Removal of *Cryptosporidium* by in-line filtration: effects of coagulant type, filter loading rate and temperature

Nicholas R. Dugan and Daniel J. Williams

ABSTRACT

Eight pilot-scale runs were performed to evaluate the impacts of coagulant type, filter loading rates and temperature on the removal of seeded *Cryptosporidium* oocysts by in-line filtration. The coagulant was either aluminum sulphate or ferric chloride, the filter loading rate was either 5 or 10 m/h and temperature was controlled at 20 or 4.5°C. *Cryptosporidium* samples were collected on an hourly basis during each run and analysed by solid phase cytometry. Depending on the operating conditions, observed average *Cryptosporidium* removals ranged from 1.0 log (std dev.=0.2 log) to >4.2 log (std dev.=0.1 log).

Key words | *Cryptosporidium*, direct filtration, solid phase cytometry

Nicholas R. Dugan (corresponding author)
Daniel J. Williams
United States Environmental Protection Agency,
Water Supply and Water Resources Division,
26 West Martin Luther King Drive,
Cincinnati,
OH 45268,
USA
Tel: +1 513-569-7239
Fax: +1 513-569-7172
E-mail: *dugan.nicholas@epa.gov*

INTRODUCTION

In-line and direct filtration are conventional treatment processes used to treat low turbidity, low colour, raw waters. In-line filtration is defined as coagulation and filtration, while direct filtration is defined as coagulation, flocculation and filtration (Cleasby *et al*., 1999). Typically, these treatment processes require lower coagulant doses, generate less sludge, require less space and result in lower capital, operation and maintenance costs (Cleasby *et al*., 1999). Because filtration is the only physical barrier, direct and in-line filtration have less overall removal capacity than treatment plants that incorporate a clarification step. As a result, the United States Environmental Protection Agency's (US EPA) Surface Water Treatment Rule (Federal Register, 1989) granted 0.5 log less *Giardia* removal credit for direct and in-line filtration than it did for treatment processes that included clarification. It is possible that the US EPA (2001*a*,*b*) will apply the same reasoning to in-line and direct filtration under the Long Term 2 Enhanced Surface Water Treatment Rule (LT2ESWTR). An analysis of the Information Collection Rule (ICR) database (US EPA, 1999) indicates that approximately 10 million people in the United States depend on direct and in-line filtration for their treated water. The ICR only collected data from water systems serving more than 100,000 customers. As a result, the true number of people receiving water from direct or in-line filtration plants may be somewhat higher.

Relatively few drinking water publications have examined the removals of seeded *Cryptosporidium* oocysts and surrogate parameters through direct and in-line filtration. The existing work is summarized in Table 1. Ongerth & Pecoraro (1995) maintained a target concentration of 5000 oocysts/l in the raw water for the duration of their 8-h runs. Nieminski & Ongerth (1995) seeded to attain a target concentration of 1.5×10^4 oocysts/l for three detention times, approximately 90 min, in each of their pilot-scale runs. In their full-scale runs, Nieminski & Ongerth (1995) added a spike of 1×10^7 oocysts. Patania *et al*. (1995) seeded 100 to 1000 oocysts/l for the duration of their runs. Edzwald *et al*. (1996) and Edzwald & Kelley (1998) spiked oocysts to achieve an influent concentration of 7200 oocysts/l for a pilot-scale run. Swaim *et al*. (1996) seeded 1×10^4 oocysts/l for the duration of three pilot-scale runs. Depending on operating conditions, 1.3 to 4.7 log *Cryptosporidium* removals were observed in the previously described studies. Although raw water temperatures sometimes varied significantly, none of the studies listed in Table 1 explicitly attempted to control for, and

Table 1 | Reported *Cryptosporidium* removals through pilot- and full-scale direct and in-line filtration

Authors	Treatment process	Coagulants	Temperature	Filter loading rates (m/h)	Log removals
Ongerth & Pecoraro (1995)	Direct filtration (pilot-scale)	Alum	14–20°C	12.5	1.5–3.1
Nieminski & Ongerth (1995)	Direct filtration (pilot- and full-scale)	Alum and anionic polymer (pilot-scale) Polyaluminium chloride and cationic polymer (full-scale)	Not reported	14.4 (pilot-scale) 12 (full-scale)	1.3–3.8 (pilot-scale) 2.6–2.9 (full-scale)
Patania *et al*. (1995)	In-line filtration (pilot-scale)	Alum, anionic polymer and cationic polymer	6.9–16°C	12.5–20	1.5–4.0
Edzwald *et al*. (1996) Edzwald & Kelley (1998)	In-line filtration (pilot-scale)	Ferric chloride and cationic polymer	5°C	7.3	4.7
Swaim *et al*. (1996)	In-line filtration (pilot-scale)	Ferric chloride and cationic polymer	18–19°C	29–38	3.0–4.4

examine the impacts of, temperature on *Cryptosporidium* removals.

In the current study, eight pilot-scale in-line filtration trials were performed to augment this existing body of data. In each trial, 1×10^5 oocysts/l were seeded into the plant influent for the duration of the run. These seeded *Cryptosporidium* concentrations were higher than the steady-state levels used in any of the previously mentioned studies. The goal of the higher influent *Cryptosporidium* concentrations was to increase the probability of detecting oocysts in the filter effluent. Increasing the probability of oocyst detection would reduce the statistical uncertainty associated with estimating log removals when no oocysts are found. The impacts of alum versus ferric chloride coagulation, with the doses optimized at 20°C, were investigated at filter loading rates of 5 and 10 m/h. These four trials were repeated at 4.5°C, using the coagulant doses established at 20°C. The results of the cold water trials provided an indication of *Cryptosporidium* removals under sub-optimal coagulation conditions.

METHODS

Figure 1 shows a schematic of the pilot plant. The treatment process consisted of *Cryptosporidium* addition, in-line mixing, coagulant addition, rapid mixing and filtration. The plant was designed to run at a flow rate of 0.45 l/min. The low flow rate permitted the seeding of *Cryptosporidium* at high concentrations while maintaining representative filter loading rates. This plant was the same as the one described by Lytle *et al*. (1998) and Dugan *et al*. (2001), with the exception that a bypass was installed around the flocculation and sedimentation basins. The applicability of very low flow rates was discussed by Dugan *et al*. (2001). In direct comparison trials, the sedimentation and filtration removals of turbidity, particles and spores were found to be within 0.2 log of those observed at a larger pilot plant. The absolute dimensions and flow rates of this second, larger, plant were over 10 times greater than those of the 0.45 l/min plant utilized for the current study. The large plant was also located at the US EPA and has produced data for several studies (Stevens *et al*., 1989; Goldgrabe *et al*., 1993; Wang *et al*., 1995; Miltner *et al*., 1995). On the basis of these comparison trials, it was felt that the 0.45 l/min plant could yield physical removal results representative of those observed in larger pilot-scale facilities.

