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Modeling of Water Age in the Drinking Water Distribution System of Trondheim Kommune

Masteroppgave i vann og miljøteknikk
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Preface

This master's thesis was written during the spring semester of 2020, and completes my Master of Science in Water and Environmental Technology at the Norwegian University of Science and Technology (NTNU). It counts for 30 ETCS credits and has the subject code TVM 4905 Water and Wastewater Engineering.

The main objective of this thesis was to provide estimates of water age in the drinking water distribution system (DWDS) of Trondheim kommune, and to assess the accuracy of model estimates by conducting a tracer study. Due to restrictions created by the pandemic situation, the scope of the thesis was adjusted so that it could be completed without performing any field work whatsoever. This resulted in additional hydraulic and statistical analyses. Luckily, after substantial cooperative efforts, as well as thorough health, safety and environment (HSE) planning, it became possible to conduct the tracer study three weeks before deadline. Due to this, the thesis became broader than planned, which may have limited the depth of some analyses.

I thank my supervisors Marius, Michael and Cynthia for excellent follow-up, guidance and help during the past two semesters. The Matlab and R scripts made by Marius and Michael, respectively, have been very important for this work. I also thank Tone, Endre, Charuka, Trine and Thuat for the effort they put into making measurement equipment available for field work during a challenging spring with pandemic-related restrictions. Furthermore, I thank Tore, Ole Martin, Arve, Geir and other co-workers at VIVA for a great cooperation on the tracer study. Lastly, I thank Odd Atle, Rannveig and the rest of Trondheim Bydrift, for making the water network model available for this research, and for their consistent willingness to cooperate with students and researchers.

I hereby declare that there are no external or additional resources that have not been referenced within the document.

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Abstract

Drinking water provision is fundamental to human activities, and has major direct and indirect effects on society, economy and the environment. Drinking water quality is affected by raw water quality, as well as processes and incidents occurring in treatment and distribution. Using digital water network models, a wide range of investigations can be made to a drinking water distribution system (DWDS), with no risk for water users and no disruption of system operation. Water age is considered as an indicator that captures all system-specific degradation of water quality, and it is also one of the most easily derived results from water network model simulations. The main purpose of this Master's thesis was to provide estimates of the water age distribution in Trondheim kommune. This was done using EPANET, Matlab, and the water network model (WNM) of Trondheim kommune. Furthermore, the accuracy of model estimates has been assessed by conducting a chemical tracer study using sodium chloride (NaCl). Microbial water sample data was compared with mean water age values from a 120-day model simulation, and a statistical test was used to study the effect of site and season on heterotrophic plate count (HPC) data. Results from the tracer study showed a large difference between simulated and measured tracer peak arrival times. Statistical tests indicated a strong positive correlation (Pearson's $R = 0.78$) between average water age and HPC counts, and that these counts are significantly affected by site and season. A clear implication, based on results from the tracer study, is that the WNM of Trondheim kommune requires further calibration if it is intended for use in future water quality modeling.

Keywords

Water age, water network modeling, water quality, EPANET, drinking water distribution, tracer study

Sammendrag

Drikkevannsforsyning er helt grunnleggende for menneskelig aktivitet, og innvirker sterkt i samfunn, økonomi og miljø. Drikkevannskvalitet påvirkes av råvannet, vannbehandling, samt prosesser og hendelser ute på vannettet. Gjennom bruk av digitale vannettsmodeller kan et vidt spekter av analyser utføres på vannettet, uten risiko for abonnenter, og uten forstyrrelser i driften. Én av disse analysene går på vannalder, eller kumulativ hydraulisk oppholdstid. Vannalderen er interessant fordi den enkelt kan estimeres ved hjelp av modeller, samt at den fungerer som indikator for mange prosesser i vannettet som påvirker vannkvalitet. Hovedformålet med denne oppgaven er å gi estimater av vannalderfordelingen i Trondheim kommunes vannett, samt å tallfeste nøyaktigheten til disse. Dette har blitt utført ved bruk av programmene EPANET og Matlab, samt vannettsmodellen til Trondheim kommune. I tillegg har det, for første gang i Trondheim kommune, blitt utført en fullskala "tracer-studie" med natriumklorid (NaCl). Resultater fra studien viser et stort avvik mellom den simulerte og målte utbredelsen til traceren i vannettet. I tillegg gir studien empiriske data om vannets reisetid mellom vannbehandlingsanlegget og flere punkter i nettet, som igjen kan benyttes til fremtidig kalibrering av modellen. Videre har statistiske metoder blitt brukt til å teste korrelasjonen mellom vannalder og kimtall, som var signifikant og positiv ved de studerte vannprøvepunktene (Pearsons $R = 0.78$). I tillegg ble det vist at kimtall påvirkes betydelig av årstid, samt beliggenheten til vannprøvepunktet. Basert på resultatene i tracer-studien, og dersom vannettsmodellen skal benyttes til vannkvalitetsanalyser i fremtiden, anbefales det at en ny kalibrering blir gjennomført.

Nøkkelord

Vannalder, vannettsmodellering, vannkvalitet, EPANET, drikkevannsdistribusjon, tracer-studie

Contents

List of Abbreviations

CFU colony-forming units

DBP disinfectant by-product

DMP demand multiplier pattern

DWDS drinking water distribution system

DWTP drinking water treatment plant

EPS extended period simulation

HPC heterotrophic plate count

MLE maximum likelihood estimation

NOM natural organic matter

PRV pressure reducing valve

WNM water network model

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1 Introduction

The sustainable development goals (SDGs) issued by the United Nations in 2015, are to a great extent connected with drinking water and urban water infrastructure [51]. It can be argued that all of the 17 goals are water-related, but only a few have been selected here: Goal 3, good health and well-being, cannot be achieved without providing people with safe drinking water. Goal 6, clean water and sanitation, requires well-working water systems and safe raw water sources. Goal 9, industry, innovation and infrastructure, requires planning with a lifetime perspective, and that new technologies are implemented in the production and management of technical systems [51, 29]. One example of such technology is the digital water network model (WNM), and there is a growing interest in the international water sector for modeling of drinking water distribution system (DWDS) water quality [4]. Models are especially important for DWDSs due to their complex topology, frequent growth and change, and sheer size. It is not uncommon for a system to supply hundreds of thousands of people, so the potential impact of water utility decisions can be large [22]. In Norway, after the contamination incidents at Askøy in the summer of 2019, drinking water quality issues have been high up on the public agenda. Norwegian municipalities are now working to address the increased focus on distribution system water quality [6]. There are high costs related to improvements in infrastructure, and the Norwegian Consulting Engineers' Association (RIF) are providing annual estimates of this in their "State of The Nation" reports. In 2019, it was estimated that the cost of necessary upgrades in Norwegian drinking water infrastructure from now until 2040, is approximately 220 billion (220×10^9) Norwegian kroner [29].

Trondheim Bydrift is already using a WNM for assessing both hydraulic capacity and drinking water quality aspects. The DWDS of Trondheim kommune has also been the subject of considerable research activity connected with water quality issues. Three studies were published in 2018-2019, all addressing microbiological conditions, but with a differing scope. The first study, by Waak et al. [55], compared the occurrence of *Legionella spp.* and bacteria in the DWDS of Trondheim with that of a DWDS in the United States. The main point of comparison was that no residual disinfection is maintained in Trondheim, while a chloramine residual is maintained in the U.S. DWDS. It was found that *Legionella* occurred frequently in biofilm at the sampling sites in Trondheim, whereas no *Legionella* was detected in the biofilm samples from the U.S. DWDS [55]. In the Norwegian drinking water regulations [15], it is not required to monitor *Legionella*, but it may still present a public health concern [55]. In 2008, Bruaset [6] conducted a research project aimed at modeling selected water quality aspects in the DWDS of Trondheim kommune using EPANET, a well-known water network modeling tool. The objective was to study the causes of drinking water quality changes in the DWDS, as well as how these can be prevented [6]. The WNM simulations resulted in graphs indicating the development and spatial distribution of corrosion and biofilm in the DWDS. A calibration approach for the biofilm model was suggested, but not conducted [6].

For ongoing and future biological water quality research in Trondheim kommune DWDS, model estimations of the travel time and water age in the system can be a useful contribution. Using EPANET, this thesis provided estimates of the water age distribution in the whole DWDS. These age estimates were compared with historical heterotrophic plate count (HPC) data, to study if there was a correlation between water age and HPC counts. Additional statistical tests were performed to study the effects that seasonal variations, as well as the spatial distribution of sampling sites, have on the HPC counts. Furthermore, a Matlab® script was used to identify a set of water pipes in the WNM that may be extra suitable for biological sampling due to unique predicted flow conditions. Hem and Østerhus [16] assessed that models will become increasingly important in the prediction of how water quality incidents develop in the DWDS, but the value of these predictions is reduced if the accuracy of the model is unknown. In this thesis, and for the first time in Trondheim, an accuracy assessment was made of the WNM by conducting a tracer study and comparing the observations with model predictions.

1.1 Objectives and research questions

The main objective in this master's thesis is to provide estimates of the water age distribution in the DWDS of Trondheim kommune, and to assess the accuracy of model estimates by conducting a tracer study. An additional objective is to use the WNM to identify water pipes that can be especially relevant for future biological research. These are the research questions:

1. Why is it relevant to perform drinking water quality modeling in the DWDS of Trondheim kommune?
2. How is water age distributed in this DWDS, and what is the reliability of these estimates?
3. How does water age relate to microbiology in the DWDS?
4. Which water pipes should be targeted for biological sampling in the future?

1.2 Structure of the thesis

The thesis is structured with an initial literature review that establishes a basis needed for studying the production and distribution of drinking water in a Norwegian municipality, as well as water quality considerations and water network modeling. After that, the geographical area under study, including the DWDS, is presented. Only publicly available information has been used in these first sections. Further, the methods used in modeling, field work, and statistical analyses, are described. Finally, results are presented and discussed, before the thesis ends with some concluding remarks.

2 Literature review

Running a water network model (WNM) and getting results from it is not a particularly demanding task for someone who is experienced in using computers. However, building an accurate model, calibrating it well, and producing reliable and useful results will require a substantial background knowledge. If the model is also intended for use in assessing the aspects of drinking water quality and drinking water distribution system (DWDS) operation, knowledge is required from the fields of water chemistry, microbiology, urban water systems, statistics, and others. The following literature review is aimed at providing background information on these areas, comprehensive enough to address the research questions and the objectives presented in this thesis. Furthermore, it presents elements from reports and publications that form a scientific and regulatory context.

2.1 Drinking water quality

On the global scale, diseases related to contaminated drinking water and poor water quality put a major burden on human health [25]. This is one of the areas where differences in development between countries become especially visible. Maintaining a satisfactory drinking water quality is key in ensuring a low health risk for the water consumer. Water quality models are used as tools for investigating several quality parameters and processes occurring in a DWDS. The World Health Organization (WHO) [25] has listed the following general elements in drinking water quality:

- Microbial aspects
- Disinfection
- Chemical aspects
- Radiological aspects
- Acceptability aspects: taste, odor and appearance

The Norwegian drinking water quality regulations (Drikkevannsforskriften) are based on the requirements set by the EU directive 98/83/EF [39], which was updated and altered in 2015. All the previously listed aspects are dealt with in the Norwegian regulations [15], and will be further explained in the following subsections. Some of them only in brief, and others in greater detail due to their relevance to this thesis.

2.1.1 Microbial aspects

Microbes are single-celled organisms which are not visible to the human eye due to their small size. The main taxonomic groups are bacteria, archaea, viruses, protozoa, fungi and algae. In practice, some multi-cellular organisms are also counted as microbes during early stages of their lives [42]. Pathogens are defined as microorganisms that pose a risk to human health and may cause disease. Several of these occur naturally in water and soil, and several others originate from human or animal sources [33]. The majority of microorganisms found in water are normally not pathogenic and play important roles in maintaining ecological balance [24].

Microorganisms can exist in DWDSs either suspended in the bulk flow, in sediments and corrosion products, or as biofilm attached to different surfaces [3, 7]. Biofilm exists in all DWDSs [7], and can cover surfaces in multiple layers or in limited thin patches of colonies. The growth surfaces are typically the walls of pipes and tanks, sediments, and suspended particles [10]. According to Flemming et al. [10], more than 95 % of the microbiome in a DWDS is normally found in the biofilm, due to a high surface-to-volume ratio and harsh environmental conditions in the water flow. The four most important factors affecting biofilm formation are [7]:

- Access to biodegradable material and nutrients
- Water temperature
- Hydraulics and water demand variations
- Corrosion rates

The microorganisms in biofilm consume biodegradable material that is supplied by the drinking water. Both biodegradable dissolved organic carbon (BDOC) and assimilable organic carbon (AOC) are parameters which indicate the potential for biofilm growth [7]. Normally, the consumption by biofilm reduces the available nutrition in drinking water as the water becomes older, and the bio-stability increases over time [7]. In this way, the drinking water may become increasingly bio-stable further out in the peripheral areas of the system [7]. This may not be the case in DWDSs that maintain a residual disinfectant. According to Bruaset and Hem [7], due to the suppressing effect that residual disinfection has on microbial activity, the biofilm growth will be very limited until the disinfectant concentration drops. A possible consequence of this is that areas of a DWDS, with a high water age and low concentration of the residual, experience elevated growth rates of biofilm and bacteria [7].

One problematic property of biofilm is that it can aid in pathogen survival, because they can get adsorbed, protected and in some cases grow there [7]. Another problem is that some microorganisms found in biofilm can oxidize or reduce iron and sulfur compounds, potentially creating local corrosion attacks on pipes. The consequences of this are mainly structural, but also the aesthetic quality of drinking water can be impacted such as when the drinking water becomes brown due to large amounts of iron oxide (rust) [7]. The aesthetic aspects are dealt with later in the subsection about acceptability aspects.

Heterotrophic plate count (HPC) (ISO 6222:1999), named "kimtall" in Norwegian, is defined as the number of colony-forming units (CFU) observed in 2 mL of water, spread on a nutrient agar plate, and incubated at 22 °C for 3 days. HPC has been widely used in the water industry worldwide for over a century [3]. It detects a wide spectrum of heterotrophic microbes, including fungi and bacteria, and measures their ability to grow on rich growth media under a given temperature and incubation time [25]. This type of analysis is principally simple, does not require advanced training or facilities, and results are directly available in relatively short time [25]. HPC has been shown to have little value as indicator for pathogens but is still considered useful in microbial water quality monitoring [25]. According to WHO [25] and Blokker et al. [3], it serves to indicate:

- The effectiveness of water treatment and disinfection
- The integrity and cleanliness of the DWDS
- Presence and expansion of biofilm
- Microbial consequences of unplanned events within the DWDS

A general operational goal in drinking water supply is to maintain low HPC numbers throughout the DWDS. Stable low numbers indicate that there is little abnormal microbial activity taking place, and high numbers may indicate potential water quality problems. Still, as stated by WHO [25], there is no evidence of association between gastrointestinal illness and the ingestion of drinking water containing HPC organisms.

2.1.2 Disinfection

The main goal of drinking water disinfection is to inactivate pathogenic microorganisms, and it is very commonly performed using chlorine compounds [25]. Other disinfection technologies in use are, for instance, ozonation and UV irradiation. Disinfection can be performed both during and after a drinking water treatment plant (DWTP) process. Post-DWTP disinfection is often performed using a residual disinfectant such as chlorine or chloramine, with the aims of providing persistent protection against low-level contamination and reducing growth of microorganisms throughout the DWDS [25]. WNMs are often used to study the chlorine residual decay in DWDSs [4]. In addition to possible taste and odor, one drawback of using chemical disinfectants and maintaining a residual disinfectant is the possibility of disinfectant by-product (DBP) formation. These compounds are known to have possible negative health effects on water consumers [25]. Still, as it is written in the drinking water quality guidelines of WHO [25], "*Disinfection should not be compromised in attempting to control disinfection by-products.*" The perspective on this statement will vary depending on context. For instance, in the Netherlands, which is a highly developed industrialized country, the public opinion has been that chemical disinfection provides more problems than benefits, and that other methods should be used to ensure the hygienic safety of drinking water [36].

2.1.3 Chemical aspects

Water quality models can be used to study chemical aspects in both raw water sources and DWDSs. Regarding health concerns arising from water chemistry, adverse effects are often due to long term exposure. On the other hand, substances such as nitrate (NO_3^-) can have more immediate negative impacts on health [25]. According to WHO [25], out of all the chemicals that may end up in drinking water, only a small proportion can be considered as pollutants with adverse health effects. In addition, there are many inorganic chemical water constituents that are essential parts of human nutrition, for which maximum guideline values have been established. This is because prolonged exposure to high concentrations, even though these constituents are considered essential, can be harmful [25]. In Table 1, an overview is given for important sources of health-relevant organic and inorganic chemical water constituents.

Among inorganic water constituents, there are many dissolved salts and heavy metals that are commonly monitored and have regulated maximum values. Organic man-made compounds such as pharmaceuticals, petroleum

Table 1: Categories and sources of chemical water constituents. From WHO [25] (modified)

Categories	Examples of sources
Naturally occurring	Soils, rocks, eutrophication in water bodies
Industrial and residential	Mining, manufacturing, sewage, solid waste, urban runoff, spills
Agriculture	Manures, fertilizers, pesticides, herbicides, fungicides
Chemicals used in water source protection	Larvicides

distillates, detergents, phthalates, fluorinated hydrocarbons and many more have become increasingly relevant over time due to their widespread use, and are also monitored [25, 11].

As mentioned in the previous subsection, the use of chemical disinfectants is associated with the formation of organic and inorganic DBPs [25]. According to WHO [25], the major DBPs formed during chlorination are trihalometanes, haloacetic acid, haloketones and haloacetonitriles. These can be formed when chlorine compounds react with natural organic matter (NOM) in the water. The NOM molecules can therefore be seen as precursors of DBP molecules [25]. This explains why NOM-reducing measures, in connection with both the raw water source and pre-treatment, is an effective strategy for lowering the DBP levels. According to WHO, other basic strategies include:

- Changing process conditions (e.g. pH during chlorination)
- Using a disinfectant with lower DBP formation potential
- Using disinfection methods with little or no reactivity (e.g. UV)
- Removing DBPs prior to distribution

2.1.4 Acceptability aspects

The most important acceptability criteria concerning drinking water quality are taste, odor and appearance. These affect consumer perception about water quality, which may differ from risk and safety. Discoloration of drinking water is one example of a situation where the water consumer may judge that the water is unsafe, even though it may have no direct consequences to health. Drinking water producers can therefore be affected, because the trust of consumers is dependent not only on the hygienic safety, but also on taste, odor and aesthetic aspects [25].

Taste and odor in drinking water can be caused by organic and inorganic chemical compounds, as well as biological processes. It may develop during distribution and storage due to microbial activity, but corrosion and leaching in pipe materials is also relevant. Elevated turbidity or color may be caused by changes in raw water quality. If an abnormal situation occurs in a DWTP or in a DWDS, such as a malfunction in dosing equipment or a rapid change in flow direction, it is likely that both appearance, taste and odor can be affected [25]. A WNM can be a useful tool in investigating such events.

2.2 Drinking water production and distribution

The water network modeling conducted in this thesis has required a theoretical basis for drinking water production and distribution. A modern public drinking water supply is expected to deliver sufficient amounts of hygienically safe, clean and appetizing water to the customer at all times, with adequate pressure [24]. In order to achieve the required functions, it is necessary to have DWTPs and DWDSs that are properly designed, operated and maintained. A DWTP can be defined as the technical facility that performs treatment of raw water, and delivers treated drinking water [21]. A DWDS can be defined as the technical facilities that receive drinking water from the DWTP and supply it to points where the water can be extracted [21].

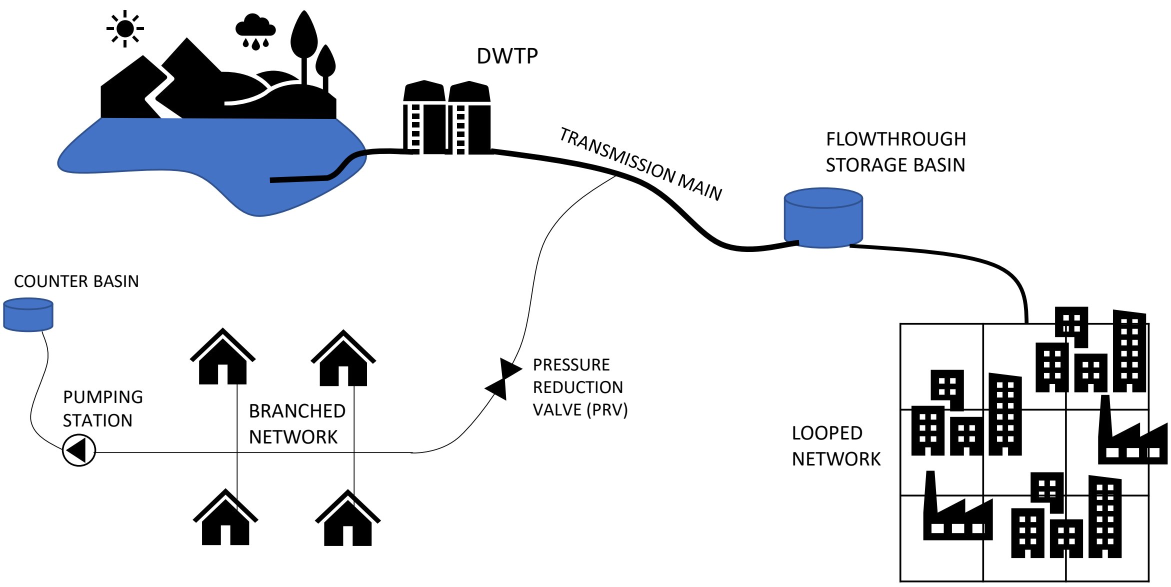


Figure 1: Simplified layout of a DWDS, including selected main components. By Jon K. Rakstang (2020)

2.2.1 Drinking water production

The majority of public DWTPs in Norway extract raw water from surface sources, and mainly from freshwater lakes. Approximately 80 % of the total annual volume of produced drinking water is extracted from lakes, 10 % from rivers and streams and 10 % from groundwater reservoirs [24]. A well-known issue connected with raw water from surface waters in Norway is the level of NOM, such as humic substances, which can give a brown color to the drinking water and may create other water quality issues. The occurrence of NOM is highly dependent on the location of the raw water source, and there exist DWTPs both with and without NOM removal in Norway [24]. As an example, the VIVA DWTP in Trondheim kommune does not have NOM removal because the level of NOM in the raw water source has so far not crossed tolerability limits or limit values. Nevertheless, due to expected alterations in the raw water source as an effect of climate change, it has been recommended in the current master plan to build an additional treatment step. This will most likely be a coagulation/filtration process, which has NOM removal as one of its main functions [44].

