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Citation for the original published paper (version of record):
Bergion, V., Sokolova, E., Samuelsson, A. et al (2025). Modelling the combined impacts of climate change and socio-economic development on waterborne pathogen transport. Water Research, 283. http://dx.doi.org/10.1016/j.watres.2025.123802

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Water Research 283 (2025) 123802

Contents lists available at ScienceDirect

Water Research

journal homepage: www.elsevier.com/locate/watres

Modelling the combined impacts of climate change and socio-economic development on waterborne pathogen transport

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ARTICLE INFO

Keywords:
Drinking water
Hydrological model
Hydrodynamic model
Microbial water quality
Representative concentration pathways
Shared socio-economic pathways

ABSTRACT

Protection of our water resources is essential to provide future generations with safe drinking water, recreational opportunities, and reliable ecosystem services. Climate and land use changes exert pressure on the quality of our water resources. Additionally, societal development may generate both positive and negative impacts on future water quality. Thus, decisions made today will impact the water quality of tomorrow. In this paper, we address the issue of future microbial water quality by combining Representative Concentration Pathways and Shared Socio-economic Pathways with projections of societal development, either downscaled to a local level or assessed by local organisations. We use Lake Vomb in Sweden (providing approximately 330 000 persons with drinking water) to illustrate our novel approach of assessing the impact of climate change and societal development on future microbial water quality. The approach includes norovirus, *Escherichia coli* (as an indicator organism), and *Cryptosporidium*. Further, we combine hydrological and hydrodynamic fate and transport modelling to simulate future water quality in the tributaries and at the drinking water intake. Future simulations are compared to a baseline scenario representing the current situation. Results show that climate change will reduce future water quality. However, we can also see that societal development significantly impacts microbial water quality, potentially counteracting the increases in microbial concentrations induced by climate change. Therefore, drinking water supply management must adapt to both future climate and societal development.

1. Introduction

Climate change, characterized by its shifting precipitation patterns and rising temperatures, directly influences the quantity (Hagemann et al., 2013) and quality (Arheimer et al., 2005; Delpla et al., 2009) of our water sources. Paired with this environmental transformation is an ongoing societal development and population growth, which exacerbates climate change and also has a direct bearing on our water sources (Arnell, 2004). This impact is predominantly detrimental, as evidenced by increased anthropogenic pollution and water use, along with the negative effects of land use changes (Mello et al., 2020). At the same time, societal progress can also have beneficial outcomes, such as enhanced wastewater treatment and positive behavioural changes of decreased chemical use and proper disposal of pharmaceuticals (Kotchen et al., 2009).

Microbial water quality is affected by the changes in water transport patterns caused by increased precipitation due to climate change (Hunter, 2003) and by the changes in the pollution sources influenced by societal development (Islam et al., 2018). The changes in population and future behavioural trends related to water and food consumption will impact the extent of potential pathogen load in water sources emanating from wastewater (Demeter et al., 2021) and from agricultural activities (Coffey et al., 2020). Rising temperatures will also affect the survival of some microorganisms (Hellberg and Chu, 2016). While it is impossible to predict the exact trajectory of societal development, scenarios can furnish a framework that allows us to acknowledge and prepare for this uncertain and multifaceted future (Delpla and Rodriguez, 2014; Islam et al., 2018).

The Intergovernmental Panel on Climate Change (IPCC) uses Representative Concentration Pathways (RCPs) to describe the uncertainties related to future climate change projections. In addition to the RCP scenarios, IPCC's work on climate change encompasses future societal developments on a global scale – the global Shared Socio-economic Pathways (SSPs) (IPCC, 2023). The global SSP scenarios have been adapted to continental (Europe) applications (Kok et al., 2019), the Barents region (Nilsson et al., 2017, 2015), and the Baltic Sea

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https://doi.org/10.1016/j.watres.2025.123802
Received 11 October 2024; Received in revised form 20 April 2025; Accepted 9 May 2025
Available online 10 May 2025

(Zandersen et al., 2019), as well as on national levels, e.g., New Zealand (Frame et al., 2018) and Japan (Chen et al., 2020). Additionally, SSP scenarios have been utilized in the context of drinking water-related fields, such as agriculture (Mitter et al., 2020), surface water quality and quantity (Dau and Adeloye, 2021; Islam et al., 2018; Jin et al., 2018; Wijngaard et al., 2018), groundwater quantity (Buhay Bucton et al., 2022; Pinsri et al., 2022), water systems (Alizadeh et al., 2022; Wang et al., 2022), and health (Rohat, 2018). Furthermore, investigations into the impacts of climate change combined with land use changes have been conducted without the use of SSP scenarios for two small watersheds in Ireland (Coffey et al., 2015), and a review of potential futures for urban water management has been carried out by Sivagurunathan et al. (2022). While it is possible to adapt the global SSP scenarios to local contexts using literature or by engaging local stakeholders (Nilsson et al., 2017), there are no specific recommendations in Sweden on how to accomplish this.

The storyline for the SSP1, based on van Vuuren et al. (2017), was described as "Green growth" aligning with the concept of sustainable development. In SSP1, no actions are taken in terms of political climate initiatives. Nonetheless, the greenhouse gas emissions are mitigated, mainly due to efficiency, technological development, and reduction in overall consumption. Land use change is governed by protection and restoration of natural areas, promoting reforestation. Population growth is low. Efficiency in agriculture, reduced waste, and reduced impact from food consumption enables transition of agricultural land into forests and urban areas.

The storyline for SSP5, based on Kriegler et al. (2017), was described as a high emission pathway called "Fossil-fuelled development" with unmitigated climate change. The rapid societal development is done using fossil fuel resources. Rapid development and an increasingly globalized economy provide inclusion and increased capability to adapt to climate change. Land use change is unregulated to a large extent, but a decline in deforestation is projected. Consumption overall is high, and the impact from food consumption is high due to a meat-rich diet. Population growth is high in the OECD countries, but on a global level, the population peaks at 2050 and then declines to 2100. Additionally, increase in migration from low-income countries to high-income countries is expected, resulting in a high population growth in Sweden.

SSP scenarios have been downscaled and used for water quality modelling of nutrients (e.g., nitrogen, phosphorous) (Jin et al., 2018; Mack et al., 2019; Molina-Navarro et al., 2018) as well as microbial pollution (Hofstra and Vermeulen, 2016; Iqbal et al., 2019; Islam et al., 2018). Modelling water quality can assist in assessing the impacts of pollution sources on water supply, environmental protection, and the risk assessment under the SSP scenarios (Iqbal et al., 2019; Islam et al., 2018; Jin et al., 2018). Combining the SSP scenarios with local water quality modelling to be used as a tool for predicting future microbial risks has the potential to inform local water utilities on decisions related to long term planning for water resources.

The aim of this paper was to investigate the effects of projected future climate conditions and socio-economic development on the microbial water quality in Lake Vomb in Sweden, a drinking water source for 330 000 consumers. The objectives were:

- to formulate scenarios representing the individual and combined effects of climate change and socio-economic development on the microbial water quality in Lake Vomb for two time periods, 2040–2050 and 2090–2100, based on Representative Concentration Pathways (RCPs) and Shared Socio-economic Pathways (SSPs).
- to model the effects of the formulated scenarios on the fate and transport of the pathogens *Cryptosporidium* and norovirus and the faecal indicator bacteria *Escherichia coli* in the catchment of and in Lake Vomb combining a hydrological model (ArcSWAT) and a hydrodynamic model (MIKE 3 FM).