Raw water from the Ohio River was used as source water for all runs and was stored in a 1.9×10^4 l stainless steel tank. The raw water characteristics for all eight runs

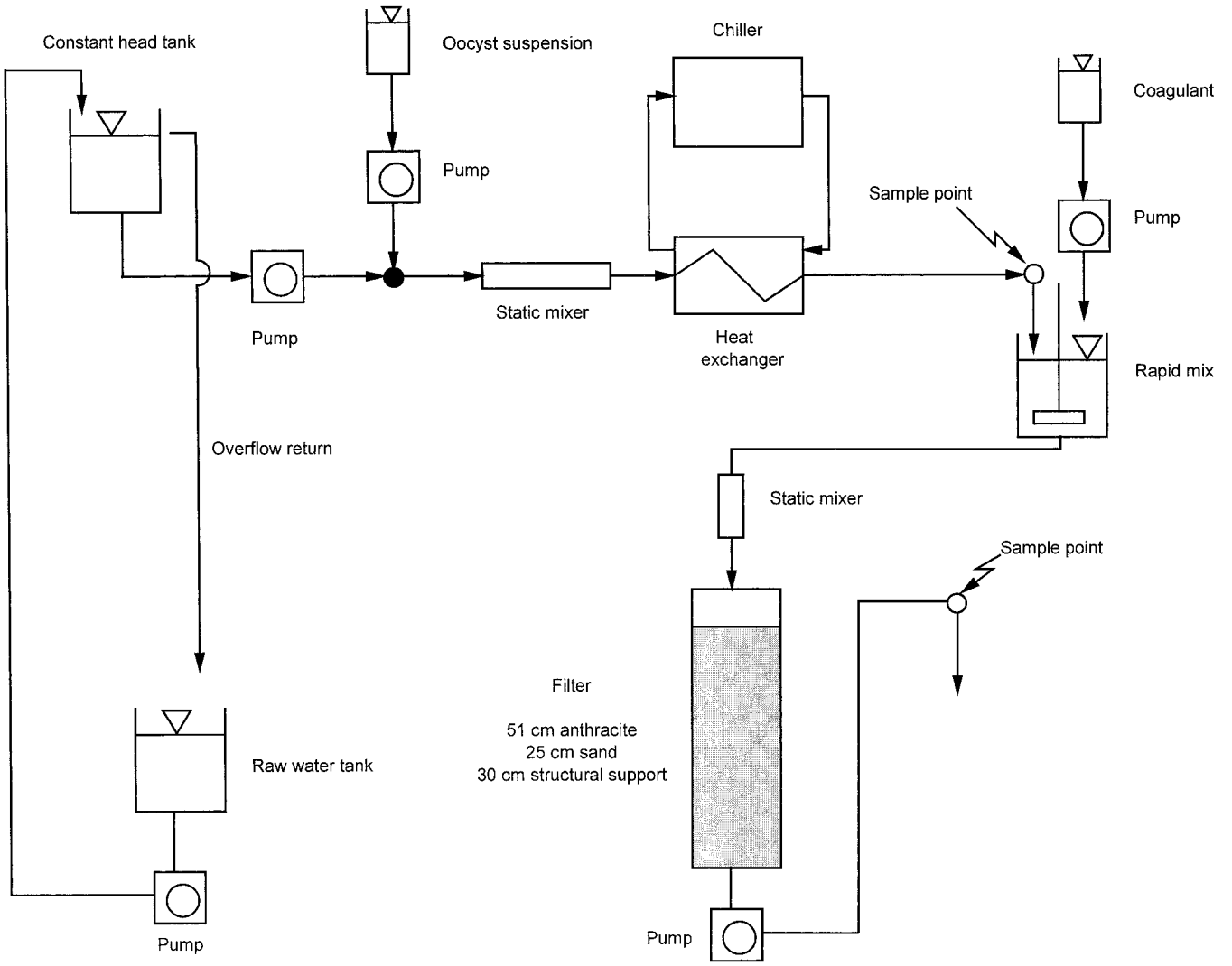

Figure 1 | Schematic of pilot plant.

are listed in Table 2. The source water was pumped to a constant head tank located at the front of the pilot plant. A gear pump (Cole-Parmer, Vernon Hills, IL), connected to the constant head tank, provided the influent water for pilot plant operation.

Unfixed, viable *Cryptosporidium parvum* oocysts were used in all trials. The oocysts, originally obtained from Harley Moon (National Animal Disease Center, Ames, Iowa) were propagated at the US EPA in female C57/BL6 mice. The oocysts were removed from the fecal material of infected mice by sieving and sucrose gradient centrifugation. Purified oocysts were stored in 0.01 M phosphate buffered saline (PBS) solution (GIBCO BRL, Grand Island, NY) with penicillin (100 units/ml) and streptomycin (100 μg/ml) at 4°C. During a pilot plant run, the oocysts were maintained in suspension in PBS solution at a concentration of 4.5×10^4 oocysts/ml. The feed suspension was constantly stirred to maintain an even oocyst distribution, and was sampled three times per run to verify oocyst density. The oocysts were fed to the raw water ahead of an in-line mixer (KoFlo, Cary, IL), prior to rapid mixing and coagulant addition. The feed rate of

Table 2 | Water quality parameters

Run	Turbidity (NTU)	Particles (#/ml×10^3)	TOC (mg/l)	pH	Alkalinity (mg/l as $CaCO_3$)	Total hardness (mg/l as $CaCO_3$)
1	0.26	7.7	2.2	8.1	81	150
2	0.20	7.6	2.1	8.1	76	150
3	0.22	8.2	2.2	8.0	80	150
4	0.22	7.8	2.2	8.0	80	150
5	0.22	6.6	1.9	7.9	84	160
6	0.26	10	2.3	8.0	73	150
7	0.28	10	—	—	—	—
8	0.26	9.6	1.5	7.9	74	150

Cryptosporidium suspension was 1.0 ± 0.2 ml/min in order to achieve a concentration of 1.0×10^5 oocysts/l in the raw water.

For runs at 20°C, the water temperature was governed by ambient temperature, which was maintained at 20 ± 1°C by the building ventilation system. For cold water runs, raw water was chilled to 2.5°C at the rapid mix using a shell and tube heat exchanger (Exergy, Hanson, MA). Raw influent water flowed through the tube side, while a 50/50 ethylene glycol/deionized water mixture was circulated counter current through the shell side. The glycol/water mixture was cooled in a recirculating chiller (Cole-Parmer, Vernon Hills, IL).

