CHALMERS
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Risk Based Economic Evaluation of Water Supply Safety Measures

A Case Study at Kvarnagården Drinking Water Treatment Plant

Master's thesis in Infrastructure and Environmental Engineering

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ABSTRACT

The awareness regarding the importance of a reliable and safe drinking water supply has been continuously growing during the past few years. Consequently, the focus on Water Safety Plans (WSPs), as suggested by the World Health Organization, has also increased. As the use of risk assessments increase, the need of decision support models, to valuate and compare alternative measures, is also increasing. With the purpose to show how risk assessment and decision support analysis can be combined, a case study was performed to evaluate the risk reduction and the benefit of the newly installed ultrafiltration (UF) membranes at the Kvarnagården Drinking Water Treatment Plant (DWTP) in Varberg, Sweden. The purpose was to determine the societal value of the UF membranes, by identifying and monetizing their associated costs and benefits. Quantitative Microbial Risk Assessment (QMRA) was used to estimate the health risk prior and after installation of the UF membranes. Including, sources of faecal contamination from onsite sewer systems around the water source, as well as more extreme events such as sewage pipe-breaks. *Campylobacter*, Norovirus and *Cryptosporidium* were used as reference pathogens in the calculations. The estimated health risk reduction was included in a Cost-Benefit Analysis (CBA) with additional aspects, e.g. aesthetic water quality benefits, to assess if the installation of UF membranes is beneficial from a societal point of view. The net present value (NPV) of the improved treatment plant was calculated using a discount rate of 3.5 % and 50-year time horizon. The results show that UF membranes reduce the microbial health risks significantly. In addition, the monetised mean NPV of the membranes was estimated to +47 MSEK, with a 5th and 95th percentile of -45 and +117 MSEK, respectively. Considering the uncertainties, the likelihood of having a positive NPV, i.e. the measures being societal beneficial, is 82 % for the Varberg case study. It should be notes that the aesthetic water quality improvements contribute significantly to the positive NPV. Furthermore, not all benefits have been possible to monetise, which means that there are additional benefits that should be taken into consideration when discussing the effects of the UF membranes.

Keywords: cost-benefit analysis, decision support, drinking water quality, quantitative microbial risk assessment, ultrafiltration, willingness to pay

Riskbaserad Samhällsekonomisk Analys av Åtgärder Inom Dricksvattenförsörjningen
En Fallstudie vid Kvarnagårdens Dricksvattenverk
Examensarbete inom infrastruktur och miljöteknik
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SAMMANFATTNING

Medvetenheten kring vikten av en pålitlig och säker dricksvattenförsörjning har under de senaste åren växt kontinuerligt. Som en följd av detta har också fokus på riskbaserat helhetstänk i linje med det Världshälsoorganisationen förespråkar också ökat. Det ökande antalet riskbedömningar som genomförs har också ökat behovet av beslutsstödsmodeller, för att värdera och jämföra olika åtgärder. Med syftet att visa hur riskbedömningar och beslutsstödsanalyser kan kombineras utfördes en fallstudie vid Kvarnagårdens dricksvattenverk i Varberg. I fallstudien utvärderades de nyligen installerade ultrafiltrens (UF) förmåga att reducera hälsorisken och den samhällsekonomiska nytta detta innebär. Syftet var att identifiera och monetärisera de kostnader och nyttor som UF-membranen är förknippade med i en samhällsekonomisk kostnads-nyttoanalys (KNA). Kvantitativ mikrobiell riskanalys användes för att uppskatta den hälsomässiga risken för dricksvattenkonsumenterna innan och efter installation av UF-membranen. De fekala föroreningskällor som ingick i analysen utgjordes av exempelvis enskilda avlopp intill vattenkällorna, men också mer sällsynta händelser som läckage från avlopp till följd av rörbrott. De referenspatogener som användes i beräkningarna var *Campylobakter*, Norovirus och *Cryptosporidium*. Utöver den reducerade hälsorisken, beaktades i KNA-beräkningarna också andra nyttor såsom vattnets estetiskt förbättrade kvalitet, i syfte att utvärdera huruvida UF-membranen är en lönsam investering ur ett samhällsekonomiskt perspektiv. Analysen gjordes med hänsyn till en tidsperiod på 50 år och en diskonteringsränta på 3,5%. Resultatet visar att UF-membranen har reducerat den mikrobiella risken signifikant och således säkerställt dricksvattenverkets rening av det dricksvattnet som distribueras ut till konsumenterna. UF-membranenens nettonuvärde (nyttor minus kostnader) beräknades till +47 MSEK, med 5- och 95-percentiler på -45 respektive +117 MSEK. Sannolikheten för ett positivt nettonuvärde uppskattats till 82%, där det framförallt var de estetiska nyttorna som bidrog till det positiva nettonuvärdet. Slutligen bör det även nämnas att alla nyttor inte har monetäriserats, vilket innebär att det finns ytterligare nyttor att ta hänsyn till i diskussionen kring UF-membranens samhällsekonomiska effekt.

Nyckelord: betalningsvilja, beslutstöd, dricksvattenkvalitet, kvantitativ mikrobiell riskanalys, kostnads-nyttoanalys, ultrafiltrering

ACKNOWLEDGEMENTS

Almost a year has passed since I began this thesis and I would like to thank my two supervisors, Andreas Lindhe and Viktor Bergion, as well as my examiner, Lars Rosén, at Chalmers University of Technology for all of your invaluable input and support during this time. I know you are all very busy and I've truly appreciated the time you've spared to aid me in my project. In addition, I would also like to thank my supervisor at VIVAB, Alexander Keucken, for your support and the provided data and connections, necessary to conduct the analysis.

It has been a pleasure working with you all!

Nils-Petter Sköld
Gothenburg June 2018

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LIST OF ABBREVIATIONS

As Low As Reasonably Practical	**ALARP**
Cost Benefit Analysis	**CBA**
Disability Adjusted Life Years	**DALY**
Drinking Water System	**DWS**
Drinking Water Treatment Plant	**DWTP**
Gross Domestic Product	**GDP**
Health-Related Quality of Life	**HRQL**
On-site Wastewater Treatment System	**OWTS**
Quality Adjusted Life Years	**QALY**
Quantitative Microbial Risk Assessment	**QMRA**
Ultrafiltration	**UF**
Ultraviolet radiation	**UV**
Water Safety Plan	**WSP**
Willingness To Pay	**WTP**
Years Lived with Disability	**YLD**
Years of Life Lost	**YLL**

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1 INTRODUCTION

Leonardo da Vinci is said to have stated that "water is the driving force of all nature". It is a statement that is probably more accurate today, than ever before. Water is not only a necessity for biological life, but it is also one of the few resources which integrate all of our behavioural fields. From the meals we eat to the technical devices we buy (Water Footprint Network, 2016). It is, therefore, particularly worrying to see how clean water is becoming an increasingly scarce resource on a global scale (Arnell, 2004; Vörösmarty et al., 2010).

The decreasing availability of clean water is primarily caused by anthropogenic pollution of the water sources as well as an unsustainable water withdrawal, for either drinking water or agricultural purposes. In addition, the problems linked to water scarcity is also expected to get worse, where factors e.g. climate change and increasing urbanisation are likely to increase the regional water stresses further (SOU, 2007; SWWA, 2007; UN, 2007; Murray et al., 2012; IPCC, 2014). Studies, by e.g. Vörösmarty et al. (2010), has found that 80% of the world's population is likely to be exposed to high levels of threat to their water security in the future. In response to these issues, a number of organisations, e.g. the World Health Organization (WHO) and the United Nations (UN), has flagged the lack of clean water as a global problem (WaterAid, 2018). Where the goal to "ensure availability and sustainable management of water and sanitation for all" is the sixth of the UN's 17 sustainable development goals (UN, 2015). Furthermore, the WHO (2017b) has also suggested an approach for assuring a clean and safe quality of the drinking water, referred to as a Water Safety Plan (WSP).

The WSP approach is an approach which can vary greatly in complexity, depending on the situation. Yet, it is meant to be implemented as comprehensive risk assessments, part of a larger risk management process, to encompass all steps in a drinking water system (DWS). From the withdrawal of water at the raw water source to the final distribution of drinking water at the consumer's tap (WHO, 2017b).

1.1 BACKGROUND

As the awareness and need of risk assessments of DWSs increase, the need of additional decision support tools also increases. In addition to risk assessment, a decision analysis enables a structure approach for evaluating the alternative measures for improving and securing the safety and security of DWSs (e.g. Lindhe, 2010). The application of available tools varies from straightforward health risk assessments, e.g. Quantitative Microbial Risk Assessment (QMRA), to more comprehensive models involving e.g. Cost Benefit Analysis (CBA) (Haas et al., 2014; Bergion et al., 2018).

The two methods mentioned above, i.e. QMRA and CBA, have in this thesis been applied to evaluate the risk reduction, costs and benefits related to implementation of ultrafiltration (UF) membranes at the Kvarnagården drinking water treatment plant (DWTP) in Varberg, Sweden. It is based on a similar model developed by Bergion et al. (2018), which aims to evaluate if a measure, e.g. a new treatment step, is feasible with respect to the needed resources and the benefits they provide, through a risk based approach.

The decision to implement the UF membranes, at Kvarnagården DWTP, was decided by the municipal company VIVAB and was based on previous decision support analyses (e.g. VIVAB, 2012). These decisions support analyses did, however, mainly focus on the health-related features of the treatment, rather than on the economic aspects.

1.2 AIM & OBJECTIVES

The overall aim of this thesis was, therefore, to perform a CBA of the recently installed UF membranes, to see if the risk reduction and the societal value of their improved drinking water quality had met the desired outcomes. Furthermore, by conducting the analysis ex-post, it was hoped that additional costs and benefits would be revealed, that may have otherwise been missed in an analysis conducted ex-ante.

The specific objectives of the study were to:

- Establish a decision support model based on CBA and QMRA, which included the three steps (i – iii):
    i. Set-up a QMRA-model for Kvarnagården DWTP, including both surface water and groundwater sources;
    ii. Identify, estimate and as far as possible monetize relevant cost and benefits from the installed UF membranes at the DWTP;
    iii. Combine the QMRA results with additional benefits and costs in a CBA to estimate whether the installation of UF membranes was beneficial from a societal point of view.

- Analyse future scenarios, e.g. the effect of climate change, to determine if additional measures will be required to ensure a safe drinking water quality in the future.

- Analyse uncertainties using a probabilistic model approach, to investigate how uncertainties in the model affect the results and how these uncertainties may be most effectively reduced.

The author believes that this thesis will increase the strength of available decision support material for future decisions regarding the improvement of DWTPs, for both VIVAB but also other drinking water producers.

1.3 LIMITATIONS

The data obtained in the analyses has been solely based on literature findings and previous measurements by e.g. the drinking water producers VIVAB or the Public Health Agency of Sweden. Furthermore, focus has been on the microbial risk reduction of the UF membranes has been included. Any chemical risk reduction which may have been acquired from the UF membranes has been excluded, as the UFs 20 nm pore size is unlikely to remove any chemicals < 800 Da from the effluent (WHO, 2017a).

2 DECISIONS AND RISKS

Good decisions, or rather good outcomes, are something that are always sought for in any decision-making process. However, how these are reached, or even how these are defined, may not be as clear. Parnell et al. (2013) states that a good decision "is one that is logically consistent with our preferences for the potential outcomes, our alternatives, and our assessment of the uncertainties" and that a good outcome "is the occurrence of a favourable event – one that we like". Nevertheless, a good decision does not always result in a good outcome. Because of uncertainties one can never predict an outcome of a decision with perfect certainty on beforehand. But by consistently making good decisions, one will create an environment which is more likely to produce good outcomes than bad.

In short, decision theory describes the handling of decisions when the knowledge about the outcomes is incomplete. It is important to distinguish between *normative* decision theory, which discusses how decisions should be made, and *descriptive* decision theory, which discusses how decisions are made (Norrman, 2004). For the assessment and handling of risk, it is primarily the former which is focused on. However, as the world grows increasingly complex, the normative decision theory often requires deeper forms of analysis, i.e. decision analysis (Keeney, 1982).

Parnell et al. (2013) describes decision analysis as "a philosophy and a social-technical process to create value for decision makers and stakeholders facing difficult decisions involving multiple stakeholders, multiple (possibly conflicting) objectives, complex alternatives, important uncertainties, and significant consequences". Furthermore, Clemen and Reilly (2014) states that "the purpose of decision analysis is to help a decision maker think systematically about complex problems and to improve the quality of the resulting decision". This statement is closely linked to the social-technical process mentioned by Parnell et al. (2013), which implies that a decision analysis cannot solely focus on the technical aspects of a decision. As even a technically superior analysis may fail to be applied if it is not understood by the decision makers at hand. Rational thoughts are not always the norm, and thinking about alternatives, preferences, and uncertainty may not always come naturally to those making the decision.

For example, it is very common for people to use *anatomical decision making*[1], rather than referring to their brain or line of thought, when explaining how a decision was reached (Parnell et al., 2013). One must therefore remember that even with perfectly objective data and infallible models, human subjectivity cannot be overlooked, and must be included in any decision analysis. Subjectivity can be included by taking the knowledge and opinions of the decision makers and stakeholders into account, and not only focus on the knowledge of the subject matter experts (Parnell et al., 2013).

The socio-technical process is also connected to the fact that decision analyses does not generate decisions. They are tools, which are used for specifying objectives, generating alternatives, assert them values and uncertainties, either mathematically or via expert judgement, and then interpret the implications of the analysis. It is only when a decision analysis is understood by the decision-makers, that it can generate the possibility for them to make an informed and justified decision.

2.1 UNCERTAINTY

Burgman (2005) states that "risk assessment differs from mainstream science because it obliges us to think about the tails of distributions, to account for the full extent of possibilities". He argues that all assessments involve a mixture of uncertainties but that only some of them can be quantified. The three primary types of uncertainty are; epistemic-, aleatoric- and linguistic uncertainty (Paté-Cornell, 1996; Burgman, 2005).

[1] *Anatomical decision making*: the reference to an anatomical part of the body (e.g. gut feeling or rule of thumb) when faced with the justification of a decision.

Epistemic uncertainty regards non-random uncertainties, e.g. incertitude and systematic error. Incertitude, is the lack of knowledge about fundamental phenomena, e.g. a parameter or a model, and can be reduced by the collection of more data (Paté-Cornell, 1996; Burgman, 2005). Whereas systematic error is the difference between a true mean value and the value a mean of the measurements converges to with increasing number of samples. It can arise from deliberate judgement of an observer or consistent unintentional errors in for example the calibration of the measurement equipment. It is remarkably hard to recognise and is best dealt with by a diligent inspection of experimental procedure, comparison of estimates against scientific theory, independent studies, replication and careful attention to detail (Burgman, 2005).

Aleatory uncertainty refers to randomness and variability of samples, e.g. natural occurring differences, unpredictable changes or diversities of which cannot be reduced with increasing measurements (Paté-Cornell, 1996). Here, measurement error should also be accounted for, which are random errors, resulting in the inability to measure a true value, e.g. the speed of light (Henrion and Fischhoff, 1986), the number of pathogens released from an infected body, or the water flow of a river or lake. It is important to note here that even if aleatory uncertainties are not reducible by larger amounts of data; they are still better understood. Hence, larger sets of data, gives more reliable estimates (Burgman, 2005).

Linguistic uncertainty implies the lack of exact definitions in language, e.g. vagueness, context dependence and ambiguity (Burgman, 2005). Vagueness derives from the acceptance of borderline cases in language and can be eliminated by introducing sharp arbitrary boundaries, although these may create other problems as well. Sharp boundaries are e.g. linked to the Sorite Paradox [2] (Blackburn, 2016), which makes them sensitive to minor changes in a continuum. Context dependence, on the other hand, is the uncertainty that arise from the lack of context in a premise for it to be understood and is, therefore, easiest dealt with by specifying context correctly and unambiguously. Lastly, ambiguity refers to the fact that a word can have more than one meaning, which may result in confusion regarding the intended meaning. Consequently, this is best remediated by carefully defining the terms being used.

However, for a risk assessment it is primarily the epistemic- and aleatory uncertainties which are of interest. This is because linguistic uncertainties are very difficult to quantify statistically and are best considered via the usage of a distinct and an exact language.

2.1.1 Quantitative Uncertainty

There are primarily two schools for handling quantification of uncertainty; frequentist and Bayesian (Paté-Cornell, 1996). The frequentist statistics analyse existent data with conventional methods and defines a probability as a limiting frequency, which is determined by the outside world (Paté-Cornell, 1996; Christensen et al., 2011). The Bayesian school, on the other hand, defines a probability as a degree of belief, which only exists in people's minds (Christensen et al., 2011). Bayesian statics incorporate all information about a system by representing the uncertainty of each parameter with a probability distribution. This distribution is based on hard data as well as expert judgement (Paté-Cornell, 1996; Christensen et al., 2011).

From a completely objective perspective, the frequentist approach appears to be the most attractive method, as it only involves objective treatment of statistical samples. However, as it is only based on measured data it is very limited outside the domain of aleatory uncertainties, or when data is restricted (Paté-Cornell, 1996). On the other hand, the Bayesian approach allows quantification of both aleatory and epistemic uncertainties, as well as being applicable when available data is limited (Paté-Cornell, 1996; Aven, 2012). Because of this, Bayesian statistics are often applied in risk assessments, since much of the needed data for the assessments is often subject to uncertainties that cannot be quantified using frequentist approaches (Aven, 2016).

[2] *Sorite Paradox*: The paradox of quantitative definitions, also known as the paradox of the heap, refers to the issue of how to quantitatively define something. It is based on the following logic: If one grain of sand is not a heap and that any number of n grains of sand are not defined as a heap, then the addition of just one more grain does not make them a heap. Hence one can never get a heap, as each added grain of sand creates just as much of a non-existent heap as before.

2.1.2 Monte Carlo Simulation

Given that all uncertainties of the input variables have been quantified, regardless if a Bayesian or frequentist method have been used, one will need to assess how input uncertainties propagate to the uncertainties of the output variable(s). One of the most common ways of doing this is via the Monte Carlo method, whose name was derived as a codename from the famous casino Monte Carlo in Monaco during the Manhattan project in world war II (Harrison, 2010). It is based on the random sampling of data, over several iterations, within a set limit of values, e.g. a distribution. This random sampling allows it to both create a combined uncertainty for an output variable, as illustrated in Figure 1, as well as to backtrack which of the variables that has the highest impact on the final output's uncertainty.

