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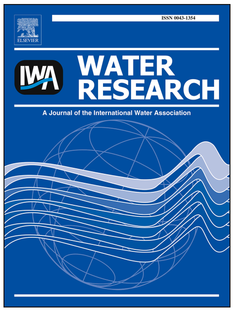

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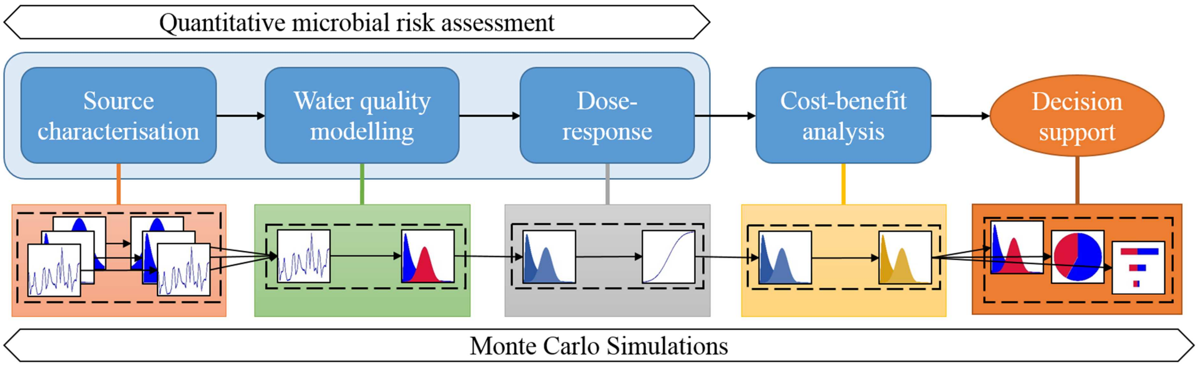
Risk-based decision model



Risk-based cost-benefit analysis for evaluating microbial risk mitigation in a drinking water system

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Abstract

Waterborne outbreaks of gastrointestinal diseases can cause large costs to society. Risk management needs to be holistic and transparent in order to reduce these risks in an effective manner. Microbial risk mitigation measures in a drinking water system were investigated using a novel approach combining probabilistic risk assessment and cost-benefit analysis. Lake Vomb in Sweden was used to exemplify and illustrate the risk-based decision model. Four mitigation alternatives were compared, where the first three alternatives, A1-A3, represented connecting 25, 50 and 75 %, respectively, of on-site wastewater treatment systems in the catchment to the municipal wastewater treatment plant. The fourth alternative, A4, represented installing a UV-disinfection unit in the drinking water treatment plant. Quantitative microbial risk assessment was used to estimate the positive health effects in terms of quality adjusted life years (QALYs), resulting from the four mitigation alternatives. The health benefits were monetised using a unit cost per QALY. For each mitigation alternative, the net present value of health and environmental benefits and investment, maintenance and running costs was calculated. The results showed that only A4 can reduce the risk (probability of infection) below the World Health Organization guidelines of 10^{-4} infections per person per year (looking at the 95th percentile). Furthermore, all alternatives resulted in a negative net present value. However, the net present value would be positive (looking at the 50th percentile using a 1 % discount rate) if non-monetised benefits (e.g. increased property value divided evenly over the studied time horizon and reduced microbial risks posed to animals), estimated at 800-1200 SEK (€100-150) per connected on-site wastewater treatment system per year, were included. This risk-based decision model creates a robust and transparent decision support tool. It is flexible enough to be tailored and applied to local settings of drinking water systems. The model provides a clear and holistic structure for decisions related to microbial risk mitigation. To improve the decision model, we suggest to further develop the valuation and monetisation of health effects and to refine the propagation of uncertainties and variabilities between the included methods.

Keywords: cost-benefit analysis (CBA), decision support, drinking water, quality adjusted life year (QALY), quantitative microbial risk assessment (QMRA), water quality modelling

1 Introduction

Risk management of drinking water systems (DWSs) is an iterative process including risk assessment and risk mitigation (i.e. risk treatment) (ISO 2009). To be effective in providing safe drinking water supply, the risk management must comprise the entire system, from catchment to consumer. If the risks are unacceptable, risk mitigation measures should be implemented, and alternatives for risk mitigation evaluated. Water Safety Plans procedures, developed by the World Health Organization (WHO), can serve as a risk management strategy for water providers (Bartram et al. 2009). However, in order to allocate societal resources for risk mitigation in an efficient manner, the economic dimension of risk levels and possible risk mitigation measures must be considered (WHO 2011).

Risks related to DWSs have been extensively discussed in the literature (e.g. Beuken et al. 2008, Keller and Wilson 1992, WHO 2011). Health risks in DWSs can be related to chemical, microbial and radiological hazards (WHO 2011). In this paper, the microbial risks are the main focus. Microbial risks in the form of pathogenic microorganisms can originate from faecal sources (Dufour et al. 2012, Ferguson et al. 2009) related to humans (municipal wastewater treatment plants (WWTPs) or on-site wastewater treatment systems (OWTSs) on private properties) or animals (wild animals, domestic grazing animals or use of manure on cropland). Pathogens in DWSs can cause endemic waterborne illness (Payment and Hunter 2001) as well as waterborne outbreaks of gastrointestinal diseases, resulting in high costs for the society (Corso et al. 2003, Larsson et al. 2014). The WHO pointed out that the societal costs for endemic waterborne illness and related gastrointestinal disease are commonly underestimated (WHO 2001).

Quantitative microbial risk assessment (QMRA) has been applied to DWSs in various settings (Haas et al. 2014, WHO 2016) in order to assess the risk in relation to an acceptable or tolerable risk level. The result from a QMRA is typically reported as probability of infection, disability adjusted life years (DALYs) or quality adjusted life years (QALYs). Both DALYs and QALYs are health metrics that combine mortality and morbidity. Drinking water producers commonly look at the (WHO) for guidance and the suggested risk levels of an annual probability of infection of 10^{-4} per person per year, and DALYs of 10^{-6} per person per year (WHO 2011).

To make informed decisions on which risk mitigation measure to implement in order to use societal resources effectively, the alternatives need to be compared. Comprehensive lists and procedures for identifying risk mitigation measures (e.g. Åström and Pettersson 2010, NZMH 2014, Rosén et al. 2010) are available. Decision support systems or decision models such as cost-effectiveness analysis (CEA) and multi-criteria decision analysis (MCDA) can aid decision makers in comparing the alternatives. If there are no regulations regarding acceptable risk levels, other evaluation methods might be needed in order to justify the implementation of risk mitigation measures. Cost-benefit analysis (CBA) provides a robust well-established decision support approach to investigate the measure that is the most profitable or least costly (if a certain risk level is required) for society (Boardman et al. 2011, Cameron et al. 2011).

Comparing mitigation measures directed at different parts of the supply system and identifying the options most profitable for society are key steps towards a holistic and sustainable risk management approach. Adopting holistic risk management also enables the multi-barrier approach emphasised by the WHO (2011). Using CBA as a basis for decision support helps to allocate monetary resources in an efficient manner providing possibilities to compare mitigation measures with interventions in other sectors (e.g. food, health care, traffic and environmental risk management). CBA facilitates optimisation of the societal resources by comparing economic metrics, such as net present value (*NPV*), and performing distributional analysis (Cameron et al. 2011). CBA also helps highlight the societal benefits of reducing microbial risks in DWSs and creates a systematic and transparent decision support tool.

Different frameworks for combining risk management, decision making process and CBA in the drinking water context have been investigated (e.g. Assmuth et al. 2016, Rizak et al. 2003). Despite the aforementioned implementations, there are few, if any, methods that use a probabilistic quantitative risk-based approach to create decision support in the form of a CBA for microbial risk management in DWSs. To include an economic dimension and to perform a CBA in this way is uncommon, even though the need is emphasised by the WHO (WHO 2001).

Aim

In this study we develop a method for creating a systematic, holistic and transparent decision support for microbial risk management in DWSs. We present a novel CBA approach from catchment to consumer. More in detail, we perform a CBA using a combination of water quality modelling and QMRA to compare microbial risk mitigation alternatives in a DWS. The methodology is exemplified using Lake Vomb in the south of Sweden. Different alternatives of removing OWTSs are compared to installation of an additional treatment step in the drinking water treatment plant (DWTP). We also highlight the choices that needs to be made in the CBA-model, and what implications these might have on the outcome of the CBA.