The Norwegian drinking water regulation [15] demands that there are sufficient hygienic barriers in place at all public DWTPs. This is to ensure that the drinking water is free from viruses, parasites, bacteria, or other organisms and compounds, which present a health risk due to their number or concentration. In addition, they are there to ensure that acceptability criteria (taste, odor, appearance) are fulfilled [21]. There are two main categories of hygienic barriers in this context. One category is the raw water source and its watershed, and the other is water treatment [11]. The Norwegian national institute of public health, Folkehelseinstituttet (FHI), has published a report [11] that assesses hygienic barriers.

Table 2 contains the key factors that are needed for assessing the raw water source. This alone is not enough to judge whether it is a hygienic barrier, because it needs to be seen in connection with the watershed [11]. For instance, there have been several cases in Norway where water intakes in lakes at depths of more than 100 m have been exposed to pollution from activities within the watershed [11].

Table 2: Factors that need to be considered when assessing if a raw water source counts as a hygienic barrier. From FHI [11] (modified)

Source type	Assessment criteria	Remarks
Lake	Depth, volume, residence time	Great depth above intake protects, large total volume dilutes pollution, long residence time facilitates pollutant breakdown
River	Flow volume	Large flow volume dilutes pollution
Groundwater, soil aquifers	Distribution of saturated/unsaturated zones, filtration effects	Residence time at least 60 days, estimate should be well-documented
Groundwater, rock aquifers	Depth, fracturing, water flow, unsaturated zone above rock	All factors have to be assessed, and combined with records that indicate the stability over time

The total assessment of whether the raw water source and its watershed is a hygienic barrier must, according to FHI [11], take into account the following:

- The potential of pollution within the watershed
- The size and other characteristics of the watershed
- The size and other characteristics of the raw water source
- Intake depth
- Climatic conditions
- Water quality analyses
- Acceptable level of risk

The other category of hygienic barriers, water treatment, is mainly focused on preventing exposure of water consumers to pathogens. The treatment technologies that can be hygienic barriers, either stand-alone or combined, are based on chlorination, UV-irradiation, ozonation, membrane filtration, and coagulation/filtration [21]. For each technology it is specified which process design parameters are required in order to document the function as a hygienic barrier. In general, bacteria and viruses need to be inactivated with a $\geq$99.9 % efficiency, and for parasites it is required to have $\geq$99 % reduction [21].

Apart from ensuring the hygienic safety, a DWTP should deliver water that is not excessively corrosive [15]. Effective corrosion control can be implemented in DWTPs and give lower corrosion rates in the DWDS. In addition to increasing the service life of pipes and valves, low corrosion rates give a lower growth potential for microorganisms. This is because the sediments and coatings created by corrosion products serve as growth surfaces for biofilm [7].

As mentioned in the subsection about microbial aspects, access to biodegradable material and nutrients is an important limiting factor for the growth rate of biofilm. If needed, effective water treatment that removes a large share of these compounds can reduce the growth potential in the DWDS greatly [7].

2.2.2 Drinking water distribution

The layout and complexity of a DWDS depends on several local factors. This subsection will mainly deal with what is relevant for larger public Norwegian DWDSs. The main function that a DWDS should fulfill is to transport and deliver drinking water, meeting all requirements for safety, capacity and quality [11]. Large mains or tunnels in the transmission system, with a large cross-sectional area, typically transfer raw water to the DWTP before transferring treated drinking water to one or more large storage facilities at a relatively stable flow rate [11]. The mains of the distribution system can tap drinking water from storage facilities, in some cases also the transmission system, and here the flow rate and flow direction can vary more, depending on the water consumption throughout the day [11]. The distribution mains make up the core structure of the distribution networks, which are either branched or looped. Every building is connected to the local network through a connection pipe, which supplies the in-house distribution system. In general, the local water utility is not responsible for in-house distribution systems. Pumping stations may be required in the DWDS, depending on local topography, and supply the necessary pressure for the transport of water when gravity alone cannot achieve this. In addition, it is common to have installations which reduce water pressure, such as pressure reducing valves (PRVs), in places where it would become too high without. Numerous manholes and valves serve as points of access and control throughout the entire system [11].

Storage basins, sometimes also referred to as equalizing basins, have mainly three functions, as described by Lindholm et al. [19]. The primary function is to even out pressure and demand variations in the supply area, especially during peak hour consumption, by having approximately 30-35% of the daily consumption volume available for tapping. Furthermore, the basin functions as a reserve for fire extinguishing water. In the event of large pipe failures or repairs in the DWDS, the basin should also provide a reserve volume equivalent to between 0.5 and 2 days of normal consumption [19]. The design of a storage basin should, according to Lindholm et al. [19], include at least two separate chambers to allow for cleaning and maintenance. The inflow and outflow should also be placed in a way that eliminates "dead zones" where the water becomes stagnant, which is problematic with regard to water quality [19].

There are several different materials available for use in drinking water pipes. The choice of material is commonly based on pressure conditions, pipe diameter, risk of corrosion and various safety considerations [19]. Cast iron, firstly gray and nowadays ductile, has been widely used for more than a century and can withstand high loads and pressures. If appropriate protection is not in place, this pipe material is known to be particularly exposed to corrosion attacks. Plastics such as PVC and PE, typically have a higher resistance against corrosion. The pipes are also light and more flexible, and are widely used in DWDSs [19]. One issue with plastic pipes is that certain chemical compounds, for instance petrochemicals from contaminated soils, can be transported from the outside, through the pipe wall, and into the drinking water. Another issue is that biodegradable organic additives can leach out from the plastic pipe materials and lead to higher microbial growth [16]. As an example, rubber gaskets in plastic pipe joints have been shown to cause significant local microbial growth [7]. Another alternative pipe material is GRP (glass-fiber reinforced plastic). Cement-based pipes, such as asbestos-cement or concrete pipes, are not widely used for DWDSs today [24].

The difference in design between branched and looped pipe networks is illustrated in Figure 1. As opposed to the branched type, looped pipe networks ensure that there are always two alternative supply directions to one connection point [19]. This means that the drinking water supply can usually remain in full service, even if pipe failure or maintenance puts a water main out of operation. For branched networks, such events have a higher probability of cutting off the drinking water supply to customers [19]. Another benefit from the looped configuration is that the fire flow capacity can be achieved with smaller pipe diameters than in a branched system, due to the two-sided supply. Lindholm et al. [19] also described a looped configuration as advantageous in terms of water quality because the water can circulate. This may not always be the case, and Blokker et al. [2] specifically pointed out that the risk of discoloration of water due to rapid re-suspension of built-up sediment can be higher in looped networks than branched ones. This was shown to be connected with the flow velocities, which can be lower in conventional looped systems than conventional branched ones, and govern the degree of self-cleaning of the pipes [2].

In Norway, due to national regulations, it is required that there is sufficient local capacity in the DWDS for extinguishing fires using drinking water. In rural and residential areas it is required that the DWDS can deliver at least $20\,\mathrm{L\,s^{-1}}$. In dense urban areas or industrial areas a minimum capacity of $50\,\mathrm{L\,s^{-1}}$ is required [24]. One issue connected with fire fighting capacity is the potentially large impact on DWDS sizing. In dense urban areas the impact may be very small due to generally large system dimensions. On the other hand, in areas with scattered settlements and low water demand the fire fighting capacity can require a large increase in system dimensions compared to the dimensions required for normal consumption only [1]. One of the consequences of increasing pipe dimensions is that the volume per kilometer of pipe will go up. A 2x increase in diameter gives a 4x increase in the cross-sectional area of the pipe, given that the wall thickness is the same. This means that doubling the diameter of a water main to provide sufficient fire flow capacity will make the volume of water per kilometer of main roughly four times as high. This will increase the travel time, and water consumers will receive drinking water with a higher age.

2.3 Water age and distribution system water quality

Water age, in the context of drinking water distribution, can be defined as the cumulative hydraulic residence time of drinking water. In a WNM, this is the time it takes for a modeled water "package" to be transported from the DWTP to a given point in the DWDS. When this value is averaged for all the water packages within a given section or point in the DWDS, the average water age can be obtained. Typical values for the average water age can range from hours up to several weeks, depending on local conditions. In parts of a DWDS that are far away from the DWTP, in long pipe sections with low flow, in storage tanks with poor mixing, or in dead-ends on pipes, the water age will typically become high [56]. The relevance of water age to water quality comes from the time-dependency of many processes occurring within a DWDS, that affect water quality parameters. Blokker et al. [3] wrote that water age is considered to be an indicator that captures all system-specific degradation of water quality. According to Machell et al. [20], water quality is to some extent a function of water age. This functional relationship can be especially visible in DWDSs that are operated with a residual disinfectant. The decay of disinfectants and the formation of DBPs is time-dependent, but the rate at which it happens is affected by several parameters including temperature and NOM [3]. In Norwegian DWDSs a residual disinfectant is generally not maintained, but in several other countries including the United States it generally is. Table 3 is adopted from an American white paper issued by USEPA (United States Environmental Protection Agency) and summarized water quality problems associated with a high water age [1]. Note that the problems related to residual disinfection can be considered less relevant for Norwegian drinking water supply.

Table 3: Drinking water quality issues associated with high water age. From USEPA [1] (modified)

Chemical issues	Biological issues	Physical issues
DBP formation	DBP biodegradation	Temperature increase
Disinfectant decay	Nitrification	Sediment deposition
Corrosion control effectiveness	Microbial regrowth, recovery or shielding	Color
Taste and odor	Taste and odor	

It is an oversimplification to say that sediment deposition is directly related to water age. That issue can be avoided by adding more emphasis on the hydraulic effects that lead to a rapid increase in age, such as low flow velocity, which is known to cause sediment build-up. Blokker et al. [3] also showed that there is not necessarily any positive correlation between water age and temperature. Since temperature conditions vary depending on the type of climate a DWDS is situated in, the temperature statement in Table 3 can still be valid.

Water age is not regulated in the Norwegian drinking water regulations (Drikkevannsforskriften) [15], and there are also no assessment criteria concerning water age in the local water construction guidelines for Trondheim kommune [47]. On the other hand, both sources recommended that the hydraulic residence time is kept low, and that stagnation over time in drinking water pipes should be avoided. A study by Machell et al. [20] in the United Kingdom stated a maximum threshold value for water age at 120 hours, or 5 days, but the DWDS in the study maintained a residual disinfectant, which is not the case in Trondheim. Furthermore, a consulting company named DHI has issued a brochure where a water age between 3 and 7 days was considered excessive, and more than 7 days was unacceptable [8]. Once again it was evident that the brochure is aimed at DWDSs where a residual disinfectant is maintained, and the threshold values are therefore not applicable in Trondheim. If, on the other hand, Trondheim kommune would activate residual disinfection during an emergency situation, threshold values become more interesting.

2.3.1 Measures for reducing the water age

There are several different approaches that can be used to reduce water age in a DWDS. Altering the mixing conditions in pipe junctions is one approach, where for instance Machell et al. [20] have suggested valve-throttling as a simple and cheap measure for reducing the maximum age in particular. When attempting this, it is important that careful considerations are made to ensure that the valve throttling does not compromise the fire fighting capacity or lead to excessive water age in other parts of the DWDS. If a water utility is planning to do renovation in an area of the DWDS where water age is an issue, the choice of new pipe diameters and looped vs branched configuration offers a possibility to reduce the water age. For instance, in the water construction guidelines of both Oslo kommune and Trondheim kommune, it is recommended to construct mainly looped systems, partly because it is expected to reduce the hydraulic residence time [27, 47]. It is unclear whether that statement will always be realistic, because of the flow reversals and larger volume commonly found in looped systems. On the other hand the occurrence of "dead ends" is lower in a looped system, which partly eliminates pipe segments where water is completely stagnant over longer periods of time. A third, principally simple approach, is to have a constant tapping of water that is discharged directly into receiving waters or a stormwater system. This will of course lead to water loss, but may be a suitable measure for pipes where stagnant water is a constant problem. A fourth approach is to adjust the operation of storage basins in order to reduce water age. Increasing the utilization of basins, for instance by changing settings in pumping stations, can be effective. Ensuring good mixing in storage basins can also make a large difference [1].

2.4 Digital modeling

Real-world systems can be represented in a simplified, mathematical version, using digital models [56]. Well-defined system boundaries can be used to isolate parts of the natural or artificial environment, so that they can be studied mathematically [30]. In order to create well-functioning models, all significant interactions and objects connected with the system under study need to be included [14]. Real-world systems may be oversimplified by models, leading to inaccurate results, but models are still useful in many applications.

Simulation is a process where the behavior of real-world systems is imitated through the functions of digital models. For instance, if it is impractical to perform experiments on a real-world system, simulation can still be performed. One example of this is when the objective is to investigate how a potential unwanted event occurring in a DWDS can impact drinking water quality. When used here, a digital model can provide useful insight at relatively low cost and without risk to human health. Furthermore, simulations can be used to predict the response of a system under varying conditions and scenarios, without disrupting its operation. For master planning purposes, such simulations can be useful for exploring possible future configuration, alteration or expansion of relevant systems [56].

Calibration is a key task in ensuring the accuracy of digital models. It can be explained as a process where the output of a model is compared with observations from the real-world system, and adjustments are then made to model parameters until the results agree adequately with the observations. Calibration may need to be performed repetitively and it can be automated [56]. According to Walski et al. [56], it is necessary to calibrate WNMs due to the following reasons:

- We seek to have confidence in the accuracy and performance of a model
- Calibration can reveal sensitive parameters and provide new insight into the system
- It can aid in uncovering missing or incorrect data (e.g., incorrect pipe diameters)

Ormsbee and Lingireddy [26] have listed seven basic steps that can be followed in model calibration:

1. Identify intended use of the model
2. Determine initial estimates of the model parameters
3. Collect calibration data
4. Evaluate model results
5. Perform macro-level calibration
6. Perform sensitivity analysis
7. Perform micro-level calibration

Validation can be described as the final step to ensure model performance and that it produces credible results. It can also aid in finding the shortcomings of the model [22]. Like calibration, it can be performed by comparing model results with observations from the real-world system. Most importantly, the observations used in validation can not be the same as the ones used in calibration, and vice versa. The principle way of doing validation is to firstly adjust system demands, initial conditions and operational rules to match the conditions during field data collection [22]. After this, the comparison of predicted versus observed is done, and an assessment of validity can be made.

2.5 Water network modeling

Water network models can approximate the hydrostatic and hydrodynamic behavior of pressurized water pipe networks. The equations used are based on continuity and energy conservation [56]. The goal is to describe the hydraulic state of a water pipe network at a given point in time, and this is done by solving the following flow continuity and headloss equations [56]:

$$\sum_{pipes} Q_i - U = 0 \tag{1}$$

This continuity equation applies for pipes (i) that meet in a junction node, where

Q_i = inflow to a node in the i-th pipe ($\frac{L^3}{T}$)

U = water used at the node ($\frac{L^3}{T}$)

For extended period simulation (EPS) it is necessary to include a water storage term [56]:

$$\sum_{pipes} Q_i - U - \frac{dS}{dt} = 0 \tag{2}$$

where

$\frac{dS}{dt}$ = change in storage ($\frac{L^3}{T}$)

The Bernoulli equation (1738) describes energy conservation, and is written below in terms of head [56]. Energy conservation is, in this case, maintained by ensuring that the sum of headloss is zero through all possible combinations of loops.

$$Z_1 + \frac{P_1}{\gamma} + \frac{V_1^2}{2g} + \sum h_p = Z_2 + \frac{P_2}{\gamma} + \frac{V_2^2}{2g} + \sum h_L + \sum h_m \tag{3}$$

where

Z = elevation (L)

P = pressure ($\frac{M}{L^2T^2}$)

V = velocity ($\frac{L}{T}$)

g = gravitational acceleration constant ($\frac{L}{T^2}$)

h_p = head added at pumps (L)

h_L = head loss in pipes (L)

h_m = head loss due to minor losses (L)

The earliest published systematic and iterative method for solving these equations in larger networks is the Hardy Cross method (1936) [56]. This is one example of a method where a continuity equation must be solved for each node in the network, and an energy equation for each loop. In order to do this, the method requires information about the water demand in each node, and the diameter, length and roughness of each pipe [56].

The information stored in a water network model is mainly based on static data about the physical assets in the DWDS. This data is often compiled from several different sources, including digital infrastructure maps and written records [35]. The accuracy and level of detail in the model is highly dependent on that data. Furthermore a water network model typically contains control rules for pumps, valves etc. and functions that represent variations in water demand [35].

2.5.1 Models as tools in drinking water quality management

WNMs can be useful tools for water utilities in managing the drinking water quality for several reasons. Proactive maintenance is a key area in this respect, because models can be used in the planning of maintenance operations such as pipe flushing. According to Bruaset and Hem [7], this is a type of maintenance operation that is effective in maintaining low levels of sediment deposition and biofilm in the DWDS. For flushing planning, models focused on biofilm formation, corrosion and sedimentation are especially relevant [6]. According to Methods et al. [22], water quality models are also useful in studying the impact of storage basins on water quality.

In the event of local loss of water pressure, there is a risk of contaminated water leaking into the drinking water pipes. WNMs can be used to predict which locations are exposed to this, as was done in Trondheim [44], and the transport of the contamination can also be simulated. In DWDSs with residual disinfection, where water quality is considered strongly connected with the residual concentration, models that simulate chemical reactions or water age are relevant [1]. For any type of DWDS, a WNM can be a useful tool for determining the placement of sensors, for instance measuring turbidity or residual chlorine concentration [16].

2.6 Calibration approaches

As stated by Ormsbee and Lingireddy [26], it is important to firstly state the intended use of a WNM before performing calibration. Network design studies have a different technical focus than water quality studies, which should be considered before selecting a calibration approach. The type of analysis (steady-state versus extended period) will also give different requirements for data collection.

The most common approach to calibrating water network models is based on field data about pressures, flows, tank levels and other operational data from the DWDS. This is then used to adjust model parameters, such as nodal demands and pipe wall roughness, in order for the model to get a better fit [5]. Boccelli et al. [5] described a manual approach, where nodal demands were adjusted proportionally to match the total system flow, and altered the pressure data from the DWTP to match real data from its control system. It is also not uncommon to use automated techniques for model calibration [56]. If necessary, the pipe wall roughness can be manually estimated by conducting fire hydrant flow tests and recording the pressures along the pipe sections that are close to the hydrant [26]. Another calibration approach is to perform tracer studies.

2.6.1 Tracer studies

Performing tracer studies is a highly relevant calibration approach in water network modeling, especially when the main focus is drinking water quality [5]. This is because they can also be used to empirically estimate the travel time and flow path of water within a DWDS [35]. The general principle is to trace a dissolved, non-reactive chemical as it is transported through the DWDS. The chemical can either be added just before the tracing procedure, or it can be done by switching off the supply of a chemical that is normally maintained at a certain concentration in the drinking water [35, 9]. The tracing is commonly performed by measuring concentration or electrical conductivity using data loggers, or taking grab samples at selected locations in the DWDS [35]. This indicates, as DiGiano et al. [9] also pointed out, that the methodology of tracer studies is not highly standardized.

In large, modern DWDSs it is not uncommon to have more than one DWTP. In those cases the complexity of conducting tracer studies is higher, and it is not possible to distinguish water from different DWTPs if only one tracer is used [9]. There are at least two different strategies that can be used in order to avoid this problem. One strategy is called a "dual tracer study", and the other strategy can be described as a "water fingerprint study" [5, 56]. In a dual tracer study, two different tracers (one for each DWTP) are used. In a study by DiGiano et al. [9] this was done by switching the coagulant at one DWTP and simultaneously turning off the fluoride supply at the other DWTP. A water fingerprint study is not directly comparable with a tracer study, but can be applied in order to obtain similar information about a DWDS. The principle is based on the differences in raw water quality parameters between each raw water source, for instance color or electrical conductivity. As Walski et al. [56] described, each raw water source will have its own "fingerprint" which can be used, for instance, to determine where the interface between two types of drinking water is located in a DWDS under certain operational conditions. The main advantage of this approach is that it requires no addition of chemicals or changes to water treatment processes [56].

There are several factors that can contribute to uncertainty in a tracer study. According to Yang and Boccelli [58], the arrival of a tracer at network nodes can be significantly impacted by demand variability, especially in peripheral areas of the DWDS. Furthermore, Blokker et al. [2] found that the travel time of drinking water on one day can be up to 30 % different compared to another. Because of this, it is important to collect information about the operation and demands within the DWDS over the tracer study time period.

2.7 EPANET

EPANET is an open source water network modeling tool, and can be used to run extended period- and steady state simulations on water pipe networks that are pressurized. It is Windows-based, and EPANET 2.0 is the current version available from USEPA. It can be used to simulate the hydraulic behavior of water, as well as changes to water quality and chemistry [32].

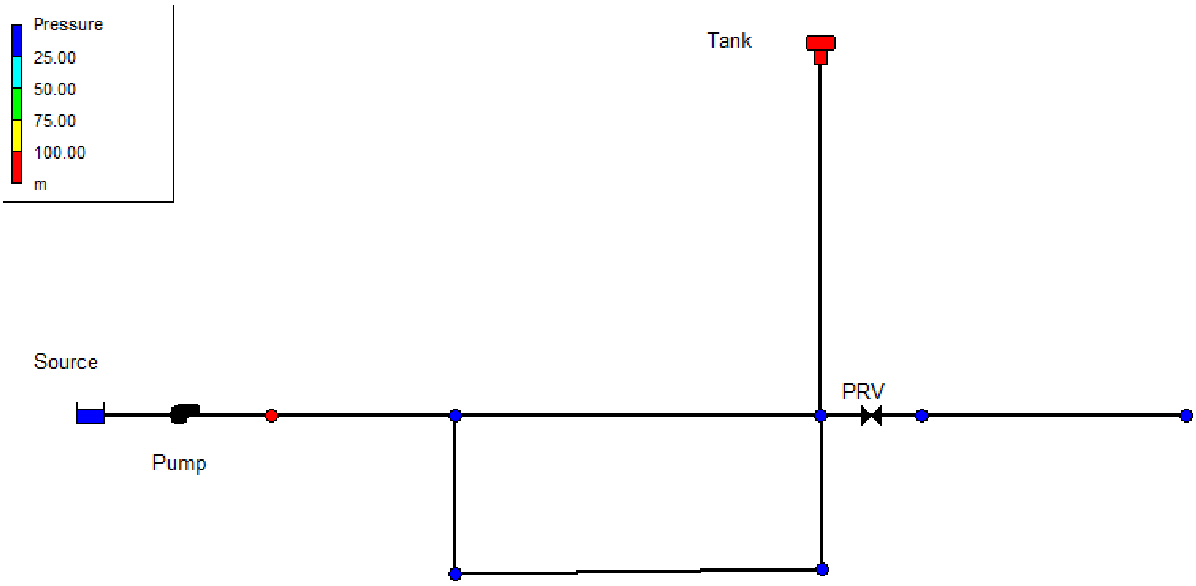


Figure 2: Screenshot from EPANET 2.0 showing components that can be included in the model.

