Feasibility of using seawater to flush toilets in the African context

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by

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Executive Summary

One-third of the world's population is estimated to live in water-stressed regions by 2025. Climate change increases temperature, frequency of heatwaves, mixed precipitation, and frequency of extreme drought, which threaten the management of freshwater supply. Cape Town has a permit to discharge up to 55 ML/d of untreated wastewater through its marine outfall sewers. Consequently, the potable water used to flush toilets is effectively lost from the urban water supply system, including the opportunity for reuse.

This study investigated the environmental impact of using alternative water sources for flushing toilets using life cycle assessment (LCA) analysis. It estimated the willingness to pay using a discrete choice experiment in Hout Bay, Cape Town.

The LCA is an environmental accounting tool that quantifies the impacts of products and services across their entire lifecycle. Therefore, it is targeted at systematic and objective decision-making. Its application in the water sector identifies hotspots, ensuring that burdens are not transmitted across the supply chain. The study was conducted in Hout Bay, a seaside suburb with an ageing infrastructure that presents an opportunity to implement dual systems when upgrading. An LCA model based on SimaPro software was used to gather and analyse the infrastructure and operational data of the water treatment and distribution stages. Two systems were compared: a dual system, which incorporates potable water for other household applications and seawater for toilet flushing, and a conventional potable water system.

In the discrete choice experiment, respondents were presented with a future scenario where the cost of drinking water increases, and, in addition, they were offered two other alternatives, recycled water and seawater, that can be provided at a discounted price. In addition to cost, colour, odour, and stain were presented as attributes. Hout Bay uses a marine outfall sewer system for wastewater management. An attribute related to alternative disposal practices was included to evaluate disposal preferences, which included maintaining the current practice, treatment and discharge and treatment and recycling.

The LCA findings indicated that seawater supply for toilet flushing conserves 26% of potable water that would otherwise be withdrawn from freshwater sources. However, this comes at the expense of a 20% increase in ecotoxicity impacts associated with background electricity production and transmission processes and the additional distribution pipelines. Despite its limited significance in the study area, ecotoxicity may have significant implications for agriculture and freshwater bodies in the regions where materials and local energy are produced. Additionally, the dual system increases the global warming potential by 45%. Fossil carbon dioxide, methane, and dinitrogen monoxide emissions in electricity production are the primary contributors. These impacts may be lowered by increasing renewable energy in the electricity grid.

The mixed logit model result indicates that the utility of alternative drinking water sources is enhanced if the water has no colour stain and odour, does not increase the monthly bill and is treated prior to discharge. Household heads earning more than R12 800 per month preferred using drinking water to flush toilets, whereas female respondents with a higher education qualification and living in a household with more than three occupants preferred alternative water sources. Monthly water bill estimates ranged from R350 to R900, contingent on water consumption habits. The findings indicate that respondents were willing to pay an additional 5-10% to improve alternative water sources' colour and stain quality. Contrastingly, a discount of up to 60% on the monthly water bill for accepting water with a slight odour. Moreover, the

respondents are also more likely to support the disposal practice of treating and discharging the wastewater if it reduces their monthly water bill by up to 7%.

In conclusion, in the context of diminishing water supply and water security assurance, implementing dual systems may be favourable despite the significant increase in impact. Furthermore, there is a willingness to accept alternative water sources provided they are supplied at a discounted rate and exhibit no colour. The disposal practice is improved to include treatment before discharge.

Acronyms

AFMBR	Anoxic fluidised membrane bioreactor
AnFMBR	Anaerobic fluidised membrane bioreactor
COP	Communities of practice
CRFWG	Cooks River Friends Working Group
CCT	City of Cpae Town
CV	Contingent valuation
CW	Constructed wetland
DCE	Discrete choice experiment
DPR	Direct potable reuse
DWS	Department of Water & Sanitation
EU	Eutrophication potential
GWP	Global warming potential
GWR	Greywater reuse
HB	Hierarchical Bayesian
HDPE	High-density polyethylene
IUWM	Integrated urban water management
IPR	Indirect potable reuse
ISO	International Standards Organisation
LCCA	Life cycle cost analysis
LCSA	Life cycle social assessment
LCI	Life cycle inventory
LCIA	Life cycle inventory assessment
MBR	Membrane bioreactor
MNL	Multinomial logit
MOS	Marine outfall sewers
MSF	Multistage flush
NBS	Nature-based solution
NPV	Net present value
OECD	Organisation for Economic Cooperation and Development
PVC	Polyvinyl chloride
RBC	Rotating biological contactor
RO	Reverse osmosis
RWH	Rainwater harvesting
PE	Person equivalents

TEV	Total economic value
UWS	Urban water system
WGM	WATERgraafsmeer
WCWSS	Western Cape Water Supply System
WTA	Willingness to accept
WTP	Willingness to pay

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1 Introduction

1.1 Background

According to Neumann *et al.* (2015), projections indicate that Egypt and countries in Western and Eastern sub-Saharan Africa will experience the highest rates of population growth and urbanisation in coastal areas by 2060, surpassing other regions worldwide. Moreover, as global warming continues, the western regions of Southern Africa are expected to become drier by the end of the 21st century, resulting in an increased frequency of droughts (Maúre *et al.*, 2018; Dosio, 2016; Engelbrecht *et al.*, 2015). Given this future context, it can be anticipated that coastal zones will be challenged to meet the increasing demand for water and sanitation services, primarily due to urbanisation.

1.2 Rationale

South Africa discharges 300 ML/d of wastewater via seven marine outfall sewers (MOS). Four of the MOS are located in Cape Town, where untreated wastewater is discharged to the sea. Consequently, the potable water used to flush toilets is effectively lost from the urban water supply system, including the opportunity for reuse.

Cape Town is expected to experience an increase in urbanisation due to the growth of informal settlements and townships. In the context of limited water resources and the effects of climate change on precipitation, providing universal access to sanitation is a challenge. Seawater, independent of the hydrological water cycle, has been used for flushing toilets for several decades in Hong Kong and has led to 20-30% savings in potable water demand (Liu *et al.*, 2019; Tang *et al.*, 2006; Li *et al.*, 2005). However, its public acceptance and environmental feasibility have not been tested in other parts of the world.

Advancements in research on wastewater treatment, with careful consideration of urban drainage and water supply systems, have been well documented (Ananda, 2019; Saagi *et al.*, 2016; Bach *et al.*, 2014; Daigger Glen, 2009). Furthermore, there has been an increase in interdisciplinary research that combines water infrastructure with economics, climate, and sociology (Bach *et al.,* 2014). Researchers have used analytical models such as life cycle assessment (LCA) to evaluate the environmental impact of implementing infrastructure for alternative water supply (Opher *et al.*, 2019; Liu *et al.*, 2016) and wastewater treatment (Li *et al.*, 2021; Gallego-Schmid & Tarpani, 2019; Opher, Tamar, & Friedler, 2016; Corominas *et al.*, 2013; Meneses *et al.*, 2010). Few researchers have modelled the integrated water supply and wastewater treatment (Liu *et al.*, 2016; Lane *et al.*, 2015) to provide a holistic evaluation.

While environmental considerations are beneficial, public perception, acceptance, and willingness to pay (WTP) for alternative water sources influence the implementation of innovative solutions to enhance water security (Bennett *et al.*, 2016; Hosking *et al.*, 2014; Cooper *et al.*, 2006). Stated preference techniques are widely used to assess public perceptions by asking individuals who may be affected by a resource change to express their preferences for the shift (Rolfe & Bennett, 2006). The application of stated preference evaluations using discrete choice experiments in the urban water supply includes improvements in water and sanitation services (Wang *et al.*, 2018; Dauda *et al.*, 2015; Hosking *et al.*, 2014), alternative water use (Thiam *et al.*, 2021; Amaris *et al.*, 2020; Lu *et al.*, 2019), and alternative water supply (Awad *et al.*, 2021; Day *et al.*, 2012; Haider & Rasid, 2002). However, few studies have evaluated households' WTP for alternative water sources, including seawater, for flushing toilets.

1.3 Project aims and objectives

1.3.1 Project aims

This project aims to assess the feasibility of using seawater to flush toilets in low-density coastal settlements. Therefore, this study seeks to answer the following questions:

- What is the environmental impact of using seawater compared with potable water for toilet flushing?
- What is the willingness to pay for alternative water sources for flushing toilets in households with access to waterborne sanitation?

1.3.2 Project objectives

The project objectives were as follows:

1) Task 1: Conduct a literature review to evaluate technical and regulatory barriers, enablers and opportunities for alternative water sources
2) Task 2: Conduct a literature review on assessing hybrid water supply using LCAs, including energy impacts.
3) Task 3: Conduct a high-level feasibility assessment of using seawater to flush toilets by modelling its environmental impact
4) Task 4: Conduct a scoping life cycle assessment (LCA) to compare the environmental impact of seawater and potable water as toilet-flushing water sources.
5) Task 5: Conduct a discrete choice experiment (DCE) to determine households' willingness to pay for alternative water sources for flushing toilets to reduce the demand for potable water.

1.4 Project limitations

The following research limitations were observed:

- The research investigation was limited to centralised sanitation systems and the waterborne sanitation level of service. Hence, the study focuses on flushing toilets but does not discern between the available toilet technologies, dual-flush, waterless, and vacuum, to name a few.
- Sanitation technologies such as septic tanks, pit latrines, waterless sanitation, urine diversion, and composting toilets are not part of the research scope.
- The research site was limited to coastal areas with marine outfall sewers as a form of disposal. In addition, the study area must be located within 30 km of the shoreline to ensure that flushing with seawater is viable (Liu *et al.*, 2016).
- Decentralised alternative water sources, such as greywater reuse and rainwater harvesting, have not been considered.
- Seawater desalination was not considered because this study sought to explore lower-quality water use for flushing toilets. Desalinated seawater is required for high-quality water requirements, such as drinking, bathing, and cooking.
- Access to primary data was limited; therefore, the comparative scoping of the LCA report was based on secondary data.

1.5 Report Outline

Chapter 1 introduces the research project, provides the necessary context, and discusses the rationale behind the study. It presents an overview of the ongoing debate and arguments on the research topic while highlighting the gaps this investigation aims to address. This subsection outlines the structure and organisation of the subsequent sections of this study.

- **Chapter 2: Review of Hybrid Water Supply Systems: Technical and Institutional Consideration.** A summary of peer-reviewed published research conducted to identify technical and regulatory barriers, enablers, and opportunities to apply alternative water use was established. The literature review fulfilled the requirements of Task 1.
- **Chapter 3: Review of the Application of Life Cycle Assessment to Alternative Water Sources.** This chapter presents comparative life cycle assessment (LCA) studies of various alternative water sources for potable and non-potable use, including reclaimed wastewater, greywater, rainwater harvesting, and seawater desalination. In addition, the methodological approach of LCA is discussed by considering system boundaries, product system lifecycle phases, sensitivity, and uncertainty analysis. The literature has been critiqued to expose knowledge gaps that require further investigation. The chapter delivers on the objectives of Task 2
- **Chapter 4: Review of the Application of Choice Modelling to Evaluate Preferences for Alternative Water Sources.** This chapter examines research on stated preferences for the willingness to pay for alternative water uses.
- **Chapter 5: Site Description:** A detailed description of the Hout Bay research site is included in this chapter.
- **Chapter 6: Life Cycle Assessment of Using Seawater To Flush Toilets In Coastal Areas.** The chapter aims to fulfil the requirements of Tasks 2 and 3. It discusses the procedure, including life cycle impact (LCI), life cycle impact assessment (LCIA), analysis, and interpretation. The LCI involves collecting data on inputs and impacts at each life cycle stage. The LCIA evaluates environmental impacts based on LCI data. The analysis and interpretation involve examining the results, identifying hotspots, and drawing conclusions.
- **Chapter 7: Using Discrete Choice Modelling to Determine The Willingness to Pay for Alternative Water Sources for Toilet Flushing**. The chapter details the choice modelling process, including the discrete choice experiment (DCE), sampling techniques, multinomial logit (MNL) model modelling, and willingness to pay. DCE was used to present the respondents with hypothetical choice scenarios. The chapter describes the sampling methods employed to gather data from a representative sample of respondents. The MNL was applied to analyse the collected data and estimate the relative importance of different attributes affecting choices. Finally, this chapter discusses the estimation of WTP by quantifying the monetary value individuals assign to specific attributes in the choice options. The objectives of Task 5 are presented in Chapter 7.
- **Chapter 8: Conclusions** This chapter highlights the primary outcomes of LCA and choice modelling analyses. This chapter discusses the significance of the project results with respect to the research questions.
- **Chapter 9: Recommendations**: Given the limitations of this project, this chapter suggests potential areas for future research.

2 Review of Hybrid Water Supply Systems: Technical and Institutional Considerations

2.1 Introduction

2.1.1 Water security

The Organisation for Economic Cooperation and Development (OECD) defines water security as managing four key risks. These include the risk of shortage and scarcity (including droughts), inadequate water quality, excess water (including floods), and inadequate access to safe water supplies and sanitation (OECD, 2021). Addressing these risks is essential to ensuring water security and safeguarding public health and well-being. Consequently, an integrated urban water management (IUWM) approach to mitigate water security is beneficial.

Sustainable urban water management encompasses concepts such as IUWM and total water cycle management, signifying its holistic approach to considering urban water systems' social, economic, environmental, and political dimensions (Van de Meene *et al.*, 2011). By integrating these aspects, sustainable urban water management seeks to address the complex challenges associated with water supply, wastewater treatment, stormwater management, and water resource conservation in urban areas. It recognises the interconnectedness of these factors and emphasises the need for integrated and adaptive strategies to ensure long-term sustainability and resilience in urban water systems.

An urban water system (UWS) incorporates the management of infrastructure related to water supply, treatment, and disposal within urban settlements. This constitutes a wide range of interconnected aspects, including the provision of water supply, effective treatment of wastewater, and overall management of water resources in urban areas (Schramm *et al.*, 2018). This holistic approach recognises the intricate interplay between the different components of the urban water system. In addition, it underscores the importance of integrated planning, sustainable practices, and efficient resource management to ensure the availability and quality of water for urban populations while minimising environmental impacts.

2.1.2 Centralised urban water systems

A centralised water supply system is a networked infrastructure in which water is sourced, treated, and distributed from a central location to multiple users. Conventional urban water systems typically source raw water from orthodox sources such as surface water and groundwater. In addition, they usually involve large-scale water treatment plants, storage reservoirs, and extensive networks of pipes for water distribution. Utility companies or municipal authorities generally manage and maintain centralised systems, ensuring a reliable and regulated water supply to meet urban or large-scale developments' potable and non-potable needs.

Climate change and anthropogenic activities have affected the availability and quality of conventional water sources. Consequently, it is necessary to consider alternative and unorthodox water sources to improve water security (Scruggs and Heyne, 2021). Focusing on domestic end use, hybrid water supply systems enable the end use of water that is fit for potable (drinking, bathing, and cooking) and non-potable (irrigating lawns and flushing toilets) purposes.

2.1.3 Aim of this chapter

This chapter focuses on alternative water sources for flushing toilets in centralised sanitation systems. Therefore, it aims to evaluate the implementation of alternative water sources for centralised urban water systems by considering technical and regulatory aspects. This chapter considers regulatory capacity as the institutional capacity to govern hybrid urban water systems. Technical elements are related to conceptual planning and infrastructure operation and maintenance. Ultimately, this chapter highlights the barriers, enablers, and opportunities for implementing hybrid water supply systems.

2.2 Alternative water sources

Furlong *et al.* (2022) assert that the degree to which hybrid UWS are successfully implemented is influenced by cost, environmental impact, governance, and community perceptions. They suggested that the difficulty of implementation reduces once the concept becomes more familiar to end users. Table 2-1 was adapted from the study by Furlong *et al.* (2022), summarising the water source and end-use combinations for potable and non-potable uses for domestic purposes. Stormwater and rainwater harvesting tanks rarely augment conventional potable water supply sources. Warrnambool, Australia, is an example of a rain tank connected directly to a dam (Furlong *et al.*, 2022).

Table 2-1 Water source and end-use combinations (Furlong *et al.*, 2022 **adapted)**

End-use	**Surface water**	**Groundwater**	**Desalinated seawater**	**Recycled water**	**Urban stormwater capture**	**Rainwater tanks**
Domestic drinking use	Very common		Becoming common	Emerging	Rare	Rare
Domestic non-potable use	Commonly sourced from the same source as drinking water			Some examples		Common

Although socio-economic and environmental considerations are necessary for implementing a hybrid UWS, the impact of climate on rainfall cannot be discounted. Resilience to climate change is imperative for ensuring the water security and sustainability of UWS. Furlong *et al.* (2022) mapped alternative water sources to centralised and decentralised UWS by considering resilience to climate change (see Figure 2:1). Water sourced from dams, rainwater tanks, and stormwater depends on rainfall and is unsustainable during droughts.

Indirect potable reuse (IPR) involves treating wastewater to a high standard. Subsequently, it is used to recharge groundwater aquifers or is released into a surface water body, such as a reservoir or lake. This type of system is implemented on a regional scale to allow for integration with other water resources and, to a large extent, is independent of rainfall. On the other hand, Direct potable reuse (DPR) is a more advanced method. It involves treating wastewater to a very high standard so that it can be safely introduced directly into the drinking water distribution system without the intermediate step of recharging a groundwater aquifer or surface water body. In both cases, the end use was potable consumption. This hybrid system is typically implemented at the municipal level and requires a dual-pipe system. It is resilient to the effects of climate change, but not more so than desalinated seawater. Seawater is independent of rain and drought and can be a source of potable water.

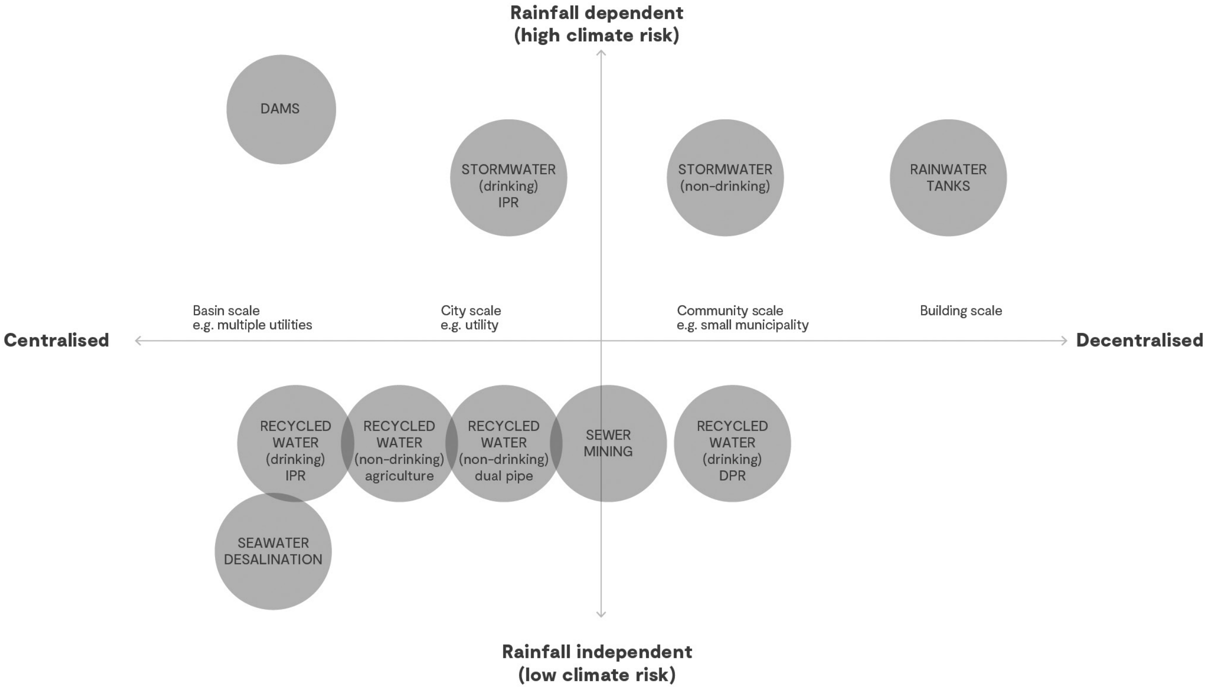


Figure 2:1 Rainfall independence of alternative water sources mapped against centralisation/decentralisation infrastructure (Furlong *et al.*, 2022)

Furlong *et al.*(2022) did not consider other alternative seawater uses. For several decades, partially treated seawater has been used as an alternative for non-potable uses, such as flushing toilets in Hong Kong (Tang *et al.*, 2006). The following subsections discuss hybrid UWS, where low-quality water sources such as recycled water and partially treated seawater are used for non-potable purposes.

2.2.1 Recycled water

Research shows that public acceptance is integral to successfully implementing hybrid WSS (Bichai *et al.*, 2011; Quezada *et al.*, 2016; Van de Meene *et al.*, 2018). For example, in Australia, recycling schemes, including dual-reticulation systems for non-drinking use, have been increasingly implemented in recent years (Bichai *et al.*, 2018). Nevertheless, the acceptance of recycled water is the main barrier to its adoption by households.

Owen and Chitonge (2022) argued that although water recycling has the potential to contribute towards alleviating water scarcity, studies have shown that public perceptions greatly influence the outcome of any water recycling scheme. Their study conducted surveys to determine the public perceptions of water recycling in informal settlements. One of the key findings of this study was the residents' trust in municipal competencies and systems. In addition, the study also found that public perceptions of tariffs influence decisions about water recycling and that reduced tariffs are not perceived as an incentive for acceptance. Further, safety concerns related to health and the "yuck factor" were dominant among those who expressed concerns about using recycled water for drinking and cooking. Owen and Chitonge (2022) proposed that rolling out a water reuse scheme starting with affluent areas increases the likelihood of acceptance among low-income communities.

Li *et al.* (2020) used evidence from a meta-analysis to investigate the social attributes of proponents of recycled water and their willingness to accept it. A meta-analysis is a quantitative literature review method whose data originate from the existing literature. Its limitation is that few authors have presented original data in the published literature. Therefore, the analysis factors are inadequate because of the lack of necessary data and strict screening conditions for meta-analyses. Nevertheless, According to the study, sociodemographic factors such as age, gender, and education level were salient variables influencing the public's willingness to accept recycled water. Younger women were more likely to accept recycled water use than older men, and individuals with higher education were more willing to accept recycled water.

Semasinghe *et al.* (2023) surveyed how information affects general attitudes toward desalinated seawater and recycled water in Australia. They found that the perception of climate change, combined with the perception of water scarcity, is a significant and influential factor that supports alternative potable water sources. Similarly, Dolnicar and Hurlimann (2011) surveyed to ascertain the sources of information factors influencing people's attitudes towards alternative water sources. This study identified various information sources that shaped respondents' attitudes towards water-related issues. The most frequently identified sources of influence were research findings (88% of respondents), followed by personal experience of water shortages (86%), consideration for future generations (84%), and advice from friends and family (83%). Other sources of influence include government campaigns, media coverage, and water utility information.

This study also explored the impact of education on the effectiveness of influential factors. It was found that for most sources of information, no statistically significant difference existed across educational groups. Notably, the study found that respondents with higher educational levels perceived the government to be more influential. Similarly, scientists were found to be more influential for respondents with a university education. Respondents without university degrees were more likely to state that nobody influenced them.

2.2.2 Partially treated seawater

Approximately 80% of the population of Hong Kong is supplied with seawater for flushing (Yue and Tang, 2011). This study explains that seawater is screened for solids removal and disinfected with chlorine or hypochlorite before being pumped into service reservoirs for distribution. Seawater for flushing toilets is provided free of charge only to households located near the seashore. Proximity to the seashore saves energy because seawater is extracted near consumer households. In contrast, most freshwater supplies in Hong Kong are pumped over long distances and undergo sophisticated treatment processes (Yue and Tang, 2011).

Tang *et al.* (2006) investigated the feasibility of using seawater, surface water, and recycled water for toilet flushing in various districts of Hong Kong. The net present value (NPV) method was used to compare the engineering costs of the cases. The NPV was calculated based on the capital cost, operation and maintenance costs, and the cost of water resources. Tang *et al.* (2006) found that using recycled water for toilet flushing costs more than using seawater or surface water. This finding can be attributed to the additional treatment process required to improve the treated effluent quality from wastewater treatment facilities. Moreover, the study found that dual water supply systems using seawater for toilet flushing had the best engineering economy, followed by surface water and recycled water. However, the study noted that using recycled water may become more cost-effective than using surface water as a demand management strategy if the price of surface water increases.

A few disadvantages of using seawater to flush toilets were reported by Tang *et al.*(2006). These include 1) the potential for seawater to corrode the plumbing system and appliances and 2) the potential for seawater to cause odours and blockages in the sewage system. Therefore, the study suggests that using seawater for toilet flushing may require installing a separate plumbing system to prevent corrosion. Proper sewage system design and maintenance to eliminate odours and blockages is pivotal.