The main scientific contribution of this paper was that we adapted the SSP scenarios to a small-scale drinking water catchment (423 km^2) and modeled the impact of microbial pollution sources under SSP and RCP scenarios (individually and in combination). The novel aspects of this study was to include several microbial pollutants, to encompass a wide range of potential microbial risks to drinking water, and employing both hydrological and hydrodynamic modelling, to represent the entire pathway from pollution source to the drinking water intake.

2. Methods

2.1. Study area

Lake Vomb is situated in the south parts of Sweden and is an essential part of the water supply system providing drinking water to the larger metropolitan area of the City of Malmö (Fig. 1). The land use (cropland 70 %, grazing 12 %, forest 17%, urban areas 1%) in the catchment area (423 km^2) surrounding the lake is dominated by agricultural activities. The main locality is Sjöbo with approximately 8400 inhabitants, and there are several additional smaller communities with 300–1500 inhabitants each. In the catchment, there are approximately 2500 on-site wastewater treatment systems (OWTSs) and six wastewater treatment plants (WWTPs) with 100–650 persons connected to each. There are three main tributaries, Björkaån, Torpsbäcken, and Borstbäcken, that have average flows of 2.3, 0.31, and 0.28 m^3/s respectively (Magnusson et al., 2008). Apart from human health risks posed to the drinking water consumers, additional health risks stem from the use of the lake for recreation. However, in this study, the main focus was on the microbial risks in the context of drinking water supply.

2.2. Simulated scenarios based on projected future changes according to RCP and SSP

The IPCC AR6 (IPCC, 2023) emphasises an integrated scenario analysis reflected in the establishment of five specific core combinations of RCP and SSP scenarios. To evaluate different future scenarios regarding the impact of land use, societal development, and population growth on microbial pollution, the IPCC scenario approach using a combination of RCP and SSP was used. The scenarios were chosen primarily based on the RCP projections, with the best-case (RCP 2.6) and the worst-case (RCP 8.5) scenarios selected to encompass the possible range. Guided by the suggested combinations of RCP and SSP (IPCC, 2023), two combinations – SSP1+RCP2.6 and SSP5+RCP8.5 – were chosen and simulated for the future time periods of 2040–2050 and 2090–2100, representing mid-term and long-term projections respectively. In addition to the combined scenarios, each RCP scenario was simulated separately, to distinguish the contribution of climate change factors only. The future scenarios were compared to a baseline scenario – the period 2009–2018 – representing the current situation. The model calibration and validation were limited by the available daily river flow observations for the period 1st of January 2008 – 17th of March 17, 2018. We assume that 10-year periods reasonably capture the interannual variability of hydrometeorological conditions. Thus, we also used 10-year periods for simulating future scenarios. Hydrological modelling was conducted for all scenarios, while the hydrodynamic modelling was performed for the most extreme scenarios. The SSP scenarios were based on the IPCC AR6 (IPCC, 2023), while the available meteorological data (air temperature and precipitation) for projected climate change from the Swedish Meteorological and Hydrological Institute (SMHI, 2024) were based on the IPC AR5 (IPCC, 2014).

The RCP scenarios were based on SMHIs regionalisation of the global model ensembles (SMHI, 2024). The regional projections are available for two separate time periods, the middle of the century (2041–2070) and the end of the century (2070–2100). In Sweden, the average daily temperature increases relative to the reference period (1971–2000) are estimated be 2.0 C° by 2041–2070 and 1.9 C° by 2071–2100 for RCP2.6, while for RCP8.5, the corresponding increases are estimated to be 3.2 C°

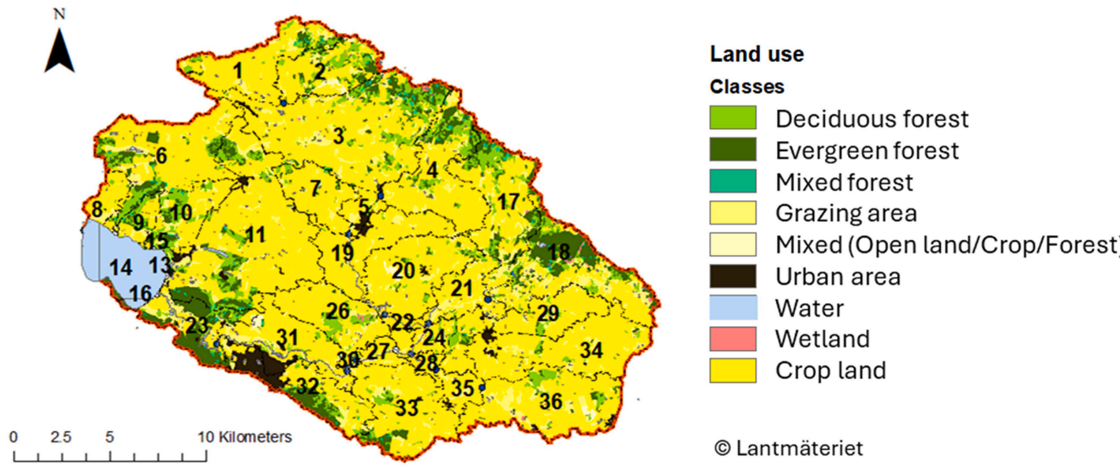

Fig. 1. Lake Vomb catchment area with land use information and location of the 36 subbasins.

and 5.0 C° (SMHI, 2025). For precipitation, the average monthly increase is estimated to be 5 mm for RCP2.6 (both time periods), and for RCP8.5, the increases are 8 mm and 13 mm for the time periods 2041–2070 and 2071–2100 respectively (SMHI, 2025).

In this paper, the SSP scenarios have been adapted to a local context using literature sources, including national, regional, and municipal planning documents and information, as well as projected quantification of population, livestock, and land use changes from the International Institute for Applied System Analysis IIASA for SSP1 and SSP5. Additionally, persons involved in water management in the local area were consulted. In Table 1, the assumed societal developments pertaining to the future water quality are described together with the baseline scenario (detailed explanation and motivation are provided in the Supporting Information (SI)). The RCP scenarios (without accounting for the SSP projections) were simulated using the settings for the baseline scenario (Table 1) with updated meteorological inputs, as described in section 1.4 in the SI.

2.3. Hydrological model

To estimate the microbial contribution (*E. coli* and *Cryptosporidium*) from agricultural land, the Soil and Water Assessment Tool (SWAT) was applied (Arnold et al., 2012). SWAT is a semi-distributed deterministic hydrological model that includes a microbial transport module to account for microbial pollutants in runoff. The meteorological input data and hydrological water flow data for the SWAT model were obtained from (SMHI, 2023), the Geological Survey of Sweden (SGU, 2023) (soil data), and the National Land Survey (NLS, 2023) (digital elevation model and land use data). Meteorological data included precipitation (daily) obtained from Sjöbo station (53450), humidity (daily average based on hourly data), wind speed (daily average based on hourly data), and temperature (daily min and max) from Hörby A station (53530), and solar radiation from Lund station (53445). Sjöbo station is located within Lake Vomb catchment, while Hörby A and Lund stations are located outside the catchment, approximately 7 km north and 20 km

Table 1
The baseline (2009–2018) and the assumed future conditions in the different scenarios: SSP1+RCP2.6 for 2040–2050 (SSP1_RCP26_40) and for 2090–2100 (SSP1_RCP26_90), SSP5+RCP8.5 for 2040–2050 (SSP5_RCP8.5_40) and for 2090–2100 (SSP5_RCP85_90). Further details regarding the values in this table can be found in the Supplementary Information.

