How can the UK statutory *Cryptosporidium* monitoring be used for Quantitative Risk Assessment of *Cryptosporidium* in drinking water?

P. W. M. H. Smeets, J. C. van Dijk, G. Stanfield, L. C. Rietveld and G. J. Medema

ABSTRACT

Quantitative Microbiological Risk Assessment (QMRA) is increasingly being used to complement traditional verification of drinking water safety through the absence of indicator bacteria. However, the full benefit of QMRA is often not achieved because of a lack of appropriate data on the fate and behaviour of pathogens. In the UK, statutory monitoring for *Cryptosporidium* has provided a unique dataset of pathogens directly measured in large volumes of treated drinking water. Using this data a QMRA was performed to determine the benefits and limitations of such state-of-the-art monitoring for risk assessment. Estimates of the risk of infection at the 216 assessed treatment sites ranged from $10^{-6.5}$ to $10^{-2.5}$ person^{-1} d^{-1}. In addition, *Cryptosporidium* monitoring data in source water was collected at eight treatment sites to determine how *Cryptosporidium* removal could be quantified for QMRA purposes. *Cryptosporidium* removal varied from 1.8 to 5.2 log units and appeared to be related to source water *Cryptosporidium* concentration. Application of general removal credits can either over- or underestimate *Cryptosporidium* removal by full-scale sedimentation and filtration. State-of-the-art pathogen monitoring can identify poorly performing systems, although it is ineffective to verify drinking water safety to the level of 10^{-4} infections person^{-1} yr^{-1}.

Key words | *Cryptosporidium*, drinking water, QMRA, risk assessment, statutory monitoring, treatment

P. W. M. H. Smeets (corresponding author)
G. J. Medema
Kiwa Water Research,
PO Box 1072, 3430 BB, Nieuwegein,
The Netherlands
Tel.: +31 30 606 9511
E-mail: *patrick.smeets@kiwa.nl*

G. Stanfield
WRC plc,
Frankland Road, Blagrove, Swindon
Wiltshire SN5 8YF,
UK

J. C. van Dijk
L. C. Rietveld
Delft University of Technology,
PO Box 5048, 2600 GA, Delft,
The Netherlands

INTRODUCTION

Most water supplies monitor their drinking water for the occurrence (absence) of indicator organisms in a relatively small volume of water (<500 mL), but absence in this volume only may not guarantee safe drinking water. Additional approaches to safeguard drinking water quality, such as the Surface Water Treatment Rule (USEPA 2006) and Water Safety Plans (WHO 2004), have therefore been introduced. Quantitative Microbiological Risk Assessment (QMRA) (Haas *et al.* 1999) is a method that can be used to estimate the health risk associated with drinking water consumption. The risk of infection is calculated from the exposure to pathogens (the chance of ingesting one or more pathogens) and the dose–response relation (the chance of infection from the number of pathogens ingested). In most situations it is not feasible to monitor directly for the presence of pathogens since they are present in extremely low numbers. However, the UK statutory monitoring for *Cryptosporidium* has resulted in a large quantity of data from the monitoring of large volumes of drinking water. The *UK Water Supply Regulations (Amended) 1999* (DWI 1999), that were introduced as a treatment- rather than a health-based standard, require continuous monitoring of finished drinking water for *Cryptosporidium* oocysts at least 23 h per day at a rate of at least 40 L per hour. This could be

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considered the best possible pathogen monitoring programme that can be achieved given the current state of the art of pathogen analysis. The database of monitoring results provides a unique opportunity to study drinking water safety based on measured pathogens in drinking water. The monitoring is performed to ensure compliance with the UK drinking water treatment standard of less than 1 oocyst per 10 L of final water. The endpoint of QMRA is an estimated risk of infection, and Dutch drinking water regulations require a maximum risk level as low as 10^{-4} per person per year. This study investigated whether extensive monitoring of treated water as applied in the UK could be used to verify compliance to this risk level.

Since most pathogens are not monitored as extensively in drinking water, a different approach is generally used in QMRA. Pathogens are monitored in source water and their reduction through treatment is estimated so the concentration in drinking water can be calculated (Regli *et al.* 1991). An essential step in such a risk assessment is determining the reduction of pathogens by drinking water treatment (Teunis *et al.* 1997; Gibson *et al.* 1999). Generally, reduction has to be estimated from indirect information such as: process removal credits from literature or pilot experiments, removal of surrogates like turbidity, reduction of indicator organisms such as *E. coli* or modelling of disinfection processes. These indirect measurements of pathogen reduction all have their specific shortcomings. Direct measurement of pathogens before and after treatment could provide a direct measurement of treatment efficacy. For some sites *Cryptosporidium* monitoring data was collected for the source water to see if this would aid the modelling of treatment in QMRA.

Changes in conditions that are outside the limits normally experienced at a treatment works are often referred to as extreme events. The most common extreme event that is considered is heavy rainfall (storms) in the catchment of the treatment works. Abnormally high rainfall can wash pathogens from agricultural land used for grazing into the river (the source water). If there has been little or no rainfall previously, the increased numbers of pathogens scoured from the land is not offset by the increased dilution and, at least initially, the numbers of pathogens in the source water will increase. The flow rate of the river also increases, causing high turbidity and carrying pathogens more quickly to the abstraction point of the water treatment works. The concern is whether the treatment works can handle this increased microbial challenge or if the quality of the treated water will deteriorate in relation to the increased loading in the source water and how this can be modelled in QMRA.

