THESIS FOR THE DEGREE OF LICENTIATE OF ENGINEERING

Development of a Risk-Based Decision Model for Prioritizing Microbial Risk Mitigation Measures in Drinking Water Systems

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ABSTRACT

Risk management of drinking water systems is crucial since our society relies on these systems to be robust and sustainable to supply safe drinking water now and to future generations. Pathogens may spread in drinking water systems and cause waterborne outbreaks resulting in human suffering and large costs to the society. Thus, mitigating microbial risks is of great importance for provision of safe drinking water in a changing world. Since risk mitigation measures can be costly, there is a need for a transparent and holistic decision support to enable a sound and efficient use of available resources. In this thesis, a risk-based decision model that facilitates evaluation and comparison of microbial risk mitigation measures is presented. The model was developed by combining source characterisation, water quality modelling, quantitative microbial risk assessment and cost-benefit analysis. Uncertainties associated with input variables and output results were analysed by means of Monte Carlo simulations. The decision model puts emphasis on health benefits obtained from reduced microbial risks in drinking water systems and the monetisation of these effects. In addition, the approach also accounts for non-health benefits that occur because of implemented mitigation measures. Such benefits, also if they cannot be monetised, are important to include and carefully consider in the cost-benefit analysis. The probabilistic approach provides an analysis of uncertainties that need to be considered by decision makers. To conclude, this thesis underlines and illustrates the strength of combining methods from several disciplines to create a robust decision support in order to optimise societal benefits.

Keywords: decision support, water quality modelling, quantitative microbial risk assessment, cost-benefit analysis, drinking water system, pathogens, microbial risks

LIST OF PAPERS

This thesis includes the following papers, referred to by Roman numerals:

I. Bergion, V., Sokolova, E., Åström, J., Lindhe, A., Sörén, K. and Rosén, L. (2017). Hydrological modelling in a drinking water catchment as a means of evaluation pathogen risk reduction. Published in *Journal of Hydrology* **544**: 74-85.

II. Bergion, V., Lindhe, A., Rosén, L. and Sokolova, E. (2017). Combining risk assessment and cost-benefit analysis for evaluating microbial risk mitigation measures in a drinking water system. Manuscript.

Division of work between authors

In Paper I, Bergion, Sokolova and Åström were part of designing the hydrological model. Bergion created the model, performed all simulations and was the main author. Bergion, Rosén and Lindhe developed the risk framework. Åström and Sörén provided substantial inputs regarding scenario design and development.

In Paper II, Bergion, Lindhe and Rosén developed and designed the decision model. Bergion created the model, performed all calculations and was the main author. Sokolova performed the hydrodynamic modelling. Bergion, Lindhe and Rosén formulated the decision problem.

Other work and publications not appended

The author has contributed significantly to the following publications, which are not appended to the thesis (note that the author surname was Johansson before 11th July 2015):

- Åström J. and Johansson V. (2015) *GIS-based dispersion modelling of parasites in surface water sources* (in Swedish), Report 2015-07, Swedish water and Wastewater Association, Stockholm (In Swedish: *GIS-baserad spridningsmodellering av parasiter i ytvattentäkter*).

- Johansson V. and Sokolova E. (2015) *Modelling fate and transport of Escherichia Coli and Cryptosporidium spp. Using Soil and Water Assessment Tool*, In E-proceedings of the *36th* IAHR World Congress*,* The Hague, 28th June-3rd July, p1162-1169

- Johansson V., Rosén L., Lindhe A., Sokolova E, Åasröm J. and Lång, L.-O. (2015). *A decision support framework for managing microbial risks in groundwater supply systems* (*Abstract*), Oral presentation at the International Association of Hydrogeologists *42th* IAH Congress, Rome, 13-18 September.

- Bergion V., Rosén L., Sokolova E., Lindhe A., Lång, L.-O. and Sörén, K. (2016). *Comparison of mitigation measures for microbial risk reduction using cost-benefit analysis for decision support* (*Abstract*), Poster at the Nordic Drinking Water Conference, Reykjavík, 28-30 September.

- Bergion V., Rosén L., Lindhe A. and Sokolova E. (2016). *Combining Quantitative Microbial Risk Assessment and Disability Adjusted Life Years to Estimate Microbial Reduction for Cost-Benefit* Analysis (*Abstract*) Poster at the Society for Risk Analysis Annual Meeting, San Diego, 11-15 December.

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Finally, my wife Ellinor. I would not have reached this far without you. I love you!

Gothenburg, March 2017

Viktor Bergion

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LIST OF NOTATIONS

The following notations are used in the main text of the thesis:

CBA	Cost-Benefit Analysis
CEA	Cost-Effectiveness Analysis
DALY	Disability Adjusted Life Years
DWS	Drinking Water System
DWTP	Drinking Water Treatment Plant
IEC	International Electrotechnical Commission
ISO	International Organization for Standardisation
Log10	Logarithmic reduction, in this thesis reduction of pathogens, where 1 Log10 reduction = 90 % reduction, 2 Log10 reduction = 99 % reduction, etc.
MCDA	Multi-Criteria Decision Analysis
NPV	Net Present Value
OWTS	On-site Wastewater Treatment System
QALY	Quality Adjusted Life Years
QMRA	Quantitative Microbial Risk Assessment
Reduction	The term reduction incorporates all processes, e.g. removal, inactivation adsorptions, predation etc., that in some way lowers the amount of pathogens.
WHO	World Health Organization
WWTP	WasteWater Treatment Plant

1 INTRODUCTION

In this chapter the relationship between drinking water systems, health risks and possible costs for the society are described and introduced. After that the aim and scope of the thesis are presented.

1.1 Drinking water, human health and societal profitability

Potable water is essential to human health and life. However, despite halving the proportion of the world population without access to safe drinking water and basic sanitation by 2015 (i.e. reaching United Nations (UN) millennium development goal 7C), there are still over half a billion people using unimproved[1] drinking water sources (United Nations 2015). Looking ahead, the UN have adopted 17 sustainable development goals to be achieved by 2030, of which several are related to drinking water and human health (United Nations 2016). The lion's share of the work related to these goals is expected to take place in regions where managed drinking water systems (DWS) do not exist and where the water resources are exposed to hazardous and unregulated sources of pollution. Even so, to achieve these goals in a world where the climate is changing and populations are growing, require substantial efforts to manage the already existing supply systems. It is crucial to ensure that these DWS can provide the societies with reliable and safe drinking water. Risk management, including the work of estimating and evaluating risk levels as well as analysing and implementing risk mitigation measures, is a key element in securing a safe and sustainable drinking water supply for future generations.

The availability of fresh water sources is dependent on the hydrological cycle. The fundamental processes of the hydrological cycle are being affected by anthropogenic activities such as cloud seeding (Viessman et al. 2014) and activities related to climate change (Oki and Kanae 2006). Climate change and associated increase in temperature, change in precipitation patterns and in some areas increasing flood events and prolonged periods of drought will have a negative effect on the water quality and quantity (Delpla et al. 2009, Coffey et al. 2014). To ensure future water quality, assessment and adaptation to possible climate change scenarios need to be incorporated into drinking water management and into related legislation (Coffey et al. 2014).

People with access to water supply systems use them at least as frequently as other public infrastructure service, such as roads, railroads and electricity. In Sweden, as well as many other industrialized countries, constant availability and good quality of potable water distributed through drinking water supply systems is many times taken for granted. Uncritical use and reliance on technical systems is often an inadequate approach. DWS do provide a life sustaining infrastructure service, but if they fail, they can rapidly change into facilitators of waterborne

[1] Unprotected spring/dug well, small tank, tanker truck, untreated surface- and bottled water (WHO/UNICEF 2017)

diseases. Therefore, risk management of these DWS is even more essential for reducing health risks to drinking water consumers.

Waterborne outbreaks of gastrointestinal diseases and their relation to DWS have been documented throughout the history (e.g. International Water Association 2016). One much noticed and re-echoed event was the linkage between cholera outbreaks and specific drinking water wells in Soho, London, made by John Snow in the mid-19th century (The John Snow Society 2016). Even nowadays, seemingly functional DWS fail, resulting in waterborne disease. The most known and largest waterborne outbreak in more modern times occurred in Milwaukee, US in 1993, where the pathogen *Cryptosporidium* affected over 400,000 people (Mac Kenzie et al. 1994). Sweden have experienced a number of waterborne outbreaks of gastrointestinal diseases the last decades (Guzman-Herrador et al. 2015). Östersund, Sweden in 2010, with 27,000 people affected, was the largest documented waterborne outbreak in Europe (Widerström et al. 2014).

The outbreaks in Milwaukee and Östersund both resulted in substantial costs for the society. Medical treatment costs and costs due to loss of production were estimated to be SEK 778[2] million ($96.2 million) for the Milwaukee outbreak (Corso et al. 2003). The corresponding costs for the Östersund outbreak was estimated to be SEK 220 million (approximately $33.8 million[3]), including also the estimated personal cost of experiencing gastrointestinal disease (Lindberg et al. 2011).

Microbial risks posed by pathogens in DWS are always present and will continue to be present in the future. To mitigate these risks and to maintain drinking water of high quality, implementation of risk management and associated risk mitigation measures are of fundamental importance. Setting health-based drinking water quality targets should acknowledge the local conditions (social, cultural, environmental and economic) and also include institutional, technical and financial aspects (WHO 2011). Societal resources are limited and should be distributed in a fair and reasonable manner, and when allocated they need to be used effectively. Two economic decision models commonly used for evaluating risk mitigation measures and create decision support are cost-effectiveness analysis (CEA) and cost-benefit analysis (CBA) (Cameron et al. 2011). In relation to risk management, CEA can be exemplified as “How to reach a certain goal at the lowest cost”. A CBA compares all internal and external costs and benefits in order to find the most societally profitable alternative. CBA could in a similar way be described as “How to find the societally most profitable alternative looking at costs and benefits?”

Given that microbial risk mitigation measures in drinking water systems in most cases also result in non-health benefits (Hutton 2001), e.g. environmental and social, there is a need to adopt a broad approach in order to also encompass these benefits. Performing a CBA is one way of achieving this holistic decision support, emphasising the health benefits without

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[2] Converted from USD using yearly average (2003), $1= 8.09 SEK (SR 2017)
[3] Converted from USD using yearly average (2011) $1= 6.50 SEK (SR 2017)

neglecting other benefits. Quantitative microbial risk assessment (QMRA), described below in section 3.3, can provide robust input to CBA regarding the health benefits obtained by microbial risk mitigation measures (WHO 2016). To use a probabilistic quantitative microbial risk-based approach in combination with CBA to create decision support for risk management in DWS are uncommon, nevertheless, it is emphasised by the World Health Organization (WHO) (Fewtrell and Bartram 2001).

1.2 Aim and objectives

The overall aim is to develop a risk-based decision model for comparison of microbial risk mitigation measures in drinking water systems using risk assessment in combination with cost-benefit analysis to create decision support. Specific objectives are to:

- set up a framework for risk-based decision support;
- compare microbial risk mitigation measures using water quality modelling;
- combine source characterisation, water quality modelling, quantitative microbial risk assessment and cost-benefit analysis to create a risk-based decision model;
- consider uncertainties in the input data and results and describe how these are included and their effects on the coupled decision model outcomes.

1.3 Scope

The scope of the thesis is to describe the quantitative risk-based decision model for microbial risk reduction in DWS on an overarching level. Detailed information on components and methods in the risk-based decision model are described and exemplified in Paper II. An in-depth description of hydrological water quality modelling is presented in Paper I.

2 BACKGROUND

In this chapter an introduction to the risk concept and terminology is presented. Drinking water systems, microbial risks in drinking water systems and health metrics are described. The relation between drinking water systems, risk management and decision support are introduced.