In the hydraulic solver of EPANET, continuity and headloss equations are computed using a hybrid node-loop approach, also known as the "Gradient method" [32]. Some assumptions in the hydraulic solver are that the network is well-defined, all demands are known, each iteration is steady-state, and there is no transient flow in pipes [4].

The water quality solver of EPANET connects reaction kinetics with the principles of mass conservation [32]. Transport of a dissolved substance in pipes is modeled as advective transport, and reactions with the pipe wall and bulk flow can be modeled simultaneously [32]. Diffusion and longitudinal dispersion of the dissolved substances is not modeled [4]. According to the developers of EPANET, that is because these phenomena are considered as not significant under most operating conditions [32]. A significant limitation in the water quality solver of EPANET 2.0 is that it can only simulate the tracking and reaction of a single chemical constituent. For water quality problems that require a multi-species analysis, meaning that multiple chemical constituents need to be modeled simultaneously, a software extension named EPANET MSX can be used. MSX stands for Multi-species extension, and this software is open source and issued by USEPA [34].

When it comes to mixing in junctions, the resulting concentration of water constituents is by default calculated as the flow-weighted mean [32]. It is also assumed in the solver that complete mixing takes place in junctions [32]. The mixing in storage basins can be set in the program as [32]:

- Fully mixed: All water entering the basin is instantaneously mixed.
- Two-compartment mixing: The basin is divided into two compartments, where fill and draw processes occur in the first compartment. Overflow is sent to the second compartment, and it can supply the first compartment during draw. Both compartments are calculated as fully mixed. The user must specify which share of the total volume belongs to the first compartment.
- First-in-first-out plug flow (FIFO): No mixing occurs, and water parcels move through the basin segregated.
- Last-in-first-out plug flow (LIFO): No mixing, and water parcels stack on top of another. Fill and draw only from the bottom.

According to Rossman [32] the "fully mixed" setting seems to apply quite well to many storage basins operating in a fill/draw mode. Still, it is a consideration that needs to be made in each case. Which setting is the most appropriate will depend not only on the design of the storage volume, but also the inlet and outlet structure [13].

There exist both digital methods, such as computational fluid dynamics (CFD) simulations, and field-based methods, such as tracer studies, for assessing the mixing in storage basins [13].

2.8 Demand modeling

Blokker et al. [4] has identified demand modeling as one of the key factors affecting the accuracy of WNMs, and therefore also water age analyses. According to Blokker et al. [4], there are three spatial and temporal levels of demand modeling in DWDSs. The planning of DWTPs is on the highest level, and typically requires modeling of daily water demands. On the level below, the transport system with mains and storage basins is located, and requires modeling with a time resolution of one hour or less. The lowest level is the distribution network, with small diameter pipes, complex variable flow conditions and small time scales. Blokker et al. [4] has assessed that EPANET is suitable for modeling of water age on the two highest levels, and that simplifications in the program makes it too inaccurate to produce reliable water age estimates on the lowest level. This does not necessarily mean that other modeled parameters, such as pressure, are modeled with insufficient accuracy and reliability.

The allocation of demand in a WNM can, according to Blokker et al. [4], be performed in two principally different ways. The "top-down approach" describes the traditional way of allocating water demands, and it is also the simplest approach because it allows for large spatial and temporal scales. Commonly, points with water demands are assigned to specific demand multiplier patterns (DMPs), depending on location and type of water use. For instance, a DMP can be constructed based on the flow variations at a pumping station for a residential area, and all the nodes in that area can then be assigned to this DMP.

The "bottom-up approach", as described by Blokker et al. [4], starts with assigning a stochastic water demand pattern for each consumption node in the WNM. The model should preferably be of the "all nodes and all pipes type", which means that every single point of water consumption, including the small connection pipes, is represented. The opposite can be described as a "skeletonized model", where smaller diameter pipes are often excluded to reduce model complexity [1]. As described by Blokker et al. [4] the stochastic demand patterns are based on statistical water use data, on-site measurements, or a combination of the two. For instance, a pattern can be created for a single family house based on information about the appliances and residents there. Such an approach will of course require substantial amounts of water use information, as well as computational resources. One advantage of this approach, compared with the top-down approach, is that the variability in flow conditions and water demands in the periphery can be more accurately captured. For water quality considerations, this can be especially valuable. It has been shown that the bottom-up approach gives less auto- and cross-correlation for demand nodes. One effect when such correlation is high, is that the modeled flow over the day becomes more constant, has fewer flow reversals, and may in this way lead to an underestimation of the hydraulic residence time [4].

3 Study area

3.1 Trondheim kommune

Trondheim kommune is a Norwegian municipality, located in the southern part of Trøndelag. In the beginning of 2020, after merging with Klæbu kommune, the total population was approximately 205 000 [38, 37]. This number is expected to have increased by approximately 11 % in 2040 [38]. The total land surface area of Trondheim kommune is 528.6 km^2 [18]. The average annual precipitation at Risvollan in Trondheim between 1988 and 2018 was 938 mm [48]. The climate in Trondheim is, according to the Köppen-Geiger classification, cold, fully humid, and with cold summers. The average annual temperature for the normal period 1986–2015 was $5.3 \pm 1.1\,°C$ ($\pm$ std. dev.) [17].

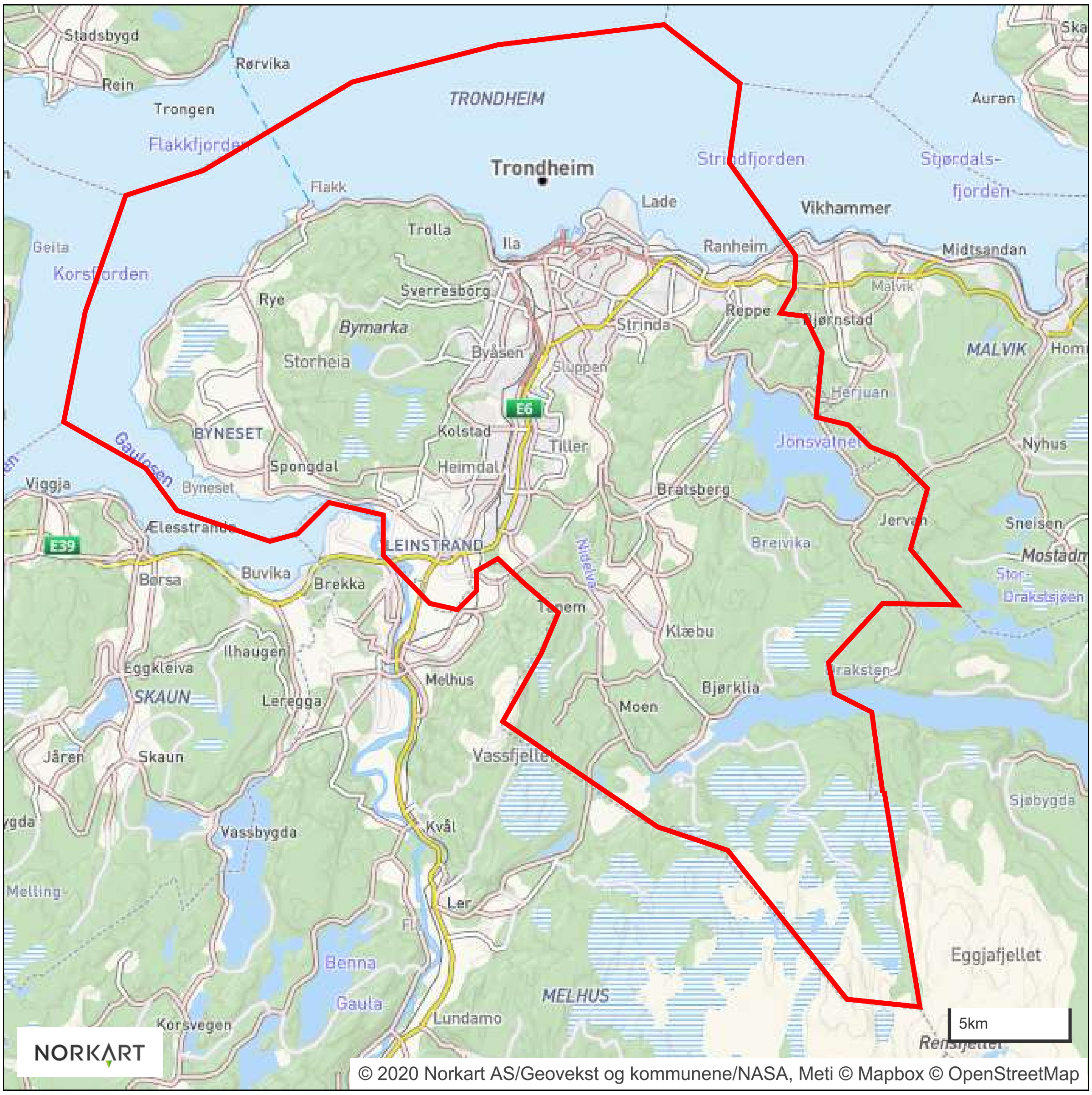


Figure 3: Overview of Trondheim kommune with districts. Administrative border in red. Source: Norkart AS (2020)

3.2 Trondheim kommune DWDS

The first public water supply system in Trondheim was opened in 1777, and used buried hollow wooden logs to transport stream water from Ilabekken to centralized wells in the city. In 1863 a system with cast iron pipes had been established, and through the 1880s it became common in the city centre to have installed water taps in kitchens [45]. The major expansions and developments of the drinking water distribution system (DWDS) have taken place after World War II. Several districts of Trondheim were established with their own separate DWDS, and these districts expanded at the same time as water customers were increasingly expecting a better drinking water quality. The first public drinking water treatment plant (DWTP) at lake Jonsvatnet was established in 1964, and from that time until 2001, all areas of Trondheim were step-wise connected to get drinking water from this lake [44]. Today's DWDS of Trondheim kommune stretches beyond the administrative borders, and is connected with both Melhus kommune and Malvik kommune. The MeTro pipeline between Melhus and Trondheim is intended to ensure a secondary water supply for both municipalities, which gives increased redundancy of the drinking water supply. The two main sources of raw water for the DWDS are large, deep lakes located below the tree line. Lake Jonsvatnet is situated in Trondheim kommune and lake Benna in Melhus kommune. Both lakes have a nearby DWTP. The VIVA plant treats raw water from Jonsvatnet and the Benna plant treats raw water from Benna. The total number of persons supplied by the DWDS of Trondheim kommune, not including Klæbu, is approximately 230,000. The Klæbu district is supplied from a groundwater source located at the Fremo-plateau. In 2015 it was reported that the total average water consumption per person per day in Trondheim was approximately 285 liters. This also includes consumption in schools, hospitals, private business, industry etc. Leakage and other losses made up approximately 29 % of the daily consumption in 2015 [44]. The temperature of drinking water in the DWDS is relatively cold (median = 7.2 °C; range = 5.2 to 8.5 °C in 2019) [54].

In 2016, it was estimated in a public report [44] that the total length of drinking water pipes in Trondheim kommune was approximately 1.700 km. Out of this approximately 50 % was privately owned and operated [44]. In 2020, after merging with Klæbu kommune and completing several construction projects, the estimate is less accurate. A rough overview of transmission and distribution mains from before the merging with Klæbu is given in Figure 4. Note that this is only a coarse illustration, and that much more accurate maps exist, but are not publicly available.

When not including the DWDS in former Klæbu kommune, there is a total of 12 storage basins in the DWDS of Trondheim kommune. The largest basin is located at Høgåsen, and placed underground in rock-blasted halls. There are also storage basins in the DWDS that are placed above ground. The total safety reserve of the 12 basins is approximately 69.300 m^3 [44].

The largest water conduit in the DWDS of Trondheim kommune is the Vikåsen water tunnel. It was finished in 1963, and is a 1100 m long rock-blasted tunnel with a cross-sectional area of approximately 10 m^2 ($B \times H$ =3.0 m×3.65 m). Approximately 80 to 90 % of the finished water from the VIVA DWTP is transferred through this tunnel, which is unlined, and a modernization is in progress to reduce the risk of contaminated water leaking in [44].

Figure 4: Overview of the drinking water sources (shaded) and large mains (blue) connected with Trondheim kommune, not including that of former Klæbu kommune. Source: Trondheim kommune (2016) [44]

3.2.1 VIVA DWTP

Vikelvdalen vannbehandlingsanlegg (VIVA) was completed in 1998 and is the main DWTP in Trondheim kommune [44]. Raw water is extracted from lake Jonsvatnet, at a depth of 50 m, and transported through a pipe to a coarse-sieving chamber, before it continues through a pipe inside a rock-blasted tunnel to VIVA. The first treatment step is injection of CO_2, which is followed by rapid filtration through crushed marble. This is performed in order to make the treated water less corrosive, and raises the pH-value. The next step is disinfection with sodium hypochlorite (NaOCl) which is produced on-site from electrolysis of NaCl brine solution. The final disinfection step is performed using UV radiation, before the finished drinking water flows by gravity into the DWDS [49, 45]. In 2007, the average flow rate of drinking water from VIVA was 900 $\mathrm{L\,s^{-1}}$. The maximum design capacity of the DWTP is 1400 $\mathrm{L\,s^{-1}}$ [49]. In the winter of 2014, a combination of strong winds, cold weather and no ice cover on lake Jonsvatnet, lead to a 3-month event where water from the surface reached the raw water intake. Because of this, the watershed and the source can not be counted as a permanent hygienic barrier, and options for additional water treatment at VIVA have been considered [44].

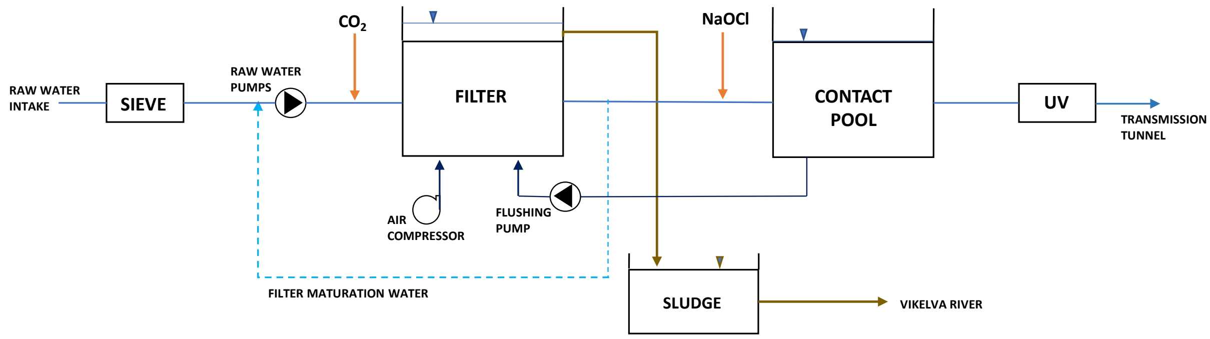


Figure 5: Illustration of the drinking water treatment system at VIVA. By Jon K. Rakstang (2020). Based on figure by Trondheim kommune [44]

3.2.2 Benna DWTP

Benna VBA (DWTP) was put into operation in the spring of 2016. The raw water is extracted from lake Benna, which has a very low content of nutrients and color. In the DWTP, it passes through a coarse-sieving, before chlorine disinfection is performed using NaOCl produced off-site. Additional disinfection is performed by UV-treatment, before the finished drinking water flows into the MeTro pipeline [44]. The maximum treatment capacity is 800 $\mathrm{L\,s^{-1}}$ and the intended normal production is 200 $\mathrm{L\,s^{-1}}$, out of which 50 $\mathrm{L\,s^{-1}}$ goes to Melhus and 150 $\mathrm{L\,s^{-1}}$ goes to Trondheim. The DWTP was built with the possibility of expanding the production capacity to 1200 $\mathrm{L\,s^{-1}}$ [46]. In the summer of 2017 it was discovered that *Copepods* and *Pallasea* had been transported from Benna, through the DWTP, and out to water customers in Melhus kommune and districts in Trondheim kommune that were receiving this drinking water. Because of that, Benna DWTP was put out of operation until October 2019, when it became necessary to put it back into operation temporarily. In the meantime, all of Trondheim kommune, Melhus kommune and Malvik kommune was supplied from VIVA DWTP [43].

3.2.3 MeTroVann

Completed in 2016, the MeTroVann project enables Melhus kommune and Trondheim kommune to exchange drinking water through a 24 km water main. This main is shown as the southernmost long stretch of pipe in Figure 4. In addition, sanitary wastewater from Melhus kommune is pumped to Høvringen wastewater treatment plant (WWTP) in Trondheim through a parallel, separate pipeline. The water main consists of 20 km GRP pipes with a diameter of 1000 mm and 1200 mm and 4 km cast iron pipes [46]. Considering the large transport capacity of the MeTro pipe and the significant difference in water consumption between Melhus and Trondheim kommune, it can be assumed that the flow velocity will depend heavily on the drinking water supply situation in those two municipalities. When Benna DWTP is in operation, it can be expected a flow in the direction of Trondheim kommune of at least 150 $\mathrm{L\,s^{-1}}$. When Benna DWTP is out of operation, and Melhus kommune receives roughly 50 $\mathrm{L\,s^{-1}}$ from VIVA DWTP, there is reason to expect very low flow velocities in the pipeline, which may contribute substantially to the water age in Melhus kommune.

4 Methods

4.1 Literature review

The literature review for this master's thesis has been conducted in two stages. The first stage was conducted in the autumn of 2019, as part of the course "TVM 4510 Water and wastewater engineering specialization project", and the main goal was to identify core literature about water network modeling and the connection between the age and quality of drinking water. In addition, literature about the use of tracer studies was reviewed in order to formulate different strategies for assessing accuracy in the water network model (WNM) of Trondheim kommune. The second stage of literature review continued on the same main topics, but was more focused around the revised research objectives for this thesis. The search engines "Scopus" and "Google scholar" have been used to find scientific publications. In addition, books, technical reports, regulations and statistical data were retrieved from the internet and the university library.

4.2 Water network modeling

The WNM was initially exported from MIKE URBAN, a commercial WNM software used by Trondheim kommune, formatted as an EPANET input file (.inp). This file has been used and modified in EPANET 2.0, through the standard user interface, and also an open-source Matlab® package named "epanet-matlab" written by Jim Uber [50]. A script written by Rokstad [31] was used as the basis for developing three scripts that are used in this thesis. The PC used is a Lenovo P52 ThinkPad with an 8th generation 2.60 GHz Intel core i7 processor, 32 GB RAM and SSD storage.

The WNM had the characteristics that Blokker et al. [3] used to describe a traditional "top-down approach" for model construction. Small diameter connection pipes were not included in the model, and clustered water demands were assigned to a set of 48 different 24-hour demand multiplier patterns (DMPs). The WNM contained approximately 9500 nodes and 10 800 links (pipes). The total number of water customer connections in 2015 was approximately 51 000, almost 100 % of the population is served by the municipal water supply [44], and 1/3 of the homes in Trondheim kommune are detached houses [43]. When comparing the number of nodes in the model with the number of detached homes in the model area, it becomes clear that water demands in the model were clustered, which is common in water network models.

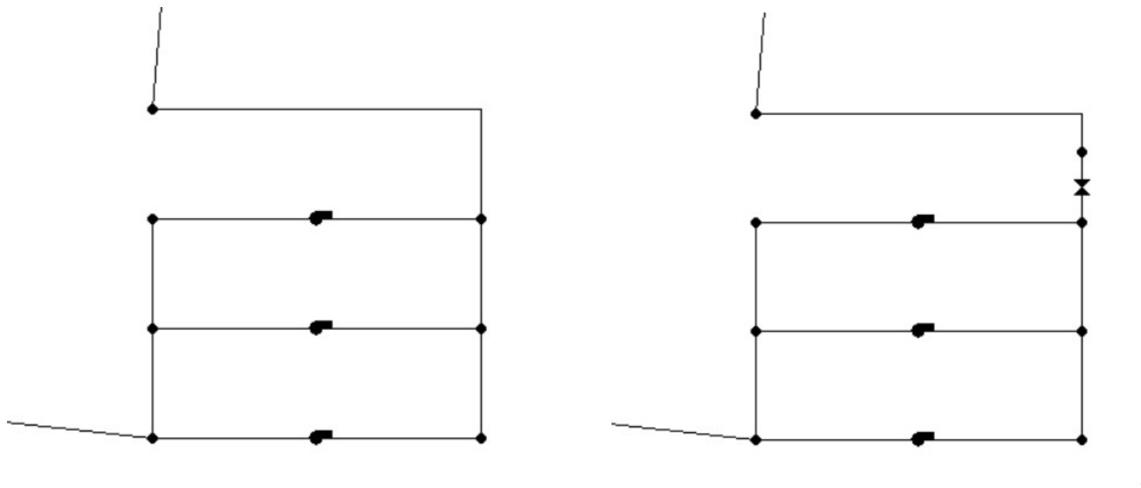
4.2.1 Various adjustments and settings in the model

Water age simulations are based on the supply situation where both drinking water treatment plants (DWTPs) are in operation. In order to model the propagation of a non-reactive tracer compound, all reaction coefficients in EPANET were set to zero. Additional nodes were also inserted closer to the inlets and outlets of selected storage basins in order to get more representative simulation results there.

Tracer simulations are based on the supply situation where only VIVA DWTP is delivering water to Trondheim. This was the situation during the days when the tracer study was conducted. Furthermore, an adaptation was done in the model to take into account that an important water main was closed off due to maintenance.