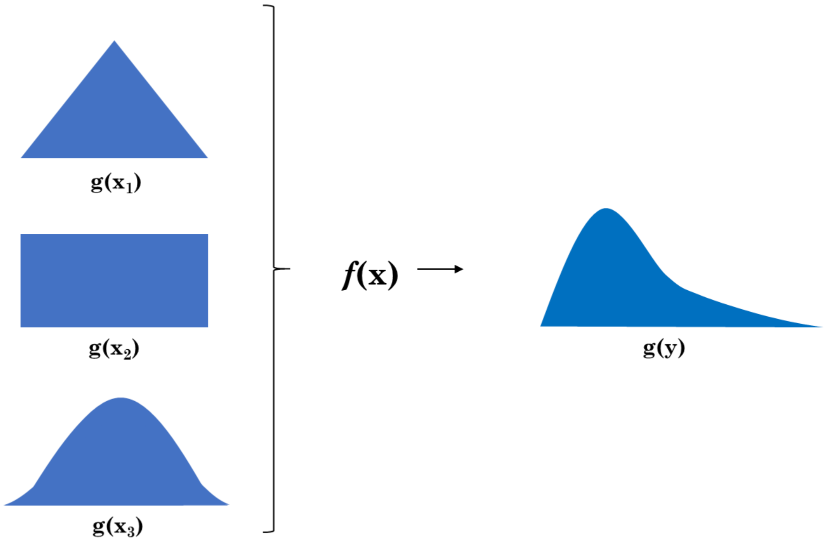

Figure 1 Conceptual illustration of a Monte Carlo simulation, based on Couto et al. (2013); combining the uncertainties of the input variables ($x_{1\text{-}3}$) to the uncertainty of the output variable (y)

2.2 Risk Management

The concept of risk and risk assessments are old terms dating back as far as 400 BC, where Athenians would offer their intellect and wisdom to assess risks before making decisions (Bernstein, 1998). Risk assessment and risk management as a scientific field is, however, still comparably young, beginning its development approximately 40 years ago (Aven, 2016). Because of this, the definition of risk is not always completely clear and may differ between applications and authors. The most common approach is, however, to observe risk as a combination of a hazard's likelihood of occurrence and its resulting consequence (Kaplan and Garrick, 1981; Burgman, 2005; ISO, 2009); where a hazard can be defined as any source or situation of potential harm (ISO, 2009).

The administration and handling of risks and uncertainties is referred to as risk management. It is a five-step process, illustrated in Figure 2 (ISO, 2009), where each step is described in their respective subheading below. In addition, it is also important to distinguish between the terms risk management and risk assessment, which are often used interchangeably. The former involves the whole five-step process, whereas risk assessment only refers to the grey-marked steps in Figure 2 (ISO, 2009).

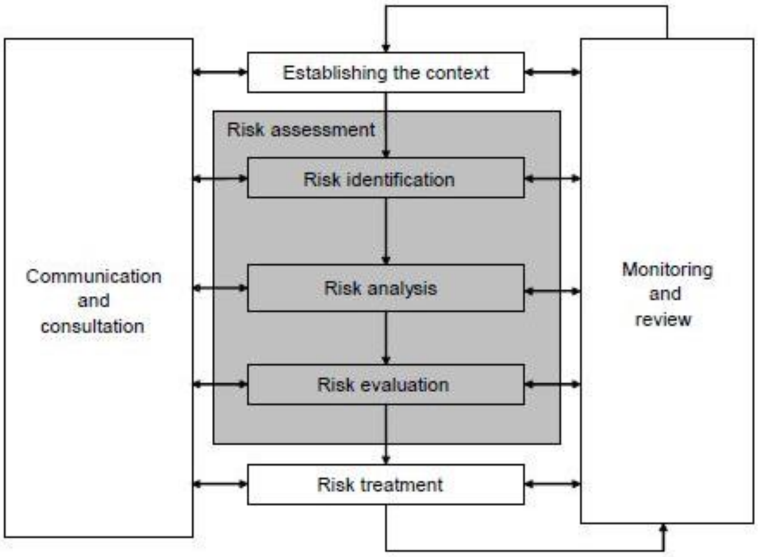

Figure 2 Overview image of a risk management process (ISO, 2009)

2.2.1 Establishing the Context

The first step of establishing context in risk, involves the consideration of relevant aspects for the management of risk, setting up the scope and the criteria for the rest of the process. It should involve both the organisation's internal context, e.g. its values and objectives, as well as the external environment in which the organisation operates, e.g. various stakeholders and political factors. The background to the specific risk being assessed should be discussed, and the risk criteria should be agreed upon (ISO, 2009). Risk criteria refers to specification of what consequences that is to be included, the expression of their probabilities, the definition of a risk and the distinction of risk levels, e.g. acceptable risk (illustrated in Figure 3). For the initiation and process of the risk assessment, it is primarily the stipulation of the risk criteria that is the most critical part. The most fundamental part of this is the definition of a risk, which is commonly expressed as the function shown in equation 1,

$$R = f(p, c) \qquad \text{Eq. 1}$$

where R is the assessed risk of a hazard, p is the probability of the hazardous event and c is its given consequence. Note that uncertainties are accounted for by representing each variable with a probability distribution. Additionally, the speciation of what parameter to assess from a hazard's consequence can vary greatly between assessments, and depends heavily on the decision maker's purpose for the risk assessment (Aven, 2012). For example, the effect of a disease can be assessed in several ways, e.g. number of lives lost or economic loss, depending on various social and political factors.

2.2.2 Risk Identification

The second step, risk identification, involves the procedure of finding, recognizing and recording risks. To identify any sources, events, situations or circumstances which may cause a negative impact on the system being assessed (ISO, 2009). If possible, one should also consider the internal risk of a systems, i.e. it's vulnerability (Lindhe, 2010). The definitions of vulnerability may vary, e.g. Zio (2015) explains it as a component or aspect of a system, which may cause large negative consequences to a system if it fails. Whereas, Johansson and Hassel (2010) interpret it as "a flaw or weakness (inherent characteristic, including resilience) in the design, implementation, operation and/or management of an infrastructure system". Yet from a practical perspective, vulnerability is handled the same way as for a hazard, described above.

A conceptual model of a risk identification is shown in Figure 3, where each risk is associated with a probability and consequence. Note that R_0 represents a more occurring event, e.g. a baseline risk, whereas R_1 to R_X represent the more extreme events, where R_X has the most extreme consequence. In addition, the system's total risk can be expressed, as a function of all the contributing hazardous events, via equation 2:

$$R_{Tot} = \int_{i=0}^{N} f(p_i, c_i) \qquad \text{Eq. 2}$$

where, R_{Tot} is the system's total risk, N is the total number of identified hazardous events, p_i is a given event's probability and c_i is the given event's consequence.

2.2.3 Risk Analysis

The third step, risk analysis, is the development of a model to better understand and estimate a risk. It provides an input to the decision makers whether a risk needs to be treated, and if so, the most suitable treatment methods (ISO, 2009). Aven (2012) states that the purpose of a risk analysis is to "aid decision-making, not to produce numbers". In other words, the analysis should be conducted and presented in a way such that the decision maker can understand its implications and meaning.

The models for illustrating the systems risk can vary greatly between different assessments and what type of system that is being analysed. A quantitative risk assessment can e.g. be estimated via a single value approach, often based on a conservative approach, or via the inclusion of variable uncertainty, as previously illustrated in Figure 1. Furthermore, systems which include more than one hazardous event, can e.g. be assessed and illustrated as risk graphs, Figure 3, or as risk matrices. The latter is primarily recommended for qualitative or semi quantitative risk assessments, where the level of uncertainty is too high for it to be quantitatively assessed (Lindhe, 2010; Duijm, 2015; Veland and Aven, 2015). Lastly, in more complex systems, logic trees can also be used to create a good overview of the events leading up to a hazardous event and where it is most likely to fail (Burgman, 2005; Pollard, 2008; Clemen and Reilly, 2014).

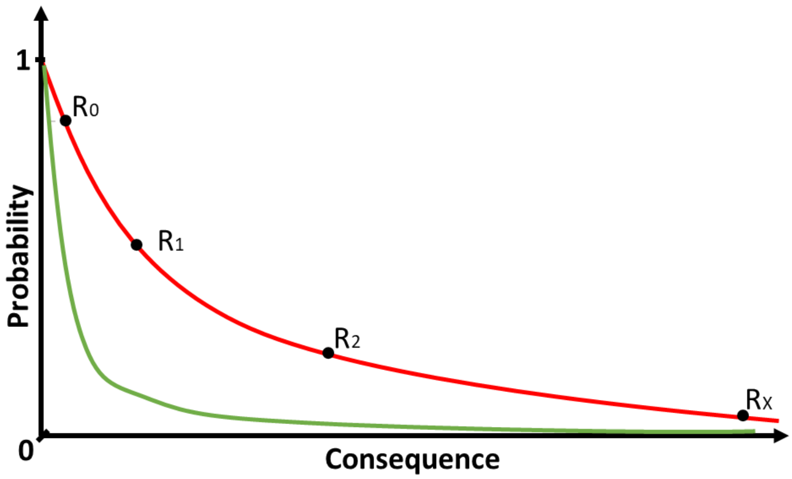

Figure 3 A conceptual model of a risk graph. The area below red line illustrates the total risk of an untreated state, where R_0 represent the most recurring and inevitable events and R_x the extremely rare but devastating cases. In contrast the area below the green line portrays a possible level of an acceptable risk and the area between the two lines is the gained or required benefit of a measure(s).

2.2.4 Risk Evaluation

The forth step of risk evaluation, determines the significance of a risk, by comparing the estimated risk-levels against the criteria from the established context (ISO, 2009). Yet, it is important to realise here that one may be naive to expect a risk to be completely eliminated. Hence, an acceptable risk-level should imply the highest value for which a total risk can be considered as non-threatening.

However, for many processes, e.g. air travel, where the consequences of a failure may be catastrophic, an acceptable risk level can be difficult to assign. This conundrum has given rise to the As Low As Reasonably Practical (ALARP) approach (Melchers, 2001; Pollard, 2008; Jones-Lee and Aven, 2011). It is an approach which establishes an interval between the acceptable- and unacceptable risk, for which one may choose to settle with, if it is practically or economically reasonable.

2.2.5 Risk Treatment

By utilizing the information regarding the system's current risk, attained from the conducted risk assessment and its corresponding steps, the decision-makers can decide about future actions. For most cases, this involves the selection of, and agreement upon, what measures that are required for remediating the risk (ISO, 2009).

2.2.6 Communication and Monitoring

The decided measure is then followed by a cyclical process, where the new level of risk is reviewed, reassessed and compared against the set criteria again, as displayed in Figure 2. It is a process that is implemented to make sure that the mediated risk do not exceed the acceptable level (ISO, 2009). Moreover, in Figure 2, one can see that this cyclical process of monitoring and reviewing is not only used at the end-step of the risk management process. It is a mechanism applied to all the risk management's respective steps, in combination with a continuous communication and consultation step.

The purpose of the monitoring and reviewing is to assure that the assessed factors, e.g. context and techniques, remain valid throughout the process. In contrast, the communication and consultation refers to the encouraged inclusion of stakeholder opinions in the process (ISO, 2009). Note how the communication and monitoring is inspired by both the social and technical approach of decision analysis discussed by Parnell et al. (2013) as well as Aven (2012) who states; a good decision is best reached by seeing "the decision-making as a process with formal risk and decision analyses to provide decision support, followed by an informal managerial judgment and review process resulting in a decision".

3 DRINKING WATER SYSTEMS

One area where the utilisation of risk assessments is of particularly importance is within the drinking water sector. Here, risk assessments should consider all three main subsystems of the drinking water system (DWS): the raw water source, the treatment facility and the supply network (Bartram et al., 2009). Further, an example of an urban water cycle is portrayed in Figure 4, also showing the impact of anthropogenic activities on a water system, e.g. wastewater discharge, farming and transportation. It should be noted that this is a schematic picture and that there can be great variations in how a DWS is setup, depending on socio-economic and geographical differences.

Looking at a global perspective, freshwater only constitute about 2.5 % of the total water volume, whereas the remaining 97.5 % consists of saltwater. Furthermore, of the existing freshwater there is only about 30 % which is available to us, as the rest is frozen in glaciers (Clement et al., 1997). Because of this, the raw water is most typically collected from either a surface- or groundwater source, where the latter can be either naturally or artificially recharged. Although, systems using a combination of both surface- and groundwater are also common. However, in more water scarce areas, e.g. Australia or the Middle-East, it has become more common with alternative sources for drinking water, e.g. desalination of seawater or recycling of waste water (Rygaard et al., 2010).

Figure 4 Conceptual view of an urban water cycle; consisting of both surface- and groundwater, as well as different contamination sources. E.g. boats, farmlands and waste water treatment plants (Lindhe, 2014).

3.1 WATER SAFETY PLANS

The access to a safe water supply is since 2003 considered a human right (WHO, 2003) and the most common approach for ensuring this, is via a water safety plan (WSP). It is a comprehensive process that aims to ensure a safe supply of water from source to tap, which is based on primarily three concepts; a system assessment, identification of control measures and management and communication plans (WHO, 2017b). WSPs are therefore often connected to extensive risk management processes which preferably considers all hazards and vulnerabilities of the DWS subsystems, i.e. the raw water source, the treatment facility and the supply network.

For sources, hazards concern anything that may cause unacceptable levels of toxic compounds, e.g. organic pollutants (e.g. PCB and PAH) or toxic metals (e.g. arsenic and lead), or pathogens (e.g. norovirus and *cryptosporidium*). In addition, other less acute water quality factors e.g. pH, hardness and natural organic matter (NOM) are also needed to be considered to ensure a good and safe drinking water quality (Thompson et al., 2007 ; WHO, 2011). The vulnerability of a water source is linked to how susceptible it is to these factors. Hence, unacceptable source conditions can either be mediated by reducing either its vulnerability, e.g. increasing its protection, or by improving the treatment processes at the treatment plant. Note how these measures are linked to the two alternative approaches of reaching an acceptable risk level in Figure 3. Either reducing a hazards probability, e.g. increasing the water source's protection, or the consequence, e.g. improving the treatment of the DWTP.

For the treatment processes, the hazards are the same as for the source, concerning unacceptable input values, where the treatment might be insufficient for the incoming contaminants. Whereas the vulnerability is linked to the treatment steps suboptimal conditions, e.g. failure or malfunctioning.

Lastly, hazards in the distribution network can involve; distribution of inadequately treated drinking water, insufficient or excessive pressure, pipe-burst, pump failure, intrusion, growth or suspension of biofilm and more. The vulnerability of the distribution is linked to its integrity to withstand these hazards.

The inability of supplying safe drinking water can result in severe disease outbreaks, with huge impacts on society. For example, in year 2004 poor water quality was estimated to account for 1.7 million deaths worldwide (Ashbolt, 2004). The majority of these deaths were situated in less developed socio-economic regions and there is still a clear trend regarding a region's economic status and the amount of deaths by diarrhoeal diseases (WHO, 2017e). That being said, this does not mean that water borne disease outbreaks are solely limited to the developing regions of the world. These types of outbreaks do still occur in well-developed, socio-economically stable, areas and it is therefore crucial that they are continuously planned for and, as far as possible, also prevented (Hrudey et al., 2003; Hrudey and Hrudey, 2007; Widerström et al., 2014; Beer et al., 2015). In Sweden, recent waterborne disease outbreaks have primarily involved *Campylobacter*, *Giardia* and *Cryptosporidium,* with examples such as the county of Gävleborg, in 2003, where approximately 3,000 were infected, or the town of Östersund, in 2010, where about 27,000 were infected (Folkhälsomyndigheten, 2018).

To estimate and predict the effect or burden of a disease outbreak on the society, several different methods can be applied. The most common are, however; Probability of infection (P_{inf}), Disability Adjusted Life Years (DALY) and Quality Adjusted Life Years (QALY). P_{inf} refers to the likelihood of a dose response from a pathogen, or chemical, resulting in an infection of a population (WHO, 2016). Whereas DALY and QALY are methods used for determining an incident's or disease's burden, given an infection, on a population's or individual person's health status. That is, not only taking the probability of infection into account, but also the severity and likelihood of a disease's symptoms (WHO, 2001).

It is important to remember that events linked to waterborne diseases does not only affect a population on a physical level, but also on psychological. It is not uncommon to see severe socio-economic problems following a disease outbreak, where the main factors primarily regard the trust and confidence for the system and the people running it (Bratanova et al., 2013). One problem with assessing these effects are, however, linked to the ambiguous definitions of the different terms

(Kelay et al., 2006). In an overview study of trust and confidence, Fife-Schaw et al. (2007) defines trust as both "a firm belief in the reliability or truth or strength etc. of a person or thing", and as "willingness to make oneself vulnerable based on a perceived similarity of the values and intentions of another". Whereas confidence is expressed as "an expectation that something will occur as anticipated". In other words, trust differs from confidence as it involves making oneself vulnerable.

Trust allows people to tolerate the growing uncertainty that arises from the increasing technological and environmental complexity. Where outbreaks and other inabilities to follow water quality standards will undermine this trust of the DWS, increase the perceived risk and decrease its acceptance (Bratanova et al., 2013; Viscusi et al., 2015). Here it is important to note that the distrust to a DWS is not solely connected to waterborne disease outbreaks, but may also rise from a lack of aesthetic quality, e.g. foul taste and smell (Doria et al., 2005). It is therefore possible that even a chemically- and microbiologically safe drinking water is perceived as unsafe, if it fails to meet a certain aesthetic quality, which in turn will lower the trust of the DWS's consumers.

An example of unwarranted mistrust for the DWSs around the developed world can be seen from the large consumptions of bottled water in relation to tap water; where the perceived risk of the tap water consumption is often much higher than the actual threat (Hu et al., 2011). This type of aversive behaviour is particularly interesting from a risk management and decision analysis perspective, as there is no scientifically proven health-benefits for choosing bottled water over tap water (Gena et al., 2008). Of course, this does not apply for instances when the tap water is known to be unsafe or aesthetically displeasing, e.g. having a foul taste or odour. Furthermore, the mistrust in a DWS may also be warranted if one e.g. is visiting a foreign country with different endemic pathogens than one may be commonly used to.

Still, for regular circumstances, the regulations for tap water, in the developed world, is in general higher than for bottled water. In addition, the latter is also almost 250 times more expensive than the former (SWWA, 2017), and has a more than 100 times higher environmental impact (Parag and Roberts, 2007; Angervall et al., 2004). Yet, there is still a considerable number of consumers who unwarrantedly chooses bottled water over tap water, due to a perceived health risk (Doria et al., 2005; Doria, 2006; Ward et al., 2009; Viscusi et al., 2015).

This unwarranted mistrust is a good example of the importance for including social factors, e.g. social capital, when assessing a DWS. Both to reduce the environmental impact of the aversive behaviour, but primarily to ensure that the consumers understand and appreciate the value of a clean tap water source; when it is available.

3.2 DRINKING WATER TREATMENT

To ensure a safe supply of drinking water and hopefully the trust of the consumers, almost all raw water sources on a municipal scale require some sort of treatment before being distributed. That is, to make sure that no contaminants reach unacceptable limits, where a contaminant can be considered as anything with a negative impact on the drinking water quality, including microbial pathogens, toxic chemicals or even natural organic matter (NOM). Because of this, a treatment chain often involves several different steps, where some may simply optimize the chemical properties of the water, e.g. changing the pH or alkalinity, whereas others are used to ensure that the distributed water does not contain any hazardous substances and is safe to drink. The latter treatment steps are commonly referred to as treatment barriers.