2.1 Introduction to the risk concept

Over centuries and between different cultures, the perception of uncertainties and the risk concept have changed and varied. In contrast to early civilizations, where uncertainties related to e.g. natural disasters, crop yields, plagues and wars often were attributed to divinity, the modern society and the rapid development of human-controlled technical systems introduced a number of mathematical tools to express uncertainties and the associated risk (Zachmann 2014). The definition of risk by Kaplan and Garrick (1981) touches upon the relation between risk and uncertainties. However, uncertainties as a term was not applied fully at that time, but introduced later (e.g. Aven 2010, Aven 2012b). Aven (2012b) also gives an overview of the development of the risk concept and definitions. The risk definition can be and have been expressed in different ways. In the latest ISO 31000 standard, risk is defined as an *effect of uncertainties on objectives* (ISO 2009a). However, in this thesis, risk is defined using the concept of *probabilities* and *consequences*. This risk concept highlights the importance of continuous and structured risk management. In section 2.5 risks posed to DWS are described and in section 2.6 the structure of risk management in DWS are explained.

2.2 Risk terminology

Given a rapid increase in the use and in the diversity of fields in which risk management has been practiced during the last two decades, the terminology has been to some extent scattered and inconsistent (Leitch 2010). In the food industry, risk analysis is commonly used as an overarching term including the entire process of estimating identifying hazards, estimating risk levels, considering whether they are acceptable or not, analysing measure for risk mitigation and implementing necessary measure (EFSA 2012, Haas et al. 2014). In more technical systems, and the approach applied in this thesis, the term risk management is commonly used to describe the same overall process (IEC 1995). The former approach is generally used by organisations that need to separate the work, and responsible parties, of determine risk levels from the decision on risk treatment. However, regardless of the framework used, the included steps and procedures are very similar and the major differences are mere lingual. In this chapter, the risk terminology and definitions used in this thesis are explained. The decision problems considered in this thesis is are to a large part managed by the drinking water procedures and the municipalities. In contrast to organisation were the distinction between decision making and prior estimation of risk levels etc. are crucial, drinking water producers and the specific municipality are commonly responsible for the entire procedure in Sweden. Therefore, risk management is used here to describe the overall process and to illustrate the basic concept and the link to decision-making. The framework and definitions by ISO/IEC standards are used (ISO 2009a, ISO/IEC 2009). Focus of the thesis lies mainly within risk assessment (Figure 1). *Risk assessment* is the term providing a common ground for most risk management frameworks, consisting of risk identification, estimations of probabilities and consequences of identified risks and risk evaluation.

The ISO/IEC standards divide the risk concept applied to organisations, roughly into three stages, *principals*, *framework* and *process*. The first stage is the initiation step where the organisation decides to embrace the risk management principles. The second stage is where the organisation commits to risk management and gives mandate to allocate the necessary resources to implement, review and continuously improve and update the risk management framework. The third stage is the process of actually creating the risk management framework and performing risk assessment adopting an iterative approach. The risk management process will be described in detail in section 2.6 in the context of drinking water. Figure 1 shows an illustration of a general risk management process and the included steps and related terms are briefly defined below.

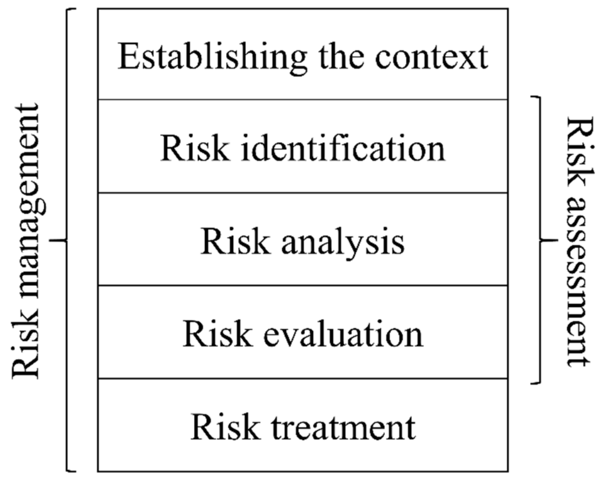


Figure 1. Risk management process, adopted from ISO (2009a)

Establishing the context or scope definition is the initial part of the risk management process. It relates to variables/factors inside or outside the organisation that are included or of importance in the risk management. It also defines the risk criterion(a) that is/are the yardstick(s) that the risk evaluation will use.

Risk assessment is the overarching term combining risk identification, risk analysis and risk evaluation.

Risk identification is the process of finding and describing risk sources, hazardous events and their consequences.

Risk analysis is the understanding of identified risks related to the magnitude (risk level) of their probabilities and consequences.

Risk evaluation compares the risk level with stated risk criteria. Tolerability and acceptability of the risk level is determined.

Risk treatment is the action taken in relation to the risk evaluation. Typically risks are responded to by: avoiding; removing; sharing; changing the probabilities and/or consequences; taking the risk to gain opportunities; and/or by taking the risk by an informed decision.

Risk(s) is/are described as the combination of probability and consequence of a hazardous event. Focus of this thesis is on risks related to the probability and consequences of water contaminated with pathogens. When risks are identified they are commonly described as a source of potential harm, a *hazard* or a *risk source*. In a DWS, hazards can be e.g. an on-site wastewater treatment system (OWTS) or a wastewater treatment plant (WWTP). A related *hazardous event* is an event where a hazard or risk source causes a consequence, e.g. wastewater is transported to a drinking water treatment plant (DWTP) and pathogen(s) present in the wastewater causes diseases to drinking water consumers. *Likelihood* is the possibility of an event to occur that can be described in general terms or using mathematical metrics such as probability. *Probability* expresses the likelihood of an event occurring using a number between 0-1, where 0 is being impossible to occur and 1 being certain to occur. *Uncertainties* (see section 2.3) are important to describe the probability of an event to happen and hence also the resulting consequences.

2.3 Uncertainties

Uncertainties are usually attributed either to natural variations in (aleatory), or to lack of knowledge of (epistemic), a system (Bedford and Cooke 2001). As described in the section 2.4, drinking water systems are complex, typically exhibiting both aleatory and epistemic uncertainties. Aleatory uncertainties, e.g. the variability of precipitation in a catchment or the presence of pathogens in a river, can be measured and statistically quantified in order to get a better understanding of the variability (NHMRC 2011). Epistemic uncertainties, e.g. lack of knowledge regarding statistical parameters describing variability, are often quantified using expert opinions (Bedford and Cooke 2001) and can be reduced by investigations. The difference between aleatory and epistemic uncertainties is not clear cut, and in a risk analysis, both types of uncertainties can be quantified using probability as a metric. However, looking at uncertainties from a decision making point of view, making the distinction between uncertainties that can be reduced (epistemic) and those that cannot (aleatory), can be of importance (Bedford and Cooke 2001). In some context the ambiguity and vagueness in language or vocabulary that is being used, can be described as a third type of (linguistic) uncertainty (Beven 2010). Frequentist methods, strictly put, are used for investigating hard data in order to derive at a point estimate for input variables only accounting for uncertainties possibly by providing a confidence interval (Bedford and Cooke 2001). A Bayesian approach adopts subjective (expert)judgements in order to establish probability distributions to describe the input variables and its uncertainties (Aven 2012a). On a practical level the difference between frequentist and Bayesian methods does not need to be substantial (Aven 2012a). However, one major theoretical difference is that frequentists aim to estimate an objective probability while the Bayesian assumes that all probabilities are subjective. The Bayesian methodology also facilitates updating of model variables as new data becomes available. In practice, the frequentist and Bayesian approaches are often mixed (Aven 2012a). In this thesis a Bayesian approach is adopted, to facilitate the inclusion of subjective estimations of statistical parameter values and associated uncertainties, based on professional judgements.

2.4 Drinking water systems

Drinking water systems or drinking water supply systems are generally divided into three parts: source water(s), DWTP(s) and distribution system(s) (Hokstad et al. 2009, Lindhe 2010) and can be extended also including a fourth part, the drinking water consumers (NHMRC 2011). The source water part consists of both the catchment area and the actual drinking water source. Catchment area is the geographical unit receiving precipitation that is transported and discharged at the catchment outlet (Soliman 1997). The terms watersheds, drainage basin and catchment area, despite small technical discrepancies, are considered synonymous; in this thesis catchment or catchment area is used as the general term. Water sources can be surface-, ground-, reclaimed waste-, storm-, brackish- and saline water (Viessman et al. 2014). Groundwater sources can also be enhanced using artificial infiltration and induced recharge. DWTPs extract raw water from the source water and divert it through a series of treatment processes, producing drinking water that is provided to consumers using a distribution system.

Meteorological conditions, soil properties etc., set the scene on what water sources that are available and can be used. Combinations of different types of water sources, multiple DWTPs and/or several separated distribution systems contribute to the diversity of DWS. In Sweden, approximately half of the produced drinking water volume originates from groundwater and the other half from surface water. In general, surface water sources are supplying DWS that have a large number of consumers, while those using groundwater sources supply a smaller number.

Sources of microbial contamination that can be introduced into the DWS are commonly described to be present in the catchment and in the distribution system. However, microbial risk mitigation measures can focus on either reducing the risk at the contamination sources or mitigation measures can be applied in the DWTS aiming at reducing the final risk posed to drinking water consumers using barriers in the treatment.

2.5 Microbial risks in drinking water systems

Microbial risks in drinking water are typically described to be pathogens present in the DWS. It can be illustrated from a water utility point of view using the risk definition earlier. What is the *probability* that drinking water consumers will be infected by pathogens spread through the DWS and what are the *consequences*, i.e. how many will be infected and what type of infection is considered or what are the economic consequences for society due to the infections. Water utilities cannot be absolutely certain that the pathogen concentration in the drinking water is zero when delivered to the consumers. Therefore, there is always a risk, even if the pathogen concentration is very low.

We can characterise waterborne pathogens differently, the most common way is to distinguish between bacteria, viruses, protozoans and helminths/trematodes. Looking at the origin of these pathogens, it can also be of importance to identify if they can be transferred only between humans or if it is possible to transfer between animals and humans (zoonotic diseases). In Table 1, some of the most common waterborne pathogens are listed, including an indication of relevant animal hosts.

Table 1. List of some common waterborne pathogens, adopted from WHO (2011) and Dufour et al. (2012)

Pathogen	**Potential animal hosts identified**
Bacteria:	
Campylobacter jejuni	Cattle, swine, poultry, dogs cats wild birds
Escherichia coli O157:H7	Cattle and other ruminants
Salmonella enterica (not S. Typhi*)*	Poultry, swine, cattle, horses, dogs, cats and wildlife
Viruses:	
norovirus	Potentially
rotavirus	None
adenovirus	None
Protozoans:	
Cryptosporidium spp.	*C. parvum*[4] can be found in cattle, and other animals
Giardia duodenalis	Cattle, beavers, porcupines, dogs and other animals

These pathogens originate predominantly from faecal sources, both animal and human. In a typical drinking water catchment, the faecal sources are; human wastewater, from OWTSs and municipal WWTPs; domestic animals, from grazing, using manure as a fertilizer and leakage from manure storage facilities; and wild animals.

Waterborne pathogens that are not related to faecal sources, e.g. Legionella, that can be present in natural waters, microbial risks related to other factors (e.g. biofilm in distribution pipes), physical drinking water quality variables (taste, odour, etc.) are not in the scope of this thesis.

[4] Other species of *Cryptosporidium* associated with various animals have been found to infect humans

2.6 Risk management of drinking water systems

In this section, the risk management process is explained in relation to DWS. The purpose is to describe the different steps of risk management in relation to the applications to DWS that are used in this thesis.

2.6.1 Establishing the context

In drinking water management, establishing the context in general terms consists of two items. First the purpose of the risk analysis and the possible decision problems are described. Second the system is described, including system boundaries, catchment area (sources of pollution), source water (characterization of source), measures for resource and source protection, water treatment system, monitoring system and distribution (also including reservoirs, internal piping, consumers and water authorities) (Hokstad et al. 2009, WHO 2011).