4.2.2 Variable speed pump representation

The EPANET version of the WNM of Trondheim kommune did not have variable speed pump functionality, and modifications were therefore made to get a more correct representation of pumping stations. Figure 6 shows how a pressure reducing valve (PRV) was inserted on the pressure side of a pumping station, and the settings of this valve were adjusted according to the pressure setpoint used in the real-world system. The valve was stored with a loss coefficient of zero, and the height was set to be the same as the height of the real-world pressure sensor that is used to regulate pumping speed. In this way the pumping station in the model delivers a constant water pressure when pumps are in operation, which is close to what happens in the real-world system. This was done for all the 8 pumping stations with variable speed drive in the drinking water distribution system (DWDS).



(a) Before

(b) After

Figure 6: Example showing before/after a PRV was inserted in the model, resembling variable speed pump operation

4.2.3 Calculation of predicted rise in conductivity

The modeled concentration rise of Na^+ and Cl^- as a consequence of tracer addition was converted to a rise in conductivity using the following simplified relationship [57] in a spreadsheet calculation:

$$EC = \frac{TDS}{K} \tag{4}$$

Where

TDS = Concentration of total dissolved solids [$\mathrm{mg\,L^{-1}}$]

K = Correction factor [$\mathrm{\mu S\,mg\,cm^{-1}\,L^{-1}}$]

EC = Electrical conductivity at 25°C [$\mathrm{\mu S\,cm^{-1}}$]

The value of the K-factor was set to be 0.51, instead of the typical value of 0.7, because 0.5 was found more suitable for the range of conductivity values expected in the DWDS. This decision was made in line with recommendations in Walton [57]. For a standard KCl solution this corresponds to a conductivity of approximately 150 $\mathrm{\mu S\,cm^{-1}}$ at 25 °C [57]. The average conductivity of raw water from Jonsvatnet in 2018 was 13 $\mathrm{\mu S\,cm^{-1}}$ at 20 °C [43], but the conductivity of drinking water is expected to be higher due to water treatment and corrosion in the DWDS. The upper limit value for conductivity in the Norwegian drinking water regulations is 250 $\mathrm{\mu S\,cm^{-1}}$ at 20 °C [15].

4.2.4 Inclusion of an additional drinking water supply

In the version of the WNM that was used in this thesis, the additional drinking water supply from Benna DWTP was not fully modeled. In order to make the WNM compatible with the supply situation during the time of writing the thesis, additions were made to this part of the model. Reservoir nodes were added to represent lake Benna, a PRV was inserted into the modeled DWTP to have correct pressure, and this is equivalent to how it had been done for Jonsvatnet by the model constructors previously. Furthermore, specific valve settings were implemented at a central pumping station in the model according to instructions given by an operational manager in Trondheim Bydrift.

4.2.5 Extended period simulation with tracer

This part of the modeling work was designed to have direct compatibility and comparability with a full-scale tracer study. For the extended period simulations with a non-reactive tracer, a node inside VIVA DWTP was set to provide a constant $20\,\mathrm{mg\,L^{-1}}$ concentration of tracer for exactly one hour, starting at 16:25 (UTC +02). This is exactly 30 minutes later than the time of the real-life tracer feed initiation (15:55), and takes into account the hydraulic residence time of approximately 30 minutes in the chlorine contact basins at VIVA DWTP. The approximate location of the stations, as well as their numbering, is shown in Figure 8. A 5-minute time-step was chosen for both the hydraulic and quality analysis in EPANET. A Matlab® script was used to run the EPANET simulations and store the results directly into a spreadsheet, see Appendix A. Equation (4) was implemented in the spreadsheet, and converted the predicted rise in NaCl concentration into an estimated rise in conductivity.

4.2.6 Extended period simulation to obtain water age

The extended period simulations were set up and run in EPANET using a Matlab® script, see Appendix A. Water age at the reservoir nodes in Jonsvatnet and Benna was set to zero by the model, and it was chosen not to track the distribution of these two different types of drinking water throughout the DWDS. In other terms, it was assumed that all of the water supplied to the DWDS had the same properties. The simulation time was firstly set to 30 days (1 month), then to 120 days (4 months), and a 10-minute time-step was used for both the hydraulic and quality analysis in EPANET.

4.2.7 Simulation and identification of pipes with flow reversals

A 24 hour simulation was run in EPANET using a Matlab® code, given in Appendix A. The flow velocity in all pipes (links) was computed and stored for each time-step, and an algorithm identified the pipes in which one or more flow reversals had occurred. In the next step, a selection was made to only include pipes where the min, max and median flow velocity matched pre-determined criteria, given in Table 7. The criteria were selected based on results in Blokker et al. [2], where simulations and field tests had been conducted to investigate self-cleaning velocities in water mains. Dutch design guidelines state that self-cleaning of water pipes is ensured if the flow velocity reaches $0.4\,\mathrm{m\,s^{-1}}$ during the day, which Blokker et al. [2] confirmed through experiments. Results in Blokker et al. [2] indicated that flow velocities below approximately $0.2\,\mathrm{m\,s^{-1}}$ are not sufficient for self-cleaning, and that fouling within pipes increased rapidly when the velocity approached $0\,\mathrm{m\,s^{-1}}$.

4.3 Tracer study

4.3.1 Experimental setup

As described earlier, the sodium hypochlorite (NaOCl) disinfectant used at VIVA is produced on-site. There are two separate production facilities for redundancy, but only one is active during normal operation. NaCl, approved for drinking water treatment, is extracted automatically from a stainless steel silo using a screw press. The salt falls into a large plastic tank where freshwater is added through a flotation valve. In order to maintain a steady production of saturated brine, the addition of salt is adjusted so that there is always a layer of undissolved salt of approximately 10 cm at the bottom of the tank. From the tanks the brine is pumped through plastic pipes by membrane pumps, and into the electrolysis units. In the tracer study, brine was extracted directly from one of the storage tanks using a membrane pump, transferred using a flexible pipe, and added to the water stream before it entered chlorine contact pool 2 as shown in Figure 7. By adding the tracer there, a thorough mixing was ensured, and the risk of contaminating the drinking water with microorganisms from the brine was reduced because of the following two disinfection processes.

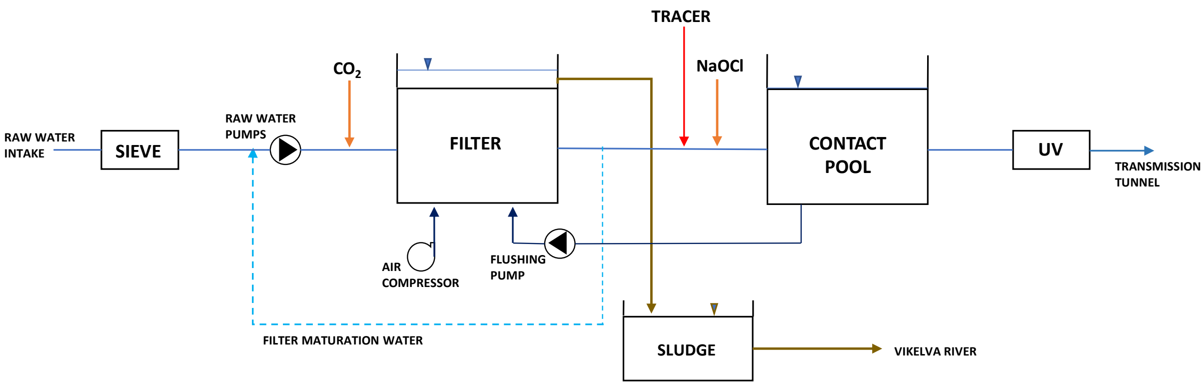


Figure 7: Illustration of the drinking water treatment system at VIVA, showing the point of tracer injection. By Jon K. Rakstang (2020). Based on diagram by Trondheim kommune [44]

It was calculated (see Appendix B) that a saturated brine volume of approximately 161.3 L would be needed for increasing the conductivity of the drinking water by 30 $\mu S\,cm^{-1}$ at 20 °C for one hour at the DWTP. This corresponds to 57.6 kg of pure NaCl. Note that this amount has been calculated assuming that the DWTP operates with a steady 800 $L\,s^{-1}$ flow rate for the whole hour that tracer is added. During tracer addition, it was necessary to adjust the tracer flow according to the changing flow rate through the DWTP.

Conductivity measurements were performed at six locations in the DWDS, indicated in Figure 8. At each station, drinking water was tapped directly from into a 20 L PP plastic box (see Figure 9) with the conductivity sensor mounted, and then discharged into the drain. Initial WNM simulations showed that it would be necessary to perform data logging for at least three days, and that the differences in tracer arrival time would allow for some of the data loggers to be moved to additional measurement stations during the experiment if needed.

4.3.2 Field work protocol

1. Conductivity probes were tested with a standard solution, before installation in measurement boxes
2. 2020-05-18: Data logging kits were installed at measurement sites, and water flow was initiated. Roughly 20 L min^{-1} was sent through each box
3. 2020-05-18: The tracer addition pump was tested and adjusted, before the tracer feed was switched on at 15:55. Pumping rate was adjusted in relation to water flow in order to give a 20 mg L^{-1} NaCl concentration. The tracer feed was switched off at 16:55. Conductivity and temperature values were recorded manually between 15:45 and 18:15, with water extracted directly after chlorine contact pool 2
4. 2020-05-18: A pumping station downstream of the DWTP was manually reduced to 50 % at 15:40, and then deactivated completely at 17:00. This was done to increase the chance of water with salt entering one of the storage basins, which is of the side-basin type. Pumping was back in full operation at approximately 21:25
5. 2020-05-19: The handheld conductivity data logger was moved from station 2 to station 5
6. 2020-05-20: The handheld conductivity data logger was moved from station 5 to station 6
7. 2020-05-25: All data loggers were disconnected after 7 days of measurement, and the field work was finished

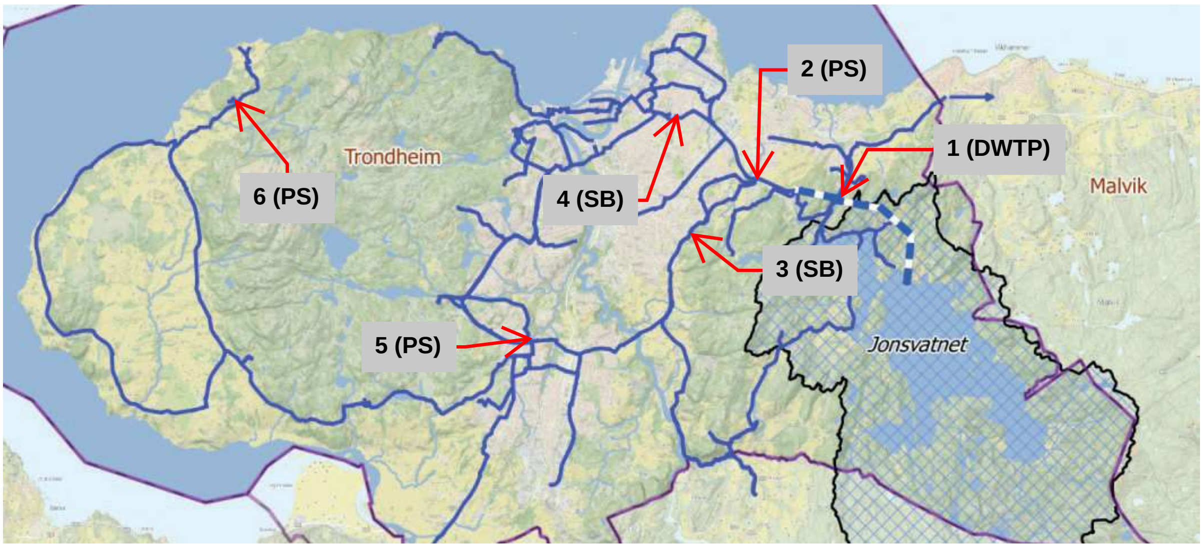


Figure 8: Approximate locations of the selected measurement stations for the tracer study. SB = Storage basin, PS = Pumping station, DWTP = Drinking water treatment plant. Source: Trondheim kommune (2016) [44]

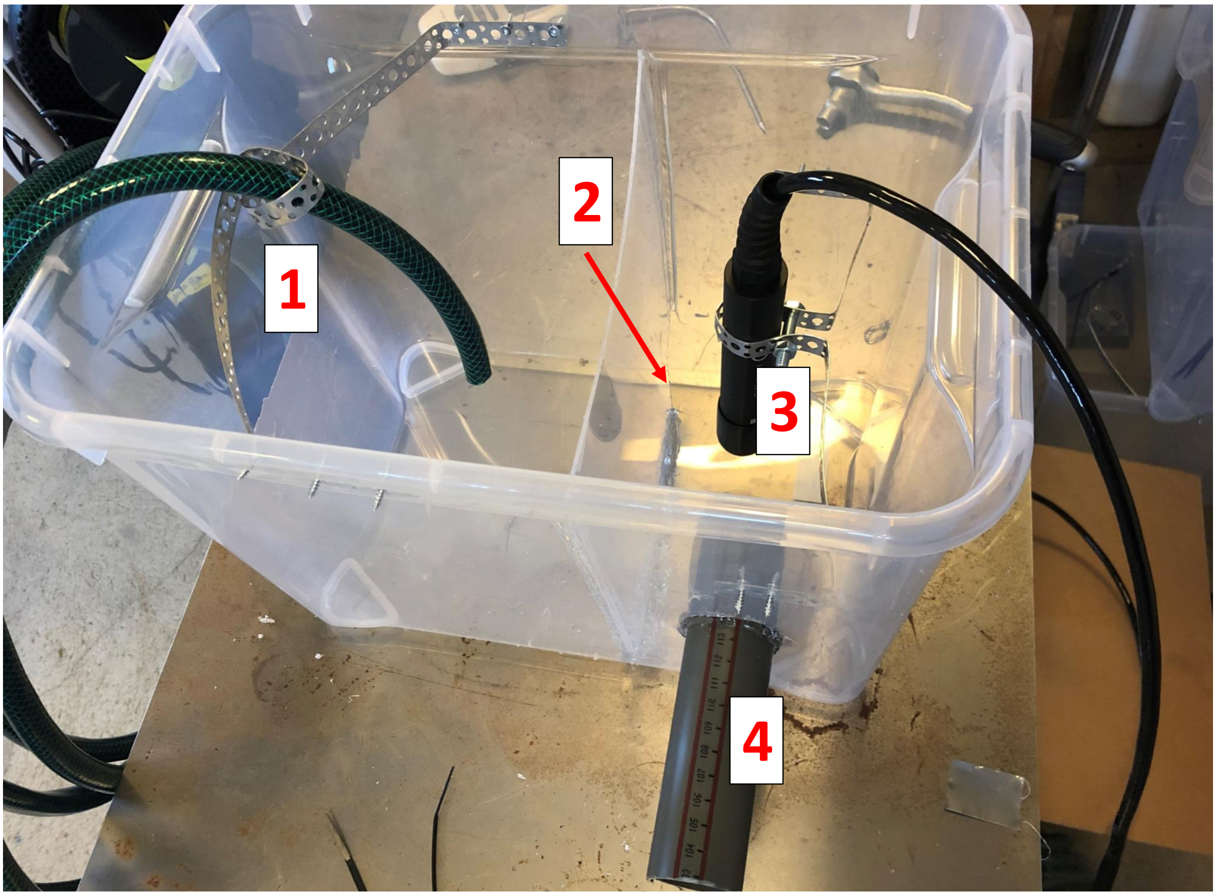


Figure 9: Photo of a 20L measurement box. 1: 1/2" flexible tube lets water free fall into the first chamber, which serves to dampen waves, release air bubbles and eliminate risk of back-flow into the DWDS. 2: Small opening in the lower part of the dividing wall lets water flow into the second chamber. 3: Conductivity and temperature probe mounted in the center of chamber two. 4: Water outlet through 40 mm PVC pipe to drain

4.3.3 Measuring equipment

The first data-logging kit had a CR200X logger and one SN-PC4EB-5713 conductivity and temperature probe. The second kit had a CR300 logger with two probes of the same type. The third kit had a handheld WTW Multi 3630 logger with a TetraCon 925-3 probe. At the DWTP a WTW LF 537 conductivity meter was used to record values manually during the start of the tracer study. All measurements in the tracer study were taken with 5-minute intervals. Figure 10 shows the first kit installed at a measurement station.

Figure 10: Photo of a data-logging kit installed and activated at one of the measurement stations

4.4 Statistical analysis of water quality data

For statistical exploration of water quality, water ages were estimated as the average value obtained from the last 48 hours of a 120-day water age simulation. Water quality records, including heterotrophic plate count (HPC) measurements in the DWDS (2016 to 2020) and raw water temperature data at VIVA (2015 only), were obtained from Trondheim Bydrift. Both Jonsvatnet and Benna have supplied the DWDS since MeTro operation started in 2015.

The open-source statistical R language [28], as well as and the open-source integrated development environment RStudio, were used in the analysis of water quality data and water age simulation data. Several packages, including "EnvStats" [23], "survival" [40, 41], and "car" [12] packages were utilized in combination with an R script written by Waak [52].

The HPC dataset contained so-called left- and right-censored data. Measurements that were below the detection limit for the HPC method were left-censored (<1 colony-forming units (CFU) mL^{-1}, or no observed colony growth). Conversely, when colony growth was excessive and essentially non-countable, HPC was sometimes reported as >300 CFU mL^{-1}, which was considered right censored. Left-censored data occurred far more often than right-censored data (216 versus 4 out of 805 data points total, respectively). Simply replacing the censored values with 0 CFU mL^{-1} and 300 CFU mL^{-1}, or removing them completely, could lead to bias.

To minimize bias with the presence of censored data, maximum likelihood estimation (MLE) and the likelihood profile were used to estimate, respectively, HPC mean averages and their 95 % confidence intervals. Currently, some MLE methods have limited implementation for datasets containing multiple types of censored data [23, 40, 23]. When only one type of censoring was permitted by the methods, left-censored data were used, and the four right-censored data points were substituted as 300 CFU mL^{-1}.

MLE/survival analysis takes censored values and estimates their "actual" or probable values using an assumed distribution. To select a distribution, cumulative distribution functions were manually inspected (Figure 11), using an R script by Waak [52]. A theoretical log-normal distribution showed good fit with HPC data, so this distribution was assumed for all subsequent statistical analyses.

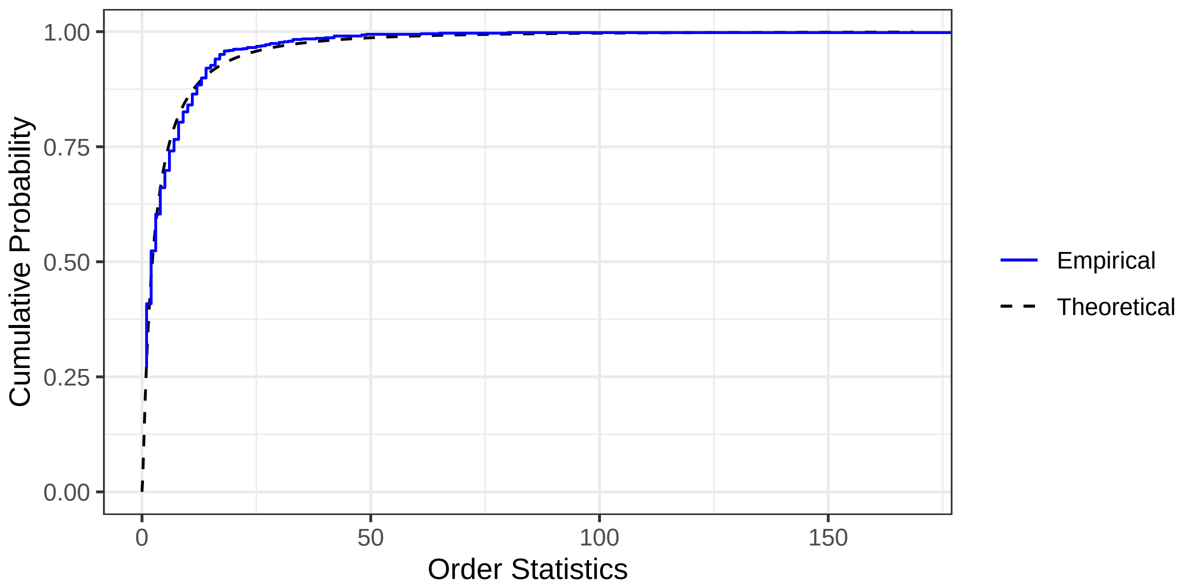


Figure 11: Cumulative distribution function plot for empirical HPC data versus the theoretical log-normal distribution.

5 Results

5.1 Tracer modeling and tracer study

The following Figures 12 and 14 to 22 are graphs presenting results from the 7-day tracer study. Measurement station numbering refers to Figure 8. Some graphs have labels indicating the approximate measured and simulated arrival time of the peak tracer concentration. Time 0 refers to 15:55 (UTC +02) on 2020-05-18, when the tracer feed was activated. A summary of simulated and measured peak arrival times is given in Table 4 below:

Table 4: Summary of simulated and measured peak arrival times of tracer at stations 2 to 6

Station	Simulated arrival time [h]	Measured arrival time [h]	Difference [h]
Station 2 (PS)	6	9.25	3.25
Station 3 (SB) inlet	6.75	10.75	4
Station 4 (SB) inlet	9	15.5	6.5
Station 5 (PS)	11	18.5	7.5
Station 6 (PS)	41	–	–

5.1.1 Measurement station 1 (DWTP)

Figure 12 below presents the simulated and measured flow rates through VIVA drinking water treatment plant (DWTP) during the first 24 hours after tracer was added. The time resolution is 5 minutes. Switching a pumping station on and off produced a distinct drop and rise in both graphs between 17:00 and 21:25 as indicated. There was a considerable difference between measured and simulated values, where simulated values are mainly over-estimating flow rates through the DWTP.