Treatment barriers can be divided into two principles; separation (e.g. filtration) and inactivation (e.g. chlorination). Separation refers to the removal of the contaminant from the water system, whereas inactivation implies a termination or transformation of the contaminant into a non-toxic state (Stanfield et al., 2005). Because of this, inactivation barriers can be very effective from a health-related perspective but often less useful for improving the aesthetic quality, as the contaminants are only transformed and not removed from the system.

In Sweden, there are no specific laws regarding a Drinking Water Treatment Plant's (DWTP's) required removal efficiencies of toxic compounds. Yet, they are required to ensure that the supplied drinking water is safe for consumption and that the limits of the Swedish National

Food Administration (SNFA) are not exceeded (Livsmedelsverket, 2017). To ensure this, SWWA (2014) recommend that a treatment plant should use a combination of both separation and inactivation for optimal removal of contaminants, if multiple barriers are being implemented. In addition, they also recommend that DWTPs which utilizes surface water as a raw water source need at least two microbial barriers to ensure a safe supply of drinking water, while DWTPs that uses groundwater as their raw water source only need at least one. The difference in the recommended number of barriers is linked to the fact that groundwater sources, in general, are less susceptible to microbial contamination. Lastly, it should also be noted that the definition of a microbial barrier have been restricted by the SNFA and SWWA (2014) to the more efficient treatment processes for microbial contamination, while the definition of a chemical barrier has yet to be defined.

3.2.1 Treatment Methods

The most common treatment methods are referred to as the in-line filtration, direct filtration and conventional treatment. The simplest of these three is the in-line filtration, which only involves a filtration step, followed by chlorination. The second method of direct filtration includes an additional flocculation step before the filtration. Whereas, the most thorough process, i.e. the conventional treatment, also utilizes a sedimentation step after the flocculation, before the filtration, to put less stress on the filters and increase the removal efficiency (LeChevallier and Au, 2004). Hence, the conventional treatment is considered as appropriate for most source waters, whereas in-line and direct filtration should only be considered as suitable for sources with good water quality (LeChevallier and Au, 2004). Note, that even if the removal efficiency may vary, none of the three methods are considered as more than a double barrier, i.e. one separation part and one disinfection.

Furthermore, for more contaminated water sources additional treatment barriers may be required for a sufficient treatment. This can involve; activated carbon, ozonation, Ultra-Violet (UV) radiation and membrane filtration (Crittenden et al., 2012).

Activated carbon is a highly active compound which works as a separator of suspended and dissolved contaminants; adsorbing them to its active sites. Ozonation functions as an inactivator, oxidising contaminants into (mostly) less toxic compounds. UV, like ozonation, works as an inactivator of contaminants, but primarily focuses on the disinfection of pathogens. The radiation is particularly good for inactivating bacteria, destroying the pathogen's DNA chains. Membrane filtration is a separative barrier and involves four types of different pore sizes; microfiltration (MF), ultrafiltration (UF), nanofiltration (NF), and reverse osmosis (RO).

As the names imply, the sensitivity of a membrane' removal efficiency is dependent on its corresponding pore size, which ranges from the MF's micro meters to the RO's less than nano meter size, as illustrated in Figure 5.

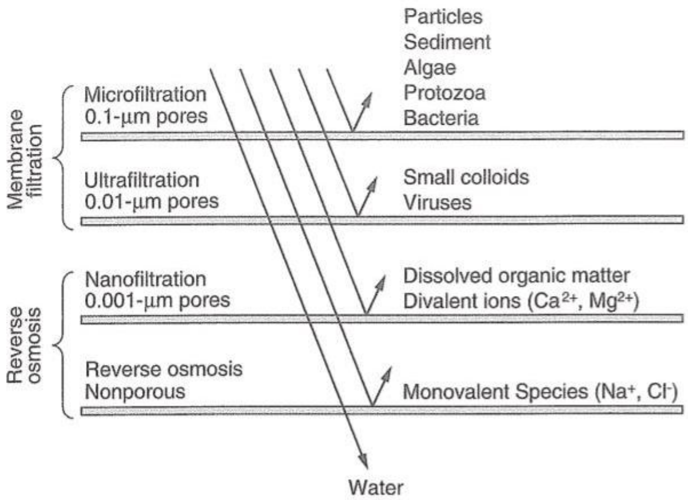

Figure 5 Hierarchy of membrane pore sizes and their corresponding barrier level (Crittenden et al., 2012).

3.2.2 Biofilm

Even if the treatment chain at a DWTP is functioning perfectly, where all pathogens are eliminated to levels well below the acceptable risk, there is still the potential of microbial regrowth and formation of biofilm in the distribution network. At a DWTP this can be somewhat assessed by utilizing a disinfectant which remains active throughout the distribution network, e.g. free chlorine or chloramine (Norton and LeChevallier, 1997; Hallam et al., 2001), or by limiting the amount of available nutrients in the distribution network via separative barriers (Bachmann and Edyvean, 2005). Conversely, the usage of strong inactivating barriers, e.g. ozonation, may enhance the regrowth of microorganisms and the formation of biofilm in the distribution network; by producing disinfectant by-products (DBPs) which are more bioavailable than their original composition.

The formation of biofilm in the distribution network, sometimes referred to as biofouling, can be formed by a single bacterial species but is typically comprised of a variation of microbial species, e.g. bacteria, fungi, algae, yeasts and protozoa. Furthermore, as it forms a protective layer for its harbouring organisms, it can increase their disinfection resistance with a factor up to 1500 times; making it very hard to mediate (Sauer et al., 2007).

From a strictly health perspective, the consequences regarding the formation of biofilm is still not certain. Negative aspects have definitely been proven, where studies have shown it to be able to harbour pathogenic organisms, e.g. bacteria, virus and protozoa, up to several weeks in the distribution network (Wingender and Flemming, 2011; Makris et al., 2013; Nya, 2015). Yet, positive aspects may also be produced from its formation. It may e.g. consume pathogens in the distribution network that would otherwise reach the consumers taps. Hence, it is as such not possible to draw any safe conclusions regarding the biofilms impact on the DWS's consumers health-related wellbeing.

Other less acute problems connected to biofouling of the distribution network can be biocorrosion, which primarily results from sulphur and iron oxidizing bacteria (Usher et al., 2014; Pizarro and Vargas, 2016). Decreased aesthetic quality, from example resuspended biofilm layers or its released metabolites (Makris et al., 2013; Nya, 2015), but also increased pump costs as biofilm may increase the systems hydrodynamic resistance and reduce turbine efficacy (Nya, 2015). Yet, similar to the health-related wellbeing, it may be possible that the formation of biofilm generates a protecting layer on the pipe-surface, which in turn produces a positive outcome of its formation.

It can be concluded that the complexity regarding the formation of biofilm still contains uncertainties that makes it hard for one to draw certain conclusions about its impact.

4 QUANTITATIVE MICROBIAL RISK ASSESSMENT

One of the most established methods for assessing the health effects of a water source, and its corresponding treatment step(s), is a quantitative microbial risk assessment (QMRA) (Haas et al., 2014). It is primarily based on three things:

- The presence of harmful substances and microorganisms in the water;
- The acceptable and infective doses of these compounds, and;
- The estimations of the exposure to the water users.

A QMRA creates a quantitative analysis of a water system's microbial risks and what treatment designs that may be required to meet an acceptable risk level. It is important to note here, that the dose response is normally, solely, based on the oral ingestion of pathogens, as the model is limited to only one exposure route and the fact that oral ingestion results in the most acute exposure (Thoeye et al., 2003).

The four key steps of a QMRA have been listed in Table 1. Here it is possible to see, predicted, similarities between the QMRA's process steps and the general steps of the risk management by ISO (2009), where each of the QMRA's four steps can almost be directly translated into ISOs first four steps respectively.

Table 1 Steps of a QMRA and their corresponding aims, based on Haas et al. (2014) and WHO (2016)

Step	Aim
1) Problem formulation	Describe the general environment and the relevant pathogens, which may cause harm to human health.
2) Exposure assessment	Identifying exposure pathways and evaluate the extent and time of an exposure for each relevant pathogen.
3) Health effects assessment	Estimate the relationship between exposure (i.e. ingested dose) and infection or illness. As well as the probability of morbidity or mortality, depending on the end-point of the assessment.
4) Risk Characterisation	Integrate the dose and exposure information to quantitatively express the public health outcomes.

The next sub-sections will discuss the four steps of a QMRA, illustrated in Table 1, in a more in-depth context, to provide the reader with the necessary information required to understand the model and its uncertainties.

4.1 PROBLEM FORMULATION

The identification of relevant pathogens can be difficult and require a good knowledge of the study area's possibly hazardous events and regular sources of contamination. In a review by Nature (2011), a total of approximately 1400 different pathogens have been identified across the globe. These pathogens can cause harm either directly, e.g. via infection, or indirectly, e.g. via release of toxins (Thoeye et al., 2003). However, their variation is not completely random, and can as such be divided into distinct categories. For pathogenic microorganisms this involves three categories; bacteria, virus and protozoa (WHO, 2016).

The smallest of these three are the viruses, which can range from approximately 0.02 – 0.8 µm in size and are essentially only composed of either a protein coated RNA or -DNA genome. Consequently, they are fully dependent on the biochemical processes of other organisms' cells for their own replication (Gelderblom, 1996).

Bacteria are unicellular prokaryotic organisms in the micron size range 0.1 – 10 µm, that exists in a great variation of structures and shapes. They reproduce via binary cell division and are for most cases capable of independent metabolic existence (Salton and Kim, 1996).

Protozoa are microscopic unicellular eukaryotic organisms and exists in the size range from 1 – 150 µm. The major difference between prokaryotes (e.g. bacteria) and eukaryotes (e.g. protozoa) is the latter's external cell membrane, protecting the cell's internal content (Yaeger, 1996). This membrane does not only allow for a more complex internal structure, but it also makes protozoans less vulnerable to disinfection, e.g. via chlorination (SWWA, 2014; WRF, 2016).

Due to the large variety of pathogens, it may be difficult to include all harmful organisms in a QMRA model. One must therefore be selective in the approach of selecting the relevant pathogens. For example, a technical approach would be to look at the most persistent pathogens, whereas an approach from health point of view would be to look at the most harmful organisms (Thoeye et al., 2003). However, to ensure that the full range of pathogens are covered, one should try to include at least one pathogen from each category of viruses, bacteria and protozoa (WHO, 2016).

4.2 EXPOSURE ASSESSMENT

The possible exposure pathways of pathogens in a QMRA can be many. Nevertheless, the most commonly assessed exposure pathway is the intentional drinking of unboiled water. Yet other pathways can also be unintentional ingestion (e.g. during recreational swimming), aerosol ingestion and food consumption (WHO, 2016). Note, that only unboiled water is of interest for QMRA, as boiled water is very unlikely to contain any pathogens.

Treated water is, however, included even if pathogens generally only occur at very low levels, often below the detection limit (Hunter et al., 2003). For example, the treatment may prove to be insufficient at suboptimal conditions (e.g. a dysfunctional disinfection step). Or the quality of the raw water source might drastically changes, e.g. via a combined sewage overflow (WHO, 2016). The role of an exposure assessment is therefore to evaluate the time, extent and effect of these possible events, as well as the general exposure during normal conditions. From a practical sense, this often involves the usage of indicator organisms, which can be divided into three distinct categories shown in Table 2.

Table 2 Definitions for indicator and index micro-organisms, based on Ashbolt et al. (2001)

Indicator Category	Definition
1) General process indicators	Organisms used for demonstrating treatment process efficiencies, e.g. total coliforms for chlorine disinfection.
2) Faecal indicator	Organisms that indicates the presence of faecal contamination, e.g. *E. coli*.
3) Index and model organisms	Organisms which indicates the presence of pathogens, as well as their respective behaviour, e.g. *E. coli* as an index for Salmonella

The uncertainties of exposure can however be many. For the index organisms, this primarily regards their representability of their respective pathogen or process (WHO, 2016). But also issues such as false positives, i.e. the presence of an index organisms in the absence of pathogens. Or reversely, false negatives, the presence of pathogens in absence of indictive organisms, must be considered (Hunter et al., 2003). Treatment processes has to be critically assessed to get an estimation of their probability of failure (Lindhe et al., 2012). For the exposure volume, uncertainties mainly regard variations in consumed amount of water. This is as rehydration can vary greatly from region to region, as it depends on a number of factors, e.g. the local climate and the population's degree of manual labour (Howard and Bartram, 2003). Yet, the general guideline by WHO (2011) is suggested to 1 liter of unboiled water per person per day, although location specific data is encouraged.

4.3 HEALTH EFFECTS ASSESSMENT

The estimated relationship between an ingested dose and infection or illness, i.e. the dose-response, is most often based on previously conducted studies' dose-response curves (i.e P_{inf} as a function of the ingested dose).

Note that the criteria for an infection may differ from study to study, but the three most commonly used principles for determining if a person's been infected are; i) observed symptoms of illness, ii) observed presence or increase of antibodies in the blood, or iii) observed presence of pathogens or similar strain (e.g. cyst, oocyst or eggs) in the subject's stools (Thoeye et al., 2003). Furthermore, dose-response studies also include a lot of aleatoric uncertainties e.g. differences regarding the test subjects degree of immunity, genetics, age and nutrition, which likely has a huge impact on the resulting infectivity of a pathogen (Jiménez, 2003). It is therefore crucial that the dose-response is expressed as ranges, with corresponding uncertainty distributions, as there is always the possibility of a single ingested pathogen infecting some individuals (Thoeye et al., 2003).

The threshold concentration, below which no infection occurs, does as such not exist and it is therefore not surprising that the most prominent uncertainties can be found at the lower end of the dose-response curve. The P_{inf} can, for this range, vary over several orders of magnitude, due to primarily the mentioned factors above but also due to a lack of sufficient test subjects (Hunter et al., 2003). Conversely, it is the lower range which is of main concern for actual real-life cases, apart from extreme scenarios or events, as pathogens tend to be present below the detection limit.

In addition, there are also studies, which have assessed the probability of illness after infection, i.e. the probability of developing symptoms. The severity and likelihood of these symptoms, as well as its sequelae (i.e. its secondary or chronic health effects), can be used to estimate an infection's disease burden; often assessed in terms of QALY or DALY (WHO, 2016).

Lastly, health effects assessment may also include the possibility of secondary transmission from person-to-person contact, as well as a populations likelihood of displaying immunity, to truly cover the health impacts of a pathogen. Yet, this may require more dynamic models, and may as such be considered exaggerated depending on the model's purpose.

4.4 RISK CHARACTERISATION

By integrating the collected dose and exposure data, it is possible to calculate the expected probability, impact and total risk of an infection in a water system (WHO, 2016). The most common methods of expressing these risks involve the previously discussed P_{inf}, DALY and QALY (WHO, 2016; Bergion, 2017). P_{inf} is a straightforward method, which assesses each respective pathogen with a probability of infection for the population. By using an Exact Beta-Poisson function, represented by an exponential function with a beta distribution in the exponent, the P_{inf} was calculated via equation 3 – 5 (Bergion, 2017):

$$P_{inf} = 1 - e^{-r \times D} \quad \text{Eq. 3}$$

$$D = C_{DW} \times V \quad \text{Eq. 4}$$

$$V = e^{Normal\ (\mu,\sigma)} \qquad \text{Eq. 5}$$

where r represents a pathogen's infectivity, described as a sample from a beta distribution with statistical parameters set for each pathogen's dose response, D is the simulated daily pathogen dose, where C_{DW} is the pathogen concentration in the distributed drinking water, and V is the estimated exposure volume; based on a Log-normal distribution with a mean value μ and a standard deviation σ. From these equations, one can then estimate a pathogen's annual probability of infection, $P_{inf\ Annual}$, via equation 6:

$$P_{inf\ Annual} = 1 - (1 - P_{inf})^{d} \qquad \text{Eq. 6}$$

where d is the expected number of days per year that the set dose, D, will occur. For example, the duration of a pathogen load to a raw water source could be estimated by assessing the expected number of days per year that people excrete pathogens to a lake or aquifer. The $P_{inf\ Annual}$ for each pathogen, n, can then be combined for all investigated pathogens to give a total annual probability of infection, $P_{Total\ inf}$, as shown in equation 7:

$$P_{Total\ inf} = 1 - \prod_{n}^{N}\left(1 - P_{inf\ Annual}\right)_{n} \qquad \text{Eq. 7}$$

The $P_{Total\ inf}$ can be used to illustrate how many people that can be expected to fall sick from water borne diseases within a certain year, i.e. by multiplying the $P_{Total\ inf}$ with the connected number of drinking water consumers. Yet the method is most often used as an indicator to see if further treatment barriers are required to ensure a safe water supply, by comparing it against a set acceptable level. Note, that there is no officially set limit for an acceptable $P_{Total\ inf}$ in Sweden, although the WHO (2016) recommend a highest level of 10^{-4} for the $P_{Total\ inf}$ during 95 % of the time.

One of the major drawbacks of a P_{inf} assessment is, however, that it does not include the severity or impact of an infection, which is better assessed using either the DALY or QALY method (WHO, 2016). The DALY is calculated as shown in equation 8 – 10; combining the Years of Life Lost (YLL), which is the incident's mortality, and the Years Lived with the Disability (YLD), which is the incident's morbidity or degree of disability. Note that the latter is also, for most cases, standardised by means of severity weights (WHO, 2001; Bergion, 2017).

$$DALY = YLL + YLD \qquad \text{Eq. 8}$$

$$YLL = \sum_{i} e \times a_i \sum_{j} d_{ij} \qquad \text{Eq. 9}$$

$$YLD = \sum_{j} N_j \times L_j \times W_j \qquad \text{Eq. 10}$$

where i is the index of different age classes represented by a, j is the index of different disabilities, e_i is the average life expectancy for each age class, d_{ij} is the number of fatalities for each age category and respective disability, N is the number of cases, L is the length of the disability and W is the disability weight, representing the severity of the disease.

QALY on the other hand is based on a function of the Health-Related Quality of Life (HRQL) over time, which is a scale from 0 – 1; representing death to full health (Robberstad, 2005). On a rough scale it is possible to consider HRQL as the inverse of the degree of disability, see Figure 6 (Gold et al., 2002). However, this does not imply that QALY is the inverse of DALY, as they may be standardised on different parameters for their respective severity weights (Sassi, 2006). Hence, an inverse relationship can only occur when weights are excluded from the calculations (Robberstad, 2005).

The change of a person's QALY from an event (e.g. infection) can be estimated via equation 11, and is also illustrated for a given year, with the red-striped area, in Figure 6.

$$\Delta QALY = \int_{t=0}^{T} HRQL(t)_e - \int_{t=0}^{T} HRQL(t)_p \qquad \text{Eq. 11}$$

where $\Delta QALY$ is the change in a person's QALY, $t = 0$ is the time from where the QALY assessment is initiated, T is the assessed end-time, e is the HRQL's function over time for the event and p is the HRQL's predicted function over time if said event had not occurred.

