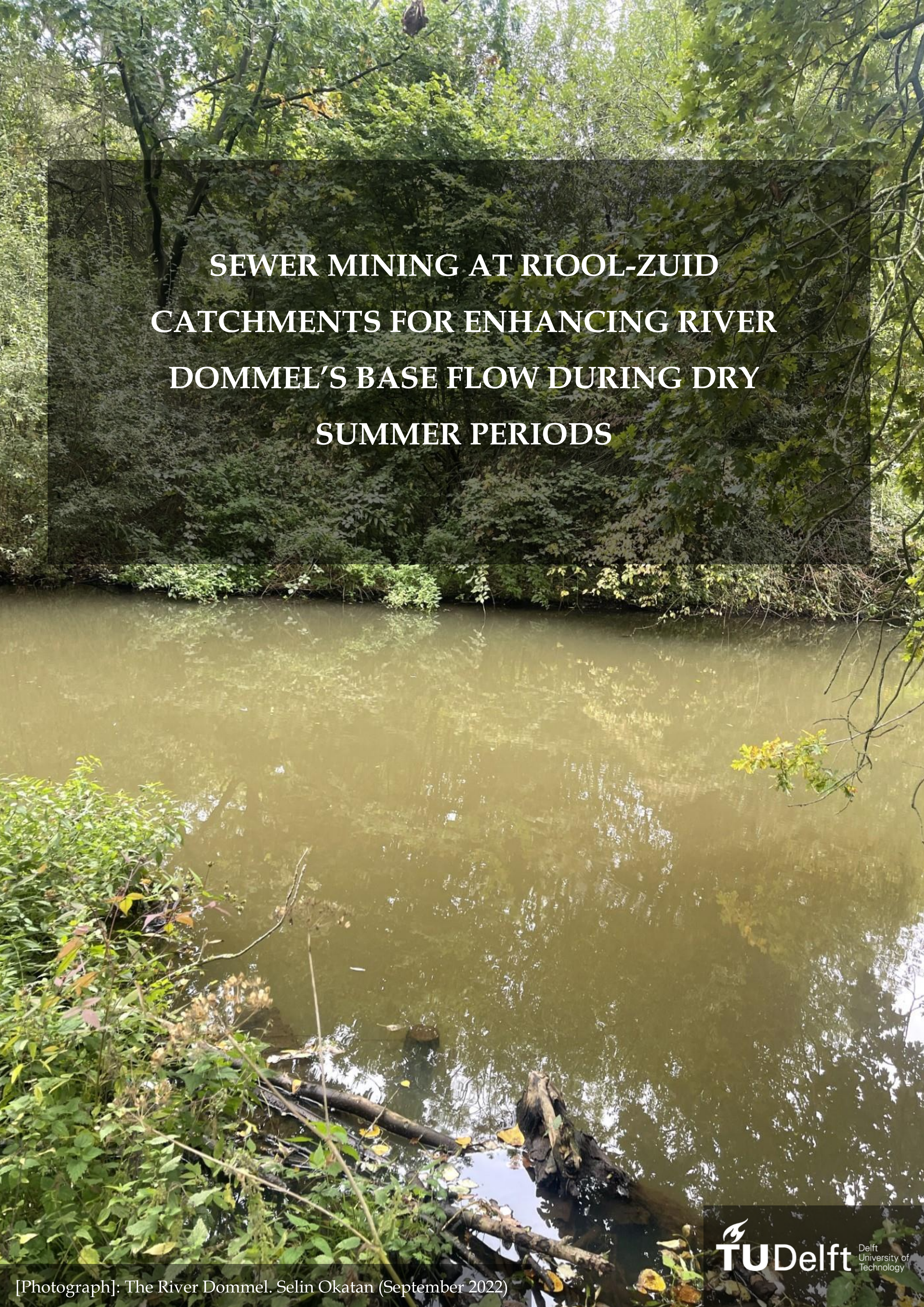
SEWER MINING AT RIOOL-ZUID CATCHMENTS FOR ENHANCING RIVER DOMMEL'S BASE FLOW DURING DRY SUMMER PERIODS



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[Photograph]: The River Dommel. Selin Okatan (September 2022)

Sewer mining at Riool-Zuid catchments for enhancing River Dommel's base flow during dry summer period

Master Thesis

by

S. Okatan

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Preface

This thesis is the completion of my effort to obtain a master's degree in Civil Engineering with a specialisation in Urban Water Management. The journey has been both challenging and rewarding and I am deeply grateful for the guidance and support I have received along the way.

First and foremost, I want to express my deepest appreciation to my supervisor, Dr. Ir. J.G. Langeveld. Your invaluable support, insightful guidance, and understanding of my personal struggles have been instrumental in bringing this thesis to fruition. Your guidance provided me with invaluable motivation, and your patience during setbacks was a source of encouragement. I consider myself fortunate to have had you as my supervisor, and I express my sincere gratitude for your exceptional mentorship. I am especially appreciative of the time you dedicated to our meetings and the thorough reviews of my drafts. Your constructive feedback and thoughtful suggestions significantly enhanced the quality of this work. Your ability to provide direction while allowing me the freedom to explore my ideas has helped me grow both academically and personally. Furthermore, your excitement for the topic and your dedication to research have been incredibly inspirational. You have not only been a mentor but also a role model. Your commitment to my success has been evident in every interaction, and for that, I am profoundly grateful.

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Finally, I am grateful to my family for their continuous support and belief in my abilities. Their love and encouragement have been my foundation throughout this journey.

To all mentioned and unmentioned individuals who have contributed to this thesis, I express deep appreciation for your guidance, support, and inspiration.

Executive Summary

The River Dommel is a modest-sized and ecologically sensitive stream located in the southern region of the Netherlands, covering a catchment area of 153,000 hectares. Its primary water source is rainwater, supplemented by a minor contribution from groundwater, particularly notable in the upstream portion of the catchment. Nevertheless, rainwater remains the primary source feeding the river, with its influence most pronounced in the midstream and downstream portions of the Dommel. In addition to that, the River Dommel receives the Eindhoven Wastewater Treatment Plant's effluent while flowing through the city of Eindhoven. Eindhoven WWTP is one of the largest wastewater treatment facilities in the Netherlands with a 750,000 Population Equivalent (p.e.) and its effluent contributes approximately 50% of the total flow in the River Dommel during dry summer periods (Weijers et al., 2012; Kamstra, 2018).

During dry summer seasons, the Rivel Dommel faces the problem of insufficient base flow and low water levels due to decreased precipitations and increased evaporation rates. Insufficient base flow and therefore low water levels causes water quality issues in the River Dommel. Moreover, inadequate water flow, compounded by warmer temperatures in the summer, exerts stress on aquatic ecosystems within the River Dommel.

The primary objective of this report is to present a comprehensive study aimed at identifying a technically and economically viable solution to feed the River Dommel base flow during dry weather flow (DWF) conditions. Possible solutions addressing these challenges are elaborated and discussed in this report. Consequently, the application of sewer mining technology is chosen to be further investigated. The idea of implementing the sewer mining technology in this study is based on the extraction of the wastewater from Riool-Zuid sewer system, followed by cleaning processes and feeding the River Dommel with reclaimed water. While there are several treatment methods that can be applied in sewer mining, a thorough review of the literature, as presented in the relevant sections of this report, has led to the conclusion that a combination of Forward Osmosis (FO) and Reverse Osmosis (RO) represents an effective and efficient approach to achieving the objectives of this study. However, it should be noted that, implementation of sewer mining technology directly impacts current sewer system conditions and potentially give rise to significant challenges such as corrosion, odour and health impacts. Therefore, the impact of the sewer mining concept on the Riool-Zuid sewer system is also investigated and discussed in this study, followed by proper risk-mitigation measures.

The analysis of sewer mining is conducted through Mega-WATS simulations, based on a number of separate scenarios composed of three different implementation locations and various extraction ratios. The pressure main plays a crucial role on the extent of sewer mining technology impact on the Riool-Zuid sewer system as it represents anaerobic conditions while the rest of the sewer system is composed of gravity pipes representing aerobic conditions. Therefore, de Meren and Aalst are chosen to reveal the impact of anaerobic conditions on the severity of water extraction from the sewer system as these locations are both located upstream of the pressure main. In addition to that, the juncture of Heeze, Sterksel and Leende domestic wastewater inflow is chosen as the third implementation location, which is located downstream of the pressure main.

The implementation of the sewer mining technology upstream of the pressure main resulted in high dissolved sulphide concentrations and therefore led to considerable corrosion and odour problems in Riool-Zuid. On the other hand, when it was implemented downstream of

the pressure main, the corrosion and odour problems were observed to be minimal. However, it is worth emphasizing that the distance between the sewer mining technology and the River Dommel plays also a crucial role in selecting the implementation location since the primary objective of this study is to feed the base flow of the River Dommel in an efficient, reliable, and cost-effective manner.

De Meren represents the nearest location to the River Dommel. Moreover, the existing infrastructure at the de Meren control station, including an adequately sized building, presents a distinct advantage. This eliminates the need for additional construction costs, and concurrently minimizes the expenses associated with the installation of piping. Despite the fact that implementing the sewer mining technology in de Meren resulted in higher dissolved sulphide concentrations and more noticeable corrosion problems in Riool-Zuid, the required nitrate dosages for corrosion control were found to be less than those required for implementing the sewer mining technology in Aalst. Regarding the water quantities, sewer mining implementation in de Meren yielded flow rates in the River Dommel ranging from 0.5446 to 0.1004 m^3/s slightly below the observed rates in Aalst, which ranged from 0.5454 to 0.1021 m^3/s. Furthermore, in terms of cost analysis, the total treatment cost for the FO-RO system was determined to be approximately 1.67 €/m^3.

In conclusion, this study reveals that FO-RO hybrid system can be applied as a sewer mining technology at the Meren control station to boost the base flow of the River Dommel during the summertime to minimize the water quality problems in River Dommel.

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Chapter 1. Introduction

The River Dommel, which flows through Eindhoven, is a sensitive water body that faces water quality challenges and disturbances to its aquatic ecosystem, especially during summer time due to insufficient base flow.

This study aims to offer an efficient, reliable, and cost-effective method for enhancing the base flow of the Dommel River, with a specific focus on mitigating water quality problems during dry summer periods. After considering various alternative solutions, sewer mining implementation is selected as the proposed solution and comprehensively examined to evaluate its advantages and potential drawbacks.

The introduction chapter is structured into subsections to explain the purpose of the study, the research objectives, and to provide an overview of this report.

1.1. Background and Problem Statement ''Insufficient Base Flow in River Dommel''

1.1.1. Context and Background Information

Water scarcity and increasing water demand are significant challenges driven by rapid population growth. Consequently, the excessive production of wastewater continues to degrade the quality of surface water sources. Access to clean water, once taken for granted, now requires advanced engineering and water management practices in urban environments. These practices encompass water treatment facilities, supply and distribution systems, and measures to prevent waterborne diseases, all of which are essential for meeting the drinking, agricultural, and industrial water needs of society. The ongoing migration of populations to urban areas in search of economic opportunities and improved living standards underscores the perpetual and critical nature of urban water management (Loucks & van Beek, 2017).

To meet the increasing demands for drinking water and sanitation, it is imperative to implement effective urban water management practices. This includes the planning, design, and operation of essential infrastructures. Additionally, it's crucial to consider aspects such as infiltration, stormwater management, and runoff control to maintain the balance of urban ecosystems. Furthermore, challenges posed by climate change should be also addressed. Moreover, developing efficient urban water systems is essential to mitigate issues like those related to dry weather flows (DWF) in urban water systems.

This study focuses on urban wastewater systems, which are comprised of two key subsystems: wastewater treatment systems and sewer systems. Wastewater systems involve various processes, such as biological, physico-chemical, or a combination of both, to treat urban wastewater. Sewer systems, on the other hand, serve the crucial role of collecting wastewater from urban areas and transporting it to wastewater treatment plants (Langeveld, 2004; Schilperoort, 2011). One primary function of sewer systems is to safeguard residents from exposure to water contaminated with faecal matter, which can lead to waterborne diseases as wastewater typically contains high concentrations of pathogens. A secondary purpose of sewer systems is to manage and transport excess stormwater from urban areas to prevent potential flooding incidents in advance (Schilperoort, 2011).

Sewer systems are typically categorized into two main types: 'Separate Sewer Systems' and 'Combined Sewer Systems.' Separate sewer systems feature two distinct water lines designed to transport stormwater and wastewater separately. In contrast, combined sewer systems collect both stormwater and wastewater within the same pipe, particularly during rainfall events.

Within combined sewer systems, it's important to consider sediment deposition and resuspension. During dry weather flow (DWF) conditions, sewer pipe velocities are notably low, resulting in the accumulation of sediment layers that reduce hydraulic capacities. These sediment deposits within sewer pipes can generate gas and unpleasant odors. Even though this study primarily focuses on DWF conditions, it should be noted that these sediments are recognized as a significant source of pollution Wet Weather Flows (WWFs), especially in the case of Combined Sewer Overflows (CSOs). CSOs occur during heavy rainfall events when excessive wastewater from the sewer system is discharged into the environment (Ashley et al., 2004) and after prolonged periods of dry weather, sediment deposits within sewer pipes may be flushed out by stormwater, leading to elevated pollution concentrations in the wastewater. Consequently, long dry weather periods followed by intense rainfall events can result in sewer system overflows, potentially causing adverse impacts on the receiving water bodies (Suarez & Puertas, 2015).

Furthermore, climate change studies have indicated an increased frequency of longer dry weather periods followed by extreme rainfall events (Miller & Hutchins, 2017). This is expected to lead to an increase in deposition of pollutants within sewer systems, subsequently impacting the water quality of receiving bodies (Regueiro-Picallo et al., 2020). Climate change presents a significant challenge that requires careful consideration within the context of urban water management.

In response to climate change, many regions are revising their design criteria to implement climate-resilient urban drainage systems (Langeveld et al., 2013; Mailhot & Duchesne, 2010) and decentralized strategies (Smith, 2009). To address these challenges, innovative solutions and adaptation strategies are being explored. These strategies primarily focus on the increased rainfall intensities, which is anticipated to increase the occurrence of CSOs and flood risks (Smith 2009; Regueiro-Picallo et al., 2009).

Conversely, climate change is also linked to a reduction in the number of rainy days, resulting in prolonged dry periods (Langeveld et al., 2013). As the majority of the wastewater treatment plants discharge their effluent into the natural water bodies, a prolonged dry periods and less rainy days will also significantly impact the water quantity and quality of the receiving water bodies.

Dry weather conditions have significant impacts on water bodies such as rivers, affecting their ecological health and functionality. One of the foremost consequences is the decrease in water levels, triggered by reduced precipitation and increased evaporation rates that can arise challenges for navigation and diminishing the available habitat for aquatic organisms. Furthermore, diminished flow rates resulting from reduced rainfall restrict the downstream flow of water, consequently affecting aquatic ecosystems. As a result, the concentration of pollutants in the water increases due to reduced dilution, threatening water quality and endangering aquatic life. Additionally, elevated water temperatures result from reduced flow and shallower depths, further causing stress on aquatic organisms and promoting algae growth. These adverse effects worsen water scarcity issues, particularly in regions reliant on rivers for various purposes such as drinking water, agriculture, and industry. Consequently, mitigating the impacts of dry weather on rivers requires comprehensive management strategies that prioritize conservation and sustainable water management practices.

1.1.2. Description of the Problem: "Insufficient Base Flow in River Dommel"

In the Netherlands, all urban wastewater undergoes comprehensive treatment. Many wastewater treatment plants discharge their effluent into ecologically sensitive surface waters. Although these plants comply with stringent effluent standards, without an additional third treatment step, the discharged effluent can still significantly impact the water quality of small and sensitive water bodies, particularly during DWFs. In essence, during prolonged dry periods, these small and sensitive water bodies receive a substantial volume of WWTP discharges, contributing up to 90% or more of the surface water quantity. Due to the enormous amount of share-out of discharged water, the surface water quality will be almost identical with the effluent characteristics which should be avoided in order to preserve the aquatic ecosystem (Dutch Water Sector, 2013).

The Dommel is a relatively small and sensitive lowland stream that is located in the southern part of the Netherlands with a 153.000 ha surface area of catchment in the Netherlands. It is mainly fed by rainwater with a small groundwater contribution that is mainly significant in the upstream of the catchment while the rainwater contribution is the greatest in the middle and downstream part of the Dommel. In addition to these two water contributors, while flowing through the city of Eindhoven, the 750,000 p.e. WWTP of Eindhoven discharges its effluent into the Dommel.

The Eindhoven WWTP treats the wastewater of a densely polluted area which results in excessive amount of water discharges into the water bodies that are located in the surroundings such as lowland rivers and creeks. The River Dommel is impacted not only by WWF but also by DWW conditions. On one hand, during WWFs, beside receiving the Eindhoven WWTP effluent, it also receives more than 200 CSOs from 10 municipalities (Weijers et al., 2012; Kamstra, 2018). On the other hand, during DWF conditions, the Eindhoven WWTP constitutes up to 50% of the River Dommel's base flow that cannot meet the European Union Water Framework Directive (WFD) standards yet (Weijers et al., 2012).

The primary objective of this study is, to enhance the base flow of the River Dommel during dry summer periods to mitigate the adverse ecological effects caused by insufficient base flow, ultimately contributing to the preservation and enhancement of the River Dommel's aquatic ecosystem and contributing to compliance with European Union Water Framework Directive standards.

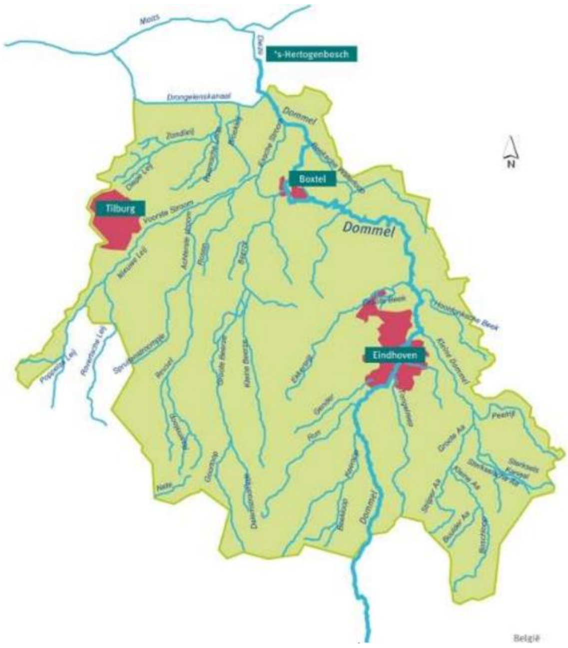


Figure 1. Stream system of the River Dommel (retrieved from Verdonk, 2016)

1.1.2.1. Overview of Wastewater System in Eindhoven

The wastewater system in Eindhoven comprises municipal sewer systems, pumping stations, transport mains, the Eindhoven WWTP, and a separate sludge processing facility. Each municipality is responsible for the collection, operation, and maintenance of its respective sewer system until it reaches the point where the responsibility shifts to the waterboard De Dommel. The waterboard De Dommel is responsible for managing and operating the wastewater transport system, encompassing the transport mains, pumping stations, and the conveyance of wastewater from the sewer outlets of each municipality to the Eindhoven WWTP (Schilperoort, 2011).

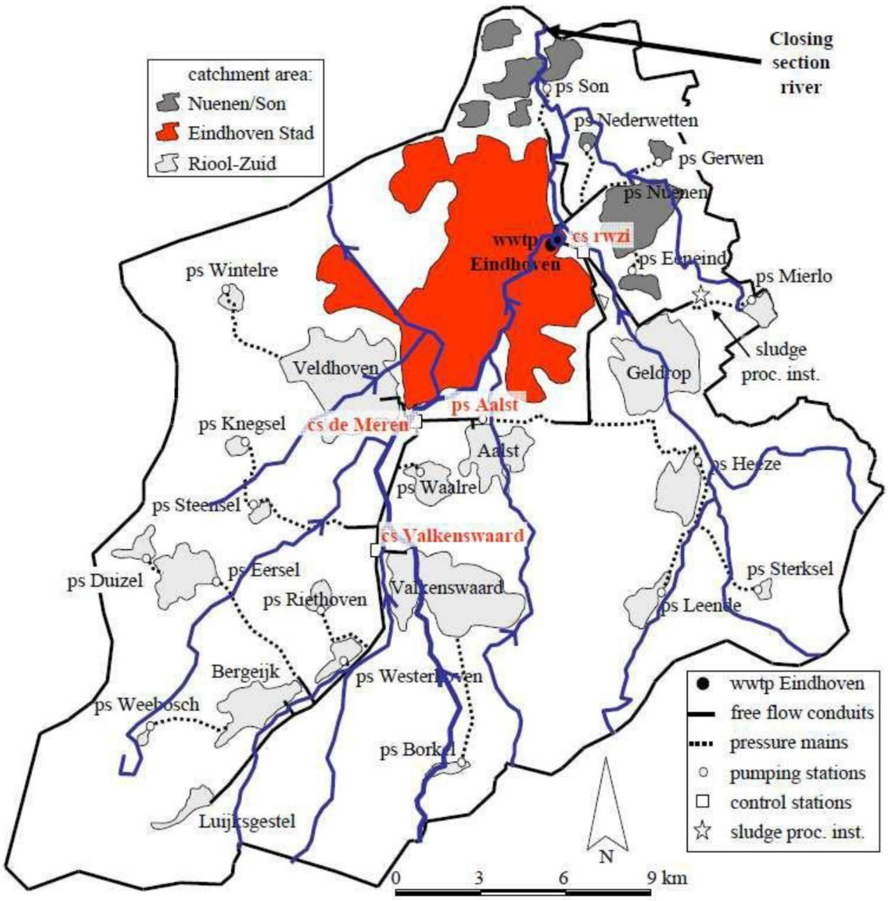


Figure 2. Wastewater system Eindhoven and catchment areas (retrieved from Weijers et al., 2012).

Eindhoven WWTP is the third-largest wastewater treatment plant in the Netherlands, with a treatment capacity of 750,000 p.e. (Amerlinck et al., 2016). It receives wastewater from three catchment areas: Riool-Zuid, Eindhoven city, and Nuenen/Son, transported from a pumping station with a capacity of 35,000 m^3/h located within the treatment area, equipped with nine pumps (Schilperoort, 2011). This treatment plant utilizes an activated sludge system based on a modified UCT configuration (Tchobanoglous et al., 2004). The received wastewater from these catchments is treated with three parallel lane with a maximum hydraulic load of 26,250 m^3/h in total in which each lane comprises a primary settler, a biological tank and four secondary clarifiers (Amerlinck et al., 2016). The influent water is exposed to both mechanical and biological treatment steps where it first through the 25x6mm bar screens at the pumping station for the large solid particle removal, followed by two parallel sand traps of 400 m^2 each and three primary clarifiers with a volume of 8,750 m^3 per clarifier (Schilperoort, 2011). In addition to that, when the influent flow exceeds the hydraulic load, the treatment plant has an additional 8,750 m^3/h capacity where the exceeded wastewater can be mechanically treated and sent to the pre-settling tank before being discharged to the River Dommel (Amerlinck et al., 2016).

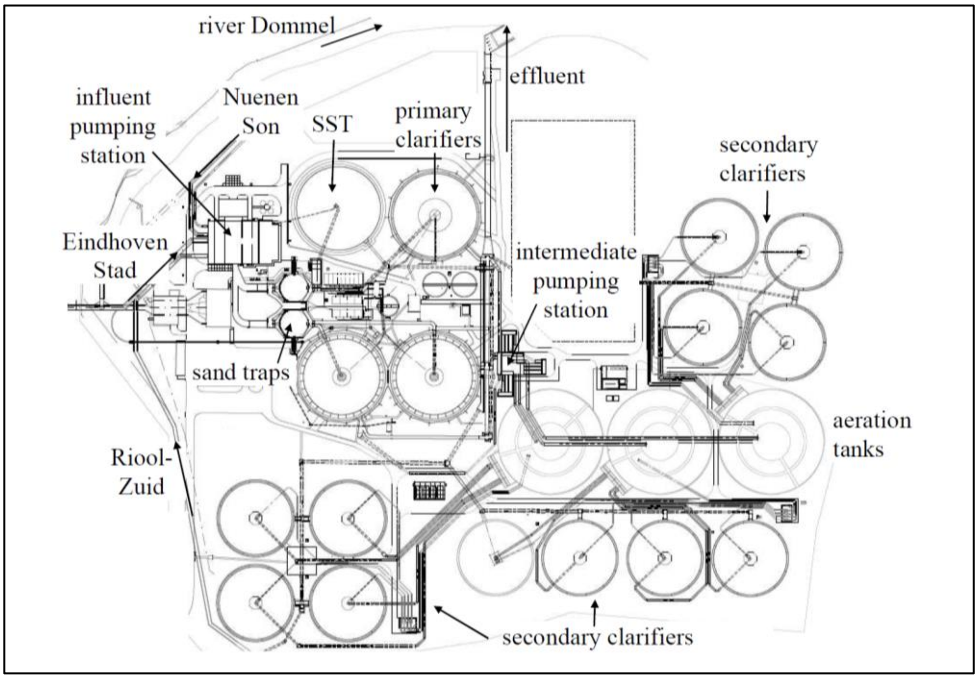


Figure 3. Layout of the Eindhoven Wastewater Treatment Plant (retrieved from Schilperoort, 2011).

The impact of the Eindhoven WWTP on the River Dommel is greatest during the dry summer time as the WWTP effluent makes up a large fraction of the total flow in the River Dommel that is approximately 50%. On the other hand, during the wet weather flow conditions the contribution of the Eindhoven WWTP to the River Dommel can be up to 90% (Schilperoort, 2011). The contributions of both DWF and WWF are illustrated by Fig. 4.

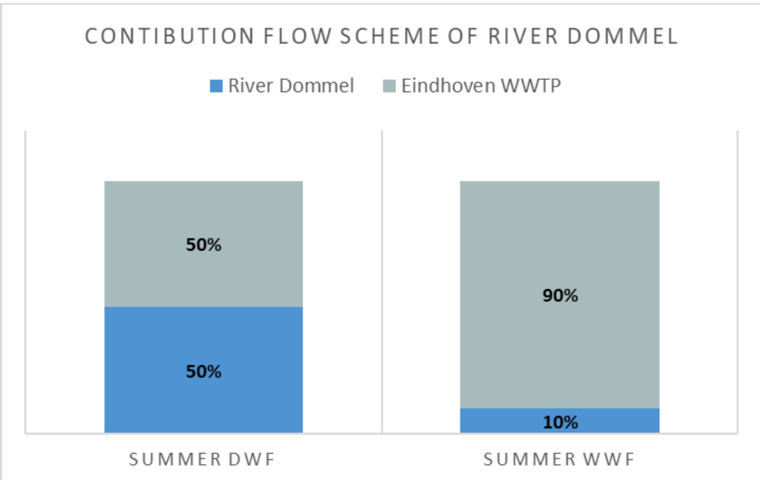


Figure 4. Summer dry weather flow (DWF) and wet weather flow (WWF) and Eindhoven Wastewater Treatment Plant (WWTP) contribution scheme to the River Dommel.

As aforementioned, the Eindhoven WWTP receives the municipal wastewater from three catchment areas: Eindhoven, Nuenen/Son and the Riool-Zuid and is composed of seven municipalities that are Geldrop-Mierlo, Heeze-Leende, Waalre, Veldhoven, Valkenswaard, Eersel and Bergeijk. During the dry weather conditions, 50% of the daily hydraulic loading of the Eindhoven WWTP is formed from the Eindhoven city, where the Riool-Zuid has a share of 40% and Nuenen/Son 10% (Schilperoort, 2011).

1.1.2.2. Characteristics of the River Dommel

According to the Provincial and Waterboard Water Plans, the River Dommel is classified as ''fish water for Cypriniformes'' which's water quality standards are described in BKMO (1994) and given in Tables 1 and 2 (Schilperoort, 2011);

Table 1. The discharge standards of Eindhoven WWTP (retrieved from Schilperoort, 2011).

**Samples are 24h volume-proportional samples.*

Parameter	maximum concentration per sample [mg/L]	maximum concentration of moving average over 10 samples [mg/L]	mean annual concentration (over 60 samples) [mg/L]	any sample
COD	125			
BOD_5	20			
TSS	30		<10	
N-total			<10	
P-total		1.0		
NH_4-N	3			
pH				$6.5 < x < 9$
temperature				< 25°C
DO				> 5 mg/L

**A total of 12 samples per year are required; one sample may exceed the stated limits but not more than 50%.*

Table 2 .Water quality demands for surface water with a function of "fish water for cypriniformes" (retrieved from Schilperoort, 2011).

Parameter	limit per sample
BOD_5	< 10 mg/L
TSS	< 50 mg/L (mean of 12 samples)
NH_4-N	< 0.8 mg/L
pH	$6.5 < x < 9$
temperature	< 3°C temperature increase with respect to natural temperature
DO	> 6 mg/L

The effluent standards for this type of surface water are, therefore, quite stringent. Even though the Eindhoven WWTP meets the effluent standards in general, during the dry weather conditions, the Eindhoven WWTP effluent shares the 50% of the River Dommel base flow that the water quality standards cannot be met.

1.2. Research Objective: "Offering a Viable Solution to Feed the River Dommel's Base Flow"

The primary objective of this research is to provide a practical solution for enhancing the base flow of the River Dommel, particularly during the dry summer periods. The study delves into various potential strategies with the goal of identifying effective measures to feed the river's base flow while ensuring strict compliance with surface water quality standards. Through comprehensive investigation and analysis, the research aims to offer a viable solution that addresses the challenge of sustaining the River Dommel's flow, thereby contributing to its environmental sustainability and ecological well-being.

1.3. Possible Solutions

In this section, three possible solutions are identified and discussed to feed the base flow of the River Dommel during dry summer time. These solutions are identified as follows:

- Rainwater harvesting and storage system in Eindhoven
- Tertiary treatment at the Eindhoven WWTP
- Sewer mining application at the catchment areas.

The potential benefits and drawbacks of each possible solution are outlined and discussed in the subsequent sections. Consequently, sewer mining was selected as the focus for further investigation, with an emphasis on the application and in-depth analysis of forward osmosis technology.

1.3.1. Rainwater Harvesting and storage system in Eindhoven

Rainwater harvesting is a potential solution involving the collection and storage of rainwater during periods of rainfall, which can then be used to feed the base flow of the River Dommel during dry summer conditions. This method, which has a long history and has been employed by various civilizations, serves as a climate adaptation strategy to address prolonged dry periods. Additionally, rainwater harvesting can be beneficial in managing the consequences of extreme rainfall events, helping to mitigate potential flooding in the catchment surroundings (Hofman and Paalman, 2014).

Rainwater is primarily harvested for non-potable purposes, such as municipal uses (e.g. watering of streets and squares, gardening, cleaning, flushing of sewer systems), industrial uses (e.g. technological processes, fire water supply, cleaning), agricultural purposes (e.g. crop growing, irrigation, animal husbandry, cleaning, agricultural production), household use (e.g. gardening, watering, toilet flushing, washing machines, car washing, cleaning), and use in trading areas, sport halls, airports, etc. (toilet flushing, fire water supply, cleaning) (Mazurkiewicz et al., 2022).

The health and hygiene criteria play a crucial role in the context of rainwater harvesting, and the treatment requirement may vary depending on the intended use of harvested rainwater. The rainwater quality depends on various factors such as rainfall intensity, the duration of the dry weather periods, the number of pollutants captured from the atmosphere and catchment areas (Sánchez et al., 2015; Hamilton et al., 2018)

The major elements that needed to be investigated for the rainwater harvesting are mainly the design principles, the scale of the application, water quality and the economic situation of the applicant (Hofman & Paalman, 2014).

Rainwater harvesting can significantly impact water management during periods of extreme precipitation. While the water quality of harvested rainwater is generally high, there might be some microbiological contamination in some cases. Moreover, small-scale rainwater harvesting systems are often considered economically impractical for low-grade applications such as drinking water, as their payback periods can exceed 60 years. In contrast, larger-scale systems tend to be more economically viable (Hofman & Paalman, 2014).

García-Ávila et al. (2023) conducted a systematic review of rainwater harvesting and storage systems from 2012 to 2022, finding that all such systems consist of four essential components:

1- Catchment area: The initial point where rainwater is collected and channelled to the gutter system.
2- Gutter(s): Designed to receive rainwater from the catchment area and convey it to the piping system.
3- Piping system: Responsible for transporting collected rainwater from the gutters to the designated storage facility.
4- Storage system: It serves as the reservoir for storing the harvested rainwater.

The classification of rainwater harvesting and storage systems is based on the collection surface types which is divided into two. The first type involves the in-situ water conservations such as small water bodies, wells, embankments that are mostly used for agricultural purposes. The second type is runoff-based systems in which the run-off water is captured and stored. This type is typically used for domestic use (García-Ávila et al., 2023; Binyam and Desale, 2015)

Rainwater harvesting is applied in the Netherlands as well and there are already several installations and projects ongoing. However, for its large-scale adoption, here is a need to establish a compelling incentive framework, accompanied by the necessity of conducting demonstration projects to explore and display its potential (Hofman & Paalman, 2014).

Rainwater harvesting is a sustainable solution for urban climate adaptation, making a notable contribution to urban water management. The collected rainwater can serve as a valuable resource for purposes such as flushing the upstream pipes in the Eindhoven area to prevent sedimentation accumulations or feeding the base flow of the Dommel during summer dry weather conditions.

Possible advantages of the rainwater harvesting for the Eindhoven area can be summarized as follows:

- An opportunity to collect and store the excess water from the Eindhoven area during short and extreme precipitation events which will decrease the load on the Eindhoven sewer system and Eindhoven WWTP
- During drought and thus water shortage periods the harvested rainwater might help to flush the upstream
- The collected and stored rainwater might help to solve the water quality problems of the River Dommel in dry periods however, it highly depends on both the quantity and quality of the harvested rainwater.