However, since the 1950s, Hong Kong has used seawater as an alternative water source to flush toilets, independent of the urban water cycle (Liu *et al.*, 2019). Moreover, Liu *et al.* (2019) found a potential 20-30% saving in using potable water when using seawater as an alternative to flushing toilets. However, they observed that flushing toilets with seawater was environmentally unfeasible for population densities less than 3 000 people/km^2 and greater than 12 000 people/km^2.

It is unclear how the disadvantages Tang *et al.* (2006) noted were managed recently. However, according to WSD (2020), the seawater supply network now covers 85% of the Hong Kong population with toilet flush water. It conserves 24% of the potable water demand. The demand for toilet flush water increased from 0.71 to 0.87 ML/d for 2016-2020. Consequently, using seawater to flush toilets has allowed Hong Kong to manage the demand for potable water supply by 3.6% instead of 7.3% (WSD, 2020). Hong Kong is actively upgrading its seawater wastewater treatment facilities to include seawater recycling for toilet flushing and supplying new developments in seawater for flushing toilets (WSD, 2020). Despite Hong Kong's success, studies on using seawater to flush toilets in other parts of the world have not yet been conducted.

2.3 Institutional

Bichai *et al.* (2018) investigated the barriers to water recycling as an innovative system for water security in arid countries. The study included three arid countries: Australia, the United Arab Emirates (U.A.E.), and Jordan. In addition, the study aimed to analyse the integration of water recycling options into the water security strategies of these countries and understand the contextual evolution of water recycling technologies in different settings. The main findings include challenges in efficient standard operating practices, legal enforcement challenges in the Middle East, and a lack of legal mechanisms for controlled implementation in Australia. Additionally, fragmented institutions that regulate water recycling can act as chokepoints in innovation systems, particularly in Australia. In contrast, the study found that Jordan faced less stringent barriers to water recycling development stages due to its wider infrastructural and institutional gap (Bichai *et al.*, 2018).

Van de Meene *et al.* (2011) found that a lack of understanding and awareness of the governance approaches required to support and enable the transition to sustainable practices was the main barrier to innovating traditional centralised urban water governance methods. These findings agree with those reported by Bichai *et al.*(2018). Furthermore, Schramm *et al.* (2018) surmised that the limitations in the ability of governance structures to adapt might be one reason for the stagnation in implementing novel technologies and concepts in urban water systems.

Recycled water projects require cooperation between wastewater and water agencies and collaboration between wholesale recycled wastewater supply and distribution departments (Mills and Asano, 1996). In their review of 19 recycled water projects in California, USA, Mills and Asano (1996) identified various institutional issues related to revenue and duplication of services. For example, conflict arose from the duplication of services when recycling water was provided within the jurisdiction of the water utility. The conflict escalated to a lawsuit whose foundation was to deliver recycled water instead of potable water, resulting in reduced revenue for the water utility. Therefore, a decrease in revenue affects the water utility's ability to invest in the water infrastructure for potential recycled water users. Another example of pricing competition in this study is when a utility invests in providing recycled water to become a regional supplier. Consequently, another utility company invests in recycled water supply to beat the regional price. Therefore, to prevent such conflicts, early planning should involve seeking a cooperative agreement that includes a fair revenue-sharing arrangement (Mills and Asano 1996).

2.3.1 Governance

Transition management provides a governance-oriented approach to enhancing urban water management by addressing governance challenges and facilitating improved decision-making processes. It is a governance approach that emphasises the need for and importance of small-scale experiments to address persistent societal problems (Porter *et al.*, 2015). The underlying assumption of transition management is that it is possible to subtly influence the direction and pace of transitions through different interventions at different levels using various instruments (Porter *et al.*, 2015). In the transition management governance framework, four types of governance activities relevant to societal transitions are identified: strategic, tactical, operational, and reflexive (Porter *et al.*, 2015). By incorporating transition management principles, cities can cultivate enhanced governance structures, encourage stakeholder participation and establish coherent policies.

An example of transition management is the proposal developed by the Cooks River Friends Working Group (CRFWG), a group of community members, academics, and government officials interested in improving the water quality of the Cooks River catchment. It aims to embed sustainable urban water management practices within the Cooks River catchment through improved governance arrangements (Bos and Brown, 2012). The Cooks River Sustainability Initiative is a bottom-up experimental governance process that took place over 10 years (2002-2011) and resulted in development of new governance rules and structures to support sustainable water practices at the political level (Bos and Brown, 2012). Figure 2:2 illustrates the phases of governance experimentation leading to the adaptation of water governance structures. The alliance brought together various stakeholders in urban water management. It helped to build a base for alternative forms of urban water management. In addition, it has been endorsed and funded by eight municipalities and has been operational since late 2011. Its mandate is to support sustainable practices in the Cooks River catchment. It provides a platform for collaboration and knowledge sharing between stakeholders, including local government, community groups, and water utilities (Bos and Brown, 2012).

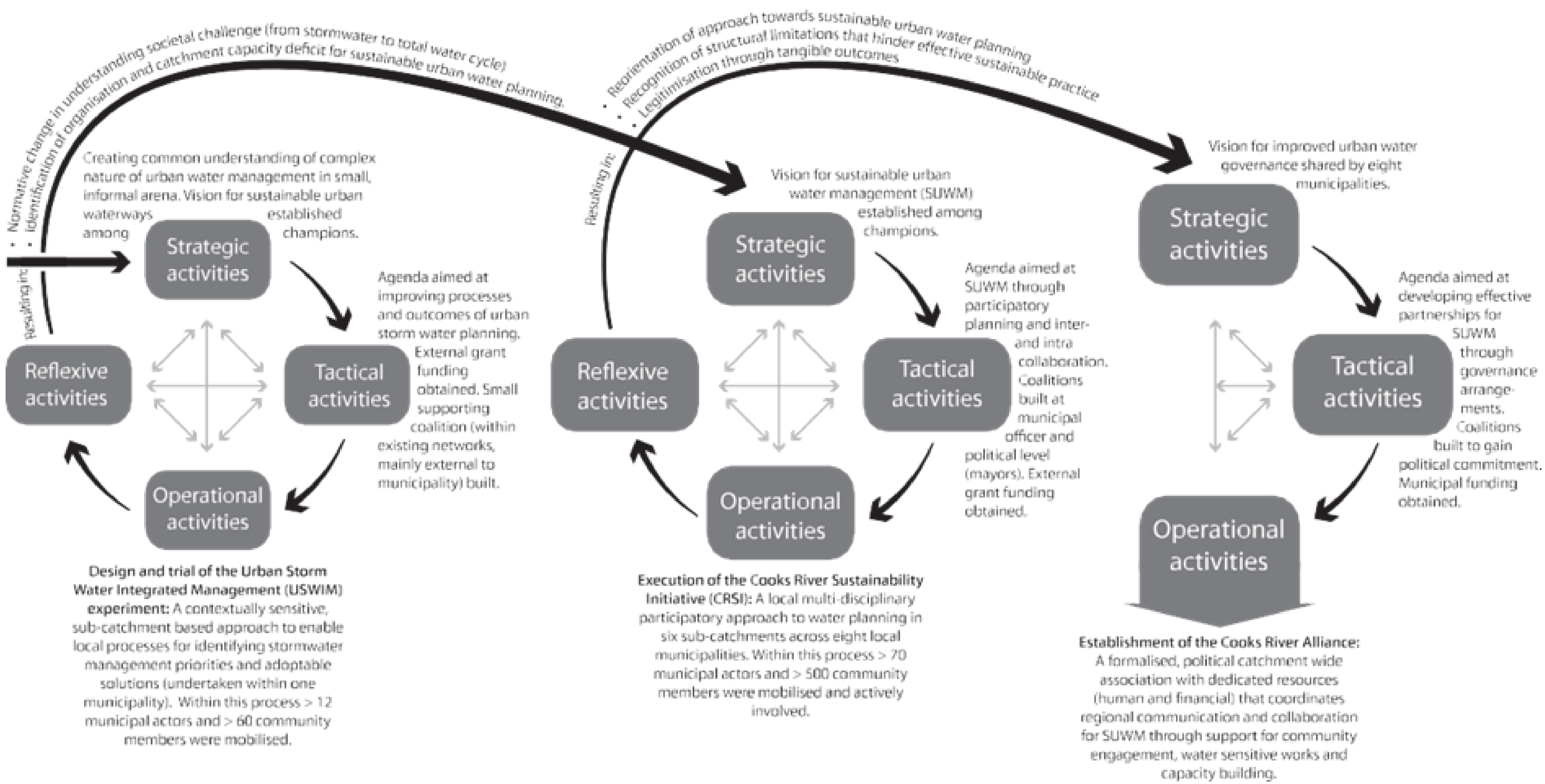


Figure 2:2: Phases of governance experimentation leading to adaptation in water governance structures in the Cooks River Catchment, Sydney, Australia (Bos and Brown, 2012)

Bos and Brown (2012) considered governance experimentation a critical factor in achieving a socio-technical transition in the urban water sector. Government experimentation refers to innovation and experimentation with governance approaches that aim to alter the configuration of decision-making, which raises issues of accountability and legitimacy. Bos and Brown (2012) proposed that governance experimentation enables social learning, which is necessary for realising sustainable urban water management practices. Additionally, new governance rules and structures can be developed and tested through governance experimentation, creating social and political capital that can change an established water governance framework.

The WATERgraafsmeer (WGM) program in Amsterdam is another example of transitional management in response to better governance of the IUWM. It was initiated in January 2010 by local municipalities to experiment with a new mode of governance to facilitate the transition to sustainable Watergraafsmeer (Porter *et al.*, 2015). Program coordinators invited local institutions, businesses, and individuals to explore sustainable measures and new business cases (Porter *et al.*, 2015). Furthermore, communities of practice (CoPs) have been established to deal with participation, knowledge and training, sustainable entrepreneurship, urban design, business cases, housing corporations, the urban water cycle, and sustainable area development. These guidelines may help design and evaluate transition experiments in the WATERgraafsmeer or other similar programs.

In response to forming integrated governance to overcome some of the management barriers discussed in Section 2.2.1, Australia instituted the Office of Living Victoria (OLV) to increase collaboration and integrated planning. The key drivers for accelerated integrated urban water management (IUWM) are the millennium drought and the environmental and social problems (Furlong *et al.*, 2016). However, it is unclear whether a transition management framework is used. Nonetheless, implementing the IUWM was supported by several critical factors, including adequate funding, institutional and individual capacities, and the government. These combined elements have fostered innovative responses and generated momentum for the successful implementation of IUWM (Furlong *et al.*, 2016). While Furlong *et al.* (2016) do not mention the specifics of these initiatives, OLV is said to have introduced governance reforms and initiatives to promote collaboration and integrated planning.

2.4 Technical

A comparative life cycle assessment (LCA) study on alternative water supply sources by Liu *et al.* (2019) suggested that electricity consumption contributes to over 80% of the environmental impacts in the urban water system. Therefore, the fuel mixes in electricity significantly influence their environmental performance. Consequently, this section focuses on energy use during the operating and maintenance life cycle stages. It highlights conceptual planning considerations for hybrid systems.

2.4.1 Conceptual planning

Mills and Asano (1996) conducted a retrospective review of 19 water recycling projects in California, USA. This study aimed to measure whether the goal of offsetting freshwater demand to augment the water supply through recycled water was achieved. They found that defining the recycled water market was critical for the success and sustainability of recycled water projects. Potential users commonly show a favourable interest in using recycled water during the early planning stages (Mills and Asano, 1996). However, there is no assurance that these users will purchase recycled water when it becomes available (Mills and Asano, 1996).

Furthermore, access to reliable data to assess recycled water yield affects the conceptualisation of the project and anticipated revenue collection. The required data relate to the volume of wastewater that can be recycled, the potential demand for recycled water, and the fluctuations in these flows and demands on a monthly, daily, and hourly basis. Poor estimates of water demand generally account for lower deliveries to users than anticipated (Mills and Asano, 1996). In addition, the reliability of water flow data is significantly affected by climate change. For instance, Singapore is likely to experience a one-meter rise in sea level by 2100, and rainfall increases by 25% from 2070 to 2099 (Ng and Teo, 2020). Consequently, since 2017, Singapore has adopted greater rainfall intensity and higher sea levels as design parameters for future drainage projects.

Mills and Asano (1996) also found that retrofitting costs are a significant factor that utilities frequently overlook. The assumption is that the cost is minor and covered by the end user. Therefore, retrofitting is not always economically feasible. To overcome the retrofitting cost barrier, Scruggs and Heyne (2021) proposed transitioning to multiple options from a suite of those available based on what is feasible for a given community. In addition, as the UWS infrastructure ages, it could be replaced with more sustainable options similar to those used for newer developments.

2.4.2 Operation and maintenance

The impact of relying heavily on unconventional alternatives for water supply mixes and the resulting electricity intensity are poorly understood (Stokes-Draut *et al.*, 2017). Therefore, Stokes-Draut *et al.* (2017) conducted a study to explore the electricity intensity of evolving water supply mixes in California's water network and to identify potential solutions to reduce electricity intensity. Table 2-2 summarises California's urban water mix energy demand for 2010. The desalination of brackish groundwater and seawater yielded the highest energy intensity. Compared with recycling water for portable use, non-potable use requires less energy.

Table 2-2: Contributors to California's urban water mix (Stokes-Draut *et al.*, 2017 **adapted)**

Water source	**Electricity intensity (kWh/m^3)[a]**
Groundwater	0.15-0.52
Surface water	0.072-0.23
Stormwater capture: non-potable	4.1
Recycled water: non-potable	0.29-1.1
Recycled water- groundwater augmentation	0.53-1.5
Desalination: brackish groundwater	0.47-1.4
Desalination: ocean	3.0-3.5

[a] Typical range for California's potable water, unless specified, includes an electricity supply for conveyance and treatment. All electrical intensities depend on the source and quality of water. Distribution of electricity is excluded as it is more affected by topography.

[b] Only one stormwater capture system was evaluated in the study. Thus, the range is not provided

The main finding was that the electricity intensity of California's urban water mix varies depending on the specific water source and site-specific conditions. They also found a significant potential to reduce the electricity intensity of water supply mixes in California through a combination of measures. These measures include increasing the use of renewable energy sources, improving energy efficiency in water treatment and conveyance, and optimising water supply mixes to reduce reliance on energy-intensive sources (Stokes-Draut *et al.*, 2017).

Singapore has implemented an integrated water-supply system that includes recycled water and desalinated seawater for potable use through technological expertise, government support, and public acceptance (Ng and Teo, 2020). Despite its success, one of the main challenges is the high energy consumption required for the water treatment processes. Approximately 0.2 kWh/m^3 of energy is required to treat rainwater, compared to 1 kWh/m^3 to treat recycled water to potable standards and 3.5 kWh/m^3 to make seawater drinkable (PUB, 2018). Consequently, Singapore is investigating alternative treatment processes and technologies to reduce the energy demand of water supply systems (Ng and Teo, 2020). These include electrode ionization, biomimicry, and low-energy ultra-permeable membranes (Ng and Teo, 2020).

Nichols (2006) compared different energy sources for alternative water supply sources. This study considered six alternative systems for the development of Sydney's water supply system until 2050: dam (not required), deep wells west of Sydney (coal-fired), desalination plant (renewable energy), desalination plant (nuclear), desalination plant (coal-fired), and water recycling plants (coal-fired). The dam option was not required because the state government of Australia had shown an interest in desalination to meet future water demand. Nichols (2006) posed that the strategy to augment water supply with desalinated water introduces a significant number of complexities in planning a response to a major disaster, such as an earthquake, and may lead to a reduction in the ability of the city to survive such a disaster.

2.5 Conclusion

Hybrid urban water systems provide resilience to ensure water security by integrating conventional and alternative water-supply sources. Water sources independent of rainfall and drought, such as desalinated seawater, partially treated seawater, and recycled water, offer increased water security. Opportunities to realise improvements in water security require an integrated urban water management approach through transitional management. By incorporating transition management principles, cities can cultivate enhanced governance structures that allow experimentation, encourage stakeholder participation by creating social and political capital, and establish coherent policies that promote collaboration and integrated planning.

Facilitating the effective integration of hybrid urban water systems involves enhancing the utilisation of renewable energy sources, refining energy efficiency in water treatment and distribution, and optimising the combination of water sources to decrease dependence on energy-intensive options.

3 Review of the Application of Life Cycle Assessment on Hybrid Water Supply Systems

3.1 Introduction

Before introducing alternative supplies, evaluating the sustainability implications of their use would be beneficial. Life cycle assessment (LCA) is an environmental evaluation technique that systematically accounts for all the inputs and outputs of a product or system from "cradle to grave" and subsequently calculates their potential environmental impacts (Finkbeiner *et al.*, 2006). As a result, LCA modelling has been applied to quantify environmental performance from raw material acquisition to resource consumption and, finally, disposal through the assessment of emissions and consumption of resources throughout the life cycle (Del Borghi *et al.*, 2013). The LCA framework has proven helpful in identifying potential hotspots, improving systems, and quantifying environmental burdens (Pillay *et al.*, 2007).

An essential characteristic of LCA is its ability to impartially compare potential alternatives for similar purposes. This characteristic enables an unbiased environmental assessment of trade-offs to be conducted. LCA was first applied to water supply in the 1990s, leading to numerous comparative LCA studies on potable water systems. The studies include fresh water, seawater, reclaimed water and groundwater either at a centralised (Chen *et al.*, 2012; Lyons *et al.*, 2009; Hsien *et al.*, 2019; Hsien *et al.*, 2019; Liu *et al.*, 2019; Liu *et al.*, 2021) or decentralised scale (Godskesen *et al.*, 2013; Opher and Friedler; 2016, Opher *et al.*, 2018, Godskesen *et al.*, 2013).

3.1.1 Aim of this chapter

This chapter summarises the methodological application of LCA studies in urban water supply by considering system boundaries, product system lifecycle phases, sensitivity, and uncertainty analysis. This review finds studies focusing on the environmental impacts associated with the life cycle of water-supply systems. It differentiates the studies considering a cradle-to-tap view from those that evaluated the holistic cradle-to-grave perspective.

3.2 Methodological Approach

3.2.1 System boundary

The selection of the system boundary is determined by an LCA practitioner based on its goal (ISO, 2006). For water systems, the 'cradle-to-grave' system boundary begins from the point of water abstraction to treatment, distribution, wastewater collection, and wastewater treatment until its end of life, when it is disposed into receiving water. However, only a few studies have considered the disposal stage (Lane *et al.*, 2016). Kobayashi *et al.* (2020) assumed that the water released into the environment after wastewater treatment in the evaluated scenarios had a similar composition and eliminated it from the study. However, if wastewater is not treated before it is discharged into the environment, this assumption does not hold. Lane *et al.* (2015) found that disposal of untreated wastewater via a marine outfall sewer (MOS) significantly impacts marine eutrophication and ecotoxicity.

Therefore, system boundaries should be tactically selected to align with the study's goal while being cognisant of resource and time constraints (Matthews *et al.*, 2014).

3.2.2 Life cycle phases

A water system has three life cycle phases: construction, operation, and decommissioning. Among the three lifecycle phases of water systems, the operational stage is the most influential owing to the chemical and, to a more significant extent, the energy requirements associated with various stages of water systems. Thus, it was included in the majority of studies. In contrast, construction and end-of-life or decommissioning stages are sometimes excluded, as they are considered to have a negligible impact relative to the system's operational phase (Vince *et al.*, 2008).

3.2.2.1 Construction Lifecycle Phase

There is yet to be a consensus on the contribution of infrastructure construction to the environmental impacts of water systems because of varying opinions regarding their significance in water systems (Xue *et al.*, 2019). For instance, several water supply LCAs exclude the construction phase, citing a minor contribution to the overall impacts compared to the operation stage (Friedrich, 2002; Vince *et al.*, 2008; Barjoveanu *et al.*, 2014). Furthermore, the contribution is assumed to be minimised by a long construction service life of 50–100 years (Zhou *et al.*, 2014). In contrast, Jeong *et al.* (2015) found that the construction phase of the surface water system significantly contributed to environmental impacts compared with the operational phase. However, a review by Corominas *et al.* (2013) suggested that the contribution of smaller treatment plants to the overall impact is much higher than that of larger plants; thus, it should not be overlooked in assessments.

In some comparative LCAs, greywater systems have contributed significantly to various impact categories. For example, the construction of a rotating biological contactor (RBC) to treat greywater has been associated with an approximately 20% contribution to freshwater, human, and marine ecotoxicity impacts and an approximately 91% contribution to metal depletion (Opher and Friedler, 2016). Similarly, the construction of membrane bioreactor (MBR) units has been linked to approximately 40-90% impacts on human health, carcinogenic potential, global warming potential, and eutrophication (Kobayashi *et al.*, 2020). Furthermore, Opher and Friedler (2016) attributed the impacts of ecotoxicity on electricity production and metal depletion to the use of steel in reinforced concrete and reticulation pipelines.

The high contribution of construction to the impact of LCAs may be attributed to using average database values for construction infrastructure components such as water treatment plants or reservoirs (Xue *et al.*, 2019). This approach can lead to overestimations during the inventory analysis phase, particularly when variations in geography and construction methods are not adequately considered (Lane *et al.*, 2015). Xue *et al.* (2019) recommend prioritising primary data from water utilities over average database values, as they offer a more precise and reliable representation of the system under study compared to estimates derived from the literature or databases. Given the potential impact of construction on future LCA models of water systems, the construction stage should be considered.

3.2.2.2 Operational Lifecycle Phase

The operational phases of a product system include chemicals, chemical transportation, energy, and maintenance (Lemos *et al.*, 2013). However, maintenance and chemical transportation are considered negligible contributors to this impact (Buckley *et al.*, 2009; Vince *et al.*, 2008).

Electricity is generally acknowledged as the primary contributor to the impact during the operation stage. Fossil-fuel-based electricity has the highest environmental impact compared to other sources (Raluy *et al.*, 2005; Tarpani *et al.*, 2021). One study suggested that fossil-fuel-based electricity results in higher greenhouse gas emissions and, thus, a higher climate change potential than other sources (Raluy *et al.*, 2005). For example, the climate change potential of desalination using a Brazilian electricity mix (63% hydropower) is nine times that of a South African mix (92% coal) (Tarpani *et al.*, 2021). Therefore, countries that rely on fossil fuels for electricity generation are bound to contribute more to greenhouse gas emissions than those that use cleaner sources (Tarpani *et al.*, 2021). However, the electricity mix in an area depends on the available natural resources. Hence, no energy source may be deemed the most universally feasible across regions because of resource availability and technological advancement variance.

The complete eradication of fossil fuels may be impossible owing to their wide availability and cost relative to other sources, most notably in developing countries (Nalule and Mu, 2020). However, studies have shown that reducing the contribution of fossil fuels to the energy mix may improve the environmental performance of alternative water supplies by lowering their environmental burden (Liu *et al.*, 2016; Tarpani *et al.*, 2021). For example, Lemos *et al.* (2013) replaced an electricity mix A (34% natural gas, 23% hard coal, 15% hydropower, 12 % wind power, 8% oil, 4% hydropower) with mix B (33% natural gas,9% hard coal, 28% Hydropower,12% wind power 2% other renewable energy) to assess the impact of fossil fuels. Their findings showed a 32 -72% reduction in air emissions.

Lane *et al.* (2015) found that chemical consumption considerably affects the operational stage. However, Vince *et al.* (2008) found that the impact of energy consumption exceeds that of chemical consumption. As a result, some studies have not specified the contribution of chemicals to this impact.

3.2.3 Lifecycle Impact Assessment (LCIA)

The life cycle impact assessment (LCIA) phase investigates the contribution of the product system's inputs and outputs to environmental issues relevant to the study (Rosenbaum *et al.*, 2018). The three mandatory stages in the impact assessment phase are as follows:

1) the selection of impact categories relevant to the study,
2) classification where impact categories are linked to inventory data and,
3) the quantification of impact, a stage known as characterisation.

International Standards Organisation (ISO) 14040 defines an impact category as a "class representing environmental issues of concern to which life cycle inventory (LCI) analysis results may be assigned" (ISO, 2006). As such, several LCIA methods originating from different countries may be utilised. These include Traci (USA), CML, Ecoindicator 94, Ecoindicator 99 (Netherlands), and Recipe (Netherlands) (Rosenbaum *et al.*, 2018; Acero *et al.*, 2016). These methods have different impacts; thus, one should choose a technique relevant to their study. Moreover, the choice of impact categories to evaluate in one's assessment should be informed by the goal and scope of the study (ISO, 2006).

Midpoint impact assessment methodologies are commonly preferred over endpoint damage-oriented methods owing to their lower uncertainty (Bare *et al.*, 2000). When selecting an impact assessment methodology, it is essential to tailor it to the intended audience and the goals of the LCA study. Existing life cycle impact assessment (LCIA) methodologies, widely used in developed countries, may not fully capture the impacts relevant to other geographies (Leske & Buckley, 2003). Goedkoop *et al.* (2008) proposed that the Recipe Midpoint offers a more comprehensive global perspective, making it particularly suitable for developing countries' geographies.