	Baseline	SSP1_RCP26_40	SSP1_RCP26_90	SSP5_RCP85_40	SSP5_RCP85_90
Land use (ha [% rel. baseline])[a]					
Crop land	28,627	−94.320 [−0.3 %]	−129.61 [−0.5 %]	0 [0 %]	−844.34 [−3 %]
Pasture	5114.9	−94.320 [−2 %]	−129.61 [−3 %]	−126.27 [−2 %]	−126.27 [−2 %]
Urban area	471.33	+188.64 [+40 %]	+259.23 [+55 %]	0 [0 %]	0 [0 %]
Forest	6940.2	0 [0 %]	0 [0 %]	126.27 [−2 %]	970.61 [+14 %]
Technological development[b]					
OWTS[c]					
Removal - all organisms	0.72 Log10	1.00 Log10	2.00 Log10	2.00 Log10	3.00 Log10
WWTP[d]					
Removal - *E. coli*	2.37 Log10	3.37 Log10	4.37 Log10	3.37 Log10	4.37 Log10
Removal - *Cryptosporidium*	1.38 Log10	1.70 Log10	2.40 Log10	1.70 Log10	2.40 Log10
Removal - Norovirus	1.02 Log10	1.50 Log10	2.00 Log10	1.50 Log10	2.00 Log10
Water					
Water use (L person^{-1} day^{-1})	140	140	130	130	120
Manure storage					
Removal - *E. coli*	No removal	2.00 Log10	3.00 Log10	2.00 Log10	2.00 Log10
Removal - *Cryptosporidium*	No removal	2.00 Log10	3.00 Log10	2.00 Log10	2.00 Log10
Agricultural and human populations (% rel. baseline)					
Livestock	Table S3	−6 %	−15 %	+11 %	−6 %
PE[e] in WWTP and OWTS	Table S2	+28 %	+61 %	+44 %	+144 %

[a] The baseline column provides the initial area in hectares of the different land-use and the SSP1+RCP2.6 and SSP5+RCP8.5 columns provide the change for each scenario (hectares [% relative Baseline]).
[b] The Log10 removal and water use are reported as absolute numbers, i.e., not as change relative the baseline scenario.
[c] On-site wastewater treatment system.
[d] Wastewater treatment plant.
[e] Person equivalent, i.e. population that is contributing to the wastewater load in the area.

west of the catchment, respectively.

To adjust for altered meteorological conditions in the RCP scenarios the precipitation and air temperature inputs were changed. The change in average monthly precipitation and temperatures (mean, minimum, and maximum) between the baseline period (2009–2018) and time periods for the future scenarios (2040–2050 and 2090–2100) were used to define model input, similar to the procedures in Mohammed et al. (2019). The changes were divided into four separate seasonal periods representing winter, spring, summer and fall (December-February, March-May, June-August and September-November). Daily precipitation was altered using the change in percent, while the temperatures were adjusted using the change in absolute values. Detailed information is presented in Section 1.4 in the Supplementary Information.

The model was calibrated and validated for the baseline scenario using available flow measurements (2009–2017) obtained from SMHI for the water gauge station in Eggelstad (2125), which was active until February 2018. The calibration (2009–2013) and validation (2014–2017) for monthly simulated water flow was conducted using the SWAT-Cup 5.1.6.2 Generalized Likelihood Uncertainty Estimation (GLUE) approach with 1000 iterations, and the resulting Nash-Sutcliff model efficiency coefficient (NSE) was 0.83 for both calibration and validation. The model parameterisation (presented in Tables S7 and S8) obtained from the best iteration from the monthly calibration was used to validate the daily simulated water flow. The validation (2014–2017) of daily simulated water flow resulted in an NSE of 0.59. Additionally, we performed water flow simulations using hourly precipitation data from the Hörby A station (53530). However, as this gauge is located outside the catchment and no corresponding high-resolution flow observations are available for calibration or validation, the results are included for illustrative purposes only and are not used in further analyses (Figure S2).

The SWAT hydrological model simulated the spread of microbial pollutants from livestock grazing and fertilisation with manure. Livestock statistics (SBA, 2022) were available for four years (2010, 2013, 2016, and 2020), and the average value for these four years was assumed to be representative for the entire baseline period (2009–2018). Microbial pollutants originating from livestock were *E. coli* (faecal indicator) and zoonotic *Cryptosporidium.* Based on literature (Sokolova et al., 2018) and communication with the Swedish Veterinary Agency, zoonotic *Cryptosporidium, C. parvum*, was only assumed to be present in calves under 1 year with a 10 % prevalence and an oocyst concentration of 2.04×10^7 oocysts/g in manure. Furthermore, Swedish Veterinary Agency (SVA, 2021) reports that at Swedish farms where *C. parvum* is prevalent in cattle, the shedding animals are mainly between 0–6 weeks. As these young animals reside primarily indoors, it was assumed that all shed zoonotic *Cryptosporidium* oocysts were added to the manure storage and not on grazing areas. The number of animals in the catchment and assumptions regarding *E. coli* and *Cryptosporidium* excretion can be found in Tables S3 and S6. Manure from grazing animal types was assumed to enter the catchment either when animals were grazing on pasture (Table S4) (using the grazing management operation in the SWAT model) or when stored manure (mixed from all animal types) from winter housing was spread on crop land (Table S5) (using the fertiliser application management operation in the SWAT model). Animals not grazing, as well as grazing animals during winter housing, were assumed to contribute to the total accumulated mixed manure that was spread for fertilisation. Wild animals were not accounted for in the model due to limited information.

There is no module in SWAT for adding specific pathogen loads from OWTSs. Coffey et al. (2010) assumed that each OWTS can be represented by a hectare where fertilising operation in SWAT is used for microbial pollutant contribution. In this study, wastewater from WWTPs and OWTSs was instead added as point source contributions to each of the three tributaries, as in Bergion et al. (2018). This approach was chosen due to short transportation time within the catchment resulting in very low microbial reduction in the tributaries (Sundahl et al., 2008). The point source contributions from OWTSs and WWTPs were assumed to enter the lake directly and were added to the microbial input to the hydrodynamic modelling. Note that the WWTP contribution only includes the regular effluent, and the potential overflow events were not considered in this study. The amount of wastewater was based on the number of persons connected to OWTSs and WWTPs in the tributaries' catchments. Each OWTS was assumed to serve on average 2.5 persons and each person was assumed to contribute 0.14 m^3 wastewater per day. Number of persons connected to the WWTPs and OWTSs are presented in Table S2. The WWTPs were all located in the catchment of the Björkaån, while the OWTSs were divided among all three tributaries. Microbial pollutants in wastewater were *E. coli* (indicator organism), *Cryptosporidium,* and norovirus. Microbial concentrations in wastewater effluent from WWTPs in the baseline scenario were assumed to be 2.83×10^5 number/L, 5.46 number/L, and 7.13×10^4 number/L for norovirus, *Cryptosporidium,* and *E. coli,* respectively, based on studies from Sweden and Norway (Table S1). Microbial concentrations in wastewater effluent from OWTS were assumed to be 3.23×10^5 number/L, 2.50×10^1 number/L, and 3.18×10^6 number/L for norovirus, *Cryptosporidium,* and *E. coli,* respectively, based on the literature values for WWTP influent (Table S1 in Supplementary Information) and assuming a 0.72 Log10 reduction (Bergion et al., 2018).