Moreover, it should be noted that, the design principles of the rainwater harvesting depend on the forecasted precipitations which can be changed over time in reality and may result in unexpected consequences such as either insufficient collected rainwater or water quality problems in the storages. Besides, during the dry periods in summer, the Dommel base flow needs to be fed continuously and the harvested rainwater might not meet the quantity requirements.

In conclusion, rainwater harvesting would be a great source to meet the small-scale water requirements for example to flush the toilets, watering the gardens and up to some extend for the irrigation purposes. The possible challenges that might come up with the rainwater harvesting can be listed as follows:

- High construction costs
- Forecasted precipitation
- Additional infrastructure and pumps are required
- A need for continuous control and monitoring system.
- Water quality

1.3.2. Tertiary Treatment at Eindhoven WWTP

Tertiary treatment is another solution that can be applied at the Eindhoven WWTP to minimize the impact of the effluent discharge on the aquatic ecosystem of the River Dommel especially during the dry weather flows. It is applied to the effluent of the secondary treatment for the further reduction of organics, turbidity, nitrogen, phosphorus, metals and pathogens (Gerba et al.,2019).

The tertiary treatment processes mostly comprise a kind of physicochemical treatment like coagulation, filtration, activated carbon process, reverse osmosis and additional disinfection. Even though the tertiary treatment target is to minimize the impact of the WWTP effluent discharges on the natural water sources such as rivers and lakes, the effluent of the tertiary treatment is commonly used for irrigation purposes where the wastewater is aimed to be reused [23]. However, within this project, the aim is to boost the River Dommel base flow during summer in order to minimize the water quality problems that may arise due to slower flow velocities (Mareddy, 2017).

The tertiary treatment is also described as the process of treating wastewater after secondary treatment with a purpose of removing the contaminants that are not removed by the secondary treatment. Therefore, more advanced treatment technologies are being used for tertiary treatment which make the effluent cleaner. The existing secondary biological treatment processes can be either expanded for the stabilization of the chemicals that need oxygen and/or for nitrogen and phosphorus removal, or, as aforementioned, physical-chemical treatment methods like carbon adsorption, flocculation/precipitation, advanced membrane filtration, ion exchange, dechlorination, and reverse osmosis can be also applied as tertiary treatment (Mareddy, 2017).

It should be highlighted that significant amount of BOD and suspended particles contained in wastewater are removed during primary and secondary treatment. Yet, in an increasing number of situations, this degree of treatment has proven insufficient to prevent damage to receiving waterways or provide reusable water for industrial and/or domestic recycling. As a result, extra treatment processes have been added to wastewater treatment facilities to allow for additional organic and solid removals, as well as for the removal of nutrients and/or hazardous compounds.

Advanced wastewater treatment is defined as any method that is designed to deliver a better-quality effluent than is commonly obtained by secondary treatment processes or which involves unit operations not often seen in secondary treatment and tertiary treatment falls under the advanced wastewater treatment methods. Any treatment technique that includes the addition of unit operations to the flow scheme after standard secondary treatment is referred to as tertiary treatment. This can be achieved by simply implementing a filter for suspended solid removal to the effluent of typical secondary treatment or adding various unit processes for organic, suspended solids, nitrogen and phosphorus removal which can be regarded as more complicated solution (Mareddy, 2017).

Choosing the most appropriate technology/method for the tertiary treatment is challenging and complex as all the aforementioned technologies have certain strengths and weaknesses. In addition to that, choosing the most convenient technology highly depends on the quality of the final effluent required and therefore depends on the purpose of the effluent water use, which in this study is to minimize the Eindhoven WWTP effluent impact on the River Dommel's water quality, therefore, on the aquatic life, during dry summer periods. For example, if the effluent is received by bathing waters, UV irradiation is known to be principally chosen method to reduce the bacteria. However, chlorine had been used earlier despite its potential negative impacts on the environment such as bleaching and direct toxicity to the aquatic environment of the receiving waters. In addition to that, phosphorus concentrations may need to be lowered in some cases, particularly if the receiving waters are lakes or rivers (EU Urban Wastewater Directive 91/271/EEC) as increased phosphorus levels in the waterbodies may lead to eutrophication. Precipitation as $FePO_4$ or $Fe_3(PO_4)_2$ with the addition of $FeCl_3$ or $Fe_2(SO_4)_3$ is a common way of eliminating dissolved phosphorus. In case of a requirement for further solid removal, sand filters or other clarifiers can also be included in the tertiary treatment. The residuals are commonly collected and mixed with the other sludges on site for further treatment and disposal (Mudge et al., 1964).

1.3.3. Sewer Mining at Eindhoven Area Catchments

Sewer mining is one of the wastewater recycling inventions to deal with the water scarcity problem and play with the urban water cycles. It is known to be an efficient solution for small-scale wastewater treatments in which less space is required to install the treatment units. Unlike the conventional wastewater treatment plants, sewer mining is typically a compact and sometimes even a portable advanced treatment plant in which the clean water is extracted from the wastewater from an existing sewer in order to enable reuse at the point of demand. In many cases the extracted water is used for public space irrigations and toilet flushing (Hadzihalilovic, 2009; McGhie et al., 2009, Sydney Water, 2006). Besides, the treated water from sewer mining is also being used for non-potable domestic water purposes in some cases such as laundry water (Marleni et al., 2013).

The water reclamation methods used are generally conventional sewage treatment, multimedia filtration, microfiltration (MF) or ultrafiltration (UF), and reverse osmosis (RO) (Bartels et al., 2005; Raffin et al., 2013; Shang et al., 2011). For the municipal wastewater treatment, primary settlers, activated sludge processes, and secondary settlers (with the sludge stabilized in a digester prior to dewatering and disposal) are known to be the conventional units/methods, however, depending on the required effluent concentrations, further treatment methods may need to be applied. RO applications on the other hand, are known to have high rejection rates manageable costs, and ease of operation. Yet, various forms of fouling reduce the efficiency, performance and salt rejection capacity. Membrane fouling forms can be categorized as particulate fouling, organic fouling, inorganic fouling (e.g.scaling) and biofouling. Both the extend and form of the fouling are highly related to the wastewater content and concentrations. In order to deal with fouling, various pre-cautions can be taken such as pre-treatment before membrane and regular cleaning (e.g. backwashing, chemicals). For example, for RO, pre-treatment is essential to cope with biofouling (Kramer, 2019).

The previous studies on sewer mining reveal that membrane bioreactors are the most common treatment method selected to treat wastewater from sewers (Hadzihalilovic, 2009; McGhie et al., 2009, Marleni et al., 2013). The wastewater coming from the sewer system is then passing the membranes to extract the water from the wastewater. In most cases, the concentrated water is discharged to the sewer system back to send it to the closest wastewater treatment plant.

As aforementioned sewer mining is based on the extraction of the wastewater from the sewage, followed by cleaning processes to reuse the wastewater. However, it may result in several sewer problems such as blockages when sludge disposal of the sewage extraction is disposed back to the sewer networks. Therefore, sewer mining has the potential to be a remarkable contributer to the blockages in the sewer pipes. Besides, the sewer mining may result in changes in the sewage composition in the sewer system that can lead to differentiations of the biochemical transformation processes such as an increase in hydrogen sulphide concentrations, therefore, odour and corrosion problems in the sewer systems (Tchobanoglous et al., 2004). This will be further investigated within this project.

A possible treatment unit for the sewer mining can be an application of two sub-units composed of a membrane bioreactor (MBR) and a Reverse Osmosis (RO) unit that. They can be both implemented as individual packaged modules and form one compact system. The MBR sub-unit can be divided into several compartments where the treatment sub-processes take place. These compartmants can be also used for buffering purposes to enable a variation of the treated sewage volume and serve as:

- A primary tank equipped with a coarse filter at the end, designed to remove floating and settling substances.
- Denitrification tank dedicated to the removal of nitrate.
- Nitrification tank featuring an aeration system with dissolved oxygen (DO) sensors to facilitate the oxidation of organic materials and the nitrification of ammonium nitrogen.
- Membrane tank where the permeate is directed to the RO unit.
- A final settling tank for concluding the treatment process, allowing for the removal of sludge from the bottom.

This sewer mining modules can also be equipped with sensors, data collection instruments and control devices which can be intagrated into an ICT smart platform (Karagiannidi et al., 2016).

Another possible treatment technology for the sewer mining is applying FO. This technology is based on an osmotic gradient driving force. In FO, the feed solution is driven through the membrane by a draw solution that has a higher ion concentration than the feed solution. Unlike the other membrane filtration technologies, FO requires less energy as the driving force is the osmotic pressure and no additional pressure is required (Lutchmiah, 2014). When it is used with RO as a hybrid system, the RO membrane allows high rejections of all the contaminants that are concentrated in the reject stream. This technology has a low fouling tendency and low energy is required for the permeation where high volumes of clean water can be obtained (She et al., 2016). To extract water from sewage, however, a high ion concentration in the draw solution of FO is necessary. Because of the high ion concentration, a high pressure RO is required, which uses a significant amount of energy (Holloway et al., 2007; Kramer et al.,2015). Furthermore, since the flux of the FO is quite low, the implementation of this technology becomes limited (Qin et al.,2010; Kramer et al., 2015).

Depending on the wastewater concentrations, the FO system can be also used in combination with an anaerobic digestion system. Using the combination of FO and anaerobic treatment was introduced by Ansari within the sewer mining concept (Ansari et al., 2017). This combination may allow the recovery of both clean water from the FO process and the energy contained in the FO reject system (Ferrari et.al., 2019).

Another innovative sewer mining concept can be built by ceramic nanofiltration membranes for pre-treatment before the RO. The ceramic membranes are known be adventageous due to their high mechanical strength, high chemical and thermal resistance (Kramer, 2019; Weber et al., 2003).

Kramer (2019) reported that both ceramic tight ultrafiltration (UF0 and ceramic nanofiltration (NF) membranes can be fed by filtrated domestic sewer and both of them could be operated for 1-4 days without any cleaning. On average 81% of the organic matter was rejected by both of the membranes. Moreover, the pressure drop increase in the Membrane fouling simulator (MFS) fed with ceramic NF permeate was low during an operation of 14 days. According to this study, these results were comparable with the increase in pressure drop of an MFS fed with Dutch drinking water. The advantages of ceramic membranes can be summarized as follows (Kramer,2019):

- High mechanical strength
- High chemical and thermal resistance
- Can deal with high pressures, high temperatures and chemicals in high concentrations
- Therefore makes vigorous chemical cleaning in the membranes
- Long life time (more than 15 years)
- Less irreversible fouling than polymeric membranes
- Suitable for direct municipal sewage treatment

The disadvantage of ceramic membranes is stated to be more expensive than polymeric membranes.

1.3.4. Selected Solution: Sewer Mining with FO&RO Hybrid System

In conclusion, out of discussed possible solutions, sewer mining has been chosen for further investigation. The reasoning for this choice is, with rainwater harvesting, a huge storage capacity would be needed which will be discussed in results section. Secondly, application of the tertiary treatment in Eindhoven WWTP would only feed the downstream of the River

Dommel. However, the purpose of this study is to find a solution that will help to feed the upstream of the River Dommel.

There are multiple treatment technologies that can be used for the sewer mining process, however, within this study, the FO-RO hybrid system has been chosen. The application of FO in sewer mining processes represents a technically feasible solution for wastewater treatment. However, it should be noted that the choice of the draw solution, reconcentration unit, membrane, and FO design plays a vital role in optimal driving force and cost-efficiency. The current vulnerability of the process is associated with potential fouling issues and the cost of FO membranes. Nonetheless, as a sustainable and energy-neutral concept, the application of forward osmosis in sewer mining holds promise as a wastewater treatment solution (Lutchmiah, 2014). In addition to that, when utilized together with RO as a hybrid system, the RO membrane demonstrates remarkable rejection capabilities for concentrated contaminants in the reject stream. This technology exhibits a low fouling tendency and demands low energy for permeation, which can result in the efficient production of large volumes of clean water (She et al., 2016). Further discussion will be provided in Chapter 3. Results.

1.4. Research Question and Sub-questions

Based on the literature studies and research objective, the research question (**RQ**) is revised as:

" Is sewer mining a feasible solution to Enhance River Dommel's Base Flow?"

Following sub questions (SQs) have been defined to answer the research question:

1. How much water can feasibly be extracted from the selected catchment areas to sustain the River Dommel's flow during dry periods? (**SQ1**)
2. How effective and efficient is the integration of FO&RO hybrid system for sewer mining? (**SQ2**)
3. How can the adverse impacts of sewer mining on the sewer system be effectively evaluated? (**SQ3**)
4. What are the optimal locations for implementing sewer mining technology? (**SQ4**)
5. How can potential hydrogen sulfide-related challenges, such as corrosion and odour, be effectively managed in sewer mining operations? (**SQ5**)
6. What is the contribution of the sewer mining implementation to River Dommel base flow? (**SQ6**)
7. What is the economic viability of FO-RO hybrid system as a sewer mining solution? (**SQ7**)

1.5. Report Structure

This thesis report is divided into various chapters. The first chapter, Chapter 1, serves as the introduction section, providing insights into the thesis study, the addressed problem, the location where the problem occurs, and the resulting consequences. This is followed by the research objective, possible solutions to mitigate the problem addressed, research question, and sub-questions. The second chapter, Chapter 2, provides the theoretical framework , methodology and the literature review and in order to define the sewer mining implementation concept followed by the research method used, the research area and its properties, investigation of the possible adverse impacts of sewer mining implementation on the Riool-Zuid sewer system, mitigation measures for these adverse impacts, data processing and research scenarios for further investigation. Chapter 3 reveals the results of the sewer mining implementation based on created scenarios, required chemical dosages to mitigate its adverse effects and cost analysis. Chapter 4 provides the discussions of the literature study results and comparisons with the outcomes of this thesis study results. Chapter 5 is composed of conclusions to manifest whether the selected solution, ''sewer mining implementation'', is an efficient, reasonable and feasible concept. The last chapter, Chapter 6, discloses the recommendations.

Chapter 2. Theoretical Framework, Methodology and Literature Review

This research aims to propose a feasible solution for addressing water quality challenges in the Dommel River, particularly during dry summer conditions, by increasing its base flow. In this chapter, the research methodology is outlined, starting with a description of the chosen research method. Afterward, the potential solutions are described, a selection for in-depth investigation is made, and the potential adverse impacts are assessed along with mitigation measures. Following this, data processing procedures are detailed and the research scenarios developed are presented for the investigation.

2.1. Sewer Mining Implementation

This section comprises three key sub-sections aimed at elucidating sewer mining process. The first sub-section delves into the sewer mining concept, explaining the steps involved in wastewater extraction from the Riool-Zuid sewer system and detailing the membranes utilized for treatment. The second sub-section gives insights into water/wastewater quantities, providing an overview of the water/wastewater mass balance dynamics associated with sewer mining. The provided illustration in this sub-section further clarifies the relationship between wastewater extracted from the sewer system, water sent to River Dommel, and concentrated water reintroduced into the sewer system. In the third sub-section, the treatment capacities of sewer mining technology are explored, focusing on the efficiency of forward osmosis and reverse osmosis membranes in removing COD, BOD, TSS, nitrogen and phosphorus from wastewater.

2.1.1. Conceptual Framework and Membrane Selection

The primary objective of this study is to enhance the base flow of the River Dommel by introducing high-quality water. The implementation of sewer mining in this study involves extracting wastewater from the existing sewer system, treating it to recover water, and then directing this treated water to feed the base flow of the River Dommel. The sequential steps of this process are outlined below:

1. The wastewater is extracted from the Riool-Zuid sewer system
2. The extracted wastewater is sent to the sewer mining technology (pre-treatment + membrane filtration)
3. The treated wastewater, permeate water, is collected and sent to the River Dommel to boost its base flow
4. The rejected water of the treatment units is sent back to the Riool-Zuid sewer system.

The focus of this research's sewer mining concept is comprised of a Forward Osmosis (FO) and Reverse Osmosis (RO) Hybrid System. The raw wastewater taken from Riool-Zuid's sewage system will be directed to a FO&RO Hybrid System. The system's treated water, known as permeate, is intended to be sent to the Dommel River in order to boost its base flow during dry summer periods. The rejected water, known as concentrate water, will be returned to the Riool-Zuid sewage system.

The need for a pre-treatment system prior to the FO&RO hybrid system is essential to be determined for an efficient, reliable, and cost-effective treatment concept design. When conventional treatment systems are taken into account, pre-treatment methods are applied in order to remove the large particles, suspended solids, and other materials that may cause fouling or damage which can decrease the membrane's lifetime and therefore, can increase the costs. The pre-treatment methods that are commonly used are known as screening, settling, filtration, and chemical treatment. Pre-treatment prior to membrane filtration decreases the fouling risk and increases the membrane efficiency and lifetime to a certain extent. For example, RO and NF are prone to fouling and require pre-treatment to increase longevity and minimize costs (Sutzkover-Gutman et al., 2010; Kim et al., 2011). However, when the FO membranes are taken into account, extensive pre-treatment systems for FO may be superfluous. Yet it it relies on the performance and membrane design of the FO (Lutchmiah, 2014).

The study results of Yang et al., 2019 given in Fig. 5 and Fig. 6 illustrate that the pre-filtration of raw sewage has little effect on FO membrane flow recovery. In other words, the suspendend solids are not expected to cause a relevant damage like irreversible fouling. According to this study, pre-treatment is not necessary required in the FO-based sewage treatment.

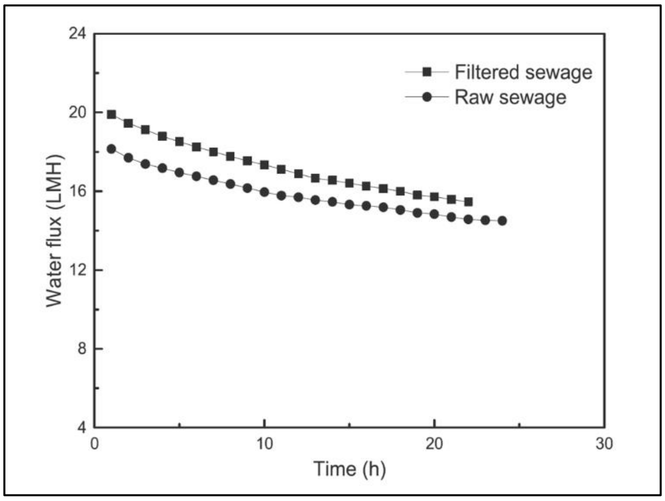


Figure 5. Water flux f the Forward Osmosis (FO)-based sewage concentration affected by sewage pre-treatment (retrieved from Yang et al., 2019).

Membrane flux recovery after physical cleaning.

Conditions	Original flux, LMH	Recovered flux, LMH	Efficiency, %
15 °C + 280 ml/min	19.7 ± 0.1	19.1 ± 0.1	96.95 ± 1.0
25 °C + 280 ml/min	20.1 ± 0.1	19.73 ± 0.07	98.16 ± 0.84
35 °C + 280 ml/min	20.55 ± 0.15	20.20 ± 0.2	98.30 ± 1.68
35 °C + 210 ml/min	20.35 ± 0.05	19.13 ± 0.07	94.0 ± 0.58
35 °C + 140 ml/min	19.48 ± 0.08	18.73 ± 0.02	96.15 ± 0.49
Filtered sewage	20.2 ± 0.2	19.89 ± 0.05	98.47 ± 1.22

Figure 6. Forward Osmosis Membrane flux recovery after physical cleaning (retrieved from Yang et al., 2019).

The sewer mining technology chosen to be applied within this study is an FO&RO Hybrid System is considered for each scenario in Riool-Zuid. Even though the study of Yang et al., revealed that the pre-treatment is not necessarily required, within this study a pre-treatment is still included to remove the large particles and suspended solids from the raw sewage in order to keep the water fluxes at the maximum and minimize the cleaning requirement. Therefore, in this study, the wastewater from the catchment first flows to the pre-treatment unit. Fine sieves have been reported to efficiently remove the largest fraction of suspendend solids (i.e. cellulose fibres) from raw sewage (Lutchmiah, 2014; Kramer 2019; Ruiken et al., 2013). As an pre-treatment unit '' Reko zeefbocht'' is planned to be used which has an capacity of 300 m³/h and a lifespan of 20-25 years.

Fig. 7 illustrates the flow scheme of the planned FO&RO Hybrid System.

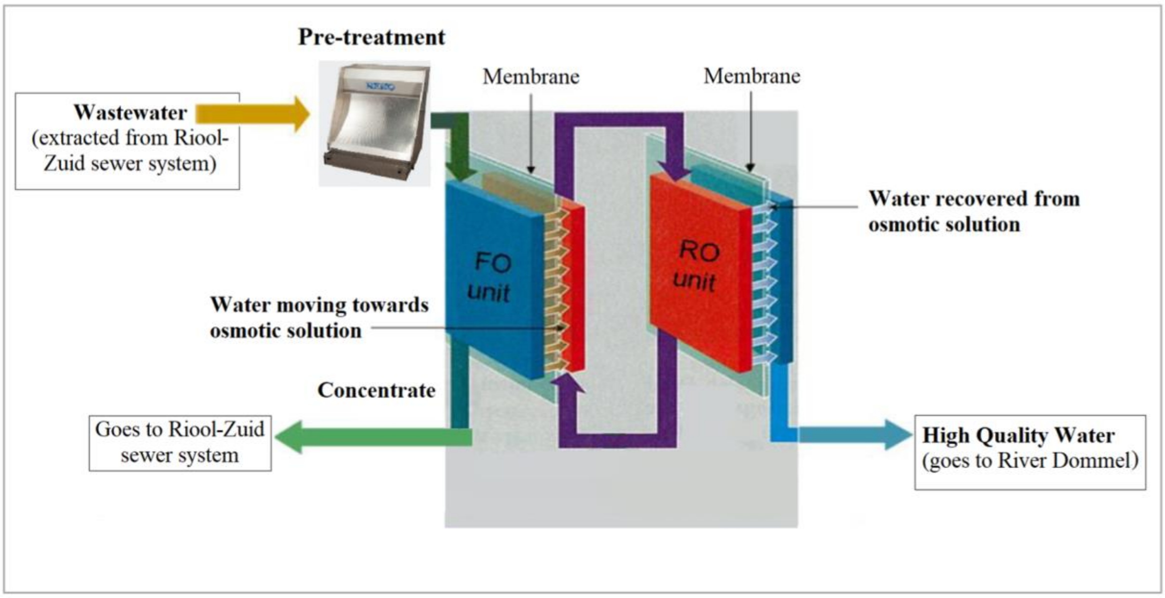


Figure 7. Sewer Mining Technology applied for Riool-Zuid catchment (Adapted from Lutchmiah, 2014).

2.1.2. Mass Balance Dynamics in Sewer Mining

As it is already illustrated in Fig. 7, the pre-treated water goes to the FO unit in which the wastewater is exposed to the osmotic solution. Filtrated water goes afterwards to the RO unit where the water is exposed to osmotic pressure and flows through the RO membranes. The filtrated water, which is called ''permeate'' is then collected and sent to the River Dommel in order to feed its base flow during summer. On the other hand, the concentrate water of the process is sent back to the Riool-Zuid sewer system. **The recovery rate** of the FO/RO process is assumed to be **60%** which means 40% of the extracted water goes back to the sewer system. These percentages depend on the wastewater extraction ratios which will be further discussed in 2.7.Research Scenarios section. However, to provide a comprehensive understanding of the water/wastewater circulation beforehand, consider the following example:

Assuming a wastewater extraction ratio of 90% and a wastewater flow rate of 500 m³/h at the extraction point of the sewer system, the following schema (Fig.8) can be derived:

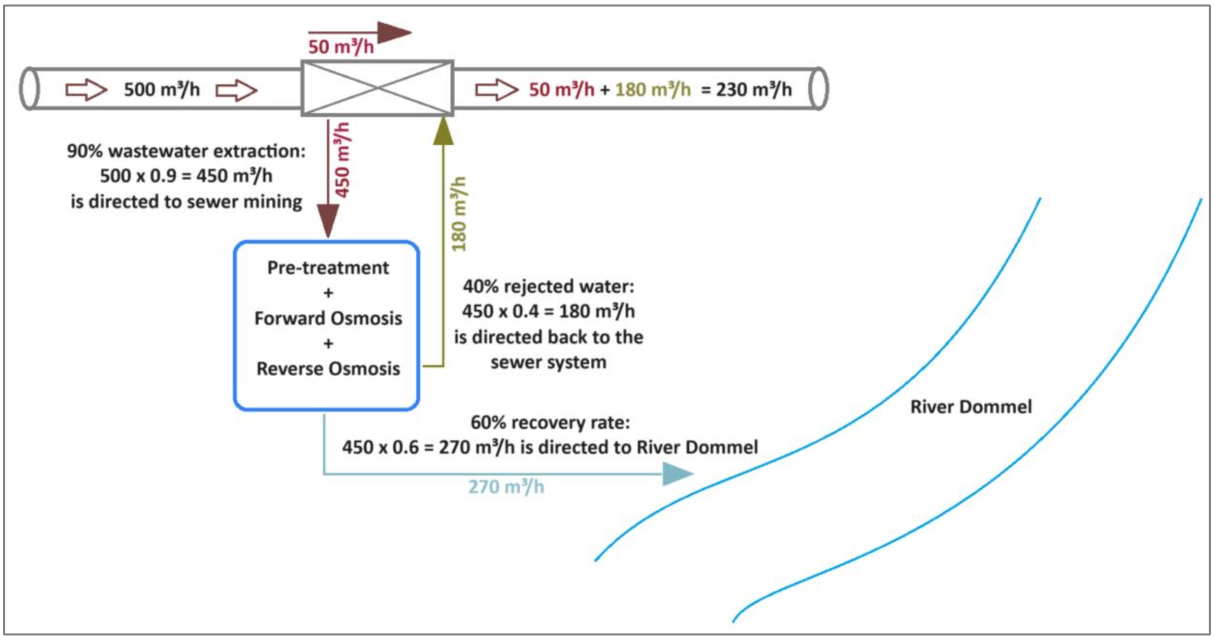


Figure 8. An example of water/wastewater circulation in Riool-Zuid catchment with sewer mining application.

The Fig.8 illustrates that with a total flow of 500 m^3/h, extracting **90%** of the wastewater to feed the base flow of the River Dommel, and considering a 60% recovery rate in the sewer mining application, the effective quantity of water contributing to the base flow of the River Dommel would be 270 m^3/h. This accounts for **54%** of the initial total flow (which was regarded as 500 m^3/h).

2.1.3. Treatment Capacities of Sewer Mining Technology

Regarding the wastewater concentrations, NH_4 and COD concentrations of the Riool-Zuid wastewater are retrieved from the study of Langeveld et al., 2017 which are given in Fig. 9.

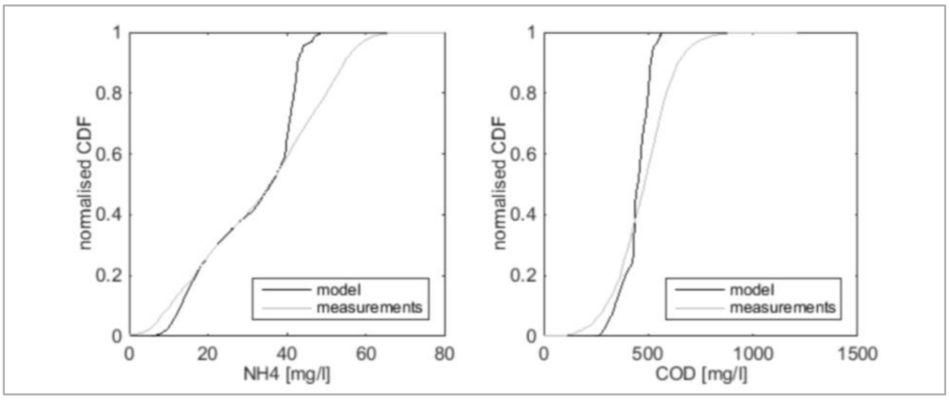


Figure 9 . Normalized Cumulative Density Functions (CDFs) of Empirical Sewer Water Quality Model Results and measurement for NH4 (left graph) and COD (right graph) in Eindhoven Wastewater Treatment Plant (WWTP) influent for the Riool Zuid catchment (retrieved from Langeveld et al., 2017).

Considering the study results given in Fig.9 (Langeveld et al., 2017), in this study, the COD concentration of the raw Riool-Zuid wastewater is regarded as **675 mg/L**. After applying the FO and RO hybrid membrane filtratation system, the concentrate is assumed to be two times more concentrated. In other words, the COD concentration of the concentrate is assumed to be **1350 mg/L**.

Forward Osmosis (FO) membrane technology emerges as a promising avenue for effectively removing and recovering nutrients from wastewater, marking a significant stride towards sustainable wastewater treatment practices. Recent advancements underscore its potential as a low-fouling membrane process, showcasing remarkable efficiency in nutrient removal. Various studies have demonstrated the high removal efficiencies of FO membranes for key nutrients like nitrogen and phosphorus, crucial for mitigating environmental pollution and facilitating resource recovery.

For instance, Nguyen et al. (2013) observed impressive removal rates, with FO membranes removing approximately 96% of NH4+ - N, 98% of PO3-4-P, and achieving a 99% removal of dissolved organic carbon (DOC). Similarly, Gao et al. (2018) reported significant COD rejection rates of 97%-99% in feed water. Hafiz et al. (2019) found FO membranes to exhibit total phosphorus rejection rates of 99% and ammonium rejection rates of 97%. Schneider et al. (2019) explored Aquaporin InsideTM TFC flat sheet FO membranes, noting rejection rates of up to 96.95% for total ammonia nitrogen (TAN), 95.87% for total Kjeldahl nitrogen (TKN), and 99.83% for total phosphorus (TP). Premanik et al. (2019) also documented COD rejection rates exceeding 97% and phosphate rejection rates surpassing 98% (Jafarinejad, 2021).

In summary, FO membranes provide a robust platform for nutrient separation, leveraging their selective permeability and resistance to fouling. Furthermore, the technology holds promise for achieving sustainable nutrient recovery objectives in wastewater treatment. As research progresses, optimizing FO processes, enhancing membrane performance, and addressing operational challenges will further enhance its efficiency and broaden its applicability in nutrient removal and recovery systems.FO technology demonstrates notable potential as a low-fouling membrane process, offering high removal efficiencies for nitrogen (up to 90%) and phosphorus (exceeding 95%) from wastewater. These efficiencies underscore the efficacy of FO membranes in mitigating environmental pollution and facilitating resource recovery.

According to the existing literature, forward osmosis (FO) treatment demonstrates remarkable efficacy in rejecting various contaminants. Notably, studies by Gao et al. (2018) indicate substantial COD rejection rates ranging from 97% to 99%. Given the inherent correlation between COD and BOD (Biochemical Oxygen Demand), a similar rejection rate of approximately 97% to 99% can be reasonably extrapolated for BOD removal. Considering the nature of FO membranes the total suspended solid (TSS) removal can likely be exceeding 90%. Regarding Ammonium Nitrogen (NH4-N), studies by Nguyen et al. (2013) and Hafiz et al. (2019) reveal rejection rates of approximately 95% to 97%. Similarly, for Phosphorus (P), studies by Nguyen et al. (2013), Hafiz et al. (2019), and Schneider et al. (2019) showcase rejection rates nearing 98% to 99%. These findings ,also included in Table 3, underscore the efficacy of FO treatment in achieving high rejection rates for a spectrum of contaminants, thus highlighting its potential as a reliable solution for water purification.

The characteristics of wastewater, including its composition and pollutant concentrations, play a crucial role in determining treatment requirements and process efficiencies. For instance, in wastewater with a COD concentration of 675 mg/L, as considered to be the

COD concentration of Riool-Zuid sewage, the BOD concentration can be expected to fall within the range of 300 to 400 mg/L.

TSS concentrations in domestic wastewater typically span from 100 to 400 mg/L. Taken the COD concentration of 675 mg/L into account, it is reasonable to expect TSS concentrations to range from 200 to 500 mg/L, with variations influenced by the presence of solid matter and the effectiveness of treatment processes.

NH_4-N concentrations in untreated domestic wastewater exhibit variability due to factors such as human activities and environmental conditions. Typically, NH_4-N concentrations range from 10 to 50 mg/L. In wastewater with a COD concentration of 675 mg/L, NH_4-N concentrations within the range of 10 to 60 mg/L can be considered reasonable.

Total Phosphorus (TP) concentrations in domestic wastewater are subject to fluctuations influenced by dietary habits and cleaning products. Untreated wastewater typically contains P total concentrations ranging from 5 to 20 mg/L. With a COD concentration of 675 mg/L, P total concentrations ranging from 5 to 25 mg/L can be expected.

These wastewater concentrations provide valuable insights into the expected pollutant concentrations in wastewater based on a COD level of 675 mg/L. The final assumptions made regarding the wastewater concentrations in Riool-Zuid are given in Table 3 with the treatment capacities of both FO and RO membranes in order to give a good insight into the concentrations that can be expected after implementing the FO&RO hybrid system as a sewer mining technology.

These wastewater concentration estimations offer valuable insights into anticipated pollutant levels, based on a COD concentration of 675 mg/L. For a comprehensive overview of expected wastewater compositions in Riool-Zuid, the final assumptions are summarized in Table 3, inclusive of treatment capacities for both FO and RO membranes. This table provides a full picture of expected pollutant concentrations following the integration of the FO&RO hybrid system as a sewer mining technology, facilitating a deeper understanding of potential treatment outcomes.