2.6.2 Risk identification

There are large number of different microbial risks that can be present in a DWS. Performing a risk identification is the process of identifying these underlying hazards or hazardous events. Table 2 lists some of the microbial hazards/hazardous events that might be present in DWS.

Table 2. List of possible microbial risks in drinking water systems, adaption from Rosén et al. (2007) and Beuken et al. (2008)

In the catchment
Discharge of treated wastewater
Sewage overflows
Manure application
Runoff from agriculture and urban areas
Wild animals
Accident with vehicles containing faecal waste tanks
In the drinking water treatment plant
Failure in treatment technology affecting microbial barriers
Ineffective reduction of pathogens in microbial barriers
Erroneous operation procedures
In the distribution system
Intrusion of pathogens to reservoirs and pipes
Cross connections with wastewater pipes

2.6.3 Risk analysis

Risk analysis of microbial risks can be performed using qualitative, semi-quantitative and/or quantitative methods. A strictly qualitative risk analysis lists the possible hazards and hazardous events and categorises the probabilities and consequences in a descriptive manner. Semi-quantitative risk analysis extends the categories to be numerical. In a quantitative risk analysis, both probabilities and consequences attributed to each hazardous event are described using values that can be combined to calculate the risk. The risk is thus seen as a combination of the probability and consequences of relevant hazardous events. In a mathematical context, the probability density function of a hazardous event, f_i, is combined with a consequence function representing the consequences of that event, C_i. The risk (R_i) related to a hazardous event (i) is then calculated as:

$$R_i = \int C_i f_i ds$$

In order to rank risks in drinking water settings, both semi-quantitative and quantitative methods are suggested (NHMRC 2011, WHO 2011). Semi-quantitative methods are commonly illustrated using risk matrixes to illustrate the ranked categories (Hokstad et al. 2009, Lindhe 2010, NHMRC 2011, WHO 2011). Quantitative risk analysis of microbial risks are commonly performed using the Quantitative Microbial Risk Assessment (QMRA) (Haas et al. 2014).

A four-step procedure for QMRA in water contexts has been suggested (WHO 2016). The steps are problem formulation, exposure assessment, health effects assessment and risk characterisation. A fifth, unifying, risk treatment (management) step can be combined with the four initial QMRA steps (Haas et al. 2014). First, presence of waterborne pathogens in the drinking water system is identified and formulated into a problem. It is possible to specify risk mitigation measures in this stage to be included later in the risk treatment. Second, the present pathogens (hazards) and their routes of exposures (hazardous events), including possible barriers in the system, are identified and estimated. Third, the estimated pathogen concentration in the drinking water, the drinking water consumption rate and dose-response relations are combined in order to estimate the health effects in the population. Finally, the risks are characterised through combining the exposure assessment (probability of infection) and the health effect assessment (consequences) to calculate the risk level[5]. The fifth step of risk treatment, further discussed in section 2.6.4 below, relates to risk acceptability criteria (RAC), tolerable risk and measures for risk mitigation.

Health metrics

In QMRA, *probability of infection* and *Disability Adjusted Life Years* (DALYs) are two health metrics commonly used (WHO 2016). These two are also used in the Swedish QMRA-model developed for drinking water producers (Abrahamsson et al. 2009, Åström et al. 2016).

Probability of infection refers directly to the dose-response relation of each specific pathogen. Based on controlled infection studies, e.g. for *Cryptosporidium* (DuPont et al. 1995) and norovirus (Teunis et al. 2008), the probability that a person will be infected given a certain dose is estimated. Infectious dose varies due to variations in infectivity between and within pathogen species as well as individual susceptibility in the population (WHO 2016). However, for practical reasons a population dose response relation is commonly used.

[5] In drinking water applications the probability of infection is sometimes used to describe the risk and describing the health consequences are sometimes omitted from the analysis.

DALY is a health metric that combines Years of Life Lost (YLL) (mortality) and the Years Lived with Disability (YLD) (morbidity) and is a well-established metric used by the WHO to estimate the burden of disease (WHO 2001). *Quality Adjusted Life Years* (QALYs) is a third health metric combining mortality and morbidity that is available. In contrast to DALYs the weights used in QALYs are based on quality of life estimates instead of disability weights (Sassi 2006). In its simplest forms QALY can be described as the inverse of a DALY. However, the relation is a bit more complicated due to that different elicitation methods are commonly used for establishing quality of life weights for QALYs and disability weights for DALYs and that DALYs are often calculated using age weighting functions that are not used in QALYs (Sassi 2006). If no age weights are used in the DALY calculation or if age weights are used in QALY calculations the inverse relationship gets even closer (Robberstad 2009). The concept and relationship between DALY and QALY is illustrated in Figure 2.

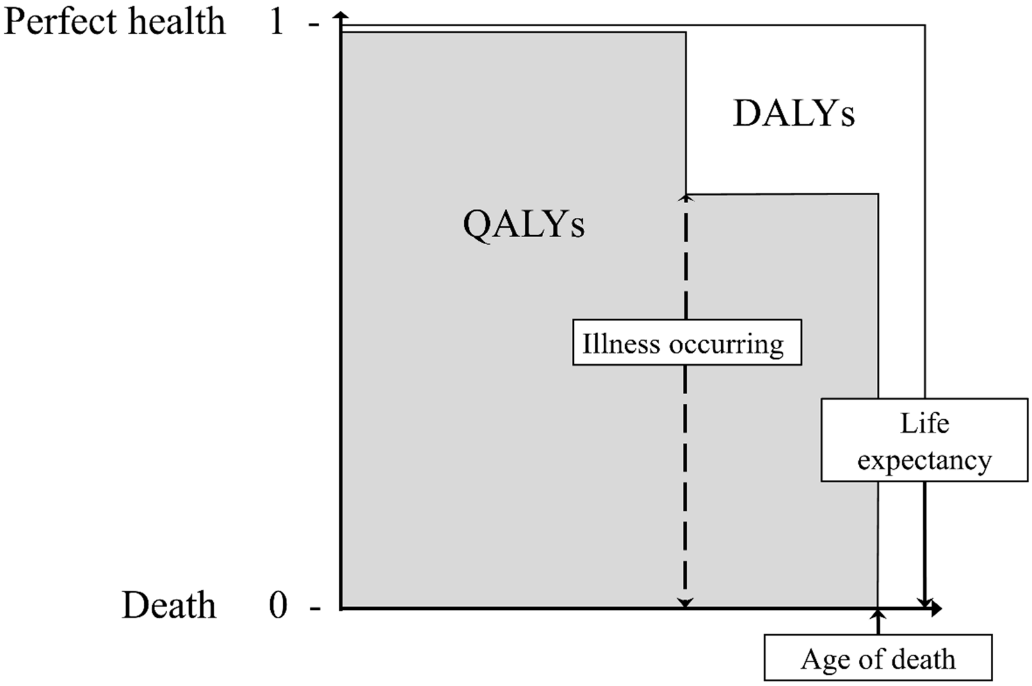


Figure 2. Conceptual relation between DALYs and QALYs is illustrated. White area represents the DALYs and the grey area represents the QALYs experienced during a lifetime. Adopted from Robberstad (2009).

2.6.4 Risk evaluation

Tolerability

The initial task for risk managers is, based on the risk analysis, to perform a risk evaluation to see if there is a need for risk treatment. Decision to implement risk mitigation measures is normally initiated based on comparison to the risk criteria defined in the *establishing the context* phase. A DWS could be found to have negligible risks[6], i.e. a satisfactory system. DWS could also be evaluated to have risks that are acceptable/tolerable, risks that are unacceptable, and/or risks that need to be evaluated on their tolerability. A system with only acceptable/tolerable risks could also be seen as satisfactory. Satisfactory systems can stay in their present state and be handled with the principle of monitoring and continuous improvement according to the risk

[6] One could argue that there are systems with no risks, however it is most unlikely and not considered.

management framework. If unacceptable risks or risks that are not tolerable are present, the system is unsatisfactory and measures need to be taken. Tolerability/acceptability could also be based on changes in legislation/policies and changes in the risk perceptions of various stakeholders.

The as low as reasonably possible (ALARP) principle divides risks into three different categories: acceptable, unacceptable and those in the ALARP region (HSE 1992). Risks in the unacceptable category need to be dealt with no matter what the costs or other efforts necessary to reduce them, the acceptable category can be handled within the everyday routines. The risks falling within the ALARP region need to be assessed in each case. Variables other than consequences and likelihoods, such as cost (Melchers 2001), time and physical difficulty reducing the risk, can be taken into account when adopting the ALARP approach (HSE 1992).

The risk acceptability criteria defines what risk levels that can be accepted (Rosén et al. 2010). In a drinking water context, it is also referred to as the “tolerable burden of disease” and “reference level of risk” (WHO 2011). Acceptable risks are below the RAC and the risks above the RAC are either unacceptable or in the ALARP region (Melchers 2001). Risks above the RAC need either to be treated (i.e. reduced) or to be tolerated. Different approaches on how to define acceptable or tolerable risk levels are discussed in e.g. Hunter and Fewtrell (2001) for the context of water related infectious diseases and Rosén et al. (2010) for the context of managing drinking water supply systems as a whole.

(WHO 2011) promotes a health-based approach for estimating RAC, incorporating financial, technical and institutional resources and the local situation regarding economic, environmental, epidemiological, social and cultural aspects. Setting health-based targets should adopt a holistic approach reflecting that drinking water is only one of many routes for exposure of contaminants or pathogens (WHO 2011). Health-based targets can be measured in health outcome, water quality, performance targets or specified technology targets. To set local risk tolerability levels, a DALY of 10^{-6} could be used as a point of departure (WHO 2011). In Sweden there are no health based RAC for drinking water.

Identification of risk mitigation measures

Corresponding to each identified hazardous event, there can be none, one or several measures for reducing the risk. One measure can affect more than one hazardous event (Lindhe A. et al. 2013). Measures can remove the risk source, alter the uncertainties of the hazardous event, alter the consequences of the hazardous event and/or divide the risk between several parties (ISO 2009a). Measures for risk mitigation need to be identified and characterised. Each alternative can consist of one or a combination of several measures (ISO 2009a). There is also the *reference alternative* keeping the system in its present state as well as the alternative to take an informed decision to accept a change in the system that will result in a higher risk level.

(WHO 2011) advocates the principle of *multiple-barriers* to create a resilient system, supporting that several measures could be implemented in different stages of the drinking water system. In case one or several of the barriers are failing, there are other that have the possibility to compensate for this. The measures can be hands-on, implementing best available technologies (BAT) or new technological application, they can be newly developed methods or established methods transferred from other drinking water systems (Niewersch and Burgess 2010). Education, training, communication, information, legislation and research are other examples of drinking water system upgrades (Åström and Pettersson 2010, WHO 2011). Identification of possible measures needs to be adapted for each individual DWS, although there are suggestions on available risk mitigation measures (Åström and Pettersson 2010, Ball et al. 2010, Menaia et al. 2010, Niewersch and Burgess 2010, NZMH 2014). There is scarce information on methods or suggestions in the literature on how to identify new or how to optimise local tailor-made measures. To identify measures, drinking water managers and experts should be involved, and it is also beneficial to include multi-, trans- and cross-disciplinary competences and to communicate with stakeholders and people with knowledge of the particular DWS (Rosén et al. 2010).

Decision analysis

When selecting which alternative(s) to implement, there are different decision support systems, decision rules and decision models available. Cost-effectiveness analysis (CEA), as mentioned in the introduction, is used to identify the alternative that achieve an objective to the lowest cost. In a CEA benefits do not need to be expressed in monetary units, rather they are investigated in relation to reaching a certain risk level.

CBA evaluates if measures are societal profitable and compares costs and benefits of each measure. The principle of Cost-Benefit Analysis (CBA) has been used for centuries although the terms of costs and benefits were introduced in the early 20th century (Persky 2001). CBA has been used within a wide range of fields, such as environmental policies, infrastructure projects, soil remediation, and company investment strategies. Terms such as benefit-cost analysis, policy evaluation, project appraisal and socio-economic analysis are more or less synonymous to CBA (Atkinson 2008). If cost and benefits are estimated from a societal point of view, instead of a personal or company perspective, it can sometimes be referred to as a social CBA (SCBA)[7] (Boardman 2011). However the term CBA will be used as an umbrella term in this thesis, though emphasis is to have a societal point of view.