Alum ($Al_2(SO_4)_3 \cdot 14\ H_2O$) (Delta Chemical, Baltimore, MD) or ferric chloride ($FeCl_3 \cdot 6\ H_2O$) (Fisher Scientific, Pittsburgh, PA) was fed to the rapid mix operating at a fluid velocity gradient (G) of 1200 sec^{-1} and a detention time of 90 sec. Rapid mixing was achieved by rotating paddles, and the coagulant feed lines were arranged to discharge near the paddle tips. The water then flowed by gravity through an in-line mixer prior to filtration. The estimated detention time and G value for the in-line mixer were 2.4 sec and 150 sec^{-1}, respectively. Separate trials to evaluate the impact of the mixer on filtration performance under conditions similar to those used in this study indicated that the mixer had no appreciable impact.

Filters were 3.8 cm (inside diameter) glass columns (Corning Glass Works, Corning, NY), filled with 76 cm of media over 30 cm of graded gravel structural support. The media was 51 cm of anthracite over 25 cm of sand. The effective diameters of the sand and anthracite were 0.44 mm and 1.0 mm, respectively. The sand and anthracite had uniformity coefficients of 1.4 and 1.3, respectively. The filters were operated at loading rates of 5 or 10 m/h. To minimize temperature variations through the plant during cold water runs, 1.7 cm of closed cell foam rubber insulation (Grainger, Chicago, IL) was applied to the outsides of all tubing and filter columns. Temperatures at the rapid mix and filter effluent were monitored using thermocouple thermometers. Filter effluent temperatures during cold water runs were 6.5°C, so the average of the rapid mix (2.5°C) and filter effluent temperatures was 4.5°C.

The alum dose of 20 mg/l was determined by trial and error at the pilot scale. The plant was run at 20°C with a 5 m/h filter loading rate at successively higher alum doses. The 20 mg/l concentration was the lowest alum dose that yielded filter effluent turbidities less than 0.1 nephelometric turbidity units (NTU). The same procedure was

then used to determine the 10 mg/l ferric chloride dose. *Cryptosporidium* was not seeded during any of the trial and error tests used to determine coagulant doses. Alum and ferric chloride were the only coagulants investigated during this study. The exclusive use of either alum or ferric chloride was not necessarily representative of coagulation practice at direct filtration facilities. Many plants use lower doses of inorganic salts combined with a polymer. The advantage of exclusively using alum or ferric chloride was that the results would not depend on proprietary chemical formulations specific to a certain manufacturer. The coagulant doses were not optimized for the lower temperature of 4.5°C. Instead, the same coagulant doses used at 20°C were also used at 4.5°C. The higher effluent turbidities and particle counts at 4.5°C (Tables 3–10, Figures 2 and 3) indicated that the cold water coagulation conditions were not optimal. As a result, the cold water log removals cannot be directly compared to those at 20°C. Instead, the cold water data help to quantify the relative impacts of non-optimal coagulant selection and dosing on *Cryptosporidium* removal.

Natural organic matter removals were not used as a criterion for coagulant dose optimization. However, raw and filter effluent total organic carbon (TOC) data were collected during seven of the eight runs (Tables 3–10). Coagulation pH and ultra-violet light absorbance at 254 nm (UV_{254}) were not measured during the eight runs with *Cryptosporidium*. However, these parameters were measured in other in-line filtration work performed with similar raw water qualities, coagulants and coagulant doses. In those studies, UV_{254} removals averaged 37% with a standard deviation of 2.7%, and coagulation pH was 0.6 to 0.8 units lower than raw pH.

The pilot plant was flushed with Ohio River water prior to each run. The flushing procedure took 2 days and consisted of running the plant without *Cryptosporidium* addition using the coagulant and filter-loading rate anticipated for the next run. On the first day of the flushing procedure, the plant was run and the filter was backwashed three times during an 8-h period. The filter was backwashed with tap water (0.5–0.8 mg/l free chlorine residual) for 10 min at 50% bed expansion. On the second day, the plant was run without backwashing. Filter effluent samples were collected three times on the second day and analysed for background levels of *Cryptosporidium* oocysts. At most, one of the three samples could test positive for *Cryptosporidium*. The number of oocysts in the positive sample had to be less than or equal to 2 per 100 ml (the standard volume of filter effluent analysed during filtration experiments). If the background samples met these criteria, the plant was considered ready for the next run. If not, the flushing procedure was repeated until the criteria were met.

Each of the eight filtration experiments or runs lasted 8 h. *Cryptosporidium* feed was initiated when the plant started up. The first sample was collected 2 h after start-up and initiation of *Cryptosporidium* feed. As a result, filter ripening occurred while *Cryptosporidium* was being fed. Sampling then continued at hourly intervals, for a total of seven sampling events. Duplicate *Cryptosporidium* samples were collected at three of the seven events. Raw water *Cryptosporidium* samples were collected after the point of oocyst addition, immediately prior to the rapid mix. Raw water samples were collected by temporarily diverting most of the raw water flow from the rapid mix to the sample bottle. Filling a sample bottle with raw water took just over 30 s, during which time the water level in the filter was not significantly affected. Raw and filter effluent *Cryptosporidium* samples were collected in high-density polyethylene (HDPE) bottles. Raw (250 ml) and filter effluent (1 l) spore samples were collected in autoclaved 1 l HDPE bottles. Turbidity, particle and spore samples were collected during *Cryptosporidium* sampling events, at the same locations used for *Cryptosporidium*. Turbidity and particles were collected at each of the seven sampling events. Spore samples were collected at three of the sampling events, with duplicates collected at one of these three events.

All *Cryptosporidium* samples collected from seeded raw or filter effluent water were evaluated by solid-phase cytometry using a laser scanning device (ChemScan RDI, Chemunex, Paris, France). This technique was found to be faster and easier to perform than centrifugation and concentration of pellets. Raw and filter effluent volumes of 100 μl and 100 ml, respectively, were sampled from their containers, membrane filtered, and treated with fluorescein conjugated antibody stain (Waterborne Inc., New

Orleans, LA). The laser scan of the membranes excited fluorescence from the stained oocysts. The locations of these fluorescent responses were recorded in computer memory. Each set of stored coordinates was then analysed microscopically to confirm the presence of *Cryptosporidium*. Negative control samples were also run through the entire analytical process to monitor for cross contamination. The method used during these analyses follows the detailed procedure described by Lindquist *et al*. (2001) with two exceptions: (1) the samples were not treated with 4,6-diamidino 2 phenyl-indole dihydrochloride (DAPI); and (2) the samples were treated with 90 μl of goat serum (Sigma Chemical, St Louis, MO). DAPI was not used because oocyst nuclei were not visualized during the analyses. The goat serum helped to reduce non-specific binding of the fluorescent antibody (Rodgers *et al*., 1995).