Table 3. Concentrations before and after FO and RO filtrations
*(*1: Schilperoort, 2011; *2: Schneiders et al., 2009)*

Parameter	Concentrations in Riool-Zuid [mg/L]	Rejection FO		Concentrations after FO [mg/L]	Rejection RO		Concentrations after RO [mg/L]	Eindhoven WWTP Discharge Standards *1	Water Quality Demands for cypriniformes
		Min	Max		Min	Max			
COD	**675**	97%	99%	**20.25**	95%	99%	**1.01**	125	
BOD_5	**400**	97%	99%	**12.00**	90%	99%	**1.20**	20	**<10 *1**
TSS	**500**	>90%		**50.00**	90%	99%	**5.00**	30	**<50 *1**
NH_4-N	**65**	95%	97%	**3.25**	90%	99%	**0.33**	3	**<0.8 *1**
TP	**20**	98%	99%	**0.40**	90%	99%	**0.04**	1	**<0.3 *2**

The cleaning of the FO membranes can be done by physically by water flushing without the use of chemical cleaning reagents (Lutchmiah, 2014). The cleaning cycle of the FO membranes in this study is assumed to be one time in a week. Depending on the fluxes, it can be also extented up to one time in a month.

Further information regarding FO membrane types, membrane areas, fluxes, and required number of FO membrane modules are given in section 3.5. Cost Analysis.

2.2. Research Method used: ''Mega-WATS Modelling''

This section reveals the research method that is applied in this study which is modeling of the sewer network processes to identify the potential sulphate-related issues in Riool-Zuid sewer system which tend to arise after implementing the FO-RO system as a sewer mining technology. In other words, the FO-RO system is intended to be fed with the wastewater extracted from Riool-Zuid catchments based on multiple scenarios. As the wastewater extraction from an existing sewer system impacts the current flows in the system, the adverse effects are inevitable.

The assessment of sulphide and corrosion problems within the Eindhoven sewer system network is conducted using the Mega-WATS Model. Originally developed in the mid-90s by Hvitved-Jacobsen, Vollertsen, and Nielsen, it has since been further advanced and adapted into the Mega-WATS sewer process model by Vollertsen (The WATS Guys, n.d.). The WATS of Mega-WATS stands for the Wastewater Aerobic/Anaerobic Transformations in sewers. This model simulates the biological and chemical processes in sewer networks and predicts how wastewater constituents are transformed and degraded when the wastewater flows from its source to the final treatment. This model helps the users to assess the impacts from the collection system, corrosion of the sewer assets, impacts on the treatment works, to manage the odours and hydrogen sulphide in sewer networks and to predict the future developments in the sewer networks. This model typically aims to analyse the generation, concentration, fate of the hydrogen sulphide the concrete corrosion caused by hydrogen sulphide, concrete corrosion caused by hydrogen sulphide, odour problems caused by hydrogen sulphide and VOCs emission into the urban atmosphere, the wastewater quality at inflows to WWTPs and solving these problems by enabling hydrogen sulphide and VOC control strategies (Hvitved-Jacobsen et al., 2013).

Utilizing the sewer system data input and geographical mapping, Mega-WATS provides the sulphide concentrations in the pipes. facilitates the determination of sulphide concentrations within the pipelines. Sulphide concentrations surpassing predefined thresholds are visualized in distinct colours within the model. Users have the flexibility to define these thresholds, extending the capability to other criteria such as corrosion assessment throughout the network. By zooming in, users gain a more detailed view of pipe structures and gain insights into specific parameters, including corrosion, COD (Chemical Oxygen Demand), fatty acids, biomass, pH, or sulphate levels within the pipes. The model also generates graphs to illustrate how the selected parameter changes over time. Moreover, this model helps to find the most appropriate solutions such as adding some chemicals like iron to deal with the sulphide problems. The user can also see the impact of chemical dosing through model and determine the optimum dosage to manage sulphide. In addition to that, the model allows the user to illustrate the impact of the iron dosage on pH.

2.3. Research Area and Its Properties

This section provides comprehensive insights into the case study area, the 'Eindhoven Wastewater System.' It comprises detailed calculations regarding the available water quantities for extraction from the catchment areas and an examination of the wastewater characteristics within the Eindhoven area sewer system.

2.3.1. Case Study Area "Eindhoven"

This study primarily focuses on addressing the summer-time water quality issues in the River Dommel, particularly within the Eindhoven area. It involves an in-depth investigation of the Eindhoven area wastewater system to mitigate water quality problems associated with summer dry weather flows in the River Dommel.

The following schema, Fig.10, illustrates the contributing cathment areas of the Eindhoven wastewater system;

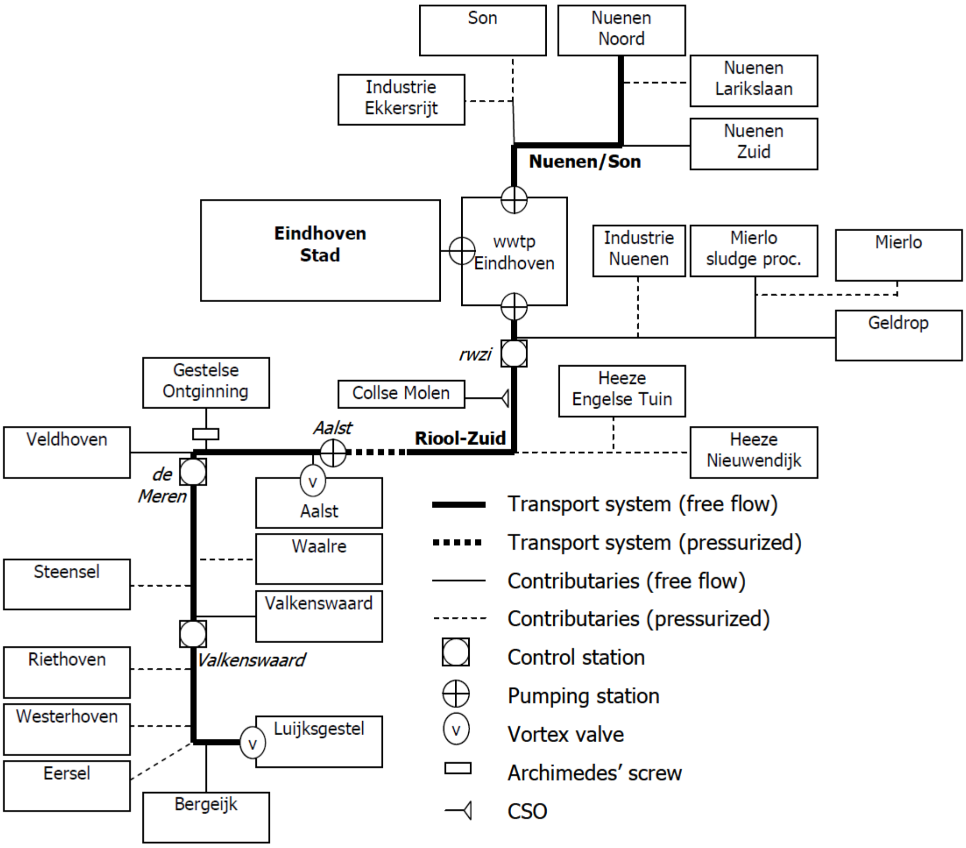


Figure 10. Eindhoven area wastewater system contributors' scheme (retrieved from Schilperoort, 2011).

Eindhoven area wastewater system is composed of 10 municipalities and municipal sewer systems that all the collected wastewater is transferred to the Eindhoven WWTP. The total municipal surface area adds to approximately 600 km^2 where the impervious area corresponds to the 7.1% that the stormwater is being discharged to the sewer system (Schilperoort, 2011).

The total number of the inhabitants in the Eindhoven area is roughly 425,000 and the half of the inhibitors (roughly 210,000) live in the Eindhoven city itself that makes the Eindhoven city the largest contributor. On the other hand, Heeze-Leende, Son en Breugel, Waalre, Eersel and Bergeijk are the smallest contributors with the number of inhabitants between 15,000 and 20,000 (Schilperoort, 2011). Moreover, the water received from the industries, mostly from Son en Breugel and Geldrop-Mierlo, corresponds to approximately 150,000 p.e. of wastewater that is also sent to the Eindhoven WWTP. Unlike these municipalities, almost no industrial water is discharged to the sewer system from Nuenen (Schilperoort, 2011).

The sewer systems in the Eindhoven area wastewater system are mostly combined sewers (approximately 78%) and the rest is composed of separate sewers. As aforementioned, the largest contribution of wastewater comes from the Eindhoven city with roughly 210,000 inhabitants' municipal wastewater, more than 65,000 p.e. industrial wastewater and stormwater of 20 km^2 of impermeable area. Secondly, another large amount of wastewater is being discharged from the Riool-Zuid catchment area that is composed of the wastewater of seven municipalities, approximately 14,000 p.e. industrial wastewater and stormwater collected from 17 km^2 of impermeable area. The collected wastewater is discharged to the transport main of Riool-Zuid.

It should be also noted that Nuenen/Son is the last catchment area where the approximately 38,000 inhabitants' municipal wastewater, nearly 14,000 p.e. industrial wastewater and 4.6 km^2 impermeable area's collected stormwater of Son en Breugel and Nuenen are conveyed to Eindhoven WWTP via the Nuenen/Son transport main.

2.3.2. Wastewater contribution of catchments to Riool-Zuid

In this study, the focus will be on the Riool-Zuid catchment areas that are illustrated in Fig 10. The contributors of Riool-Zuid are as follow:

- Bergeijk (Villages of Luyksgestel, Riethoven and Westerhoven are included)
- Eersel
- Steensel
- Valkenswaard
- Veldhoven
- Waalre
- Aalst
- Geldrop
- Mierlo
- Heeze
- Sterksel
- Leende

In order to find out how much water can be extracted from the selected catchment areas; the calculations were made based on the water consumption per inhabitant and the populations of these areas. Water consumption per inhabitant is regarded as **120 L/d**. As the focus of this study is to deal with the dry weather flow impact on the river Dommel in summer time, rainwater flow factor is not taken into account.

The population of the Veldhoven is the greatest in the Riool-Zuid catchment area. Therefore, the calculations made are elaborated below for the Veldhoven to give an insight into the results shown in Table 4.

Veldhoven

Population = 45,500 [27]
$\mathbf{Q_{DWF}}$ = 45,500 x 120 / 1000/ 24 = **227.50 m³/h, (0.063 m³/s)**

Table 4. Riool-Zuid catchment area corresponding nodes, number of inhabitant and dry weather flows (Q_{DWF}) (The population of each catchment are retrieved from Allecijfers.nl, 2022).

Catchment Area	Node ID	Inhabitants	Q_{DWF} [m³/h]	Q_{DWF} [m³/s]
Bergeijk	27.TSL8	18,754	93.77	0.0260
Eersel	27.TP148	19,528	104.32	0.0290
Steensel		1,335		
Valkenswaard	ID_000492	31,213	156.07	0.0434
Velthoven	38.RS_DE_MEEREN	45,500	227.50	0.0632
Waalre	33.TS78	17,544	87.72	0.0244
Aalst	38.TS49	2,222	11.11	0.0031
Geldrop (1/2)	Nod_A*HLRZ00970970970 *1	28,367	70.92	0.0197
Mierlo	*2	11,695	58.48	0.0162
Heeze + Sterksel + Leende	Nod_A*HLRZ00957957957	15,745	78.73	0.0219
Geldrop 1/2+Mierlo	Nod_A*HLGDRW1290 *3			0.0359
Total			**959.52**	**0.2665**

*1 The half of the Geldrop wastewater is received by the node "Nod_A*HLRZ00970970970"
*2,3 The wastewater of the Mierlo is collected together with the other half of the wastewater of Geldrop and are both received from the node Nod_A*HLGDRW1290

The total amount of water that can be extracted from the catchment areas of Riool-Zuid is therefore, calculated as **959.52 m³/h**.

The observed mean DWF of the Riool-Zuid is reported to be 1,840 m³/h (Schilperoort, 2011). Table 5 illustrates the mean DWFs and the peak factors of Eindhoven sewer system.

Table 5. The mean Dry Weather Flow (DWF) and related peak factors of Riool-Zuid, Eindhoven city and Nuenen/Son catchment areas (retrieved from Schilperoort, 2011).

		Riool-Zuid	Eindhoven Stad	Nuenen/Son	wwtp Eindhoven
mean dry weather flow	[m³/h]	1,840	2,330	360	4,530
peak factor: minimum	[-]	0.60	0.68	0.55	0.68
peak factor: maximum	[-]	1.27	1.20	1.34	1.19

The theoretical DWF reported in the same study (Schilperoort, 2011) are given in Table 6. As the table reveals that there is a 24% difference between the observed mean DWF and the theoretical DWF of the Riool-Zuid area.

Table 6. Observed mean and theoretical Dry Weather Flows of Riool-Zuid, Eindhoven city, Nuenen/Son catchment areas and Eindhoven Wastewater Treatment Plant (WWTP) (retrieved from Schilperoort, 2011). ([1]DWF: Dry weather flow, [2]p.e.: Population Equivalent)

	observed mean DWF[1] [m³/h]	p.e.[2] [#]	theoretical DWF [m³/h]
Riool-Zuid	1,840	257,000	1,390 (-24%)
Eindhoven Stad	2,330	268,000	1,450 (-38%)
Nuenen/Son	360	52,000	280 (-23%)
wwtp Eindhoven	4,530	577,000	3,130 (-31%)

The total DWF of the Riool-Zuid area is calculated as **959.52 m³/h** in this study, which deviates from both the observed mean DWF and the theoretical DWF values presented in the tables 5 and 6. This discrepancy can be attributed to several factors such as:

- There is a WWTP sludge processing installation in the Geldrop-Mierlo area which adds nearly 2,500 m³/day to the municipal sewer system (Schilperoort, 2011) which is not included in this study.
- The industrial wastewater is not included in the calculations which were reported as approximately 67,031 p.e. (Schilperoort, 2011).

However, when the contribution of the industries are taken into account, the total industrial water contribution of the Riool-Zuid is calculated as:

Table 7. The Dry Weather Flows (DWFs) of Riool-Zuid and the corresponding industrial water contributions.

Catchment Area	Q_{DWF} [m³/h]	Industrial water (p.e.)
Bergeijk	93.77	7,721
Eersel	104.32	9,010
Steensel		
Valkenswaard	156.07	7,230
Velthoven	227.50	6,330
Waalre	87.72	2,000
Aalst	11.11	
Geldrop (1/2)	70.92	30,250
Mierlo	58.48	
Heeze + Sterksel + Leende	78.73	4,490
Geldrop 1/2+Mierlo		
Total Q_{DWF}	**959.52**	**67,031**

In addition to that, when the contribution of the WWTP sludge processing installation in Geldrop-Mierlo is also included, the total wastewater in Riool-Zuid is found as given in Table 8:

Table 8. Total wastewater flow in Riool-Zuid.

Contributer	Q_{DWF} [m³/h]
Households of Riool-Zuid	959.52
WWTP Sludge processing	104.17
Industry water in Riool-Zuid	363.08
Total	**1426.77**

By including the industry water and WWTP sludge processing intallation contribution, the total flow in the Riool-Zuid is calculated as approximately **1,426 m³/h** which is only %2.6 more than the theoritical DWF given in Table 6 which is considered to be an acceptable variation.

2.3.3. Wastewater Characteristics of Eindhoven Sewer System

While the Dutch wastewater infrastructure in the Netherlands is considered to be well-developed for wastewater collection, transportation, and treatment, it is important to note that many surface waters in the Netherlands still encounter water quality issues that require further efforts to meet established surface water quality standards (Schilperoort, 2011). Even when effluent meets defined discharge requirements, water quality problems in receiving water bodies might remain. These issues may develop as a result of flow changes throughout time, resulting in the disturbance of aquatic ecosystems in surface waters.

The main pollutants that can cause water quality problems are Total Suspended Solids (TSS), Chemical Oxygen Demand (COD) and nutrients like Nitrogen (N) and Phosphate (P). The environmental impact of the micro pollutants should also not be underestimated. However, in this research, it has been taken out of context as the micro pollutant removal is another wise topic which is demanding a lot of time.

In order to give a clear overview, the TSS and COD concentrations of the wastewater collected from Riool-Zuid, Eindhoven city and Nuenen/Son catchment areas during the dry weather flows are shown in the table below;

For a comprehensive overview, the Table 9 presents the TSS and COD concentrations of wastewater collected from the catchment areas of Riool-Zuid, Eindhoven city, and Nuenen/Son during dry weather flows.

*Table 9 . Mean Dry Weather Flow (DWF) Wastewater Characteristics of Riool-Zuid, Eindhoven city, Nuenen/Son and total inflow of the Eindhoven Wastewater Treatment Plant (WWTP) (*TSSeq: Total Suspended Solid equavalent, CODeq: Chemical Oxygen Demand Equavalent, CODfeq: Filtered/soluable COD equavalent *) (retrieved from Schilperoort, 2011).*

		Riool-Zuid	Eindhoven Stad	Nuenen/Son	wwtp Eindhoven
Q	mean DWF [m^3/h]	1,840	2,330	360	4,530
	peak factors [-]	0.60 - 1.27	0.68 - 1.20	0.55 - 1.34	0.68 - 1.19
TSS_{eq}	mean DWF [kg/h]	432	691	69	1,191
	peak factors [-]	0.50 - 1.31	0.52 - 1.39	0.32 - 1.85	0.50 - 1.34
COD_{eq}	mean DWF [kg/h]	1,247	1,550	156	2,953
	peak factors [-]	0.55 - 1.28	0.58 - 1.33	0.45 - 1.51	0.58 - 1.27
$CODf_{eq}$	mean DWF [kg/h]	473	467	71	1,011
	peak factors [-]	0.59 - 1.25	0.64 - 1.25	0.53 - 1.34	0.61 - 1.21

However, it should be noted that in this study, some variations in the wastewater concentrations might occur, depending on the scenarios developed (e.g. higher concentrations due to rejected water received from the sewer mining technology, the adverse impacts of water extraction from the sewer system, lower velocities and higher retention time) Therefore, the possible fluctuations in wastewater characteristics will be also taken into account in this study and the method will be further discussed in the following chapters.

2.4. Impact Assessment of Sewer Mining on Riool-Zuid Sewer System

Wastewater extraction from a sewer system has undoubtedly consequences due to the fact that it changes the flows of the existing system which may lead to sulphur related problems such as corrosion and odour. In order to give an insight into the possible problems, this section elaborates the sulphur cycle, concrete corrosion and odour problems. The extend of these problems depend on the wastewater extraction ratios and the technology used for the sewer mining which determines the quantity and quality of the rejected water.

2.4.1. Sewer Processes and Sulphur Cycle

The sewer system processes are not only based on removal and transformation of organic substances, which are electron donors, but also based on the transformation of electron acceptors, which controls the redox conditions. Dissolved oxygen, nitrate and sulphate are regarded as significant electron acceptors in the sewer systems and are known to be the determinants of the process conditions. More specifically, these electron acceptors pave the way for aerobic, anoxic and/or anaerobic processes. As a result of transformation of these electrons, water, molecular nitrogen and hydrogen sulphide arise. When dissolved oxygen and nitrates are absent, anaerobic conditions develop. In these circumstances, the external electron acceptor is often sulphate, which results in the formation of hydrogen sulphide (Hvitved-Jacobsen et al., 2013).

Process Conditions	External Electron Acceptor	Typical Sewer System Characteristics
Aerobic	+ Oxygen	Partly filled gravity sewer Aerated pressure sewer
Anoxic	– Oxygen + Nitrate or nitrite	Pressure sewer with addition of nitrate
Anaerobic	– Oxygen – Nitrate and nitrite + Sulfate (+CO_2)	Pressure sewer Full-flowing gravity sewer Gravity sewer with low slope and deposits

Figure 11 . Electron acceptors and corresponding conditions for microbial redox processes (respiration processes) in sewer networks (retrieved from Hvitved-Jacobsen et al., 2013).

In sewer systems, pressure mains reflect the anaerobic condition. In addition to that, when local temperature increases, the gravity sewers with full flows can simulate anaerobic conditions as well. The anaerobic bioprocesses, such as sulphate reduction and fermentation, primarily take place in specific parts of the sewer such as biofilms and sediments which are illustrated in Fig.12 given below.

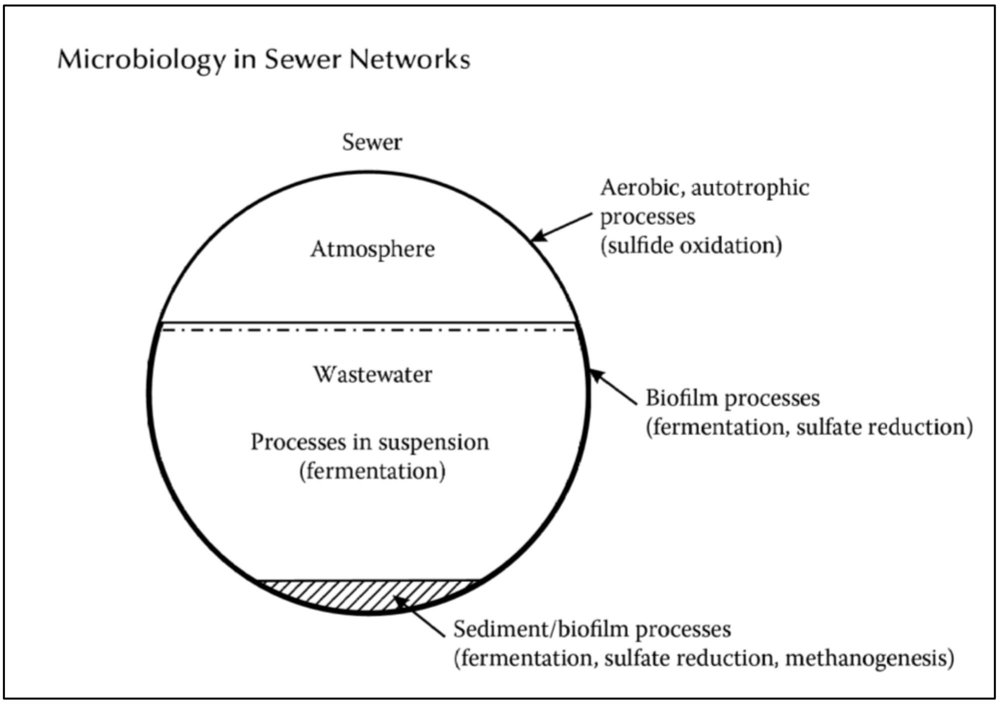


Figure 12 . Illustration of the major microbial processes in a gravity sewer under anaerobic conditions in the wastewater phase. Aerobic processes may take place at sewer walls exposed to the atmosphere. (retrieved from Hvitved-Jacobsen et al., 2013).

The sulphate reducing bacteria is capable to convert the sulphate into hydrogen sulphide with a conversion ratio of approximately 1:1 under anaerobic conditions when the sulphide production from sulphur containing proteins is neglected (Hvitved-Jacobsen et al., 2013). However, it should be highlighted that, according to Nielsen and Hvitved-Jacobsen 1988, the maximum rate of sulphide production in sewer systems by sulphate reducing bacteria occurs at a concentration of sulphate in the order of 3-5 g SO_4-S/m^3 depending on the site- and species-specific aspects. Which means, higher sulphate concentrations do not boost the sulphide production rate. This limitation can be explained by the rate limitation of sulphate diffusion into the biofilm on the sewer walls that is produced by sulphate reducing bacteria. In other words, the sulphate concentrations below 3-5 g SO_4-S/m^3 can cause rate limitation and above these concentrations a more or less constant rate can take place when there are no other external factors that can have an impact on the rate limitation (Hvitved-Jacobsen et al., 2013).

In terms of biofilm penetrations, there are two possibilities: the biofilm is either fully or partially penetrated, corresponding to either a fully process-effective or partially process-effective biofilm (illustrated in Fig.13). Sulphate concentrations are mostly greater than 5-15 g S/m^3 in all types of municipal wastewaters and due to complete penetration, these high concentrations do not restrict sulphide production in relatively thin biofilms (Hvitved-Jacobsen et al., 2013; Nielsen and Hvitved-Jacobsen 1988). When the sulphate is penetrated into deeper sediment layers in sewer sediments, the sulphate reduction potential may increase in case sulphate concentration in the bulk water phase increases.

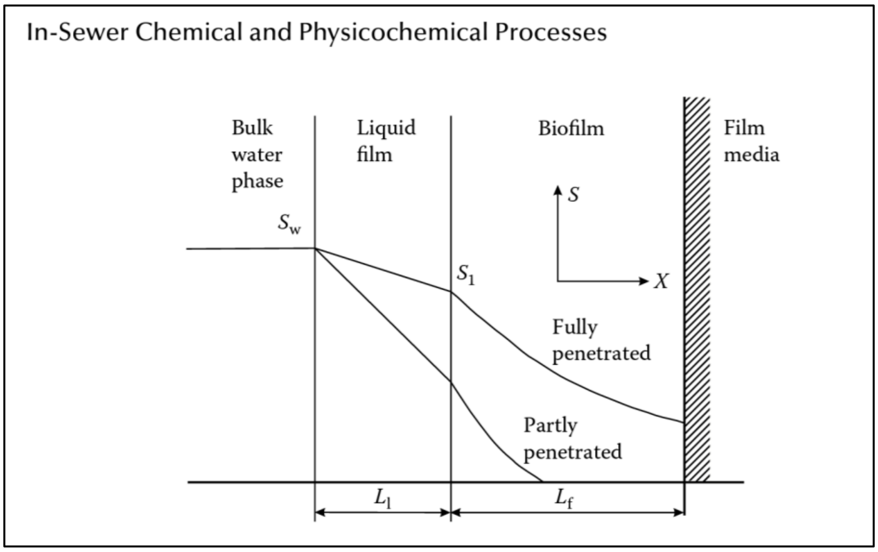


Figure 13: Substrate profiles for a biofilm with full and partial penetration of a single substrate, respectively ; S: Dissolved substance, X: Particulate substance, f: Biofilm, w: Water phase, L_l: Liquid film thickness, L_f: Biofilm thickness (retrieved from Hvitved-Jacobsen et al., 2013).

Due to the fact sulphate-reducing bacteria are slow growing microorganisms, they typically reside in the stationary components of the sewer systems like biofilms and sediments as they can be washed out when they enter the suspended water phase. Sulphate reduction can also take place in wastewater phase in case the biofilm is detached from the pipe surfaces and suspends (Rudelle et al., 2011; Hvitved-Jacobsen et al., 2013).

The main pathways of the sulphur cycle in the sewer systems are illustrated in Fig.14 and Fig.15. It should be highlighted that the extent of the formation of problematic components are determined by the transformation and transportation rates. The sulphide related problems arise when the hydrogen sulphide takes place in the sewer atmosphere. Hydrogen sulphide is a week acid and through chemical reactions with other substances like heavy metals and forms precipitates.

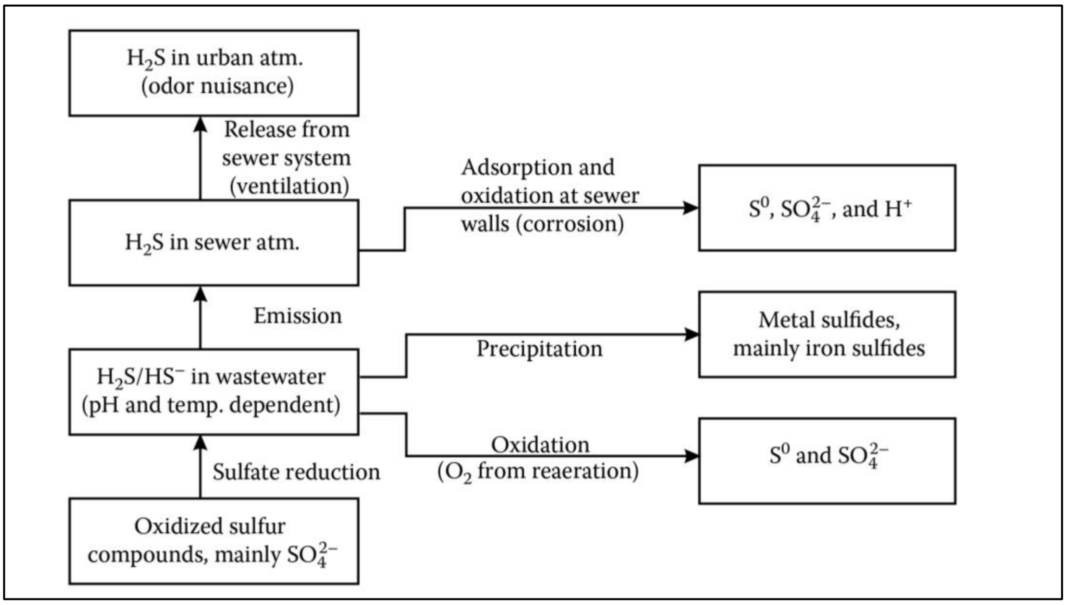


Figure 14: The main sulphur cycle transformation and transportation pathways in sewer networks (retrieved from Hvitved-Jacobsen et al., 2013).

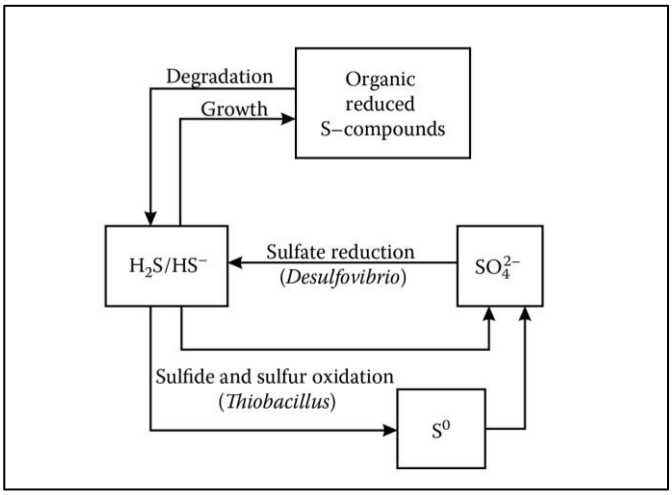


Figure 15: Biological sulphur cycle under aerobic and anaerobic conditions in sewer system (retrieved from Hvitved-Jacobsen et al., 2013).

Fig.15 not only includes the aerobic oxidation of sulphur but also the anaerobic processes in which sulphur is reduced. In an aerobic environment, biomass growth and degradation are the most important factors. Because anaerobic sulphate-reducing bacteria (mostly *Desulfovibrio* and *Desulfotomaculum*) grow slowly and can be flushed out of the sewer system when they enter the water phase. They may, however, survive in biofilm (sewage slimes) and sewer sediments (deposits). As a result, sulphate reduction occurs predominantly in the biofilm and sediments. However, detached (anaerobic) biofilm may produce a modest quantity of sulphide in the wastewater phase, generally less than 10% of the overall amount (Hvitved-Jacobsen et al., 2013). Furthermore, as it can be observed from Fig.16 that the production of sulphide is primarily produced through sulphate reduction and the breakdown of the sulphur containing organic materials. However, the contribution of the sulphate reduction process to sulphate production in sewer systems is quantitively greater than the degradation of sulphur containing organic material.

Fig. 16 illustrates the extended version of the sulphur related processes that are precisely associated with aerobic transformation whereas anaerobic sulphide production is not depicted. More specifically, Fig.16 is a representation of an aerobic gravity sewer environment that receives anaerobic wastewater from a pressure main with sulphide production. Therefore, it can be concluded that the presence of sulphide and VOCs in a gravity sewer is basically a consequence of a transmission from an upstream location. As a result, this example can be regarded as an indication that the impacts of the sewer processes can be observed at different locations of the sewer system from where the sewer processes originally proceed (Hvitved-Jacobsen et al., 2013; Nielsen et al. 2008).

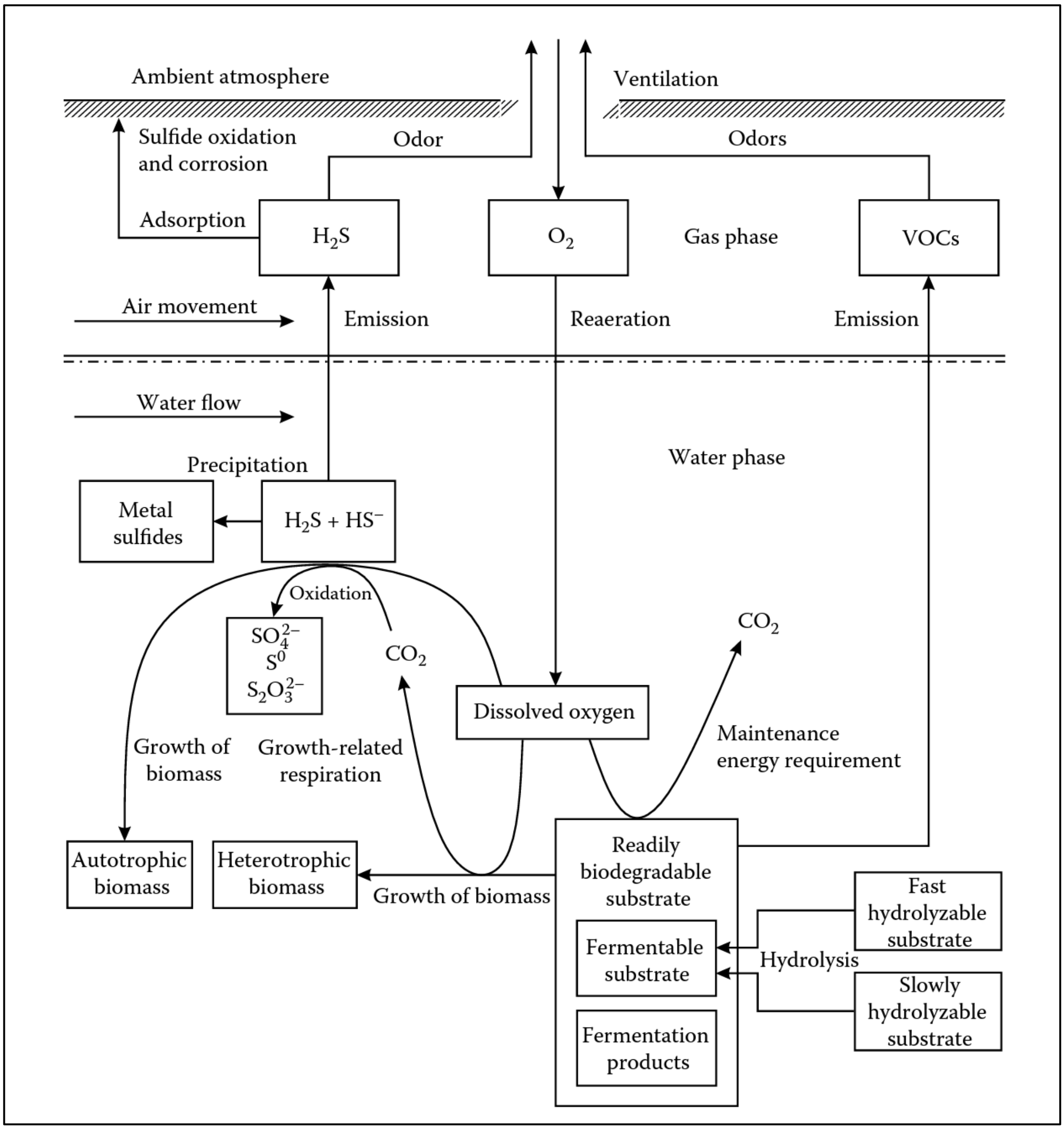


Figure 16: An elaborated concept for aerobic, heterotrophic transformations of wastewater organic matter in a sewer including aerobic processes of sulfur cycle and sewer gas phase (retrieved from Hvitved-Jacobsen et al., 2013).