Costs and benefits that occur when implementing risk mitigation measures in drinking water systems can be divided into health benefits/costs and non-health benefits/costs (Moore et al. 2010). Investments, operational, capital, maintenance, additional and external costs, e.g. due to negative effects on human health and ecosystem services can be described as cost categories. Reduction in operation cost, reduction in capital expenditure, improvements in water supply service levels, improved aesthetic qualities public goodwill, external benefits, e.g. due to

[7] Swedish Environmental Protection Agency SEPA (2008d) describe SCBA as it *identifies and quantifies all consequences a measures has on different groups in the society*. Socio-economic consequences are described as positive (socio-economic benefits) and negative (socio-economic costs) consequences. Monetised and non-monetised consequences should be included in a SCBA (SEPA 2008c), and preferably a rough estimation of the non-monetised consequences should be performed (SIKA 2005).

improved health, increased provision of ecosystem services and social benefits can be described as benefit categories (Baffoe-Bonnie et al. 2008).

Expressing the monetary values of costs and benefits is not always straight forward. If there are market prices for costs or benefits these prices could be used as monetary values. Values from non-market goods can be categorised as both use (direct use, indirect use and option values) and non-use (existence, bequest and altruistic values) values. When non-market goods, such as environmental or health benefits, are monetised a so called shadow price is commonly used. The shadow price is a price that should reflect the non-market goods value and can be estimated using various methods. Stated preferences and revealed preferences are different concepts for estimating a shadow price. Stated preferences, e.g. contingent valuation methods and choice experiment methods investigate people's preferences when choosing between hypothetical alternatives. Revealed preferences seeks so find a surrogate metric for valuing a non-market good. Revealed preferences incorporates different methods such as hedonic price method, travel cost method and cost of illness. A detailed review of economic valuation for water resource management can be found in Birol et al. (2006).

Multi-criteria decision analysis (MCDA) is a method that can cope with complex decision problems. MCDA can help prioritise the available measures evaluating appropriate criteria, without converting these criteria into monetary units.

2.6.5 Risk treatment

Risk treatment is, as the risk management work as a whole, a continuous iterative process of deciding upon appropriate measures for mitigating the risk, and thereafter assessing whether the residual risk is tolerable or not. If the residual risk is not tolerable, further measures need to be implemented until the risk can be tolerated (ISO 2009b). Implementing measures for risk mitigation in DWS can be a substantial investment, and the discussions and decisions should be made with a holistic perspective with respect to risk as well as, for example, economic conditions, implementation time and the ability to monitor the effects (WHO 2011). The decision analysis provide vital input in the form of decision support do aid decision makers.

Monitoring

Monitoring and review are essential for a sustainable risk management. Changes in policies, objectives, goals or stakeholder preferences and/or risk perceptions need to be monitored. These changes can be triggered by various actors such as pressure groups, research, media, politicians, etc. Physical changes in the DWS (both long term and acute) altering the pathogen prevalence situation, pollution sources, transport routes, treatment process, the distribution system and/or the consumer susceptibility for infections are also variables that could be monitored. These changes in DWS could be within (internal) or outside (external) the risk managers control. Pursuing opportunities related to research, investments and collaboration will most certain render a need for a risk assessment or a review of the already existing one.

2.7 Risk management in relation to a decision making process

A schematic illustration of the decision making process is displayed in Figure 3 (Rosén et al. 2010, Aven 2012a). The stakeholder values, goals, criteria and preferences initiate a decision process. First, the decision problem is identified and formulated and different decision alternatives are developed. Second, risk and decision analyses are performed characterising the decision alternatives. Third, the managers review the decision alternatives by comparing results from the risk and decision analyses. Finally, the decision makers conclude upon a decision. Commonly, the decision makers are identified in the initial step of the decision making process. A decision making process in relation to CBA (Baffoe-Bonnie et al. 2008, SEPA 2008a, Aven 2012a) and in relation to risk management (Rosén et al. 2010) have also been described. The risk assessment provide essential input to the risk- and decision analysis, connecting the risk management framework (Figure 1) to the decision making process described in Figure 3 (Rosén et al. 2010, Aven 2012a).

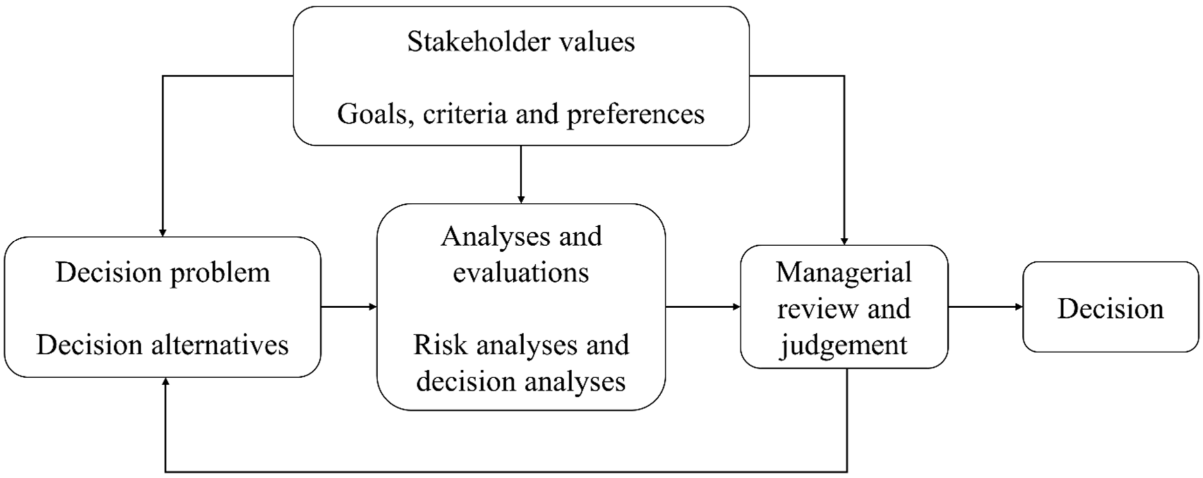


Figure 3. Decision making process, adopted from (Aven 2012a)

3 METHODS

In this Chapter, the specific methods used to establish the decision model is presented. Hence, the methods constitute a necessary toolbox focusing on source characterisation, water quality modelling, microbial risk assessment, cost benefit analysis and how to consider uncertainties. In chapter 4, a possible chain of methods is suggested to encompass the decision model (adopted from Paper II). An appropriate combination of methods needs to be adapted reflecting the local settings in the DWS.

3.1 Source characterisation

There are several both qualitative and quantitative methods, e.g. pathogen sampling, epidemiologically based methods etc. that can be used for source characterisation. In this section, methodology for quantification of pathogen sources based on prevalence is described.

The description divides the pathogen sources into OWTSs, WWTPs and animals sources. Three factors governs the pathogen source: population size, the prevalence of the disease in the population and the concentration of pathogens in faecal matter. The method is applied for each pathogen that is to be included in the risk assessment. In the QMRA methodology implemented in the Swedish QMRA-tool, sometimes three reference pathogens are adopted to represent protozoan, bacterial and viral pathogens.

The prevalence of pathogens in the human population is calculated[8] as:

$$P_{human} = \frac{I \cdot U \cdot D}{365 \cdot 100{,}000} \qquad (1)$$

where P_{human} is the prevalence, I is the incidence (per year/100,000 inhabitants), U is the factor of underreporting, and D is the number of days when excretion occurs during infection. Incidence was expressed by a gamma distribution adopted from incidence data between 2006 and 2016 reported by the Public Health Agency of Sweden (PHAS 2017). The number of infections that are reported included in the incidence represents only a fraction of the actual infections present in the population. Underreporting is illustrated (Figure 4) in the form of a report pyramid (Haas et al. 2014).

[8] In Paper I, a factor accounting for asymptomatic infection (A) was also included: $P_{human} = \frac{I \cdot U}{365 \cdot 100{,}000(1-\mathrm{A})}$

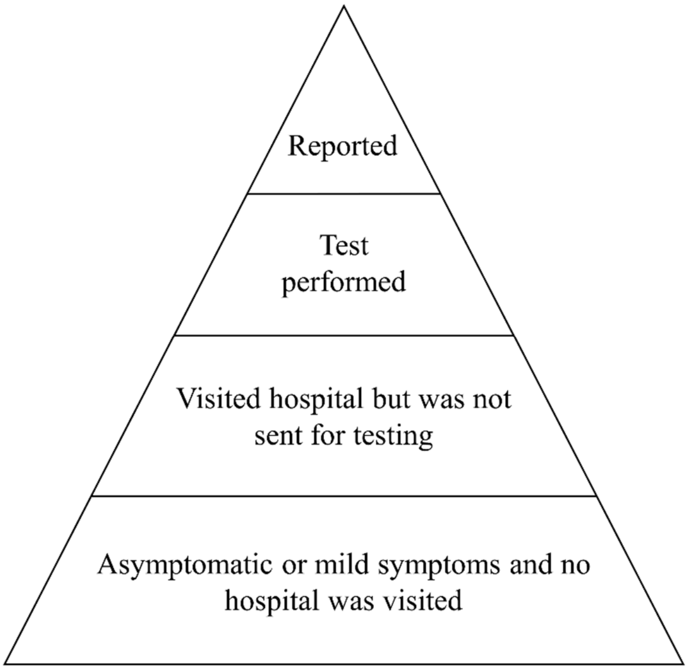


Figure 4. Illustration of the clinical report pyramid, adapted from Haas et al. (2014).

The pathogen concentration[9] in the OWTSs and the WWTPs discharge is calculated as:

$$C_{path} = \frac{P_{human} \cdot F \cdot C_{human}}{W \cdot 10^{R}} \qquad (2)$$

where C_{path} is the pathogen concentration in wastewater per day either from OWTSs or WWTPs, P_{human} is the prevalence in humans, F is the faecal production per person per day, C_{human} is the pathogen concentration in faeces from infected individuals, W is wastewater production per person and day and R is the Log10 reduction of pathogens in OWTSs or WWTPs, respectively. The term *reduction* incorporates all processes, e.g. removal, inactivation adsorptions, predation etc. that in some way lowers the amount of pathogens.

For animal sources, similar calculations can be executed using prevalence as base. Site-specific data are to be preferred when estimating the pathogen load, however, there are few studies that actually quantify pathogens in animal faeces (Dufour et al. 2012). Extensive reviews of animal pathogen prevalence (e.g. Ferguson et al. 2009, Dufour et al. 2012) cannot serve as a starting point when adopting local estimates. However, they can be used in order to develop and evaluate models or methods. Equation 3 can be used for calculating the pathogen concentrations in manure either applied during grazing or via manure application as fertilizer.

$$\mathrm{C}_m = \frac{\sum_{i=1}^{n} \mathrm{E}_i \cdot \mathrm{P}_i \cdot \mathrm{V}_i \cdot \mathrm{N}_i \cdot \mathrm{T}_i}{\sum_{i=1}^{n} \mathrm{V}_i \cdot \mathrm{N}_i \cdot \mathrm{T}_i} \qquad (3)$$

[9] For OWTSs, the source characterisation was slightly adjusted in Paper I to be adapted to the SWAT model

Variable C_m is the mean pathogen concentration in manure, E_i is the pathogen excretion rate in infected animals, P_i is the prevalence, V_i is the manure production per day, N_i is the number of animals in the area, T_i is the number of days for manure accumulation each year and i represents different domestic animal categories ($i=1 \ldots n$). Depending on local legislation, routines and procedures, the annual manure load from grazing animals and from applying manure as fertiliser needs to be distributed accordingly. In Paper I, further details on the calculations of animal faecal contribution can be found.

