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Climate change impact on infection risks during bathing downstream of sewage emissions from CSOs or WWTPs

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ABSTRACT

Climate change is expected to influence infection risks while bathing downstream of sewage emissions from combined sewage overflows (CSOs) or waste water treatment plants (WWTPs) due to changes in pathogen influx, rising temperatures and changing flow rates of the receiving waters. In this study, climate change impacts on the surface water concentrations of *Campylobacter*, *Cryptosporidium* and norovirus originating from sewage were modelled. Quantitative microbial risk assessment (QMRA) was used to assess changes in risks of infection. In general, infection risks downstream of WWTPs are higher than downstream CSOs. Even though model outputs show an increase in CSO influxes, in combination with changes in pathogen survival, dilution within the sewage system and bathing behaviour, the effects on the infection risks are limited. However, a decrease in dilution capacity of surface waters could have significant impact on the infection risks of relatively stable pathogens like *Cryptosporidium* and norovirus. Overall, average risks are found to be higher downstream WWTPs compared to CSOs. Especially with regard to decreased flow rates, adaptation measures on treatment at WWTPs may be more beneficial for human health than decreasing CSO events.

1. Introduction

Concentrations of human pathogens in surface waters are determined by a several processes. The pathogen input is important, but once released in the aquatic environment, pathogens are diluted, reduced by die-off and (temporarily) reduced by sedimentation.

Domestic wastewater is a major source of human pathogens to surface waters. Commonly, wastewater is treated by WWTPs before it is discharged into the surface waters. However, in the case of combined sewer systems, the capacity of the sewer systems and WWTPs may be exceeded during periods of high rainfall, and, untreated wastewater will be discharged directly into the surface waters. Since climate change predictions show an increase in intensive precipitation events (KNMI, 2014b), an increase in CSOs is expected in the Netherlands. Recent research on the effects of climate change on the frequency, duration and volume of CSOs supports this (Abdellatif et al., 2015; Bi et al., 2015; Nie et al., 2009; Semadeni-Davies et al., 2008). An increase in CSOs could result in increased influx of microbial pathogens and other pollutants into receiving waters.

Besides the changes in influx, climate change is also expected to influence river flow rates. In winter time, precipitation will increase river flow rates (Middelkoop et al., 2001). During dry summers, periods of low discharge will occur more often. This is mainly true for surface water-dominated rivers, such as the river Meuse (de Wit et al., 2007), as they cannot rely on a relatively stable groundwater fed base flow. Based on parameters, including general water quality variables, nutrients, heavy metals and metalloids, a case study on the impact of summer droughts on the water quality of the Meuse river (Van Vliet and Zwolsman, 2008) indicates a general degeneration of the water quality of the Meuse river during droughts. They concluded that the reduction of the dilution capacity of point source effluents was one of the reasons for the decline in water quality. The effects of changing flow rates may amplify or counteract the change in pathogen influx through CSOs.

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Most of the 715 official bathing water locations in the Netherlands are not in contact with wastewater from outlets of WWTPs or CSOs, as shown by completing bathing water profiles (EEA, 2015). An inventory of the accessible bathing water profiles (86%) shows that, of the bathing water locations in the Netherlands, only 11% and 10% could be influenced by wastewater discharges from WWTPs and CSOs respectively (Anonymous, 2016). At some of the locations with high wastewater discharge, additional wastewater treatment is applied to the emission or high emissions are monitored and communicated to the bathers.

About two-third of recreational activities in the Netherlands takes place at official bathing sites (Schets et al., 2011), so the implication is that people frequently bathe at unofficial sites as well. In Amsterdam, Greven and Jakobs (2015) found that 5%, of the people surveyed with regard to their swimming habits, occasionally swim in the canals. Another possibility is that people jump, fall or get pushed into the canal (Schets et al., 2008). Depending on the exact location of bathing, this section of the bathing population may experience increased exposure risks to surface water contaminated with pathogens from outlets of WWTPs and CSOs. Exposure to surface water contaminated with human pathogens, including viruses, bacteria or parasites, may lead to infection and subsequent illness, such as gastroenteritis or skin, ear and eye infections (Brunkard et al., 2011; Schets et al., 2010).

If climate change increases pathogen contributions of CSO events and affects flow rates, an increase in risk of infection is expected during recreation in close proximity of a WWTP or CSO. The main aim of this study is to quantify this change in risk for the Netherlands. Quantitative microbial risk assessment (QMRA) is used to determine the risk of gastroenteritis when exposed to surface water contaminated with norovirus, *Campylobacter* and *Cryptosporidium* originating from wastewater under current and future scenarios. This selection of pathogens was based on human disease burden, data availability and pathogen characteristics.

In previous QMRA studies (Sterk et al., 2015, 2016), the assumption was that people swim randomly over the summer. However, as discussed in these papers, better estimates of probabilities of human exposure to surface waters could improve predictions of the infection risks. One improvement would be to weigh the chance that people will be swimming based on the conditions of a certain day. Intuitively, one could say that higher temperatures result in more water recreation. However, besides water temperature, factors like air temperatures, precipitation, sunshine and economic factors like amount of leisure time will largely determine whether or not people will be bathing. For example, data from Statistics Netherlands shows that recreation takes place more often during the weekends than on a weekday (Centraal Bureau voor de Statistiek 2009). In this study, change in risks of infection are not only based on changes in dose, but also on bathing behaviour.

2. Methods

2.1. Sewer system model

Calculations of rainwater fluxes into the sewer system were based on a standard rainfall-runoff model used for sanitary works in the Netherlands. The sewer system itself was modelled using a simple reservoir model (RIONED, 2004; van de Herik and van Luytelaar, 1989). Fig. 1 shows a schematic overview of the model.

Part of the precipitation (P) is temporarily stored at the surface in depressions and puddles (S_0). The maximum amount of depression storage (S_0Max) is determined by surface type and its inclination. For a summary of model variables see Table 1. A distinction is made between four different surface types, namely impervious paved surface (like bitumen), pervious paved surface (like paving bricks), roofs and unpaved surface. Distinction in inclination is made into sloped surfaces (>4%), flat surfaces, and flat and wide surfaces (flow length to reach sewer system >100 m).

The depression storage (S_0) on the impervious areas can be emptied (during dry weather conditions) due to evaporation. The potential evaporation (ET_{pot}) depends on temperature and radiation (Hiemstra, 2011), but the actual evaporation (ET_{act}) is limited by the volume of water available in the storage:

$$ET_{act} = \begin{cases} ET_{pot} & S_0 > ET_{pot} \\ S_0 & S_0 \le ET_{pot} \end{cases} \quad (1)$$

For the pervious areas, the storage volume can also be emptied by infiltration.