3.2.4 Uncertainty analysis

The quality of an LCA output depends on the data used (Weidema & Wesnæs, 1996). Because LCAs are data-intensive, numerous potential sources of uncertainty originate from the model used. Uncertainty can arise from measurement errors during data collection and the selected impact assessment methodology (Xue *et al.*, 2019). In addition, combining data sources, including direct measurements and database values, as is the norm in the inventory analysis phase, could also be a source of uncertainty. Buckley *et al.* (2009) conducted their study according to the data quality requirements prescribed in ISO 14044. By contrast, Goga *et al.* 2019, Lam *et al.* 2017, Liu *et al.* 2016, and Opher and Friedler 2016 neither conducted an uncertainty analysis nor specified the quality of the data utilised in their inventories based on the ISO requirements. However, Buckley *et al.* (2009) acknowledged reduced transparency due to data gaps.

3.2.5 Sensitivity analysis

Sensitivity analysis was used to assess the most influential parameters of a study and evaluate their effects on the impact assessment results. Most studies have considered water system components, including pipeline materials, water mix, electricity production, and consumption, without considering that the water source and environmental impacts are sensitive to changes in electricity mixes (Zhou *et al.*, 2011; Opher and Friedler, 2016). Environmental impacts can be significantly reduced using renewable energy sources or low-carbon electricity mixes (Tarnacki *et al.*, 2012). For instance, Zhou *et al.* (2011) found that switching from a coal-based electricity mix to a nuclear-based blend resulted in a 70% reduction in environmental impact.

Liu *et al.* (2016) conducted a sensitivity analysis of water provision in coastal areas, focusing on physical conditions such as distance from the shoreline, distance from a freshwater abstraction source, and effective population density. Their study aimed to evaluate the impact of these factors on water availability and provision in the coastal regions. The effective population density had the highest impact on the LCA results of using seawater to flush toilets; for all scenarios, the consequences decreased as the effective density increased for density below 12 000 persons/km^2. Furthermore, they found that the impact increased with increasing distance from the sea because of the increased pipeline length required for seawater abstraction and wastewater collection. In another study, Liu *et al.* (2019) evaluated the sensitivity of growing water demand, the amount of water used for toilet flushing, and energy recovery from wastewater treatment to the impacts of climate change and the replacement of coal with natural gas and nuclear power. However, they found additional parameters to make a minimal contribution to the environmental effects.

3.3 Comparative LCA Studies on Alternative Water Sources- Cradle-to-Tap

The cradle-to-tap approach considers water extraction from sources such as rivers, lakes, and groundwater aquifers. It includes the necessary infrastructure and processes to treat water to make it safe for drinking or other uses. The environmental impact of unconventional sources such as greywater reuse, water recycling, rainwater harvesting, stormwater harvesting, and seawater has also been modelled using LCA.

3.3.1 Potable use

Several LCA studies have indicated that desalination contributes between 50% and 80% of the total impacts in a water supply system (Tarpani *et al.*, 2021; Liu *et al.*, 2019; Liu *et al.*, 2016; Del Borghi *et al.*, 2013). Overall, desalination has the highest impact across all impact categories, owing to energy consumption and production (Goga *et al.*, 2019; Hsien *et al.*, 2019; Liu *et al.*, 2019; Shi *et al.*, 2019; Liu *et al.*, 2016; Loubet *et al.*, 2014; Zhou *et al.*, 2014; Lyons *et al.*, 2009; Vince *et al.*, 2008). However, Tarpani *et al.* (2021) found that indirect potable reuse through aquifer recharge has a much higher impact than desalination. Nevertheless, energy consumption highly depends on the technology used and the treated water quality (Goga *et al.*, 2019).

Mannan *et al.* (2019), Tarnacki *et al.* (2012), and Raluy *et al.* (2006) compared different desalination methods, including multistage flush (MSF) and reverse osmosis (RO). The analyses found that RO is the most environmentally friendly desalination method because of its lower energy consumption than other methods. Furthermore, there was no significant difference in the environmental impact when RO was compared with long-distance water transfer from a surface water source. Moreover, including the construction phase negates the lower energy consumption of the transferred surface water.

Another study by Tarpani *et al.* (2021) concentrated only on the water production and supply stages and found that RO is more environmentally unfavourable than indirect potable reuse through managed aquifer recharge and rainwater harvesting. Similarly, other studies have compared the impacts of potable water systems produced from different water sources. Godskesen *et al.* (2013) compared the environmental effects of surface water, stormwater harvesting, groundwater, and seawater desalination. They found that seawater desalination had the highest environmental impact, followed by groundwater, surface water, and stormwater harvesting. Considering the significant effect of freshwater withdrawal owing to high levels of groundwater abstraction, the impacts of desalination, stormwater, and rainwater harvesting, which do not involve water withdrawal from the natural environment, were comparatively lower.

3.3.2 Non-potable use

Greywater is collected separately from showers, baths, and hand basins and can be treated and recycled for non-potable use (Kobayashi *et al.*, 2020). Various chemical and biological greywater treatments have been compared using LCA, including rotating biological contactors (RBC), sequence batch reactors (SBR) (Yoonus & Al-Ghamdi, 2020), membrane bioreactors (MBR) (Kobayashi *et al.*, 2020; Lam *et al.*, 2017), constructed wetlands (CW) (Kobayashi *et al.*, 2020), and anoxic fluidised membrane bioreactors (AFMBR) (Lam *et al.*, 2017).

In a study by Opher and Friedler (2016), the effects of different wastewater treatment approaches were compared. These include 1) activated sludge treatment with no reuse, 2) wastewater reuse for gardening and toilet flushing, and 3) RBC greywater treatment and reuse. The study examined these approaches on a smaller scale involving 40 households and a larger scale involving 320 households, with the latter being designated for toilet flushing and gardening purposes. The results indicated that on a scale of 320 households, greywater treatment and reuse demonstrated the least significant impacts, followed by the 40-household scale and wastewater reuse. Conversely, the highest level of impact was associated with wastewater treatment without the option of reuse.

Kobayashi *et al.* (2020) conducted a comparative LCA study between nature-based constructed wetland (NBS CW) systems and energy-dependent membrane bioreactor (MBR) greywater systems, both coupled with reuse, at different scales. The scales examined in the study were expressed in person equivalent (PE) households, ranging from small-scale (5 PE) to larger scales representing neighbourhoods (350 PE) and communities (3 500 PE). The comparison specifically focused on the performance of these systems in treating greywater for various purposes, including flush toilets, laundry, and irrigation. The contributions of different combinations of greywater treatment and reuse applications to the impact categories varied in the study, highlighting that no single system was universally deemed more favourable.

However, specific observations emerged from the analysis. At the community level, the membrane bioreactor (MBR) system demonstrated the lowest global warming potential (GWP) and eutrophication potential (EUP) compared with the constructed wetland (CW) system. On the other hand, at the neighbourhood scale, the GWP of MBR was lower than that of CW. These findings highlight the importance of considering scale-specific factors and impact categories when evaluating greywater treatment and reuse system performance.

Lam *et al.* (2017) compared aerobic and anaerobic greywater systems with chlorinated seawater for flushing toilets in their study. The assessment was conducted within a building scenario, considering different systems' sustainability and environmental performance. The anaerobic fluidised membrane bioreactor (AnFMBR) system was the most sustainable option for domestic buildings, followed by seawater flush toilets and MBR systems. The AnFMBR system was identified as the most environmentally friendly because of its potential for retrofitting using resource recovery and water recycling technologies. The findings of Lam *et al.* (2017) highlight the importance of considering infrastructure and distribution aspects when assessing the overall sustainability of different greywater systems. The findings suggest that the AnFMBR system presents a promising solution for domestic buildings, offering sustainable wastewater treatment and the potential for resource recovery and water recycling.

3.4 Comparative LCA Studies on Alternative Water Sources – Cradle to Grave

The cradle-to-grave approach considers the entire water supply system, including its abstraction, production, use, and disposal phases. This broader scope allows for a more comprehensive assessment of environmental impacts as it considers all stages of a product's life, including raw material extraction, manufacturing, distribution, use, and end-of-life treatment.

3.4.1 Potable use

Loubet *et al.* (2014) found that only a few consolidated LCAs have been conducted with a cradle-to-grave perspective where water production, wastewater treatment, and disposal are jointly considered in one assessment. Lane *et al.* (2015) compared the environmental impacts of a traditional freshwater water system with minimal rainwater harvesting to those of a potential future system based on alternative water sources, including rainwater harvesting, desalination, and wastewater reuse. Interestingly, traditional and diverse systems significantly impact the wastewater collection and treatment stages (Lane *et al.*, 2015).

3.4.2 Non-potable use

Liu *et al.* (2019) used the LCA approach to evaluate the environmental benefits of using seawater as an alternative water source for flushing toilets. They compared seawater to alternative water sources, including seawater desalination, wastewater reuse, and on-site greywater reclamation, with potable water supply as the baseline. The evaluation was limited to environmental factors for Hong Kong and China research sites. Moreover, Liu *et al.* (2019) found a potential 20-30% saving in using potable water when using seawater as an alternative to flushing toilets. However, they observed that flushing toilets with seawater was environmentally unfeasible for population densities less than 3 000 people/km^2 and greater than 12 000 people/km^2.

3.5 Conclusion

Integrating alternative water sources into water-supply systems is pivotal for conserving freshwater resources. However, it is imperative to acknowledge the absence of a universally favourable alternative source across studies. The feasibility of hybrid water systems is intricately tied to factors such as geography, water sources, topography, and methodological parameters, emphasising the need for caution in direct study comparisons owing to varying influences across applications.

Within the realm of alternative water sources, a nuanced distinction arises between studies that adopt a cradle-to-tap view and those that embrace a holistic cradle-to-grave perspective. Despite these inherent complexities, insights from past research converge on certain conclusions. Applying energy-intensive processes, such as desalination, to treat alternative water sources is unsuitable for non-potable purposes. Moreover, it underscores the primary role of energy generation as a driver of environmental impact.

4 Review of the Application of Choice Modelling in the Preference of Alternative Water Sources

4.1 Introduction

4.1.1 Environmental valuation

Environmental valuation is the process of assigning monetary value to environmental goods and services that are not traded in the market and incorporating them into economic decision-making (Milne, 1991). These public goods are goods or services that are non-excludable and non-rivalrous in consumption. Milne (1991) described non-excludability as a characteristic of a public good, where, once it is made available, it can be accessed by everyone. It is challenging or even impossible to prevent anyone from using it. Furthermore, his description of non-rivalry implies that one person's use of a good does not diminish its availability for others. Examples of public goods that illustrate this concept include urban green spaces, flood control measures and clean water. Governments often supply these goods because private markets struggle to do so efficiently. This inefficiency is due to the free-rider problem, in which individuals can enjoy the benefits of the good without contributing to its cost.

4.1.2 Valuation methods

Environmental resources' total economic value (TEV) is often considered the sum of use, option, and existence values (Milne, 1991). Use value is the value of the direct benefits that people derive from using the resource. Second, the option value is the value people place on the option to use the resource in the future. Lastly, existence value is the value people place on the mere existence of the resource, regardless of whether they use it. This definition was expanded by Börger (2012) and is summarised in Table 4-1.

Table 4-1: Classification of values of an environmental good according to the concept of TEV(Börger, 2012)

Total economic value	Use values	Direct use value	e.g. recreation benefits of a forest
		Indirect use value	e.g. ecosystem functions
	Nonuse values	Bequest value	e.g. habitat protection for future generations
		Existence value	e.g. the existence of whales
		Option value	e.g. future medical use of a plant species
		Quasi-option Value	e.g. the still unknown and unlikely use

Various methods have been employed to assess the value of environmental goods and capture their non-market worth. Unlike traditional economic analyses, which depend on market prices to determine the value of goods and services, environmental goods are typically not traded in markets, and their values are not represented in market prices. Therefore, non-market valuation methods are used to estimate the value of environmental goods. These approaches encompass stated preference techniques, such as contingent valuation and choice experiments, and revealed preference methods, such as the travel cost method and hedonic pricing (Milne, 1991). These methods strive to estimate the value individuals place on environmental goods by analysing their behaviour or expressed preferences. However, since this chapter focuses on stated preferences, revealed preference valuation methods will not be discussed further.

4.1.2.1 Contingent valuation method

The contingent valuation (CV) method is a stated preference method used to estimate the economic value of environmental goods and services. This involves asking people how much they would be willing to pay (WTP) or accept (WTA) compensation for a change in the provision of an environmental good or service. For example, a survey might ask people how much they would be willing to pay to preserve a wetland or how much compensation would be required to accept the construction of a new dam. The responses to these questions were then used to estimate the economic value of an environmental good or service.

The premise of the contingent valuation method is that people's willingness to pay or accept compensation reflects their preference for environmental goods or services. However, this method has been criticised for its potential to elicit hypothetical or insincere responses and for its sensitivity to the framing of the question and other survey design factors (Milne, 1991). Despite these limitations, the contingent valuation method remains a widely used tool for estimating the economic value of environmental goods and services.

4.1.2.2 Choice experiment

The choice experiments involved presenting people with a series of hypothetical scenarios in which they must choose between options that vary in their attributes and levels. For example, a choice experiment might ask people to choose between two different wetland conservation programs that differ in cost, location, and number of protected species (Milne, 1991). Responses to these choices are used to estimate the economic value of an environmental good or service.

The choice experiment was based on the idea that people's choices reflect their preferences for the attributes of the environmental good or service. This method has several advantages over other stated preference methods, such as the contingent valuation method, including its ability to estimate multiple attributes' value simultaneously and account for the interactions between attributes. However, this method has limitations, such as its sensitivity to the design of choice sets and the potential for hypothetical bias (Carson and Louviere, 2011). Despite these limitations, the choice experiment is a widely used tool for estimating the economic value of environmental goods and services.

4.1.3 Aim of this chapter

This chapter explores the application of choice modelling to estimate the utility of alternative water sources in urban areas. Furthermore, the chapter reviews choice experiments conducted in urban water supply to elucidate the most significant attributes and impact of socio-economic factors influencing the willingness to pay for the utility of water supply.

4.2 Stated Preference for alternative water sources

Brouwer *et al.* (2023) evaluated the attributes of water and sanitation service levels for slum and non-slum dwellers in Dhaka, Bangladesh. Theirs is one of the few studies that modelled the circularity of water, considering urban water supply, wastewater treatment, and stormwater management. Other studies have evaluated the willingness to pay for the non-use of urban water supplies. For example, Kobel and Del Mistro (2015) investigated WTP to improve water and sanitation services in informal settlements by the "non-poor" and non-user residents in Cape Town.

The following sections review the application choice experiments for evaluating the stated preference for alternative water sources in non-slum urban areas. The findings are discussed under the categories of centralised and non-centralised water sources.

4.2.1 Centralised alternative water sources

Alternative water sources for domestic use have become imperative as the demand for supply from conventional water sources such as rivers and groundwater has increased to meet rising urbanisation. Research on the application of choice modelling in water supply has mainly focused on surface water, which depends on the hydrological cycle. For example, Haider and Rasid (2002) used choice modelling to determine consumer preferences for alternative surface water supply sources in the context of deteriorating water supply quality. They found that water supply pressure and taste were the most significant attributes.

Other studies have elicited consumer preferences for alternative water supplies and water conservation to meet future demand. Blamey *et al.* (1999) considered surface water sources, including large-scale wastewater recycling and reduced water use coupled with wastewater recycling. They found a favourable preference for increasing the use of recycled wastewater, particularly as a replacement for potable water supplies for industrial use. However, respondents indicated a negative preference for the indoor use of recycled wastewater. Furthermore, water supply restrictions had an adverse effect on the stated choices.

On the other hand, in addition to surface water and recycled wastewater, Awad *et al.* (2021) included storage methods (no storage, aquifer injection or a reservoir) and conservation policies (rebates versus restrictions). They found a significant preference for additional water storage, either in the form of aquifer injection or a new reservoir. There was a strong aversion to creek water as an alternative water source. However, the study did not measure end-user attributes, such as reliability, taste, and quality, similar to the study by Blamey *et al.* (1999).

Conversely, Bennett *et al.* (2016) conducted a choice modelling study to estimate households' WTP for recycled wastewater to meet the water supply demand. The focus of this study was to determine the end-use preferences for recycled wastewater, including industrial, open-space irrigation, domestic use, and environmental flows. At the time of the research, 20,000 households had used recycled wastewater for domestic use, such as flushing toilets and watering gardens (Bennett *et al.*, 2016). The respondents preferred using recycled wastewater for industrial use, which is similar to the findings of Blamey *et al.* (1999).

4.2.2 Decentralised alternative water sources

Some studies have considered integrating central water supply infrastructure with decentralised household water facilities such as rainwater harvesting (RWH) and greywater reuse (GWR) in the context of secure and long-term availability of urban water supply. For example, Lu *et al.* (2019) conducted a discrete choice experiment (DCE) to elicit consumer preferences for using decentralised household water facilities. They found that the early adoption of decentralised water facilities is positively related to neighbours' adoptions, and the pressure of water scarcity increases households' willingness to share a decentralised facility.

Amaris *et al.* (2020) investigated the preference for GWR for six indoor uses (toilet flushing, garden irrigation, clothes washing, washing hands, bathing or showering, and drinking). Their study found that the GWR for toilet flushing was the most preferred indoor use, with drinking being the least preferred. In addition, they found that the odour attribute was the most significant compared to the colour attribute. In addition, previous knowledge of GWR is significant in the preference for an alternative water source.

4.3 Conclusion

There is limited application of choice modelling to preferences for urban water supply. Most studies have focused on regional water supply options influencing water supply policies. Few studies have been framed within the context of water use behaviour by investigating preferences for alternative water sources at a domestic level.

5 Site Description

5.1 Site Locality

Hout Bay is a harbour town located in the Western Cape Province of South Africa on the Atlantic seaboard of the Cape Peninsula, approximately 20 km south of Cape Town.

The town is surrounded by mountains, namely by the Table Mountain National Park to the north, Oranjekloof to the north-west, bordered by Little Lion's Head, Karbonkelberg, Kaptein's Peak and The Sentinel to the west, the Vlakkenberg, Koorsteenskopber and Constiaberg to the east and the southern Atlantic Ocean to the south.

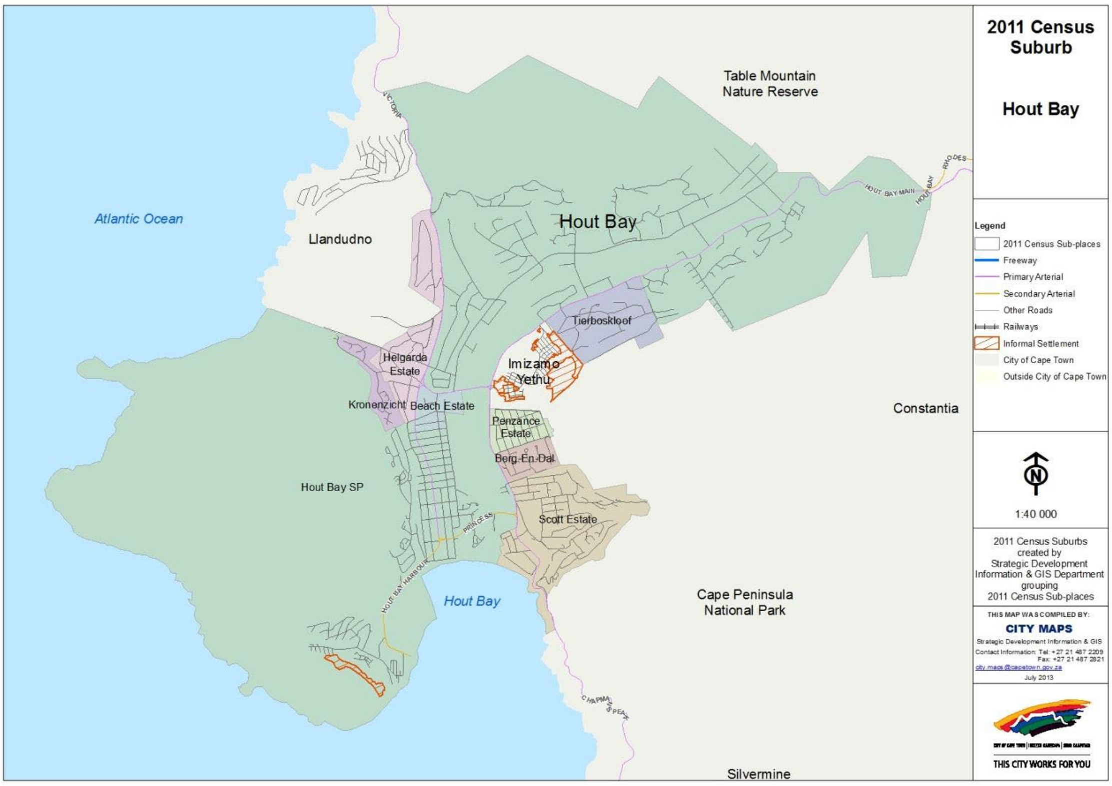


Figure 5:1: Hout Bay site locality
(Source:2011_Census_CT_Suburb_Hout_Bay_Profile**/ 21 November 2022 23:50)**

Hout Bay has several suburbs, including the Beach Estate, Berg-En-Dal, Helgarda Estate, Hout Bay SP, Kronenzicht, Penzance Estate, Scott Estate, and Tierboskloof. Hangberg, previously classified as a coloured area under the apartheid government, is located on the mountain slopes between Hout Bay Harbour and The Sentinel Mountain (see Figure 5:1). Imizamo Yethu is an informal settlement situated in Hout Bay.

5.2 Demographics

Following apartheid zoning, the Hout Bay Valley was predominantly a white-only community with a few coloured people who worked in the harbour. The coloured community was segregated from the white population and was located in Hangberg, close to the harbour. Furthermore, in 1988 and 1989, black informal settlers began to occupy state and private property in the area. The rapid growth of informal settlers, although resisted by the original residents of Hout Bay, compelled the local authority at the time to formalise their settlement on 18 hectares of land (Tony, 2021). According to this resettlement scheme, 400 households have received serviced housing. However, once Imizamo Yethu was established as a residential area, more informal settlers continued to dominate.

SALDRU (2006) conducted a survey to understand the influence of migration on two neighbourhoods in Hout Bay, Hangberg, and Imizamo Yethu. This study found that the white population lived in well-secured and formal properties in Hout Bay. In contrast, a substantial portion of the coloured population lived in the Hout Bay Harbour area, including Hangberg, an informal settlement. In addition, SALDRU (2006) identified the following distinct types of housing in Hangberg:

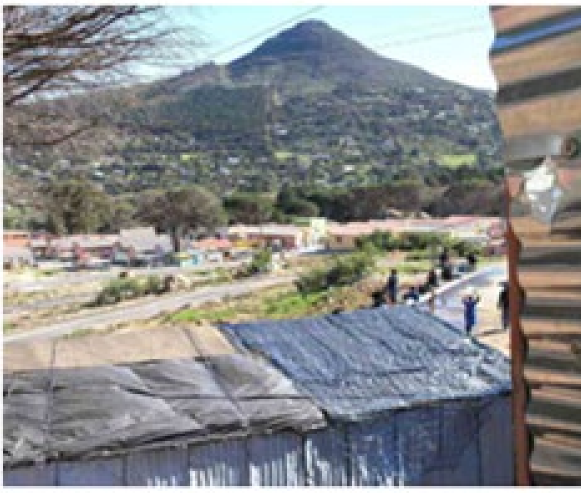
a) Sections of flats that are clearly demarcated,
b) Informal settlements with shacks built in the sand behind the flats,
c) The compound section comprises blocks of flats located in the area that initially belonged to a fishing company close to the sea
d) Free-standing houses or bungalows

Formal housing (flats) in Hangberg

Informal housing in Hangberg

Figure 5:2: Type of housing in Hangberg (Source: (SALDRU, 2006) pg 16)

The black population lives in Imizamo Yethu. Figure 5:3 illustrates the types of housing development.



Formal housing in Imizamo Yethu

Informal housing in Imizamo Yethu

Figure 5:3: Type of housing in Imizamo Yethu (Source: (SALDRU, 2006) pp 12&13)

SALDRU (2006) described the distinct types of dwellings as follows:

a) Two and 3-bedroom brick houses were arranged linearly along access roads. These houses were built through a project funded by Irish philanthropist Niam Mellon. Backyard shacks are found behind most brickhouses.
b) Informal shacks were built close to the main road in the early 1990s.
c) Informal development of shacks that lack organised arrangements and access roads. Informal development occurred after the 2004 fire, which damaged a substantial portion of the shacks at that time.

According to the 2011 census, the population in Hout Bay is estimated to be 17 900 (https://www.statssa.gov.za/ 15 November 2022), and demand for housing and essential services remains challenging for Hangberg. In the 2011 census, demographic statistics for Hangberg were included in Hout Bay. On the other hand, Imizamo Yethu is a rapidly growing settlement. In 2008, it had a population of 8 052 persons; by 2011, the population had almost doubled to 15 538 persons (STATSSA, 2011). Table 5-1 summarises the demographics of Hout Bay and Imizamo Yetho based on the 2011 census.