3.2 Water quality modelling

Below, three different approaches to water quality modelling are presented. Each modelling approach represents a method that can be used to investigate the fate and transport of pathogens in water.

Factors important to the fate of pathogens are water/osmotic pressure, temperature, pH, solar radiation, and nutrients (inorganic and organic) (Ferguson et al. 2003). Transport of pathogens in catchments is affected mainly by adsorption/desorption to particles and hydrologic-, mechanical- and biologic movement (Ferguson et al. 2003). Transport in groundwater systems are highly affected by pathogen adsorption to particles, hence the pathogen attributes adsorption and pH are of great importance (Åström et al. 2016). Hydrological surface water modelling investigates transport from sources on land to and within the river course. Hydrodynamic surface water modelling estimates transport in water bodies, both within rivers and lakes. Groundwater modelling, using analytical or numerical models, investigates reduction of pathogen during groundwater transport. If the several types of models are combined, they can describe the transport of pathogens, from both point and non-point sources on land and in water, to the drinking water intake. Both hydrological modelling (Oliver et al. 2016, Bergion et al. 2017) hydrodynamic modelling (Sokolova et al. 2015) and groundwater modelling (Pang 2009) can aid in microbial risk assessment of drinking water systems.

Hydrological modelling

Hydrological modelling of pathogen fate and transport can be performed using various models (Dorner et al. 2006) and can be helpful in analysing microbial risks for water quality management (Coffey et al. 2010a). The Soil and Water Assessment Tool (SWAT) was ranked highest on the performance of microbial contamination modelling (Coffey and Cummins 2007). The SWAT model has been used to assess the fate and transport of various microbial contaminants, e.g. faecal coliforms (Parajuli et al. 2009, Cho et al. 2012), *E. coli* (Coffey et al. 2010a, Kim et al. 2010, Bougeard et al. 2011) and *Cryptosporidium* (Coffey et al. 2010b, Tang et al. 2011, Jayakody et al. 2014, Bergion et al. 2017). SWAT is a deterministic semi-distributed process-based hydrological model describing the hydrological cycle and the water transport in catchments (Nietsch et al. 2011). A sub-model for pathogen loading is incorporated and is coupled to the hydrological cycle (Sadeghi and Arnold 2002). The SWAT model is based on geographic information system (GIS) and can be combined with ArcGIS (Winchell et al. 2013) and QGIS (Dile et al. 2016) interfaces. In Paper I the SWAT model was used to estimate the pathogen reduction in different microbial risk mitigation scenarios, adopting Stäket catchment as a case study.

Hydrodynamic modelling
Hydrodynamic modelling can provide information on fate and transport of pathogens within water bodies. In Paper II, hydrodynamic modelling was performed using the MIKE Powered by DHI MIKE 3 FM model. This model solves three-dimensional incompressible Reynolds averaged Navier-Stokes equations invoking the assumptions of Boussinesq and of hydrostatic pressure (DHI 2011).

Groundwater modelling
To estimate the pathogen inactivation during groundwater transport, groundwater transport and inactivation models can be used. In Paper II a groundwater virus transport model was implemented to represent the pathogen reduction occurring in artificial infiltration. Moreover, the methodology can be applied to natural groundwater systems as well. The model has been incorporated into the Swedish QMRA-tool (Åström et al. 2016) and is based on reduction from dilution, attachment and inactivation (Schijven et al. 2006, Pang 2009).

3.3 Quantitative microbial risk assessment

The Quantitative Microbial Risk Assessment (QMRA) is a widely used methodology adopted for quantifying the health effects of the microbial risk mitigation measures. In this section the application of QMRA in drinking water and the QMRA-tool developed for Swedish drinking water producers is presented.

The methodology is based on the relation between certain levels of exposure (i.e. pathogen dose) and health effects. The daily dose is calculated as:

$$D = C_{DW} \cdot V \qquad (4)$$

where D was the daily pathogen dose from drinking water, C_{DW} was the pathogen concentration in drinking water and V was the volume of ingested drinking water per person per day. The C_{DW} was estimated from the water quality model output and the Log10 removal in DWTP barriers. Probability density functions for pathogen concentration are used.

The volume is calculated (Equation 5) using a log-normal distribution (Westrell et al. 2006):

$$V = exp^{Normal\,(\mu,\sigma)} \qquad (5)$$

where Normal (μ, σ) was a normal distribution (μ = -0.299 and σ = 0.57). A dose-response function, e.g. Exact Beta Poisson, Exponential, etc. is assigned to each pathogen. To illustrate an example, the Exact Beta-Poisson function is shown in Equation 6.

$$P_{inf} = 1 - exp^{-r \cdot D} \qquad (6)$$

where P_{inf} is the daily probability of infection, r is a sample from a beta distribution with statistical parameters set for each pathogen and D is the simulated daily pathogen dose that was ingested. As an example parameters for r , i.e. the Beta(α,β) distribution, norovirus has been described to have (α=0.04, β=0.055) (Teunis et al. 2008).

The annual probability of infection is calculated using Equation 7.

$$P_{Annual} = 1 - \left(1 - P_{inf}\right)^{365} \quad (7)$$

Where P_{annual} is the annual probability of infection per person. The P_{annual} is calculated using bootstrap technique, where each iteration of the annual probability of infections forms the base for a probability density function.

Separate probabilities of infection for each pathogen can be added[10] to estimate the total probability of infection from an arbitrary pathogen. An example the use of three reference pathogens as described in section 3.1 and adopted in Paper II is calculated as:

$$P_{Total\_inf} = 1 - \left(1 - P_{Annual\_virus}\right) \cdot \left(1 - P_{Annual\_bact}\right) \cdot \left(1 - P_{Annual\_prot}\right) \quad (8)$$

where P_{Total_inf} is the total annual probability of infection, P_{Annual_virus} , P_{Annual_bact} and P_{Annual_prot} was the annual probabilities of infection due to the reference virus, bacteria and protozoa, respectively.

P_{annual} for each pathogen can be converted into QALYs lost using a simple method (Equation 9) adopting literature values based on a US study of QALYs lost per infection. As an example, in Paper II the number of QALYs lost per infection was assumed to be 0.0035, 0.0163 and 0.0009 for *Cryptosporidium, Campylobacter* and norovirus, respectively (Batz et al. 2014).

$$QALY_{Year} = I \cdot Q \qquad (9)$$

Where $QALY_{Year}$ is the QALY lost per infection, I is the number of infections in the drinking water consumers per year and Q is the number of QALYs lost per infection. The QALYs from each pathogen is added to estimate the total sum of QALYs.

[10] This implies that the different events are independent. Since pathogen often originates from faecal contamination one could argue that the presence of one pathogen could increase the probability for the presence of another, resulting in a positive correlation that has not been accounted for.

P_{annual} can also be converted into *DALYs* per person and year using Equations 10, 11 and 12 (WHO 2001).

$$DALYs = YLL + YLD \qquad (10)$$

where *YLL* is calculated as:

$$YLL = \sum_i e \cdot (a_i) \sum_j d_{ij} \qquad (11)$$

and *YLD* as:

$$YLD = \sum_j N_j \cdot L_j \cdot W_j \qquad (12)$$

where *i* is the index of different age classes, *j* is the index for different disabilities, $e{\cdot}(a_i)$ is the average life expectancy for that age category, d_{ij} is the number of fatalities for each age category for respective disability, *N* is the number of cases, *L* is the length of the disability and *W* the disability weight to represent the severity of the disease. The DALYs from each pathogen is added to estimate the total sum of DALYs.

3.4 Cost-benefit analysis

If the decision is bound to render costs and benefits over several years (time horizon), the costs and benefits from each year are added together using an appropriate discount rate. The Net Present Value (*NPV*) of a certain measure is comparing the costs and benefits discounted into a present value (Baffoe-Bonnie et al. 2008). Note that the terminal value, i.e. the costs and/or benefits that will occur after the studied time horizon, can be included as a benefit in the last year of the time horizon. In a CBA the (*NPV*) of each measure is calculated (Equation 13) in order to compare decision alternatives.

$$NPV = \sum_{t=0}^{T} \frac{(B_t)}{(1+r)^t} - \sum_{t=0}^{T} \frac{(C_t)}{(1+r)^t} \qquad (13)$$

The variable *T* is the time horizon[11], *B* is the benefits during year *t*, *C* is the costs during year *t*, and *r* is the discount rate.

Benefits can be split up in to arbitrary constituents depending on application. It can be useful to have one constituent aggregating the non-monetised benefits. To provide an example, in Paper II the benefits were estimated as:

[11] The time horizon of a CBA is usually the expected life time of the implemented measure, although if costs and/or benefits are likely to occur far into the future, a longer time horizon could be considered (Baffoe-Bonnie et al. 2008). To identify the time horizon, investigations of when the influence of the discount rate renders all costs and benefits as insignificant can be helpful (Cameron et al. 2011).

$$B_T = B_H + B_E + B_O \qquad (14)$$

where B_T was the total benefits, B_H was the benefits estimated from reduced negative health effects in drinking water consumers, B_E was the benefits from increased treatment efficiency of nutrients and B_O was other benefits. In the application presented in Paper II B_O was not monetize, while the B_H and B_E were monetised. Health benefits, when monetising a QALY, are calculated as:

$$B_H = Q_{Red} \cdot V_{QALY} \qquad (15)$$

where Q_{red} is the amount of reduction in QALYs achieved from the mitigation measure and V_{QALY} is the value of a QALY. The value of a QALY can be based on estimates from literature investigating the willingness to pay for a QALY. In Paper II the V_{QALY} was estimated using willingness to pay for a QALY in related to reimbursement for pharmaceuticals (Svensson et al. 2015). V_{QALY} was assigned both a high (SEK 1,220,000) and low (SEK 700,000) value in Paper II.

Environmental benefits when using a simplified approach are calculated as:

$$B_E = N_{red} \cdot C_N + P_{red} \cdot C_P \qquad (16)$$

where N_{red} and P_{red} is the expected reduction (kg) in nitrogen and phosphorous discharge due to each measure and C_N and C_P is the value of the cost for discharging one kg nitrogen or phosphorous to the recipient, respectively. In Paper II N_{red} and P_{red} was based on increased nutrient reduction in WWTPs in comparison to OWTSs and C_N and C_P were based on literature estimates (SEPA 2008b).

B_O is generally difficult to monetised using quantitative measures. However, to illustrate the importance of these benefits, an analysis of how large they need to be in order to produce a positive *NPV* is included as a part of the decision support.

Costs can be derived from e.g. literature, previous implementation of measures, obtained from relevant stakeholders etc. and are estimated for each measure. In Paper II costs were based on estimates from literature based on previous investments (Kärrman et al. 2012) and information from relevant stakeholders.

3.5 Uncertainties

Uncertainty analysis comprises of the estimation of uncertainties of input variables and investigations of how the estimated uncertainties of input variables affect the output of a model. Using a Monte Carlo simulation approach, multiple iterations (e.g. 10,000) are conducted, sampling from the input probability distribution. The output will be a probability distribution incorporating the probability distributions of the input.

Sensitivity analysis investigates how changes in different input variables affect the output. Local sensitivity analysis can be calculated using Equation 17 as suggested by Burgman (2005). It presents the sensitivity as the percent change in output value due to the percent change in one input value at a time. This does not give any information regarding the uncertainty of the results, only on the results sensitivity to different input variables.

$$Sensitivity = \frac{\Delta V}{\Delta I} \cdot \frac{I}{V} \qquad (17)$$

where ΔV is the change in output value, V is the original output value, ΔI is the change in input variable and IP is the original input variable value.

In Paper I, the hydrological SWAT-model provides limited possibilities regarding uncertainty analysis due to its deterministic approach. A local sensitivity analysis of the SWAT-model altering input variables showed that hydrological variables related to the runoff, the plant available water in soil and soil evaporation processes were the most influential on the river water flow. The water transport and flow govern the transport of pathogens.

