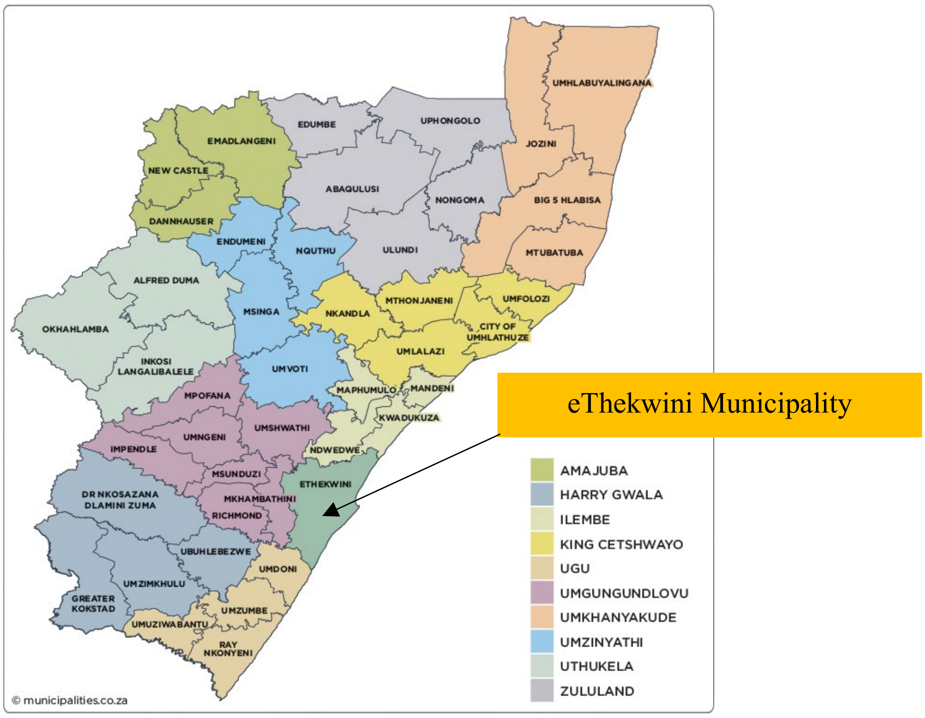

Figure 3-1: KZN Municipalities (*Municipalities of South Africa, 2021)*

Figure 3-2 magnifies the eThekwini Municipality layout first presented in **Figure 3-1** and also shows the approximate position of the case study.

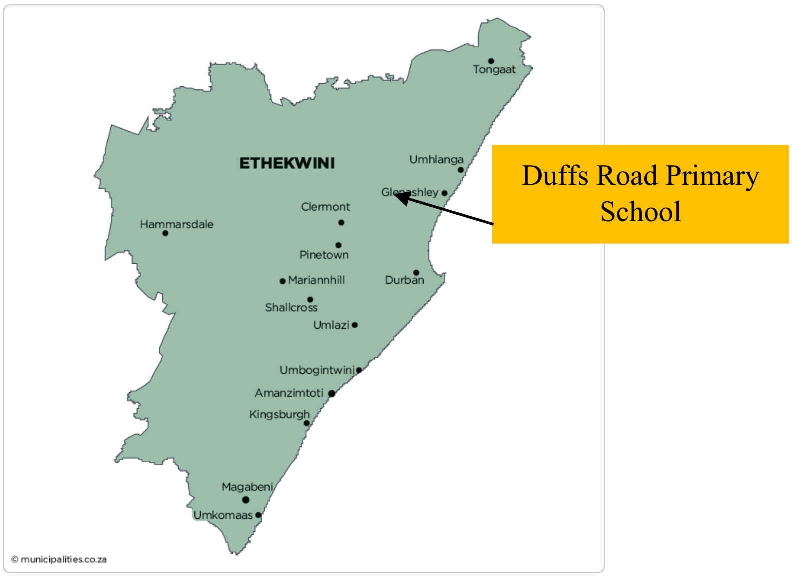

Figure 3-2: eThekwini Municipality

3.3. Case Study Site Description

Duffs Road Primary is a 44-year old schooling institution that educates learners from Grade R to Grade 7. The site location characteristics and features are tabulated in Section 3.4, while the physical layout of the case study is shown in **Section 3.5.1**. Located in the Duffs Road KwaMashu region, the primary school is approximately 11 854.25 m^2 (or 0.01 km^2) in area (Google Earth Pro, 2021). The school consists of several buildings that are further described in **Section 3.5.2**.

It is also a public, non-fee-paying school and primarily utilizes municipal water supply to cater to the water consumption demands. The municipal water usage points have been highlighted in **Section 3.5.3**. The water consumption at Duffs Road Primary is discussed in **Section 3.6** by analyzing the demographics and water activities in the school setting. Initially, the school's layout was much smaller, consisting of two classroom blocks and one administration block. However, in April 2019, newly constructed blocks provided more resource facilities. The recent renovation also resulted in the increased water and energy consumption shown in the findings chapter. In terms of RWH potential, the presence of several blocks would provide additional catchment areas, thereby promoting higher rainwater usage.

3.4. Site Location

Table 3-2 summarizes the site location characteristics.

Table 3-2: Site Location Features

Site Location	
Street address	2 Robin Road, Duffs Road
City	Durban
Province	KwaZulu Natal
Country	South Africa
Latitude	29° 44’ 39” South
Longitude	31° 00’ 24” East
Average elevation above mean sea level	58.11 m

3.5. Site Layout

3.5.1. Physical Layout

The physical layout of Duffs Road Primary (in reference to the north point) is shown in **Figure 3-3**. Also shown are the site boundary (in red) and the approximate location of the water meter.

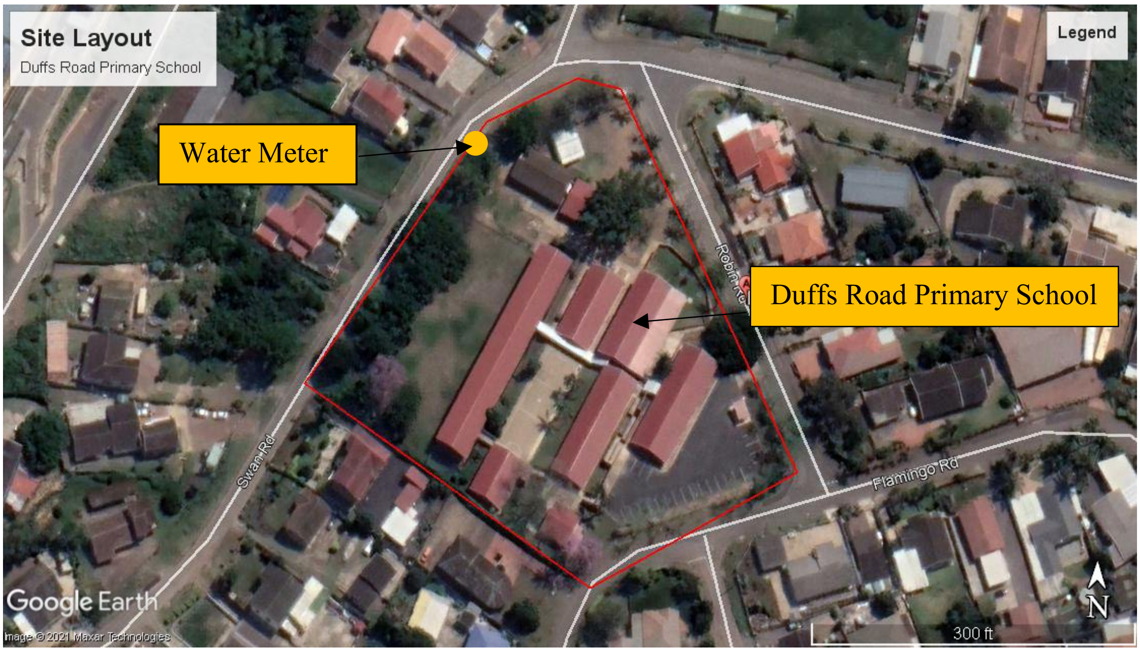

Figure 3-3: Duffs Road Primary School Layout *(Google Earth Pro, 2021)*

3.5.2. Block and Roof Layout

Figures 3-4 and **3-5** display the block and roof layout, respectively.

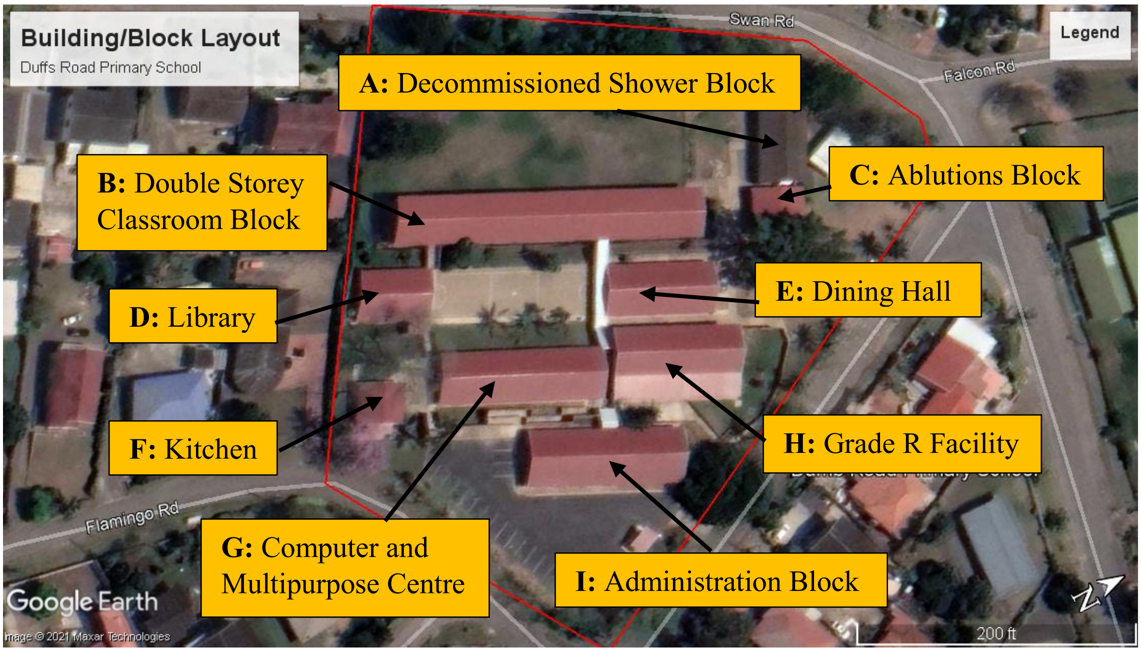

Figure 3-4: Block layout for Duffs Road Primary *(Google Earth Pro, 2021)*

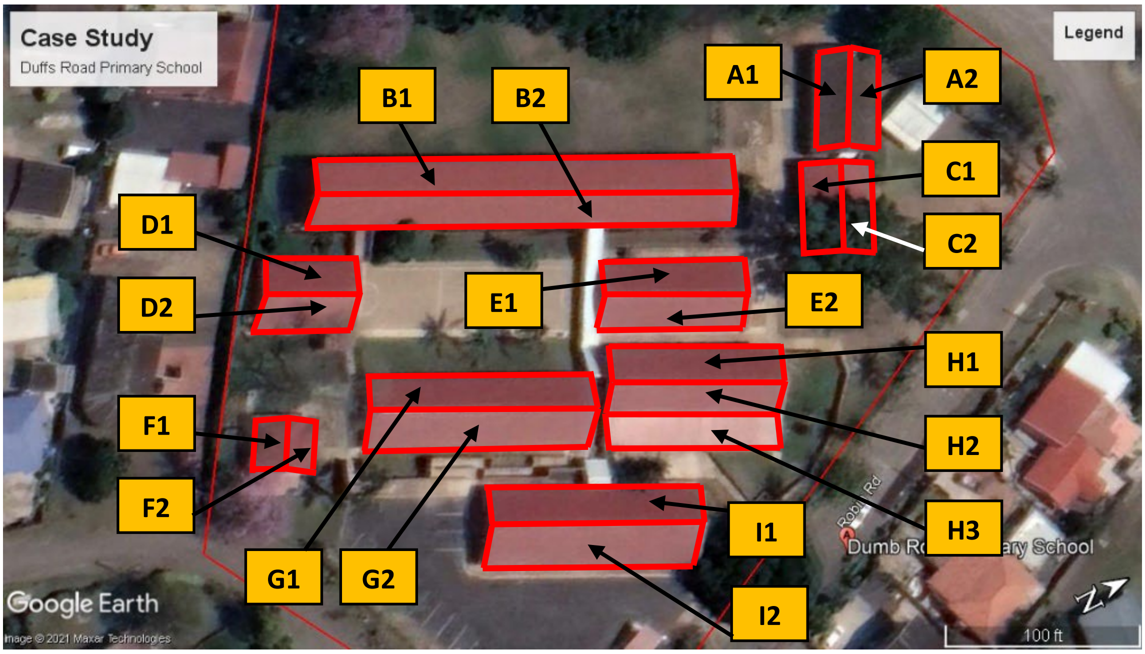

Figure 3-5: Roof Segments for the Case Study *(Google Earth Pro, 2021)*

The school blocks are located on different levels of land, which provides a unique landscape for the case study. Blocks on higher elevations would help set up gravity-fed RWHS that could potentially provide harvested rainwater to other buildings at lower elevations. It was also noted that all the blocks are single storey except for Block B.

3.5.3. Municipal Water Usage Points

The municipal water usage points were determined according to the block layout described in **Section 3.5.2**. These points have been tabulated as follows.

Table 3-3: Municipal Water Usage Points

Location	Block	Water Infrastructure
Decommissioned Shower Block	A	None (removed)
Double Storey Classroom Block	B	Taps and faucets x2 Toilet x1
Ablutions Block	C	Learner taps and faucets x4 Learner urinal x1 Male learner toilets x4 Female learner toilets x6 Teacher toilet (unisex) x1 Teacher toilet tap x1
Library	D	Tap and faucet x1
Dining Hall	E	None
Kitchen	F	Tap and faucet x1
Computer and Multipurpose Centre	G	None
Grade R Facility	H	Taps and faucets x3 Learner toilets (unisex) x3 Teacher toilet (unisex) x1
Administration Block	I	Female toilet taps x2 Female toilets x3 Male toilet taps x2 Male toilet x1 Kitchen tap x1

Note: Water infrastructure is limited to taps, faucets, and toilets.

3.6. Water Consumption

3.6.1. Demographics

In order to understand the municipal water usage and consumption for the case study, it became necessary to look at the demographics and water activities of the school, i.e., the people who use the water and the purpose for which water is used. According to the principal of Duffs Road Primary, Mr G. Maharaj, both the learner body and the staff component have remained relatively constant for the past four years (2018 – 2021). However, it was anticipated that the learner body contributed to the most significant water consumption based on the population sizes shown in **Tables 3-4** and **3-5**.

Table 3-4: Learner Demographics (2021)

Learners	Male	Female	Sub Total
Grade R	39	41	80
Grade 1	50	51	101
Grade 2	41	51	92
Grade 3	46	42	88
Grade 4	44	48	92
Grade 5	42	38	80
Grade 6	41	50	91
Grade 7	40	49	89
Total	*343*	*370*	**713**

Table 3-5: Staff Demographics (2018 – 2021)

Staff	Male	Female	Sub Total
Educators	4	17	21
Office administrators	0	1	1
Cleaning	1	1	2
Security	1	0	1
Food handling	0	2	2
Total	*6*	*21*	**27**

The total population for the case study is 740 people, with learners constituting 96.4% of the demographics. The specific water activities are discussed next.

3.6.2. Water Activities

It has been mentioned that certain water activities like toilet flushing, hand washing, showering, and irrigation result in significant water consumption in schools, in general. This fact is highlighted in **Figure 3-6**, which shows the (general) representative water usage per activity in a high school of 1200 learners (Price, 2009).

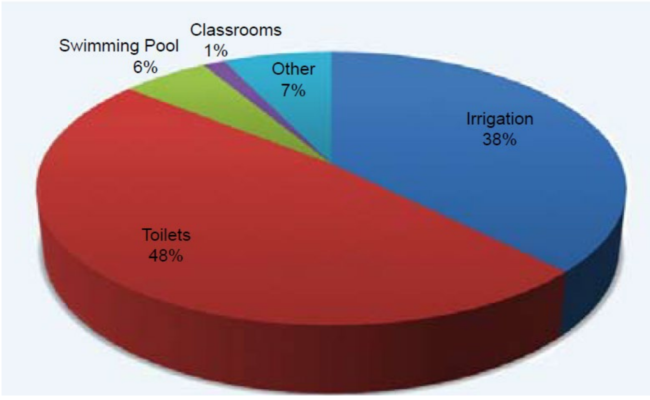

Figure 3-6: Typical Water Use in a High School with 1200 learners
(Price, 2009)

However, for the chosen case study, the main water activities specifically include toilet flushing, hand washing, food preparation, dishwashing, and general cleaning. Learners that attend Duffs Road Primary are provided with daily lunch; hence, municipal water is consumed during the food preparation and dishwashing. As Duffs Road Primary has a decommissioned shower block, no water usage occurs through this activity. Furthermore, the school does not have a swimming pool or sprinklers; hence, recreational water use and regular irrigation do not happen. A summary of the water activities and specific water users per activity is shown in **Table 3-6**.

Table 3-6: Water Consumption at Duffs Road Primary

Water Activity	Water User
Toilet flushing	The entire staff and learner bodies
Hand washing	The entire staff and learner bodies
Food preparation	Food handlers
Dishwashing	Food handlers
General cleaning and maintenance	Cleaners

3.7. Rainwater Harvesting Considerations

3.7.1. Catchment Areas

The catchment areas were obtained using the 'polygon ruler function' on Google Earth Pro (2021). The catchments were then divided into roof segments per block and labeled accordingly. The roof segments were numbered from top to bottom and left to right in reference to the building orientation shown in **Figure 3-5**. It was noted that all the blocks had symmetrical catchment areas due to the gable roof design. **Table 3-7** summarizes the catchment areas for Duffs Road Primary School.

Table 3-7: Roof Areas

Location	Block	Roof Segment	Area (m^2)
Decommissioned Shower Block	A	A1	94.91
		A2	94.91
Double Storey Classroom Block	B	B1	344.90
		B2	344.90
Ablutions Block	C	C1	105.64
		C2	105.64
Library	D	D1	85.10
		D2	85.10
Dining Hall	E	E1	116.89
		E2	116.89
Kitchen	F	F1	37.26
		F2	37.26
Computer and Multipurpose Centre	G	G1	190.39
		G2	190.39
Grade R Facility	H	H1	140.52
		H2	140.52
		H3	139.09
Administration Block	I	I1	198.62
		I2	198.62

It was further noted that all the roofs could act as catchment areas, with specific design considerations shown in Chapter 4. In addition, the Grade R facility has a shelter that could also serve as a catchment area, thereby increasing the rainwater supply.

3.7.2. Existing Gutters and Downpipes

The existing gutter and downpipe dimensions were measured and recorded manually. It was found that the gutters and downpipes for all the blocks displayed similar characteristics in terms of their non-standardized sizes, cross-sectional areas, and materials. These characteristics are summarized in **Table 3-8**.

Table 3-8: Existing Gutter and Downpipe Features

Block	Gutter dimensions (width x depth)	Downpipe diameter (mm)	Material
A	250mm x 200mm	210mm	Polyvinyl chloride (PVC)
B	250mm x 200mm	210mm	PVC
C	250mm x 200mm	210mm	PVC
D	250mm x 200mm	210mm	PVC
E	250mm x 200mm	210mm	PVC
F	250mm x 200mm	210mm	PVC
G	250mm x 200mm	210mm	PVC
H	250mm x 200mm	210mm	PVC
I	250mm x 200mm	210mm	PVC

The existing gutters and downpipes required checks to verify their sufficiency in rainwater conveyance. These confirmations are described in **Section 4.6.1** and shown in **Section 5.4.1** of Chapters 4 and 5, respectively.

3.7.3. Existing Tanks

Duffs Road Primary has several existing tanks present on site. The purpose of the existing tanks is to provide non-potable water during municipal water shortages. However, none of the tanks present on the premises are connected to pumps or used systematically. Manual transfer of rainwater by the cleaning staff for mopping occurs, and the tanks are a recent addition to the premises. The placement and sizing of these tanks would need to be checked to verify their efficiency. Thus, it was noted that the existing tanks could potentially be incorporated into the rainwater harvesting design should their sizing and positioning be sufficient. These aspects are discussed in the methodology. The approximate locations of the existing tanks and toilet facilities are shown in **Figure 3-7**. **Table 3-9** summarizes the current tank sizes and uses in relation to the block layout.

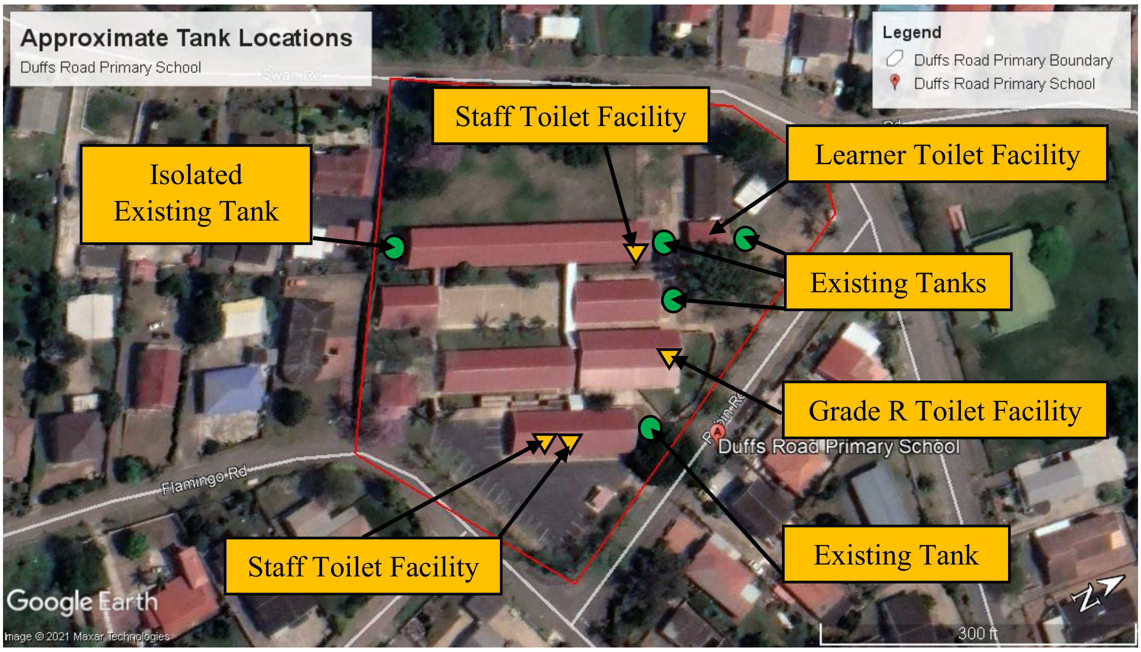

Figure 3-7: Existing Tank Locations *(Google Earth Pro, 2021)*

Table 3-9: Existing Tank Characteristics

Block	Tank Capacity (kL)	Water Use
B	5	Cleaning
B	5	Cleaning
C	2.5	Cleaning
E	2.5	Cleaning
I	2.5	Cleaning

3.7.4. Pre-filtration Mechanisms

Due to the presence of several trees on site, pre-filtration mechanisms are recommended. Other contaminants like sands and silts are also likely to be present. Thus, the pre-filtration tools would aim to remove larger, unwanted pollutants from the harvested rainwater before it is stored. However, the proposed pre-filtration components would not be sufficient against microbial contaminants. Therefore, harvested rainwater for potable uses would require disinfection, which is discussed further in **Section 3.7.7**. The pre-filtration mechanism characteristics are tabulated according to their placement within the RWHS and their specific function (**Table 3-10**).

Table 3-10: Pre-filtration components

Pre-filtration mechanism	Component placement	Function
First-flush diverter	Along the downpipe	Removal of initial rain-washed water from the catchment surface
Leaf eater	Along the downpipe	Removal of leaves and larger debris washed from the gutters
Tank screen	Below the tank lid	Removal of smaller contaminants like silts and sands by sifting

3.7.5. Tank Sizing Considerations

Storage tanks are usually manufactured in standardized sizes (Mathe, 2019) and aim to collect most or all the rainwater from the catchment surfaces. Tanks may be oriented vertically or horizontally depending on spatial factors. Selected vertical tank sizes applicable to the case study rainwater supply are presented in **Table 3-11**.

Table 3-11: Selected Tank Sizes (Eco Tanks, 2021; Jojo, 2021)

Tank Product Brand	Tank Capacity (kL)	Diameter (mm)	Height (mm)	Minimum Platform Width (mm)
Eco Tanks	2.5	1410	2050	1510
Jojo	2.4	1420	1700	1520
Eco Tanks	5	1800	2200	1900
Jojo	5.5	1900	2240	2000
Eco Tanks	10	2300	2800	2400
Jojo	10	2200	3150	2300
Eco Tanks	15	2480	3280	2580
Jojo	15	2600	3260	2700
Eco Tanks	20	2600	4450	2700
Jojo	20	2600	4270	2700

Since the proposed system would be implemented in a school setup, the tanks should be non-obstructive yet large enough to collect the potential rainwater supply on-site. It is also recommended that tanks be placed on level surfaces, properly designed tank stands, or concrete plinths of at least 85mm in depth (Jojo, 2021). Furthermore, platform widths should be, at minimum, 100mm wider than the tank width (Jojo, 2021).

3.7.6. Tank Placement Considerations

The optimal placement of the proposed rainwater tanks is vital for efficient system performance. Generally, tanks should be placed near the water application point to reduce water travel, piping, and pumping distances (Maharaj, 2020; Team Poly Water Tanks, 2018). Utilizing points that allow for easy connection to existing downpipes is another consideration, and spacing considerations for the tank dimensions and platform parameters also need to be deliberated (Maharaj, 2020; Team Poly Water Tanks, 2018). Unobtrusive tank placement and aesthetics are other factors for optimal tank placement. The detailed tank placement considerations at Duffs Road Primary are shown as a part of the RWHS tank design in **Section 4.6.3**.

3.7.7. Disinfection

The most cost-effective and easily applicable disinfection option for harvested rainwater is chlorination, which should utilize 0.4-0.5 mg of free chlorine to disinfect one litre of rainwater (Helmreich and Horn, 2009). Alternatively, the Australian Department of Health (2014) recommends an amount of chlorine required for disinfection in millilitres (ml) based on specified chlorine concentrations (1, 2, or 5 mg/l) and tank volumes (kL). The chlorine amounts (in ml) are shown in **Table 3-12**.

Table 3-12: Amount of chlorine required for disinfection (Australian Department of Health, 2014)

	Stipulated chlorine concentration (mg/l)		
Tank Capacity (kL)	1	2	5
2	16 ml	32 ml	80 ml
5	40 ml	80 ml	200 ml
10	80 ml	160 ml	400 ml
16	128 ml	256 ml	640 ml
20	160 ml	320 ml	800 ml

Any of the required concentrations can be adopted based on product characteristics.

3.7.8. Pump

In cases where a gravity-fed RWHS cannot be implemented, a pump will be required to distribute the collected rainwater to the water application point. Since the water activity at Duffs Road Primary is not continuous, a constant pressure supply pump is unnecessary. Instead, a booster pump (that automatically operates when rainwater is required) would be sufficient in direct supply to the application points (no header tanks). The pump would also need to be connected to the tanks and water infrastructure through connector kits and Polycop piping (Jojo, 2021). The appearance of a standard booster pump is shown in **Figure 3-8**, and the pump characteristics are summarized in **Table 3-13**.

Figure 3-8: Booster Pump *(Jojo, 2021)*

Table 3-13: Booster Pump Characteristics (Jojo, 2021)

Motor power	0.37 kW
Pressure	3.8 bar
Flow rate	34 l/min
Impeller	Brass
Shaft	Stainless steel
Start/stop function	Automatic
Other features	Dry running and overcurrent protection

The pump was selected as it was a recommended product of Jojo (2021) and specifically tailored to function with all Jojo products.

3.8. Limitations and Assumptions

The following limitations and assumptions regarding the case study were identified.

- Catchment areas were obtained using Google Earth Pro (2021); therefore, human error and inaccuracies may be present due to the scale.
- From the discussed water activities, non-potable rainwater can only be used for cleaning (mopping/wiping) and toilet flushing.
- Block C (the ablutions block) is the only toilet facility available for learners from Grades 1 to 7 (85.5% of the demographics); hence, it is likely the location where most of the toilet flushing occurs.
- Block C is obstructed by an overhanging tree that will likely reduce rainwater collection and increase contamination by leaves. It was recommended that the tree be removed to utilize the catchment area of the ablutions block entirely.
- Due to the presence of the tree discussed above, the dimensions of Block C were checked on site.
- Downpipes are located at the corners of all the buildings, which limits the tank placement options. Furthermore, tanks cannot be positioned along the lengthier building spans due to spatial and aesthetic reasons.
- The topography of the case study displays an elevation profile that generally decreases from the administration block to the double-storey classroom block (Google Earth Pro, 2021). Hence, gravitationally fed systems would only function accordingly.
- The lower level of the double-storey classroom block rests on the lowest level of land on site. Furthermore, it has the largest catchment area implying it could supply the highest amount of rainwater. Hence, rainwater collected from this building would need to be pumped to any water application point, thereby reducing energy efficiency.

CHAPTER 4: METHODOLOGY

4.1. Introduction

This chapter outlines the systematic procedures used to determine the aims and objectives of the research. Firstly, the investigation approach was drafted, which included the preliminary methods, literature review analysis, and case study selection. Thereafter, specific scenarios applicable to the case study were described. Following the scenario account is the framework of the collected case study data. This data included the municipal water and energy consumption at Duffs Road Primary School and the historical rainfall data and proportion of water used for toilet flushing. The predicted rainwater supply and potential water, cost, and carbon emission savings from RWH were then discussed. Next, the design considerations of the proposed RWHS were deliberated, culminating in the economic and environmental analyses. An LCA was also performed as an extension of the ecological analysis. Lastly, the limitations and assumptions of the proposed methods were highlighted.

4.2. Research Method

4.2.1. Preliminary Approach

This research's methodological approach included a combination of statistical, calculative, and fieldwork data with a critical assessment of the information obtained. The methodology resonated with the aims and objectives stated in **Section 1.4** of Chapter 1 and included the following preliminary steps.

1. The project planning and proposal drafting (research development).
2. The formation of the aims and objectives of the study (see **Chapter 1**).
3. A theoretical understanding via the literature review (see **Chapter 2**).
4. The case study selection and layout (see **Chapter 3**).
5. Data collection (see **Section 4.4**).
6. Data analysis and rainwater harvesting system design (see **Sections 4.5**, **4.6**, **4.7**, **4.8**, and **4.9**).
7. Presentation of the findings (see **Chapter 5**).
8. Conclusion, recommendations, and final write-up (see **Chapter 6**).

4.2.2. Literature Review

A theoretical understanding of RWHS was provided through the literature review. In order to understand the need for such systems, the water situation in South Africa was first discussed. The major rainfall patterns that subsequently lead to the replenishment of freshwater resources provided the basis for water availability and the various threats and challenges that have emerged in the country.

Factors such as population growth, urbanization, industrialization, and accelerated climate change have had adverse effects on water resource accessibility, thereby promoting efficient water management and conservation. One such measure is rainwater harvesting, which itself has many aspects. Hence, the detailing of the types and components of rainwater systems were discussed, followed by the different design parameters and considerations. The focus then shifted to LCA, its background, software tools, and relation to RWH. Finally, the optimization of RWHS was highlighted by analyzing the performance and economic and environmental concerns.

Hence, to obtain a practical understanding of RWH system optimization, RWHS would be designed, analyzed, and implemented in a specific situation. The details of the chosen context were documented in Chapter 3 (the case study).

4.2.3. Case Study

It was decided that Duffs Road Primary School would be the subject of the research. As the school falls under the jurisdiction of eThekwini, general information pertaining to the municipality was laid out. A brief description of the school was then presented, followed by the site location and layout. The school blocks and roof segments were labelled in this section, and the municipal water usage points were documented according to the block layout. The municipal water consumption was then discussed in terms of the school demographics and water activities.

It was found that toilet flushing was the main activity for which harvested rainwater could be used. Thus, several RWHS would be designed for the school. Consequently, the potential and existing rainwater considerations were deliberated. These included the catchment areas, the existing gutters and downpipes, the existing storage tanks on site, proposed pre-filtration mechanisms, tank sizing, placement considerations, proposed disinfection, and the pump outline. All information pertaining to the case study was obtained through the school's principal, Mr G. Maharaj.

4.3. Case Study Scenarios

The adoption of case-specific scenarios would provide opportunities to optimize RWH systems. Three rainwater-harvesting setups were applicable to the case study and are described in the following subsections. It was noted that the case study scenarios were potential options in a generalized context.

4.3.1. Scenario 0: Business as Usual

The business as usual scenario refers to an unchanged water situation at Duffs Road Primary. The municipal water consumption would not incorporate RWH systems in any manner, and the municipal bills would reflect the total water and energy usage and costs at the school. Thus, the business-as-usual situation would provide a baseline comparison to other scenarios that implement RWH systems, thereby gauging the potential water, cost, and carbon emission savings. Further information regarding this scenario is shown in **Sections 4.4.1** and **4.4.2**.

4.3.2. Scenario 1: Pumped RWHS

The first RWH scenario would implement one or more RWHS that utilize identical pumps to convey water from the storage points to the water usage points. Since several blocks and roof segments are present on-site, potential RWHS were named accordingly. This means that the RWH systems would be designated according to the roof segments that provide the rainwater supply (see **Section 3.7.1**). As previously mentioned, the recommended RWHS would be used for toilet flushing. Hence, the buildings that would need to be supplied include the double-storey classroom block (Block B), the ablutions block (Block C), the Grade R Facility (Block H), and the administration block (Block I). However, any block could theoretically provide the rainwater supply. The chosen configurations (discussed in **Section 4.5.2**) would dictate the final supply. To simplify the analysis, RWHS would be used to supply water to the ablutions block (Block C), as this is the toilet facility used by most people (85.5% of the demographics).

4.3.3. Scenario 2: Gravity-fed RWHS

The second RWH scenario aimed to reduce the pump energy and carbon footprint by initiating gravity flows for the harvested rainwater, thereby optimizing Scenario 1. Similar to the concepts stated in **Section 4.3.2**, RWHS corresponded to the block and roof segments and were used for toilet flushing. The buildings that enclose toilets (and so require rainwater supply) have been highlighted in **Section 4.3.2**, with Block C again being the supply target. It was also necessary to know which buildings were on the same level (topographically) and which were on higher or lower ones to identify the systems that can initiate gravity flows. **Table 4-1** shows the school block elevations (above sea level) obtained from Google Earth Pro (2021). Also shown are specific notes concerning the blocks and their relative height/position to other blocks.

Table 4-1: Block elevations and notes

Block	Height above sea level (m)	Note
A	54	Lowest catchment on site
B	56	The second storey is at the same height as Block C
C	56	The building is elevated above the lower level of Block B
D	58	Considerable distance away from any toilet facility
E	58	Same elevation as Block D
F	60	Considerable distance away from any toilet facility
G	60	Same elevation as Blocks F and H
H	60	Same elevation as Blocks F and G
I	61	Highest catchment on site

4.4. Case Study Data Collection

In order to implement RWH systems and the proposed design scenarios, case study data was first needed. This data included the municipal water and energy usage at Duffs Road Primary and the historical precipitation depths of the area. As the research focuses on harvested rainwater used for toilet flushing, the water consumed for this activity (the water demand) was also necessary. Hence, the subsequent sections indicate the steps taken to obtain the required data. All data was formulated into an EXCEL spreadsheet.

4.4.1. Municipal Water Consumption Data

The municipal water usage at Duffs Road Primary was needed to estimate the potential water savings from the proposed RWHS. The municipal water bills (obtained through the school's principal) were used to extract the necessary monthly water consumption data. Certain accounts provided three months' worth of records; hence, to obtain the water consumption for the required month, subtraction of the other months' data was performed. The water consumption figures were obtained for the years 2018, 2019, and 2020. It was noted that the water tariffs for the school increased during this time, with the latest rates shown in **Table 2-1**. In addition to the standard tariffs (in R/kL), a fixed daily water rate (in R/day) was also applied to the school water usage. A summary of the eThekwini water rates is shown in **Table 4-2**. Selected municipal bills are presented in **Appendix A**.

Table 4-2: Periodic Water Tariffs (eThekwini Municipality)

Period	Standard Water Rate (R/kL)	Fixed Charge Rate (R/day)
January 2018 – June 2018	25.21	38.91
July 2018 – June 2019	29.12	44.94
July 2019 – June 2020	33.35	51.46
July 2020 – December 2020	36.52	56.35

From the municipal bills, it was noted that the annual water consumption increased over the period of 2018 to 2020. Considering the fact that the COVID-19 pandemic affected school attendance in 2020, the facility was not at full capacity. However, water consumption for 2020 was higher than in 2019 and 2018 despite fewer people being on the premises. However, the school principal acknowledged the possibility of water leaks during 2020 due to recent renovation damages. Thus, it was assumed that municipal bills accurately represented the school water consumption, particularly for the years 2018 and 2019. The average water consumption was then calculated from 2018 through 2020. The municipal water consumption and costs are shown in **Section 5.2.1.**

4.4.2. Municipal Energy Consumption Data

The municipal energy consumption values were obtained in the same way as the water consumption data discussed in **Section 4.4.1**, i.e., via the municipal bills. Energy rates included the standard tariff (R/kWh) and a monthly service charge (R/month). Energy consumption data were included in the analysis to gauge the amount of electricity used on the premises under Scenario 0 (business as usual). Thereafter, the application of Scenario 1 (pumped RWH system implementation) would present the energy effects of the proposed RWH systems. Energy-related aspects are further deliberated in **Sections 4.7** and **4.8**. **Table 4-3** shows the eThekwini electricity rates for the analysis period (2018 – 2020).

Table 4-3: Periodic Energy Tariffs (eThekwini Municipality)

Period	Standard Energy Rate (R/kWh)	Service Charge (R/month)
January 2018 – June 2018	1.6024	209.68
July 2018 – June 2019	1.7120	224.02
July 2019 – June 2020	1.9358	253.30
July 2020 – December 2020	2.0562	269.05

The municipal bills indicated an increase in the yearly energy consumption from 2018 to 2019 and a decrease from 2019 to 2020. This fluctuation is likely attributed to the energy use for the new construction that occurred during 2019. The energy consumption and associative costs for the three years were then averaged and are displayed in **Section 5.2.2**.

4.4.3. Historical Rainfall Data

The case study region's historical rainfall depths (mm) were needed to estimate the likely monthly rainfall supply at Duffs Road Primary. Therefore, the eThekwini Datafeeds website (eThekwini Municipality, 2021a) was used to source the precipitation values. It was found that the ‘Newlands Reservoir No. 3’ rain gauge was the most appropriate station to be used for the historical data accumulation, given its proximity to the case study area. The particulars of the chosen rain station are presented in **Table 4-4**.