2.4. Hydrodynamic model

To simulate the water circulation in Lake Vomb, a three-dimensional time-dependent hydrodynamic model MIKE 3 FM (MIKE Powered by DHI) was used. The MIKE 3 FM model is based on the numerical solution of three-dimensional incompressible Reynolds averaged Navier–Stokes equations using Boussinesq and hydrostatic assumptions. The model consists of continuity, momentum, temperature, salinity, and density equations, and is closed using a turbulent closure scheme. The hydrodynamic model set-up for Lake Vomb was previously described in detail in Sokolova et al. (2018). The modelling domain was described using a flexible computational mesh consisting of triangles in the horizontal direction (253 nodes and 426 elements). Vertically, the lake was approximated by layers; the top part of the lake down to −1 m was described using two layers with thickness varying depending on the water level, while the other layers had a fixed thickness of 1 m. The model was set up to account for the inflows to and outflows from the lake, for hydrometeorological conditions, and to simulate the heat exchange with the atmosphere. The water density was formulated as a function of temperature.

For the baseline scenario (2009–2018), the water flows and water temperature in the tributaries were specified using the output from the S-HYPE model provided by SMHI through the Vattenwebb tool (https://vattenwebb.smhi.se/). The initial conditions and the downstream boundary conditions in the lake were specified using the measured daily data for water level at the outflow from the lake obtained from Kävlingeån Water Conservation Organisation. Records of ice coverage (data from SMHI) at Lake Vomb stopped in season 2013/2014, therefore the data for another lake in the vicinity, Lake Östra Ringsjön, were used to describe ice coverage afterwards. The meteorological data were obtained from SMHI: the daily precipitation data from the station Vomb (station 53410), and the hourly data for wind (speed and direction), air temperature, relative humidity, and cloudiness from the station Hörby A (53530).

For the future scenarios, the water flows in the tributaries were specified using the output from the SWAT model. The water temperature in the tributaries was described as zero gradient. It was assumed that the lake is not covered by ice in the future. Inputs for future precipitation and average air temperature were the same as for the hydrological model.

The hydrodynamic model was validated by plotting the simulated and observed surface water temperature (Figure S3).

To simulate the microbial fate and transport in Lake Vomb, the microbial water quality model ECO Lab was coupled to the hydrodynamic model of the lake. The microbial concentrations in the tributaries were specified using the output from the SWAT model. The microbial decay was described using the first order decay equation with the decay coefficients of 0.57 day^{-1} for *E. coli* (Sokolova et al., 2012), and 0.03 day^{-1} for *Cryptosporidium* and norovirus as was assumed earlier in Bergion et al. 2018. *Cryptosporidium* oocysts were assumed to be unattached to particles and to settle with the settling velocity of 0.03 m day^{-1} (Medema et al., 1998) as was used in Bergion et al. (2018).

3. Results and discussion

3.1. Baseline conditions

Simulated water flow (monthly mean) (Figure S1b) shows that the lowest water flows occur during the summer periods. This highlights an inverted relationship between water flow and microbial concentrations, indicating that high water flows dilute the microbial concentrations in the tributaries (Fig. 2a). When looking at simulated and measured daily water flow, there are some peaks during the summer, but the highest peaks are observed during the fall/winter period (Figure S1a). In some years, a clear snowmelt peak occurs (visible both in simulated and measured flows) in spring (e.g., year 2010) (Figure S1); however, due to the lack of snow this phenomenon does not always occur in the southern parts of Sweden where the catchment is located.

The microbial concentrations simulated by the SWAT model for the baseline scenario are presented for the tributary Björkaån in Figure 2; results for the tributaries Torpsbäcken and Borstbäcken are presented in Figures S5 and S6. The concentrations of microorganisms in Lake Vomb at the water intake for the local drinking water treatment plant, simulated by the MIKE3 FM model, are presented in Fig. 3. The tributaries' contribution to the concentrations at the water intake, in decreasing order, was Björkaån, Torpsbäcken, and Borstbäcken, which aligns with the order of magnitude of their water flows. Simulated mean, minimum, and maximum concentrations of microorganisms are reported in Table 2. The concentration of *Cryptosporidium* in the tributaries was highly variable, and during the manure application and grazing season, heavy rains contributed to high concentrations on specific days (Fig. 2c). These variations are levelled out by the lake processes, resulting in a concentration pattern of *Cryptosporidium* at the water intake that is more similar to that of other microorganisms (Fig. 3c).