Uncertainties in the hydrodynamic modelling were estimated based on variations in the calculated log10 reduction. The model was used to simulate a long period of time (5 years) in order to consider variations in the meteorological and hydrological conditions that determine the spread of pathogens from the source to the drinking water intake.

The QMRA-tool incorporates uncertainty features for Monte Carlo simulations. The input values of the log10 reduction in the DWTP barriers were assigned using probability density functions.

Uncertainty analysis was performed using the Spearman's rank correlation. Equation 18 reflects the contribution of each input variable uncertainty to the output uncertainty and is calculated as the Spearman's correlation coefficient.

$$\rho = 1 - \frac{\left(6 \cdot \sum d_i^2\right)}{n\left(n^2 - 1\right)} \qquad (18)$$

where ρ is the correlation coefficient, d is the rank difference between the input and output and n is the number of correlation sets. A ρ close to 1 shows a high importance and ρ close to 0 shows low importance.

All uncertainties are not suitable to model using probability distributions. Instead different scenarios can be used investigating variations in these variables. In Paper II, two levels of proportion of the OWTSs contribution to the pathogen load (50% and 75%), discount rate (1% and 3.5%), and the value of a QALY (700,000 and 1,220,000 SEK) were investigated in different scenarios.

4 RESULTS

In this chapter a short review of the appended papers is presented showing the key findings. After the specific results from the papers are presented and key aspects related to the risk-based decision model are highlighted.

4.1 Paper I

In Paper I, a risk management framework was developed to describe the relation between risk management and CBA as a decision support. The role of hydrological modelling in the risk management framework was also described. Pathogen reduction in different microbial risk mitigation measures was estimated using the hydrological model SWAT. Stäket catchment north-west of Stockholm was used as a case study. Key results showed how microbial risk reduction can be quantified using hydrological modelling and that the combination of water quality modelling, QMRA and CBA constitute vital methods for creating decision support.

In more detail, fate and transport modelling of *Cryptosporidium* and the indicator bacteria *E. coli* was performed for the Stäket catchment to analyse four mitigation scenarios (M1-M4). Scenarios M1 and M2 simulated a 50 m vegetative filter strip in connection to cropland and grazing areas, respectively. In scenario M3, all underperforming OWTSs were restored and assigned a microbial reduction of 2 Log10 units. In scenario M4, microbial reduction by the WWTPs was increased by one Log10 unit. Results showed that M2 and M3 did not result in a significant reduction of *Cryptosporidium* or *E. coli,* but M1 and M4 did. The magnitude of microbial reduction differed between sub-basins. For scenario M1, the reduction in different sub-basins ranged from 0 to 0.41 and from 0 to 0.46 Log10 reduction for *Cryptosporidium* spp. and *E. coli*, respectively. For scenario M4, the reduction ranged between 0 and 1 Log10 reduction for both *Cryptosporidium* spp. and *E. coli*. Looking at the catchment outflow, M4 resulted in the highest microbial reduction.

4.2 Paper II

In Paper II, the *risk assessment and decision analysis* part of the risk management framework was described in detail. Source characterisation, hydrodynamic modelling, QMRA and CBA were described in a risk management and drinking water context. This set of methods was combined to create a decision support model in order to compare microbial risk mitigation measures. Uncertainties in input data and results were considered using Monte Carlo simulations. Lake Vomb served as a case study to illustrate the risk-based decision model.

Four decision alternatives (A1-A4) for microbial risk mitigation were investigated. Three alternatives (A1-A3) connected OWTSs in the catchment area (25, 50 and 75 %, respectively) to the municipal WWTP. In A4, a UV-treatment was installed in the DWTP. None of the alternatives resulted in a positive *NPV*, based on the included costs and benefits. Nevertheless, the analysis showed that if non-monetised benefits would reach SEK 800-1200 per connected OWTS and year, the 50th percentile of the *NPV* would be positive looking at a 1% discount rate. Alternative A1 (25 % of OWTSs connected to the WWTP) achieved the highest *NPV* closely followed by A4 (UV-treatment in the DWTP). However, comparing the microbial risk in terms of infection probability to the WHO guidelines, only A4 would reduce the risk enough when looking at the 95th percentile. The derived decision support illustrated the importance of the distributional and sensitivity analyses, in particular when looking at the scenario-based sensitivity analysis. Investigating such variables as discount rate, assumptions on OWTSs contribution to total pathogen load, the monetisation of health effects, and how large non-monetised benefits need to be in order to achieve a positive *NPV*, provide valuable information for decision makers.

4.3 Risk management framework

The risk management framework presented in Paper I is shown in Figure 5. The framework illustrates the decision making process in relation to the risk management framework (see section 2.2), as presented by IEC/ISO (IEC 1995, ISO 2009a), and the role of CBA in this context. The framework should be seen as a point of departure for comparing microbial risk mitigation measures in DWS using CBA as a decision model.

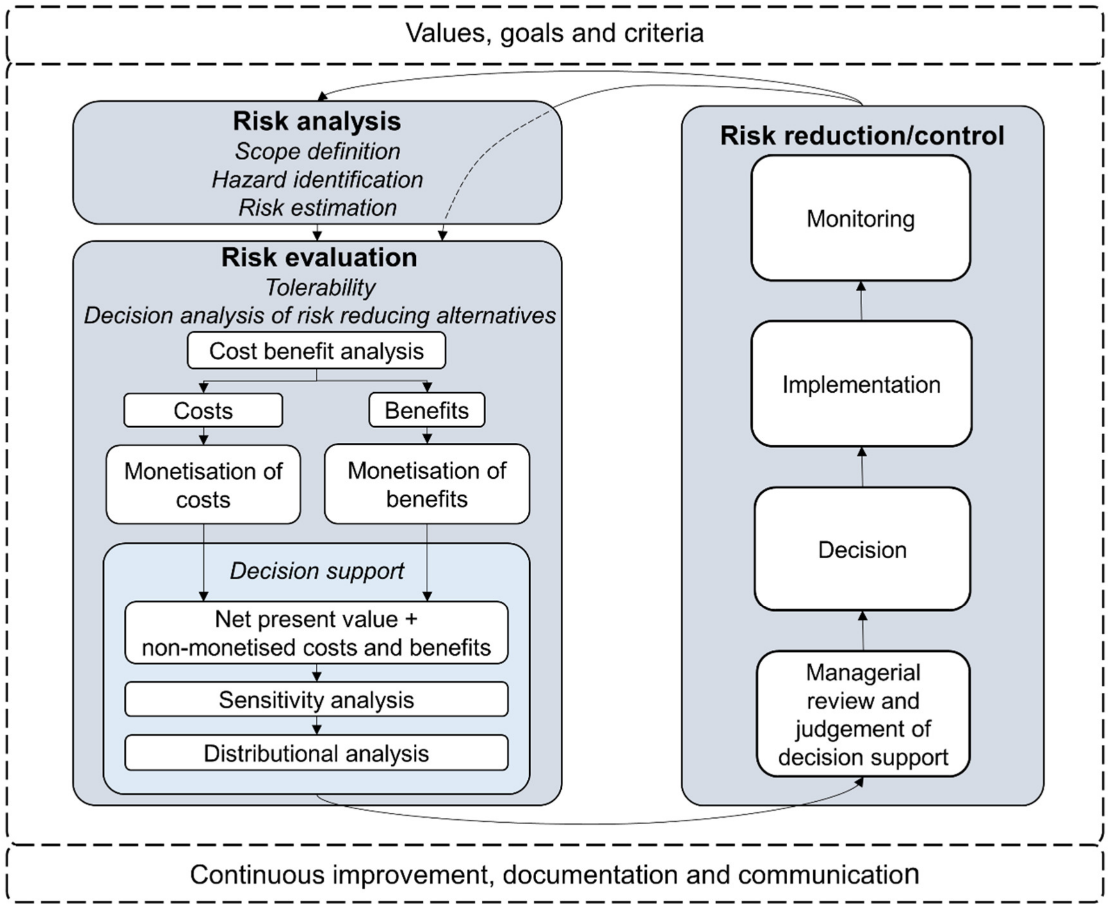


Figure 5. The role of CBA in risk management and decision making. Risk management framework from Paper I.

The preconditions for a dynamic and transparent framework are the values, goals and criteria set by various stakeholders, as well as continuous improvement, documentation and communication. All these criteria form the foundation of the framework.

The main compartments in the framework are risk analysis, risk evaluation and risk reduction/control. Hydrological modelling is a useful method in the risk analysis, performing risk estimation looking pathogen transport. In Paper I, risk reduction mitigation measures implemented in the catchment were compared using hydrological modelling; and the risk management framework was used to illustrate the role of hydrological modelling in risk assessment. Water quality modelling and QMRA were highlighted as useful methods in terms of estimating health effects of risk mitigation measures. The risk management framework forms the basis for the developed decision model and illustrates the applied risk approach

4.4 Risk-based decision model for microbial risk mitigation in drinking water systems

In order to link risk assessment with decision analysis and provide decision support, different methods need to be combined. Here, an overview of how to combine and link different methods is presented in accordance with the focus of this thesis (Figure 6). The description is partly generic, but it is devised based on the case study of Lake Vomb. Each individual method is described more in detail in Chapter 3 and commented below. The set of methods may vary depending on the specific application, i.e. preconditions, specific hazards and other aspects. Combining several methods to describe various parts of the DWS, from the source characterisation all the way to the health effects on the drinking water consumers provides a clear structure. It also gives the opportunity to tailor each compartment using the best available method for that particular application. To establish the risk reduction, the chain of methods needs to be applied to each risk mitigation measure.

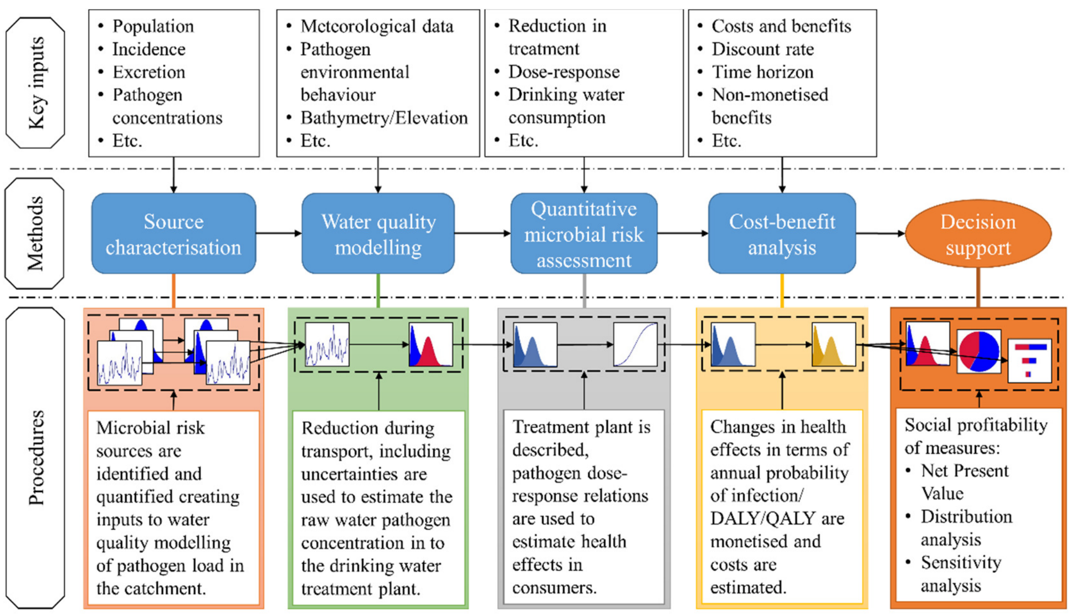


Figure 6. Illustration of the chain of methods that were combined in the decision model, adjusted from Paper 2.

In Paper II, for source characterisation, an incidence-based quantification method was used. Quantification can alternatively be made by other epidemiological metrics or sampling and there is also the possibility to use semi-quantitative or qualitative methods.