Table 4-4: Rain Station Details (eThekwini Municipality, 2021a)

Name	Newlands Reservoir No. 3
Location	Newlands/KwaMashu
Latitude	29° 46’ 18” South
Longitude	30° 58’ 56” East
Elevation above sea level	163.86m
Period investigated	2001 – 2020

Using Google Earth Pro (2021), the coordinates of the rain station (29º 46' 18" South; 30º 58' 56" East) indicate an approximate 4 km distance from the case study coordinates (29º 44' 39" South; 31º 00' 24" East), thereby justifying the rain station choice. **Table 4-4** also indicates that historical rainfall data for twenty years (2001 – 2020) was used for the analysis. It was noted that certain months had partially/completely missing rainfall depths and, thus, were not used as a part of the investigation. The annual precipitation depths for the investigated period are shown in **Section 5.2.3**. Selected daily precipitation values are displayed in **Appendix B** (all years can be found in the EXCEL spreadsheet).

4.4.4. Demand for Toilet Flushing

The water activity considered for the potential rainwater usage was established as toilet flushing. However, the municipal water bills only provided the total water consumption and not the portion of the water used for the selected activity. Thus, a survey was developed and distributed amongst the classes to ascertain the learners' toilet usage. The school teachers were asked to obtain this information from their respective classes based on set options provided in the survey. This was done for easier recording purposes. Through the survey, the number of toilet flushes per day could be determined and extrapolated to a weekly, monthly, or yearly usage based on the number of school days in any given month. The number of litres used for a toilet flush was also needed. Hence, the water meter was checked before and after a single flush (with no other water use occurring in that time) to determine this value (6 litres). Thus, the product of the number of toilet flushes, the number of school days, and the water consumed per flush gave the quotient of water used for toilet flushing. Absentee flushing behaviour was assumed.

Recalling that the ablutions block (Block C) would be the rainwater supply target for the research, this is the only toilet facility available for the 633 learners from Grades 1 to 7 (85.5% of the demographics). Hence, choosing this block as the supply focus relates to likely usage by most people on site. The other toilet facilities (see **Section 3.5.3**) may be used unpredictably by staff members; however, with Block C, toilet usage by learners from Grades 1 to 7 is set and foreseeable. Hence, the water demand in that facility may be constant as opposed to the other toilet facilities (water demand per block). While it was acknowledged that the survey might not ideally determine the proportion of water used for toilet flushing, it would estimate the water demand for toilet flushing. The survey details are shown in **Appendix C,** and the findings of the toilet flushing water demand at Duffs Road Primary are shown in **Section 5.2.4**.

4.5. Potential RWH Implementation

4.5.1. Estimated Monthly Rainfall

Section 4.4.3 highlighted the procedures used to obtain the historical rainfall data. The section also mentioned missing precipitation values (mm) that were not used in the analysis. The months with missing data are tabulated according to year as follows:

Table 4-5: Missing Rainfall Depths

Year	Months Missing Data
2001	September, October
2002	March, April
2005	December
2006	January – June
2012	October – December
2013	January – March
2015	May, June, October, November

To resolve the issue of missing historical data, weighted averages (based on similar rainfall depths per month) were applied to the historical precipitation data to estimate the likely monthly rainfall at Duffs Road Primary. It was noted that certain months like March and October had weighted averages applied over seventeen years. In contrast, other months like July and August had weighted averages calculated over the whole twenty-year period. September was averaged over nineteen years, while the remaining months had weighted averages analyzed over eighteen years. Once the weighted averages for the historical rainfall data were calculated, the final precipitation values (mm) were used to estimate the potential rainwater supply on-site. The findings of this part of the research are shown in **Section 5.3.1**.

4.5.2. Potential Rainwater Supply Volume

The potential rainwater supply at Duffs Road Primary was calculated using the estimated monthly rainfall (**Section 4.5.1**), the catchment areas on site (**Section 3.7.1**), and the runoff coefficients described in **Section 2.3.3**. The supply volume is expressed through **Equation 4-1**, as adapted from Worm (2006):

$$V = R \times A \times Cr \qquad \text{Eq. 4-1}$$

Where:

V = Potential rain supply volume (m^3)
R = Estimated monthly rainfall (mm)
A = Catchment area (m^2)
Cr = Runoff coefficient (no units)

The estimated monthly rainfall was multiplied by a conversion factor of $10^{-3,}$ and the catchment areas were all obtained using Google Earth Pro (2021). For the chosen case study, it was noted that the rainfall captured on the school block roofs would not be collected in its entirety; hence, the usage of runoff coefficients. The runoff coefficient was selected according to the roof material, with **Table 4-6** highlighting several types of roof materials and their corresponding runoff coefficients.

Table 4-6: Runoff coefficients based on roofing materials (Maharaj, 2020; Worm, 2006)

Roof Material	Runoff Coefficient
Galvanized iron sheets	> 0.9
Aluminium sheets	0.8 – 0.9
Tiles (glazed)	0.6 – 0.9
Flat cement roof	0.6 – 0.7
Organic (thatched roofing)	0.2

Since all the school blocks have tiled roofs, the runoff coefficient was taken as 0.9, suggesting that all the buildings would collect 90% of the roof rainwater. Although an obstructive tree near Block C was indicated in the case study, the research was conducted as if it were removed, allowing for the full utilization of Block C's roof area. The rainwater supply was calculated according to the design scenarios (**Section 4.3**) and selected catchments.

4.5.2.1. Supply Scenario 1

The potential rainwater supply was initially calculated for all the designated school blocks using **Equation 4-1**. However, it was deemed impractical to utilize all the catchment areas to supply rainwater to Block C (ablutions block), given the remoteness of certain blocks. Hence, for Scenario 1, it was decided that rainwater be collected and pumped from Block B (lower level of the double storey classroom block) to Block C above it. Furthermore, Blocks A (decommissioned shower block) and C could also be used for rainwater provision to the ablutions block through pumping. The findings for Supply Scenario 1 are shown in **Section 5.3.2.1**.

4.5.2.2. Supply Scenario 2

The second supply scenario was RWH implementation through gravity-fed systems. For this situation, Blocks A and B could not be used given the topographical layout. Hence, it was decided that gravity-fed systems utilizing the catchments of Blocks E (dining hall), H (Grade R facility), and I (administration block) would be practical in terms of non-obtrusiveness and proximity to the supply target block. Blocks D (library), F (kitchen), and G (computer and multipurpose room) were not considered since gravity-fed systems would be unfeasible given the location of these buildings. **Section 5.3.2.2** displays the results of this subsection.

4.5.3. Supply and Demand Comparison

The available monthly rainwater supply (according to the chosen configurations in **Section 4.5.2**) was compared to the toilet flushing demand (**Section 4.4.4**) for both case scenarios. Bar graphs were used to show excess or deficit supply and demand per month. It was noted that months having a surplus rainwater supply could be used for alternate months with a shortage. However, should this event be impossible, then municipal water supply would be needed to supplement the RWH system configurations.

It was found that for both case scenarios, the cumulative (yearly) supply was less than the estimated toilet flushing demand (obtained through the survey). The only month in which the supply exceeded the demand was December (for both scenarios). Considering all the blocks as supply sources also resulted in a deficit; hence, it was noted that municipal water would still be required to supplement the toilet flushing demand at Block C. The significant findings are shown and discussed further in **Section 5.3.3**.

4.5.4. Potential Water Savings

The potential water savings were obtained directly from the chosen rainwater supply configurations and defined as the on-site rainwater used in place of municipal water for toilet flushing. The total potential water savings were estimated according to all the school blocks and then according to each case scenario. The findings are presented in **Section 5.3.4**. The potential water savings were then used directly to estimate the cost savings and the carbon emission reduction savings

4.5.5. Potential Cost Savings

The potential cost savings were estimated according to the most recent eThekwini water tariffs, the potential water savings, and the number of school days for any given month. Hence, the cost savings were multi-stepped since the municipal water costs for the case study were not limited to consumption alone. Initially, the product of the water usage charge (R36.52/kL) and the physical savings (**Section 4.5.4**) were calculated for every month for all the buildings on site. Following this, the number of school days for each month (Department of Education, 2019) was multiplied by the fixed charge rate (R56.35/day). These values were then summed with the initial cost savings for all the blocks. Thereafter, the same procedures were used for the chosen case scenarios and configurations. A general equation for the cost savings has been provided using abbreviated terminology:

$$CS = WS \times UR + NSD \times FR \qquad \text{Eq. 4-2}$$

Where:

CS = Cost Savings (R)
WS = Water Savings (kL)
UR = Usage Rate (R/kL)
NSD = Number of School Days
FR = Fixed charge Rate (R/day)

It was noted that the fixed charge tariff was applied over the school term/sessional dates (Department of Education, 2019) and not per building. The potential cost savings for each case scenario are shown in **Sections 5.3.5.1** and **5.3.5.2**.

4.5.6. Potential Carbon Emission Savings

The processes of centralized water supply have associated carbon footprints. Thus, the potential water savings (kL) arising from the proposed RWH catchments could result in reduced carbon emissions that would have been produced under the municipal water supply. For the eThekwini Municipality (and this case study research), water is collected at Inanda Dam before being redirected to Wiggins Waterworks. Thereafter, the municipal water undergoes various distribution, collection, and treatment processes, all of which have associated impact factors (kg/kL). The impact factors are used to estimate the amount of carbon dioxide (kg CO_2) emitted per kL of supply (Friedrich et al., 2009a). This concept is highlighted in **Table 4-7**.

Table 4-7: Carbon emission impact factors under municipal water supply (Friedrich et al., 2009a)

Unit	Water Volume Produced (Mega litres – ML)	Impact factor (kg CO_2/kL)	Total Carbon Impact (kg CO_2)
Inanda Dam	52	0.0511	2657.2
Wiggins Waterworks	52	0.219	11 388
Distribution	52	0.139	7228
Subtotal	**-**	**0.4091**	**21 273.2**
Collection	24	0.15	3600
Primary Treatment	24	0.112	2688
Total	**-**	**0.6711**	**27 561.2**

The impact factors considered for centralized supply included only the Inanda Dam, Wiggins Waterworks, and Distribution Units; hence, the carbon emission impact factor amounted to 0.4091 kg CO_2/kL and not 0.6711. The carbon emission savings (kg CO_2) generated by the proposed RWH systems were calculated using **Equation 4-3**:

$$Carbon\ emission\ savings = water\ savings \times 0.4091 \qquad \text{Eq. 4-3}$$

The carbon emission savings (reductions), per case scenario, are shown in **Sections 5.3.6.1** and **5.3.6.2**, respectively.

4.6. Rainwater Harvesting System Design

4.6.1. Gutter and Downpipe Analysis

The existing gutters and downpipes at Duffs Road Primary required checks to confirm whether those RWH subsystems would adequately convey rainwater from the catchment to the storage tanks. The verifications were performed using the Roof Drainage guidelines (Chapter 11) of the Southern African Institute of Steel Construction Handbook (2013), also known as the Red Book. Initially, a rough sizing for both the gutters and downpipes was adopted, after which a more detailed design procedure was followed. The intricate design procedure included risk assessment, discharge, and critical depth analyses for the gutters and downpipes. Figures used from the Red Book are shown in **Appendix D**.

Rough Sizing Guide

A rough guide to the sizing of gutters and downpipes incorporated the use of *Figures 11.1 and 11.2* from *Section 11.2* of the Red Book (2013). The recommended cross-sectional gutter/downpipe areas (y-axes) could be estimated based on the contributory roof area in m^2 (x-axes) and a five-minute rainfall intensity for a summer rainfall region (200 mm/h). Since the gutters and downpipes at the school are the same for all the buildings, the largest catchment (Block B) was used to check the minimum sizes required. In this way, the building that could potentially supply the highest rainwater volume (strain) would be used as the worst-case scenario, thereby approving the sizing adequacy for the other smaller blocks. The catchment areas have already been discussed in **Section 3.7.1**. The cross-sectional areas were then checked against the actual sizes (measured manually).

Detailed Design Procedure – Risk Assessment

The first check of the detailed design procedure involved the risk analysis of the functional failure of the RWH subsystems due to high rainfall intensity. Hence, **Equation 4-4** was used to calculate the risk of such an event based on the design life of the system and the return period (recurrence) of the risk:

$$P = (1 - \left(1 - \frac{1}{T}\right)^N) \times 100 \qquad \text{Eq. 4-4}$$

Where:

P = Risk of subsystem functional failure (%)
T = Recurrence interval (years)
N = System design life

It was recommended that a conservative return period of 50 years be used in this analysis, while the system design life was taken as the storage tank guarantee since the storage tank is the most costly component of an RWHS. Hence, the design life of the proposed RWHS was estimated as ten years since storage tanks of all sizes have the same guarantee (Eco Tanks, 2021; Jojo, 2021).

Detailed Design Procedure – Design Discharge

The peak discharge produced from a five-minute rainfall intensity and design recurrence interval was calculated using **Equation 4-5**:

$$Q = 0.278 \times I_T \times A \times 10^{-6} \qquad \text{Eq. 4.5}$$

Where:

Q = Peak discharge/flow (m^3/s)
I_T = Five-minute rainfall intensity (mm/h)
A = Contributing roof area (m^2)

The five-minute rainfall intensity (I_T) was assumed using *Table 11.3* (Red Book, 2013) based on the closest location to the case study region (Durban). The contributory roof area was taken as the catchment of Block B (as before in the rough sizing guide) since this building would maximize the discharge (worst-case scenario). The peak discharge was then used to check the optimal gutter and downpipe sizes.

Detailed Design Procedure – Gutter Design

In order to simplify the analysis, the gutters were assumed to be rectangular sections. The width and depth of the existing channels were measured by hand, and Equation 4-6 was used to ascertain the critical depth of water flow (i.e., the optimal water depth in the gutter that will allow flow into the downpipe). This depth was then checked against the gutter depth to verify that no spillage would occur. **Equation 4-6** has been presented as follows:

$$h_c = (Q^2/(w^2 \times g))^{1/3} \qquad \text{Eq. 4-6}$$

Where:

h_c = Critical water depth (m)
Q = Peak flow/discharge (m^3/s)
w = Gutter width
g = gravitational acceleration (9.81 m/s^2)

Detailed Design Procedure – Downpipe Design

Since the gutters and downpipes are intricately connected at Duffs Road Primary, the downpipe design considerations included some of the parameters assessed for the gutter design. According to the Red Book, the maximum depth of water flow (h) in a gutter may be taken as twice the critical depth (h_c), with h and Q (peak discharge) used in **Equation 4-7**.

$$d = 0.692 \times Q^{0.5}/h^{0.25} \qquad \text{Eq. 4-7}$$

Where:

d = Recommended downpipe diameter (m)
Q = Peak flow/discharge (m^3/s)
h = Water depth required for the gutter discharge to enter the downpipe (m)

In cases where unusually long gutters are present, $h = 2.2h_c$ (Red Book, 2013). The recommended downpipe diameter was compared to the existing downpipe diameter (at Block B). Generally, no replacement of the RWH components would be necessary if the current dimensions exceeded the recommended ones. The findings of the gutter and downpipe analysis are shown in **Section 5.4.1**.

4.6.2. Pre-filtration Mechanisms

The pre-filtration mechanisms were selected and incorporated into all the proposed RWHS. These RWH system components have been discussed in terms of their placement and functionality in **Section 3.7.4**. General depictions of the pre-filtration mechanisms have been provided in **Figures 4-1**, **4-2,** and **4-3**.

Figure 4-1: First-flush diverter kit
(Eco Depot, 2021)

Figure 4-2: Leaf eater subsystem
(Eco Tanks, 2021)

Figure 4-3: Tank screen component

(Jojo, 2021)

4.6.3. Storage Tank Design

The storage tank design included considerations for the tank sizes, platforms, and placement relative to the selected RWH configurations. As before, the storage tank designs were conducted under the two case study scenarios, i.e., pumped and gravity-fed RWHS. However, the general procedure remained the same. Initially, tanks were selected according to the maximum rainwater supply (**Section 4.5.2**), which occurred in November for both scenarios. Tank sizes applicable to the case study supply have been provided in **Section 3.7.5**. The next consideration involved tank placement. **Figure 4-4** indicates the chosen locations for tank placement according to each scenario.

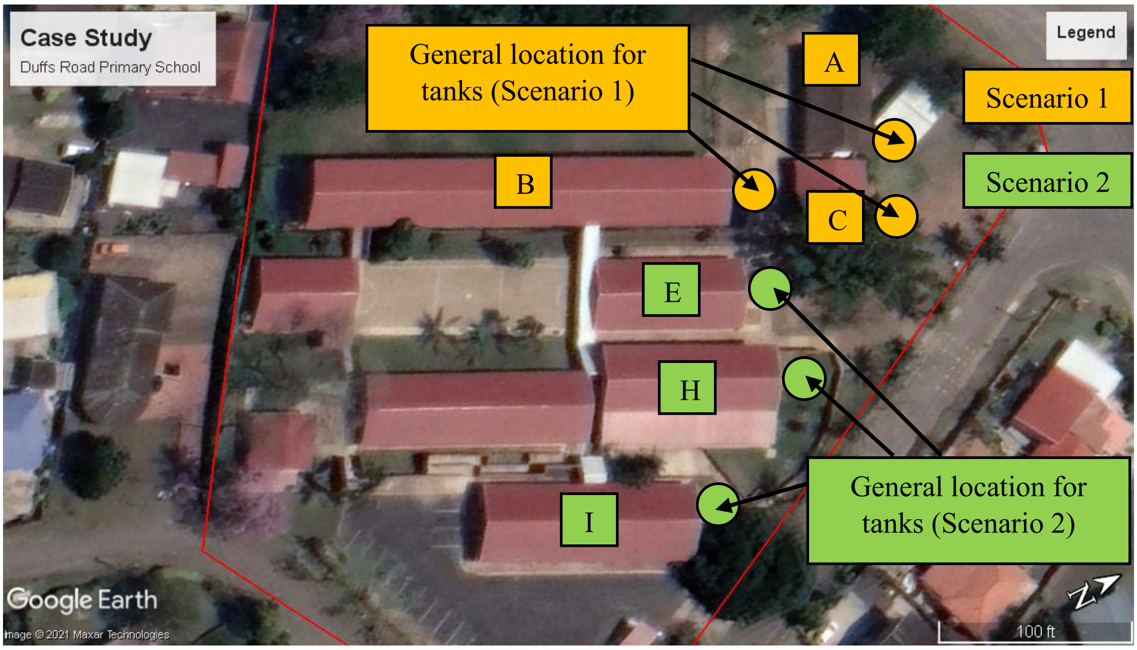

Figure 4-4: Proposed tank positions (approximate) *(Google Earth Pro, 2021)*

The existing tanks (**Section 3.7.3**) were shown to be insufficient for the rainwater supply; hence, they would be replaced by the chosen tanks. Only vertical, aboveground tanks were considered for practical and spatial reasons. The tank dimensions were also checked against the proposed placements to verify spatial concerns. Present obstructions like the trees near Blocks C and I would be removed for system optimization.

The placement of the proposed tanks was also selected based on practicality, non-obtrusiveness, and accessibility. From **Figure 4-4**, the positioning of the tanks (for Scenario 1) is shown to be in close proximity to Block C (supply target), thereby reducing pumping distances and operation. For Scenario 2, Blocks E and H are relatively close to Block C, while supply from Block I could be diverted along the site boundary.

Furthermore, the selected tank positioning (for both scenarios) would not be intrusive to the daily school activities and usage points, i.e., walkways, corridors, etc. Gutters could also be adjusted to maximize the rainwater flow into the storage units, thereby utilizing entire roof catchments. The Polycop piping needed for water transference could be aboveground or below. However, excavation, installation, and maintenance costs are not considered in this study. Placement also accounted for the downpipe locations.

The final design consideration involved the tank base/platform. Recommendations for a minimum platform thickness of 85mm, a width 100mm wider than the tank diameter, and proper leveling (using plinths or tank stands) have already been discussed in **Section 3.7.5**. Tank thicknesses could be designed according to storage capacity (weight); however, this design procedure was not used in the research. The tank design outcomes are displayed in **Section 5.4.2** according to each design scenario.

4.6.4. Amended Savings

As RWH tanks are manufactured in standard sizes, with the maximum being 20 kL (Eco Tanks, 2021; Jojo, 2021), the chosen tank sizes limited the potential water, cost, and carbon emission savings. Therefore, once the tank sizes were selected per case study scenario, the monthly water savings and its associated cost and carbon reductions were amended accordingly. The procedures used to calculate the water, charge, and carbon emission savings were the same as those described in **Sections 4.5.4, 4.5.5,** and **4.5.6**. It was noted that all the amended savings produced slight reductions to the original savings calculated, and augmenting municipal supply would be required to cater to the demand. The amended savings are shown in **Section 5.4.3.**

4.7. Economic Analysis

The benefits of the proposed RWHS (i.e., the water cost savings) were analyzed economically. The investigation included the capital costs of the systems per case scenario and the pump operation expenses for Scenario 1. Repayment periods for the proposed designs were then estimated using the ratio of the capital costs and the water cost savings per annum. Lastly, the economic findings of the two scenarios were compared in terms of their capital costs and repayment periods.

4.7.1. Capital Costs

The capital costs of the planned RWHS (under the two case scenarios) were calculated by summing the expenses of the system components. The prices of these subsystems were sourced using online catalogues, while the installation and maintenance costs (considered negligible) were excluded from the analysis. The capital costs for the pumped and gravity-fed RWH systems are shown in **Sections 5.5.1.1** and **5.5.1.2**, respectively.

4.7.2. Pump Operation Costs

The pump usage costs were investigated under Scenario 1, with operational activity only occurring during toilet flushing. Essential parameters like the number of toilet flushes, cistern refill time, and pump motor power were identified. The following procedure was used to estimate the yearly pump operation expenses:

1. The daily number of toilet flushes was estimated using the toilet flushing demand survey (**Section 4.4.4)** and scaled up to an annual figure based on the number of school days in a month (2019 sessional dates). This value was then proportioned to the number of flushes that would use up the available supply per annum.
2. The yearly pump operation/usage was equated to the proportioned number of toilet flushes (i.e., the number of flushes consuming the rainwater supply).
3. A cellphone stopwatch was used to check the cistern refill time for a toilet in Block C. All the toilets were identical; hence, their cistern refill times were assumed to be constant. The refill time was recorded in seconds (s) and converted to hours (h).
4. The yearly pump operation time (h) was taken as the product of the proportioned number of toilet flushes and the cistern refill time (h).
5. The yearly pump energy consumption (kWh) was calculated as the product of the pump's motor power (kW) and annual pump operation time (h).

6. The pump operation costs (per annum) were determined using the yearly energy consumption of the pump (kWh) multiplied by the latest energy rate (R/kWh).

The pump usage was estimated in general terms and not according to a specific pump per building. Once the pump operation costs were calculated per annum, they were compared to the existing energy usage at Duffs Road Primary to gauge whether the pump energy input/contribution was significant. After that, a comparison of the pump costs and system savings was performed. The pump operation expenses are presented in **Section 5.5.2**.

4.7.3. Payback Period

The return periods (in years) for both RWH designs were calculated using the ratio of the related capital costs (R) and system cost savings (R/year). Hence, for Scenario 1, the capital costs for a pumped RWHS were divided by the amended expense savings generated by that system. In contrast, Scenario 2 saw the gravity-fed capital costs divided by the gravity-fed amended cost savings. A general formula for the repayment period is shown in **Equation 4-8,** and the results are documented in **Section 5.5.3**.

$$Payback\ period = \frac{Capital\ costs\ of\ the\ RWHS}{Amended\ cost\ savings\ of\ the\ RWHS} \qquad \text{Eq. 4-8}$$

4.7.4. Economic Summary of the Pumped and Gravity-fed Systems

Using the information and findings obtained under **Sections 4.6.4, 4.7.1,** and **4.7.3**, the capital costs and repayment periods for the case-specific RWH systems were compared. This comparison was undertaken to ascertain which system design performed better and the overall economic feasibility of the proposed RWHS. It was noted that the number of subsystems and the designed cost savings were factors that influenced economic viability. The findings of this subsection are displayed and discussed further in **Section 5.5.4**.

4.8. Environmental Analysis

The economic factors of the proposed RWHS were only a part of the systems' feasibility. Environmental considerations like the system carbon footprint and the pump carbon footprint were also undertaken to ascertain system viability per case scenario. Furthermore, the period before carbon emission reduction was estimated for both case situations using the systems' carbon footprints and the carbon emission savings. Finally, an environmental comparison was performed between the pumped RWHS (Scenario 1) and the gravity-fed systems (Scenario 2).

4.8.1. System Carbon Footprint

The system carbon footprints for the RWH systems (designed under the two case scenarios) were assessed by analyzing the specific components like the storage tanks, pumps (Scenario 1), Polycop piping, and pre-filtration mechanisms. These components were tabulated according to their masses (kg), materials, and carbon emission factors (kg CO_2/kg material). The masses and materials of the subsystems were obtained using the same online catalogues used to source the prices. The carbon emission factors were obtained from research. Once the component carbon footprints were estimated, the system carbon footprint was calculated by summing the subsystem carbon footprints. The results of the system carbon footprint analyses are shown in **Section 5.6.1** under both design contexts. The formula for the component carbon footprint is illustrated in **Equation 4-9.**

$$CO_2\ footprint = component\ mass \times CO_2\ emission\ factor \qquad \text{Eq. 4-9}$$

4.8.2. Pump Carbon Footprint

The presence of pumps under Scenario 1 required electrical input for the harvested rainwater distribution. Consequently, carbon emissions were generated in the process; thus, to quantify the pump carbon emission production, the pump operation output per annum was analyzed. The pump operation characteristics have already been discussed in **Section 4.7.2**. The following procedure was used to estimate the RWH carbon footprint from pump operation. (See **Section 5.6.2** for the findings).

1. From the approach provided in **Section 4.7.2**, steps 1 – 5 were re-examined. This step included the estimations of the proportioned number of toilet flushes per year, the yearly pump operation, cistern refill time, the annual pump operation time, and the pump's energy consumption per year.
2. The annual carbon emissions from pump operation (kg CO_2) were taken as the product of the yearly pump energy consumption (kWh) and the electricity carbon emission factor for every kWh of energy used (kg CO_2/kWh).

As before, the pump carbon emissions were calculated according to toilet flushing usage and not a particular pump within an RWHS. This calculation was done since the toilet flushing usage would occur as each tank in the RWH system had its rainwater depleted. The annual carbon emissions produced by the pump were then compared to the yearly carbon emission savings under Scenario 1 to ascertain if the system was energy-efficient.

4.8.3. Period before Carbon Emission Reduction

Much like the period of return on the RWH investment, the period before carbon emission reduction indicated the time taken before the RWH system would begin to lower its initial carbon footprint (through its savings). These figures were calculated (in years) for both case scenarios. A general relationship for this parameter is given in **Equation 4-10** as the system carbon footprint (kg CO2) ratio to the amended carbon emission savings (kg CO_2/year). The results of this part of the methodology are shown in **Section 5.6.3**.

$$Carbon\ emission\ reduction\ period = \frac{System\ carbon\ footprint}{Amended\ carbon\ emission\ savings} \quad \text{Eq. 4-10}$$

4.8.4. Environmental Summary of the Pumped and Gravity-fed Systems

Similar to the economic comparison of the pumped and gravity-fed systems described in **Section 4.7.4**, an environmental comparison was also performed for the two scenarios. The findings from **Sections 4.6.4, 4.8.1,** and **4.8.3** were used to determine which system was more ecologically viable. Specific components like the pumps and tanks were shown to be significant factors against environmental feasibility due to their masses and material build. The results of the comparative assessment are shown in **Section 5.6.4**.

4.9. System Life Cycle Assessment

4.9.1. SimaPro Software

An LCA investigation of the RWH systems proposed under the two research scenarios (pumped and gravity-fed) was performed using the SimaPro software application. This software is a Netherlands-based computer package developed by PRé Consultants for LCA methodology analyses (Silva et al., 2017; SimaPro, 2021). It is regarded as a pioneer for LCA software practices and includes a user interface that models unit or system processes from matrices (Silva et al., 2017). Furthermore, it can be used to compare products or systems like the components of scenario-based RWHS. Hence, it was chosen as the software tool for this research.

The general layout/user interface of the SimaPro software is segmented according to the four major phases of an LCA, namely the goal and scope definition, inventory analysis, impact assessment, and interpretation of the findings. These phases are further subdivided into categories like processes and product stages (among others), which all have their purposes described in a project. In setting up a project, data applicable to the investigation is collected/processed here. The convenience of program Wizards is also present to help aid and facilitate the setup of LCA models. Moreover, SimaPro includes library databases like Ecoinvent 3 (among others), containing life cycle inventory (LCI) information for processes like energy production, material manufacturing, chemical generation, transport considerations, metal production, and waste treatment scenarios. These generalized processes were used to construct the system boundaries in this investigation.

SimaPro Faculty Version 9.2.0.2 was used in conjunction with the ISO LCA standards and guidelines for this research. The prominent standards for an LCA are the ISO 14040 and the ISO 14044 standards, which indicate (in turn) the Principles and Frameworks and the Requirements and Guidelines in the LCA setting (PRé Consultants, 2016a). The Ecoinvent library database was also selected for the investigation based on its expansive records, incorporation in the Wizard, and popularity of use in other studies (Ghimire et al., 2014; Ghimire et al., 2017; Goga, 2016; Wissing, 2020; Yan et al., 2018).

4.9.2. Goal and Scope Definition

The first phase of the LCA involved the goal and scope definition that described the purpose or intended application of the study and the research boundaries or margins (Goga, 2020). The goal definition also included the intended audience for this study, while the scope specified the functional unit and any system considerations as suggested by the ISO standards (PRé Consultants, 2016a).

4.9.2.1. Goal Definition

The definition of the study's goal aligned with the aims and objectives stipulated in Chapter 1, specifically **Objectives 4** and **5** (see **Section 1.4**). The purpose of this LCA was to assess the environmental impacts of the RWH system components during the assembly phase (cradle-to-gate) for both case study scenarios. Furthermore, the ecological effects from the operation phase of Scenario 1 would also be assessed and then compared to its assembly phase. The intended audience for this research included professionals in the water and environmental engineering fields, researchers investigating RWH and LCA, and governmental authorities looking to promote RWH. Ideally, the findings presented in this study could potentially be used as a guide for RWH adoption and its environmental feasibility in the LCA setting. Thus, by assessing the ecological impacts of RWH systems through LCAs, future implementation would be multi-faceted, accounting for considerations beyond water, cost, and carbon emission savings.

4.9.2.2. Scope Definition

The scope definition allowed for an initial layout of the methodological choices/options, assumptions, and restrictions/limitations. It was noted that these initial stipulations could be changed depending on information availability (PRé Consultants, 2016a). The functional unit, reference flows, initial system boundaries, and modelling type were laid out in this section. A general representation of the RWH system and the components considered for the LCAs are shown in **Figure 4-5**.

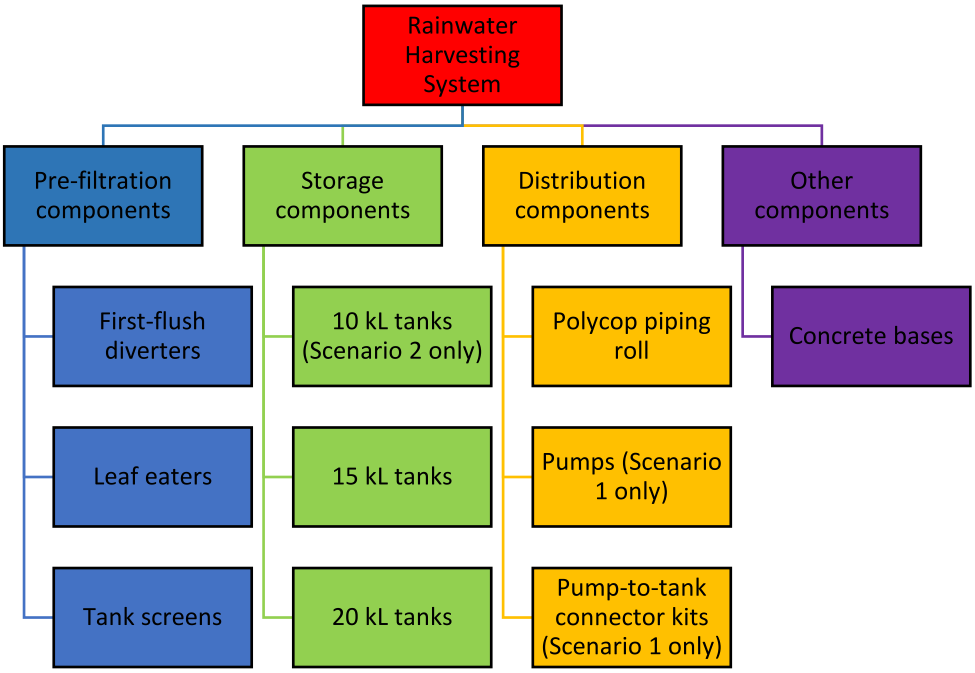

Figure 4-5: RWH components considered in the LCA analyses (both scenarios)

It was noted that the RWH systems under the two case study scenarios would function identically, i.e., to collect, store and use volumes of rainwater. The choice of the functional unit was selected by looking at what system/product was being analyzed. Hence, the functional unit was taken as a single RWH system collecting one cubic metre (1 kL) of rainwater (of an unspecified but constant quality for the two scenarios).

Each RWH system included several subsystems, each having its own unit numbers, masses, and (dominant) material types. Specific unit numbers, materials, and masses were tabulated in **Sections 4.7** and **4.8** and are revisited in **Section 4.9.3**. It was noted that Scenario 1 and 2 had minor differences in terms of their subsystem compositions and system performances (savings); hence, separate LCAs would account for these changes.

Figure 4-6, shown next, illustrates the component LCA phases, component reference flows, and system boundary (black box) considered for this investigation.

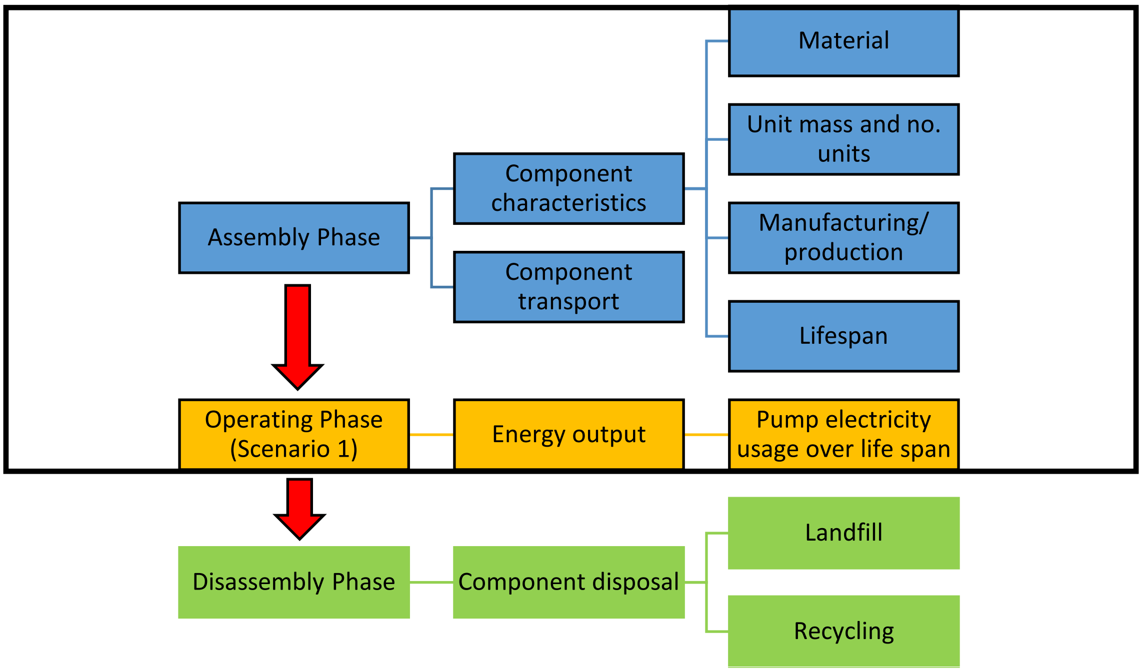

Figure 4-6: LCA phases, reference flows, and system boundary for both case study scenarios

Figure 4-6 shows that the assembly and operation of the components of the RWH systems were the main phases investigated in SimaPro. The reference flows referred to the RWH elements during the LCA phases. Further information on the quantification of the reference flows (displayed in **Figure 4-6**) is shown in **Section 4.9.3**. It was also noted that several considerations, limitations, and assumptions were present for the LCA investigation and are listed as follows:

- The RWH subsystems needed to complete the RWH systems were considered for the LCAs and the not existing features like gutters, downpipes, and roofs since those were already present at the school before the proposed system implementation.
- It was noted that the components would collectively allow for slightly varied water savings for the two RWH systems (pumped and gravity-fed).
- The LCAs were performed in relation to the physical benefits introduced by the RWH systems. This meant that the environmental impacts from the production of the system components' were estimated in conjunction with the water-saving benefits generated over the component lifespans.

- Thus, the component unit masses were scaled down according to the functional unit of 1 kL of rainwater for both scenarios investigated. The same was done for the electricity usage and transport impacts.
- Since the Ecoinvent 3 library database was used for the investigation, it became necessary to identify the type of modelling that would be adopted. It was decided that attributional (allocational) modelling would be used as it assesses the environmental impact of a product/function/system in its life cycle (PRé Consultants, 2016a).
- Furthermore, attributional modelling is appropriate for comparing two products having the same functional unit (PRé Consultants, 2016a). Recalling that the investigation ponders the component environmental impacts for every cubic metre of municipal water saved through rainwater substitution, attributional modelling was found to be most suitable.
- The specific type of attributional modelling adopted was the 'cut-off by classification' as this model accounts for mass allocation.
- The Faculty version 9.2.0.2 of SimaPro was used in this research, which did not have features supporting parameterization and Monte Carlo/uncertainty analyses. Hence, system processes were always selected over unit processes of the same name/type since unit processes contained uncertainty information (PRé Consultants, 2016a).
- Weighting was also excluded from the investigation due to subjectivity and much contention in literature (Goga, 2016; PRé Consultants, 2016a).
- The component material characteristics like material, unit mass, and the number of units were already obtained from **Sections 4.7** and **4.8**. They were assumed to represent the dominant subsystem constituents, e.g., storage tanks were considered to be made mainly of LLDPE.
- It was assumed that the production of all component materials incorporated all the energy and transport processes in the database models.
- Due to the lack of South African-specific data, the component materials were often modelled in a global/rest-of-world context. But transport vehicles and electricity data were taken from South African (ZA) Ecoinvent databases.
- The transport distances were taken as the routes travelled by standard trucks between warehouses (point of manufacturing) and the case study area (place of

use). These estimated distances were needed to evaluate the environmental loads generated from transporting the RWH components (in tonnes kilometres or tkm).