In both anaerobic and aerobic conditions, the sulphur cycle takes place such as in the biofilm, the sewer deposits, the water phase and at the air-solid surfaces. The biomass growth and degradation are relevant in an aerobic environment (Hvitved-Jacobsen et al., 2013). The anaerobic sulphate reducing bacteria are slow growing and in case they remain in the biofilm and/or sewer sediments, they can remain in the system. However, as aforementioned once they enter the water phase, they can be washed out. Therefore, the sulphate reduction predominantly occurs in the biofilm and sediments. Nonetheless, the detachment of biofilm can still result in sulphide production in wastewater waste. However, the contribution is less than 10% of the total sulphide production. The oxidation of sulphide to sulphate, elemental sulphur and thiosulphate can take place biologically by autotrophic bacteria and by chemical & biological processes which is considered to be the most significant. (Nielsen et al. 2006; Hvitved-Jacobsen et al., 2013). To sum up, the existence of sulphide in aerobic water phase of sewer can be as a result of:

- Sulphide diffusion from the anaerobic sewer biofilm or the sediments into the aerobic part of the biofilm and potentially into the water phase.
- Transportation of sulphide produced from the upstream of the sewer network to the aerobic part of the sewer network. For example, this situation can be observed when the sulphide production takes places in the pressure main located in the upstream of a gravity sewer system.

As aforementioned, the sulphide oxidation can arise from both biological and chemical processes (Wilmot et al. 1988; Nielsen et al. 2003). The biological sulphide oxidation processes are carried out most likely by chemoautotrophic bacteria as the sulphide oxidation releases energy that helps the bacteria to exploit carbon for growth (Hvitved-Jacobsen et al., 2013). The major products and intermediates that have been identified by sulphide oxidation are elemental sulphur (S^0), thiosulfate ($S_2O_3^{2-}$), sulphide (SO_3^{2-}) sulphate (SO_4^{2-}). The corresponding reactions of the oxidation stoichiometry are as follow (Hvitved-Jacobsen et al., 2013):

$$2HS^- + O_2 \rightarrow 2S^0 + 2OH^-$$

(2.1)

$$2HS^- + 2O_2 \rightarrow S_2O_3^{2-} + H_2O$$

(2.2)

$$2HS^- + 3O_2 \rightarrow 2SO_3^{2-} + 2H^+$$

(2.3)

$$2HS^- + 4O_2 \rightarrow 2SO_4^{2-} + 2H^+$$

(2.4)

Stoichiometric studies of aerobic sulphide oxidation in wastewater systems were carried out under sewer network-relevant conditions including oxidation at relatively low DO concentrations (Nielsen et al. 2003; Hvitved-Jacobsen et al., 2013). The sulphide oxidation stoichiometry can be quantified using the reaction coefficient (Rc) described below:

$$Rc = \Delta S^{2-}/\Delta O_2$$

(2.5)

where:

Rc = reaction coefficient (mol S / mol O_2)
ΔS^{2-} = transformed amount of dissolved sulphide (mol S)
ΔO_2 = consumed amount of dissolved oxygen (DO) (mol O_2)

The reaction coefficient is therefore the ratio between the dissolved sulphide concentration and dissolved oxygen concentration required and represents either the biological sulphide oxidation or chemical sulphide oxidation. The reaction coefficient for chemical sulphide oxidation varies between 0.8-0.9 mol S / mol O_2 while for biological sulphide oxidation it is mol 2 S / mol O_2(Hvitved-Jacobsen et al., 2013).

Even though the kinetics of sulphide oxidation is complex, the biological oxidation does not include the biomass. The sulphide oxidation is highly dependent on the pH, temperature and the catalysis. The study of (Nielsen et al. 2003, 2006, 2008) have performed the kinetics of sulphide oxidation in specific conditions that represents the sewer network conditions. According to the result of these studies, sulphide oxidation is significant even at low DO concentrations (Hvitved-Jacobsen et al., 2013).

Biofilm and sediments are known to be the places where the sulphate reduction primarily takes place. In gravity sewer systems, the biofilm thickness is commonly more than 1-2 mm in which the surface is considered to be aerobic and the deeper layers as anaerobic. Depending on the DO concentration in the wastewater and the degradability of the organic material present, the aerobic surface layer of the biofilm is in the order of 0.5 mm. The sulphide production generally takes place in the anaerobic segment of the biofilm and diffuses into the anaerobic part. The existence of sulphide and DO oxygen in the aerobic part creates a convenient environment for sulphide oxidizing bacteria and leads to internal sulphur cycle which is illustrated in Fig.17. When the sulphide production takes place in the deep part of the gravity sewer's biofilm, the oxidation process can occur either in the aerobic upper layer of the biofilm or in the water phase. The interaction between these two processes is illustrated in Fig 17.

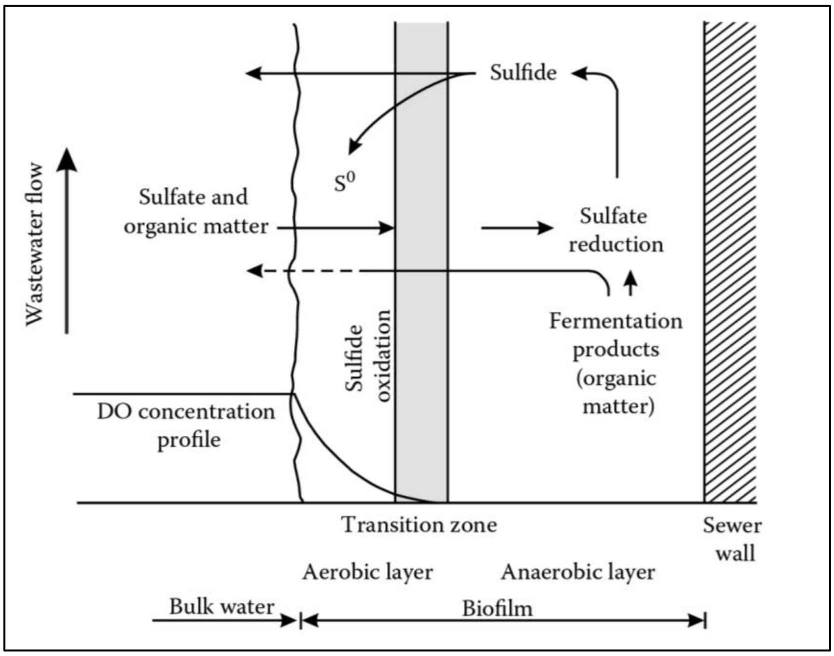


Figure 17: Aerobic and anaerobic process interactions in a gravity sewer biofilm illustrating the internal sulfur cycle. (retrieved from Hvitved-Jacobsen et al., 2013).

On the other hand, the sulphide produced in the pressure mains located in the upstream of the sewer system can diffuse into the biofilm and result in going through oxidation process. Moreover, studies about the stoichiometry of sulphide oxidation in biofilms have revealed that the main product is elemental sulphur and sulphate and thiosulphate are either not formed or found at relatively low concentrations (Hvitved-Jacobsen et al., 2013; Nielsen et al. 2005). Taking the results of these investigations into account, the sulphide oxidation in wastewater phase is most likely dominated by biological processes and therefore, chemical sulphide oxidation can be neglected.

The production of sulphide processes has two types that are the reduction of sulphate and degradation of sulphur containing organic matter (illustrated in Fig.16). In regards to the production quantity, only sulphate reduction is important. However, under anaerobic conditions, degradation of sulphur containing organic matter with certain types of protein form odorous volatile sulphur compounds like mercaptans (Hvitved-Jacobsen et al., 2013).

Sulphide production takes place under anaerobic conditions in which DO, nitrate or other oxidized inorganic nitrogen compounds are absent. Sulphides are often not found in bulk water at DO values greater than 1 mg/L (Carrera et al., 2016). **Anaerobic conditions** arise typically in **full-flowing gravity sewers and pressure mains**. When the aerobic wastewater with moderate DO concentrations enters these parts, the DO is quickly depleted, often after 5-30 minutes. This depletion time is dependent on both the DO concentration level and the aerobic respiration rate of the wastewater. Even though the sulphide production strictly needs anaerobic conditions, sulphide can be transported to and temporarily exist in the part of the sewer system where the DO is not zero (Hvitved-Jacobsen et al., 2013). The Fig.18 is also a great example for this condition, in which, the sulphide production takes place in the inner part of the sewer biofilm and then transported to the upper aerobic layer or into the bulk water phase. Besides, it should be highlighted that the biomass production under anaerobic conditions is relatively low, typically with a yield constant, Y of 0.05–0.1 g COD / g COD (Hvitved-Jacobsen et al., 2013)

The sulphide problems in sewer networks mostly occurs in case of high temperatures. However, these problems may also arise in pressure mains during winter under mild climate temperatures, for example, when the wastewater temperature is around 5°C to 12°C (Hvitved-Jacobsen et al. 1995; Nielsen et al. 1998; Hvitved-Jacobsen et al., 2013). The sulphide production at low temperatures is typically low and the sulphide production is considered to be important when the anaerobic **residence time exceeds 0.5 to 2h** (Hvitved-Jacobsen et al., 2013).

The residence time depends on the hydraulic conditions of the sewer which are determined by the type of the sewer system. The combined sewer systems are affected by the weather conditions as the combined sewer systems received the rain water as well. Therefore, the hydraulic conditions vary during dry weather flows and wet weather flows. On the other hand, the separate sewer system flows can fluctuate during the day depending on the time as during night less inflow is received and during the day time more wastewater will be loaded to the sewer system. But in any case, sewer sediments are expected to be formed and resulted in biofilm formations.

The biofilm thickness and anaerobic residence time are both affect the biological activity of the sulphate reducing bacteria. When the wastewater velocity is less than 0.8 m/s, the biofilms tend to form thicker and a greater deposition of sediments take place. On the contrary, when the velocities are higher the biofilm is thinner due to the shear forces resulting in less resistance to mass transfer (Carrera et al., 2016). The sulphide build-up is considered significant when the anaerobic time typically exceeds 0.5- 2h (Hvitved-Jacobsen, 2002).

Even though the sulphide related problems cannot be assessed with a simple and general criterion, some certain parameters can still give an indication. For example, sulphide is considered to be not commonly occur in bulk water phase of a gravity sewer system when the DO concentrations are above >1 g O_2 m^{-3} (Norsker et al., 1995; Hvitved-Jacobsen et al., 2013; Carrera et al., 2016). In addition to that sulphide problems generally do not arise when the DO concentrations are above 0.2 - 0.5 g O_2 m^{-3}(Hvitved-Jacobsen et al., 2013). However, it should be considered that the sulphate related problems depend on the balance between various wastewater quality parameters where the severity of the problems is determined by the sewer system design and the local climate. To sum up, the sulphide formation is influenced by the present sulphate concentrations, quality and quantity of biodegradable organic matter, local temperature, pH, the ratio of area/volume of the sewer pipes, wastewater velocity and anaerobic residence time.

Another factor that impacts the sulphide oxidation rate is the pH which is commonly an important factor for biological processes. The pH is highly related to the distribution of H_2S and HS^- which is illustrated by Fig 18.

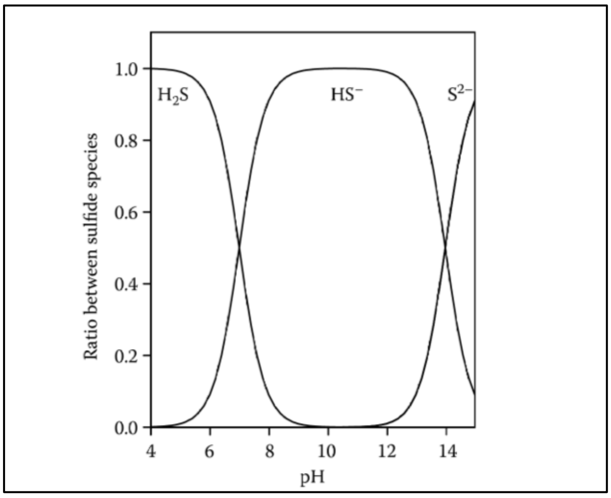


Figure 18: C–pH diagram for $H_2S(aq)$ ⇔ HS^- + H + and HS^- ⇔ $S2^-$ + H^+ (retrieved from Hvitved-Jacobsen et al., 2013).

It should be highlighted that pH has the ability to control the hydrogen sulphide emission to the sewer atmosphere and therefore, can control the corrosion and odour problems as well. In addition to that, in order to control the sulphide in sewer systems, iron salts are also known to be effective.

As aforementioned, the biofilm plays a crucial role in sulphide production in the pressure mains. In gravity mains, even though the water phase conditions are dependent on the reaeration effect and sulphate reduction rate that determines the conditions to be either aerobic or anaerobic. However, the deeper part of the biofilm is typically anaerobic. When the sulphide production takes place in de deeper part of the biofilm in gravity mains, the oxidation step can occur either in the anaerobic upper layer of the biofilm or in the water phase (illustrated in Fig.17). Even though the final product of the oxidation process is sulphate, elemental sulphur is generally stable and leads the stoichiometry and kinetics of the transformation. The oxidation processes of hydrogen sulphide, on the other hand, is known to be responsible for the concrete corrosion. This oxidation process takes places on the sewer walls once the hydrogen sulphide is released to the sewer atmosphere.

2.4.2. Sulphide build-up problems in sewer systems

Sulphate reducing bacteria produces in anaerobic zones of the sewer system which can either take place in the pressure mains or in the stagnant parts of the system. In the gravity sewers, the anaerobic conditions can lead to H_2S emissions into the sewer atmosphere or precipitations in the liquid phase. Sulphide build-up in sewer systems can cause significant issues that are; (I) corrosion of sewer structures, (II) odour problems and (III) health impacts on sewer workers (US EPA 1974; Nielsen et al. 1992; Carrera et al., 2016). This section elaborates the major sulphate related problems in the sewer networks.

2.4.2.1. Corrosion of sewer structures

Concrete corrosion is expensive for communities since it demands more frequent rehabilitation and pipe replacement (Sydney et al.1996; Carrera et al., 2016) and hydrogen sulphide is known to be the primary source for concrete corrosion in sewer pipes.

When the hydrogen sulphide is formed under anaerobic conditions and is exposed to aerobic conditions, it can oxidize to sulphuric acid (H_2SO_4). Sulphuric acid has a high potential to react with the alkaline cement in the concrete material and deteriorate the compound bonds of the concrete pipes. Concrete corrosion is closely associated with the hydrogen sulphide formation and emission to the sewer atmosphere. Which means, when the sulphide stays in the water phase, no corrosion problem related to sulphide will arise as the concrete corrosion is a consequence of hydrogen sulphide existence in the gas phase, followed by the adsorption process in the liquid film of the sewer pipe's concrete surface (Hvitved-Jacobsen et al., 2013).

The condensate layer of H_2S reacts with the oxygen in the sewer atmosphere and by microbial reactions and oxidizes to sulphuric acid (H_2SO_4) that is a highly corrosive component that interacts with the binder in the concrete pipe and forms ettringite and gypsum (Roberts et al.2002; Hvitved-Jacobsen et al., 2013).

The following expression (Hvitved-Jacobsen et al., 2013) represents the aforementioned chemical reaction:

$$2H_2S + 2O_2 \rightarrow H_2SO_4$$

(2.6)

The corrosion reaction between H_2SO_4 and alkaline cement of the concrete pipes can be expressed simply with the following equation (Hvitved-Jacobsen et al., 2013):

$$H_2SO_4 + CaCO_3(cement) \rightarrow H_2O + CO_2 + CaSO_4(gypsum)$$

(2.7)

Fig.19 illustrates the principles of concrete corrosions in the sewer pipes to provide an apparent insight into corrosion process in the sewer systems:

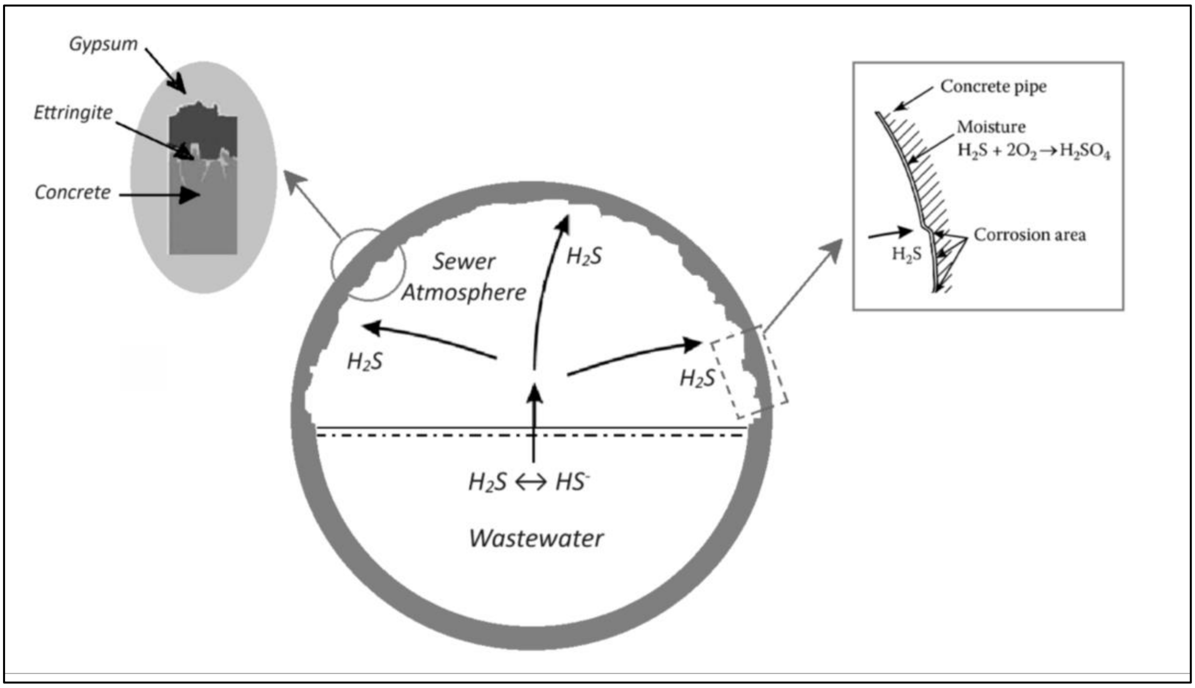


Figure 19: Principles of concrete corrosion in a sewer pipe (adapted from Roberts et al.2002; Hvitved-Jacobsen et al., 2013).

Since hydrogen sulphide is very insoluble in water, it condenses quickly around the sewer crown and produces a condensate layer on the concrete surface. However, it should be noted that, there is a significant variance in corrosion rates over the length of the sewer pipe line. Yet, the spatial variation is not consistent. In a number of cases, a much greater corrosion rate was recorded at the sewer's crown (Vincke et al. 2001; Wells et al., 2009), while other investigations discovered more prominent corrosion just above the wastewater level (Mori et al., 1992; Wells et al., 2009). Corrosion rate can be also affected by concrete mixture design parameters (Wells et al., 2009) like cement and coarse aggregate content, and the water and cement ratio. For instance, higher cement and lower water/cement ratios resulted in higher degrees of deterioration (Hewayde et al., 2007) and concretes with reduced porosity and permeability likely to have lower corrosion rates in sewer pipes (Islander et al. 1991).

Even though the H_2S oxidation can be simply represented by the Equation 2.6., it is suggested that various chemical pathways involving sulphur compounds with thiosulphate and elemental sulphur as possible intermediates take place in corroding concrete (Parker 1945; Islander et al., 1991; Jensen et al., 2009a; Hvitved-Jacobsen et al., 2013). Experimental evidence supports the hypothesis put forth by Jensen et al. (2009a) that elemental sulphur can degrade into two fractions: a rapid and likely amorphous form, S^0_{fast}, and a slowly degradable form, S^0_{slow}. The presence of slowly biodegradable fraction may cause delayed corrosion (Hvitved-Jacobsen et al., 2013).

Aerobic bacteria that oxidize hydrogen sulphide to sulfuric acid are members of the aerobic and autotrophic *Thiobacillus* family (Parker 1945; Milde et al. 1983; Sand 1987; Hvitved-Jacobsen et al., 2013). Many of these bacteria are active at low pH levels and have the capacity to produce sulfuric acid solutions up to 7%. *Acidithiobacillus thiooxidans*, used to be known as *Thiobacillus thiooxidans* and *Thiobacillus concretivorus*, is the most prevalent bacterial species on severely corroded concrete surfaces. The activation of this bacteria requires a pH level between 0.5 and 5, therefore, is also dependent on the other species of the *Thiobacillus* family that are capable of decreasing the pH levels. *A. thiooxidans* are not only capable of using the sulphide but also can use both thiosulphate and elemental sulphur as energy sources (Hvitved-Jacobsen et al., 2013).

A corrosion rate of **4-5 mm per year** has been found in situations of severe concrete corrosion (Mori et al. 1991) and highly related to the hydrogen sulphide adsorption rate and the alkalinity of the concrete material. Hvitved-Jacobsen et al. (2013) suggests a correction factor for the prediction of concrete corrosion rate and highlights that the concrete corrosion is a complicated phenomenon influenced by several process-related variables as well as sewage network design. Even though a helpful empirical equation was given for forecasting a sewage pipe's yearly corrosion depth, it is stated the suggested empirical equation fails to reflect dynamic corrosion events in larger sewage networks that combine gravity sewers, pressurized systems, and pumping stations, for instance. Additionally, some unforeseen short-term alterations and consequences may also arise and impact the concrete corrosion rate.

As aforementioned, the hydrogen sulphide in de sewer air is caused by the emission of H_2S that is afterwards adsorbed by the liquid layer on the inner surface of the sewer pipe. The H_2S adsorption rate is relatively higher on the concrete surface than the emission rate from the water phase, resulting in low H_2S concentrations in the sewer atmosphere. Therefore, when the pipe material is not corroding, for example, plastic pipes, the H_2S concentration in the sewer atmosphere is relatively higher than the sewer air of the concrete pipes with corrosion (Nielsen et al. 2008). As a result, even if the concentration of H_2S in the sewer air is modest, it is critical that a high corrosion rate is attainable (Hvitved-Jacobsen et al., 2013).

2.4.2.2. Odour Problems

The majority of compounds causing odors are produced in anaerobic environments. Known causes of odour issues include the development of both hydrogen sulphide (H_2S) and volatile organic molecules (VOCs). As previously stated, odour problems are generally caused by anaerobic conditions, and when sewer systems are taken into account, pressure mains reflect the anaerobic conditions. Furthermore, when exposed to high temperatures, gravity sewers with full flows can also simulate anaerobic conditions (Hvitved-Jacobsen et al., 2013).

In manholes, gravity sewers, and wet-wells of pumping stations, hydrogen sulphide is likely to be released from the water phase and contaminate the atmosphere. The locals, especially in metropolitan areas, may complain about this gas because of its distinctive rotten egg smell. Additionally, the presence of H_2S may be linked to a number of issues with human health (Carrera et al., 2016. Fig.20 illustrates an outline of central interfacial related exchange and reactions of volatile compounds in a sewer atmosphere (concrete pipe).

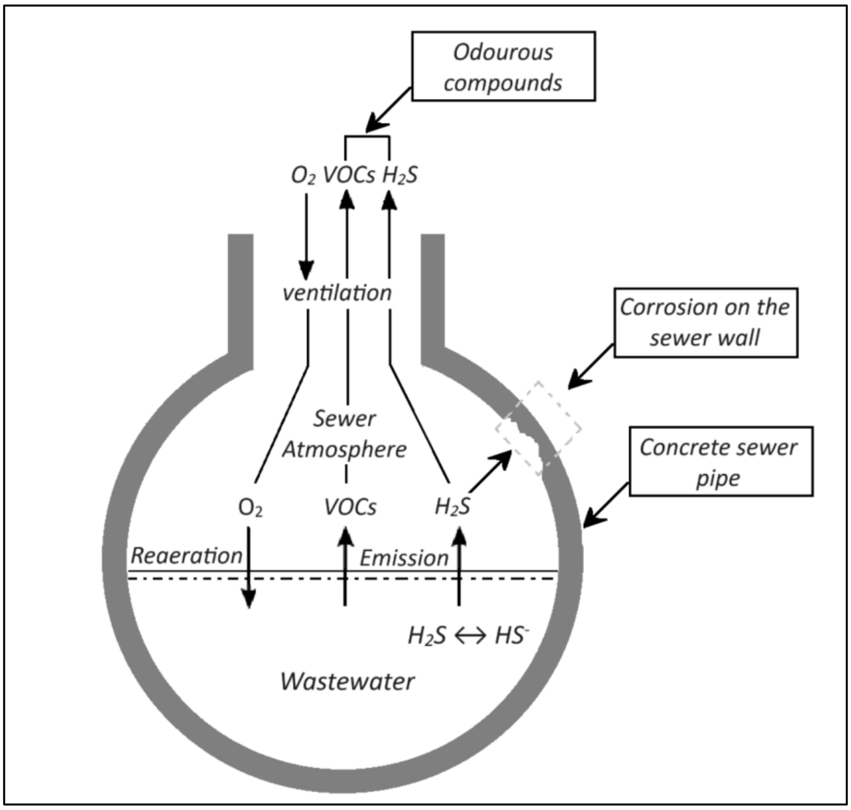


Figure 20: Illustration of central interfacial related exchange and reactions of volatile compounds in the sewer atmosphere of a concrete pipe (adapted from Hvitved-Jacobsen et al., 2013).

It should be also mentioned that, the key contributor to maintaining a high redox potential is the presence of dissolved oxygen in wastewater from sewage networks. The air-water oxygen transfer process called ''reaeration'' (illustrated by Fig.20) is in reality the only means to give oxygen to the water phase in the sewer systems. Ventilation can be either natural or forced. Natural aeration can take place in case the manholes have small openings or through other types of leaks in the sewer system. Forced ventilation on the other hand, is carried out by blowers or fans. The potential for aerobic and anaerobic microbial activities, and subsequently the transformation and removal of wastewater compounds, is determined by the degree of reaeration relative to the microbial removal of DO in the water phase.

It has not been possible to establish a direct link between an odorous compound's molecular structure and how it is perceived due to the complexity of odour detection. However, it is now apparent that odour detection by humans is an objective phenomenon, and it is typical for most people to be able to tell the difference between a pleasant and an unpleasant odour.

Odour detection can be carried out either by analytical or by sensory measurements and the odour description is based on its concentration, intensity, character ad persistency. Sensory measurements are conducted by making use of either the human nose or electronic detectors and thereby correlate with the impact of the odour (Sneath & Clarkson 2000; Stuetz et al. 2000; Hvitved-Jacobsen et al., 2013). Although sensory measures are helpful, analytically based measurements are frequently chosen in modelling (Hvitved-Jacobsen et al., 2013).

The quantity of hydrogen sulphide in the sewer atmosphere or surrounding environment can be regarded as a direct and more accurate indicator of odour issues. However, it is suggested to initially estimate the hydrogen sulphide in the wastewater phase of the sewer system to indicate its potential risk. Yet, it should be highlighted that the ratio of hydrogen sulphide concentrations in the sewer atmosphere and water phase might not be constant due to nonequilibrium circumstances a number of sinks for hydrogen sulphide in the atmosphere. The first noticeable **rotten egg smell** of the hydrogen sulphide is reported to be **0.01 - 1.5 ppm** which regarded as the ''odour threshold''. When the hydrogen sulphide concentration is between **3-5 ppm**, its odour becomes stronger and more unpleasant. The odour of hydrogen sulphide concentrations **exceeding 30 ppm** are described as sweet or sickeningly sweet (OSHA, 2023). It is stated that hydrogen sulphide loses its **characteristic odour** roughly at **50 ppm**, which means the possibility for immediate detection drastically decreases. Because it is often undetectable by odour above 50 ppm, it can be life-threatening for the sewer workers. Therefore, sensors and warning systems for its detection must be in constant operation when working in sewer systems. In this regard, it is crucial to note that H_2S is frequently observed to build in places where gas flow and ventilation are limited, such as pumping stations and manholes (Hvitved-Jacobsen et al., 2013).

2.4.2.3. Health Impacts

Hydrogen sulphide release into the sewer atmosphere has severe health effects including immediate death depending on its concentrations and exposure time. At sewer drops, pumping stations and hydraulic jumps, the hydrogen sulphide emissions into the sewer atmosphere are expected to be more severe.

In order to give an insight into the hydrogen sulphide concentrations at sewer atmosphere, Hvitved-Jacobsen et al., 2013, put forwards the following example; at a **pH of 7**, the atmospheric partial pressure of H_2S in equilibrium with a water phase of sulphide (H2S + HS-) is about equivalent to **130 ppm** (g S /m^3) $^{-1}$ which can cause eye irritation, drowsiness, respiratory problems and might even result in death when the exposure time exceeds 48 hours (OSHA, 2023).

The severity of the symptoms and health impact of hydrogen sulphide not only depends on its concentrations in the atmosphere but also depends on exposure time. Table 10 summarizes human health related effects of hydrogen sulphide in the atmosphere based on its concentrations in the air [ppm] and exposure times.

Table 10 . Health hazards of Hydrogen Sulphide (H_2S) based on its concentrations in the air [ppm] and Severity Category and Health and Safety Impact description for the related concentrations for up to 5 minutes of H_2S exposure (adapted from OSHA, 2023; Bertelsmann et al., 2019).

H_2S Concentration in Atmosphere (ppm)	**Symtomps / Health Impact**	**Severity Category**	**Health and Safety Impact Description**
		for up to 5 min H_2S Exposure	
0.01-1.5	Odor threshold	No health and safety concequences	Very low / None
2-5	**Prolonged exposure:** Nausea, watery eyes, headaches or loss of sleep. Possible airway problems to asthma patients.	No health and safety concequences	Very low / None
20	Fatigue risk, loss of appetite, headache, irritability, poor memory, dizziness.	First aid case	Low
50-100	**1 hour of exposure:** Mild conjunctivitis, airway irritation. Possible digestive discomfort and loss of appetite.	OSHA recordable incident	Moderate
100	**2-15 minutes of exposure:** Coughing, eye irritation, loss of smell **15-30 minutes of exposure:** Altered breathing and drowsiness. **1 hour of exposure:** Throat irritation. **>48 hours of exposure:** Death risk.	OSHA recordable incident	Moderate
100-150	Loss of smell.	OSHA recordable incident	Moderate
200-300	**1 hour of exposure:** Severe conjunctivitis and airway discomfort. **Prolonged exposure:** Pulmonary edema risk.	OSHA recordable incident	Moderate
500-700	**5 minutes of exposure:** Staggering and collapse. **30 minutes of exposure:** Serious eye damage. **30-60 minutes of exposure:** Death.	Injury resulting in hospitalization or permanent disability	High
700-1000	**1-2 breaths:** Immediate unconsciousness, knockdown or instantaneous collapse. **Within minutes:** Respiration failure and death.	Fatality	Very High
1000-2000	**Nearly instant** death.	Fatality	Very High

2.5. Mitigation measures to reduce the adverse effects of hydrogen sulphide

This section elaborates the possible mitigation measures to tackle the possible adverse effects of the hydrogen sulphide (H_2S) in order to cope with its corrosion, odour and health risks as previously elaborated and discussed in the Section 2.4. In order to choose an appropriate mitigation method for hydrogen sulphide, it is crucial to identify where in the system and when the control of sulphide is feasible. Fig. 10 can be useful to determine for identifying strategic points in the sulphur pathways where the implementation of sulphide control measures is applicable. The visual representation in Fig. 10 delineates distinct junctures in the sulphur pathways, providing potential starting points for mitigation methods. These methods may involve actions such as closure, reduction, or modification of hydrogen sulphide generation or transfer processes (Hvitved-Jacobsen et al., 2013).

The mitigation measures can be categorized into three primary groups. The first approach involves inhibiting or reducing sulphide formation by attenuating the general biological activity of the sulphate-reducing biomass. This can be achieved through methods such as pH increase, biofilm removal, the application of biocides, or adding oxygen or nitrate into the sewer system. The second mitigation measure focuses on reducing the generated sulphide, accomplished by introducing electron acceptors such as oxygen and nitrate. Alternatively, it involves initiating the chemical oxidation of sulphide by introducing chemical oxidants or inducing sulphide precipitation through the addition of iron salts. The third mitigation strategy centres on addressing the potential occurrence of hydrogen sulphide in the gas phase. This can be achieved by either reducing its concentration through an increase in the wastewater's pH to minimize the molecular and volatile form of H_2S or by diluting it through the application of forced ventilation. All these strategies fall under the category of process-related controls (Hvitved-Jacobsen et al., 2013).