Table 5-1: Hout Bay Demographics (https://www.statssa.gov.za/ 15 November 2022)

Description	Hout Bay	Imizamo Yethu
Population	17 900	15 538
Population density	631 persons/km^2	27 227 persons /km^2
Formal dwelling	94.0%	23.1%
Number of households	5 963	6 010
House own/ paying off	58.9%	23.1%
Working Age (15 – 64 years old)	67.5%	75.2%

5.3 Water supply and sanitation infrastructure

The City of Cape Town (CCT) is a water service authority responsible for providing water to its customers, including households, industries, and agriculture. Surface water is the primary water source in this area, and Hout Bay has local water resource reserves. However, in 2000, CCT commissioned a 20 ML reservoir to augment the water supply from the bulk regional water scheme. The augmentation scheme includes water resources from the Steenbras, Wemmershoek and Voelvlei Dams (see Figure 5:4).

However, the project's scope considered only water resources local to Hout Bay. Raw water is abstracted from three Table Mountain dams: de Villiers, Alexandra, and Victoria. It is gravity-fed to the Constantia Nek Water Treatment Works. The facility has a treatment capacity of 3 ML/day. Raw water is treated according to the SANS 241 standard regulation in South Africa, and the process includes coagulation, pH level adjustment, flocculation, settlement, filtration, and chlorine disinfection.

The treated water is then pumped into the 20 ML Constantia Nek reservoir, which is stored before being distributed to the Hout Bay consumers. Water consumption in these households and industries generates sewage, which is reticulated to collector pump stations. From there, sewage is pumped to the marine outfall, which consists of three components: a pretreatment facility, a pump station, and an outfall pipeline. The outfall pump station has a capacity of 9.8 ML/day, and the outfall pipeline is anchored to the seabed at a depth of approximately 39 m and discharges 2.2 km from the coastline (CCT, 2018).

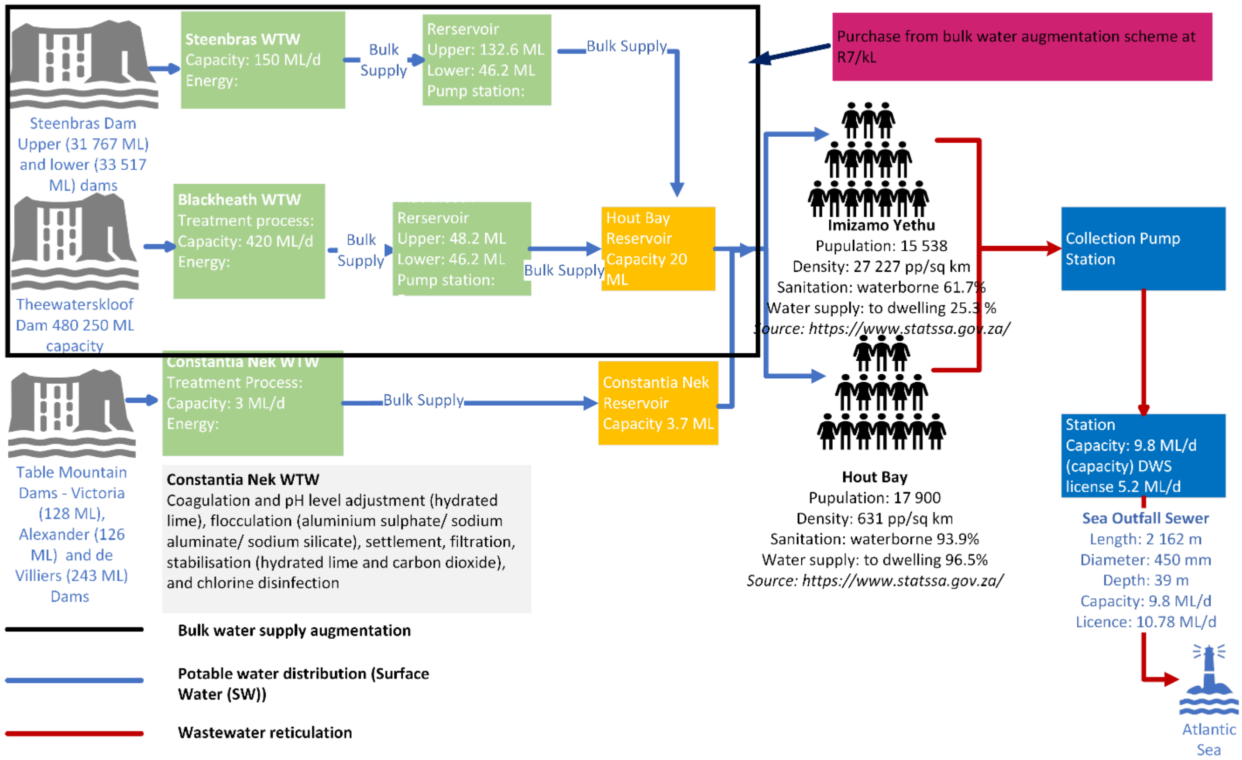


Figure 5:4: Water and sanitation level of service to Hout Bay and Imizamo Yethu

Unlike many urban residential areas, sewage is not transported to a conventional wastewater treatment plant but disposed of into the sea untreated. In the case of Hout Bay, it is due to limited land that a wastewater treatment facility cannot be constructed. Moreover, the proximity of the residential area to the sea made the sea outfall option cheaper than conveying sewage to a treatment facility further away because of the energy consumption associated with this option (Tony, 2021).

5.3.1 Water supply level of service

As shown in Table 5-2, approximately 99% of the Hout Bay residents have access to water, of which 97% have water inside their dwellings. However, approximately 74% of the Imizamo Yethu population had access to water, whereas 23% relied on vendors. In addition, only 25% of those with access to water have it inside their dwellings, coinciding with the 23% with access to formal dwellings.

Table 5-2: Water Supply Level of Service (https://www.statssa.gov.za/ 15 November 2022)

Water Supply Sources	Hout Bay	Imizamo Yethu
Regional/Local water scheme	98.6%	73.6%
Borehole	0.30%	1.40%
Spring	0.10%	0.20%
Rainwater tank	0.10%	0.30%
Dam/Pool/Stagnant water	0.10%	0.10%
River/Stream	0%	0%
Water vendor	0.30%	0.20%
Water tanker	0.20%	23.4%
Other	0.40%	0.70%
Total	100%	100%

The service levels for water and sanitation in the 2011 census are summarised in Table 5-3. Approximately 61.7% of the population in Imizamo Yethu have access to flush toilets connected to the sewerage system. However, only 23.2% of the population lives in formal dwellings, and a further 25.3% has access to piped water inside the dwelling. Coupled with the limited formal housing development observed by SALDRU (2006), it is incongruent that 61.7% of the Imizamo Yethu could access waterborne sanitation.

Table 5-3: Summary of Water and Sanitation Services (https://www.statssa.gov.za/ 15 November 2022)

Description	Hout Bay	Imizamo Yethu	Total
Population	17 900	15 538	33 438
Population density	631 persons/km^2	27 000 persons/km^2	
Formal dwelling	94.0%	23.2%	61%
Piped water inside the dwelling	96.5%	25.3%	63%
Flush toilet connected to the sewerage system	93.9%	61.7%	79%

6 Life Cycle Assessment of Using Seawater to Flush Toilets in Coastal Areas

This study was conducted according to the globally accepted Life Cycle Assessment (LCA) methodology of the ISO 14044. The assessment stages are illustrated in Figure 6-1. It comprises four interrelated phases: goal and scope definition, life cycle inventory analysis, life cycle impact assessment, and interpretation.

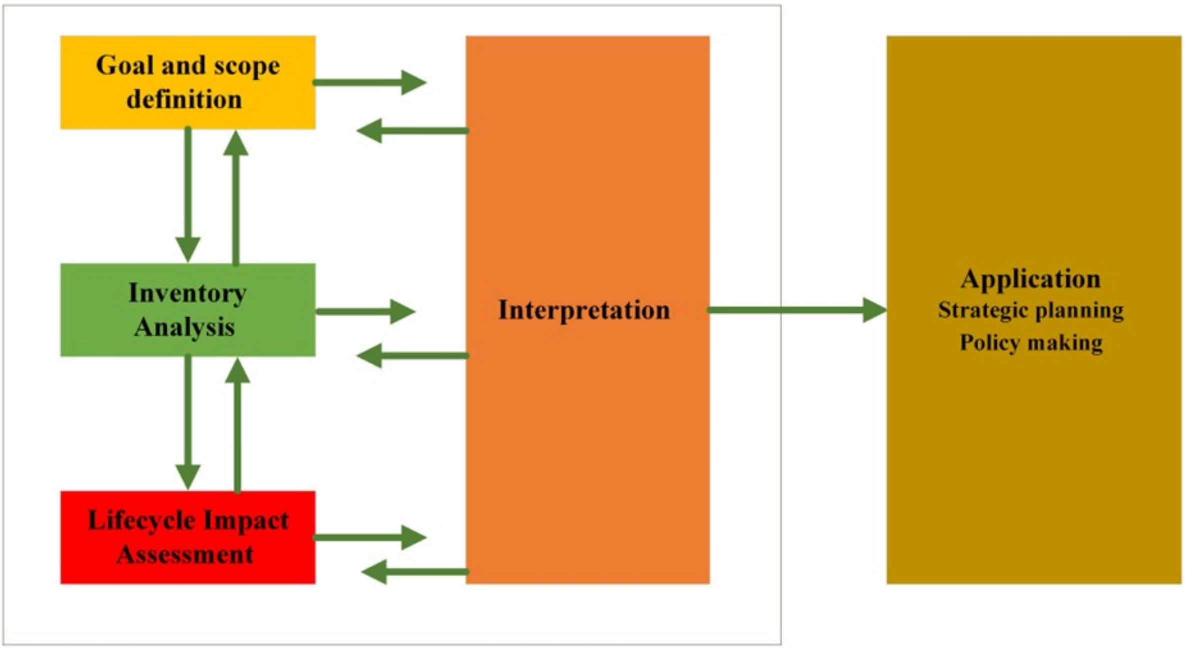


Figure 6:1: Life Cycle Assessment summary (adapted from ISO, 2006)

6.1 Goal of the study

This study considers a base case where surface water is provided to flush toilets in coastal areas serviced by marine outfall sewers (MOS). Given the increasing urbanisation and limited water supply resources, the goal is to assess the feasibility of using seawater to flush toilets to increase access to waterborne sanitation within centralised systems. As a result, this study compared the environmental impacts of supplying water from the base case of a single source (surface water) to a dual source (surface water and seawater). The first option represents a surface water system in which potable water is produced from local surface water sources, such as dams, and is used for all household applications. The alternative hybrid system consists of two water sources: potable water and partially treated seawater, which are supplied to households separately.

6.1.1 Target audience and application of findings

The findings of this study are beneficial to coastal water utilities that aim to diversify their water sources and enhance local water security. Owing to the academic nature of this study, the findings will be released to the public through conferences and reports.

6.2 Scope of Study

The scope definition phase outlines the critical elements of the study. It articulates the level of detail required at each stage. This section defines all methodological choices and critical parameters following ISO 14044 regulations, including the product system, function and functional unit, system boundary, life cycle phases, impact assessment methodology, and impact categories considered.

6.2.1 Product system

The product system includes water abstraction from local dams, treatment, storage, and distribution to the end-user. MOS is used as a wastewater management option in Hout Bay. Therefore, the post-consumption phase was similar to that of the considered scenarios. According to Opher and Friedler (2016), similar processes can be eliminated in comparative scenarios without interfering with the reliability of the findings. Therefore, the post-consumption phase was excluded from the product system.

The local water supply from the Table Mountain dams was insufficient to meet the needs of Hout Bay. Consequently, Hout Bay receives water from the integrated Western Cape Water Supply System (WCWSS), which consists of six dams owned and run by the Department of Water and Sanitation (DWS) and the City of Cape Town (CCT). For this analysis, we assumed that all the surface water used in the system came from Table Mountain dams. The scenarios considered in this study were as follows.

6.2.1.1 Scenario A

Figure 6:2 depicts Scenario A, representing the base case in which a single-source surface water system was utilised.

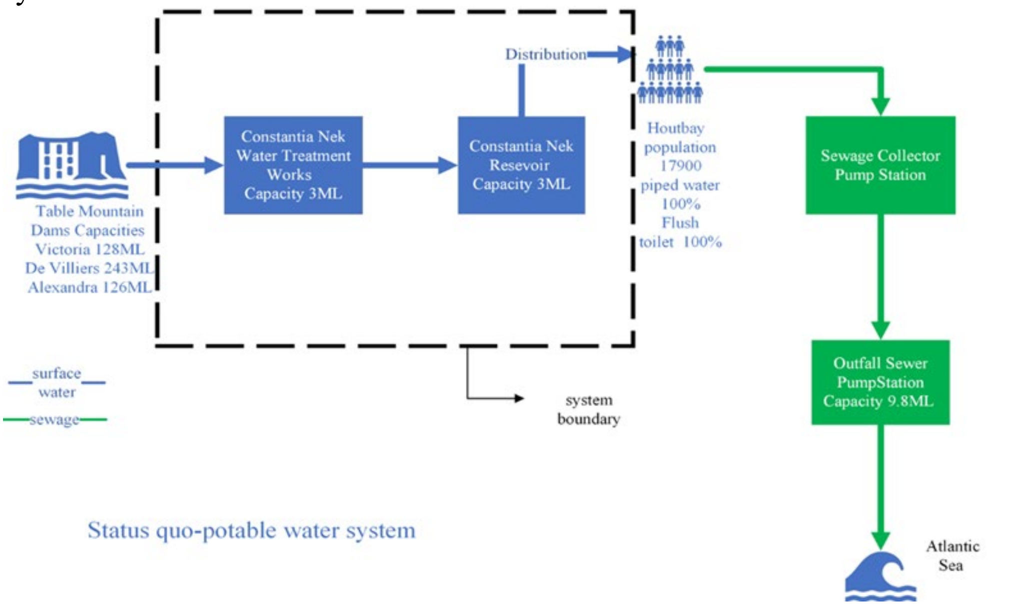


Figure 6:2: Single source surface water system

This scenario assumes that raw surface water is extracted solely from three Table Mountain dams in the Nature Reserve, Devilliers, Victoria, and Alexandria, and treated to potable standards at the Constantia Water Treatment Works (CNWTW), which has a hydraulic treatment capacity of 3 ML/d (CCT, 2018). Raw water is conventionally treated to potable standards through various processes, including flocculation, sedimentation, and chlorination. The potable water produced was pumped into the Hout Bay reservoir and delivered to households through a distribution network. The potable water supplied to Hout Bay is utilised for all household end-uses, including bathing and cooking, and non-potable applications, such as toilet flushing, do not require high-quality water.

6.2.1.2 Scenario B

Figure 6:3 illustrates Scenario B, representing a hypothetical hybrid system comprising potable water and partially treated seawater separately supplied to households for potable use and toilet flushing. As in Scenario A, potable water was assumed to be sourced, treated, and distributed.

An additional water source for flushing is seawater extracted from the sea, screened, and partially treated with chlorine to limit microbial growth to a quality that is sufficient for toilet flushing. Due to saline water's corrosive nature, partially treated seawater is best transported through thermoplastic pipelines such as polyvinyl chloride (PVC) and high-density polyethylene (HDPE) (Liu *et al.*, 2019). In this study, it was assumed that all pipelines conveying seawater were composed of HDPE.

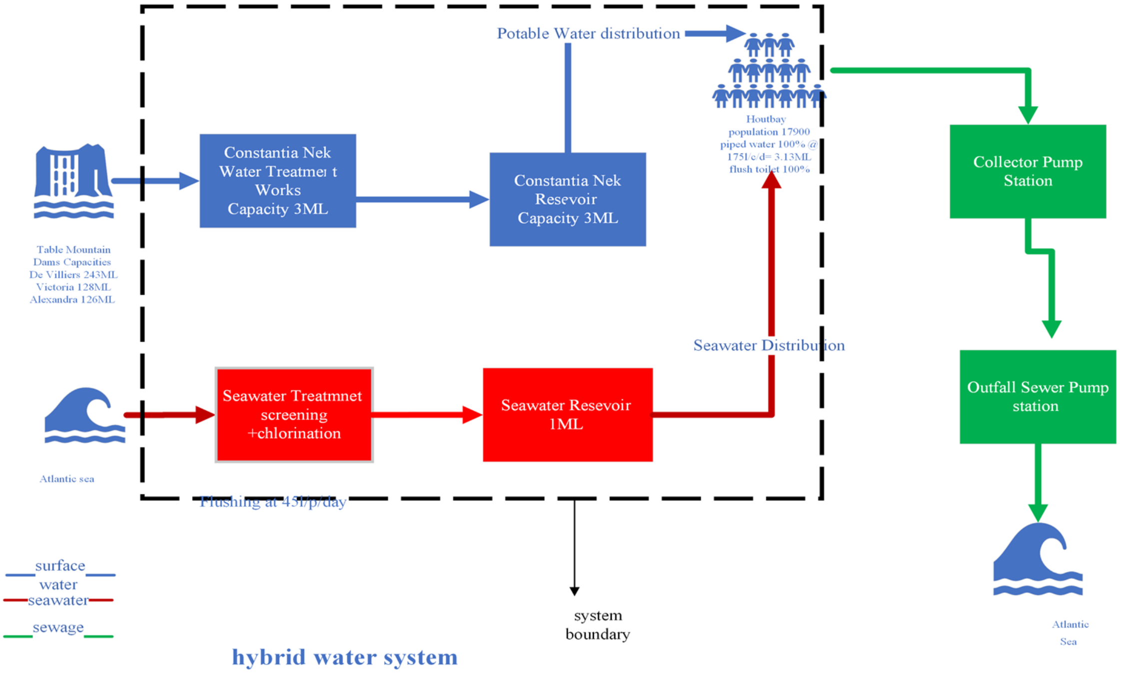


Figure 6:3: Hybrid water supply system

6.2.2 Function and functional unit

The two systems have identical functions in abstracting, treating, and supplying treated water of the desired quality that applies to the intended purpose of the end user. The functional unit quantitatively links the function of the product system to the inputs and outputs. It provides a basis for comparing alternative products or systems (Rebitzer *et al.*, 2004). This study takes the functional unit as 1 m^3 of treated water delivered to the end-user.

6.2.3 System boundary

The system boundary specifies the processes included in the assessment and those excluded from the study based on the study goal. It consists of product system infrastructure construction and operational life cycle phases. The stages considered include infrastructure materials, transport, and disposal. However, the energy and machinery required for construction were not included because of the data gaps. In addition, previous studies assumed that the overall impact of end-of-life infrastructure is negligible.

Components of the distribution infrastructure and electromechanical equipment, including valves and flowmeters, are not expected to contribute significantly to the construction of the entire system, owing to their relatively small weight compared to the structure. However, their impact may be substantial in studies focusing on distribution systems (Hajibabaei *et al.*, 2018).

The average transportation of raw materials and chemicals from suppliers to consumers is considered. A local study by Buckley *et al.* (2009) found that transporting chemicals and construction materials had a negligible impact, contributing less than 5% of the total implications for potable water production. Consequently, the impact of transportation was not considered in this analysis.

Table 6-1: System boundary conditions

Stages	Water system
Included	• Raw material extraction, civil and mechanical infrastructure production, pipelines, water treatment works, and pumps. • Operational and maintenance data, including chemical production and consumption, energy production and consumption during water treatment. • Storage and distribution.
Excluded	• Transportation of construction elements and chemicals • End of life of the different components of the water cycle • Sewage collection • Wastewater management and disposal

6.2.4 LCIA methodology and impact categories selection

Global Life Cycle Impact Assessment (LCIA) methodologies are better equipped to represent geographies that are often not considered in country- or region-specific methods with American and European focus. This study used the ReCiPe 2016 methodology. Several local and international studies have used this methodology in their analyses (Liu *et al.*, 2021; Goga *et al.*, 2019; Liu *et al.*, 2019; Opher and Friedler, 2016).

The results are expressed using midpoint indicators because their modelling uncertainty is lower than that of the endpoint indicators (Bare *et al.*, 2000; Huijbregts *et al.*, 2017). The following impact categories were obtained from the ReCiPe midpoint suite of impact categories: freshwater consumption, global warming, terrestrial ecotoxicity, freshwater ecotoxicity, marine ecotoxicity, fossil resource depletion, mineral resource depletion, and land use. They were selected because they align with the study aim and system boundary.

Landu and Brent (2006) found that freshwater withdrawal profoundly impacts local ecosystems more than toxicity and energy impacts, considering the region's prevailing water scarcity. Because this study focuses on increasing water supply reliability, it is crucial to quantify the impact of water consumption; hence, the water footprint is of significant interest to the target audience.

Additionally, the analysis considered the environmental impacts of the water supply chain on other aquatic ecosystems. Consequently, ecotoxicity is significant because chemical and material production processes release toxins into the aquatic environment. The depletion of mineral and fossil resources is also important because it is associated with water systems' water material and energy production processes (Bonton *et al.*, 2012). Water systems require the construction and installation of various types of infrastructure. Consequently, it is imperative to consider the effects of land occupation and transformation on biodiversity because of infrastructure construction within the water supply chain. Finally, global warming was considered because of its significance in global and local policies on carbon emissions, such as the Paris Agreement.

Other impact categories were not evaluated because they were assumed to have negligible significance, considering the defined system boundary and scope of this study. These impacts include ionisation radiation, ozone formation (human health, terrestrial ecosystem), fine particulate matter formation, terrestrial acidification, freshwater and marine eutrophication, human carcinogenic and noncarcinogenic toxicity, and stratospheric ozone depletion.

6.3 Methodology

6.3.1 Data quality requirements

6.3.1.1 Time-related coverage

The study data were initially planned to be collected within five years. However, due to the limited primary data availability, the analysis relied on Ecoinvent data.

6.3.1.2 Technological coverage

This study assumes that there are no significant technological differences in the treatment processes utilised in the Ecoinvent conventional water treatment and seawater treatment processes employed by Liu *et al.* (2016). Therefore, the datasets used in this study were not modified.

6.3.1.3 Geographical coverage

This study used as much South African data as possible. When South African data were unavailable, the global datasets were used.

6.3.1.4 Allocation

Allocation is the division of the environmental load among products derived from a unit process or multi-input process. There seems to be no consensus regarding the allocation rules for the environmental load on products and co-products derived from a single process (Klöpffer & Grahl, 2014). The LCA standard promotes allocation avoidance. This study allocated 100% of the impact to the output. The default allocation by classification procedure in Ecoinvent 3.5 is utilised (Weidema *et al.*, 2013).

6.4 Life Cycle Inventory (LCI)

The second phase of LCA is inventory analysis, which involves systematic collection, organisation, and data analysis. This stage is often regarded as resource-intensive because of the rigorous data acquisition and validation procedures typically employed in LCA studies (Curran, 2013).

The initial phase of the LCI involved identifying the unit processes required for the LCI model of supplying 1 m^3 of potable water to households in Hout Bay within the foreground and background system boundaries. This encompasses the infrastructure for raw water, water treatment plants, pipelines, mechanical equipment, such as pumps, and operational requirements, including the chemical and energy inputs required during water treatment and distribution. The data sources for each stage were identified by scanning literature and databases.

The next section discusses the steps followed in inventory analysis. It encompasses data collection and the modification of datasets to suit the study area.

6.4.1 Planning and data collection

The planning and data collection process was adapted from Wenzel *et al.* (2001). Data collection was a lengthy process that involved communicating with CCT employees and visiting the treatment plant to learn more about the water system utilised in Hout Bay.

CNWTW was constructed in 1928 (CCT, 2018). Owing to the age of the facility, the water utility could not provide as-built information on infrastructure construction. However, operational inputs were obtained, including materials, chemicals, and energy consumption. The background processes associated with producing chemicals, energy, and materials were obtained from the Ecoinvent database.

After identifying and assessing the relevant processes and data sources, a spreadsheet was created to document the inputs of materials and energy and the outputs of emissions and waste for each unit process. The data were obtained from the Ecoinvent database and structured according to the format of the database. The data were then annualised and weighted based on the functional unit to determine the required quantities for the selected system boundary.

6.4.2 Water consumption

6.4.2.1 Scenario A

Water is the raw material of interest in this LCA study. As water moves along different stages of the urban water system, its quality and quantity can be altered. The water required in each scenario was calculated using a daily consumption rate of 175 L/c/d. However, at this consumption rate, the capacity of the existing water treatment plants is insufficient to fulfil the water needs of 100% of the Hout Bay community.

6.4.2.2 Scenario B

The hybrid system uses two raw water sources: surface water and seawater. The water requirement was calculated based on the consumption rate of 175 L/c/d. A study in Cape Town found that medium-income households use a maximum of 45 litres per person per day for waterborne sanitation, assuming a flush toilet that consumes 9 litres per flush and an average of five flushes per person daily (Viljoen, 2015). In this scenario, the flush water was assumed to be extracted from the sea. Based on this assumption, 130 L/c/d was used for all household applications except toilet flushing.