Note that QALY differ from DALY in the sense that it is possible to achieve positive value-changes, i.e. to gain QALY, from an event, e.g. a medical procedure (Robberstad, 2005). In comparison, a change in DALY can only be more or less negative. It is because of this possibility of assessing a positive change in QALY, either by prolonging a patient's life, by increasing the patient's HRQL, or sometimes even both, that QALY is commonly used in the healthcare sector. The assessment of changes in a patient's QALY, be it negative or positive, allows the decision makers to determine the profitability of different medical procedures and compare them against each other (Annemans, 2017).

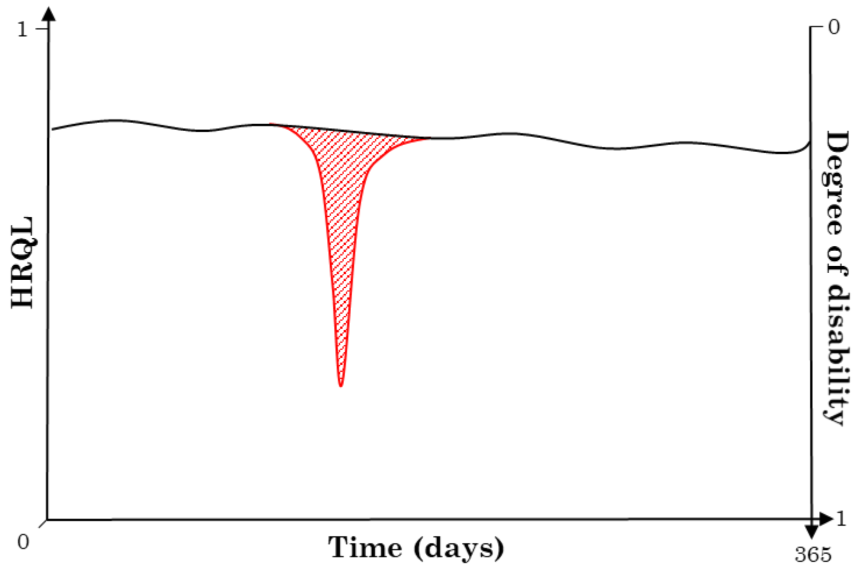

Figure 6 A conceptual graph of how a negative event (i.e. the red line) may impact a person's health within a certain year, in comparison to an unaffected circumstance (i.e. black line). The said events change of a person's QALY is also illustrated with the red-striped area.

5 COST-BENEFIT ANALYSIS

The purpose of a socio-economic assessment is to estimate a measure's societal value in monetary terms, to determine if a measure or intervention is profitable from a societal perspective or not. That is, if it's benefits are greater than its costs (ISO, 2009; Cameron et al., 2011). Note that an intervention in a system, e.g. an improved treatment barrier, often produces more than one benefit. It is therefore vital for the assessment tries to include as many of the resulting costs and benefits as possible in the model.

Examples of societal benefits can be reduced time of sickness or increased social capital (Hutton, 2001; Cameron et al., 2011). While the cost can be divided into capital and recurrent cost. The former refers to costs that will only occur once, e.g. an investment, and the latter refers to costs that occur on a regular basis, e.g. maintenance or staff (Cameron et al., 2011).

In practice a socio-economic assessment is typically conducted using a Cost-Benefit Analysis (CBA). It is a widely used method that can be applied in almost all economic assessments. Yet, it is particularly attractive from a societal perspective, as it allows the inclusion of many kinds of variables. In comparison to other economic analysis', e.g. Cost Efficiency Analysis (CEA) or Cost Utility Analysis (CUA), which primarily focuses on one variable at a time, e.g. reduced number of infections to cost, a CBA instead tries to include all affected parameters and variables at once.

In its simplest form, a CBA purely aggregates the total estimated benefits and the total estimated costs respectively, for all involved stakeholders, to see if the formers total value is greater than the latter. However, if an intervention is expected to provide benefits over a longer period of time, also known as a time horizon, a discount rate should also be applied. The discount rate is a variable which illustrates how the present value of future costs and benefits will decrease over time (Hanley and Barbier, 2009). The estimation of a long-term CBA is given as a Net Present Value (NPV) and is calculated as shown in equation 12 (Bergion, 2017).

$$NPV = \sum_{t=0}^{T} \frac{B_t}{(1+r)^t} - \sum_{t=0}^{T} \frac{C_t}{(1+r)^t} \qquad \text{Eq. 12}$$

where T is the established time horizon, t is a given year, B_t is that year's aggregated benefit, C_t is that year's aggregated cost and r is the discount rate.

5.1 VALUE DISCOUNTING

It is important to note that the discounting of a measure's is not linked to the inflation of the monetary value, but rather the measure's opportunity cost and associated time preference. The former refers to the fact that the investment in one measure likely prohibits the investment in other solutions which may be more profitable. Whereas, the latter refers to the impatiens of humans and that people tend to want benefits to come sooner rather than later (Hanley and Barbier, 2009). Furthermore, the assessment of a discount rate for a measure's cost or benefit can normally be considered in either of two ways; descriptively or prescriptively (Söderqvist, 2006).

A descriptive discount rate tries to assess how people will actually behave and will, as such, be heavily based on the interest of the market and the possible loss in opportunity cost. It is said to be based on the practical ethics of society and does as such try to apply a discount rate which is as close to the market's rate of interest as possible[3]. The utilization of a descriptive discount rate is therefore often explained by saying that the best option for the future is for us to focus on the generation of the economic growth today, to create the required wealth for the generations of tomorrow. A solution that, as such, will allow the future generations to tackle the problems on their own, rather than us selecting preventive measures today, that may be less profitable in the long run (Söderqvist, 2006).

[3] The state where the discount rate is equal to the rate of interest is also known as the Ramsey condition, named after the economist Frank Ramsey

The prescriptive discount rate on the other hand states that markets in practice rarely, or never, are perfect and therefor questionable to be used as a measurement of a measure's future monetary value. The invisible hand[4] may not always function well and there may be goods and services which are located outside of the market-priced regions that still increases the peoples' wellbeing (Söderqvist, 2006; McIvor, 2009). A prescriptive approach also questions the ethics of a society treating its future generations in the same manner as its individuals of today, who show great shortcomings in their consideration. A prescriptive approach, therefore, argues that there are good reasons for a society to choose a discount rate whose social rate- and time preference size is based on ethical rather market-sized values (Söderqvist, 2006).

At first glance, the prescriptive approach may appear as more subjectively based than the descriptive assessment. However, as stated by Baum (2009) in his analysis of the two methods, it is impossible for an analyst to describe a society's discount rate without inserting her values into the analysis.

In other words, both approaches are based on a subjective analysis of a measure's future value and the arguments for selecting a specific discount rate should therefore be clearly stated. One common method of doing this is via equation 13, which gives the decision makers a good illustration of how the analysists may have assessed the selected discount rate (Söderqvist, 2006; Hanley and Barbier, 2009; HM Treasury, 2018):

$$r = \rho + \eta \times g \qquad \text{Eq. 13}$$

where r is the assessed discount rate, ρ is the estimated rate of time preference, η is the consumption elasticity and g is the estimated growth rate. Based on this equation, the HM Treasury (2018), as well as Trafikverket (2018), has recommended a discount rate of 3.5 % to be applied in governmental projects (where $\rho = 1.5$ %; $\eta = 1.0$; $g = 2$ %). Note that the suggested discount rates may be considered as more prescriptive, than descriptive, as the mean annual Swedish growth rate (g), between year 1990 and 2015, has been calculated to 3.5 % rather than 2 % (Gunér, 2017).

5.2 MONETARY VALUATION

In addition of asserting a discount rate to the present value of a benefit, or cost, one must also assess the monetary value of that variable's good or service. This can be particularly difficult for intangible variables that lacks a direct market price, e.g. social capital. It is therefore common that Willingness To Pay (WTP) studies and surrogate market prices are implemented to assess these variables values (Hutton, 2001; ISO, 2009; Cameron et al., 2011; Young and Loomis, 2014). The four most common methods for monetary valuation are listed 1 – 4 below, in their recommended order of use (Hutton, 2001):

1. *Market valuation* is the market's stated value of a good or a service and is the most effective method for monetary valuation, if a market price is available. Hence, it can only be applied if a market price is already set (e.g. reduced sick-leave or operation costs).

2. *Contingent valuation* is the most efficient valuation method, if no market price is available, and is based on the public's WTP for a non-market good or service in a hypothetical market. It is a method which allows the public to monetise a wide range of variables that would otherwise lack market value (e.g. increased quality of life or amenity). Albeit the benefits are many, with also reproducible results, it

[4] Refers to the term invented by Adam Smith, which argues that "individuals' efforts to pursue their own interest may frequently benefit society more than if their actions were directly intending to benefit society"

can be a complex and time-consuming method to set up and is as such not always possible to conduct.

3. *Revealed preference* tries to analyse the relationship between levels of services (e.g. water supply) and the prices of the marketed goods (e.g. house price) or services. However, this is less recommended than the contingent valuation as it based on a larger number of uncertainties with many methodical problems, e.g. large sample size requirements, omitted variable bias and more.

4. *Household production* is the least preferred, of the four mentioned, valuation method as it only refers to a WTP for an improvement based on previous aversive expenditures (e.g. household water-filters or boiling of water). Hence, it is likely that other factors, which generate value, are excluded.

Note that, even if it is not preferred, non-monetised benefits could also be included in a CBA. For example, if a beneficial variable is generated from an intervention, yet it deems too hard to assign it with a monetary value. One can still apply it to the assessed CBA to illustrate what possible value the variable may be required to generate, to produce a positive outcome (Bergion, 2017).

5.3 THE VALUE OF WATER

Clean water is in many ways a priceless resource, both from an anthropocentric as well as a biocentric view. It is a necessity for the survival of any biological life and considered one the most vital assets in any society today. Yet, for a CBA, it is not very useful to assess a resource as invaluable and one should therefore try to, as far as possible, value even the invaluable resources.

The author has therefore tried to identify factors that can be assessed to determine the economic value of a safe and clean water from a societal perspective. It is similar to the valuation of the environment's economic value by Hanley and Barbier (2009), and based on the principles of social cost-benefit analysis of drinking water by Cameron et al. (2011) and the findings of Hasselström et al. (2014) and Söderqvist and Wallström (2017). The three identified factors for the monetary valuation of water were recognised by their relationship to it and by how:

- Clean water is directly linked to the *health-related wellbeing* of any water consumer. By preventing waterborne disease outbreaks, the benefit of the clean water can be assessed by determining the prevented cost of sickness and cost of care of said waterborne disease.

- Clean water is highly associated with the *social capital* of a society, where it has a great recreational value. A clean water source can both be used for recreational activities as well as creating an aesthetically pleasing environment. In addition, the distribution of a contaminant-, colour-, taste- and odour-free drinking water also provides a major social capital and trust for the drinking water consumers.

- Clean water is an essential part in many *industrial processes* and thus a requirement for many functioning industries. Like the health-related value of clean water, the industrial value can be estimated by assessing the prevented additional measures, e.g. filtration-steps, that would have been required by the industry to utilize the water in their processes if the water was not clean enough. Or by assessing the industrial appeal to remain or create industries in the area, which require clean water for their processes. Furthermore, for industries that are directly dependent on a clean water source, e.g. fishing or farming, the value of that water source's purity can be seen as closely related to the local industries' revenue, as they would have a challenging time existing without it.

Note that the identified factors for monetizing the value of clean water are linked to both the sources' value as well as the distribution's value. This thesis will, however, solely assess the value of the distribution of clean water, as an UF membrane will not improve the quality of a DWTP's raw water sources.

5.4 THE VALUE OF HEALTH

The previously discussed procedure of attempting to turn a person's health status into one single index, e.g. QALY or DALY, section 4.4, allows analysists to compare a variation of different hazardous events, e.g. ingestion of pathogens or chronic diseases, against a variety of preventive measure or treatments to assess the best solution to the problem (Klarman et al., 1968; Mason et al., 1998; Whitehead and Ali, 2010; Söderqvist and Wallström, 2017).

In health-care related areas, QALYs are often used to determine the viability of a possible intervention or medicine. In these procedures, the value of a QALY is determined after the required cost of its improvement, i.e. how many QALYs one can expect to gain from an intervention or measure. For example, if a measure produces ten QALYs at a price of 1 MSEK, it will generate a value of 100,000 SEK per QALY. In these circumstances, the value of a QALY can be considered low if the cost per QALY is less than 100,000 SEK, medium if it ranges between 100,000 SEK to 500,000 SEK, high in the range between 500,000 SEK to 1 MSEK and very high if it exceeds 1 MSEK (Carlsson et al., 2006).

Note that this type of approach does not take an effected person's social and societal factors into account, as it only assesses the cost of a medical treatment against its gained QALYs. It is an approach that is more suitable for a Cost Efficiency Analysis (CEA) or a Cost Utility Analysis (CUA) and is as such not very useful in a socio-economic assessment, using CBA, which attempts to have a more holistic view of a measure's impact (Haninger and Hammitt, 2011). Notwithstanding, the discussed arbitrary limits may tell us what the societal WTP, i.e. what the society as a whole may be willing to pay, for a QALY and should be considered as the very high range of around 1 MSEK.

Furthermore, the societal value of health should describe the expected value a healthy person contributes to the society, what health-related costs, e.g. medical treatment, that are avoided by her good health and also the value of the mere avoidance of discomfort that arise from her being ill. Note, that the actual societal value may differ from the societal WTP. The author has suggested three factors, based on the principles of Cameron et al. (2011), that should, optimally, be included in the monetary valuation of a hazard's effect on a person's societal health-related wellbeing. These factors are the individual burden, the domestic burden and the societal burden, where:

- The *individual burden* is linked to an event's morbidity and can be assessed as discussed in section 4.4 (in terms of lost QALYs). It is, intuitively, the most generally included factor in any monetary valuation of health.

- The *domestic burden* is an event's impact on the effected person's close environment and can, for example, be assessed as the environment's required time of care. It is less often included in the monetary valuation of health, as it is harder to assess, but may have an equally substantial impact on the last factor of the societal burden.

- The *societal burden* is an event's effect on the societies production. In an optimal analysis this factor includes both the effected person's as well as her close environment's lost production.

Note that the domestic- and societal burden may be very hard to assess with a single index, e.g. QALY, where the latter also includes a few ethical concerns, e.g. if it is a valid approach to valuate a person's health based on her income and wealth. Yet from a societal view, it is vital that these variables are at least considered when assessing the health-related wellbeing with a monetary value.

One suggested way of estimating the societal burden of a hazardous event is by assessing the single index's value against its relationship with the Gross Domestic Product (GDP) per capita (Cameron et al., 2011; Nimdet et al., 2015). However, the actual correlation between a person's health-related wellbeing and her production may not be as easy to determine. In a study, Nimdet et al. (2015) found that the ratio between the WTP for a QALY and the GDP per capita varied between 0.05 to 5.40, with an average QALY per GDP ratio of 1.43. A variation that most likely is explained by the difficulty of assessing a QALY with a monetary value. Note that the societal value of QALY often exceeds the society's GDP per capita, because of the mentioned factors above. A healthier person will not only be more likely to contribute more to the society's generation of wealth, than if she was impaired, but will also avoid additional medical or other health-related costs. In addition, her good health will also put less stress on her close environment, and their production, as they will need to spend less time giving her health-related care, than if she was impaired. The societal value of a QALY should as such be valued higher than the GDP per capita, as a person's health will affect more than one person.

There are, however, issues of asserting a single health index with a monetary value using WTP studies, as illustrated by Haninger and Hammitt (2011). In their study it was shown that the WTP for an improved QALY was much higher for events with short durations and mild consequences, than for long-lasting events with severe consequences. The former's value was estimated to 49 MSEK per QALY and the latter to 1.3 MSEK per QALY. A result that shows that the individual WTP per QALY is very dependent on the quantity of expected QALYs (Nimdet et al., 2015). In addition, the study also shows that the social value of an avoided health-related hazard may be much higher than the hazard's actual health-related total cost.

Economic health-valuation is applicable to both QALY and DALY, yet, this thesis will be restricted to QALY. This is as QALY is more generally applied in the economic valuations of health related wellbeing, whereas DALY is more generally used for international comparisons of disease outbreaks (Whitehead and Ali, 2010; Söderqvist and Wallström, 2017).

6 Study Area

The UF membranes analysed in this study were first planned by VIVAB in September 2012. Primarily to improve the safety of the drinking water but also due to increasing levels of Natural Organic Matter (NOM), e.g. humic substances, being observed in the lake Stora Neden (Keucken, 2011; VIVAB, 2018). The NOM may affect the aesthetic quality of the drinking-water in a negative way, even if it is not a health risk per say, causing taste, colour and odour problems. In addition, it may also affect the industries connected to the DWS in a negative way, by contaminating their processes (Keucken, 2017). The UF membranes were as such not only implemented at the Kvarnagården DWTP in November of 2016 to ensure a safer water quality, by adding an additional treatment barrier, but also to produce a more aesthetically pleasing drinking water for its consumers.

6.1 Kvarnagården DWTP

The Kvarnagården DWTP is located in the eastern part of Varberg, which is a municipality with approximately 62,000 inhabitants located in the district of Halland in the south-western part of Sweden (SCB, 2017). The DWTP is the main water supplier to the municipality and serves around 99 % of the inhabitants with treated drinking water (VIVAB, 2016). The raw water is collected from two raw water sources, Figure 7, the lake Stora Neden and the aquifer Ragnhilds Källa.

To this day, no major reported waterborne disease outbreaks has occurred in the municipality of Varberg. A fact that is most likely linked to the DWTP's good raw water quality. During recent years, *E. Coli* and *Clostridium perfringens* has only been observed on a few occasions at the DWTP's inlet, where almost all observations has been measured to below one Colony-Forming Unit (CFU) per 100 mL (VIVAB, 2012; VIVAB, 2016). However, there has been one occasion, where *E. Coli* was measured to 15 CFU per 100 mL, which could indicate either a larger rarer contamination source or simply a measurement error (VIVAB, 2012).