The goal of this study was to test whether elaborate end-product testing can provide valuable information for quantitative risk assessment. On the one hand, risk of infection was estimated directly from the treated water monitoring. On the other hand, *Cryptosporidium* reduction by treatment was assessed by comparing *Cryptosporidium* in the source and treated water. Finally the impact of peak events in the source water on treated water quality was studied.

METHODS

For this study, the results of statutory *Cryptosporidium* monitoring at 216 UK water treatment works from 1/4/2000 until 31/3/2002 were obtained. The daily sampled volume and the oocysts concentration in that volume were used in the data analysis.

In addition, data from source water monitoring at 8 UK water treatment works from 5/1/1993 to 20/4/2004 was obtained. These samples were taken at irregular intervals, typically between one week and one month. The treatment schemes at these sites are presented in Table 1. All sites apply coagulation, sedimentation, filtration, GAC filtration and chlorination. At sites A and E ozonation is also used and at site B there is also dissolved air flotation.

To study the effect of source water peaks one UK water company (not described in Table 1) submitted *Cryptosporidium* monitoring data from one of its water treatment works that was fortuitously collected during a period of extreme rainfall. At this treatment works, the river water is abstracted for treatment. The first stage of treatment consists of bank-side storage of around 2 d. The initial coagulation stage is a process that uses sand and polyelectrolyte with aluminium sulfate. After settlement the water is passed through a bank of rapid gravity filters. Ozone is

Table 1 | Treatment processes at sites A–H

Site	Treatment
A	Coagulation, sedimentation, GAC filtration, ozone, chlorination
B	Impoundment, coagulation (polyelectrolyte), sedimentation, dissolved air flotation, filtration, GAC filtration, chlorination
C	Coagulation (polyelectrolyte), sedimentation, filtration, GAC filtration, chlorination
D	Impoundment, coagulation (polyelectrolyte), sedimentation, filtration, GAC filtration, chlorination
E	Coagulation, (polyelectrolyte) sedimentation, filtration, GAC filtration, ozone, chlorination
F	Coagulation, (polyelectrolyte) sedimentation, filtration, GAC filtration, chlorination
G	Coagulation, (polyelectrolyte) sedimentation, filtration, GAC filtration, chlorination
H	Coagulation, sedimentation, filtration, GAC filtration, chlorination

added and, after contact, the water passes through GAC filters before final chlorination. Statutory monitoring for *Cryptosporidium* is carried out on the final water from this works. During the period, immediately before and for two weeks after, additional source water monitoring was carried out in the form of one 10 grab sample each day.

The methods used for *Cryptosporidium* monitoring and analysis in drinking water in connection with the *UK Water Supply Regulations (Amended) 1999* are strictly controlled by the UK Drinking Water Inspectorate (DWI). The methods are detailed in supplements to the Regulations and include a requirement to provide a high level of security such that a "chain of evidence" is produced to allow the results to be admissible in a court of law. The DWI stipulated methods include Sampling and Transportation of Samples (Part 1), Laboratory and Analytical Procedures (Part 2), Validation of New Methods (Part 3) and Requirements for the Inter-laboratory Proficiency Scheme (Part 4). All laboratories undertaking *Cryptosporidium* analysis in connection with the regulations must take part in the inter-laboratory scheme. In addition, the laboratories are regularly inspected by inspectors from the DWI and are awarded a license of proficiency that can be revoked at subsequent inspections if standards have fallen below the required standard. The regulations are given in full on the DWI internet site (www.dwi.gov.uk). Basically the method of sampling and analysis involved filtration at the sampling point using an IDEXX Filter Max© filter or a Pall Life Sciences Envirochek™ filter. On return to the laboratory the filters go through an elution process. The eluent is then treated with centrifugation and immuno-magnetic separation to concentrate the oocysts. The concentrate is then treated with an immuno-fluorescent reagent and examined microscopically. The requirements stipulated in the DWI Standard Operating Procedures (SOP) are extremely strict about each of the stages of the analyses in an attempt to retain the "chain of custody", maximise the efficiency of recovery and provide the best means of comparing results from different laboratories. For example, the SOP for microscopic examination requires that, where an initial result of 0.5 oocysts per 10 L is obtained, the microscopic examination should be checked by another approved microscopist in the same laboratory. If the results of the initial analysis suggests more than 0.7 oocysts per 10 L or there is doubt about identification of some "oocysts", the microscopic examination should be checked by one of the DWI approved microscopists from another organisation as well as being examined by another microscopist in the same laboratory. The recovery of the method is approximately 30–60%; recovery is not determined for individual samples (DWI 2006). On the one hand, such recovery leads to an under-estimation of *Cryptosporidium* concentrations: on the other hand, only part of the counted oocysts are human pathogenic, viable and infectious (Aboytes *et al.* 2004). Since there is insufficient data to quantify these effects, it was assumed that the net effect on the assessment was negligible.