- The operating phase for Scenario 1 included the electricity used by the pumps and was compared to the assembly phase.
- As the water savings, assemblies, and operation of the RWH systems differed for the two scenarios, a direct comparison of the two systems was not performed. However, it could be inferred that Scenario 2 would have fewer environmental impacts due to the lack of pumps/electricity consumption.
- The life spans of the components were assumed based on product guarantees/warrantees and average values found in research.
- The saving benefit for both systems/scenarios was only calculated for one year (see Section 4.6.4). Hence, it was assumed that these annual water savings would be constant over the life spans' of the RWH subsystems. However, in reality, this is not the case due to climate change and erratic rainfall patterns.
- The disposal scenarios for the components were outside the scope of research as stipulated in the aims and objectives (cradle-to-gate). The reason for excluding the decommissioning/disassembly phase was that once RWH systems are decommissioned, they would no longer function towards the benefit of water-saving; hence, they could not be considered in that context.

4.9.3. Inventory Analysis

The second phase of the LCA involved relevant data collection pertaining to the components of the two RWH systems. SimaPro was particularly useful in this regard as the software contained life cycle inventories for a wide variety of materials and processes. Generally, the processes for component material production were used for the LCAs; however, some information like component materials, unit masses, and unit numbers were also needed. Hence, it became vital to identify and distinguish between the information provided on SimaPro and the information required to complete the LCAs.

4.9.3.1. Foreground and Background Data

The data needed for an LCA can be recognized as either foreground or background data (PRé Consultants, 2016a). Foreground data refers to the data that describes a specific product or system, while background data concerns the production of generic materials. (PRé Consultants, 2016a). Much of the foreground information was already obtained during the environmental analysis (Section 4.8). Subsystem characteristics like materials and masses were obtained from online catalogues; however, the impacts from transporting the components over specific distances (i.e., from the place of manufacturing to the site of use) were still needed. For this part of the collection, several assumptions were made, including transport type and delivery points. It was assumed that a nonspecific transport truck would transfer RWH components from the manufacturing warehouses to the school. Hence, the Google Maps Application was used to estimate these distances. A summary of the relevant data needed for the LCAs is shown in **Table 4-8.**

Table 4-8: Relevant LCA data for both case scenarios

Required Information for the LCAs	Data Type	Reference
Component material	Foreground	Section 4.8
Component material production	Background	SimaPro software, Ecoinvent databases
Component unit mass (kg)	Foreground	Section 4.7
Component unit number	Foreground	Section 4.7
Component lifespan (years)	Foreground	Online catalogues and research
RWH system water savings per annum (kL/year)	Foreground	Section 4.6.4
RWH system water savings over component lifespan (kL)	Foreground	Multiplication of the two preceding rows
Component transport distance (km)	Foreground	Google Maps
Transport process for the components	Background	SimaPro software, Ecoinvent databases
Component transport impact (tkm)	Foreground	Multiplication of transport distances and component unit masses
Electricity usage per annum (Scenario 1) (kWh/year)	Foreground	Sections 4.7.2 and 4.8.2
Electricity process (Scenario 1)	Background	SimaPro software, Ecoinvent databases
Electricity usage over component lifespan (kWh) (Scenario 1)	Foreground	Multiplication of electricity usage per annum and pump lifespan

Hence the LCA would show the environmental impacts per component (over their respective lifespans) during the assembly phase (both scenarios) and operational phase (Scenario 1 only). It was further noted that all the component masses (kg), transport impacts (tkm), and electricity usage (kWh) were scaled down to the functional unit (1 kL of harvested rainwater). Therefore, the input data into SimaPro was expressed as kg/kL for all the component unit masses, tkm/kL for the transport impacts, and kWh/kL for the electricity usage by the pumps. The component lifespans and water savings generated in that time are shown in **Table 4-9**. The inventory data for Scenarios 1 and 2 are shown in **Table 4-10** and **Table 4-11**, respectively.

Table 4-9: Component lifespans and lifetime water savings

Component	Assumed lifespan (years)	Scenario 1 lifetime water savings (kL)	Scenario 2 lifetime water savings (kL)
Storage tanks	20	9535.11	9049.61
Concrete bases	50	23 837.77	22 624.03
Other	10	4767.55	4524.81

Table 4-10: Inventory input data for Scenario 1

Subsystem	Assembly Phase		Operation Phase
	Material data (kg/kL)	Transport data (tkm/kL)	Electricity data (kWh/kL)
First-flush diverter	1.05×10^{-4}	6.64×10^{-6}	0
Leaf eater	1.47×10^{-4}	9.29×10^{-6}	0
Tank screen	2.10×10^{-5}	1.33×10^{-6}	0
Storage tank (15 kL)	2.83×10^{-2}	1.79×10^{-3}	0
Storage tank (20 kL)	3.78×10^{-2}	2.39×10^{-3}	0
*Concrete base	4.20×10^{-5}	4.87×10^{-7}	0
Polycop piping roll (22mm x 50m)	1.05×10^{-3}	4.40×10^{-6}	0
Pump body	1.42×10^{-3}	8.96×10^{-5}	0.822
Pump shaft	2.83×10^{-4}	1.79×10^{-5}	
Pump impeller	1.89×10^{-4}	1.19×10^{-5}	
Pump-to-tank connector kit	2.52×10^{-4}	1.59×10^{-5}	0

Table 4-11: Inventory input data for Scenario 2

Subsystem	Assembly phase	
	Material data (kg/kL)	**Transport data (tkm/kL)**
First-flush diverter	1.11×10^{-4}	6.99×10^{-6}
Leaf eater	1.55×10^{-4}	9.79×10^{-6}
Tank screen	2.20×10^{-5}	1.40×10^{-6}
Storage tank (10 kL)	1.99×10^{-2}	1.26×10^{-3}
Storage tank (15 kL)	2.98×10^{-2}	1.89×10^{-3}
Storage tank (20 kL)	3.98×10^{-2}	2.52×10^{-3}
*Concrete base	4.40×10^{-5}	5.13×10^{-7}
Polycop piping roll (22mm x 50m)	1.11×10^{-3}	4.64×10^{-6}
Polycop piping roll (22mm x 25m)	4.42×10^{-4}	1.86×10^{-6}

*Note: the material data for the concrete bases is in m^3/kL and not kg/kL since the library inventories for concrete were defined in volume units and not masses.

4.9.3.2. SimaPro Model Setup

The modelling setup on SimaPro was a multi-stage process that utilized the foreground data collection in **Section 4.9.3.1** as the primary input for the software. Initially, the Wizard was used to set up the two RWH systems in terms of their component subassemblies. However, it was found that the Wizard limited the number of subassemblies to five, and since there were more than five components present per scenario, the Wizard was not used further. Thus, SimaPro was used conventionally by inputting information and data under the headings segmented in the user interface. The system descriptions were first specified according to the goal and scope, and selected libraries were chosen as the inventory databases. These included the 'Ecoinvent 3 – allocation, cut-off by classification – system' and the 'Methods' libraries. Thereafter, new product stages were created for each component.

These product stages were named according to the component and scenario and quantified by material type and production (Ecoinvent database), unit mass (foreground data), and transport process (Ecoinvent library). The exception was concrete, which could only be quantified in terms of volume and not mass. Once the subassemblies were set up for all the components, they were combined under the primary assembly or RWH system. The life cycles were set up similarly to the assemblies, except for the electrical contribution of the pumps. This part included an energy process over the lifespan of the first system scenario. As Scenario 2 had no operational inputs, the life cycle referred only to the assembly phase. The decommissioning phases for the two systems were not included; hence, no disposal, disassembly, or reuse scenarios were defined. Once all the components were modelled, SimaPro would automatically calculate the impacts using the ReCiPe 2016 Midpoint Hierarchist already defined in the program.

4.9.4. Impact Assessment

The third phase of the LCA involved the environmental impacts generated from the assembly and operation of the two RWH systems. These impacts were represented as emissions and extractions that produced adverse effects/problems in the environmental setting (PRé Consultants, 2016). Obligatory steps like classification and characterization were needed for this part of the LCA to comprehend and assess the scale and importance of the impacts produced by the systems (PRé Consultants, 2016). Optional elements like normalization, ranking, grouping, and weighing were all excluded due to subjectivity.

4.9.4.1. Classification

The inventory data outlined in **Section 4.9.3** was automatically assigned to various impact categories by use of the ReCiPe 2016 Midpoint Hierarchist Method and the generated inventory table. This step (known as classification) was performed to assess a component's contribution to varying environmental problems in the assembly and life cycle phases (PRé Consultants, 2016). These problems are usually termed impact categories/indicators, all of which have their own definitions and impact units. It was noted that all available impact categories were analyzed under the ReCiPe Midpoint Method and are discussed in the following subsection.

4.9.4.2. Impact Categories

The SimaPro software displays the environmental effects of products and systems in terms of 18 impact categories for the ReCiPe Midpoint Method. The environmental impacts from the assembly of the considered components were assessed and analyzed for both RWH systems/scenarios. Furthermore, the operational output of the pumps in Scenario 1 was evaluated over their lifespan to gauge the amount of electricity used by the system and then the subsequent impacts. The assembly and operational stages for

Scenario 1 were also compared. As the active (operating) phase for Scenario 2 involved no electrical contributions (and thus impacts), the assembly phase was solely analyzed as discussed previously. It was noted that the components involved in the assemblies of both system scenarios were similar, with each system also generating comparable water savings per annum. The major impact categories (described next) include global warming, stratospheric ozone depletion, ionizing radiation, ozone formation, fine particulate matter formation, terrestrial acidification, eutrophication, ecotoxicity, human carcinogenic toxicity, human non-carcinogenic toxicity, land use, resource scarcity, and water consumption.

Global warming

Global warming (or climate change) is the temperature increase in worldwide climate systems due to human activities like industrialization and energy production. Greenhouse gases (GHGs) like CO_2 are the leading cause of these temperature rises, resulting in other implications like rising sea levels and erratic rainfall patterns (Hauschild and Huijbregts, 2015). The build-up of more GHGs in the earth's lower atmosphere prevents the sun's radiation from being reflected into space (Wissing, 2020). Hence, more radiation is absorbed into the earth's atmosphere resulting in the global warming phenomenon (Hauschild and Huijbregts, 2015). Therefore, it can be seen that global warming is a midpoint indicator for the disastrous consequences on human health and ecosystem stability. In SimaPro, the impacts from global warming are represented in kg CO_2 equivalents (eq). The CO_2 equivalents refer to the effects or (global warming potential) of other GHGs in reference to the CO_2 gas effects (Goga, 2016). The outcomes for this specific impact category also provide a check for the pump emissions of **Section 4.8.2**.

Stratospheric ozone depletion

Stratospheric ozone depletion refers to the diminishment of the ozone particles that layer the stratosphere. This layer maintains and regulates many conditions on Earth, like the absorption of harmful ultraviolet sun radiation (Goga, 2016; Wissing, 2020). However, several gases like chlorofluorocarbons (CFCs) and halocarbons destabilize these regulatory processes, subsequently leading to detrimental human and ecosystem health impacts (Hauschild and Huijbregts, 2015). The unit of measurement for stratospheric ozone depletion is kg CFC-11 equivalents, which relates the ozone depletion potential of other gases to a reference point.

Ionizing radiation

According to the World Health Organisation (2021), ionizing radiation refers to the radioactive energy released by atoms in the form of electromagnetic waves or particles. Radioactive (atom-disintegrating) particles that cause ionizing radiation are termed radionuclides and are used as a reference for ionizing radiation measurement. In SimaPro, the unit of measure is a kilo Becquerel cobalt 60 (kBq Co-60). The word Becquerel refers to a single disintegration per second, while cobalt 60 refers to the synthetic radioactive isotope of the same name. In general, radiation sources can be from natural emanations in air, water, and soil (like Radon) or manufactured devices (like X-ray machines and microwaves).

Ozone formation

Ozone formation refers to the impacts of ozone formed in the troposphere. Solar radiation is often a natural catalyst for this photochemical process, whereby ozone diffusion (or spread) occurs into the lower atmosphere (Wissing, 2020). While ozone is necessary for regulatory functions in the stratosphere, higher concentrations produced at "ground-level" can be damaging to living organisms. Thus, the environmental impacts associated with the formation of ozone particles are analyzed in two settings: human health and terrestrial ecosystems. Ground-level ozone can also be formed by air contaminants like nitrogen oxides (Hauschild and Huijbregts, 2015). Therefore, the unit of measurement for both ozone formation settings is nitrogen oxide equivalents (kg NO_x eq) that exist as atmospheric pollutants.

Fine particulate matter formation

Human activities like industrialization and energy production are usually associated with widespread airborne emissions. Many of the emissions are minuscule and reactive in nature, resulting in complex mixtures of hazardous particles (Goga, 2016). These microscopic particles, when inhaled, can pose threats to humans. Severe health effects include respiratory issues like lung cancer and chronic medical conditions like diabetes and birth abnormalities/deformities (Hauschild and Huijbregts, 2015). Therefore, fine particulate matter formation is categorized as an impact category within the SimaPro software. The environmental effects were measured in kilograms of particulate matter equal to or less than 2.5 µm (kg PM2.5 eq).

Terrestrial acidification

Terrestrial acidification refers to the deposit and leaching of inorganic substances into soils resulting in their altered chemical properties (Hauschild and Huijbregts, 2015). Substances like nitrogen and sulphur (generated in excess by fossil fuel combustion, vehicle emissions, energy generation, and industrial processes) often change the natural alkalinity (ability to neutralize acid) and pH of ground conditions. Consequently, these activities lead to disrupted and damaged ecosystems. As was the case for other impact categories, human activities tend to accelerate land acidification. However, natural events like volcanic eruptions and ocean emissions may also increase the acidity of soils (Hauschild and Huijbregts, 2015). In SimaPro, the environmental burden of terrestrial acidification was represented by kilograms of sulphur dioxide equivalents (kg SO_2 eq).

Eutrophication

Eutrophication is the process by which water bodies receive extra nutrients (due to nitrogen and phosphorous inputs), resulting in algal blooms, oxygen diminutions, and disturbed bionetworks. Similar to terrestrial acidification, eutrophication is a natural process. However, industrial, agricultural, and sewage discharges are often associated with introducing excess nitrates and phosphates into ecosystems. For SimaPro, eutrophication impacts were displayed for the two major types of water systems, namely freshwater and marine bodies. The freshwater eutrophication impact was expressed in kilograms of phosphorous equivalents (kg P eq), while the marine burden was represented in kilograms of nitrogen equivalents (kg N eq).

Ecotoxicity

Ecotoxicity can be defined as the detrimental or poisonous effects on organisms and ecosystems by physical, chemical, or biological stressors and emissions (Goga, 2016; Hauschild and Huijbregts, 2015; Wissing, 2020). Thus, the causes of ecotoxicity are wide-ranging, encompassing natural and manufactured (artificial) considerations. In SimaPro, ecotoxicity is further categorized as terrestrial (land), freshwater, and marine. Although these three impact categories refer to broad ecosystems, they are measured in terms of the same unit, i.e., kilograms 1,4 dichlorobenzene (kg 1,4-DCB). The 1,4-DCB denotes the organic compound often found in pesticides and disinfectants (Hauschild and Huijbregts, 2015).

Human toxicity

Human toxicity is a sphere of ecotoxicity pertaining solely to human beings. Again, physical, chemical, and biological stressors may indicate the damaging effects on humankind; however, many studies suggest chemical stressors as being the most prominent indicators (Hauschild and Huijbregts, 2015). The toxic effects on humans were categorized as either carcinogenic or non-carcinogenic. Human carcinogenic toxicity refers to the impact levels of toxins, pollutants, and contaminants that have the potential to cause cancers in humans (Hauschild and Huijbregts, 2015). Non-carcinogenic impacts, though still harmful to human health, were then defined as non-cancer causing. As was the case for ecotoxicity, human toxicity was measured in kg 1,4-DCB in the SimaPro software.

Land use

Another category linked to human activities is the way in which land is used. Human developments and settlements may consist of farms, towns, and cities with varied purposes and unconsidered environmental impacts. The ecological burdens amounting from land use are usually classified by land degradation and quality (Hauschild and Huijbregts, 2015). The reference unit in SimaPro is square-metre-year equivalents (m^2a crop eq).

Resource scarcity

Resources form the backbone of human/societal development as every product or process utilizes critical raw materials. However, resources like fossil fuels, metals, and water are not unlimited. The effects of over-population and development have placed immense strain on these abiotic resources, depleting them to the point of insufficiency/shortage. In SimaPro, resource scarcity is divided into mineral (underground) and fossil (transformed biomass) resource scarcity with units kg Copper (Cu) eq and kg oil eq, respectively.

Water consumption

The consumption of water (an abiotic resource) is closely linked to the preceding impact category. According to Colvin et al. (2016), every product and process has a water footprint in some capacity. However, water is finite and contaminable, and its usage in the agricultural, domestic, and industrial contexts further adds to the strain on the reserve. The consumption of water is measured in volumetric terms, i.e., m^3.

Table 4-12: Summary of the impact categories

Impact Category	General Description	Unit of measurement
Global warming	Temperature increase across the planet	kg CO_2 eq
Stratospheric ozone depletion	Stratosphere ozone layer damage by particular gases	kg CFC-11 eq
Ionizing radiation	Radioactive energy released by atoms	kBq Co-60 eq
Ozone formation – human health	The formation of ozone particles in the troposphere	kg NO_x eq
Fine particulate matter formation	Tiny, volatile particles formed from human activity	kg PM2.5 eq
Ozone formation – terrestrial ecosystems	The formation of ozone particles in the troposphere	kg NO_x eq
Terrestrial acidification	Chemical alteration of soil properties	kg SO_2 eq
Freshwater eutrophication	Excess nutrients in freshwater bodies	kg P eq
Marine eutrophication	Excess nutrients in seawater	kg N eq
Terrestrial ecotoxicity	Toxic impacts on land organisms and ecosystems	kg 1,4-DCB
Freshwater ecotoxicity	Toxic impacts on freshwater organisms and ecosystems	kg 1,4-DCB
Marine ecotoxicity	Toxic impacts on seawater organisms and ecosystems	kg 1,4-DCB
Human carcinogenic toxicity	Cancer-causing toxins	kg 1,4-DCB
Human non-carcinogenic toxicity	Non-cancer causing toxins	kg 1,4-DCB
Land use	Effects of repurposed land	m^2a crop eq
Mineral resource scarcity	Underground resource depletion	kg Cu eq
Fossil resource scarcity	Transformed biomass depletion	kg oil eq
Water consumption	Excess water usage, depletion, and contamination	m^3

4.9.4.3. Characterization

The final mandatory step of the impact assessment phase was characterization, which converted the inventory data (inputs) into environmental impacts by use of material characterization factors of the ReCiPe Method (Goga, 2016; Wissing, 2020). In other words, the findings of the component environmental effects were computed and presented according to the impact categories and their corresponding units.

Similar to the classification step, characterization was automatically computed by the SimaPro software; thus, the specifics of the ReCiPe characterization factors were not further described. The underlying limitation of characterization is that impact categories cannot be compared due to different units (PRé Consultants, 2016). Therefore, an optional step called normalization could be performed to compare different impact categories (PRé Consultants, 2016). However, as the goal of the LCA was only to assess the component environmental impacts in the assembly phase (both scenarios) and operation phase (Scenario 1), a comparison between impact categories was not needed. Thus, normalization was excluded from the investigation.

4.9.5. Interpretation

The final phase of the LCA was the interpretation of the findings by use of the 'network' and 'analyze' button functions. In SimaPro, the LCI input data was first represented by network diagrams that displayed RWH components as subassemblies of the two RWH systems (separate main assemblies). The networks accounted for all the component material, electrical, and transport data and the corresponding environmental burdens from those inputs per kL of harvested rainwater. Flow arrows allowed for organizational links between components and their input data. 'Thermometer' displays were also included as impact gauges for the RWH subsystems. However, these networks only displayed one impact category at a time. Therefore, the findings were also presented as bar graphs, which revealed the component impact contributions per impact category. This display was helpful in assessing a particular component's impact against other elements in the same category. However, it could not be used to compare component effects across the impact categories (e.g., the storage tank effects for global warming could not be compared to the storage tank effects for stratospheric ozone depletion since they have different units). Although normalization was not performed, impact categories with the same units could be compared (see **Table 4-12**). All findings for the LCA analyses are shown and discussed in **Section 5.7**.

4.10. General Limitations and Assumptions

Several limitations and assumptions (not previously discussed) were found:

- Municipal bills were assumed to represent the total school water usage. Hence, water leakage was not considered in the investigation.
- Historical rainfall data was assumed to be representative of future rainfall patterns despite climate change and erratic rainfall.
- The usage of a survey (for the toilet flushing demand) is based on many people's assumptions; hence, inaccuracies may be present from their over/under-estimation of toilet use. Human errors from data recording may also be present.
- Supply scenarios were subjective and case-specific, as they were done according to existing block catchment sizes on site. Hence, a comparative investigation with another school (in the same area) was not done as many variables like total water consumption, and available supply would need to be similar.
- Potential cost savings were estimated for the 2021 municipal rates and are subject to change due to annual tariff increases. Hence future cost savings may fluctuate for the same amount of water savings.
- The impacts of municipal water provision were assumed to be constant.
- Tank sizing was done according to the maximum potential supply of particular school blocks. Certain months with less rain could result in the underutilization of tanks; however, if smaller tanks were used to counteract under-utilization, the maximized savings for other months would be lost.
- The chosen tank sizes would limit the savings for specific months, and more tanks could not be used due to spatial reasons.
- It was noted that component prices might vary between wholesalers/retailers and over time. Hence, implementation costs are not rigid. Furthermore, one can obtain lower-priced components at alternative stores to reduce capital costs and improve (reduce) payback periods.
- Replacement of components was considered; however, the study used different lifespans for the various components. It was noted that the lowest lifespan (10 years) could not be applied to all the subsystems since environmental impacts per savings would be limited and not reflect the actual effects per RWH subsystem.
- It was reiterated that the LCA findings displayed the environmental burdens from component construction and pump use per kL of harvested rainwater.

4.11. Summary

The information provided in this chapter detailed the procedures used to answer the research questions and obtain the study aim and objectives. The methodology incorporated aspects of the literature review and the case study, thereby justifying the chosen considerations. Initially, the case study scenarios were outlined followed by a description of the school data needed. This information included municipal water and energy consumption (from the bills), historical rainfall data (eThekwini Datafeeds), and the toilet-flushing water consumption (survey). The potential implementation scenarios were then laid out in terms of water, cost, and carbon emission savings. Design considerations like tank sizing, gutter, and downpipe checks, and amended savings were also documented. System viability was then described through economic and environmental analyses and included LCAs of the proposed systems. Finally, the limitations and assumptions of the methodology were listed. It was noted that while certain assumptions may have limited the study, they would not invalidate the results.

CHAPTER 5: RESULTS AND DISCUSSION

5.1. Overview

This chapter provided the research outcomes according to the methods of the preceding chapter. Initially, the case study data required for the analysis was gathered, which included the school water and energy consumption, historical rainfall information for the region, and the toilet flushing demand. Thereafter, the prospective rainwater harvesting performance was assessed through the potential rainwater supply and its relation to the demand. The fundamental performance indicators were the possible water, cost, and carbon emission savings introduced by scenario-specific RWH systems.

The system design for both scenarios was also executed and encompassed the gutter-downpipe analysis, storage tank design, and the amended savings. The economic analysis of the RWH systems then occurred, which comprised of the capital costs, pump operation expenses, system repayment periods, and system scenario comparisons. Similarly, an environmental analysis was done involving the system carbon footprints, pump carbon footprint, the period before carbon emission reduction, and comparative performance of the system scenarios from an ecological standpoint. Finally, the results of the LCA were showcased as an extension of the environmental inquiry. An overview of the findings sections and their descriptions is shown in **Table 5-1**.

Table 5-1: Chapter Overview

Section	Brief Description
Section 5.2	Collected data needed for the RWH system performance and design
Section 5.3	Potential system performance findings
Section 5.4	Component design according to potential harvestings and case study data
Section 5.5	Economic feasibility analyses for the proposed RWH systems
Section 5.6	Environmental viability analyses for the proposed RWH systems
Section 5.7	LCA findings

5.2. Case Study Data

5.2.1. Municipal Water Consumption Data

The municipal water consumption was obtained as described in **Section 4.4.1**. The average municipal water usage at Duffs Road Primary School for the period 2018 – 2020 is presented in **Figure 5-1**, and associative costs are shown in **Table 5-2**.

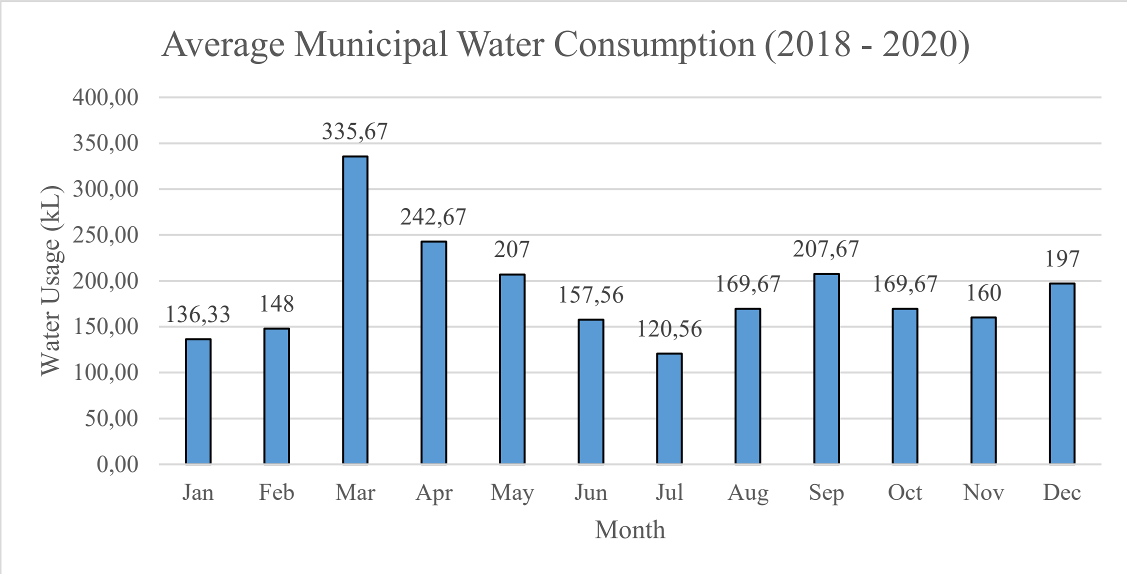

Figure 5-1: Case Study Municipal Water Consumption

Table 5-2: Average Municipal Water Costs at Duffs Road Primary

Month	Water Use (kL)	Municipal Expenses (R)
January	136.33	5562.81
February	148.00	6276.38
March	335.67	11266.02
April	242.67	9291.21
May	207.00	7699.68
June	157.56	6738.51
July	120.56	5695.82
August	169.67	7192.21
September	207.67	8564.67
October	169.67	7272.83
November	160.00	6914.96
December	197.00	8771.33
Total	**2251.78**	**91 246.43**

5.2.2. Municipal Energy Consumption Data

The municipal bills were used to acquire the electricity consumption at the school. The process has been described in **Section 4.4.2**, while the findings are shown in **Figure 5-2** and **Table 5-3**. It was noted that the average energy consumption values were used in the data collection.

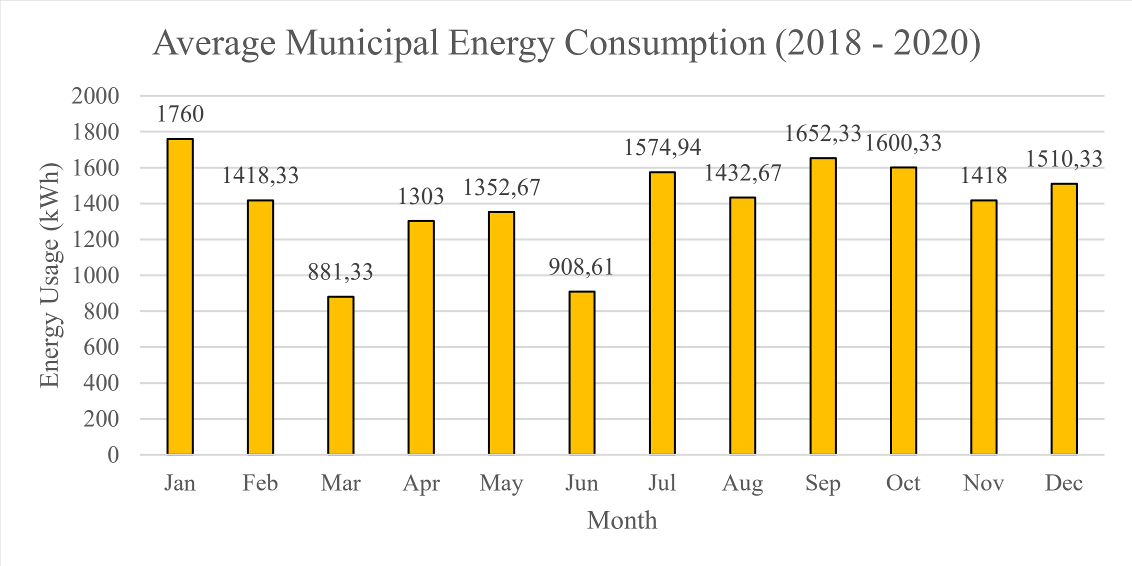

Figure 5-2: Case Study Municipal Energy Consumption

Table 5-3: Average Municipal Electricity Expenses at Duffs Road Primary School

Month	Energy Use (kWh)	Electrical Expenses (R)
January	1760.00	3457.30
February	1418.33	2764.03
March	881.33	1643.95
April	1303.00	2535.25
May	1352.67	2643.87
June	908.61	1761.72
July	1574.94	3291.60
August	1432.67	2913.03
September	1652.33	3459.09
October	1600.33	3337.30
November	1418.00	2964.91
December	1510.33	3153.08
Total	**16 812.55**	**33 925.13**

5.2.3. Historical Rainfall Data

The historical rainfall data was obtained by the method described in **Section 4.4.3** using the eThekwini Datafeeds Website (2021). This website provided graphical representations of the past rainfall values near the case study region. Selected daily precipitation depths are shown in **Appendix B**. **Table 5-4** shows the monthly historical rainfall data for the case study region over the period of 20 years (2001 – 2020).

Table 5-4: Monthly Historical Rainfall (2001 – 2020)

Year	Month											
	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
2001	19.2	19.2	3.2	10.2	4.4	0	9.8	0.2	-	-	87.4	197.8
2002	109.6	61.2	-	-	3	18.6	141	48.2	31.2	21.2	37	1.2
2003	0	12.4	45.6	25.2	16	26.6	0.4	16.4	66.6	17	86.8	48.4
2004	121.4	66.4	49.2	46.8	0.6	2.6	53.6	21.6	39.2	43.8	63.8	36.6
2005	63.6	53.4	64.4	10.8	2.4	40.6	0	20.6	28	36.8	37.8	-
2006	-	-	-	-	-	-	0.2	0	60.4	84.2	67.6	109.4
2007	51.6	42.6	105.4	85	1.6	40.4	0.8	11	28.8	98	109.8	55.2
2008	124.2	125.6	11.8	80.8	6.6	14.2	0.4	0.4	41.2	33.2	70	77.4
2009	81.4	28.4	51.6	17	43.4	1.8	2.2	24.6	34.6	52.6	53.4	115.8
2010	52.2	50	9.4	22	10.8	9.2	0.8	3.6	6	49.8	77.2	100.8
2011	166.6	6.6	51.8	67	26.8	36.6	78.8	28.2	24.4	38.2	205.4	55.8
2012	40.4	27.8	152	32.8	2.6	4	10	0	0	-	-	-
2013	-	-	-	84.6	60.8	30.6	9.2	19.4	26.6	71.2	8	49
2014	109.2	19.4	27.8	16	7	2.6	2.4	8.6	18	65.2	43	50
2015	29.4	20.2	22.4	1.2	-	-	20.2	4.2	8.8	-	-	21.2
2016	55.2	15.4	66.2	12.6	160.4	2	238.4	52.8	68	88	98.2	41.6
2017	67	104.4	29	47.6	108.2	0.4	12.4	25.6	52.8	40.6	177.6	163.4
2018	12.2	81	91.6	27.6	37	0	12.2	27.4	62.2	43.2	15.6	3.4
2019	9.2	21.6	91	97.6	9	7	4.8	9.2	29.8	28.8	116.8	211.6
2020	59.8	148.6	57.4	83	12.6	8.6	24	4.8	34.6	75.6	199.4	72.2

It was noted that the years 2001, 2002, 2005, 2006, 2012, 2013, and 2015 had missing data for certain months. This occurrence was taken into account for the average monthly rainfall estimation (see **Section 5.3.1**). The annual and mean rainfall depths were then calculated by summing and averaging the monthly precipitation values shown in **Table 5-4**. These values are shown in **Table 5-5**.

Table 5-5: Total and Average Rainfall Depths (2001 – 2020)

Year	Total	Average
2001	351.4	35.1
2002	472.2	47.2
2003	361.4	30.1
2004	545.6	45.5
2005	358.4	32.6
2006	321.8	53.6
2007	630.2	52.5
2008	585.8	48.8
2009	506.8	42.2
2010	391.8	32.7
2011	786.2	65.5
2012	269.6	30.0
2013	359.4	39.9
2014	369.2	30.8
2015	127.6	16.0
2016	898.8	74.9
2017	829	69.1
2018	413.4	34.5
2019	636.4	53.0
2020	780.6	65.1

Recalling that the annual rainfall in South Africa averages around 495 mm per year (Colvin et al., 2016), the yearly totals presented in **Table 5-5** generally correspond to this figure. Some years (2011, 2016, 2017, and 2020) showed values significantly higher than the average annual precipitation; however, this is substantiated by more significant rainfall occurrence in the eastern parts of the country. Furthermore, the years with missing historical information (designated in red font in **Table 5-5**) are also similar to the average yearly precipitation, except for the 2012 and 2015 years.

Besides the comparison to the average yearly rainfall, the case study rainfall totals were also compared to each other. When considering the years with missing values, it became essential to liken them to years without missing data. An example would be the 2002 and 2003 totals. While 2002 had missing rainfall data, it provided a yearly sum greater than the 2003 total (which had no lost data that year). Hence, the years with missing data were still deemed helpful in this regard.

5.2.4. Demand for Toilet Flushing

It has been highlighted in Chapters 3 and 4 that the proposed RWH implementation at Duffs Road Primary would cater for toilet flushing, specifically the toilet flushing demand at Block C (ablutions block). Using the method described in **Section 4.4.4**, a survey was conducted (the results of which are shown below). It was noted that two classes (A and B) were present per grade (14 classes in total). However, due to the pandemic, each class was split into two batches, with the attendance of one batch every alternate day.

The flushing behaviour of absentees was assumed according to the general flushing behaviour of those present, and the number of toilet flushes per day was adjusted accordingly. It was found that most people flushed once a day; hence, absentees were assumed to do the same. Pertinent information from the survey is shown in **Table 5-6**.

Table 5-6: Toilet Usage at Block C

Toilet Use Option	Number of pupils	Number of flushes per day (toilet use option x pupils)
Zero	62	0
Once a day	221	221
Twice a day	156	312
Thrice a day	100	300
Four times a day	54	216
Five times a day	16	80
Six times a day	3	18
Seven times a day	1	7
Eight times a day	1	8
More than eight times	0	0
Total	**614**	**1162**
Adjusted	**633**	**1181**

It was further noted that the water consumption used for a single flush amounted to 6 litres (when reading the water meter). This value was assumed to be the same for all the toilets in Block C. The total number of flushes per day (1181) was also constant and applied to the school sessional dates. The number of school days in any given month was assumed to be the same for that month every year. However, the number of school days per month varied based on the 2019 sessional dates (Department of Education, 2019). The 2020 sessional school dates were not used since they were continuously revised due to the COVID-19 pandemic. The findings for this section are tabulated below according to the month, sessional dates, flushes per day, single flush consumption, water demand at Block C, and the total municipal water usage (see **Section 5.2.1**).

Table 5-7: Toilet Flushing Demand at Block C

Month	No. School Days	No. flushes/day	Flush consumption (kL)	Demand at Block C (kL)	Total Water Usage (kL)
Jan	17	1181	0.006	120.46	136.33
Feb	20	1181	0.006	141.72	148.00
Mar	11	1181	0.006	77.95	335.67
Apr	19	1181	0.006	134.63	242.67
May	22	1181	0.006	155.89	207.00
Jun	10	1181	0.006	70.86	157.56
Jul	17	1181	0.006	120.46	120.56
Aug	21	1181	0.006	148.81	169.67
Sep	15	1181	0.006	106.29	207.67
Oct	23	1181	0.006	162.98	169.67
Nov	21	1181	0.006	148.81	160.00
Dec	5	1181	0.006	35.43	197.00
Total	**201**	**14172**	**-**	**1424.29**	**2251.78**

5.3. Potential RWH Implementation

5.3.1. Estimated Monthly Rainfall

The procedure used to obtain the monthly rainfall at Duffs Road Primary was described in **Section 4.5.1** and involved weighted averages of the 20-year historical rainfall data (**Section 5.2.3**). Weighted averages were used to overcome the issue of missing data for certain months and improve the accuracy of the precipitation depth estimation for the case study area. The significant findings are shown in **Table 5-8** and **Figure 5-3**. It was noted that the scores referred to the number of years used for the weighted average.