Cryptosporidium feed suspension samples were treated with fluorescent antibody and goat serum, and analysed on a hemacytometer using the procedure described by Dugan *et al*. (2001). The feed suspension samples counted by hemacytometer were used to spike previously unseeded tap and raw water samples as positive controls to estimate recovery efficiencies. These spikes were performed separately from, and after the conclusion of, each pilot plant run. The spikes were always performed in triplicate. The spiked, positive control samples were included with raw (after seeding) and filter effluent samples collected from the pilot plant for analysis by solid phase cytometry. Recovery efficiencies in raw water ranged from 47% to 132%, with a mean of 69% and a standard deviation of 27%. Recovery efficiencies in tap water ranged from 82% to 111%, with a mean of 94% and a standard deviation of 10%. The recovery efficiencies for each run are listed in Tables 3–10. Tap water was used as a proxy for filter effluent water in spike recovery tests because the particulate concentrations were comparable and the chances of oocyst contamination were exceedingly low. Pilot-scale *Cryptosporidium* log removals were based on measured raw water samples (after seeding) versus measured filter effluent samples.

All spore analyses were performed using methods developed by Rice *et al*. (1996). Turbidity was measured by the nephelometric method (*Standard Methods,* 1998) (Hach 2100N, Loveland, CO). Particle count results were averaged from three measurements made with a light obscuration instrument (*Standard Methods,* 1998) (Hiac-Royco 9064, HR-LD150 detector, Grants Pass, OR). Particle removals were reported for the sum of all particles with diameters between 1 and 150 μm. TOC was measured using the UV-persulphate method (*Standard Methods,* 1998) (Tekmar-Dohrmann, Mason OH). Alkalinity and hardness were measured by titration (*Standard Methods,* 1998).

RESULTS

For each run, *Cryptosporidium* log removals as a function of filter bed volumes treated are summarized in Figures 2 and 3. TOC removals, *Cryptosporidium* recoveries, rates of head loss development, filter effluent turbidities and particle counts (1–150 μm), as well as *Cryptosporidium* and spore log removals for each run are presented in Tables 3–10. All data are presented as a function of the treated water volume in order to normalize for the effects of two different loading rates. The empty bed (no media) volume of the filter was 0.86 l. In terms of empty bed volumes, the filter flow rates at surface loading rates of 5 and 10 m/h were 7.0 and 14 empty bed volumes/h. *Cryptosporidium* log removals were always based on measured raw water samples (after seeding) versus measured filter effluent samples. Log removals that incorporate non-detects in the filter effluent are preceded with a 'greater than' (>) symbol. A quantity of 1 oocyst/sample volume was substituted for zero in log removal calculations involving non-detects. Although they are numbered sequentially in this paper, the runs were actually performed in random chronological order to eliminate a potential source of bias.

Cryptosporidium removals averaged >4.2 log ($\sigma = 0.1$ log; σ = standard deviation) with alum coagulation at 5 m/h and 20°C (Table 3, Figure 2(a), run 1). With ferric chloride coagulation, *Cryptosporidium* removals at 5 m/h and 20°C (Table 4, Figure 2(b), run 2) averaged >4.1 log ($\sigma = 0.1$ log). At the 95% confidence level ($P = 0.05$), this average was not significantly different from the average oocyst removal observed with alum

Table 3 | *Cryptosporidium* removals at 5 m/h, 20°C and 20 mg/l alum (run 1) (raw pH=8.1; raw TOC=2.24 mg/l; filter effluent TOC=1.97 mg/l; head loss development=1.7 cm/h; raw water *Cryptosporidium* recovery=75%; tap water *Cryptosporidium* recovery=90%)

Bed volumes filtered	Log removals (±std dev.)		Filter effluent turbidity (NTU)	Filter effluent particle counts[1] (#/ml)
	Cryptosporidium	Spores		
14	$>4.2 \pm 0.095$		0.066	360
21	>4.3	0.60	0.068	310
28	>4.1	1.3 ± 0.050	0.061	340
35	$>4.1 \pm 0.065$		0.063	350
41	>4.2		0.074	280
48	>4.2	1.1	0.061	320
55	$>4.0 \pm 0.15$		0.064	330

[1]1–150 μm size range.

under the same conditions. *Cryptosporidium* removals were constant with respect to treated water volumes in both runs.

At 10 m/h and 20°C, *Cryptosporidium* removals achieved with alum (Table 5, Figure 2(c), run 3) and ferric chloride (Table 6, Figure 2(d), run 4) coagulation averaged 1.9 log ($\sigma = 1.3$ log) and >4.1 log ($\sigma = 0.2$ log), respectively. These averages were significantly different ($P = 0.05$). Over the course of the runs, oocyst removals declined by 2.8 log during alum coagulation and 0.2 log during ferric chloride coagulation.

At 5 m/h and 4.5°C, *Cryptosporidium* removals with alum (Table 7, Figure 3(a), run 5) and ferric chloride (Table 8, Figure 3(b), run 6) averaged 2.5 log ($\sigma = 0.6$ log) and >4.2 log ($\sigma = 0.1$ log), respectively. These averages were significantly different ($P = 0.05$) from each other. Oocyst removals declined by 1.3 log during alum coagulation and 0.2 log during ferric chloride coagulation.

At 10 m/h and 4.5°C, *Cryptosporidium* removals with alum (Table 9, Figure 3(c), run 7) and ferric chloride (Table 10, Figure 3(d), run 8) averaged 1.0 log ($\sigma = 0.2$ log) and 1.4 log ($\sigma = 0.4$ log), respectively. These averages were not significantly ($P = 0.05$) different from each other. Over the course of the runs, oocyst removals during alum and ferric chloride coagulation declined by 0.6 log and 0.9 log, respectively.