The mitigation of sewer gas-related issues should ideally be integrated into the planning and design phase of sewer networks. However, practical experience indicates that this is frequently deferred until problems manifest and necessitate remedial action. Control methodologies can be classified based on the type of control procedures employed, adding a practical dimension to the more systematic approach of process mitigation (Hvitved-Jacobsen et al., 2013):

- **Design procedures for active control of sulphide problems:** Typically implemented during the design phase of the sewer network, these procedures aim to diminish the conditions conducive to the formation of sulphide and VOCs.
- **Design procedures for passive control of sulphide problems:** Included in the design phase, these procedures focus on reducing the impact of sewer gases without necessarily altering their formation.
- **Operational procedures for the control of sewer gas problems:** Implemented after the identification of a problem, these procedures aim to decreases the formation or associated effects through the use of mitigation methods.

This study focuses on the Riool-Zuid sewer system which has been already built, therefore, the focus will be on applying the mitigation measures to operational procedures. Interventions such as air dosing, pure oxygen infusion, or nitrate addition might be explored to prevent the formation of anaerobic conditions or their associated effects. These techniques raise the redox potential, inhibiting anaerobic activities such as sulphide production. However, it should be noted that adding oxygen and nitrate to wastewater may boost the biological activities especially in case of warm climate conditions and might not be a cost-effective solution.

When the sulphide is already generated, a feasible mitigation measure that can be taken into account is the chemical precipitation of the sulphide. The precipitation of sulphide is commonly achieved using metal salts such as ferrous iron (iron II) and ferric iron (iron III). Despite the complex kinetics involved in the reaction between sulphide and a mixture of iron II and iron III, the practical application of iron salts for sulphide control is known to be efficient, straightforward, and cost-effective. Notably, optimal results are typically achieved when introducing iron salts upstream of sewer sections experiencing anaerobic conditions, allowing sufficient residence time for the relatively slow autotrophic microbial process. This process involves the oxidation of sulphide to elemental sulphur by ferric iron. However, it is crucial to acknowledge that the precipitation process is pH-dependent. Iron II sulphide (FeS) exhibits limited efficiency in precipitation at pH values below approximately 8. Effective sulphide control using iron salts may necessitate pH adjustment by introducing a base, such as $Ca (OH)_2$. Furthermore, it is noteworthy that Fe (III) demonstrates a higher reaction rate, rendering it more efficient than Fe (II) (Hvitved-Jacobsen et al., 2013).

Another practical method involves the chemical oxidation of sulphide, achieved by introducing chlorine compounds (e.g., NaOCl), hydrogen peroxide (H_2O_2), ozone, and permanganate. Both NaOCl and H_2O_2 are taken into consideration in this study. However, the required dosages were massive and therefore are excluded from this study. This will be further discussed in Section 3.3.

In conclusion, this study focuses on the possible adverse effects of sewer mining technology in Riool-Zuid sewer system, which is already constructed. Therefore, the emphasize is on the application of mitigation measures to operational procedures. Specifically, chemical dosing to the Riool-Zuid sewer system, involving Iron (II), Iron (III), and NO3-N, is discussed in further detail in Section 3.3.

2.6. Data Processing – Storm Water Management Model (SWMM)

The data required for Mega-WATS (e.g., pipe characteristics, IDs and coordinates of each pipe and node) is retrieved from Storm Water Management Model (SWMM). However, some adjustments were applied to the invert levels of various pipes as unlike the SWMM modelling, the Mega-WATS is more sensitive to the order of upstream and downstream invert levels. In addition to that, various pipes were still resulting in negative slopes, therefore, the invert levels of these pipes were changed using reasonable assumptions based on the previous and next pipes' invert levels. Fig. 21 illustrates the upstream and downstream contributors of the Riool-Zuid part of the Eindhoven Wastewater System.

Fig. 21 is composed of overmany nodes and pipes. Since the principal focus is on the main pipeline, the data of the each catchment area is simplified and regarded as a single simple contributer connected to the main pipeline. Therefore, only the connection pipes that carries the wastewater from the catchments to the main pipeline are taken into account with the total wastewater amount of each catchment area. However, as it can be also seen from Fig 21. that, unlike the other catchments, Mierlo is not directly connected to the main pipeline. In fact, it is connected to the pipeline of the Geldrop. In other words, the domestic wastewater of the Mierlo is intermingling with the domestic wastewater of Geldrop before being transported to the main pipeline. Fig. 21 further demonstrates that Geldrop is connected to the main pipeline from two different locations. Therefore, another assumption made in this project is connecting one-half of the Geldrop catchment to the sub-connection point. The other half is assumed to enter the main pipeline from the other connection point just after mixing with Mierlo's household wastewater.

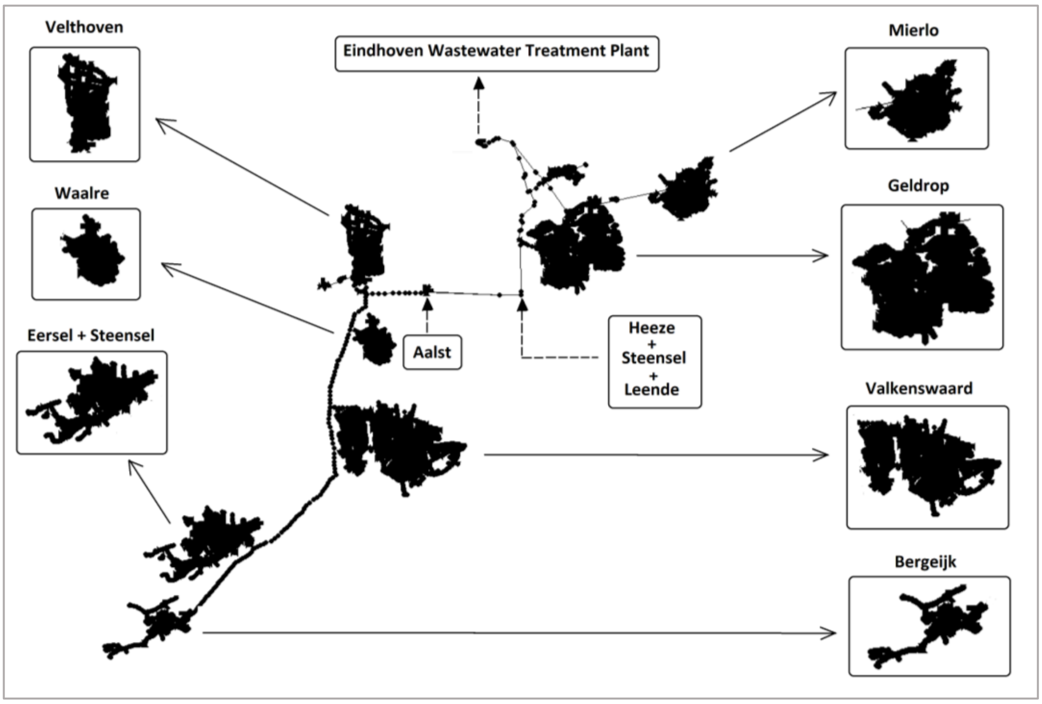


Figure 21. Riool-Zuid wastewater system contributors in SWMM.

The data retrieved and adjusted through SWIMM model is afterwards trasmitted to Mega-WATS together with the wastewater flows and relevant concentrations of each contributers.

2.7. Research Scenarios

The domestic wastewater flow from the Riool-Zuid catchments to Eindhoven WWTP is approximately **960 m³/h** during dry weather periods, according to the calculations given in Section 2.2.2. In this thesis study, the contribution of the industrial wastewater is neglected. It should be noted that 960 m³/h represents the **100%** of domestic wastewater flow available for sewer mining application. However, the full-extraction of wastewater should be avoided to keep the Riool-Zuid sewer system operating properly and reliably during summer time.

In order to assess the sewer mining technology impact on the dissolved sulphide concentrations in the Riool-Zuid sewer system, Mega-WATS model is chosen as a simulation tool (c.f. Section 2.1.). The simulations are done using a variety of scenarios created based on the wastewater extraction percentages and the location of the sewer mining installation.

The Aalst pumping station is located at Riool-Zuid and is connected to two pressure mains. The rest of the Riool-Zuid sewer system is composed of gravity pipes. Pressure main conditions play a crucial role in dissolved sulphide concentrations. Therefore, the scenarios are also based on locating the sewer mining technology **at upstream and downstream of the pressure main** in order to observe how the dissolved sulphide concentrations are affected by the sewer mining application and to what extent when the wastewater is exposed to a pressurized environment.

The first scenario created is installing the sewer mining technology **in Aalst**, **the second scenario in de Meren** which are both located at the **upstream of the pressure main** and **the third scenario downstream of the pressure main**. Within these scenario, four sub-scenarios were applied for each in order to see how the extracted wastewater amount can impact the dissolved sulphide concentrations in the sewer system. Therefore, the Mega-WATS modelling simulations are carried out with **40%, 60%, 80%** and **90%** for each scenario. The scenarios created are given in Table 11.

Table 11 . Scenarios created based on the location of the Sewer Mining Technology installation and domestic wastewater extraction ratios.

Scenario	Extraction
Scenario 1. Sewer Mining in Aalst	No Extraction
	40%
	60%
	80%
	90%
Scenario 2. Sewer Mining in de Meren	No Extraction
	40%
	60%
	80%
	90%
Scenario 3. Sewer Mining at downstream of the Pressure Main	No Extraction
	40%
	60%
	80%
	90%

Each scenario includes the full-flow conditions as well, in which no domestic wastewater is extracted from the Riool-Zuid sewer system and is called as ''**no extraction**'' scenario. Comparing the sub-scenarios with no extraction conditions enables a better understanding into the extent of the extraction ratio's impact on the sewer system.

Chapter 3. Results

The results section is composed of four sub-sections that are Mega-WATS results, required chemical dosages, sewer mining implementation results and cost analysis. The Mega-WATS results section illustrates and elaborates the simulation results of the modelling based on dissolved sulphide concentrations of each scenario. Required chemical dosages section gives an insight into the required Fe (II), Fe (III) and NO_3-N concentrations for the mitigation of the increased dissolved sulphide concentrations in the Riool-Zuid sewer system which is a consequence of wastewater extraction. Sewer mining implementation results covers the water quantities that can feed the River Dommel base flow during dry periods in summer. And the last part, cost analysis, gives an insight into CAPEX and OPEX costs of the sewer mining technology.

3.1. Mega-WATS Results

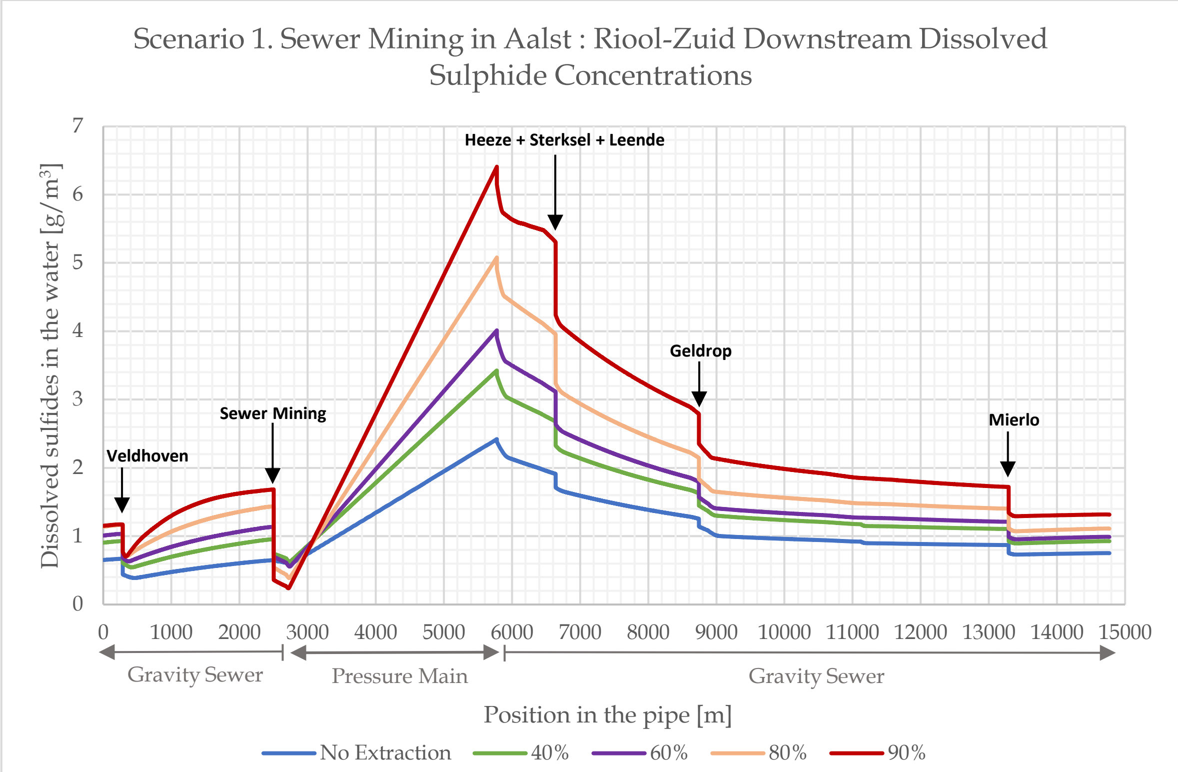
As aforementioned in section '' 2.2. Research method used: 'Mega-WATS Modelling'', this section reveals an insight into the possible sulphide-related problems. The primary emphasis is understanding the sulphur cycle in Riool-Zuid sewer system. Therefore, this section is based on dissolved sulphide concentrations to provide an insight into the transformation of H_2S and H_2SO_4 and hence into the corrosion and odour problems in Riool-Zuid that may arise as a result of the implementation of sewer mining technology.

The dissolved sulphides in sewage are composed of three species that are S^{2-}, HS^- and H_2S (Zhang et al., 2022). Sulphide concentrations are classified as **'minor'** if they are less than **0.5 g S/m³**, '**medium**' if they are between **0.5** and **2 g S/m³**, and 'considerable' if they are greater than **2 g S/m³** (Hvitved-Jacobsen et al., 2013). On the other hand, in terms of problems commonly reported, sulphide concentrations of **0.5**, **3**, and **10 g S/m³** are considered **low**, **moderate**, and **high**, respectively. However, it should be emphasized that a classification of "minor" or "medium" hazard should not be interpreted as meaning "no need for control" (Hvitved-Jacobsen et al., 2013). **Within this study, the dissolved sulphide concentration threshold for the Mega-WATS simulation** is set to **2.5 g S/m³** for the investigation of the sulphide related problems in Riool-Zuid.

The Mega-WATS modelling simulation results of dissolved sulphide concentrations given in this section are based on each scenario and sub-scenario which were explained explicitly in section 2.7. Research Scenarios.

3.1.1. Scenario 1.: Sewer Mining in Aalst

This section includes the Mega-WATS Modelling results of sulphide concentrations when the sewer mining technology is placed and applied in Aalst which is upstream of the pressure main of the Riool-Zuid part of the Eindhoven sewer system. The assumption of sewer mining implementation in Aalst is based on the idea that the corresponding catchment areas' wastewater is extracted with various extraction ratios and the rejected water is directed back to the rest of the sewer system. The dissolved sulphide concentrations in Riool-Zuid of this scenario are represented in Graph 1.



Graph 1. Dissolved sulphide concentrations of Riool Zuid downstream pipeline after implementing the Sewer Mining Technology in Aalst for no extraction, 40%, 60%, 80% and 90% domestic wastewater extraction ratios.

Graph 1. Illustrates that the significant elevation in dissolved sulfide concentrations occurs within the pressure main as a result of anaerobic conditions and the dissolved sulphide concentration patterns are all the same for each situation. The highest rise in dissolved sulphide concentrations is obtained with 90% water extraction, whereas the lowest increase is obtained with the full flow condition, which includes no water extraction.

Dissolved sulphide concentrations exhibit noticeable declines at additional wastewater inflow points. These inflow points are the locations that are connected to the catchment areas. However, it should be highlighted that the first sharp decline is also related to the pipe conditions as it corresponds to the location where the wastewater flows from the pressure main into the gravity pipe. The second sharp decline is observed where the Heeze, Sterksel and Leende catchments are connected to the system. The 3rd and 4th sharp declines are related to the catchments of Geldrop and Mierlo, respectively.

The simulation results are in line with the literature studies, discussed in Section 2.4.1, related to sulphur cycles in sewer systems. First of all, a higher water extraction ratio leads to increased water concentration and reduced wastewater velocities within the pipes. This condition results in greater exposure of microorganisms to substrate and an extended hydraulic residence time. It's noteworthy that hydraulic residence time is significantly influenced by wastewater velocity, given that all other pipe parameters remain constant across each scenario and sub-scenario. In addition to that, as aforementioned, the pressure main represents anaerobic conditions, therefore, the highest increases were expected to be observed in the pressure main. The dissolved sulphide concentrations show a downwards pattern in the gravity pipes, which is reasonable due to the aerobic conditions in the gravity pipes. Under aerobic conditions the transformation of dissolved sulphides takes place as discussed in Section 2.4.1 and Section 2.4.2.

Fig. 22 further illustrates the Mega-WATS simulation results for dissolved sulphide concentrations exceeding 2.5 mg/L in Riool Zuid after implementing the Sewer Mining Technology in Aalst with a wastewater extraction of 90%.

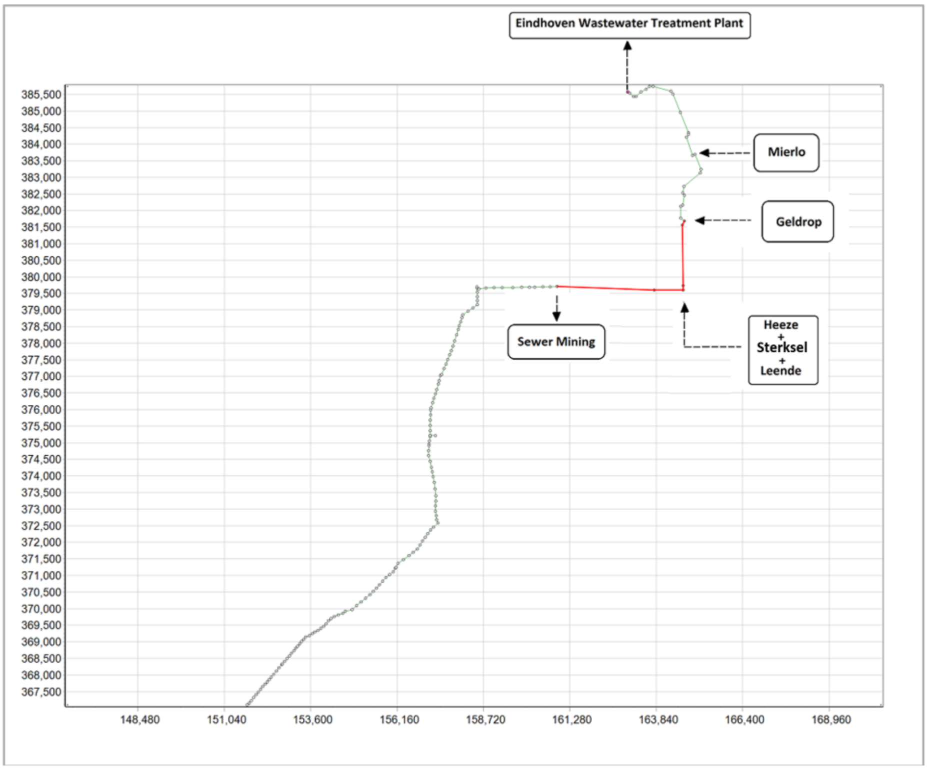
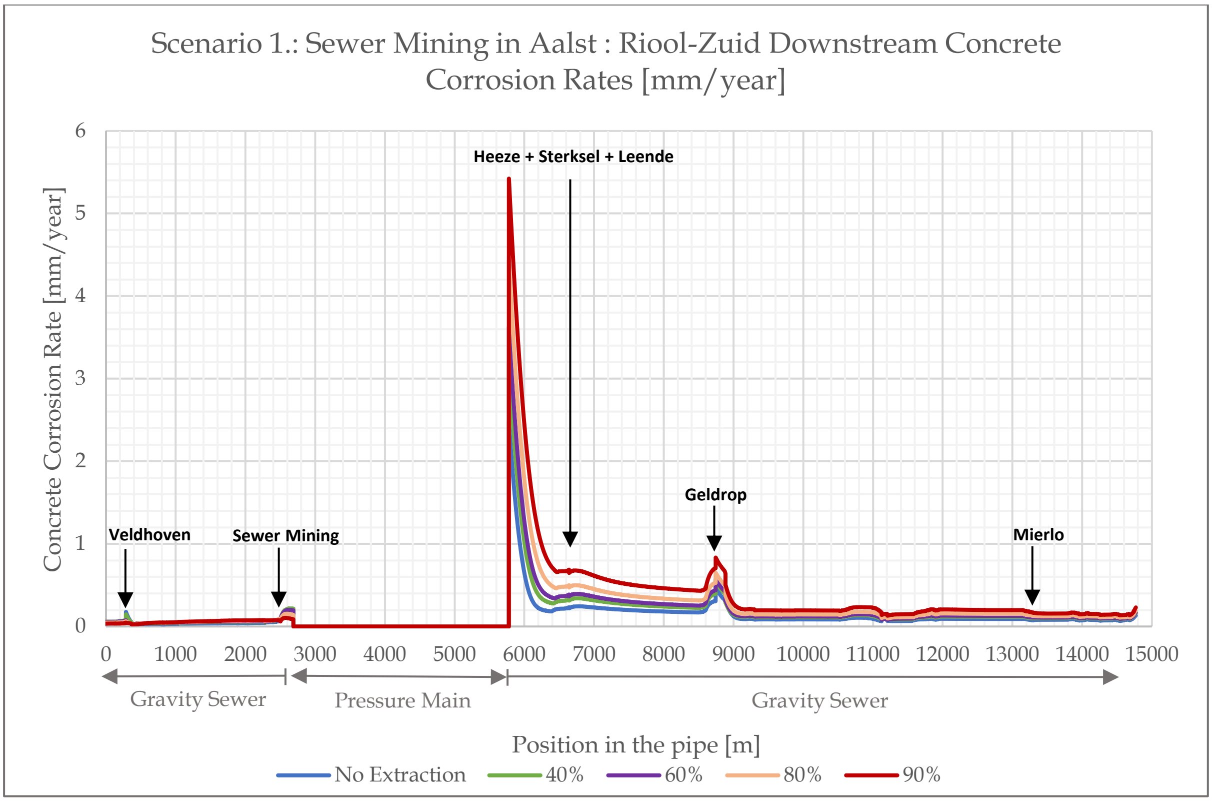


Figure 22. Dissolved sulphide concentrations exceeding 2.5 mg/L (highlighted with red lines) in Riool Zuid after implementing the Sewer Mining Technology in Aalst with a wastewater extraction of 90%.

Graph 1. and Fig.22 show that as the dissolved sulphide concentration increases in the pressure main, a greater number of downstream pipes exhibit concentrations exceeding 2.5 mg/L. As discussed in Section 3.2.1.1, the downstream pipes may still face the risk of corrosion and odour issues, even when the dissolved concentrations are below 2.5 mg/L. However, the extent of these problems depends on various factors explained in Section 2.4.2.1 and Section 2.4.2.2.

Graph 2 provides a clearer understanding of the corrosion issues arising from wastewater extraction in Riool-Zuid based on 40%, 60%, 80%, 90% wastewater extraction ratios and under no extraction conditions.



Graph 2. Concrete corrosion rate [mm/year] of Riool Zuid downstream pipeline after implementing the Sewer Mining Technology in Aalst for no extraction, 40%, 60%, 80% and 90% domestic wastewater extraction ratios.

As previously discussed in Section 2.4.2.1, in instances characterized by severe concrete corrosion, studies have reported a corrosion rate ranging from **4 to 5 mm per year** (Mori et al., 1991; Hvitved-Jacobsen et al., 2013). In addition to that, concrete corrosion rate, such as **2 to 3 mm per year** on a pipe surface, is frequently observed, leading to the deterioration of a sewer within a few years of operation. Therefore, it is crucial to possess comprehensive knowledge for managing issues in existing sewers and for designing new sewer networks with a reduced risk of future sulphide attacks on constructions (Vincke et al. 2000; Vollertsen et al. 2008; Jensen et al. 2009b, 2011; Nielsen et al. 2012; Hvitved-Jacobsen et al., 2013).

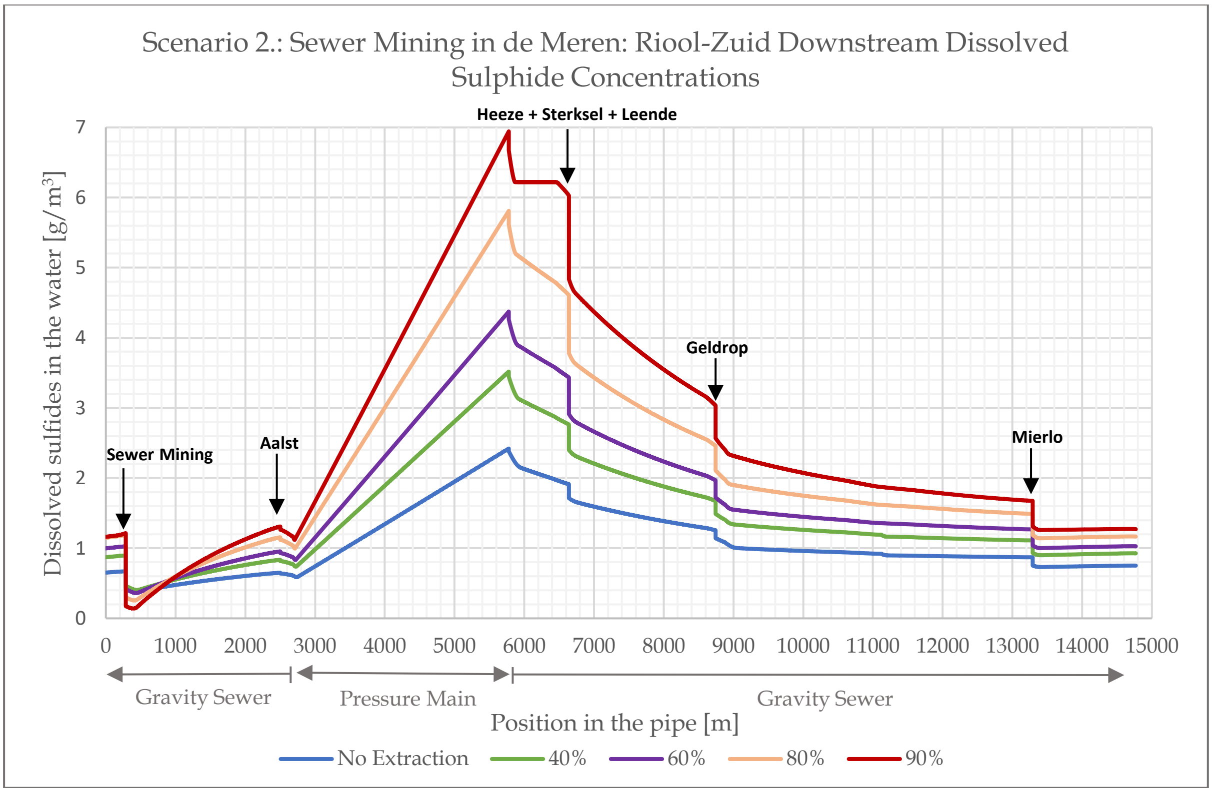
Taking the Mega-WATS simulation results into account, it can be concluded that the highest corrosion rate is observed with the 90% wastewater extraction from Riool-Zuid sewer system. Graph 2. clearly illustrates that the highest corrosion rate, **5.42 mm/year**, takes place in the downstream gravity pipe from the pressure main in Riool-Zuid sewer system. According to the literature study as discussed in the previous paragraph, this corrosion rate tends to result in severe concrete corrosion. In the **40%**, **60%**, and **80%** sub-scenarios, the same gravity pipe exhibits the highest corrosion rates of **3.19 mm/year**, **3.62 mm/year** and **4.43 mm/year**, respectively. Hence, it is imperative to implement precautionary measures to mitigate corrosion problems, as detailed in Section 3.3.

The corrosion rates in the remaining sections of the downstream pipeline for the 40%, 60%, 80%, and 90% sub-scenarios exhibit a range of approximately 0.48-0.09 mm/year, 0.54-0.1 mm/year, 0.65-0.11 mm/year and 0.84-0.14 mm/year, respectively.

3.1.2. Scenario 2.: Sewer Mining in de Meren

This section presents the Mega-WATS Modeling results for sulphide concentrations following the implementation of sewer mining technology in de Meren. The control station of de Meren has an existing infrastructure with a constructed building and integrating the sewer mining technology within the same framework may mitigate the additional construction costs. For this location, only the relevant contributions are taken into account. In other words, only the domestic wastewater originating from Bergeijk, Eersel, Steensel, Valkenswaard, Veldhoven, and Waalre is extracted from Riool-Zuid and directed to the sewer mining technology based on various extraction ratios. The rejected water from the FO&RO hybrid system is subsequently directed back into Riool-Zuid downstream, where it converges with the domestic wastewater from Aalst, and afterwards integrates with the wastewater streams from Heeze, Sterksel, Leende, Geldrop, and Mierlo. It should be noted again that this study focuses on the wastewater coming from the households of the contributing areas where the industrial wastewater contribution is neglected.

Graph 3. illustrates the dissolved sulphide concentrations in the downstream of the Riool-Zuid sewer system for 40%, 60%, 80% and 90% domestic wastewater extraction in de Meren and for no extraction conditions.



Graph 3. Dissolved sulphide concentrations of Riool Zuid downstream pipeline after implementing the Sewer Mining Technology in de Meren for no extraction, 40%, 60%, 80% and 90% domestic wastewater extraction ratios.

The outcomes of this scenario exhibit analogous trends to those observed in Scenario 1.: sewer mining implementation in Aalst based practical assumption. As depicted in Graph 3, there is a positive correlation between the quantity of water extracted from the system and the concentrations of dissolved sulphides in the wastewater phase. Furthermore, notable shifts in trend occur at consistent locations along the primary pipeline, mirroring the patterns observed in Scenario 1.. The noticeable decline corresponds to the juncture where wastewater transitions from the pressure main to the gravity main. Subsequent noticeable declines are consistently observed at points where water from Heeze, Sterksel and Leende, Geldrop, and Mierlo domestic wastewater enter the sewer system.

Figure 23 illustrates the outcomes of Mega-WATS simulations concerning dissolved sulphide concentrations surpassing 2.5 mg/L in Riool Zuid after implementing the Sewer Mining Technology in de Meren with a wastewater extraction of 90%.

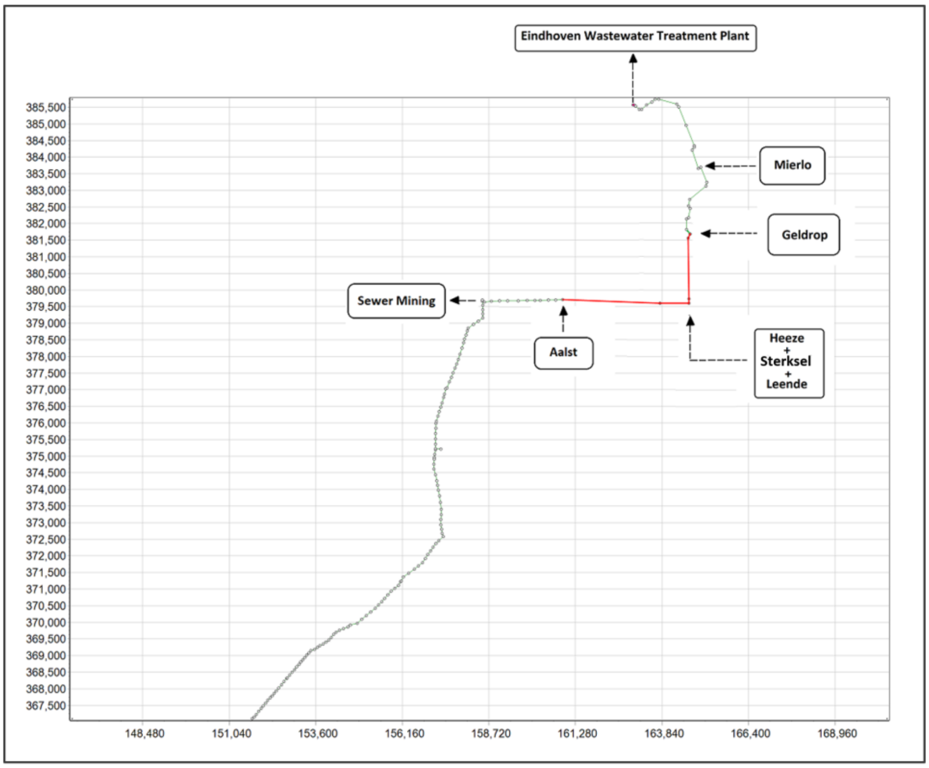
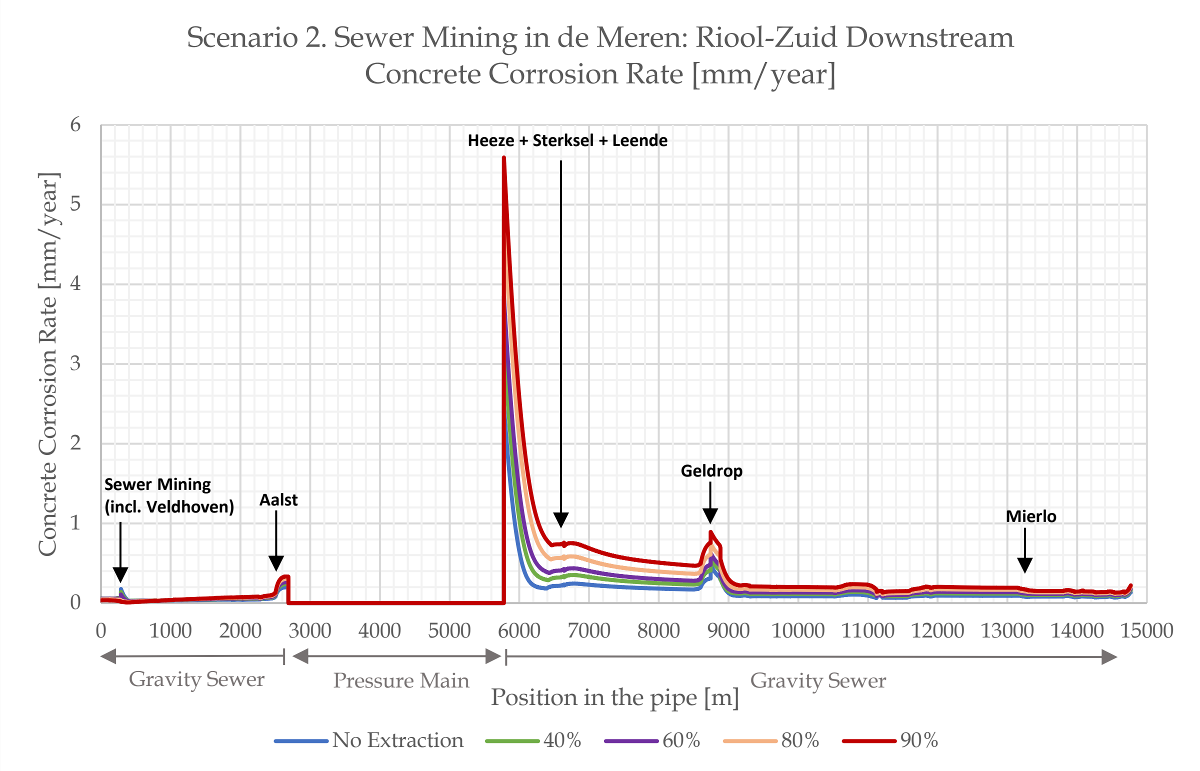


Figure 23. Dissolved sulphide concentrations exceeding 2.5 mg/L (highlighted with red lines) in Riool Zuid after implementing the Sewer Mining Technology in de Meren with a wastewater extraction of 90%.