In an earlier study of Lake Vomb, a simulated mean of 0.1 Log10 #

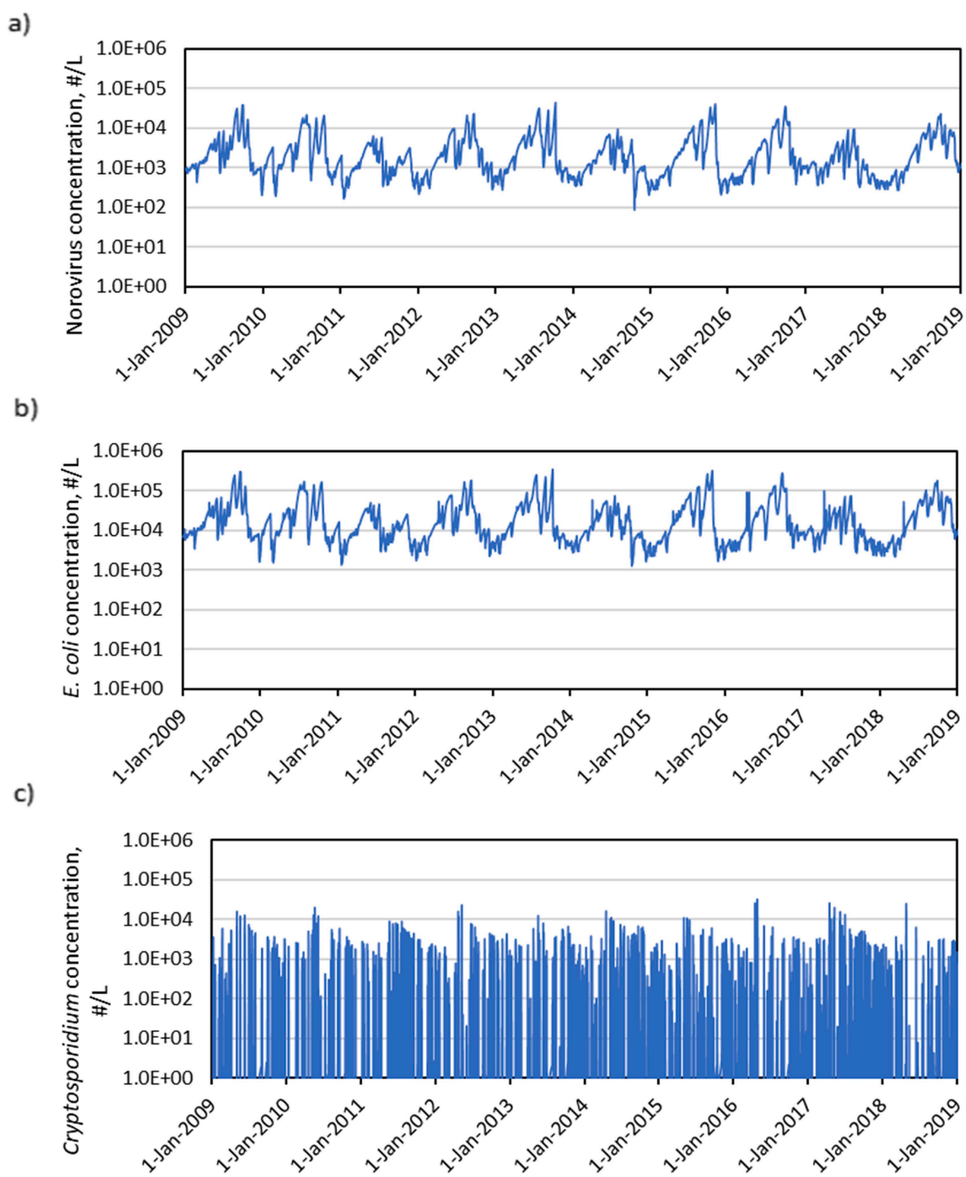

Fig. 2. Concentrations of a) norovirus, b) *E. coli*, and c) Cryptosporidium in the tributary Björkaån as simulated by the hydrological model for the baseline scenario.

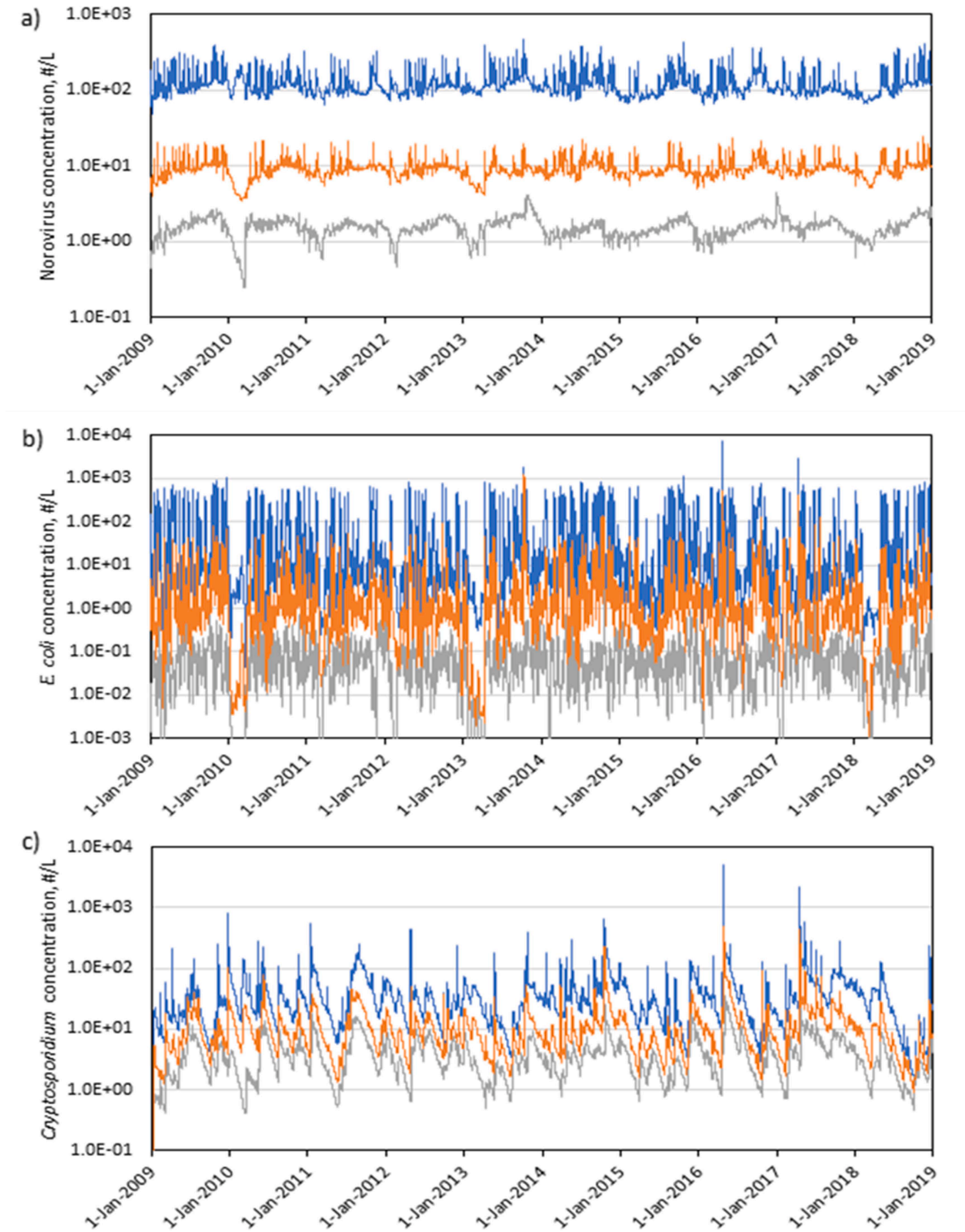

Fig. 3. Concentrations of a) norovirus, b) *E. coli*, and c) Cryptosporidium at the water intake in Lake Vomb – impact of the tributaries Björkaån (blue), Torpsbäcken (orange), and Borstbäcken (grey), as simulated by the hydrodynamic model for the baseline scenario.

Cryptosporidium oocysts/L and 1.5 Log10 # norovirus gene copies (GC)/L were reported for Björkaån (Bergion et al., 2018). Another previous study reported *E. coli* concentrations (min, max, mean) in Björkaån of (1.8, 5.2, 3.9 Log10 #/L) measured as ssrA gene copies using qPCR (Chuquimia et al., 2020). Chuquimia et al. (2020) also reported a mean *E. coli* concentration analysed with culture method of 2.8 Log10 CFU/L (mean of three sampling occasions for all three tributaries). The microbial concentrations in our study were high compared to earlier studies in the Lake Vomb area. However, looking at the routine water quality measurements (summer periods 2020–2024) at a recreational swimming site in Lake Vomb (located 1 km from the Björkaån outlet and 1.7 km from the Torpsbäcken outlet), the range (min, max, mean) of *E. coli* concentrations (<1.0, 4.2, 2.9 Log10 #/L) was more similar to our results, especially considering the die-off and dilution effects of Lake Vomb. The high simulated microbial concentrations in our study may stem from assumptions related to the microbial inputs, e.g., the OWTS and the WWTP concentrations were based on measured concentrations in wastewater from large municipal WWTPs, which may not be representative for the wastewater stemming from OWTSs and small WWTPs in the study area. All WWTPs in the Vomb catchment area had less than 650 connected person equivalents. The order of magnitude (greatest to smallest) of the source contribution to the microbial concentrations in

Table 2
Simulated microbial concentrations in the tributaries to and at the water intake in Lake Vomb.