Cryptosporidium removals averaged >4.1 to >4.2 log in those runs where oocyst removals declined by ≤0.2 log (runs 1, 2, 4 and 6). In those runs, *Cryptosporidium* removals were defined as not impaired. *Cryptosporidium* removals averaged 1.0 to 2.5 log in those runs where oocyst removals declined by 0.6 to 2.8 log (runs 3, 5, 7 and 8). In these runs, *Cryptosporidium* removals were defined as impaired. Impaired filtration removals of seeded *Cryptosporidium* oocysts tended to correlate with deteriorating filter effluent turbidities and particle counts. Turbidity increased by 0.2 to 0.5 NTU, and particles increased 4.6- to 32-fold over the course of those runs where *Cryptosporidium* removals were impaired. Effluent turbidities and particles either declined or remained stable in those runs where *Cryptosporidium* removals were not impaired. Effluent turbidities were less than 0.1 NTU in three of the four runs where *Cryptosporidium* removals were not impaired. Effluent turbidities were greater than 0.1 NTU in all of the runs where *Cryptosporidium* removals were impaired. The runs where *Cryptosporidium* removals were not impaired included the operating conditions (20°C, 5 m/h, runs 1 and 2) at which coagulant

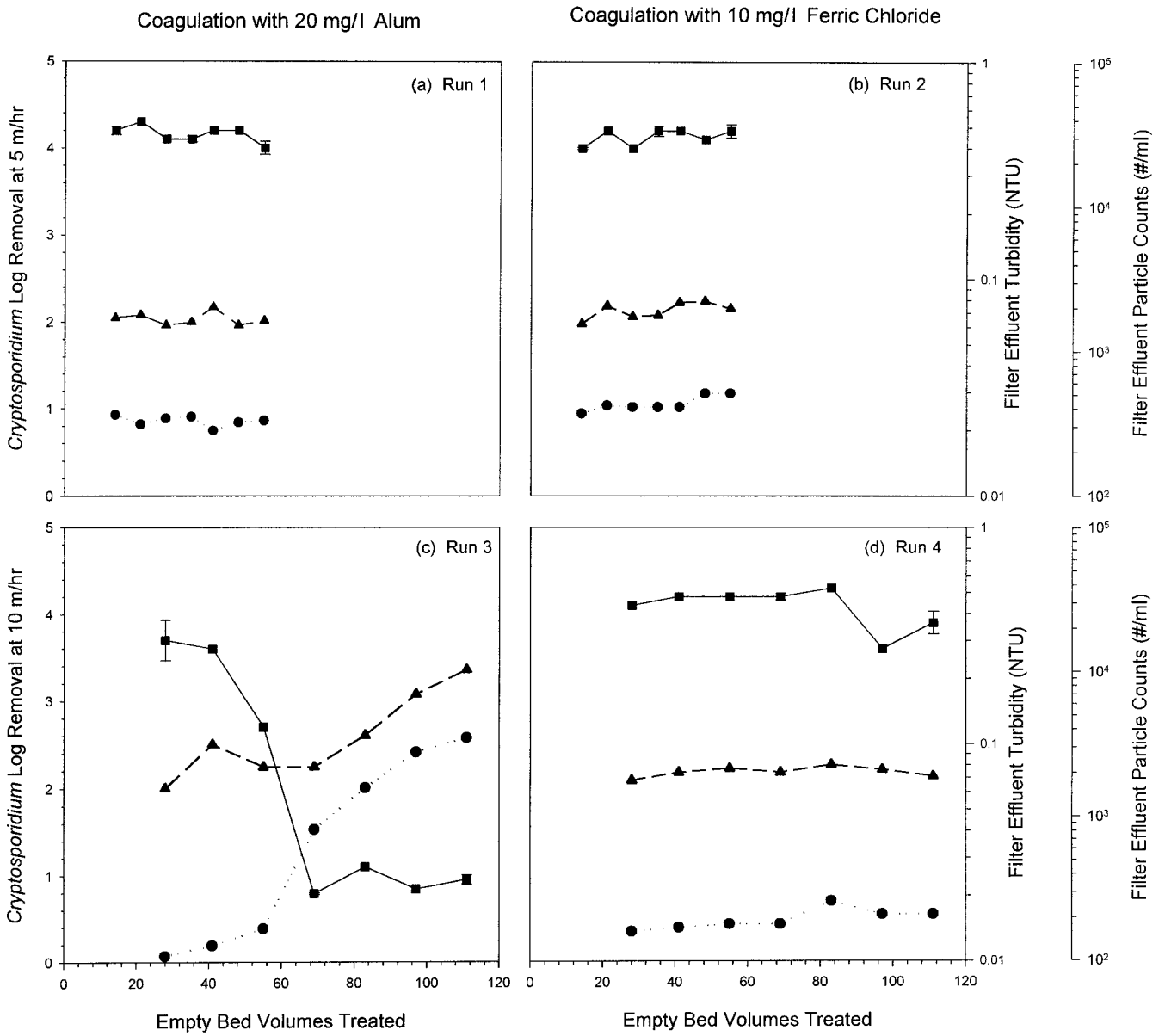

Figure 2 | *Cryptosporidium* removals and filter performance at 20°C: (a) 5 m/h, 20 mg/l alum; (b) 5 m/h, 10 mg/l ferric chloride; (c) 10 m/h, 20 mg/l alum; (d) 10 m/h, 10 mg/l ferric chloride; —■— *Cryptosporidium* log removal (left Y axis); – –▲– – filter effluent turbidity (1st right Y axis); · ·●· · filter effluent particle counts (2nd right Y axis).

doses were determined. *Cryptosporidium* was not added to the pilot plant during coagulant dose determination. The filter effluent turbidities during coagulant dose determination and runs 1 and 2, where *Cryptosporidium* was seeded, did not differ significantly. Consequently, it appears that oocyst addition did not affect coagulation performance.

Spore and *Cryptosporidium* removals did not correlate well. Combining log removal data from all eight runs, the observed correlation coefficient between these two variables was 0.24, with a significance probability of 0.26. These numbers indicate a degree of correlation that was low and statistically insignificant. Average removals of spores ranged from 1.0 to 1.1 log in those runs where

Table 4 | *Cryptosporidium* removals at 5 m/h, 20°C and 10 mg/l ferric chloride (run 2) (raw pH=8.1; raw TOC=2.12 mg/l; filter effluent TOC=2.01 mg/l; head loss development=2.3 cm/h; raw water *Cryptosporidium* recovery=58%; tap water *Cryptosporidium* recovery=86%)

Bed volumes filtered	Log removals (±std dev.)		Filter effluent turbidity (NTU)	Filter effluent particle counts[1] (#/ml)
	Cryptosporidium	Spores		
14	$> 4.0 \pm 0.021$		0.062	370
21	> 4.2	1.0	0.075	420
28	4.0	1.1 ± 0.15	0.067	410
35	$> 4.2 \pm 0.12$		0.068	410
41	4.2		0.078	410
48	> 4.1	0.90	0.079	510
55	$> 4.2 \pm 0.16$		0.073	510

[1]1–150 μm.