2 Risk-based decision model

The suggested approach for combining the methods for QMRA and CBA is presented as a decision model in Figure 1. The four major compartments are: (i) source characterisation, (ii) water quality modelling, (iii) dose-response, and (iv) CBA. The source characterisation provides input to the water quality modelling, and the water quality modelling provides input to the dose-response. The QMRA framework, including (i), (ii) and (iii), describes the entire risk chain in the DWS and provides input for the CBA. Epistemic uncertainties (associated with lack of knowledge) and aleatory uncertainties (associated with natural variations) in all compartments are incorporated into the model by means of Monte Carlo (MC) simulations. The combination of methods aims to enable an estimation of the microbial risk in the DWS as well as an estimation of the effect of risk reduction measures and their societal profitability. Hence, the decision model can serve as a tool within the water safety plan framework. When analysing different mitigation measures, each compartment of the decision model needs to be executed. Detailed method descriptions of each compartment are presented in sections 3.2-3.4. It should be noted that this decision model is generic, and the applied methods in each case study should be selected to fit the specific context of the analysed DWS.

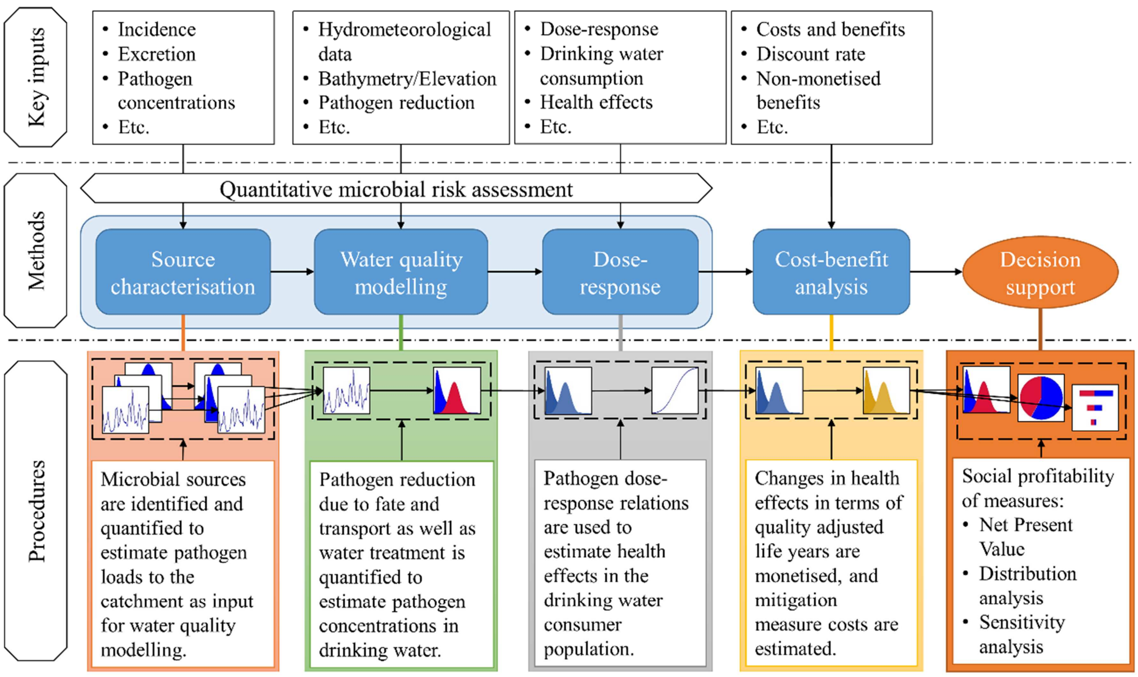

Figure 1. Risk-based decision model combining methods for evaluating and comparing microbial risk mitigation measures.

3 Methods

3.1 Lake Vomb

Lake Vomb is a small lake in Scania, the southernmost part of Sweden, providing 330,000 consumers with drinking water. The average water depth is 6.6 m, and the maximum depth is 16 m. Three major tributaries discharge into Lake Vomb: Borstbäcken, Torpsbäcken and Björkaån draining 26, 42 and 340 km^2, respectively. There are approximately 2800 OWTSs in the catchment (Norwegian Water BA 2009) posing a risk to the drinking water source. Other sources of microbial risks are e.g. WWTP, fertilisation using manure, grazing animals, wild animals. Raw water is extracted from Lake Vomb and artificially infiltrated into a glaciofluvial aquifer and then treated using conventional treatment consisting of rapid sand filtration and chlorination (Norwegian Water BA 2009). Figure 2 illustrates the case study area.

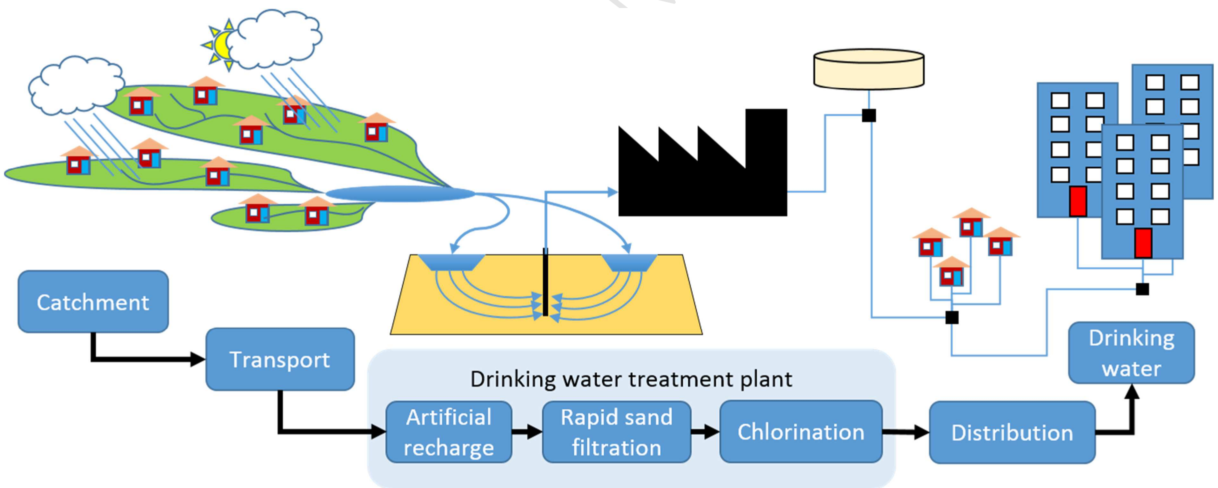

Figure 2 Schematic illustration of Lake Vomb drinking water system.

Microbial risk mitigation alternatives in different parts of the DWS were chosen to illustrate how the risk-based decision model can be used. The mitigation alternatives also reflect the contemporary trends in Sweden regarding OWTSs management and an increase in installation of UV-disinfection in DWTPs. Three of the analysed alternatives represent connection of different proportions (25, 50 and 75 %, respectively) of the OWTSs in the catchment to the municipal WWTP. The costs for the alternatives were based on connection of clusters of closely located OWTSs. However, the pathogen load from these OWTSs was assumed to be removed evenly across the different types of OWTSs and geographically across the catchment area. This assumption was made because of the short transport time in the catchment (Sundahl et al. 2008). The fourth alternative was to install UV-disinfection at the DWTP at Lake Vomb. The four decision alternatives and one reference alternative were analysed:

- Reference alternative (A-Ref) – Continuation of the present state.
- Alternative 1 (A1) – Connecting 25 % (621) of the OWTSs to the local WWTP.
- Alternative 2 (A2) – Connecting 50 % (1240) of the OWTSs to the local WWTP.
- Alternative 3 (A3) – Connecting 75 % (1861) of the OWTSs to the local WWTP.
- Alternative 4 (A4) – An additional barrier, UV-disinfection, is installed at the DWTP.

3.2 Quantitative microbial risk assessment (QMRA)

The QMRA methodology (Haas et al. 2014) was used for quantifying the health effects related to the reference alternative and the microbial risk mitigation alternatives. All inputs for the QMRA are listed in Table 1; input distributions represent both epistemic uncertainties and aleatory uncertainties.

3.2.1 Source characterisation

Human pathogens in wastewater from OWTS were quantified as described by Ottoson and Stenström (2003). It was assumed that the population was large enough to have pathogens present continuously, and that the entire pathogen load was evenly spread throughout the catchment. Three reference pathogens were used, one for each of the pathogen groups: *Cryptosporidium* for protozoa; *Campylobacter* for bacteria; and norovirus for viruses. The pathogen concentration ($C_{Tributary}$, *pathogens/L*) of each reference pathogen in each tributary was calculated as:

$$C_{Tributary} = \frac{I_{Path} \cdot U \cdot D_{Path} \cdot FP \cdot C \cdot P}{365 \cdot 10^5 \cdot F_{Tributary}} \qquad (1)$$

where I_{Path} (*infections/year/10^5 inhabitants*) was the incidence; U (*no unit*) was the factor for underreporting; D_{Path} (*days*) was the duration of excretion; FP (*g*) was the faecal production per person per day; C (*pathogens/g*) was the pathogen concentration in faeces when infected; P (*persons*) was the number of persons that are using OWTSs in the sub-basin; and $F_{Tributary}$ (*L/day*) was the average daily flow for each tributary.

Table 1 Inputs for the quantitative microbial risk assessment.