The dissolved sulphide concentrations exceeding 2.5 mg/L show in general a similar pattern to the Scenario 1. results and are observed in the same pipes. However, the observed dissolved sulphide concentrations in the current scenario (Scenario 2: Sewer Mining in de Meren) exhibit a slight elevation in comparison to those observed and elaborated in Scenario 1.: Sewer Mining in Aalst. This can be explained by the increased concentrations in the upstream. In other words, when the sewer mining technology is implemented in de Meren, concentrated water will flow through a longer distance in the gravity pipes and be more exposed to aerobic conditions before entering the pressure main.

To enhance comprehension of the corrosion challenges associated with wastewater extraction in the Riool-Zuid sewer system, a dedicated graphical representation based on each sub-scenario is presented in Graph 4.



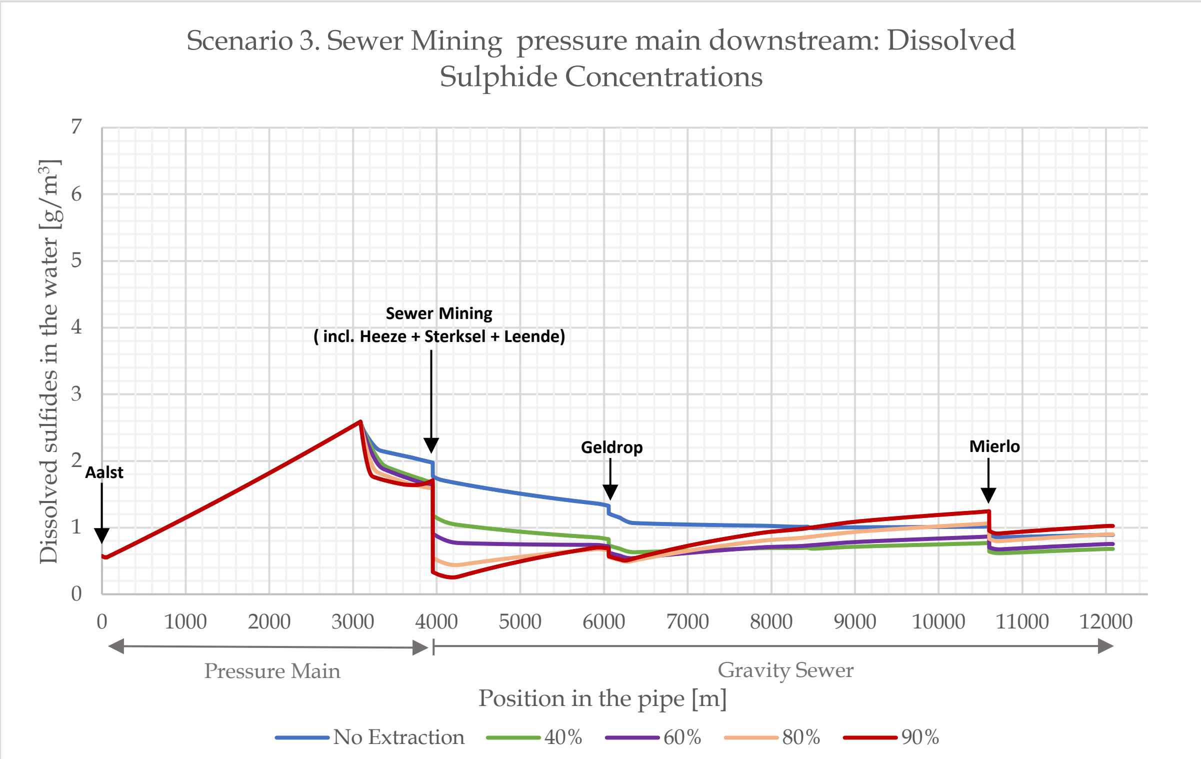
Graph 4. Concrete corrosion rate [mm/year] of Riool Zuid downstream pipeline after implementing the Sewer Mining Technology in de Meren for no extraction, 40%, 60%, 80% and 90% domestic wastewater extraction ratios.

According to the Mega-WATS concrete corrosion rates depicted in Graph 4, the highest corrosion rate occurs with 90% wastewater extraction from the Riool-Zuid sewer system, specifically in the downstream gravity pipe of the pressure main, consistent with the findings in Scenario 1.. However, there is a slight overall increase in corrosion rates when implementing the sewer mining technology in de Meren. More precisely, the highest concrete corrosion rates for 40%, 60%, 80%, and 90% wastewater extraction from the Riool-Zuid sewer system are found as **3.24 mm/year**, **3.85 mm/year**, **4.85 mm/year** and **5.59 mm/year**, respectively. As discussed earlier, these corrosion rates pose a risk of severe deterioration of a sewer pipe within a few years of operation. Therefore, it is imperative to consider precautionary measures, which will be expounded upon in Section 3.3.

The corrosion rates in the remaining sections of the downstream pipeline for the 40%, 60%, 80%, and 90% sub-scenarios exhibit a range of approximately 0.56-0.09 mm/year, 0.62-0.1 mm/year, 0.71-0.11 mm/year and 0.88-0.13 mm/year, respectively.

3.2.3. Scenario 3.: Sewer Mining at downstream of the Pressure Main

This section provides the WATS Modelling results on sulphide concentrations following the implementation of sewer mining technology downstream of the pressure main, specifically at the juncture where the domestic wastewater from Heeze, Sterksel, and Leende enters the Riool-Zuid sewer system. The dissolved sulphide concentrations are illustrated in Graph 5.



Graph 5. Dissolved sulphide concentrations of Riool Zuid downstream pipeline after implementing the Sewer Mining Technology at downstream of the pressure main (juncture of Heeze, Sterksel and Leende) for no extraction, 40%, 60%, 80% and 90% domestic wastewater extraction ratios.

In contrast to Scenario 1. and Scenario 2., the application of sewer mining technology downstream of the pressure main in this scenario (Scenario 3) effectively mitigated significant elevations in dissolved sulphide concentrations. This outcome is rationalized by the fact that concentrated water (as a result of water extraction and concentrate water of the FO&RO hybrid system), which would otherwise undergo anaerobic conditions resulting in dramatic increases in sulphide concentrations in the wastewater phase, is not exposed to such conditions. This mitigation may consequently reduce the risk of corrosion in the downstream section of the Riool-Zuid. Notably, the upstream pipes and pressure main experience full-flow conditions (no extraction), given that wastewater extraction occurs at downstream of the pressure main.

Fig. 24 demonstrates that the dissolved sulphide concentration surpasses 2.5 mg/L in a singular pipe, specifically the downstream gravity pipe from the pressure main. While the pressure main exhibits a dissolved sulphide concentration slightly above the threshold, a notable and rapid decrease is evident upon entry into the gravity pipe, as highlighted in Graph 5.

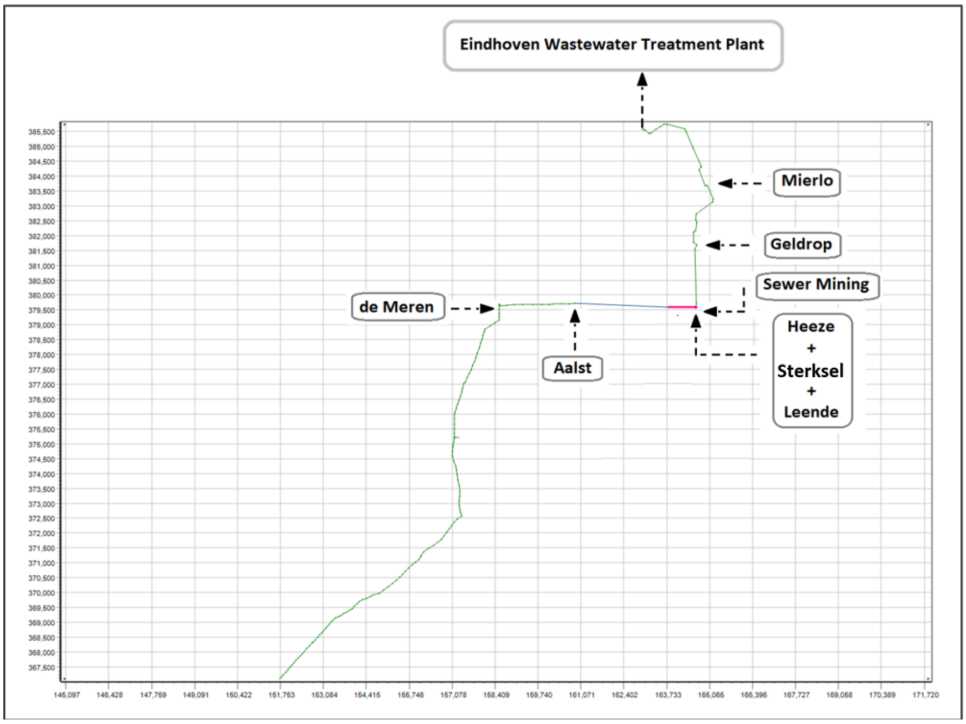
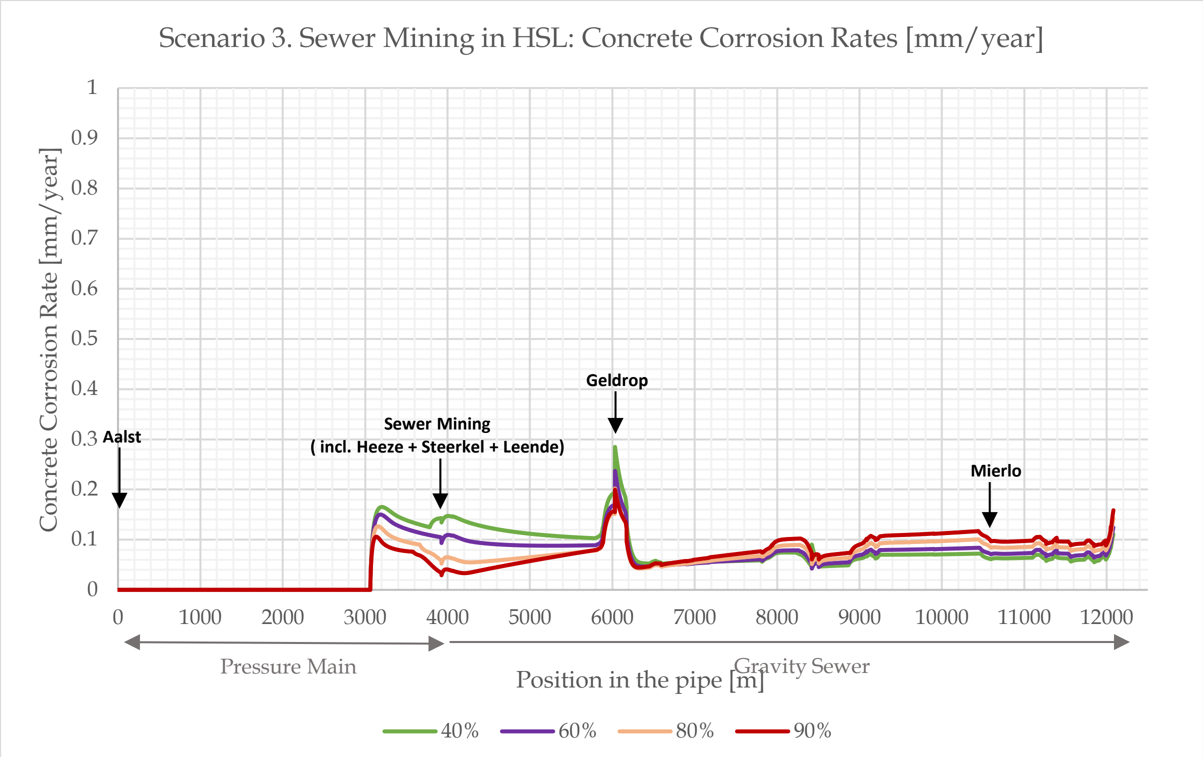


Figure 24. Dissolved sulphide concentrations exceeding 2.5 mg/L (highlighted with red lines) in Riool Zuid after implementing the Sewer Mining Technology at downstream of the pressure main (juncture of Heeze, Sterksel and Leende domestic wastewater inflow) with a wastewater extraction of 40%, 60%, 80% and 90%.

Graph 6 provides an understanding into potential corrosion issues in the downstream pipes of the Riool-Zuid sewer system.



Graph 6. Concrete corrosion rate [mm/year] of Riool Zuid downstream pipeline after implementing the Sewer Mining Technology at downstream of the pressure main (juncture of Heeze, Sterksel and Leende domestic wastewater inflow) with a wastewater extraction of 40%, 60%, 80% and 90%.

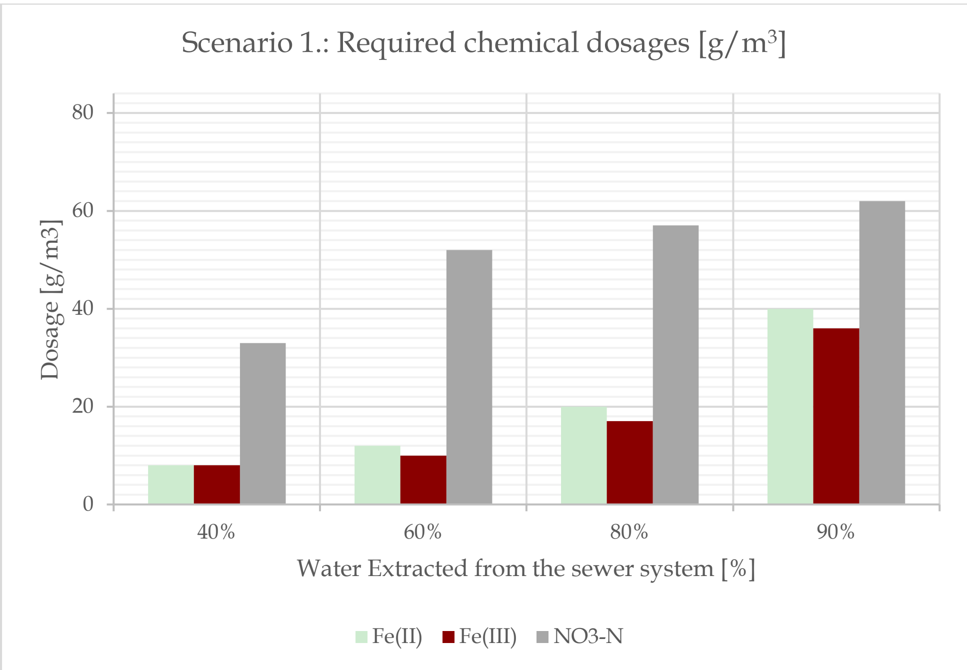
The observed concrete corrosion rates in this scenario exhibit a range between **0.03 mm/year** and **0.28 mm/year**. These corrosion rates are notably lower than the corrosion depths observed in the previous scenarios.

The substantial reduction in corrosion rates in this scenario is primarily attributed to the avoidance of wastewater extraction, preventing increased concentrations from entering the pressure main and, consequently, circumventing anaerobic conditions conducive to sulphate reduction and the formation of dissolved sulphides in the wastewater phase. As discussed in Section 2.4.1. and Section 2.4.2.1., the transition of formed dissolved sulphides, particularly H_2S, to aerobic conditions, as observed in the flow from pressure main to gravity main, initiates corrosion processes in the concrete pipes. While the anticipated corrosion depths may be deemed acceptable, prudent management and pipe replacement plans are imperative for assessing the necessity of mitigation measures to address the envisaged corrosion risks. In this study, mitigation measures for this scenario are thoroughly considered and discussed in Section 3.3.

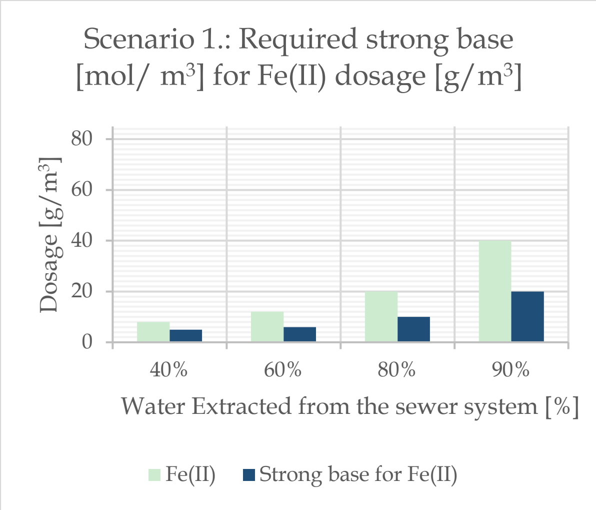
3.3. Required Chemical Dosages

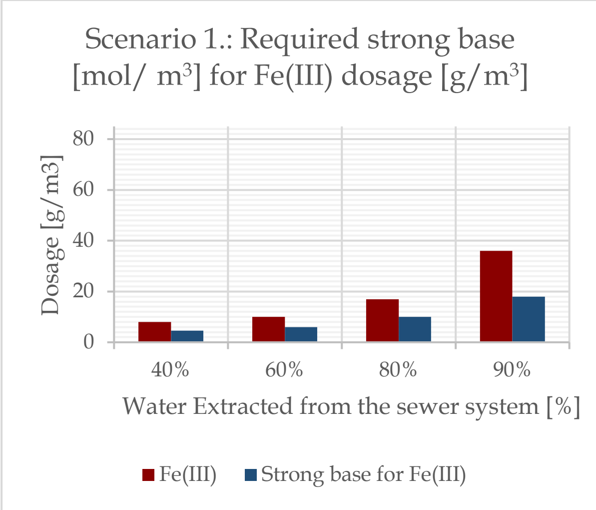
The extraction of wastewater from the sewer system induces an increase in dissolved sulphide concentrations, subsequently leading to concrete corrosion, as discussed in Section 2.4. To mitigate the corrosion problems, dosing strategies involving ferrous iron (Fe (II)), ferric iron (Fe (III)), nitrate (NO_3-N), hydrogen peroxide (H_2O_2), and sodium hypochlorite (NaOCl) can be employed within the sewer system.

This section presents the findings regarding the optimal chemical dosages necessary to prevent corrosion in the pipelines under three scenarios (outlined in Section 2.7) based on water extraction ratios of 40%, 60%, 80%, and 90%. The determination of optimal dosages involved a trial-and-error approach using Mega-WATS simulations. The required chemical dosages for Fe (II), Fe (III), and NO_3-N are presented in the following graphs, which also include the strong base requirements. As previously stated, the efficient control of sulphide through the application of iron salts may require the elevation of pH, accomplished by the addition of a strong base, such as calcium hydroxide (Ca $(OH)_2$). This necessity was underscored in the present study by Mega-WATS simulations, where the addition of iron salts led to a significant decrease in pH values and an associated inefficacy in reducing corrosion rates. Furthermore, due to the substantial quantities required, hydrogen peroxide and sodium hypochlorite were initially considered but are subsequently excluded from the following graphs.

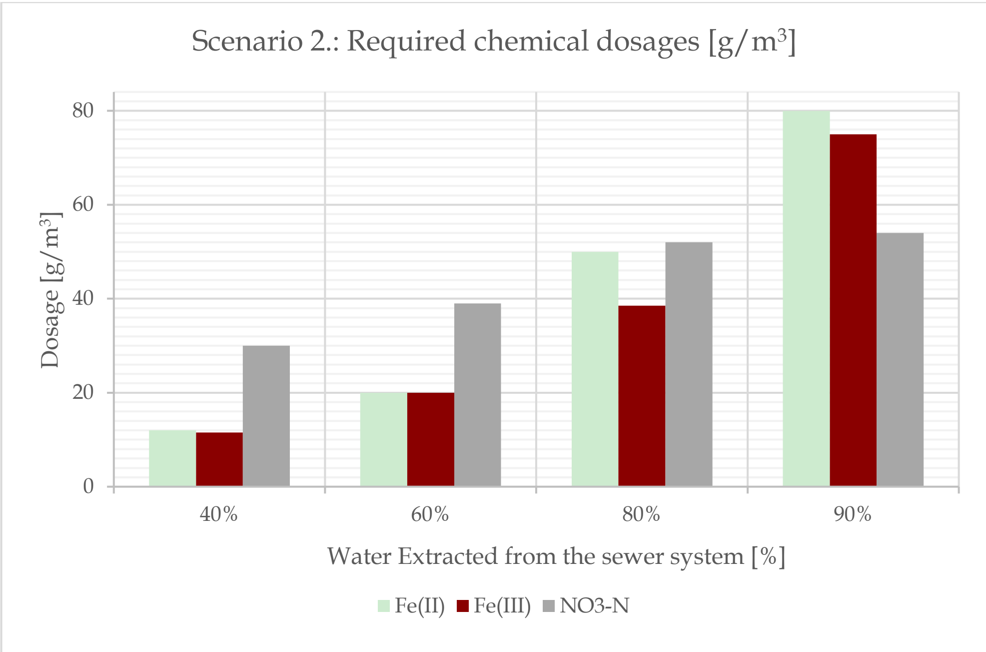


Graph 7. Required chemical dosages in Scenario 1.: Sewer Mining in Aalst for 40%, 60%, 80% and 90% of wastewater extraction from Riool-Zuid sewer system.

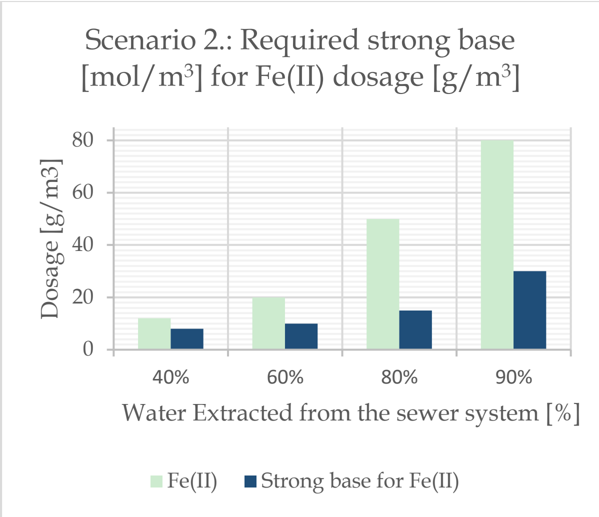


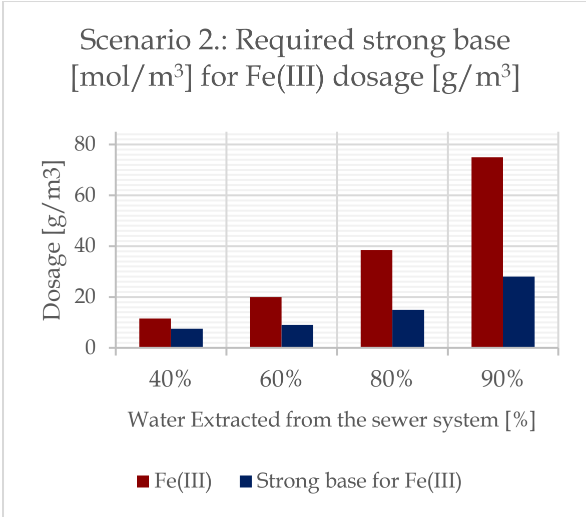


Graph 8 (left) and Graph 9 (right): Required strong base dosages for Fe (II) (left Graph) and Fe (III) (right Graph) in Scenario 1.: Sewer Mining in Aalst for 40%, 60%, 80% and 90% of wastewater extraction from Riool-Zuid sewer system.

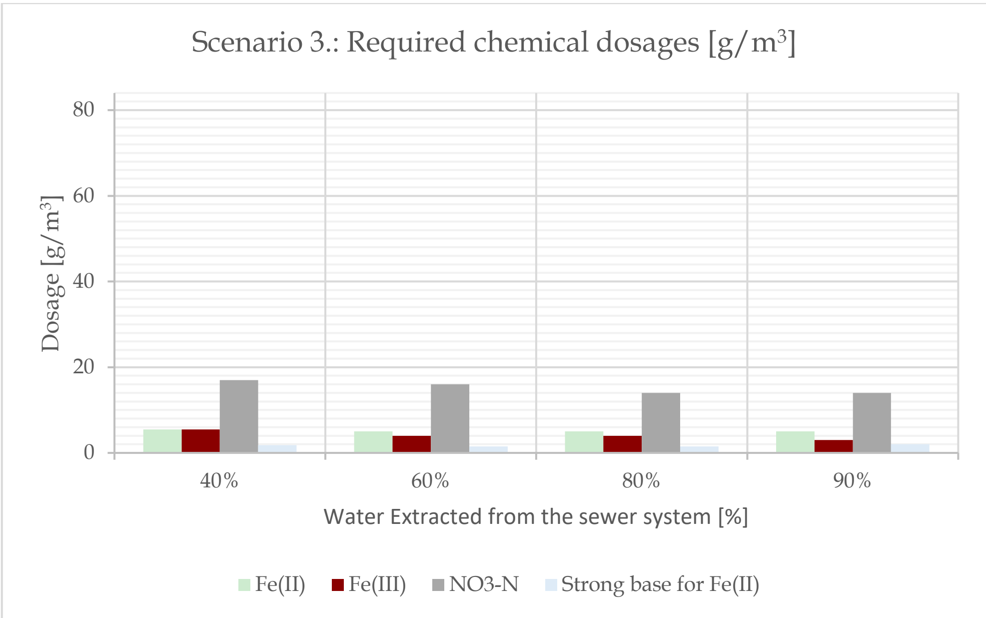


Graph 10. Required chemical dosages in Scenario 2.: Sewer Mining in de Meren for 40%, 60%, 80% and 90% of wastewater extraction from Riool-Zuid sewer system.

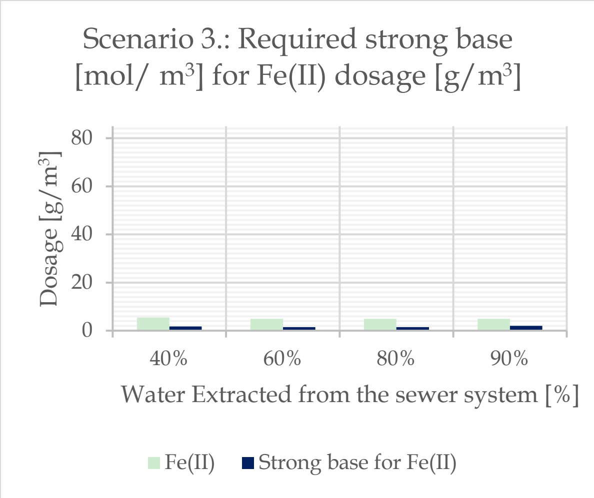


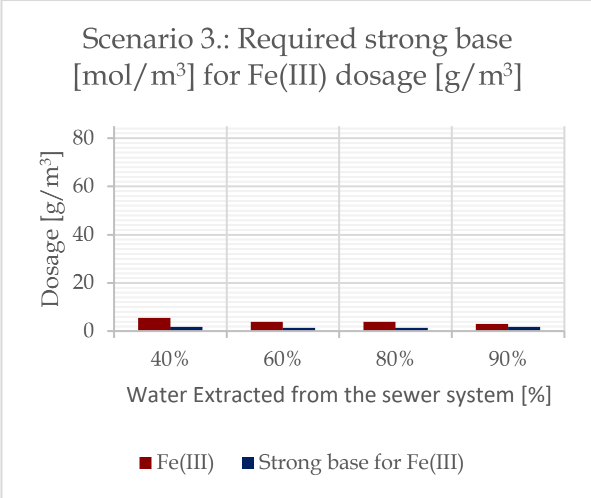


Graph 11 (left) and Graph 12 (right): Required strong base dosages for Fe (II) (left Graph) and Fe (III) (right Graph) in Scenario 2.: Sewer Mining in de Meren for 40%, 60%, 80% and 90% of wastewater extraction from Riool-Zuid sewer system.



Graph 13. Required chemical dosages in Scenario 3.: Sewer Mining at downstream of the pressure main (juncture of near the juncture of Heeze, Sterksel and Leende domestic wastewater inflow) for 40%, 60%, 80% and 90% of wastewater extraction from Riool-Zuid sewer system.





Graph 14 (left) and Graph 15 (right): Required strong base dosages for Fe (II) (left Graph) and Fe (III) (right Graph) in Scenario 3.: Sewer Mining at downstream of the pressure main (juncture of near the juncture of Heeze, Sterksel and Leende domestic wastewater inflow) for 40%, 60%, 80% and 90% of wastewater extraction from Riool-Zuid sewer system.

The necessary chemical dosages for each scenario align with the Mega-WATS simulation results elucidated in Section 3.2. Despite the higher concentrations of dissolved sulphides observed in "Scenario 2: Sewer mining in de Meren" compared to "Scenario 1: Sewer mining in Aalst" the essential nitrate dosages were found to be lower in Scenario 2: de Meren than in Scenario 1: Aalst. On the contrary, the required Fe(II) and Fe(III) dosages were found to be higher in Scenario 2.

In the context of Scenario 3, where sewer mining is implemented near the juncture of Heeze, Sterksel, and Leende domestic wastewater inflow, the required chemical dosages are notably minimal. This observation aligns with this scenario's characteristic of low concrete corrosion rates. The necessity of providing iron salts or nitrate is highly dependent on the adopted management strategies. While the introduction of iron salts and nitrate has the potential to extend the lifespan of concrete pipes, the corrosion depths elucidated in Section 3.2.3 may be deemed acceptable.

When chemical dosing results of sewer mining in de Meren is taken into account, the required chemical dosages do not show a big difference. Even though NO_3-N is known to be more expensive than iron salts, dosing Fe (II) and Fe (III) in this study, impact the pH levels of the sewer system significantly and require strong base for pH control. On the other hand, dosing NO_3-N seemed to be more practical as it does not require any strong base addition to control the pH.

The impacts of NO3-N on corrosion rates and pH in the Riool-Zuid, as related to sewer mining in de Meren, are detailed in Appendix A.

3.4. Sewer Mining Implementation results

As aforementioned in previous sections, the ultimate objective of this study is to boost the base flow of the River Dommel during dry summer periods. In pursuit of this aim, the contribution of each scenario is meticulously computed and presented in the ensuing tables, detailing the flows within Riool-Zuid towards Eindhoven WWTP and the flows transmitted from the sewer mining technology to the River Dommel, expressed in various units. Tables 12 provides the wastewater flows from Riool-Zuid to Eindhoven WWTP and the water flows from the sewer mining technology to the River Dommel for each scenario. Tables 13 represents the total flows in River Dommel and the contribution of sewer mining technology in each scenario. It is imperative to acknowledge that in computing the reduction ratio of the Eindhoven WWTP contribution, a meticulous consideration is given to the wastewater extracted from the Riool-Zuid system. This extraction is systematically subtracted from the flows attributed to the Eindhoven WWTP.

Table 12. Water flows in Riool-Zuid sewer system in each scenario.