Oocyst concentrations were plotted in a Complementary Cumulative Distribution Function (CCDF) graph of exceeded concentration on a log–log scale. Non-detects are not shown but determine the starting point of the graph. To determine the impact of possible oocysts concentrations represented by the non-detects, three approaches were used to extrapolate the measured oocyst concentrations to below

detection limits. For the minimal estimated risk of infection, these concentrations were set to 0, assuming that no oocysts were present in the drinking water produced on that day. For the maximum estimate, these samples were set to the detection limit (1 oocyst per 1000 L). For the best estimate the concentration in the non-detect samples was extrapolated linearly on a log–log scale (log–linear) as

$$\log_{10}(\text{frequency}) = a + b\log_{10}(C_{\text{crypto}})$$

where C_{crypto} is the *Cryptosporidium* concentration in oocysts L^{-1}. Parameters a and b were determined from the lowest 10 measured concentrations. If the number of positive samples was between 2 and 10, all samples were used for the extrapolation. Single or no positives were not extrapolated.

Source water was generally monitored in 1–10 L samples. Recovery in the source water can be influenced by water quality changes such as high turbidity (Nieminski *et al.* 1995; Wohlsen *et al.* 2004). In this study the recovery for all samples was assumed equal since the recovery was not assessed for individual samples. The reduction of *Cryptosporidium* at the assessed treatment sites A–H relies on the physical removal by sedimentation and filtration. Chlorination has a very limited effect on *Cryptosporidium* under normal operating conditions. Sites A and E apply ozonation which could inactivate oocysts. Since the inactivated oocysts are not removed from the water, they are counted by the analysis. Therefore this assessment only evaluated the physical removal of *Cryptosporidium* by treatment. Average concentrations before and after treatment at each site were calculated as the total number of oocysts counted in the total sample volume. Log removal was calculated from the average concentrations. The number of *Cryptosporidium* in the source water and the total volume of the treated water samples determine the maximum level of reduction that can be demonstrated at a site. The demonstrable *Cryptosporidium* reduction by treatment works was calculated by assuming that only one *Cryptosporidium* was counted in the total treated water sample volume.

Risk of infection was calculated both as a point estimate based on average values and as a varying risk using Monte Carlo simulation. Treated water *Cryptosporidium* concentrations were determined from the data. Drinking water consumption in the UK was modelled with a Poisson distribution for the number of 190 mL glasses per day (mean of 2.81 glasses) (Mons *et al.* 2007). For the point estimate the mean consumption of 0.53 L was applied. A beta-Poisson dose–response model ($\alpha = 0.115$, $\beta = 0.176$) was used to calculate the risk of infection when exposed to a concentration of *Cryptosporidium* (Teunis *et al.* 2002):

$$P_{\text{inf\_d}} = 1 - \left(1 + \frac{C_{\text{crypto}} \times consumption}{\beta}\right)^{-\alpha}$$

where $P_{\text{inf\_d}}$ is the daily risk of infection (person^{-1} d^{-1}) and *consumption* is the volume of daily consumed unboiled drinking water (L). The yearly risk of infection $P_{\text{inf\_y}}$ (person^{-1} yr^{-1}) is generally approximated by 365 times the daily risk of infection. However, at daily risks above $10^{-3.5}$ (person^{-1} d^{-1}) this approach significantly overestimates the actual risk. Therefore the yearly risk of infection was calculated as

$$P_{\text{inf\_y}} = 1 - (1 - P_{\text{inf\_d}})^{365}$$

The Monte Carlo simulation was performed by 100 000 independent draws from monitored *Cryptosporidium* concentrations and the applicable type of extrapolation.

RESULTS AND DISCUSSION

Overview of monitoring results

The treated water results from 216 sites, including sites A–H, were collected. Table 2 provides an overview of these results.

Figure 1 shows the frequency of observed concentrations is almost linear on a log–log scale. The minimal, maximal and log–linear extrapolation of negative sample results are also represented in the graph. For each concentration the exposure to *Cryptosporidium* ($C_{\text{crypto}} \times$ *frequency*) was calculated to show the relative impact of the observed concentrations. The exposure at relatively low concentrations between 0.001 and 0.01 oocysts L^{-1} seems to be five times higher than the exposure at high concentrations between 0.5 and 1 oocyst L^{-1}, since the latter rarely occur.

Table 2 | Overview of statutory UK *Cryptosporidium* monitoring results

Number of sites	216	
Number of samples	97,997	
No. positive	5353	(5.5%)
Total sample volume	115,303,050	L
Total no. oocysts counted	24919	
Average oocysts concentration	0.000,216	oocysts.L^{-1}
No. non-compliance (>0.1 oocysts.L^{-1})	18	(0.018%)

For sites A–H 4214 samples of source water and 5579 samples of finished water, representing 133 500 and 6826 549 L of tested water, respectively, were collected. Each of the 8 sites had analysed approximately 850 000 L of finished water in 700 samples of 1200 L over a period of 23 months. Table 3 provides an overview of the monitoring results.

The period of source water monitoring varied per site and did not (completely) overlap with the study period of finished water monitoring. Table 4 compares the overlapping source water monitoring period to all source water monitoring. All the source water data was used in the analysis of treatment efficacy, since it provided a better representation of source water concentration and variation than the limited number of samples in the overlapping period.