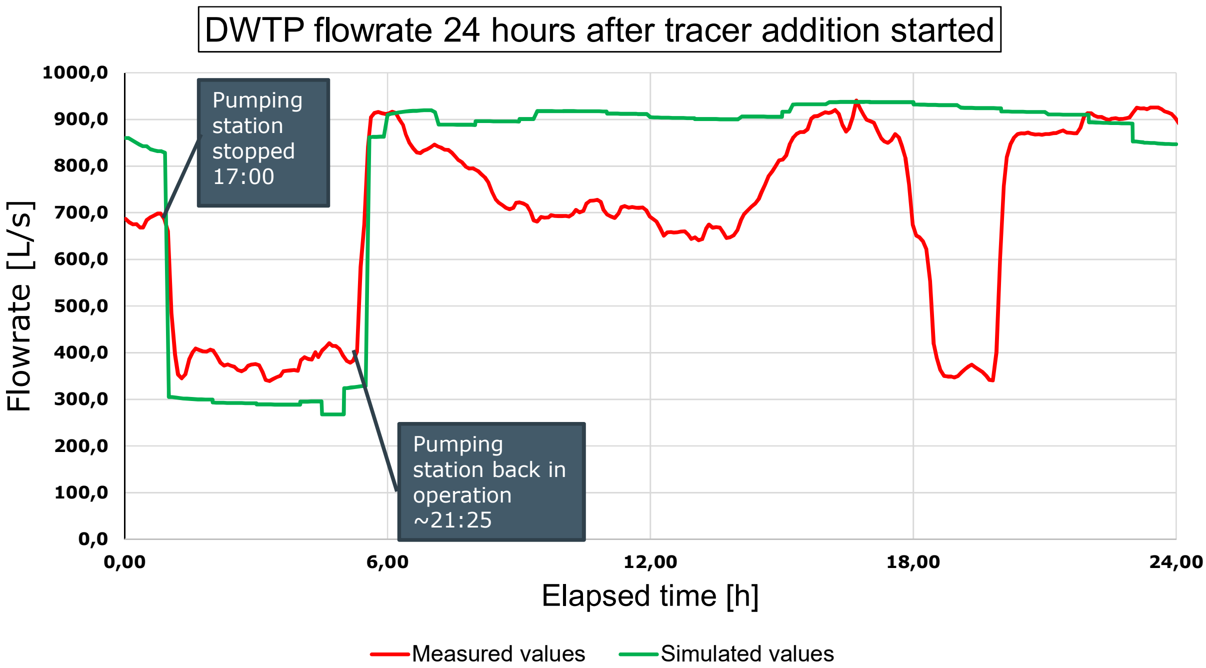


Figure 12: Simulated and measured flow rates in the DWTP during the first 24 hours after the tracer study was initiated

Figure 13 shows simulated and measured accumulated flow through VIVA DWTP during the first 24 hours after tracer feed was switched on. Accumulated volumes were estimated using 5-minute resolution flow rate data, and the trapezoidal rule. The graph shows that the model predicted a flow volume more than twice as big as measured by the end of the first 24 hours, which is considerable.

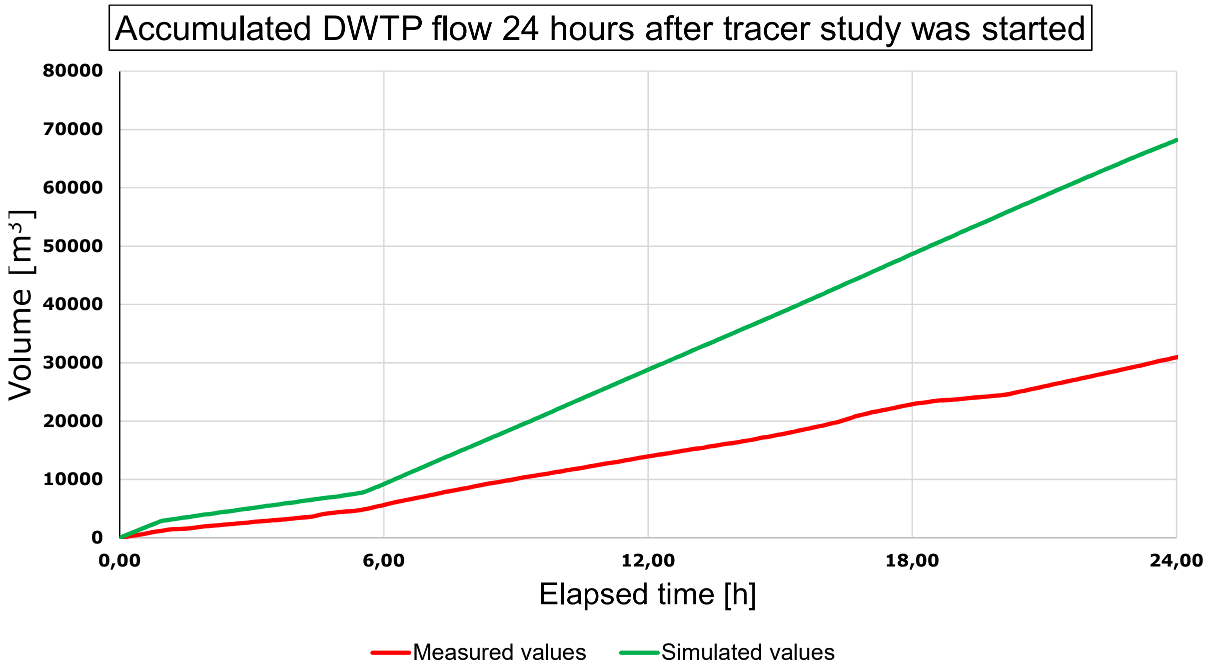


Figure 13: Simulated and measured accumulated flow volume through VIVA DWTP. The trapezoidal rule was used to calculate volumes from flow rates

Figure 14 below presents the simulated and measured conductivity values in water extracted after chlorine contact basin 2 at VIVA DWTP. Note that contact basin 2 only receives half of the water flow passing through the DWTP. In addition, a 1-minute time resolution flow rate graph for that location in the DWTP has been added. It is clearly visible that the hydraulic residence time, as well as the mixing conditions, in the contact basin has made the tracer pulse far less "square" shaped than simulations predicted. It also comes clear that the measured conductivity rise was much higher, at maximum twice as high as predicted. This is easily explained by looking at the measured flow rate through contact basin 2, which was roughly half of the total flow rate through the DWTP. Shortly after the sampling point, the two streams of water are joined, meaning that the tracer concentration will be lower in water leaving the DWTP.

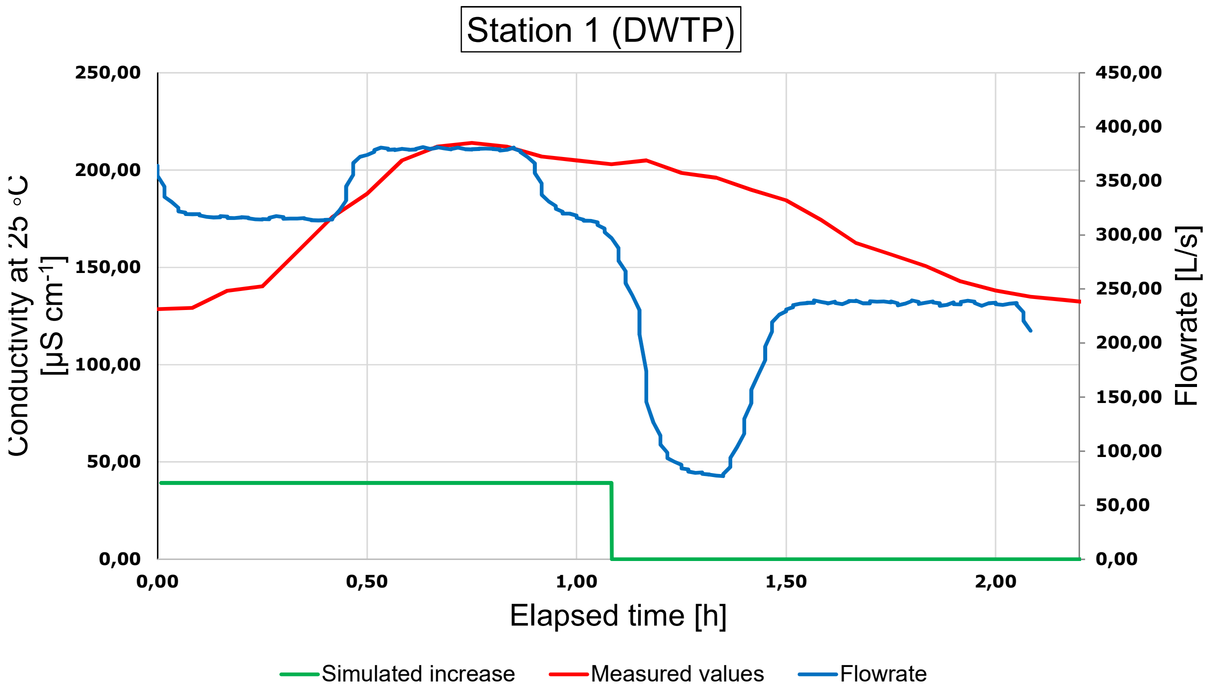


Figure 14: Simulated and measured conductivity values, as well as flow rate, at the sampling site after contact basin 2 in the DWTP

5.1.2 Measurement station 2 (Pumping station)

Figure 15 below presents the simulated and measured conductivity values at station 2. Both graphs have distinct peaks, although the simulated peak is more "square" shaped, and there was a 3.25 hour difference between simulated and measured arrival time. The measured peak conductivity value was approximately $10\,\mu S\,cm^{-1}$ lower than predicted. The measured graph was cut off after 16 hours because the data logger was moved to station 5.

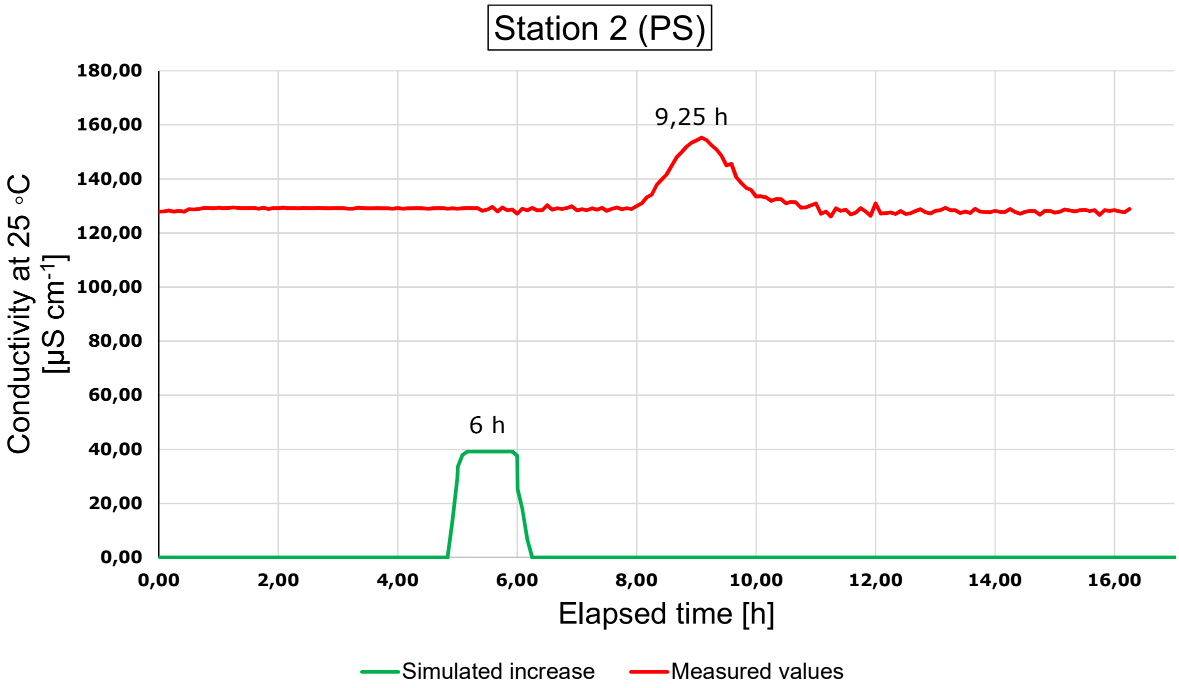


Figure 15: Simulated and measured conductivity values at measurement station 2 (Pumping station)

5.1.3 Measurement station 3 (side basin)

Figure 16 shows the simulated and measured conductivity values at measurement station 3, which is a storage basin of the side basin type. Figure 17 also shows measured conductivity values, in combination with the inflow/outflow graph. Both figures show a fill/draw behavior which is characteristic for a side basin. Figure 17 shows that a water volume with fairly high tracer concentrations was pumped into the basin approximately 10 hours after the tracer feed was switched on at the DWTP. It can also be observed that the subsequent conductivity peaks occurred mainly when water was leaving the storage facility, and that the conductivity range was varying with approximately 8 μS cm^{-1} for several days after tracer arrived.

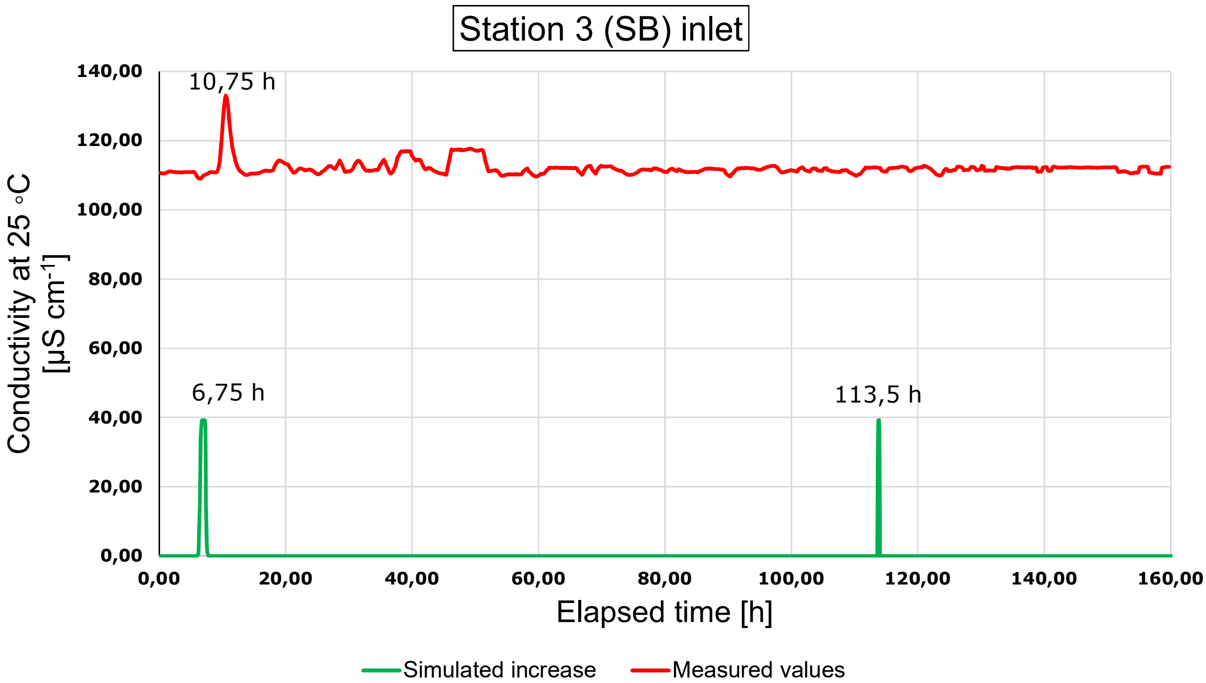


Figure 16: Simulated and measured conductivity values at the inlet of measurement station 3 (side basin)

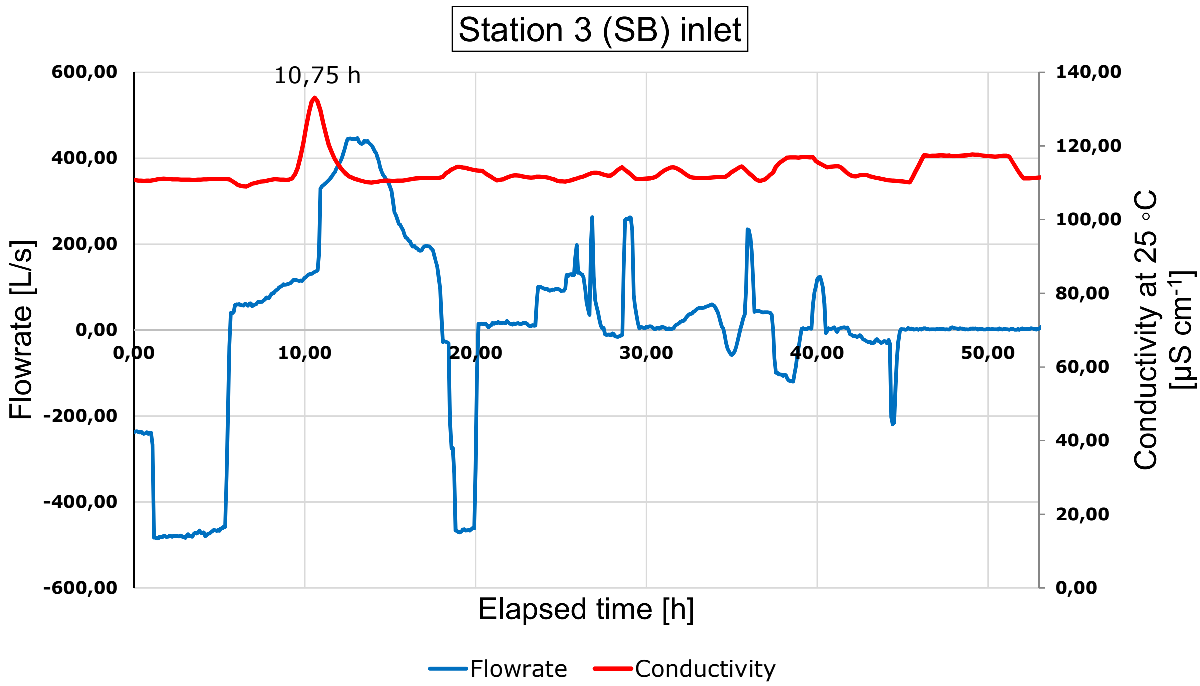


Figure 17: Measured conductivity values and flow rate at the inlet of measurement station 3 (side basin)

5.1.4 Measurement station 4 (flow-through basin)

Figures 18 and 19 show the simulated and measured conductivity values at the inlet and outlet of station 4, a flow-through storage basin with 3 separate tanks. The shape of the first peaks in Figure 18 are fairly similar, but the arrival time differs by 6 hours. The measured values also indicate a drop in conductivity after 5 days, before it rises again. This may have resulted from disturbances in the measuring equipment or other uncertainties. In Figure 19 there are no distinct peaks on the graphs, but time labels are placed where both graphs have a sudden and similar increase.

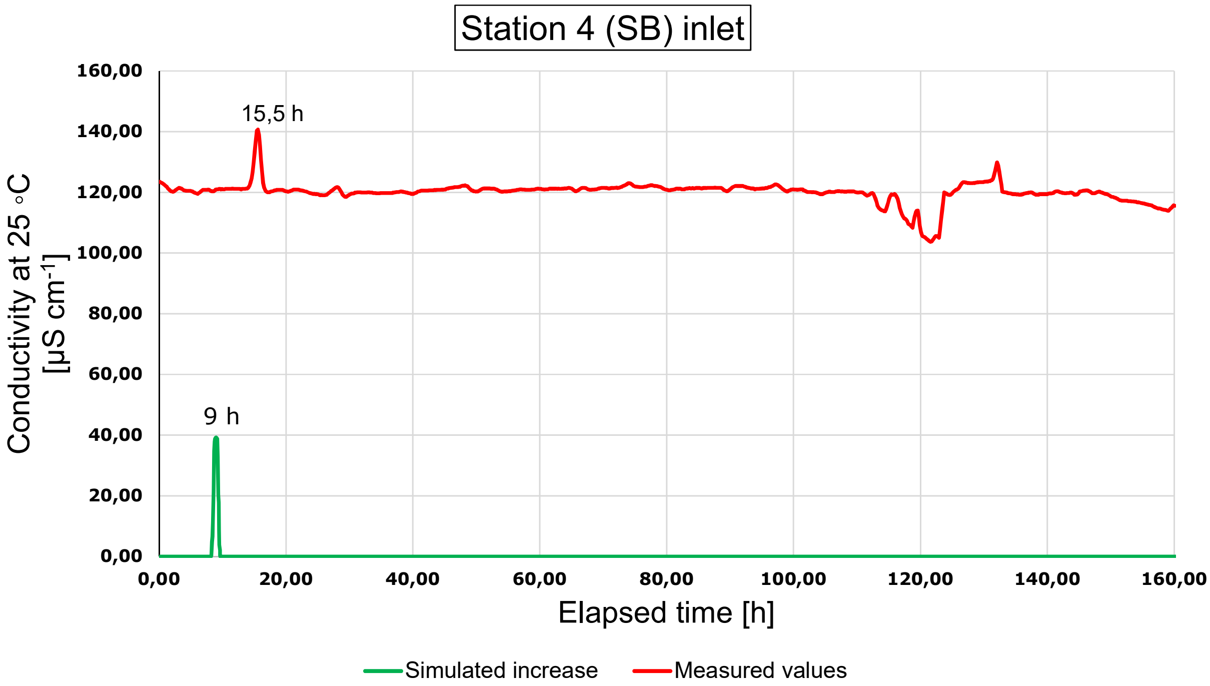


Figure 18: Simulated and measured conductivity values at the inlet of measurement station 4 (flow-through storage basin)

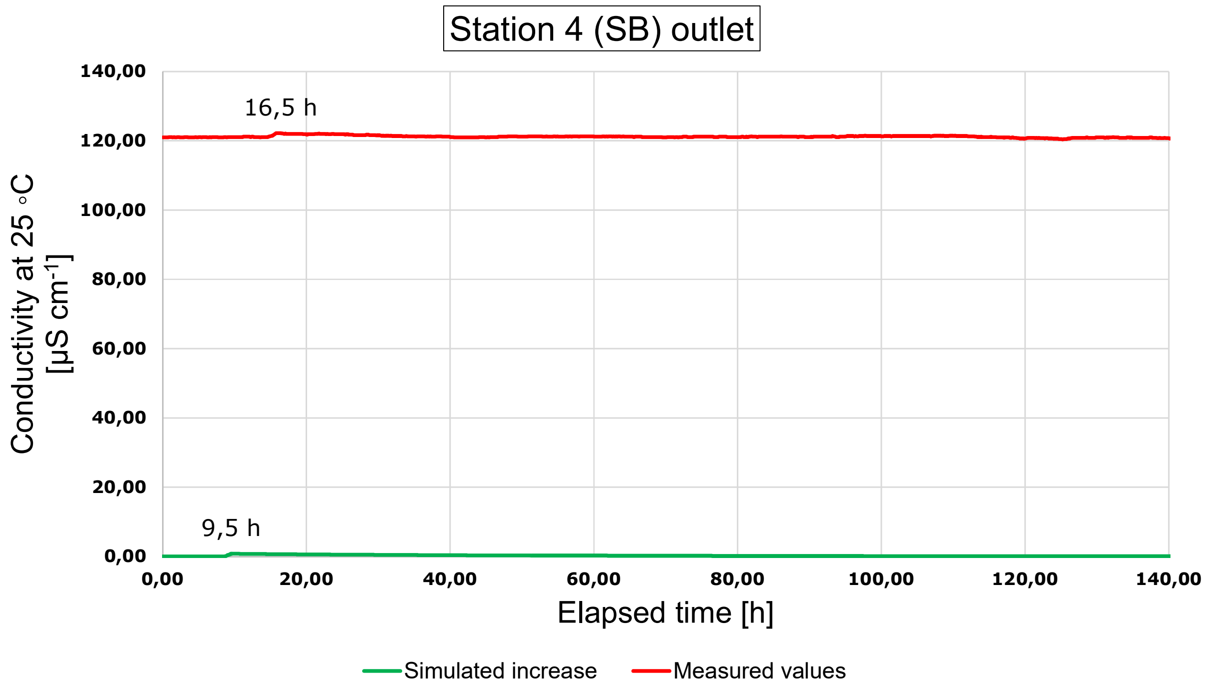


Figure 19: Simulated and measured conductivity values at the outlet of measurement station 4 (flow-through storage basin)

5.1.5 Measurement stations 5 (pumping station)

Measured conductivity values in Figure 20 indicate that the tracer peak arrived at station 5 immediately after the data logger was activated. Tracer peak arrival was 7 hours later than simulations predicted. The measurements were limited at this station, because the equipment was moved to station 6 the next day.