Input	Unit	Abbr.	Distr. [s]	Parameters
		General inputs		
Pers. connected to OWTSs [a]	#	P	PV	ARef=6215, A1=4661, A2=3107, A3=1554, A4=6215
Factor of underreporting [b]	-	U	PV	38
Faecal production [c]	g	FP	N	P05=115.7, P95=144.2
Red. in on-site sewage [d]	Log_{10} red.	R_{OWTS}	PV	0.72
Depth of unsat. zone [e]	m	Dp_{UZ}	T	Min=1.2, Mode=1.5, Max=2.4
Length of satur. zone [e]	m	L_{SZ}	T	Min=100, Mode=150, Max=250
Drinking water consumption [f]	L/day	WI	e^N	N(μ=-0.299, σ=0.57)
Average flow Björkaån [g]	L/day	F_{Bjorka}	PV	$3.21*10^8$
Average flow Torpsbäcken [g]	L/day	F_{Torps}	PV	$3.46*10^7$
Average flow Borstbäcken [g]	L/day	F_{Borst}	PV	$2.16*10^7$
		Inputs for norovirus		
Incidence [h]	Inf./10^5inh/y	I_{Noro}	G	α=6.25, β=10.6
Days excreting [i]	days	D_{Noro}	LN	P01=13, P99=27, Loc=0
Path. conc. in faeces if inf. [j]	Log_{10} path./g	C_{Noro}	N	P01=5, P99=9
Red. lake transp. Björkaån [k]	Log_{10} red.	R_{Bjork_Noro}	N	μ=4.5, σ=0.12, Tr: Min=0
Red. lake transp. Torpsbäcken [k]	Log_{10} red.	R_{Torp_Noro}	N	μ=4.5, σ=0.11, Tr: Min=0
Red. lake transp. Borstbäcken [k]	Log_{10} red.	R_{Borst_Noro}	N	μ=4.6, σ=0.10, Tr: Min=0
Red. in unsatur. zone [l]	Log_{10} red./m	R_{UZ/m_Noro}	T	Min=0.05, Mode=0.3, Max=1
Red. in satur. zone [m]	Log_{10} red./m	R_{SZ_Noro}	LN	μ=6.6587, σ=5.5366
Red. in conv. treatm. [n]	Log_{10} red.	R_{CT_Noro}	T	Min=0.4, Mode=0.5, Max=0.6
Red. in UV-treatm. [o]	Log_{10} red.	R_{UV_Noro}	PV	4.2
Infectivity [p]	-	m_{Noro}	EBP	α=0.04, β=0.055
DALYs per infection [q]	DALY/inf	D_{Noro}	PV	0.000716
QALYs per infection [r]	QALY/inf	Q_{Noro}	PV	0.0009
		Inputs for *Campylobacter*		
Incidence [h]	Inf./10^5inh/y	I_{Camp}	G	α=64.6, β=1.27
Days excreting [i]	days	D_{Camp}	LN	P01=13, P99=27, Loc=0
Path. conc. in faeces if inf. [j]	Log_{10} path./g	C_{Camp}	N	P01=4, P99=10
Red. lake transp. Björkaån [k]	Log_{10} red.	R_{Bjork_Camp}	N	μ=5. 5, σ=0.40, Tr: Min=0
Red. lake transp. Torpsbäcken [k]	Log_{10} red.	R_{Torp_Camp}	N	μ=5.6, σ=0.38, Tr: Min=0
Red. lake transp. Borstbäcken [k]	Log_{10} red.	R_{Borst_Camp}	N	μ=5.9, σ=0.29, Tr: Min=0
Red. in unsatur. Zone [l]	Log_{10} red./m	R_{UZ/m_Camp}	T	Min=0.05, Mode=0.5, Max=1
Red. in satur. Zone [l]	Log_{10} red./m	R_{SZ/m_Camp}	T	Min=0.001, Mode=0.05, Max=1
Red. in conv. treatm. [n]	Log_{10} red.	R_{CT_Camp}	T	Min=0.2, Mode=0.25, Max=0.3
Red. in UV-treatm. [o]	Log_{10} red.	R_{UV_Camp}	PV	5.3
Infectivity [p]	-	m_{Camp}	EBP	α=0.024, β=0.011

DALYs per infection [q]	DALY/inf	D_{Camp}	PV	0.00328
QALYs per infection [r]	QALY/inf	Q_{Camp}	PV	0.0163
		Inputs for *Cryptosporidium*		
Incidence [f]	Inf./10^5inh/y	I_{Crypt}	G	α=5.43, β=0.228
Days excreting [g]	days	D_{Cryp}	LN	P01=5, P99=30, Loc=0
Path. conc. in faeces if inf. [j]	Log_{10} path./g	C_{Cryp}	N	P01=7, P99=9
Red. lake transp. Björkaån [k]	Log_{10} red.	R_{Bjork_Cryp}	N	μ=4.5, σ= 0.12, Tr: Min=0
Red. lake transp. Torpsbäcken [k]	Log_{10} red.	R_{Torp_Cryp}	N	μ=4.6, σ=0.10, Tr: Min=0
Red. lake transp. Borstbäcken [k]	Log_{10} red.	R_{Borst_Cryp}	N	μ=4.6,σ=0.095, Tr: Min=0
Red. in unsatur. zone [l]	Log_{10} red./m	R_{UZ/m_Cryp}	T	Min=0.05, Mode=0.5, Max=1
Red. in satur. zone [l]	Log_{10} red./m	R_{SZ/m_Cryp}	T	Min=0.001, Mode=0.05, Max=1
Red. in conv. treatm. [n]	Log_{10} red.	R_{CT_Cryp}	T	Min=0.4, Mode=0.5, Max=0.6
Red. in UV-treatm. [o]	Log_{10} red.	R_{UV_Cryp}	PV	3
Infectivity [p]	-	m_{Cryp}	EBP	α=0.115, β=0.176
DALYs per infection [q]	DALY/inf	D_{Cryp}	PV	0.00267
QALYs per infection [r]	QALY/inf	Q_{Cryp}	PV	0.0035

a) 2.5 persons (Åström and Johansson 2015) per on-site wastewater treatment system.
b) Mead et al. (1999)
c) Wyman et al. (1978)
d) Log_{10} reduction in on-site sewage was estimated using standard values for four types of systems, achieving 100 % 99 %, 95 %, 50 %, respectively (SEPA 1991, 2002, 2003, 2008b), and the proportions of OWTS types.
e) Personal communication with B.M. Pott at Southern Sweden Water Supply (Sydvatten)
f) Westrell et al. (2006)
g) SMHI (2017)
h) Based on yearly incidence data 2006-2016 (PHAS 2017). Norovirus and *Campylobacter*: Best fit using Chi-Squared, *Cryptosporidium*: Best fit using Kolmogorov-Smirnov. For norovirus: incidence was based on per season incidence and data from 2006-2012 was based on national data due to lack of regional estimates.
i) Norovirus: The Min/Median of the range of days shedding from Atmar et al. (2008) was chosen as P01/P99 respectively, *Campylobacter* and *Cryptosporidium*: Reported triangular Min/Max from Petterson et al. (2016) was chosen for P01/P99 respectively.
j) Norovirus: From Marshall et al. (2001) as reported in (Westrell 2004), *Campylobacter* and *Cryptosporidium*: Reported triangular Min/Max from Petterson et al. (2016) was chosen for P01/P99 respectively.
k) Distributions fitted from three years (2005-2007) of simulated daily Log_{10} reduction from the hydrodynamic modelling. Tr=Truncation
l) Estimation based on expert judgement.
m) Reduction calculated using 10,000 MC iterations using a groundwater transport model estimating the removal due to attachment, inactivation and dilution (Åström et al. 2016, Schijven et al. 2006).
n) Norwegian Water BA (2009)
o) Calculated using Equation 7. UV-dose (fluence) was set to 400 J/m^2. The inactivation constant (k) and intercept of the fluence axis (b) were set to (k/b) (0.106/0), (0,293/0) and (0,225/1.087) for norovirus, *Campylobacter* and *Cryptosporidium* respectively (Hijnen et al. 2006). Not to exceed the experimental range (Hijnen et al. 2006), the maximum Log_{10} reduction was used for norovirus and *Cryptosporidium*.
p) Norovirus: Teunis et al. (2008), *Campylobacter*: Teunis et al. (2005), *Cryptosporidium*: Teunis et al. (2002).
q) Norovirus and *Cryptosporidium*: based on a re-analysis from Kemmeren et al. (2006), *Campylobacter*: based on Havelaar and Melse (2003).
r) Batz et al. (2014)
s) PV=point value, N=Normal distribution, T=triangular distribution, e^N=exponential with a normal distribution in the exponent, G=Gamma distribution, LN=LogNormal distribution, EBP=Exact Beta-Poisson distribution.