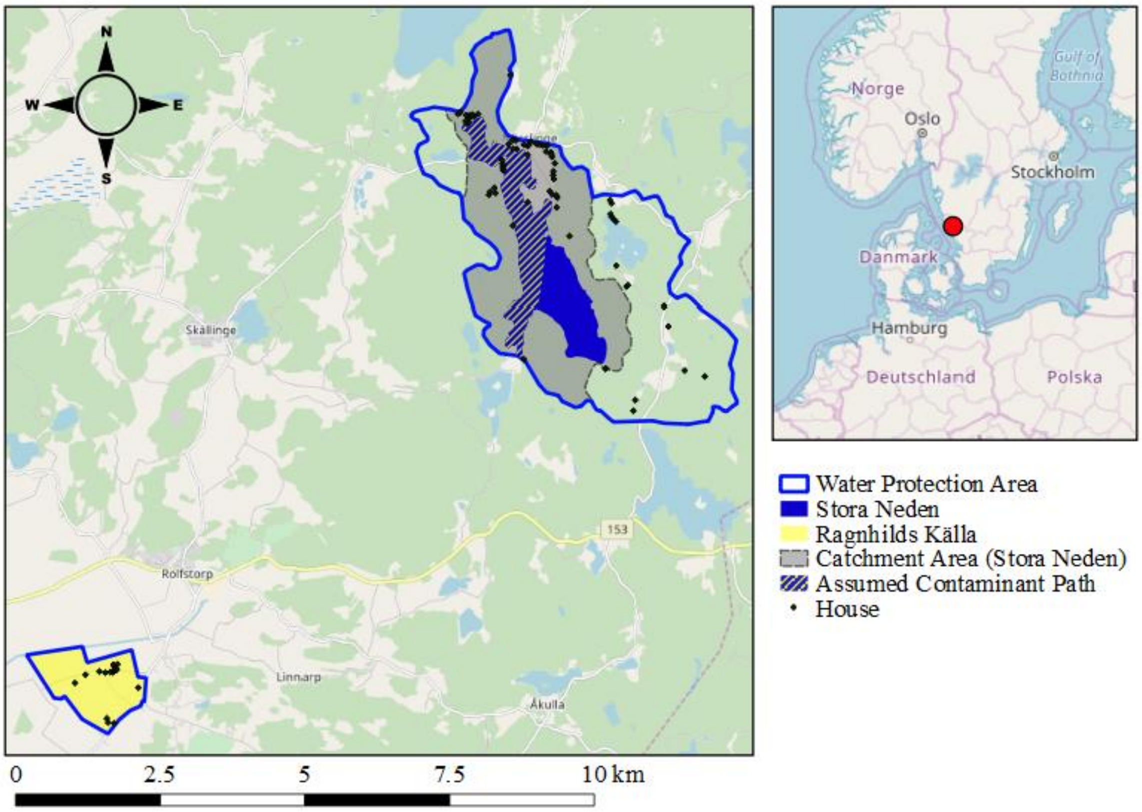

Figure 7 Map over Kvarnagården DWTP's raw water sources; the lake Stora Neden and the aquifer Ragnhilds källa; scale 1:125,000. Note that only the houses within the (grey) catchment area was assumed to have OWTSs discharging wastewater to the lake.

Before the installation of the UF membranes the treatment consisted of one separation step using rapid sand filter, followed by two disinfection steps, UV light at 250 Jm^{-2} and a chloramine dosage, before the distribution network. It is important to note here, that neither the rapid-sand filter nor the chloramine are considered as microbial barriers by the Swedish National Food Agency (SNFA, 2014). A rapid sand filter could, in theory, be considered a valid microbial barrier if it was combined with a coagulation step (i.e. direct filtration treatment), whereas the chloramine should primarily be considered as an inhibitor of microbial regrowth in the distribution system (Norton and LeChevallier, 1997). In addition, SWWA (2009b) states that for a UV disinfection to be considered as a good barrier it needs to operate at a minimum of 400 Jm^{-2} (e.g. Sommer et al., 2004) and that 250 Jm^{-2} should rather be considered as an acceptable level.

In late 2016, the treatment chain was improved, as the 20 nm nominal pore size UF membrane was added as an additional barrier; after the rapid sand filter and before the UV light. This addition did not only improve the drinking water quality and safety directly, but also indirectly. The much more transmittable UF permeate allowed a greater UV strength to be applied at much lower energy levels than what was possible with the previous rapid sand filter filtrate. Hence, the UV disinfection unit was able to be increased to the 400 Jm^{-2} that it was originally dimensioned for (VIVAB, 2012).

6.2 STORA NEDEN

The primary raw water source for the Kvarnagården DWTP is the lake Stora Neden, which constitutes for about 80±5 % of the daily treated raw water volume. The lake has a surface area of approximately 2.9 km^2, a maximum depth of 57 m and an average depth of 19.3 m (SMHI, 2016).

Stora Neden has been rated to a moderately good ecological status, due to its somewhat elevated levels of nutrients and its acidification classification. Furthermore, the lake’s chemical status has been rated as good to not good, particularly due to elevated levels of mercury and polybrominated diphenyl ether, which are considered as environmentally hazardous compounds (VISS, 2017). However, as none of the mentioned problems are at levels high enough to affect human health, the lake can be considered as a good raw water source from a strictly human health perspective.

The possible sources of microbial contamination to the lake, identified from the open geo-data archives provided by Lantmäteriet (2018), involved:

- The 50 houses connected to On-site Wastewater Treatment Systems (OWTS), within the main catchment area of the lake.

- The agricultural field in the northern part of the catchment area that is in direct contact with the lake, which could be a susceptible source for example runoff fertilizer and manure.

- The main road, which crosses the northern part of the lake, should also be considered as a possible, albeit unlikely, source of contamination; via e.g. spillage of hazardous goods from passing transports.

Note that the water protection area for the lakes Stora Neden, established in 2007, does not allow any transport of hazardous goods or the appliance of manure as fertilizer in areas were run-off may contaminate the surface waters (Länsstyrelsen, 2007).

Furthermore, in lack of a more detailed hydrodynamic model, e.g. a finite difference model, a compartmental model was applied instead. It was based on average depth of the lake and the assumed contamination path, yellow striped area in Figure 7, which was chosen as it is the most likely path for the majority of possible contamination sources.

6.3 RAGNHILDS KÄLLA

The secondary raw water source to Kvarnagården DWTP is the groundwater aquifer Ragnhilds Källa, which constitutes for about 20±5% of the treated raw water volume. It is an artesian aquifer, i.e. confined aquifer with groundwater pressure level above ground, meaning it does not require any additional pressure for water abstraction. Artesian aquifers also have a reduced risk of contamination from infiltration because of both their confinement but also the upward pressure gradient.

Ragnhilds källa is considered a good raw water quality source for all measured quality levels of nitrate, pesticides, chloride and sulphates (VISS, 2018). Furthermore, it is not exposed to any chemical risks and its quantity levels has been evaluated to remain good till at least 2021 and likely after that as well (VISS, 2018).

Data regarding the aquifer's exposure to microbial risks has been limited. Apart from the evaluation report of the aquifer's water protection area by Sweco (2001), little information could be found. The identification of contamination sources has therefore been based on the findings of the Sweco report, where five possible microbial hazards was identified:

- The most extreme source of contamination would be the event of a pipe burst in the sewage network connected to the small number of houses and farms nearby the aquifer. Possibly leaking pathogens straight into the aquifer.

- Infiltration, or direct runoff, of pathogens from the appliance of manure as fertilizer on the nearby fields.

- Spillage of hazardous goods, e.g. manure, from passing transports.

- Leakage of pathogens from the breakdown of a manure tank located nearby the water protection area.

- Leakage of faecal substances from a nearby poultry farm.

Similarly to Stora Neden, precautionary measures have been taken to reduce the possibility of the mentioned hazards above (Sweco, 2001). The appliance of manure as fertilizer has been restricted to areas where it should not affect the aquifer. Transports containing hazardous goods must be equipped with the necessary protection to be allowed passage. Plus, all sources of possible leakage are inspected and maintained on a regular basis.

7 METHOD

The health-related risk reduction of the installed UF membrane at Kvarnagården DWTP was estimated via a QMRA-model, described in section 7.1, assessing the *total number of infections* and *reduced number of lost QALYs per year*. The latter was converted into monetary terms, using WTP studies, and analysed in combination with other benefits and costs in a CBA-model, described in section 7.2, to determine if the UF membranes are beneficial from a societal point of view. In addition, the sensitivity of the CBA model was also analysed in section 7.3, to see the impact of the different variables on the uncertainties in the model results. Lastly, two scenario analyses, section 7.4, were assessed to determine if any further treatment steps or measures may be required to ensure a good drinking water quality in the future

All calculations were performed as Monte Carlo simulations, using the Microsoft Excel add-in software Palisade @RISK, version 7.5.1 (10,000 iterations).

7.1 QMRA-MODEL

The gained health effects from the implemented UF membranes were estimated using three reference pathogens, each representing their respective pathogen-category; *Campylobacter* (bacteria), Norovirus (virus) and *Cryptosporidium* (protozoa). This selection was based on their low infection doses, their prevalence in Swedish waters and their connection to previous waterborne disease outbreaks in Sweden (Lindberg and Lindqvist, 2005; Folkhälsomyndigheten, 2015).

The risks were estimated based on the conceptual model seen in Figure 3, where the consequence was set as an event's expected number of lost QALYs and the probability was set as the respective hazard's likelihood of occurring within a given year. Detailed information about the used input variables data can be seen in Table 3 and the estimated probabilities for each event is illustrated in Appendix A & B.

7.1.1 Raw Water Source Characterisation

Based on the information regarding the regulations of Stora Neden's water protection area, only two sources of contamination were included in the microbial risk assessment of the lake. The first contamination source was set as the accidental crash of a truck transporting untreated cattle-manure into the lake. It was assessed to illustrate a more extreme and rare contamination event, like R_X in Figure 3, which could have devastating consequences. The second source of contamination was set as the discharge of pathogens from the OWTSs surrounding the lake. This, second source, was also considered as the baseline risk, meaning that it could be expected to occur on a yearly basis, like R_0 in Figure 3,

Given a release of pathogens to the lake, from any of the two sources, the resulting intake concentrations of pathogens at the DWTP were estimated via equation 14.

$$C_{In_{SN}} = \frac{w_{Source}}{Q_{SN} \times 10^{L+O}} \times (1 - r_{GW}) \qquad \text{Eq. 14}$$

where $C_{In_{SN}}$ was the DWTP's intake concentration of pathogens (# L^{-1}) during the duration of a pathogen release[5] to the lake Stora Neden, w_{Source} was the expected daily pathogen load (# day^{-1}) from the released source, Q_{SN} was the daily water flow (L day^{-1}) of Stora Neden, L was the lake's natural Log-removal, O was the OWTSs Log-removal, and R_{GW} was the groundwater ratio at the DWTP. Note that O was only included for the calculations of intake concentration of pathogens released from the lake's OTWSs, i.e. the baseline risk. Furthermore, L was primarily based on the expected dilution in the lake, as the run time for a contaminant to reach the DWTP's intake is

[5] Considered in the risk characterisation step (7.1.3), when assessing a pathogen's annual probability of infection; equation 6.

expected to be approximately six hours; giving very little time for any deterioration to occur (Bertrand et al., 2012).

Furthermore, the pathogen load of the manure transport was estimated to last for one day and was calculated via equation 15 & 16:

$$w_{Crash} = C_{manure} \times V_{manure} \quad \text{Eq. 15}$$

$$C_{manure} = \sum_{n=i}^{N} C_{cattle_i} \times p_i \times r_{cattle_i} \quad \text{Eq. 16}$$

where w_{Crash} was the load of pathogens (# d^{-1}) released from an accidental crash of manure transport, C_{manure} was the released manure's concentration of pathogens (# L^{-1}), V_{manure} was the released volume of manure (L d^{-1}), i was the type of cattle, C_{cattle_i} was the concentration of pathogens in the cattle type's manure given an infection, p_i was the pathogen's (i.e. cryptosporidium's) prevalence among the said type of cattle and r_{cattle_i} was the ratio of the cattle type in the mixed manure, i.e. juvenile or adult cows.

In comparison, the expected load of the OWTSs was estimated to last for the duration of pathogenic excretion, given an infection, for each respective pathogen and was calculated via equation 17 and 18:

$$w_{OWTS} = C_{Faecal} \times m_{Faeces} \times I \quad \text{Eq. 17}$$

$$I = i \times U \times P \quad \text{Eq. 18}$$

where w_{OWTS} was the expected load of pathogens (# d^{-1}) released from the OWTSs given an infection, C_{Faecal} was the faecal concentration of pathogens (# g^{-1}) given an infection, m_{Faeces} was a person's daily production of faeces (g p^{-1} d^{-1}), I was the expected number of infected people per year living within the catchment area of Stora Neden, i was the pathogens' geometric mean of incidence per person per year in the district of Halland, U was the underreporting factor for each respective pathogen's incident data, as only a small portion of the actual infections are likely to be reported in the medical databases (Gibbons et al., 2014) and P was the number of people connected to the OWTSs in connection to the raw water source[6].

Based on the reports regarding the water protection area of Ragnhilds Källa and its set regulations, the aquifer appeared to be absent of any baseline risks. However, two other sources of pathogenic contamination were still included in the assessment. The first was the infiltration of pathogens from a leakage of the manure tank in the area to a sensitive part of the aquifer; assessed as an either detected or undetected event. The second was the event of a sewage pipe-burst in one of the OWTSs in connection to the aquifer during an infectious load from one household.

The resulting intake concentration of pathogens at the DWTP, given a release of pathogens to the aquifer's area was estimated via equation 19:

$$C_{In_{RK}} = \frac{C_{Source}}{10^{U+S}} \times r_{GW} \quad \text{Eq. 19}$$

where $C_{In_{RK}}$ was the DWTP's intake concentration of pathogens (# L^{-1}) from the aquifer Ragnhilds Källa during the duration of the contaminant release, C_{Source} was the source's released concentration of pathogens (# L^{-1}), U was the Log-removal of the aquifer's unsaturated zone and S was the Log-removal of the aquifer's saturated zone.

[6] Based on the expected number of houses with OWTSs in connection to the raw water source area and the expected number of people per household in the municipality of Varberg.

Note that $C_{In_{RK}}$ was calculated based on concentration rather than load of pathogens, as the estimations of the aquifer's saturated Log-removal includes the contaminant dilution in the groundwater flow. Furthermore, to illustrate that the sewage pipe is likely in near, or in direct, contact with the aquifer's saturated zone; the unsaturated zone was excluded for the events associated with a sewage pipe-burst.

The expected concentration of pathogens in the manure[7] were estimated from literature values, while the sewage concentration of pathogens given an infectious load was assessed via equation 20:

$$C_{Pipe-burst_{Inf.}} = \frac{w_{pipe-burst}}{Q_{OWTS}} \qquad \text{Eq. 20}$$

where $C_{Pipe-burst_{Inf.}}$ was sewage concentration of pathogens given an infectious load, $w_{pipe-burst}$ was the load from the number of infected households (h_i) connected to the leaking sewage pipe, calculated as w_{OWTS} (Eq. 17), and Q_{OWTS} was the OWTSs sewage flow, based on the connected peoples' combined wastewater production; assumed to be equal the to the drinking water consumption.

7.1.2 Treatment Chain

The expected concentration of pathogens being distributed, before ($C_{DW_{Previous}}$) and after ($C_{DW_{Current}}$) the installation of the UF membranes, given one of the mentioned events, was calculated via equation 21 and 22:

$$C_{DW_{Previous}} = \frac{C_{In}}{10^{(B_{RS} + B_{UV250})}} \qquad \text{Eq. 21}$$

$$C_{DW_{Current}} = \frac{C_{In}}{10^{(B_{RS} + B_{UV400} + B_{UF})}} \qquad \text{Eq. 22}$$

where C_{In} was the DWTP's intake raw water concentration of pathogens, i.e. any of the above source characterised concentrations, B was the log-removal of the different treatment barriers, RS was the rapid sand filter, UV was the UV radiation and UF was the UF membranes. Note that the UV strength was increased from 250 Jm^{-2} to 400 Jm^{-2} for the current treatment and that chloramine was excluded from the treatment chain, as its main purpose is to hinder the regrowth of pathogens rather than inactivating them (Norton and LeChevallier, 1997).

The log-removal of the rapid sand filters were based on the findings of Smeets et al. (2006), albeit somewhat modified to better fit with the standards of SWWA (2015); stating that the removal efficiency of the rapid sand filter should be considered as poor. Hence the minimum log-removal was set to zero for all pathogens and the maximal removal efficiency was limited to the mean elimination capacity of the sand filters in the study.

The UV-inactivation capacity is based on the results of Hijnen et al. (2006), where the log-removal was assumed to the maximum removal efficiency for the assessed UV strength. Furthermore, an additional hazardous event, or rather an internal vulnerability was also assessed looking into the impact of a possibly dysfunctional UV barrier during similar circumstances of a baseline event. The probability of failure for the UV light was assumed to be equal to that of the UF membranes, i.e. 0.5 % (VIVAB, 2017).

The UF membranes' log-removal was assigned with respect to a conservative approach to not overestimate their benefits. It was based on data from the manufacturer, X-Flow (2018), the operational routines at Kvarnagården DWTP and the previously mentioned QMRA-model by

[7] assumed to be same as for the transported manure

Åström et al. (2017), where the minimum separation efficiency was assigned for all pathogens as a sharp cut-off value. Uncertainties were disregarded, as the log-removal of the UF is solely based on its pore-size. Hence, events linked to a barrier's availability, which could affect other treatment steps, e.g. power failure, would not decrease the UF's removal efficiency but rather the water flow. The only viable inclusion of a UF failure would be a membrane breakdown. Yet this was considered too unlikely, given the operational routines at Kvarnagården DWTP, to be included from a practical sense, as it would not give much information about the societal benefits of the membranes.

7.1.3 Risk Characterisation

The expected number of infections for each event was estimated from equations 3 – 7; multiplying the number of drinking water consumers with the total annual probability of infection. Furthermore, given each pathogens' $P_{inf\ Annual}$, the pathogens' respective number of infections were used to calculate their annual total burden on the municipality of Varberg's DWS as:

$$QALY_{Tot} = \sum_{n=i}^{N} I_i \times \Delta QALY_i \qquad \text{Eq. 23}$$

where $QALY_{Tot.}$ was the municipality's total change in QALYs for each hazardous event, N represented the total number of assessed pathogens, I_i was a pathogen's number of annually infected drinking water consumers, and $\Delta QALY_i$ was a pathogen's effect on the infected consumer's QALY. Here it is also possible to note how the $\Delta QALY_i$ value is connected to the pathogens respective underreporting factor, where a higher morbidity gives a lower level of underreporting.

By asserting the consequence of an event as its $QALY_{Tot.}$, given its occurrence, and combining this with the event's probability of occurrence the microbial risk of each event was calculated via equation 1. In addition, the total microbial risk for all events were calculated via a simplified version of equation 2:

$$R_{Tot} = \sum_{x=1}^{X} (c_x - c_{x-1}) \times p_x \qquad \text{Eq. 24}$$

were R_{Tot} was the total microbial risk, X was the total number of assessed events, c_x was the $QALY_{Tot}$ of a specific event and p_x was the event's probability of occurrence. Note that c_1 represent the consequence of the baseline risk, while the most severe event was represented with c_X. In addition to this, all hazardous events with lower consequences than that of the baseline risk were also excluded from the model, as they would not affect the total risk in any substantial way.

Finally, the expected risk reduction of the UF membranes was calculated as:

$$R_{Reduced} = R_{Previous} - R_{Current} \qquad \text{Eq. 25}$$

where $R_{Reduced}$ was the expected reduction of a risk, $R_{Previous}$ was the estimated risk before the instalment of the UF membranes and $R_{Current}$ was the estimated risk after the UF membranes had been installed. Note that equation 24 can be used to calculate both the individual risk reduction of a specific event, as well as the total risk reduction of the system.

Table 3 Detailed information regarding the input variables data for the quantitative microbial risk assessment.