Infiltration capacity, $f_c(t)$, was calculated using the infiltration model of Horton (Horton, 1941). In this model, infiltration capacity decreases exponentially while water is present on the surface. During dry periods, the infiltration capacity is restored to its initial value. At the start of the rain event f_c equals the maximum infiltration capacity (f_b [L T^{-1}]). The change in infiltration capacity is determined by:

$$\frac{df_c(t)}{dt} = \begin{cases} k_a(f_c(t) - f_e) & P + S_0 > 0 \\ k_h(f_c(t) - f_b) & P + S_0 = 0 \end{cases} \quad (2)$$

where f_e is the minimum infiltration capacity [L T^{-1}], and k_a [T^{-1}] and k_h [T^{-1}] are constants determining the exponential decrease and increase respectively. As for evaporation, actual infiltration f(t) is limited by the volume of water available.

Since not all excess water (Xs) reaches the sewer system immediately, the volume of excess precipitation stored dynamically at the surface is modelled as an additional compartment. Change in dynamic storage (h [L]) is determined by:

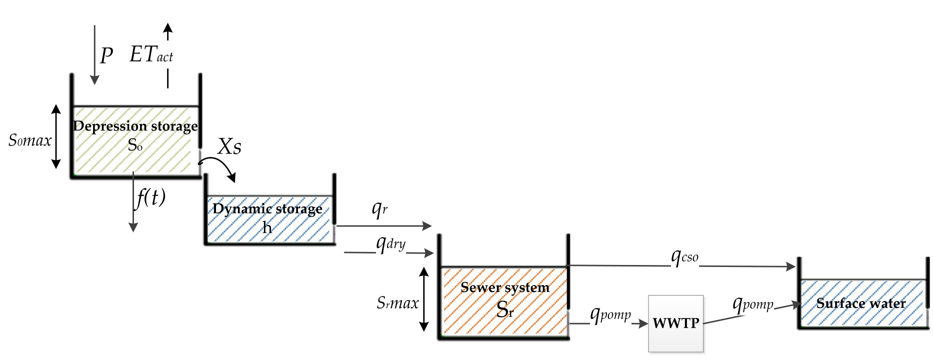

Fig. 1. Schematic overview of rainfall-runoff model combined with the reservoir model. P is precipitation; ET_{act} is evapotranspiration; f{t} is infiltration rate; X_s is excess precipitation water; q_r is the water flux entering the sewer system; q_{dry} is dry weather flow, q_{cso} and q_{pomp} are discharges from a CSO event and to the WWTP respectively.

Table 1
Variables for model, conservative policy values (RIONED, 2004).

Surface type	Inclination	c (min^{-1})	S_0Max (mm)	Infiltration capacity ($mm\ h^{-1}$)		Increase/decrease rate (h^{-1})	
				f_b	f_e	k_a	k_h
Impervious- paved	Sloped	0.5	0.0				
	Flat	0.2	0.5				
	Flat and Wide	0.1	1				
Pervious-paved	Sloped	0.5	0.0	2.0	0.5	3.0	0.1
	Flat	0.2	0.5	2.0	0.5	3.0	0.1
	Flat and Wide	0.1	1	2.0	0.5	3.0	0.1
Roof	Sloped	0.5	0.0				
	Flat	0.2	2				
	Flat and Wide	0.1	4				
Unpaved	Sloped	0.5	2	5	1	3.0	0.1
	Flat	0.2	4	5	1	3.0	0.1
	Flat and Wide	0.1	6	5	1	3.0	0.1

$$\frac{dh}{dt} = Xs - q_r \tag{3}$$

$$q_r = c \cdot h \tag{4}$$

where q_r is the water entering the sewer system [$L\,T^{-1}$] and c [T^{-1}] is the delay factor depending on the inclination of the surface.

The sewer system is modelled as a compartment and dynamic storage is neglected, i.e. there is no travel time within the sewer system:

$$\frac{dS_r}{dt} = q_r + q_{dry} - q_{pump} - q_{cso} \tag{5}$$

where S_r is the volume stored in the sewer system [L] with a maximum value of S_rMax, q_{dry} [$L\,T^{-1}$] is the influx of wastewater from households, q_{pump} [$L\,T^{-1}$] is the flux to the WWTP and q_{cso} [$L\,T^{-1}$] is the flux directly discharged into surface waters.

The capacity of the overflow in the sewer systems is often quite large, with values of up to 30 mm/h (Overeem, 2014). When this capacity is exceeded, excess water is discharged into the street, consequently resulting in urban flooding (de Man et al., 2014). However, this is beyond the scope of this study and the overflow capacity in the model is set to infinity.

Pathogens were assumed to enter the sewer system only through the dry-weather flow, C_{dry} [# L^{-1}].

$$C_{dry} = q_{dry} \cdot N_{dry} \tag{6}$$

where N_{dry} is the number of pathogens shed by the population connected to the sewer system (Nr_{people}). N_{dry} is determined similarly to the method of Dorner et al. (2004):

$$N_{dry} = Nr_{people} \cdot prev \cdot M_f \cdot C_m \tag{7}$$

where M_f [M] is the amount of faeces produced per person, prev [-] is the pathogen prevalence in the population and C_m [# L^{-1}] the pathogen concentration in faeces.

M_f is set at 125 g per day (Wyman et al., 1978).

Prev was derived by random sampling from a beta-distribution, describing the probability that humans are infected:

$$prev \sim \beta(\alpha, \beta) \tag{8}$$

Parameters, α and β are based on an inventarisation of De Wit et al. (2001), which gives the prevalence of pathogens in a group of cases with gastroenteritis and a healthy control group (see Table 2). They determined that the standardized gastroenteritis incidence was 283 per 1000 person-years. This ratio is used to determine the incidence of pathogens in a population of both healthy and ill individuals:

$$\alpha = s + 1 \tag{9}$$

$$\beta = n - s + 1 \tag{10}$$

where s is the number of infected humans and n is the number of samples.

Pathogen concentrations were determined by random sampling the gamma distribution representing shedding intensity in $\log_{10}$ units of pathogens per gram of fresh weight faeces:

$$C_m \sim \Gamma(r, \gamma) \tag{11}$$

with shape and location parameters, r and γ, r and γ are calculated by using mean (μ) and standard deviation (σ):of literature data for pathogen concentrations in human faeces (Table 2):

Table 2
Pathogen prevalence in the human population (values for the healthy population, values for the group with gastroenteritis between parenthesis) and concentration in the faeces [#/g].