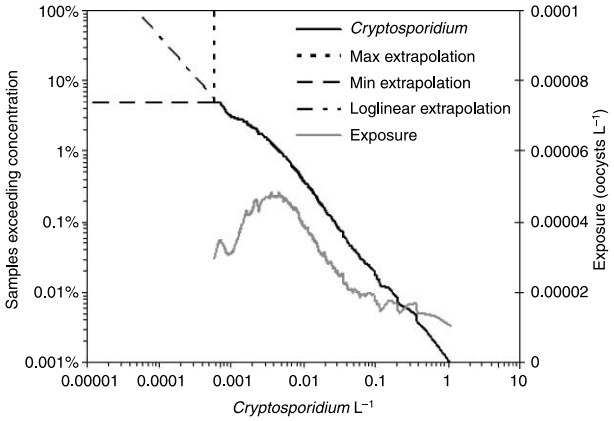

Figure 1 | Complementary Cumulative Frequency Distribution (CCDF) of observed *Cryptosporidium* concentrations (black), exposure due to these concentrations (grey) and three approaches to extrapolate the 94.5% negative samples (dashed).

Seasons may have an impact on both the concentrations of *Cryptosporidium* in the source water and treatment efficacy. Between 25–50% of the source water samples were positive for *Cryptosporidium*. The positive monitoring results for all sites were plotted against the day of the year in Figure 2. The only seasonality observed is a decrease of *Cryptosporidium* in August and a slight increase of positives from September to December for most sites.

Seasonal variations, like temperature or algal blooms, could lead to compromised treatment, resulting in *Cryptosporidium* in the drinking water. *Cryptosporidium* was detected in 5.5% of all treated water samples. Figure 3 shows how these samples were spread over the year. From January to March relatively little *Cryptosporidium* were detected, whereas the number of positive samples doubles in July.

The source water event data collected for a single site during the period before and after rainfall are shown in Figure 4. The only detection of *Cryptosporidium* in the final water occurred on day 32 when one oocyst was detected in approximately 1200 L.

QMRA based on treated water monitoring

The risk from the combined monitoring results of all sites was assessed under the assumption that each site produces the same amount of drinking water. The mean $P_{\text{inf\_d}}$ in Table 5 is the average risk of infection determined by Monte Carlo simulation. The point $P_{\text{inf\_d}}$ is the point estimate of average risk based on mean *Cryptosporidium*

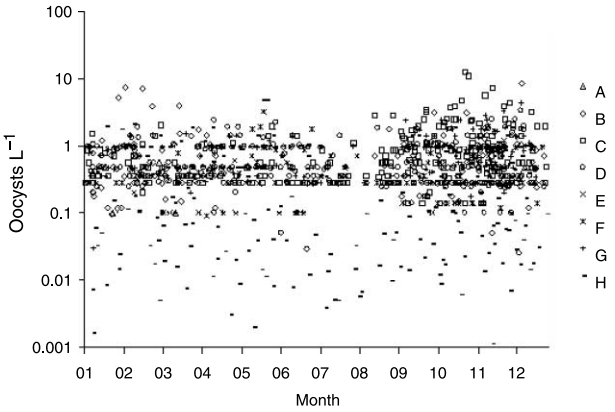

Figure 2 | Yearly observations of *Cryptosporidium* in source water (non-detects are not shown).

Table 3 | Results of source water and finished water monitoring at sites A–H

	Source water				Treated water			
Site	No. samples	% positive	Oocysts counted	Volume analysed (L)	No. samples	% positive	Oocysts counted	Volume analysed (L)
A	470	2	8	2746	702	0.57	4	817,989
B	643	18	179	3561	691	0.43	6	828,326
C	489	43	701	1289	710	0.42	3	820,731
D	1025	24	275	3756	698	0.29	2	863,874
E	25	44	36	292	699	0.72	5	948,163
F	467	20	99	2083	698	0.86	6	834,655
G	397	39	191	1335	695	2.01	14	889,946
H	698	31	332	118,473	686	2.92	37	822,861

concentration, drinking water consumption and dose–response. The point estimate results in a slightly higher estimation of risk for all approaches. The estimated risk varies by a factor of 4 (0.6 log), depending on the approach for extrapolation of measured *Cryptosporidium* concentrations below the detection limit. The difference between the log–linear and minimal approach is less than 0.03 log. The uncertainty caused by the negative samples has very little impact on this risk assessment. The yearly average risk of infection is 2.8×10^{-2} person^{-1} yr^{-1}.

The risk of infection was also assessed for individual sites based on their monitoring results. As an example, Figure 5 shows the distributions of *Cryptosporidium* concentrations in treated water for sites 22–28. Several typical distribution forms were observed. Sites 24 and 25 show a typical distribution where over 10% of the samples is

Table 4 | Average and maximum *Cryptosporidium* concentration (oocysts L^{-1}) in source water during treated water study period and all source water monitoring

	Source water monitoring during treated water study period			All source water monitoring results		
Site	No. samples	Average concentration	Maximum concentration	No. samples	Average concentration	Maximum concentration
A	16	0.0008	0.10	470	0.0029	0.57
B	7	0.0592	1.54	643	0.0503	8.48
C	2	–	–	489	0.5438	420.00
D	28	0.0158	0.42	1025	0.0732	3.33
E	4	0.3095	1.20	25	0.1233	1.20
F	13	0.0021	0.40	467	0.0475	3.21
G	15	0.0567	1.00	397	0.1430	4.29
H	119	0.0021	1.00	698	0.0028	4.76