Section	Input Variable	Symbol	Pathogen Type	Distribution Type [a]	A	B	C	Source
Raw Water Source Char. Endemic Risk	Households connected to OWTS at Stora Neten	-	-	-	-	-	-	Lantmäteriet (2018)
	People per household	-	-	Triangular	1.0	2.0	7.0	SCB (2017)
	Incidence per person per year	i	Camp.	Ext Value	$9.9\ 10^{-4}$	$1.3\ 10^{-4}$	-	Folkhälsomyndigheten (2018) [b]
			Noro.	Ext Value	$3.3\ 10^{-4}$	$1.5\ 10^{-4}$	-	
			Cryp.	Uniform	0	$3.8\ 10^{-4}$	-	
	Underreporting factor	U	Camp.	-	-	-	-	Haagsma et al. (2013)
			Noro.					Lindqvist et al. (2001)
			Cryp.					Lindqvist et al. (2001)
	Faecal pathogen concentration (# g^{-1})	C_{Faecal}	Camp.	Triangular (Log_{10})	4	6	10	Petterson et al. (2016a) [c]
			Noro.		5	8	11	Based on various authors [d]
			Cryp.		6	7	9	Petterson et al. (2016a) [c]
	Faecal production (g p^{-1} d^{-1})	m_{Faecal}	-	Triangular	200	800	1000	Fine and Fordtran (1992)
	Flow Stora Neten (L d^{-1})	Q_{SN}	-	Gamma	$1.3\ 10^{8}$	$2.3\ 10^{7}$	-	SMHI (2016) [b]
	Log removal OWTS	O	All	Triangular	1.0	2.0	3.0	USEPA (2002)
	Log-removal in surface water	S	All	Triangular	2.5	3.5	4.5	Dilution; contaminant path area Figure 7
	Groundwater ratio	r_{GW}	-	Triangular	15%	20%	25%	VIVAB (2011)

Section	Input Variable	Symbol	Pathogen Type	Distribution Type [a]	A	B	C	Source
Raw Water Source Char. Risk Events	Manure pathogen concentration of juvenile cattle (# g^{-1})	$C_{cattle_{Juv}}$	Cryp.	Log-normal	38155	22500		Ferguson and Kay (2012)
	Manure pathogen concentration of adult cattle (# g^{-1})	$C_{cattle_{Adu}}$	Cryp.	Log-normal	3830	46		Ferguson and Kay (2012)
	Mean pathogen prevalence in juvenile cattle	p_{Juv}	-	-	-	-	-	Ferguson and Kay (2012)
	Mean pathogen prevalence in juvenile cattle	p_{Adul}	-	-	-	-	-	Ferguson and Kay (2012)
	Cattle ratio (Juvenile:Adult)	r	-	Beta (Alt)	10	20%	0;1	Assumed
	Volume of manure transport (L)	V_{manure}	-	Triangular	$1.0\ 10^4$	$2.0\ 10^4$	$3.0\ 10^4$	Harrigan (2011)
	Household connected to broken OWTS at Ragnhilds Källa	h_i	-	-	-	-	-	Assumed
	Waste water production (L p^{-1} d^{-1})	-	-	Uniform	160	200	-	SWWA (2009a)
	Log-removal unsaturated zone	U	Camp.	Triangular	0.9	4.4	8.4	Ho et al. (1992) Åström et al. (2017)
			Noro.		0.6	3.1	5.9	
			Cryp.		0.9	4.4	8.4	
	Log-removal saturated zone	S	Camp.	Triangular	1	2.2	3.3	Sinton (1980) Åström et al. (2017) [e]
			Noro.		2.9	3.2	4	
			Cryp.		1	2.2	3.3	

Section	Input Variable	Symbol	Pathogen Type	Distribution Type [a]	A	B	C	Source
Treatment Chain Log-Removal	Rapid sand filter	*RS*	Camp.	Triangular	0	0.1	0.6	Smeets et al. (2006)
			Noro.		0	0.1	0.8	
			Cryp.		0	0	1.8	
	UV (250 m^{-2})	*UV250*	Camp.	-	-	-	-	Hijnen et al. (2006)
			Noro.	-	-	-	-	
			Cryp.	-	-	-	-	
	UV (400 m^{-2})	*UV400*	Camp.	-	-	-	-	Hijnen et al. (2006)
			Noro.	-	-	-	-	
			Cryp.	-	-	-	-	
	UF (20 nm)	UF	All	-	-	-	-	X-Flow (2018)
Risk Character.	Infectivity	*r*	Camp.	Beta	0.024	0.011	-	Teunis et al. (2005)
			Noro.		0.04	0.055	-	Teunis et al. (2008)
			Cryp.		0.115	0.176	-	Teunis et al. (2002)
	Unboiled drinking water consumption	*V*	-	Log-normal	-0.299	0.570	-	Westrell et al. (2006)
	Drinking Water Consumers	-	-	-	-	-	-	SCB (2017)
	Duration of pathogen excretion (d)	*d*	Camp.	Triangular	15	34	42	Petterson et al. (2016a) [c]
			Noro.		14	29	45	Petterson et al. (2016b)
			Cryp.		5	10	30	Petterson et al. (2016a) [c]

Section	Input Variable	Symbol	Pathogen Type	Distribution Type [a]	A	B	C	Source
Risk Character.	Duration of a detected event [e] (d)	d	All	-	-	-	-	Assumed
	Duration of an un detected event [f] (d)	d	All	Triangular	1	7	14	Assumed
	Lost QALY per infection	$\Delta QALY$	Camp.	-	-	-	-	Batz et al. (2014)
			Noro.	-	-	-	-	
			Cryp.	-	-	-	-	

a) Triangular (A = min; B = mode ; C = max), Ext Value (A = α; B = β), Uniform (A = min; B = max), Gamma (A = α; B = β), Log-normal (A = mean; B = StDev), Beta (Gamma (A = α; B = β), Beta (Alt) (A = α; B = P50; C = min; max)
b) @Risk fitted distribution, using Akaike Information Criterion (AIC)
c) Various authors = Chan et al. (2006); Atmar et al. (2008); Petterson et al. (2016b); Newman et al. (2016); and Teunis et al. (2015)
d) Using the virus transport model, where ground water flows were based on information from SGU (2018) and the outtake flow recorded by Sweco (2017)
e) Detected events were considered as the crash of a manure tank into Stora Neten, the detected leakage from the manure tank at Ragnhilds Källa and the dysfunctional UV barrier
f) Undetected events were considered as the pipe-burst of an OWTS nearby Ragnhilds Källa and the un detected leakage from the manure tank at Ragnhilds Källa

7.2 CBA-MODEL

The UF membranes' NPV was calculated by comparing their quantified benefits, section 7.2.1, against their capital and recurrent cost, section 7.2.2, using equation 12. The discount rate was set to 3.5 %, as recommended by HM Treasury (2018) and the Swedish transport administration (Trafikverket, 2018), for a time horizon of 50 years.

7.2.1 Benefits

The identified benefits of the UF membranes have been divided into four sections. The first three sections include the three perspectives for valuing water, discussed in section 5.3, while the fourth includes the reduced biofilm in the distribution network, which does not fit any of the discussed perspectives specifically.

7.2.1.1 Health-Related Benefits

The first benefit of the UF membranes was analysed as the improvement of the health-related wellbeing of Varberg's drinking water consumers. It was linked to the UF membranes' microbial risk reduction and was assessed as the reduction of lost QALYs per year.

The value of the improved QALYs was evaluated to, as far as possible, resemble a societal rather than individual value of QALY. It was set as a triangular distribution of 0.7 (min), 1.0 (mode) and 1.3 (max) MSEK and was based on the findings of Ryen and Svensson (2015) and the Swedish governments revealed preference value for a QALY (Svensson et al., 2015). . Put in perspective to the Swedish gross domestic product (GDP); the value of a QALY fell in the range between 1.5 and 3 times the 2015 Swedish GDP per capita (Gunér, 2017), which is within an acceptable range of the findings of Nimdet et al. (2015).

7.2.1.2 Social Benefits

The second benefit was linked to the social capital of Varberg's drinking water consumers. It was analysed as the UF membranes improvement of the drinking water's aesthetic quality, e.g. odour and taste. The aesthetic improvements was identified, albeit not quantified, by VIVAB's customer service. The customer service office had received an increasing number of verbal appreciations regarding the municipality's drinking water quality, since the UF membranes installation in 2016. A positive feedback which could likely explained by the UF membranes more than 50% reduction of NOM in the distributed drinking water (Keucken, 2017; VIVAB, 2017).

The monetary value of the aesthetic benefit was set to a triangular distribution of 200 (min), 400 (mode) and 600 (max) SEK per household per year; for a total of 20,000 households. The monetary valuation was based on the findings of three contingent valuation studies (Brox et al., 1996; Polyzou et al., 2011; Beaumais et al., 2014), one household production study (Lanz and Provins, 2016), and the municipality of Varberg's 2017 water bill[8] of 1850 SEK per apartment per year.

7.2.1.3 Industrial Benefits

The third benefit is associated with the industrial value of clean water. In the municipality of Varberg, the local industries' that are dependent on clean water for their processes are dominated by the municipality's Nuclear Power Plant (NPP), Ringhals. That is, that even if there are definitely other industries dependent on the distribution of clean water from the Kvarnagårdens DWTP, the NPP is considered the DWTP's major client; receiving approximately 10^6 m^3 per year, i.e. about 20 % of the DWTP's distributed volume.

The NPP is dependent on ultra-pure water for its cooling processes, which is why it uses RO membranes, as an additional desalination step, for the distributed water from the Kvarnagårdens DWTP (Keucken, 2017). However, before the installation of the UF membranes,

[8] The water bill includes the supply of drinking water and the collection of waste water but excludes the collection of storm water

the increasing levels of NOM at Stora Neden began to put too much pressure on the RO membranes at Ringhals, forcing the NPP to work well below its original design mode. This situation has since then been mediated by the installation of the UF membranes at Kvarnagården DWTP. The implemented UF membranes has lowered the Total Organic Carbon (TOC) levels of the RO membranes' effluent by tenfold, from 200 – 500 ppb to 25 – 30 ppb, since their implementation. A solution which has reassured that the NPP is now back to producing electricity at its original design mode (Bengtsson, 2018).

However, for the framework of this thesis, it has not been possible to monetise this variable. Instead, the industrial value for the UF membranes was solely included as a soft and non-monetised variable in the result's

7.2.1.4 Reduced Biofilm

The last and forth noticed benefit of the UF membranes at Kvarnagårdens DWTP, was the decreased levels of biofilm in the distribution network (VIVAB, 2017). It is a possible benefit which is relevant for all of the other categories, as discussed in section 3.2, and is as such not categorised in any of the above categories specifically.

At the moment little is known about the actual impact of the reduced biofilm in Varberg's distribution network. Although research is currently being conducted, at the University of Lund, to determine its long-term effects. Furthermore, as the factors surrounding the impact of biofouling are so complex, it is not even certain that its reduction will result in a positive outcome. Albeit, no negative consequences has yet to been seen, since the initial operation of the UF membranes in November 2016 (VIVAB, 2017).

The reduced biofilm in the distribution network was henceforth only considered as a non-monetised variable, similar to the industrial benefits.

7.2.2 Costs

The costs of the UF membranes were based on their direct market value and estimated by VIVAB to 108 MSEK for the investment, i.e. its capital cost, and about 2 MSEK per year in maintenance, i.e. its recurrent costs. Note that these values are considered as certain, as they were assessed post the instalment of the membranes and does as such not include any variability or uncertainty in the analysis.

7.2.3 Non-Monetised Variables

If the output's minimum value (NPV_{min}) is negative, the required cumulative value of the non-monetised benefits, to ensure a positive NPV, can be calculated as:

$$B_{Non-Monetised} > \frac{-NPV_{min}}{\sum_{t=0}^{T} \frac{1}{(1+r)^t}} \qquad \text{Eq. 26}$$

where $B_{Non-Monetised}$ is the cumulative non-monetised benefits required value for making the UF membranes profitable, NPV_{min} is the minimum, and negative, value of the result's distribution, t is any given year within the time horizon T and r is the result's assessed discount rate.

Note that a similar analysis could be conducted for non-monetised costs, if the minimum NPV was positive, to see how high these costs would have to be to threaten a positive outcome.

7.2.4 Impact of Discount Rates

Lastly, the impact of the discount rate on the NPV was also assessed, as recommended by e.g. Hanley and Barbier (2009), to see how the result may change with this variable. The recommended discount rate of 3.5 % (HM Treasury, 2018; Trafikverket, 2018) was therefore compared against a lower and higher discount rate, of 1.5 % and 5.5 %, respectively. The former would be appropriate if all benefits were considered as health-related (HM Treasury, 2018), whereas the latter was assessed to see how a more descriptive discount rate, i.e. one that better corresponds with the interest of the market (g = 4%), may affect the result (Söderqvist, 2006; Gunér, 2017).

7.3 Sensitivity Analysis

The model's sensitivity was analysed, utilizing the @RISK software's monte carlo simulations, to determine the impact of each respective input variable. The sensitivity was assessed in two analyses. The first analysis assessed the input variables' change in output mean, where one compared how a variable's input value, from its P5-value (low input) to its P95-value (high input), would change the mean of the result; given that the other variables remained unchanged.

The second analysis calculated the input variables' respective spearman rank. The spearman rank determines a variable's correlation to the final outcome of the result. The rank can vary between -1 and 1, where values close to |1| are of a major importance for the results, while values close to zero indicate a low importance for the result. Furthermore, a positive rank implies a direct correlation with the result, i.e. that the result increases with the variable, whereas a negative value implies an inverse relation to the result, i.e. that the result decrease with an increase of the variable. Note that it is only applicable to monotonic functions, i.e. functions which are either continuously decreasing or increasing (Stover, 2018).

7.4 Scenario Analyses

Two future scenarios were assessed to determine if any further treatment steps or measures may be required to ensure a good drinking water quality in the future. The first scenario analysed the possible effects of climate change, while the second scenario analysed the possible consequences of a petroleum leakage to any of the DWS's raw water sources.

The scenarios were analysed, as literature studies, to address the mentioned limitation of the study, section 1.3, and to widen the scope of the thesis beyond the assessed microbial risks. Note that they may not cover the whole extent of possible events, yet they do highlight two important aspects, which may likely affect the DWS in the future.

8 RESULTS

The estimated effects of the implemented UF membranes at Kvarnagårdens DWTP are presented in the below three sections. The first section, 8.1, illustrates the results of the QMRA and the UF membranes' effect on the microbial risks in the DWS of the municipality of Varberg. In the second section, 8.2, the results of the CBA and the UF membranes' socio-economic profitability are presented. The third section, 0, includes a sensitivity analyses, where both the respective input-variables are assessed, as well as the impact of different scenarios.

8.1 QMRA

The annual probability of infection ($P_{Total\ inf}$) of the previous and current treatment setup was analysed and in Table 4 the percentile values (P5, P50 and P95) and the corresponding number of infections per year. From these results, it is clear that the P95-value of the previous treatment is much greater than the recommended value of 10^{-4} by the (WHO, 2016).

Table 4 Estimated percentile values for the $P_{Total\ inf}$ and the corresponding expected number of infections per year. Unacceptable levels, according to WHO's (2016) recommended level, are coloured in red.

Situation	$P_{Total\ inf}$ (Expected Infections) P5	$P_{Total\ inf}$ (Expected Infections) P50	$P_{Total\ inf}$ (Expected Infections) P95
Before UF	$5.0\ 10^{-9}$ (<< 1)	$4.0\ 10^{-5}$ (2)	$3.8\ 10^{-2}$ (2,400)
After UF	$3.0\ 10^{-13}$ (<< 1)	$6.1\ 10^{-10}$ (<< 1)	$1.1\ 10^{-7}$ (<<1)

Given each pathogen's annual probability of infection, the mean total health improvement of the UF membranes, illustrated by the green area in Figure 8, was calculated using equation 25, to 0.62 (P5 = 2 x 10^{-4}; P95 = 2) gained QALYs per year. Note that some events have been excluded from the risk graph since they had a lower consequence than that of the baseline risk and would not affect the total risk in any substantial way.

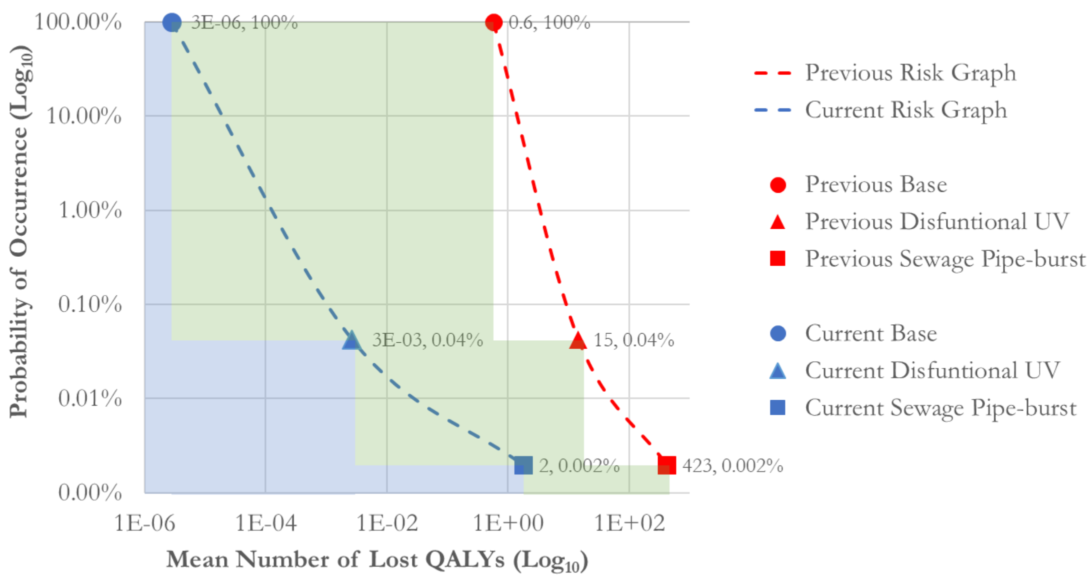

Figure 8 Estimated risk graph of the hazardous events contributing to the total risk; plotting each event's probability of occurrence against its mean consequence, with or without the UF membranes, i.e. current or previous. The total risk reduction of the UF membranes is illustrated by the green area, while the total remaining risk is illustrated by the blue area.

8.2 CBA

The expected NPV of the UF membranes is illustrated in Figure 9, where the mean NPV was calculated to +47 (P5 = -45 ; P95 = +117) MSEK, at a discount rate 3.5% and a time horizon of 50 years. Note that the aesthetic benefits was the main contributor to the UF membranes NPV, with a mean value of 190 (P5 = 125; P95 = 255) MSEK, while the health-related benefits was calculated to a mean value of 13 (P5 = 0.005; P95 = 49) MSEK.