Pathogen:	Prevalence			Concentration			Removal WWTP	
	s	n	Ref	Mean	SD	Ref	Mean[c]	Ref
Campylobacter	4 (9)	665 (700)	(De Wit et al., 2001)	6E + 05	8E + 05	(Lin et al., 2008)	1.0 Log_{10}	(Koenraad et al., 1994)
Cryptosporidium	1 (14)	673 (706)	(De Wit et al., 2001)	2E + 07[a]	5E + 07[a]	(Chappell et al., 1996)	1.4 Log_{10}	(Hoogenboezem et al., 2000)
norovirus	35 (114)	669 (709)	(De Wit et al., 2001)	5E + 09[b]	1E + 10[b]	(Chan et al., 2006)	2.4 Log_{10}	(van den Berg et al., 2005)

[a] Based upon total excretion nos.and days of excretion per dose, hence only deviation based on dose.
[b] Read from graph, combination of GI and GII.
[c] Averages for two or three WWTPs combined.

$$r = \frac{\mu^2}{\sigma^2} \tag{12}$$

$$\gamma = \frac{\sigma^2}{\mu} \tag{13}$$

The motivation for including pathogen estimates at the source over the use of literature data of concentrations in sewage effluent or at CSO was to include dilution in the sewer system in the calculations and to ensure that results for the WWTP and CSO would correspond to the same system.

Pathogen decay and attachment to sewer pipes or sediments were neglected during the transport in the sewer system. The first was assumed negligible because of the fast transport in the sewage system and the latter was not included because applicable data on the process was not available. The number of pathogens in the sewer system (N_s) was divided over the flow routes, that is proportional to the water flow and based on the assumption of instant mixing for all fluxes. The change in the number of pathogens in the sewer system is:

$$\frac{dN_S(t)}{dt} = C_{dry}(t)\cdot q_{dry}(t) - C_{pump}(t)\cdot q_{pump}(t) - C_{cso}(t)\cdot q_{cso}(t) \tag{14}$$

where $C_{pump,}$ and C_{CSO} are the concentration of pathogens in the flux to the WWTP, and directly to the surface waters, respectively.

$$C_{pump} = C_{cso} = \frac{N_s}{S_r + S_{in}} \quad if\ S_r > 0 \tag{15}$$

where Sin [L] is the water volume entering the sewer system that is not initially stored. This means that when Sr is zero, incoming pathogens are pumped out directly towards the WWTP.

The pathogen concentration entering the WWTP (e.g. C_{pump}), is multiplied by the fraction of pathogens passing the WWTP (Log_{10} F_{wwtp}) to determine the concentration discharged to the surface water (C_{wwtp}). F_{wwtp} is based on literature values (see Table 2).

2.2. Dilution in surface water

Pathogen influx to the surface waters, through either the WWTP or by CSO, are diluted to the final pathogen concentration to which swimmers could be exposed. This mixing will not be instantaneous. To calculate the cross-sectional maximum concentration in the surface waters at X metres downstream of the emission, the same approach was used as by de Nijs and de Greef et al. (1992). They used a model describing the blending of two water courses (Fischer et al., 1979). One water course is the river with a discharge of q_w, in which the initial concentration C_w is assumed zero. The other is the sewage flux.

Concentration at complete mixing is:

$$C_\infty = \frac{C_w \cdot q_w + C_{in}\cdot q_{in}}{C_w + C_{pump}} \tag{16}$$

where $C_{in}\cdot q_{in}$ equal $C_{wwtp}\cdot q_{pump}\cdot A$ for the WWTP and $C_{cso}\cdot q_{cso}$ A in case of the CSO.

The reflection equation for the blending of two watercourses of equal widths is then calculated according to:

$$C(x,y) = C_\infty \cdot \sum_{n=-3}^{+3} erf\left[\frac{y\prime + \frac{1}{2} + 2n}{\sqrt{4x\prime}}\right] - erf\left[\frac{y\prime + \frac{1}{2} - 2n}{\sqrt{4x\prime}}\right] \tag{17}$$

where x′ and y′ are the transformed x and y and n equals the Mannings coefficient (see Table 3).

The amount of removal of pathogens during the transport depends on the travel time (e.g. the flow rate). Die-off during transport is described as a first order decay reaction, where the decay rate in water (μ_w) during transport was determined based upon the water temperature (T), using:

$$\mu_W(t) = \frac{\ln 10}{10^{a_0 + a_1 T}} \tag{18}$$

a_0 ($\log_{10}$ day) and a_1 ($\log_{10}$ day $^\circ C^{-1}$) are inactivation rate parameters: 2.3 and −0.035 for norovirus, 0.53 and −0.017 respectively for *Campylobacter* and 3.1 and −0.078 respectively for *Cryptosporidium* (Schijven et al., 2013). Water temperatures were not available, but in the calculations the average week air temperature was used as an approximation of the water temperature. Available surface water and air temperature time series (data not shown) indicated that this was a reasonable approximation, especially during summer.

Another process that influences the concentration of pathogens in the surface water is its sedimentation and resuspension from the sediments (Drummond et al., 2014; Searcy et al., 2006). Since both sedimentation and resuspension of pathogen to the sediments is dependent on flow velocity, climate change could also affect these processes (Sterk et al., 2013). However, because the surface water compartment in the model is reduced to a simple reservoir with a constant flow velocity and a general since estimates for sedimentation and resuspension rates is not available, nor how they would change under climate change, is was chosen not to include an estimate for these rates in the model.

2.3. Risk assessment

The risks of infection are calculated assuming that recreation takes place at 1.000 m distance from sewage emission. It was assumed that bathing only takes place during the bathing season (1 May to 1 October).

2.3.1. Recreational behaviour vs. risk

To include the likelihood of recreation under certain climate conditions in the risk assessment, an empirical relation was derived based on visitor data from recreational sites. Daily visitor data during the bathing season at five recreational sites was available for the period 2000–2014. This data was combined with data on

Table 3
Influx variables, constants and intermediate calculations for blending of two watercourses.

Influx variables			
w	Width of the surface water		[m]
h	Depth of the surface water		[m]
Constants			
n	Manning coefficient	0.025	[m]
v_w	pathogen concentration surface water supply	$1.14\cdot 10^6$	[$m^2\ s^{-1}$]
g	Gravitation constant	9.81	[$m\ s^{-2}$]
Intermediate calculations			
$\bar{u}$	Average longitudinal stream velocity	$=\frac{q_w+q_{in}}{w\cdot h}$	[$m\ s^{-1}$]
R_h	Hydraulic radius	$=\frac{w\cdot h}{w+2h}$	[m]
CZ_f	Chezy coefficient	$=\frac{\frac{1}{2}R_h^{1/6}}{n}$	
u*	Shear stress velocity	$=\frac{\overline{u}}{CZ_f}\cdot\sqrt{g}$	
D	Transverse dispersion coefficient	$=0.6\cdot h\cdot u^*$	
L_{mix}	Length of mixing zone	$=0.4\cdot\overline{u}\cdot\frac{w^2}{D}$	[$mm\ h^{-1}$]
Transformation of x and y scales			
x’	Transformation of x	$=x\frac{D}{\overline{u}\cdot w^2}$	[−]
y’	Transformation of y	$=\frac{y}{w}$	[−]

temperature, precipitation, hours of sunshine from the nearest measurement station of the Royal Netherlands Meteorological Institute (KNMI, 2015). Vacation dates, based on primary school vacation periods per region (Government), as well as weekends and holidays were recorded in the database.