Three different approaches to estimate the pathogen reduction during transport from risk sources have been investigated and applied in this thesis. In Paper I, hydrological modelling was used to describe the reduction achieved by the different risk mitigation measures. In Paper II, the pathogen reduction, due to various processes in the Lake Vomb, was estimated by hydrodynamic modelling. By using the hydrodynamic modelling, it was possible to estimate the decrease of pathogen concentration due to transport in the lake from the point of the discharge to the point of the lake water intake. Source pathogen concentration and the modelled lake water intake pathogen concentration were compared and the Log10 reduction was presented as a probability density function. The third approach was conducted when estimating the reduction from the artificial infiltration in the QMRA used in Paper II. This simplified analytical groundwater modelling of the pathogen reduction could also be seen as a step modelling the pathogen reduction between pathogen sources and raw water intake in natural groundwater sources.

The risk assessment in Paper II was conducted using QMRA. Based on barrier performance in DWTP, pathogen concentrations in drinking water, and dose-response relations, the health risk in the drinking water population was estimated for each microbial risk mitigation measure. The health risk reduction achieved by each measure can be described using different metrics. In the Vomb case study, QALYs gained by each measure were estimated. The microbial risk in terms of health effects can also be expressed as pathogen concentration in drinking water, the probability of infection or other health metric such as e.g. DALYs and VSL.

In the final compartment the positive health effects in terms of gained QALYs were monetised and incorporated into a CBA of each alternative. Environmental benefits due to increased reduction of nutrients in WWTPs compared to OWTSs were also estimated and included. The costs were estimated based on a literature review of actual water and wastewater investments made in Sweden (Kärrman et al. 2012). Other decision analysis models, such as CEA or MCDA, can alternatively be used depending on the previous methods adopted. If qualitative methods were used, MCDA would be an alternative, and if water quality or health targets were to be reached, a CEA would be appropriate.

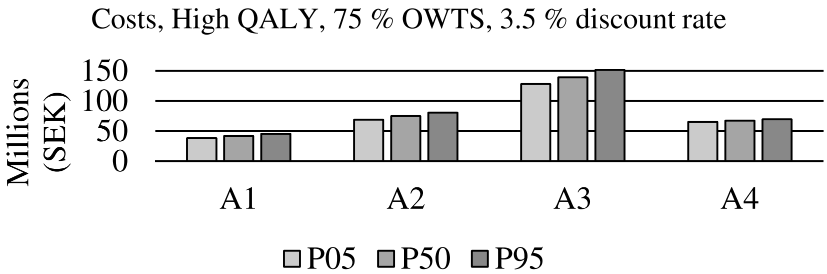
4.5 Cost-benefit analysis

In this section parts from the results from the CBA in Paper II are presented and an additional example. To complement and to further highlight the potential of the decision model, an additional example of results from a CBA is described using an illustrative example. These additional results exemplifies a situation when there is large ambiguity regarding which alternative to choose when comparing *NPV*, due to that uncertainties are included using Monte Carlo simulations.

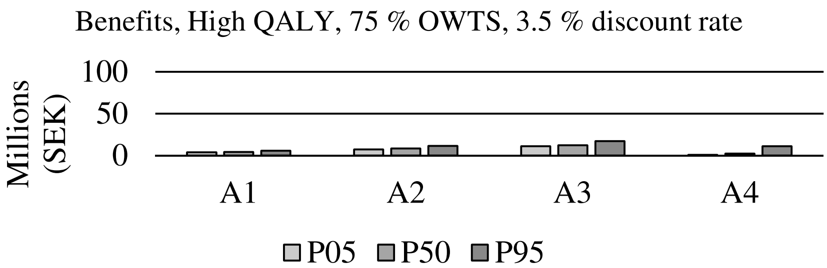
Results from Paper II

In Paper II, four different measures for risk mitigation were evaluated. The 5th, 50th and 95th percentiles of the costs, benefits and *NPV*s of the scenario with a high value of a QALY, assuming OWTSs contributed 75 % of the total pathogen load and a discount rate of 3.5 % are shown in Figure 7. A time horizon of 100 years was used. Complete results are presented in Pater II, where both; discount rates of 3.5 % and 1 %; a low (700,000 SEK) and high (1,220,000 SEK) value of a QALY; and OWTSs contributed 75 % and 50 % of the total pathogen load, are included.

a)



b)



c)

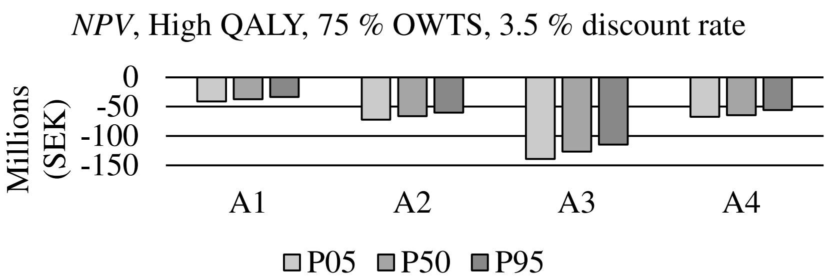


Figure 7. The costs (a), benefits (b) and NPV (c) for the scenario with a high value of a QALY (1,220,000), assuming OWTSs contributed with 75 % of the total pathogen load to Lake Vomb and a discount rate of 3.5 % are presented for the 5th, 50th, and 95th percentiles.

The order of the alternatives based of the 50th percentiles *NPV* was A-ref, A1, A4, A2 and A3. No alternative resulted in positive *NPV*. However, the ranking of the *NPV*s can illustrate which alternative that should be decided upon in order to chose the option with the least negative *NPV*.

Non-monetised benefits can substitute a major part of the benefits and they need to be identified and a rough estimate of how large they need to be in order to achieve a positive *NPV* is provided. Non-monetised benefits for A1-A3 were identified:

- Perceived value for private OWTS owners not being responsible for treating their wastewater.
- Benefits of removing the possible risk of direct contamination from OWTSs to private wells.
- Reduction of CO_2 emission when sludge transportation trucks do not need to drive and empty closed tanks and three compartment septic tanks.
- Positive health effects for humans from higher water quality for recreational activities in Lake Vomb.
- Positive health effect for animals (both domestic and wild) from higher water quality in the catchment and in Lake Vomb.
- Reduction in traffic accidents and related risks since heavy traffic is reduced in the catchment area.

Analysis shows that if these non-monetised benefits was estimated to be SEK 800-1200 per connected OWTS per year, using a discount rate of 1 %, the *NPV* for A1-A3 would be positive, looking at the 50th percentile.

In A4, the following non-monetised benefits were idenitfied:

- Less disinfection by-products due to lower dosage[12] in chlorination
- Reduced handling and storage of chlorination chemicals

The costs in A1-A3 were solely taken by the private OWTS owners in the form of a connection fee and an increase in yearly costs for water and wastewater services. The monetised benefits for A1-A3 were divided between drinking water consumers and persons living in or visiting the Vomb catchment. Drinking water consumers received 11 % of the benefits, whereas the persons living in or visiting the Vomb catchment received 89 % of the benefits, using a low value of a QALY. The corresponding distribution for a high value of a QALY was 18 % and 82 %, respectively. In A4, all costs and benefits were assigned the drinking water consumers.

The scenario-based sensitivity analysis of the CBA investigated the changes of the *NPV*s resulting from choosing a discount rate of 1%. The ranking of the alternatives changed to A-Ref, A1, A2, A4 and A3.

Sensitivity analysis looking at the Spearman's rank correlation coefficient of the CBA showed that for A1-A3 the adopted cost per meter pipe was the most influential input on the *NPV* followed by the estimated pipe length and then the QALYs. For A4, the QALYs was the most influential input followed by the cost for installing UV-treatment.

[12] Since UV-disinfection will increase the reduction of pathogens, the chlorination dose can be lowered.

Additional example

The outcome from the CBA approach performed in Paper II can vary substantial between cases. The purpose with the suggested decsion model is to be able to see differences between alternatives investigation e.g. uncertainties or scenarios. To further highlight the decision support rendered from the decision model, an additional example of how the results from a CBA can look (Figure 8). The results are not from any of the presented papers.

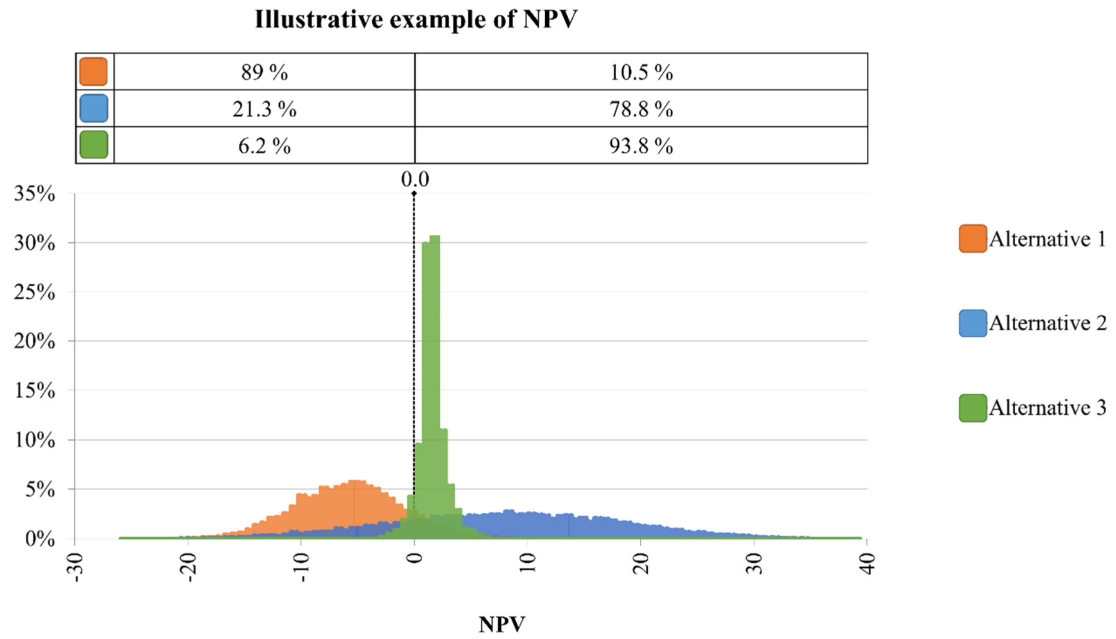


Figure 8. Three fictive alternatives are illustrated by their NPV. The results display the proportion of the 10,000 iterations resulting in each NPV. Percentages on top of the diagram represents the proportion of the iterations resulting in a negative (left hand percentage) or positive (right hand percentage) NPV.

The 50th percentiles of the *NPV* were -5.8, 8.3 and 1.5 million SEK for Alternative 1, 2 and 3, respectively. However, the probability of obtaining a positive *NPV* is highest for Alternative 3, even though the alternative only has the second highest *NPV*. Alternative 2 has a high probability of a positive *NPV*, it has the highest *NPV* looking at the 50th percentile. At the same time, Alternative 2 is also associated with the largest uncertainties. Risk averse decision makers may choose Alternative 3 instead of Alternative 2, since it is associated with the highest probability of a positive *NPV* and the lowest uncertainty range. Alternative 1 have both the smallest probability of obtaining a positive *NPV* and the lowest 50th percentile NVP, making it the least profitable alternative when comparing these three alternatives.

5 DISCUSSION

In this thesis, several methods have been combined to create decision support for comparing microbial risk mitigation measures. In these methods, there are uncertainties and various limitations in the input variables affecting how the different methods can be combined and how the outputs can be interpreted. Decision criteria and risk acceptability criteria are discussed in relation to the results.

Source characterisation

Several crucial assumptions need to be made in order to estimate the pathogen load from various sources. For example, information on prevalence and excretion of pathogens and their variability, both in human and animals, is scarce (Xiao and Herd 1994, Chappell et al. 1996, Ferguson et al. 2009). The high variability in the available estimates has a direct impact on the output of the decision model. The variability of pathogen concentration in human faeces especially, since it ranges over several Log10 units. As new information becomes available, the model inputs should be revised adopting the Bayesian approach.