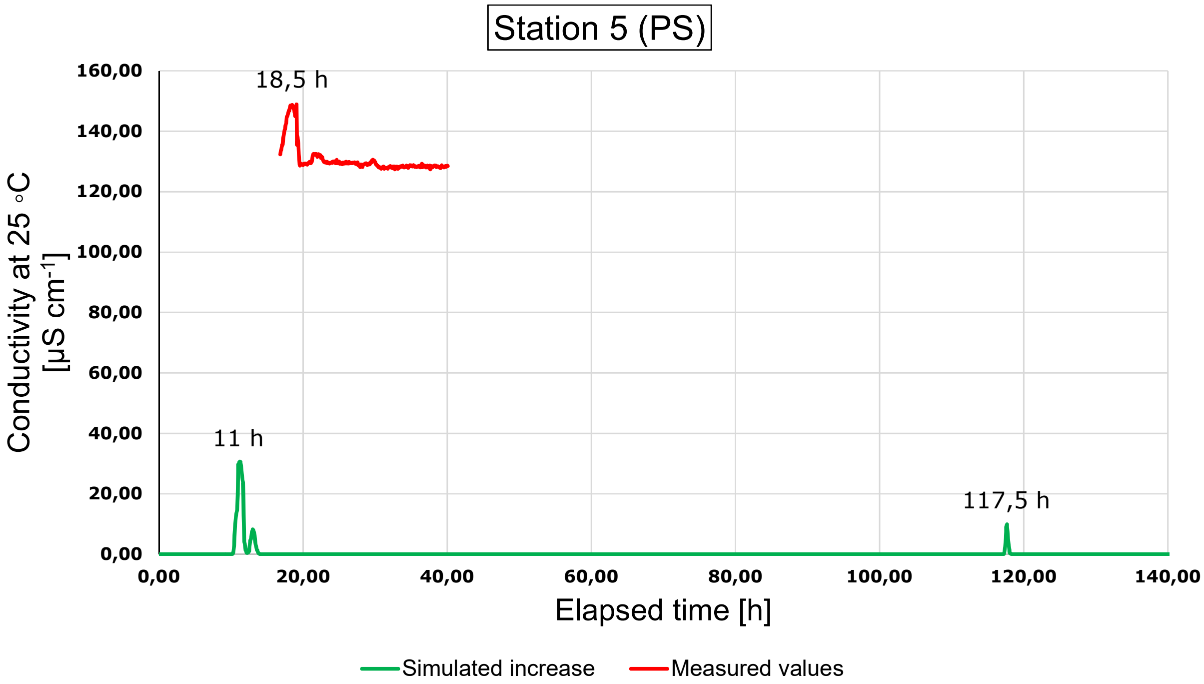


Figure 20: Simulated and measured conductivity values at measurement station 5 (Pumping station)

5.1.6 Measurement station 6 (pumping station)

Figure 21 shows that the predicted peaks in conductivity were not detected at station 6 during the measurements there. Furthermore, measurements indicate that conductivity had a slow and relatively steady increase.

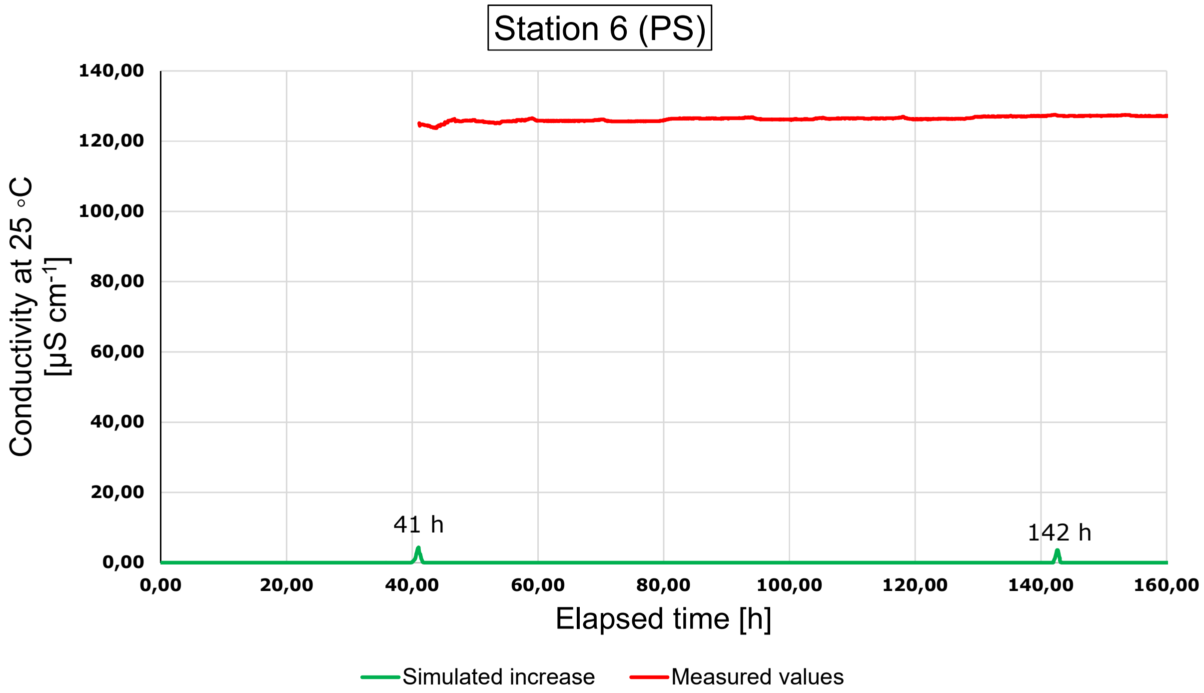


Figure 21: Simulated and measured conductivity values at measurement station 6 (Pumping station)

5.1.7 Sensitivity analysis

Figure 22 uses station 4, the flow-through basin, to illustrate how sensitive the simulations are to assumptions in the water network model (WNM). The red graph is the simulated tracer arrival, before two modifications were performed in the WNM. The green graph is the result of:

1. Taking into account that an important pumping station was manually deactivated between 17:00 and 21:25 on the day of tracer addition
2. Closing a water main from one of the storage basins, according to descriptions given by Trondheim Bydrift

These two modifications resulted in a peak arrival time difference of approximately 30 minutes, which is considerable.

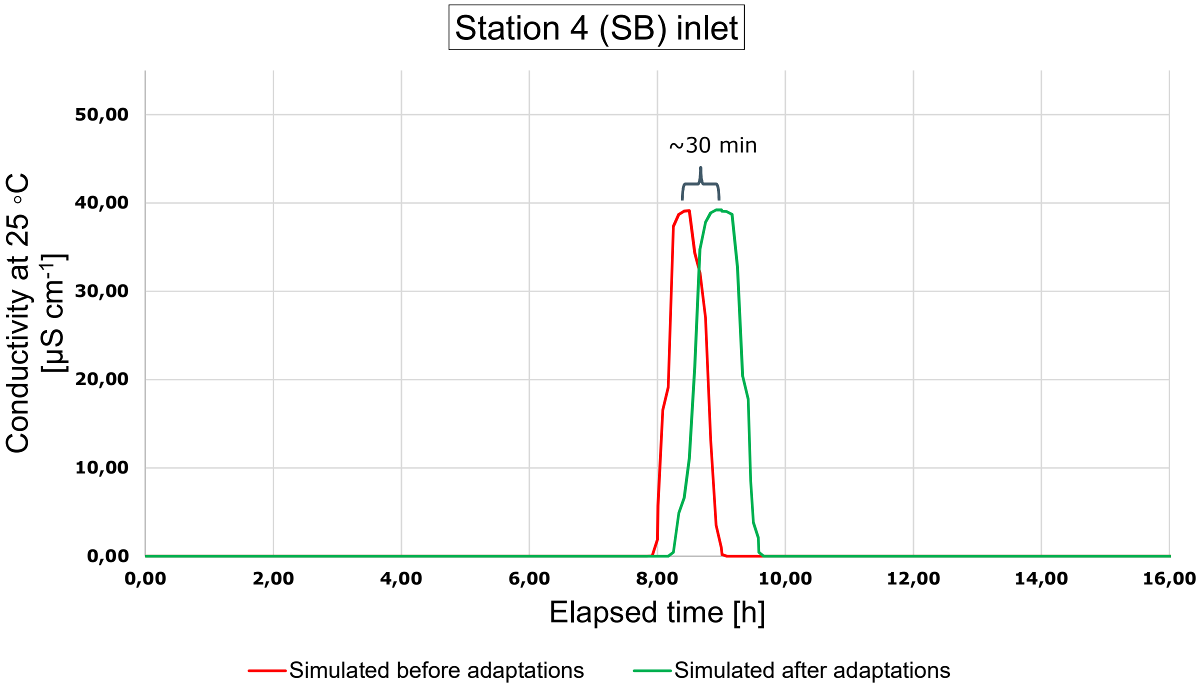


Figure 22: Tracer arrival time at station 4 (storage basin) as a consequence of adjusting settings in the water network model

5.2 Estimated water age distribution

An extended period simulation was run in EPANET using a Matlab® script, see Appendix A, in order to provide estimates of the water age distribution. The simulation time was set to 120 days. Figures 23 and 24 present the resulting average and maximum age values, which are calculated based on the last 48 hours of the simulations. Maps were produced using QGIS 3.4.15. The color coding is inspired by the water age maps that can be extracted directly from EPANET. The legend in Figure 24 has been added a purple water age bin at 120 days, in order to highlight nodes where the water age is not converging.

A stability analysis of the water age results is presented in Figures 25 and 26. In Figure 25, four different nodes were selected in order to illustrate the variations in water age stability between two different storage basins and two different junctions in the network. The identity and location of the nodes is not disclosed due to confidentiality reasons. Figure 26 puts emphasis on the two nodes in Figure 25 that have not stabilized with a 30-day simulation time, and gives a comparison with the behavior observed when the simulation time was increased to 120 days.

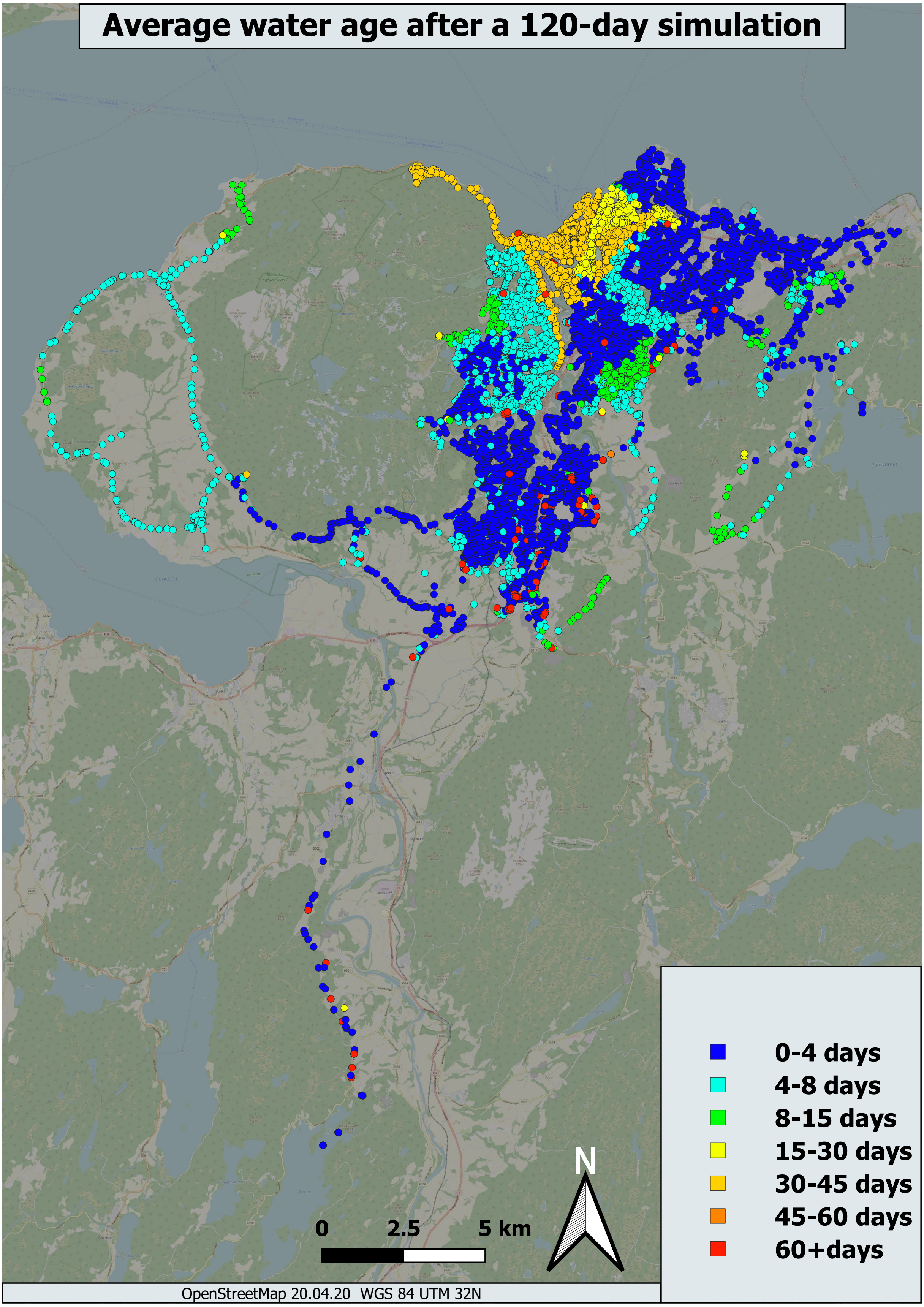


Figure 23: Estimated average water age in the drinking water distribution system after 2880 h (120 days) simulation. Average values were calculated for the last 48 hours of the simulation

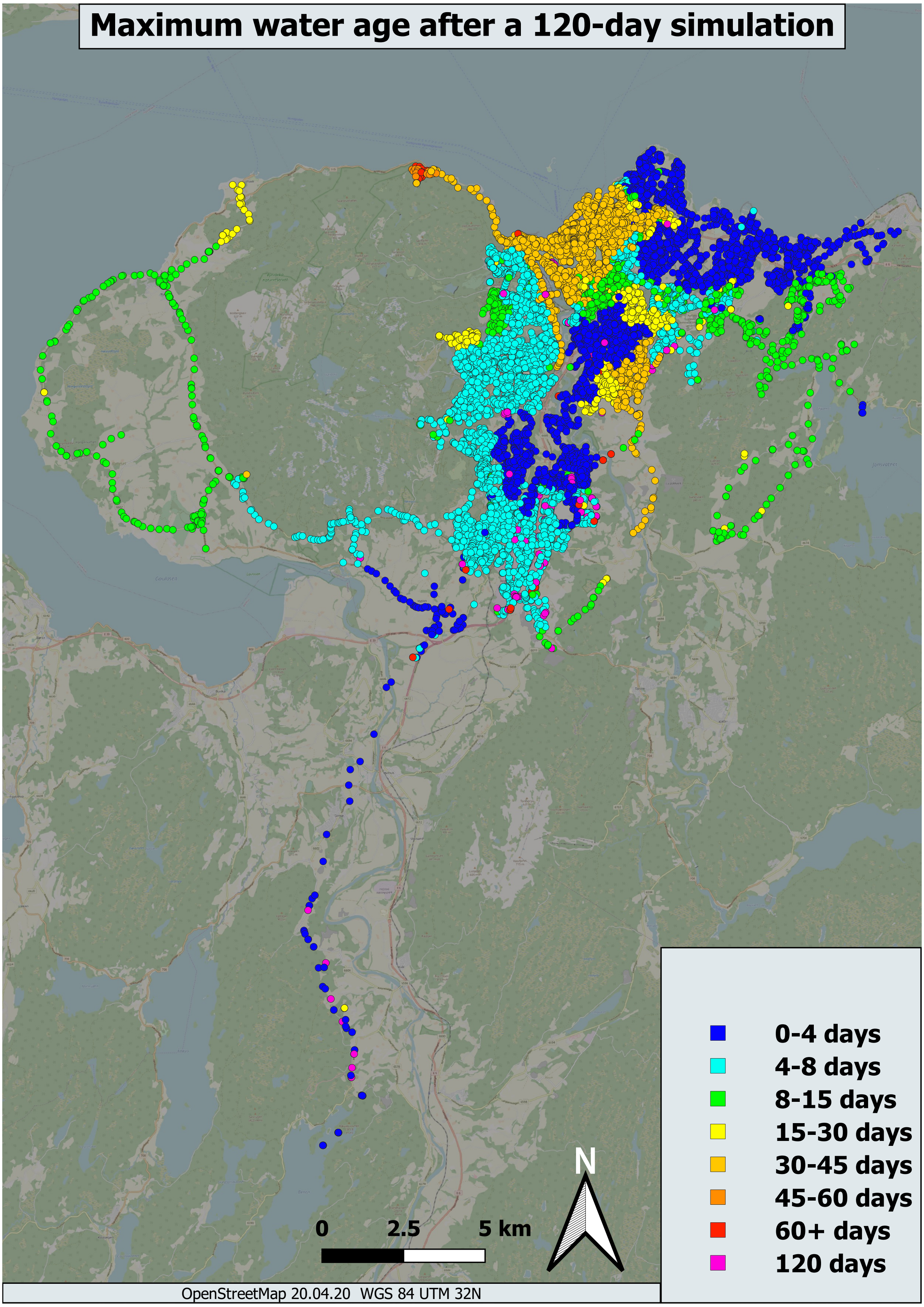


Figure 24: Estimated maximum water age in the drinking water distribution system after 2880 h (120 days) simulation. Maximum values were calculated for the last 48 hours of the simulation

5.2.1 Stability analysis of water age simulations at four different nodes

Figure 25 [A] below, associated with a storage basin, stabilizes after approximately 400 simulated hours, with a water age varying between 120 and 135 hours. Figure 25 [C], a junction in a residential area, reaches stability at approximately the same time, but has a much larger variation in age, around 100 hours. Figures 25 [B] and [D] have not stabilized after 720 hours (30 days) of simulated time.

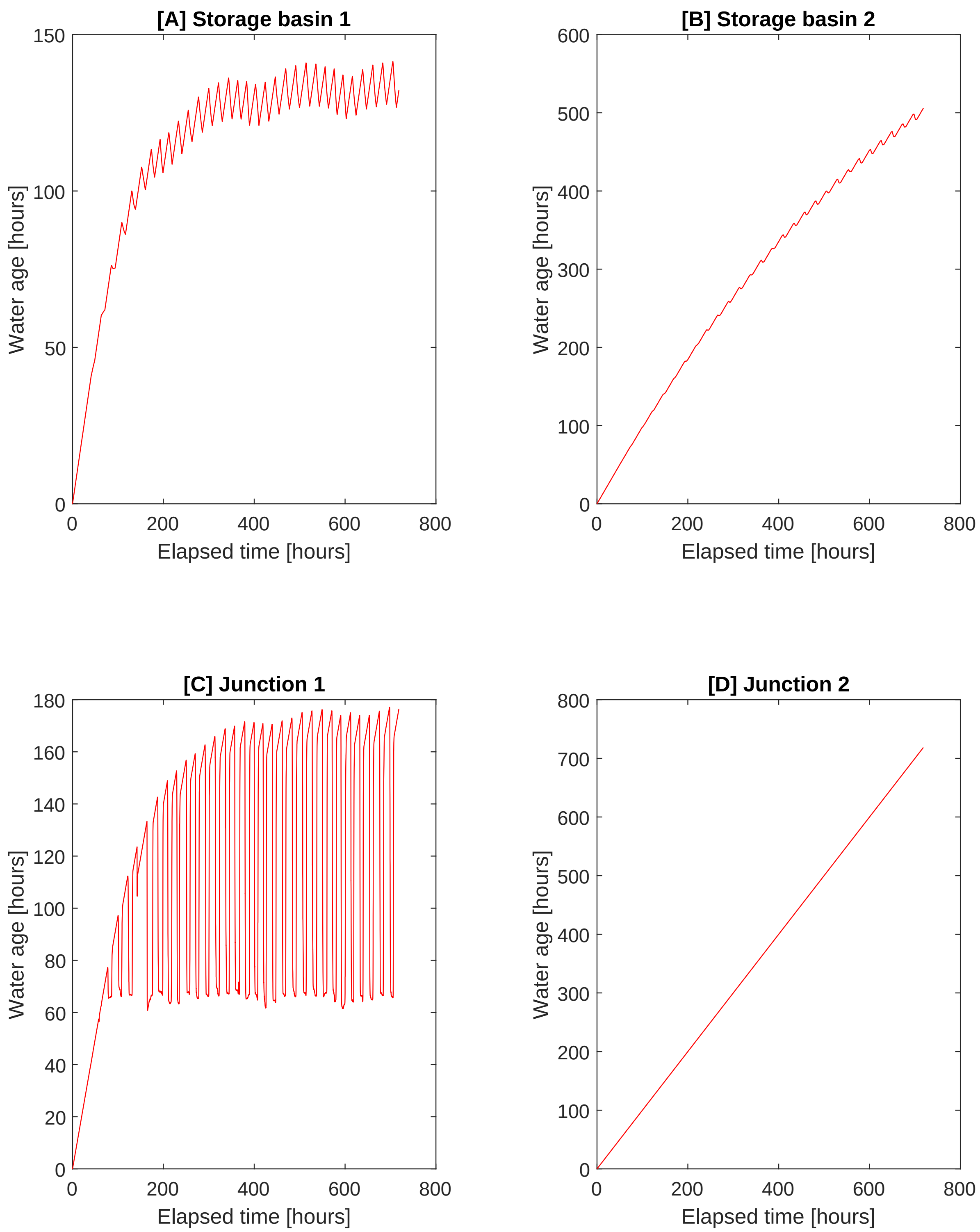


Figure 25: Estimated water age after a 720 h (30 day) simulation at four selected nodes. [A] and [C] show a stabilized behavior, whereas [B] and [D] do not.