An exploration of the data revealed that the daily number of visitors to a swimming location was highly over-dispersed. Therefore this outcome was modelled via (mixed-effects) negative binomial models in R (R Core Team, 2015). The clustering of the data within the swimming locations was adjusted for by adding a random intercept for each location. Two types of model selection were performed, on the fixed effects part only. First, a forward model selection based on the Akaike information criterion (AIC) was carried out by starting with a model that only included the random effects. Second, a backward model selection based on the AIC was carried out, starting with all fixed effects and a random intercept per location. The resulting model was used in the Monte Carlo simulation of the QMRA to weigh days in summer for the amount of recreation.

2.3.2. QMRA

The ingested dose, D was calculated using:

$$D = C_{max}\, V \tag{19}$$

where $C_{max}(x, X)$, [# L^{-3}] is the cross-sectional maximum concentration in the surface waters at X metres downstream of the emission predicted under the scenario examined and V [L^3] is the individual volume of water that was consumed (Schets et al., 2011).

The risk of infection (R_{inf}) per exposure event was calculated using the following dose response relationship (Teunis et al., 2008):

$$R_{inf} = 1 - {}_1F_1(\alpha_i, \alpha_i + \beta_i; -\mathrm{D} \tag{20}$$

where ${}_1F_1$ is the hypergeometric distribution and α_i and β_i are the parameters of the Beta-distribution, (0.038 and 0.022 respectively for *Campylobacter*, 0.106 and 0.295 respectively for *Cryptosporidium* and 0.04 and 0.055 respectively for norovirus (Schijven et al., 2013)).

Risks of infection with *Campylobacter*, *Cryptosporidium* and norovirus were calculated (Mathematica 9.0.1 Wolfram Research Inc., Champaign, USA) using Monte Carlo simulations with random sampling of 10 000 values. In order to evaluate effects of climate change for current and future summers under the different climate scenarios, a fixed seed number was used in the Monte Carlo simulations.

2.4. Scenarios

2.4.1. Climate change

KNMI data on hourly precipitation, temperature and radiation measured at the Bilt from 1981 to 2010 was used as a reference scenario (KNMI, 2015). This 30-year period is assumed to capture the natural variation. Values for temperature and radiation were used to calculate the Makkink reference evapotranspiration (Hiemstra, 2011).

Changes in sewage volumes and sewer overflows were calculated under the most conservative (G_L) and most extreme (W_H) climate scenarios developed by the KNMI. These scenarios predict changes in air temperature, precipitation, wind and sea level in 2050 and 2085 compared to the period 1981–2010 (KNMI, 2014b). The scenarios predict an air temperature rise of about 1.0–1.3 °C and 2.3–3.7 °C for 2050 and 2085 respectively. Predictions for precipitation show that on average, and according to all scenarios, winters will become wetter and extreme precipitation quantities will increase. Changes in precipitation for summer are more difficult to predict, hence they are given with a large uncertainty band.

The reference data was transformed (Bakker, 2012; Bessembinder, 2012; KNMI, 2014a) under the G_L scenarios, using the lower limit of the prediction band for changes of precipitation in summer and under the W_H scenarios with the upper limit to create the most conservative and extreme predictions. These transformations resulted in a 30 year-time series, with each year representing a possible scenario for the climate in 2050 or 2085.

In general, climate change is expected to decrease river flow during summers (low flow period) and increase river flow during the high flow period (end of winter/spring). For example, river flow in the Rhine is expected to decrease by 8% on average for 2071–2100 relative to 1971–2000, with an increase of 1% during the high flow period and a decrease of 37% during the low flow period (van Vliet et al., 2013). This effect could be even stronger for rivers such as the Meuse, that cannot rely on a stable groundwater influx (Van Vliet and Zwolsman, 2008). In the current model the receiving water is fictive and discharge is kept constant. To include the possible effect of change in dilution through changes in river flow, the scenarios are also evaluated with discharges of +10 and −40%.

2.4.2. Leisure time

To determine the probability of recreation at bathing sites in the Netherlands, weekends and vacation periods have to be included as well. Summer vacation in the Netherlands takes place at differing times per region and per year. It was assumed that there will be no change in the duration of summer vacation in the future, and the same six-week period was assumed (week 28 to week 33) for every year.

2.4.3. Sewer systems in the Netherlands

The model is evaluated for a fictive sewer system, with characteristics typical of the Netherlands (see Table 4). The most common type of sewer system in the Netherlands (66% in 2013) is the combined improved system (Rijkswaterstaat-WVL, 2015).

Dry-weather flow was assumed to be equal to the average water use in the Netherlands, which was $7.8 \cdot 10^8$ m^3 in 2012 (CBS). With $1.7 \cdot 10^7$ inhabitants nationally, this would equal approximately 5.3 L per person per hour. A drainage area of 1.000 m2 is modelled assuming a flat area with equal division into impervious-paved, pervious paved, pervious-unpaved surfaces and flat and sloped roofs.

For the reference scenario of receiving waters, dimensions of a medium sized river were used (de Nijs and de Greef, 1992). The initial concentration of the river was set to zero.

Table 4
Influx variables for fictive improved combined sewer system used in the model.

Parameter	Value
$q_{pompMax}$ (mm h^{-1})	0.7
S_rMax (mm)	9
q_{dry} (L h^{-1} pp^{-1})	5.3
Nr_{people} (CBS)	500/km^2 (2014)
Surface area (m^2)	1000
Surface type	
	20% Impervious- paved (flat)
	20% Pervious- paved (flat)
	20% Roof (flat)
	20% Roof (sloped)
	20% Pervious- unpaved (flat)
w (m)	15
h (m)	2
q_w (m^3 s^{-1})	0.099
C_w (nr mm^{-1})	0

3. Results

3.1. Sewage fluxes

Sewage fluxes at the WWTP are relatively constant throughout the year, fluxes at the CSO are non-existent during most of the year with only a few peaks (data not shown). Table 5 shows the frequency, volume per year and total duration of overflows for both the reference and the future scenarios. Frequency was defined by a daily interval (RIONED, 2004). If there are several overflow events separated by a period shorter than one day, this is counted as one single event. All scenarios showed an increase in frequency, volume and duration. With the highest increase for the most extreme 2085W_H scenario, for which the frequency was more than twice as high and mean flux increased about 200%. The mean flux at the WWTP (data not shown) hardly changed, only +0.14% for the 2085W_H scenario compared to the reference.