Table 5 | *Cryptosporidium* removals at 10 m/h, 20°C and 20 mg/l alum (run 3) (raw pH=8.0; raw TOC=2.17 mg/l; filter effluent TOC=1.92 mg/l; head loss development=5.7 cm/h; raw water *Cryptosporidium* recovery=58%; tap water *Cryptosporidium* recovery=93%)

Bed volumes filtered	Log removals (±std dev.)		Filter effluent turbidity (NTU)	Filter effluent particle counts[1] (#/ml)
	Cryptosporidium	Spores		
28	3.7 ± 0.47		0.063	110
41	3.6	1.8	0.10	130
55	2.7	1.9 ± 0.0	0.079	170
69	0.79 ± 0.015		0.079	830
83	1.1		0.11	1,600
97	0.84	1.6	0.17	2,800
111	0.95 ± 0.11		0.22	3,500

[1]1–150 μm.

Cryptosporidium removals were not impaired, and from 0.7 to 1.8 log in those runs where *Cryptosporidium* removals were impaired.

The rates of head loss development correlated primarily with increases in filter loading rates and the type of coagulant. For a given coagulant and temperature, increasing the loading rate to 10 m/h was associated with a 2.6 fold ($\sigma = 0.5$) increase in the rate of head loss development. Changing from alum to ferric chloride increased the rate of head loss development by an average of 9.6% ($\sigma = 18\%$) at a given temperature and loading rate. Assuming linear head loss development and no

Table 6 | *Cryptosporidium* removals at 10 m/h, 20°C and 10 mg/l ferric chloride (run 4) (raw pH=8.0; raw TOC=2.15 mg/l; filter effluent TOC=2.00 mg/l; head loss development=5.0 cm/h; raw water *Cryptosporidium* recovery=47%; tap water *Cryptosporidium* recovery=94%)

Bed volumes filtered	Log removals (±std dev.)		Filter effluent turbidity (NTU)	Filter effluent particle counts[1] (#/ml)
	Cryptosporidium	Spores		
28	4.1		0.068	160
41	>4.2	1.0	0.074	170
55	4.2	1.1 ± 0.0	0.077	180
69	$>4.2 \pm 0.076$		0.074	180
83	>4.3		0.080	260
97	3.6	1.1	0.076	210
111	$>3.9 \pm 0.26$		0.071	210

[1]1–150 μm.

Table 7 | *Cryptosporidium* removals at 5 m/h, 4.5°C and 20 mg/l alum (run 5) (raw pH=7.9; raw TOC=1.94 mg/l; filter effluent TOC=1.75 mg/l; head loss development=1.7 cm/h; raw water *Cryptosporidium* recovery=65%; tap water *Cryptosporidium* recovery=90%)

Bed volumes filtered	Log removals (±std dev.)		Filter effluent turbidity (NTU)	Filter effluent particle counts[1] (#/ml)
	Cryptosporidium	Spores		
14	2.9 ± 0.54		0.14	140
21	3.1	2.1	0.099	180
28	3.0	1.8 ± 0.20	0.10	150
35	2.5 ± 0.044		0.087	240
41	2.0		0.094	350
48	2.1	1.5	0.15	580
55	1.6 ± 0.12		0.64	870

[1]1–150 μm.

breakthrough, projected filter run times to reach a terminal head loss of 2.5 m ranged from 98 to 130 h at 5 m/h. Projected filter run times at 10 m/h ranged from 32 to 39 h.

The non-impaired *Cryptosporidium* removals were at the high end of those previously reported for direct and in-line filtration (Table 1), and compare favourably with total log removals in conventional treatment processes that included clarification prior to filtration (LeChevallier *et al*., 1991, 1992; Patania *et al*., 1995; States *et al*., 1997; Dugan *et al*., 2001). The >4.1 to >4.2 logs of non-impaired *Cryptosporidium* removals were comfortably higher than