3.2.2 Water quality modelling

The pathogen concentrations at the raw water intake (C_{RW}, *pathogens/L*) were calculated as:

$$C_{RW} = (C_{Borst} \cdot 10^{-(R_{OWTS}+R_{Borst})}) + (C_{Torp} \cdot 10^{-(R_{OWTS}+R_{Torp})}) + (C_{Bjorka} \cdot 10^{-(R_{OWTS}+R_{Bjork})}) \qquad (2)$$

where C_{Borst}, C_{Torp} and C_{Bjorka} (*pathogens/L*) were the pathogen concentrations in the tributaries; R_{OWTS} (*no unit*) was the Log_{10} reduction in the OWTSs; and R_{Borst}, R_{Torp} and R_{Bjork} (*no unit*) were the Log_{10} reduction due to transport in Lake Vomb from the tributary to the raw water intake. Reduction in the catchment from OWTS discharge until entering Lake Vomb was conservatively assumed to be negligible due to the longevity of pathogens and the rapid transport (Sundahl et al. 2008) in the catchment.

Hydrodynamic modelling was performed to simulate the fate and transport of pathogens from the point of entering Lake Vomb from the three tributaries to the raw water intake. Due to the linearity of the hydrodynamic model, a constant load was used to estimate the pathogen reduction. Decay of the pathogens was calculated as:

$$C_t = C_0 \cdot e^{-wt} \qquad (3)$$

where C_t (*pathogens/L*) was the concentration at t; C_0 (*pathogens/L*) was the initial concentration; w (*1/day*) was the decay rate; and t was the time step. The w value was set to 0.23 for *Campylobacter* and 0.03 for *Cryptosporidium* and norovirus, based on the estimates of half-life of pathogens in environment. For *Campylobacter,* the median half-life was estimated from various literature sources (Catalao Dionisio et al. 2000, Cook and Bolster 2007, Hendricks 1971, McGee et al. 2002, Medema et al. 1997, Nasser et al. 2003, Ottosson and Stenström 2003, Rhodes and Kator 1988, Terzieva and McFeters 1991). For *Cryptosporidium* and norovirus the same half-life was used, estimated as the median half-life given in literature (Medema et al. 1997, Nasser et al. 2003, Ottosson and Stenström 2003). *Cryptosporidium* was also reduced due to settling in the lake. In the MIKE 3 FM, the settling velocity for *Cryptosporidium* was specified as 0.03 m/day, which is the settling velocity previously suggested for free oocysts (Medema et al. 1998). It was conservatively assumed that *Cryptosporidium* oocysts released into the lake were not attached to particles.

Comparing the input pathogen concentration at each tributary with the resulting pathogen concentration at the raw water intake, the daily Log_{10} reduction due to transport in the lake was calculated. Three year time-series of daily Log_{10} reductions were used to estimate the variability in the daily Log_{10} reduction for the three different transport paths. The estimated Log_{10} reductions for each tributary are presented in Table 1.

The pathogen concentration in drinking water (C_{DW}, *pathogens/L*) was calculated as:

$$C_{DW} = C_{RW} \cdot 10^{-(R_{UZ}+R_{SZ}+R_{CT}+R_{UV})} \quad (4)$$

where R_{UZ} (*no unit*) was the total Log_{10} reduction in the unsaturated zone; R_{SZ} (*no unit*) was the total Log_{10} reduction in the saturated zone; R_{CT} (*no unit*) was the Log_{10} reduction by the conventional treatment at the DWTP; and R_{UV} (*no unit*) was the total Log_{10} reduction by the UV-disinfection. The chlorination step was assumed not to contribute to the microbial removal because of a small dose and that chloramine was used as disinfection agent[1].

The Log_{10} reduction in the unsaturated zone (R_{UZ}, *no unit*) was calculated as:

$$R_{UZ} = R_{UZ/m} \cdot Dp_{UZ} \quad (5)$$

where $R_{UZ/m}$ (Log_{10}/m) was the Log_{10} reduction per meter, and Dp_{UZ} (*m*) was the depth of the unsaturated zone.

For *Campylobacter* and *Cryptosporidium*, the Log_{10} reduction in the saturated zone (R_{SZ}, *no unit*) was calculated as:

$$R_{SZ} = R_{SZ/m} \cdot L_{SZ} \quad (6)$$

where $R_{SZ/m}$ (Log_{10}/m) was the Log_{10} reduction per meter in saturated zone, and L_{SZ} (*m*) was the length of the saturated zone. For norovirus, the Log_{10} reduction in saturated zone (R_{SZ}, *no*

[1] Personal communication the Southern Sweden Water Supply (Sydvatten).

unit) was estimated using a groundwater transport model (Åström et al. 2016, Schijven et al. 2006).

The estimated Log_{10} reductions in conventional treatment (R_{CT}) are presented in Table 1.

The Log_{10} reduction by the UV-disinfection (R_{UV}, *no unit*) was described as a first order disinfection model and calculated as:

$$R_{UV} = -x \cdot f - b \quad (7)$$

where x (cm^2/mJ) was an inactivation constant; b (*no unit*) was the interception of the fluence axis; and f (mJ/cm^2) was fluence.

3.2.3 Dose-response

To estimate the health effects in the form of infections due to the pathogens in the drinking water, the pathogen daily dose (D, *pathogens/day*) was calculated as.

$$D = C_{DW} \cdot WI \quad (8)$$

where WI (L) was the daily ingested volume of drinking water per capita in Sweden (Westrell et al. 2006). All three reference pathogens were assigned the Exact Beta-Poisson dose-response function. An Exact Beta-Poisson function can be represented by an exponential function with a beta distribution in the exponent (Equation 9); this approach has been reported to be representative in infection studies (Teunis et al. 2005, Teunis et al. 2002, Teunis et al. 2008).

$$P_{inf} = 1 - e^{-m \cdot D} \quad (9)$$

where P_{inf} (*probability*) was the daily probability of infection for each pathogen; m (no unit) was the infectivity; and D was the simulated daily pathogen dose that was ingested.

The annual probability of infection (P_{annual}, *probability*) was calculated as:

$$P_{annual} = 1 - \prod_{1}^{365} \left(1 - P_{inf}\right) \qquad (10)$$

A bootstrap technique was used to sample 365 random P_{inf} values for each iteration calculating the annual probability (Equation 10). This is necessary since the daily probability of infection is not constant for one year. P_{annual} was used to calculate the QALYs lost (*QALYs*, *QALYs lost per person per year*):

$$QALYs = P_{annual} \cdot Q \qquad (11)$$

where Q (*QALYs/infection*) was the amount of QALYs reported per infection for each pathogen. All infections were assumed to result in QALYs. P_{annual} was also used to calculate *DALYs* per person as suggested by the WHO (Havelaar et al. 2000, Kemmeren et al. 2006, WHO 2001). The Swedish population age structure of 2010 from the European database (EU 2010) was used to characterise the drinking water consumer population.

Three separate probabilities of infection[2] for the three pathogens were summarised into the total probability of infection (P_{annual_tot}, *probability*) calculated as:

$$P_{annual\_tot} = 1 - \left(1 - P_{annual\_noro}\right) \cdot \left(1 - P_{annual\_camp}\right) \cdot \left(1 - P_{annual\_cryp}\right) \quad (12)$$

where P_{annual_noro}, P_{annual_camp} and P_{annual_cryp} (*probabilities*) were the annual probabilities of infection due to norovirus, *Campylobacter* and *Cryptosporidium* respectively.

3.3 Cost-benefit analysis (CBA)

A CBA was performed to compare the economic negative effects (costs) with the positive effects (benefits) for each alternative. All inputs for the CBA are presented in Table 2; input distributions represent both epistemic uncertainties and aleatory uncertainties. To enable a

[2] This implies that the different events are independent. Since pathogens often originate 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.

comparison of the alternatives' societal profitability, the net present value (*NPV*) was calculated and a distributional analysis was performed for each alternative. The *NPV* (*SEK*) was calculated as:

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

where *B* (*SEK*) and *C* (*SEK*) were the benefits and costs for each year *t* during the time horizon *T* (*years*); *r* (*%*) was the discount rate used. *T* was set to 100 years, representing the expected life-time of the mitigation alternatives.

The procedure used in this study for taking into account the project-specific costs and benefits, as well as externalities, follows the basic concept of CBA given by e.g. Boardman et al. (2011), among others. The total annual benefits (B_{tot}, *SEK*) were calculated as:

$$B_{tot} = B_{health} + B_{environmental} + B_{other} \qquad (14)$$

where B_{health} (*SEK*) were the benefits estimated from reduced negative health effects to drinking water consumers; $B_{environmental}$ (*SEK*) were the benefits from reduced nitrogen (N) and phosphorous (P) discharge to recipient water bodies due to increased treatment efficiency; and B_{other} (*SEK*) were other benefits.