Table 5-8: Weighted Average of Historical Rainfall Data

Year	Month											
	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
2001	19.2	19.2	3.2	10.2	4.4	0	9.8	0.2	-	-	87.4	197.8
2002	109.6	61.2	-	-	3	18.6	141	48.2	31.2	21.2	37	1.2
2003	0	12.4	45.6	25.2	16	26.6	0.4	16.4	66.6	17	86.8	48.4
2004	121.4	66.4	49.2	46.8	0.6	2.6	53.6	21.6	39.2	43.8	63.8	36.6
2005	63.6	53.4	64.4	10.8	2.4	40.6	0	20.6	28	36.8	37.8	-
2006	-	-	-	-	-	-	0.2	0	60.4	84.2	67.6	109.4
2007	51.6	42.6	105.4	85	1.6	40.4	0.8	11	28.8	98	109.8	55.2
2008	124.2	125.6	11.8	80.8	6.6	14.2	0.4	0.4	41.2	33.2	70	77.4
2009	81.4	28.4	51.6	17	43.4	1.8	2.2	24.6	34.6	52.6	53.4	115.8
2010	52.2	50	9.4	22	10.8	9.2	0.8	3.6	6	49.8	77.2	100.8
2011	166.6	6.6	51.8	67	26.8	36.6	78.8	28.2	24.4	38.2	205.4	55.8
2012	40.4	27.8	152	32.8	2.6	4	10	0	0	-	-	-
2013	-	-	-	84.6	60.8	30.6	9.2	19.4	26.6	71.2	8	49
2014	109.2	19.4	27.8	16	7	2.6	2.4	8.6	18	65.2	43	50
2015	29.4	20.2	22.4	1.2	-	-	20.2	4.2	8.8	-	-	21.2
2016	55.2	15.4	66.2	12.6	160.4	2	238.4	52.8	68	88	98.2	41.6
2017	67	104.4	29	47.6	108.2	0.4	12.4	25.6	52.8	40.6	177.6	163.4
2018	12.2	81	91.6	27.6	37	0	12.2	27.4	62.2	43.2	15.6	3.4
2019	9.2	21.6	91	97.6	9	7	4.8	9.2	29.8	28.8	116.8	211.6
2020	59.8	148.6	57.4	83	12.6	8.6	24	4.8	34.6	75.6	199.4	72.2
mean	65.1	50.2	54.7	42.7	28.5	13.7	31.1	16.3	34.8	52.2	86.4	78.4
scores	18	18	17	18	18	18	20	20	19	17	18	18
weighted	64.3	38.4	52.3	45.9	14.1	8.7	12.1	14.9	35.1	49.2	88.8	64.0

From **Table 5-8**, it can be seen that weighted averages were applied to 18 years of data for the months January, February, April, May, June, November, and December. Similarly, March and October had weighted averages estimated over 17 years, while September received a weighted average rating over 19 years of records. July and August were the only months in the 2001 – 2020 period that did not have missing precipitation values; hence, a weighted mean was calculated for the entire 20-year period. It was noted that the weighted averages were estimated based on similar depths for any given month. Generally, the arithmetic average overestimated the local monthly rainfall depths for the case study area. Thus, the weighted average analysis improved the case study rainfall accuracy by assigning more importance to months of similar depths.

Figure 5-3 (presented next) summarizes the likely rainfall depths at Duffs Road Primary.

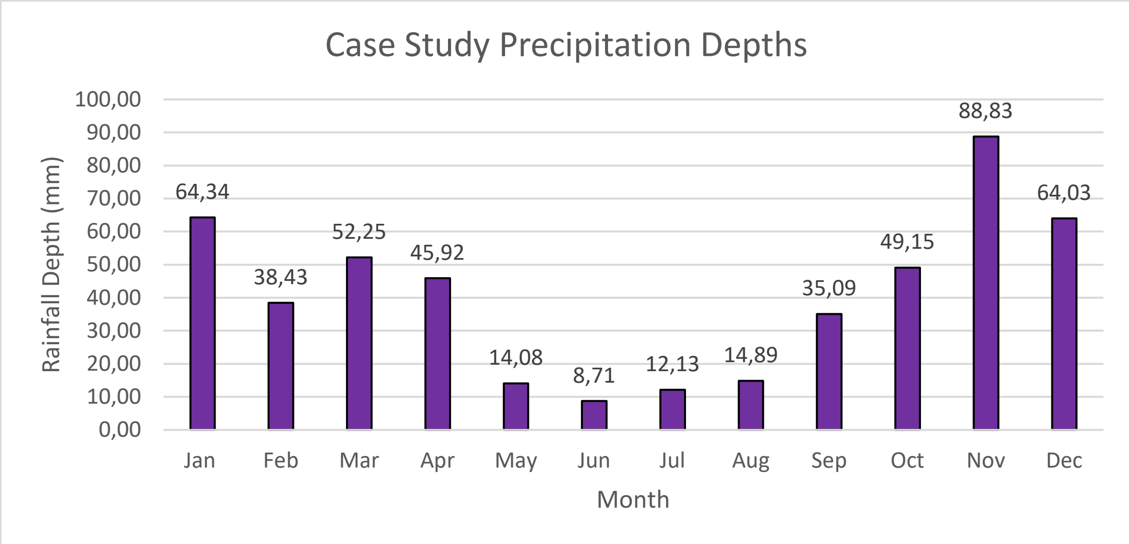

Figure 5-3: Estimated Monthly Rainfall at Duffs Road Primary

The weighted averages that provided the precipitation estimates in **Figure 5-3** suggest higher rainfall at Duffs Road Primary during summer (November, December, January, February, and March). This occurrence corresponds to the rainfall patterns in KwaZulu Natal, often regarded as a summer rainfall region (Red Book, 2013). The winter months like May, June, and July have much less rainfall suggesting the implemented RWH systems would be most effective in the summer since rainwater collections would be maximized. More specifically, the rainfall depths range from 8.71 mm in June to 88.83 mm in November. The approximations represented in **Figure 5-3** were then used to ascertain the potential rainwater supply volume.

5.3.2. Potential Rainwater Supply Volume

The rainwater supply that could theoretically be collected at Duffs Road Primary was estimated based on the precipitation depth estimates (**Section 5.3.1**), block/catchment areas (**Section 3.7.1**), and the stipulations of the case study scenarios (**Section 4.3**). These findings are recorded in the subsequent subsections and were obtained according to the methodology of **Section 4.5.2**.

5.3.2.1. Supply Scenario 1

The first supply scenario involved rainwater collections from Blocks A, B, and C that would be provided to Block C (toilet flushing ablutions) using pumps. The results are presented in **Table 5-9**.

Table 5-9: Potential Rainwater Collection (Scenario 1)

Month	Rainfall (mm)	Area (m^2)	Coefficient	Supply (m^3)
January	64.34	1090.9	0.9	63.17
February	38.43	1090.9	0.9	37.73
March	52.25	1090.9	0.9	51.29
April	45.92	1090.9	0.9	45.08
May	14.08	1090.9	0.9	13.82
June	8.71	1090.9	0.9	8.55
July	12.13	1090.9	0.9	11.91
August	14.89	1090.9	0.9	14.62
September	35.09	1090.9	0.9	34.45
October	49.15	1090.9	0.9	48.26
November	88.83	1090.9	0.9	87.21
December	64.03	1090.9	0.9	62.87
Year/Total	**487.84**	**-**	**-**	**478.97**

5.3.2.2. Supply Scenario 2

The second scenario utilized Blocks E, H, and I for the same purpose as Scenario 1 and excluded pumping. The findings are presented in **Table 5-10**.

Table 5-10: Potential Rainwater Collection (Scenario 2)

Month	Rainfall (mm)	Area (m^2)	Coefficient	Supply (m^3)
January	64.34	1051.15	0.9	60.87
February	38.43	1051.15	0.9	36.36
March	52.25	1051.15	0.9	49.43
April	45.92	1051.15	0.9	43.44
May	14.08	1051.15	0.9	13.32
June	8.71	1051.15	0.9	8.24
July	12.13	1051.15	0.9	11.48
August	14.89	1051.15	0.9	14.09
September	35.09	1051.15	0.9	33.19
October	49.15	1051.15	0.9	46.50
November	88.83	1051.15	0.9	84.04
December	64.03	1051.15	0.9	60.58
Year/Total	**487.84**	**-**	**-**	**461.52**

5.3.3. Supply and Demand Comparison

The comparative scenario analysis of the potential rainwater supply and the toilet flushing demand was done in accordance with the guidelines of **Section 4.5.3**.

5.3.3.1. Scenario 1

Figure 5-4 and **Table 5-11** highlight the supply-demand comparisons under Scenario 1.

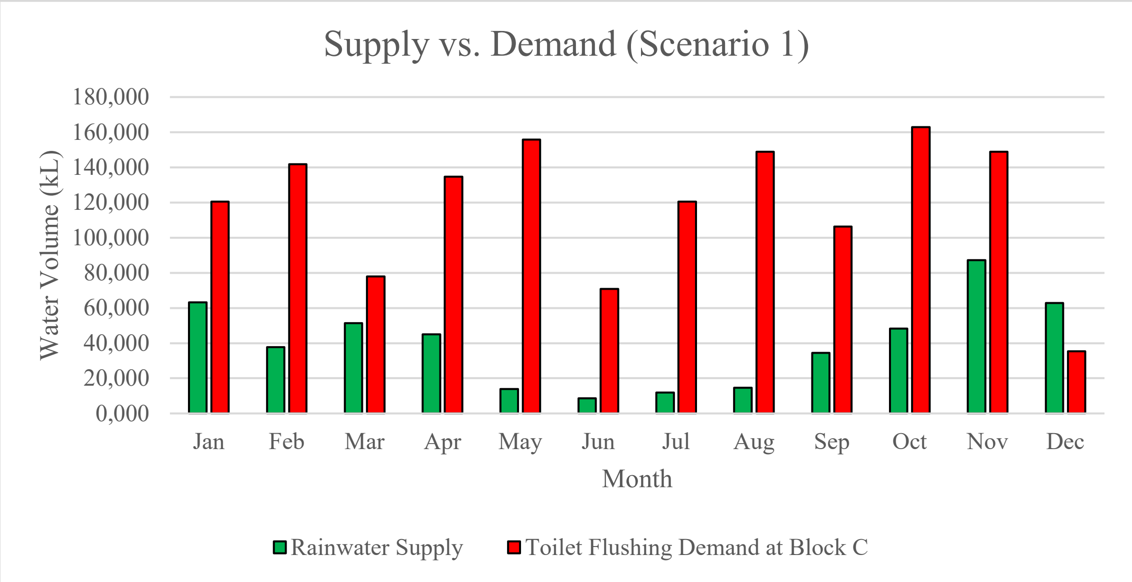

Figure 5-4: Supply vs. Demand for Scenario 1 (pumped RWHS)

Table 5-11: Deficit/Excess Rainwater for Scenario 1

Month	Supply (kL)	Demand (kL)	Deficit/Excess
January	63.17	120.46	-57.29
February	37.73	141.72	-103.99
March	51.29	77.95	-26.65
April	45.08	134.63	-89.55
May	13.82	155.89	-142.07
June	8.55	70.86	-62.31
July	11.91	120.46	-108.55
August	14.62	148.82	-134.19
September	34.45	106.29	-71.84
October	48.26	162.98	-114.72
November	87.21	148.81	-61.59
December	62.87	35.43	+27.44
Total	**478.97**	**1424.29**	**-945.32**

5.3.3.2. Scenario 2

Figure 5-5 and **Table 5-12** juxtapose the supply-demand occurrence for Scenario 2.

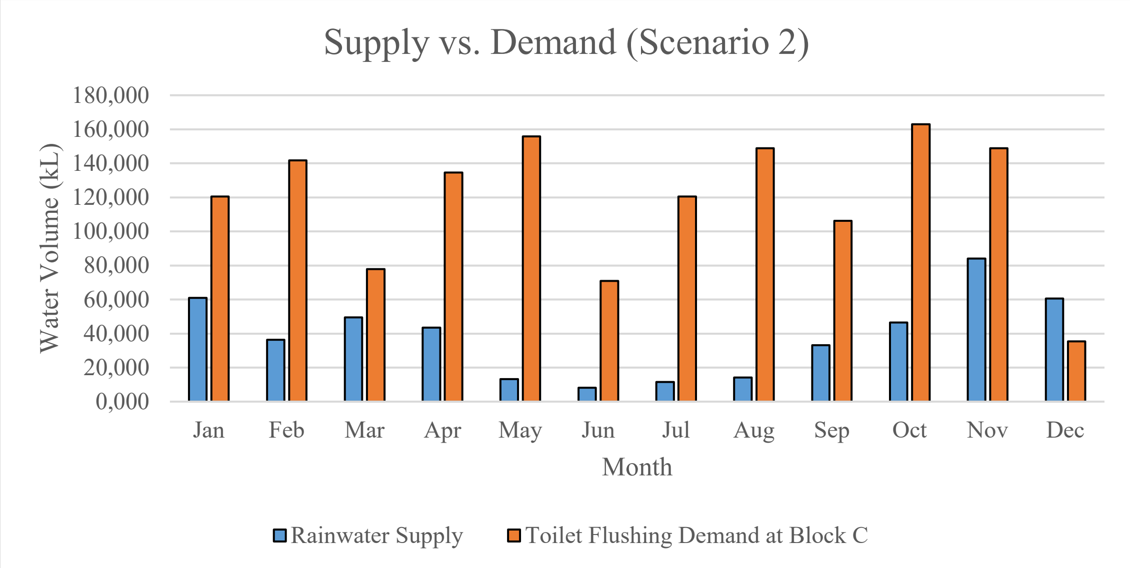

Figure 5-5: Supply vs. Demand for Scenario 2 (gravity-fed RWHS)

Table 5-12: Deficit/Excess Rainwater for Scenario 2

Month	Supply (kL)	Demand (kL)	Deficit/Excess
January	60.87	120.46	-59.59
February	36.36	141.72	-105.36
March	49.43	77.95	-28.52
April	43.44	134.63	-91.20
May	13.32	155.89	-142.57
June	8.24	70.86	-62.62
July	11.48	120.46	-108.98
August	14.09	148.81	-134.72
September	33.19	106.29	-73.10
October	46.50	162.98	-116.48
November	84.04	148.81	-64.77
December	60.58	35.43	+25.15
Total	**461.52**	**1424.29**	**-962.77**

Figures 5-4, 5-5, and **Tables 5-11, 5-12** indicate deficit rainwater occurrences for all months except December. Thus, it can be seen that municipal water supplements would be required for both case scenarios. A yearly water total of 945.32 kL would still be necessary for toilet flushing at Block C under Scenario 1 once all the rainwater is used. Similarly, 962.77 kL of municipal water would be needed to supplement the gravity-fed RWH systems (Scenario 2). It was also found that incorporating collections from all blocks/catchments would result in a yearly shortage of 209.17 kL. This issue, coupled with the impracticable use of all catchments, suggested that the RWHS would not be implemented to replace municipal networks but rather to reduce the municipal water usage in the schooling setup. Therefore, potential water savings (and its associated costs and carbon emission reductions) would still be possible through RWH.

5.3.4. Potential Water Savings

The potential water savings introduced by the RWH system implementation (under Scenarios 1 and 2) were described in **Section 4.5.4** and shown in **Table 5-13**.

Table 5-13: Water Savings from Pumped and Gravity-Fed RWH Systems at Duffs Road

Month	Water Savings (kL) – Scenario 1	Water Savings (kL) – Scenario 2
January	63.17	60.87
February	37.73	36.36
March	51.29	49.43
April	45.08	43.44
May	13.82	13.32
June	8.55	8.24
July	11.91	11.48
August	14.62	14.09
September	34.45	33.19
October	48.26	46.50
November	87.21	84.04
December	62.87	60.58
Total	**478.97**	**461.52**

Scenario 1 indicates higher water savings than Scenario 2 (17.45 kL difference) expectedly due to the rainfall depths, catchment sizes, and chosen RWH configurations. It has been reiterated that higher rainfall depths occur during the summer months (November, December, January, and February), thereby clarifying the larger savings here as opposed to months like June and July (winter). Since the water cost savings and carbon emission reductions are proportional to the physical municipal savings, it follows that RWH systems for Scenario 1 would have higher cost and carbon emission savings than Scenario 2. This affirmation is shown next, albeit separately, according to each scenario.

5.3.5. Potential Cost Savings

The cost savings were calculated using the methodology of **Section 4.5.5**. A tariff of R36.52/kL was applied to the water savings, while a fixed charge rate of R56.35/day was applied to the number of school days (2019 sessional term dates).

5.3.5.1. Cost savings under Scenario 1

The monthly cost savings from the pumped RWH systems are shown in **Table 5-14**.

Table 5-14: Scenario 1 Cost Savings

CS = WS x UR + NSD x FR (**Equation 4-2**)

Month	Water Savings (kL)	Rate (R/kL)	School days	Rate (R/day)	Cost Savings (R)
January	63.17	36.52	17	56.35	3264.98
February	37.73	36.52	20	56.35	2504.99
March	51.29	36.52	11	56.35	2493.14
April	45.08	36.52	19	56.35	2717.05
May	13.82	36.52	22	56.35	1744.49
June	8.55	36.52	10	56.35	875.64
July	11.91	36.52	17	56.35	1392.97
August	14.62	36.52	21	56.35	1717.26
September	34.45	36.52	15	56.35	2103.35
October	48.26	36.52	23	56.35	3058.41
November	87.21	36.52	21	56.35	4368.36
December	62.87	36.52	5	56.35	2577.64
Total	**478.97**	**-**	**201**	**-**	**28 818.26**

From **Table 5-14**, it can be seen that the highest water savings occur in the summer months, which, in turn, yield higher cost savings. However, the cost savings are also proportional to the number of school days present per month since this is when water consumption by the users/pupils occurs. Thus, for the approximate 201 days of school time in a year, municipal water consumption can be reduced by 478.97 kL/annum, which is equivalent to R28 818.26 in municipal water savings per year from the RWH systems.

5.3.5.2. Cost savings under Scenario 2

The monthly cost savings for the gravity-fed RWH systems are shown in **Table 5-15**.

Table 5-15: Scenario 2 Cost Savings

Month	Water Savings (kL)	Rate (R/kL)	School days	Rate (R/day)	Cost Savings (R)
January	60.87	36.52	17	56.35	3180.92
February	36.36	36.52	20	56.35	2454.78
March	49.43	36.52	11	56.35	2424.88
April	43.44	36.52	19	56.35	2657.06
May	13.32	36.52	22	56.35	1726.09
June	8.24	36.52	10	56.35	864.26
July	11.48	36.52	17	56.35	1377.12
August	14.09	36.52	21	56.35	1697.81
September	33.19	36.52	15	56.35	2057.51
October	46.50	36.52	23	56.35	2994.19
November	84.04	36.52	21	56.35	4252.31
December	60.58	36.52	5	56.35	2493.98
Total	**461.52**	**-**	**201**	**-**	**28 180.90**

The yearly cost savings that could be generated from gravity-fed RWH systems amounted to R28 180.90 for municipal water savings of 461.52 kL/annum. These cost savings are R637.36 lower than the savings introduced by Scenario 1. However, this case scenario still presents the same correlations between savings, seasonal rainfall depths, catchment areas, and the RWH system layout configurations. Furthermore, the cost savings introduced per scenario were not the final justification for an RWH system type (pumped or gravity-fed) as indicated by the economic and environmental analyses (see **Sections 5.5** and **5.6**).

5.3.6. Potential Carbon Emission Savings

The final aspect of the RWH system performance for this study involved the potential carbon emission savings generated from the substitution of rainwater over municipal supply. The methodology provided in **Section 4.5.6** was used to obtain the carbon emission savings per case study scenario. It was noted that the physical water savings discussed in **Section 5.3.4** were fundamental to the carbon emission savings estimation.

5.3.6.1. Carbon emission reduction under Scenario 1

Table 5-16 highlights the carbon emission savings generated from the pumped RWH systems. A carbon emission factor of 0.4091 kg CO_2/kL water was applied to all the water savings.

Table 5-16: Carbon emission savings for Scenario 1

Carbon emission savings = 0.4091 x water savings (**Equation 4-3**)

Month	Water Savings (kL)	Carbon emission reductions (kg CO_2)
January	63.17	25.84
February	37.73	15.44
March	51.29	20.98
April	45.08	18.44
May	13.82	5.65
June	8.55	3.50
July	11.91	4.87
August	14.62	5.98
September	34.45	14.09
October	48.26	19.74
November	87.21	35.68
December	62.87	25.72
Total	**478.97**	**195.95**

It was noted that the carbon emission savings (generated through rainwater use in place of municipal supply) ranged from 3.50 kg CO_2 in June to 35.68 kg CO_2 in November. This finding correlated with the monthly rainfall received in the region and the amount of rainwater collected per month since the emission decreases were proportional to the water savings. The yearly carbon emission reductions were estimated at 195.95 kg CO_2. Hence, the RWH systems proposed under Scenario 1 could potentially reduce carbon emissions that would typically be produced during municipal water provision.

5.3.6.2. Carbon emission reduction under Scenario 2

Table 5-17 was used to organize the findings for the carbon emission savings of Scenario 2. As before, a carbon emission factor of 0.4091 kg CO_2 per kL of municipal water was used in the analysis.

Table 5-17: Carbon emission savings for Scenario 2

Month	Water Savings (kL)	Carbon emission reductions (kg CO_2)
January	60.87	24.90
February	36.36	14.87
March	49.43	20.22
April	43.44	17.77
May	13.32	5.45
June	8.24	3.37
July	11.48	4.70
August	14.09	5.76
September	33.19	13.58
October	46.50	19.02
November	84.04	34.38
December	60.58	24.78
Total	**461.52**	**188.81**

As was the case for Scenario 1, Scenario 2 also presented minimum carbon emission savings in June (3.37 kg CO_2) and a maximum in November (34.38 kg CO_2). The annual carbon emission savings of 188.81 kg CO_2 were approximately 7.14 kg less than Scenario 1's emission savings. Thus, in terms of the water, cost, and carbon emission savings, the proposed RWH systems for Scenario 1 slightly outperformed Scenario 2. However, this was only due to the chosen configurations (catchments) chosen per case scenario. With the potential performance of the RWH systems assessed in this section, procedural design aspects like the gutter and downpipe analysis, storage tank setup, and amended savings were explored next.

5.4. Rainwater Harvesting Design

5.4.1. Gutter and Downpipe Analysis

The investigation of the existing gutters and downpipes at Duffs Road Primary involved the procedure described in **Section 4.6.1**. Important aspects included the rough sizing check, risk assessment, design discharge, gutter design, and downpipe design. These various considerations applied to both case scenarios concurrently.

Table 5-18 summarizes the outcomes of the conveyance analysis as conducted using the guidelines of the Red Book (2013).

Table 5-18: Gutter and Downpipe Findings for Block B (worst-case scenario)

Rough Sizing Check		
Parameter	**Value and Unit**	**Reference**
Summer rainfall intensity (I_T)	200 mm/h	Figures 11.1 and 11.2 (Red Book, 2013)
Maximum contributory catchment area (A)	344.90 m^2	Block B, Duffs Road Primary School (Google Earth Pro, 2021)
Recommended gutter cross-sectional area	0.048 m^2	Figure 11.1 (Red Book, 2013)
Actual gutter cross-sectional area	0.05 m^2	Block B, Duffs Road Primary School
Recommended downpipe cross-sectional area	0.034 m^2	Figure 11.2 (Red Book, 2013)
Actual downpipe cross-sectional area	0.035 m^2	Block B, Duffs Road Primary School
Risk Assessment		
Recurrence interval (T)	50 years	Section 11.3.1 (Red Book, 2013)
System design life (N)	10 years	Storage tank guarantee (Jojo, 2021)
Risk of functional failure (P)	18.3%	Section 11.3.1 (Red Book, 2013)
Detailed Design Procedure		
Five minute rainfall intensity (I_T)	340 mm/h	Table 11.3 (Red Book, 2013)
Peak discharge (Q)	0.033 m^3/s	Section 11.3.3 (Red Book, 2013)
Gutter width (w)	250 mm	Block B, Duffs Road
Critical water depth (h_c)	0.120 m	Section 11.3.3 (Red Book, 2013)
Water depth required for the discharge to enter the downpipe (h)	0.264 m	Section 11.3.3 (Red Book, 2013)
Recommended downpipe diameter	174 mm	Section 11.3.4 (Red Book, 2013)

The existing gutters and downpipes on-site were deemed sufficient for rainwater transference since the actual dimensions were greater than the required ones. Ultimately, the findings are recommendations for the worst-case conditions and are not mandatory.

5.4.2. Storage Tank Design

The storage tank design encompassed the procedural elements of **Section 4.6.3**. The tank sizes, platform sizes, and positioning are represented for both case scenarios and were generally sized according to the maximum potential rainwater supply and the short span block distances obtained from Google Earth Pro (2021). The exceptions were Blocks A and C (Scenario 1), which had tanks positioned along their long spans.

5.4.2.1. Scenario 1

The chosen tank layout for Scenario 1 is shown in **Table 5-19**.

Table 5-19: Tank Configuration for Scenario 1

Block	Maximum Potential Supply	Chosen Tank Volumes	Platform length for chosen tank sizes	The length available for positioning
A	15.18 kL	1 x 15 kL	2.7 m	15 m
B	55.15 kL	2 x 20 kL + 1 x 15 kL	8.1 m	10 m
C	16.89 kL	1 x 15 kL	2.7 m	16.5 m

Table 5-19 indicated that Scenario 1 could potentially amass 85 kL of rainwater in any month, which slightly limited the collection for November. Hence, the water, cost, and carbon emission savings required amendment (see **Section 5.4.3**). More specifically, **Table 5-19** shows that the maximum rainwater collection would be reduced by 0.18 kL for Block A, 0.15 kL for Block B, and 1.89 kL for Block C due to the chosen tank volumes/capacities.

It can also be seen from **Table 5-19** that the tank positioning is possible since the platform lengths are less than the required lengths (span distances) for all the blocks. It was further noted that the existing downpipes (located at block corners) would convey rainwater directly into nearby tanks for Blocks A and C. As for Block B, only two existing downpipes could be used to divert water into two outer tanks, which would overflow into a middle tank when storing the peak discharge. This means that the two 20 kL positioned near the downpipes of Block B would collect the initial rainwater supply while the 15 kL placed in between would collect the overflow. Overall, spatial concerns regarding the placement of the tanks were satisfied.

5.4.2.2. Scenario 2

A similar layout was applied to the second case scenario, with **Table 5-20** highlighting the tank configuration specifics.

Table 5-20: Tank Configuration for Scenario 2

Block	Maximum Potential Supply	Chosen Tank Volumes	Platform length for chosen tank sizes	The length available for positioning
E	18.69 kL	1 x 15 kL	2.7 m	10 m
H	33.59 kL	1 x 20 kL + 1 x 10 kL	5.1 m	14 m
I	31.76 kL	1 x 20 kL + 1 x 10 kL	5.1 m	10 m

It was found that the tank configurations and positioning were satisfactory for Scenario 2. The platform lengths were found to be less than the building span distances, and the existing downpipes could be connected directly to the tanks (for all the blocks). As before, the chosen tank volumes corresponded to the maximum potential rainwater supply with slight constraints for collection. This meant that Scenario 2 would collect 75 kL of rainwater at maximum, thereby limiting catchment collections to 15, 30, and 30 kL for Blocks E, H, and I, respectively. An in-depth analysis of the amended savings is discussed further in the next section.

5.4.3. Amended Savings

As previously mentioned in **Section 5.4.2**, the water, cost, and carbon emission savings were amended slightly based on the chosen tank sizes for the case study scenarios. These savings were estimated as described in **Section 4.6.4**. The amended savings are shown in **Tables 5-21** and **5-22** for Scenarios 1 and 2. For Scenario 1, only the month of November required amended water, cost, and carbon emission savings since this was the only month in which the potential rainwater supply exceeded the RWH system capacity (85 kL). The same can be said for Scenario 2, except the RWH system capacity was 75 kL. All other findings previously discussed in **Sections 5.3.4, 5.3.5,** and **5.3.6** were still applicable to this part of the research. Again, the amended water, cost, and carbon emission savings for Scenario 1 were slightly higher than those of Scenario 2. However, as before, these differences were due to the chosen configurations per scenario.

Table 5-21: Amended Savings for Scenario 1

Month	Water Savings (kL)	Cost Savings (R)	Carbon emission reductions (kg CO_2)
January	63.17	3264.98	25.84
February	37.73	2504.99	15.44
March	51.29	2493.14	20.98
April	45.08	2717.05	18.44
May	13.82	1744.49	5.65
June	8.55	875.64	3.50
July	11.91	1392.97	4.87
August	14.62	1717.26	5.98
September	34.45	2103.35	14.09
October	48.26	3058.41	19.74
November	*85.00*	*4287.55*	*34.77*
December	62.87	2577.64	25.72
Total	**476.76**	**28 737.45**	**195.04**

*Note: annual water saving efficiency (WSE) is (476.76/1424.29[demand])*100 = 33.5%*

Table 5-22: Amended Savings for Scenario 2

Month	Water Savings (kL)	Cost Savings (R)	Carbon emission reductions (kg CO_2)
January	60.87	3180.92	24.90
February	36.36	2454.78	14.87
March	49.43	2424.88	20.22
April	43.44	2657.06	17.77
May	13.32	1726.09	5.45
June	8.24	864.26	3.37
July	11.48	1377.12	4.70
August	14.09	1697.81	5.76
September	33.19	2057.51	13.58
October	46.50	2994.19	19.02
November	*75.00*	*3922.35*	*30.68*
December	60.58	2493.98	24.78
Total	**452.48**	**27 850.94**	**185.11**

*Note: annual WSE is (452.48/1424.29)*100 = 31.8%*

5.5. Economic Analysis

5.5.1. Capital Costs

The capital costs of the pumped and gravity-fed RWH systems were calculated based on the component prices obtained from online catalogues. The methodology used was as described in **Section 4.7.1**.

5.5.1.1. Scenario 1

Table 5-23 categorizes the pumped RWH system components according to their unit prices, the number of units required, total component expenses, and online catalogue references. The number of component units required correlated to the block layout and the chosen RWH configurations. Detailed calculations were shown in EXCEL.

Table 5-23: Capital costs for the RWH system (Scenario 1)

Component	Unit price (R)	No. units required	Component expenses (R)	Reference
First-flush diverter	509	4	2036	Makro (2021)
Leaf eater	519	4	2076	Makro (2021)
Tank screen	355	5	1775	Makro (2021)
Storage tank (15 kL)	28 999	3	86 997	Step Building Supplies (2021)
Storage tank (20 kL)	32 500	2	65 000	Step Building Supplies (2021)
Concrete base	1690.65 *(per m^3)*	3	5238.06	Household Quotes (2021)
Polycop piping roll (22mm x 50m)	458	1	458	Leroy Merlin (2021)
0.37 kW booster pump	1799	3	5397	Makro (2021)
Pump-to-tank connector kit	299	5	1495	Makro (2021)
Total capital costs of the pumped RWH system			**R170 472.06**	

It was noted that the storage tanks constituted 89.2% of the total capital costs for the system under Scenario 1, which resembled the findings discussed in the literature review.

5.5.1.2. Scenario 2

The capital costs of the gravity-fed RWHS are shown in **Table 5-24**.

Table 5-24: Capital costs for the RWH system (Scenario 2)

Component	Unit price (R)	No. units required	Component expenses (R)	Reference
First-flush diverter	509	5	2545	Makro (2021)
Leaf eater	519	5	2595	Makro (2021)
Tank screen	355	5	1775	Makro (2021)
Storage tank (10 kL)	13 986	2	27 972	Step Building Supplies (2021)
Storage tank (15 kL)	28 999	1	28 999	Step Building Supplies (2021)
Storage tank (20 kL)	32 500	2	65 000	Step Building Supplies (2021)
*Concrete base	1690.65 *(per m^3)*	3	5005.25	Household Quotes (2021)
Polycop piping roll 1 (22mm x 50m)	458	2	916	Leroy Merlin (2021)
Polycop piping roll 2 (22mm x 25m)	239	1	239	Leroy Merlin (2021)
Total capital costs of the gravity-fed RWH system			**R135 046.25**	

It was noted that while the RWH system for Scenario 1 provided higher water and cost savings, Scenario 2's system would be cheaper to implement. More specifically, the gravity-fed RWH system would be R35 425.81 less than the pumped RWH system, which, again, can be attributed to the catchment configurations. Furthermore, the Scenario 2 system was found to have fewer components (no pumps), thereby allowing for reduced capital expenses. It was also shown that the storage tanks carried 90.3% of the capital costs, which is similar to the cost proportions for Scenario 1 (89.3% of the capital expenses). As a side note, the concrete prices were calculated by volume using the cubic platform widths, lengths, and depths highlighted in **Section 5.4.2**.

5.5.2. Pump Operating Costs

With regards to Scenario 1, the pumps located at Blocks A, B and C would operate immediately (and at random) after toilet flushing in Block C (depending on which block has available rainwater supply). The pump functioning costs were calculated according to the method provided in **Section 4.7.2**. A summary of the findings is presented in **Table 5-25**. The parameters described were assumed to be constant every year.

Table 5-25: Economic Outcomes from Pump Operation

Parameter	Value
Number of school days in a year	201 days
Number of flushes in a day at Block C	1181 flushes/day
Number of flushes in a year at Block C	237 381 flushes/year
Yearly toilet flushing demand for Block C	1424.29 kL/annum
Annual rain supply for Block C (amended)	476.76 kL/year
Supply-demand ratio	33.47%
The yearly proportion of flushes using rainwater supply only	79 459 flushes/year
Annual pump operation (usage)	79 459 times a year
Cistern refill time	0.013 hours (48 seconds)
Yearly pump operation time	1059.46 hours (a year)
Pump motor power	0.37 kW
Annual energy consumption from pumping	392 kWh/year
Energy tariff	R2.0562/kWh
Yearly pump operating costs	**R806.03/annum**

Recalling that the average yearly municipal consumption for the period 2018 to 2020 is 16 812.55 kWh (or R33 925.13/year in costs) (see **Section 5.2.2**), the implementation of a pumped RWH system would cause a 2.3% electricity consumption increase and a 2.4% cost increase at the school. Thus, the pumps would not present a significant increase in the municipal energy usage at Duffs Road Primary. Furthermore, the cost of running the system pumps (R806.03/annum) was found to be substantially less than the annual cost savings introduced by the RWH system (R28 737.45/year).

5.5.3. Payback Period

The repayment periods were calculated using the ratio of the capital costs and annual (amended) system cost savings as stipulated in **Section 4.7.3**.

5.5.3.1. Scenario 1

The repayment period was calculated using **Equation 4-8**, as follows:

$$Payback\ period = \frac{Scenario\ 1\ capital\ cost}{Scenario\ 1\ system\ savings} = \frac{170\ 472.06}{28\ 737.45} = 5.93\ years$$

Hence, it was found that it would take almost 6 years (5 years, 11 months) for the system savings to match the investment costs, after which the RWH system would begin to save on municipal water expenses. The repayment period was considered reasonable since it was less than the ten-year guarantees of the storage tanks (which were the most costly components of the system).

5.5.3.2. Scenario 2

For Scenario 2, the repayment period was calculated as follows:

$$Payback\ period = \frac{Scenario\ 2\ capital\ costs}{Scenario\ 2\ system\ savings} = \frac{135\ 046.25}{27\ 850.94} = 4.85\ years$$

Scenario 2 presented a repayment period of 4 years and 10 months, which was approximately one year shorter than the repayment period for Scenario 1. As before, the repayment period was practical.

5.5.4. Economic Summary of Pumped and Gravity-fed Systems

A comparative analysis of the scenario-specific RWH systems was performed as described in **Section 4.7.4**. **Table 5-26** highlights the economic performances of the RWH systems implemented under the two case study scenarios.

Table 5-26: Economic outcomes from RWH implementation

RWH outcome	Scenario 1	Scenario 2	Optimal system
Amended water savings	476.76 kL/year	452.48 kL/annum	Scenario 1
Amended cost savings	R28 737.45/annum	R27 850.94/year	Scenario 1
Capital costs	R170 472.06	R135 046.25	Scenario 2
Return period	5.93 years	4.85 years	Scenario 2

5.6. Environmental Analysis

5.6.1. System Carbon Footprint

The system carbon footprint was estimated using the procedure of **Section 4.8.1**. General considerations that were applied to both scenarios are shown in **Table 5-27**. The carbon emission factors were obtained from Winnipeg (2012) and Sustainable Concrete (2018). The references for the component masses and materials were the same as those used to source the component prices (see **Table 5-23** and **Table 5-24**). Jojo (2021) was used as the reference in cases where the material type was not given.

Table 5-27: Characteristics of the RWH components

Component	Material	Unit mass (kg)	Carbon emission factor (kg CO_2/kg material)
First-flush diverter	Polyvinyl chloride (PVC)	0.5	2.22
Leaf eater	Polypropylene (PP)	0.7	1.95
Tank screen	Stainless steel	0.1	6.15
Storage tank (10 kL)	Linear low-density polyethylene (LLDPE)	180	1.01
Storage tank (15 kL)	LLDPE	270	1.01
Storage tank (20 kL)	LLDPE	360	1.01
*Platform/base	Concrete	2400 *(per m^3)*	0.0721
Polycop piping roll 1 (22mm x 50m)	PP	5	1.95
Polycop piping roll 2 (22mm x 25m)	PP	2	1.95
0.37 kW booster pump (body)	Cast iron	6.75	1.51
0.37 kW booster pump (shaft)	Stainless steel	1.35	6.15
0.37 kW booster pump (impeller)	Brass (copper-zinc alloy)	0.9	6.18
Pump-to-tank connector kit	PP	1.2	1.95

5.6.1.1. Scenario 1

Table 5-28 shows the carbon footprint for Scenario 1, calculated in conjunction with the information provided in **Table 5-27**.

Table 5-28: System Carbon Footprint (Scenario 1)

Component carbon footprint = component mass x carbon emission factor (**Equation 4-9**)

Component	Number of units required	Total component mass (kg)	Carbon emission factor (kg CO_2/kg material)	Component carbon footprint (kg CO_2)
First-flush diverter	4	2	2.22	4.44
Leaf eater	4	2.8	1.95	5.46
Tank screen	5	0.5	6.15	3.08
Storage tank (15 kL)	3	810	1.01	818.10
Storage tank (20 kL)	2	720	1.01	727.20
*Platform/base	3	7435.8	0.0721	536.12
Polycop piping roll (22mm x 50m)	1	5	1.95	9.75
0.37 kW booster pump (body)	3	20.25	1.51	30.58
0.37 kW booster pump (shaft)	3	4.05	6.15	24.91
0.37 kW booster pump (impeller)	3	2.7	6.18	16.69
Pump-to-tank connector kit	5	6	1.95	11.70
Total carbon footprint for Scenario 1				**2188.02 kg CO_2**

It was noted that the pump had a total mass of 9 kg with assumed proportions of 75% making up the pump body (6.75 kg), 15% the shaft (1.35 kg), and 10% the impeller (0.9 kg). The concrete base masses were determined from the volumes used to calculate the prices and the density of concrete (2400 kg/m^3). The total carbon footprint for the pumped

RWH system was estimated at 2188.02 kg CO_2. It was found that the storage tanks contributed towards the most carbon emissions (70.6% of the system carbon footprint), while the concrete bases constituted 24.5% of the carbon impacts. As these components were the heaviest parts of the RWH system, this outcome was expected.

5.6.1.2. Scenario 2

Table 5-29 displays the system carbon footprint for Scenario 2, calculated as before.

Table 5-29: System Carbon Footprint (Scenario 2)

Component	Number of units required	Total component mass (kg)	Carbon emission factor (kg CO_2/kg material)	Component carbon footprint (kg CO_2)
First-flush diverter	5	2.5	2.22	5.55
Leaf eater	5	3.5	1.95	6.83
Tank screen	5	0.5	6.15	3.08
Storage tank (10 kL)	2	360	1.01	363.60
Storage tank (15 kL)	1	270	1.01	272.70
Storage tank (20 kL)	2	720	1.01	727.20
*Platform/base	3	7105.32	0.0721	512.29
Polycop piping roll 1 (22mm x 50m)	2	10	1.95	19.50
Polycop piping roll 2 (22mm x 25m)	1	2	1.95	3.90
Total carbon footprint for Scenario 2				**1914.64 kg CO_2**

Table 5-29 shows that the total carbon footprint for Scenario 2 is 1914.64 kg CO_2. This impact is 273.37 kg less than the emissions produced under Scenario 1, suggesting that the gravity-fed RWH system is more environmentally friendly. Hence, the absence of specific components like the pump makes this system more ecologically viable as there are no pump emissions to counteract and exceed the carbon emission savings that would

be introduced. As before, the highest carbon emission contributors came from the tanks and the concrete platforms, which make up 71.2% and 26.8% of the emissions, respectively. These percentages are similar to those calculated for Scenario 1 (70.6% for the tanks and 24.5% for the concrete platforms).