6.4.3 Single source: Surface water

Owing to the small capacity of the treatment plant, all treatment units are housed in a single building; therefore, the construction material quantities incorporate both the building and treatment process units. Building impacts are often not considered because of their minor contribution (Buckley *et al.*, 2009). The structural elements of civil infrastructure can ordinarily be determined from a bill of quantities. However, a bill of quantities could not be obtained within the resources and time allocated to this project. Hence, all the infrastructure processes were based on the Ecoinvent database.

6.4.3.1 Potable water treatment construction

After assessing various transformation infrastructure processes in the Ecoinvent database, the process was selected to match the conventional water treatment employed at the CNWTW closely. The infrastructure process of the construction of the water treatment plant used is "*Waterworks, capacity $1.1E^{10}$l/year | water works construction, capacity $1.1E^{10}$l/year, conventional treatment.*"

The dataset represents the construction of a medium-sized conventional treatment plant, including stages such as intake pumping, coagulation, flocculation, clarification, filtration, ozonation, and the material input of equipment such as pumps. The demolition of the plant at the end of its design life was considered in the dataset. However, the energy and equipment used in this study were not included because of the lack of data. The pumps had a lifetime of 13 years, whereas the civil structures had a 60-year lifetime. In addition, the dataset was modified to represent local infrastructure by assuming that the concrete used in its construction was produced in South Africa.

The capacity of the waterworks in the Ecoinvent dataset was that of a medium-sized plant (30 000 m^3/d), whereas that of the CNWTW was a small-capacity plant (3 000 m^3/d). The medium-sized plant was scaled to represent the CNWTW using Equation 1.1, adapted from the literature (Lane *et al.*, 2015).

$$\boldsymbol{I_{TP} = I_{DS} \times \left(\frac{C_{TP}}{C_{DS}}\right)^{0.65}} \qquad \textbf{6-1}$$

Where I_{TP} = quantity of required inventory item (I) for the treatment plant model used here
I_{DS} = quantity of inventory item (I) in the data source
C_{TP} = volumetric capacity (C) of the treatment plant in our scenario
C_{DS} = volumetric capacity (C) of the treatment plant modelled in the data source

According to Ecoinvent, this dataset's primary source of uncertainty originates from using cost-scaling factors beyond the intended applicability range. Nevertheless, this uncertainty has been accounted for using Ecoinvent's technological correlation" data quality indicators.

Table 6-2 lists the inventory for constructing the 1.01 E6 m^3/ year (CNWTW), assuming a lifetime of 60 years based on a functional unit of 1 m^3 of water.

Table 6-2: Construction Inventory for the CNWTW per functional unit

Raw Material	Ecoinvent process	Amount [kg/m³]
Aluminium	Aluminium, wrought alloy(ROW)| market for aluminium, wrought alloy | Cut-off, U	0.0000428
Charcoal	Charcoal {GLO}| market for charcoal | Cut-off, U	0.00000196
Concrete	Concrete, normal strength {ZA}| market for concrete, normal strength | Cut-off, U	0.0000186
Copper	Copper, cathode {GLO}| market for copper, cathode | Cut-off, U	0.0000132
Glass fibre reinforced plastic.	Glass fibre reinforced plastic, polyester resin, hand lay-up {GLO}| market for glass fibre reinforced plastic, polyester resin, hand lay-up | Cut-off, U	0.000121
Polyethylene	Polyethylene, linear low density, granulate {GLO}| market for polyethylene, linear low density, granulate | Cut-off, U	0.00000258
Polyurethane	Polyurethane, rigid foam {GLO}|market for polyurethane, rigid foam | Cut-off, U	0.0000107
Polyvinyl chloride	Polyvinylchloride, bulk polymerised {GLO}| market for polyvinylchloride, bulk polymerised | Cut-off, U	0.0000728
Steel	Reinforcing steel {GLO}| market for reinforcing steel | Cut-off, U	0.00158
Sand	Silica sand {GLO}| market for silica sand | Cut-off, U	0.000264

Raw Material	Ecoinvent process	Amount [kg/m^3]
Steel	Steel, low-alloyed, hot rolled {GLO}| market for steel, low-alloyed, hot rolled | Cut-off, U	0.000749

6.4.3.2 Potable water distribution construction

The water distribution system transports potable water from treatment plants to consumers. It involves a complex system of civil and mechanical components, including storage reservoirs, pumps, valves, water meters, and pipelines. This study assumes that storage reservoirs and pumps are part of the construction dataset for potable water treatment plants because the water is expected to be directly conveyed from the plant to the distribution line. Consequently, distribution infrastructure includes the production and installation of pipelines. Table 6-3 presents the length and mass of the pipelines installed in Hout Bay.

Table 6-3: Pipe distribution of the single source surface water system

Material	Length[m]	Mass [kg]
Concrete	10	165
Fibre cement	72 783	1 328 705
HDPE	14 514	57 022
Other	396	8 922
Steel	851	26 300
PVC	44 310	381 250

Approximately 0.3% of the distribution network material is unknown. The analysis did not consider this to avoid overaccounting for its impact. While the potential impact of doing so lowers the certainty of the findings, it is assumed to be insignificant, given that it constitutes less than 0.5% of the total pipe network length.

Fibre cement, known as asbestos cement, was commonly used as a pipe material in Hout Bay from 1965 to the 1990s. Approximately 55% of the pipeline length in Hout Bay comprises fibre cement. However, its use was discontinued because of the potential health risks of inhaling fibre cement dust (van Zyl, 2014). Despite these concerns, fibre cement pipes are generally considered safe in water-supply systems. Consequently, van Zyl (2014) reported that existing fibre cement pipelines in South Africa are typically not excavated out of concern for public health. However, these pipelines should be replaced with alternative materials once they reach their designed lifespan.

The Ecoinvent database does not support asbestos cement, so an alternative dataset using synthetic fibres instead of asbestos fibres was used based on the study by Hajibabaei *et al.* (2018). The dataset was *fibre cement corrugated slab {GLO}| market for fibre cement corrugated slab | Cut-off, U*

Thermoplastic pipes, including PVC and HDPE, were introduced into the distribution network of Hout Bay in 2000. Currently, the network comprises 33% PVC and 11% HDPE pipes. Plastic pipes are preferred over steel and concrete owing to their durability, lightweight properties, and ease of installation (van Zyl, 2014).

The material production process *Polyethene, high density, granulate {GLO}| market for polyethene, high density, granulate | Cut-off*, represents the production of polyethene. Polyvinyl chloride was used as the material dataset for the PVC pipe, and the suspension was

polymerised. The dataset was *(ROW)| Polyvinylchloride production, suspension polymerisation | Cut-off, U =*. However, it was assumed that the processing of this material into plastic pipes, known as extrusion, occurred locally. Therefore, a South African electricity mix was used in the extrusion process.
Steel and concrete are materials commonly used in pipe construction; however, they are not widely used in the distribution network of Hout Bay. Only 0.6% of the total length of the pipes in the network was made of steel, whereas less than 0.1% was made of concrete. The Ecoinvent dataset *Steel, low-alloyed, hot rolled {GLO}| market for steel, low-alloyed, hot rolled | Cut-off, U* was used.

6.4.3.2.1 Distribution pipeline inventory

The inputs of the distribution network infrastructure process were determined by dividing the total material mass by the service life of the pipe materials and annual water distribution. The service lives for concrete, fibre cement, HDPE, PVC, and steel were obtained from the literature and were 50, 60, and 20 years, respectively (Liu *et al.*, 2019). The mass of each pipe was calculated by multiplying the mass per unit length of the pipe material specified in the pipe catalogues with the length of the pipe material for each diameter. For PVC, HDPE, steel, and concrete, the websites of local pipe manufacturers such as Sizabantu, Leroy Merlin, and Rocla were consulted. Table 6-4 lists the types of pipelines and the production infrastructure for each functional unit.

Table 6-4: Inputs of the distribution infrastructure

Materials	Ecoinvent Process	Amount [kg/m^3]
HDPE	Polyethylene, high density, granulate {GLO}| market for polyethylene, high density, granulate | Cut-off, U	0.00125
PVC	Polyvinylchloride, bulk polymerised {GLO}| market for polyvinylchloride, bulk polymerised | Cut-off, U	0.00706
Steel	Steel, low-alloyed, hot rolled {GLO}| market for steel, low-alloyed, hot rolled | Cut-off, U	0.00146
Asbestos cement	Fibre cement corrugated slab {GLO}| market for fibre cement corrugated slab | Cut-off, U	0.0297
Concrete	Concrete, 30MPa {ZA}| concrete production, 30MPa, with cement, CEM II/B-V | Cut-off, U	0.00000367

In addition to the materials used for pipes, the installation process significantly contributes to the impact of distribution networks (Simion *et al.*, 2024; Hajibabaei *et al.*, 2018). In a study comparing different pipelines, Simion *et al.* (2024) found that installing HDPE pipelines contributed 14-42% to environmental impacts. However, several studies on urban water systems have not considered the impact of the pipe installation phase on the construction of distribution systems (Kobayashi *et al.*, 2020; Lane *et al.*, 2015; Liu *et al.*, 2019; Opher and Friedler, 2016).

6.4.3.2.1 Pipe Installation

The pipe installation phase consisted of several processes considered in this assessment, including trench excavation, gravel bedding, pipe laying, and sand backfilling. Our analysis assumes that all required materials are available on-site so that no bedding material will be borrowed from other areas. The pipe installation process was based on the data from Simion *et al.* (2024) and Hajibabaei *et al.* (2018).

6.4.3.3 Operation

The operational inventory of the surface water system is presented in Figure 6:4. The following subsections discuss the process units in detail.

Figure 6:4: Operation processes within the system boundary

6.4.3.4 Chemical consumption

The conventional surface water treatment chemicals used to achieve drinking water quality were determined based on the 2021 Water Treatment Process Audit Report. The CNWTW employs a traditional treatment process that begins with coagulation and pH level adjustment with hydrated lime, followed by flocculation with aluminium sulphate and sodium aluminate, settlement, filtration, stabilisation using hydrated lime, and disinfection with chlorine. However, the quantity of sodium silicate used was not provided in the report; therefore, it was calculated using stoichiometry. All chemical production data were sourced from Ecoinvent, representing the global or rest of the world production averages owing to the unavailability of local production datasets in the database. Table 6-5 lists the chemicals used to treat each functional unit.

Table 6-5: Chemical inputs of the single-source surface water system

Chemical	Ecoinvent Process	Mass[kg/m^3]
Lime	Lime, hydrated, loose weight(ROW)| market for lime, hydrated, loose weight | Cut-off, U	0.0367
Aluminium sulphate	Aluminium sulphate, without water, in 4.33% aluminium solution state {GLO}| market for aluminium sulphate, without water, in 4.33% aluminium solution state | Cut-off, U	0.00519
Sodium aluminate	Sodium aluminate, powder {GLO}| market for sodium aluminate, powder | Cut-off, U	0.000866
Sodium Silicate	Sodium silicate, without water, in 37% solution state(ROW) | Market for sodium silicate, without water, in 37% solution state | Cut-off, U	0.00123
Chlorine	Chlorine, liquid ROW | market for chlorine, liquid | Cut-off, U	0.00360

6.4.3.5 Energy consumption

Table 6-6 summarises the monthly energy usage in water production and distribution processes from 2020 to 2023. The CNWTW operates seasonally and is affected by changes in the water levels within the Table Mountain dams. Electricity consumption was notably lower in 2022 and 2023 than in 2020 and 2021, respectively. Therefore, our study uses the consumption rates of 2020 and 2021 as they represent the maximum electricity consumption.

Table 6-6: Electricity consumption of the Hout Bay single-source system

Month	**Electricity consumption in kWh**			
	2020	**2021**	**2022**	**2023**
January	9 684	14 990	10 849	5 466
February	8 366	13 299	9 558	4 899
March	10 957	16 019	11 754	6 519
April	12 077	14 691	10 259	6 293
May	14 426	16 633	8 850	6 649
June	16 001	16 541	9 344	8 803
July	16 307	19 809	10 609	12 579
August	17 369	20 836	11 491	11 915
September	15 447	18 910	9 841	12 198
October	14 132	17 160	7 435	12 710
November	14 795	15 181	6 471	5 841
December	16 045	12 557	6 177	-

The average annual energy consumption during this period was 181 MWh. Based on an annual water delivery of 900 035.25 m^3 in the surface water system scenario, the average electricity consumption was 0.201 kWh/m^3. The electricity mix dataset *Electricity, high voltage {ZA}| electricity market, and high voltage | Cut-off, U* were used to account for the local generation and transmission of electricity within South African geography.

6.4.3.6 Distribution energy

The study could not differentiate between the energy consumption of the distribution system and that of the treatment plant because the utility does not measure it separately. Therefore, the energy consumed in the water distribution is a part of the overall water production process and is included in the total electricity consumption provided in Table 6-6.

6.4.4 Hybrid source: Surface water

6.4.4.1 Construction

The hybrid system supplemented surface water with seawater for toilet flushing. Thus, the amount of surface water abstracted, treated, and distributed was less than that in the base-case scenario (using surface water as the single source). The surface water hybrid-source processes represent the water treatment works and distribution network infrastructure. The infrastructure inputs of this new system, including the waterworks infrastructure and distribution networks, were assumed to be the same as those of the base-case surface water system. However, the inventories per functional unit differed because of differences in annual water consumption.

Table 6-7 summarises the construction input inventories for the water treatment plant of the hybrid source surface water system.

Table 6-7: Inputs of the water treatment works infrastructure per functional unit

Material	Ecoinvent process	Amount [kg/m^3]
Aluminium	Aluminium, wrought alloy(ROW)| market for aluminium, wrought alloy | Cut-off, U	0.0000455
Charcoal	Charcoal {GLO}| market for charcoal | Cut-off, U	0.00000208
Concrete	Concrete, normal strength {ZA}| market for concrete, normal strength | Cut-off, U	0.0000198
Copper	Copper, cathode {GLO}| market for copper, cathode | Cut-off, U	0.0000140
Glassfibre reinforced plastic.	Glass fibre reinforced plastic, polyester resin, hand lay-up {GLO}| market for glass fibre reinforced plastic, polyester resin, hand lay-up | Cut-off, U	0.000129
Polyethylene	Polyethylene, linear low density, granulate {GLO}| market for polyethylene, linear low density, granulate | Cut-off, U	0.00000273
Polyurethane	Polyurethane, rigid foam {GLO}|marketfor polyurethane, rigid foam | Cut-off, U	0.0000114
Polyvinyl chloride	Polyvinylchloride, bulk polymerised {GLO}| market for polyvinylchloride, bulk polymerised | Cut-off, U	0.0000771
Steel	Reinforcing steel {GLO}| market for reinforcing steel | Cut-off, U	0.00167
Sand	Silica sand {GLO}| market for silica sand | Cut-off, U	0.000280
Steel	Steel, low-alloyed, hot rolled {GLO}| market for steel, low-alloyed, hot rolled | Cut-off, U	0.000793

6.4.4.2 Distribution Infrastructure

This section details the inventory of the hybrid source system distribution network.

Table 6-8: Inputs of Hybrid Water Source Distribution Infrastructure

Materials	Ecoinvent Process	Amount [kg/m^3]
HDPE	Polyethylene, high density, granulate {GLO}| market for polyethylene, high density, granulate | Cut-off, U	0.00132
PVC	Polyvinylchloride, bulk polymerised {GLO}| market for polyvinylchloride, bulk polymerised | Cut-off, U	0.00748
Steel	Steel, low-alloyed, hot rolled {GLO}| market for steel, low-alloyed, hot rolled | Cut-off, U	0.00155
Asbestos cement	Fibre cement corrugated slab {GLO}| market for fibre cement corrugated slab | Cut-off, U	0.0315
Concrete	Concrete, 30MPa {ZA}| concrete production, 30MPa, with cement, CEM II/B-V | Cut-off, U	0.00000389

6.4.4.3 Operational Inventories

The chemicals and energy consumed in the hybrid surface water system were lower than those in the single-source system because of the reduced water volume that required treatment. The operational inventories of the water production dataset are listed in Table 6-9.

Table 6-9: Chemical inputs of the surface water system of the hybrid system

Chemical	Ecoinvent Process	Amount [kg/m^3]
Lime	Lime, hydrated, loose weight (ROW)| market for lime, hydrated, loose weight | Cut-off, U	0.0273
Aluminium sulphate	Aluminium sulphate, without water, in 4.33% aluminium solution state {GLO}| market for aluminium sulphate, without water, in 4.33% aluminium solution state | Cut-off, U	0.00386
Sodium aluminate	Sodium aluminate, powder {GLO}| market for sodium aluminate, powder | Cut-off, U	0.000643
Sodium Silicate	Sodium silicate, without water, in 37% solution state (ROW) | Market for sodium silicate, without water, in 37% solution state | Cut-off, U	0.000913
Chlorine	Chlorine, liquid ROW | market for chlorine, liquid | Cut-off, U	0.00270

6.4.5 Hybrid source: Seawater system

6.4.5.1 Construction inventory

The hypothetical seawater system comprises various components, including an intake pump station, a treatment unit, a storage reservoir, and distribution pipelines (Liu *et al.,* 2016).

6.4.5.1.1 Intake and treatment works construction

The analysis was based on the same seawater works as in the study by Liu *et al.* (2016). Similarly, this study assumed that the seawater treatment station includes a pump station integrated with chlorine dosing. Additionally, two screens, measuring 5 mm and 3 mm aperture widths, were included for debris removal.

The Ecoinvent infrastructure transformation process for a pump station with a capacity of 1 766 m^3/d and a 70-year design life was selected to represent the seawater treatment infrastructure. This dataset was scaled down to represent a 1 000 m^3/d pump station required to pump seawater, the inventory of which is presented in Table 6-10.

Table 6-10: Inventory for the intake pump station and treatment unit

Material	Ecoinvent Process	Amount	Unit
Aluminium	Aluminium, cast alloy {GLO}| market for aluminium, cast alloy | Cut-off, U	0.00000827	kg/m^3
Aluminium	Aluminium, wrought alloy {GLO}| market for aluminium, wrought alloy | Cut-off, U	0.0000176	kg/m^3
Brass	Brass (ROW)| market for brass | Cut-off, U	0.00000148	kg/m^3
Bronze	Bronze {GLO}| market for bronze | Cut-off, U	0,00000247	kg/m^3
Iron	Cast iron {GLO}| market for cast iron | Cut-off, U	0.0106	kg/m^3
Cement	Cement, unspecified {ZA}| market for cement, unspecified | Cut-off, U	0.1241	kg/m^3
Concrete	Concrete, normal strength {ZA}| market for concrete, normal strength | Cut-off, U	0.000535	m^3/m^3
Copper	Copper, cathode {GLO}| market for copper, cathode | Cut-off, U	0.00202	kg/m^3
Glass	Flat glass, coated (ROW)| market for flat glass, coated | Cut-off, U	0.000759	kg/m^3
Concrete	Lean concrete (ROW)| market for lean concrete | Cut-off, U	0.0000378	m^3/m^3
PVC	Polyvinylchloride, bulk polymerised {GLO}| market for polyvinylchloride, bulk polymerised | Cut-off, U	0.00274	kg/m^3
Steel	Reinforcing steel {GLO}| market for reinforcing steel | Cut-off, U	0.0422	kg/m^3
Steel	Steel, low-alloyed, hot rolled {GLO}| market for steel, low-alloyed, hot rolled | Cut-off, U	0.00756	kg/m^3
Rubber	Synthetic rubber {GLO}| market for synthetic rubber | Cut-off, U	0.000000987	kg/m^3

6.4.5.2 Distribution Construction

The seawater distribution system includes the seawater reservoir and the pump necessary to pump water from the treatment station to the reservoir.

6.4.5.2.1 Storage

The seawater storage reservoir is assumed to be constructed of reinforced concrete with a capacity of 1 000 m^3, which coincides with the daily flush water requirement for the residents of Hout Bay. The 1 000 m^3 /d *seawater storage ZA| water storage construction | Cut-off, U* dataset was adapted from the *Water Storage (ROW)| water storage construction | Cut-off, U* dataset representing a 2 500 m^3/d storage reservoir using Equation 6-1. Table 6-11 lists the inventory inputs for constructing a seawater storage reservoir.

Table 6-11: Inputs of the seawater storage reservoir infrastructure per functional unit

Material Input	Ecoinvent Process	Unit	Amount
Concrete	Concrete, normal strength {ZA}| market for concrete, normal strength | Cut-off, U	m^3/m^3	0.00132
Concrete	Lean concrete {ROW} | market for lean concrete | Cut-off, U	m^3/m^3	0.0000550
Steel	Reinforcing steel {GLO}| market for reinforcing steel | Cut-off, U	kg/m^3	0.118
Steel	Steel, low-alloyed, hot rolled {GLO}| market for steel, low-alloyed, hot rolled | Cut-off, U	kg/m^3	0.00852

6.4.5.2.2 Distribution pipeline network

The seawater distribution network was estimated based on existing potable water distribution networks. It was assumed that the pipe distribution network was made of HDPE because the corrosive nature of seawater can reduce the life of concrete and iron pipes. The amount of material required per functional unit over a 60-year design life is presented in **Table 6-12**.

Table 6-12: Inputs of the seawater distribution network per functional unit

Material	Ecoinvent Process	Amount [kg/m^3]
HDPE	Polyethylene, high density, granulate {GLO}| market for polyethylene, high density, granulate | Cut-off, U	0.0501

6.4.5.3 Operation Inventory

6.4.5.3.1 Chemical consumption

Chlorine is the only chemical required for the treatment of seawater and is of an appropriate quality for toilet flushing. This chemical reduces the microbial activity of raw seawater. A chlorine dosage of 6 mg/L was adopted from a study by Leung *et al.* (2012).

6.4.5.3.2 Energy consumption

The treatment and distribution stages of water consumption require energy. Raw seawater also undergoes a screening process prior to chlorination that requires power. The energy needed for the treatment was calculated using data from the literature. We assumed an energy consumption of 0.0025 kWh/m^3 for the seawater production (Leung *et al.* 2012). As in the other processes, the electricity production process utilised was: *Electricity, low voltage {ZA}| market for electricity, low voltage | Cut-off, U was used to represent the energy.*

6.4.5.3.3 Distribution energy

The energy required to distribute seawater was based on the estimates. The daily seawater requirement or flow rate (Q) was 0.00932 m^3/s. Based on the continuity equation for flow rate (Q) with a velocity of 1.2 m/s, the estimated pipeline diameter is 100 mm, following design standards, for a length of 4 727 m. The frictional head was calculated based on the Darcy–Weisbach equation, and minor losses due to pipe bends and materials were assumed to be 30% of the major losses (Liu *et al.*, 2019). Using the Bernoulli equation, the total head, including elevation differences, major losses due to friction, minor losses, and household pressure head, was 248 m. The power consumption required to overcome the energy head is 37.8 kW at 60% efficiency, assuming the pump operates as expected. Therefore, the energy consumption for the continuous pumping period was 907.2 kWh. Therefore, the annual consumption of distributed energy was 331 MWh.

Table 6-13: Distribution network properties

Properties	Unit	Amount
Flowrate (Q)	m^3/s	0.00934
Pipe diameter	m	0.1
Total head	m	248
Pump power consumption (P)	kW	37.8
Annual energy consumption	MWh	331

A summary of the seawater operation inventory is presented in Table 6-14.

Table 6-14 Seawater system operation inventory

Input	Amount	Units
Chlorine	0.006	kg/m^3
Energy	5.81E-05	kWh/m^3

All the chemical, energy, and infrastructure input processes described above were used to build a comparative system inventory for the water treatment and distribution stages.

6.5 Life Cycle Impact Assessment (LCIA)

The LCIA constitutes the third phase of LCA, and its primary objective is to offer more comprehensive insights into the inventory table generated during the inventory analysis phase. LCIA translates the inventory into environmental impact categories pertinent to the evaluated product or service. It converts elementary flows from inventory into potential impacts on ecosystems, humans, and resources (Matthews *et al.*, 2014). This conversion is essential because elementary flows, although representing quantities emitted or resources used, cannot be directly compared with one another regarding their relative contribution to environmental impact (Rosenbaum *et al.*, 2018). This study considered three mandatory stages of the LCIA specified in ISO 14044, highlighted as follows.