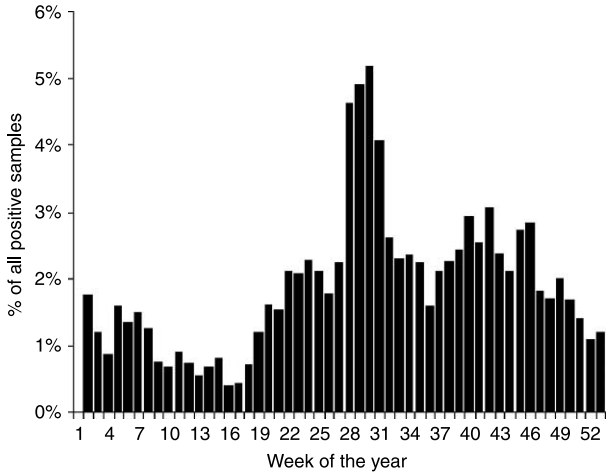

Figure 3 | Yearly distribution of all treated water samples containing *Cryptosporidium*.

positive. The frequency of occurrence decreases with increasing concentrations. Site 23 has a similar distribution, but is susceptible to peak events, shown by a higher increase of observed concentrations at decreasing frequencies. Site 22 is an example of a site with highly variable concentrations. Sites 27 and 28 show less variation at 1% and 10% positive samples, respectively. Finally at site 26 only one oocysts was found in one sample, so it could not be extrapolated.

The risk of infection was estimated with Monte Carlo analysis using the three different approaches to deal with non-detect samples. Figure 5 shows the resulting frequency of concentrations for the log–linear extrapolation. The assessed daily risks for these sites are presented in Table 6. Since the minimal assessment of risk at sites 22, 23, 24 and 25 is relatively high it is hardly impacted by the way the negative samples are interpreted. At sites 27 and 28 extrapolation of the measured *Cryptosporidium*

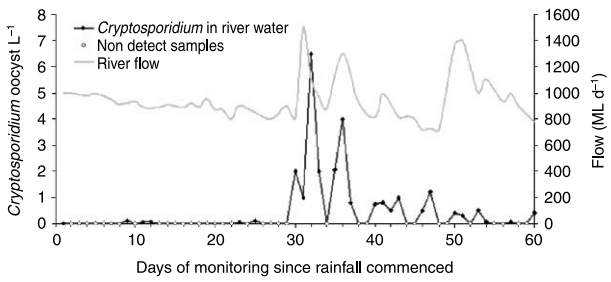

Figure 4 | Observation of peak contamination of *Cryptosporidium* in river water in relation to flow. A single oocyst was detected in a 1200 L treated water sample on day 32 (not shown).

Table 5 | Daily risk of infection for the combined 216 UK sites included in the study

Extrapolation	Monte Carlo simulation mean P_{inf_d}	Point estimate of P_{inf_d}
Minimal	7.45×10^{-5}	8.08×10^{-5}
Log–linear	7.98×10^{-5}	8.75×10^{-5}
Max.	26.96×10^{-5}	28.20×10^{-5}

concentrations increases the assessed risk by one or two log units. The few "extra" oocysts from the extrapolation have a strong impact on the low average concentration at these sites. Secondly, the few positive samples are all very low and show little variation. When this little variation is extrapolated below the detection limit, this results in a large percentage of the water being just below the detection limit (steep extrapolation in Figure 5). Site 26 shows that a single detected oocyst is sufficient to achieve an average yearly risk of infection exceeding 10^{-4}. When no *Cryptosporidium* is detected, assumptions on how to interpret these negative samples can still yield a risk exceeding 10^{-4} person^{-1} yr^{-1} (or 2.7×10^{-7} person^{-1} d^{-1}).

The frequency of minimal estimates of daily risk of infection at all 187 sites where *Cryptosporidium* was found (Figure 6) ranges from $10^{-6.5}$ to $10^{-2.5}$ person^{-1} d^{-1}.

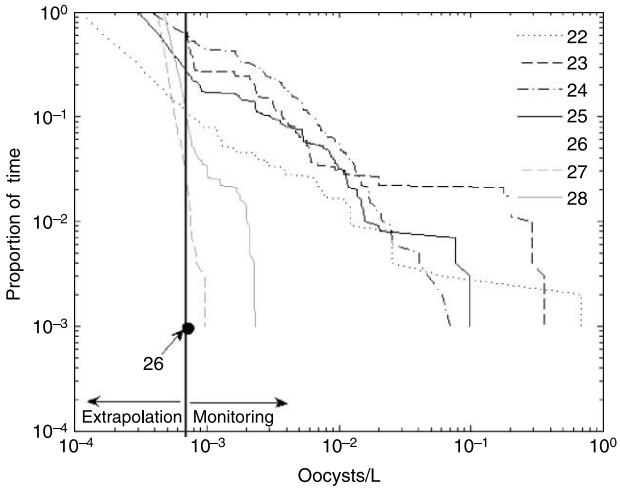

Figure 5 | Monitored *Cryptosporidium* concentrations in treated water at sites 22–28 including extrapolation by Monte Carlo simulation.