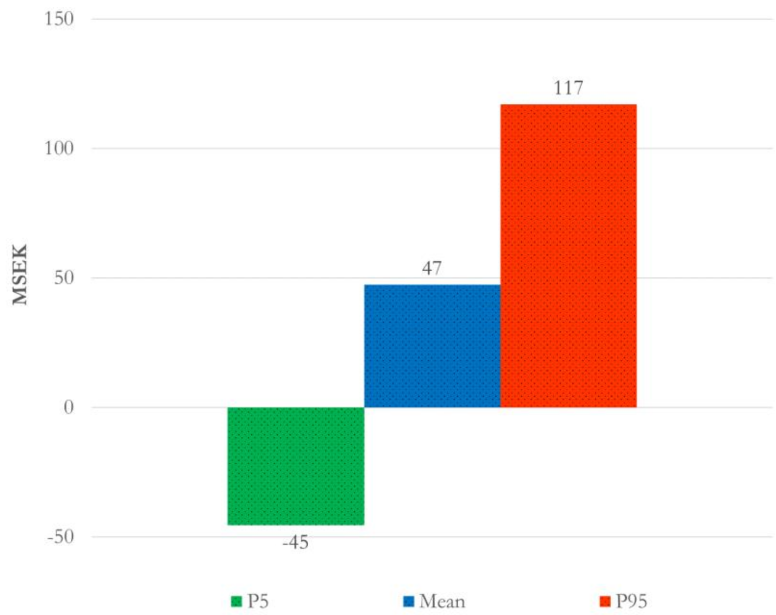

Figure 9 The bar chart shows the calculated NPV's P5-, mean and P95-value in their respective colours

Furthermore, the probability of an NPV above zero SEK, i.e. that the investment in the UF membranes was a profitable expenditure, was estimated to 82 %, see Table 5. The impact of different discount rates was also compared in Table 5, where it is clear that the discount rate has a significant impact on the result and that the implementation of a descriptive discount rate (5.5 %) has a negative effect on the profitability of the UF membranes.

Table 5 NPV of improved QALYs and aesthetic benefit at three different discount rates, 1.5%, 3.5% and 5.5%, showing their respective P5, mean, and P95 values, as well as their certainty of being profitable.

Discount Rate	P5	Mean	P95	Probability of NPV > 0
5.5 %	-50	5	54	52%
3.5 %	-45	47	117	82%
1.5 %	48	162	264	100%

Note that only a part of the expected benefits have been quantified and valued and if the cumulative value of the non-monetised benefits exceeds 2.4 MSEK per year, at a discount rate of 3.5 %, the outcome of a positive NPV (> 0) has a probability of 1. Similarly, for a discount rate of 5.5%, the cumulative value of the non-monetised benefits would have to exceed 4.0 MSEK per year to ensure a positive outcome. In perspective, the mean annual value for the health-related and aesthetic benefits was calculated to 0.6 and 8 MSEK per year, respectively

8.3 Sensitivity Analyses

Two sensitivity analyses of the CBA model are presented below. The first analysis, Figure 10, illustrates the variables' change in output mean. The second analysis, Figure 11, shows the variables' spearman rank.

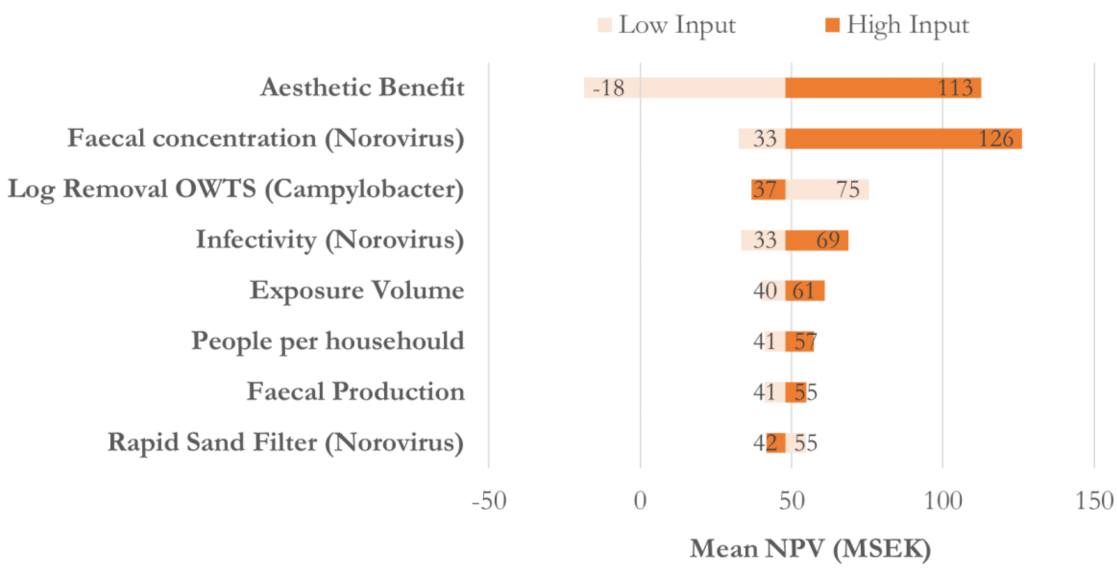

Figure 10 Ranked input variables based on their effect on the output's mean NPV, at both high (P95) and low (P5) input values from their assigned distributions. Baseline is equal to the mean result of 47 MSEK.

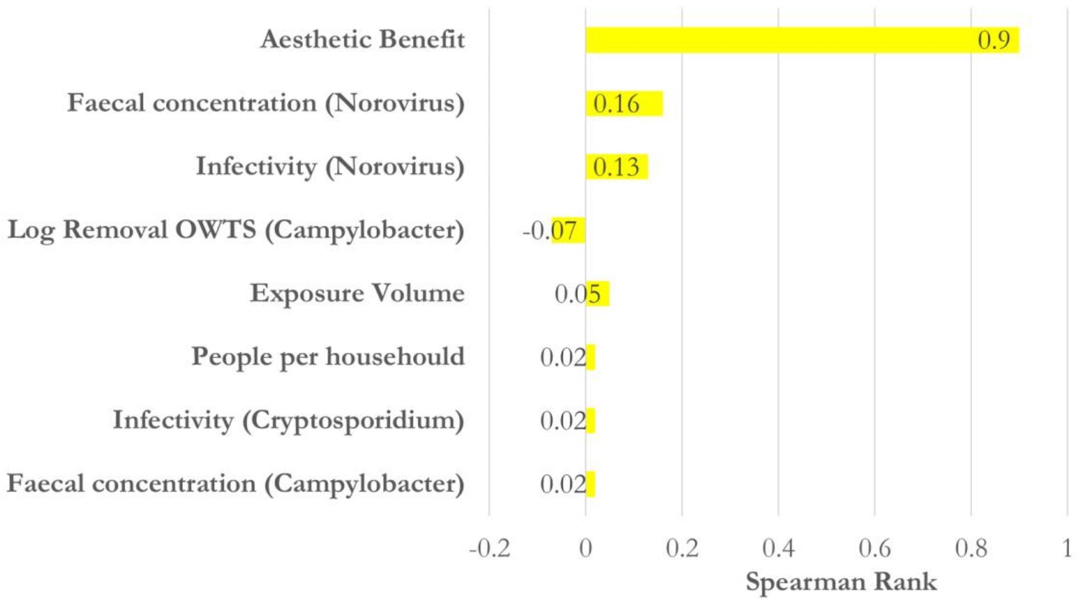

Figure 11 Spearman ranks of the input variables that contribute the most to the result.

From the sensitivity analyses, Figure 10 and Figure 11, it appears that the pathogen related parameter that contributes most to the uncertainty in the NPV is related to Norovirus. There are primarily three variables that contribute to the large uncertainty of the results, i.e. NPV. These are the aesthetic benefit, the faecal concentration of pathogens and the pathogens infectivity (i.e. dose response).

8.4 SCENARIO ANALYSES

The result of the literature studies for the two scenario are presented in section 8.4.1 and 8.4.2, where the possible effects of climate change and the possible consequences of a petroleum leakage to any of the DWS's raw water sources, were assessed respectively.

8.4.1 Climate Change

The future expected changes in climate, due to primarily anthropogenic global warming, will likely cause longer periods of heat and shorter but more extreme durations of precipitation in the future; increasing number of areas which will suffer from droughts as well as floods (Pearce et al., 1989; SOU, 2007; IPCC, 2014).

For the raw water sources in the DWS of Varberg, the effect of climate change will likely lead to an increased average temperature and precipitation (SOU, 2007; SMHI, 2018). The Geological Survey of Sweden (SGU) has estimated that the yearly average groundwater recharge, of Ragnhilds Källa, can be expected to either stay unchanged or increase by up to 10%, by the year 2100 (Rodhe et al., 2009). However, even if the annual average groundwater recharge may be unaffected by the climate changes, SGU still expects to see changes to the monthly variations of groundwater levels. In their study, Sundén et al. (2010) predicts that one can expect to see increased groundwater levels during the years' early months, i.e. January – April, but lower levels during the summer months, i.e. May – September. Note that the change in the yearly groundwater pattern is linked to the shift in precipitation from snow to rain in the winter months, i.e. November – Mars, and the increased dry periods during the summer months.

In addition, increased levels of precipitation will also increase the likelihood of contaminating the raw water sources. During heavy rainfall, it is not unlikely that toxic chemicals, pathogens or NOM may be transported with the run-off to the raw water sources (Eikebrokk et al., 2004; SOU, 2007; SWWA, 2007; Aschermann et al., 2016). For the lake Stora Neden, the increasing temperature from climate change, may also effect the lake's turn-over time (Köhler et al., 2013). During a lake's turn-over, contaminants in the sediments, removed from the system, may resuspend back into the water.

However, because of the improved microbial treatment at Kvarnagårdens DWTP and the already established water protection areas of the raw water sources, it is possible to reassure that neither the increased risk of contamination from microbial pathogens nor toxic chemicals should be the primary focus of an assessment on the effects of climate changes on the DWS. Instead, this scenario analysis was rather focused on the rise of NOM, in the lake Stora Neden, and the UF membranes possibility of handling this issue.

As mentioned in Chapter 0, NOM is not a toxic compound in itself, but may have a negative effect on the water quality both before, during and after the treatment. Keucken (2017) mentions four distinct problems with increased levels of NOM in the water:

i. It has a negative effect on water quality relevant to colour, taste, and odour;
ii. It increases the disinfectant dose requirements, resulting in potential harmful DBPs production;
iii. It promotes biological growth in the distribution system; and
iv. It may increase levels of complex heavy metals and adsorbed organic pollutants.

Based on the findings of Keucken (2017), it appears as the implemented treatment of NOM, using a combined setup of UF membranes with in-line coagulation, is currently sufficient for meeting the drinking water standards. Still, this may not apply in the future, as the setup's maximum removal capacity is limited to a dosing rate of 1.6 mg $Al_2(SO_4)_3$ L^{-1} (Keucken et al., 2017). Keucken et al. (2017) has found that this improved treatment of NOM may be insufficient for meeting the Swedish requirement standards as early as year 2031, at a 15 % groundwater ratio. Albeit, it appears to be a viable setup beyond year 2040 at a groundwater use of 25 %. In other words, one solution to the rising levels of NOM in Stora Neden could as such be to keep increasing the groundwater ratio at the DWTP, to meet the Swedish requirements.

To see how this possible change of the UF membranes time horizon may impact the model and the profitability of the UF membranes. The mean NPV were assessed at two addition time horizons, 15 and 25 years; representing the 15 % and 25 % groundwater ratio respectively. The results showed that 15 years would not be a sufficient time period for the membranes to generate the required benefits, as its mean NPV was calculated to -38 MSEK. Yet, the somewhat longer time horizon of 25 years may possibly be sufficient as its mean NPV was calculated to 2 MSEK.

The solution of consistently increasing the groundwater ratio, for the next 50 years, may, however, be considered unsustainable. As a result, four types of additional treatment measures were investigated, to see if any of them would be suitable to mediate the problems of rising NOM-levels in the future. The result of the study can be seen in Table 6, which shows 13 different studies recommended treatment steps for reducing NOM in the drinking water.

Table 6 Result of literature study of possible measures to assist, or replace, the UF membranes treatment of increasing NOM-levels.

Study	Activated Carbon	NF Membranes	Anion Exchange Resins (AER)	Alternative Coagulation
Aschermann et al. (2016)	x		x	
Plourde-Lescelleur et al. (2015)	x	x		
Särkkä et al. (2015)				x
Black and Bérubé (2014)	x			
Aslam et al. (2013)				x
Al-Naseri and Abbas (2009)	x			
Ødegaard et al. (2009)		x		
Cornelissen et al. (2008)			x	
Humbert et al. (2008)	x		x	
Matilainen et al. (2006)	x			
Lebeau et al. (1998)	x			
Jacangelo et al. (1995)	x	x		
Vickers et al. (1995)	x			

From Table 6, it is clear that the most commonly suggested and investigated measure for the reduction of NOM, in drinking water, is the utilization of various activated carbon methods. Even though combinations with other measures, e.g. AER, are often recommended. In addition, some studies in Table 6 also recommended a combination of the suggested measures with UF membranes. This was, however, not included in the table, as the UF membranes are already implemented at the DWTP.

8.4.2 Petroleum Leakage

Apart from dealing with the rising NOM-levels of the intake water, Kvarnagårdens DWTP must also produce a chemically safe drinking water for its consumers. One drawback of the UF membranes is, however, that they may not be sufficiently able to remove the possible contamination of chemical pollutants, as mentioned in chapter 1 (Limitations). Even with the additional coagulation step, before the UF membranes, which was specifically added for the purpose of removing dissolved chemicals from the drinking water. There is still a relatively large probability of dissolved contaminants reaching the distribution network given their release to any of the raw water sources.

For example, given that the contaminants from a petroleum spillage reaches the DWTP's water intake, only the larger pollutants, i.e. molecules with ten or more equivalent carbon (EC), would likely be removed in the treatment chain (WHO, 2008). This scenario analysis was therefore conducted to assess the possible consequences of such a spillage in either of the DWTP's two raw water sources, as well as the consequences of spilled contaminants reaching the distribution network.

It is important to note here, that even if the consumption of petroleum products can be very harmful, the distribution of petroleum products is more likely to cause taste and especially odour problems long before it reaches those levels. As the odour problems likely would deteriorate any consumption of the drinking water, it is very unlikely that an accident linked to the distribution of petroleum products will cause any health-related problems (WHO, 2008; WHO, 2017d). Instead it will rather affect the aesthetic quality of the drinking water.

That being said, it is still crucial that a spillage is remediated and preferably before it reaches the distribution network. The consequence of an accidental petroleum spillage has therefore been assessed by the expected cost of remediation, at either Ragnhilds Källa or Stora Neden, as well as the expected cost of an interrupted drinking water supply; given that the contaminants reach the distribution network.

The most common ways to treat the release of petroleum products to surface waters is via the utilisation of (sorbent) booms and skimmers (ITOPF, 2014a; ITOPF, 2014b; ITOPF, 2014c). The mean cost of such a remediation was assessed to 0.7 MSEK with a CI90% of 0.4 MSEK – 0.9 MSEK, based on the findings of Cohen (2010) and assuming a spilled volume in the same range as for the manure transport in Table 3.

For the remediation of a petroleum contaminated groundwater aquifer the possible treatment methods are many. The most common approach is the utilization of the so called pump and treat method (Brusseau and Maier, 2004). It is a method, which as the name suggest, pumps in clean water to the aquifer, while it extracts and treats the contaminated water. By continuously pumping back treated water to the aquifer, the system's contaminants are flushed and treated. Various complementary actions, such as heating, may be performed to enhance the remediation. Based on the findings of the National Research Council (1997), the mean cost for remediating Ragnhilds Källa, given a contamination, was estimated to 9.2 MSEK, with a CI90% from 6.7 MSEK – 11.7 MSEK.

Lastly, given that an accidental spill was not sufficiently remediated, the consequence of an interrupted distribution network was based from the findings of Bolander and Martinsson (2018). From this study, the daily cost of interruption was estimated to vary somewhere between 300 – 600 SEK per drinking water consumer per day, for an interruption between one to three days. The mean total cost of interruption was assessed to 56 MSEK, with a CI90 % of 28 MSEK – 93 MSEK.

However, even if this scenario analysis show that the consequences from an accidental spillage of hazardous goods could cost more than 100 MSEK. The probability of such an event occurring is very unlikely, due to the raw water sources' implemented water protection areas and corresponding protocols. The author, therefore, recommends that if an additional treatment step is to be installed at the DWTP, to reduce the consequences of a possible petroleum spillage, it should also be able to remediate the problem of increasing NOM-levels in Stora Neden. This is as the latter is much more likely to cause problems in the future, than the former.

Based on the findings of WHO (2017a) and the suggested measures for the treatment of NOM, Table 6. Only one treatment method appeared to function properly for the removal of both ionically charged organic molecules, e.g. humic acids, and uncharged organic molecules, e.g. benzene. The suggested measure for the mediation of an accidental petroleum spillage have therefore been suggested as the implementation of an additional treatment step at the DWTP using activated carbon.

9 DISCUSSION

The discussion has been divided into four parts to distinguish the topics of the discussion. The first part, section 9.1, includes a discussion on the ex-post approach and what was anticipated, compared to what was realized. In the second part, section 9.2, the structure of the model and its underlying uncertainties are discussed. The third part, section 9.3, focuses on the meaning and implication of the results, whereas the fourth part, section 9.4, deals with what possible measures that could be added for securing a good quality drinking water in the future.

9.1 EX-POST ANALYSIS

CBAs are commonly conducted before the implementation of a measure, comparing a number of different options, to aid the decision makers selection of the most economically viable solution to a problem. In contrast, this thesis differs from the general approach as the CBA was conducted post the measure's installation. Rather than opting for the best measure, this thesis used a CBA to solely assess a measure's societal economic value.

The ex-post analysis was, therefore, conducted to see if the UF membranes at the Kvarnagårdens DWTP was a profitable investment, from a societal point of view. In addition, it was also hoped that an ex-post analysis would further increase the understanding regarding the applicability QMRA and CBA in combination to evaluating water safety measures.

Note, that the ex-post assessment of the UF membranes may have been conducted too shortly after their installation in 2016, as additional benefits may have yet to be observed. Furthermore, additional and better data could have strengthened the result's robustness and decreased its uncertainty. From the method's valuation of possible benefits, section 7.2.1, it was clear that lack of data primarily affected three variables:

i) The aesthetic quality of drinking water – Actual statistical data, rather than just the verbal appreciation, regarding the change in complaints since the UF membranes installation could have given a more accurate evaluation of the consumers WTP for its improvement. The consumers' WTP is both linked to their trust and social capital of the system, as well as the degree of improvement (Polyzou et al., 2011; Beaumais et al., 2014; Söderqvist and Wallström, 2017). In addition, this variable is particularly interesting due to its high spearman rank of 0.9, Figure 11; making it the most influential variable of the final result.

ii) The effect of decreased biofilm – More detailed information regarding the distribution network's biofilm, apart from its apparent decrease, could have made it possible to monetise this variable. More importantly, improved data regarding the exact effects of decreased biofilm in the distribution network could have certified it as a benefit. Whereas now, as its effects are still unknown, it cannot definitely be considered as beneficial. On the bright side this variable is currently being investigated by the University of Lund, whose results may make it possible to include the reduction of biofilm in future CBAs of DWSs.

iii) The value of improved operation capacity and decreased maintenance at Ringhals nuclear power plant – This variable was left non-monetized as Ringhals had not made any investigations on the monetary value of the NPP's improved intake water. This is particularly unfortunate as the monetarisation of this variable likely could have a made an extensive impact on the final result.