1) **Impact category selection:** Impact categories, characterisation indicators, and models were selected using ReCiPe midpoint hierarchical methodology. The following impact categories were of interest based on the goal and scope of the analysis and geography, as mentioned in the scope definition: global warming potential, land use, stratospheric ozone depletion, fossil resource scarcity, human toxicity, carcinogenicity, and freshwater consumption.
2) **Classification:** In the classification phase, inventory results were assigned to the impact categories they contributed to. This was performed using the SimaPro software based on expert preprogrammed classification parameters.
3) **Characterisation:** Characterisation is the final mandatory step of an LCIA. The software also conducted this analysis based on the ReCiPe midpoint characterisation factors. In this stage, the elementary flows within the LCI were evaluated in terms of their contribution to environmental impacts. Each elementary flow E, categorised within a specific impact category C (which represents a pertinent environmental concern), is multiplied by its respective characterisation factor CF. The results are summed up for all relevant emissions or resource extractions. This produces an impact score IS for the environmental impact category, expressed as a standard unit shared by all elementary flows within the same impact category (Matthews *et al.*, 2014), as shown in Equation 6-2.

$$\boldsymbol{IS_C = \sum_I CF_i * E_i} \qquad \textbf{6-2}$$

6.5.1 Hypothesised impact categories.

The following section discusses the impact categories hypothesised to be significant in this study based on a previous investigation by Liu *et al.* (2016), who conducted a similar comparative study of alternative water sources.

6.5.1.1 Global warming

High levels of greenhouse gases, including carbon dioxide, methane, nitrogen oxides, and chlorofluorocarbons, trap infrared radiation from the sun, thereby preventing the Earth's surface from reflecting it (Hauschild *et al.*, 2018).This results in an unnatural increase in atmospheric temperature, commonly known as global warming (Pennington *et al.*, 2004). Human activities drive global warming, and their effects include temperature rise, desertification, the proliferation of extreme weather events, and rising sea levels. The global warming potential is typically quantified in kilograms of CO_2 equivalents (kg CO_2eq) (Acero *et al.*, 2016).

6.5.1.2 Land Use

Land use impacts include human activities in the soil, such as agricultural and forestry production, urban development, and mineral extraction. Although these activities are crucial for the development and sustenance of human life, they can have detrimental consequences for biodiversity and affect the ecological function of the soil (Hauschild *et al.*, 2018). The impact on land use is quantified as the product of area and time in years, given in m^2a crop eq.

6.5.1.3 Ecotoxicity

Substance emissions may be toxic, depending on the quantity released and the potential to reach and impact organisms (Hauschild *et al.*, 2018). Industrial chemical emissions are associated with all production processes and may affect organisms in terrestrial, freshwater, and marine ecosystems (Hauschild *et al.*, 2018). In the ReCiPe methodology, these impacts are characterised relative to 1.4 dichlorobenzene (1.4 DCB).

6.5.1.4 Fossil resource depletion

The impact of fossil resource depletion results from the consumption of non-renewable resources such as fossil fuels. In ReCiPe, this was measured relative to a kilogram of oil equivalent (kg oil eq.).

6.5.1.5 Water consumption

The freshwater consumption impact category quantifies potential water shortages experienced by users (humans and ecosystems) when water is consumed in a particular region. This is performed using scarcity indicators and is characterised based on the volume measured in cubic meters (Hauschild *et al.*, 2018).

6.6 Life cycle impact assessment characterisation results

Figure 6:5 illustrates the potential impact contribution of the comparative analysis of dual- and single-source systems. The dual system uses seawater for toilet flushing and potable water for all other purposes. It has a 25% lower freshwater consumption potential. However, this advantage is accompanied by a higher environmental impact in five of the seven impact categories analysed; hence, the dual system was used as the reference point. These categories include global warming (38%), terrestrial, freshwater, and marine ecotoxicity (20%-33%), land use (24%), and fossil resource scarcity(44%). However, the contribution of the single-source system to the impact of mineral resource scarcity was 4% greater than that of the dual-source system.

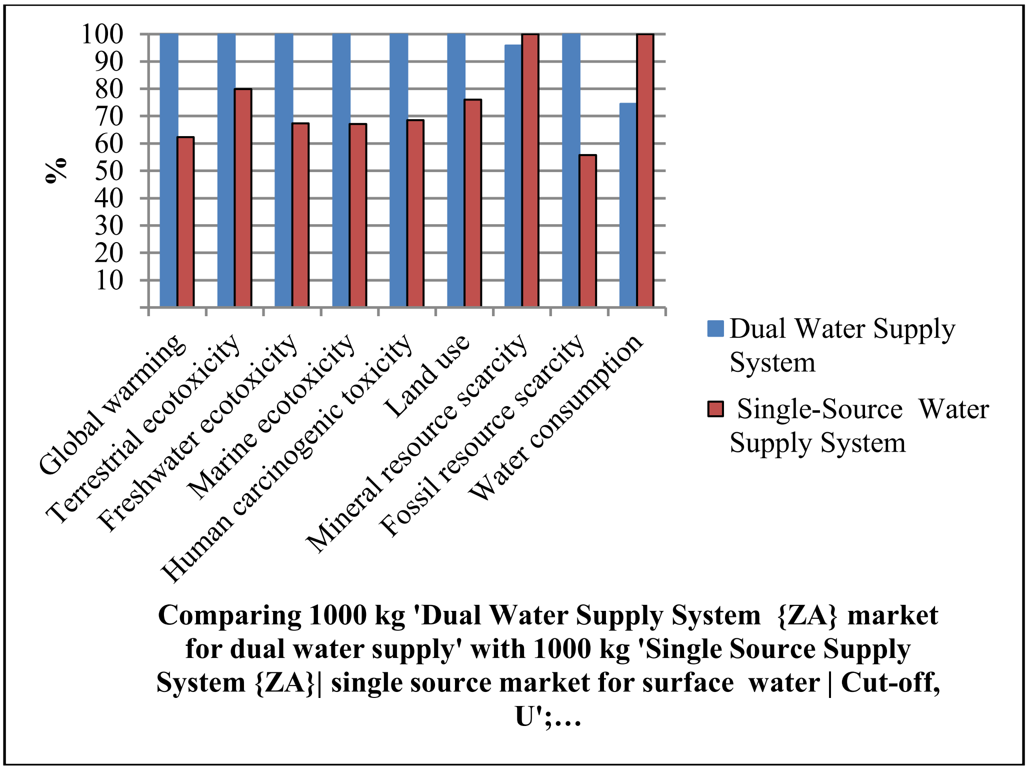


Figure 6:5: Comparison of characterisation results of the dual and single source systems

The absolute characterisation results comparing dual- and single-source water systems for the impact categories of interest are presented in Table 6-15. The following sections discuss each impact category in detail, highlighting their main contributing processes.

Table 6-15: Absolute characterisation results of the ReCiPe midpoint (H) methodology

Impact category	Unit	Dual water source system	Single water source system
Global warming	kg CO2eq.	0.6740	0.4200
Terrestrial Ecotoxicity	kg 1.4DCB	1.3100	1.0500
Freshwater Ecotoxicity	kg 1.4DCB	0.0202	0.0136
Marine Ecotoxicity	kg 1.4DCB	0.0276	0.0185
Land use	m^2a crop eq.	0.0192	0.0146
Mineral Resource Scarcity	kg Cu eq.	0.0107	0.00112
Fossil Resource Scarcity	kg oil eq.	0.2050	0.1140
Water consumption	m^3	0.7200	0.9660

6.6.1 Global Warming

Our analysis revealed that the dual system's global warming potential (GWP) is approximately 38% greater than that of the surface water system. This impact is attributable to the dual pumping of separate water streams into the households. The effect of distribution is primarily associated with electricity generation and distribution. Fossil fuels dominate the electricity mix in South Africa. Consequently, it has a high potential for carbon dioxide, methane, and nitrogen dioxide emissions, contributing 71% of the GWP impact for the dual system and 57% for the single-source system.

Lime, a chemical used in conventional water treatment, can contribute approximately 9% to global warming effects in the surface water system compared to 3% in the dual system, where a lower volume of water is treated to potable quality.

The dual system had a 3% global warming impact linked to ethylene production. In contrast, it contributed 1.75% to the surface water system scenario. This difference may be due to the additional HDPE pipeline used to distribute the seawater in the dual system. The single-source system consists of more infrastructure components than the simple seawater treatment of a dual-source water system. Consequently, the infrastructure process associated with cement and clinker production required for concrete production contributed 3.5% of the global warming impact on tap water production, compared to approximately 1.8% in the dual system.

6.6.2 Freshwater consumption

The dual system utilises a supplementary seawater supply for waterborne sanitation, thereby reducing freshwater withdrawal. Consequently, this could lead to a 25% lower freshwater withdrawal rate than that of the conventional single-source system. Water extraction from marine environments within LCA circles does not affect the environment (Opher *et al.*, 2018). Hence, it is considered to be an environmental benefit because it offsets the depletion of freshwater resources.

6.6.3 Terrestrial ecotoxicity

A significant finding is the dual system's 20% higher potential contribution to terrestrial ecotoxicity than the single-source system. This finding underscores the need for effective mitigation strategies, mainly because both scenarios show that terrestrial ecotoxicity is primarily associated with background processes in materials and chemical production processes on a global scale. Processes such as copper smelting, brake wear emission treatment, and electricity production are critical areas of potential improvement.

6.6.4 Freshwater ecotoxicity

The dual system had a higher freshwater ecotoxicity impact than that of the surface water system. The effect of freshwater ecotoxicity is linked to the mining industry. The treatment of coal spoils (47.4%) and hard coal mining operations (9.94%) of the dual system had higher impacts than those of the single-source system (37.1% and 7.71%, respectively). Other contributing processes (1-9%) were associated with treating sulfidic tailings from copper mining, basic oxygen furnace slag, and coal slurry, with each process emanating from different geographies where materials and chemicals are sourced, as assumed in this study.

Improvements in the mining industry's waste treatment methods can reduce the impact of toxicity. Sulphide tailings can be repurposed in infrastructure as a cement replacement. Similarly, the impact of coal spoils can be reduced by implementing more environmentally friendly mining practices and technologies and rehabilitating coal ash to produce fabricated soils.

6.6.5 Marine Ecotoxicity

The dual system had a higher marine ecotoxicity potential than that of the single-source water system. It was found that 48% of this potential came from the spoil from hard coal mining, compared to 38% from the single-source water system. In addition, the treatment of hard coal ash accounted for 10.2% of the potential of the dual system, whereas that of the single-source water system was 8%. Various processes with contributions ranging from 1-8% were associated with sulfidic tailings from copper mines, basic oxygen furnace slag, and coal slurry treatment in both scenarios. The impact of marine ecotoxicity may be reduced, as suggested for freshwater ecotoxicity, given that it emanates from mining activities.

6.6.6 Land use

The land-use impact was dominated by the background process of sand extraction required for construction, accounting for 36% of the dual system and 33% of the single-source system. The subsequent dominant processes were coal mining, coal preparation, loss of vegetation, and infrastructure construction, which accounted for 7% and 11.7% of the dual- and single-source water system land use impacts, respectively. The sawing of logs and veneer wood used in the construction contributed less than 2% to both systems. Land use impacts can be mitigated by rehabilitating areas affected by industrial and infrastructure development to restore their natural environment once their operational life has ended, enabling the land to support the ecological functions of plant and animal growth.

6.6.7 Fossil resource scarcity

Water systems have contributed to the decline in fossil resource reserves. The resources of interest in this analysis include coal used in electricity production and ethylene associated with plastic manufacturing. Hard coal mine operation processes contributed 71% to dual supply and 64. 7% of the single source system, ethylene production contributed 12 % and 8%, respectively. Finally, the preparation of petroleum and coal contributes to less than 5% of the fossil resource scarcity in each scenario.

6.6.8 Mineral resource scarcity

The extraction of minerals from the natural environment to produce various components or materials influences the scarcity of mineral resources. In the dual system, iron ore and ferronickel extraction contributed 23.4% and 18.2%, respectively, to this impact, whereas in a single-source conventional water system process, these processes contributed 21.3% and 17%, respectively. The bauxite used in steelmaking accounts for 20% of the total impact of single-source systems compared to 13% in the dual system, owing to its use in steel pipelines and reinforcing steel. Other processes, such as mining and beneficiating copper and iron ores, contributed negligibly to this impact in both systems.

6.7 Comparison of the potential impact of the urban water stages

Figure 6:6 illustrates the environmental impact profiles of different stages in a single-source water system. This study compared the combined water treatment and distribution processes to the water production process for a functional unit of 1 m^3 to determine the impact of the distribution stage alone. The water production and distribution stage had the most significant impact and was used as the reference point. The distribution stage accounted for less than 50% of the potential effects of global warming (38%), freshwater ecotoxicity (38%), marine ecotoxicity (38%), mineral resource scarcity (45%), and fossil resource scarcity (46%). However, it significantly impacts land use (51%) and terrestrial ecotoxicity (70%) more significantly. Therefore, in a single-source system, the water treatment stage has a more significant impact than the distribution stage.

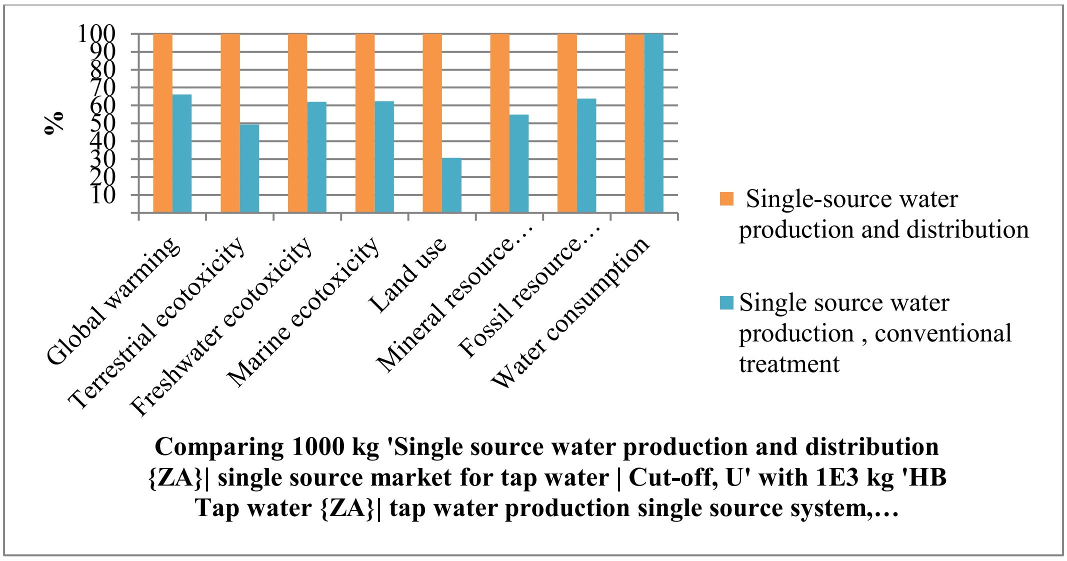


Figure 6:6: Comparison of the different stages of the Hout Bay System

Figure 6:7 shows the comparative environmental impact profile of the dual system's water production, distribution, and production stages. Relative to the water production and distribution processes, the water production process contributed 20-40% of the impacts in different categories. On this basis, the distribution stage alone contributed 60-80% of the effects. This indicates that the water distribution in the dual system is the primary source of the impact, mainly because of the additional pumping energy required to transport the treated seawater to the storage reservoir.

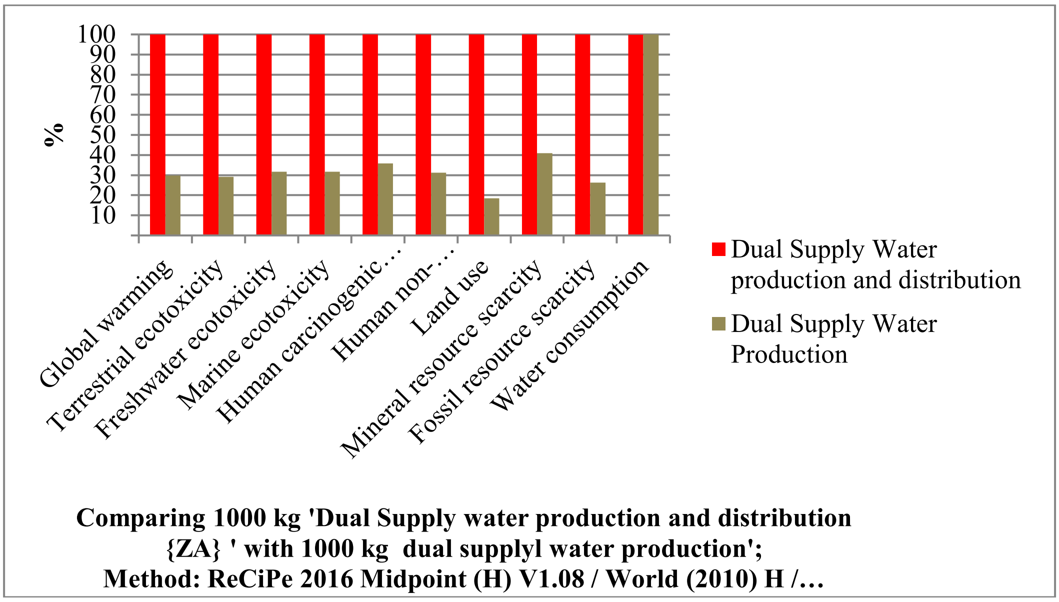


Figure 6:7: Comparison of characterisation results at different stages of the dual system.

6.8 Validation of the characterisation results

LCA methodologies carry a degree of uncertainty. Therefore, an additional analysis was conducted using the IMPACT World+ methodology to understand the impact of using different methodologies on the results. The chosen alternative was the only global midpoint method available for SimaPro. Figure 6-8 shows the relative characterisation results. Based on the impact profile, the IMPACT World+ assessment corroborates the results obtained using the ReCiPe methodology. Dual systems have more significant potential impacts on long- and short-term climate change, fossil and nuclear energy use, freshwater ecotoxicity, land transformation, and occupation.

By contrast, conventional single-source water systems significantly impact mineral resource use and water scarcity. However, the absolute results differ, as illustrated in Table 1-16 (ReCiPe) and Table 1-17 (IMPACT World+), likely because of the varying characterisation models and impact modelling utilised by different methodologies (Huijbregts *et al.*, 2017). For example, ReCiPe assesses freshwater by the volume extracted from a country or region. In contrast, IMPACT World+ assesses water resource depletion on a global scale.

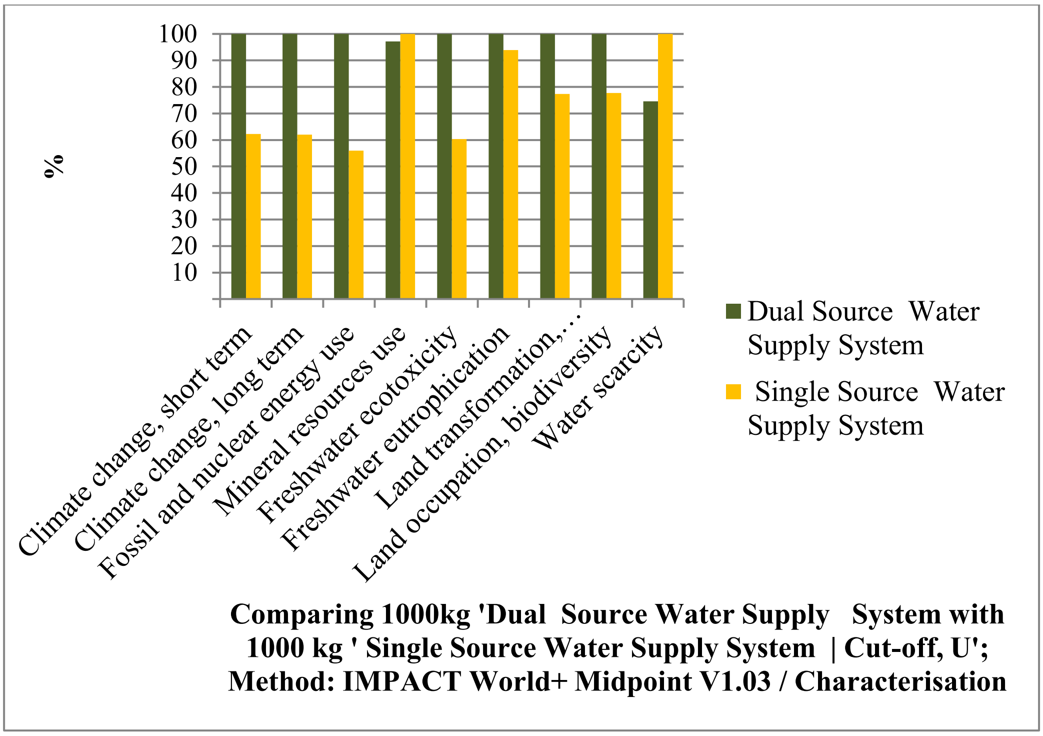


Figure 6:8: Characterisation results using the IMPACT World + method

The absolute characterisation results obtained using IMPACT World + are presented in Table 6-16.

Table 6-16: Absolute characterisation results using IMPACTWorld + methodology.

Impact category	Unit	Dual sources system	Single-source water system
Climate change short term	kgCO_2eq	0.674	0.42
Climate change long-term	kg1.4DCB	0.659	0.409
Fossil and nuclear energy	kg1.4DCB	9.89	5.54
Mineral resource use	kg1.4DCB	0.00839	0.00863
Land transformation, biodiversity	m^2yr, arable	0.00133	0.00103
Land occupation, biodiversity	m^2yr, arable	0.0119	0.0926
Water scarcity	m^3world eq	27.4	36.8

6.9 Discussions

Our study found that the dual systems yielded higher environmental impacts than the single-source systems, except for freshwater consumption and mineral resource depletion. Implementing a dual water system in Hout Bay could reduce freshwater consumption by 25% but increase other impacts by 20-44%. Similarly, Lane *et al.* (2015) found that using a mix of alternative sources reduced the freshwater extraction stress index by 62% but increased other impacts by 4-93%. In addition, Godskesen *et al.* (2013) found that after normalisation, a seawater source significantly reduced the impact of freshwater depletion, making it more favourable than a groundwater source system. Thus, freshwater withdrawal impact is a significant parameter in water systems relative to a person's global emissions.

Our findings suggest that seawater distribution significantly affects the dual systems more than the treatment stage. This outcome aligns with the study of Liu *et al.* (2016), which suggests that dual systems using seawater for flushing toilets may not be feasible in coastal areas far from the shore because of energy consumption in the intake and disposal pipelines. However, when all stages of an urban water system are considered, activated sludge wastewater treatment is the primary impact source for single-source surface water systems compared to distribution systems and potable water treatment (Lane *et al.*, 2015; Liu *et al.*, 2019). Given that our scenario did not assess the impact of wastewater treatment, the potable water treatment process was responsible for approximately 60-80% of the effects on surface water systems. Moreover, detailed direct comparisons with literature could not be made.

The results of this study suggest that the impacts of water systems extend beyond the effects of land use and water consumption, which water utilities typically manage. The results demonstrated that chemical and energy production background processes beyond the purview of the water utility have significant environmental implications. Consequently, increased sustainability in these sectors is crucial for mitigating the water-related impacts. Previous studies have suggested that electricity-related impacts may be reduced by incorporating renewable energy sources, such as wind and photovoltaic energy, into the electricity grid (Goga *et al.*, 2019; Kobayashi *et al.*, 2020; Raluy *et al.*, 2005). However, an in-depth analysis of the lifecycle of these systems is necessary to avoid burden shifting.

Several factors limited the scope of this study. First, owing to a lack of data, the study relied heavily on the Ecoinvent database, using the global average instead of primary datasets. This approach may have led to an overestimation of the impacts, increasing the uncertainty of the findings by not accounting for variations in processes across different locations (Xue *et al.*, 2019). Second, the analysis focused solely on the water supply aspect of the urban water system, excluding the wastewater management stages that could have ecotoxicity and eutrophication effects on marine life. Additionally, the study only addressed environmental impacts, excluding financial and social sustainability, which should be assessed through life cycle cost analysis (LCCA) and social assessment (LCSA) for more comprehensive findings that better support sustainable water management and are acceptable to communities. Finally, the assessment only considered water sources applicable to centralised scenarios, leaving out other alternative water sources, such as rainwater harvesting and greywater, applicable for on-site reuse.

7 Using Discrete Choice Modelling to Determine The Willingness to Pay for Alternative Water Sources for Toilet Flushing

7.1 Introduction

The research aim was to investigate people's willingness to pay to use alternative water sources to flush toilets. To this end, the discrete choice experiment methodology approach by Hanley *et al.* (2001) was used to investigate the research objectives. The chapter begins by describing the characteristics of the decision problem as recommended by Holmes et al. (2017). It then explains the stated choice survey's aim and details how the stated choice survey was defined, refined, and constructed. The data collection process follows, after which the models used to analyse and interpret the data are discussed.