Health benefits (B_{health}) for A1-A4 were calculated as:

$$B_{health} = \Delta QALYs \cdot QALY_B \cdot DWC_t \qquad (15)$$

where *ΔQALYs* (*QALYs*) were the QALYs gained per person in year *t* in relation to the reference alternative (*QALYs A-ref*) for each mitigation alternative; $QALY_B$ (*SEK/QALY*) was the monetary value per QALY; and *DWC* (*persons*) was the number of drinking water consumers in year *t*. The value of a *QALY* is further discussed in the sensitivity analysis, Section 3.4.

Environmental benefits for A1-A3 (in A4, environmental benefits were assumed to be zero) were calculated as:

$$B_{Environmental} = N \cdot SEK_N + Php \cdot SEK_P \qquad (16)$$

where N (*kg*) was the increased nitrogen removal; SEK_N (*SEK/kg*) was the monetary value per kg nitrogen removed; Php (*kg*) was the increased phosphorus removal; and SEK_P (*SEK/kg*) was the monetary value per kg phosphorous removed.

Other benefits (B_{other}) were not monetised using quantitative measures. However, to illustrate the importance of these benefits, an analysis of how large they need to be to produce a positive *NPV* was conducted within the sensitivity analysis.

Investment costs were added to the first year of the CBA. For A1-A3, the investment costs ($C_{Investments}$, *SEK*) were calculated as:

$$C_{Investments} = C_{WWTP} + C_{Pump} \cdot Pumps + C_{Con\_WWTP} \cdot OWTSs + C_{Pipe} \cdot WP \qquad (17)$$

where C_{WWTP} (*SEK*) was the cost for expanding the WWTP; C_{Pump} *(SEK)* was the cost per pump; *Pumps* (*#*) was the number of pumps needed; C_{Con_WWTP} (*SEK*) was the connection cost per OWTS; *OWTSs* (*#*) was the number of OWTS connected; C_{Pipe} (*SEK/m*) was the cost per pipe meter; and *WP* (*m*) was the pipe length for each alternative.

Table 2 Inputs for the cost-benefit analysis.

Input	Unit	Abbr.	Distr. [m]	Parameters
		General input		
Time horizon	y	T	PV	100
Discount rate	%	r	PV	High value=3.5, Low value=1%
OWTSs connected	#	$OWTSs$	PV	A1=621, A2=1,240, A3=1864, A4=0
Persons per OWTS	#	P_{OWTS}	PV	2.5 (as reported in Table 1)
Wastewater pipe length (A1) [a]	m	WP_{A1}	T	Min=8*10^3, Mode=8.8*10^3, Max=9.5*10^3
Wastewater pipe length (A2) [a]	m	WP_{A2}	T	Min=1.4*10^4, Mode=1.5*10^4, Max=1.6*10^4
Wastewater pipe length (A3) [a]	m	WP_{A3}	T	Min=2.8*10^4, Mode=3*10^4, Max=3.2*10^4
Drinking water consumers	#	DWC	PV	330,000
Population increase	#/year	PI	PV	2300
		Investment cost		
Number of pumps required [b]	Pumps	$Pumps$	PV	A1=1, A2=10, A3=20, A4=0
Cost for expanding WWTP (A1) [a]	SEK	C_{WWTP_A1}	LN	P05=5*10^6, P95=6*10^6, Location=0
Cost for expanding WWTP (A2) [a]	SEK	C_{WWTP_A2}	LN	P05=10*10^6, P95=11*10^6, Location=0
Cost for expanding WWTP (A3) [a]	SEK	C_{WWTP_A3}	LN	P05=1.6*10^7, P95=1.7*10^7, Location=0
Cost per pump installation [b]	SEK	C_{Pump}	N	P05=4.5*10^4, P95=5.5*10^4, Location=0
Cost per meter pipe [b]	SEK/m	C_{Pipe}	LN	P05=3,150, P95=3,850, Location=0
Cost for installing UV treatm. [c]	SEK	C_{UV}	LN	P05=5.3*10^7, P95=5.7*10^7, Location=0
Cost for connecting to WWTP [i]	SEK/OWTS	C_{Con_WWTP}	PV	1.29*10^5
		Annual costs		
Water use per person per day [e]	m^3/p/y	WU	PV	58.4
Cost per m^3 water used [f]	SEK/m^3	C_{Water}	PV	26.36
Cost for water services [f]	SEK/y/Con	C_{Con_Year}	PV	2,792
Cost for OWTS maintenance [g]	SEK/y	C_{Sludge}	PV	1,118
Lifetime of an OWTS [h]	y	$OWTS_{Life}$	PV	25
Cost for re-investing in OWTS [b]	SEK	C_{OWTS_RI}	PV	1.3*10^5
Cost for UV treatm. maintenance [d]	SEK/y	$C_{UVmaint.}$	LN	P05=4.2*10^5, P95=4.4*10^5, Location=0
		Benefits		
Benefit/QALY avoided (High) [j]	SEK/QALY	$QALY_{B\_H}$	PV	1.22*10^6
Benefit/QALY avoided (Low) [j]	SEK/QALY	$QALY_{B\_L}$	PV	7*10^5
Benefit per kg N avoided [k]	SEK/N	SEK_N	PV	22.91
Benefit per kg P avoided [k]	SEK/N	SEK_P	PV	53.06
Increase, N removal/year (A1) [l]	kg N/y	N_{A1}	T	Min=3,550, Mode=4,000, Max=4,450
Increase, N removal/year (A2) [l]	kg N/y	N_{A2}	T	Min=7,100, Mode=8,000, Max=7,900
Increase, N removal/year (A3) [l]	kg N/y	N_{A3}	T	Min=10,700, Mode=12,000, Max=13,300
Increase P removal/year (A1) [l]	kg P/y	Php_{A1}	T	Min=580, Mode=650, Max=720
Increase P removal/year (A2) [l]	kg P/y	Php_{A2}	T	Min=1,160, Mode=1,300, Max=1,440
Increase P removal/year (A3) [l]	kg P/y	Php_{A3}	T	Min=1,750, Mode=2,000, Max=2,150

a) Total wastewater pipe length and cost for expanding the municipal WWTP for each alternative was derived from personal communication with P. Fröjd at Sjöbo municipality and by using expert judgements.
b) Cost per pump, amount of pumps and the pipe cost per meter were based on Swedish literature (Kärrman et al. 2012).
c) Cost for installing UV-treatment was based on personal communication with B.M. Pott at Southern Sweden Water Supply (Sydvatten)
d) Based on the cost per litre for medium sized drinking water treatment plants (Cotton et al. 2001).
e) Åström and Johansson (2015)
f) Since stormwater is not included, the fee for connecting OWTS to the municipal WWTP was reduced (Sjöbo Municipality 2016b).
g) Sludge removal cost (968 SEK/year) (Sjöbo Municipality 2016a) and electricity cost (150 SEK/year) (expert judgement).
h) Wastewater guide (2016)
i) Connection fee (101,450 SEK) (Sjöbo Municipality 2016b), application fee (2,550 SEK) (Sjöbo Municipality 2016c), and excavation and plumbing on own property (25,000 SEK) (expert judgement).
j) Svensson et al. (2015)
k) SEPA (2008a)
l) Based on: 41% non-functioning (zero reduction) OWTSs (SEPA 2004); triangular distributions representing the nitrogen (Min=20, Mode=30, Max=40) and phosphorous (Min=60, Mode=70, Max=80) percentage removal in OWTSs (SEPA 2015); point values estimating the nitrogen (70) (SEPA 2017) and phosphorous (96) (SEPA 2013) percentage removal in WWTP. Triangular distributions were derived using MC simulations.
m) PV=point value, T=triangular distribution, LN=LogNormal distribution, N=Normal distribution

Annual costs (*ΔC_{Annual}*, *SEK*) for A1-A3 when connected to the municipal WWTP in relation to having an OWTS were calculated as:

$$\Delta C_{Annual} = C_{Annual\_WWTS} - C_{Annual\_OWTS} \quad (18)$$

where C_{Annual_WWTP} (*SEK*) was the annual cost per property when connected to the municipal WWTP; and C_{Annual_OWTS} (*SEK*) was the annual cost per property when having an OWTS.

The C_{Annual_WWTP} was calculated as:

$$C_{Annual\_WWTP} = C_{Water} \cdot WU \cdot P_{OWTS} \cdot OWTSs + C_{Con\_Year} \cdot OWTSs \quad (19)$$

where C_{Water} (*SEK/m*^{3}) was the cost for water use; *WU* (*m*^{3}*/p/year*) was the water use per person and year; P_{OWTS} (persons) was the number of persons per OWTS; OWTSs (*#*) was the number of OWTSs connected to the WWTP; and C_{Con_Year} (*SEK/year/OWTS*) was the annual connection fee per OWTS per year.

The C_{Annual_OWTS} was calculated as:

$$C_{Annual\_OWTS} = C_{Sludge} \cdot OWTSs + \frac{C_{\mathrm{Re}inv} \cdot OWTSs}{OWTS_{Life}} \quad (20)$$

where C_{Sludge} (*SEK/OWTS*) was the annual cost for sludge removal per OWTS per year; C_{Reinv} (*SEK*) was the cost for re-investing in a new OWTS; and $OWTS_{Life}$ (*years*) was the expected life time of an OWTS.