5.2.2 Analysis of water age simulation instability at two nodes

Figures 26 [A] and [C] show that a 4× increase in simulation time did not make the water age stabilize at Junction 2. In both simulations, the age at that node increased at exactly the same rate as the elapsed simulation time. The explanation for that was found in the WNM, where this specific node has a water demand of zero. Figures 26 [B] and [D] indicate that the water age in Storage basin 2 became fairly stabilized after extra simulation time was added. This slow convergence may be due to limited through-flow in the model-version of the basin, or that convergence takes extra time because the node is in a peripheral area of the drinking water distribution system (DWDS).

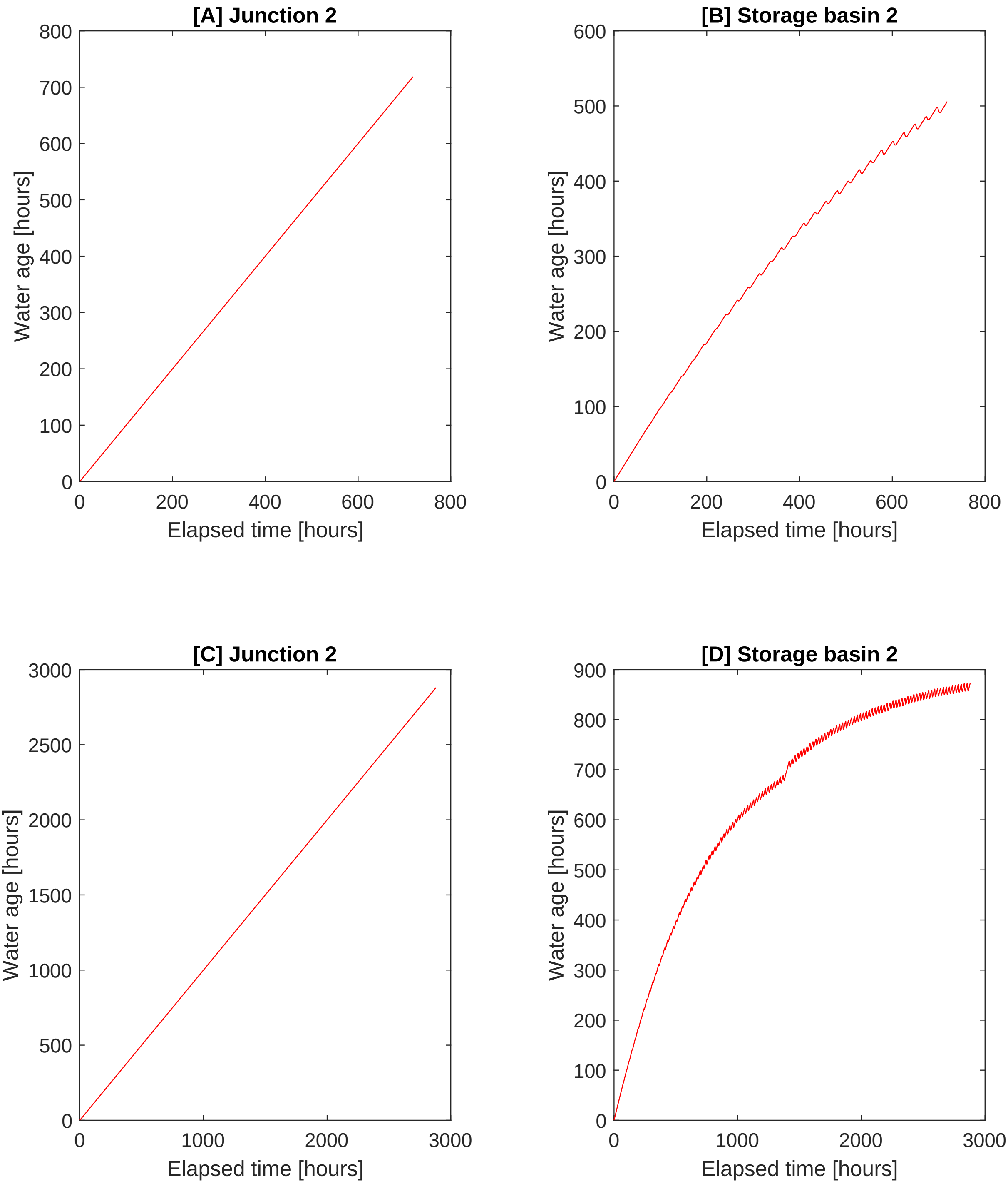


Figure 26: Comparison of water age stability for Junction 2 and Storage basin 2 with two different simulation times. [B] and [D] indicate that Storage basin 2 is reaching a stable solution. [A] and [C] indicate that the water age at Junction 2 is not converging

5.3 Statistical analysis of water quality data

Heterotrophic plate count (HPC) may be influenced by water temperature, water age, and other water quality parameters [3]. A brief statistical analysis was conducted to assess various possible effects and relationships related to HPC.

5.3.1 Effects of season and HPC collection site

In the R script [52], it was convenient to define two 6-month seasons for the HPC data (autumn and spring were ambiguous and monthly data were inconsistently available at some sites). Because the raw water temperature was expected to have an important influence on DWDS water temperature, seasons were roughly defined based on raw water temperatures (Figure 27; plotted using an R script by Waak [53]). Raw water temperatures from lake Jonsvatnet were used for 2015, as the annual pattern is fairly consistent from year to year. These typically stayed below 5 °C throughout the year. There was an apparent seasonal variation, where the temperature was fairly stable at 4 °C from June until December ("summer") and then became more dynamic—dropping to as low as 1 °C—within the period of December to May ("winter").

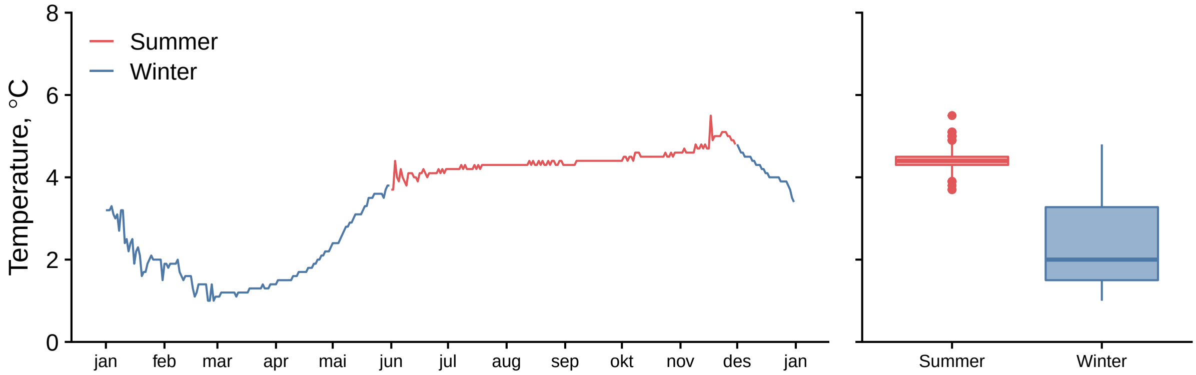


Figure 27: Raw water temperatures in Jonsvatnet from 2015

The seasonal effects may also vary at different sites of interest in the DWDS, more or less related to water age at each site (but potentially influenced by other factors, such as placement above or below ground for storage basins, which may have different dampening effects on water temperature). Several sites were chosen initially based on a list of 14 regular water sampling sites, and then further limited to sites with adequate seasonal data available and that also likely only represented water from Jonsvatnet for the period of 2016 to 2020. Benna water was excluded to keep the regression straightforward, and also because limited data exist about when and to what extent Benna water interacts with Jonsvatnet water.

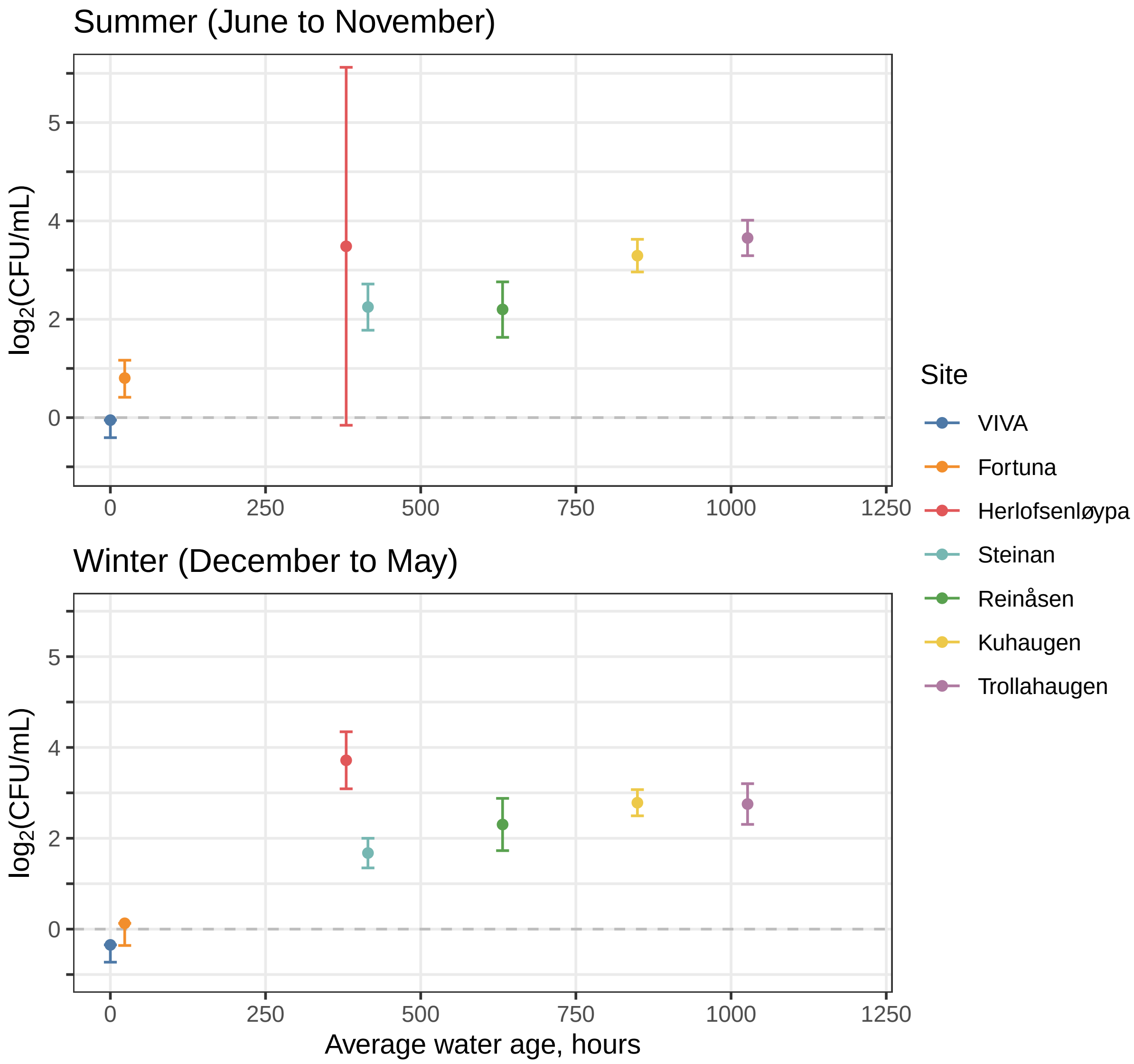


Figure 28: HPC (mean ± 95 % confidence limits) versus average water age (mean of the last 48 hours of the 120-day simulation). Dashed line indicates the lower detection limit for the HPC method. All locations except VIVA are storage basins.

After visualizing the data by site and by season (Figure 28), a likelihood-ratio chi-squared (χ^2) analysis of deviance was used with a parametric survival regression model from the "survival" package [12, 40, 52] to determine whether HPC sample collection site, season, and/or site/season cross effects were significant predictors of HPC (Table 5). Note that this test does not indicate predictor strength (size of the effect). It only indicates whether the effect is significant. The test results indicated that:

1. HPC likely varies significantly among sampling sites, since P <2.2 × 10^{-16} (***)
2. HPC likely varies between summer and winter, since $P = 0.01$ (*)
3. HPC likely varies by site by season ($P = 0.004$ (**)), meaning that some sites are significantly affected by season

Table 5: Likelihood-ratio chi-squared (χ^2) test for site and season as predictors of HPC

Effect Type	LR χ^2	Df	Pr(>χ^2)	Significance
Site	189.244	7	<2.2 × 10^{-16}	***
Season	6.016	1	0.014	*
Site:Season	20.753	7	0.004	**
Significance codes for P values:	0 '***'	0.001 '**'	0.01 '*'	

5.3.2 Pearson's correlation test on HPC and water age data

Based on initial visualization of the data (Figure 28), there appeared to be a positive correlation between higher mean water age and mean HPC, so a one-sided Pearson's product-moment correlation was used to test this hypothesis (Table 6, using a script from Waak [52]). Two conclusions were drawn from the results:

1. The low P value of 0.00046 indicated a significant positive correlation between HPC and water age
2. The correlation coefficient (R) of 0.78 indicated a strong relationship between HPC and water age (95 % confidence interval for Pearson's R was $[0.51, 1.00]$)

Table 6: Pearson's product-moment correlation of mean water age and mean HPC

Parameters	Values
t value for hypothesis test	4.367
df (degrees of freedom)	12
P value	0.0005
95 % confidence interval for correlation	0.507, 1.000
Correlation (Pearson's R)	0.784

Water age estimates were expected to have inaccuracies, but it was also expected that the general ordering of water age from lowest to highest at each site was stable. Therefore, Spearman's rank correlation was used to provide a more robust, but also less powerful, alternative estimate of correlation. Like in Pearson's test, the Spearman's rank correlation coefficient indicated a significant and strongly positive correlation (Spearman's $\rho = 0.65$, $P = 0.0062$).

5.4 Pipes with flow reversals and low velocities

A 24 hour simulation was run in EPANET using a Matlab® script, see Appendix A, in order to predict which pipes in the water network model experienced a combination of flow reversal and low flow velocities during a day of normal DWDS operation. The values for minimum, maximum, and median flow velocities in Table 7 were used to select which of those pipes to highlight. Figure 29 gives a coarse overview of the results, and indicates the approximate location of the highlighted pipes.

Table 7: Selection criteria for min, max and median values of flow velocity

Parameter	Selection criteria
Min. velocity	$0\ \mathrm{m\,s^{-1}}$
Max velocity	$0.2\ \mathrm{m\,s^{-1}}$
Median velocity	$>0\ \mathrm{m\,s^{-1}}$

Figure 29: Overview map highlighting the pipes (in black) that had flow-reversals and met specified flow velocity requirements in the model

6 Discussion

This section is structured around the research questions:

1. Why is it relevant to perform drinking water quality modeling in the DWDS of Trondheim kommune?
2. How is water age distributed in this DWDS, and what is the reliability of these estimates?
3. How does water age relate to microbiology in the DWDS?
4. Which water pipes should be targeted for biological sampling in the future?

6.1 The relevance of drinking water quality modeling in Trondheim kommune

In a DWDS supplying roughly 230 000 people, operation and management practices can potentially have a large impact on public health. This is also true for the biological processes taking place in raw water sources and the distribution system. Although less serious than health concerns, public perception of the aesthetic water quality is relevant in this respect as well. If, for instance, the water utility decides to shut down a water main for maintenance, altered flow conditions in the local DWDS may mobilize biofilm and particles, giving discolored water. This risk may be reduced if a WNM is used in advance to predict the altered flow conditions, and pipe flushing is done in the affected area before the water main gets closed off.

When combined, hydraulic and water quality modeling may be a powerful tool in maintenance planning, system optimization, or risk assessments to mention a few. Selecting locations for water quality sensors (e.g. turbidity), was mentioned by Hem and Østerhus [16] as a process where models can be useful. Trondheim kommune has already used a digital WNM for several years, which means that there is personnel in both the water utility, local consulting companies, and the university, with the skills needed to use it. On the subject of risk assessment and management, a drinking water quality model can be particularly useful for modeling the propagation of a pollutant in the DWDS. Such incidents, where unwanted compounds enter the DWDS unintentionally or as a consequence of sabotage, can be more effectively dealt with if the propagation over distance and time can be predicted by the model. During discussions with the operators at VIVA, this was brought up as an important aspect.

6.2 Estimated water age distribution

The water age maps in Figures 23 and 24 show that the major share of nodes in the WNM have a water age, both average and maximum, below 8 days. The stability analysis indicated that it is necessary, at least for Storage basin 2 in Fig. 26, to have a simulation time of 120 days or more in order to get fairly stable water age values. Therefore, the 120-day average water age map in Fig. 23 is considered as the main result for general descriptions of the water age distribution in the DWDS. In light of the tracer study, it is reasonable to expect that the water age has been underestimated to some extent. No attempt has been made here to quantify this, although it should be done if the work is continued.

If residual disinfection would be implemented in the future by Trondheim kommune, for instance during an emergency situation affecting microbial water quality, Figure 23 indicates that certain areas of the DWDS can lose the suppressing effect of the disinfectant due to breakdown processes happening over time. Further assessments would be needed to determine which water age value would be the maximum acceptable value in that situation, but it is not unlikely that the nodes colored in yellow, orange and red will be exposed. It is worth noting that the central city areas "Midtbyen, Ila and Øya", which are old parts of Trondheim and may have a considerable share of older pipes, are colored in yellow and orange. It is reasonable to expect more internal corrosion in older pipes, which creates favorable conditions for biofilm, and the combination of high water age and biofilm can reduce the effect of residual disinfection. Such information could be used by Trondheim kommune to assess the need for emergency disinfectant boosters at certain locations in the DWDS.

6.3 Tracer modeling and tracer study

Conductivity can be expected to rise by approximately $30\,\mu S\,cm^{-1}$ when the NaCl concentration increases by $20\,mg\,L^{-1}$ [35]. Figure 5 predicts a rise in conductivity of $39\,\mu S\,cm^{-1}$ with exactly the same increase in NaCl concentration. This suggests that K, the correction factor in Equation (4), may be set too low. Figure 15, for instance, indicated that this is true. At that station, measured peak values of conductivity were approximately $10\,\mu S\,cm^{-1}$ lower than simulated. No changes were made to the correction factor, but this information could be used to calibrate it.

It was shown that the simulations have consistently underestimated tracer peak arrival times, and this is most likely caused by several different factors. It can be seen in Figure 13 that the measured water flow volume into the DWDS is significantly lower, roughly half of what the model predicts when 24 hours have passed. This explains why the model underestimates peak arrival times, because a higher water flow gives higher flow velocities in the model, thereby reducing peak arrival times. Another factor can be the starting conditions in the model, such as initial filling levels in storage basins. If the model assumes different filling levels than what is occurring in reality, it can have an effect on flow. The $200\,\mathrm{L\,s^{-1}}$ difference between starting flow rates in Figure 12 may be explained by that. One modification that may reduce the difference in basin level starting conditions, would be to start the tracer simulation when the model has been run for several simulation days in advance, similar to what was done for the water age maps. Seasonal variations or variations in the activity level in Trondheim, may also be contributing to the observed difference in flow rates. It is not unlikely that the pandemic, which lead to a reduced overall activity level in society, made the total water consumption decrease enough to impact the results in this study.

The conductivity graphs from the tracer addition session at VIVA (Figure 14) clearly illustrated the difference between the idealized behavior in the model, and the realistic behavior in the field, related to the addition and mixing of tracer at the DWTP. While the simulated graph was perfectly steady and dropped down immediately after tracer feed cut-off, the real-world graph stretched longer in time and had a gradual development. This difference in the "stretching" of the graphs is likely to have contributed to the difference in simulated versus measured peak arrival times. If a contact basin would be included in the model, equivalent to how the DWTP is built, it can be expected that the deviation between simulated and observed will be smaller. In this thesis, only a simplified approach was used to account for residence time in the contact basin. The starting time of the simulation was given a 30 minute delay, based on information about theoretical residence time in that specific contact basin. Another issue that is shown in Figure 14 is the large difference between the predicted and observed increase in conductivity. At most, the measured rise in conductivity was more than twice as high as predicted. This was due to the choice of measurement node in the model, which did not take into account that there are two separate filtration lines, supplying two partly separated contact basins. The blue graph in Figure 14 shows the flow rate through contact basin 2, which was roughly 50 % of the total DWTP flow rate seen in Figure 12. As a result of this, the tracer concentration was much higher at the sampling point, before it was later reduced when the main water lines were joined in the DWTP. The issues related to measurements could have been eliminated by selecting a sampling point further downstream in the DWTP. Still, the advantage of keeping the original sampling point was that more accurate information was obtained about residence time in basin 2, as well as the mixing conditions between contact basins 1 and 2.

The storage basin connected with measurement station 3 was specifically chosen because it is a side basin. In the WNM, the chosen mixing model was FIFO (first-in-first-out plug flow). Figures 16 and 17 show the simulated and measured conductivity values, which display significant differences. Only two distinct changes in conductivity were predicted by the model, whereas the measurements show frequent distinct changes over several days. Figure 17 combines the measured conductivity with the measured inflow and outflow at this storage basin. It shows how tracer-containing water was firstly pumped in, and that the conductivity increased every time water was being drawn out of the facility. An attempt was made to simulate again with the LIFO (last-in-first-out plug flow) mixing model, and compare the results, but this produced no significant changes. Still, it may be useful to consider a different mixing model, and do further simulations to make new assessments.