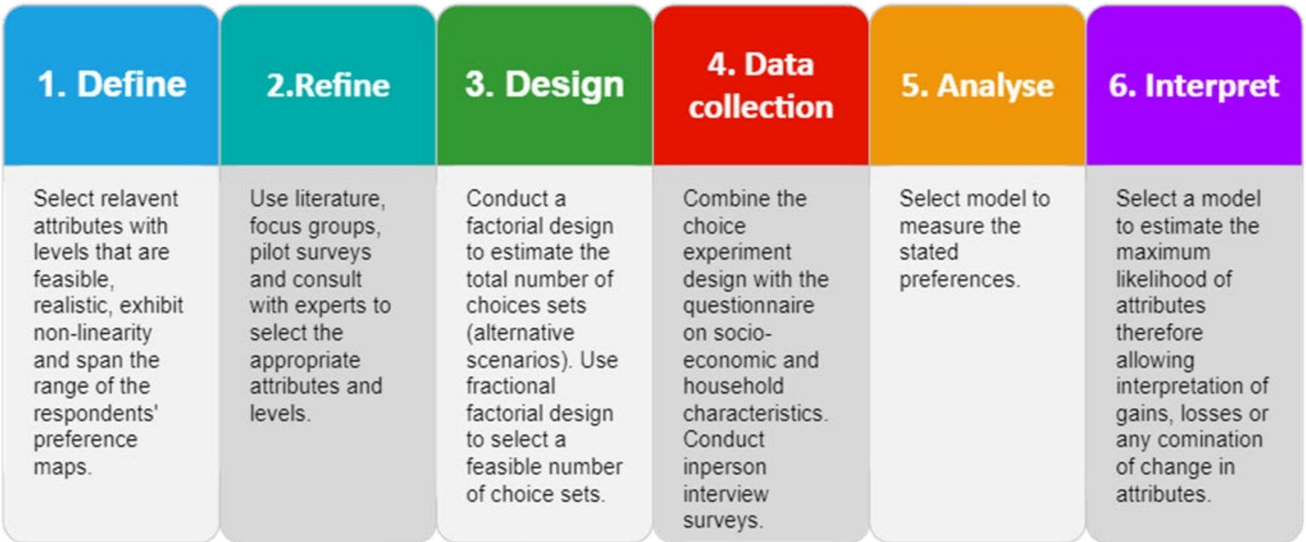


Figure 7-1: Choice experiment methodology (adapted from Hanley *et al.* 2001)

7.1.1 Characteristics of the decision problem

7.1.1.1 Water supply

The City of Cape Town's (CCT's) water strategy centres on inclusion, drought resilience, and sustainability (CCT, 2020). Inclusion was related to ensuring safe access to water and sanitation for all residents. The 2040 vision for a drought-resilient city includes water conservation, reliable and diverse water sources, and shared regional water resources. Water-sensitive design is at the forefront of ensuring sustainable development of the City. Currently, the conventional water supply sources are groundwater (4%) and surface water (96%), and a limited amount of recycled water is used for industrial, commercial, and landscaping purposes. Recycled water is treated wastewater that can be reused for non-drinking water purposes. In the future, CCT plans to diversify the water supply mix to include groundwater (7%), surface water (75%), desalination (11%), and recycled water (7%) (CCT, 2020). Recycled water is planned at the city scale to recharge the Cape Flats Aquifer and augment the drinking water supply source. The intention is to blend recycled water with surface water before conventional treatment to produce drinking water.

7.1.1.2 Housing demand

The Human Settlements Strategy aims to address the housing requirements of its most vulnerable residents. The primary focus is on residents classified as the "gap market" (households earning R22,000 per month or less) and those living in inadequate shelters (CCT, 2021). From 1996 to 2016, the population of Cape Town rapidly increased. Consequently, expansion has increased formal settlements, comprising 60% of informal housing (CCT, 2021). Simultaneously, the remaining formal structures have additional dwellings in backyards, further compounding the demand for water and sanitation.

7.1.1.3 Ageing reticulation infrastructure

The water supply is distributed via approximately 10 700 km pipe network with numerous pump stations and reservoirs. The water used by domestic, industrial, and commercial users is collected via an approximately 9 300 km sewerage pipe network and pumped to numerous wastewater treatment facilities (CCT, 2018). Cape Town is engaged in a pipeline replacement program to address the ageing infrastructure and minimise pipe bursts, water leaks, and sewage spills. Each year, this initiative involves the replacement of approximately 30 km of sewer lines and 40 km of water mains (CCT, 2018).

7.1.1.4 Marine outfall sewers

South Africa discharges 300 ML/d of wastewater via seven marine outfall sewers (MOS). Consequently, the potable water used to flush toilets is effectively lost from the urban water supply system, including the opportunity for reuse. Four of the MOS are located in Cape Town, where untreated wastewater is discharged to the sea. Hout Bay, Sea Point, Camps Bay, and Robben Island have licences to discharge a combined total of approximately 55 ML/d of wastewater into the sea, and they were constructed between the 1970s and the 2000s.(CCT, 2018).

7.1.2 Survey aim

Replacing the ageing infrastructure allows exploring dual supply networks for drinking water and lower-grade water sources for other uses. For example, using different water sources can reduce the need for clean drinking water to flush toilets by 20%. The saved water can support the demand for drinking water for new housing developments. Furthermore, the City of Cape Town encourages the use of recycled water by pricing it at R10,78 less than the lowest monthly usage (R19,42 for $\leq$ 6kl) drinking water tariff, providing an opportunity to incentivise alternative water options. Therefore, understanding the stated preferences and willingness to pay for using alternative water sources for flushing toilets can help optimise water supply strategies and assist in policy reforms. Consequently, the survey used a discrete choice experiment to elicit preferences for using alternative water sources for flushing toilets.

7.2 Stated choice survey

We hypothesised that socioeconomic factors influence preferences and willingness to pay (WTP) for alternative water sources in flushing toilets. This section details the design of the discrete choice experiment, the administration of the survey, and the choice models used for parameter estimation.

7.2.1 Stated choice task construction and additional survey components

The stated choice survey comprises three sections, as illustrated below and summarised in the following subsections.

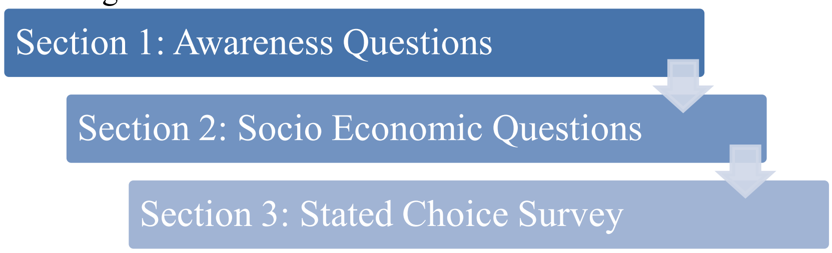


Figure:7:1: Stated choice survey layout

7.2.1.1 Section 1: Awareness questionnaire

A set of statements was presented to the respondents to gauge their knowledge of the water supply instead of explaining it, which could create bias. The statements were adapted from Bennett *et al.* (2016), and a 3-point Likert awareness scale was used. These statements covered the following themes: water supply, recycled water use, seawater use for flushing, and housing development needs. The modified awareness statements are as follows:

1. Most of Cape Town's water supply is collected in dams. The water is then treated and supplied as drinking water to households and businesses.
2. Water that goes down the drains from baths, sinks, and toilets is gathered through pipes and then cleaned and recycled for other uses.
3. In Cape Town, the recycled water is used for watering parks, in some factories and during building projects.
4. Seawater has been used in flushing toilets since the 1950s in Hong Kong.
5. In Hout Bay, water that goes down the drains from baths, sinks, and toilets is gathered through pipes and pumped into the sea.
6. In the future, most new housing developments in Cape Town will take place in areas such as Hout Bay, Khayelitsha and Atlantis.

7.2.1.2 Section 2: Socioeconomic characteristics

We collected evidence on household socioeconomic characteristics (Table 71)- to test whether these factors significantly affected the stated choices. Awad *et al*. (2021) conducted a choice experiment on consumer preferences for alternative water supplies and conservation policies. Their survey demographics included data on where respondents grew up to account for preference differences across generations and life experiences with water. We found that during the focus groups, the participant's knowledge of the 2015-2018 drought and their reuse of water to flush toilets made it easier to describe their preferences for the attributes of alternative water sources. Therefore, the survey included knowing whether the respondent lived in Hout Bay before 2018 to account for the influence that the experience of living through drought may have on preferences for alternative water sources.

Table 7-1: Socioeconomic Characteristics

Description	Options
Age	18 – 34 / 35 - 64 / > 65
Gender	male /female
Education	no schooling/completed matric/ diploma/degree
Ethnicity	white/black/coloured/Asian/other
Religion	Muslim/Christianity/Judaism/Hinduism/Buddhism/no religious affiliation/other
Income per month	R1 – R3 200 R3 201 - R12 800 R12 801 – R25 600 R25 601 – R102 400 >R102 401
Tenure (property ownership)	owner/renter
Household size	respondent to state the number of people living in the home 1/2/3/ >3
Suburb	Hout Bay/ Hangberg/ Imizamo Yethu
Length of stay	before 2018/ after 2018

7.2.1.3 Section 3: Stated choice survey

The stated choice survey presented three alternatives: seawater, recycled water, and drinking water (the current flushing conditions). Decentralised alternatives, such as greywater reuse, are unsustainable water sources during droughts and have not been considered. Similarly, rainwater harvesting has been excluded because of its decentralised nature and reliance on rainfall. Centralised water supply sources, such as recycled water and seawater, have been considered for their drought resilience (Furlong *et al.*, 2022; Li *et al.*, 2005).

7.2.1.3.1 Attributes

Focus groups were conducted to investigate the attributes associated with seawater and recycled water as alternative sources of flushing toilets. Furthermore, the attribute levels were categorical and supported by previous studies to describe clearly how each level's outcomes would impact the respondents. Moreover, the cost attribute was of economic consequence and based on current water pricing strategies.

The stated choice survey design included colour and odour attributes to test whether it resulted in findings similar to those of Amaris *et al.* (2021). The colour consisted of three levels, depicted by images from the Apha-Hazen scale. It was used to explain the range of the yellow colour level in the water. No colour, slight colour, or moderate colour are the levels for the attributes of colour. Similar to the study by Amaris *et al.* (2021), odour was described by the strength of a chemical odour in water. Three odour levels were used: none, slight, and moderate.

Some residents of Hong Kong complained about the stain remaining in the toilet bowl when using seawater to flush toilets (Li *et al.*, 2005). Furthermore, during the focus groups, some participants highlighted that the stain left by using greywater to flush toilets during the Cape Town 2015-2018 drought had a negative impact on their experience of using it as an alternative water source. Therefore, the stain attribute was included in the survey design with levels of none, slight, and moderate, which refer to the cleaning effort required to remove the stain.

The focus group discussions raised concern for the environment as a pivotal attribute for the preference for alternative water sources. Environmental impact was not part of the survey objectives. However, it was included, given the current public displeasure with discharging raw sewage into the sea (Overy, 2020). This attribute was framed as a preference for changes in sewage disposal practices rather than the environmental impact of different disposal practices. Ultimately, odour, colour, stain, cost, and wastewater disposal were considered and are summarised in Table 7-2.

Table 7-2: Water Supply Attributes

Attributes	Levels	Coding	Expected sign
Odour Intensity Chlorine odour (strong, sharp odour similar to the scent of bleach)	No odour: no noticeable odour Slight odour: noticeable but does not overpower other odours Moderate: odour is recognisable	Dummy coding Base = No odour [β_{ODOUR}]	Negative (-)
Water stain The water stain on the toilet bowl after flushing	None: no stain Mild: slightly noticeable little effort to remove Moderate stain: noticeable and may require cleaning effort to remove.	Dummy coding Base = No stain [β_{STAIN}]	Negative (-)
Water colour The water colour for flushing toilets	No colour Slight discolour Distinctly yellow	Dummy coding Base = No colour [β_{COLOUR}]	Negative (-)
Cost increase or decrease to monthly water bill cost	-R150, -R100, -R50, 0, +R25, +75	[β_{COST}]	Negative (-)
Wastewater disposal Disposal of all used water in the household	Discharged to sea without treatment Treatment and discharge to sea Treatment and reuse for toilet flushing	Dummy coding Base = No treatment [$\beta_{DISPOSAL}$]	Positive (+)

The baseline monthly water cost was established across different income groups. The cost of supplying 20% of the water consumed as either recycled water or seawater is estimated. The baseline monthly bill was discounted by 20%, and seawater and recycled water costs were added independently to estimate the future water bill. The future water bill for each alternative was compared with the baseline cost to establish a range for the cost attribute.

Viljoen (2016) studied household water use behaviour in Cape Town based on a sample of five representative suburbs, including informal settlements, low-income areas and middle/high-income areas. Data on the number of people per household and the associated monthly water use across different income categories were obtained from the findings of Viljoen (2016). The City of Cape Town's 2021/22-tiered tariff charges for potable water supply were used to estimate a typical monthly bill. The baseline monthly bill ranged from R350 to R900, and similar results were obtained using water use data based on plot size from Viljoen (2016). The CCT 2021/22 recycled water supply tariff of R8,00/kL was used to estimate the recycled water supply cost.

Tang *et al.* (2006) calculated the unit cost of seawater supply to be 5,88 HKD/kL based on the seawater quality that only required screening and chlorination as pretreatment before use. In this study, we assumed the same conditions in the absence of seawater-quality data. We applied a 6% escalation over 17 years and an exchange rate of 2,33 ZAR/HKD to produce a unit of cost R37,70/kL.

7.2.2 Discrete choice experiment design

The survey was based on an optimal d-efficient design, where stain, colour, odour, and water disposal attributes were modelled as nonlinear effects, and the cost attribute was coded as a continuous variable. The design included three labelled alternatives, each with an alternative specific constant (ASC), to encompass all unobserved factors that the attributes failed to capture. Ngene was used to generate a d-efficient design without priors, and a d-efficient design of 72 choice sets with a Derror of 0.002268 was achieved. SurveyEngene® was used to administer the survey. The design choice sets were subdivided into 12 blocks; thus, each respondent was presented with six choice sets to eliminate cognitive overburden (Rose *et al.*, 2009). Figure 7:2 shows a typical choice card that was presented to the respondents. The order of the alternatives was randomised to eliminate bias, and the choice sets did not include an opt-out option.

	Recycled Water Supply	Drinking Water Supply	Seawater Supply
Water stain	Moderate: noticeable with some effort to remove	None: No stain	Moderate: noticeable with some effort to remove
Colour of the flush water	Slight discolour	No colour	Distinctly yellow
Cost increase or decrease to monthly water bill	-R50	+R25	R0
Smell intensity of chlorine	Slight smell	No smell	No smell
Disposal of wastewater	Treat before discharge to sea	Discharge to sea without treatment	Treat before discharge to sea
Which would you choose?	Recycled Water Supply	Drinking Water Supply	Seawater Supply

Figure 7:2: Typical choice card

7.2.2.1 Data collection

Pre-testing methods were used to gather quantitative and qualitative data to finalise the preliminary survey design. Qualitative data were collected through focus groups and peer reviews, which were used to discuss concepts and language and help clarify scenarios (Johnston *et al.*, 2017). Quantitative data were collected through a pilot study.

7.2.2.2 Pilot survey

A pilot survey of the target population of the primary sample was conducted to gather quantitative data. The pilot survey was used to test survey comprehension, ease of answering choice sets, and completion times. Orm (2010) recommends a minimum pilot survey size of 50. A total of 132 responses were collected.

7.2.2.3 *Survey administration and sampling*
A private company was contracted to conduct in-person surveys of the heads of households in Hout Bay. The interviewees were trained to ensure they understood the scope, intention and permissible aspects of conducting the interviews.

An exogenous stratified random sampling strategy (ESRS) was implemented (Rose and Bliemer, 2013). The sampling strategy divides the sample population into mutually exclusive groups representing a portion of the total population. Louvier *et al.* (2002) recommend choosing any characteristic common to the population except choice. Hout Bay is comprised of three distinct suburbs. Accordingly, the population was stratified based on the location and access to flushing toilets. A random sample was drawn from each stratum to ensure a random sampling.

Some challenges were encountered during the pilot survey sampling. The door-to-door strategy was ineffective in the Hout Bay suburb because few people were willing to answer their doorbells. In the case of Hangberg and Imizamo Yethu, the personal safety and risk of tablet theft hampered the randomised door-to-door approach. The survey administration strategy was revised to intercept respondents from the shopping malls in Hout Bay.

7.2.2.4 *Sampling size*
The estimated sample size depends on the model type, the number of alternatives, attributes, and levels, the discrete choice experiment design, and likely parameter estimates (*a priori)* (Rose and Bliemer, 2013). Additionally, clarification on what needs to be measured to obtain a statistically significant result must be considered (Orme, 2010). All these factors must be considered when estimating the sample size.

Without prior parameter estimates, the sample size was estimated using Orme's rule of thumb (2005). The population set has been stratified according to households with private and shared sanitation services, and the sample size estimate was conducted using the rule of thumb equation (1),

$$\frac{nta}{c} \geq 500 \tag{7-2}$$

where n is the number of respondents, t is the number of choice sets, a is the number of alternatives per choice set (excluding non-alternatives), and c is the number of analysed cells. According to Orme (2010), when considering the main effects, c equals the largest number of attribute levels. If two-way interactions are allowed, c equals the largest product of the levels of any two attributes (Orme, 2010).

The rule of thumb equation was used to estimate the minimum sample size; therefore, the minimum sample size was 300, based on respondents being exposed to six choice sets. It was stratified as illustrated in Table 7-3.

Table 7-3: Preliminary Estimate of Total Survey Sample Size

Suburb	Households connected to sewerage system*	Proportion	Sample size
Hout Bay	4 350	48%	144
Hangberg	1 002	11%	33
IY	3 714	41%	123
Total	9 066	100%	300

* Census 2011 data

7.2.3 Model estimation

7.2.3.1 Random Utility Model

The classical multinomial logit model (MNL), linked to random utility theory, forms the basis of the modelling framework (Lancsar *et al.*, 2017). The utility equation is given by

$$\mathbf{U_{njt} = V_{njt} + \epsilon_{njt}} \qquad \textbf{7-3}$$

where U_{njt}, is the maximum utility for respondent *n,* presented with a choice set *t* flush water alternative j, which comprises a deterministic component V_{njt} and error term ϵ_{njt}. The Equation quantifies the deterministic component as follows:

$$\mathbf{V_{nj} = \delta_j + f(\beta, x_{nj}, z_n)} \qquad \textbf{7-4}$$

where δ is the constant associated with the alternative, $f(\,)$ indicates the assumptions of the model by including β, which explains the role that alternatives, choices and respondent characteristics influence the observed choice, x_{nj}, the choice sets and z the characteristics of the respondent. However, the MNL model has several limitations.

1) The error term and preference parameter are constant for all respondents.
2) The error term is distributed independently and identically across all alternatives and respondents using type I value distribution (Gumbel).

The following equations provide the utility model for the three alternatives.

$$\begin{aligned}
&\mathrm{U\ (seawater)} = \delta_{seawater} \\
&+ \beta_{\mathrm{stain}}\left(\mathrm{stain}_{\mathrm{seawater}_{\mathrm{none}}} == 1\right) + \beta_{\mathrm{stain}}\left(\mathrm{stain}_{\mathrm{seawater}_{\mathrm{slight}}} == 2\right) \\
&+ \beta_{\mathrm{stain}}\left(\mathrm{stain}_{\mathrm{seawater_moderate}} == 3\right) \\
&+ \beta_{\mathrm{colour}}(\mathrm{colour}_{\mathrm{seawater}_{\mathrm{none}}} == 1) + \beta_{\mathrm{colour}}(\mathrm{colour}_{\mathrm{seawater}_{\mathrm{slight}}} == 2) \\
&+ \beta_{\mathrm{colour}}(\mathrm{colour}_{\mathrm{seawater_moderate}} == 3) \\
&+ \beta_{\mathrm{cost}}\mathrm{cost}_{\mathrm{seawater}} \\
&+ \beta_{\mathrm{odour}}(\mathrm{odour}_{\mathrm{seawater}_{\mathrm{none}}} == 1) + \beta_{\mathrm{odour}}(\mathrm{odour}_{\mathrm{seawater}_{\mathrm{slight}}} == 2) \\
&+ \beta_{\mathrm{odour}}(\mathrm{odour}_{\mathrm{seawater_moderate}} == 3) \\
&+ \beta_{\mathrm{disposal}}(\mathrm{disposal}_{\mathrm{seawater}_{\mathrm{none}}} == 1) + \beta_{\mathrm{disposal}}(\mathrm{disposal}_{\mathrm{seawater}_{\mathrm{treat}}} == 2) \\
&+(\mathrm{disposal}_{\mathrm{seawater_reuse}} == 3)
\end{aligned} \qquad \textbf{7-5}$$

$$\begin{aligned}
&\mathrm{U\ (recycled\ water)} = \delta_{recycledwater} \\
&+ \beta_{\mathrm{stain}}\left(\mathrm{stain}_{recycledwater_{\mathrm{none}}} == 1\right) + \beta_{\mathrm{stain}}\left(\mathrm{stain}_{recycledwater_{\mathrm{slight}}} == 2\right) \\
&+ \beta_{\mathrm{stain}}\left(\mathrm{stain}_{ecycledwater_\mathrm{moderate}} == 3\right) \\
&+ \beta_{\mathrm{colour}}(\mathrm{colour}_{recycledwater_{\mathrm{none}}} == 1) + \beta_{\mathrm{colour}}(\mathrm{colour}_{recycledwater_{\mathrm{slight}}} == 2) \\
&+ \beta_{\mathrm{colour}}(\mathrm{colour}_{recycledwater_\mathrm{moderate}} == 3) \\
&+ \beta_{\mathrm{cost}}\mathrm{cost}_{recycledwater} \\
&+ \beta_{\mathrm{odour}}(\mathrm{odour}_{recycledwater_{\mathrm{none}}} == 1) + \beta_{\mathrm{odour}}(\mathrm{odour}_{recycledwater_{\mathrm{slight}}} == 2) \\
&+ \beta_{\mathrm{odour}}(\mathrm{odour}_{recycledwater_\mathrm{moderate}} == 3) \\
&\boldsymbol{+ \beta_{\mathrm{disposal}}(\mathrm{disposal}_{recycledwater_{\mathrm{none}}} == 1) + \beta_{\mathrm{disposal}}(\mathrm{disposal}_{recycledwater_{\mathrm{treat}}} == 2)} \\
&\boldsymbol{+\beta_{\mathrm{disposal}}\ (\mathrm{disposal}_{recycledwater_\mathrm{reuse}} == 3)}
\end{aligned} \qquad \textbf{7-6}$$

$$\mathrm{U\ (drinking\ water)} = \delta_{drinkingwater}$$

$$+ \beta_{\text{cost}} \text{cost}_{rdrinkingecycledwater}$$
$$+ \boldsymbol{\beta}_{\textbf{disposal}} (\textbf{disposal}_{\boldsymbol{drinkingwater}_{\textbf{none}}} == \mathbf{1}) + \boldsymbol{\beta}_{\textbf{disposal}} (\textbf{disposal}_{\boldsymbol{drinkingwater}_{\textbf{treat}}} == \mathbf{2})$$
$$+ \beta_{\text{disposal}} (\text{disposal}_{rdrinkingwater\_\text{reuse}} == 3) \qquad \mathbf{7\text{-}7}$$

The cost parameter was modelled as a continuous variable, and the rest of the other parameters were modelled as categorical variables.

Model estimation

Model estimation aims to measure the parameter values that best explain stated choice data. There are several statistical methods for estimating choice models. The maximum log-likelihood estimation (MLE) is the most commonly used method (Louviere *et al.*, 2000). MLE aims to find the parameter values that maximise the log-likelihood function, which is typically achieved using optimisation algorithms. The optimisation involves iteratively adjusting the parameter values to increase the log-likelihood until a maximum is reached.

7.2.3.2 Mixed Logit

Discrete choice experiments that include several observations per respondent increase the potential for correlation of observations by any particular respondent. Consequently, this induces a bias in the parameter t-ratio estimate of the MNL model, which is restricted to the condition of independent irrelevant alternatives and the fixed iid error term. The mixed multinomial logit model (MMNL) allows for the relaxation of MNL conditions and can be interpreted in two ways. The first is the random parameter logit (RPL) model, which focuses on taste heterogeneity. Second, the error component logit (ECL) considers the correlation between the utility of the choice of alternatives.

Heterogeneity in cost attributes was anticipated, given household heads' variation in reported income. In addition, this study's DCE included repeated choices, increasing the propensity of complex substitution patterns known to the respondent. Therefore, this study investigated the impact of taste heterogeneity and the correlation between the utility of choice of the alternatives using the RPL version of the mixed logit (ML).

Random Parameter Logit

The random parameter logit (RPL) model allows for preference heterogeneity across respondents by assuming that the parameters of the utility function are continuous and randomly distributed across the population. Therefore, random parameters allow for correlation across alternatives (measured by the attributes) and across choice sets (Hensher, 2011). The standard deviation of the parameters represents the degree of unobserved heterogeneity. In contrast, the mean values indicate preference heterogeneity relative to another attribute of an alternative or choice context.