For A4, the investment cost was the installation of UV treatment (C_{UV}, *SEK*), and the annual cost was the maintenance of the UV treatment ($C_{UVmaint}$, *SEK*).

A distributional analysis was performed by assigning costs and benefits to private OWTS owners, drinking water consumers, or inhabitants/visitors of the catchment of Lake Vomb.

3.4 Uncertainty and sensitivity analysis

To acknowledge epistemic and aleatory uncertainties, MC simulations were used in the QMRA and the CBA calculations. The model was divided into the following compartments: source characterisation; water quality modelling in the DWTP; dose-response in the QMRA; and the CBA. An adaptation of the local sensitivity analysis, which investigates the change in output by varying one input variable at a time, keeping all other input variables constant, as suggested by Schijven et al. (2013), was used. For the compartments with monotonic behaviour i.e. source characterisation, water quality modelling in the DWTP, and the CBA, the Spearman's rank correlation analysis was used (Mokhtari and Frey 2005). For the compartment with non-monotonic behaviour, i.e. dose-response in QMRA, scatter plots were used (Frey and Patil 2002). All results from the uncertainty and sensitivity analysis are reported in the supplementary material.

For investigating the uncertainties not suitable to model using probability distributions, scenarios were defined. Scenarios were used for the different values of a QALY, discount rate, and the proportion of OWTSs contribution to the total pathogen load. The low and high values of a QALY were assumed to be 700,000 and 1,220,000 SEK based on the government implied willingness to pay for a QALY (Svensson et al. 2015). For the discount rate, two scenarios were used: 1 % or 3.5 %. The Swedish Transport Administration recommends 3.5 % for infrastructure projects in the traffic sector (STA 2016); this value is commonly used in other sectors as well. Since the time horizon was long and decisions affect several future generations, a low discount rate was also used. The pathogen load to Lake Vomb from OWTSs (L_{OWTS}, *pathogens/day*) was calculated as:

$$L_{OWTS} = C_{Tributary} \cdot F_{Tributary} \qquad (21)$$

The total pathogen load (L_{total}, *pathogens/day*) originates from many sources (e.g. WWTP, fertilisation using manure, grazing animals, wild animals) and was illustrated as:

$$L_{total} = L_{OWTS} + L_{other} \tag{22}$$

where L_{other} (*pathogen/day*) was the load from all other pathogen sources. Since the OWTSs load to Lake Vomb in relation to the total pathogen load was unknown, two scenarios (L_{OWTS} = 75 % and L_{OWTS} = 50 % of L_{total}) were investigated.

To estimate in what range the non-monetised benefits in A1-A3 would have to be in order to render a positive *NPV*, a calculation of non-monetised benefits to reach break-even ($NPV \geq 0$) was performed.

3.5 Software

For the source characterisation and CBA calculations, the MC simulations were performed using *Microsoft Excel*, *@RISK* version 7.5.1. For the drinking water treatment performance, virus groundwater transport model and the dose-response relationship, the MC simulations were performed using *Analytica* release 4.1.6.30. For the hydrodynamic modelling, the model for Lake Vomb was developed using *MIKE 3 FM* (*MIKE Powered by DHI*), which is a deterministic three-dimensional numerical model that solves the incompressible Reynolds averaged Navier-Stokes equations invoking the assumptions of Boussinesq and hydrostatic pressure (DHI 2011). The period 2005-2007 was simulated using the observed hydrometeorological data.

Uncertainties were propagated between the different model compartments to calculate the probability distributions of the final results of the CBA. Using 10,000 MC iterations, the resulting probability distribution of the output of one model compartment was then used as an input in the next model compartment. The propagation of uncertainties and the combination of methods are illustrated in Figure 3.

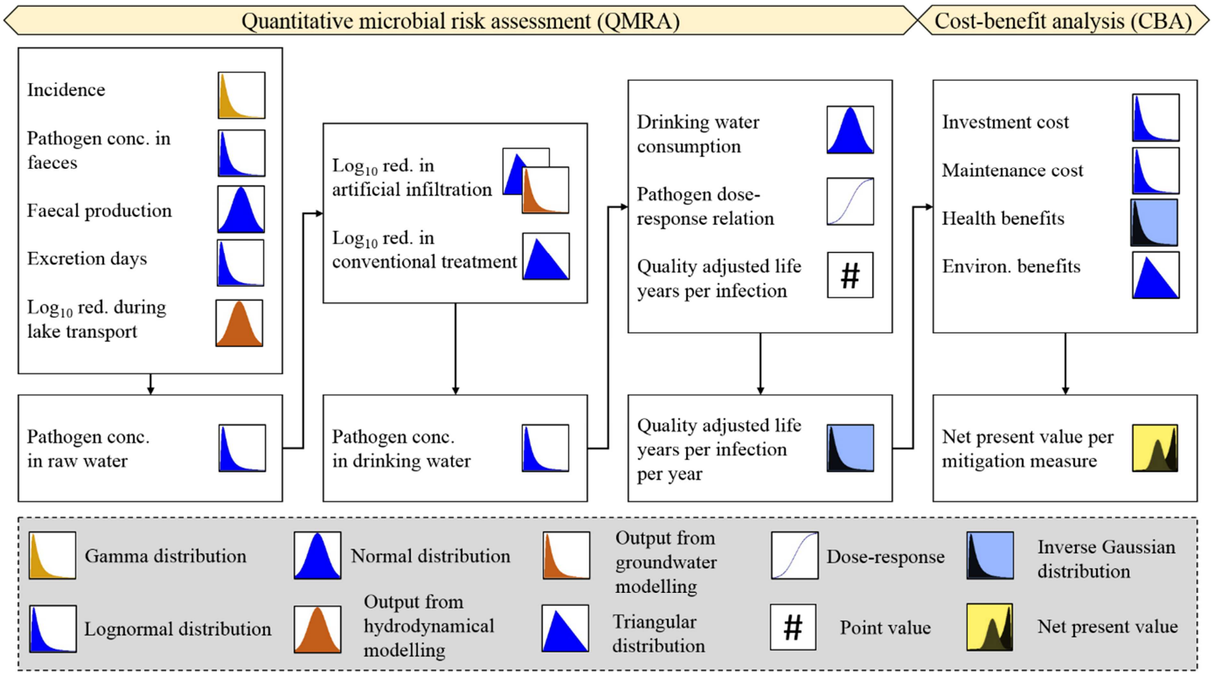

Figure 3 Schematic illustration of how the different methods are combined in the model.

4 Results

All results, unless stated otherwise, are from the scenario assuming that OWTSs are contributing 75 % of the total pathogen load to Lake Vomb. The complete results from the source characterisation (Table S1), dose-response (Table S2), CBA (Table S3), and uncertainty and sensitivity analysis (Tables S4-S8, Figures S1-S5) are presented in the supplementary material.

4.1 Cost-benefit analysis

The costs, benefits and *NPV* for the scenario with a high value (1,220,000 SEK) of a QALY and a discount rate of 3.5 % are presented (Figure 4) for the 5th, 50th, and 95th percentiles.