Figures 18 and 19 show the simulated and measured conductivity at station 4, a flow-through storage facility with 3 basins. The inflow graphs differ from each other in several ways, but the outflow graphs display a similar behavior. The distinct peak that was measured at the inlet gave very little response at the outlet, which indicates that the tracer was thoroughly mixed into the large basin volumes.

The results from measurement station 6 in Figure 21 cannot be used to make a certain assessment of tracer peak arrival time. There are several potential causes of this, related to both system design and day-to-day operation. When water from VIVA is delivered to station 6, it may pass through 3 large storage basins on its way. This will lead to a significant dilution of the limited tracer "package". Furthermore, station 6 was supplied from Benna VBA up until 6 days before the tracer study was initiated. This means that water from Benna was most likely still present to some extent in two storage basins associated with station 6, and its conductivity may have interfered with the field measurements. It may also be the case that a tracer peak had already gone past station 6 when the data logger was activated.

The two modifications that were done in the WNM, were intended to make the model more adapted to the operational conditions during the tracer study. Figure 22 shows that the modifications resulted in 30 minutes less underestimation of tracer peak arrival time at station 4, and the difference between simulated and measured arrival time decreased by approximately 7 %. A review of this result, as well as the other tracer study results, indicates that calibration will be needed to reduce WNM inaccuracies. This would add value to future water quality simulations, and also improve conditions for a future model validation.

6.4 Statistical analysis of water quality data

The consistent underestimation of tracer arrival time, most likely leading to underestimation of water age, should be taken into consideration in the following discussion about statistical analyses. This is because the underestimation was not corrected before the average water age data was used in statistical analyses. The results are almost certainly affected by this, but if consistent underestimations has only lead to a linear shift of the water age data, it is likely that the conclusions from correlation analysis were not affected.

The results from Pearson's correlation analysis showed that there was a strong, positive correlation between simulated mean water age and calculated mean HPC at the selected sampling sites. Spearman's rank coefficient gave the same conclusion as well. It was also shown that the sampling site itself had a significant effect on HPC, and that the season (summer or winter) affected some sites more than others. Once again, as WHO [25] and Blokker et al. [3] pointed out, it is important to note that HPC is not a direct indicator for the safety and quality of drinking water, but it can give information about microbial processes that have direct effects on the safety and quality. Growth and dispersal of *Legionella* bacteria is one such process [55]. In follow-up research, further investigations can be conducted to see if water age is a contributing factor at locations where water quality problems occur. This can, for instance, be locations where pathogenic *Legionella* is suspected to be causing problems.

6.5 Pipes with flow reversal and low velocities

The results in Figure 29 showed that the WNM predicts several of the pipes in the DWDS to have the specified flow conditions in Table 7 during the 24 hour simulation period. Some of the highlighted pipes appear to be very short, in the order of a few meters, whereas other pipes stretch over several hundred meters. It is likely that some pipes were excluded from the group of highlighted pipes due to minor instabilities in the simulation. For instance, if a spike in flow velocity was recorded at only one timestep, due to a simulation instability, that pipe would be excluded from the group of highlighted pipes regardless of the flow velocity it had outside that unstable timestep. The Matlab® script could have been modified to allow some spikes in flow velocity to occur without pipes becoming excluded, but this was not done due to time limitations in the thesis work.

For future microbiological investigations in the DWDS, especially those related to biofilm, the highlighted areas may be extra interesting to target for sampling. Blokker et al. [2] found that pipes that match the flow velocity criteria in Table 7 are likely to have excessive biofilm formation and sediment deposition. Due to model inaccuracies uncovered by the tracer study, it is recommended that a new calibration is performed before using this method to target pipes. For the water utility, the information can also be used as input to the planning of pipe flushing, and flushing tests on a selection of the highlighted pipes offers opportunity to evaluate the accuracy of these predictions.

7 Concluding remarks

In this thesis, water age was estimated in the drinking water distribution system (DWDS) of Trondheim kommune using a water network model (WNM). The resulting 120-day water age maps showed that the majority of nodes in the model are below 8 days on average, and that some areas of the DWDS can have significantly higher age values. A full-scale tracer study was conducted for the first time in Trondheim kommune, and used in assessment of model predictions, as well as providing empirical data about the travel time of water to specific locations in the DWDS. The tracer study showed that simulations have consistently underestimated tracer peak arrival times, which gives reason to expect that the water age simulations are also exposed to underestimation. Performing additional calibration of the model is expected to reduce this issue, and may also present an opportunity to validate the model.

Furthermore, statistical methods indicated that there was a significant positive correlation between mean water age and heterotrophic plate count (HPC) for the tested water sampling sites. It was shown that site and season were significant predictors of HPC at seven monitoring sites of varying water age in the Trondheim DWDS. This forms a basis for further studies into the relation between microbiology and water age, as well as other hydraulic parameters that can be obtained from the water network model.

Model simulation results indicated that a number of pipes in the DWDS may have flow conditions that make them more exposed to fouling and sedimentation. Due to the inaccuracies uncovered in tracer simulations, it could be wise to put more emphasis on the method than on the highlighted pipe map. The method for locating such pipes can, preferably after the model has been calibrated, be further developed and used to target specific pipes for biofilm and sediment sampling.

8 Outlook

The tracer study results indicate that it would be worth it to calibrate the water network model (WNM) further, if water age analyses are to be continued. Keeping the model up to date with structural changes in the drinking water distribution system (DWDS) will also be important if a high level of accuracy in the model is to be maintained. In case results from this thesis are to be used in calibration, the following steps can be considered:

1. Perform an extensive mapping of how operational conditions in the DWDS were during the tracer study time period, and make adjustments to the model based on this, so that initial conditions in new simulations are closer to reality
2. Adjust water demands in the model, compare simulation results with tracer study results, and assess whether this provides sufficient calibration
3. Collect statistical data about how water demand is distributed over time in Trondheim kommune, long enough to be comparable with water age simulations, and use the data as a basis for recalculating water age

If water age studies are to be continued in Trondheim kommune, it would also be a good idea to perform simulations for all three possible supply scenarios. Future tracer studies can also be adapted to different supply scenarios, and potentially increase the knowledge about how each scenario affects the DWDS. As an example, if Benna is to temporarily supply the entire DWDS in the future, a "water fingerprint study" focusing on color can be conducted when the supply situation is altered.

Through the experimental work in this thesis, it has been shown that a full-scale tracer study can be conducted efficiently in Trondheim, yielding clear results at a low tracer-dosage, and that it is not particularly resource-intensive. If another tracer study is to be performed in the future, the following aspects could be useful to consider:

- Conductivity measuring equipment can most likely be connected with the SCADA system found at every pumping station and storage basin. This could be a practical and cost-effective way to greatly increase the number of measurement stations. This would also allow for real-time monitoring of data, and gives greater opportunity to compare conductivity with measured flow rates. In a "water fingerprint study," measuring equipment for color could provide similar functionality.
- Local tracer studies at a smaller scale, e.g. in residential areas, can be worth considering. This method may be better suited for peripheral areas, such as Byneset, where no conductivity increase was detected in the field experiment.

For future research and development connected with the WNM, implementation of stochastic demand patterns could be a good idea. This could be combined with an effort to expand and transition the model into an "all nodes and all pipes" type, as described by Blokker et al. [4]. Smart residential water meters could provide specific statistical water consumption data if implemented in the future. This leads on to the next subject, which is the inclusion of weekly, monthly or annual demand variation patterns. One area where such model functionality could be useful is water quality sampling, which is done on a regular basis by Trondheim Bydrift. Adding seasonal variation functionality to the model could make it better equipped to capture the seasonal variation that was found within the microbial water quality samples. Having water temperature measurements at many parts of the DWDS, for instance through the SCADA system, would add value to analyses regarding seasonal variations. Lastly, it is worth mentioning that Blokker et al. [3] used an advanced non-linear, multivariate statistical technique called "self-organizing maps" to analyze the effects between water age, temperature, and a wide range of quality parameters. This approach may perform better than linear regression, so using a similar approach in this thesis would have taken statistical analysis one step further than what was done here.

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Appendix A: Matlab® code

Script for tracer modeling

This script is based on a script written by Rokstad [31]. It has been anonymized by replacing real station names with “station name”, and by replacing the file path with “file path”.

```
% https://github.com/OpenWaterAnalytics/epanet-matlab
% Specifiy path and filename:
fileNameOrigModel = 'model file path';
T=5420;

% Start epanet toolkit:
global EN_CONSTANT
ENMatlabSetup('epanet2','epanet2.h');

% Open model:
[~] = ENopen(fileNameOrigModel, 'report.hyd', 'binTest');

% Initialise:
[~]=ENopenH();
[~]=ENinitH(0);

% Set quality type:
ENsetqualtype(1,'CHEM','mg/L','');

% Set trace node to be a setpoint booster:
ENsetnodevalue(3115, 7, 2);

% Get number of links and nodes:
[errcode, linkCount]=ENgetcount(2);
[errcode, nodeCount]=ENgetcount(0);

x=cell(nodeCount,1);
y=cell(linkCount,1);
for i=1:1:nodeCount
[~,x{i}]=ENgetnodeid(i);
end
for i=1:1:linkCount
[~,y{i}]=ENgetlinkid(i);
end

% dT=zeros(1,T);

%Get initial pressures:
%Pumping station 2 is switched off from 16:55 to 21:25
for i=1:1:T

    if sum(dT(1,1:i))/3600>=1
        if sum(dT(1,1:i))/3600<=5.5
            ENsetlinkvalue(10648, 11, 0);
        else
            ENsetlinkvalue(10648, 11, 1);
        end
    else
        ENsetlinkvalue(10648, 11, 1);
    end

    [~]=ENrunH();
     fprintf('Model timestep %i\n',i);

```

```
    [errcode,f_2(i)]=ENgetlinkvalue(10648,8);
    [errcode,f_6(i)]=ENgetlinkvalue(5102,8);

    if i<=T
        [~,t]=ENnextH();
        dT(i) = t;
    end
end

disp(max(cumsum(dT))/3600);

% Solve hydraulics:
[~] = ENsolveH();

% Reset dT
dT=zeros(1,T);

% Open and initialise quality:
[~]=ENopenQ();
[~]=ENinitQ(0);

for i=1:1:T

    %For the first hour,VIVA trace node has a concentration of 20.
    %For the rest of the run, concentration is 0.

    if sum(dT(1,1:i))/3600<=1
        ENsetnodevalue(3115, 5, 20);
    else
        ENsetnodevalue(3115, 5, 0);
    end

    [~]=ENrunQ();

    [errcode,station name(i)]=ENgetnodevalue(372,12);
    [errcode,station name(i)]=ENgetnodevalue(246,12);
    [errcode,station name(i)]=ENgetnodevalue(7,12);
    [errcode,station name(i)]=ENgetnodevalue(9496,12);
    [errcode,station name(i)]=ENgetnodevalue(9497,12);
    [errcode,station name(i)]=ENgetnodevalue(9498,12);
    [errcode,station name(i)]=ENgetnodevalue(9499,12);
    [errcode,station name(i)]=ENgetnodevalue(3115,12);


    fprintf('Model timestep %i\n',i);

    if i<=T
        [~,t]=ENnextQ();
        dT(i) = t;
    end
end

% make tables with tracing results:
station name = table(cumsum(dT)'/3600,station name');
station name = table(cumsum(dT)'/3600,station name');
station name = table(cumsum(dT)'/3600,station name');
station name = table(cumsum(dT)'/3600,station name');
station name = table(cumsum(dT)'/3600,station name');
station name = table(cumsum(dT)'/3600,station name');
station name = table(cumsum(dT)'/3600,station name');
station name = table(cumsum(dT)'/3600,station name');
```

```
station name = table(cumsum(dT)'/3600,station name');
station name = table(cumsum(dT)'/3600,station name');

%Write the results to the excel file:
writetable(station name,'path.xlsx','Sheet',1);
writetable(station name,'path.xlsx','Sheet',2);
writetable(station name,'path.xlsx','Sheet',3);
writetable(station name,'path.xlsx','Sheet',4);
writetable(station name,'path.xlsx','Sheet',5);
writetable(station name,'path.xlsx','Sheet',6);
writetable(station name,'path.xlsx','Sheet',7);
writetable(station name,'path.xlsx','Sheet',8);
writetable(station name,'path.xlsx','Sheet',10);
writetable(station name,'path.xlsx','Sheet',9);


% Close hydraulics:
ENclose();

% Unload libraries
ENMatlabCleanup();
```

Script for 2880h water age simulation

This script is based on a script written by Rokstad [31]. It has been anonymized by replacing the file path with “file path”

```
% https://github.com/OpenWaterAnalytics/epanet-matlab
% Analysis time must be preset in the .inp file
% The 720 h simulation must be run first.

% Specifiy path and filename:
fileNameOrigModel = 'model file path';
T_2=21721;
%10768
% Start epanet toolkit:
global EN_CONSTANT
ENMatlabSetup('epanet2','epanet2.h');

% Open model:
[~] = ENopen(fileNameOrigModel, 'report.hyd', 'binTest');

% Set quality type:
ENsetqualtype(2,'AGE','mg/L','');

% Get number of links and nodes:
[errcode, linkCount]=ENgetcount(2);
[errcode, nodeCount]=ENgetcount(0);

% make list of node indices and IDs:
x=cell(nodeCount,1);
for i=1:1:nodeCount
[~,x{i}]=ENgetnodeid(i);
end

%Get initial pressures:
for i=1:1:T_2
    [~]=ENrunH();

    if i<=T_2
        [~,t]=ENnextH();
        dT(i) = t;
    end
end

% Solve hydraulics:
[~] = ENsolveH();

% Reset dT:
dT=zeros(1,T_2);

% Create result table for water age at all nodes:
ages=zeros(T_2,nodeCount);

% Open and initialise quality:
[~]=ENopenQ();
[~]=ENinitQ(0);

for i=1:1:T_2

    [~]=ENrunQ();

    % Store age for all nodes in table:
    for j=1:1:nodeCount
```

```
        [errcode, ages(i,j)]=ENgetnodevalue(j,12);
        end

        % Store elapsed time in hours:
        elapsed_time(i)=sum(dT(1,1:i))/3600;

        fprintf('Model timestep %i\n',i);

        if i<=T_2
            [~,t]=ENnextQ();
            dT(i) = t;
        end
end

disp(max(cumsum(dT))/3600);

% Calculate the average age and max age for all nodes during the last 48 h:
avg_age(1:nodeCount)=mean(ages(T_2-361:T_2,1:nodeCount));
max_age(1:nodeCount)=max(ages(T_2-361:T_2,1:nodeCount));
%407
% Get X and Y for all nodes:
[errcode,vx,vy,vertx,verty] = getNodeXY('model file path','epanet2','epanet2.h');

% Make one array with X,Y,avg_age and one with X,Y,max_age:
plot_avg_age_2=zeros(nodeCount,3);
plot_max_age_2=zeros(nodeCount,3);

plot_avg_age_2(1:nodeCount,1)=vx(1:nodeCount,1);
plot_avg_age_2(1:nodeCount,2)=vy(1:nodeCount,1);
plot_avg_age_2(1:nodeCount,3)=avg_age(1,1:nodeCount);

plot_max_age_2(1:nodeCount,1)=vx(1:nodeCount,1);
plot_max_age_2(1:nodeCount,2)=vy(1:nodeCount,1);
plot_max_age_2(1:nodeCount,3)=max_age(1,1:nodeCount);

% Covert arrays to tables:
avg_age_tab_2 = table(plot_avg_age_2(1:nodeCount,1),plot_avg_age_2(1:nodeCount,2),plot_avg_age_2(1:nodeCount,3));
max_age_tab_2 = table(plot_max_age_2(1:nodeCount,1),plot_max_age_2(1:nodeCount,2),plot_max_age_2(1:nodeCount,3));

% Write tables to .txt files ready to be exported to QGIS:
writetable(avg_age_tab_2,'file path.txt');
writetable(max_age_tab_2,'file path.txt');

% Store the age versus cumulative simulation time for selected nodes:
basin_1_1440h(1:T_2,1)=elapsed_time(1,1:T_2);
basin_1_1440h(1:T_2,2)=ages(1:T_2,9511);

junction_1_1440h(1:T_2,1)=elapsed_time(1,1:T_2);
junction_1_1440h(1:T_2,2)=ages(1:T_2,6633);

% Create data and 2-by-2 tiled chart layout
x_2 = elapsed_time(1,1:T_2);
% y1 = basin_2(1:T,2);
y2_1440h = basin_1_1440h(1:T_2,2);
% y3 = junction_2(1:T,2);
y4_1440h = junction_1_1440h(1:T_2,2);
tiledlayout(2,2)

```

```
% Top plot 1
ax1 = nexttile;
plot(ax1,x_1,y4,'Color',[1,0,0])
title(ax1,'[A] Junction 2')
xlabel(ax1,'Elapsed time [hours]')
ylabel(ax1,'Water age [hours]')

% Top plot 2
ax2 = nexttile;
plot(ax2,x_1,y2,'Color',[1,0,0])
title(ax2,'[B] Storage basin 2')
xlabel(ax2,'Elapsed time [hours]')
ylabel(ax2,'Water age [hours]')

% Bottom plot 1
ax3 = nexttile;
plot(ax3,x_2,y4_1440h,'Color',[1,0,0])
title(ax3,'[C] Junction 2')
xlabel(ax3,'Elapsed time [hours]')
ylabel(ax3,'Water age [hours]')

% Bottom plot 2
ax4 = nexttile;
plot(ax4,x_2,y2_1440h,'Color',[1,0,0])
title(ax4,'[D] Storage basin 2')

xlabel(ax4,'Elapsed time [hours]')
ylabel(ax4,'Water age [hours]')

% Close hydraulics:
ENclose();

% Unload libraries
ENMatlabCleanup();
```

Script for finding pipes with flow reversal and low flow velocities

This script is based on a script written by Rokstad [31]. It has been anonymized by replacing the file path with "file path"

```matlab

% https://github.com/OpenWaterAnalytics/epanet-matlab
% Specifiy path and filename:
fileNameOrigModel = 'model file path';
T=184;

% Start epanet toolkit:
global EN_CONSTANT
ENMatlabSetup('epanet2','epanet2.h');

% Open model:
[~] = ENopen(fileNameOrigModel, 'report.hyd', 'binTest');

% Set analysis time of 24 hours:
ENsettimeparam(0, 24*3600);

% Initialise:
[~]=ENopenH();
[~]=ENinitH(0);

% Get number of links and nodes:
[errcode, linkCount]=ENgetcount(2);
[errcode, nodeCount]=ENgetcount(0);

% make list of link indices and IDs:
x=cell(linkCount,1);
for i=1:1:linkCount
[~,x{i}]=ENgetlinkid(i);
end

% Make arrays for flow rates, diameters and velocity:
flow=zeros(T,linkCount);
diam=zeros(T,linkCount);
flow_vel=zeros(T,linkCount);

%Get initial pressures:
for i=1:1:T
    [~]=ENrunH();

    % Record link flows and diameters in arrays:
    for j=1:1:linkCount
        [errcode,flow(i,j)]=ENgetlinkvalue(j,8);
        [errcode,diam(i,j)]=ENgetlinkvalue(j,0);
    end

     fprintf('Model timestep %i\n',i);

    if i<=T
        [~,t]=ENnextH();
        dT(i) = t;
    end
end

disp(max(cumsum(dT))/3600);

% Calculate flow velocity in each pipe (m/s):
area=(diam.*diam).*0.25.*3.1416./1000000; % m2
```

```
flow_vel=(flow./1000)./area;

% Mark the links that have flow reversals, with a nonzero value:
rev_links=diff(sign(flow_vel),1,1);

% Return indices and timesteps of these:
[zC_row,zC_col]=find(rev_links);

% Remove duplicate indices:
indices=unique(zC_col);

% Calculate the mean velocity of these links throughout the day:
median_vel=median(abs(flow_vel(1:T,indices)));
min_vel=min(abs(flow_vel(1:T,indices)));
max_vel=max(abs(flow_vel(1:T,indices)));

% Make vector with indices of links chosen for plotting:
chosenlinks=[];
p=1;

% Keep only links with appropriate min, max and median velocity, put them in the vector:
for i=1:1:size(median_vel,2)
    if (min_vel(i) > 0) && (max_vel(i) < 0.2) && (median_vel(i) > 0)
        chosenlinks(p)=indices(i);
        p=p+1;
    end
end

% Print index of chosen links:
fprintf('The indices of chosen links are: [');
fprintf('%g ', chosenlinks);
fprintf(']\n');

% Make vector with one element for every link. Interesting links stay zero
% other links get dim color
plotvect=zeros(1,linkCount);
plotvect(1:linkCount)=0.8;
plotvect(chosenlinks)=0;

%Set plotting rules for nodes and links:
    PData.c = 'gray';
    PData.logtransv = 'n';
    PData.logtransl = 'n';
    PData.vmin = 0;
    PData.vmax = 0;
    PData.lmin = 0;
    PData.lmax = 1;
    PData.lwidth = 2;
    PData.vsize = 0;
    PData.tsize = 0;
    PData.legend = 'n';
    PData.DLLName = 'epanet2';
    PData.Hname = 'epanet2.h';

% Plot map where chosen links are highlighted:
NetworkFrame([],[],plotvect,'model file path',PData,[],[]);
% Close hydraulics:
ENclose();
% Unload libraries
ENMatlabCleanup();
```

Appendix B: Calculation of NaCl brine amounts

Water flow at VIVA

Normal	800,00 L/s
High	1000,00 L/s

Saturated brine is used at VIVA, max temp 15 c	
Approximate NaCl concentration	357,2 g/L

Tracer study: increase NaCl by	20 mg/L

Addition of salt at average:	16000 mg/S
	16,00 g/s
Required brine flow (Average)	0,045 L/s
	0,45 dL/s
	2,7 L/min

Addition of salt at high:	20000 mg/S
	20,00 g/s
Required brine flow (High)	0,056 L/s
	0,56 dL/s
	3,4 L/min

NORMAL	Needed brine volume, 1h experiment	161,3 L
	Needed salt amount, 1h experiment	57,6 kg
HIGH	Needed brine volume, 1h experiment	201,6 L
	Needed salt amount, 1h experiment	72 kg

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Temperature(C)	**Solubility**(gms/l)
$0°C$	356.5
$10°C$	357.2
$20°C$	358.9
$30°C$	360.9
$40°C$	363.7
$60°C$	370.4
$80°C$	379.3
$100°C$	389.9

Table 2.1: Solubility of NaCl in water variation with temperature.

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