The iid assumption is relaxed in the model, and the error term distribution can take any form. In addition, it overcomes the MNL limitations of substitution patterns and correlations in unobserved factors over time (Train, 2002). A few interdependent factors need to be considered when selecting a random parameter, such as the selection of the appropriate random parameter and its distribution and accounting for the correlation of choice situations in cases where respondents are required to observe multiple choice sets. This study used the Likelihood Ratio test statistic to determine the significance of the selected random parameter.

Modelling choice probabilities with ML has become widely common given the advent of computational advances despite the model being known for several years. ML is an integral mixed function of logit choice probabilities with a mixing distribution. The integral function implies that the ML choice probability is not a closed-form expression.

$$\boldsymbol{P_{n,i} = \int L_{n,i}\,(\beta) f(\beta|\theta) d\beta} \qquad \textbf{7-8}$$

Where $L_{n,i}(\beta)$ is the logit probability at a given set of coefficients, β;

$$\boldsymbol{L_{n,i} = \frac{e^{V_{n,i}(\beta)}}{\sum_{j=1}^{J} e^{V_{n,j}(\beta)}}} \qquad \textbf{7-9}$$

and $f(\beta|\theta)$ is the density function of co-efficient β, which depends on a vector of parameters θ that define the density of the random coefficients. Therefore, the ML model is a weighted average of logit choice probabilities, which are evaluated at different values of β, and each choice probability is weighted by density f(β | θ). The deterministic utility portion is expressed in the term $V_{n,i}(\beta)$ and if utility is linear in β then $V_{n,i}(\beta) = \beta' x_{n,i}$. Then, the mixed logit probability takes the form:

$$\boldsymbol{L_{n,i} = \frac{e^{\beta' x_{n,i}}}{\sum_{j=1}^{J} e^{\beta' x_{n,j}}}} \qquad \textbf{7-10}$$

The ML model does not estimate the β coefficients that appear in the utility expression because they are integrated out of the choice probability (Train, 2002). ML estimates the mean and variance of the distribution and not the θ value. Therefore, the choice probability can be expressed in the form;

$$\boldsymbol{P_{n,i} = \int L_{n,i}\,(\beta) f(\beta|b, W) d\beta} \qquad \textbf{7-11}$$

where *b* and *W* are the selected coefficient distribution's mean and variation, respectively. If the mean value significantly differs from zero, then the sample population can be inferred to demonstrate taste variation. If this is not the case, then the MMNL model collapses into an MNL model.

There are several distribution patterns of the density function $f(\beta)$ that can be modelled, such as normal, log-normal, uniform and triangular (reference). While applying the distribution forms to continuous variables is standard practice, a discrete distribution can be applied, particularly when the attributes are non-continuous. This identical distribution across respondents results in a model segmented according to the latent covariate constants (Hensher, 2011).

The selection of the distribution form accounts for *a priori* theoretical restrictions and empirical properties of the model estimation. For example, an attribute such as cost theoretically has a negative sign and would follow a negative log-normal distribution. In other cases, modelling the data may support some distribution forms better than others. When there is no apparent restriction in the parameter sign, then, *a priori,* the normal distribution can be considered. However, the disadvantage of the normal distribution is that extreme observations can be underestimated when compared to the mean. Consequently, it influences the model estimates. Therefore, a balanced view of these disadvantages is required when selecting a normal distribution form.

Triangular and uniform distributions are both bounded, which may reduce the model sensitivity to extreme data points, as is the case with normal and log-normal distributions. The uniform distribution is specifically bounded between the intervals of 0 and 1. On the other hand, the triangular distribution truncation is dependent on the analyst.

In this study, cost was the only continuous attribute and was modelled *a priori* with a negative log-normal distribution. The parameter was modelled in this way to overcome the issue of non-convergence or significantly large estimates when a parameter with *an a priori* negative sign is included in the model as a positive log-normal. This statement holds because the negative log-normal parameter of an attribute is equivalent to a positive log-normal parameter for the negative attribute (Hensher, 2011). The colour, odour, and disposal attributes were fixed using the same covariate interactions from the MNL base model. Stain was introduced as a random parameter with a uniform distribution.

7.2.3.3 Marginal willingness to pay

The choice experiment enables the determination of trade-off measures between attributes, including price. These measures can be utilised to estimate the monetary value that individuals are willing to pay to modify their attribute levels. Importantly, this estimation ensures that individuals maintain the same level of satisfaction even after the change. Ultimately, these estimates provide insights into the compensating variation associated with attribute changes, which meet the study objectives.

The proposed DCE design includes non-continuous attributes with dummy variables; thus, the ratio of marginal utilities is not well-defined (Champ *et al.*, 2017; Louviere *et al.*, 2000). Rescaling the marginal utilities of specific non-cost attributes using the cost coefficient simplifies the estimation of willingness to pay (WTP). This approach directly estimates WTP, eliminating the need to calculate WTP as the ratio of partial derivatives (Louviere *et al.*, 2000). By incorporating the cost coefficient into the rescaling process, estimating individuals' monetary valuations for attribute changes becomes more straightforward. It avoids the additional step of calculating the ratios. This method provides a more direct and efficient means of estimating WTP in choice modelling scenarios and can be calculated in the model space, where β_k is the attribute (Scarpa and Rose, 2008).

$$\boldsymbol{WTP_k = \frac{\beta_k}{-\beta_{cost}}} \qquad \textbf{7-12}$$

7.3 Results

7.3.1 Sample statistics

The survey was administered over two weekends in March, specifically on the 16th-17th and 23rd-24th. A total of 316 surveys were collected, leaving 239 after screening. The completed surveys took an average of 7 minutes to complete, with a standard deviation of ± 5 minutes. The characteristics of the sample sizes are presented in Table 7-4.

Table 7-4: Sample Socioeconomic Characteristics

Characteristics	N =239 No.	N =239 %	Binary coding	Characteristics	N =239 No.	N =239 %	Binary coding
Age				**Income (R)**			
18-34	82	34	0	1-3 200	25	10	0
35-64	143	60	1	3 201-12 800	108	45	0
Above 64	14	6	0	12 801-25 600	71	30	1
Gender				25 601-102 400	35	15	1
Male	136	57	0	Above 102 400	0	0	1
Female	103	43	1	**Tenure**			
Education				Owner	168	70	1
No schooling	20	8	0	Renter	71	30	0
Completed matric	127	53	0	**Household size**			
Diploma	53	22	1	1	14	6	0
Degree	39	16	1	2	55	23	0
Ethnicity				3	74	31	1
Black	96	40	0	Above 3	96	40	1
White	83	35	1	**Stay**			
Coloured	57	24	0	Before 2018	157	66	1
Asian	2	1	0	After 2018	82	34	0
Other	1	0	0	**Suburb**			
Religion				Hangberg	28	12	1
Muslim	28	12	1	Imizamo Yethu	93	39	0
Christian	164	69	1	Hout Bay	118	49	1
Judaism	4	2	1				
Hinduism	1	0	1				
Buddhism	1	0	1				
Other	14	6	1				
None	27	11	0				

The inclusion of modelling covariates occurs through the binary coding of a population segment. Therefore, categorical covariates were coded as 1 or 0 to facilitate modelling. Hout Bay and Hangberg were coded 1 to align with the municipal spatial classification of the Hout Bay region. Middle- and high-income categories were coded 1, while individuals with diplomas or degrees were coded 1 to segment the education covariate.

All religious affiliations were coded 1, and households with three or more occupants were coded 1. Amaris *et al.* (2020) selected females for their study. Similarly, we chose females, as in other studies such as Li *et al.* (2020), who found them more willing to accept alternative water sources.

Individuals were required to state their water supply awareness by answering six statements, as discussed in Section 7.2.1.1. The results are shown in Figure 3. Respondents were generally aware of water supply sources, potable and recycled water use, and wastewater generation, as indicated by Questions 1, 2, 3, and 5. In contrast, 75.7% and 61.5% of the respondents were unaware of the use of seawater for flushing toilets (Question 4) and local wastewater disposal practices (Question 6), respectively.

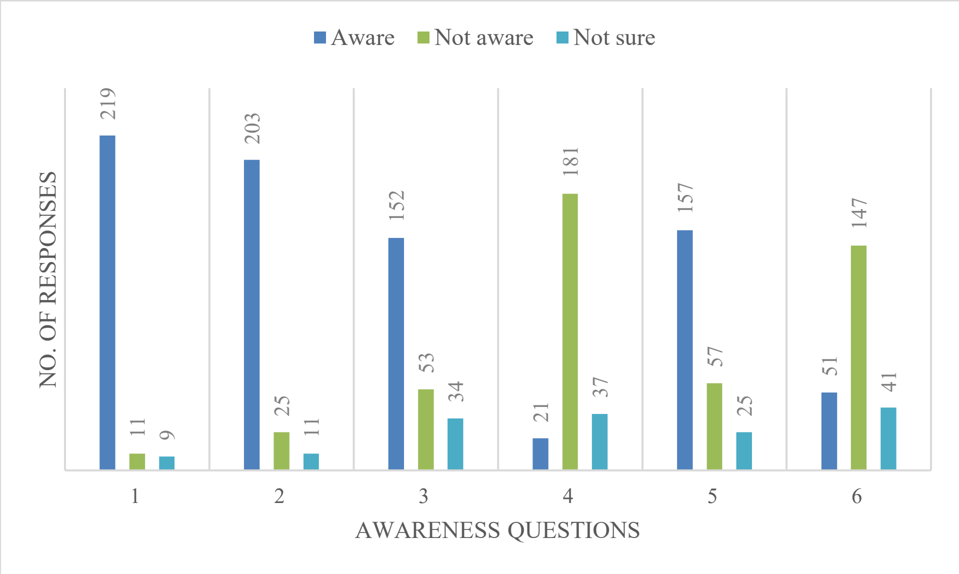


Figure 7:3: Summary of Awareness Responses

Table 7-5 summarises the frequencies of the choices made between the three alternatives. A total of 1 434 observations were made, based on the presentation of six choice sets to 239 respondents. Seawater was selected the most (57.6%), followed by recycled water (32.2%). Drinking water was selected the least number of times (10.2%).

Table 7-5: Selection of Alternatives

	Seawater	Recycled water	Drinking water
Times available	1 434	1 434	1 434
Times chosen	827	461	146
Percentage chosen (%)	57.6	32.2	10.2

7.3.2 Model results

A covariate mixed logit (ML) was used to investigate the preferences. Furthermore, WTP was estimated in the model space of the covariate ML. The stated preference results are presented, followed by the WTP estimation.

7.3.2.1 Stated Preferences

Base and covariate ML models were used to investigate the stated preference and the influence of socioeconomic variables. Table 7-6 summarises the results.

Table 7-6: Mixed Logit Multinomial Model Preference Parameters

	Base ML		Covariate ML	
LL(final)	-835.25		-833.15	
Adjusted ρ^2 vs observed shares	0.3511		0.3512	
Estimated parameters	18		20	
	Estimate	**t-ratio**	**Estimate**	**t-ratio**
μ_seawater	4.9606	4.7520***	5.1892	4.4416***
σ_seawater	-3.8890	-7.7050	-3.8206	-7.5819
μ_recycled water	3.2244	3.5053***	3.6226	3.4712***
σ_recycled water	1.0302	3.4619	0.9396	3.0554
μ_drinking water		*base*		*base*
β_stain_none	0.0835	0.2470	0.0672	0.2018
β_stain_slight	-0.6142	-1.8393*	-0.6195	-1.8757*
β_stain_moderate		*base*		*base*
β_colour_none		*base*		*base*
β_colour_slight	-0.4529	-1.3178	-0.4379	-1.2872
β_colour_moderate	-1.3599	-3.5114***	-1.3202	-3.4623***
μ_log_cost	-4.0547	-20.1547***	-4.0852	-19.1648***
σ_log_cost	1.3659	6.0046	1.3318	5.5654
μ_odour_none	2.5818	2.6516***	2.2850	2.1865**
σ_odour_none	-3.6060	-2.6249	-3.3485	-2.0409
μ_odour_slight	-0.9012	-1.1386	-1.1523	-1.3938
σ_odour_slight	4.5788	3.2941	4.7283	3.4506
μ_odour_moderate		*base*		*base*
β_disposal_none		*base*		*base*
β_disposal_treat	0.4734	2.2144	0.4645	2.2349
β_disposal_treatreuse	0.4307	1.4830	0.4230	1.4962
Covariates				
γ_seawater_income			-0.9951	-1.1574
γ_recycled water_income			-1.3274	-2.1036**

Significance codes: 98% = ***,95%= **, 90% =*

The signs of the estimates indicate utility (+ sign) or disutility (− sign) for any of the alternatives and attributes.

The magnitude of the beta sign highlights the difference from the base point of the given attribute or alternative. In this case, drinking water was selected as the base alternative, and for the nonlinear attributes, moderate odour, moderate stain, no colour, and discharge without treatment were selected as the reference points for their respective attributes.

The base ML model results have the expected signs and monotonic direction. The random parameters introduced to the model include odour, cost and the alternative specific constants (ASC), while the rest were fixed. The model fit improved significantly with the introduction of the random parameters, which indicates heterogeneity in preferences for the alternatives, colour, and costs amongst the respondents. As anticipated, alternative water sources were preferred to the drinking water option, which is congruent with the frequency of choices made (see Table 7-5). Furthermore, the ASC parameters are significant, implying that the model does not capture unobserved factors related to alternatives. In other words, factors about the options remain unknown because they have not been explicitly presented in the attributes that describe the different options.

A slight stain decreases the utility of the alternative water sources. In addition, if the alternative is offered with a moderate stain, all things being equal, the utility of the drinking water alternative increases. Regarding the odour parameter, the utility for the alternative increases and is significant if the water has no odour. As expected, the cost parameter is highly significant and reduces the alternatives' utility. Finally, respondents tend to prefer the disposal practice of treat and discharge compared to treat and reuse. On the whole, there is a preference to move away from the status quo practice of discharging untreated wastewater into the sea.

The covariate model was anticipated to improve the ML model fit in explaining the respondents' preferences. Income was the most significant covariate, which provided a marginal log-likelihood improvement of approximately two units. Respondents earning more than R12 800 prefer drinking water to flush toilets over seawater and recycled water. The disutility of recycled water is significant and more than that of the seawater alternative.

7.3.2.2 Analysis of Covariates' Influence on Preferences for Alternative Water Sources

We explored the effect of covariates on preferences for seawater and recycled water by estimating a regression model using the posterior means of preference parameters for these alternatives as dependent variables. Posterior means are the average values of individual-level preference parameters obtained from the posterior distributions in an ML model. These means represent the most likely preferences for each individual based on their choice data, combined with prior information or assumptions. Essentially, they summarise how much a person values specific attributes or alternatives, accounting for individual variability. For example, in a mixed logit model, the posterior mean for the seawater alternative reflects each individual's estimated preference for seawater as an alternative water source derived from their observed choices and the overall population trend.

Analyzing posterior means through regression offers a transparent and interpretable way to understand the effects of covariates on individual preferences, highlighting heterogeneity without complicating the main choice model. This approach allows for a straightforward exploration of how factors like demographics or attitudes influence preferences, offering actionable insights for policy and marketing. It also avoids overfitting and the computational complexity that can arise from introducing interaction terms (gammas) directly into the mixed logit model. By decoupling the analysis of covariates from the primary model estimation, this method provides flexibility, enhances model interpretability, and facilitates validation, making it a practical alternative to embedding gammas within the mixed logit framework.

The results are summarised in Table 7-7 and Table 7-8, respectively. We now discuss the results of the seawater regression model. As discovered in the covariate ML model, income is highly significant and decreases the utility of the seawater alternative. Respondents who are female and have a tertiary education are more likely to accept the seawater alternative.

Table 7-7: Regression model of the seawater vs income gender and education

	Estimate	Std. Error	t value
(Intercept)	2.1402	0.2277	9.401
Income	-1.8247	0.3336	-5.469***
Gender	0.5897	0.2906	2.029*
Education	0.8905	0.3413	2.609***

Significance codes: >98% = ***,95%= **, 90% =*

Next, we discuss the results of the recycled water regression model. As expected, respondents earning more than R12 800, all things being equal, prefer the drinking water alternative for flush toilets. Respondents who are female, have a tertiary education and have more than three occupants in their household are more likely to accept the recycled alternative.

Table 7-8: Regression model of the recycled water vs income, gender, education and household size

	Estimate	Std. Error	t value
(Intercept)	1.6967	0.2994	5.666
Income	-1.9082	0.3328	-5.733***
Gender	0.5555	0.2885	1.9268*
Education	0.8544	0.3388	2.522**
Hsize	0.7158	0.3176	2.254*

Significance codes: >98% = ***,95%= **, 90% =*

7.3.2.3 Willingness to Pay

All willingness to pay estimates are presented in South African Rand values (ZAR). We found that, on average, the respondents were willing to pay R46,10 for an improvement in moderate colour for the alternative water sources. In contrast, respondents are more likely to accept an alternative with a slight odour at a discount of R187,37, and this finding is highly significant. Interestingly, all things being equal, respondents were more likely to accept compensation for changes in disposal practices. We found the treat and discharge parameter significant at a discount of R22.45 to the respondents' current water bill.

Table 7-9: Willingness to Pay Estimation

	WTP ML	
LL(final)	-873.29	
Adjusted ρ^2 vs observed shares	0.3214	
Estimated parameters	19	
Parameters	**Estimate**	**t-ratio**
μ_seawater	3.1552	2.1836**
σ_seawater	2.8876	6.5685
μ_recycled water	1.6370	1.1234
σ_recycled water	1.0261	0.9936
μ_drinking water	**base**	
β_stain_none	12.25	0.7992
β_stain_slight	26.91	1.6393*
β_stain_moderate	**base**	
β_colour_none	**base**	
Parameters	**Estimate**	**t-ratio**
β_colour_slight	-1.12	-0.0755
β_colour_moderate	46.10	2.4953**
β_cost	-0.0159	-9.0454
μ_odour_none	-100.62	-1.3630
σ_odour_none	105.96	1.1779
μ_odour_slight	-187.37	-3.3761***
σ_odour_slight	238.82	4.0417
σ_odour_moderate	**base**	
β_disposal_none	**base**	
β_disposal_treat	-22.45	-2.2011***
β_disposal_treatreuse	-14.01	-1.1141
Covariates		
γ_seawater_income	-1.5090	-2.4945***
γ_recycled water_income	-1.5042	-3.0802***

Significance codes: >98% = ***,95%= **, 90% =*

In contrast to the odour parameter, where respondents are more likely to accept compensation, the respondents are willing to pay for improvement in the colour of the alternative water source. We found that willingness to pay to improve the slight stain parameter to be approximately R26,91.

7.4 Discussion

The utility of alternative water sources in recycled water and seawater increases if the water is not distinctly yellow, has no odour, does not increase the monthly bill, and is treated before it is discharged. In addition, household heads with middle-to-high incomes preferred using drinking water to flush toilets. Furthermore, females with a tertiary education are more likely to accept seawater as a water source for flushing toilets. On the other hand, females with tertiary education and living in households with more than three occupants tend to prefer the recycled water alternative for flushing toilets.

The monthly water bill estimates ranged from R350 to R900, depending on household water consumption habits. Therefore, our results indicate that respondents are willing to pay 3-9% extra to improve a moderate stain of the alternative water source. Similarly, a 5-10% increase in the utility bill for improvements of the moderate colour attribute. The result indicates significant sensitivity to the odour parameter. It can be expected that a 20-60% discount on the utility for accepting water that has a slight odour. Amaris *et al.* (2020) found in their study, which looked for the preferences for urban greywater reuse and their qualitative attributes, that odour and colour were significant. This study's findings are similar to those of Amaris *et al.* (2020) regarding the odour and colour parameters.

Furthermore, the results of this study show that a discount of up to 7% is expected for changes in disposal practices. Since the disposal question was not framed in a manner that protects the environment, respondents may have perceived no direct benefit from changes to the disposal practices. This finding highlights the need to improve awareness around the environmental benefits of changing disposal practices to increase household willingness to pay for the investment in new infrastructure.

Moreover, the cost parameter was highly significant and decreased the utility of the alternatives. Therefore, this indicates that all things being equal, the preference for adopting alternative water sources is more likely to increase if offered at a discount to the current water utility bill.

Pro-environmental behaviour refers to actions taken by individuals or groups that contribute to the conservation and sustainable use of environmental resources, including water. It is generally accepted that high income, higher education qualifications and gender (females) are pre-dispositioned to report pro-environmental behaviours and intentions (Gul *et al.*, 2024; Kumarasamy and Dube, 2016; Owen and Chitonge, 2022; Prins *et al.*, 2023; Sacolo and Abidoye, 2017). This study contributes to this knowledge in the African context. Contrary to general research findings, this study found that middle to high-income does not increase pro-environmental behaviours. Further, our findings indicate that household size is a contributing factor to the adoption of alternative water sources.

8 Conclusion

In conclusion, this study investigated the environmental impact of using a hybrid system with a seawater supply for flushing toilets and the willingness to pay for alternative water sources for toilet flushing in catchments with marine outfall sewers as a wastewater disposal practice.

Our study found that seawater supply for toilet flushing conserves a significant 26% of potable water that would otherwise be withdrawn from freshwater sources. However, this comes at the expense of a 20% increase in ecotoxicity impacts associated with background electricity production and transmission processes and the additional distribution pipelines. In addition, based on the normalised results, the study found that freshwater depletion was the second most impactful category after human health impacts caused by carcinogens. In the context of diminishing water supply and water security assurance, implementing dual systems may be favourable despite the significant increase in impact.

Moreover, the LCA scope boundary system was limited due to a lack of data. Consequently, the research relied on the Ecoinvent database and used global averages instead of primary datasets. This approach may have overestimated impacts and increased uncertainty by not accounting for process variations across different locations. Additionally, the analysis focused solely on the water supply aspect of the urban water system, excluding the wastewater management stages that could have ecotoxicity and eutrophication effects on marine life. However, these limitations provide potential opportunities for further research to build on our study and address the remaining gaps.

Regarding investigating stated preferences, the project was limited by sample size and location. This study's findings indicate a preference for using alternative water sources over drinking water to flush toilets. Furthermore, there is a willingness to pay for improvement in colour and stain. On the contrary, there is a willingness to accept a discount if the water has a slight odour and the wastewater is treated before discharging to the sea. All things considered, the preference for alternative water sources decreases if it increases the monthly utility bill.

In addition, females with a higher education qualification and more than three occupants in the household are more likely to adopt alternative water sources. On the other hand, respondents with medium to high income prefer flushing with drinking water. These results can provide inputs for estimating the demand for alternative water sources and inform integrated water supply policies. The insights provided can assist policymakers, practitioners, and decision-makers make informed choices regarding integrated water supply planning.

Overall, this study contributes to the current knowledge base of stated preferences in environmental economics, specifically regarding water supply and the environmental impact of hybrid water supply systems.

9 Recommendations

The following recommendations are proposed to build on the findings of this research project;

1) LCA analysis
    a. We recommend including a wastewater management stage to determine the most feasible end-of-life process by considering wastewater treatment technologies capable of treating saline sewage. Liu *et al.* (2016) found that the addition of the SANI® (sulphate autotrophic-denitrification nitrification) treatment process improves the feasibility of using seawater for toilet flushing as it lowers the population density from 3 000 to 1 100 people/km^2 and doubles the maximum distance from the sea to 60 km.
    b. Additionally, future studies should evaluate the impact of using renewable energy on water systems, as electricity significantly contributes to the water impact.
2) Choice modelling analysis
    a. The significant ACS emphasises the significance of unobserved factors associated with the alternatives and suggests room for improvement in survey design. The scope can be expanded to include attributes related to the service's water supply level and the environmental impact of discharging untreated wastewater to the sea.
    b. In addition to Hout Bay, MOS disposal practices are used in Durban, Gqeberha, Sea Point, Camps Bay, and Robben Island. Therefore, it is recommended that the sample size be increased to generate results that can provide national input.
    c. More complex model distribution can be explored to explore the sample's heterogeneity extent. This can be achieved by using the Hierarchical Bayesian model approach. The Hierarchical Bayesian (HB) approach in choice modelling helps understand preference differences by combining data from all participants while still capturing individual differences. It estimates each person's choices as part of a larger group, showing both shared patterns and unique preferences.

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Annexure: Knowledge Dissemination

Table 1: Knowledge Dissemination

Initiative	**Comment**
1. Conferences	
IMESA 2021 15 September Conference	Presented paper on Critical review: Feasibility of using seawater to flush toilets)
IWA 2025 Conference	Two papers were approved for presentation at the conference to be held in March 2025. One paper was approved for a poster. Submitting a journal paper for the conference
2. Stakeholder Engagement	
CCT	Invited CCT and various other government and private sector stakeholders to a workshop hosted by Future Water Institute
3. Social Responsiveness	
Radio Interview	https://omny.fm/shows/mid-morning/how-do-we-diversify-water-sources
Television Interview	https://youtu.be/hCK5jIp7sEY?si=-5iuPmmL8xAMWRkw
UCT News article	https://www.news.uct.ac.za/article/-2024-08-13-womens-month-south-africans-mixed-reaction-to-adopting-alternative-water-sources