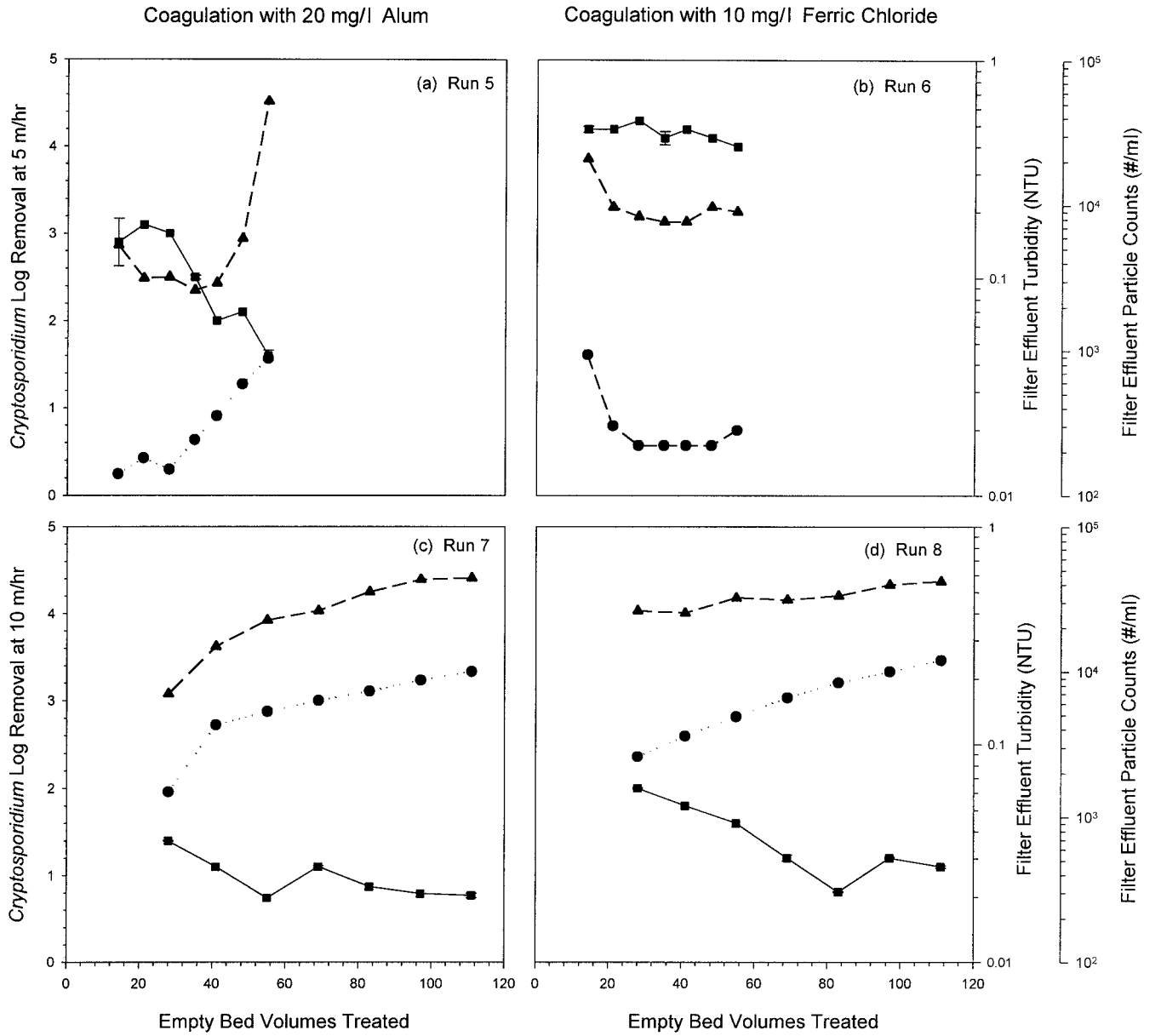

Figure 3 | *Cryptosporidium* removals and filter performance at 4.5°C: (a) 5 m/h, 20 mg/l alum; (b) 5 m/h, 10 mg/l ferric chloride; (c) 10 m/h, 20 mg/l alum; (d) 10 m/h, 10 mg/l ferric chloride; —■— Cryptosporidium log removal (left Y axis); – –▲– – filter effluent turbidity (1st right Y axis); · ·●· · filter effluent particle counts (2nd right Y axis).

the 2.0 log credit the US EPA grants for direct filtration under the Interim Enhanced Surface Water Treatment Rule (Federal Register, 1998), or the 2.5 logs that the US EPA is considering under the LT2ESWTR (USEPA, 2001*a*,*b*).

The exclusive use of either alum or ferric chloride was not necessarily representative of coagulation practice at direct filtration facilities. Many plants use lower doses of inorganic salts combined with a polymer. The advantage of using alum and ferric chloride was that the observed

Table 8 | *Cryptosporidium* removals at 5 m/h, 4.5°C and 10 mg/l ferric chloride (run 6) (raw pH=8.0; raw TOC=2.27 mg/l; filter effluent TOC=2.00 mg/l; head loss development=1.9 cm/h; raw water *Cryptosporidium* recovery=63%; tap water *Cryptosporidium* recovery=111%)

Bed volumes filtered	Log removals (±std dev.)		Filter effluent turbidity (NTU)	Filter effluent particle counts[1] (#/ml)
	Cryptosporidium	Spores		
14	4.2 ± 0.070		0.35	930
21	4.2	1.0	0.21	300
28	> 4.3	0.95 ± 0.050	0.19	220
35	> 4.1 ± 0.15		0.18	220
41	> 4.2		0.18	220
48	4.1	0.90	0.21	220
55	4.0 ± 0.0		0.20	280

[1]1–150 μm.

Table 9 | *Cryptosporidium* removals at 10 m/h, 4.5°C and 20 mg/l alum (run 7) (head loss development=4.3 cm/h; raw water *Cryptosporidium* recovery=57%; tap water *Cryptosporidium* recovery=82%)

Bed volumes filtered	Log removals (±std dev.)		Filter effluent turbidity (NTU)	Filter effluent particle counts[1] (#/ml)
	Cryptosporidium	Spores		
28	1.4 ± 0.014		0.17	1,500
41	1.1	1.0	0.28	4,300
55	0.74	0.95 ± 0.15	0.37	5,300
69	1.1 ± 0.023		0.41	6,300
83	0.87		0.50	7,300
97	0.79	0.70	0.57	8,700
111	0.77 ± 0.052		0.58	10,000

[1]1–150 μm.

results did not depend on proprietary chemical formulations specific to a certain manufacturer. Rather than serving as a recommendation for the use of certain coagulants and doses, the results of this study should be interpreted as helping to define the range of *Cryptosporidium* removals that could be achieved with direct filtration. The data imply that a direct filtration plant, optimized for turbidity control, is capable of achieving high *Cryptosporidium* removals. Because the coagulants and doses in this study were optimized for 20°C and 5 m/h,

Table 10 | *Cryptosporidium* removals at 10 m/h, 4.5°C and 10 mg/l ferric chloride (run 8) (raw pH=7.9; raw TOC=1.54 mg/l; filter effluent TOC=1.36 mg/l; head loss development=4.6 cm/h; raw water *Cryptosporidium* recovery=132%; tap water *Cryptosporidium* recovery=109%)

Bed volumes filtered	Log removals (±std dev.)		Filter effluent turbidity (NTU)	Filter effluent particle counts[1] (#/ml)
	Cryptosporidium	Spores		
28	2.0 ± 0.015		0.41	2,600
41	1.8	0.80	0.40	3,600
55	1.6	0.80 ± 0.10	0.47	4,900
69	1.2 ± 0.066		0.46	6,600
83	0.81 ± 0.010		0.48	8,400
97	1.2 ± 0.022	0.60	0.54	10,000
111	1.1 ± 0.030		0.56	12,000

[1]1–150 μm.

Cryptosporidium removals and filter effluent quality tended to decline as temperatures dropped and loading rates rose. Consequently, the data obtained under these conditions should help to quantify the relative impacts of non-optimal coagulant selection and dosing. A full-scale direct filtration facility would obviously vary its coagulant selection and dose to maintain consistently high effluent quality.

CONCLUSIONS

Eight pilot-scale runs evaluated the removal of seeded *Cryptosporidium* oocysts by direct filtration under varying operating conditions. The pilot plant achieved greater than 4 log *Cryptosporidium* removals in several of those runs. High *Cryptosporidium* removals were associated with low filter effluent turbidities and particle counts. *Cryptosporidium* control tended to deteriorate in those runs when water temperatures and filter loading rates deviated from the values at which the coagulants had been optimized. Average removals in those runs were never higher than 2.5 log. The lower *Cryptosporidium* removals were associated with higher filter effluent turbidities and increasing particle counts.

ACKNOWLEDGEMENTS

The authors thank Christopher Parrett for performing numerous spore analyses, and Samuel Hayes and James Owens for producing the oocysts used in these studies. Any opinions expressed in this article are those of the authors and do not necessarily reflect the official positions and policies of the US EPA. Any mention of products or trade names does not constitute endorsement or recommendation for use by the US EPA.