WATER FLOWS IN RIOOL-ZUID							
Scenario		Extraction [%]	Flows in Riool-Zuid to Eindhoven WWTP		Water Flows to the River Dommel		
			[L/s]	[m^3/s]	[L/s]	[m^3/s]	[m^3/h]
1.	Sewer Mining in Aalst	40	221.1	0.2211	45.4	0.0454	163.3
		60	198.5	0.1985	68.0	0.0680	244.9
		80	175.8	0.1758	90.7	0.0907	326.6
		90	164.4	0.1644	102.1	0.1021	367.4
2.	Sewer Mining in de Meren	40	221.9	0.2219	44.6	0.0446	160.4
		60	199.6	0.1996	66.9	0.0669	240.8
		80	177.3	0.1773	89.2	0.0892	321.1
		90	166.1	0.1661	100.4	0.1004	361.3
3.	Sewer Mining at downstream of the Pressure Main	40	215.9	0.2159	50.6	0.0506	182.0
		60	190.6	0.1906	75.9	0.0759	273.2
		80	165.3	0.1653	101.2	0.1012	364.4
		90	152.6	0.1526	113.9	0.1139	410.0

For example, when considering a contributing flow of **361.3 m³/h**, if ‘’rainwater harvesting’’ solution had been chosen to provide the same flow, the required daily storage would need to be:

361.3 x 24 = 8,671.2 m^3

Table 13. Minimum and maximum total flows in River Dommel for each scenario.

TOTAL FLOW IN RIVER DOMMEL							
Scenario		Extraction [%]	[m^3/s]		Contribution of Sewer Mining		Contribution of Eindhoven WWTP is decreased by
			Min	Max	For min flow	For max flow	
1.	Sewer Mining in Aalst	40	0.5454	1.0454	9%	5%	-3.6%
		60	0.5680	1.0680	14%	7%	-5.4%
		80	0.5907	1.0907	18%	9%	-7.3%
		90	0.6021	1.1021	20%	10%	-8.2%
2.	Sewer Mining in de Meren	40	0.5446	1.0446	9%	4%	-3.6%
		60	0.5669	1.0669	13%	7%	-5.4%
		80	0.5892	1.0892	18%	9%	-7.1%
		90	0.6004	1.1004	20%	10%	-8.0%
3.	Sewer Mining at downstream of the Pressure Main	40	0.5506	1.0506	10%	5%	-4.0%
		60	0.5759	1.0759	15%	8%	-6.1%
		80	0.6012	1.1012	20%	10%	-8.1%
		90	0.6139	1.1139	23%	11%	-9.1%

It should be noted that the following values are taken into account to estimate the total wastewater flows in Riool-Zuid and the total water flows in River Dommer:

- In summer WWTP of Eindhoven constitutes approximately 50% of the total flow in the river Dommel by 1.25 m^3/s (Schilperoort, 2011).
- The base flow of the River Dommel is assumed as minimum 0.5 m^3/s and maximum 1.0 m^3/s during dry summer periods.

3.5. Cost Analysis

The cost analysis is carried out for all the scenarios (except the theoretical assumption/worst-case scenario) for the sewer mining implementation and chemicals required for the corrosion control, separately.

The investment costs of FO and RO membranes depends on the membrane area per module, fluxes and the membrane unit cost. The total membrane area required varies per scenario as each scenario results in different flows. In addition to the FO membrane costs, pre-treatment, RO, energy, draw solution consumption, installation, service, personnel, maintenance, cleaning and CIP are also significant in order to determine the total costs of the sewer mining system. Furthermore, the sewer mining location also plays a role for the costs as it effects the required pipe lengths. The piping costs are not included in this study. For the cost analysis, the FO membrane cost is taken into account at 150 €/m^3 (Cornelissen, personal communication, December 23, 2022).

Table 14. Forward Osmosis (FO) Membrane specifications and required number of modules for each scenario (Membrane specifications: [1] Aquaporin Inside® HFFO14 Module Datasheet; [2,3] TUD Water Treatment Course, 2016, Chapter 10; [4] Lutchmiah, 2014; [5] Aquaporin Inside® HFFO2 Module Datasheet)*

FO Membrane Analysis						
Available FO Modules		Hallow Fibre FO (HFFO14)*1	UF-MF Capillary Membrane *2	UF-MF Tubular Membrane *3	8040FO-MS-P*4	Hallow Fibre FO (HFFO2) *5
Membrane Area [m^2/module]		13.8	60.0	41.0	14.4	2.3
Flux [L /m^2/h]		11 ± 1.5	60-183	3.3 ± 0.4	8 ± 2	11 ± 1.5
Total flux per module [m^3/h/module]		0.1518	3.6	0.1353	0.1152	0.0253
Number of modules required per scenario						
Scenario 1. Sewer Mining in Aalst	40%	1793	76	2012	2363	10759
	60%	2690	113	3018	3544	16138
	80%	3586	151	4024	4726	21517
	90%	4034	170	4526	5316	24207
Scenario 2. Sewer Mining in de Meren	40%	1764	74	1979	2324	10583
	60%	2646	112	2968	3486	15874
	80%	3528	149	3958	4648	21166
	90%	3969	167	4453	5229	23812
Scenario 3. Sewer Mining at downstream of the Pressure Main	40%	2001	84	2245	2636	12003
	60%	3001	127	3367	3954	18005
	80%	4001	169	4489	5272	24006
	90%	4501	190	5050	5931	27007

*Table 15. Forward Osmosis (FO) Membrane costs for each scenario (*FO unit prices: [6]Cornelissen, 2022; [7]Nyman, 2023)*

FO Membrane Costs				
Available FO Modules		Hallow Fibre FO (HFFO14) *6	8040FO-MS-P *6	Hallow Fibre FO (HFFO2) *7
Scenario 1. Sewer Mining in Aalst	40%	€3.7M	€5.1M	€5.8M
	60%	€5.6M	€7.7M	€8.6M
	80%	€7.4M	€10.2M	€11.5M
	90%	€8.4M	€11.5M	€13.0M
Scenario 2. Sewer Mining in de Meren	40%	€3.7M	€5.0M	€5.7M
	60%	€5.5M	€7.5M	€8.5M
	80%	€7.3M	€10.0M	€11.3M
	90%	€8.2M	€11.3M	€12.7M
Scenario 3. Sewer Mining at downstream of the Pressure Main	40%	€4.1M	€5.7M	€6.4M
	60%	€6.2M	€8.5M	€9.6M
	80%	€8.3M	€11.4M	€12.8M
	90%	€9.3M	€12.8M	€14.4M

Table 16. Estimated CAPEX and OPEX for Forward Osmosis (FO) and Reverse Osmosis (RO) Hybrid System.

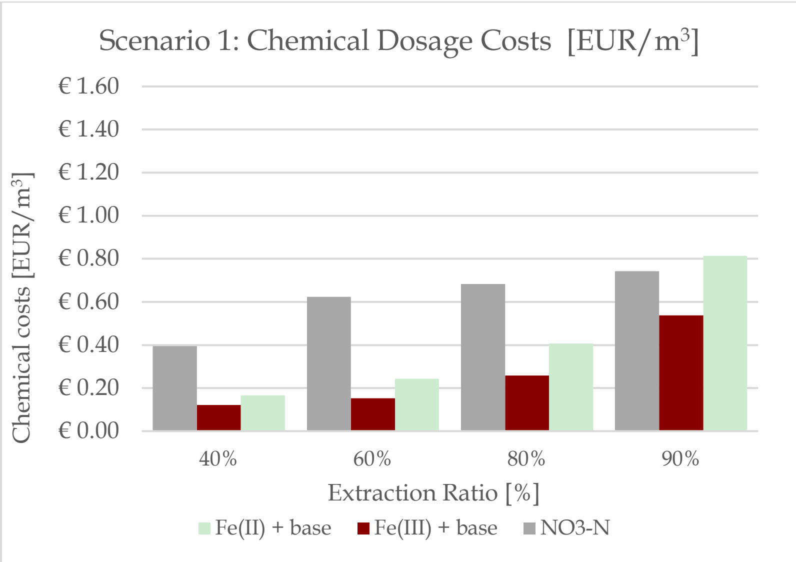
The Estimated CAPEX and OPEX Costs for FO&RO Hybrid System			
Parameter	Costs per m^3	Additional Information	Reference
Capex			
Pre-treatment	<€0.01	Reko zeefbocht, 300 m3/h, lifetime: 25 years	Reko, 2023.
Installation	€ 1.05	FO&RO hybrid system - 5 year depreciation	Cornelissen, 2023, KWR Water; Lutchmiah, 2014.
$Total_{CAPEX}$	€ 1.06		
Opex			
Recirculation Pumps + RO Energy	€ 0.33		Zou et al., 2016; Lutchmiah, 2014.
Service costs	€ 0.03		Cornelissen, 2023, KWR Water; Lutchmiah, 2014.
CIP	€ 0.05		Cornelissen, 2023, KWR Water.
Other costs	€ 0.21	NaCl, personel, insurance etc.	Cornelissen, 2023, KWR Water; Lutchmiah, 2014; Park et al., 2020.
$Total_{OPEX}$	€ 0.61		

The most significant cost driver is the expense associated with FO membranes, which accounts for the largest share of the overall costs. Based on cost estimates, the total treatment expenditure is approximately **1.67 €/m³.**

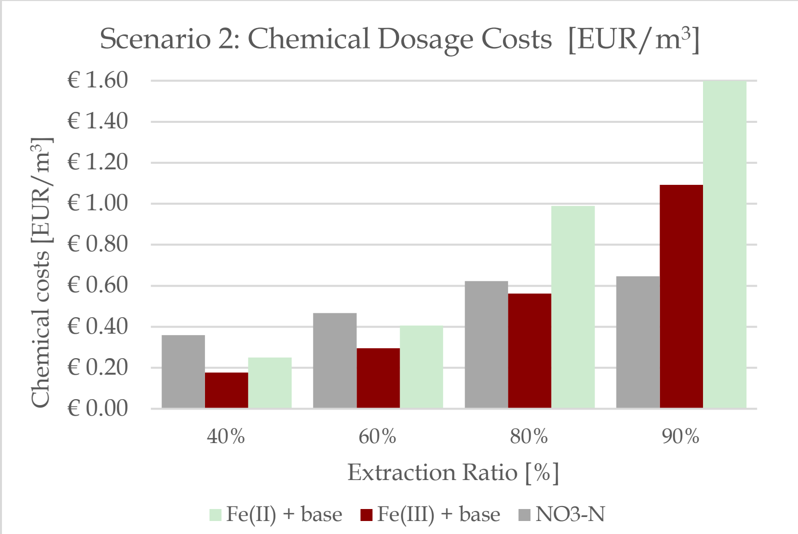
Comparing costs with other studies and membranes can be challenging due to the notable variations that may occur depending on the location and the timing of the treatment prototype installation. Factors such as membrane unit costs, electricity expenses, personnel requirements, and maintenance expenditures can exhibit significant differences. However, this study includes a cost analysis to provide insight into the level of cost-effectiveness of the chosen treatment method, ''FO&RO Hybrid System''.

Perez et al. (2022) reported wastewater reclamation costs using a UF-RO system ranging from 0.570 to 1.096 €/m³, with UF and RO membrane costs estimated at 52 €/m² and 25 €/m², respectively. In another study, Corzo et al. (2018) explored the implementation of an FO-NF demonstration plant for wastewater reuse and found a treatment cost of 0.96 €/m³. Iglesias et al. (2010) estimated the cost of a large-scale UF-RO process to be in the range of 0.35-0.45 €/m³. Vinardell et al. (2020) identified the cost of water reclamation by an FO-RO system minimum as 0.81 €/m³ for a 50% water recovery rate, and 1.01 and 1.27 €/m³ for recovery rates of 80% and 90%, respectively. It is important to note that in this study, the costs of FO and RO membranes are considered to be 55 and 24 $/m^2, respectively, based on the findings of Teusner et al. (2017) and Valladares Linares et al. (2016). Furthermore, Lutchmiah (2014) reported a water production cost of 0.65 €/m^3 for an FO-RO system. Notably, membrane module prices have experienced significant increases over time, which may account for the cost variations observed between these studies and the results obtained in this thesis study.

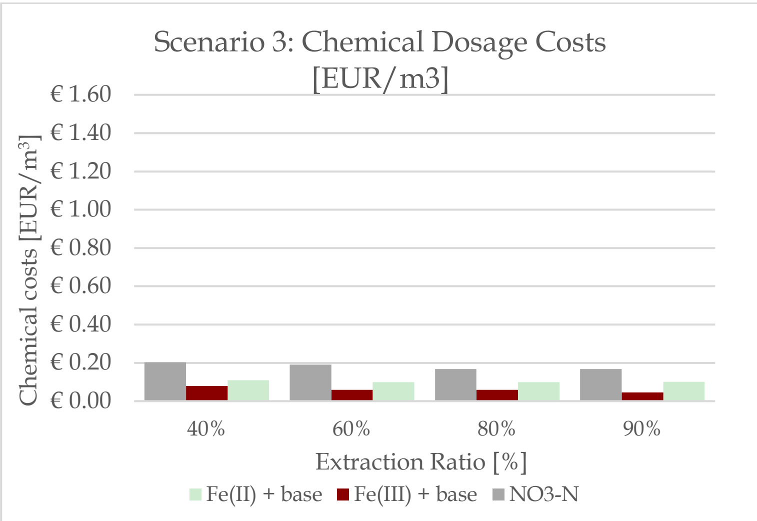
As discussed in the previous sections, the implementation of sewer mining leads to increased risk of corrosion in the Riool-Zuid downstream. Therefore, iron (II), iron (III) and nitrate are considered to be injected into the Riool-Zuid as a mitigation measure. The required chemical dosages were given in 3.3. Required Chemical Dosage section. Based on these findings, the chemical costs for each scenario are illustrated by Graph 18, 19 and 20. Unit prices were obtained from various sources: for Fe (II) and strong base from Labshop, for Fe (III) from Laboratoriumdiscounter, and for nitrate from Amazon as of 2023.



Graph 16. Chemical dosage costs [EUR/m^3] in Scenario 1: Sewer Mining in Aalst.



Graph 17. Chemical dosage costs [EUR/m³] in Scenario 2: Sewer Mining in de Meren.



Graph 18. Chemical dosage costs [EUR/m³] in Scenario 3: Sewer Mining at downstream of the pressure main.

The chemical unit costs vary per scenario as the required chemical dosages that are different in each scenario. When higher extraction ratios are implemented, the more chemicals are required to be injected into the Riool-Zuid, and therefore, the higher chemical unit cost were observed.

The chemical unit cost graphs reveal that the highest Fe (II) and Fe (III) unit costs were observed with Scenario 2: Sewer Mining in de Meren while the highest NO_3-N costs were obtained in Scenario 1: Sewer Mining in Aalst. In both cases, the highest costs were found with the wastewater extraction ratio of 90%.

Chapter 4. Discussion

The discussion section integrates the outcomes of the literature review followed by a comparative analysis with the findings of this study. This section will explore and discuss the following key aspects:

1- Impact of sewage flow rate on sulphide formation
2- Hydraulic Retention Time (HRT) influence on sulphide generation
3- Chemical Oxygen Demand (COD) levels and sulphide production
4- Anaerobic conditions and sulfide generation dynamics in sewer systems
5- Sulphide concentrations and concrete corrosion: Investigating the correlation
6- Effectiveness of Nitrate Dosing in Suppressing Hydrogen Sulfide Formation

4.1. Impact of Sewage Flow Rate on Sulphide Formation

The research by Liang et al. (2019) involved both experimental and modeling approaches to investigate sulfide formation in large-scale deep tunnel sewer systems. They focused on a real Sewage Conveyance System (SCS) located in Hong Kong, collects and conveys the sewage ($1.14\times10^6 m^3$/day) from both sides of the Victoria Harbour to the Stonecutters Island Sewage Treatment Works (SCISTW) through a system of 23.3 km in length and interconnected sewer tunnels depths ranging from 70 to 160 meters. The study aimed to assess sulfide inputs and outputs within the SCS through a seven-day field investigation conducted in two stages, under an average temperature of 28.7 °C. They conducted mathematical modeling using the Biofilm-Initiated Sewer Process Model (BISM) to elucidate detailed profiles of sulfide formation across various tunnel sections within the SCS.

The bio-kinetic and stoichiometric parameters of the BISM were calibrated and validated using the data obtained during Stages 1 and 2 of the field investigation, respectively. Their analysis indicated that the flowrates predicted by the BISM, without parameter adjustment, closely matched the measured flowrates at the outlet of the SCS. Furthermore, they observed a close alignment between predicted and measured flowrates in Stage 2.

Their findings revealed a trend where dissolved sulfide concentrations typically decreased with increasing flow rates, often reaching their lowest levels around peak flow rates. The reported correlation coefficient (R) of -0.26 suggested a negative correlation between dissolved sulfide concentrations and flow rates.

Liang et al., 2019 concluded that despite an increase in flowrate levels from 80% to 130%, the overall sulfide production rate in the SCS remained relatively stable, with reported changes of less than 6%. However, the concentrations of dissolved sulfide in the sewage at the outlet of the SCS decreased from 2.39 mg S/L to 1.69 mg S/L due to the dilution effect of the stormwater and the suppressed sulfide production by the DO introduced by the stormwater.

This study has determined that the overall sulfide production rate remains mainly unaffected by variations in flowrate. However, it is noteworthy that lower flowrates corresponded to higher outlet sulfide concentrations, a trend consistent with the observations made in this thesis.

To further contextualize these findings, it is useful to take into account the correlation between flowrate and sulfide concentration, as investigated in various scenarios through extraction rates in this thesis study. These scenarios revealed that the higher wastewater extraction rates resulted in higher dissolved sulphide concentrations in the sewer system of Aalst which is closely related to flow rates. The dissolved sulfide concentrations found in all scenarios align with the findings of Liang et al., 2019, indicating that lower flowrates correspond to higher outlet sulfide concentrations.

4.2. Hydraulic Retention Time (HRT) Influence on Sulphide Generation

The study conducted by Liang et al. (2019) revealed a significant linear correlation (R2=0.61) between normalized sulphide production rates and Hydraulic Retention Time (HRT), underscoring a noteworthy association between these variables. Furthermore, the investigation highlighted that longer HRTs can foster the development of anaerobic conditions, a crucial factor influencing sulfide formation and oxidation (Hvitved-Jacobsen et al., 2013). Conversely, the correlation with A/V (Area to Volume) ratios was found to be weak (R2=0.01).

As previously discussed in section 2.4.1. Sewer Processes and Sulphur Cycle, the production of sulphide becomes significant when the anaerobic residence time exceeds 0.5 to 2 hours (Hvitved-Jacobsen et al., 2013). Both the thickness of biofilms and anaerobic residence time play crucial roles in influencing the activity of sulphate-reducing bacteria. When wastewater velocity drops below 0.8 m/s, thicker biofilms tend to form, leading to greater sediment deposition. Conversely, higher velocities result in thinner biofilms due to increased shear forces, reducing resistance to mass transfer (Carrera et al., 2016).

According to the findings of this study, it can be inferred that higher rates of wastewater extraction from the sewer system correspond to higher concentrations of dissolved sulphides in the pressure main. This correlation is logical, as the extraction ratio not only determines wastewater velocity but also affects hydraulic residence time within the sewer pipes. Both wastewater velocity and hydraulic residence time are critical for microbial processes, as discussed in 2.4.1. Sewer Processes and Sulphur Cycle.

The extraction ratios not only impact wastewater velocity but also determine residence time, which is known to significantly influence the degree of wastewater transformation. Since catchment sizes and sewer pipe characteristics (such as diameter, length, and slope) remain constant across each scenario in this thesis project, changes in water extraction rates solely affect residence time.

Simulation results align with the findings of the study conducted by Liang et al. (2019) and the literature studies discussed in Section 2.4.1. Higher water extraction ratios lead to increased water concentration and decreased wastewater velocities within the pipes. This condition exposes microorganisms to substrate for longer periods and extends hydraulic residence time. Notably, hydraulic residence time is greatly affected by wastewater velocity, assuming all other pipe parameters remain constant across scenarios and sub-scenarios.

4.3. Chemical Oxygen Demand (COD) Levels and Sulphide Production

The field investigation conducted by Liang et al. (2019) was designed to quantify both sewage and dissolved sulfide inputs and outputs within the Sewage Conveyance System (SCS). Their study revealed that sulfide formation within the system was notably influenced by elevated concentrations of sulphate and Chemical Oxygen Demand (COD), supporting earlier findings by Zhang et al. (2008).

Moreover, Liang et al. (2019) observed that increases in soluble COD concentrations in the sewage ranging from 133 to 304 mg COD/L, led to a notable increase in sulfide production rates within the SCS, reaching up to 42%. Concurrently, these increases also elevated the concentrations of dissolved sulfide at the Sewage Conveyance and Initial Sewage Treatment Works (SCISTW) by approximately 32%.

In addition to the findings of the study conducted by Liang et al. (2019), the relationship between sulfide production rate and COD was also investigated by Tanaka and Hvitved-Jacobsen (2001) using a pilot plant pressure sewer system. The study focused on understanding the anaerobic transformations of organic matter in wastewater. The study found that COD components, particularly readily biodegradable substrate and fermentable, readily biodegradable substrate, were better predictors of sulfide production rates in pressure sewers compared to traditional dissolved COD measurements. This suggests that the biodegradability of specific COD components plays a significant role in sulfide generation in sewer systems.

In conclusion, based on the findings from Liang et al. (2019) and Tanaka and Hvitved-Jacobsen (2001), the correlation between COD concentrations and sulfide production in sewage systems emerges as a significant relationship worthy of attention. Liang et al. (2019) demonstrated that elevated concentrations of sulphate and COD within the SCS notably influenced sulfide formation, corroborating earlier research by Zhang et al. (2008). The study revealed that increases in soluble COD concentrations led to a notable rise in sulfide production rates within the SCS, resulting in concurrent increases in dissolved sulfide concentrations at the SCISTW. These trends were further supported by the observations of Tanaka and Hvitved-Jacobsen (2001), who found that specific COD components, particularly readily biodegradable substrate and fermentable, readily biodegradable substrate, were better predictors of sulfide production rates in pressure sewer systems. The relationship between average dissolved COD and sulfide production rates, found in this study, underscores the importance of understanding the biodegradability of COD components in predicting sulfide generation within sewer systems. This collective evidence emphasizes the critical role of wastewater quality, particularly COD concentrations, in influencing sulfide production, thus highlighting the necessity for effective management strategies to mitigate sulfide-related issues in sewage systems.

In light of the findings from this thesis, particularly the scenarios involving sewer mining implementation in Aalst and de Meren, where higher COD concentrations are observed due to the intake of concentrated water from the FO&RO system, the observed increase in sulfide concentrations aligns logically. It is important to note that while increased COD levels play a significant role in influencing dissolved sulfide concentrations, other factors such as flow rates and HRTs also undergo changes due to wastewater extractions from the sewer system.

Nonetheless, these observations remain consistent with the outcomes of literature studies, emphasizing the multifaceted nature of sulfide generation dynamics in sewer systems.

4.4. Anaerobic Conditions and Sulfide Generation Dynamics in Sewer Systems

In sewer systems, sulfide production predominantly occurs under fully anaerobic conditions, particularly in rising mains (pressure pipes) after pump stations. Upon discharge into partially filled gravity pipes, hydrogen sulfide (H_2S) transfer from the liquid phase to the pipe's headspace can occur. Subsequently, H_2S in the gas phase may transfer to exposed concrete pipe surfaces, where it undergoes oxidation to form sulfuric acid and other sulphur species (Zhang et al., 2008).

The study conducted by Nielsen et al. (2008) undertook an extensive field investigation to examine the formation and fate of sulfide in a force main and its downstream gravity sewer. The objectives of this study were twofold:

1- To gather data on sulfide formation in the force main and its subsequent fate in an aerobic gravity sewer.
2- To assess the applicability of the WATS model in simulating sulfide formation and fate in sewer systems based on field study outcomes.

The field study carried out by Nielsen et al. (2008) focused on an intercepting sewer serving approximately 4000 person equivalents, comprising a force main followed by a gravity sewer. Samples were collected at the pumping station and discharge manhole to study anaerobic sulfide transformations in the force main. In the gravity sewer section, samples were collected from three manholes along a 402-meter distance to investigate aerobic sulfide transformations. According to this study, significant sulfide formation was observed in the force main, however, through the gravity pipes, sulfide concentrations decreased by an average of 30% due to aerobic sulfide oxidation and hydrogen sulfide emission.

Furthermore, WATS model predicted that approximately 90% of the decrease in sulfide concentration in the gravity sewer was due to sulfide oxidation, with only a small fraction entering the sewer atmosphere, contributing to odour and corrosion. Despite this, the model forecasted concrete corrosion rates of up to 1.2 mm/year in the gravity sewer section.

In conclusion, the study conducted by Nielsen et al. (2008) underscored a significant increase in sulfide concentrations within the force main during transport, followed by a notable decrease in sulfide concentrations downstream of the force main outlet in the gravity sewer, with concentrations typically around 70% of the initial concentration. The field study further confirmed the effectiveness of the WATS model in predicting sulfide concentrations in both force mains and gravity sewers.

Furthermore, the results of this thesis study are in line with these findings, as evidenced by the sharp decrease in dissolved sulfide concentrations downstream of the pressure main. This decrease reflects the transition from anaerobic conditions in the pressure main to aerobic conditions in the downstream gravity pipes. As detailed in Section 2.4.2.1, the subsequent exposure of hydrogen sulfide to aerobic conditions facilitates its oxidation to sulfuric acid, explaining the observed decrease in dissolved sulfide concentrations in the downstream gravity pipes. Additionally, the findings of this thesis also demonstrate a significant increase in dissolved sulfide concentrations within the pressure main, consistent with the establishment of anaerobic conditions conducive to sulfide production. This is further supported by the observed changes in sulphate-reducing bacteria biomass, suggesting a direct proportional relationship between dissolved sulfide concentrations and sulphide-reducing biomass.

In summary, the results of this thesis align with previous literature studies, highlighting the critical role of anaerobic conditions in sulfide production within sewer systems and emphasizing the importance of understanding sulfide dynamics for effective corrosion mitigation strategies.

4.5. Sulphide Concentrations and Concrete Corrosion: Investigating the correlation

Based on the study conducted by Sun et al. (2014), which introduced a rapid, non-destructive methodology for monitoring sulfide-induced corrosion of concrete in sewer systems, it can be concluded that there is a significant correlation between sulfide concentrations and corrosion rates. The methodology developed in the study, based on measuring the H_2S uptake rates of concrete at various corrosion stages, demonstrated good reproducibility and reliability in assessing the corrosion process.

The results indicated that severely corroded concrete segments exhibited higher sulfide uptake rates compared to less corroded segments. This finding suggests a direct relationship between the severity of concrete corrosion and the activity of biological sulfide oxidation. Additionally, temperature fluctuations were shown to have a stronger effect on the uptake rate of heavily corroded coupons, further emphasizing the influence of environmental factors on corrosion rates.

Moreover, the corrosion rate estimated from the H_2S uptake results aligned well with corrosion rates observed in real sewer systems under similar conditions. This suggests that the methodology developed by Sun et al. (2014) can effectively predict corrosion rates based on sulfide concentrations, providing valuable insights into factors affecting sulfide-induced concrete corrosion.

In comparison to the findings of Sun et al. (2014), this thesis study investigated the correlation between dissolved sulphide concentrations and corrosion rates in sewer systems under varying wastewater extraction scenarios. While Sun et al. (2014) focused on monitoring sulphide-induced corrosion using H_2S uptake rates, this study examined the impact of different water extraction rates (90%, 80%, 60%, and 40%) on both sulphide concentrations and corrosion rates. The outcomes indicated that higher dissolved sulphide concentrations corresponded to increased corrosion rates, consistent with the notion of sulfide-induced corrosion highlighted in the study of Sun et al. (2014). Specifically, in both Scenario 1: Sewer mining in Aalst and Scenario 2: Sewer mining in de Meren, where higher wastewater extraction rates were applied, elevated dissolved sulphide concentrations were observed and subsequently higher corrosion rates. This supports the premise that sulphide levels play a significant role in exacerbating concrete corrosion in sewer systems, as suggested by Sun et al. (2014). However, it's noteworthy that this thesis study provides additional insights into the specific impact of varying water extraction rates on sulphide concentrations and corrosion rates, contributing to a more comprehensive understanding of corrosion processes in sewer infrastructure.

4.6. Effectiveness of Nitrate Dosing in Suppressing Hydrogen Sulfide Formation

The study carried out by Bentzen et al. (1995) focused on the controlled dosing of nitrate to prevent hydrogen sulfide (H_2S) formation in a sewer network and its subsequent effects on the treatment process. The results revealed that nitrate dosing was highly effective in suppressing hydrogen sulfide levels in a rising main. During nitrate dosing, the average concentration of hydrogen sulfide at the works inlet was 0.24 mg/l, with a daily variation ranging from 0 to 0.4 mg/l. In contrast, without nitrate addition, the background level of hydrogen sulfide averaged 4.2 mg/l, with daily variations between 1 and 10 mg/l. Importantly, nitrate dosing achieved these reductions without significant breakthroughs of nitrate to the treatment works.

Furthermore, nitrate dosing led to increased removal of soluble BOD across the rising main, although no significant changes in total BOD or COD were observed at the treatment plant during the dosing period. The nitrification process in the biological filters at the treatment plant also showed improvement during nitrate dosing.

The study highlighted the formation of anaerobic conditions in the rising main, accompanied by high concentrations of H_2S at the works inlet and in the primary tank overflow. Nitrate dosing effectively suppressed septicity and eliminated hydrogen sulfide at the works inlet, with the average H_2S level reduced by 95% during the dosing period compared to the post-dosing period.

Despite the low levels of nitrate entering the primary tanks during the dosing period, there was a significant reduction in sulfide concentration in the primary tank overflow. The average concentration of sulfide decreased from 5.4 mg/l after nitrate dosing ceased to 2 mg/l during the dosing period, representing a 63% reduction primarily attributable to septicity suppression in the rising main.

Studies have demonstrated that nitrate, when present in concentrations ranging from 10 to 40 mg-N/L, can effectively reduce sulfide levels to as low as 0.2–3 mg-S/L in rising main sewers spanning lengths between 2.4 and 5 km (Bentzen et al., 1995; Saracevic et al., 2006; Jiang et al., 2009). For instance, Rodriguez-Gomez et al. (2005) observed significant reduction in sulfide production with just 5 mg-N/L of nitrate in sewage within a 61 km long gravity sewer conveying reclaimed water. Similarly, field trials conducted in a 6.7 km combined sewer network confirmed the efficacy of nitrate in sulfide control (Mathioudakis et al., 2006; Jiang et al., 2009).

Furthermore, it's notable that nitrate dosing does not exert immediate or long-term inhibitory or toxic effects on sulphate reduction by sewer biofilms. Biofilm studies within a laboratory setting, receiving nitrate additions, revealed sustained sulfidogenic activity over several months, even in the presence of nitrate. Sulfide accumulation continued unabated until the third or fourth nitrate dose, typically at 30 mg-N/L. Notably, the rapid development of anoxic sulfide oxidation by nitrate-reducing sulfide-oxidizing bacteria (NR-SOB) emerged as the primary mechanism for sulfide control in sewers receiving nitrate additions (Mohanakrishnan et al., 2009; Jiang et al., 2009)

Another relevant study conducted by Ganigue et al. (2011) delves into the practice of chemical dosing for sulfide control in Australia. Their findings shed light on the preferential use of nitrate in smaller systems, with approximately 25% of the sites situated in pipes with average dry weather flow exceeding 1 ML/day. In terms of pipe dimensions, over 60% of the sites employing nitrate dosing feature diameters ranging between 0.15 and 0.3 m. Notably, nitrate, albeit effective, is identified as a costly chemical (de Haas et al., 2008), typically dosed as $NaNO_3$ or Ca $(NO_3)_2$ (Zhang et al., 2008). Unlike oxygen, nitrate does not inhibit sulphate-reducing bacteria (SRB) activity in the short- or long-term and does not diminish SRB abundance in sewer biofilms. Instead, nitrate introduction at the start of rising mains elevates SRB activity in downstream biofilms (Mohanakrishnan et al., 2009). While sulfide can undergo biological oxidation by nitrate-reducing sulfide-oxidizing bacteria (NR-SOB), chemical sulfide oxidation remains minimal (Ganigue et al., 2011). Furthermore, nitrate also serves as an electron acceptor like oxygen for other heterotrophic bacteria, leading to subsequent nitrate consumption. These biological processes primarily occur within the biofilm matrix (Mohanakrishnan et al., 2009). Consequently, the efficacy of nitrate dosing is not dependent on the pipe's surface area-to-volume (A/V) ratio, which makes it suitable for application in both small and large pipe networks. However, the feasibility of employing nitrate in larger systems may be constrained by its prohibitive cost (Sharma et al., 2011; Ganigue et al., 2011).

Upon comparing the results obtained from the literature review with the outcomes of this thesis investigation, it is evident that nitrate dosing emerges as an effective strategy for suppressing hydrogen sulfide formation and consequently mitigating concrete corrosion issues within the sewer system (refer to APPENDIX A). Although the study also carefully analysed the effectiveness of Fe (II) and Fe (III) dosing, it was observed that in some cases, their application requires considerable amount of strong base dosing to regulate pH levels in the sewer system. Moreover, optimizing the dosage of iron and strong base often required multiple dosing points in certain sub-scenarios. In contrast, nitrate dosing demonstrated greater feasibility, typically requiring only a single dosing point to address corrosion concerns. Furthermore, while Fe (II) and Fe (III) are generally considered more cost-effective than nitrate, the cost analysis conducted in 3.5 Cost Analysis, the Scenario 2: sewer mining in de Meren, with a 90% wastewater extraction ratio revealed that nitrate costs were lower than those of Fe (II) and Fe (III). However, it is important to note that unit costs differ across scenarios and sub-scenarios due to varying wastewater extraction ratios and application areas, resulting in variations in sulphide concentrations, COD levels, flow rates, HRTs and, consequently, the required chemical dosages.

In summary, nitrate dosing emerges as a potent method for mitigating hydrogen sulfide formation without necessitating additional pH control measures. Although initial impressions may suggest higher costs compared to Fe (II) and Fe (III) dosing, in some cases, such as in Scenario 2: Sewer mining in de Meren with a 90% wastewater extraction ratio, nitrate dosing turned out to be a feasible and economically advantageous solution. Moreover, in other scenarios, despite potentially higher expenditure, its simplicity of application with a single dosing point and absence of the need for supplementary strong base addition render it a more practical and straightforward approach than using iron for hydrogen sulfide suppression.