Pathogen sampling can also be used to quantify the pathogen load and can be the most accurate method for describing local concentrations. Since it is tedious and expensive to perform pathogen sampling, it is rarely done. However, there are currently rapid developments in the field of pathogen analysis and advanced techniques such as metagenomics will be more available and affordable in the near future (Koch 2016). Developments like this will provide important input to decision models like the one presented in this thesis.

Using a Bayesian approach enables the combination of measured data and expert judgements. This approach is necessary in order to incorporate both information from sampling and the available expert knowledge.

When illustrating the decision model in the case of Lake Vomb (Paper II), simplifications regarding the pathogen load were had to be made. An important assumption was that the pathogen load was evenly distributed geographically and temporally, aiming at representative of the endemic level of infection. This assumption can be valid for a large WWTP and it can be argued that a large number of OWTSs also can reach a load that represents a similar endemic situation. However, the decision model or rather the included methods should be further developed to enable scenario modelling of possible disease outbreaks in the population. This means that should be possible to analyse scenarios including periods of both low and high pathogen loads, rather than using a mean pathogen load and uncertainties about that value.

In the applications presented in this thesis, reduction of pathogens in OWTSs and WWTPs has been included in the source characterisation. Reduction in WWTPs (e.g. Ottoson et al. 2006) and OWTSs (e.g. SEPA 2002) has been studied. However, the OWTSs have been assumed to be fully functional and if there are available local information on the performance, e.g. proportion of underperforming or old OWTSs in a catchment etc., it will provide useful input.

Water quality modelling

A large part of the total pathogen reduction occurs during transport from the source to the raw water intake. Estimating this reduction requires extensive input data and the associated uncertainties are typically large. In Paper I, a deterministic approach was used for modelling pathogen transport within the catchment, and the input variability was not considered using a probabilistic methodology. In Paper II, a different approach to water quality modelling was used. The Log10 reduction was calculated for several years using the hydrodynamic model for Lake Vomb. Based on the reduction over time, an uncertainty distribution was fitted to represent variations in the specific variable. This approach does, however, not include a detailed sensitivity analysis of the variables of the hydrodynamic model. Possible approaches for this purpose can be to perform local sensitivity analyses and scenario-based analyses. However, if stochastic modelling is implemented (as described below) other methods are available as well.

In the QMRA performed in Paper II, a third approach to water quality modelling was used as part of estimating the reduction in the artificial groundwater infiltration part of the DWTP. A simple analytical stochastic groundwater modelling was performed using Monte Carlo simulations, considering uncertainties in variables and their effect on the pathogen reduction.

In order to perform more detailed uncertainty and sensitivity analysis of water quality modelling, a stochastic approach has been suggested by e.g. Benham et al. (2006). Looking at future development of the decision model, stochastic water quality modelling should be investigated. Adopting a stochastic approach would include uncertainties in the model input variables and uncertainties in the final outputs would be described. However, since these are highly computationally demanding, also simpler approaches may be of interest to not restrict possible applications of the decision model.

Quantitative microbial risk assessment

The QMRA is widely used, and it is promoted by the WHO to be used in water safety management (WHO 2016). However, it should be highlighted that the methodology, and the Swedish QMRA-tool in particular, is built upon few dose-response relations adopted from specific empirical infection studies for each pathogen. In the Swedish QMRA-tool (Abrahamsson et al. 2009, Åström et al. 2016), it is possible to investigate a high and a low infectious dose for *Cryptosporidium* and *Campylobacter*. For norovirus no such sensitivity analysis is possible. In the case of Lake Vomb, norovirus constitutes the major part of the microbial risk and the assumptions on infectivity can have a large impact on the health effects, hence additional infectious doses of norovirus need to be investigated. The dose-response relations used in QMRA need to be up to date and if possible multiple levels of pathogen infectivity should be investigated.

In general, there are many assumptions in the QMRA methodology and the result should not be interpreted as exact numbers. However, the results will give an indication on what mitigation measures provide small or large risk reduction. A model output is directly dependant in the quality of the model input. Nevertheless, a QMRA will help to understand the decision problems and does map the health risk reductions and give an indication of their magnitude.

The calculations to estimate the DALY were based on the probability of infection. Despite using a widely accepted methodology, there are large uncertainties in DALY calculations; accounting for these uncertainties would constitute further improvement of the decision model. The same discussion is even more valid for the QALY calculations; the number of QALYs per infection was estimated based on the information from the USA. It could be possible to adopt Swedish quality of life estimates for QALY calculations in order to develop the method.

Cost-benefit analysis

Estimation of the monetary value of a QALY was adopted from a different context, i.e. the governmental implied willingness-to-pay for reimbursement of pharmaceuticals. This application should be seen as a first rough approximation; and there is a need to further develop methods for estimating and choosing how to monetise the health benefits. The monetary value of a QALY used in this these was estimated using a societal perspective, i.e. effects beyond the health sector were accounted for. Thus, one could argue that this monetisation of health effects can be applied to any type of setting when comparing alternative options for optimisation of societal benefits. Monetising health effects is a difficult task, and the values adopted should therefore always be clearly stated and a sensitivity analysis performed (ASCC 2008). Specific applications might need to divide the health benefits into several categories, e.g. reduced medical and hospitalisation costs, reduced discomfort from being ill, reduction in production loss etc., to provide a more detailed analysis. Information on costs may be adapted from occurred waterborne disease outbreaks. Thus, a possible approach is to use this information to estimate the possible benefits from mitigation measures.

As all benefits were not included in the case study presented here, the analysis regarding the additional benefits required to achieve a positive *NPV* provides important information in the decision support. Given the difficulties of monetising non-market goods, this approach provides a straightforward and illustrative way of putting the *NPV* from the CBA in relation to the non-monetised benefits.

Other factors might influence the decision, resulting in choosing an alternative not based on the *NPV* from the CBA. Looking at environmental targets regarding levels of nitrogen and phosphorous discharged into water courses, alternatives (A1-A3) exemplified in Paper II provide substantial reductions of these contaminants. However, A4 in the same study does not result in any reduction of nutrient discharge to recipients. The benefits from reducing these nutrients in A1-A3 are included in the CBA. However, there might be requirements regarding these nutrients that are affected by the measures. Regardless of whether an acceptable risk is sought, if water quality guidelines are achieved or if environmental targets need to be met, a CBA provides useful decision support in order to compare decision alternatives.

Uncertainty and sensitivity analysis

The sensitivity analysis highlights the variables most influential on the outputs of the different methods used in the decision model. Variables that should be investigated further are identified, and if possible, the uncertainties related to these variables should be reduced. The uncertainty analysis investigates the output uncertainty. For the Lake Vomb case study (Paper II), concentration in faecal matter, estimated pipe length and pipe cost per meter were the variables that had the highest impact on the uncertainty of the outputs. Local sensitivity analysis (Equation 17) is suitable for deterministic modelling and more simple non-probabilistic models. Monte Carlo simulation facilitates a global sensitivity and uncertainty analysis (Equation 18) making it possible to simultaneously analyse the contribution to the total uncertainty from each specific input variable represented by an uncertainty distribution. These analyses are essential procedures in the decision model and to describe uncertainties provides a more robust and detailed decision support.

Decision support for the drinking water sector

Models can never fully describe the reality. They are limited by the information and knowledge of the reality that is used to compile them. Nevertheless, models are useful in helping us understand and structure problems and focus on those parts that are of importance in the specific application. Models can also pinpoint where additional information and knowledge is needed. The decision model presented in this thesis provides a clear structure for how risk assessment and decision analysis in the form of CBA can be combined. The combination provide a transparent and holistic decision support that aims to optimise the societal benefits. The combination of methods, integrating several scientific disciplines, provides a novel approach for comparing microbial risk mitigation measures. Even though the approach is promoted in the literature, it is rarely applied. The decision model constitutes a risk-based approach that will map the microbial risks in the DWS and provide useful information for drinking water utilities. Furthermore, implementing the decision model in the drinking water sector will aid decision makers to use societal resource efficiently when mitigating microbial risks and enables an integrated water resource management.

6 CONCLUSIONS AND FURTHER RESEARCH

The focus of this thesis was to present a comprehensive risk-based decision model from source to tap for comparing microbial risk mitigation measures, combining risk assessment and CBA. The main conclusions from Paper I, Paper II and the thesis are:

- The risk-based decision model illustrate the possibility to combine methods and that results from each individual method is useful as well as the overall results from the entire decision model.

- The presented combination of water quality modelling, quantitative microbial risk assessment and cost-benefit analysis provides a comprehensive description of the drinking water system and a practical approach to evaluate and compare possible microbial risk mitigation measures.

- Water quality modelling have been implemented into the decision model and was shown to comprise a useful part of risk assessment when microbial risk mitigation measures are compared.

- There are large uncertainties in the input variables of the decision model that need to be taken into account. Uncertainty and sensitivity analysis provided information on which input variables are important for the results of the risk assessment and the cost-benefit analysis. Input variables that contribute with large uncertainties to the decision model should be investigated further to reduce uncertainties in the derived decision support.

- Results from the cost-benefit analysis in the form of net present value, uncertainty- and sensitivity analyses as well as distributional analyses can be combined with other information, e.g. relevant legislation, in order to provide additional decision support when risk mitigation measures are compared.

- The risk-based decision model opens for the possibility to use alternative methods for risk assessment and decision models, making it flexible in order to be tailored to different drinking water systems and different types of decision problems. If an even more holistic analysis is needed including, for example, local social effects and inherent environmental values, other methods such as multi-criteria analysis can be useful.

The structure of the decision model has been established providing a robust platform. Nevertheless, there are components of the decision model that will need to be improved, providing more detailed descriptions or experimenting with alternative methods in various parts:

- There is a need to develop procedures on how to decide upon which methods that should be used in each compartment of the decision model, depending to the local setting of the DWS.

- Model input that varies over time, both within a year, e.g. incidence and water flow etc., and over longer time periods, e.g. population increase and climate change, have not been fully included the presented model. A linear population increase based on population projections for Sweden in general was included. However, variations on a sub-yearly basis have not been included. For development of the model, methods for including these short-term variations need to be developed. Long-term changes, e.g. climate change, need to be included as well.

- Adopting methods for describing epidemic outbreaks in the source characterisation, in addition to the endemic approach that is used presently, would be an important improvement, since waterborne outbreaks can render extensive costs to the society (Corso et al. 2003, Lindberg et al. 2011). In catchments with a small population contributing to the pathogen load (as for Lake Vomb in Paper II), this can be even more crucial.

- Further developments regarding estimation of health benefits are needed. The possibility to divide health benefits into more detailed posts and to investigate if costs of occurred events can be used for estimating health benefits should be investigated. Furthermore, the value of a QALYs was estimated using a linear approach. However, there are studies that show non-linear relationships between individuals willingness to pay for a QALY or similar health metrics. Both severity of the illness as well as duration of the illness show non-linear correlation with willingness to pay for a QALY (Haninger and Hammitt 2011, Ryen and Svensson 2015). This need to be acknowledged in further development of methods used for monetising health benefits.

- Large efforts were made to describe the health benefits obtained due to risk reduction, while the environmental benefits were included using a more simplified approach. Nonetheless, including the environmental benefits illustrates a key element of the CBA, i.e. the possibility to include other benefits apart from the target risk reduction. Non-monetised benefits and legislation related to aspects affected by the mitigation measures (e.g. environmental legislation, the European bathing water directive, etc.) constitute important factors in decision making, as they may provide valid grounds to depart from decisions taken based solely on the results from the CBA. These procedures need to be formally described and presented in more detail.

- To broaden the decision analysis, a combination of CBA and multi-criteria analysis could be investigated. This would broaden the approach further and making the decision support even more transparent and holistic.

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