Note that the health-related variable was not included in the list of variables above, as it is very difficult to obtain robust data regarding a DWS's probability of infection from a pathogen, regardless if the analysis is conducted before or after the measure's installation. Pathogen concentrations tend to be difficult to measure in the intake water, as they are often well below the detection limit. Furthermore, the usage of indicator organisms, e.g. total coliforms, for the detection of pathogens is prone to produce both type I and type II statistical errors, although they may be useful to detect changes in water quality trends (Harwood et al., 2005; WHO, 2017c).

The QMRA model was, as such, instead constructed to assess the estimated load of pathogens to the raw water sources using data, e.g. incidents of infection, which are independent on whether the assessment was conducted pre- or post-instalment. Hence, the modelling of the UF membranes' effect on the DWS's consumers health-related wellbeing could not be improved by a post-instalment assessment. That being said, there are still a number of factors that need to be examined regarding the structure of the model and its most sensitive variables, as discussed below, section 9.2.

9.2 MODEL STRUCTURE

The discussion regarding uncertainties in the structure of the model have been divided into two parts. The first part, section 9.2.1**Error! Reference source not found.**, examines the u ncertainty generated from the variables with quantified uncertainties, i.e. the input variables with assigned distributions. Here, the three variables with the highest spearman rank, Figure 11, are investigated to see if it is possible to reduce their uncertainties further. The second part, section 9.2.2, discusses the possible impact of the unquantified uncertainties, e.g. assumed point-value variables.

9.2.1 Quantified Uncertainty

Based on the result of the CBA, it is not surprising to find that the variable with the highest correlation to the final result, is the UF membranes aesthetic benefit to the drinking water. However, there is also great incertitude regarding this variable. As discussed in the previous section, 9.1, better data regarding the actual improvement of the aesthetic benefit of the UF membranes could have reduced the uncertainty of this variable greatly. Yet, an actual WTP study of Varberg's drinking water consumers would have given an even more accurate value of this variable. However, due to the vast complexity and time-requirement for such a study (e.g.Gunatilake et al., 2007; Cameron et al., 2011), it was not included in this thesis.

The second variable, to consider improving, is the faecal concentration of pathogens given infection. More specifically, the faecal concentration of Norovirus. All investigated pathogens had great variations in their faecal concentration range. Although for Norovirus, the values of the investigated studies varied from the lower limits of 100 viruses per gram found by Teunis et al. (2015), to the upper limits of 1.6 10^{12} Norovirus per gram found by Atmar et al. (2008). In addition, by assuming a Log-normal distribution of the interquartile range found by Lee et al. (2007), 1.7 10^{6} – 1.7 10^{10} Norovirus per gram, the distribution's P99-value would be equal to 1.4 10^{15}. This latter study by Lee et al. was, however, excluded from the average concentration assumed in Table 3, due to these possibly extreme values. Albeit, it clearly shows the uncertainty of the variable. However, based on the findings of the different studies, one can safely assume that a major part of the faecal concentration's variation is due to aleatory uncertainty, rather than epistemic uncertainty. A realisation, which makes any further reductions of this variable's uncertainty very difficult.

Similarly, the third variable, i.e. the pathogens dose-responses, also seem to be based on aleatory uncertainty and natural variation rather than systematic errors. Any further reduction of uncertainties can therefore only be reached by the conduction of more specific studies for the local area of interest, i.e. areas within or close to Sweden. For example, further studies could be conducted to limit the dose-response to the most generic type of Norovirus in Sweden, or by only using Swedish inhabitants for the dose-response tests. However, based on the setup of the QMRA model, section 7.1, it is not certain that more specific data, for this variable, will improve the result in any significant manner. Due to the model's more general approach, e.g. the usage of three reference pathogens to represent their respective pathogen-category, there are already a number of underlying uncertainties which will not be reduced by more specific studies.

9.2.2 Unquantified Uncertainty

Apart from the quantified uncertainties in the input variables' assigned uncertainty distributions, discussed above, there are also a number of unquantified uncertainties in the model's underlying assumptions and single-value variables.

Perhaps, one of the most questionable assumptions of the model is the assumption of the certain value of the maintenance cost. It had been estimated to approximately 2 MSEK by VIVAB, post the instalment of the membranes, and it includes the DWTP's additional cost of staff, chemicals and electrical power, for the UF membrane operation. Based on this information, there are a number of arguments why the maintenance cost should not be considered as certain, e.g. the cost of any of these underlying variables may change drastically with time. However, the inclusion of such a variability would be, more or less, solely based on an arbitrary expectation of the change of price over time, e.g. ± 20 %. Furthermore, in comparison to other quantified and unquantified uncertainties of the model, the assessed maintenance cost appeared to be the model's most certain variable apart from the actual investment cost. It was as such not assigned any variability, as any factors which may change its future value will most likely also affect a number of other variables at the same time.

Another important aspect of the model is the representativeness of the assumption that all the people who may excrete pathogens to Stora Neden via the OWTSs, will do so at the same time. That is, that the number of expected people to fall sick from a reference pathogen within a certain year, based on that specific pathogen's geometric mean of incidents, will fall sick at the same time. For rural areas, e.g. the area within the catchment area of Stora Neden, this assumption, of simultaneous excretion, may be representative. Yet, for densely populated areas, this assumption could greatly overestimate the intake concentrations at the DWTP during a contamination event. For these type of instances it may, therefore, be better to assume a pathogen prevalence among the inhabitants connected to the sewage system, as conducted by Bergion et al. (2018). However, due to the small number of households, i.e. 50 houses, within the main catchment area. The expected number of people to fall sick from the reference pathogens each year was estimated to around 1 – 5 people per year for Campylobacter, 2 – 10 for Norovirus and 1 – 5 for cryptosporidium. Further, due to the pathogens infectivity, one can also expect that all the members of a household will fall sick within a brief time period after the first infection. Henceforth, by comparing the expected number of people who will excrete pathogens per year, against the number of people per household in Varberg, i.e. between 1 – 7 people per household (SCB, 2017). It is possible to assess that the representability of the original assumption, of a simultaneous excretion, is valid.

In addition to the previous argument, it also possible to call into question the representability of the pathogens' geometric mean of incident, for the whole municipality of Halland, for the small area of 50 households surrounding the lake Stora Neden. However, based on the underlying uncertainties regarding this variable e.g. the underreporting factor, it is considered the best available option for determining the likelihood of sickness within the lake's main catchment area. However, even if the geometric mean of incidence variable per say may be considered as reliable, there is still great uncertainty regarding its underreporting factor. As shown in Table 3, it can vary from a factor of 17 to 67, depending on the disease's severity (Mead et al., 1999; Lindqvist et al., 2001; Haagsma et al., 2013; Gibbons et al., 2014). Hence it is not unlikely that the selection of an underreporting factor may have a much larger impact than any variation in the geometric mean by itself.

Lastly, it should also be noted that the compartmental model used to estimate the reduction of pathogens in the lake, during the transport from source to intake, may be very over simplified and contains a lot of incertitude. The incertitude could have been reduced using hydrodynamic models, e.g. a finite difference model, although this was not within the framework of the thesis. Furthermore, the compartmental model was also assumed to be sufficient, as it is primarily the dilution which reduces the concentration of pathogens during their transport (Bergion et al., 2018); especially because of the lake's short run-time of six hours (Bertrand et al., 2012).

9.3 Implications of Results

The discussion regarding the implication of the results have been divided into two parts. The first part, section 9.3.1, discusses the result of the QMRA, while the second part, section 9.3.2, discusses the result of the CBA

9.3.1 QMRA

The mean annual P_{inf} for the assessed pathogens in the previous system, i.e. before the UF membranes installation, was calculated to 10^{-2}. It is a value which exceeded the acceptable risk of 10^{-4}, recommended by WHO (2016), by a factor of 100 and is equal to approximately 600 infections in a year. In addition, the annual probability of exceeding the acceptable risk was estimated to 42% and an annual probability of a major waterborne disease outbreak, infecting more than 10,000 people, was estimated to 1.5%. The probability of a major waterborne disease outbreak may at first be considered low in comparison to the high probability of exceeding the acceptable risk. Yet, the former's low probability, may be explained by the municipality of Varberg's stringent regulations and water protection areas, which greatly reduces the likelihood of any larger pathogen loads to the raw water sources. Whereas, the latter's, high probability, may be linked to the inadequate number of treatment steps, which makes it possible for even small pathogen loads to reach distribution network. Lastly it should be noted that even if no outbreaks have been recorded in Varberg; undetected outbreaks, below the threshold of infection, may still have occurred previously (Haas et al., 2014).

Since the UF membranes were installed in 2016, the mean annual P_{inf} has been reduced to 10^{-8} and the annual probability of exceeding a P_{inf} of 10^{-4} is currently less than 0.01%. It is, therefore, clear that the UF membranes has effectively safeguarded the DWTP's treatment efficiency.

Furthermore, based on the result illustrated in Figure 8, it is clear that the baseline risk is the main contributor to the total microbial risk and thus also the event which brings the most value to the UF membranes microbial risk reduction. Consequently, other less extreme events, than those covered by the base-line event, e.g. microbial contamination from wildlife or manure run-off, were not included, as they presumably would bring little contribution to the total risk.

9.3.2 CBA

The contribution of the gained QALYs to the UF membranes NPV may at first be considered low. However, as discussed above, section 9.3.1, the UF membranes will only generate health-related value, i.e. reduce the likelihood of infection, during more extreme conditions. In other words, the UF membranes will not generate any significant health benefits unless the raw water quality is deteriorated, which is quite unlikely due to the raw water sources implemented water protection areas. However, given a situation with high levels of pathogens in the water source, the UF membranes will have a completely different and better possibility to provide a safe drinking water compared to the previous treatment at Kvarnagården.

Even if the estimated value of a QALY did not dominate the final result of this study, its value in relation to other studies should be mentioned. The Swedish governmental revealed preference value, 0.7 – 1.2 MSEK, of a QALY (Svensson et al., 2015) may be lower than the actual WTP found by Söderqvist and Wallström (2017), of about 2 MSEK per QALY, but also the actual cost of a lost QALY. The Swedish Civil Contingencies Agency (MSB, 2014) report found that the total cost of the 2011 *Cryptosporidium* outbreak in Östersund[9] ranged between 141 to 221 MSEK. Further, by combining this cost with the findings of Batz et al. (2014)[10], the value of a lost QALY would range between 1.5 to 2.3 MSEK, or 3.5 to 5.5 times the 2015 Swedish GDP per capita, for a water borne disease outbreak. It is therefore possible that the value of a QALY may be valued higher in the drinking water sector than in the general domain, as the failure to distribute a sufficiently treated drinking water may have such extensive consequences. However, the societal

[9] 27,000 people were estimated to have been infected during the outbreak

[10] A *cryptosporidium* infection is estimated to decrease a person's health with 0.0035 QALY

value of the UF membranes is not solely linked to the health-related benefits. The NPV value of Kvarnagården DWTP's UF membranes is positive with a probability of 82 %, primarily due to its aesthetic improvements, whose benefits contributed a lot more than the health-related benefits

Furthermore, based on the result of the first scenario analysis, section 7.2.3, it is possible that the value of the non-monetised benefits is enough to cover all the UF membranes expenses. Depending on the discount rate, this varies from 2.4 MSEK per year at the set discount rate of 3.5 % to 4 MSEK per year for the more descriptive discount rate of 5.5 %. This required value of the non-monetised benefits may be put in perspective to the maintenance cost of the UF membranes at Kvarnagården, 2 MSEK per year, to get an estimate of what the improved water quality could be worth at the nuclear power plant, Ringhals. Furthermore, by taking the improved energy production into account as well, it is not unlikely that the non-monetised benefits may cover the NPV's minimum value.

However, it should be noted that for some cases, the inclusion of industrial benefits may raise some principal questions. Is it e.g. reasonable to include the benefits for the private sector in a solely municipal expenditure? The answer to this question will of course vary from case to case. Yet, for infrastructural projects, using a general societal perspective, the author would say it is a valid inclusion, although distributional analyses should be conducted to see whom stands to gain or pay for the benefits. Notwithstanding, the distribution of clean water does increase many companies' willingness to invest in a region. Further, as this is likely to generate an increased tax revenue, the author does think that the inclusion industrial benefits should be considered valid.

9.4 OUTLOOK

The major problems for Kvarnagårdens DWTP in the future, will most likely be linked to the rising levels of NOM in the lake Stora Neden. Keucken (2017) has stated that the current UF membranes will not be able to handle these issues past the year 2040, at the DWTP's current groundwater ratio.

Based on these findings, two treatment methods are suggested to be investigated further at the DWTP. The first method involves the possibility to exchange the current UF membranes to NF membranes. Primarily as it has been proven to function properly, for the removal of NOM, at the Görvälnverket DWTP by Keucken (2017), but also by other authors, as illustrated in Table 6. Second, since NF may be a too expensive investment, an alternative treatment method could be the utilization of activated carbon filters. These have proven to function well for the removal of NOM, as illustrated in Table 6, and can be expected to have a lower investment and maintenance cost, than the NF membranes. In addition, the activated carbon filters would also mediate the consequences of a petroleum spillage, in comparison to the NF membranes, where organic pollutants with EC < 10 would still pass through membranes.

This is, however, a too complex issue to be solely determined via literature studies and pilot studies are recommended to determine the best option for Kvarnagårdens DWTP, to handle the rising NOM-levels.

10 CONCLUSION

The upgraded treatment chain at Kvarnagården's DWTP has been a sound investment from both a health-related and an economic perspective. The mean NPV for the monetised benefits and costs was assessed to +47 (P5 = -45; P95 = +117) MSEK. From the results and the scenario analyses it was also clear that:

- The current risk of a waterborne disease outbreak, due to an insufficiently treated drinking water, can be considered as negligible.
- The aesthetic benefits appeared to be significantly higher than the health-related benefits; clearly showing the importance of including this factor in future economic assessments of drinking water.
- There is an 82 % probability of obtaining a NPV > 0 SEK, indicating that the installation of the UF membranes was a beneficial investment
  - The value of the non-monetised benefits has the possibility to ensure a NPV > 0 SEK if their cumulative value exceed 2.4 MSEK per year.
- Additional measures may be required in the future to ensure a distribution of high-quality drinking water and primarily mediate the rising levels of NOM in the DWTP's raw water sources.

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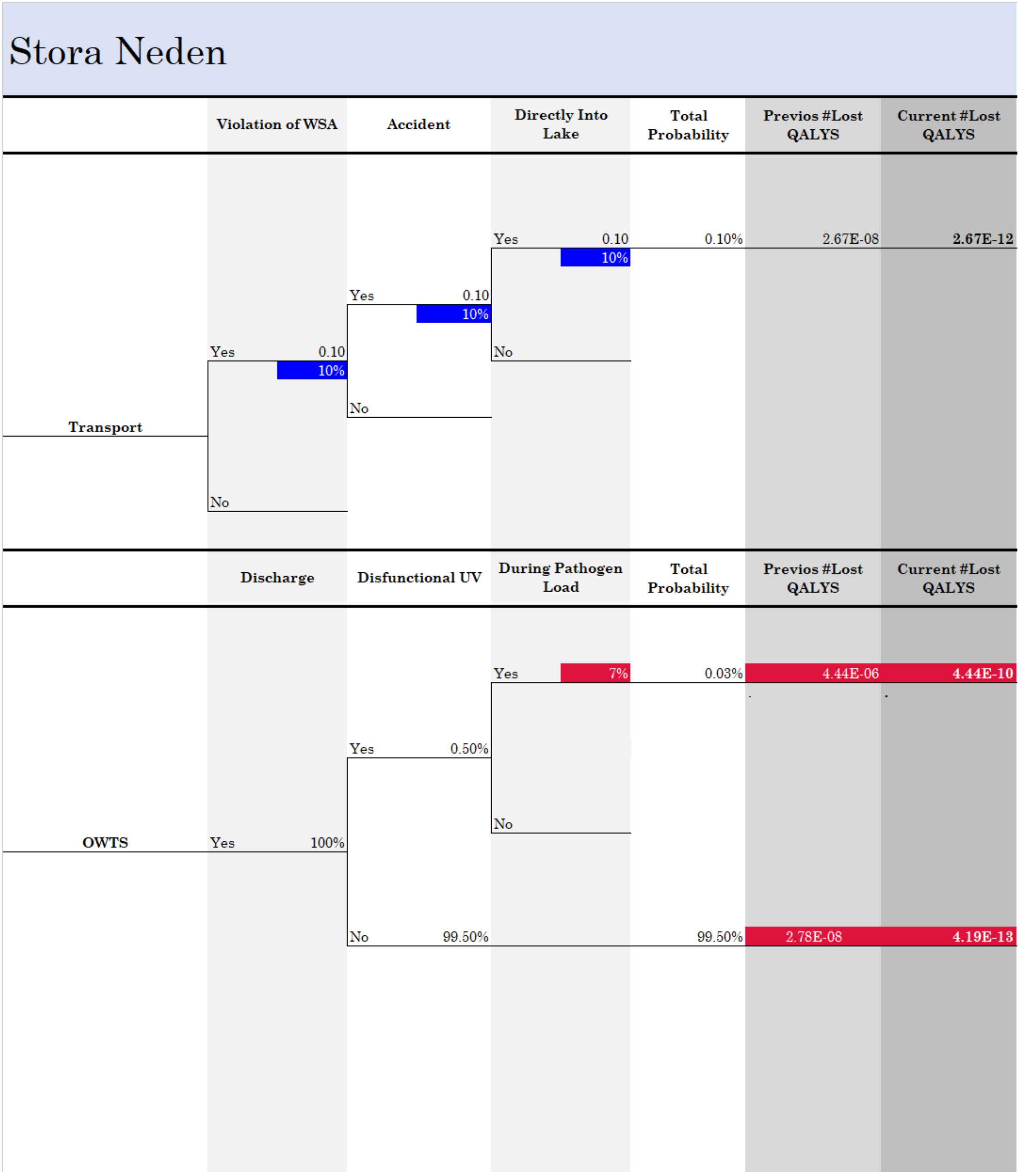
APPENDIX A

Event tree analysis for the different hazardous events connected to microbial pollution of Kvarnagårdens DWTP. Blue cells indicate mean value of a beta-distribution, with P20 of 0.8 om the mean and P80 of 1.2 of the mean. The probability of a dysfunctional UV barrier during a pathogen load was calculated by dividing the expected number of days that the raw water source would be exposed to a pathogenic load per year by number of days in a year.



APPENDIX B

Event tree analysis for the different hazardous events connected to microbial pollution of Kvarnagårdens DWTP. Blue cells indicate mean value of a beta-distribution, with P20 of 0.8 om the mean and P80 of 1.2 of the mean. Note that probability of pipe-failure is also divided by ten. The probability of a pipe-burst during a pathogen load was calculated by dividing the expected number of days that the OWTS would be exposed to a pathogenic load per year by number of days in a year and multiplying it with probability of a households being infected in a given year.