NOMENCLATURE

DAPI	4,6-diamidino 2 phenyl-indole dihydrochloride
$P = 0.05$	95 percent confidence level
$Al_2(SO_4)_3 \cdot 14H_2O$	aluminum sulphate
cm	centimetre
°C	degrees Celsius

$FeCl_3 \cdot 6H_2O$	ferric chloride
G	fluid velocity gradient
$>$	greater than
HDPE	high density polyethylene
h	hour
ICR	Information Collection Rule
$<$	less than
$\leq$	less than or equal to
l	litre
l/min	litres per minute
LT2ESWTR	Long Term 2 Enhanced Surface Water Treatment Rule
m/h	metres per hour
μg/ml	micrograms per millilitre
μl	microlitre
μm	micrometre
mg/l	milligrams per litre
ml	millilitre
ml/min	millilitres per minute
mm	millimetre
min	minute
M	molar
nm	nanometres
NTU	nephelometric turbidity units
oocysts/l	oocysts per litre
oocysts/ml	oocysts per millilitre
%	percent
PBS	phosphate buffered saline
$\pm$	plus or minus
sec^{-1}	reciprocal seconds
sec	seconds
σ	standard deviation
TOC	total organic carbon
US EPA	United States Environmental Protection Agency
units/ml	units per millilitre
UV	ultraviolet
UV_{254}	ultraviolet light absorbance at 254 nanometres

REFERENCES

Cleasby, J. & Logsdon, G. 1999 Granular bed and precoat filtration. In *Water Quality and Treatment,* 5th Edn. McGraw-Hill, New York.

Dugan, N., Fox, K., Owens, J. & Miltner, R. 2001 Controlling *Cryptosporidium* oocysts using conventional treatment. *J. Am. Wat. Wks Assoc.* **93**(12), 64–76.

Edzwald, J., Kelley, M., Dunn, H., Kaminski, G. & Malley, J. 1996 Control of *Cryptosporidium* by coagulation, flotation and filtration. *Proceedings: AWWA Water Quality Technology Conference.* Boston, Massachusetts.

Edzwald, J. & Kelley, M. 1998 Control of *Cryptosporidium*: from reservoirs to clarifiers to filters. *Wat. Sci. Technol.* **37**(2), 1–8.

Federal Register 1989 *National Primary Drinking Water Regulations; Filtration, Disinfection, Turbidity,* Giardia lamblia*, Viruses,* Legionella *and Heterotrophic Bacteria; Final Rule*. 54:124:27486 (June 29).

Federal Register 1998 *National Primary Drinking Water Regulations: Interim Enhanced Surface Water Treatment; Final Rule*. 63:241:69478 (December 16).

Goldgrabe, J., Summers, R. & Miltner, R. 1993 Particle removal and head-loss development in biological filters. *J. Am. Wat. Wks Assoc.* **85**(12), 94–106.

LeChevallier, M., Norton, W. & Lee, R. 1991 *Giardia* and *Cryptosporidium* in filtered drinking water supplies. *Appl. Environ. Microbiol.* **57**(9), 2617–2621.

LeChevallier, M. & Norton, W. 1992 Examining relationships between particle counts and *Giardia*, *Cryptosporidium* and turbidity. *J. Am. Wat. Wks Assoc.* **84**(12), 54–60.

Lindquist, H., Ware, M., Stetler, R., Wymer, L. & Schaefer, F. 2001 A comparison of four fluorescent antibody-based methods for purifying, detecting and confirming *Cryptosporidium parvum* in surface waters. *J. Parasitol.* **87**(5), 1124–1131.

Lytle, D., Fox, K., Miltner, R., Speth, T., Schock, M., Kropp, K., Shukairy, H., Fichter, L. & Dugan, N. 1998 The design of a mini-scale conventional/filtration water treatment plant for *Cryptosporidium* research. *Proceedings, AWWA Annual Conference,* Dallas, Texas.

Miltner, R., Summers, R. & Wang, J. 1995 Biofiltration performance. Part 2, Effect of backwashing. *J. Am. Wat. Wks Assoc.* **87**(12), 64–70.

Nieminski, E. & Ongerth, J. 1995 Removing *Giardia* and *Cryptosporidium* by conventional treatment and direct filtration. *J. Am. Wat. Wks Assoc.* **87**(9), 96–106.

Ongerth, J. & Pecoraro, J. 1995 Removing *Cryptosporidium* using multi-media filters. *J. Am. Wat. Wks Assoc.* **87**(12), 83–89.

Patania, N., Jacangelo, J., Cummings, L., Wilczak, A., Riley, K. & Oppenheimer, J. 1995 *Optimization of Filtration for Cyst Removal*. AWWA Research Foundation, Denver, Colorado.

Rice, E., Fox, K., Miltner, R., Lytle, D. & Johnson, C. 1996 Evaluating plant performance with endospores. *J. Am. Wat. Wks Assoc.* **88**(9), 122–130.

Rodgers, M., Flanigan, D. & Jakubowski, W. 1995 Identification of algae which interfere with the detection of *Giardia* cysts and *Cryptosporidium* oocysts and a method for alleviating this interference. *Appl. Environ. Microbiol.* **61**(10), 3759–3763.

Standard Methods for the Examination of Water and Wastewater 1998 19th Edn. American Public Health Association/American Water Works Association/Water Environment Federation, Washington, DC.

States, S., Stadterman, K., Ammon, L., Vogel, P., Baldizar, J., Wright, D., Conley, L. & Sykora, J. 1997 Protozoa in river water: sources, occurrence and treatment. *J. Am. Wat. Wks Assoc.* **89**(9), 74–83.

Stevens, A., Moore, L. & Miltner, R. 1989 Formation and control of non-trihalomethane disinfection by-products. *J. Am. Wat. Wks Assoc.* **81**(8), 54–60.

Swaim, P., Heath, M., Patania, N., Wells, W. & Trussell, R. 1996 High-rate direct filtration for *Giardia* and *Cryptosporidium* removal. *Proceedings: AWWA Annual Conference*. Toronto, Canada.

US EPA 1999 *ICR Auxiliary 1 Database, version 3.0*. EPA 815-C-99-003. Washington, D.C.

US EPA 2001*a* Draft Long Term 2 Enhanced Surface Water Treatment Rule Preamble Language. *http://www.epa.gov/OGWDW/lt2/lt2_preamble.pdf*.

US EPA 2001*b* Draft Long Term 2 Enhanced Surface Water Treatment Rule Regulatory Language. *http://www.epa.gov/OGWDW/lt2/lt2_reglanguage.pdf*.

Wang, J., Summers, R. & Miltner, R. 1995 Biofiltration performance. Part 1, Relationship to biomass. *J. Am. Wat. Wks Assoc.* **87**(12), 55–63.

First received 18 November 2002; accepted in revised form 20 January 2003