Fig. 2 shows the increase in CSO volume per season. The highest increase was seen during the winter period, and the lowest increase during summer.

3.2. Pathogen concentrations

Table 6 gives the change in pathogen concentration at the outlet of the WWTP and CSO for the different climate scenarios. The highest concentrations were estimated for norovirus, and the lowest for *Campylobacter*. No change was found in average concentration at the outlet of the WWTP. Change in concentration at the outlet of the CSO varies from a small increase of max 3.3%, to a decrease of 9.0%, depending on the scenario and the pathogen.

Table 7 gives the change in pathogen concentration 1000 m downstream of the WWTP and CSO. Downstream of the WWTP, concentrations were lowered under the different climate scenarios, up to 46% for *Campylobacter,* whereas a limited decrease for norovirus was observed and no change in the case of *Cryptosporidium*.

Concentrations 1000 m downstream of the CSO again showed a decrease for *Campylobacter*, although the maximum decrease was less than for the WWTP. On the other hand, concentrations of *Cryptosporidium* and norovirus increased, in the worst-case scenario, up to 24% and 13% respectively.

Fig. 3 shows, based on temperature and precipitation time series of the reference climate scenario and for different scenarios of q_w, the average pathogen concentration at the outlet of the WWTP and CSO during overflow events, and the diluted concentration at 100 m and 1000 m downstream. The concentration decreased further downstream, and trends are similar for contributions of the WWTP and CSO. A decrease in flow rate resulted in an increase in concentrations for all pathogens at a distance of 100 m. 1000 m downstream, a decreased flow rate increased *Cryptosporidium* and norovirus concentrations, but resulted in a 70% decrease in *Campylobacter*.

3.3. Risk of infection

Both forward and backward model selection procedures yielded the same final model, which showed that the number of visitors was best predicted by average and maximum temperature (T_{avg}, T_{max}), radiation (RAD) and three categorical variables: precipitation during 24 h (*precip*), weekend and vacation:

$$\ln(recreants) = Int + 0.12{*}T_{max} + 0.17\ T_{avg} + 0.00089\ RAD \tag{21}$$

with temperature in °C, radiation in J/cm^2 and where the intercept (*int*) depends upon the categorical variables:

$$Int = -2.6 - 0.3\ (precip = yes) + 0.45\ (weekend = \text{yes}) + 0.12\ (vacation = yes) \tag{22}$$

Infection risks calculated for the different climate change scenarios are given in Table 8. Current (1981–2010) mean risks were found to be highest downstream of the WWTP. The *95*th percentile of the risk downstream of the CSO equals 0 for all pathogens (data not shown) However, as regards maximum risks, the risks downstream of the CSO are slightly higher.

Downstream of the WWTP, climate scenarios show a decrease in risk for *Campylobacte*r infection for swimmers and little or no change in risks for human *Cryptosporidium* and norovirus infection. Downstream of the CSO, changes in the risks of infection vary largely per scenario and pathogen (i.e. −42, 5% to +8.3%).

A decrease in flow rate decreased the risk of *Campylobacter* infection, while it increased the risk of infection for *Cryptosporidium* and norovirus (see Table 9). Changes in infection risks for norovirus were much smaller than for *Campylobacter* and *Cryptosporidium.*

4. Discussion

For the current scenario, average pathogen concentration calculated with the sewage model at the outlet of the WWTP were in the order of 10^2, 10^3, and 10^6 per litre of surface water for *Campylobacter*, *Cryptosporidium* and norovirus, respectively. Concentrations at the outlet of the CSO during an overflow event were one or two log higher, namely 10^3, 10^4 and 10^8 per litre for *Campylobacter*, *Cryptosporidium* and norovirus, respectively.

The range of pathogen concentrations detected in sewage influents and effluents at WWTPs in the Netherlands can be found in Table 10. Measurements at the outlet of the CSO for the Netherlands are not available yet, but de Man et al. (2014) measured concentrations in urban flood water from a combined sewer.

A comparison of the model output with the literature data shows that predictions of *Campylobacter* and *Cryptosporidium* concentrations in sewage effluent are in the same range as measured. However, predicted norovirus concentrations are two log higher than the maximum concentrations found. Since the only differences between the pathogens within the sewage system are prevalence and influx concentrations, this likely constitutes the deviation. Differences may be caused by differences in the method used to determine norovirus concentrations; To determine the concentration in faeces qPCR was applied (Chan et al., 2006) while Lodder and de Roda Husman (2005) used conventional RT-PCR to determine the concentration in sewage.

Except for *Campylobacter,* concentrations measured in urban flood water are much lower than concentrations predicted during a CSO event. An explanation could be that urban flood water is not a true representative of CSO water, since it may not yet have been fully mixed with the dry weather flow. However, measurements at CSOs in other countries do not correspond with the predicted values either. Measurements of sewage overflows in the US show no or little *Cryptosporidium*, with concentrations in the range of 0.59–1 #/L (Arnone and Walling, 2006). During a CSO event

Table 5
Change in yearly frequency, volume and duration of overflows for the different scenarios.

CSO	1981–2010	2050G_L	2050W_H	2085G_L	2085W_H
Frequency	5 events/year	+11%	+69%	+25%	+111%
Volume	51.2 m^3/year	+9%	+98%	+28%	+195%
Duration	12.5 h/year	+14%	+75%	+20%	+118%

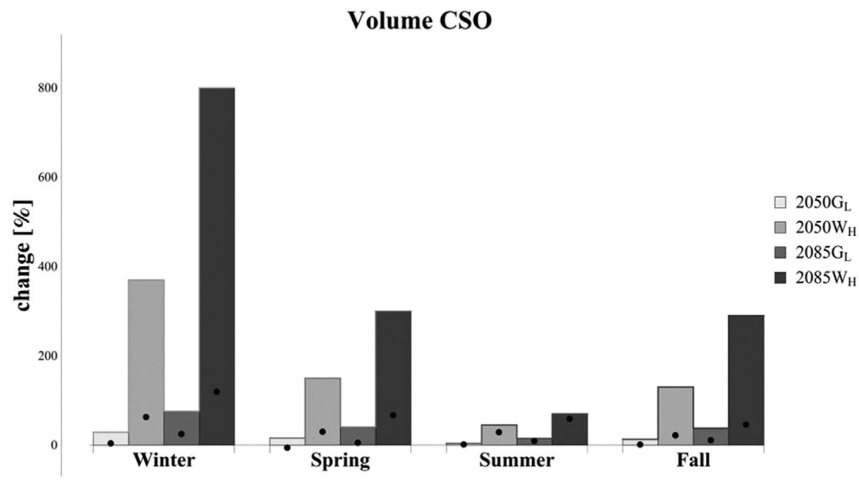

Fig. 2. Mean relative change in overflow volume for future scenarios per season. Black dots give relative change for the 95% level of the volumes during overflow.