5.6.2. Pump Carbon Footprint

It has been alluded to in the previous section that the presence of pumps under Scenario 1 would generate carbon emissions (through its operation). The procedure used to calculate the pump carbon footprint is shown in **Section 4.8.2**. The findings encompassing the pump carbon footprint are shown in **Table 5-30**.

Table 5-30: Environmental Outcomes from Pump Operation

Parameter	Value
Number of school days in a year	201 days
Number of flushes in a day at Block C	1181 flushes/day
Number of flushes in a year at Block C	237 381 flushes/year
Yearly toilet flushing demand for Block C	1424.29 kL/annum
Annual rain supply for Block C (amended)	476.76 kL/year
Supply-demand ratio	33.47%
The yearly proportion of flushes using rainwater supply only	79 459 flushes/year
Annual pump operation (usage)	79 459 times a year
Cistern refill time	0.013 hours (48 seconds)
Yearly pump operation time	1059.46 hours/annum
Pump motor power	0.37 kW
Annual energy consumption from pumping	392 kWh/year
Carbon emission factor for electricity	0.957 kg CO_2/kWh (Spalding-Fecher, 2011)
Yearly carbon emissions from pumping	**375.14 kg CO_2/annum**

It was found that the yearly carbon emissions produced from the pumps amounted to 375.14 kg CO_2 per year. When comparing this value to the carbon emissions savings that could be introduced by the Scenario 1 RWH system (195.04 kg CO_2), the pump emissions counteracted and exceeded the system savings by 180.10 kg CO_2. Hence, the pumped RWH system would produce excess carbon emissions, thereby reducing its ecological performance and feasibility.

5.6.3. Period Before Carbon Emission Reduction

The timeframe in which the proposed RWH systems would reduce carbon emissions (that would usually be generated through municipal supply) was calculated as the ratio of the system carbon footprint and the yearly (amended) carbon emission savings for each scenario. The method has been described in **Section 4.8.3**.

5.6.3.1. Scenario 1

The period before carbon emission reduction (**Equation 4-10**) was calculated as follows:

$$Reduction\ period\ = \frac{Scenario\ 1\ footprint}{Scenario\ 1\ carbon\ savings} = \frac{2188.017}{195.041} = 11.22\ years$$

This period of 11 years and 3 months would be the time taken to reduce the initial carbon footprint of the system components (excluding the pump). The pump would always generate carbon emissions through its electrical usage.

5.6.3.2. Scenario 2

The carbon emission reduction period was calculated as follows:

$$Reduction\ period\ = \frac{Scenario\ 2\ footprint}{Scenario\ 2\ carbon\ savings} = \frac{1914.644}{185.110} = 10.34\ years$$

Scenario 2 presents a period approximately one year shorter than Scenario 1, indicating better environmental performance.

5.6.4. Environmental Summary of the Pumped and Gravity-fed Systems

A summary of the environmental performances of the RWH systems has been provided in **Table 5-31**, which contrasts the outcomes of the case study scenarios.

Table 5-31: Environmental outcomes from RWH implementation

RWH outcome	Scenario 1	Scenario 2	Optimal system
Amended water savings	476.76 kL/annum	452.48 kL/annum	Scenario 1
Amended carbon emission savings	195.04 kg CO_2/year	185.11 kg CO_2/year	Scenario 1
System carbon footprint	2188.02 kg CO_2	1914.64 kg CO_2	Scenario 2
Period before emission reduction	11.22 years	10.34 years	Scenario 2

5.7. System Life Cycle Assessment

The environmental burdens generated from system assembly and use were displayed according to the components and selected impact categories for the two RWH scenarios. It is important to note that environmental burdens from 1 kg of CO_2 eq are not equivalent to the environmental loads of one unit of another impact category. This is the case for all impact categories having different units. As 18 impact categories were present, only a few network diagrams were shown in this part of the report and were chosen based on subjective importance. The remaining impact category diagrams can be found in **Appendix F**. As previously mentioned, Scenarios 1 and 2 were not directly compared due to differences in component composition and potential water savings. However, the operational impacts were compared to the assembly impacts for Scenario 1 only.

5.7.1. Global Warming Burdens

The global warming burdens of the RWH system elements were calculated for the two case study scenarios and are discussed in the following subsections

5.7.1.1. Scenario 1 Impacts

Figure 5-6 presents the most prominent RWH components and their corresponding global warming impacts per kL of harvested rainwater. It was noted that the pre-filtration mechanisms and transport inputs were not included in the network diagram display due to their negligible contributions.

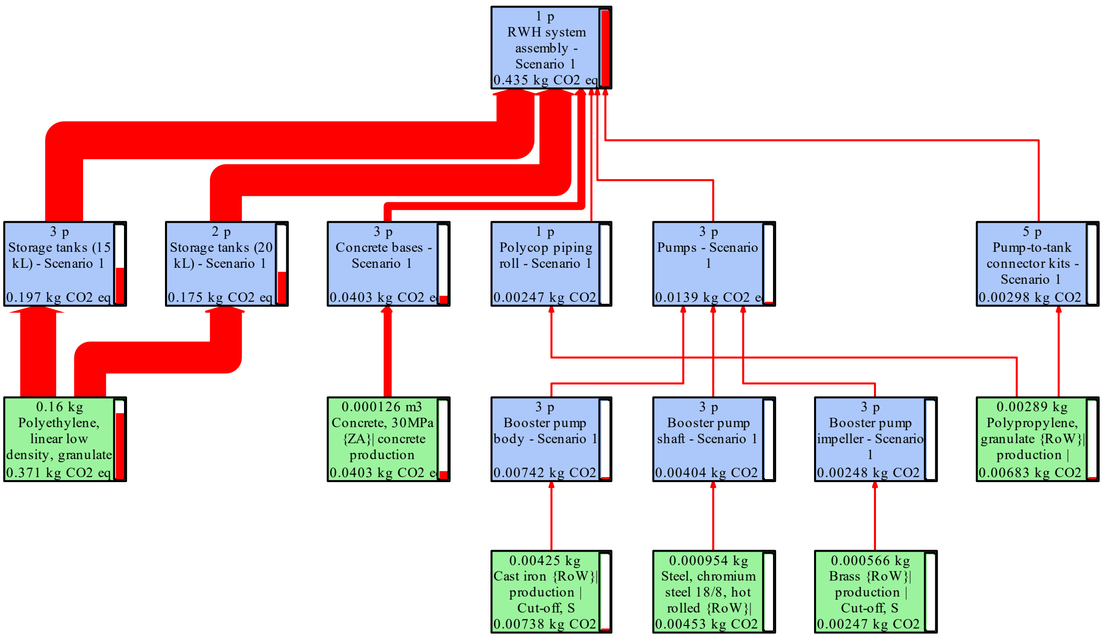

Figure 5-6: Global warming burdens per kL of harvested rainwater (Scenario 1 assembly)

It was found that for every kL of rainwater harvested under the first scenario, 0.435 kg of CO_2 eq would be generated (from the component assemblies). The most notable global warming impacts were shown to come from the production of the storage tanks and concrete bases (indicated by the thicker red lines). The three 15 kL tanks were shown to produce 45.3% of the total emissions (0.197 kg CO_2 eq), while the two 20 kL tanks generated 40.3% of the emissions (0.175 kg CO_2 eq). However, the component assemblies were not the only consideration for Scenario 1. The operational outputs of the pumps were also assessed during their lifespans, having environmental burdens in all the impact categories. The global warming effects from lifetime pump operation are shown in the next network diagram along with the assembly phase highlighted previously.

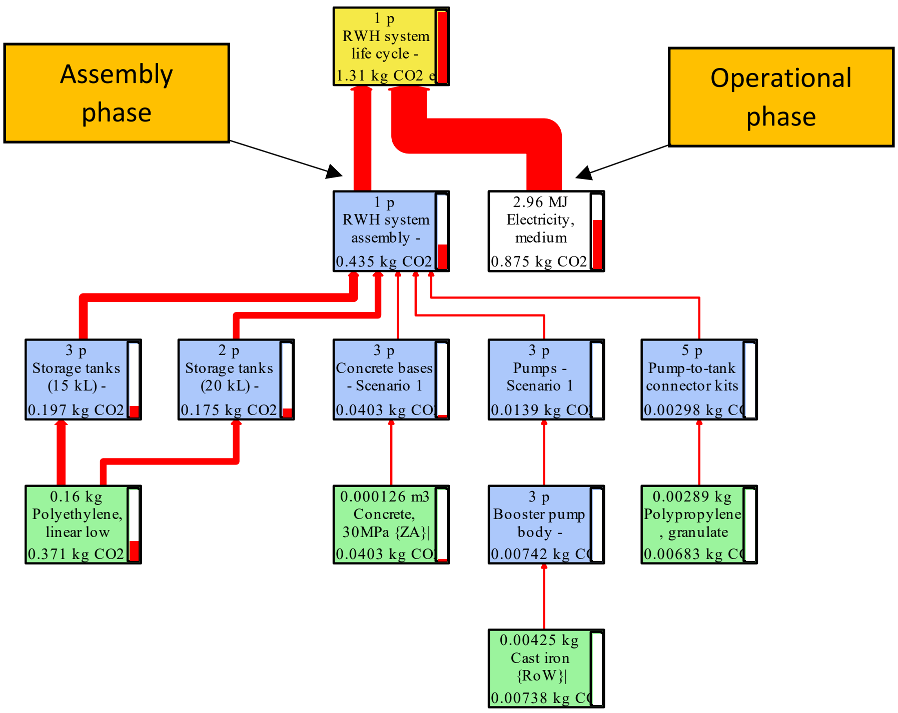

Figure 5-7: Global warming impacts from assembly and operation (Scenario 1)

It is evident from **Figure 5-7** that the assembly of the RWH system has a lower global warming effect than its electrical operation. More specifically, the assembly of all the components generated 0.435 kg CO_2 eq (per kL of harvested rainwater), while the pump lifetime operation produced twice that effect (0.875 kg CO_2). This value was then used to check the yearly carbon emissions for system harvestings of 476.76 kL/annum. Recalling that the yearly pump emissions were calculated as 375.14 kg CO_2 (see **Table 5-30**); SimaPro estimated these pump carbon emissions to be 417.16 kg CO_2 (0.875 kg CO_2

multiplied by 476.76 kL). The 42 kg difference was attributed to variances in the SimaPro modelling method and the empirical method (**Section 4.8.2**).

5.7.1.2. Scenario 2 Impacts

For Scenario 2, no electrical output was needed for system operation. Hence, only the assembly network diagram was computed using SimaPro. This diagram (depicted in **Figure 5-8**) showed the environmental burdens of component construction (per kL of harvested rainwater). A direct comparison between Scenarios 1 and 2 was not performed due to their differences in annual water savings/harvestings and component compositions.

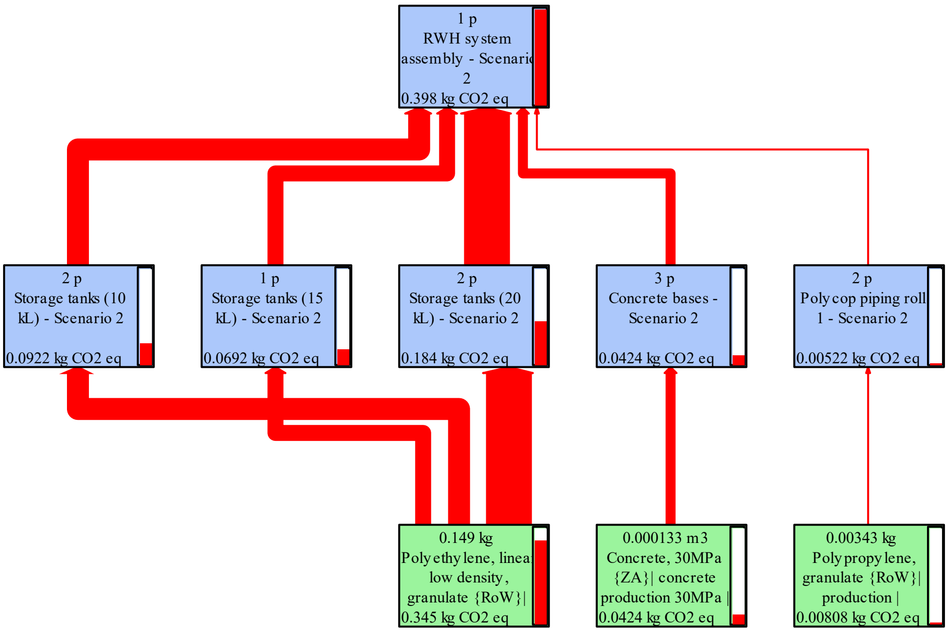

Figure 5-8: Global warming burdens per kL harvested rainwater (Scenario 2 assembly)

The total global warming effects of the component assemblies (per kL of harvested rainwater) were found to be 0.398 kg CO_2 eq for Scenario 2. **Figure 5-8** indicates that the largest influence on global warming originated from tank fabrication. As two 20 kL storage tanks were adopted under this system configuration, their global warming impacts amounted to 0.184 kg CO_2 eq or almost half (46.2%) of the total burdens. The remaining noteworthy contributions were due to the two 10 kL tanks and the single 15 kL tank, which contributed to 23.2% (0.0922 kg CO_2 eq) and 17.4% (0.0692 kg CO_2 eq) of the total impacts, respectively. As before, the pre-filtration component effects and transport impacts were negligible in comparison to the other components discussed and were not shown in the network diagram.

5.7.2. Toxicity Burdens

The toxicity effects of the two RWH systems were displayed according to five impact categories: terrestrial, freshwater, and marine ecotoxicity, and human carcinogenic and non-carcinogenic toxicity. The reasoning for this grouping was to assess and compare these impacts as they were all of the same unit (kg 1,4-DCB). Unlike the global warming impacts, network diagrams were not used to display the ecological burdens as these illustrations could only show one impact category at a time. Hence, numerical figures were displayed as impact percentages. The total burdens (in kg 1,4-DCB) for all the toxicity categories are summarized in **Section 5.7.5**.

5.7.2.1. Scenario 1 Impacts

The toxicity burdens for Scenario 1 assembly are shown in **Figure 5-9**. The data for the toxicity impact categories were transposed into EXCEL to visually represent the impact percentages in relation to each other.

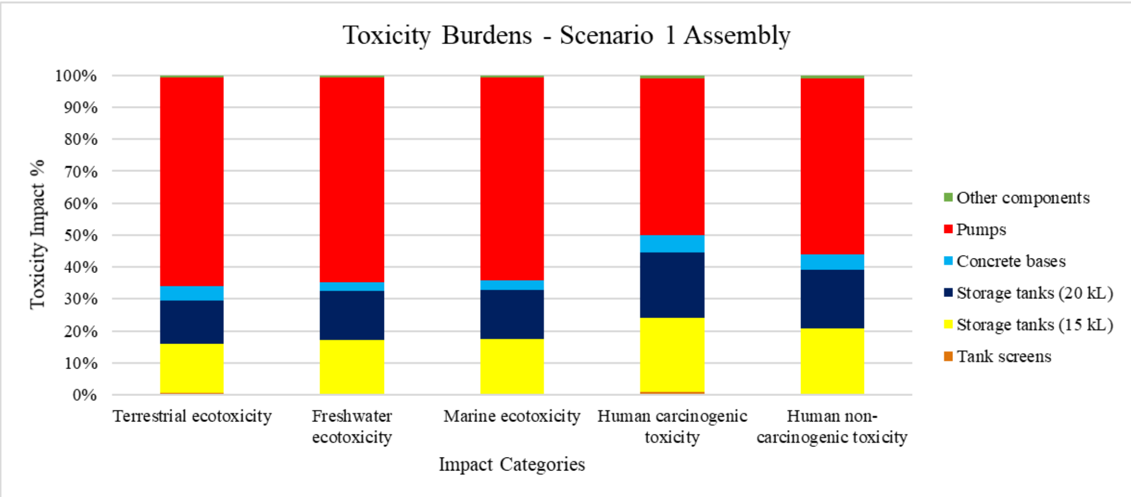

Figure 5-9: Toxicity burdens per kL harvested rainwater (Scenario 1 assembly)

According to the analysis performed on SimaPro, it was found that the construction of the pumps dominated the environmental burdens for all five toxicity impact categories. This finding can be seen in **Figure 5-9**, with the pumps (indicated as red bars) contributing towards half (or more) of the toxicity effects. Ecotoxicity impact percentages for the pumps ranged from 63.4% (marine) to 65.2% (terrestrial) of the total impacts, while the human toxicity pump impacts were slightly lower at 49.2% (carcinogenic) and 55.1% (non-carcinogenic). For the storage tanks (15 and 20 kL), ecotoxicity impact percentages ranged from 13.6% to 17.4% of the total burdens.

On the other hand, the human toxicity impacts from tank manufacturing had a higher impact range between 18.4% and 23.1%. Hence, it was shown that while the pumps bore most of the environmental burdens, the impacts were greater for the ecotoxicity categories as opposed to the human toxicity ones. In other words, pump assemblies were found to be more damaging in the terrestrial, freshwater, and marine contexts. Components like the pre-filtration mechanisms, Polycop piping roll, and pump-to-tank connector kits were shown to have negligible impacts (less than 2% across all five toxicity categories).

It was also noted that the toxicity burdens of the assembly phase needed to be compared to those of the operation phase (pump electricity usage) as per the stipulated objectives. Thus, a summary of these findings is shown in **Table 5-32** as obtained from the SimaPro network diagrams.

Table 5-32: Total toxicity impacts for the assembly and operational phases (Scenario 1)

Impact Category	Assembly impact (per kL harvested rainwater)	Operational impact (per kL harvested rainwater)
Terrestrial ecotoxicity	2.2 kg 1,4-DCB	0.92 kg 1.4-DCB
Freshwater ecotoxicity	0.0368 kg 1,4-DCB	0.0261 kg 1,4-DCB
Marine ecotoxicity	0.0471 kg 1,4-DCB	0.0356 kg 1,4-DCB
Human carcinogenic toxicity	0.023 kg 1,4-DCB	0.0599 kg 1,4-DCB
Human non-carcinogenic toxicity	0.561 kg 1,4-DCB	1.04 kg 1,4-DCB

Generally, it can be seen that the total assembly impacts are higher than the operational impacts (per kL of harvested rainwater) for the ecotoxicity impact categories. Conversely, the human toxicity categories suggested higher operational burdens. While the differences between assembly and operational impacts were relatively small for the freshwater and marine categories, it was noted that the 'human carcinogenic' group had operational impacts 2.6 times larger than the assembly burdens.

On the other hand, the terrestrial ecotoxicity operational burdens were 2.4 times less than the assembly burdens. Hence, these two categories (human carcinogenic and terrestrial) displayed the largest impact ranges, albeit in contrasting contexts. Overall, it was found that the assembly impacts under the terrestrial classification accounted for the largest ecological burdens (2.2 kg 1,4-DCB). The lowest impacts were shown to come from the assembly phase of the human carcinogenic category (0.023 kg 1,4-DCB).

5.7.2.2. Scenario 2 Impacts

The toxicity burdens for Scenario 2 assembly were obtained from SimaPro in a similar fashion to the findings for Scenario 1. The results are shown in **Figure 5-10**.

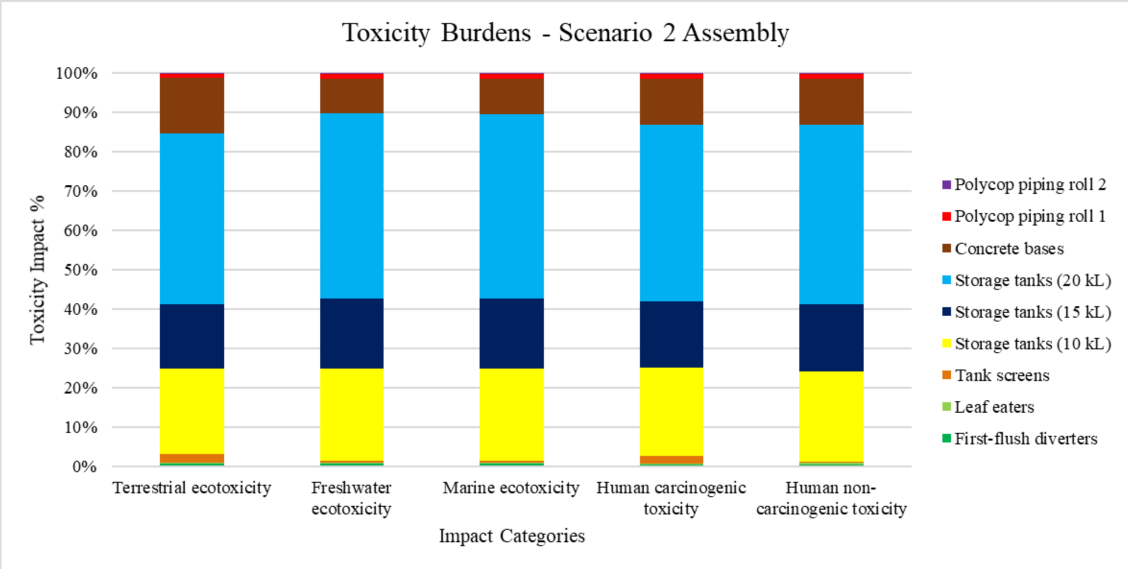

Figure 5-10: Toxicity burdens per kL harvested rainwater (Scenario 2 assembly)

As before, the toxicity burdens were grouped according to the three ecotoxicity categories and the two human toxicity categories. It is evident from **Figure 5-10** that the two 20 kL storage tanks (indicated as light blue bars) presented the greatest burdens across all five toxicity categories. It was found that the impact percentages for the 20 kL storage tanks ranged from 43.3% (terrestrial) to 47.1% (freshwater) of the total impacts. The second-highest impact providers were the two 10 kL tanks (yellow bars). These tanks contributed between 21.7% (terrestrial) and 23.5% (freshwater and marine) of the total toxicity burdens. The effects from the single 15 kL tank (dark blue bars) were in the range of 16.2% (terrestrial) and 17.7% (freshwater) of the total impacts. The last significant contributions were due to the concrete bases having 8.76% (freshwater) to 14.2% (terrestrial) of the entire burdens.

It was also noted that the freshwater and marine groups had similar impact percentages for all the RWH components, whereas the terrestrial group displayed more significant impacts from the tank screens (orange bars). The human carcinogenic group was the only other category to display higher burdens from the tank screens but otherwise, had similar impact percentages to the non-carcinogenic category. Overall, the most negligible impacts came from the Polycop piping rolls and pre-filtration components like the leaf-eaters and first-flush diverters.

5.7.3. Resource Scarcity Burdens

The resource scarcity burdens were displayed according to the mineral resource and fossil resource impact categories. However, due to them having different units, these categories were not grouped and compared. The network diagrams for the Scenario 1 assembly and operation phases were obtained from SimaPro and shown in **Section 5.7.3.1**. Similarly, the network diagrams of the assembly phase of Scenario 2 were displayed and discussed (for the two scarcity types) in **Section 5.7.3.2**.

5.7.3.1. Scenario 1 Impacts

The mineral resource burdens (per kL of rainwater harvested) from the assembly phase of Scenario 1 are shown in **Figure 5-11**.

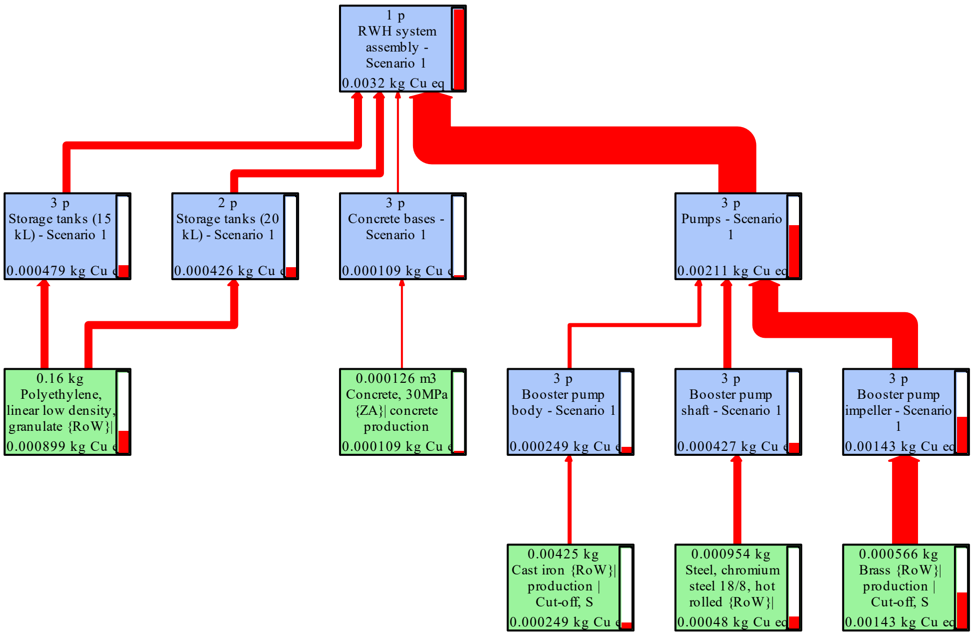

Figure 5-11: Mineral resource burdens per kL harvested rainwater (Scenario 1 assembly)

The total mineral resource burdens were estimated as 0.0032 kg Cu eq per kL of rainwater harvested by the Scenario 1 RWH system. For this impact category, 65.9% of the burdens (or 0.00211 kg Cu eq) were attributed to the pump assemblies. As the pumps were made up of cast iron, stainless steel, and brass it was inferred that these materials had a greater effect on mineral scarcity when compared to LLDPE (tanks) and concrete (bases). Concerning the tank and base components, it was found that the three 15 kL tanks contributed towards 14.9% of the burdens, the two 20 kL tanks made up 13.3% of the effects, and the concrete bases 3.4% of the total impacts.

The mineral resource operational impacts (per kL of harvested rainwater) were also assessed for Scenario 1 and are shown in **Figure 5-12**.

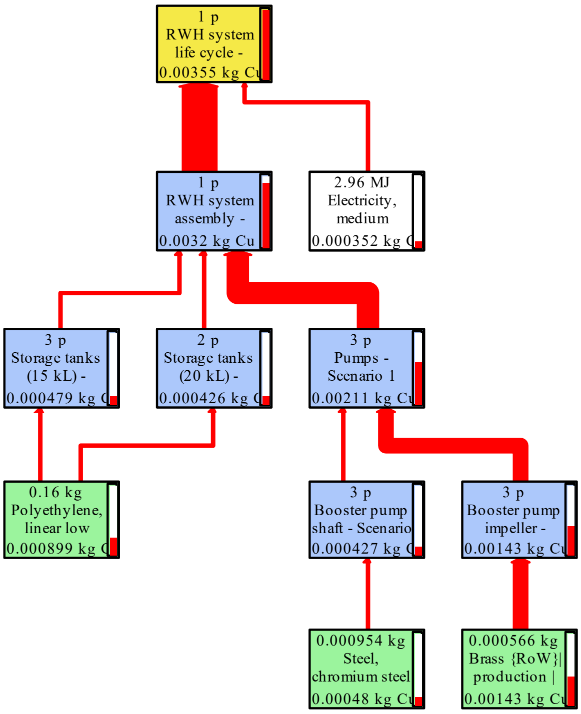

Figure 5-12: Mineral resource impacts from assembly and operation (Scenario 1)

Figure 5-12 illustrates the assembly burden (blue box) in relation to the operational burden (white box) for the mineral resource impact category. It is clear that the total assembly impacts (0.0032 kg Cu eq) are much greater than the operational burdens (0.000352 kg Cu eq) accounting for 90.1% of the life cycle impacts. With respect to the other resource scarcity category (i.e. fossil), the burdens of the assembly and operation phases were represented by one network diagram (see **Figure 5-13** overleaf).

It can be seen from **Figure 5-13** that the assembly and operational burdens in the 'fossil resource scarcity' context are closely matched. Of the total fossil resource impacts (0.535 kg oil eq), the assembly burdens amounted to 0.277 kg oil eq, while operational burdens were estimated to be slightly lower at 0.257 kg oil eq. Hence, the impact proportion was slightly skewed towards the assembly part, which generated 51.8% of the total impacts for this category. Much of the assembly impacts were attributed to the storage tanks since the three 15 kL tanks and two 20 kL tanks accounted for 94.2% of the assembly burdens, collectively. Individually, these impact proportions were 49.8% for the 15 kL tanks and 44.4% for the 20 kL tanks. Other component impacts were minor for this category.

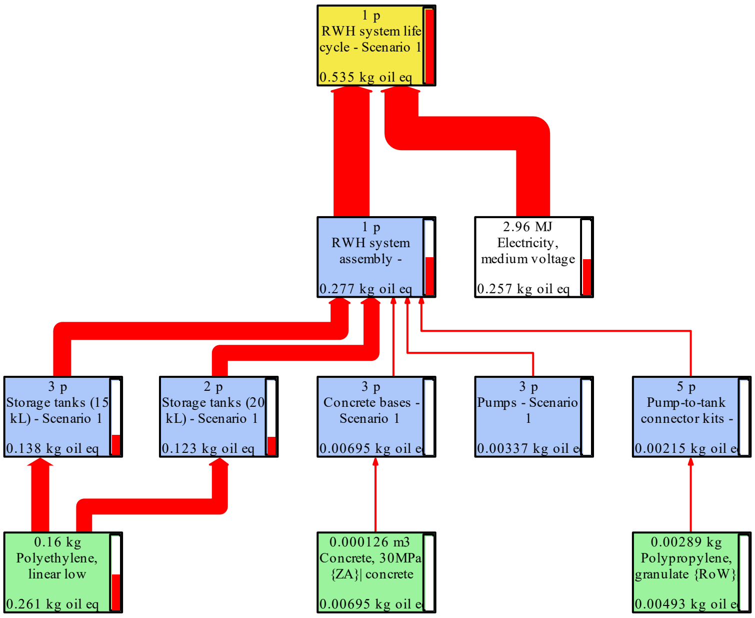

Figure 5-13: Fossil resource impacts from assembly and operation (Scenario 1)

5.7.3.2. Scenario 2 Impacts

For Scenario 2, the mineral resource scarcity impacts (kg Cu eq) were first represented in a network diagram in **Figure 5-14** followed by the fossil resource scarcity impacts (kg oil eq) in **Figure 5-15**. The findings displayed the burdens per kL of rainwater harvested under Scenario 2.

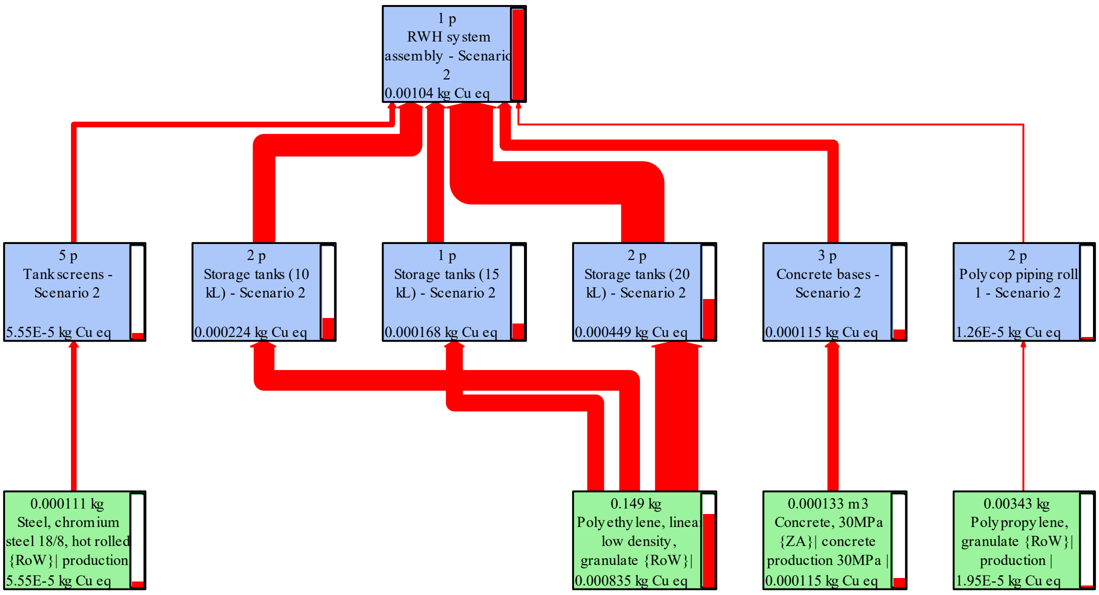

Figure 5-14: Mineral resource burdens per kL harvested rainwater (Scenario 2 assembly)

The components contributing towards high levels of mineral resource depletion were the two 20 kL storage tanks (43.2% of the total burdens), the two 10 kL storage tanks (21.5% of the amassed effects), and the single 15 kL tank (16.2% of the overall impacts). Hence, the five storage tanks collectively contributed towards 80.9% of the mineral resource impacts. This outcome was expected since higher environmental burdens were linked to components with greater masses in the previous section of the research (**Section 5.6**). However, the relatively light tank screens also produced prominent impacts in this category (5.3% of the total burdens). Other notable burdens came from the three concrete bases having impacts of 0.000115 kg Cu eq (11.1% of the total effects).

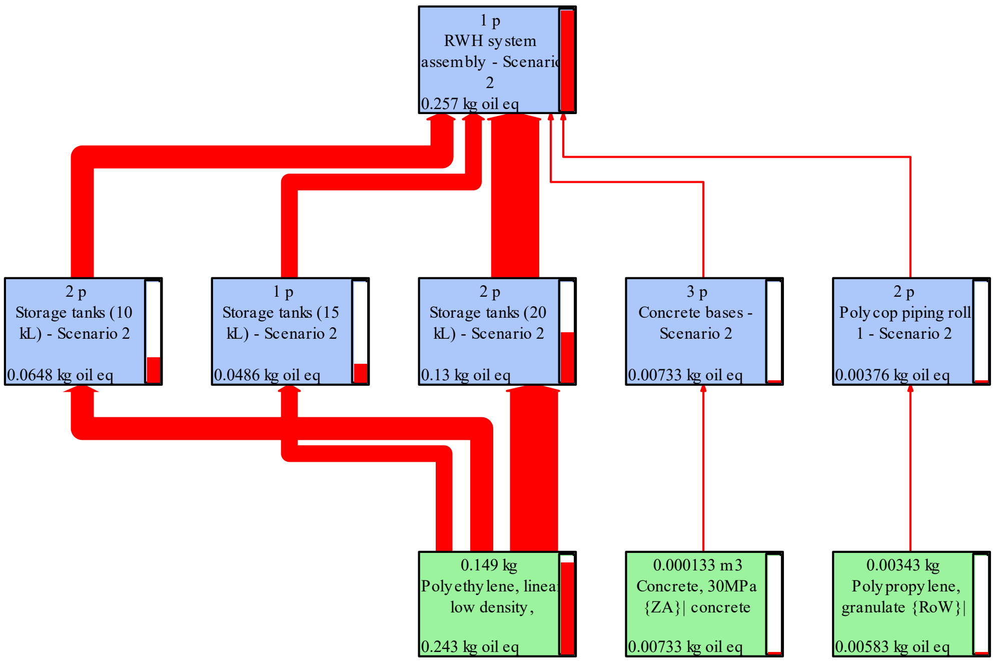

Figure 5-15: Fossil resource burdens per kL harvested rainwater (Scenario 2 assembly)

From **Figure 5-15**, the total fossil resource burdens were estimated at 0.257 kg oil eq per kL of harvested rainwater. The majority of these impacts originated from the two 20 kL storage tank assemblies (LLDPE production), which accounted for approximately half (50.6%) of the total ecological burdens. The two 10 kL tanks were the next major impact contributors, generating 0.0648 kg oil eq (or 25.2% of the fossil resource burdens). The single 15 kL tank was then shown to have 18.9% of the environmental effects. Minor contributions from the three concrete bases (2.9% of the burdens) and the two 22mm x 50m Polycop piping rolls (1.5% of the effects) were also included in the network diagram. The rest of the components were comparatively insignificant; hence, were not displayed.

5.7.4. Water Consumption Burdens

The water consumption impacts presented a unique perspective in the RWH context as the burdens were measured in m^3, which was the same as the functional unit (kL). The overarching concept for this impact category was that water would be consumed in order to construct/assemble the RWH components and operate the pumps, which, in turn, would allow for water-saving benefits. Hence, the viability of the RWH systems would be contingent on the ratio of the water consumption impacts per kL of harvested rainwater.

5.7.4.1. Scenario 1 Impacts

The water consumed (per kL of harvested rainwater) for the assembly and operation of the first RWH system was presented in a single network diagram (**Figure 5-16**).

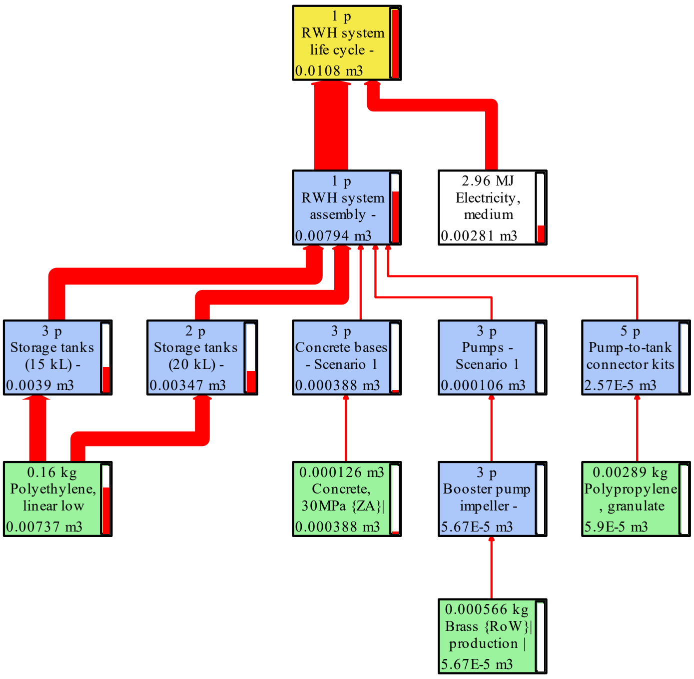

Figure 5-16: Water consumption from assembly and operation (Scenario 1)

For Scenario 1, the water consumed for component assemblies (0.00794 m^3) outweighed the water consumed for the lifetime pump operation (0.00281 m^3) for every kL of rainwater reaped. Hence, 73.5% of the water consumption burdens were due to the construction of the various RWH components. Thus, it was noted that almost three times more water would be needed to construct the system components than that for lifetime electricity production. Regarding the system components themselves, the storage tanks were found to be responsible for most of the ecological burdens with the three 15 kL tanks generating 49.1% of the assembly effects and the two 20 kL tanks having 43.7% of the assembly impacts. In terms of the total impacts (assembly and operation), tank or LLDPE production would collectively consume 68.2% of the water required to build and operate this system.