Table 6 | Risk assessment results for 7 individual sites

			Daily risk			
Site number	**No. of samples**	**% positive**	**Minimal**	**Log–linear**	**Max.**	**Yearly risk minimal**
22	417	7.7	3.8×10^{-4}	4.25×10^{-4}	8.06×10^{-4}	1.5×10^{-1}
23	201	53.7	1.6×10^{-3}	1.60×10^{-3}	1.73×10^{-3}	4.4×10^{-1}
24	453	53.9	9.6×10^{-4}	1.01×10^{-3}	1.08×10^{-3}	2.9×10^{-1}
25	459	29.0	4.8×10^{-4}	6.20×10^{-4}	7.59×10^{-4}	1.9×10^{-1}
26	454	0.2	4.7×10^{-7}		3.43×10^{-4}	1.7×10^{-4}
27	454	0.9	2.6×10^{-6}	1.67×10^{-4}	3.44×10^{-4}	9.3×10^{-4}
28	454	7.3	3.1×10^{-5}	2.07×10^{-4}	3.50×10^{-4}	1.1×10^{-2}

The minimal risk estimate for the 29 sites where no *Cryptosporidium* was detected could be reported as $< 5 \times 10^{-7}$ person^{-1} d^{-1}and the maximum risk estimate is approximately 3.4×10^{-4} person^{-1} d^{-1}. Risk levels were equally spread among sites with different sources (river, spring, groundwater or undefined), although the highest risks ($10^{-3.5}$ to $10^{-2.7}$ person^{-1} d^{-1}) were not observed for groundwater and springs. There was no correlation between treatment throughput, number of samples or total monitoring volume and the risk of infection.

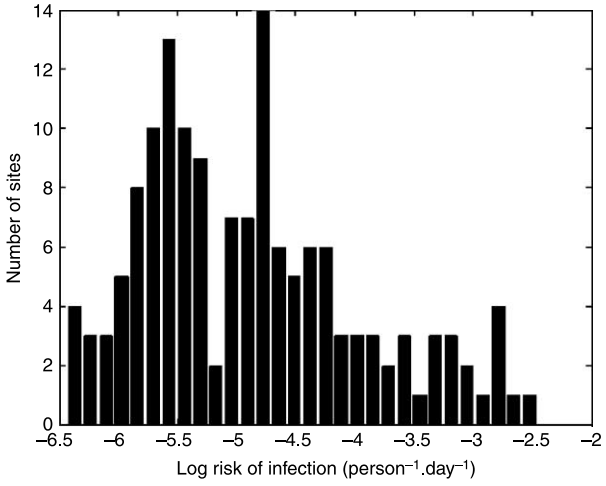

Figure 6 | Frequency of minimal estimates of log of risk of infection at the 187 sites where *Cryptosporidium* was found. At 29 sites no *Cryptosporidium* was detected.

Cryptosporidium removal

Table 7 and Figure 7 provide an overview of the overall *Cryptosporidium* removal at sites A–H. The average treated water concentration is very similar for most sites, so the observed log removal is dominated by the source water concentration. Due to the small number of oocysts found in treated water, the assessed log reduction is slightly less than the demonstrable log reduction for most sites.

Since the physical treatment processes at sites A, C, D, E, F, G and H are of the same type one would expect similar log removal. A literature study resulted in a best estimate of 3.2 log removal of *Cryptosporidium* by conventional treatment (sedimentation, filtration) with a range of 1.4–5.5 log (Hijnen *et al.* 2005*a*). Indeed 1.8–5.2 log removal was determined in this study. Site B could have provided more removal with the additional dissolved air flotation: however, this was not observed. Since treated water concentrations are similar at all sites, the removal appears to be related to the source water concentration. False positives or contamination in the laboratory could lead to such an observation. However, this seems extremely unlikely for the analysis of *Cryptosporidium* given the strictness of the analysis procedures and the quality assurance regime demanded by the DWI. The following explanations for the differences in observed removal at these sites are discussed:

1. The treatment design and operation is tailored to the water treated to comply with the drinking water standard.

Table 7 | Overall log removal at the treatment sites and demonstrable log removal

Site	Average source concentration	Average treated concentration	Log removal	Demonstrable log removal
A	0.0029	0.000 0049	2.78	3.38
B	0.0503	0.000 0072	3.84	4.62
C	0.5438	0.000 0037	5.17	5.65
D	0.0732	0.000 0023	4.50	4.80
E	0.1233	0.000 0053	4.37	5.07
F	0.0475	0.000 0072	3.82	4.60
G	0.1430	0.000 0157	3.96	5.10
H	0.0028	0.000 0450	1.79	3.36

2. *Cryptosporidium* is accumulated in the treatment up to a steady state where sporadically some *Cryptosporidium* are released, independent of the source water concentration.
3. Short high peaks of *Cryptosporidium* in the source are the cause of the positive samples in treated water. These peaks are not demonstrated by the infrequent monitoring of source water but are found in the continuous treated water sampling.

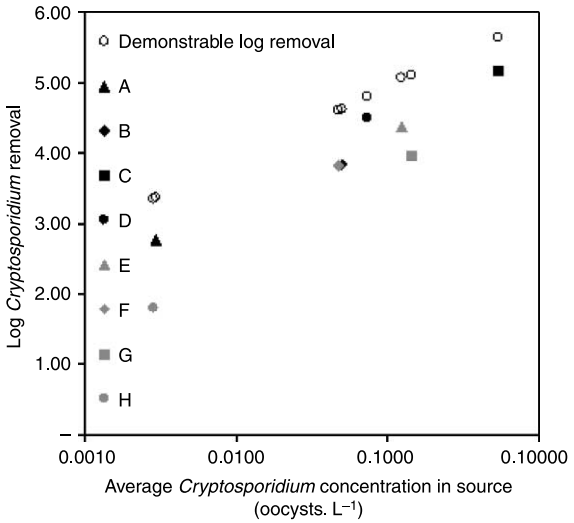

Figure 7 | Log reduction based on *Cryptosporidium* monitoring and demonstrable log reduction (if no *Cryptosporidium* had been found) at eight drinking water treatment sites in the UK.