Table 6
Change in average pathogen concentration at the outlet of the WWTP and CSO; 95% level between parenthesis (−indicates change <0.1%).

	1981–2010	$2050G_L$	$2050W_H$	$2085G_L$	$2085W_H$
WWTP					
Campylobacter	5.1E+02 (1.3E+03)	–	–	–	–
Cryptosporidium	5.6E+03 (2.0E+04)	–	–	–	–
Norovirus	1.7E+06 (4.3E+06)	–	–	–	–
CSO					
Campylobacter	2.2E+03 (4.8E+03)	+3.3% (+2.8%)	−0.1% (−2.1%)	+0.5% (+0.1%)	−5.5% (−1.2%)
Cryptosporidium	5.5E+04 (1.4E+05)	+2.6% (+4.4%)	−0.6% (+3.3%)	−1.6% (+3.2%)	−4.5% (+2.5%)
Norovirus	1.6E+08 (3.2E+08)	+1.1% (−1.0%)	−3.7% (−1.7%)	−2.6% (−3.7%)	−9.2% (−5.5%)

Table 7
Change in average pathogen concentration 1000 m downstream of the WWTP and CSO; 95% level between parenthesis (−indicates change <0.1%).

	1981–2010	$2050G_L$	$2050W_H$	$2085G_L$	$2085W_H$
WWTP					
Campylobacter	1.8E-01 (4.7E-01)	−14.2% (−14.2%)	−32.6% (−32.7%)	−16.4% (−16.4%)	−45.9% (−46.0%)
Cryptosporidium	4.0E+01 (1.4E+02)	–	–	–	–
Norovirus	1.2E+04 (2.9E+04)	−0.7% (−0.7%)	−1.9% (−1.9%)	−0.8% (−0.9%)	−3.0% (−3.1%)
CSO					
Campylobacter	8.4E-01 (2.3E+00)	−15.1% (−16.1%)	−23.6% (−22.2%)	−11.0% (−14.1%)	−32.2% (−33.3%)
Cryptosporidium	4.1E+02 (1.5E+03)	−0.2% (0.6%)	+12.1% (+10.7%)	+6.6% (+10.6%)	+23.7% (+16.3%)
Norovirus	1.1E+06 (3.2E+06)	−3.5% (−12.3%)	+5.9% (−1.1%)	+2.1% (−0.9%)	+12.5% (+4.5%)

Rodríguez et al. (2012) detected an average norovirus concentration of 42.8 RT-PCR MPN/L in the outfall. Aslan et al. (2011) could not detect norovirus in four samples at CSO pumping stations. During CSO events in Germany a mean concentration of *Campylobacter* of 10^6 MPN/L was found (Rechenburg and Kistemann, 2009).

An explanation for the differences between the model results and these observations is that model concentrations are based on complete mixing. In reality, CSO concentrations may be highly variable and given that observed CSO concentrations in the monitoring studies are based on only a few samples, peak values could have been missed and require risk based sampling strategies.

A reduction of pathogens due to die-off or sedimentation within the sewer system is not included in the model either. Since the main improvement for the improved combined sewer systems is the reduction of contaminants by settling in a sedimentation basin, especially the latter could be considerably important. On the other, hand, since concentrations found for *Campylobacter* during a CSO event by Rechenburg and Kistemann (2009) are even three log higher, and similar to what they found in sewage influent, differences could also be caused by pathogen seasonality which is not included in the model, or differences in geographical location.

Without validation, absolute model results should be interpreted with caution. However, the model can be used to estimate the relative effects of climate change.

Results for the climate change scenarios show an increase in CSO frequencies as well as volume and duration. The largest volume increases are found in winter and are weaker during the bathing season. Volume increases during the bathing season in summer were much smaller than found during a case study in Canada; Bi et al. (2015) found a 15–500% increase in the volume discharged by CSOs in 2050 compared to 2013 based on eight rainfall events measured from May to October (i.e. the bathing season) in Canada. Possibly because the climate scenarios are evaluated using hourly data, CSO events under very intensive rain events, which occur