Tributary	*E. coli*, Log_{10} #/L mean (min – max)	*Cryptosporidium*, Log_{10} #/L mean (min – max)	Norovirus, Log_{10} #/L mean (min – max)
Björkaån	4.4 (3.1 – 5.5)	2.6 (−2.0 – 4.5)	3.5 (1.9 – 4.6)
Torpsbäcken	4.7 (2.9 – 5.9)	2.7 (−2.2 – 4.6)	3.7 (1.8 – 4.9)
Borstbäcken	4.2 (3.4 – 5.5)	2.7 (−2.7 – 4.5)	3.2 (1.1 – 4.5)
Lake Vomb (at intake)	1.9 (−1.2 – 3.7)	1.8 (−2.5 – 3.5)	2.1 (1.7 – 2.6)

Björkaån was as follows (Figure S7): *Cryptosporidium* (Fertilisation – OWTSs – WWTPs), *E. coli* (OWTSs – Fertilisation – WWTPs – Grazing), and norovirus (OWTSs – WWTPs). The order of magnitude is valid for all seasons except for *E. coli* during summer, when the contribution from WWTPs was greater than from fertilisation (Figure S7). Looking at the total microbial load to the lake from all tributaries, rather than the concentrations in the tributaries, the order of magnitude (greatest to smallest) of the source contribution changed only for *E. coli*: OWTSs – WWTPs – Fertilisation – Grazing (Table S13). In summary, the main sources of *E. coli* and norovirus were OWTSs and WWTPs, while the main source of *Cryptosporidium* was fertilisation. There were no larger seasonal variations in the source contribution.

3.2. Effect of future scenarios

The Log10 change in the microbial load to Lake Vomb from the tributaries (Fig. 4, a and b) and the change in the concentrations at the water intake (Fig. 4, c and d) are shown for the 50th and 95th percentiles, based on simulated results from the SWAT hydrological model and the MIKE3 FM hydrodynamic model, respectively. The RCP scenarios, without SSP scenarios, indicated that the impact of climate change on the microbial load from the tributaries will primarily increase extreme microbial loads (95th percentile, Fig. 4b). Although there were slight increases in the microbial load to the lake for the 50th percentile, this is not visible in the diagram (Fig. 4a). For the microbial concentration at the water intake, there was an increase for both the 50th and 95th percentiles by the end of the century (Figs. 4c and 4d). The increases in microbial load to the lake for the RCP scenarios were mainly for *E. coli* and *Cryptosporidium*, and not norovirus (Fig. 4b). This can be attributed to increased precipitation that, combined with grazing and manure application activities in the area, increased the microbial content in runoff from agricultural areas (norovirus is not prevalent) and the number of runoff events.

The SSP scenarios indicate that, based on the assumptions in the future scenarios (Table 1), the development of society may counteract the effects of climate change and reduce the microbial load from the tributaries and the microbial concentrations at the water intake in the lake (Fig. 4). In the case of SSP5, although the population increase is larger than in SSP1, reduction in water use and, consequently, less wastewater, combined with increased removal of microorganisms in both OWTSs (the dominant source of wastewater in the area) and WWTPs, leads to approximately 0.5–1 higher Log10 reduction of the microbial load and concentrations in the lake compared to SSP1 (Fig. 4). In both SSP1 and SSP5 scenarios, there was generally a greater reduction in *Cryptosporidium* 95th percentiles compared to *E. coli* and norovirus loads to the lake (Fig. 4); however, this difference is less pronounced in SSP5. The greater reduction in *Cryptosporidium* load arises from the assumption that there was a trend of reducing livestock in the area and that manure treatment (2 Log10 units) was introduced both for SSP1 and SSP5 in 2040, and for SSP1 microbial removal in manure increased to 3 Log10 units in 2090 (Table 1). In SSP5, there is an increase of livestock during the period of 2040–2050 followed by a reduction in 2090–2100 (Table 1).

Human health risks in relationship to the SSP scenarios are posed by both recreational activities in the Lake and consumption of drinking water produced from the lake water. However, for the drinking water, in addition to the microbial barriers in the drinking water treatment plant, there are barrier effects within Lake Vomb, i.e., the decrease of concentrations due to in-lake processes (dilution and microbial decay) before the pollutant reaches the water intake. These barrier effects and the pathogen reduction at the drinking water intake were calculated for Borstbäcken, Torpsbäcken, and Björkaån (Table 3) for the baseline and

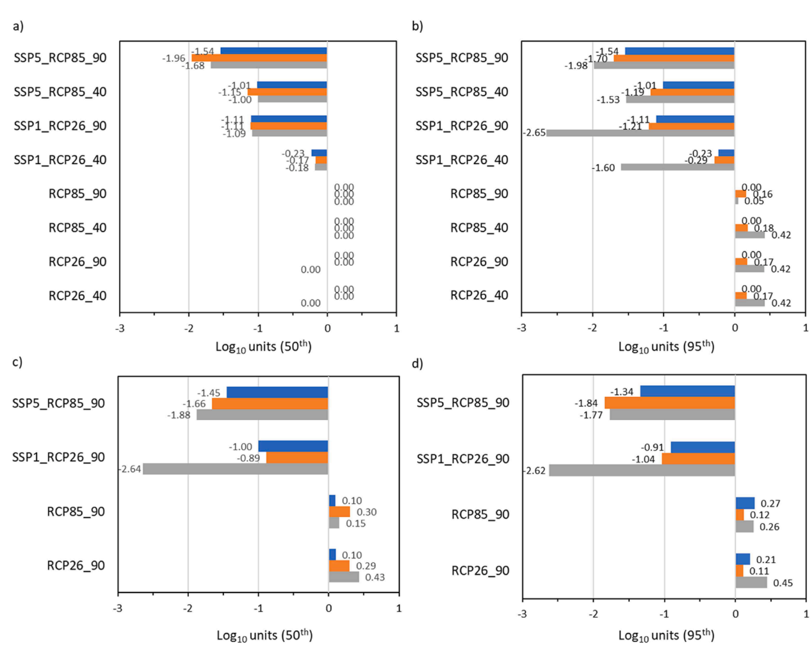

Fig. 4. Simulated future changes presented as the difference between each future scenario and the baseline, expressed in Log_{10} units: 50th (a) and 95th (b) percentiles of microbial load from the tributaries to Lake Vomb as simulated using the hydrological model, and 50th (c) and 95th (d) percentiles of microbial concentration at the water intake in Lake Vomb as simulated using the hydrodynamic model. Norovirus, *E. coli*, and Cryptosporidium are represented by blue, orange, and grey, respectively.

Table 3
Climate change effects on Lake Vomb function as a microbial barrier between tributary inflow and drinking water intake (due to in-lake processes) in terms of Log10 reduction of the concentrations. Future scenarios, in which the season with maximum reduction changes in comparison to the baseline scenario, are marked in grey.