4. Short "failure" of treatment occurs at a similar frequency at all sites.
5. Sedimentation and filtration are more effective to remove high concentrations of micro-organisms than low concentrations.

Design and operation

Sedimentation is generally optimised for the local situation. Regular jar tests are performed to determine optimal dosing of coagulant and coagulant aid and rapid and slow mixing energy. The removal of suspended matter or turbidity is thus optimised to reach a low turbidity after sedimentation. Polluted source water with high turbidity requires a better and more frequent optimisation than clean source water with low turbidity. The better optimised system also provides more *Cryptosporidium* removal, resulting in a relationship between source pollution and *Cryptosporidium* removal. Dugan *et al.* (2001) found that optimisation of sedimentation can improve *Cryptosporidium* removal from less than 2 to more than 5 log. In addition, a filter receiving higher turbidity will show breakthrough of turbidity after the filter, so the filter can be backwashed in time. When turbidity is low before filtration, breakthrough may go unnoticed. Finally both sedimentation and filtration perform optimally within a range of suspended solids concentrations. Insufficient material can be present to form

suitable flocks for sedimentation, while ripening of filters also requires sufficient suspended matter. Thus higher pollution drives treatments to improve their performance to meet both *Cryptosporidium* and other standards.

Accumulation

Filtration processes capture particles (including *Cryptosporidium*) in the filter bed. Periodically the filter is backwashed to remove accumulated particles when the head loss over the filter has increased too much or when breakthrough of turbidity is observed. Although cleaner water will result in longer run times, the total loading of the filter bed before backwashing will be similar for filters treating differently polluted waters. So, regardless of the water quality, the amount of *Cryptosporidium* accumulated in the filter bed during a filter run is independent of the amount of *Cryptosporidium* in the source water. Due to change of flow, some of the accumulated *Cryptosporidium* may detach from the filter material and leave the filter with the filtrate. Thus the number of *Cryptosporidium* after a filter could be more related to the loading of the filter than to the actual source water concentration. In the UK a system referred to as "slow start up" is being used by water companies to bring a filter back on line after backwashing. This procedure has been shown to reduce the risk of residual oocysts in the filter being washed into the filtrate as the filter compacts after backwashing (WHO 2004). Still, some form of attachment and release might occur at a low level.

Peaks in source water

The data for sites A–H were analysed to determine whether a recorded peak concentration in the source water had led to *Cryptosporidium* in the treated water. The source and treated water samples were combined by date. This showed that 57 treated water samples were positive, but unfortunately source water samples on the same date were only available on 5 occasions and 4 of these were negative for *Cryptosporidium*. The remaining single positive sample did not show a particularly high concentration. Neither were peaks in the source water observed in the periods preceding the 57 positive treated water samples. Comparing the average source water concentration to the maximum concentration in Table 4 shows that peaks of 10–200 times the average concentration have been recorded. So, although these peaks could potentially lead to peaks in the treated water, this could not be confirmed from the UK statutory monitoring data.

The reported source water peak event in Figure 4 was studied in detail. A single oocyst was detected in the treated water on day 32. Because of the bank-side storage for 2 d the "paired" source water value was a no detect in 10 L on day 29. However, due to mixing or preferential flow in the bank-side storage, the peak of 65 oocysts in 10 L in the river water on day 32 may have led to an increased concentration at the intake of the treatment works on the same day. The fact that none of the treated water samples on subsequent days 33–40 was positive for *Cryptosporidium* implies that the single detected oocyst was not related to the peak in the source water. These findings suggest that the detection of oocysts in treated water is not always related to peaks in source waters.

Short treatment failure

The failure of equipment (e.g. a dosing pump, valve), installations (e.g. defective filter nozzle) or erroneous operation leading to decreased treatment performance is referred to as treatment failure. The occurrence of treatment failure is related to equipment age, maintenance and operational procedures. For the studied sites the frequency at which treatment failure occurs could be similar. However, if failure of treatment occurred at the same frequency at sites with different source concentrations this would lead to higher peaks at the more polluted sites. These sites only found low concentrations of one or two oocysts per sample, just like the less polluted sites. Therefore it is not likely that the similar occurrence of *Cryptosporidium* in the treated water of the studied sites is a consequence of treatment failure.

Reduction related to microbial density

Some studies have observed that, at high concentrations of micro-organisms before slow sand filtration, more removal was found (Hijnen *et al.* 2005*b*, 2006). Removal of spores of sulfite-reducing *Clostridia* ranged over three log units at full scale, and in some cases the concentration after filtration exceeded the influent concentration. They concluded that the high DEC values assessed during short-term dosing

experiments most likely are not predictive for full-scale conditions. They attributed this observed relation between micro-organism concentration and its removal to accumulation and release in the filter, as explained above.