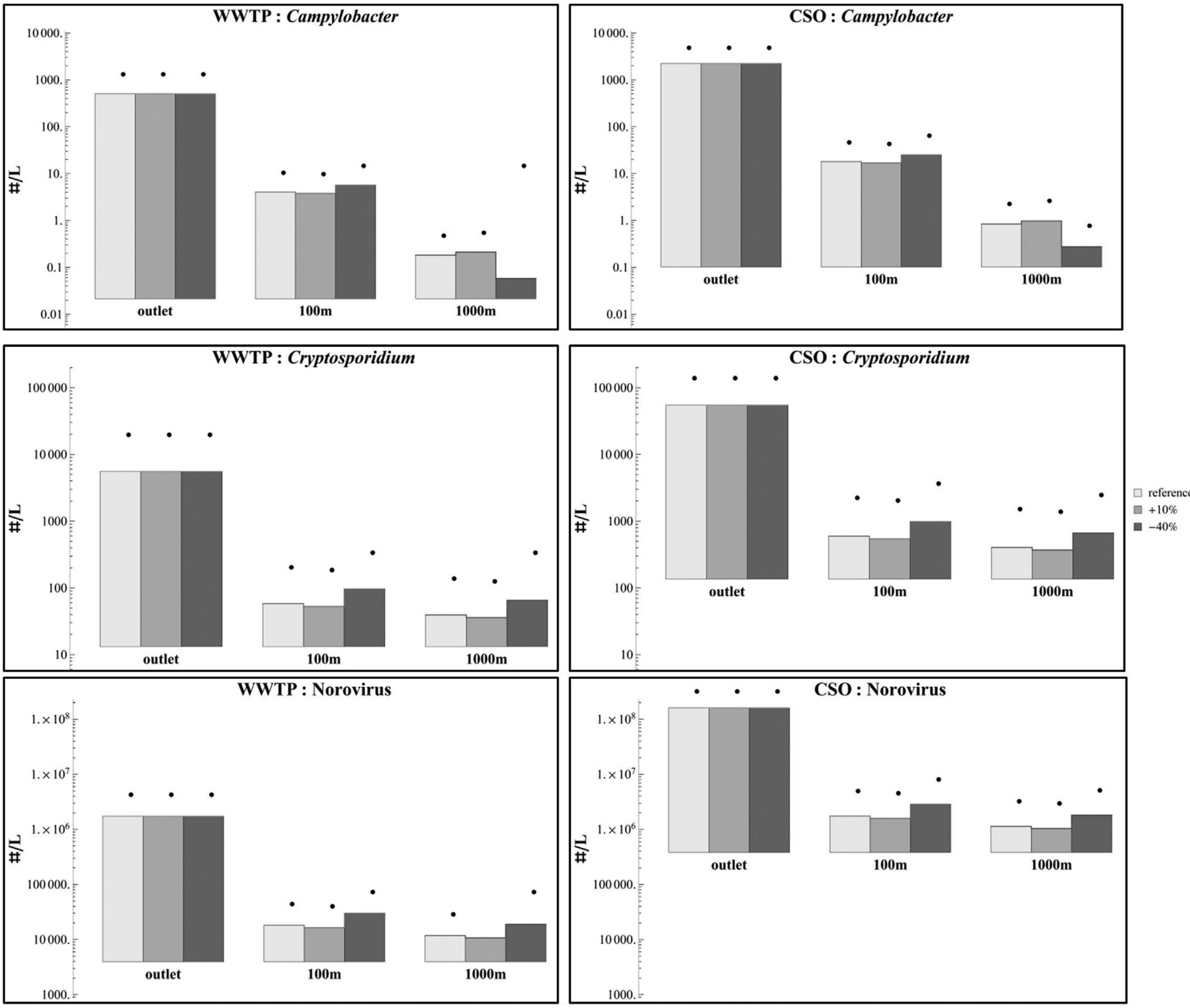

Fig. 3. For the reference scenario: average pathogen concentration in surface water at the outlet, 100 m and 1.000 m downstream for the WWTP (let pane) and CSO (right pane), for the different values of $q_{w.}$ Black dots represent the 95% level.

Table 8
Current average risk of infection per person for norovirus, Campylobacter and Cryptosporidium at 1.000 m distance from WWTP (A) and CSO (B) and change in percentage of average risk for future scenarios; maximum value between parentheses (−indicates change <0.1%).

	1981–2010	$2050G_L$	$2050W_H$	$2085G_L$	$2085W_H$
WWTP					
Campylobacter	2.5E-03 (6.5E-02)	−12.0% (−15.4%)	−32.0% (−38.5%)	−16.0% (+3.1%)	−44.0% (−47.7%)
Cryptosporidium	8.7E-02 (4.6E-01)	–(−4.3%)	–	–	–
Norovirus	4.9E-01 (6.0E-01)	–	–	–(−1.7%)	–
CSO					
Campylobacter	4.0E-04 (2.9E-01)	+5.0% (+24.1%)	−35.0% (+3.4%)	−2.5% (−)	−42.5% (−31.0%)
Cryptosporidium	6.0E-03 (6.4E-01)	−3.3% (−4.7%)	−21.7% (+3.1%)	+6.7% (−)	−33.3% (−)
Norovirus	1.2E-02 (6.8E-01)	–(−1.5%)	−25.0% (−1.5%)	+8.3% (−1.5%)	−16.7% (−)

more often in summer, may have been underestimated. However, another case study in Canada did confirm a variation in change per season. Fortier and Mailhot (2014)revealed a slight increase in duration of CSO events, but with a decrease in July and August and an increase in May and October. Moreover, Abdellatif et al. (2015) found an annual increase for CSO volume, duration and frequency in northwest England, but a decrease during the bathing season.

Climate change effects on the pathogen concentrations in the CSO fluxes vary per pathogen and scenario, and the change in concentrations can be explained by the interplay of increased

Table 9
Average risk of infection per person for norovirus, Campylobacter and Cryptosporidium at 1.000 m distance from the WWTP(A) and CSO (B) and change in percentage of average risk different scenarios of q_w; maximum value between parentheses (− indicates change <0.1%).

	1981–2010	1981–2010	1981–2010
	$q_w = 0.099\ m^3\ s^{-1}$	$q_w + 10\%$	$q_w - 40\%$
WWTP			
Campylobacter	2.5E-03 (6.5E-02)	+16.0% (+24.6%)	−67.6% (-58.5%)
Cryptosporidium	8.7E-02 (4.6E-01)	−6.9% (−)	+37.9% (+2.2%)
Norovirus	4.9E-01 (6.0E-01)	−	+4.1% (+1.7%)
CSO			
Campylobacter	4.0E-04 (2.9E-01)	+22.5% (+20.7%)	−60.0% (−48.3%)
Cryptosporidium	6.0E-03 (6.4E-01)	− (−1.6%)	+11.7% (−3.1%)
norovirus	1.2E-02 (6.8E-01)	− (−1.5%)	−

by the decrease in dilution. This finding is important for water regulations (Anonymous, 2006) based on indicators like E.coli, as it indicates that, even though the source could be similar, the effects of climate change on indicators may not correspond with more resistant pathogens like *Cryptosporidium* and norovirus.

Comparing risks downstream of the WWTP and CSOF reveals that, even though sewage is treated at the WWTP, average risks of infection downstream of the WWTP are significantly higher for all pathogens. Pathogen concentrations downstream of the WWTP are high year-round, while concentrations downstream the CSO are only increased a few times per year. However, when recreation coincides with such an event, the risks downstream of the CSO are higher than downstream of the WWTP.

An evaluation of the climate change scenarios shows that downstream of the WWTP, following the changes in concentration, infection risks are decreased for *Campylobacter*, and do not change

Table 10
Pathogen concentrations measured in sewage influent, effluent and urban flood water.

Pathogen	Origin	Concentration	Ref
Campylobacter (MPN/L)	Combined sewer flood water	10^1– >$1.5 \cdot 10^3$	(de Man et al., 2014).
	Influent	$2.0 \cdot 10^2$–10^4	(Koenraad et al., 1994)
	Effluent	>$3 \cdot 10^1$–10^3	(Koenraad et al., 1994)
Cryptosporidium(#/L)	Combined sewer flood water	10^{-1}–10^1	(de Man et al., 2014).
	Influent	$4 \cdot 10^1$–$2.9 \cdot 10^4$	(Hoogenboezem et al., 2000)
	Effluent	4–$1.4 \cdot 10^3$	(Hoogenboezem et al., 2000)
norovirus (pdu/L)	Combined sewer flood water	$5 \cdot 10^2$–$4 \cdot 10^4$	(de Man et al., 2014).
	Influent	$5 \cdot 10^3$–$9 \cdot 10^5$	(Lodder and de Roda Husman, 2005)
		10^3–10^6	(van den Berg et al., 2005)
	Effluent	$9 \cdot 10^2$–$8 \cdot 10^3$	(Lodder and de Roda Husman, 2005)
		10^1–10^4	(van den Berg et al., 2005)

dilution within the sewage system and an increased number of pathogens from the dry weather flow that are disposed by the CSO. Fluxes at the outlet of the WWTP are much more stable, and are hardly affected by climate change. What is more, climate change has no effect on the pathogen concentrations in the fluxes at the WWTP.