Chapter 5. Conclusion

The Eindhoven Sewer System consists of Eindhoven, Nuenen/Son and the Riool-Zuid. The wastewater of these catchments, collected and conveyed to the Eindhoven WWTP, is subsequently discharged into the River Dommel after treatment. Managing contaminants in the River Dommel is generally feasible during periods of rainfall when the base flow benefits from natural dilution. However, during dry summer periods, the River Dommel experiences reduced base flow rates, rendering it susceptible to the influx of pollutants from upstream sources and the Eindhoven WWTP, resulting in water quality issues.

This chapter is dedicated to exploring the potential of mitigating water quality issues in the River Dommel during dry summer periods by feeding its base flow with treated wastewater extracted from the Riool-Zuid part of the Eindhoven sewer system. Within this context, the responses to the primary research question and sub-questions will be reviewed. Below, the main conclusions with respect to the sub questions are listed.

1- **How much water can feasibly be extracted from the selected catchment areas to sustain the River Dommel's flow during dry periods?? (SQ1)**

The domestic wastewater flow from the Riool-Zuid catchments to Eindhoven WWTP is estimated at around 960 m^3/h during dry weather periods, calculated based on the 2022 population and an assumed water consumption rate of 120 L/d per inhabitant. This volume constitutes the entirety of the domestic wastewater flow from the Riool-Zuid catchments to the sewer system. To ensure the proper and reliable functioning of the Riool-Zuid sewer system, this study employs a maximum wastewater extraction ratio of 90%. It's important to note that the extraction ratios in practical scenarios represent the amount of extracted water from the upstream and do not directly correlate with a percentage of the total flow within the Riool-Zuid sewer system.

2- **How effective and efficient is the integration of FO&RO hybrid system for sewer mining? (SQ2)**

The primary objective of this research is to identify an effective solution for mitigating the water quality challenges faced by the River Dommel during summer dry weather flows, with a focus on the Riool-Zuid catchment area of the Eindhoven wastewater system. The possible methods to mitigate these challenges were identified as: rainwater harvesting and storage system in Eindhoven, tertiary treatment at the Eindhoven WWTP and sewer mining application at the catchment areas. The decision was to utilize sewer mining technology in the Riool-Zuid catchments. Within this approach, the wastewater is extracted from Riool-Zuid, treated by FO-RO hybrid system and the permeate water of the system is subsequently directed to the River Dommel to increase its base flow. The literature studies discussed in Section 2.1.3, along with the treatment capacity analysis of the FO&RO hybrid system, demonstrate a high efficacy and efficiency of the selected method for sewer mining. This approach proves highly effective in removing COD, BOD, TSS, nitrogen, and phosphorus, thereby enabling compliance with water quality standards suitable even for cypriniformes fish habitat.

3- **How can the adverse impacts of sewer mining on the sewer system be effectively evaluated? (SQ3)**

The impact of the sewer mining implementation is further explored with Mega-WATS model simulations through an analysis of various scenarios that were developed. These scenarios were based on three locations: de Meren, Aalst and the juncture of Heeze, Sterksel and Leende domestic wastewater inflow. In addition to that, for an in-depth analysis, various wastewater extraction ratios were applied: 40%, 60%, 80% and 90% and compared with the no extraction conditions.

The Riool-Zuid sewer system consists of gravity pipes and two parallel pressure mains. The catchment areas of de Meren and Aalst are located at upstream of the pressure main. Heeze, Sterksel and Leende, on the other hand, are located at downstream of the pressure mains. Therefore, the impact of sewer mining technology on Riool-Zuid sewer system is analysed both for gravity and pressure mains, which represent aerobic and anaerobic conditions, respectively.

4- **What are the optimal locations for implementing sewer mining technology? (SQ4)**

The implementation of sewer mining technology upstream of the pressure main led to high dissolved sulphide concentrations, consequently causing considerable corrosion and odour problems in Riool-Zuid. Conversely, when the technology was implemented downstream of the pressure main, these problems were notably reduced. However, it is essential to underscore the significance of the distance between the sewer mining technology and the River Dommel, as the primary objective of this study is to enhance the River Dommel's base flow efficiently, reliably, and cost-effectively.

De Meren, as the closest site to the River Dommel, offers the advantage of existing infrastructure, including a suitably sized building, which eliminates the need for extra construction costs and minimizes expenses related to piping installation.

While the adoption of sewer mining technology in de Meren resulted in increased corrosion rates and more significant corrosion challenges in Riool-Zuid, leading to higher chemical expenses, the absence of construction costs and reduced infrastructure requirements may still render it more cost-effective. However, to thoroughly assess cost-effectiveness, a more comprehensive cost analysis is advisable for future investigations.

5- **How can potential hydrogen sulfide-related challenges, such as corrosion and odour, be effectively managed in sewer mining operations? (SQ5)**

This study, focusing on an existing sewer system, has employed chemical dosing strategies to mitigate corrosion issues. Chemicals recognized for their effectiveness in corrosion control include ferrous iron (Fe (II)), ferric iron (Fe (III)), nitrate (NO_3-N), hydrogen peroxide (H_2O_2), and sodium hypochlorite (NaOCl). The Mega-WATS simulations revealed that excessive quantities of hydrogen peroxide and sodium hypochlorite were required to address the corrosion problems in Riool-Zuid. Therefore, hydrogen peroxide and sodium hypochlorite were initially considered but subsequently excluded as viable options. Consequently, the use of Fe (II), Fe (III), and NO_3-N appeared to be a more practical approach. While Fe (II) and Fe (III) chemicals are more economical than NO_3-N, they necessitate the addition of a strong base for pH control. Moreover, optimizing their dosage often requires multiple dosing points. Conversely, NO_3-N proves more practical due to its single dosing point requirement and the absence of additional pH control measures.

6- What is the contribution of the sewer mining implementation to River Dommel base flow? (SQ6)

In the context of water quantities contributing to the River Dommel, the implementation of sewer mining in the de Meren resulted in total flow of ranging from **0.5446** to **1.0446 m^3/s** in River Dommel, slightly below the observed flow rates in Aalst, which ranged from 0.5454 to 1.1021 m^3/s. When implementing sewer mining technology in de Meren with wastewater extraction ratios of 40%, 60%, 80%, and 90%, the water quantities feeding the River Dommel are **160.4 m^3/h**, **240.8 m^3/h**, **321.1 m^3/h** and **361.3 m^3/h**, respectively.

7- What is the economic viability of FO-RO hybrid system as a sewer mining solution? (SQ7)

The total treatment cost for the FO-RO system with a pre-treatment unit was estimated to be approximately 1.67 €/m^3. The dominating factor is therefore the price of the FO membrane, which is one of the major costs for the sewer mining concept. The FO and RO membrane costs are regarded as 120 €/m^2 and 95 €/m^2, respectively. However, it should be noted that, the chemicals required for the corrosion control are also high and needs to be taken into account.

Chapter 6. Recommendations

In light of the comprehensive assessment of this thesis study, the following recommendations can be considered for future research:

1- **Further Investigation of the FO-RO hybrid system on a Lab-Scale:** While the literature studies have shed light on the efficiency of the FO-RO hybrid system as a sewer mining technology, it is advisable to establish a lab-scale pilot plant for the FO-RO hybrid system. This pilot plant would be supplied with actual wastewater from the Riool-Zuid catchments, offering a deeper understanding of its precise efficiency and potential drawbacks. This approach is crucial as the actual wastewater may contain unforeseen pollutants, not considered in this study due to the neglect of industrial wastewater contributions. Moreover, conducting draw solution analysis for optimization, alongside evaluating the impacts of system shutdowns and determining the optimal cleaning method and cycle, are recommended.

2- **Exploration of Alternative Methods:** Implementing tertiary treatment at the Eindhoven WWTP might be more advantageous in terms of water quantity. This is due to the fact that the effluent flow from the Eindhoven WWTP is approximately twelve times higher than the water that can be supplied through sewer mining technology in de Meren to the River Dommel. While this solution may result in implementation higher costs, it offers a continuous solution to the River Dommel to meet the standards. However, it should be highlighted that the focus in this study is to feed the upstream of the River Dommel. Therefore, application of the tertiary treatment to feed the upstream of the River Dommel is challenging. As a second recommendation, exploring alternative methods could involve the use of a different type of membrane with a higher recovery rate than FO, and an analysis of its potential impact on the sewer system.

3- **Enhanced Data Collection:** Conducting up-to-date measurements of wastewater concentrations in Riool-Zuid is recommended to enhance the quality and accuracy of research findings. Additionally, measuring water quality parameters upstream and downstream of the Eindhoven WWTP discharge point separately can provide a more comprehensive assessment of the impact of sewer mining implementation.

4- **Long-term Monitoring:** Following the implementation of sewer mining technology at the Meren control station, regular monitoring of hydrogen sulphide levels in the Riool-Zuid sewer system is recommended. This measure ensures that chemical dosages remain sufficient to mitigate corrosion, odour, and health risks effectively.

5- **Extensive Cost Analysis:** Conducting cost analyses based on current prices is essential, considering the tendency for membrane module prices to fluctuate over time. Additionally, accounting for pipe and infrastructure costs, which were not factored into this study, is crucial for a comprehensive cost assessment.

6- **Sustainability and Environmental Impact of Sewer Mining:** Exploring the sustainability aspects and environmental impact of sewer mining is recommended for future research studies. This entails conducting a thorough assessment of the ecological footprint associated with sewer mining operations, including energy consumption, greenhouse gas emissions, and overall resource utilization. Furthermore, evaluating the potential ecological disturbances and habitat alterations resulting from the implementation of sewer mining technologies is crucial. This comprehensive analysis will contribute to the development of environmentally conscious and sustainable approaches to wastewater management in urban areas.

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APPENDIX

A. Mega-WATS Model Results: Impact assessment of dosing NO_3-N

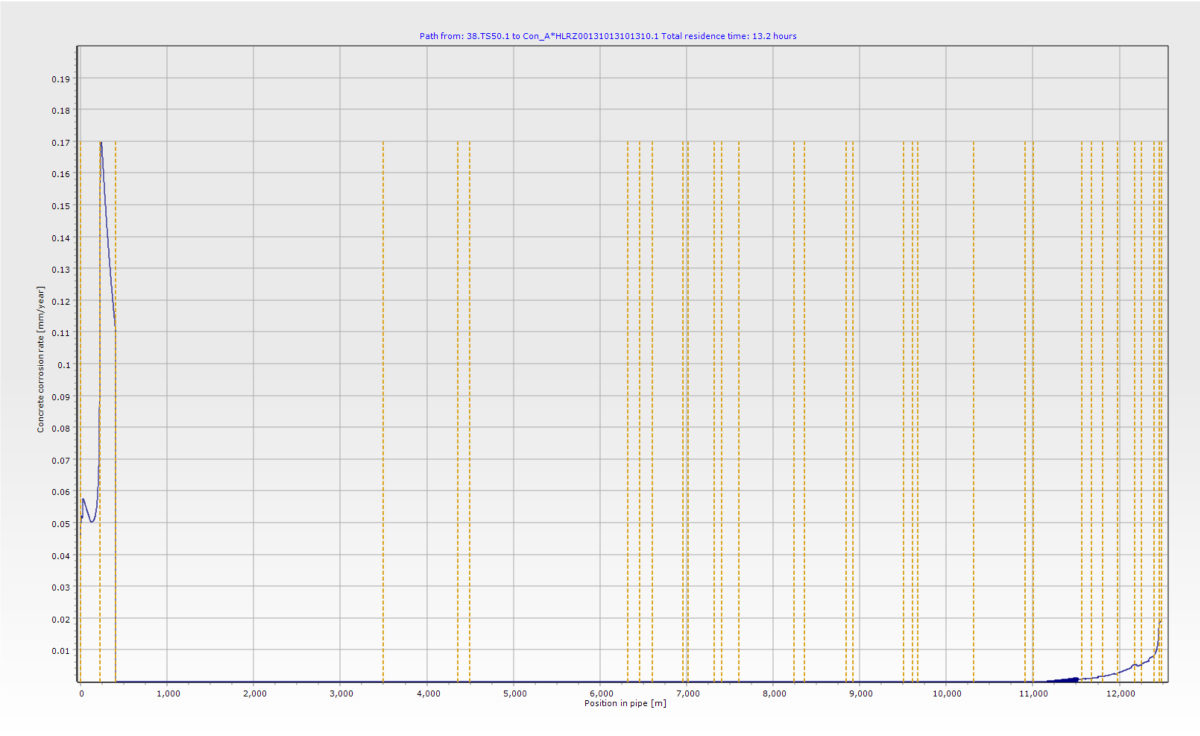

Figure A1. Concrete corrosion rates [mm/year] in Riool Zuid downstream pipeline after implementing the Sewer Mining Technology in de Meren with a wastewater extraction of 90% and with NO_3-N dosing.

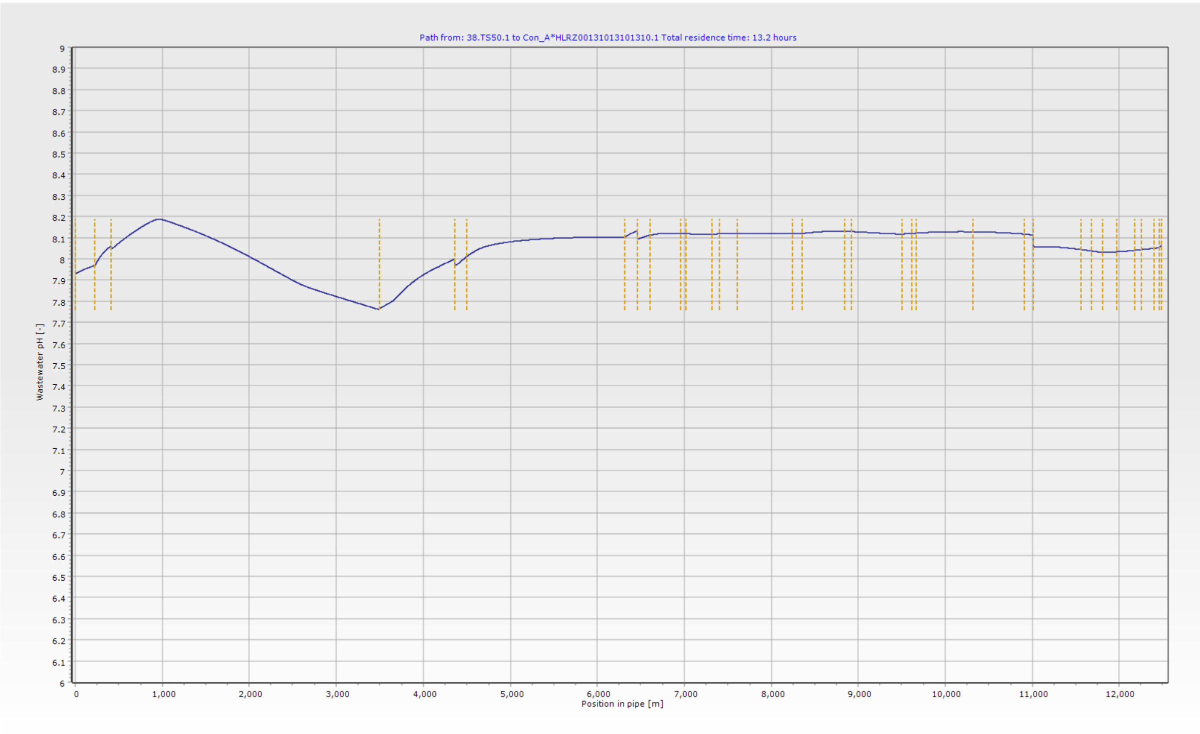

Figure A2. pH in Riool Zuid downstream pipeline after implementing the Sewer Mining Technology in de Meren with a wastewater extraction of 90% and with NO_3-N dosing.

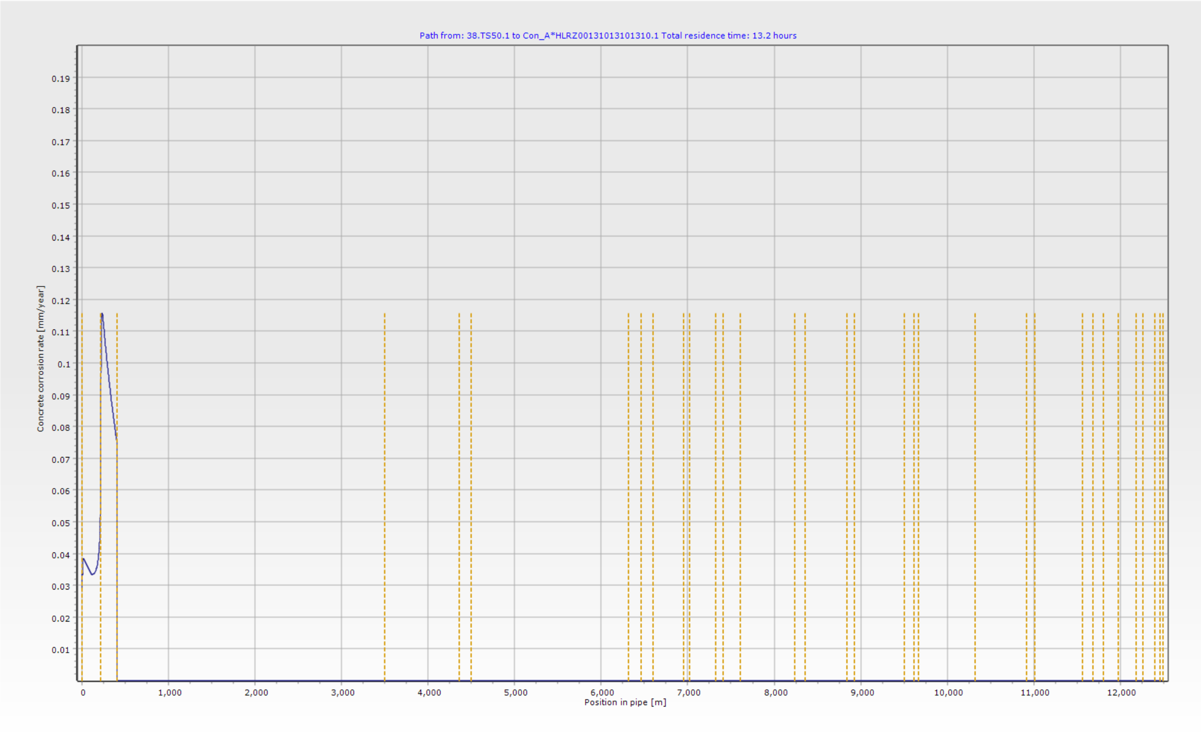

Figure A3. Concrete corrosion rates [mm/year] in Riool Zuid downstream pipeline after implementing the Sewer Mining Technology in de Meren with a wastewater extraction of 80% and with NO_3-N dosing.

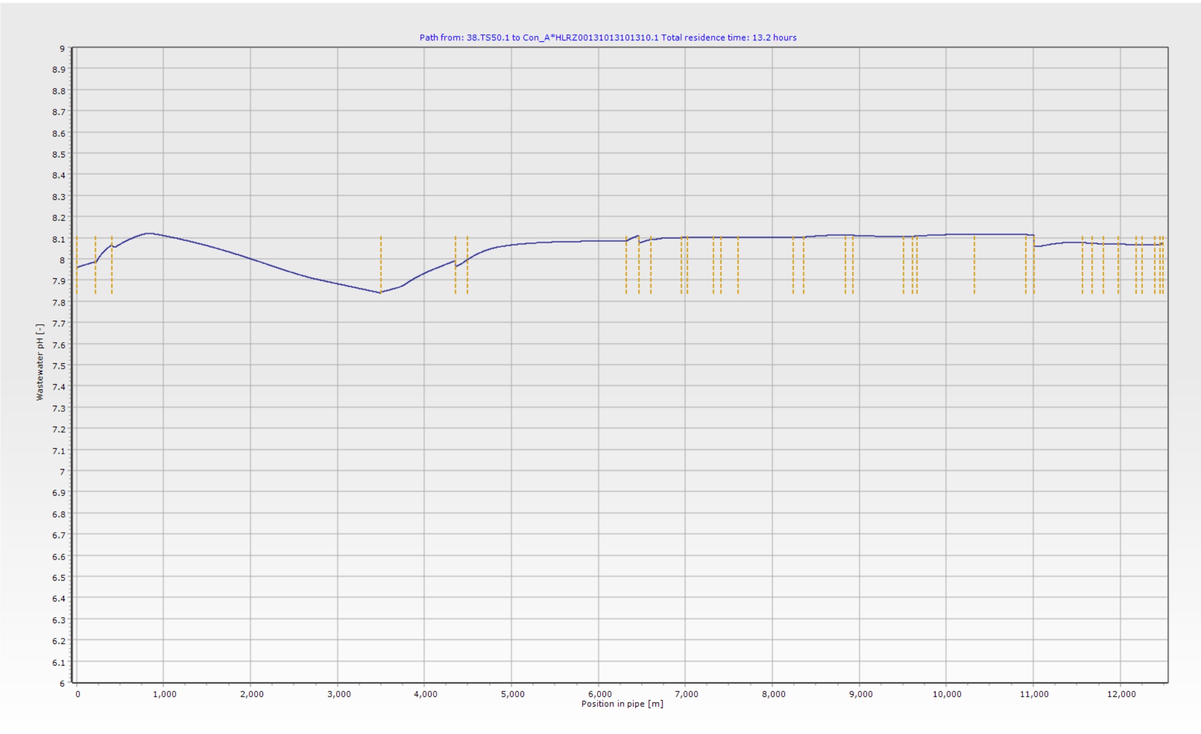

Figure A4. pH in Riool Zuid downstream pipeline after implementing the Sewer Mining Technology in de Meren with a wastewater extraction of 80% without chemical dosing.

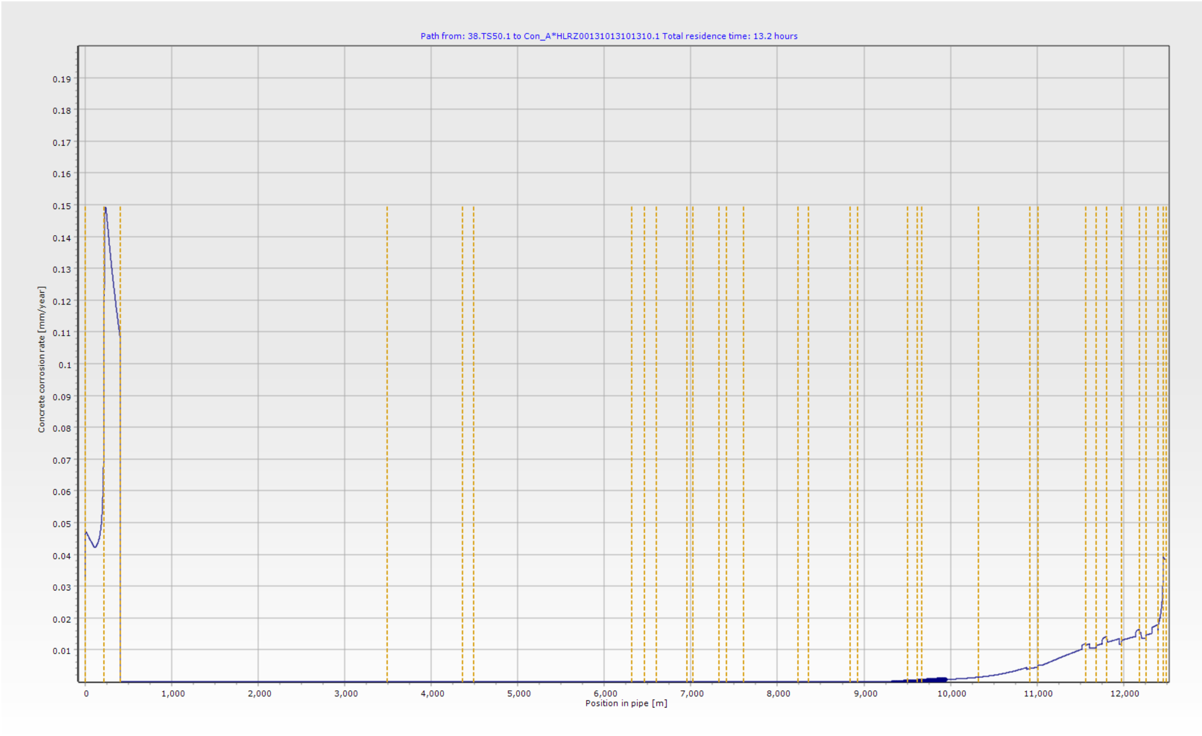

Figure A5. Concrete corrosion rates [mm/year] in Riool Zuid downstream pipeline after implementing the Sewer Mining Technology in de Meren with a wastewater extraction of 60% and with NO_3-N dosing.

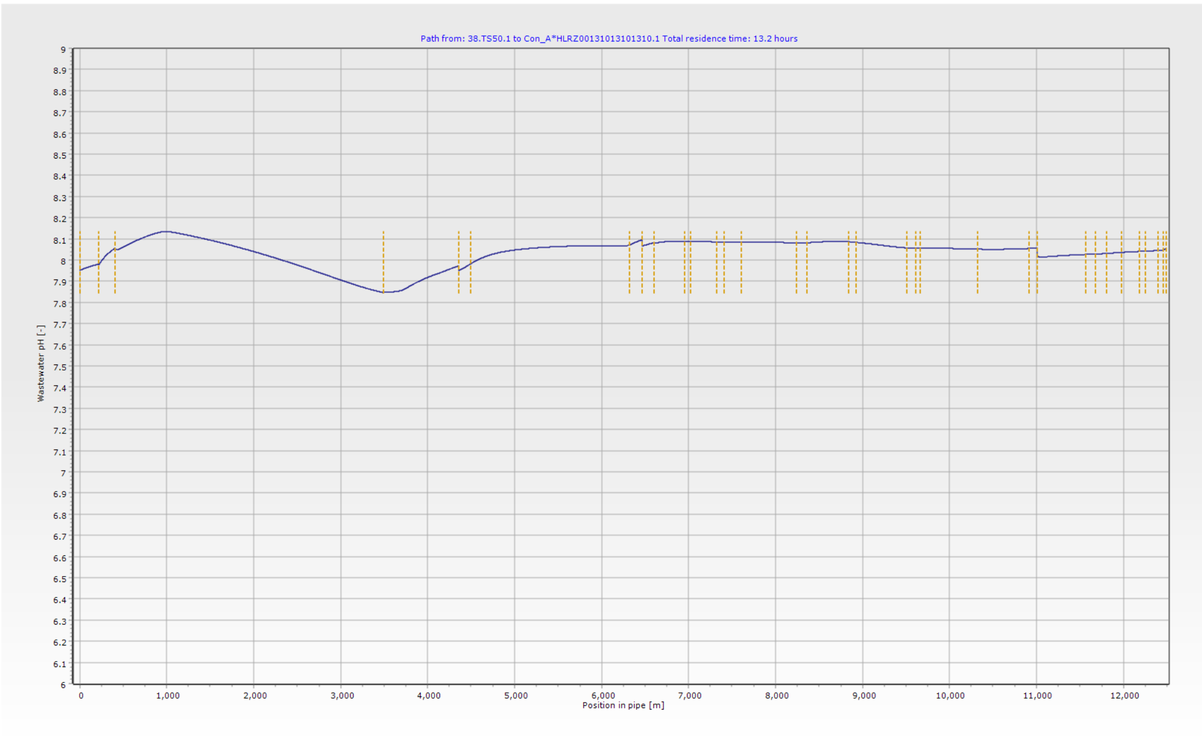

Figure A6. pH in Riool Zuid downstream pipeline after implementing the Sewer Mining Technology in de Meren with a wastewater extraction of 60% without chemical dosing.

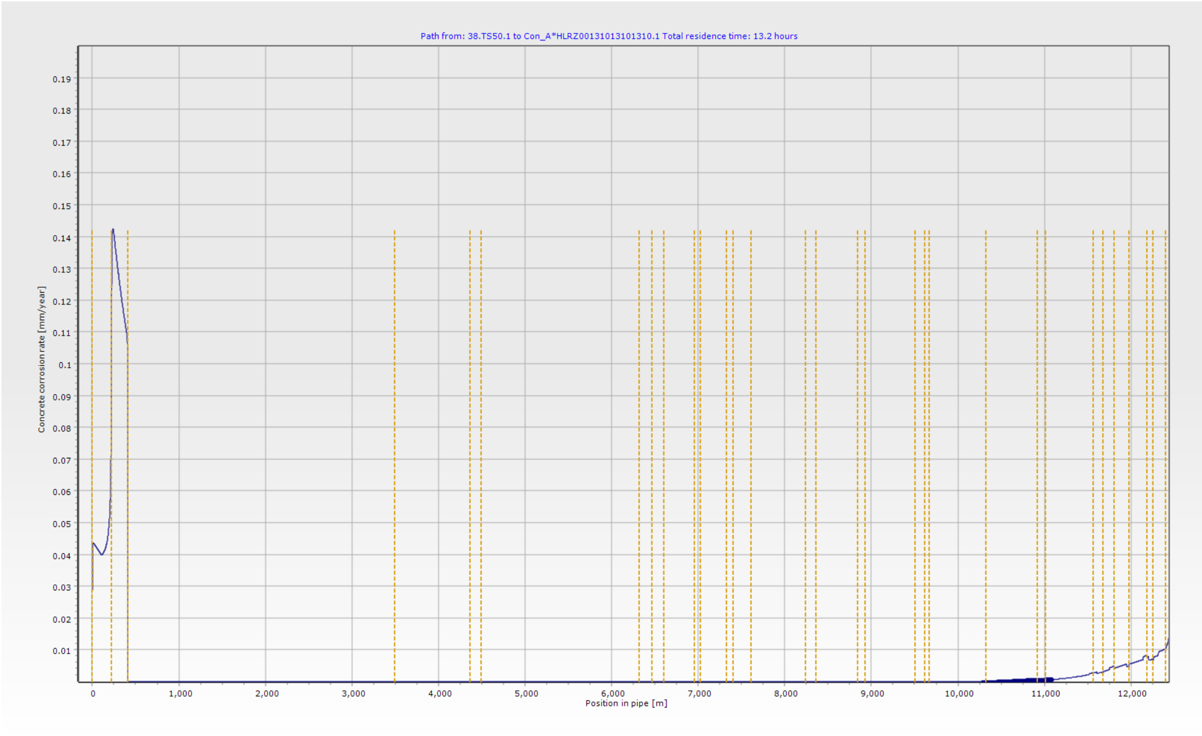

Figure A7. Concrete corrosion rates [mm/year] in Riool Zuid downstream pipeline after implementing the Sewer Mining Technology in de Meren with a wastewater extraction of 40% and with NO_3-N dosing.

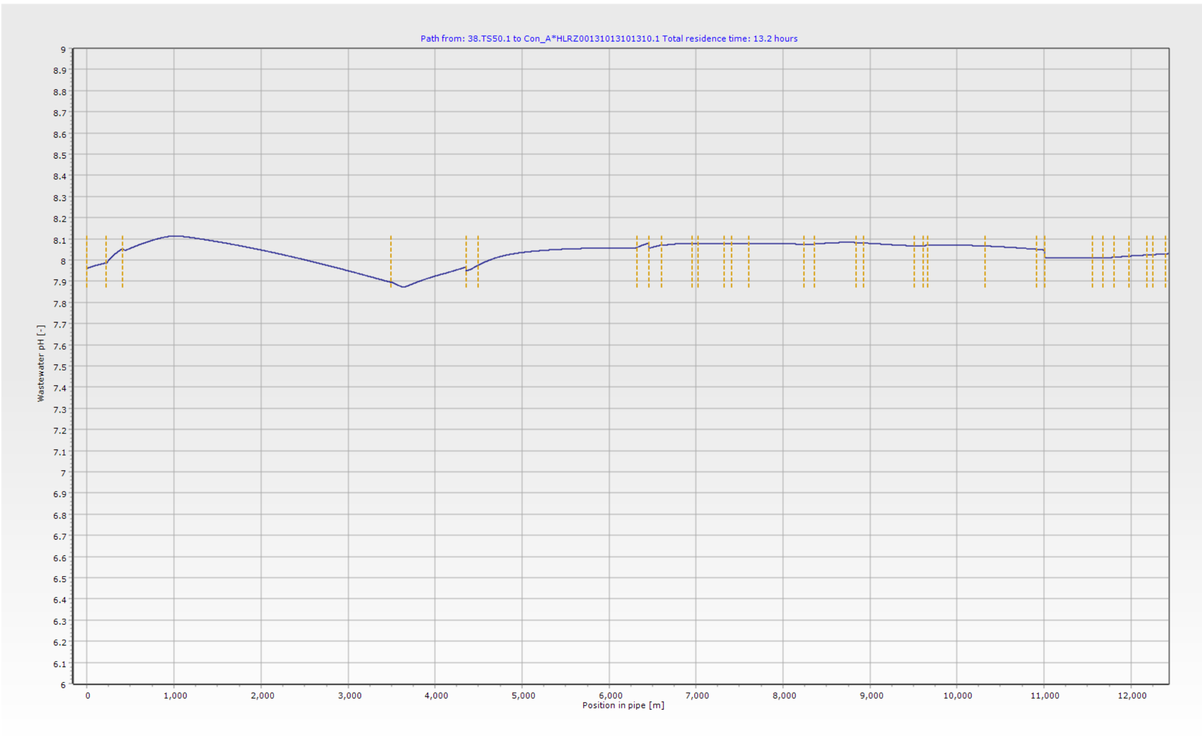

Figure A8. pH in Riool Zuid downstream pipeline after implementing the Sewer Mining Technology in de Meren with a wastewater extraction of 40% without chemical dosing.