Modelling treatment in QMRA

In QMRA, removal by treatment is modelled as a "removal-credit" or by a distribution of removal values. According to the LTSESWTR (USEPA 2003) a conventional treatment (coagulation, sedimentation and filtration) provides an average of 3 log *Cryptosporidium* removal when the treatment complies with the rule. The point estimates of removal at sites A–H show that in practice 1.8–5.2 log removal is achieved. So the type of treatment provides insufficient information to determine appropriate removal value(s) for *Cryptosporidium*. Local information can verify substantial higher removal, leading to a lower risk estimate. Information about removal of turbidity, particles or surrogate organisms could support a choice of removal value, although this could not be verified in this study. The distribution of *Cryptosporidium* in treated water might be related to its distribution in source water. Therefore the frequencies of *Cryptosporidium* in source and treated water were combined in one figure for each site. In the example in Figure 8, treated water at site G shows less variation that at H. The source water varies more at site H. For these sites, a single removal-credit could be used to model removal by treatment in QMRA. However, these were the poorest performing sites. Since the number of positive samples after treatment was very low for sites A–F, variability could not be quantified. Since source water events do not lead directly to *Cryptosporidium* in the treated water, a single removal credit isn't appropriate for these sites. This leaves the question of how to model sedimentation–filtration in QMRA. One option is to relate the removal to the source water concentration. This leads to little variation in the treated water. However, both large and little variation in treated water was observed in this study. In order to improve knowledge about treatment efficacy this study could be expanded with additional information, such as design characteristics, additional source water data from other sites and on-line measurement of surrogates (turbidity) to match *Cryptosporidium* monitoring. Such a full-scale study at many sites could then lead to effective models of treatment efficacy for QMRA.

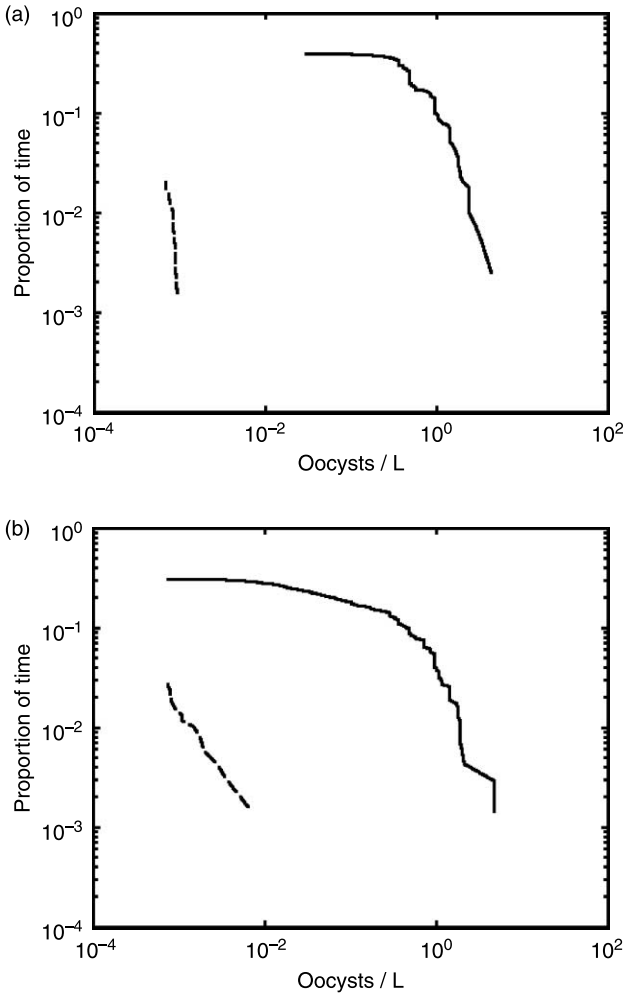

Figure 8 | Observed source water (line) and treated water (dashed) concentrations for sites G (a) and H (b).

CONCLUSIONS

State-of-the-art *Cryptosporidium* monitoring, as required in the UK, can be used to identify effectively those water supplies that do not reach a health-based target below 10^{-4} person^{-1} yr^{-1}. At that level, the impact of non-detect samples on the uncertainty of the risk estimate is limited. However, such monitoring is insufficient to verify a risk below the 10^{-4} level. When few *Cryptosporidium* oocysts are detected, the interpretation of the non-detect samples has a strong impact on the assessed risk level. Since such

extensive monitoring is unable to verify risk levels below 10^{-4} and such monitoring is not feasible for all pathogens, QMRA will need to rely on source water monitoring and modelling of removal by treatment. The study has shown that removal of *Cryptosporidium* by conventional treatment systems can range from 1.8–5.2 log. More removal was found at higher *Cryptosporidium* concentrations in source water. This may be due to better design and operation of such plants to meet treatment standards or due to accumulation and release of *Cryptosporidium* in the treatment processes. The use of a single average "removal-credit", as applied in the Surface Water Treatment Rule, can both underestimate or overestimate removal leading, respectively, to unnecessary actions or a false sense of safety. This underpins the need to perform site-specific assessments of treatment performance. Since extensive microbial monitoring is not feasible for most sites, a treatment model needs to be developed that appropriately describes the treatment's ability to deal with peak events in source water, but also predicts the number of *Cryptosporidium* that break through treatment during nominal operation.

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