Concentrations downstream of the WWTP decrease only for *Campylobacter*. Since concentrations at the outlet and fluxes do not change, this decrease can be explained by the increase in die-off of *Campylobacter* in the surface water during transport.

Since *Cryptosporidium* and norovirus are more persistent and less temperature sensitive than *Campylobacter* (Eq. (13)), they are not/less affected by the rise in temperature. *Campylobacter* concentrations downstream of the CSO show a smaller decrease for the climate scenarios. Besides the increase in die-off, CSO concentrations changed and CSO volumes increased. This sum of effects led to an increased concentration for *Cryptosporidium* and norovirus in the worst-case scenario.

Differences in persistence could also explain how the decrease in flow rate increased *Cryptosporidium* and norovirus concentrations, but resulted in a decrease of *Campylobacter* 1.000 m downstream of the WWTP and CSO. When low flows reduce stream capacity for dilution this may result in greater pathogen loading. However, low flows may also result in an increased amount of time between discharge and abstraction, allowing more time for pathogen decay to occur (Boxall et al., 2009). As also shown by an analysis of the effects of climate change in a QMRA tool (Schijven et al., 2013): increased inactivation because of longer residence time will, from a certain distance onwards, decrease infection risks for exposure to waterborne pathogens that are very temperature-sensitive (like *Campylobacter*), whereas slowly inactivating pathogens (like *Cryptosporidium* and norovirus) will mainly be affected for *Cryptosporidium* and norovirus. Risks downstream of the CSO do not follow the same trend as the changes in concentration. Since concentrations are only elevated a few times per year, recreational pattern, as included by Eq. (21), may counteract with the change in concentrations.

The effect of flow rate on the risk of infection is much higher compared with the effects of the increase in CSO events. For *Campylobacter*, a decrease in flow rate will decrease the risks of infection. However, for *Cryptosporidium* this largely increases the risks. For norovirus, changes in the risk of infection are not that significant.

Norovirus concentrations produced by the model are high, and since norovirus exhibits a much higher dose-response relation compared with the bacteria and parasites (Kirby et al., 2015), the level of changes in concentration may not affect the risk that much.

It is important to note that norovirus concentrations are overestimated by the model, and because influx values for norovirus are based on detection by PCR, which reveals the presence of viral RNA or DNA but does not indicate the infectivity of the virus (de Roda Husman et al., 2009), overestimation of the risk could be even larger. However, when concentrations are lower, risks may respond stronger to concentration changes, consequently this means that climate changes for norovirus are indistinct.

Combining the results of the climate change scenarios and evaluation of the effect of flow rates suggests that climate change could pose an increased health risk during recreation of pathogens originating from sewage, in the event of the more persistent pathogens, not because of the increased frequency and volume of CSO, but because of the decrease in dilution caused by low flow rates during summer. In this study, however, this effect is only examined by including scenarios for constant surface water fluxes. In reality, surface water fluxes will respond to intense rainfall events also. Further examination of the combination of river water flowrates with sewage discharge is needed to improve estimates of changes in infection risks.

Another improvement would be to include pathogen deposition to the sediments in the model. Even though the storage of micro-organisms in sediments can decrease their concentration in the surrounding water, it also permits the microbial agents to be released again into the environment through natural or human-made resuspension events. Since bacteria and faecal indicators survive longer in sediments than suspended in the water (Burton et al., 1987; Davies et al., 1995), resuspension of pathogens in sediments during high flow could give rise to high concentration peaks in the water.

Interventions such as increasing sewer capacity, decreasing paved surface, green roofs or raingardens (vegetated depressions that are designed to capture drained runoff), would successfully reduce the contribution of CSOs (Montalto et al., 2007). However, they do not affect the contribution from the WWTPs. The trend is to upgrade combined sewer systems to separated or improved-separated systems and, even though this will take time, eventually it will automatically reduce risks from CSOs but will not benefit surface water quality downstream of WWTPs. Furthermore, since the contribution of pathogens is much larger from WWTPs compared to CSOs, improving treatment efficiency at WWTPs could be a more successful intervention in reducing pathogen concentrations in surface waters.

In this model evaluation the choice was made not to restrict qCSO and it therefore does not include urban flooding. In reality, a maximum water volume can be processed by the sewer system and excess water can result in urban flooding. An inventory along communities in the Netherlands (RIONED, 2015), regarding the frequency of urban flooding showed that an increase in frequency of urban flooding had already been noticed. Because of this, many communities are currently taking adaptation measures. These adaptation measures mainly focus on the effects on water volumes to reduce inconvenience of urban flooding and often do not take account of consequences for microbial risks to health. Autixier et al. (2014) showed that rain gardens can successfully reduce the volume of runoff and volume of CSOs, although these reductions could run counter to source water protection objectives for managing CSOs. They postulate that reductions in runoff volumes could correspond to increases in CSO contaminant concentrations because of decreased dilution of raw sewage and accumulation of in-sewer deposits. On the other hand, such intervention methods may be successful in decreasing risks posed by urban flooding (de Man et al., 2014). So a comparison of the effects of adaptation measures on risks from both exposure pathways should be subject to further studies to determine the most efficient intervention method with respect to public health.

5. Conclusions

- Frequency, volume and duration of CSO in the Netherlands are expected to increase, but changes during the bathing season are minor.
- Concentrations downstream of WWTPs and CSOs decreased by up to 46% for *Campylobacter*, that, because of its temperature sensitivity, was mainly affected by the increase in die-off caused by rising temperatures.
- Concentrations of *Cryptosporidium* and norovirus downstream of WWTPS hardly changed, while downstream of CSOs concentrations increased by up to 24 and 13% respectively.
- Fortunately, the concurrence of CSO events and bathing habits avert a large increase in infection risks for those pathogens.
- However, a decrease in flow rates could significantly increase the infection risks of pathogens that are relatively stable like *Cryptosporidium* and norovirus.
- Overall, average risks are found to be higher downstream WWTPs compared to CSOs. Especially with regard to decreased flow rates, adaptation measures applied to treatment at WWTPs may be more beneficial for human health than decreasing CSO events.

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