5.7.4.2. Scenario 2 Impacts

The water required for Scenario 2 implementation is shown in **Figure 5-17**.

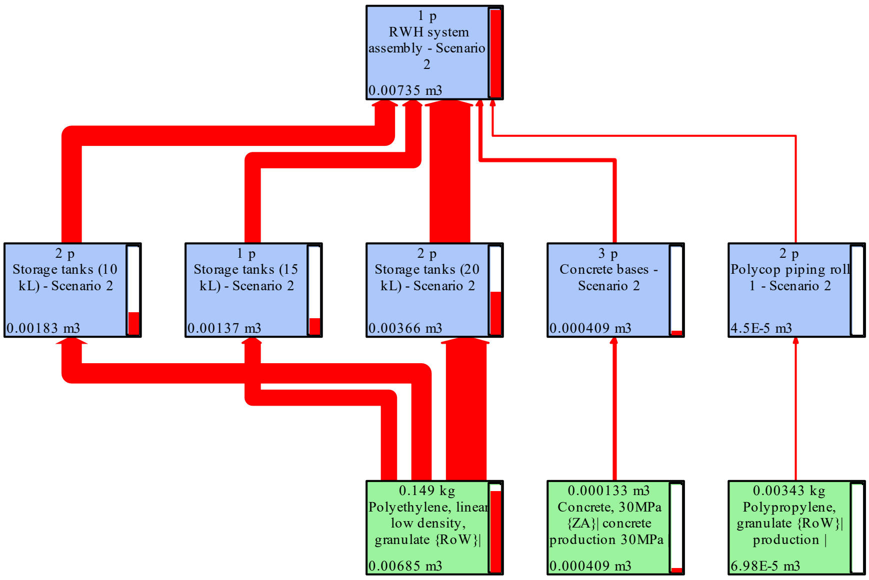

Figure 5-17: Water consumption per kL harvested rainwater (Scenario 2 assembly)

The total water needed to assemble the gravity-fed system was found to be 0.00735 m^3 per kL of collected rainwater. As this value was less than 1 m^3, the system was deemed practically viable since more water would be saved by the system than the amount needed for its implementation. A similar finding occurred for the first scenario; however, the consumption values for the two scenarios were not compared. From **Figure 5-17**, it is evident that the storage tanks contribute to the more significant impacts. In particular, the two 20 kL tanks (having the greatest masses) account for 0.00366 m^3 of water usage for LLDPE manufacturing. This value translates to almost half (49.8%) of the total assembly impacts for this category. The two 10 kL tanks (24.9% of the impacts) and the one 15 kL tank (18.6% of the burdens) also indicate relatively larger water consumption for LLDPE production. The other components like the concrete bases, Polycop piping rolls, and pre-filtration mechanisms had combined impacts of 6.7% of the total burdens. A summary of the total impacts is documented in the next section, highlighting the assembly and operational system burdens in relation to the case scenario and the 18 impact categories.

5.7.5. Summary of the LCA Findings

Summaries of the LCA results are shown in **Tables 5-33** (Scenario 1) and **5-34** (Scenario 2) according to each impact category and total system burden (per kL of harvested rainwater). The element contributing towards the largest impact proportion is also displayed.

Table 5-33: Scenario 1 Environmental Impacts

Impact category	System impact per kL of rainwater harvested (assembly phase)	System impact per kL of rainwater harvested (operational phase)	Highest individual contributor (impact proportion)
1. Global warming	0.435 kg CO_2 eq	0.875 kg CO_2 eq	Pump energy (66.8%)
2. Stratospheric ozone depletion	1.09×10^{-7} kg CFC-11 eq	6.9×10^{-7} kg CFC-11 eq	Pump energy (86.3%)
3. Ionizing radiation	0.00765 kBq Co-60	0.0396 kBq Co-60	Pump energy (83.8%)
4. Ozone formation – human health	0.00094 kg NO_x eq	0.00385 kg NO_x eq	Pump energy (80.4%)
5. Fine particulate matter formation	0.00055 kg PM2.5 eq	0.00251 kg PM2.5 eq	Pump energy (82.0%)
6. Ozone formation – terrestrial ecosystems	0.000997 kg NO_x eq	0.00386 kg NO_x eq	Pump energy (79.5%)
7. Terrestrial acidification	0.00136 kg SO_2 eq	0.00834 kg SO_2 eq	Pump energy (85.9%)
8. Freshwater eutrophication	0.000117 kg P eq	0.000633 kg P eq	Pump energy (84.4%)
9. Marine eutrophication	7.19×10^{-6} kg N eq	3.84×10^{-5} kg N eq	Pump energy (84.2%)
10. Terrestrial ecotoxicity	2.2 kg 1,4-DCB	0.92 kg 1,4-DCB	Pump construction (45.8%)
11. Freshwater ecotoxicity*	0.0368 kg 1,4-DCB	0.0261 kg 1,4-DCB	Pump energy (41.5%)
12. Marine ecotoxicity*	0.0471 kg 1,4-DCB	0.0356 kg 1,4-DCB	Pump energy (43.0%)
13. Human carcinogenic toxicity	0.023 kg 1,4-DCB	0.0599 kg 1,4-DCB	Pump energy (72.2%)
14. Human non-carcinogenic toxicity	0.561 kg 1,4-DCB	1.04 kg 1,4-DCB	Pump energy (64.9%)
15. Land use	0.00445 m^2a crop eq	0.009 m^2a crop eq	Pump energy (66.9%)
16. Mineral resource scarcity	0.0032 kg Cu eq	0.000352 kg Cu eq	Pump construction (59.4%)
17. Fossil resource scarcity*	0.277 kg oil eq	0.257 kg oil eq	Pump energy (48.1%)
18. Water consumption	0.00794 m^3	0.00281 m^3	3 x 15 kL storage tanks (36.1%)

Table 5-34: Scenario 2 Environmental Impacts

Impact category	System impact per kL of rainwater harvested (assembly phase)	Highest contributor (impact proportion)
1. Global warming	0.398 kg CO_2 eq	2 x 20 kL storage tanks (46.3%)
2. Stratospheric ozone depletion	9.49 x 10^{-8} kg CFC11	2 x 20 kL storage tanks (45.7%)
3. Ionizing radiation	0.00671 kBq Co-60	2 x 20 kL storage tanks (46.9%)
4. Ozone formation – human health	0.000837 kg NO_x eq	2 x 20 kL storage tanks (44.7%)
5. Fine particulate matter formation	0.000436 kg PM2.5 eq	2 x 20 kL storage tanks (46.3%)
6. Ozone formation – terrestrial ecosystems	0.000888 kg NO_x eq	2 x 20 kL storage tanks (45.0%)
7. Terrestrial acidification	0.00109 kg SO_2 eq	2 x 20 kL storage tanks (45.7%)
8. Freshwater eutrophication	8.2 x 10^{-5} kg P eq	2 x 20 kL storage tanks (42.9%)
9. Marine eutrophication	5.89 x 10^{-6} kg N eq	2 x 20 kL storage tanks (43.8%)
10. Terrestrial ecotoxicity	0.728 kg 1,4-DCB	2 x 20 kL storage tanks (43.3%)
11. Freshwater ecotoxicity	0.0126 kg 1,4-DCB	2 x 20 kL storage tanks (46.9%)
12. Marine ecotoxicity	0.0163 kg 1,4-DCB	2 x 20 kL storage tanks (47.1%)
13. Human carcinogenic toxicity	0.0111 kg 1,4-DCB	2 x 20 kL storage tanks (44.9%)
14. Human non-carcinogenic toxicity	0.239 kg 1,4-DCB	2 x 20 kL storage tanks (45.6%)
15. Land use	0.00386 m^2a crop eq	2 x 20 kL storage tanks (36.0%)
16. Mineral resource scarcity	0.00104 kg Cu eq	2 x 20 kL storage tanks (43.2%)
17. Fossil resource scarcity	0.257 kg oil eq	2 x 20 kL storage tanks (50.4%)
18. Water consumption	0.00735 m^3	2 x 20 kL storage tanks (49.7%)

Network diagrams for impact categories not shown previously are displayed in **Appendices F-1** to **F-14** under appropriate headings.

The 18 impact categories listed in **Table 5-33** and **Table 5-34** were numbered for ease of reference. The corresponding burdens from the assembly and operational phases were also displayed in adjoining columns. The final column indicated the part of the RWH system generating the highest proportion of the burdens. For Scenario 1 (**Table 5-33**), the findings generally suggested that the electrical energy usage by the system pumps accounted for the largest impact proportions (across most LCA categories). These percentages ranged from 41.5% of the total burdens (freshwater ecotoxicity) to 86.3% (stratospheric ozone depletion). It was further noted that where the pump energy impacts were predominant, the operational burdens outweighed the total and individual assembly effects. This result can be seen for impact categories 1-9 and 13-15 (12 out of 18 groups).

Although greater operational burdens frequently correlated to electricity garnering the higher impact proportions, there were a few exceptions (denoted by *) where the larger assembly impacts were still surpassed by the pump usage effects. When analyzing freshwater and marine ecotoxicity (categories 11 and 12) and fossil resource scarcity (impact category 17), the total assembly effects were higher than operational burdens yet energy usage was still found to be the highest impact contributor. This was due to the pump electricity having greater ecological impacts than any individual component assembly. Visual aids depicting this outcome are shown in **Appendices F-10** and **F-11**, and **Figure 5-13**.

The findings for impact category numbers 10 (terrestrial ecotoxicity), 16 (mineral resource scarcity), and 18 (water consumption) revealed that the total assembly impacts outweighed the electrical operational burdens of the RWH system. Furthermore, the production/manufacturing of certain individual components had the highest impact proportions. For the terrestrial ecotoxicity grouping, pump construction was found to generate 45.8% of the total burdens while the energy usage accounted for 29.5% of impacts. In terms of mineral resource scarcity, pump manufacture was at 59.4%, and electricity usage was 9.9% of the total effects. As the pumps were made of cast iron (body), stainless steel (shaft), and brass (impeller), the greatest burdens stemmed from brass production (see **Appendix F-9** and **Figure 5-12**). For the final impact category (water consumption), neither pump construction nor usage had the greatest burdens. Rather the assembly/production of the three 15 kL storage generated the majority of the environmental concerns (36.1%) followed by the two 20 kL tanks (32.1%) and pump usage (26%). (See **Figure 5-16**).

The ecological burdens for Scenario 2 (**Table 5-34**) summarized the total system impacts per kL of harvested rainwater much like for the first scenario. However, unlike Scenario 1, the second RWH system did not have operational burdens from electrical pump usage; hence, only the total assembly impacts were present. It is evident from **Table 5-34** that the two 20 kL storage tanks accounted for the highest individual burdens for all 18 impact categories ranging from 36% (in land use) to 50.4% of the total effects (for fossil resource scarcity). In other words, the two 20 kL tanks made up (at minimum) over a third and at most, slightly over half the total environmental concerns for Scenario 2. The other components made up the remaining ecological burdens based on their masses and material type.

5.8. Comparison of the results with other studies

Once more, it was noted that RWH implementation at schools was an uncommon focal point of previous studies (in both local and global contexts), thus necessitating the need for this research. However, outcomes like the design, performance, and viability of the proposed RWH systems needed to be analyzed in relation to previous studies. Therefore, the findings uncovered in this research were compared to other papers of a <u>similar</u> scope to verify/assess the likenesses and differences of the results. However, not all findings could be compared due to distinctive viewpoints presented in this study.

Energy-intensive RWH systems were found to be a pivotal point for optimization in several studies. Recalling the research by Ward et al. (2012) previously discussed in the literature review, their study entailed RWH implementation for toilet flushing at a university office block in the UK. A significant outcome of their paper showed pump energy consumption at 0.54 kWh per m^3 of harvested rainwater. For this research, the pump energy consumption was estimated at 392 kWh/annum for water collections of 476.76 kL/year (**Section 5.5.2**). Hence, for every kL (m^3) of rainwater harvested, the pump energy consumption was calculated as 0.82 kWh/m^3. This value is 0.28 kWh/m^3 higher than that of Ward et al. (2012). The slight difference was attributed to catchment area size [1500 m^2 (Ward et al., 2012) vs. 1091 m^2 (this study)] and rainfall conditions that could have yielded higher harvestings and, therefore, lower ratios. Another reason for the difference may be due to pump output characteristics having dissimilar electricity usages.

Regarding the CO_2 emissions, a similar situation to the energy outcomes occurred. In this study, the pumping components of the RWH system (Scenario 1) generated a total carbon footprint of 375.14 kg CO_2 (**Section 5.6.2**). The system harvestings were 476.76 kL per annum as before, with the pump producing 0.79 kg CO_2 for every m^3 of rainwater collected and used. Ward et al. (2012) estimated their CO_2 emission ratio as 0.56 kg CO_2 eq per m^3 of harvested rainwater. As before, the difference (0.23 kg CO_2) was likely due to pump characteristics and site-specific conditions that are highly influential on the results. A final remark on this study involved the pump electricity usage in relation to the normal energy usage. For the current research, it was discovered that the presence of the pumps would increase the municipal energy consumption by 2.3% given that the average annual energy usage is 16 812.55 kWh and yearly pump usage is 392 kWh (**Section 5.5.2**). Hence, the fraction of energy represented by pump usage would still amount to approximately 2.3% of the total energy usage by the entire school (all blocks). Ward et al. (2012) found that the pump represented only 0.07% of the building energy consumption. However, their analysis considered a single building, whereas this research included the entire school; hence, the significant difference in percentages. In any case, it was found RWH pumps would not significantly increase the total energy consumption of the selected area of study.

In another paper by Vieira et al. (2014), the energy intensities of many RWH studies were reviewed and collated. The research found that system energy intensities ranged from 0.20 to 1.40 kWh per m^3 of harvested rainwater. Recalling that the system energy intensity for this study was 0.82 kWh/m^3, the numerical value is approximately halfway within that range, thereby offering some validation of the results. Vieira et al. (2014) go on to suggest that gravity-fed systems could potentially provide full water supplies in appropriate water-activity contexts. As the present research focuses on toilet flushing alone, a relation can be made for the system being in an appropriate water-activity context. However, the designed gravity-fed system would only partially contribute to water usage at Duffs Road Primary, for several reasons stated previously.

In correspondence to WSE and payback periods, a paper undertaken by Stec and Zeleňáková (2019) was examined. Their research provided insight into two toilet flushing RWH systems implemented at different academic facilities in Poland and Slovakia. It was shown by the authors that a favourable WSE (of 29%) was present in the Slovakian facility. Conversely, an 18% WSE was estimated for the Polish region, where it was also deemed unfeasible. For this study, WSE was at 33.5% for the Scenario 1 system and 31.8% for Scenario 2 (**Section 5.4.3**). Furthermore, the proposed systems would have payback periods of 5.93 years (Scenario 1) and 4.85 years (Scenario 2) as opposed to return periods of over 30 years in the study by Stec and Zeleňáková (2019). Hence, this research allowed for more economically feasible systems having improved water-saving efficiencies (WSEs) and lower payback periods. However, specific WSE percentages are not set factors for system viability as suggested by Juliana et al. (2019). For their research, the WSE of an RWH system in a public facility in Palembang (Indonesia) was also calculated as 29%. However, the authors deemed this ratio inviable in the context of their study as opposed to Stec and Zeleňáková (2019). Hence, feasible WSEs (like many RWH considerations) are case-dependent and subjective from the authors' perspectives.

Further analysis of the research by Juliana et al. (2019) showed another RWH system (implemented at a different public facility) having a return period of 7.5 years, with a tank capacity of 1050 litres. When comparing this to the current research, it can be seen that much higher tank capacities (85 000 litres in Scenario 1 and 75 000 litres in Scenario 2) yielded only slightly lower returns periods of 5.93 years (Scenario 1) and 4.85 years (Scenario 2). Recalling that payback periods are dependent on system costs and effective harvestings, this outcome was likely due to those differences. Where Juliana et al. (2019) estimated their RWH system WSE at almost 60% for this second facility, the WSEs for the two systems in this research were just over half of that (33.5% and 31.8%). Overall, the return periods for the two studies were deemed feasible.

The viability of an RWH system was also assessed in a study by Marteleira and Niza (2018). For their research, a proposed RWH system was tested using a holistic feasibility assessment tool (RaINvesT) at a university campus in Portugal, with findings relevant to payback periods and energy intensities being the chief outcomes. Marteleira and Niza (2018) were able to show the economic feasibility of the RWH system since it had a low return period (12 years). Furthermore, the energy produced from their system operation was estimated at 0.013 kWh per cubic metre of harvested rainwater. This value is even lower than the range uncovered by Vieira et al. (2014) (0.2 – 1.4 kWh/m^3) and the value obtained for this research (0.82 kWh/m^3), suggesting a very highly energy-efficient system. The return period of 12 years was also accepted as viable by the authors, thereby validating the lower return periods of this research (5.93 years and 4.85 years for Scenarios 1 and 2, respectively).

For the LCA part of this research, a comparison was made with two separate studies by the same leading author. Ghimire et al. (2017) assessed the environmental impacts (cradle-to-grave) of a four-storey commercial RWH system in relation to the usual municipal supply in the building. The main water activity considered was toilet flushing with general findings suggesting lower impacts from RWH as opposed to municipal water provision. For the current research, the existing municipal supply was not a part of the LCA, although carbon emission reductions were assessed for the two scenarios (**Section 5.4.3**). For the RWH system specifically, Ghimire et al. (2017) highlighted pump construction and pump energy usage as having the greatest burdens across most impact categories, with the exceptions of ozone depletion, freshwater withdrawal, and eutrophication groupings. A similar outcome was present for the current research (Scenario 1), which showed pumping energy as the highest contributor for 15 out of 18 impact categories. The remaining impact groups (terrestrial ecotoxicity, mineral resource scarcity, and water consumption) did not have the dominant impacts emanating from pump energy usage. Hence, for these two studies, the only category showing fewer burdens from pump operation was water consumption/freshwater withdrawal.

The second paper (Ghimire et al., 2014) analyzed the impacts of domestic and agricultural RWH systems for Back Creek Watershed in Virginia, USA. Similar to the findings by Ghimire et al. (2017) and the current research, electricity from pumping brought the highest impacts for both RWH systems. However, certain components like the storage tank, piping, and the pump itself were responsible for significant burdens in certain categories, particularly for the domestic RWH system. This finding resembles the outcomes from Scenario 2 (pump energy-free), which indicated the 20 kL storage tanks having the highest burdens for all impact categories.

5.9. General RWH implementation at schools

The methodology and findings of this study presented an overview of the viability of RWH implementation at Duffs Road Primary School. However, these considerations for RWH could be extended to other schools through a model or flow chart process. Important aspects of RWH implementation included the potential harvestings, municipal cost savings, municipal carbon emission reductions, tank sizing, economic viability, energy usage, and environmental impacts.

Figure 5-18 shows the generalized considerations for adopting RWH systems at schools in the eThekwini Municipality.

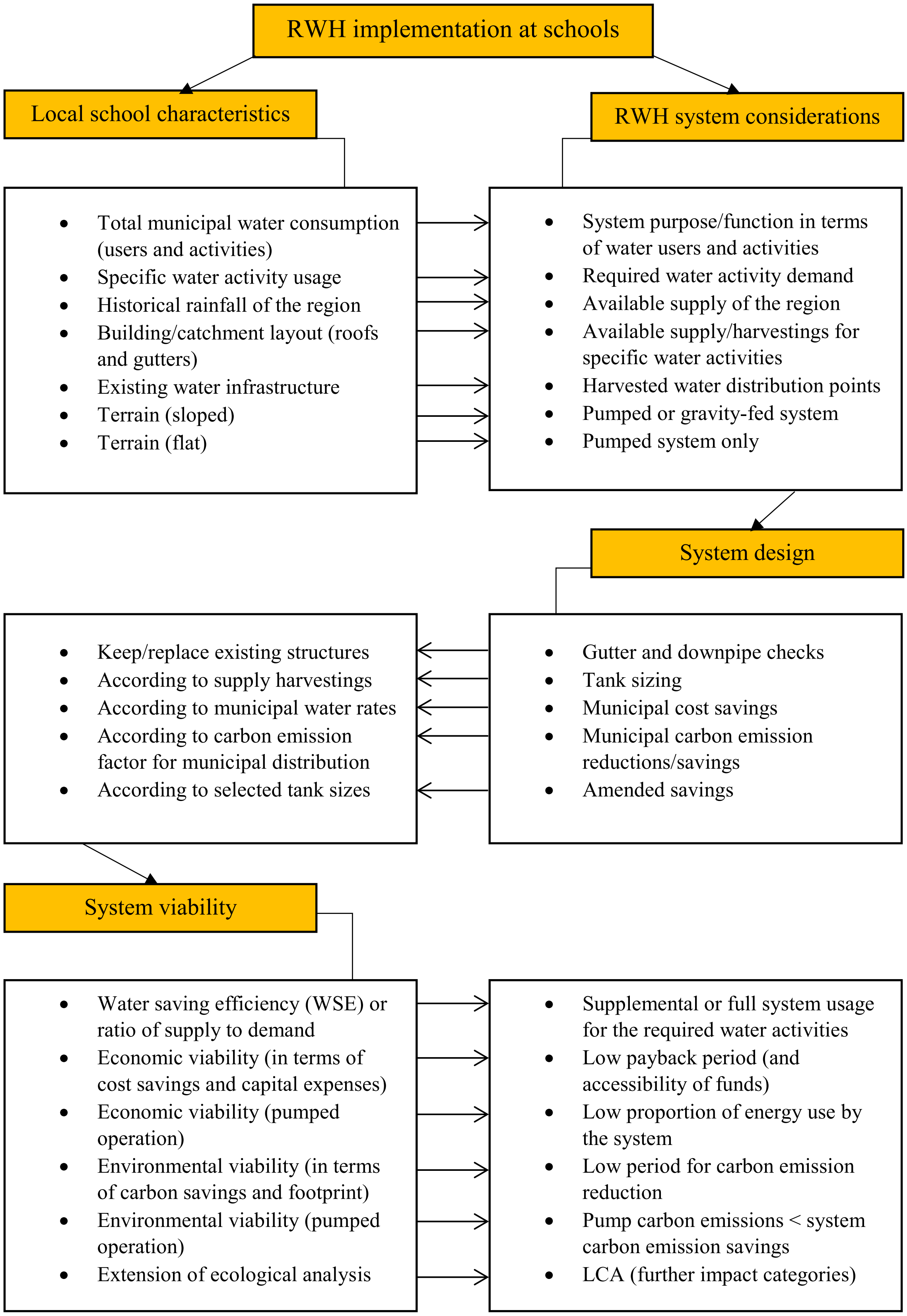

Figure 5-18: Generalized considerations for RWH implementation at schools

Discussion of the flow chart process

From **Figure 5-18**, it can be seen that the local school characteristics and RWH system considerations are intrinsically linked as shown by the diagrammatic arrows. The total municipal water consumption at any school (due to its water users and activities) plays a significant role in how the RWH system would function. For this research, the RWH systems were configured to supply water for toilet flushing at Block C for Grades 1 to 7. However, such systems may not necessarily be limited to one water activity or supply target or even a particular demographic in other contexts/schools. Hence, in identifying the specific water usage per activity, the required rainwater (demand) per activity can be gauged, further outlining the system purpose.

Historical rainfall data was also shown to be a vital local characteristic for RWH implementation at schools. Such systems are primarily dependent on rainwater and historical data (along with catchment sizes) provide an estimation of the likely supply in the school region. **Figure 5-18** also highlights existing water infrastructure as an important local consideration as it indicates where the harvested water would be needed e.g., at a toilet facility. Finally, the site topography factor would be required to identify if a pumped or gravity-fed system would be applicable. Although flat terrains would not be able to initiate gravity flows unless higher tank platforms are utilized, arguments for aesthetic appeal may detract from using this option. Schools located in hilly regions (like this case study) have the option for either pumped or gravity-fed systems. Thus, system design considerations then come to the forefront.

In **Figure 5-18**, the major design considerations pertained to the gutters, downpipes, and tank sizes. By performing gutter and downpipe checks, one can gauge whether these components can be incorporated into the RWH system or if they need replacement. Tank sizing considerations usually account for the largest potential collections, thereby maximizing (or amending) cost savings and carbon emission reductions. It was noted in **Figure 5-18** that the cost savings and carbon emission savings were contingent on municipal water rates and carbon emission factors, respectively.

System viability was also a major point under general considerations of school RWH systems. By estimating the ratio of rainwater supply to demand, school RWH systems could potentially replace municipal supply for its intended purpose (water activity). Even if this situation were not possible, RWH systems could still supplement centralized supply and provide savings benefits and reduced municipal strains. Higher cost savings would also assist in lowering the system payback periods, providing further arguments for RWH implementation. Since lower repayment periods would indicate a quick return on the initial/capital investment, economic feasibility would be present. With regards to pumped systems only, economic viability could also be assessed by identifying if the RWH system (and pump specifically) would significantly increase the energy consumption at a school. This idea is highlighted in **Figure 5-18**, and also adopted in this study (**Section 5.5.2**).

The environmental feasibility of general RWH systems was also a key aspect in **Figure 5-18**. Similar to the economic considerations, ecological system viability was linked to the period in which carbon emissions could be reduced. Systems that can reduce their carbon footprints (which come from component production) in short spaces of time would be the optimized solution. However, pumped systems would still generate environmental burdens through electrical operation. Hence, to determine if a pumped system is viable, it is important to relate the system carbon savings to the carbon emissions generated.

This concept can theoretically be applied across many environmental (impact) categories; however, in this research, only the global warming burdens were analyzed to this degree initially. Thus, LCAs were shown to provide a further context of environmental burdens. In terms of RWH, the LCAs indicated the environmental burdens amounting from the construction of the components and pump operation. For this research, it was found that higher burdens came from electricity use rather than component manufacturing for most impact categories. Hence, gravity-fed systems would be the optimal RWH system implemented at schools, should the local site characteristics permit it.

Research findings in the wider context

Recalling **Section 2.5.3.3** of the literature review, context specificity plays an influential role in the benefits of RWH systems. Even from **Figure 5-18**, it is evident that many local characteristics like historical rainfall patterns, catchment sizes, site topography, and water consumption influence how RWH systems are implemented at educational institutes. For example, schools may be located in regions of high rainfall; however, they might have few catchments that limit the rainwater collection. Conversely, larger schools having more buildings and catchment areas may be located in regions of lower rainfall frequency. Thus, the effectiveness of such systems is dependent on several conditions, which in turn, dictate how RWH systems should be implemented.

Nonetheless, the findings of this research were upscaled to a proportion (25%) of schools (in the eThekwini Municipality) that were assumed to have similar characteristics to the selected case study. It was also assumed that these schools (primary and secondary) had access to basic water and sanitation services by the eThekwini Municipality. In doing so, the RWH benefits could be viewed in a wider context and serve as a conservative measure against water challenges (like depletion, scarcity, and municipal supply strains). As this research focused on optimizing RWH systems in the eThekwini Municipality, the remaining provincial and national schools were not considered in the broader scope. This was largely due to the spatial and temporal rainfall variability in the country (see **Section 2.2.1**).

At a national level, RWH systems implemented at Duffs Road Primary may be ineffective at schools in the western parts of the country due to rainfall variability. Even within KZN there exists spatial rainfall variability to some degree (see **Figure 2-1)**. Furthermore, a proportion of schools were already assumed to have comparable characteristics to Duffs Road Primary in order to implement RWH systems that would generate similar benefits. Hence, for these reasons, it was deemed sufficient to consider schools within the eThekwini Municipality only.

It was found that two major school districts (Umlazi and Pinetown) were under the jurisdiction of the eThekwini Municipality in terms of municipal water supply (KZN Department of Education, 2021; Municipalities of South Africa, 2021). Unfortunately, further information regarding the number of schools per district was unavailable from the KZN Department of Education website. Hence, an alternate online website was used. For the Umlazi District, the school total was found to be 590, 506 (or 86%) of which were public schools (Our Schools.Social, 2021). Similarly, the Pinetown District was shown to have 587 schools in the region with a higher proportion of public schools (535 or 91%) as per the online source (Our Schools.Social, 2021). In any case, it was noted that the RWH systems could be implemented at both public and private schools.

Recalling the findings of this study, physical water savings of the pumped system amounted to 476.76 kL/annum, which was equivalent to R28 737.45 in cost savings and 195.04 kg of CO_2 reductions, every year. Thus, should similar yearly savings be present for a quarter of the municipal schools (approximately 294), then the total savings within the eThekwini Municipality could be as much as **140 285.25 kL/year**, garnering close to

R8.46 million in annual municipal cost savings. The upscaled carbon emission savings would be around **57 391 kg CO_2/annum**; however, these advantages are likely to be overshadowed by the yearly pump operating emissions. Hence, gravity-fed systems would be the optimal solution and (assuming that the findings could still be applied to the same proportion of schools) would allow for savings benefits and viability in the ecological and economic settings. More specifically, the upscaled benefits of the gravity-fed system were estimated at **133 142.42 kL** in water savings, **R8.20 million** in cost savings, and **54 469 kg** in CO_2 reductions every year.

Given that the eThekwini Municipality already has water provision challenges due to migration, population growth, and urbanization (see **Section 2.2.4.7**), the execution of RWH systems at schools would provide ample benefits to counteract these effects. Physical water savings of **130** to **140 000 kL** may alleviate the strains of municipal supply and allow for better water security in the province (and country) overall. According to Umgeni Water (2021), which supplies bulk water to eThekwini (Sutherland et al., 2014); 472 million m^3 of water is provided to 6.7 million people every year. By proportioning this figure to the eThekwini population (3.99 million people), it can be approximated that the municipality provides 281.1 million kL of water per annum. Hence, the proportion of water saved/reduced could be as much as **0.1%** if RWH systems were implemented at schools in the eThekwini Municipality. This is a substantial reduction in municipal supply given that water is used for many activities beyond toilet flushing in schooling contexts.

Cost savings between **R8.20 and R8.46 million every year** may also provide the incentive and budgetary input by municipal/governmental authorities for system implementation. Furthermore, these outcomes were only upscaled for toilet-flushing systems but could be extended to other water activities. However, it must be noted that higher levels of disinfection would be required for more potable applications and uses, whereas harvested rainwater used for toilet flushing requires minimal (if any) treatment.

Therefore, further studies involving school-implemented RWH systems must be undertaken. In this way, the benefits of harvested rainwater can be seen beyond toilet-flushing contexts. Furthermore, RWH may not just be beneficial but crucial in certain schools that lack proper water and sanitation services. By increasing hygiene and sanitation, factors like school attendance, academic performance, and general quality of life may fare better. However, having the necessary infrastructure to collect and store water on-site may not be financially possible for some schools. Thus, governmental/municipal subsidies are encouraged for these situations, as the aforementioned benefits would be gained by the Department of Education and the eThekwini Municipality. Not only have the outcomes of this study shown RWH systems to be ecologically and economically feasible (in terms of payback periods); they may also avert the current water scarcity crisis in the country. Ultimately, schools that have the potential to save more (based on their topography, catchment sizes, rainfall, and demand) should be prioritized along with disadvantaged schools in order to provide optimal outcomes.

5.10. Summary

The findings presented in this chapter were obtained according to the methods and procedures described in **Chapter 4**. All the calculated research outcomes were represented using tables and figures and discussed to show their significance. Firstly, the case study data involving municipal water and electricity consumption at the school, historical rainfall of the area, and toilet flushing demand (at Block C) were presented. Following these sets of results were the potential benefits that could be introduced through system implementation (for both scenarios). Design considerations like conveyance checks and tank sizing were also performed.

The viability of the proposed systems was then conducted through economic and environmental analyses. Important concepts included the capital expenses, pump operating costs, payback periods, carbon footprints, pump carbon emissions, and carbon emission reduction periods. Pen-ultimately, the environmental analysis was expanded through the LCA, which identified component assembly and pump operation burdens across 18 impact categories. Finally, using the benefits of RWH implementation at Duffs Road Primary, the results were roughly upscaled across many schools in the eThekwini Municipality to gauge their implication on a broader scale. Ultimately, all the findings were able to answer the research questions and satisfy the aim and objectives of the study. A summary of the sections and objectives they fulfilled is shown in **Table 5-35**.

Table 5-35: Sections that fulfil the research objectives

Section	Objective satisfied
Section 5.2	1
Section 5.3	2 and 3
Section 5.4	1, 2, and 3
Section 5.5	3
Section 5.6	3
Section 5.7	4 and 5

CHAPTER 6: CONCLUSION

6.1. Concluding Remarks

6.1.1. System Design and Performance

The research presented in the preceding chapters assessed the feasibility of RWH systems at Duffs Road Primary School (the case study). The outcomes were obtained according to the stipulated aims and objectives of the first chapter (**Section 1.4**), while the entire case study (including the school layout) was documented in **Chapter 3**. Two types of RWH systems were proposed, the first being a pumped system utilizing the catchments of Blocks A, B, and C (Scenario 1). The second system (Scenario 2) was configured to collect rainwater from Blocks E, H, and I and, due to the terrain and building elevations, allowed for a gravity-fed system. In both cases, the RWH systems aimed to supply water for toilet flushing at Block C (the ablutions building) only. Hence, a survey was conducted to estimate the yearly water consumption at this location (**1424.29 kL/annum**).

Based on the historical rainfall data, catchment areas, selected configurations, and runoff coefficients, the total water harvesting potential (supply) for Scenario 1 initially amounted to **478.97 kL/year**. A slightly lower figure of **461.52 kL/annum** was estimated for Scenario 2's system. Once these values were calculated, several standardized tanks were selected to match the maximum monthly supply. For Scenario 1, **two 20 kL** and **three 15 kL** tanks were chosen, thereby allowing for adjusted water harvestings/savings of **476.76 kL per year**. The modified savings for the second scenario were estimated at **452.48 kL per annum** due to the selection of the **two 10 kL** tanks, the **two 20 kL** tanks, and the **single 15 kL** tank. Consequently, the annual municipal water cost savings (according to the eThekwini rates) amounted to **R28 737.45** and **R27 850.94** for Scenarios 1 and 2, respectively. As the two RWH systems would also lower carbon emissions usually produced from the municipal water supply, the yearly carbon emission savings were calculated as **195.04 kg CO_2** for the pumped system and **185.11 kg CO_2** for the gravity-fed system.

The viability of the two systems was then assessed in the economic and environmental settings. The WSE of the pumped system was first estimated at **33.5%**, while that of the second system was **31.8%**. While these proportions may seem low (accounting for approximately a third of the demand), both systems were deemed viable due to the significant municipal cost savings discussed beforehand. Furthermore, previous studies have suggested lower WSEs as still being feasible, especially if widespread implementation were to occur. The reduced burdens on municipal supply and the likelihood of increased municipal rates over time provided added benefits for the proposed RWH systems.

Overall, the pumped RWH system showed the advantage of higher water, cost, and carbon emission savings. For the period of carbon emission reduction, values of **11.22 years** and **10.34 years** were obtained for Scenarios 1 and 2, respectively. These estimations were deemed as satisfactory timeframes in which the system carbon footprints would be reduced, thereby providing a level of ecological feasibility for both systems. However, for Scenario 1 the significant tradeoff was increased carbon emissions generated from pump usage (**375.14 kg CO_2/annum)** that would counteract and exceed the potential carbon emission savings by **180.10 kg CO_2 every year**. Scenario 2, on the other hand, presented slightly lower water, cost and carbon emission savings. Yet, the absence of pumps meant carbon emission savings were not opposed or inhibited like in Scenario 1. Hence, this finding suggested environmental viability for Scenario 2 only.

Economic feasibility was also assessed according to energy costs, system expenses, and payback periods. Although the expenses associated with pump operation fared better than its carbon emission counterpart, RWH pump operation would still increase the current municipal energy costs by **2.4%**. On the other hand, the gravity-fed system would not present any increases to the school energy consumption, suggesting an improved RWH system. Additionally, pump operation expenses were estimated to be around **R806.03 a year**. Comparing this to the yearly water savings of **R28 737.45** further demonstrates the economic feasibility of the pumped system. In terms of the repayment periods, it was found that the pumped system would require a period of **5.93 years** before having effective municipal cost savings. Similarly, the second scenario's payback period was estimated at **4.85 years**. This slightly lower value was due to the gravity system having fewer components and, thus, a lower implementation/capital cost. Thus, it was concluded that the gravity-fed system would be the optimized choice for Duffs Road Primary as it provided both economic viability and environmental feasibility in terms of carbon emissions only. The LCA results provided further substantiations for ecological viability beyond the carbon emission/global warming setting.

6.1.2. LCA Outcomes

Irrespective of the benefits like municipal water, cost and carbon emission reductions, the construction of RWH components would always generate environmental burdens. This outcome was expected; however, the research showed what those burdens were in relation to the water-saving benefit in the chosen case study setting (see **Section 5.7**). The SimaPro analysis assessed the environmental burdens of both systems across 18 impact categories (with global warming being one of them).

Firstly, a calculation using the SimaPro software was done to check the pump carbon emissions. Recalling that these emissions were previously calculated as **375.14 kg CO_2/annum**, SimaPro estimated them at **417.16 kg CO_2/year**. The 42 kg difference was attributed to SimaPro using global models for its material data as opposed to South African specific ones. In any case, both values are in a close range of each other, thus offering some corroboration of the 'design and performance' findings discussed in the previous subsection.

It was reiterated that Scenario 1 had the benefit of higher water savings when compared to Scenario 2. However, Scenario 1 was also comprised of more RWH components that were likely to have higher environmental impacts during the assembly and usage phases of the RWH systems. Thus, this outcome presented a tradeoff between environmental performance and water savings for this specific case study. Although greater masses generally indicated more severe environmental burdens, the production of material types like brass (in the pump impellers) were exceptions in some impact categories. For example, the terrestrial ecotoxicity and mineral resource scarcity groupings indicated pump construction as the prevalent source of burdens and yet the pumps as a whole were much lighter than the storage tanks and concrete bases.

Besides mass and material considerations, pumping energy was also a key aspect of the LCA. For Scenario 1, the majority of the environmental impacts were due to electrical pump usage (with the exception of three categories: terrestrial ecotoxicity, mineral resource scarcity, and water consumption). Hence, for the other 15 impact categories, the ecological burdens generated during the operational phase accounted for between **41.5 to 86.3% of the total burdens**. In other words, pump operation would overshadow the effects of manufacturing all the system components. Therefore, it follows that a pumped RWH system would be environmentally inviable in comparison to a gravity-fed system even though these two systems were not directly compared in the LCA. As the second scenario did not have any pumps, the highest burdens were found to come from the manufacturing of the 20 kL storage tanks (**36 to 50.4% of the total burdens**), with impact proportions generally decreasing according to component masses. Thus, it was concluded that while the environmental burdens from the LCA were proportional to component masses (both scenarios), many impacts were mostly dependent on energy usage and significantly reliant on material constituents (Scenario 1).