	Baseline			
	Mean	Seasonal Min – Max (Range)	Min	Max
Borstbäcken				
Norovirus	3.02	2.10 – 3.24 (1.14)	Winter	Fall
E. coli	5.25	4.30 – 5.66 (1.33)	Winter	Fall
Cryptosporidium	2.06	1.73 – 2.33 (0.40)	Winter	Spring
Torpsbäcken				
Norovirus	2.75	1.89 – 3.00 (1.11)	Winter	Fall
E. coli	4.02	3.28 – 4.41 (1.13)	Winter	Fall
Cryptosporidium	1.65	1.46 – 1.78 (0.32)	Winter	Spring
Björkaån				
Norovirus	1.45	0.84 – 1.68 (0.82)	Winter	Summer
E. coli	2.61	2.06 – 2.97 (0.91)	Winter	Summer
Cryptosporidium	1.00	0.80 – 1.18 (0.38)	Winter	Spring
	RCP26_90			
	Mean (ΔBaseline)	Seasonal Min – Max (Range)	Min	Max
Borstbäcken				
Norovirus	2.83 (−0.19)	1.62 – 3.46 (1.84)	Winter	Fall
E. coli	5.13 (−0.12)	4.21 – 5.86 (1.65)	Winter	Summer
Cryptosporidium	1.84 (−0.22)	1.62 – 1.99 (0.37)	Winter	Spring
Torpsbäcken				
Norovirus	2.91 (+0.16)	1.82 – 3.24 (1.42)	Winter	Fall
E. coli	4.09 (+0.09)	3.09 – 4.46 (1.37)	Winter	Summer
Cryptosporidium	1.42 (−0.23)	1.36 – 1.47 (0.11)	Winter	Spring
Björkaån				
Norovirus	1.98 (+0.44)	0.85 – 2.42 (1.57)	Winter	Summer
E. coli	3.14 (+0.53)	2.08 – 3.71 (1.63)	Winter	Summer
Cryptosporidium	1.01 (+0.01)	0.80 – 1.16 (0.36)	Winter	Spring
	RCP85_90			
	Mean (ΔBaseline)	Seasonal Min – Max (Range)	Min	Max
Borstbäcken				
Norovirus	2.99 (−0.03)	1.63 – 3.53 (1.90)	Winter	Summer
E. coli	5.26 (+0.01)	4.14 – 5.95 (1.81)	Winter	Summer
Cryptosporidium	1.85 (−0.21)	1.40 – 2.15 (0.75)	Winter	Spring
Torpsbäcken				
Norovirus	3.00 (+0.25)	1.72 – 3.36 (1.64)	Winter	Summer
E. coli	4.15 (+0.13)	3.00 – 4.57 (1.57)	Winter	Summer
Cryptosporidium	1.43 (−0.22)	1.15 – 1.55 (0.40)	Winter	Fall
Björkaån				
Norovirus	2.09 (+0.64)	0.74 – 2.59 (1.85)	Winter	Summer
E. coli	3.25 (+0.64)	1.98 – 3.88 (1.90)	Winter	Summer
Cryptosporidium	0.80 (−0.20)	0.40 – 0.97 (0.57)	Winter	Spring

the 2090–2100 projections as this period contained the most extreme scenarios. For the contribution from Björkaån and Torpsbäcken, the barrier effects were generally stronger in the RCP scenarios in comparison to baseline, with few exceptions (Table 3). For the contribution from Borstbäcken, the barrier effects were generally weaker in the RCP scenarios in comparison to baseline (Table 3). It should also be noted that the barrier effect varies between the different tributaries. In the case of implementing mitigation measures, understanding these geographical differences can aid decision makers in ensuring that the effect of the measure is optimised. The ranges of the seasonal variations in barrier effects were larger for the RCP-scenarios compared to the baseline simulations. The minimum barrier effects were reduced in the RCP scenarios, with few exceptions, and the maximum barrier effects increased for almost all scenarios and microbial contaminants (Table 3). These increases in the ranges and changes of the min/max of barrier effects indicate that due to climate change effects, there will be a larger uncertainty and increased variability in the microbial concentrations at the drinking water intake in the future. These uncertainties, possible barrier effects, and the pathogen removal at pollution source need to be considered by the municipal drinking water utilities in their long-term planning to safeguard the health of the drinking water consumers.

In literature, SSP1 has often been combined with RCP4.5 (Iqbal et al., 2019; Islam et al., 2018) to examine climate projections. In this study, we aimed to compare the best- and worst-case scenarios. The best-case scenario for Sweden would involve a combination of RCP2.6 and SSP1. In the Swedish context, SSP5 is the only scenario that leads to the climate projection RCP8.5, making it the worst-case scenario. Future research using combinations of RCPs and SSPs should consider the ongoing assessment of actual climate change and adjust the combinations accordingly. It is important to note that political decisions and other actions could eliminate the possibility of following the low emission pathways. However, the available input and databases for creating local scenarios are limited, particularly towards the end of the century. Population growth projections for the Lake Vomb area only exist for two decades on a local scale, and land use change projections are only available for certain small areas near the urban areas of the Sjöbo municipality (Sjöbo Municipality, 2022).

The modelled RCP scenarios indicate that climate change will lead to an increase in microbial load to the lake, consistent with other studies (e.g. Coffey et al., 2015). Moreover, when considering the combination of RCP with SSP scenarios, the socio-economic changes are projected to result in a greater reduction of microbial load compared to the increases caused by climate change. Similar results have been reported previously (Iqbal et al., 2019; Islam et al., 2018). Jin et al. (2018) reported that different aspects of socio-economic change can both increase (e.g., increasing populations) and decrease (e.g., improved wastewater treatment) nutrient levels in river water. While climate change may lead to an increase in available surface water in some regions (Hagemann et al., 2013), socio-economic changes can result in an increased demand for drinking water, creating a potential water deficit (Jin et al., 2018; Wijngaard et al., 2018). Therefore, the management of drinking water supply must adapt to both future climate and socio-economic changes.

3.3. Uncertainties and future research

The fact that many variables are difficult to address, and that there will always be remaining uncertainty, makes it important to be transparent with the assumptions and simplifications that have impacted studies (Sivagurunathan et al., 2022). In this study, we have scaled down the global SSP scenarios using literature and projections within different fields (e.g., population increase, technology development, agriculture, land use) related to microbial water quality. We have contacted local water government and other national authorities, but their input has been limited given that they have not yet worked with these questions using this approach. The Swedish Veterinary Agency cannot provide any information regarding the future prevalence of zoonotic pathogens, so this aspect was not included in this study, but could be relevant, if such information becomes available. The adaptation of the global SSP scenarios to a local scale could also involve stakeholders in a structured way (Alizadeh et al., 2022). Stakeholder involvement could facilitate adapting the storyline and provide technical input on aspects that will impact future water quality.

Hazardous events are an essential part of drinking water risk management (Bergion et al., 2021), and patterns and frequency of heavy rainfall events are expected to increase due to climate change in the south parts of Sweden (Bergstrom et al., 2012). Our approach for future meteorological data may underestimate some of the effects on microbial concentrations that are induced by heavy rainfall events. This is because the future data on precipitation were only scaled based on the regional climate models provided by the SMHI. More advanced meteorological modelling (Pfahl et al., 2017) could be used to overcome this. However, such approaches were out of the scope of this paper. Similarly, it may be possible for other catchments to conduct hydrological modelling with a finer temporal resolution to include e.g., sewer overflows (Derx et al., 2023) and achieve a better model performance. The use of finer temporal resolution in hydrological models, as illustrated by the hourly simulations in Figure S2, should ideally be explored in catchments with sub-daily meteorological data and high-resolution flow observations for calibration and validation. However, for the Lake Vomb catchment, sub-daily precipitation data from within the catchment are unavailable (Figure S2 uses data from a station located outside the area), and no sub-daily flow observations exist to support such a high-resolution model setup. As such, it is not recommended to calibrate and validate hydrological models at a temporal resolution different from the simulation time-step (Adla et al., 2019; Setti et al., 2022).