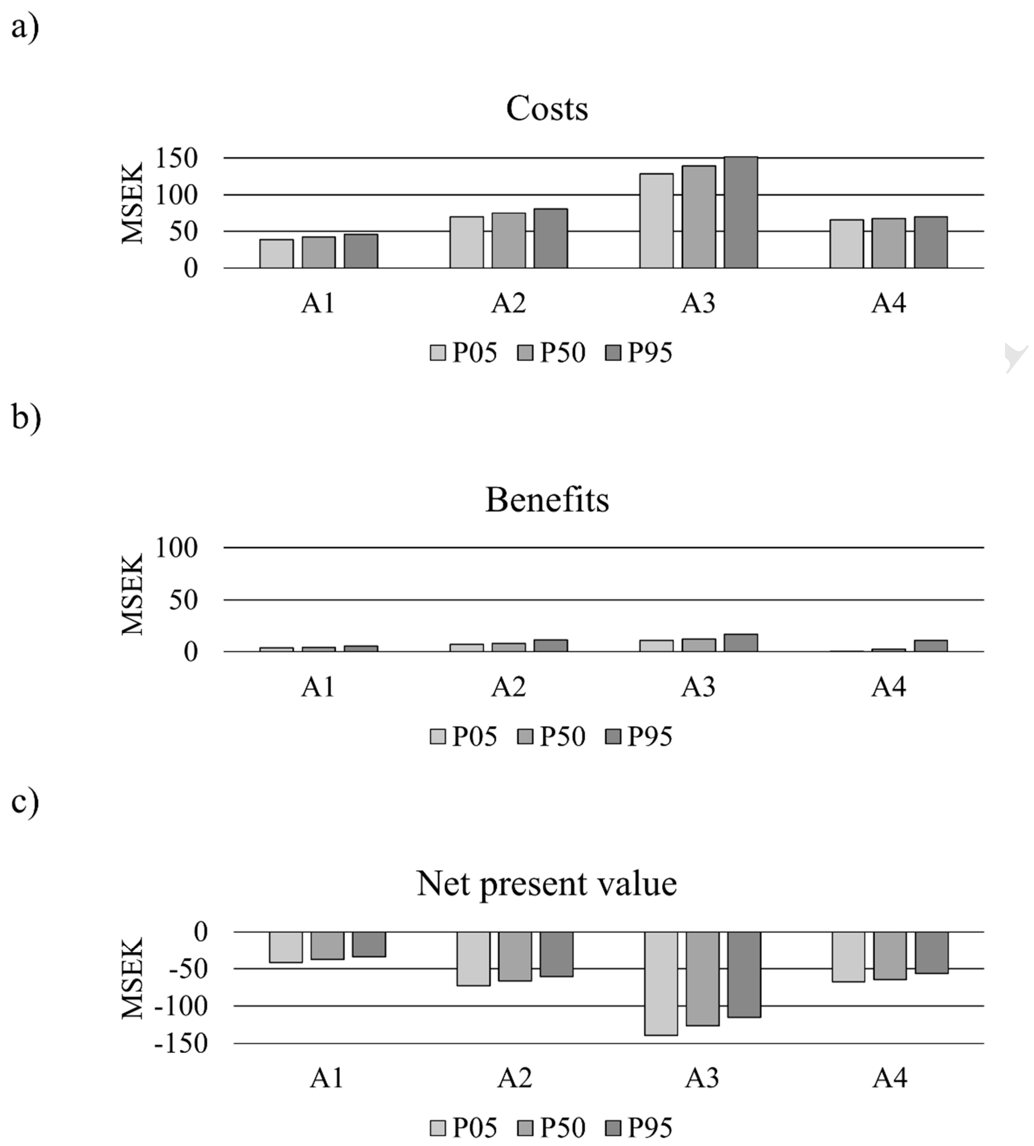

Figure 4. The costs (a), benefits (b) and NPV (c) for the scenario with the high value of a QALY (1,220,000 SEK) and the discount rate of 3.5 % are presented for the 5th, 50th, and 95th percentiles in million SEK (MSEK).

For A1-A3, the costs would be solely taken by the owner of the OWTS that will be connected to the WWTP (installation of pipes on their own property and a connection fee). For A4, the costs would be solely taken by the drinking water producers initially and eventually by the drinking water consumers through a higher drinking water fee.

The benefits were distributed between the drinking water consumers (health benefits) and the inhabitants and visitors of the catchment area of Lake Vomb (environmental benefits). For A1-A3, the drinking water consumers received 11 or 18 % and the inhabitants and visitors received 89 or 82 % of the benefits, using a low or high valuation of a QALY, respectively. For A4, the benefits were solely attributed to the drinking water consumers.

4.2 Uncertainty and sensitivity analysis

For A1-A3 using 3.5 % discount rate, the *NPV* sensitivity to the inputs in the CBA were (top three in descending order): cost per pipe meter (C_{Pipe}), wastewater pipe length (*WP*), and $\Delta QALYs$ (A1-A3). For A4 using 3.5 % discount rate, the *NPV* sensitivity to the inputs in the CBA were (top three in descending order): $\Delta QALYs$ (A4), cost for installation of UV treatment (C_{UV}) and cost for maintenance of UV treatment ($C_{UVmaint}$).

The concentration of norovirus in raw water (C_{RW_Noro}) was the most sensitive to the following inputs (top three in descending order): concentration in faeces (*C*), incidence (I_{Noro}), and days excreting (D_{Noro}). The concentration of *Campylobacter* in raw water (C_{RW_Camp}) was the most sensitive to the following inputs (top three in descending order): concentration in faeces (*C*), Log_{10} reduction in Björkaån (R_{Bjork_Camp}), and Log_{10} reduction in Torpsbäcken (R_{Torp_Camp}). The concentration of *Cryptosporidium* in raw water (C_{RW_Cryp}) was the most sensitive to the following inputs (top three in descending order): concentration in faeces (*C*), days excreting (D_{Cryp}), and incidence (I_{Cryp}).

The concentration of norovirus in drinking water (C_{DW_Noro}) was the most sensitive to the foellowing inputs (top three in descending order): Log_{10} reduction in saturated zone (R_{SZ_noro}), raw water concentration (C_{RW_Noro}), and Log_{10} reduction per meter in unsaturated zone (R_{UZ/m_Noro}). The concentrations of *Campylobacter* (C_{DW_Camp}) and *Cryptosporidium* (C_{DW_Cryp}) in drinking water were the most sensitive to the following inputs (top three in descending order): Log_{10} reduction per meter in saturated zone (R_{SZ_Camp} and R_{SZ_Crypp}), saturated zone vertical length (L_{SZ}), and raw water concentration (C_{RW_Camp} and C_{RW_Cryp}).

The dose-response relationship between the concentrations of pathogens in drinking water (C_{DW}), infectivity (m), drinking water consumption (WI) and the probability of infection (P_{inf}) was illustrated using scatter plots from the 10,000 MC simulations.

A scenario-based analysis was performed to analyse the effects on the final *NPVs* from uncertainties regarding the QALY valuation, discount rate, and the OWTSs contribution to the total pathogen load. The rank order of the 50th percentiles for the *NPV* (A1>A4>A2>A3) does not change depending on the level of OWTSs contribution to the total pathogen load nor the QALY valuation. However, with a low discount rate (1 %), the rank order changes to A1>A2>A4>A3.

Benefits that have not yet been monetised and included in the CBA that might alter the rank order of *NPV* for the alternatives were identified. For alternatives A1-A3, non-monetised benefits are:

- positive health effects for humans from improved water quality for recreational activities in Lake Vomb;
- positive health effect for animals (both domestic and wild) from improved water quality in the catchment and in Lake Vomb;
- perceived value for private OWTS owners not being responsible for treating their wastewater;
- increased market value of the properties connected to the municipal water and wastewater system;
- benefits of removing the possible risk of direct contamination of private wells by OWTSs;

- possibility to recycle nutrients when wastewater is treated at the WWTP;
- reduction of CO_2 emission when sludge transportation trucks do not need to empty closed tanks and three compartment septic tanks;
- reduced traffic accidents and related risks since heavy traffic is reduced in the catchment area.

For alternative A4, non-monetised benefits are:

- less disinfection by-products due to lower dosage in chlorination;
- reduced handling and storage of chlorination chemicals.

For A1-A3, these additional benefits need to be 800-1200 SEK or 1800-2400 SEK per OWTS per year for 1% and 3.5% discount rate, respectively, to give a positive *NPV* (50^{th} percentile). These ranges apply for both the high and low valuation of a QALY.

5 Discussion

The aim of this study was to present an approach for comprehensive decision analysis using CBA of microbial risk mitigation measures in DWSs, and including Lake Vomb as a case to illustrate the assumptions needed and the associated variabilities and uncertainties. Below we discuss the QMRA, the CBA, the uncertainties, and the overall applicability of the decision model.

5.1 Quantitative microbial risk assessment

5.1.1 Source characterisation

Pathogen concentration in faeces and the pathogen excretion duration are subject to large variability. In this study, it was assumed that the catchment was large enough to have pathogens present at all times, evenly distributed geographically. However, if a smaller catchment is to be described, it will be important to account for temporal and geographical variations of the pathogen prevalence. If persons are infected, there will be high pathogen concentrations in the OWTSs effluents, otherwise there will be no pathogens present. One way forward is to combine the data on incidence with binary probability density functions. This would capture the on/off characteristics of infections and enable the use of the decision model on smaller systems, even on a single OWTS.

To acknowledge the ambiguity (a factor of 100 between values) and the lack of information in the underreporting factor, it was assigned a point value and not included in the MC simulations. However, the factor for underreporting is uncertain, and further investigations on how to describe this input need to be conducted. The factor for underreporting is important, since increased underreporting results in a corresponding increase of the estimation of the pathogen concentration at the raw water intake.

The estimated pathogen load to Lake Vomb can be validated. The estimated concentrations of *Cryptosporidium* in the tributaries (0.36-1.4 oocysts/L) in this study are in agreement with the values reported by other studies, e.g. the mean of 0.62 oocysts/L in an Australian river (Swaffer et al. 2014). The estimated concentrations could also be validated by monitoring the local pathogen concentrations in the catchment; however, this is tedious and expensive. Instead, based on the factor for underreporting and the incidence of norovirus (since it was the pathogen causing the main part of the loss of QALYs), we made an estimate of the annual infections in the drinking water consumer population, confirming that the waterborne infections only represented a small proportion of the total infections calculated from the incidence. Chosen values and associated probability distributions should be regarded as a possible, but not necessarily the optimal, representation of the pathogen source characteristics.