6.1.3. General RWH implementation at schools

The implementation of RWH systems in a wider context involved the application of this study's findings to a proportion of schools within the eThekwini Municipality also receiving a centralized water supply. Taking into account case-specific factors (like topography, rainfall, catchment areas, water activities, and demand), these conditions were found to be highly influential on the results at Duffs Road Primary. Nonetheless, a quarter of the schools in the eThekwini Municipality were estimated to have these optimal conditions in order to implement RWH systems of similar capacities to those proposed for Duffs Road Primary. Consequently, it was found that the centralized water supply by the eThekwini Municipality could be reduced by **130 to 140 000 kL every year (0.1%)**. If a greater percentage of schools were found to have the ideal conditions for RWH implementation, then the savings would also increase.

This outcome necessitates the need for more research on RWH in schooling environments as municipal supply strains would be reduced and water security improved. From an economic standpoint, municipal cost savings could be as much as **R8.20 to R8.46 million every year**, acting as a catalyst for implementation. As many schools (like Duffs Road Primary) are non-fee-paying institutions, millions in cost savings could be diverted to other needs within the Department of Education. Environmentally speaking, gravity-fed systems would be the optimized choice, as the emissions generated through pumping would be eliminated. Annual carbon emission savings would range from **54 000 to 57 000 kg CO_2**, ushering a drive towards environmental feasibility. However, global warming impacts are not the only environmental burdens; hence, future studies may involve impact category comparisons between municipal supply and RWH provision. Overall, RWH systems may not just serve as a conservative measure but could be indispensable in disadvantaged schools.

6.2. Recommendations and Future Scope

6.2.1. Recommendations

Recommendations were made based on the outcomes of this research and are listed as follows:

- The proposed design of the two types of RWH systems would allow for a WSE of 33.5% in Scenario 1 and 31.8% in Scenario 2. Hence, it is recommended that should RWH systems be implemented at the school, they are used in conjunction with the existing municipal supply in order to cater to the remaining water demand on site.
- Both the pumped and gravity-fed systems were deemed economically viable; however, only the gravity-fed system was ecologically feasible when concerning carbon emissions. Thus, it is recommended that a gravity-fed system be prioritized at Duffs Road Primary.
- The cost, carbon footprint, payback period, and the period before carbon emission reduction of Scenario 2 are less than those of Scenario 1. Therefore, this outcome provides further proof of the optimal RWH system.
- The adoption of a gravity-fed system would not garner any energy usage, operating costs, or operating emissions, unlike the pumped system. Once again, the gravity-fed system is favoured and recommended.
- The LCA indicates environmental burdens across several impact categories. However, a gravity-fed system (absent of pumps) would have lower burdens than a pumped system. Although impacts would be generated from the assembly of system components (like the tanks), it is recommended that more environmentally friendly materials and processes be used to lower the overall impacts.
- Widespread RWH implementation at schools in the eThekwini Municipality would offer upscaled advantages like water and cost savings and would allow for reduced strains on municipal networks. Furthermore, disadvantaged schools that do not have access to basic water and sanitation services may benefit from the water supply through RWH systems (although proper treatment and disinfection for potable uses would be needed). It is recommended that such schools (and schools likely to garner higher savings based on their local characteristics) be prioritized for governmental RWH subsidies.

6.2.2. Further Studies

The following ideas could be extended from the current research for future studies:

- Increases in water tariffs will provide higher municipal water cost savings for the same physical amount of water harvested and used. Hence, systems with similar capacities/characteristics specified in this study may have improved economic viability in the future.
- Consumption data was only used for the last three years since these bills were readily available compared to years further back. Unfortunately, municipal data may become obsolete over minor time intervals depending on the frequency of increased rates. Hence, it may be more effective to use projected rates rather than current ones to avoid research that becomes quickly outdated.
- In-depth survey data could be gathered to identify how many flushes occur per toilet facility per day. In this way, the proportion of toilet-flushed water per block can be determined. Tanks can then be sized according to catchment supply and facility demand. In this study, the savings introduced per scenario indicated the rainwater that would be substituted for partial municipal supply. However, by performing a more in-depth account for water usage proportions, harvested rainwater could potentially be used for more, if not all water activities at a school. Enhanced disinfection protocols would then be necessary for potable water uses.
- LCAs could be prepared for different component materials that may lower the environmental burdens. Currently, the storage tanks in SA are made up of LLDPE; however, should an alternate material be used for production it would be beneficial to assess the environmental impacts in a consequential LCA setting. In other words, by substituting LLDPE for another material, the comparative environmental burdens could be determined to identify a material type that performs better. This concept can be extended to all RWH components; however, the most noticeable effects would probably be the tanks based on the findings of this research.
- The ecological burdens from municipal supply could be assessed in relation to the environmental effects of the proposed RWH systems. In this way, the burdens could be compared across all 18 impact categories, thereby indicating which system is more environmentally viable. Overall, more research on school RWH systems should be undertaken to verify and identify further system benefits.

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APPENDICES

Appendix A: Selected Municipal Bills – Duffs Road Primary

Below: Water bill for December 2020

Detailed Invoice

DUFFS ROAD PRIMARY SCHOOL
95089231035

ETHEKWINI MUNICIPALITY

Water and Sanitation

Reference - W9805652 2 ROBIN ROAD, DUFFS ROAD(P), DUFFS ROAD(P)

Meter No.	Previous Meter Reading		Current Meter Reading		Usage	
	Date	Reading	Date	Reading	Units	Days
BMT384	2020/11/24	24801.00	Not Read	Not Read	269.00 kl(Estimate)	44

Description	Units	Rate (R)	Amount (R)
Consumption charge for estimated consumption (2020/11/24 to 2021/01/07)	269.00 kl	36.52 /kl	9,823.88*
Fixed charge for estimated consumption (2020/11/24 to 2021/01/07)	44 days	56.35 /day	2,479.53*

Below: Electricity bill for December 2020

Business And Residential Electricity

Reference - E681707 2 ROBIN ROAD, DUFFS ROAD(P), DUFFS ROAD(P)
Business & General - Scale 1

CT Ratio	1.00000	VT Ratio	1.00000	Installed Capacity	

Meter No.	Register	Previous Meter Reading		Current Meter Reading		Reading	Usage
		Date	Reading	Date	Reading	Constant	
2307812M	Energy	Not Read	Not Read	Not Read	Not Read	1.00	Not Read

Service from 2020/12/03 to 2021/01/03 31 days Daily Average: 46.27 kWh/day

Description	Units	Rate (R)	Amount (R)
Energy charge for estimated consumption (2020/12/03 to 2021/01/03)	1434.00000 kWh	2.0562 /kWh	2,948.59*
Service charge for estimated consumption (2020/12/03 to 2021/01/03)	-	269.0500 p.m	278.02*
Percentage rebate	0.40000 kV	0.0000 /kV	0.00Cr

Below: Water and Electricity bill for May 2018

Water and Sanitation

Reference - W9805652 2 ROBIN ROAD, DUFFS ROAD(P), DUFFS ROAD(P)

Meter No.	Previous Meter Reading		Current Meter Reading		Usage	
	Date	Reading	Date	Reading	Units	Days
BMT384	2018/04/26	19565.00	2018/05/28	19691.00	126.00 kl	32

Description	Units	Rate (R)	Amount (R)
Consumption charge	126.00 kl	25.21 /kl	3,176.46*
Fixed charge	32 days	38.91 /day	1,245.22*

Sewage Disposal

Reference - W9805652 2 ROBIN ROAD, DUFFS ROAD(P), DUFFS ROAD(P)

Description	Units	Rate (R)	Amount (R)
Sewage Disposal Charges	113.40 kl	7.47 /kl	847.10*
Non-DomesticSewage disposal charges: 90.00% of 126 kl			
Sewage Disposal Fixed charge	32 days	27.67 /day	885.32*

Business And Residential Electricity

Reference - E681707 2 ROBIN ROAD, DUFFS ROAD(P), DUFFS ROAD(P)
Business & General - Scale 1
CT Ratio 1.00000 VT Ratio 1.00000 Installed Capacity

Meter No.	Register	Previous Meter Reading		Current Meter Reading		Reading	Usage
		Date	Reading	Date	Reading	Constant	
2307812M	Energy	2018/05/02	55252.00000	Not Read	Not Read	1.00	Not Read

Service from 2018/05/02 to 2018/06/05 34 days Daily Average: 29.83 kWh/day

Description	Units	Rate (R)	Amount (R)
Energy charge for estimated consumption (2018/05/02 to 2018/06/05)	1014.00000 kWh	1.6024 /kWh	1,624.83*
Service charge for estimated consumption (2018/05/02 to 2018/06/05)	-	209.6800 p.m	237.64*
Percentage rebate	0.40000 kV	0.0000 /kV	0.00Cr

Appendix B: Selected Daily Rainfall Data – EXCEL spreadsheet

Below: Daily rainfall data for the year 2020 (Newlands Reservoir No. 3)

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
1	0	0	0	0	11.4	0	0	0	0	0	26.4	2.6
2	1	0	0	3.8	0	0	0	0	3.8	9.2	29.2	0.8
3	0	0	0	0	0	0	0	0	0.2	14.2	7.4	0.6
4	0	0	0	0	0	0	0	0	0	9	0	28.8
5	0	0	0	0.2	0	0	0	0	0	13.4	0	12.8
6	0	0	0	0	0	0	0	0.2	0.4	9.2	4.6	1.4
7	0	6.4	0	2	0	0	0	0.4	0.6	0	0	0
8	0	8	0	0.6	0	0	1	0	0	0	0.8	0
9	1.2	0.2	0	0	0	0	0.2	0	0	0	0.6	0
10	9.6	1.4	1	0.2	0	0	0	0	0	0.2	10	0
11	9.8	3	7.8	0	0	0	0	0	0	0.2	40.6	2
12	0	3.2	0	0	0	3.6	21.6	0	0	0.2	12.6	2
13	0	2.2	0	0	0	0.2	0.2	0	0	0	0.2	5.2
14	0	0	0	16	0.8	0	0	0	0	0	0	2.2
15	0	0	0	26.4	0	0	1	0	5	0	0	0
16	0	0	0.6	0	0	0	0	0	1.2	0	1.6	0.6
17	0	0	28.2	0	0	0	0	0	0.2	0.2	0.8	0.6
18	0	0	0	0	0	2.4	0	0	0.4	0	0.8	0
19	0	52.6	0	2.8	0	2.4	0	1	0	0	44.2	0.4
20	5.6	55.4	0	0.2	0	0	0	0	0	0	0.4	0.2
21	0.2	0.2	0.8	0	0	0	0	0	0	0	0.2	0
22	0	0	0	0	0	0	0	0	0	2	0.6	0
23	11.2	0.2	0	0	0	0	0	0	0	0	0	3
24	2.6	1	0	0	0	0	0	0	0	0	0	0.6
25	0	0	18.4	1.8	0	0	0	0	0	0	2.8	6.4
26	0	0	0.4	0.2	0	0	0	0	0	0.2	2.2	1.6
27	18.2	0	0	6.6	0	0	0	0	0.8	5.4	12	0
28	0.4	0.8	0	0.4	0	0	0	0	9.4	0	1.2	0
29	0	14	0	13	0	0	0	0	2	0	0	0
30	0		0.2	8.8	0.4	0	0	0	10.6	0	0.2	0
31	0		0		0		0	3.2		12.2		0.4
TOTAL	59.8	148.6	57.4	83	12.6	8.6	24	4.8	34.6	75.6	199.4	72.2

Below: Daily rainfall data for the year 2019 (Newlands Reservoir No. 3)

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
1	0.2	0.2	0.2	0	0	0	0	1	3	0	3.2	0
2	0	0	0	0.2	0	0	2	0.2	0	0	0.2	2.4
3	0	0.2	3.6	5.6	0	0	0	0	0	0	0	0
4	0.2	0.2	2.8	0.2	0	0	0	0	0	0	0.6	13
5	0	0	0	0	0.6	0	0	0	0	0	9.6	3.6
6	0.2	0.2	0	6.2	0.2	0	0.4	0	23	0	1.6	0
7	0	0.2	0	6.6	0.2	0.6	0	0	3.6	0	2.2	0
8	0	0	0	0.2	0	0	0.2	0	0	0	5.6	0
9	0	0	0	0.2	0	6.2	0.2	0	0	0	0.6	0.6
10	0	0	7.8	0	0	0	0	0	0	2.4	3.4	5.6
11	0	0	45	0	0	0	0	0	0	1.2	21.2	15
12	0	0.2	10.4	0	0	0	2	0	0	0	21	87
13	0	0	5	0	0	0	0	0.4	0	0	4	70.2
14	0	0.2	0	0.6	0	0	0	0	0	4.4	3.8	3.4
15	1.4	0	0	2.6	0	0	0	0	0	0	12.6	0.6
16	1.4	0	0	0.6	0	0	0	0	0	0	5.6	0
17	0.6	0	2.8	0	0	0	0	0	0	0	0.2	0
18	0.2	0	0	0.4	0	0	0	2.2	0	0	6.6	0
19	0.2	0.4	0.8	1.2	0	0	0	0	0.2	0	0	0
20	0	8.2	1.2	0.2	0	0	0	0.2	0	0	4.4	0
21	0.2	8.4	1	0	0	0	0	0	0	0	6	1.2
22	0	0.2	2.8	0	0	0	0	0	0	5	0.4	2.8
23	0.8	0.6	3.4	0	0	0	0	0	0	1.8	0.2	0.2
24	0.6	2	1.4	3.8	0	0	0	0	0	0	0	0
25	1.2	0.2	0.8	68.6	0	0	0	0	0	0	0.8	0
26	0.4	0	0.4	0.4	0	0.2	0	0	0	0	2.6	1.4
27	0.8	0	0.8	0	0	0	0	0	0	1.8	0	0.6
28	0.4	0.2	0.4	0	0	0	0	0	0	0.2	0	1
29	0.2		0.2	0	8	0	0	1.4	0	9.6	0.4	0.6
30	0		0.2	0	0	0	0	0.4	0	0.4	0	2.4
31	0.2		0		0		0	3.4		2		0
TOTAL	9.2	21.6	91	97.6	9	7	4.8	9.2	29.8	28.8	116.8	211.6

Appendix C: Toilet Flushing Usage – Duffs Road Primary Survey

Survey: Toilet Usage at Duffs Road Primary

- This survey aims to identify the water consumption for toilet flushing at Duffs Road Primary School.
- Teachers are kindly requested to ask their class **how many times in a day their learners use any of the school toilets based on the options provided**.
- Learners will raise their hands for **one** option, and teachers will count and record the number in the appropriate column.
- The options include both urine and faecal use.
- All learner data obtained will be anonymous.
- Any absentees may be questioned on another day.

Teacher: ………………………………………………..

Grade: ……………

Class Total: ………………

Toilet Use Options	Number (Batch 1)	Number (Batch 2)	Total
Zero			
Once a day			
Twice a day			
Three times a day			
Four times a day			
Five times a day			
Six times a day			
Seven times a day			
Eight times a day			
More than eight times daily			

Thank you for your participation and assistance.

Appendix D: Gutter and Downpipe Information – Red Book

Below: Red Book (2013) – Section 11.2, Figure 11.1

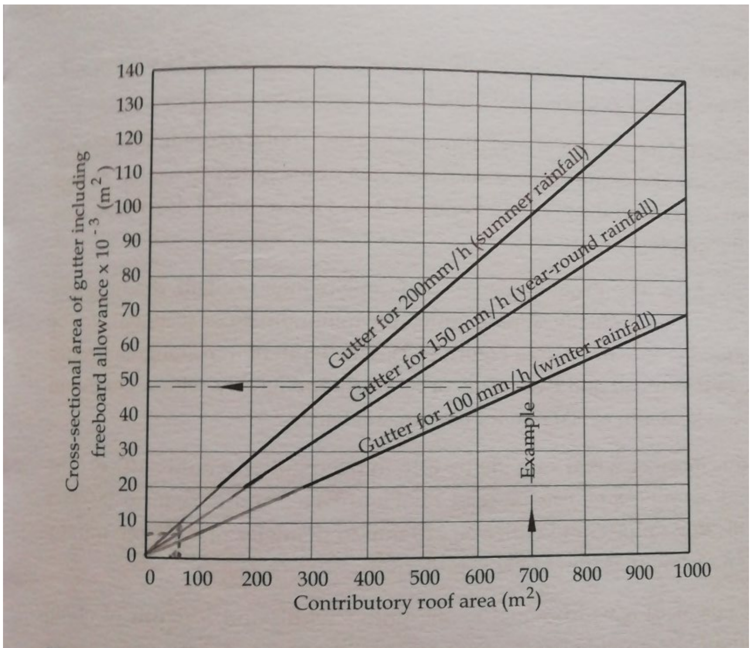


Figure 11.1 Gutter selection associated with 'rough guide'

Below: Red Book (2013) – Section 11.2, Figure 11.2

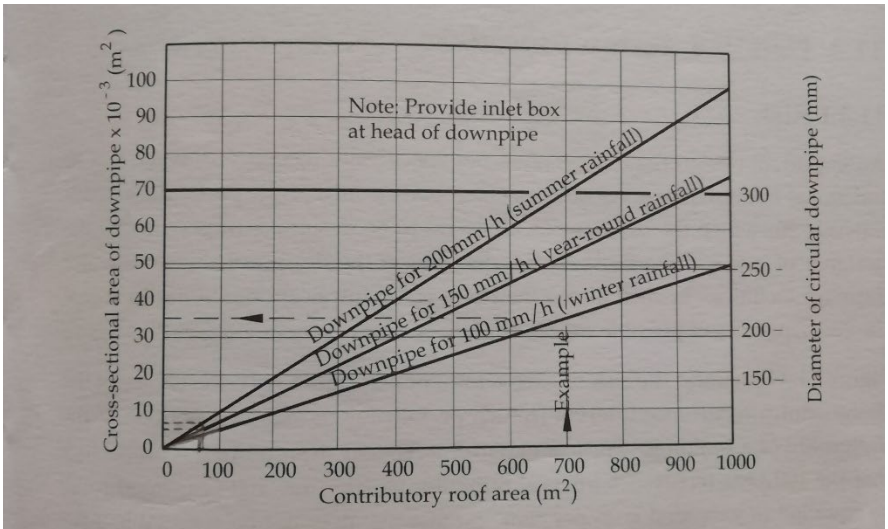
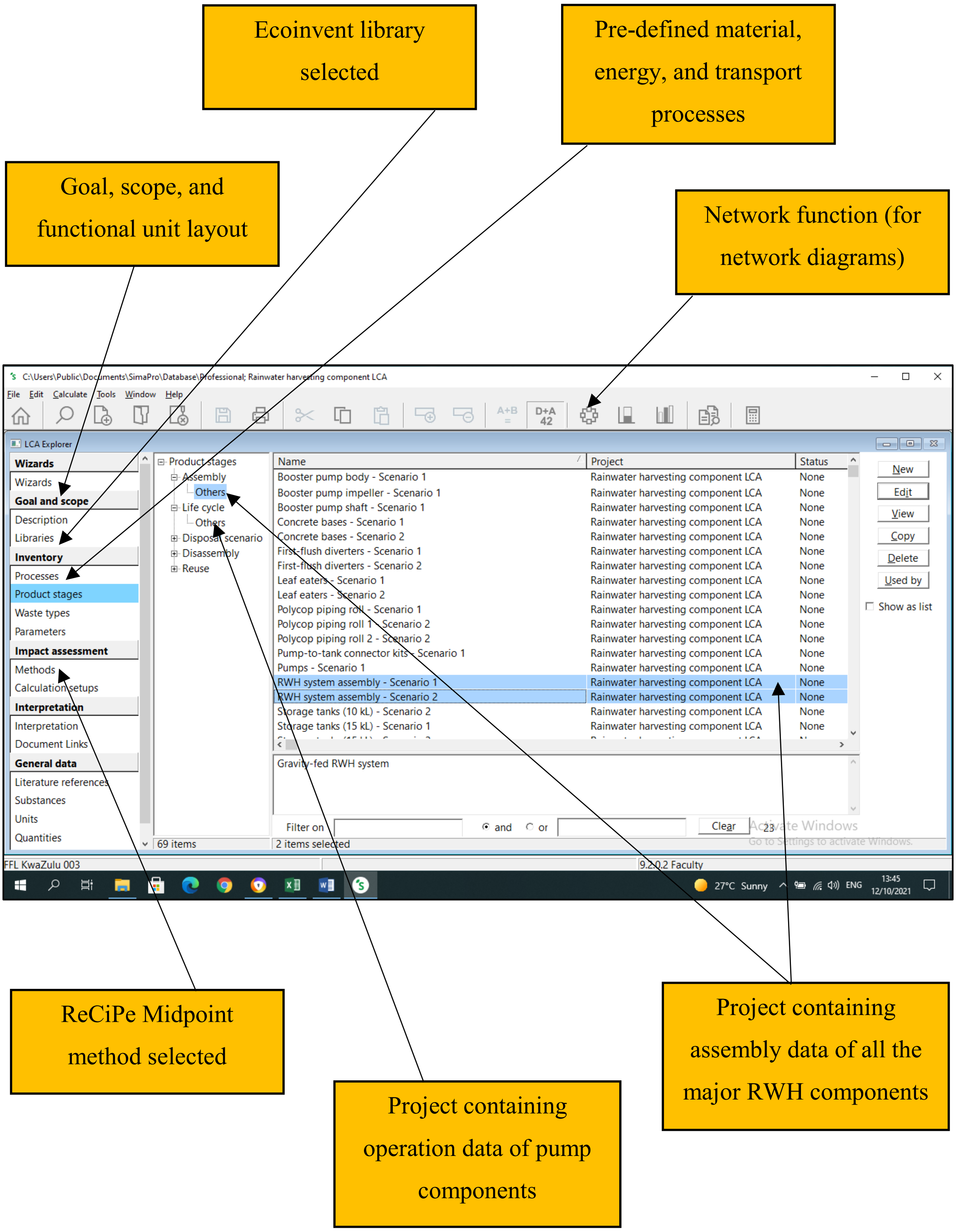


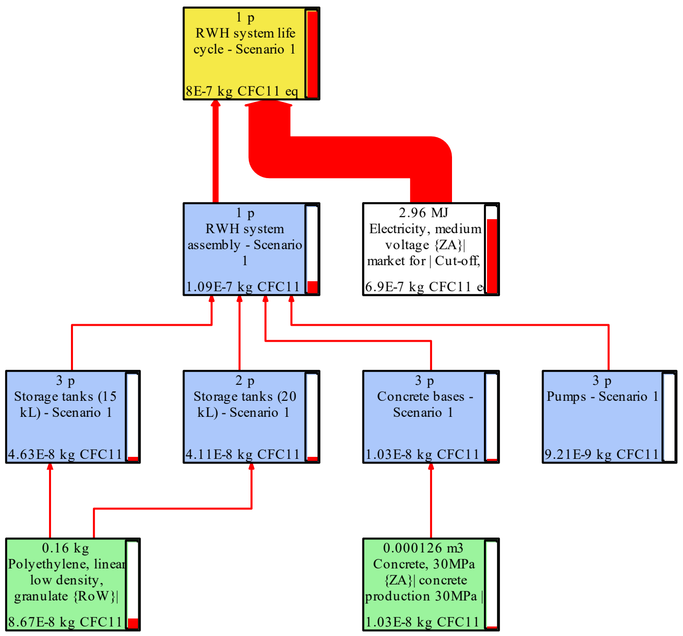
Figure 11.2 Downpipe selection associated with 'rough guide'

Appendix E: Basic SimaPro User Interface

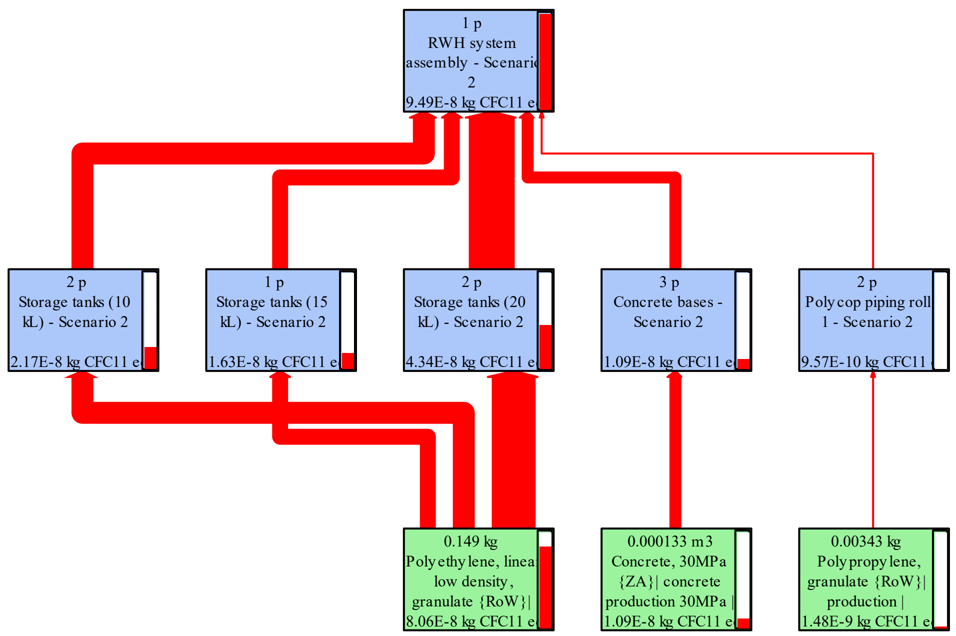


Appendix F-1: Stratospheric Ozone Depletion Burdens

Below: Assembly and operation impacts per kL harvested rainwater (Scenario 1)

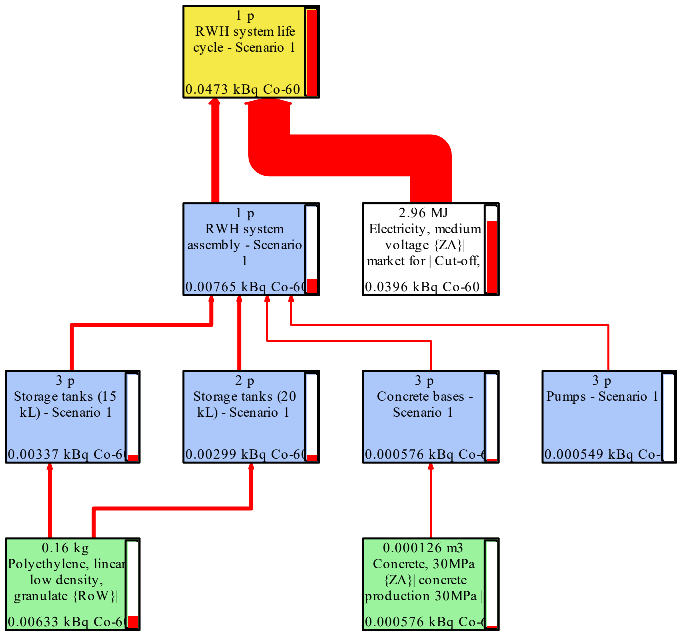


Below: Assembly impacts per kL harvested rainwater (Scenario 2)

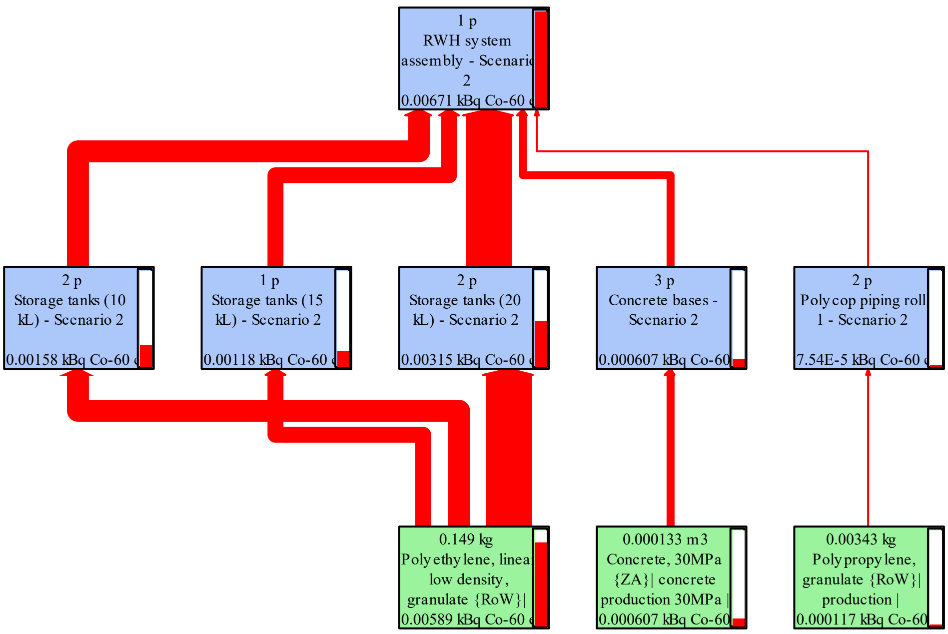


Appendix F-2: Ionizing Radiation Burdens

Below: Assembly and operation impacts per kL harvested rainwater (Scenario 1)

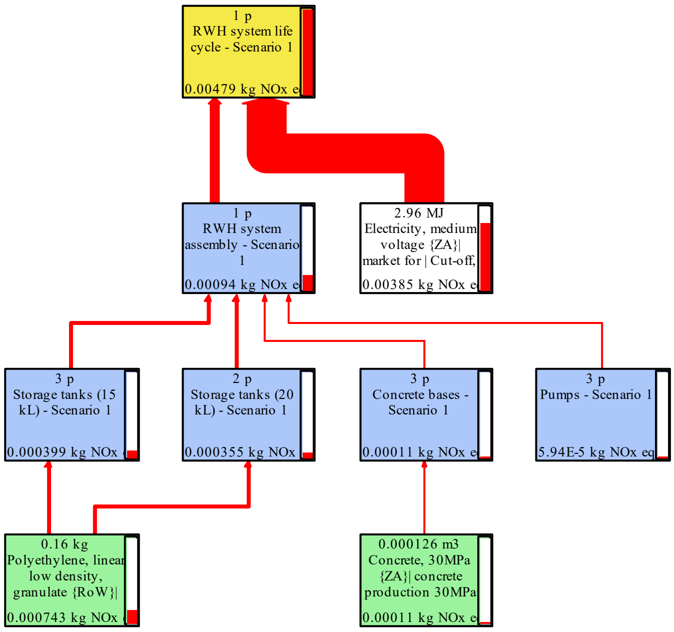


Below: Assembly impacts per kL harvested rainwater (Scenario 2)

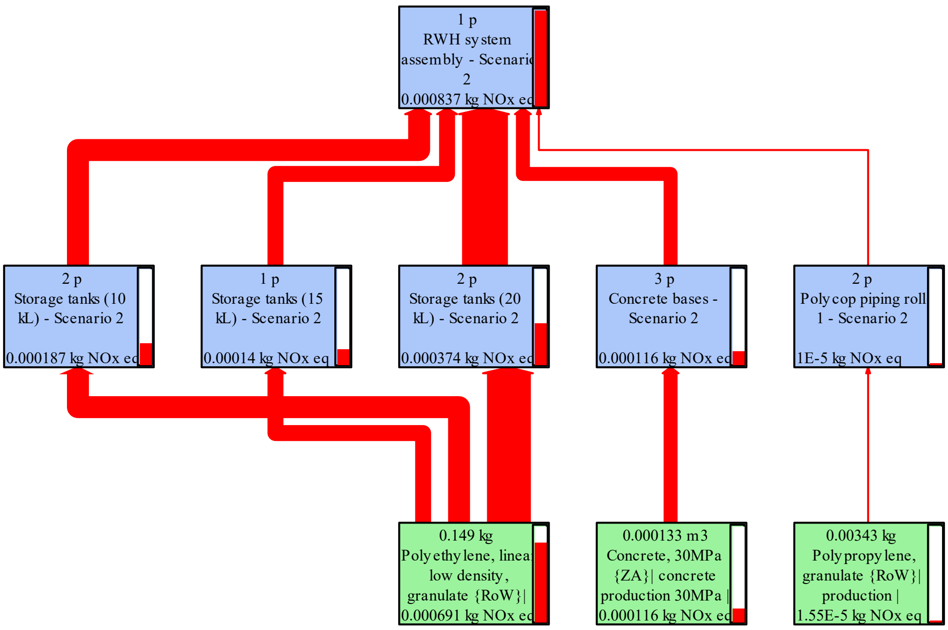


Appendix F-3: Ozone Formation Burdens (Human Health)

Below: Assembly and operation impacts per kL harvested rainwater (Scenario 1)

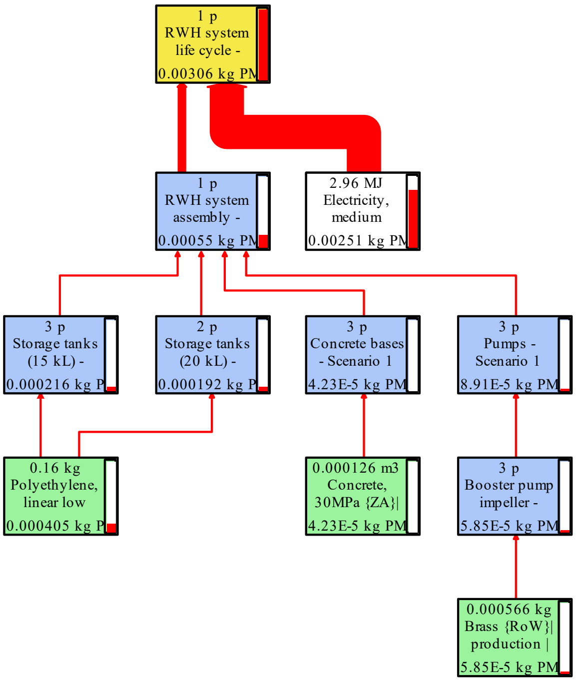


Below: Assembly impacts per kL harvested rainwater (Scenario 2)

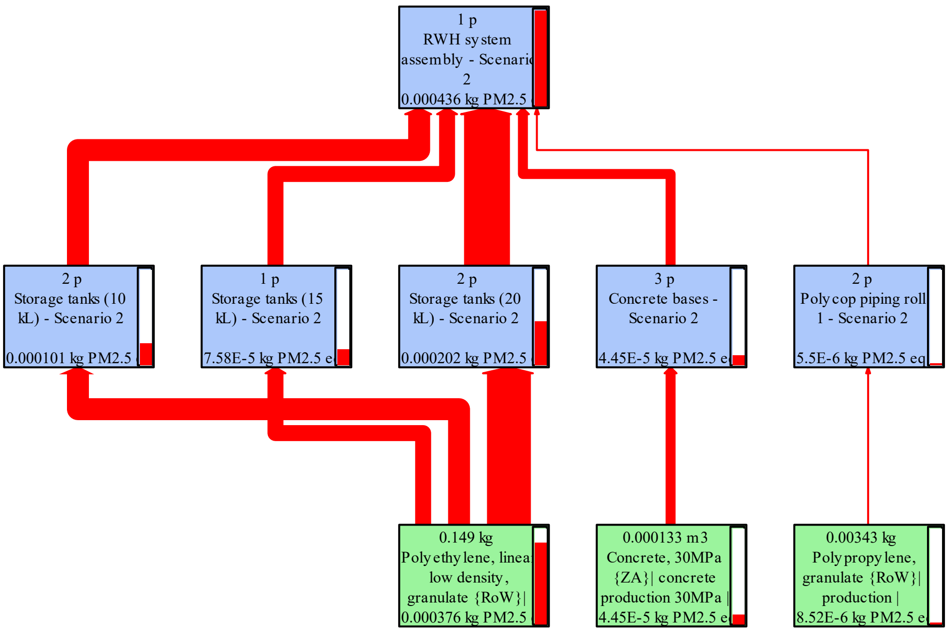


Appendix F-4: Fine Particulate Matter Formation Burdens

Below: Assembly and operation impacts per kL harvested rainwater (Scenario 1)

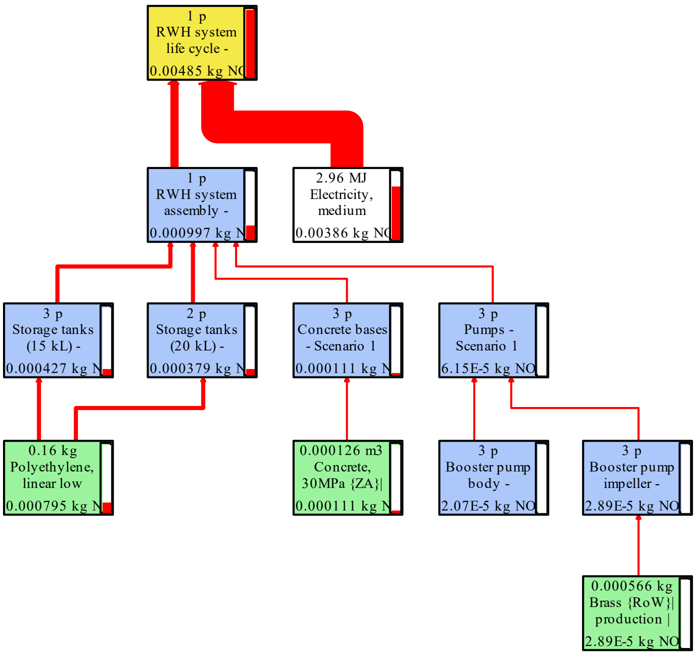


Below: Assembly impacts per kL harvested rainwater (Scenario 2)

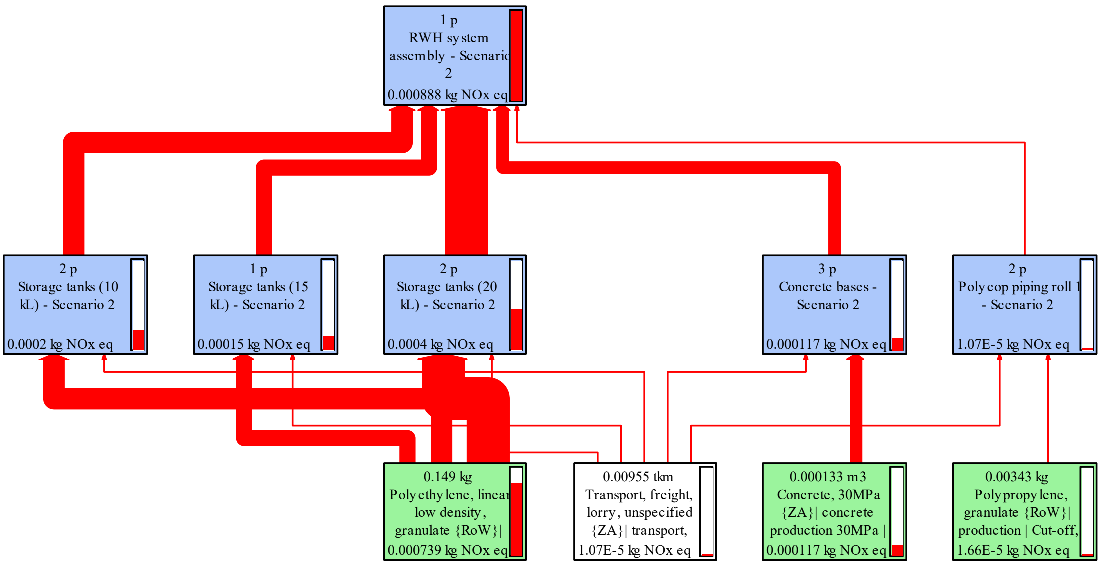


Appendix F-5: Ozone Formation Burdens (Terrestrial Ecosystems)