The model performance may also be improved; however, the performance of the hydrological model was evaluated to be satisfactory on a daily basis (NSE = 0.59) and very good on a monthly basis (NSE = 0.83) (N. Moriasi et al., 2015; Pandit et al., 2025). We conclude that this performance is sufficient for the purpose of modelling future microbial concentrations under SSP and RCP scenarios with their associated large uncertainties in assumptions. The simulated flow peaks in Figure S1 are in some cases underestimated, resulting in a conservative estimation of the microbial concentrations. The simulated flow peaks, regardless of how well they match the observed flow, will exceed the threshold value in the hydrological model for initiating run-off microbial transport. Thus, any further increase in water flow would dilute the microbial concentration. Our assumptions on land use development may not be representative, particularly for the end of the century. Currently, we assume that urban land use will only expand from existing urban areas. This expansion is likely to occur adjacent to the already established urban areas in the short projection period 2040–2050, but the projection for 2090–2100 is more uncertain. The fact that the catchment is part of a drinking water source may hinder development and therefore justify the placement of additional urban expansion near existing urban areas. It is also unlikely that new towns will be established. Agricultural land is rarely converted into urban areas due to legislation hindering conversion of agricultural land to other land use types. If such conversion occurs, it is typically done on land of the lowest agricultural quality, which is adjacent to Sjöbo. However, there is an ongoing discussion regarding the protection of natural land areas close to water. These areas have been protected from development for a significant period. The Swedish government (2018–2022) proposed a bill that weakened the restrictions on development near water. The bill was refused by the Swedish Parliament (Riksdagen), although unfolding these discussions may open for future bills aiming to weakening not only the protection of land near water, but also of agricultural land in favour of developing urban areas. Population projections for the year 2070 for both SSP1 (14.6 million) and SSP5 (19.2 million) from the IIASA (IIASA, 2018) exceed the projected population in the end of 2070 from Statistics Sweden (12.7 million) (SCB, 2020). In order to maintain consistency, we have chosen to rely on the assumptions provided by the IIASA.

The improvements in WWTP pathogen removal were motivated by the new EU wastewater directive (Directive, 2024). However, there are no regulations that target the already existing individual OWTSs or small WWTPs (< 1000 PE) unless they are identified as important in a risk assessment. We have motivated the assumed increase in OWTS removal by technology development for both SSP scenarios. For urban water management, there is a need to acknowledge the local conditions on a catchment scale. The high number of OWTSs in the Vomb catchment and the lack of regulation may result in that the future removal in these systems will be less than assumed in these SSP scenarios. If housing restrictions are loosened and the future development leads to more houses connected to OWTSs (especially close to lakes and water courses) in combination with lack of regulation for increased OWTS treatment, the impact on Lake Vomb from wastewater will increase substantially.

4. Conclusions

We conducted a scenario analysis to predict future development and climate change projections for the catchment of Lake Vomb in Sweden. The combination of two models (one hydrological and one hydrodynamic) was used to simulate norovirus, *E. coli,* and *Cryptosporidium* loads to the lake and concentrations in the drinking water intake, to investigate the impact of climate change and socio-economic development on the water quality in Lake Vomb. There are countless possible paths for development that may be explored. In this study, our goal was to outline both the best-case and the worst-case scenarios. To effectively plan for the future, it is crucial to acknowledge the diversity and uncertainty in potential future development projections related to socio-economic change. It is also essential to combine these societal projections with climate projections in order to make informed long-term decisions. By utilizing as much available information as possible, decision-makers can ensure that drinking water supply management is prepared for the challenges ahead. Based on our assumptions concerning socio-economic development, projected climate change, and the results of our water quality modelling, we have drawn the following conclusions:

- The modelling results indicate that the RCP2.6 and RCP8.5 climate scenarios (without considering socio-economic changes) will increase both the microbial loads into Lake Vomb and the microbial concentrations at the water intake with up to 0.5 Log10 units.
- The dominant source of microbial pollution was OWTSs for *E. coli* (as faecal indicator) and norovirus, while for *Cryptosporidium* it was fertilisation.
- The barrier effects in Lake Vomb (reduction in microbial concentrations due to in-lake processes) varied geographically between the tributaries, and the seasonal range of the barrier effects will expand in the future, increasing the uncertainty in microbial concentrations at the drinking water intake. Understanding both geographical and temporal future variations of the barrier effects is important for decision makers to maximise the effects of mitigation measures that may be implemented.
- By incorporating the SSP scenarios and combining them with RCP scenarios, we discovered that projected socio-economic changes have a significant impact on the microbial water quality and can counteract the microbial increases induced by climate change. In the studied scenarios, the resulting net reduction was 1 – 2.5 Log10 units relative the baseline. Thus, if proper mitigation measures are implemented as part of the socio-economic development (e.g., improved manure management and increased microbial removal in WWTPs and OWTSs), it is possible to achieve a net microbial reduction of 1 – 2.5 Log10 units relative to the baseline.
- This study focused on utilising the available data and information easily available in literature and online databases – an approach and structure that may be utilised in other catchments as well. Future studies on locally adapted SSPs are recommended to explore more advanced approaches for creating projections of hydrometeorological data and using stakeholder involvement in formulating the possible societal developments.

Declaration of generative AI and AI-assisted technologies in the writing process

During the preparation of parts of this work, the authors used a large language model GPT-4o by OpenAI to improve readability and language. After using this tool/service, the authors reviewed and edited the content as needed and take full responsibility for the content of the publication.

CRediT authorship contribution statement

V. Bergion: Writing – review & editing, Writing – original draft, Visualization, Validation, Software, Project administration, Methodology, Investigation, Formal analysis, Conceptualization. **E. Sokolova:** Writing – review & editing, Visualization, Validation, Software, Methodology, Funding acquisition, Formal analysis, Conceptualization. **A.**

Samuelsson: Methodology, Conceptualization. **E. Östberg:** Methodology, Conceptualization. **M. Bondelind:** Writing – review & editing, Validation, Methodology, Funding acquisition, Conceptualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgements

We highly acknowledge Josefine Elving and Karin Troell at the Swedish Veterinary Agency for discussions related to our microbial inputs and future societal changes. Part of the study was funded by the collaborative platform on drinking water research (DRICKS), coordinated at the Chalmers University of Technology. Funding was also recieved from the Swedish government research council for sustainable development (Formas), grant number 2017-01413, as part of the research project *ClimAQua – Modelling climate change impacts on microbial risks for a safe and sustainable drinking water system*. We also thank Sydvatten AB for providing data on water temperature in Lake Vomb.

Supplementary materials

Supplementary material associated with this article can be found, in the online version, at doi:10.1016/j.watres.2025.123802.

Data availability

Supplementary material with the data is provided online.

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