5.1.2 Water quality modelling

The $\log_{10}$ reduction during transport in Lake Vomb was estimated using hydrodynamic modelling encompassing several years of daily and sub-daily variation in hydrometeorological data. Looking at the best fit, a normal distribution was reasonable to use for describing the variability in pathogen reduction. Future development of water quality modelling within the decision model is to include probabilistic modelling. Probabilistic modelling will further facilitate risk-based modelling approaches, QMRA and holistic water resource management (Oliver et al. 2016).

The model describing the artificial groundwater recharge system was highly simplified. The pathogen reduction was based on a conceptual model describing the artificial groundwater infiltration as one system, when in reality there are many smaller sub-systems with complex flow and transport conditions between different infiltration ponds and abstraction wells. Nonetheless, the model is assumed to give a good understanding of the key processes affecting the level of reduction in the artificial infiltration.

Local investigations of the barrier efficiency at the DWTP would be preferred. Since it is not ethical to use active DWSs to directly test the reduction of pathogens, surrogate organisms can be used instead. It is also possible to use literature estimates. The Log_{10} reduction of *Cryptosporidium* by the UV-disinfection was not allowed to be higher than 3 Log_{10}, in order not to interpret results outside of the investigated range (Hijnen et al. 2006). Although this can be considered a low reduction given the efficiency of UV-disinfection towards *Cryptosporidium*, this approach is used in the QMRA-tool for drinking water producers in Sweden. However, the truncation in UV-treatment needs to be further investigated and thoroughly reviewed. Investigation of altering the UV-dose may also be of importance for future implementation of the decision model.

5.1.3 Dose-response

The estimated annual probability of infection (P_{annual}) was slightly higher than the WHO guidelines in the current situation (A-ref, 50th percentile), while the *DALYs* were under the threshold (A-ref, 5th percentile), indicating that there is ambiguity whether the microbial risks were acceptable or not. The large uncertainty and variability described in input probability distributions should be taken under consideration when interpreting the results. To get below the WHO guideline for P_{annual} looking at the 50th percentile, A2, A3 and A4 are the possible options, while A1 almost reaches the threshold. It is only A4 that meets the guideline level with respect to the 95th percentile. Even though no strict guideline level exists in Sweden, drinking water producers should be aware of the discrepancy between meeting the *DALY* or P_{annual} WHO guideline. The same pathogen concentration in drinking water can meet one target and miss the other.

P_{annual} was calculated into to QALYs using standard unit values adopted from a study from the U.S (Batz et al. 2014). It may be argued that results from the U.S. are not representative for Swedish settings. Even so, to illustrate the methodology, it was assumed that the U.S. values would be useful. However, further development of the model could use more detailed health effect quantification and implement local studies for estimating the quantity of the health risk reduction in the risk mitigation alternatives. To monetise the health effects, there are other approaches which can be implemented into the model, e.g. information from previous events, quality of life investigations, etc.

5.2 Cost-benefit analysis

None of the mitigation alternative rendered a positive *NPV*. However, the *NPV* results must be interpreted using a wider perspective in combination with other results from the CBA, such as distributional analysis and non-monetised benefits. From a socio-economic perspective, it is important to identify the alternative with the least negative *NPV* (A1). In a situation where decision makers are required to reduce the microbial risk, they will need to choose an alternative. Such a situation would occur e.g. if there is a guideline or risk level that needs to be achieved, such as the WHO recommendation of a maximum yearly probability of infection per person of 10^{-4} or a maximum DALY of 10^{-6} (WHO 2011). If looking at the 50th percentile with 1 % discount rate, both A1 and A2 resulted in higher *NPVs* than A4. Nevertheless, as noted above, only A4 would achieve the WHO recommendation of the P_{annual} with a high degree of certainty (looking at the 95th percentile).

When monetising health benefits, it is important to make sure that the underlying valuation study represents the relevant health effects. The monetisation of health benefits was based on a governmental implied willingness to pay for a QALY (Svensson et al. 2015). The values used were estimated from a societal perspective, i.e. the effects both within the health care sector (e.g. reduced medical and hospitalisation costs) and beyond the health care sector (e.g. reduced discomfort from being ill and loss of production) were accounted for.

When decision makers choose an alternative, they also accept the distribution between beneficiaries and payers associated with the decision. Even though the Kaldor-Hicks criterion[3] can be argued, the distribution of the costs and benefits will need to be communicated with stakeholders. Alternative A4 is the only alternative when the beneficiary and the payer are the same stakeholder. In decision making, distributional analysis can be of importance when applying the polluter pays principle.

5.3 Uncertainty and sensitivity analysis

The Spearman's rank correlation is inadequate (Ellouze et al. 2010) for measuring sensitivity when analysing complex relationships such as the dose-response relation in QMRA. We have used scatter plots to illustrate the relationships between drinking water pathogen concentration, drinking water consumption and the infectivity. Future research needs to investigate more advanced sensitivity analysis methods (see e.g. Mokhtari and Frey 2005).

[3] The Kaldor-Hicks criterion, simply put, state that: beneficiaries can compensate those that pay or experience negative effects. However, the compensation only needs to be possible and not realized, since it is argued that if a decision is societally profitable, the costs and benefits will eventually even out with regard to different stakeholders.

Since the total risk level in the drinking water system cannot be estimated, it was important to investigate whether the results change if the OWTSs contribution to the total risk is altered. Results showed that changing the OWTSs contribution to the total pathogen load did not change the ranking of the alternatives.

After scrutinising the CBA results, decision-makers need to consider benefits that were omitted from the monetised analysis. Even though the *NPVs* were negative for the A1-A3 alternatives, all alternatives could render a positive *NPV* (looking at the 50th percentile) if these other benefits could be valued in the range of 800-2400 SEK per connected OWTS per year. The value of the benefits when using a 3.5 % discount rate need to be approximately 1000 SEK higher than when using a 1 % discount rate.

Some factors vary over time both within a year, e.g. incidence and water flow etc., and over longer time periods, e.g. population increase, climate change etc., to mention a few. The model included a population increase based on population projections for Sweden in general. However, the inter-yearly variations have not been included. For further development of the decision model, methods for including these temporal variations and uncertainties need to be developed.

5.4 Risk-based decision model

Depending on the type of decision and the local settings, other methods than presented in this paper can be more suitable to combine in the decision model. For decisions aiming at reaching a certain guideline or threshold value, a CEA may be preferred, instead of a CBA. CBA represents a strict anthropocentric and utilitarian context, only accounting for benefits attributed to human values (Hutton 2001). If decision makers want to include intrinsic values, they need to apply methods that can consider such values as well, such as multi-criteria decision analysis (see e.g. DCLG 2009). In such multi-criteria decision models, the decision support rendered from the CBA and the QMRA can be used as input for appropriate criteria. To give some examples, the *NPV* can provide information to the economic dimension, distributional analysis and QALY assessment can provide input to the social dimension, and water quality modelling can provide input to the environmental dimension in a sustainability assessment, see e.g. Rosén et al. (2015).

The focus of this study was to describe the methodology of comparing microbial risk mitigation measures using CBA in combination with QMRA to estimate risk levels and the effect of possible mitigation measures. Benefits, in terms of the health risk reduction obtained in each alternative were described in detail. Environmental benefits were included using a more simplified approach. However, 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. These additional benefits may be of substantial importance and heavily affect the final decision.

The decision model incorporates both aleatory and epistemic uncertainties in the input probability distributions. To further develop the model and to provide additional decision support, these uncertainties can be divided. This separation would also facilitate additional decision analyses, e.g. value of information analysis.

6 Conclusions

Results from the case study showed that the alternative to connect the smallest proportion (25 %) of on-site wastewater treatment systems to the wastewater treatment plant (A1) at Lake Vomb was the most societally beneficial. However, the only alternative that would reduce the annual probability of infection to meet the WHO guidelines with a high degree of certainty (95th percentile) was installing UV-disinfection (A4). In relation to the development of the risk-based decision model, the following conclusions were drawn:

- The developed decision model is flexible and can be tailored to different drinking water systems and different types of decision problems.
- To implement the decision model, a multitude of uncertainties and variabilities needs to be addressed. However, the model provides tools to include these variabilities and uncertainties in a structured manner.
- Through the process of performing the cost-benefit analysis, aspects important for decision making that may otherwise easily be overlooked or ignored are openly displayed and assessed.
- The combination of quantitative microbial risk assessment and cost-benefit analysis provides a novel decision model that creates transparent and holistic decision support tool for microbial risk mitigation.
- For improvement of the decision model, we suggest to further develop the valuation and monetisation of health effects and the propagation of variability and uncertainty between the included methods.

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- A decision model for drinking water context, combining QMRA and CBA, was developed.
- This flexible model can be tailored to different systems and decision problems.
- The microbial risk mitigation measures were compared in a Swedish case-study.
- Microbial risk reduction was measured in QALYs and monetised.
- This novel decision model provides transparent and holistic decision support.