Below: Assembly and operation impacts per kL harvested rainwater (Scenario 1)

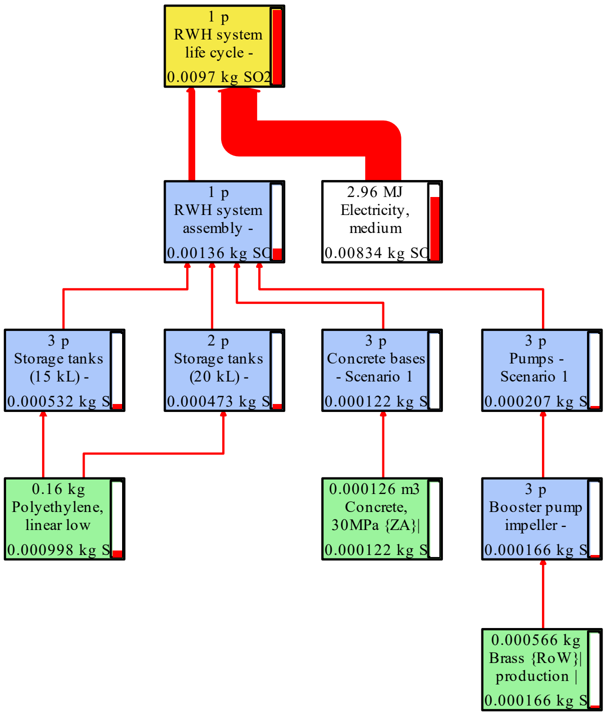


Below: Assembly impacts per kL harvested rainwater (Scenario 2)

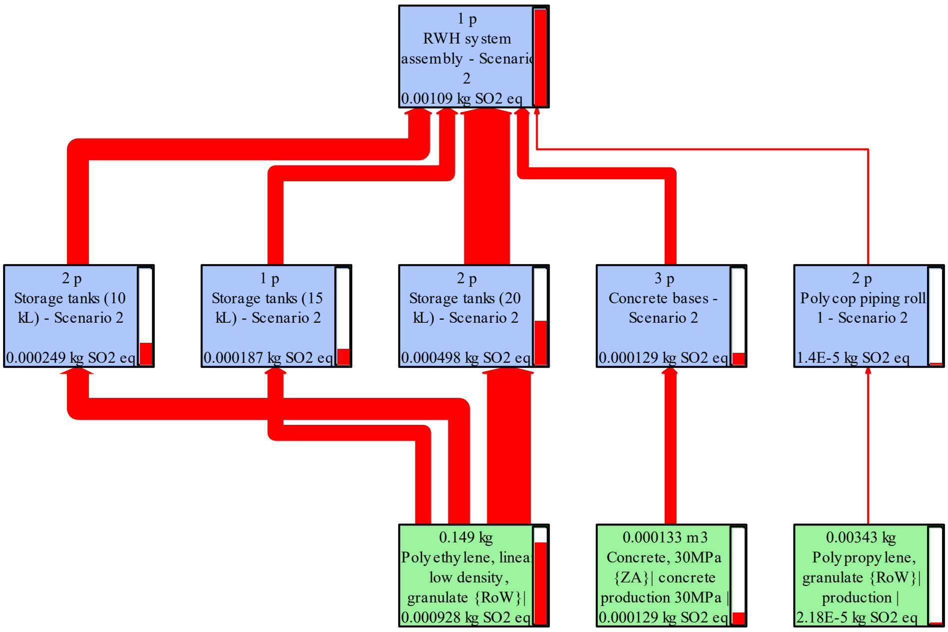


Appendix F-6: Terrestrial Acidification Burdens

Below: Assembly and operation impacts per kL harvested rainwater (Scenario 1)

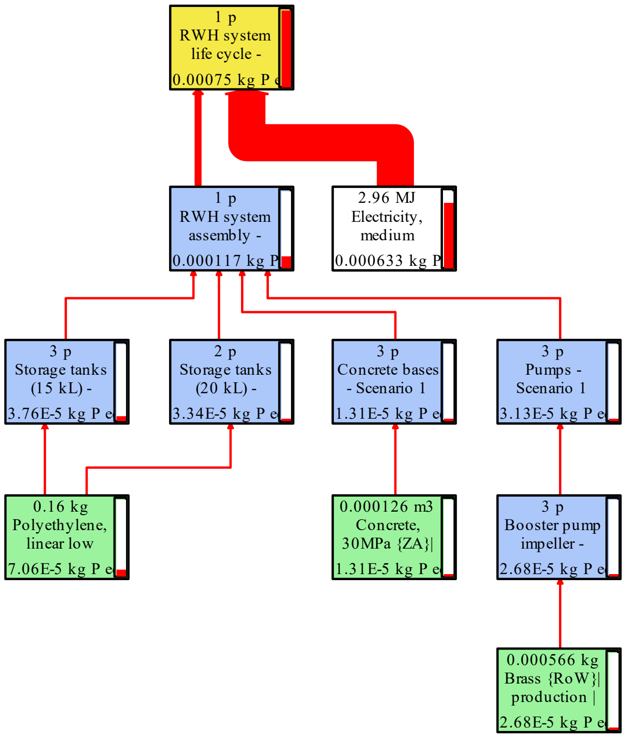


Below: Assembly impacts per kL harvested rainwater (Scenario 2)

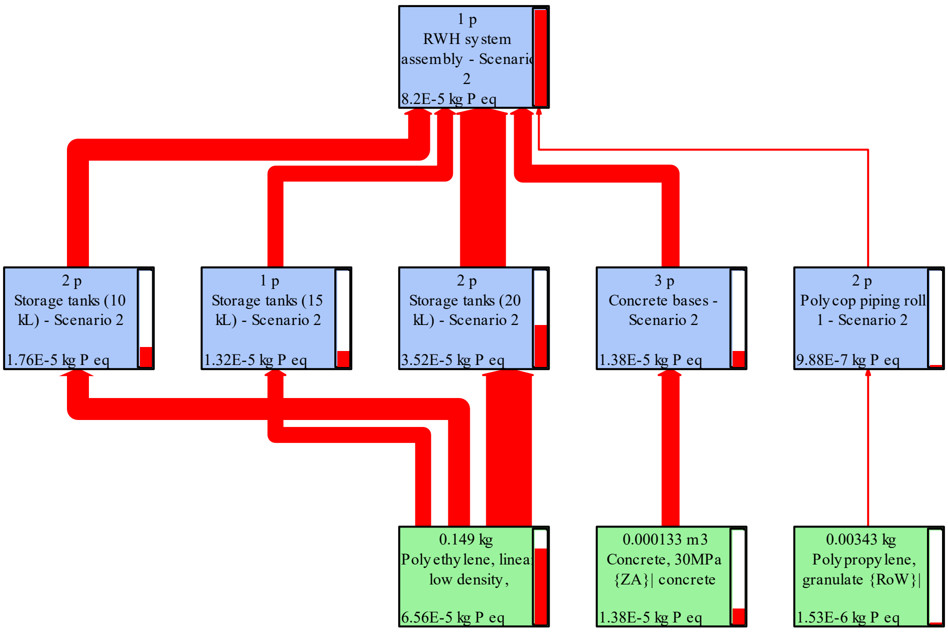


Appendix F-7: Freshwater Eutrophication Burdens

Below: Assembly and operation impacts per kL harvested rainwater (Scenario 1)

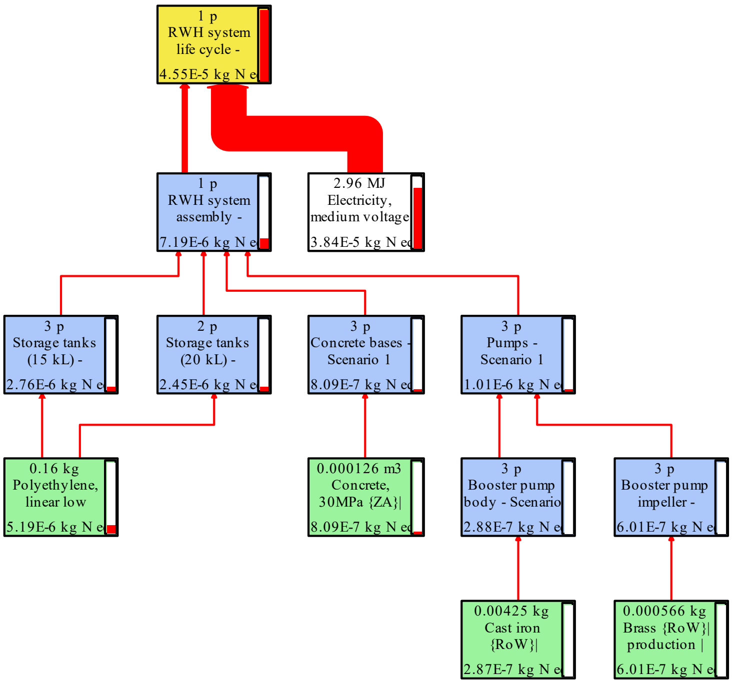


Below: Assembly impacts per kL harvested rainwater (Scenario 2)

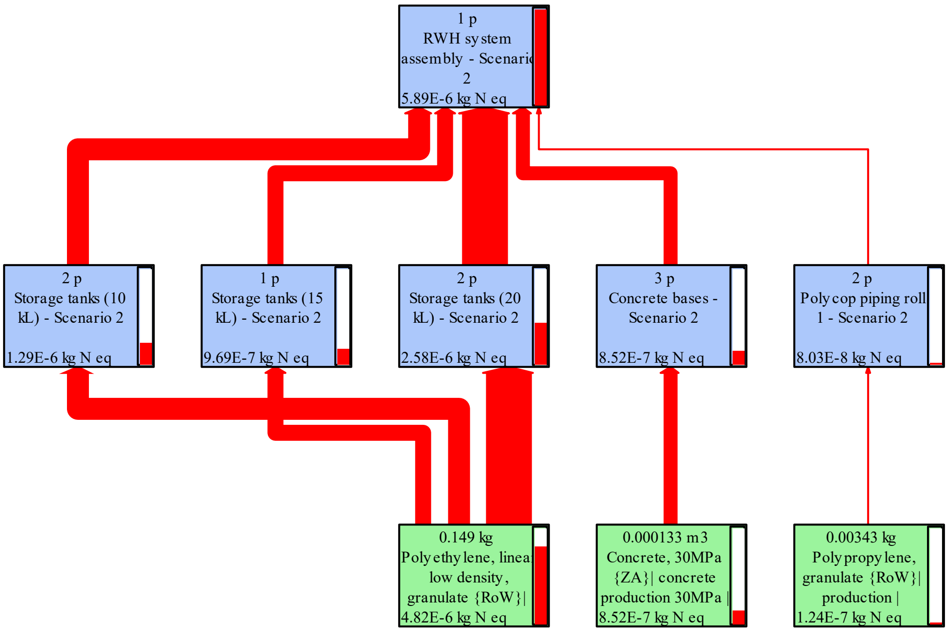


Appendix F-8: Marine Eutrophication Burdens

Below: Assembly and operation impacts per kL harvested rainwater (Scenario 1)

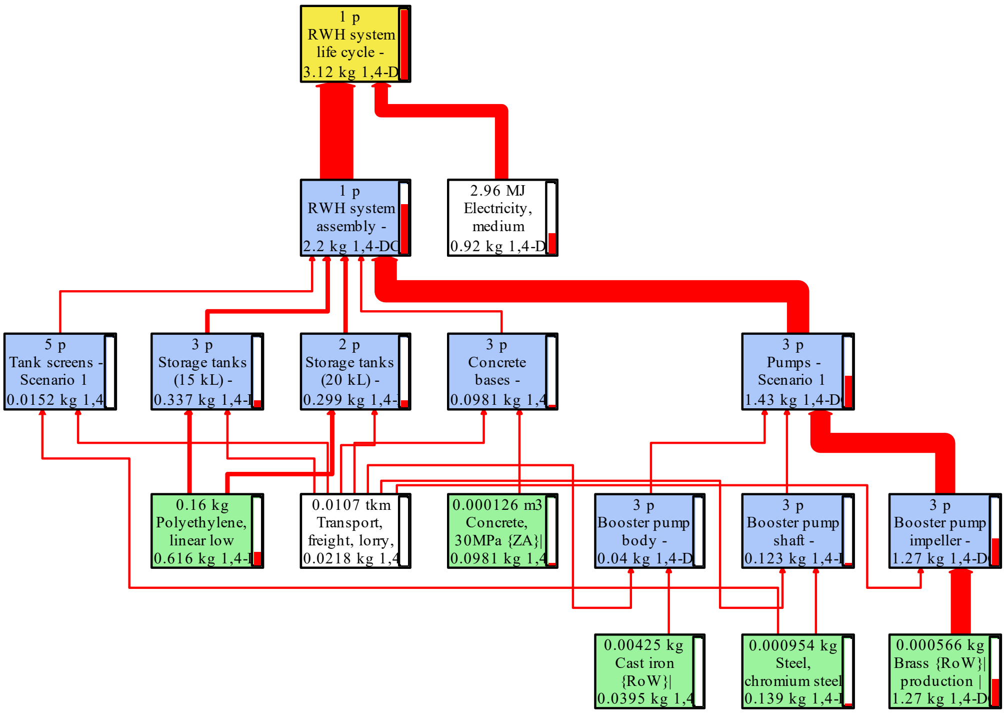


Below: Assembly impacts per kL harvested rainwater (Scenario 2)

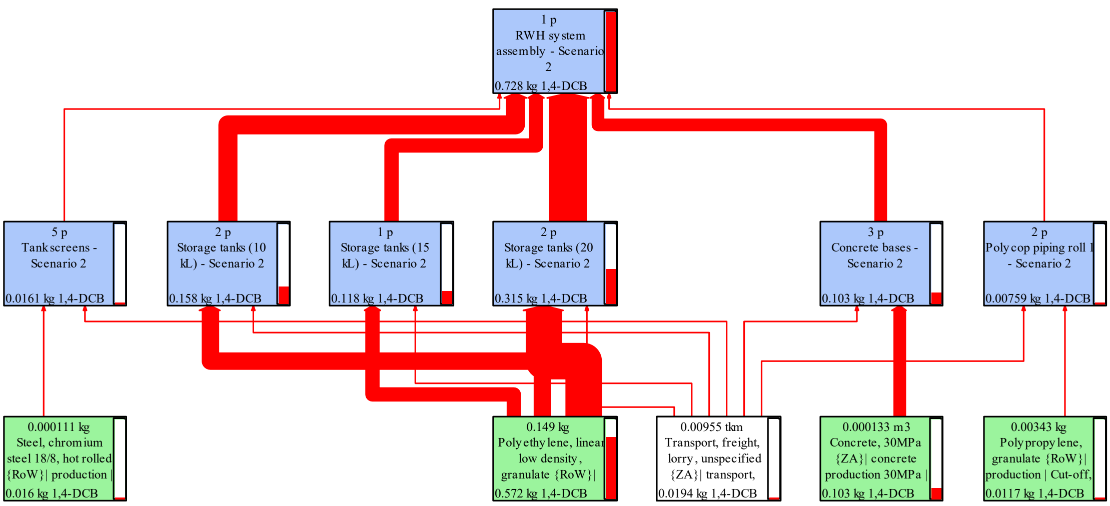


Appendix F-9: Terrestrial Ecotoxicity Burdens

Below: Assembly and operation impacts per kL harvested rainwater (Scenario 1)

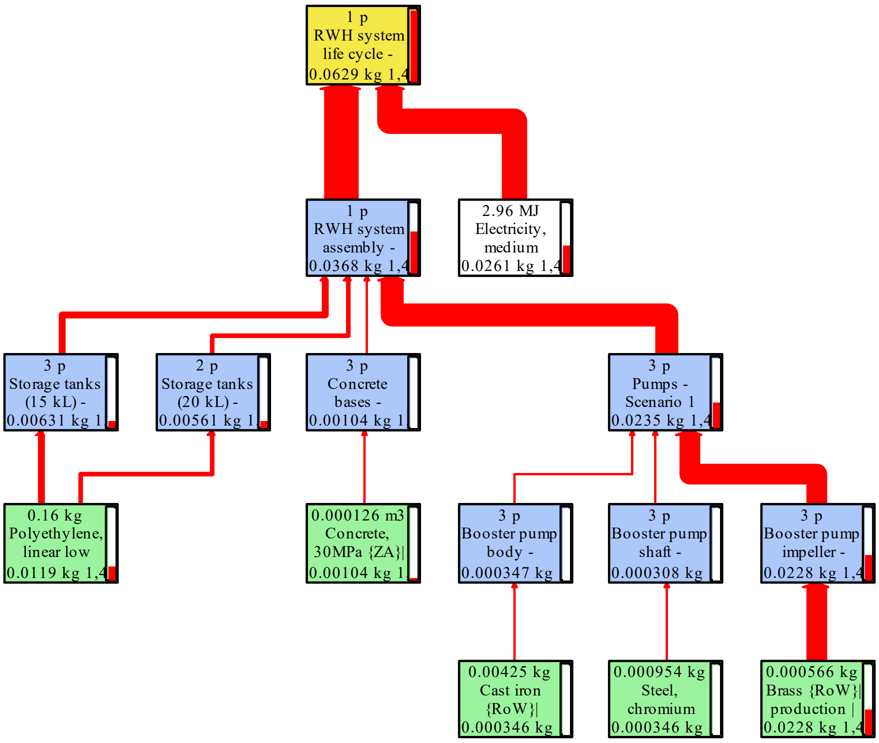


Below: Assembly impacts per kL harvested rainwater (Scenario 2)

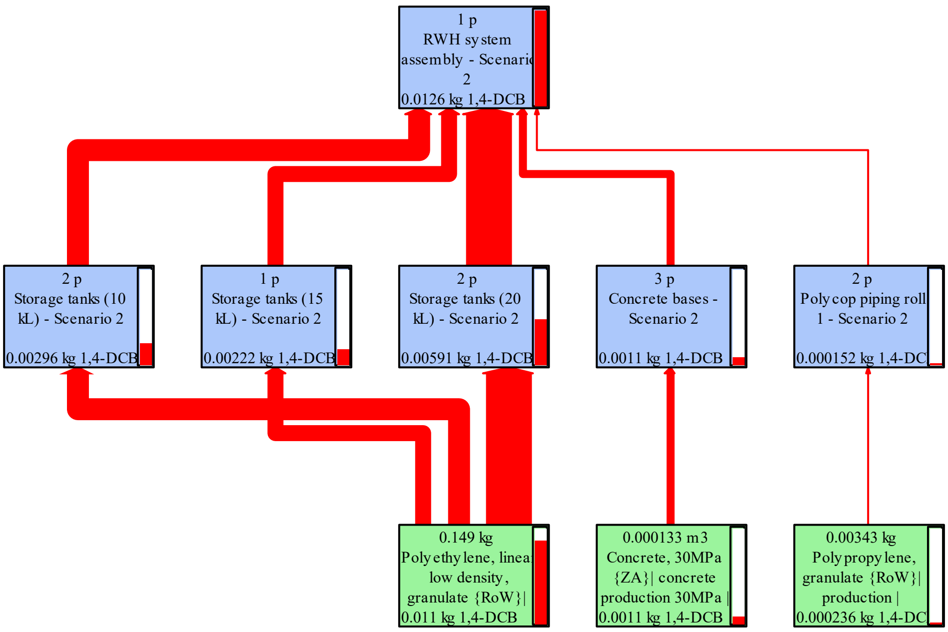


Appendix F-10: Freshwater Ecotoxicity Burdens

<u>Below: Assembly and operation impacts per kL harvested rainwater (Scenario 1)</u>

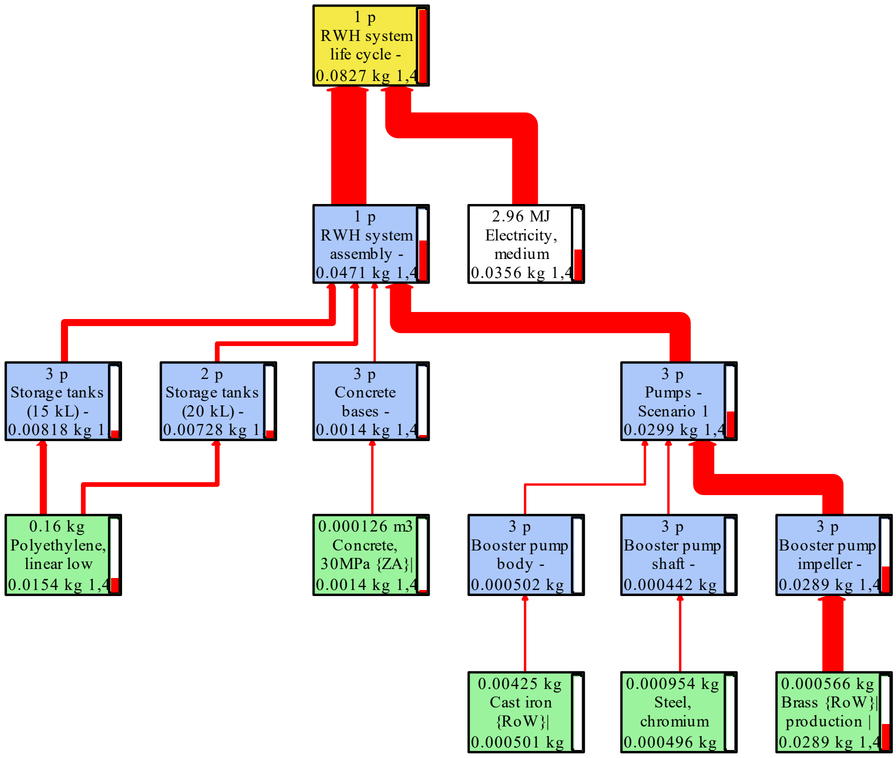


<u>Below: Assembly impacts per kL harvested rainwater (Scenario 2)</u>

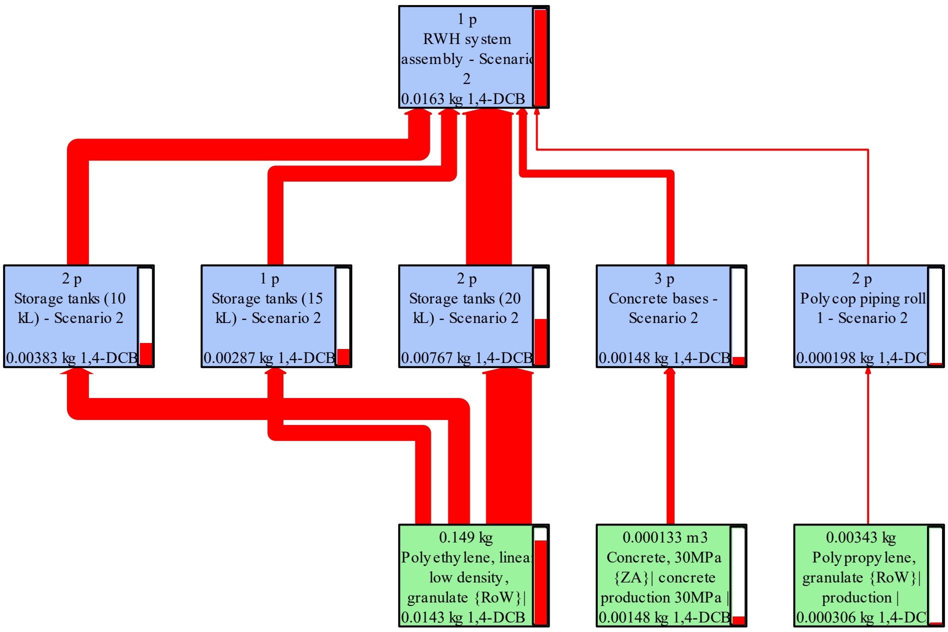


Appendix F-11: Marine Ecotoxicity Burdens

Below: Assembly and operation impacts per kL harvested rainwater (Scenario 1)

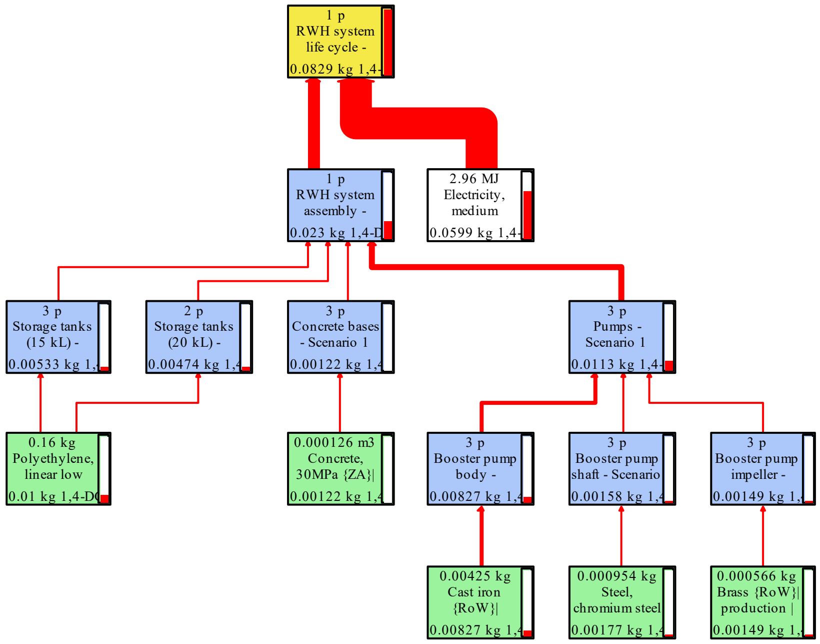


Below: Assembly impacts per kL harvested rainwater (Scenario 2)

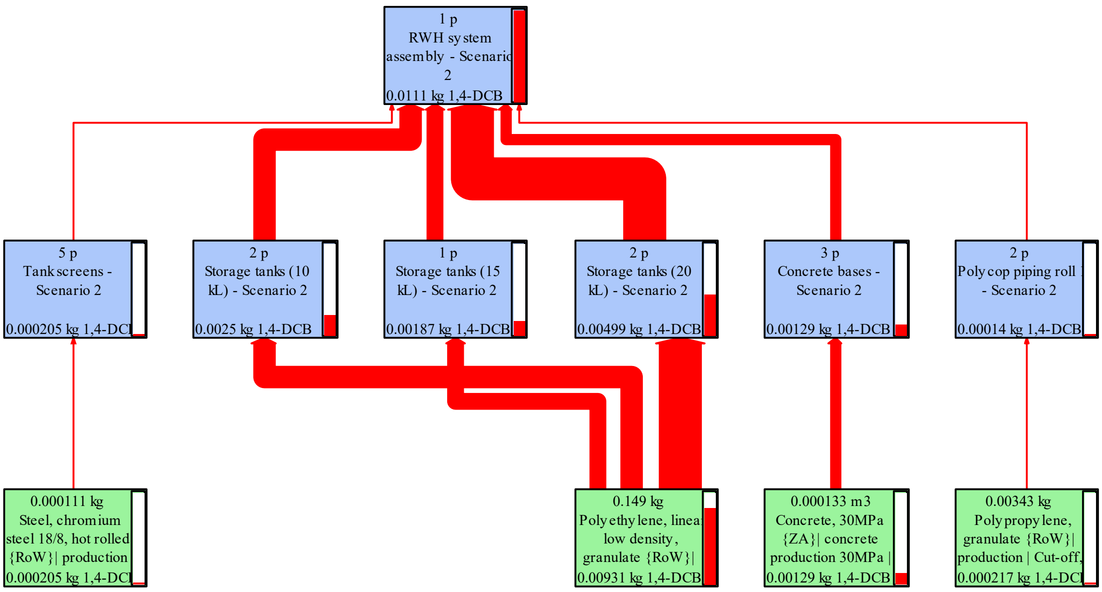


Appendix F-12: Human Carcinogenic Toxicity Burdens

Below: Assembly and operation impacts per kL harvested rainwater (Scenario 1)

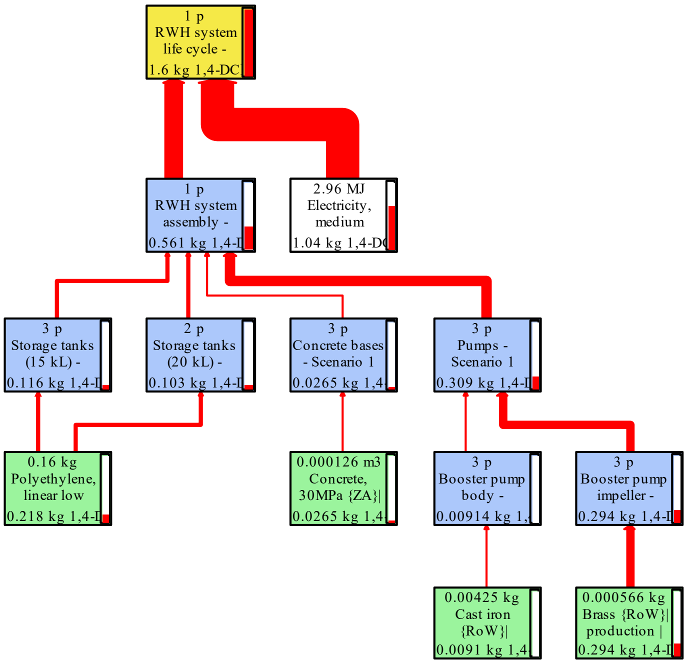


Below: Assembly impacts per kL harvested rainwater (Scenario 2)

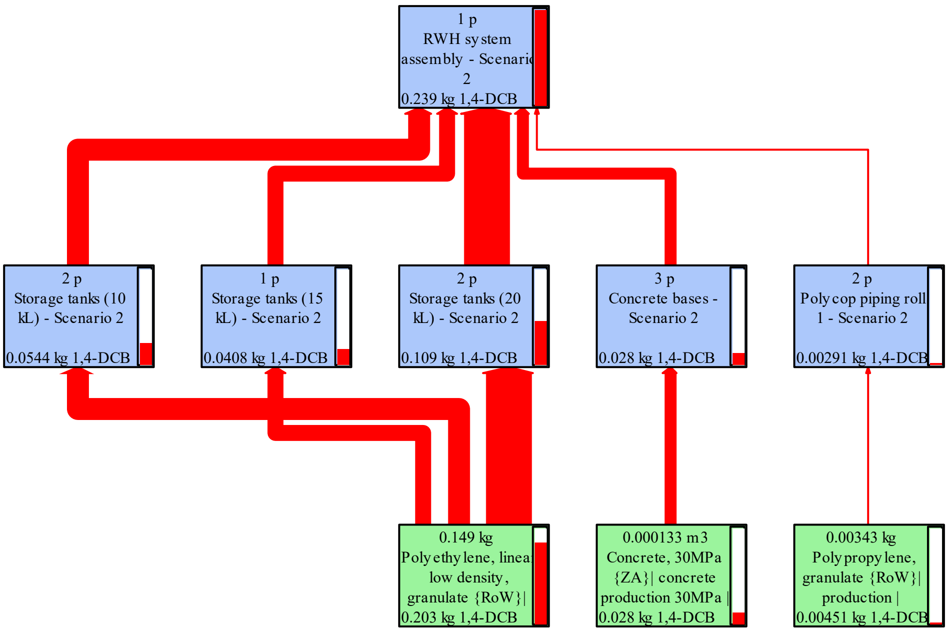


Appendix F-13: Human Non-Carcinogenic Toxicity Burdens

Below: Assembly and operation impacts per kL harvested rainwater (Scenario 1)

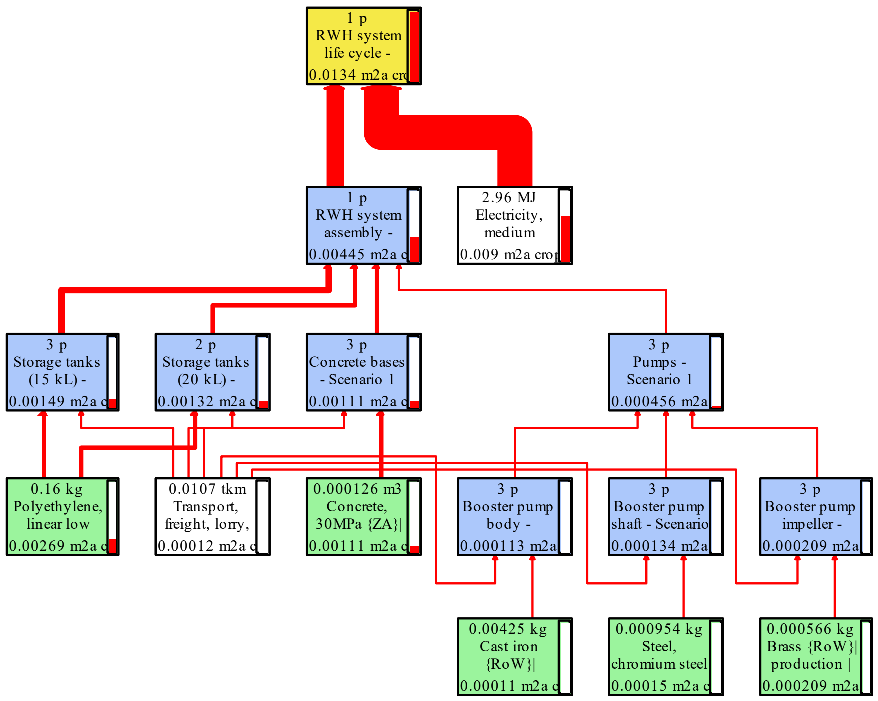


Below: Assembly impacts per kL harvested rainwater (Scenario 2)

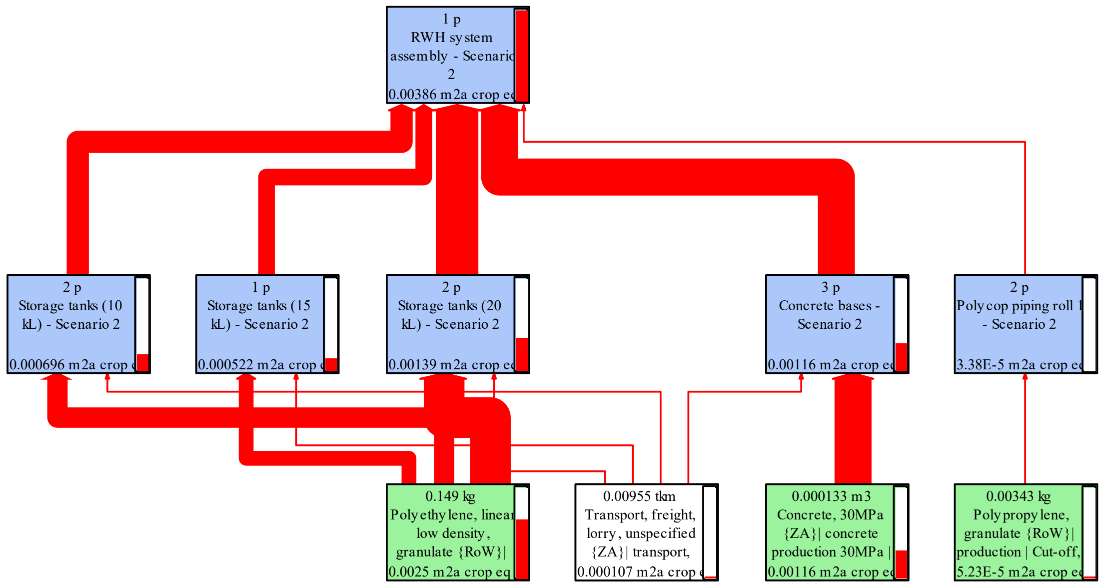


Appendix F-14: Land Use Burdens

Below: Assembly and operation impacts per kL harvested rainwater (Scenario 1)

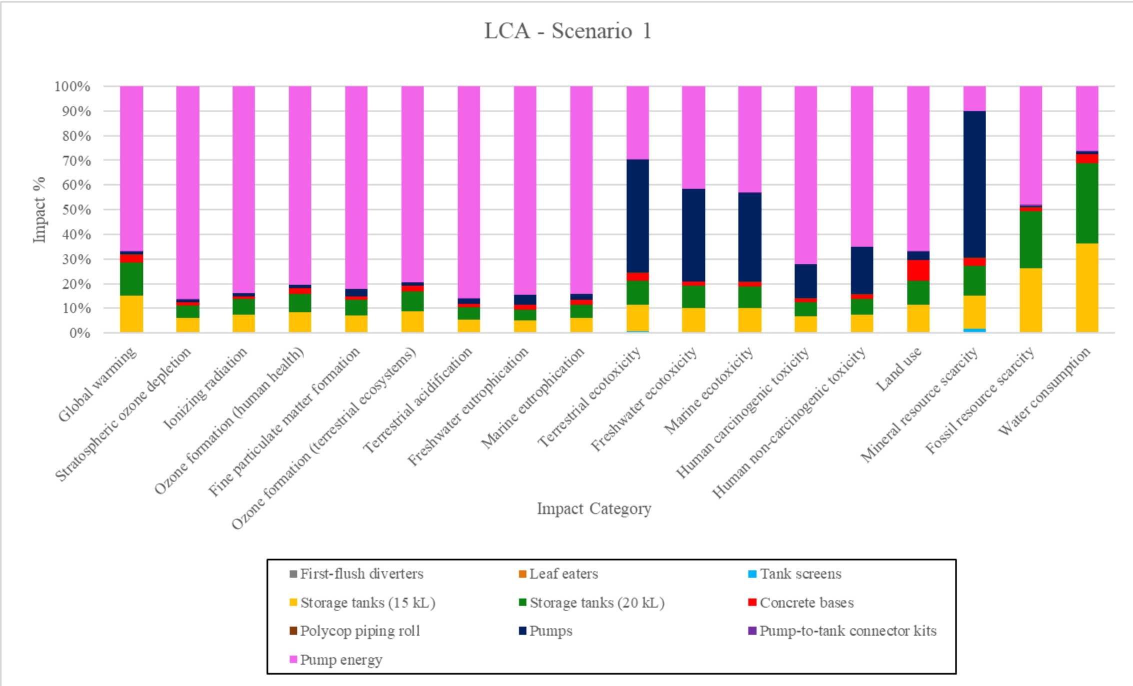


Below: Assembly impacts per kL harvested rainwater (Scenario 2)



Appendix F-15: Impact Category Graphs

Below: Environmental burdens per kL harvested rainwater (Scenario 1)



Below: Environmental burdens per kL harvested rainwater (Scenario 2)